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Indonesian coastal wetlands

The past dynamics of environment and carbon, the potential for climate change mitigation and the assessment for conservation, management and restoration

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To the pretty face my eyes stuck on

Content

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Summary

As an archipelago country, Indonesia harbors vast area of coastal wetlands such as mangrove and peatland that stores around 50 Gt carbon (C). Indonesian coastal wetlands are also rich in biodiversity, providing habitats for over 2000 species of flora and fauna. However, coastal wetlands in Indonesia potentially sustain significant loss due to inundation following the upcoming rapid sea level rise in addition to landuse change and conversion following rapid population growth in the coastal areas.

To reduce the impact of sea level rise and to ensure the sustainability of these ecosystems and their important functions, conservation, management and restoration of Indonesian coastal wetlands are necessary to be conducted. As conservation, management and restoration can be costly while sufficient resources are not always available, effective and efficient strategies are required. For this, a thorough ecological understanding is critical. However, current knowledge relating to coastal wetlands in Indonesia remains a large gap. Moreover, thorough ecological understanding is often difficult to gather as some ecosystem processes and their responses to environmental changes often occur after a long period of time. Therefore, long term ecological studies, such as palaeoecology, are required to conduct.

Thereupon, this multi-proxy palaeoecological research that includes was carried out on two Indonesian coastal wetlands, Sungai Buluh peatland in Central Sumatra and Segara Anakan lagoon in Central Java, in order to: (1) improve the insights on the environmental process and dynamics of coastal wetlands in Indonesia as well as the mechanisms and dynamics of their capacity to store C; (2) underline their potential for climate change mitigation; and (3) provide consideration to improve conservation, management and restoration of coastal wetlands in Indonesia.

This research suggests that Indonesian coastal wetlands and their capacities to sequester C are very dynamic and mainly influenced by sea level changes, climatic variabilities and human activities. These factors mainly alter the hydrological conditions and vegetation composition of the coastal wetlands, although the impacts can be site-specific. Coastal wetlands in Indonesia also have high potential for global climate change mitigation for they have considerable capacities to accumulate C compared to similar ecosystems globally. Additionally, Indonesian coastal wetlands have shown resilience to past disturbances, both natural and anthropogenic.

Based on the outcomes, some considerations to improve coastal wetlands conservation, management and restoration are provided. This includes (1) setting up priorities or classifying the importance of species, ecosystems, areas, or actions, in order to help solving the most critical problems, optimizing the efforts and allocating the resource; (2) incorporating socioeconomic needs in conservation and sustainable use; (3) selecting effective and efficient restoration approach; and (4) evaluating policy design and implementation.

Zusamenfassung

Als ein Archipel-Land beherbergt Indonesien ein großes Gebiet von Küstenfeuchtgebieten wie Mangroven und Torfmoore, die etwa 50 Gt Kohlenstoff (C) speichern. Indonesische Küstenfeuchtgebiete sind auch reich an biologischer Vielfalt und bieten Lebensraum für mehr als 2000 Arten von Flora und Fauna. Die Küstenfeuchtgebiete in Indonesien könnten jedoch aufgrund des bevorstehenden schnellen Anstiegs des Meeresspiegels zusätzlich zu Landnutzungsänderungen und -umstellungen infolge des rapiden Bevölkerungswachstums in den Küstengebieten erhebliche Verluste aufgrund von Überschwemmungen erleiden.

Um die Auswirkungen des Meeresspiegelanstiegs zu verringern und die Nachhaltigkeit dieser Ökosysteme und ihrer wichtigen Funktionen zu gewährleisten, müssen die indonesischen Küstenfeuchtgebiete geschützt, bewirtschaftet und wiederhergestellt werden. Da Konservierung, Management und Wiederherstellung kostspielig sein können, während nicht immer genügend Ressourcen zur Verfügung stehen, sind wirksame und effiziente Strategien erforderlich. Dafür ist ein gründliches ökologisches Verständnis von entscheidender Bedeutung. Das derzeitige Wissen über Küstenfeuchtgebiete in Indonesien bleibt jedoch eine große Lücke. Darüber hinaus ist ein gründliches ökologisches Verständnis oft schwierig zu erfassen, da einige Ökosystemprozesse und ihre Reaktionen auf Umweltveränderungen oft nach langer Zeit auftreten. Daher sind langfristige ökologische Studien, wie Paläoökologie, erforderlich.

Daraufhin wurde diese paläoökologische Multi-Proxy-Forschung an zwei indonesischen Küstenfeuchtgebieten, dem Sungai Buluh-Moor in Zentral-Sumatra und der Lagune Segara Anakan in Zentral-Java durchgeführt, um (1) die Erkenntnisse über den

Umweltprozess und die Umweltdynamik zu verbessern Küstenfeuchtgebiete in Indonesien sowie die Mechanismen und Dynamiken ihrer Fähigkeit, C zu lagern; (2) unterstreichen ihr Potenzial zur Eindämmung des Klimawandels; und (3) Überlegungen zur Verbesserung der Erhaltung, Bewirtschaftung und Wiederherstellung von Küstenfeuchtgebieten in Indonesien anzustellen.

Diese Untersuchung legt nahe, dass indonesische Küstenfeuchtgebiete und ihre Fähigkeit, C zu sequestrieren, sehr dynamisch sind und hauptsächlich von Veränderungen des Meeresspiegels, klimatischen Schwankungen und menschlichen Aktivitäten beeinflusst werden. Diese Faktoren verändern hauptsächlich die hydrologischen Bedingungen und die Zusammensetzung der Vegetation der Küstenfeuchtgebiete, obwohl die Auswirkungen ortsspezifisch sein können. Küstenfeuchtgebiete in Indonesien haben auch ein hohes Potenzial für eine globale Minderung des Klimawandels, da sie über beträchtliche Kapazitäten zur Akkumulation von C im Vergleich zu ähnlichen Ökosystemen weltweit verfügen. Darüber hinaus haben die indonesischen Küstenfeuchtgebiete Widerstandsfähigkeit gegenüber natürlichen und anthropogenen Störungen gezeigt.

Auf der Grundlage der Ergebnisse werden einige Überlegungen zur Verbesserung der Erhaltung, Bewirtschaftung und Wiederherstellung von Küstenfeuchtgebieten vorgelegt. Dazu gehören (1) die Festlegung von Prioritäten oder die Klassifizierung der Bedeutung von Arten, Ökosystemen, Gebieten oder Maßnahmen, um zur Lösung der kritischsten Probleme beizutragen, die Bemühungen zu optimieren und die Ressourcen zuzuweisen; (2) Einbeziehung sozioökonomischer Bedürfnisse in die Erhaltung und nachhaltige Nutzung; (3) Auswahl eines effektiven und effizienten Wiederherstellungskonzepts; und (4) Bewertung des Politikentwurfs und der Umsetzung.

Chapter 1

Introduction

Wetlands are multiple-value ecosystems that hold ecological, social and economic importance such as flood control, biodiversity and natural resource provisioning, water regulation and quality protection as well as education and recreational purposes (Boavida, 1999; Mitsch and Gosselink, 2000; Zedler and Kercher, 2005; Crooks et al., 2011). However, their unfortunate reputation for being gloomy, source of diseases, harbor of mosquitos and dangerous animals, etc. has guided the society to undervalue wetlands for centuries (Giblett, 1996; Boavida, 1999; Friess, 2016). For those disservices, wetlands are often regarded as waste lands (Ramsar Convention, 1982).

Globally, wetlands cover an estimated area of 5800 to 12800 Mha (Ramsar Convention, 1982; Matthews and Fung, 1987; Finlayson et al., 1999; Zedler and Kercher, 2005). Due to the flooded or saturated condition of these ecosystems which imped the decay process of organic matter (OM), wetlands also play important role in carbon (C) cycle by storing around 500 Gt C or one-third of total terrestrial soil C (Boavida, 1999; Mitra et al., 2003; Crooks et al., 2011; Keller, 2011). Following the growing population and economy, around half of the total global wetland coverage is estimated to be lost (Zedler and Kercher, 2005), although a more realistic estimation cannot be made owing to lack of documentation in the past (Boavida, 1999). These ecosystems were drained to meet the suitable criteria for agriculture, settlements or industrial purposes while disregarding their ecosystem services and values (Zedler and Kercher, 2005; Crooks et al., 2011).

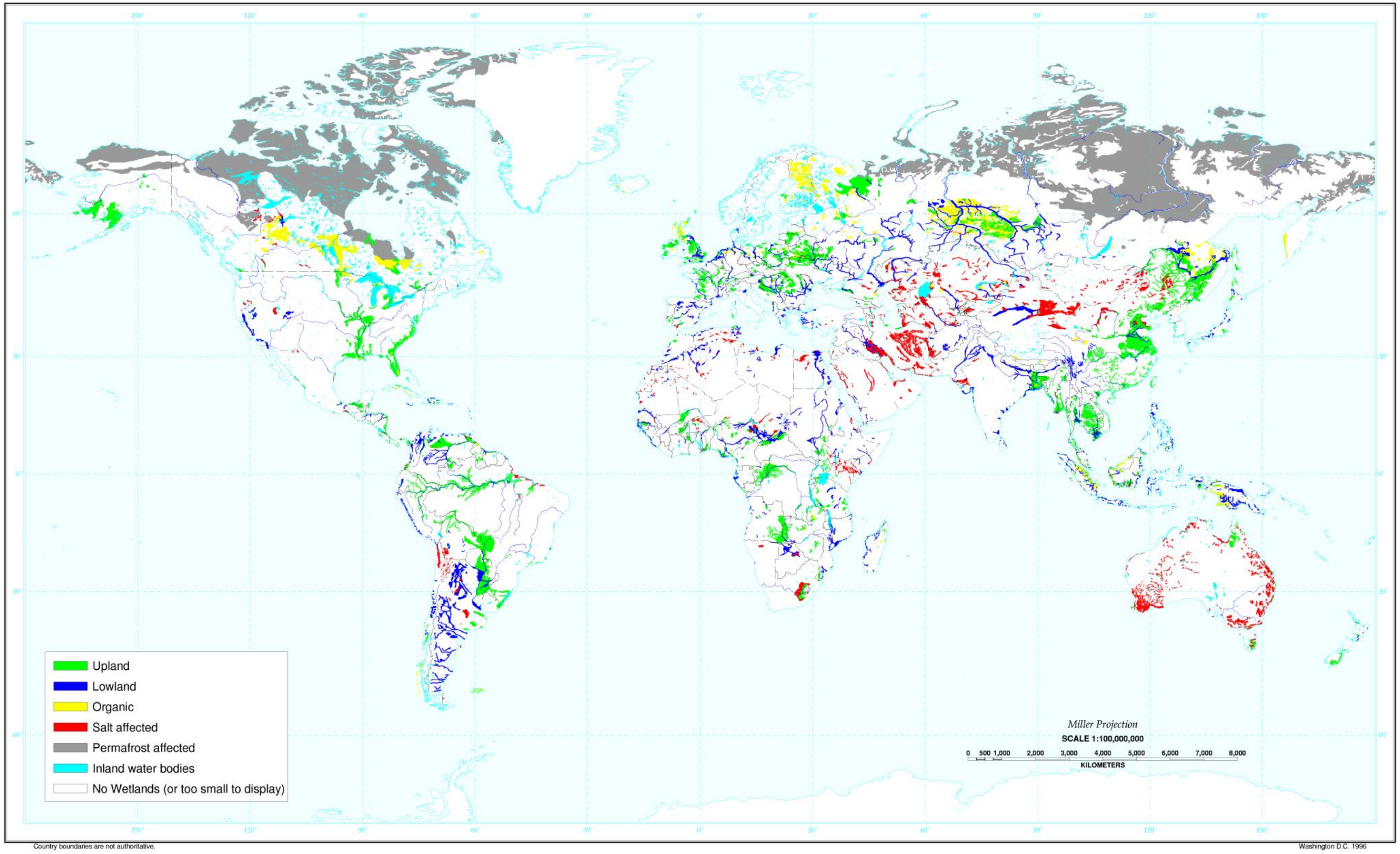

Fig. 1.1. Wetland distribution from https://www.nrcs.usda.gov/

A large proportion of global wetlands are distributed in the coastal areas that are vulnerable to impacts from sea level rise (SLR; Fig. 1.1; Webb et al., 2013). Predicted to rise 3.4 m per century and significantly accelerate due to the warming climate and ice melt (Nerem et al., 2010; Nicholls and Cazenave, 2010; Rahmstorf, 2010), SLR potentially destroys up to 20% of global coastal wetlands by the 2080s (Nicholls, 2004). This risk is increased by the growing population in the coastal areas that cause coastal ecosystem degradation around 1% per year (Zedler and Kercher, 2005). Such loss of coastal wetlands will cause devastating consequences such as carbon emissions and increased risk of storm events (Nicholls and Tol, 2006).

With over 80,000 km-long coastal line, Indonesia is projected to be the most vulnerable country to SRL (Mcleod et al., 2010; Measey, 2010; Zikra et al., 2015). A model with an assumption the rate of SLR is less than 1 cm per year predict that Indonesia will lose around 2 Mha of its coastal wetlands by 2100 (Mcleod et al., 2010). Such loss will lead to a casualty of over US$ 600 million that can increase by a factor of three if any adaptation or mitigation is not taken (Mcleod et al., 2010), without taking into account the values of ecosystem services that, in fact, beyond estimation (Boavida, 1999). Moreover, the damage on Indonesian coastal wetlands will also threaten around 50 Gt C stored as mangrove, seagrass and, mostly, peatland (Keller, 2011; Page et al., 2011; Alongi et al., 2016).

1.1. Coastal wetlands in Indonesia

1.1.1. Characteristics and classification

Wetlands are commonly defined as transitional zones between terrestrial and aquatic systems (Cowardin et al., 1979; Boavida, 1999; Margono et al., 2014a). It is also defined as the areas “where water is the primary factors controlling the environment and the associated plant and animal life" (Niering, 1985). These ecosystems are commonly characterized by their inundated lands or their high water table which are usually at or close to the surface (Cowardin et al., 1979; Margono et al., 2014a).

According to the Committee on Characterization of Wetlands (1995), wetlands have three main characterizations on the water, substrate and vegetation. To be classified as wetlands, an ecosystem needs to be inundated above the upper root layer for at least 15 days during the growing season in most years. Most wetlands also have hydric soil substrates which are saturated and lack of oxygen. Due to their saturated and low oxygen condition, many plants cannot tolerate and adapt, leaving the hydrophytic plants to occupy these ecosystems (Committee on Characterization of Wetlands, 1995). Despite of the common description of wetlands, an ecosystem cannot be classified as wetland if one amongst the three main characteristics, particularly of its hydrological criteria, is not fulfilled (Committee on Characterization of Wetlands, 1995; Boavida, 1999).

In regards to their water chemistry, wetlands can be classified as saline/brackish water and freshwater wetlands (Warner and Rubec, 1997). Saline/brackish water wetlands are the wetland ecosystems which hydrology contains dissolved salt (sodium), which are sourced or influenced from sea water, salt spray and/or inland saltwater (Warner and Rubec, 1997). On the other hand, freshwater wetlands receive very little or no dissolved salt and can further classified as minerotrophic and ombrothopic in respect of the water source (Warner and Rubec, 1997). Minerotrophic wetlands receive the influence of groundwater which usually contain dissolved minerals such as calcium, magnesium and sodium, whereas the hydrology of ombrotrophic wetlands is influenced exclusively from precipitation which contain low or no dissolved mineral (Warner and Rubec, 1997).

In Indonesia saline/brackish water coastal wetlands covered an approximate area 60 Mha, mainly of mangrove and seagrass (Alongi et al., 2016). In contrast, the coverage of Indonesian freshwater coastal wetlands is around 20 Mha, consist mainly of ombrotrophic peatland (Page et al, 2011; Osaki et al., 2016). These ecosystems are distributed mostly on the coastal area of Kalimantan, Sumatra, Papua and Java with the share of 30.8, 30.1, 29.9 and 4.7 %, respectively (Fig. 1.2; Margono et al., 2014a).

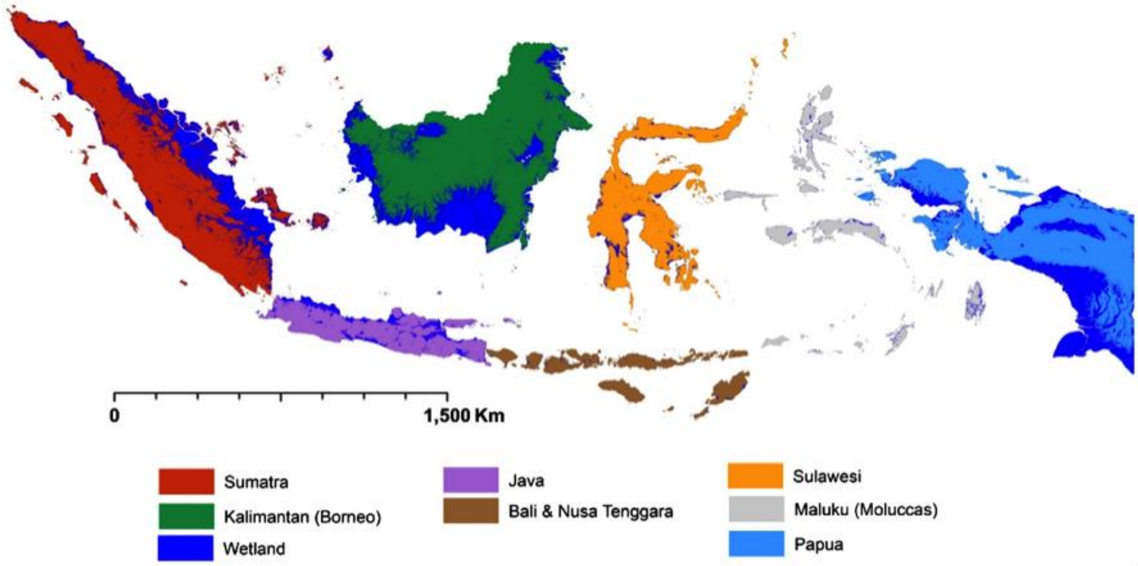

Fig. 1.2. Wetland distribution in Indonesia (Margono et al., 2014a)

1.1.2. Significance and issues

Indonesian coastal wetlands are very complex and dynamics ecosystems that play a significant role in global carbon cycle (Donato et al., 2011; Page et al., 2011; Alongi et al., 2016). Around 10% of total global peat carbon pool is stored mainly in the coastal area of Indonesia (Page et al., 2011; Yu et al., 2011). Moreover, Indonesian mangrove and seagrass ecosystems hold around 17% of global blue carbon reservoir (Alongi et al., 2016). Despite of their importance, the information regarding C sequestration in

Indonesian coastal wetlands remains a large gap (Page et al., 2004; Dommain et al., 2011; Donato et al., 2011). Although a large C proportion (49-98%) of the Indonesian coastal wetlands are stored as belowground C (Donato et al., 2011), information regarding belowground C of coastal wetlands in Indonesia is limited. As a consequence, the more realistic number of C storage in Indonesia cannot be made and the understanding on the factors controlling the C dynamics is sparse (Page et al., 2004; Dommain et al., 2011; Donato et al., 2011; Charman et al., 2013).

Fig. 1.3. Critically endangered species that inhabit Indonesian coatal wetlands: Orangutan (left; courtesy of Borneo Nature Foundation, 2017) and Sumatran tiger (right; courtesy of Jonathan Carre, 2008)

Indonesian coastal wetlands are also important mainly for their biodiversity provisioning (Sukarjo, 2004; Hutomo and Moosa, 2005; Polidoro et al., 2010; Posa et al., 2011). More than 2000 species of flora and fauna are recorder from Indonesian wetlands, although due to limited studies conducted, this number could be underestimated (Hutomo and Moosa, 2005; Posa et al., 2011). According to IUCN red list (www.iucnredlist.org/), some of these species such as *Bruguiera hainesii* (mangrove tree; Polidoro et al., 2010; Ono et al., 2016), *Betta persephone, B. miniopinna* and *B. spilotogena* (fishes; Posa et al., 2011) are critically endangered. Whereas some other species are listed as endangered, such as orangutan (*Pongo sp.*; Fig. 1.3), Sumatran tiger (*Panthera tigris sumatrae*; Fig. 1.3), proboscis monkey (*Nasalis larvatus*), rough-nose stingray (*Pastinachus solocirostris*) as well as the largest (*Rafflesia arnoldi*) and the tallest flower (*Amorphophallustitanium titanium*; www.iucnredlist.org/; Posa et al., 2011). Furthermore, Indonesian wetlands also harbor economically high value species such as ramin trees (*Gonystylus bancanus*; Posa et al., 2011), red meranti tree (*Shorea pauciflora*; Soerianegara and Lemmens, 1994), the giant tiger, Indian and banana prawns (*Penaeus monodon*, *P. indicus* and *P. merguiensis*; Sukarjo, 2004) also the giant mud, blue and red-spotted-swimming crabs (*Scylla serrata*, *Neptunus pelagicus* and *N. sanguinolentus*; Sukarjo, 2004).

Fig. 1.4. Patches of converted peatland surrounded by unconverted peat-swamp forest (http://www.desdemonadespair.net)

As to other coastal wetlands globally, the current rapidly changing environment also potentially shift the Indonesian wetlands from ‘significant carbon sink’ to ‘enormous carbon source’ (Eong, 1993; Li et al., 2007; Page et al., 2011). For instance, ENSO-related fire in 1997 alone has released over 2 Gt peat C to the atmosphere which is equal to one fourth of global annual fossil fuel emission in 2008 (Page et al., 2002; Le Quéré et al., 2009). Moreover, SLR, which globally predicted to increase around 2 m per year, potentially jeopardize around 8 Gt C stored on the lowland coastal tropical peatlands, which are mainly distributed in

Indonesia (Page et al., 2011; Whittle and Galego-Sala, 2016). In addition, mangrove deforestation released around 0.1 Gt C annually (Siikamäki et al., 2012)

Such risks are further worsened by the growing population and economy of Indonesia (Polidoro et al., 2010; Page and Baird, 2016). Only between 1990 and 2010, the development of palm oil and pulpwood (*Acacia*) plantation has converted over 8.5 Mha of Indonesian peatland (Jauhiainen et al., 2012; Miettinen et al., 2012; Fig. 1.4). The development of settlements also has reclaimed over 2 Mha of Indonesian mangrove areas during the period of 1982-2010 (Siburian and Haba, 2016). In addition, pollution and over exploitation also increase the stress on Indonesian coastal wetlands (Wong, 2004). This would not only potentially release a significant amount of C to the atmosphere, but also loss of biodiversity provisioning and ecosystem services (Polidoro et al., 2010; Page and Baird, 2016).

1.1.3. Conservation, management and restoration of Indonesian wetlands

In regards to the potential risk and threat of rapidly changing climate and ongoing severe human impact, the values and functions of coastal wetlands need to be maintained through conservation, management and restoration (Posa et al., 2011; Murdiyarso et al., 2015; Friess et al., 2016; Page and Baird, 2016). To protect and/or conserve an ecosystem, it is important to understand the values of the ecosystems (Adame et al., 2015; Friess et al., 2016). As often the needs to conserve an ecosystem intersect with socioeconomic interests, such understanding is crucial to strengthen the argument why a protection or conservation is necessary and why the investment for such action needs to be made (Maron et al., 2012; Friess et al., 2016).

In Indonesia, several efforts to conserve, manage and restore coastal wetlands have been widely conducted mainly to maintain and/or restore their C sink and biodiversity values (http://www.katinganproject.com; https://www.adb.org; Sukardi, 2010; van Eijk et al., 2010). There efforts are, however, considered to be ineffective as they failed to deliver the expected outcomes (ADB, 2006; Sukardi, 2010; van Eijk et al., 2010; Dharmawan et al., 2016), mainly due to the fragmented knowledge of the ecosystems and neglect to the socioeconomic as well as cultural aspects (van Eijk et al., 2010; Dharmawan et al., 2016).

Protected or conserved areas are often considered as "biological success and social failures" as they give positive ecological impacts but contradictory socioeconomic outcomes (Christie, 2004). This causes conflicts and tensions between local population and practitioners that consequently lead to non-cooperation, legal challenge, sabotage and even criminality (Holmes, 2013). The situation is worsened by the weak national policies for conservation and restoration (Friess et al., 2016). Lack of awareness and interests from both society and policy makers might also be responsible for the ineffectiveness of conservation and restoration efforts (Villa et al., 2014).

As the available resources are often limited, effectiveness, efficiency and accuracy of conservation, management and restoration programs are very pivotal (Rey Benayas et al., 2008; Graham et al., 2013; Adame et al., 2015; Friess et al., 2016). Adequate information of the target ecosystems such as the ecological process and their driving factors are therefore required to reach and improve the effectiveness, efficiency and accuracy of conservation and management practices (Webb et al., 2013; Adame et al., 2015; Dharmawan et al., 2016). Moreover, thorough ecological understanding is very useful to create better conservation and management strategies by identifying the risks and consequences of actions taken to the ecosystems and/or their values, as well as assessing the future ecosystem fates in regards to environmental changes such as warming climate or SLR (Birks, 2012; Webb et al., 2013).

Such information is, however, often difficult to gathered as some ecosystem processes and their responses to environmental changes occur only after a long period of time (Willis and Birks, 2006; Birks, 2012). Palaeoecology, or also known as long term ecology, can offer the solution to the temporal limitation of ecological perspective (Davies, 2007; Davies and Bunting, 2010; Bhagwat et al., 2011; Birks, 2012; Page and Baird, 2016). Extending from years to millions of years, palaeoecology can provide long temporal series of ecological records (Seppä, 2009).

Nowadays, the needs for palaeoecological information to improve ecosystem conservation and management are widely recognized (Jackson and Hobbs, 2009; Lyver et al., 2015; McCarrol et al., 2017; Nogué et al., 2017). In the Northland region, New Zealand, for instance, palaeoecological studies have informed the conservation practitioners that active restoration is necessary as the ecosystem is no longer able to naturally regenerate (Lyver et al., 2015). However, in Indonesia, the importance of palaeoecology for ecosystem conservation, management and restoration is barely acknowledged. Also, the available palaeoecological information in Indonesia is very limited (Dommain et al., 2011; Biagioni et al., 2015). Additionally, studies that integrate palaeoecological information and the ecosystem values such as C storage capacity are very sparse (Dommain et al., 2011).

1.2. Aims and objectives

This research aims to understand the environmental process and dynamics of coastal wetlands in Indonesia as well as the mechanisms and dynamics of their capacity to store

C. This research also intends to underline their potential for climate change mitigation.

These above aims raise the following research objectives:

- To identify the driver of environmental changes in Indonesian coastal wetlands
- To identify the factors controlling C accumulation of Indonesian coastal wetland
- To acknowledge how significant the C sequestration of Indonesian coastal wetlands amongst other similar ecosystem globally

The outcomes of this research are purposed to provide considerations to improve conservation, management and restoration of coastal wetlands in Indonesia which raise the additional following objectives:

- To assess the resilience of Indonesian coastal wetlands to disturbances, both natural and anthropogenic
- To create an implication to enhance the effectiveness, efficiency and accuracy of conservation, management and restoration of coastal wetlands in Indonesia

1.3. Study area

Indonesia is the largest archipelago country in the world covering around 790 million ha mainly of sea area (Cribb and Ford, 2009). This country consists of over 18000 islands scattering from the islands of Breueh and Sabang on the west to Sibir Island in Teluk Yos Sudarso, Papua (Cribb and Ford, 2009). Kalimantan, Papua, Sumatra, Sulawesi and Java are the largest main islands in Indonesia (Cribb and Ford, 2009).

The climate of Indonesia is tropical with very high humidity and moderate temperature (Syaukat, 2011). The rainfall variabilities in Indonesia are strongly influenced by the El Niño–Southern Oscillation (ENSO; Aldrian and Susanto, 2003) and the Indian Ocean Dipole (IOD; Saji et al., 1999; Fig. 1.5). ENSO is a variation of sea surface temperature (SST) and wind across the Pacific Ocean that results in changes in rising or descending moisture depending on the region (Philander, 1983). ENSO consists of two different climatic phases: El Niño and La Niña (Philander, 1983). During the period of El Niño warm-phase, Indonesia experiences lower rainfall variability than usual years, whereas La Niña cold-phase led to the higher rainfall (Philander, 1990). Similar to ENSO, IOD is a fluctuation of SST and wind across the Indian Ocean that is also differentiated into two phases: negative and positive (Saji et al., 1999). During negative IOD phases, the sea surface temperature in the western Indian Ocean is anomalously low while the eastern part is warmer than usual, resulting in an increase in precipitation over western Indonesia (Saji et al., 1999; Niedermeyer et al., 2014).

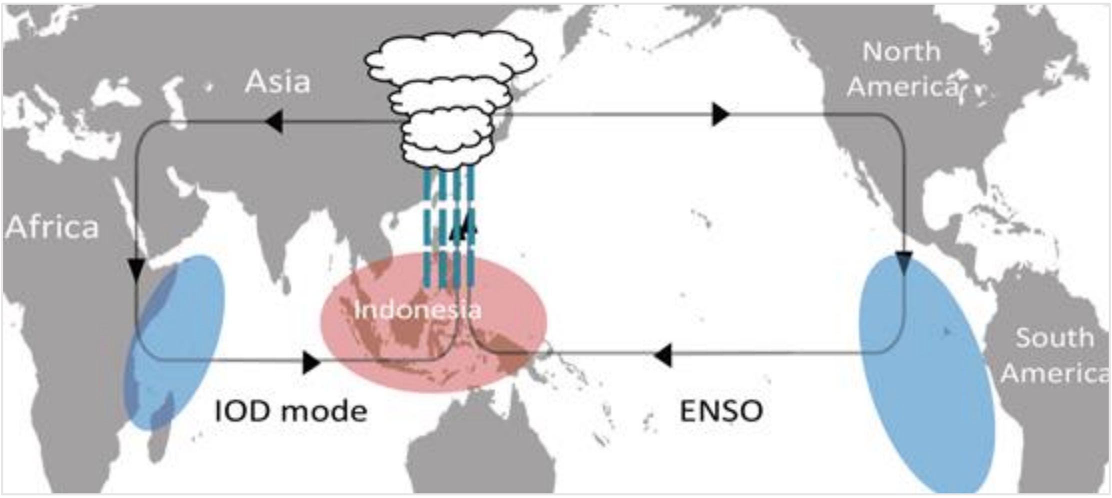

Fig. 1.5. Simplified atmospheric circulation of Indonesian climate (courtesy of Eva Niedermeyer)

Indonesia is also one amongst the most populated country in the world with more than 260 million inhabitants (The World Bank, 2016**;** http://databank.worldbank.org/) concentrated mostly in Java and Sumatra (Cribb and Ford, 2009). More than 50% Indonesian population lives in Java (Statistik Pekerjaan Umum, 2013). Sumatra, on the other hand, is inhabited by around 24% of Indonesian population, that are mostly comes from Java as a result of transmigration program conducted by Indonesian government in 1970s (Statistik Pekerjaan Umum, 2013).

Being the most populous islands in Indonesia, Java and Sumatra suffer from very severe human impact (Agus et al., 2004; van Dijk et al., 2004; Margono et al., 2014b). With a long history of human occupation and settlements, Java has subjected for a long deforestation and degradation since the 19th centuries (Boomgard, 1992; Olson and Berry, 2004). Nowadays only less than 10% of forest remains in Java (Nijman, 2003). Sumatra, on the other hand, has massively impacted only after the mass migration program in 1970s (Margono et al., 2012). Only between 1990 and 2010, more than 7 Mha of Sumatran forest cover has been lost (Margono et al., 2012). Such loss and the following consequences like erosion and extinction has raised a lot of concerns. Thereupon, numbers of conservation, management and restoration efforts have been conducted in Java and Sumatra, although the outcomes are not very pleasing (Smiet, 1990; Asean US CRMP, 1992; ADB, 2006; Sukardi, 2010; van Eijk et al., 2010; Gunawan et al., 2012).

To meet the aims and objectives of this research, two Indonesian coastal wetlands, Sungai Buluh peatland in Central Sumatra and Segara Anakan lagoon in Central Java are selected. Both wetlands are considered suitable to represent the wetlands that are under suppression of both natural and anthropogenic stressors. Moreover, both wetlands are subjected to conservation, management and restoration practices.

1.4. Methodology

1.4.1. Core sample source

This research is conducted using two cores taken from the selected coastal wetlands (Fig. 1.6). One peat core (350 cm; SB-B) was recovered from Sungai Buluh peatland using a Russian Corer (Jowsey, 1966) in 2013. Meanwhile, another sediment core (500 cm; SA-102) was taken from Segara Anakan lagoon using a 5-cm diameter Livingstone piston corer (Wright, 1967) in 2014.

A. *Sungai Buluh peatland*

Sungai Buluh peatland is a protected peat restoration area in Jambi province, Central Sumatra. Located around 19 km from the coastline, this peatland cover an area of 18000 ha with its elevation ranging between 9 and 25 m above sea level (asl). As most of Indonesian region, this peatland is covered by tropical humid climate with the mean annual temperature 26ºC. The annual precipitation is 2400 mm and influenced by ENSO (Aldrian and Susanto, 2003) and IOD (Saji et al., 1999).

Despite of its status as protected restoration area, the overall information regarding this peatland is very limited. The average water table of this peatland is 0.3 m with a fluctuation ranging from 0 to 0.7 m below the peat surface (http://space.geocities.jp/hkdkalimantan/jambi1/jambi1home.html). The vegetation covering this peatland is currently classified as secondary peat swamp forest (PSF; Melati et al., 2015) while the original forest cover was reduced by ENSO-related fire in 1997 (Tata et al., 2016). This peatland was replanted with highly commercial timber trees *Shorea pauciflora* and *Dyera polyphylla* in 2003 following the instructions from the Department of Forestry of the Jambi Province (Nurjanah et al., 2013). The surrounding area of this peatland is converted into agricultural field as well as oil palm and pulp plantation.

Around 28 km south of this peatland on the banks of Batang Hari River, an ancient Muara Jambi temple complex of Malayu Empire was found. According to the archaeological records, the 12 km^2 temple complex was built around the 9th and 14th centuries and functioned as the capital of the empire (Tjoa-Bonatz et al., 2009;

Witrianto, 2014). Malayu Empire was one of the largest kingdoms in Indonesian history that played significant role in trading particularly forest and animal products from Sumatran hinterland area (Tjoa-Bonatz et al., 2009; Witrianto, 2014).

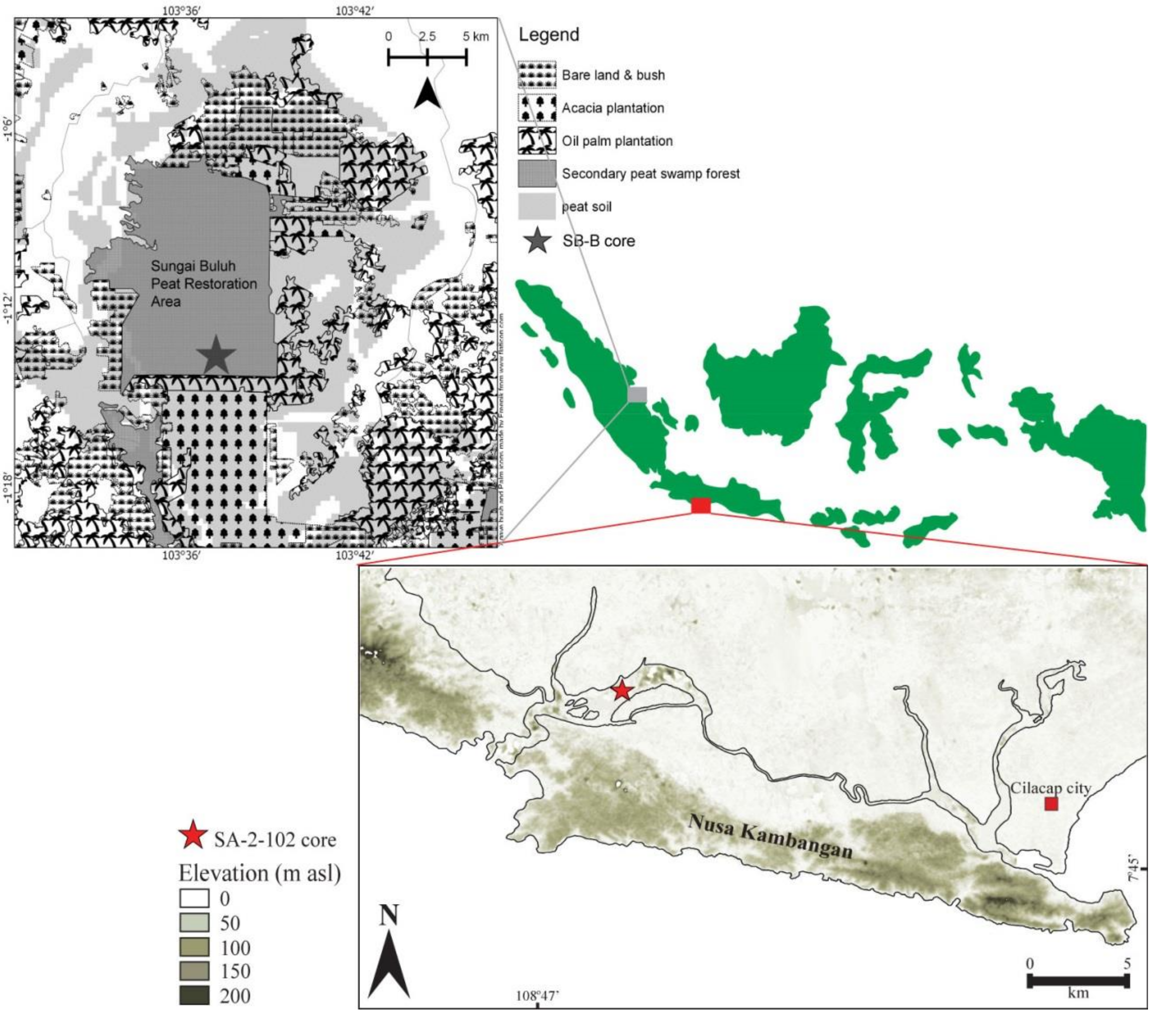

Fig. 1.6. Map of core sources. Grey square represents Sungai Buluh peatland while red square shows the location of Segara Anakan lagoon.

B. Segara Anakan lagoon

Segara Anakan is a mangrove-fringed lagoon in the southern coast of Java Island. Surrounded by around 9200 ha mangrove forest mainly of *Rhizophora apiculata*, *Avicennia corniculatum, A. alba, Nypa fruticans, Sonneratia caseolaris* and *Bruguiera gymnorrhiza*, Segara Anakan is separated from Indian Ocean by Nusakambangan Island

(White et al., 1989; Ardli and Wolff, 2009; Hinrichs et al., 2009). Within the lagoon, the tides are semidiurnal with amplitude variation ranges from 0.2 to 2.6 m (White et al., 1989). Depending on the freshwater input that comes mainly from Cintanduy River and precipitation, the lagoon salinity is ranging from 28 to 1 part per thousand (ppt; White et al., 1989; Noegrahati and Narsito, 2007; Holtermann et al., 2009). As most of Indonesian regions, the climate of this lagoon is tropical humid with mean annual precipitation of 3400 mm (www.weatherbase.com). The inter-annual rainfall variability of Segara Anakan area is influenced by ENSO (Qian et al., 2010) with a mean annual temperature of 27.2^{0}C. Only through Citanduy River flow, Segara Anakan receives around 140 m^3 s^{-1} sediment input (Holtermann et al., 2009). The sediment sourced from a wide catchment area covers around 450,000 ha of Tasikmalaya and Ciamis districts, West Java as well as Cilacap districts, Central Java (Lukas, 2017). This lagoon flows out around 555 m^3 s^{-1} sediment to the Indian Ocean through the western channel (Holtermann et al., 2009).

Nowadays, SAL is inhabited by ca. 17,300 of “Orang Laut” (Sea People) that are mostly fisherman, farmer and aquaculture farmer who lives in “Kampung Laut” (Sea Village) (Ardli and Wolff, 2009; BPS Cilacap, 2017). The population of Segara Anakan distributed around the area of Ujungalang, Ujunggagak, Klaces and Penikel (BPS Cilacap, 2017; Ardli and Wolff, 2009). This lagoon was subjected for pearling ground by the Dutch India Company in the early 17th century (Máñez, 2010).

Segara Anakan lagoon has been subjected to intensive conservation efforts since 1982 (Asean US CRMP, 1992; ADB, 2006). Mangrove rehabilitation, dam construction, vertical drainage and sediment dredging are attempted mainly to decline rapid sedimentation rate (Sukardi, 2010; Lukas, 2017). However, such efforts failed to provide the expected outcome due to inadequate ecological and socioeconomic knowledge (Sukardi, 2010; Dharmawan et al., 2016).

1.4.2. Core analyses

A. Core chronologies

For this research, the chronologies of the cores were developed based on the Accelerator Mass Spectrometry (AMS) ^{14}C and ^{210}Pb ages. Eleven samples of SB-B core were selected and sent to the AMS-laboratory at the University of Nürnberg/Erlangen, Germany and Poznan Radiocarbon Laboratory, Poland for radiocarbon dating. Meanwhile, three samples of SA-102 were selected and sent to NTUAM AMS-Laboratory, Taiwan.

The AMS ^{14}C results of SB-B core suggest that the peat core span from ca. 13,400 cal yr BP to the present. The age-depth model of SB-B is constructed using the SHCal13 calibration curve with Bayesian accumulation model, Bacon (Blaauw and Christen, 2011) script in R (R Core Team, 2014). A robust Student's T analysis (Christen and Pérez, 2009) was applied to identify the outliers prior to the construction of the age depth model.

On the other hand, the AMS ^{14}C results of SA-102 core suggest that the core covers the past 500 years with the top 300 cm of the core was too young to be dated with AMS ^{14}C analysis. Thus, the top 270 cm of SA-102 was analyzed for ^{137}Cs, ^{210}Pb and ^{214}Pb at the Laboratory for Radioisotopes (ISOLAB) in Göttingen, Germany. To calculate the ^{210}Pb age of SA-102 core, the constant rate of supply (CRS) model was used (Applebly and Oldfield, 1978; 1983). CRS model is suitable and widely used to calculate ^{210}Pb age in lakes, coastal zones or estuaries where sedimentation processes are highly influenced by anthropogenic activities (Appleby and Oldfield, 1978; Lubis, 2006). This model is applicable for sediment core with non-monotomic ^{210}Pb profile with an assumption that the flux of excess ^{210}Pb is constant (Applebly and Oldfield, 1978; 1983; Appleby, 2008). The age depth model of SA-102 was constructing by integrating both AMS ^{14}C and ^{210}Pb dates using P-sequence depositional model in OxCal v.4.2 (Ramsey, 2009). The detection of outlier was conducted using SSimple model (Christen, 1994) in OxCal prior to the construction of the age depth model.

B. Pollen and spore analysis

Pollen and spore are reproductive cells produced by angiosperms and gymnosperms (pollen) and sporophyte (spore; Reitz and Shackley, 2012). Unlike other organic material, pollen and spores are resistant to microbial, physical and chemical degradation due to their sporopollenin-wall structures (Brooks and Shaw, 1978). The recalcitrant characteristic of sporopollenin allows the pollen and spore to be preserved on the sediment layer as they are washed or blown from the plant source (Brooks and Shaw, 1978).

Pollen and spore also have unique shapes that allow them to be identified in accordance to their plant source (Goodwin, 1934). The insight on the growing vegetation when the pollen and spore were produced provides an understanding on the environmental and climatic condition (Goodwin, 1934; Prentice, 1988). Thus, the changes in pollen assemblage can be assumed to reflect the changes in vegetation composition as well as in climate and environment (Prentice, 1988).

To assess the change in past vegetation of the study areas, we applied pollen analyses on the core sample by processing the subsamples following the standard pollen and spore extraction methods (Faegri and Iversen, 1989). Prior to the extraction process, *Lycopodium* spore tablets were added to each subsample. The extracted pollen and spores were then counted and identified using the reference collection of pollen and spores of the Department of Palynology and Climate Dynamics, University of Göttingen, and other available literatures (eg. Pollen and Spore Image Database of the University of Goettingen-available at http://gdvh.uni-goettingen.de/; Mao et al., 2012; Li et al., 2012; Cole et al., 2015; Jones and Pearce, 2015).

C. *LOI, TC, TN,* $\delta^{13}C_{org}$ *and* $\delta^{15}N$

Loss-on-ignition (LOI) analysis is a common method to determine the OM content of peat (Wüst et al., 2003). It is conducted by taking peat subsamples and drying the subsamples at 105^{0}C for 24 h. Both wet and dry subsamples were weighed. The subsamples were then combusted at 550^{0}C for 4 h. The combusted subsamples were also weighed. The OM content of each subsample was calculated using following equation:

$$OM_{LOI}(\%) = \left(\frac{dry\ weight\ after\ 105°\mathrm{C} - weight\ after\ combustion}{dry\ weight\ after 105°\mathrm{C}}\right) \times 100$$

In order to trace the change in environmental condition of study areas, total carbon (TC), total nitrogen (TN) and stable isotope composition of organic carbon and nitrogen ($\delta^{13}C_{org}$ and $\delta^{15}N$) are analyzed. TC, TN, $\delta^{13}C_{org}$ and $\delta^{15}N$ were determined by analyzing core subsamples that were dried at 60^{0}C, finely ground and weighed. TC, TN and C_{org} were determined by high temperature of oxidation in a Euro EA3000 elemental analyzer. 1N HCl treatment was applied to remove carbonates, prior the determination of $\delta^{13}C_{org}$ and $\delta^{15}N$ in a Thermo Finnigan Delta Plus gas isotope ratio mass spectrometer after high temperature combustion in a Flash 1112 EA elemental analyzer. The $\delta^{13}C_{org}$ and $\delta^{15}N$ is reported as per mil (‰) deviation from the carbon isotope composition of the Vienna Pee Dee Belemnite (VPDB) with ±0.1‰ uncertainty of each measurement.

D. Charcoal analysis

Charcoal particles are produced through incomplete OM combustion (Whitlock and Larsen, 2002). This particle then distributed and deposited in accordance to their size (Whitlock and Larsen, 2002). Due to their lighter weight, smaller particles can reach a greater distribution than the larger ones (Whitlock and Larsen, 2002). Thus, the larger charcoal particles are considered to be more appropriate to reflect the local fire events (Whitlock and Larsen, 2002).

To understand the past local fire regime of the study areas, macro-charcoal analysis was performed. Subsamples were taken continuously from the core and prepared following the method developed by Rhodes (1998) and Stevenson and Haberle (2005)**.** OM in the sediment was removed using weak hydrogen peroxide (6% H_2O_2) and particles >125 µm were retained after gentle wet-sieving. All charcoal particles >150 µm were counted under a stereomicroscope and concentrations were calculated as particles cm^{-3}. Fire regime characteristics were identified using CharAnalysis (Higuera et al., 2009).

E. XRF profile

XRF scanning is a powerful and non-destructive method to trace the variation in elemental and other sediment property of a core (Croudace et al., 2006). In XRF profile, the relative changes of the elements in this profile are of interest for interpretation rather than the absolute values of concentration, as the results are shown as element count per second (cps; Kylander et al., 2011). For this research, the XRF scanning was conducted at GEOPOLAR, University of Bremen using Mo-tube on the ITRAX (CS-8)-XRF scanner (Croudace et al., 2006). Selected elements are reported in the XRF profile.

F. Carbon accumulation rate

C accumulation rates of the study areas were calculated by multiplying core bulk density (BD) with total organic carbon (TC_{org}) and sediment accumulation rate. C accumulation rates are presented as g C m^{-2} yr^{-1}. BD was calculated by taking core subsamples and subsequently dried at 105^0C for 24 h. Both wet and dry weights were weighed. BD is calculated as the dry weight (g) divided by the wet sample volume (cm^3). TC_{org} were determined by drying the subsamples at 60^0C. The dried subsamples then finely ground and weighed. 1N HCl treatment was applied to the subsamples to remove carbonates prior to TC_{org} determination by high temperature of oxidation in a Euro EA3000 elemental analyzer.

1.4.3. Data handling

Depending on the aims, pollen data of each core were groups according to the characteristic of ecosystem and the function of the taxa (e.g. mangrove, peatland, staple cultivation) or the plant categories (e.g. trees, shrubs, herbs, fern). For the interpretation of pollen data, the pollen records are divided into palynological zones following the agglomeration using constrained cluster analysis (CONISS; Grimm, 1987).

To assess the relationship between vegetation and C accumulation rate, a Principal Component Analysis (PCA) analysis was performed on both data. PCA analysis was also carried out to inspect the trend in vegetation change to assess the ecosystem resilience. Prior to each ordination, aquare root data transformation was applied to minimize the effect of over-represented taxa in the record. Such over-representation might occur due to the difference in pollen productivity and dispersal rate among taxa. For instance, the insect-pollinated plants, *Callophyllum* and *Shorea* (Lemmens and Bunyapraphatsara, 2003; Masuda et al., 2013) must have produced fewer amounts and have had a more limited dispersal of pollen than *anemophilous* taxa like Moraceae-Urticaceae (Haselhorst et al., 2013).

1.5. Outline and content of the chapters

The objectives of this research were developed into three separate manuscripts with a purpose to be published in peer-review international scientific journals. The published and submitted manuscripts are presented in the following chapters:

Chapter 2 – **Environmental dynamics and carbon accumulation rate of a tropical peatland in Central Sumatra, Indonesia**

Within this chapter, the understanding on the environmental and C accumulation dynamics of Sungai Buluh peatland is provided. The controlling factors of the peatland and its C sequestration dynamics, both natural and anthropogenic, are identified. The capacity of Sungai Buluh in storing peat C is compared with other tropical peatlands. This chapter also assesses the future fate of Sungai Buluh peatland as global C sink in regards to SLR.

Chapter 3 – **Resilience of a peatland in Central Sumatra, Indonesia to anthropogenic disturbance: improving conservation and restoration designs using palaeoecology**

The understanding on the resilience of Sungai Buluh peatland to anthropogenic disturbance is provided within this chapter. The recovery process and time of Sungai Buluh from past human disturbance are also discussed herein. This chapter also suggests some considerations to improve the efficiency and effectiveness of peatland conservation, management and restoration strategies.

Chapter 4 – Climate and land use change governed environmental dynamics and carbon accumulation in the mangrove-fringed Segara Anakan lagoon, Java, Indonesia in the Anthropocene and beyond

This chapter provides the understanding on the environmental and C accumulation dynamics of Segara Anakan lagoon. The controlling factors, both natural and anthropogenic, of the lagoon and its C sequestration dynamics are identified. The capacity of Segara Anakan in storing C is compared with other mangrove dominated estuarine ecosystems on a global scale. This chapter also assesses the future fate of Segara Anakan lagoon as C sequester in regards to the climate and human influence.

Chapter 5 – Synthesize

This chapter synthesizes and discusses the main outcomes of this research. The outlooks for the future works and research are also proposed.

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Chapter 2

Environmental dynamics and carbon accumulation rate of a tropical peatland in Central Sumatra, Indonesia

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Abstract

Tropical peatlands are important for the global carbon cycle as they store 18% of the total global peat carbon. As they are vulnerable to changes in temperature and precipitation, a rapidly changing environment endangers peatlands and their carbon storage potential. Understanding the mechanisms of peatland carbon accumulation from studying past developments may, therefore, help to assess the future role of tropical peatlands. Using a multi-proxy palaeoecological approach, a peat core taken from the Sungai Buluh peatland in Central Sumatra has been analyzed for its pollen and spore, macro charcoal and biogeochemical composition. The result suggests that peat and C accumulation rates were driven mainly by sea level change, river water level, climatic variability and anthropogenic activities. It is also suggested that peat C accumulation in Sungai Buluh is correlated to the abundance of *Freycinetia*, Myrtaceae, *Calophyllum*, Stemonuraceae, *Ficus* and Euphorbiaceae. Sungai Buluh has reasonable potential for being a future global tropical peat C sinks. However, considering the impact of rapid global climate change in addition to land-use change following rapid economic growth in Indonesia, such potential may be lost. Taking advantage of available palaeoecological records and advances made in Quaternary studies, some considerations for management practice such as identification of priority taxa and conservation sites are suggested.

Keywords: *Holocene, palaeoecology, Southeastern Asia, vegetation dynamics, stable isotopes, peatland, C accumulation rate, ecological process, controlling factor, Sumatra*

2.1. Introduction

Peatlands play a major role in the global carbon cycle by storing around 600 Gt carbon or one third of the total global carbon pool (Gorham, 1991; Yu et al., 2011). Around 18% of the global peat carbon pool is stored in tropical regions, with around 65% stored in SE Asia (Page et al., 2011; Dargie et al., 2017). In SE Asia the largest share of tropical peat carbon is stored in Indonesia, with an estimated 57.4 Gt making up 65% of the total peat carbon in the tropics (Page et al., 2011). The estimation, however, cannot be precisely made due to limited information about the peat basal ages and the rates of carbon accumulation (Page et al., 2004; Dommain et al., 2011). SE Asian peatlands are distributed mainly in the river deltas and coastal plains of the islands of Sumatra and Borneo (Dommain et al., 2011; Veloo et al., 2014). Due to land use conversion and drainage following the economic growth of Indonesia, the existence of these peatlands is endangered (Miettinen et al., 2012).

Peatlands are sensitive to changes in precipitation and temperature, and are therefore vulnerable to global climate change (Page et al., 2011). A decrease in precipitation and an increase in temperature can lower the water table in peatlands, which in turn can increase the decomposition rate of the accumulated organic material (OM; Couwenberg et al., 2010). An experimental study in a peatland in Manitoba, Canada suggests that a temperature increase of 4^{o}C would cause a 40 to 80% loss of organic carbon due to water-table lowering (Ise et al., 2008). An approximate 30% decrease in annual mean C accumulation following a three years-long 60% precipitation reduction is reported from a study in Switzerland (Bragazza et al, 2016).

Climate is, however, not the only factor controlling the rate of peat C accumulation (Charman et al., 2015; Dommain et al., 2015). For instance, an experimental study in

Mer Bleue bog, Canada reported that the change in vegetation cover could lessen the C sink's capacity by altering the OM quality from slowly-decomposed *Sphagnum* to more decomposable vascular plants such as *Chamaedaphne calyculata* and *Ledum groenlandicum* (Juutinen et al., 2010). Yet, other factors besides climate such as water-table fluctuation and vegetation type are hardly discussed (Charman et al., 2015). Differently from northern peatlands where the relationship between climate, ecosystem dynamics and C accumulation is well studied, the information for tropical regions is sparse (Page et al., 1999; Yu et al., 2011). Consequently, the mechanisms driving peat C accumulation in tropical regions are still poorly understood, in particular in Indonesia with its extensive peatlands. Such knowledge is important for an improved understanding of past global carbon cycling as well as for an assessment of the future role of tropical peatlands in the rapidly changing environment (Page et al., 2004; Charman et al., 2013).

Supporting information such as palaeovegetational and palaeoenvironmental data which is required to identify the driving factors is also lacking (Birks, 2012; Dommain et al, 2015). For instance, a palaeoecological study in the Alaskan boreal forest found that fires occurred more frequently during wetter climatic conditions, contrary to the common assumption that fire is closely related to drought (Lynch et al., 2004). During the wetter conditions, the extensive grass growth increased fuel availability which than led to more frequent fire occurrence (Lynch et al., 2004).

In order to better understand the mechanisms of C accumulation in tropical peatlands, a multi-proxy palaeoecological study of a core from a peatland in the coastal area of Sumatra has been undertaken. These specific questions are addressed: 1.What factors control C accumulation in this peatland? 2. Is there any relationship between the rate of

C accumulation and the composition of particular taxa? 3. How does the long-term (apparent) rate of C accumulation (LORCA) compare in the global context?

2.2. Study site

The Sungai Buluh peatland is situated 19 km from the coastline with an elevation ranging from 9 to 25 m above sea level (asl). The restoration area of Sungai Buluh (18,000 ha) is located in the Tanjung Jabung Regency, approximately 30 km north-east of the city of Jambi in Central Sumatra (Fig. 2.1a). The climate of the area is tropical humid, with yearly precipitation patterns mainly controlled by seasonal variation of the Asian-Australian monsoon and the Intertropical Convergence Zone (ITZC; Saji et al., 1999). Mean annual temperature is 26ºC with little variation throughout the year. Average annual rainfall is 2400 mm, with a slightly drier season corresponding to the onset of the southeast monsoon from June to September (Aldrian and Susanto, 2003; Karger et al., 2016). The inter-annual variability of rainfall is influenced by the *El Niño–Southern Oscillation (ENSO;* Aldrian and Susanto, 2003) and the Indian Ocean Dipole (IOD; Saji et al., 1999).

In general, the information on this protected peatland is sparse. The vegetation covering the restoration area is classified as secondary peat swamp forest (PSF; Melati et al., 2015). According to Tata et al., (2016) the original peat swamp forest cover was reduced by *El Niño-related fires* in 1997. Later in 2003, *Shorea pauciflora* and *Dyera polyphylla* were replanted in the Sungai Buluh area following instructions from the Department of Forestry of the Jambi Province (Nurjanah et al., 2013). The water table in this peatland fluctuated from 0.0 to 0.7 m (average 0.3 m) below the peat surface (http://space.geocities.jp/hkdkalimantan/jambi1/jambi1home.html). The area around the site is converted into agricultural field and plantations (Fig. 2.1b) such as oil palm

(*Elaeis guineensis*) and pulp wood (*Acacia* spp.). The ancient Muara Jambi temple complex is found around 28 km south of the study area, on the banks of the Batang Hari River (Fig. 2.1a). The complex covers about 12 km^2 and it dates back to somewhere between the 9th and 14th centuries (Tjoa-Bonatz et al., 2009) and was supposedly the center of the Malayu Empire (Witrianto, 2014). This ancient kingdom in Sumatra is reported to have played a significant role in trading from the 10th to the 13th centuries as an overseas gateway for goods from the Sumatran hinterland, including forest and animal products (Witrianto, 2014). Such a significant role was lost after the conquest of the Majapahit Empire in the 14th century (Locher-Scholten, 2003).

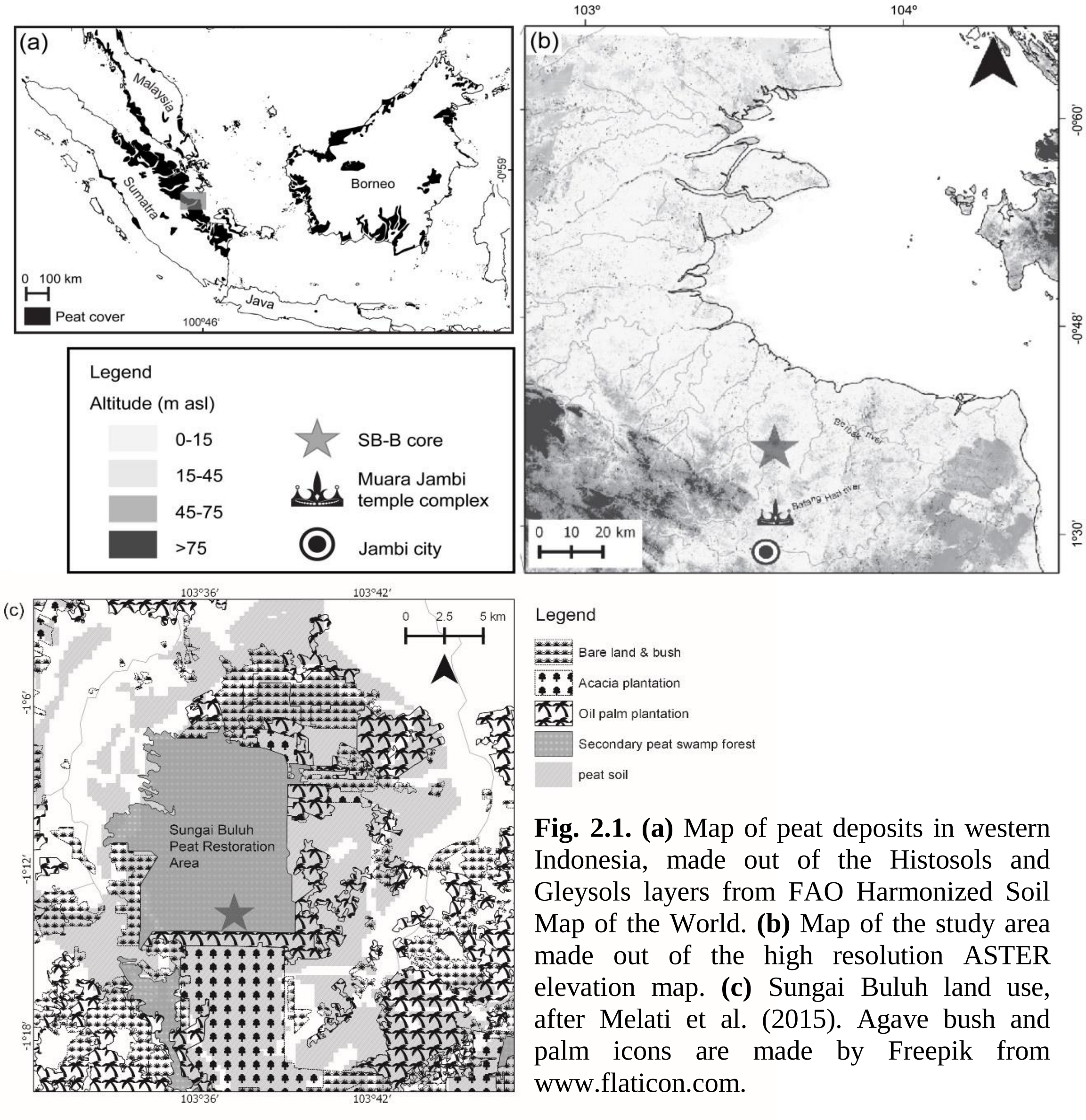

Fig. 2.1. (a) Map of peat deposits in western Indonesia, made out of the Histosols and Gleysols layers from FAO Harmonized Soil Map of the World. **(b)** Map of the study area made out of the high resolution ASTER elevation map. **(c)** Sungai Buluh land use, after Melati et al. (2015). Agave bush and palm icons are made by Freepik from www.flaticon.com.

2.3. Material and methods

In 2013, a 350 cm-long core (SB-B) was collected in the Sungai Buluh peat restoration area using a Russian Corer (Jowsey, 1966)**.** The SB-B core (1°14’10” S, 103°37’12” E; 18 m asl) was photographed and described after recovery following the SE-Asian peat classification system (Esterle and Ferm, 1994; Wüst et al., 2003). Eight samples of the peat core and three additional samples from basal clay material were selected and sent to the AMS-laboratory at the University of Nürnberg/Erlangen, Germany and Poznan Radiocarbon Laboratory, Poland for radiocarbon dating (Table 2.1).

Table 2.1. AMS radiocarbon dates from SB-B core, calibrated using SHCal13.14C calibration curve (Hogg et al., 2013). Italic font represents samples taken from basal clay mineral. Outliers are in grey.

Depth (cm)	Lab. Code	Material	pMC	Radiocarbon dates (^{14}C yr BP)	Age 2 σ (cal yr BP)
55	Poz-85957	Plant remains	106.44 ± 0.34	modern	modern
74	Erl-19233	Organic samples (1-2 cm^3)	93.38 ± 0.46	550 ± 39	529 ± 33
123	Poz-85958	Seed		1,275 ± 30	1,142 ± 75
164	Erl-19232	Organic samples (~1cm^3)	73 ± 0.37	2,528 ± 40	2,575 ± 196
167	Erl-19515	Charred fragment	79.74 ± 0.40	1,819 ± 40	1,693 ± 106
181	Erl-19514	Charred particles	69.66 ± 0.39	2,904 ± 45	2,983 ± 128
201	Erl-19231	Seed	37.97 ± 0.27	8,441 ± 62	9,419 ± 154
238	Erl-19230	Charred particles	23.53 ± 0.19	11,624 ± 66	13,409 ± 145
241	*Erl-19229*	*Bulk sediment (~2cm^3)*	*9.22 ± 0.12*	*19,151 ± 109*	*23,018 ± 388*
305	*Erl-19513*	*Organic bulk sediment (1-2cm^3)*	*15.18 ± 0.20*	*15,146 ± 105*	*18,354 ± 303*
347	*Erl-19228*	*Bulk sediment (~2cm^3)*	*22.48 ± 0.32*	*11,991 ± 115*	*13,796 ± 299*

2.3.1. Palynological analysis

A palynological analysis was conducted to reconstruct past vegetation in the Sungai Buluh peatland. Samples for palynological analysis were taken at 10 cm intervals along the core. A total of 35 subsamples were processed following standard methods (Faegri and Iversen, 1989). Two tablets of *Lycopodium* spores were added as marker to each subsample prior to pollen extraction in order to calculate the pollen and spores concentrations. Pollen and spores were identified using the reference collection of pollen and spores of the Department of Palynology and Climate Dynamics, University of Göttingen, and other available literature (e.g. Anderson and Muller, 1975; Supiandi and Furukawa, 1986; Cole et al., 2015; Jones and Pearce, 2015; Pollen and Spore Image Database of the University of Goettingen – available at http://gdvh.uni-goettingen.de/). Counting of pollen and spore grains was conducted up to a sum of 300 pollen grains. Pollen and spore concentrations were calculated as grains cm^{-3}.

The pollen taxa were grouped into the categories of "mixed-riverine forest" (MRF), representing pollen produced by lowland and riverine plants; "peat swamp forest" (PSF), representing pollen of plants commonly found in peat swamp forest; "open vegetation" (OV), representing pollen produced by herbaceous plants; "ubiquitous" (UQ), representing pollen of plants which do not have a specific ecological distribution; and "mangrove" (MG) which represents pollen produced by salt tolerant plants (e.g. Soerianegara and Lemmens, 1994; Lemmens et al., 1995; Sosef et al., 1998; Padmanaba and Sheil, 2014; Cole et al., 2015). The pollen diagram was prepared with the program C2 (Juggins, 2007).

2.3.2. Stable carbon isotope and Loss-On-Ignition (LOI) analysis

Stable organic carbon isotope ($\delta^{13}C_{org}$) analysis was undertaken to trace the source of OM and change in environmental conditions (Khan et al., 2015). In total, 69 samples were taken along the SB core, dried at 60ºC for 48 hours and finely ground. Samples were subsequently weighed (~1-1.5 mg) and treated with 1N HCl to remove carbonates prior to the analysis of organic carbon content (C_{org}). Determination of C_{org} was done using high temperature oxidation in a Euro EA3000 elemental analyzer. A similarly treated sample was used for determination of $\delta^{13}C_{org}$ in a Thermo Finnigan Delta Plus gas isotope ratio mass spectrometer after high temperature combustion in a Flash 1112 EA elemental analyzer. The $\delta^{13}C_{org}$ is reported as per mil (‰) deviation from the carbon isotope composition of the Vienna Pee Dee Belemnite (VPDB) with ±0.1‰ uncertainty of each measurement.

LOI analysis was conducted to determine the OM content of the peat. A total of 47 samples were taken along the peat section, weighed and dried at 105^0C for 24 h. It was then combusted at 550^0C for 4 h. The OM content (%) was calculated using following equation:

$$OM_{LOI} = \left(\frac{dry\ weight\ after\ 105°\mathrm{C} - weight\ after\ combustion}{dry\ weight\ after 105°\mathrm{C}}\right) \times 100$$

2.3.3. Carbon accumulation rate

To quantify peat bulk density, 92 samples were taken along the core in 2.5 cm interval (except for the interval of 185-200 cm, where a 5 cm interval was applied due to insufficient amount of material) and dried overnight at 105ºC (reported as g m^{-2}; Chambers et al., 2011). Replications were performed on 16 of the 92 samples. Additionally, 92 subsamples were taken at the same depths and analyzed for total organic carbon content (C_{org}) following the same procedure as in Section 3.2. The C accumulation rate (g C m^{-2} yr^{-1}) was calculated by multiplying the bulk density with total % C_{org} and the rate of peat accumulation per sample as determined based on the age-depth model (cm yr^{-1}). LORCA, which represents the average of carbon accumulation rate between basal age and peat surface (Tolonen and Turunen, 1996), was calculated by multiplying the average rate of peat accumulation with median dry bulk density and median C_{org} content following the equation used by Page et al. (2004).

2.3.4. Macro-charcoal analysis

In order to understand the past fire regime and its correlation to vegetation and peat carbon dynamics, macro-charcoal analysis was performed. Subsamples of 1 cm^3 were taken continuously at 0.5 cm intervals from the core at depths of 240-180 cm and in 1 cm intervals from 180-0 cm. Based on the change in sediment/peat accumulation rate, the lower part of the core with the lower accumulation rate was sampled in higher resolution. Each sample was prepared following the method developed by Rhodes (1998) and Stevenson and Haberle (2005)**.** Weak hydrogen peroxide (6% H_2O_2) was used to remove OM in the sediment and particles >125 µm were retained after gentle wet-sieving. All charcoal particles >150 µm were counted under a stereomicroscope and concentrations were calculated as particles cm^{-3}. Fire regime characteristics were identified using CharAnalysis (Higuera et al., 2009; detail parameters explained in Supplementary 1)

2.3.5. Statistical analysis

Principal Component Analysis (PCA) was carried out using R to assess a possible relationship between vegetation community composition and the peat C accumulation rate. The PCA was performed on pollen and spore percentage, C accumulation rate, peat accumulation rate, organic C content and peat bulk density. Square root data transformation was applied to pollen and spore percentage data prior to the ordination to minimize the effect of over-represented taxa in the record due to the difference in pollen productivity and dispersal rate among taxa. For instance, the insect-pollinated plants, *Callophyllum* and *Shorea* (Lemmens and Bunyapraphatsara, 2003; Masuda et al., 2013) must have produced fewer amounts and have had a more limited dispersal of pollen than *anemophilous* taxa like Moraceae-Urticaceae (Haselhorst et al., 2013).

2.4. Results

2.4.1. Stratigraphy

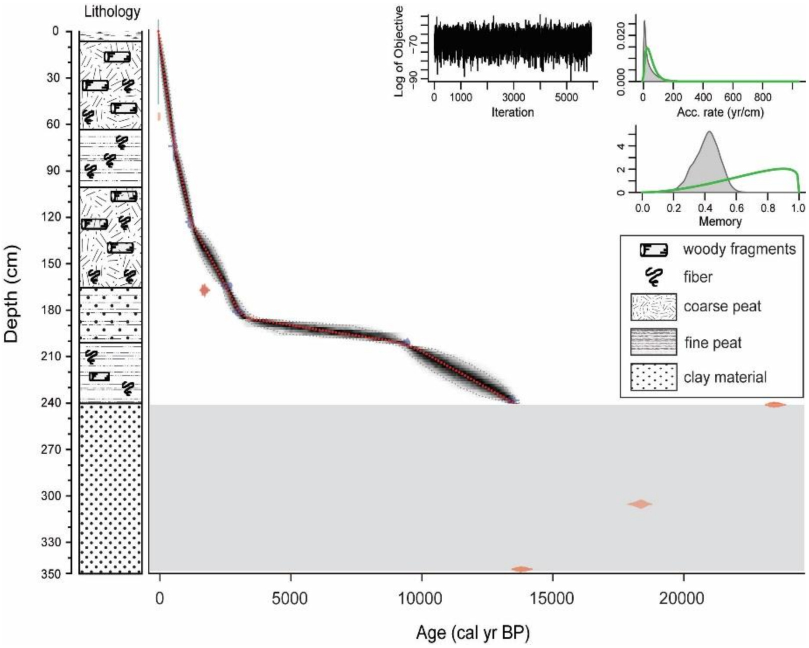

Fig. 2.2. Lithological profiles and age-depth model of SB-B core. The age depth model was constructed with Bacon script (Blaauw and Christen, 2011) in R using SHCal13 calibration curve. The central dotted red line represents the 'best' model based on the weighted mean age, while the outer grey dotted lines indicate 95% confidence inteval. Outliers and excluded dates are in red.

The 350 cm-long SB-B core consist of a peat layer and underlying mineral clay. The mineral substrate (350 to 240 cm) consists mostly of grayish brown to light olive brown clay, whereas the peat layer spans from 240 cm to the top of the core. The peat layer was further divided into coarse and fine hemic peat according to the size of the fragments and the availability of woody material, with an inserted clayey-peat layer in the depth interval of 200 to 165 cm (Fig. 2.2).

According to the results of LOI analysis, the organic matter of Sungai Buluh core can be classified as "muck" (LOI= 44%) from 240-235 cm and "peat" (LOI >45%) from 230 cm onward to the top (Wüst et al., 2003).

2.4.2. Radiocarbon chronology

The AMS radiocarbon dating results (Table 2.1) show that SB-B core comprises peat depositional history from ca. 13,400 cal yr BP to the present. The age-depth model is constructed using the Bayesian accumulation model, Bacon (Blaauw and Christen, 2011) script in R (R Core Team, 2014), using SHCal13 calibration curve (Fig. 2.2). Radiocarbon dates of samples taken from the clay part of the SB-B core are excluded from the analysis. Using a robust Student's T method (Christen and Pérez, 2009), 'Bacon' identified the samples at 55 and 167 cm as outliers. Both samples may have been a part of root that intruded into older material in the deeper layer. Wüst et al., (2008) stated that root contamination may result in several hundred years younger than other OM that deposited together in the peat matrix. In the Tasek Bera peatland, for example, modern roots were even found in 2 m depth (Wüst et al., 2003). The possibility of hiatus in the record between 200 and 165 cm was ruled out as there is no indication observed in both lithology and pollen data.

2.4.3. Palynological results

The record is divided into four palynological zones based on agglomeration using constrained cluster analysis (CONISS; in Supplementary 2; Grimm, 1987) of pollen and spore data along the core (Fig. 2.3): SB-I (240-150 cm; 13,400-2,000 cal yr BP), SB-II (150-112.5 cm; 2,000-1,200 cal yr BP), SB-III (112.5-57.5 cm; 1,200-400 cal yr BP) and SB-IV (57.5-0 cm; 400 cal yr BP to present). Zone SB-I is further divided into SB-Ia (240-200 cm; 13,400-9,000 cal yr BP) and SB-Ib (200-150 cm; 9,000-2,000 cal yr BP) following the change in lithology (clayey-peat; Fig. 2.2). The pollen concentrations are high in zone SB-I with an average of ca. 300,000 grains cm^{-3} and are lower in zone SB-II to -IV with an average of ca. 30,000 grains cm^{-3}.

The pollen assemblage of zone SB-I is dominated by riverine taxa (average 41%; all following % values are also averages), primarily *Macaranga* (9%), *Antidesma* (4%), and *Nauclea* (4%). Pollen of PSF taxa is also well represented like *Calophyllum* (3%), Stemonuraceae (5%), and *Shorea* (4%). In zone SB-II, total pollen of riverine taxa decrease, especially *Macaranga* (3%) and *Antidesma* (1%), while PSF pollen taxa such as Stemonuraceae (10%) and *Shorea* (from 6%) become more abundant. In the end of zone SB-II, the total of PSF pollen reaches its maximum value (69%). In zone SB-III, the total pollen values of PSF decrease to 50% whereas pollen of open vegetation increase to 25% marked by the increase of Piperaceae (from 1% in SB-II to 5%), *Ardisia* (from 1% in SB-II to 6%), Poaceae (from 2% in SB-II to 8%) and Grewioidea (from 1% in SB-II to 5%). In the end of SB-III both Poaceae and Grewioidea reach their maximum values (16% and 7%). In zone SB-IV, the total value of PSF pollen re-increased (60%) such as Stemonuraceae (12%), *Calophyllum* (9%), *Durio* (3%), *Garcinia* (6%) and *Ficus* (3%). At the end of this last zone, the values of *Stenochlaena* (15%) and *Nephrolepis* (11%) are markedly higher. Mangrove pollen of *Sonneratia* and Rhizophoraceae are present with low values in all zones (3%).

2.4.4. Biogeochemistry

From 240 cm (13,400 cal yr BP), C_{org} increases from 22 to 46 % until the depth of 200 cm (ca. 10,000 cal yr BP) and displays little variation around an average of 52 % until the top of the core (Fig. 2.4). $\delta^{13}C_{org}$ ranges from -30.3‰ to -29.0‰ from the bottom of the core up to the depth of 170 cm (ca. 1,700 cal yr BP) and remains stable around -30‰ until the top of the core (Fig. 2.4).

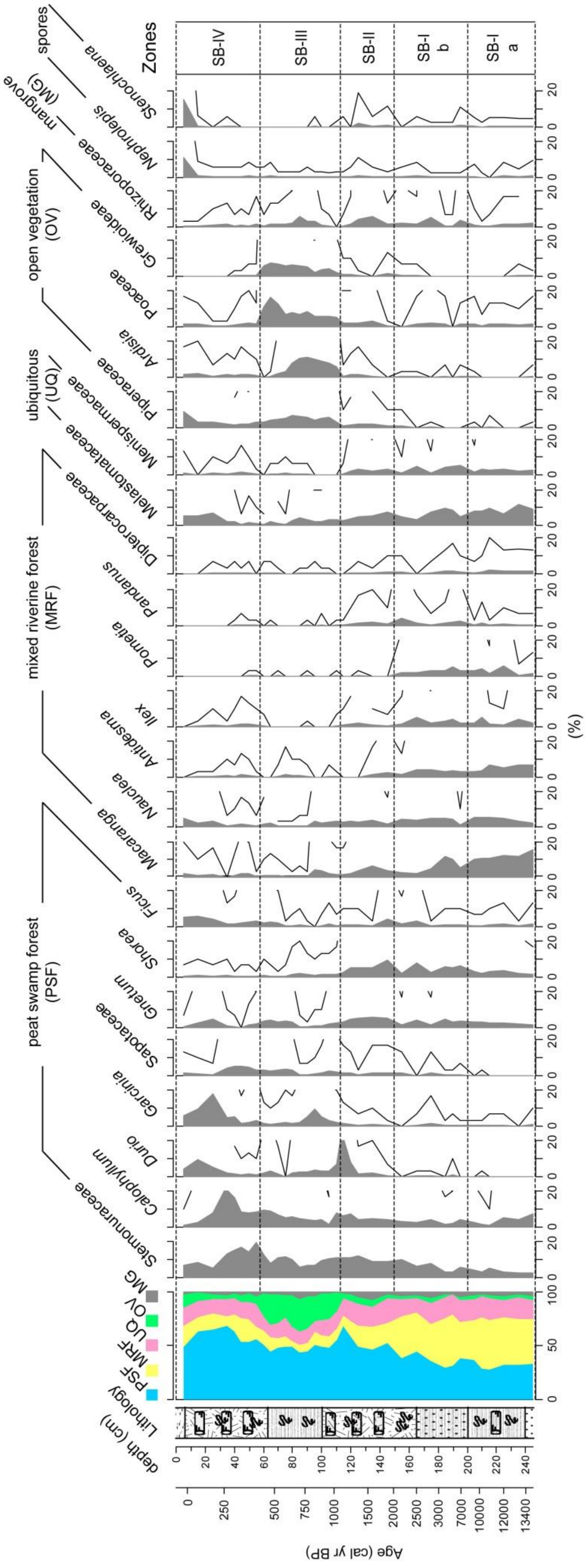

Fig. 2.3. Pollen diagram from Sungai Buluh. The complete list of taxa is comprised in Supplementary S3.

2.4.5. Peat and carbon accumulation rate

From 240-180 cm (13,400 to 3,000 cal yr BP), the peat and C accumulation rate are low (average 0.1 mm yr^{-1} and 13 g C m^{-2} yr^{-1}). Both rates increase to an average of 0.3 mm yr^{-1} and 34 g C m^{-2} yr^{-1} from 180-120 cm (3,000-1,200 cal yr BP). From 120-75 cm (1,200 to 570 cal yr BP), both rates markedly increase to an average of 0.8 mm yr^{-1} and 62 g C m^{-2} yr^{-1}. From 75 to 55 cm (570 to 400 cal yr BP), the peat and C accumulation rates rise to 1.1 mm yr^{-1} and 91 g C m^{-2} yr^{-1}. Both then increase to an average of 1.23 mm yr^{-1} and 100 g C m^{-2} yr^{-1} from 55 cm to the top part of the core (400 cal yr BP to present; Fig. 2.4).

2.4.6. Macro-charcoal and fire regime

In zone SB-I, the average concentrations of macro-charcoal (Fig. 2.4) is 90 particles per cm^3. The concentration is lower in zones SB-II and SB-III (150-57.5 cm; 2,000-400 cal yr BP), and higher in zone SB-IV (57.5-0 cm; 400 cal yr BP to present). The results of CharAnalysis show that the fire frequency in Sungai Buluh was relatively low along the record with an average of 2 fire events in 1,000 years. It reaches its highest frequency at around 12,000 cal yr BP with 5 fire events in 1,000 years. High magnitude of fire peaks (>100 particles per peak) are more frequent in zone SB-I (3 peaks).

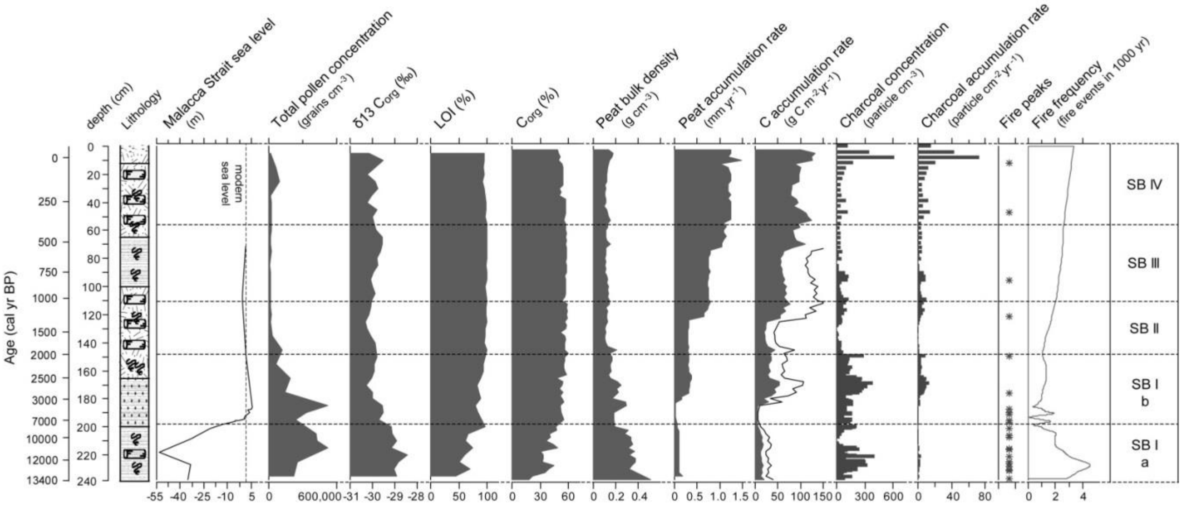

Fig. 2.4. Diagram of total pollen concentration, δ13 C_{org}, LOI, C_{org} content, peat bulk density, peat accumulation rate, C accumulation rate and charcoal analysis results. Malacca Strait sea level change data after Geyh et al. (1979).

2.4.7. Statistical analysis

The variance of pollen data is expressed by the PCA (Fig. 2.6A) with PC1 and PC2 explaining about 41% and 10% of variance, respectively. The result shows a good separation between sample group 1 and 2 according to PC1. The eigenvector shows that C accumulation rate is positively correlated to the peat accumulation rate, C_{org} and sample group B, and negatively correlated to bulk density and depth.

In order to have more clear relationship pattern between vegetation community and peat C accumulation rate, the PCA was again performed on sample group 2. Exclusion of rare taxa, whose average appearance in all samples was <1%, was done prior to the ordination to minimize the noise. The result from PCA group 2 was then further plotted C accumulation rate. The vector representing C accumulation rate is positively correlated with *Freycinetia*, Myrtaceae, *Calophyllum*, Stemonuraceae, *Ficus* and Euphorbiaceae (Fig. 2.6B).

2.5. Discussion

2.5.1. The dynamics of peatland and carbon accumulation rate

A. Initiation phase

In Indonesia, peatlands in the coastal area mostly developed around the stabilization of sea-level or 7,000 cal yr BP (Dommain et al., 2011, 2014). However, based on radiocarbon dating results (Fig. 2.2), the accumulation of OM in Sungai Buluh started earlier in the late Glacial around 13,400 cal yr BP. Early initiation of peatland in the coastal area was also reported from Palangkaraya deposit, Central Kalimantan. The Palangkaraya peatland formed around 10,000 yr BP on a podzol soil, as a result of increased groundwater due to sea-level rise (Neuzil, 1997; Page et al., 2010). It was reported that due to its slightly higher elevation (10-20 m asl), the inundation from sea level rise, even the sea level highstand (ca. 4,500 cal yr BP; +5.8 m asl; Geyh et al., 1979) did not interrupt the accumulation of OM in Palangkaraya (Neuzil, 1997).

During phases of rapid rise of sea-level around 15,000 to 7,400 cal yr BP, early development of peatlands at the low elevation of the coastal areas was not possible as marine transgression would persistently disrupt the accumulation of OM (Hanebuth et al., 2000; Page et al., 2004; Dommain et al., 2014). On the other hand, at higher elevation, marine transgression and coastal inundation would not exert such an impact and therefore the peat could accumulate earlier (Neuzil, 1997; Page et al., 2004; Dommain et al., 2014). As in Palangkaraya site, the Sungai Buluh peatland was also deposited at a slightly higher elevation area (9-25 m asl). Thus, it is plausible that the early initiation date of Sungai Buluh peatland was related to its higher elevation.

Although some studies suggest that most peatlands in coastal area in Indonesia are developed over mangrove or marine sediment (Anderson and Muller, 1975; Supiandi, 1988), the $\delta^{13}C_{org}$ result (~28‰) from Sungai Buluh suggests that the basal clay material was of terrestrial origins (Meyers, 1994). As seen from the case of Kutai peatland, Borneo (85 km inland; 24 m asl), a marine basal clay is not always a requirement for peatland development in coastal areas (Hope et al., 2005). Kutai peatland reportedly developed over a basal clay layer sourced from the river as suggested by the dominance of kaolin (Hope et al., 2005). The OM accumulation of Kutai peatland took place on a floodplain lake infilled by river impediment due to the rising sea level around 8,000 years ago (Hope et al., 2005).

Similar to Kutai peatland, it is also very much unlikely that the underlying clay mineral in Sungai Buluh was of marine origin. At the time when the accumulation of OM in Sungai Buluh started around 13,400 years ago, the sea level was 38 m lower than today (Geyh et al., 1979; Hanebuth et al., 2000) and the site was over 300 km from the shoreline (Sathiamurthy and Voris, 2006). Considering such conditions, the distribution of marine sediments by coastal inundation to Sungai Buluh site would be very unrealistic.

Additionally, the result from pollen analysis suggests that during the initiation process, the vegetation was dominated by *Macaranga*, Myrtaceae, *Nauclea*, *Pometia* and

Antidesma, which commonly are found along rivers. It is, therefore, conceivable that in the case of Sungai Buluh, the OM accumulated on a floodplain depression that was filled by a river impediment and/or rising water table due to the rapid sea level rise. This is further confirmed by the presence of reworked material in the basal clay which is commonly found in floodplains (Fig. 2.2; Bridge, 2003).

B. Topogenous phase (Zone SB-Ia)

From 12,500 to 9,000 cal yr BP, the LOI result suggests that the peat formed in Sungai Buluh can be classified as topogenous peat (LOI≤75%; Anderson, 1961; Fig. 2.4), while pollen analysis indicates that the peat-forming vegetation in Sungai Buluh was dominated by mixed-riverine forest (Fig. 2.3). During its topogenous phase, the accumulation rate of peat and C of Sungai Buluh were low (average 0.1 mm yr^{-1} and 15 g C m^{-2} yr^{-1}; Fig. 2.4).

Globally, it is agreed that the accumulations of peat and C in peatlands, which depend on the balance between plant production and OM decay, require a relatively stable water level (Keddy, 2000; Limpens et al., 2008; Schumann and Joosten, 2008; Hirano et al., 2009). Fluctuations in the water table could lead to faster aerobic OM decomposition by increasing the oxygen supply (Schumann and Joosten, 2008; Hirano et al., 2009; Sinsabaugh, 2010). Floodplains are ecosystems characterized by a strongly fluctuating water table (van Geest et al., 2005) that is related to several factors such as river stage, distance from the river, rainfall and flood events (Keddy, 2000; Jung and Burt, 2004; van Geest et al., 2005). Taking into consideration that Sungai Buluh peatland is located in a floodplain area, it is likely that the slow peat and C accumulation of Sungai Buluh between 13,400 and 9,000 cal yr BP resulted from rapid decay of OM due to a strongly fluctuating water table.

Between 9,000 and 2,500 cal yr BP, there was a deposition of clayey peat material in Sungai Buluh, whereas the proportion of mixed-riverine vegetation gradually decreased towards the present, except for *Pandanus* which is commonly found colonizing submerged or flooded river banks (Laumonier, 1997). In the early Holocene, there was a period of ineffective drainage due to channel filling following the rapid increase of sea level in Sundaland (Hanebuth, 2000; Hanebuth and Stattegger, 2003). The rise of sea level caused the river flows to back up and overflow their banks. Such a period of ineffective drainage later continued following the high precipitation in the mid-Holocene and resulted in reduced river water retention (Partin et al., 2007; Niedermeyer et al., 2014).

Considering that Sungai Buluh is located in a floodplain area, it is conceivable that the deposition of clay was caused by frequent river floods due to ineffective drainage as a result of sea level rise and a period of high precipitation. The river flood transported fine mineral material into the site and resulted in peat accumulation with admixed clay material (Fig. 2.2). Similar conditions were observed in Kutai peatland, Borneo. The abundance of *Pandanus* pollen in a peaty clay layer was reported to reflect the occurrence of frequent river floods (Hope et al., 2005).

Between 9,000-3,300 cal yr BP, the peat and C accumulation rates of Sungai Buluh were very low (on average 0.03 mm yr^{-1} and 5 g C m^{-2} yr^{-1}; Fig. 2.4). As stated by Kozlowski (2002), even for very flood-tolerant trees, floods may impair the cambial growth rate that is responsible for the growth of roots and stems. On the other hand, flooding accelerates OM decomposition in riparian ecosystems by enhancing the leaching and microbial decomposition of OM by providing moisture and nutrients to the microbial and fungal communities (Brinson, 1977; Limpens et al., 2008; Bragazza et al., 2012). Thus, low accumulation of peat and C during 9,000 to 3,300 cal yr BP was conceivably related to periodic river floods as a result of ineffective drainage due to sea level rise.

C. Ombrogenous phase (Zone SB-Ia to –IV)

Around 2,500 cal yr BP, there was a gradual change in vegetation cover. Mixed-riverine vegetation e.g. *Pandanus* and *Ilex,* was gradually replaced by PSF vegetation such as Stemonuraceae and *Durio*. Based on pollen analysis, PSF in Sungai Buluh reached its maximum extension around 1,200 cal yr BP. This is Similar to the observation at peatlands in Borneo where the abundance of *Pandanus* pollen was reported to precede the development of PSF in Kutai and Palangkaraya peatland (Morley, 1981; Hope et al., 2005).

As stated by Branß et al (2016), with every flood that occurred, natural levees grew and limited the input of sediment into the floodplain carried by overbank river flow. Moreover, sea level was reported to have gradually decreased after the highstand (4,200 yr BP) and reached the present level at around 1,000 yr BP (Sathiamurthy and Voris, 2006), accompanied by an overall decreased precipitation in Borneo and Sumatra region (Partin et al., 2007). Both might have played a major role in increased river water retention and less flooding events.

In order to maintain riverine vegetation, periodic floods or overbank flows are needed to sustain the nutrient supply into the floodplain system as well as for seed dispersal and groundwater replenishment (Hughes and Rood, 2003). On the other hand, the decay of OM releases organic acids e.g. citrate, oxalate and malate, which consequently increase the acidity of the environment (Whalen and Sampedro, 2010; Posa et al., 2011). This condition is harmful to acid-intolerant riverine vegetation (Hicks et al., 2003). Therefore, fewer flooding events and an increasing acidity in Sungai Buluh might have driven the decline of riverine vegetation.

Following the change in vegetation composition at around 2,500 cal yr BP, peat and C accumulation rates in Sungai Buluh increased to 0.3 mm yr^{-1} and 34 g C m^{-2} yr-1 and approach 0.8 mm yr^{-1} and 70 g C m^{-2} yr^{-1} around the period when the pollen of PSF reached its maximum proportion (Fig. 2.4). As the OM accumulates, the fluctuations in water table decrease (Keddy, 2000). With a more stable groundwater level, the oxygen supply decreased, leading to a decreased rate of aerobic OM decay (Brady, 1997; Hirano et al., 2009). The sea level fall after its highstand and reduced precipitation could also have played an important role as it caused a reduction in river flooding. Fewer river flooding could also have improved the cambial growth and led to increased root and stem biomass production (Kozlowski, 2002). Hence, it is conceivable that rapid peat and C accumulation in Sungai Buluh is related to a more stable water table due to accumulated OM and sea level fall. This subsequently led to peat thickening and gave rise to ombrogenous peat.

Between 1,000-450 cal yr BP, a period of forest opening is indicated by the increasing proportion of Piperaceae, Poaceae, *Ardisia* and Grewioideae pollen (Fig. 2.3). Around the same period (1,100 to 600 cal yr BP), an ancient kingdom, Malayu Empire, was reported to be occupying the banks of Batanghari River, approximately 28 km south of Sungai Buluh peatland. A study by Tjoa-Bonatz et al. (2009) reported that the people of Malayu Empire utilized wood and plant material from the “locally-available abundant resource” to built their settlements. Ergo, it is likely that the forest opening in Sungai Buluh was related to the activities of the inhabitants of Malayu Empire.

Simultaneously to the forest opening in Sungai Buluh, the C accumulation rate slightly declined to an average of 60 g C m^{-2} yr^{-1}. According to Gautam and Mandal (2016), reduced forest cover can result in significant decrease in forest productivity and consequently affect the ecosystem ability to accumulate C. Loss in forest cover could also reduce the input of lignin-rich OM such as wood or branch. This would ease the breakdown of OM and accelerat C release through decomposition (Yule et al., 2016).

Moreover, forest opening would also cause a loss of canopy protection for the forest floor. A study from Palangkaraya peatland, Borneo shows that the peat surface temperature of the open peatland area is around 5ºC higher than in the area with dense forest cover which help enhancing the oxidation process of OM (Ludang et al., 2007; Wüst et al., 2007)

At around 600 cal yr BP, there was an increase of Poacea and Grewioideae pollen accompanied by an increase of *Ficus, Calophyllum* and *Gnetum* pollen. Increasing grass commonly indicates the onset of forest regeneration (Kobayashi, 1998; Guariguata and Ostertag, 2001) which is then followed by the growth of rapidly regenerating trees such as *Ficus, Calophyllum* and *Gnetum* (Fig. 2.3; Radke and Radke, 1993; Hopewell, 2006; Manner and Elevitch, 2006). Around the same time (ca. 14th century), the Malayu Empire collapsed after being conquered by the Majapahit Kingdom from Java (Locher-Scholten, 2003 Witrianto, 2014). According to historical records, the so-called “new regime” moved the entire Malayu Empire to the hinterland area of Saruaso (ca. 350 km from Muara Jambi temple) soon after the conquest. This suggests that the forest recovery in Sungai Buluh was related to the site abandonment.

Following the forest recovery process, the peat and C accumulation rates of Sungai Buluh increased to averages of 1.1 $m-^{2}$ $yr-^{1}$ and 92 g C $m-^{2}$ $yr-^{1}$. Both rates increased to averages of 1.23 m-^{2} yr-^{1} and 100 g C m-^{2} yr-^{1} after the PSF recovered. During the forest recovery process, the increase of peat and C accumulation rates might have been related to rapid seedling establishment and rapid fine root biomass accumulation of fast growing saplings (Guariguata and Ostertag, 2001, Feshe et al., 2002). The increase in forest cover would also lead to greater net productivity and simultaneously boost the C accumulation (Gray et al., 2016). Additionally, an increase in forest cover will raise the supply of lignin-rich OM that might slow down the OM decomposition (Yule et al., 2016).

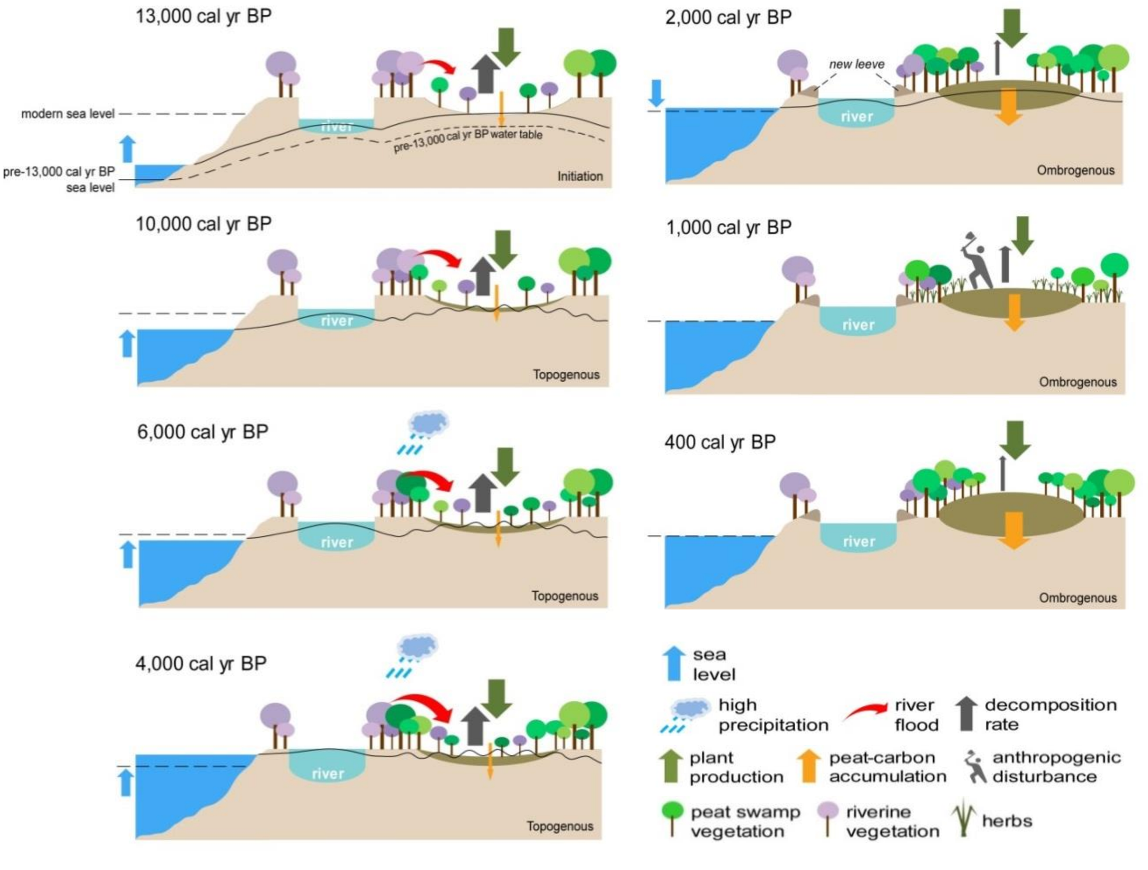

Fig. 2.5. Summary of the carbon accumulation dynamic in relation to the changes in environment. The arrows indicate the direction of the process flow. The thickness of the arrows indicate the magnitude of the process.

2.5.2. Factors controlling C accumulation rate

Peatland C accumulation is the result of the balance between OM productivity and decay (Clymo, 1984; Whittle and Gallego-Salla, 2016). This balance can be affected and altered by several factors which can be various and site specific. Our palaeoecological analysis indicates that the most important drivers of C accumulation rates in Sungai Buluh were changes in sea level and river water level, climatic variability, human activities and vegetation dynamics (visualized in Fig. 2.5).

A. Sea level change and river influence

The influence of sea level and river on the peat C accumulation rate of Sungai Buluh is in agreement with findings from other tropical peatlands. For example, similar to Sungai Buluh, sea level rise in the early Holocene accommodated the initiation of Palangkaraya peatland.in Borneo (Neuzil, 1997). However, in contrast to Sungai Buluh, sea level regression led to termination of peat and C accumulation in Palangkaraya peatland around 5,000 years ago. It caused groundwater depletion that subsequently resulted in rapid OM decay (Neuzil, 1997).

The occurrence of river flood in Sungai Buluh resulted in a decreased rate of peat C accumulation by impairing plant productivity and enhanced OM decay. River influence is also reported to have played an important role in controlling peat and C accumulation in Pastaza-Marañón, Peru. In lowland Amazonia, river migration interrupted peat accumulation by causing peat erosion in some sites and peat burial on some others (Lähteenoja et al., 2009; 2012).

B. Climate

Climatic variability is known to play an important role in the dynamics of peat and C accumulation rates (Page et al., 2004; Dommain et al., 2011; 2014). In Sungai Buluh, enhanced precipitation during the mid-Holocene increased the occurrence of river flood and subsequently reduced peat and C accumulation rates by reducing productivity and enhancing OM decomposition. In Sebangau peatland, Borneo, reduced precipitation between 20,000 and 13,000 cal yr BP resulted in increased OM oxidation that led to a decline in peat and C accumulation rate (Page et al., 2004).

Reduced precipitation can lead to drought and subsequently cause fires that recently are responsible for massive peat and C loss (Page et al., 2002). The information regarding past fire activities in tropical peatlands is, however, very sparse (Anshari et al., 2001; Hope et al., 2005; Biagioni et al., 2015; Cole et al., 2015). Only the data from Kutai peatland comprises direct relations of peat C accumulation and fire. The fire occurrence in Kutai peatland was reported to be related to human activities and El Niño phase (Hope et al., 2005; Dommain et al., 2014).

To better understand the relationship between peat C accumulation and fire occurrence, macro-charcoal analysis was conducted in Sungai Buluh. It is interesting to see that the past occurrence of fire in Sungai Buluh, with an average of 2 fires per 1,000 years and a maximum frequency of 5 fires per 1,000 years in around 12,000 cal yr BP (Fig. 2.4), was very low compared to the recent fire occurrence of 6-10 fires per year as observed in Palangkaraya peatland (Heil, 2007).

Although not impossible, the probability of charcoal redisposition from the surrounding area is small as long-distance charcoal transportation to the sites can be restricted by the dense-canopy of PSF (Muller, 1963). Therefore, the presence of charcoal in Sungai Buluh likely reflects the occurrence of local fire. However, it is difficult to evaluate to what extent the fire burned in Sungai Buluh and whether it burned only on the tops of the trees or also combusted the peat layer.

In PSF, fire can be ignited from a severe period of drought or by lightning strike on the fire-prone surface such as dried branch or leaves (Hoscilo et al., 2011; Page and Baird, 2016). Even though a single fire can consume a deep layer of peat, peat swamp forest has a high humidity that acts as fire suppression barrier and protects both forest and the peat surface from burning (Wüst et al., 2007; Wösten et al., 2008; Page and Baird, 2016). Thus, it is very likely that the past fire occurrence in Sungai Buluh did not have a significant impact on peat and C accumulation. A similar result is also reported from a study in the Air Hitam peatland, Sumatra, where the increase of fire events during the past El Niño years did not affect the peat accumulation (Biagioni et al., 2015).

C. Human activities

Past human activities are often related to the occurrence of fire (Anshari et al., 2001; Hope et al., 2005; Hunt and Premanthilake, 2012) which seems not to be the case in Sungai Buluh. Instead, past human activities altered the vegetation composition of Sungai Buluh, which consequently caused a decline in C accumulation rate due to a reduction in productivity and deposition of more decomposable OM. Due to limited record of past human activities in tropical peatlands, the impact of human activities on peat and C accumulation rate in Sungai Buluh is difficult to compare. None of the available information on past human activities in the tropics which are inferred from the presence of charcoal directly related to either peat or C accumulation (Anshari et al., 2001; Hope et al., 2005, Yulianto et al., 2005; Hunt and Premanthilake, 2012).

D. Vegetation composition

It has been shown that changes in vegetation composition could result in alteration of C accumulation (Charman et al., 2015). Based on the PCA analysis, the peat and C accumulation rates of Sungai Buluh peatland are positively correlated to the abundance of *Freycinetia*, Myrtaceae, *Calophyllum*, Stemonuraceae, *Ficus* and Euphorbiaceae (Fig.

2.6). Such correlation is, however, difficult to explain. We hypothesize that it might be related to the high biomass productivity and/or lignin content of the aforementioned taxa. For example, *Calophyllum*, Myrtaceae and *Ficus* are reported to have high lignin content which help impeding decomposition process of OM (Rodrigues et al., 1998; Dhanya et al., 2013; Yang et al., 2015).

Table 2.2. PCA eigenvectors of pollen sample for depth, bulk density, C_{org}, peat accumulation rate and C accumulation rate.

Parameter	Principal component eigenvector		
	PC1	PC2	PC3
Depth	-0.90741	0.42024	0.8828
C_{org}	0.99652	-0.08332	0.4274
Bulk density	-0.97579	0.21872	0.6476
Peat accumulation rate	0.88081	-0.47348	0.9048
C accumulation rate	0.88335	-0.46871	0.7851

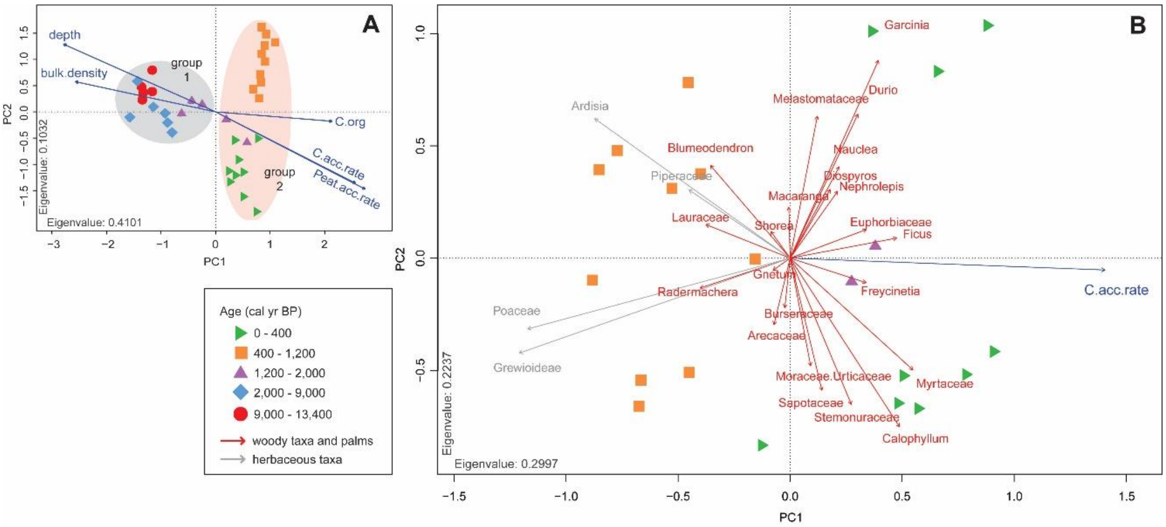

Fig. 2.6. PCA ordination diagram (PC1 and PC2) of Sungai Buluh. The vectors for depth, peat and C accumulation rate, C_{org} and bulk density are indicated by blue arrows radiating from the center (eigenvector values are in Table 2.2). Taxa indincating forest opening are in grey.

2.5.3. A pantropical comparison

Compared to other peatlands in tropical regions, the LORCA of Sungai Buluh is moderate (55 g C m^{-2} yr^{-1}; Fig. 2.7). It is lower than for most peatlands in Africa and some sites in SE Asia such as Siak Kanan and Teluk Keramat, but higher in comparison to most sites in South America and some other sites in SE Asia such as Air Hitam (Fig. 2.7a).

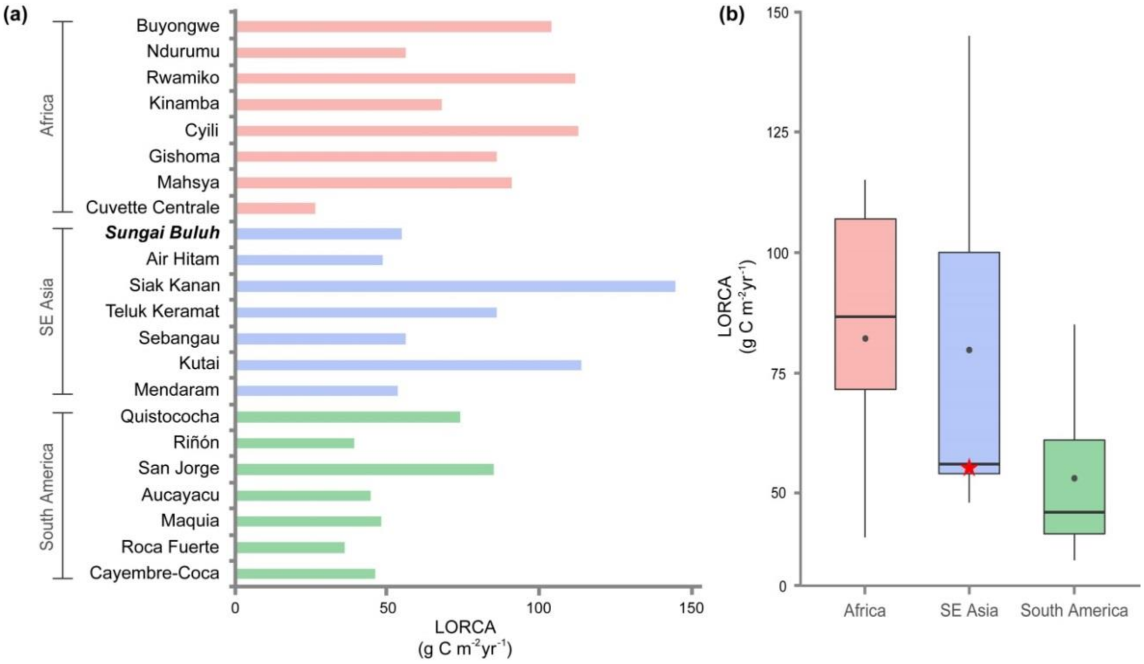

Fig. 2.7. **(a)** Comparison of LORCA of peatlands in tropical regions since the initiation date (sites correspond to Table 2.3). Different bar colors indicate the regions: blue for SE Asia, green for South America and pink for Africa. **(b)** LORCA comparison of peatland between the regions. Average values are represented in dark grey dots. Value of Sungai Buluh site is shown in red star symbol.

In Africa, peats are mainly formed by the accumulation of *Cyperus papyrus*, an herbaceous perennial plant that grows rapidly in swampy areas (Panujen, 1996; Saunders et al., 2013), except for Cuvette Centrale site, Congo where the peats are formed by hardwood vegetation and *Raphia* palms (Dargie et al., 2017). Meanwhile, SE Asian and South American peatlands are formed mainly by woody vegetation and palms (Anderson and Müller, 1975; Wüst et al., 2003; Lähteenoja and Page, 2011; Biagioni et al., 2015). Cayembre-Coca in Ecuador is an exception, where peat is formed by the accumulation of slow-growing cushion plants (Chimner and Karberg, 2008).

C. papyrus is a fast growing plant with enormous biomass productivity and lignin content (Saunders et al., 2013; Brink and Achigan-Dako, 2012). *C. papyrus* is also very resilient and rapidly mature as it takes only 6 months to reach maturity after aboveground removal due to extensive exploitation (van Dam et al., 2011). Thus it is likely that higher the LORCA in African peatlands is related to the accumulation of *C. papyrus*.

In South America, most peatlands included in this study are strongly influenced by river processes such as meandering, avulsion and migration that can subsequently result in peat erosion (Lähteenoja et al., 2009; 2012). Such process can also cause rapid OM decay due to fluctuating water table resulting from the variation of river water level (Schumann and Joosten, 2008; Lähteenoja and Page, 2011). This might explain why the peat C accumulation rate in South America is relatively lower than in the other tropical regions.

In SE Asia, the processes that control C accumulation seem to be more complicated and difficult to generalize. For instance, in Sungai Buluh, the presence of a river and the proximity to the shoreline caused the area to be subject to river inundations that led to slow peat C accumulation during the period of sea level rise and enhanced precipitation in the early to mid-Holocene. Meanwhile in Siak Kanan, ombrogenous peat rapidly accumulated since the initiation date around 6,000 years ago (Diemont and Supardi, 1987). Similarly, Kutai peatland which initiated around 8,000 years ago has developed to its ombrogenous phase since 6,500 cal yr BP (Hope et al., 2005). The high C accumulation in Siak Kanan and Kutai therefore appears to be related to the long ombrogenous phase.

Nevertheless, it is interesting to see that the C accumulation rate in upland peatlands of Africa can actually exceed the rates of those located in tropical lowlands. Highland vegetation is commonly lack of woody plants such as tree or tree-like shrubs. Even when shrubs are present, they tend to be dwarfed (Mani and Giddings, 1980). Moreover, warmer temperature in lowland areas is often related to the increase of net primary productivity (Yu et al., 2010; Charman et al., 2013).

Table 2.3. List of peatlands in tropical regions compared in this study.

No	Site name		Location	Coordinate	Altitude (m asl)	Peat basal date (^{14}C yr BP)	Peat basal calibrated age (cal yr BP)	Carbon measurement	Reference
1	South-East Asia	Sungai Buluh, Sumatra	coastal	$1^0$14’S $103^0$37’E	24	11,624 ± 66	13,419	C analysis	This study
2		Air Hitam, Sumatra	inland	$2^0$2’S $102^0$39’E	57	7,770 ± 101	8,562	C analysis	Biagioni et al., 2015
3		Siak Kanan, Sumatra	coastal	$0^0$42’N $102^0$01’E	8	5,220 ± 220	5,986	C analysis	Neuzil, 1997; Dommain et al., 2011
4		Teluk Keramat, Borneo	coastal	$1^0$25’N $109^0$08’E	2	4,040 ± 170	4,521	C analysis	Neuzil, 1997; Dommain et al., 2011
5		Sebangau, Borneo	inland	$2^0$19’S $113^0$54’E	15	22,620 ± 310	26,000	C analysis	Page et al., 2004
6		Kutai, Borneo	inland	$0^0$12’S $116^0$15’E	24	7,500 ± 220	8,285	0.064 g C/cm^3 carbon density	Hope et al., 2005; Dommain et al., 2014
7		Mendaram, Borneo	coastal	$4^0$22’N $114^0$21’E	55	2,655 ± 35	2,776	C analysis	Dommain et al., 2015
8	South America	Quistococha, Peru	inland	$3^0$50’S $73^0$19’W	90-130	2,300 ± 25	2,335	C analysis	Lähteenoja et al., 2009
9		Riñón, Peru	Inland	$4^0$54’S $74^0$00’W	90-130	1,700 ± 25	1615	C analysis	Lähteenoja et al., 2009
10		San Jorge, Peru	inland	$4^0$03’S $73^0$11’W	90-130	2,850 ± 35	2945	C analysis	Lähteenoja et al., 2009
11		Aucayacu, Peru	inland	$3^0$56’S $74^0$23’W	124	7,963 ± 35	8,870	C analysis	Lähteenoja et al.,

									2012
12		Maquia, Peru	inland	6^{0}19’S 74^{0}48’W	135	2,032 ± 22	1,975	C analysis	Lähteenoja et al., 2012
13		Roca Fuerte, Peru	inland	4^{0}26’S 74^{0}49’W	115	4,505 ± 24	5170	C analysis	Lähteenoja et al., 2012
14		Cayembre-Coca, Equador	inland	0^{0}48’S 78^{0}48’W	3970	2,630 ± 40	2,770	C analysis	Chimner and Karberg, 2008
15	Africa	Buyongwe, Rwanda/ Burundi[a]	inland	2°29'S 29°58'E [a]	1373 [a]	11,770 ± 70	13,730	C analysis	Panujen, 1996
16		Ndurumu, Rwanda/ Burundi [a]	inland	2°29'S 29°58'E [a]	1373 [a]	7,380 ± 50	8,130	C analysis	Panujen, 1996
17		Rwamiko, Rwanda/ Burundi [a]	inland	2°29'S 29°58'E [a]	1373 [a]	7,720 ± 60	8,440	C analysis	Panujen, 1996
18		Kinamba, Rwanda/ Burundi [a]	inland	2°29'S 29°58'E [a]	1373 [a]	10,110 ± 50	11,710	C analysis	Panujen, 1996
19		Cyili, Rwanda/ Burundi [a]	inland	2°29'S 29°58'E [a]	1373 [a]	2,495 ± 50	2,610	C analysis	Panujen, 1996
20		Gishoma, Rwanda/ Burundi [b]	inland	1°25'S 29°32'E [b]	2556 [b]	1,600 ± 50	1,510	C analysis	Panujen, 1996
21		Mashya, Rwanda/ Burundi [b]	inland	1°25'S 29°32'E [b]	2556 [b]	1,855 ± 40	1,800	C analysis	Panujen, 1996
22		Cuvette Centrale, Congo [c]	inland	1°S - 2°N 16 - 18°S [c]	305-366 [c]	7,993 ± 40	8,835	C analysis	Dargie et al., 2017

[a] Coordinate and elevation of Akanyaru swamp complex. The sites are located nearby. The coordinate and elevation of each site is not mentioned.

[b] Coordinate and elevation of Virunga swamp complex. The sites are located nearby. The coordinate and elevation of each site is not mentioned.

[b] Coordinate and elevation are in range.

2.6. Summary and conclusion

Our results from the palaeoecological investigation of Sungai Buluh peatland in Central Sumatra revealed that the peatland has developed since its initiation date (ca. 13,400 cal yr BP) from mixed-riverine forest-dominated topogenous peat to peat-swamp forest-dominated ombrogenous peat. We also found that the rate of C accumulation in Sungai Buluh, which varied depending on the balance between the ecosystem productivity and decay, was dynamic overtime and driven mainly by sea level change, river water level, climatic variability, anthropogenic activities and vegetation dynamics.

Such results are in agreement with other studies, although our comparison shows that those factors could be distinct and unique for each site and could impact each site differently. Sea level regression, for instance, resulted in cessation of peat and C accumulation in Palangkaraya, but contrarily led to increased peat and C in Sungai Buluh. Therefore, it is important to identify the site-specific processes and factors as well as the impacts of different processes and factors on C accumulation rate.

In Sungai Buluh peatland, the peat and C accumulation is positively correlated to *Freycinetia*, Myrtaceae, *Calophyllum*, Stemonuraceae, *Ficus* and Euphorbiaceae. This assessment, however, needs to be confirmed by more similar studies as well as supporting ones, such as pollen productivity and dispersal rate studies. In the tropics, such correlation might not be as clear as *Sphagnum*-peat C accumulation as in the northern peatlands. Regardless, such information is valuable as it can help to set priority-taxa in peat carbon management and peatland restoration project.

Amongst other tropical peatlands globally, Sungai Buluh has moderate LORCA, suggesting a reasonable potential for future global tropical peat C sequestration. Considering the pressure from rapid economic growth that led to land use conversion and natural resource exploitation, its status as a protected area elevates the potential of Sungai Buluh peatland as global tropical carbon sink. However, as many other coastal environments, Sungai Buluh is unlikely to be able to escape the threat and damage of sea level rise owing to warming climate which is predicted to increase around 1.6 m per century (Rohling et al., 2008; Jevrejeva et al., 2010). The rise of sea level will lead to land inundation and saltwater intrusion that can negatively impact the rate of peat C accumulation, particularly in SE Asia where most of the peatlands are located in lowland coastal area (≤5 m asl; Whittle and Gallego-Salla, 2016). Although direct seawater inundation in Sungai Buluh peatland is very unlikely due to its slightly higher elevation, the history shows that it might be once again affected by river flooding due to the ineffective drainage. Thus, a mitigation of this potential damage on the peatland needs to commence. In the meantime, consideration for priority and alternatives sites for peatland conservation and restoration purpose also needs to be given.

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Supplementary 1

Detail parameter applied to macro-charcoal data in CharAnalysis

The raw charcoal data are interpolated into 32 years (the median temporal resolution) and converted into charcoal accumulation rates as particle cm^{-2} yr^{-1}. The charcoal accumulation rates are then separated into a background and peak components using a 1,000-yr window locally weighted regression robust to outliers. The peak component (C_{peak}) is defined as residual after removing the noise (resulting from distant fires, redeposition, or analytical error) using a locally defined Gaussian mixture model (Gavin et al., 2006; Higuera et al., 2008). Peaks exceeding the 99th percentile threshold of the noise distribution are identified as fire peaks. Identified peaks are subjected to a “Poisson minimum count” to evaluate whether the identified-peak charcoal count has more than 5% chance to emerge from the same Poisson-distributed population as the counts of the previous five samples, and thus reduced the counting variance bias. Fire frequencies over time are calculated based on the frequency of fire peaks per 1,000 years. Peak magnitude as a measure of total charcoal deposition per event (Higuera et al., 2009), is used to reflect fuel consumption per fire and/or fire size. A total of 23 fire peaks are detected along the core. The local signal-to-noise index (SNI) represents the standard deviation units which separate the charcoal peak identified to the mean of the modelled noise distribution of the peak charcoal series (Kelly et al., 2011). SNI values for SB-B macro-charcoal data are fluctuating around 3, indicating a good separation of noise and signal (peak) populations (Kelly et al., 2011). The mean fire return interval (FRI) is 598 yr (425-784 yr).

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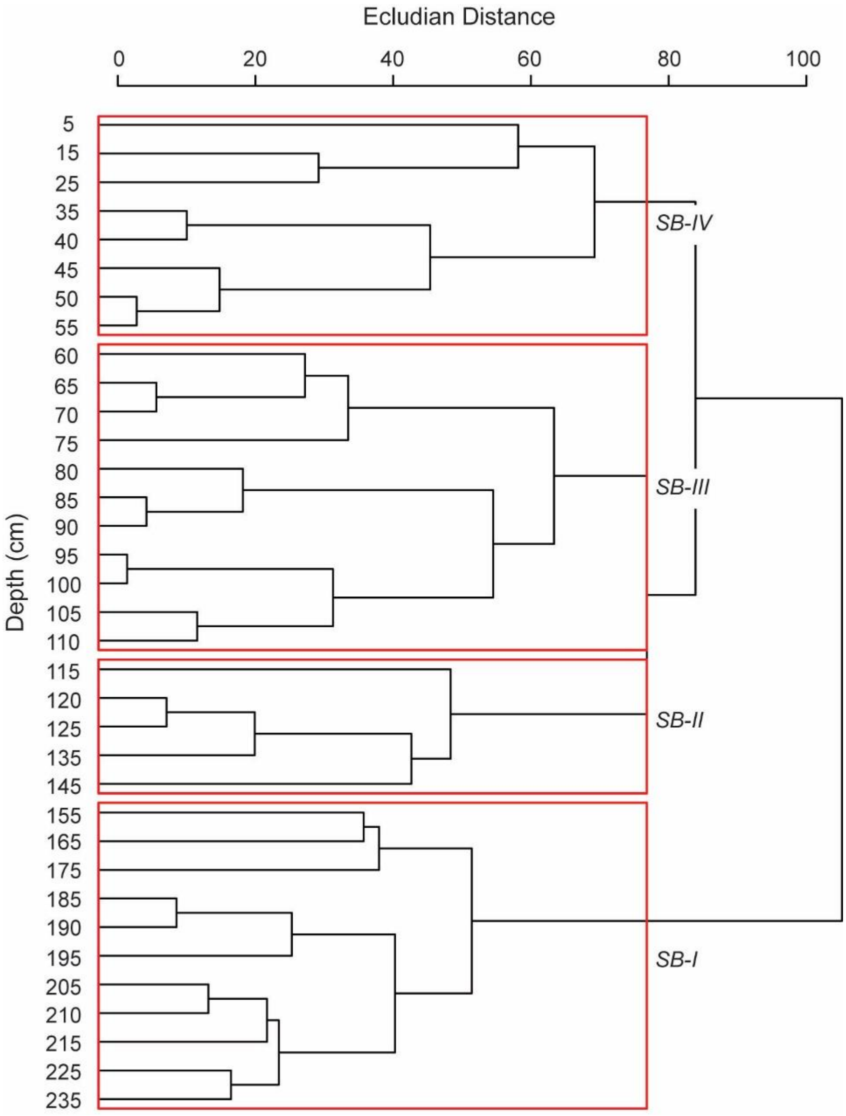
Supplementary 2

Cluster analysis result of the pollen and spore data.



Supplementary 3

Complete list of plant taxa identified from pollen and spore analyses

Peat-swamp forest

Plant family	***Pollen taxon***	**Plant family**	***Pollen taxon***
Anisophylleaceae	*Combretocarpus*	Ebenaceae	*Diospyros*
Annonaceae		Gnetaceae	*Gnetum*
Arecaceae		Lauraceae	
Bignoniaceae	*Radermachera*	Loranthaceae	
Burseraceae	*Canarium*	Malvaceae	Bombacoideae
	Burseraeae type		*Durio*
Chrysobalanaceae	*Parastemon*	Moraceae-Urticaceae	*Ficus*
Clusiaceae	*Calophyllum*	Pandanaceae	*Freycinetia*
	Garcinia	Sapindaceae	Sapindaceae type
Euphorbiaceae	*Blumeodendron*	Sapotaceae	*Palaquium*
	Euphorbiaceae type		Sapotaceae type
	Neoscortechinia	Stemonuraceae	Stemonuraceae type
Dipterocarpaceae	*Shorea*		Stemonurus

Mixed-riparian forest

Plant family	***Pollen taxon***	**Plant family**	***Pollen taxon***
Anacardiaceae	*Campnosperma*	Phyllanthaceae	*Antidesma*
Aquifoliaceae	*Ilex*	Polygalaceae	
Cannabaceae	*Celtis*	Rubiaceae	*Canthium*
Cornaceae			*Nauclea*
Euphorbiaceae	*Macaranga*		*Randia*
Dipterocarpaceae type	Dipterocarpaceae		Rubiaceae type
			Timonius
Myrtaceae		Sapindaceae	*Pometia*
Olacaceae	*Anacolosa*	Verbenaceae	
Pandanaceae	*Pandanus*		

Ubiquitus

Plant family	***Pollen taxon***
Melastomataceae	
Menispermaceae	
Moraceae-Urticaceae	Moraceae-Urticaceae type

Open vegetation

Plant family	***Pollen taxon***	**Plant family**	***Pollen taxon***
Convolvulaceae		Piperaceae	*Piper*
Malvaceae	Grewioideae	Poaceae	
Onagraceae		Primulaceae	*Ardisia*

Mangrove

Plant family	***Pollen taxon***
Lythraceae	*Sonneratia*
Rhizophoraceae	

Spores

Plant family	***Spore taxon***	**Plant family**	***Spore taxon***
Blechnaceae	*Stenochlaena*	Nephrolepidaceae	*Nephrolepis*
Davalliaceae	*Davallia*	Polypodiaceae	
Lycopodiaceae		Pteridaceae	
Monolete		Trilete	

Chapter 3

Resilience of a peatland in Central Sumatra, Indonesia to past anthropogenic disturbance: improving conservation and restoration designs using palaeoecology

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Abstract

1. Tropical peatlands, which provide important functions such as biodiversity provisioning and carbon (C) storage, are currently threatened by land-use conversions. Thus, conservation and restoration efforts are needed to maintain their functions. Conservation concepts aiming to separate human from ecosystems are no longer conceivable. Therefore, understanding peatland resilience to human disturbance, that is the ability of peatland ecosystems to maintain their structure and function despite perturbations and to return to their pre-disturbance states, can assist with integrating human needs into conservation strategies and improving restoration effectiveness.
2. Understanding ecosystem resilience is often impeded by a lack of long-term data, which can be obtained from palaeoecological studies. Located close to the archaeological remains of the Malayu Empire, the Sungai Buluh peatland in Sumatra, Indonesia provides an opportunity to study the resilience of a tropical peatland to past human disturbance. We subjected a 250 cm-long peat core to palynological, charcoal and C content analyses to delineate the anthropogenic impact on the peatland and the ecosystem's response.
3. The results revealed that extensive human activities in Sungai Buluh such as logging, grazing/cut-and-carry, and wild-harvesting started soon after humans occupied the vicinity of the peatland ca. 1000 cal yr BP. Even without fire use and cultivation, these activities were able to alter vegetation composition and decrease the peatland's C sequestration capacity.
4. Following site abandonment after the demise of the Malayu Empire at ca. 600 cal yr BP, the palaeoecological record suggests that the Sungai Buluh peatland recovered

in terms of both floristic composition and C sink function, with the latter recovering faster (ca. 60 years) than the former (ca. 170 years).

5. *Synthesis:* The palaeoecological record from Sungai Buluh provides the first evidence of tropical peatland recovery following human disturbance, which can help improve present peatland conservation/restoration strategies. The design of peatland wise-use strategies can mimic the ‘resilience-friendly’ human activities identified in this study. Consideration should also be given to selecting rapidly regenerating taxa for cost-and-effort-efficient restoration strategies. Additionally, the 170-year recovery time of the Sungai Buluh peatland suggests that the 60-year timeframe currently allocated in most tropical peatland restoration projects may be insufficient.

Keywords: *palaeoecology and land-use history, ecosystem function and services, human activities, natural recovery, peat carbon sink, sustainable use, restoration design, local policy*

Second language abstract (Bahasa Indonesia)

1. Keberadaan lahan gambut tropis yang memiliki peran penting dalam menjaga keanekaragaman hayati dan penyimpanan karbon saat ini semakin terancam karena adanya kegiatan alih fungsi lahan. Upaya konservasi dan restorasi sangat diperlukan untuk menjaga keberlangsungan fungsi ekologis lahan gambut tersebut. Namun, konsep konservasi yang bertujuan memisahkan aktivitas manusia dari lingkungan akan kurang sesuai jika diterapkan di era dominasi manusia seperti sekarang. Oleh karena itu, pemahaman daya lenting (*resilience*) ekosistem lahan gambut terhadap gangguan aktivitas manusia sangat penting untuk mengintegrasikan kebutuhan masyarakat lokal ke dalam strategi konservasi serta meningkatan efektivitas upaya restorasi lahan gambut.
2. Namun, pemahaman terhadap daya lenting ekosistem pada umumnya terhambat oleh ketidaktersediaan informasi jangka panjang yang semestinya dapat diperoleh melalui studi palaeoekologi (ekologi di masa lampau). Lahan gambut Sungai Buluh di Sumatra merupakan area ideal untuk mempelajari daya lenting ekosistem lahan gambut tropis terhadap aktivitas manusia di masa lampau karena berlokasi dekat dengan situs arkeologi Muara Jambi, peninggalan Kerajaan Melayu Jambi. Kami manggunakan sampel bor sepanjang 250 cm yang diperoleh dari lahan gambut Sungai Buluh untuk analisis kandungan polen, partikel arang (*charcoal*) dan karbon, guna menggambarkan dampak antropogenik pada lahan gambut dan respon ekosistemnya.
3. Hasil studi menunjukkan bahwa aktivitas manusia di lahan gambut Sungai Buluh antara lain penebangan kayu, penggembalaan ternak/mengarit, serta panen hasil

hutan, sudah dimulai sejak masyarakat Kerjaan Melayu Jambi menempati daerah sekitar lokasi tersebut pada tahun 1000 SM. Aktivitas tersebut mampu mengubah komposisi tutupan vegetasi dan mengurangi daya serap karbon lahan gambut, meskipun tanpa melakukan pembakaran lahan dan agrikultur.

4. Setelah Kerajaan Melayu runtuh dan situs Muara Jambi ditinggalkan pada tahun 600 SM, studi palaeoekologi menunjukkan bahwa komposisi flora serta daya serap karbon lahan gambut Sungai Buluh kembali pulih. Pemulihan daya serap karbon lahan gambut Sungai Buluh memerlukan waktu lebih-kurang 60 tahun, lebih cepat dari waktu yang dibutuhkan untuk pemulihan komposisi flora (lebih-kurang 170 tahun).
5. *Sintesis:* Studi palaeoekologi di Sungai Buluh menghadirkan bukti pertama dari pemulihan lahan gambut tropis terhadap aktivitas manusia, dan bukti tersebut penting untuk meningkatkan strategi konservasi serta restorasi lahan gambut. Rancangan pemanfaatan lahan gambut secara berkelanjutan dapat meniru/mengikuti aktivitas manusia di masa lampau yang tidak mengganggu daya lenting ekosistem. Pertimbangan dalam hal pemilihan taksa yang cepat beregenerasi juga harus diberikan demi strategi restorasi yang lebih efisien dalam hal biaya dan usaha. Selanjutnya, alokasi waktu selama 60 tahun untuk restorasi lahan gambut (sebagian besar program restorasi lahan gambut di Indonesia saat ini) sangat tidak mencukupi/memadai jika dibandingkan dengan kisaran waktu 170 tahun yang dibutuhkan lahan gambut Sungai Buluh untuk pulih dari aktivitas manusia di masa lampau.

3.1. Introduction

Tropical peatlands are important for water regulation (Rieley, 2007) and harbour a rich biodiversity possessing specialised adaptations to waterlogged, acidic, and nutrient-poor conditions (Yule, 2010; Posa et al., 2011). These ecosystems also play a significant role in the global carbon (C) cycle, as they store 18 % of the total global peat C pool (Page et al., 2011; Dargie et al., 2017). Around 65 % of the global tropical peat C pools are located in Indonesia and Malaysia, primarily distributed along the coast of Sumatra and Borneo, Indonesia (Page et al., 2011; Dargie et al., 2017). Due to rapid population increase and economic growth in Indonesia over the past few decades, pressure on these peatlands has escalated quickly (Brady, 1997). Only from 1990-2010, 4.5 Mha of peatland cover in Indonesia has been converted to plantation and urban areas (Miettinen et al., 2012). Such conversions jeopardise the survival of specialised taxa, disturb the important hydrological functions of these ecosystems and increase the rate of C release (Page et al., 2009; Petrenko et al., 2016; Wijedasa et al., 2017). Additionally, these conversions reduce the significant economic value of peatlands as a source of food and building materials or as a C sink, a service that is worth up to $22 per tonne CO_2 sequestered (van Beukering et al., 2008), by virtue of the destruction of the peat-swamp forest which provide these ecosystem services. Thus, strategies to maintain and sustain the ecosystem functions of these peatlands through conservation and restoration are required.

Conservation is an effort to preserve the existing ecosystems and prevent them from further degradation (Gerber, 2010), whereas restoration actively reverses the degradation in order to strengthen the ecosystems resilience (Lake, 2013). In Indonesia, several conservation and restoration efforts on peatlands are already underway.

However, expectations are low for these efforts, primarily due to the fragmented knowledge of the target ecosystems and the neglect of the socioeconomic and cultural aspects (Giesen & van der Meer, 2009; Page et al., 2009; van Eijk et al., 2009; Graham et al., 2017).

In order to effectively sustain the ecosystem properties (attributes that characterise the ecosystems e.g. soil properties, biodiversity, primary productivity; Bastian et al., 2012) and services (directly or indirectly beneficial to human well-being e.g. food provisioning, carbon sequestration, natural hazard protection; Bastian et al., 2012) and/or to reverse the degradation of an ecosystem, it is essential to first understand the impact of past and present disturbances on the ecosystem while also assessing the ecosystem's resilience (Chapin III et al., 2009).

The term "resilience" has been variously defined in the ecological literature (e.g. Gunderson, 2000; Carpenter et al., 2001). For this study, resilience is defined as the ability of an ecosystem subject to disturbances to maintain its structure and function (Holling, 1973), and return to its pre-disturbance state. Resilience comprises both resistance (the capability to absorb perturbations and resist shifts of its structure) and recovery (the capability to return to its previous condition after a perturbation; Holling, 1996). Ecosystem resilience can be altered by adding/removing stressors or drivers of change (Côté & Darling, 2010).

Human impact is one factor that can reduce ecosystem resilience, resulting in a failure of the affected ecosystem to recover (GBRMPA, 2009). For these reasons, old conservation concepts tried to separate humans from the ecosystem (Miller & Hobbs, 2002; Chapin III et al., 2009; Sörlin & Warde, 2009; Ramutsindela, 2014). However, in our human-dominated world, it is almost impossible to separate humans from ecosystem processes (Laurila-Pant et al., 2015). Consequently, humans need to be considered as part of the ecosystem and included in conservation strategies (Miller & Hobbs, 2002; Gorenflo & Brandon, 2006; Laurila-Pant et al., 2015). Understanding ecosystem resilience in response to human activity provides a valuable basis for the integration of humans into the design of ecosystem conservation and restoration strategies.

In general, ecosystem resistance can be assessed when disturbance severity increases, whereas ecosystem recovery, which is often a slow process that might take decades or even centuries, can only be observed once the disturbance is alleviated (Dobson et al., 1997; Lake, 2013). As a result, understanding ecosystem resilience, particularly in response to ongoing human-induced disturbances, requires long periods of observation and/or extensive time-series information (Leslie & McCabe, 2013; Mumby et al., 2014). Thus, understanding resilience is often hindered by a lack of long-term data (Martin et al., 2013; Cole et al., 2015), which can be obtained from palaeoecological studies (Bhagwat et al., 2011; Birks, 2012; Page & Baird, 2016). This is particularly true for tropical peatlands, where very little is known about the effects of past anthropogenic disturbances (Hope et al., 2005; Biagioni et al., 2015; Dommain et al., 2015). In SE Asia, evidence of human activities in peatlands is only reported in a limited number of studies (Anshari et al., 2001; Yulianto et al., 2005; Hunt & Premathilake, 2012; Cole et al., 2015).

Aiming to assess the resilience of a tropical peatland to human disturbance, we conducted a study on a natural archive from Sungai Buluh peatland in Sumatra, Indonesia. Due to its vicinity to the Muara Jambi temple complex, the archaeological remains of the former Malayu Empire, Sungai Buluh peatland provides a unique opportunity to understand the impact of anthropogenic disturbances on peatland ecosystems. Four main questions arise: (1) What anthropogenic activities were conducted by the Malayu Empire in Sungai Buluh? (2) How did those activities impact the peatland? (3) Did the peatland recover from those disturbances? (4) If so, how long did it take for the peatland to recover?

The outcomes of this study are intended to help improve present-day peatland management, conservation and restoration practice such as: (1) Designing peatland conservation and sustainable use; (2) Enhancing the effectiveness and efficiency of peatland restoration in terms of cost and effort; and (3) Improving peatland restoration frameworks and local government policy.

3.2. Study site

3.2.1. Environmental and archaeological setting

The Sungai Buluh peatland is located in Central Sumatra, approximately 19 km from the coastline and 30 km north-east of the city of Jambi (Fig. 3.1). It covers an area of 18,000 ha. The study area is covered by secondary peat-swamp forest resulting from regrowth following selective cutting during the 1960s and -70s based on the selective-logging concession granted by then President of the Republic of Indonesia (Nurjanah et al., 2013). Sungai Buluh peatland is surrounded by an area converted to agricultural fields and plantations, for example, pulp wood (*Acacia* spp.) and oil palm (*Elaeis guineensis*; Melati et al., 2015). The peat-swamp forest canopy was reduced due to the *El Niño* related fires in 1997 (Tata et al., 2016). The area was planted with *Shorea pauciflora* and *Dyera polyphylla* trees in 2003 following the instructions from the Department of Forestry of Jambi Province (Nurjanah et al., 2013).

The elevation of Sungai Buluh ranges from 9 to 25 m above sea level (asl) with a tropical humid climate. The average yearly temperature is 27ºC (WorldClim – Global Climate Data, http://www.worldclim.org/), with mean annual rainfall amounts of 2400 mm (Aldrian & Susanto, 2003). Precipitation patterns of the area are influenced by seasonal variations of the Asian-Australian monsoon and the Intertropical Convergence Zone (ITZC; Saji et al., 1999). Meanwhile, the inter-annual rainfall variation is affected by the *El Niño–Southern Oscillation* (*ENSO;* Aldrian & Susanto, 2003) and the Indian Ocean Dipole (IOD; Saji et al., 1999).

Around 28 km south of Sungai Buluh peatland, the ancient Muara Jambi temple complex is found on the banks of the Batanghari River (Fig. 3.1). It is reported to be a remnant of the Malayu Empire and covers about 1200 ha. The Malayu Empire was one of the largest empires in Indonesian history and renowned as the largest pepper centre in Sumatra (Witrianto, 2014). This empire is reported to have been established prior to the 7th century and is suspected to be part of the Srivijaya Kingdom (McKinnon, 1985; Wolters, 1986; Rahman, 1993). An inscription found in Karang Brahi (Merangin, Jambi) suggests that the Malayu Empire was previously based in the upper Batanghari (McKinnon, 1985; Wolters, 1986; Andaya, 2001).

Muara Jambi, the capital of the Empire, was built later between the 9th and 10th century (McKinnon, 1985; Tjoa-Bonatz et al., 2009). This temple complex served as an administrative and ceremonial centre, a settlement, and an important international trading centre (McKinnon, 1985; Tjoa-Bonatz et al., 2009). Muara Jambi was surrounded by some small settlements, with the five largest complexes ranging from 1-140 ha in size (Fig. 3.1; Alam, 2014). In the 14th century, the Empire was conquered by a Javanese kingdom, the Majapahit. The Empire then became a vassal of Majapahit and was ruled by King Adityawarman (Witrianto, 2014; Kozok & van Reijn, 2010). In 1347 AD, King Adiyawarman declared his independence from Majapahit and moved the Empire to Saruaso, Minangkabau-West Sumatra (Witrianto, 2014; Kozok & van Reijn, 2010). The settlements of the Muara Jambi complex and on the lower Batanghari were later destroyed completely by the Javanese in 1377 AD (McKinnon, 1985; Wolters, 1986).

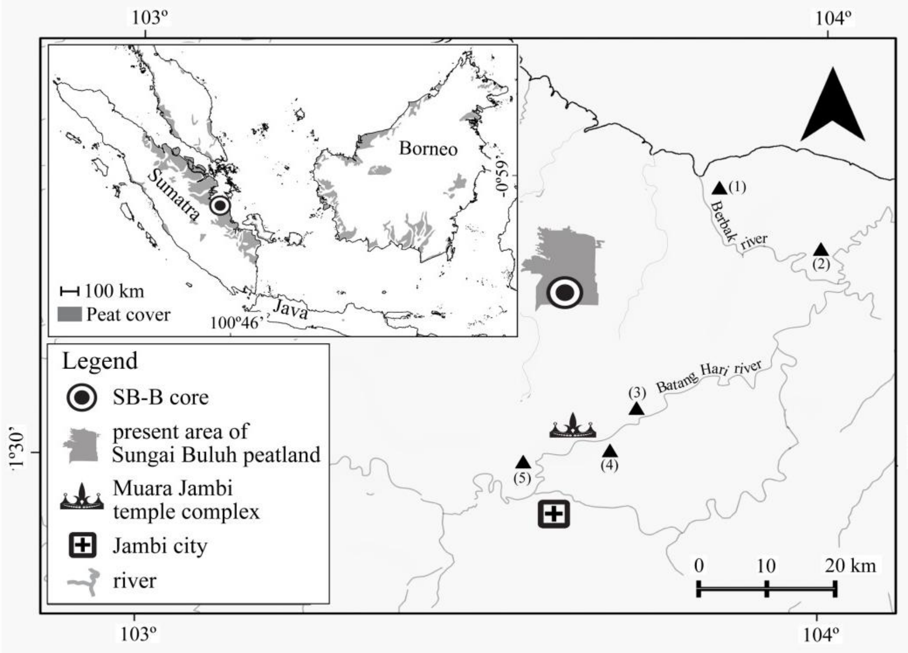

Fig. 3.1. Map of the study site. Black triangles represent the five largest settlement complex (1-140 ha) that existed around the period of the Malayu Empire: (1) Lambur, (2) Kota Kandis (3) Jebus (4) Suak Kandis (5) Candi Teluk (McKinnon, 1985; Alam, 2014).

3.2.2. Palaeoecological setting

The palaeoecological setting of Sungai Buluh peatland is described in detail by Hapsari et al. (2017), in which, the peat accumulation rate (PAR; annual deposition rate of more than 40 % organic matter; OM; expressed as mm yr^{-1}) is calculated separately from the carbon accumulation rate (CAR; annual deposition rate of grams of carbon per unit peat area per unit time; expressed as g C m^{-2} yr^{-1}). The differentiation between PAR and CAR as in Hapsari et al. (2017) is similarly applied in this study.

The Sungai Buluh peatland initiated around 13,400 cal yr BP and initially formed as a topogenous peat deposit (LOI ≤ 75 %; Anderson, 1961) with a mixed-riverine forest dominating the site. During its topogenous phase, Sungai Buluh peatland was strongly influenced by the flood pulse of the adjacent river caused by rising sea levels in the early Holocene and a period of increased precipitation in the mid-Holocene. This caused strong fluctuations in the water table of Sungai Buluh, increased oxygen supply into the ecosystem and triggered rapid aerobic OM decay. Consequently, during its topogenous phase, the PAR and CAR of the Sungai Buluh peatland were low.

Coinciding with the regression of sea level and a reduction of precipitation in the late Holocene, the influence of the adjacent river on the Sungai Buluh peatland was reduced. The reduction in flood frequency suppressed the growth of riverine vegetation by reducing nutrient supply and groundwater replenishment. Moreover, increasing acidification of the environment due to the accumulation of decaying OM harmed the riparian vegetation. This allowed the hydrophytic peat-swamp vegetation to flourish and resulted in the establishment of a peat-swamp forest at around 1200 cal yr BP. Reduced flooding also led to a more stable water level which limited the oxygen supply and thwarted OM decomposition. In turn, this accelerated OM accumulation and supported the development of an ombrogenous peatland, as suggested by high accumulation rates of peat and C and the absence of river sediment input from 1200 cal yr BP onwards.

3.3. Materials and methods

To examine whether the people of the Malayu Empire conducted any activities in the Sungai Buluh peatland and whether the peatland maintained its structure (i.e. floristic composition) and function as a C sink despite the anthropogenic disturbances, and returned to its pre-disturbance condition, we assess the changes in past vegetation, CAR and fire regimes using the SB-B core taken from Sungai Buluh peatland (Hapsari et al., 2017; detailed methods are provided in **Appendix S1**). The past vegetation of Sungai Buluh was examined using pollen and spore data with the assumption that the assemblages of pollen and spores in the core generally reflect the vegetation of the surrounding area at the time of their deposition (Bradley, 1999). Pollen and spores were extracted using the standard method (Faegri & Iversen, 1989), counted to 300 pollen grains and identified to the finest taxonomical classification possible using the available literature (e.g. Pollen and Spore Image Database of the University of Goettingen-available at http://gdvh.uni-goettingen.de/; Anderson & Muller, 1975; Jones & Pearce, 2015). The identified pollen taxa are grouped according to their life form: (1) trees/shrubs and (2) herbs/subshrubs (i.e. vegetation that has characteristics of both herbaceous and woody plants). The pollen and spore data is presented in the pollen diagram prepared using C2 software (Juggins, 2007). Only the taxa with notable changes during the occupation period of the Malayu Empire and those that serve a special purpose (i.e. species representative of a specific use or changes in land use/cover) are shown.

The Sungai Buluh CAR (g C m^{-2} yr^{-1}) was calculated using the following equation (Tolonen & Turunen, 1996), where BD is the peat bulk density (g m^{-3}), % C_{org} is the total organic C content, and PAR is the peat accumulation rate (mm yr^{-1}).

$$\boldsymbol{CAR = BD} \times \%\, \boldsymbol{C_{org}} \times \textbf{PAR}$$

The past fire regime was examined using macro-charcoal analysis. This analysis is based on the assumption that macroscopic (>150 µm) charcoal particles represent the occurrence of local fires, as such particles tend to deposit near their originating fires, being too large for many forms of transport to carry them far (Higuera et al., 2009). For this analysis, the subsamples were prepared using the method developed by Rhodes (1998) and Stevenson & Haberle (2005). Large charcoal particles (>150 µm) were counted and analysed using CharAnalysis (Higuera et al., 2009). The past fire frequency was inferred from the identified fire peaks. These peaks were isolated by dividing charcoal accumulation rate (CHAR) into 'background' and 'peak' components and separating the 'peaks' into 'fire peaks' and 'noise' using a defined threshold (detail presented in **Appendix S1**).

The results are subsequently divided into three zones: I. Pre-Malayu Empire, II. Malayu Empire and III. Post- Malayu Empire, whose boundaries are based on the occupation period in Muara Jambi. These boundaries chronologically follow the SB-B age-depth model based on calibrated ^{14}C ages (Fig. 3.2; **Appendix S1**). As for the SB-B chronology, this study expresses the age in years before present (yr BP).

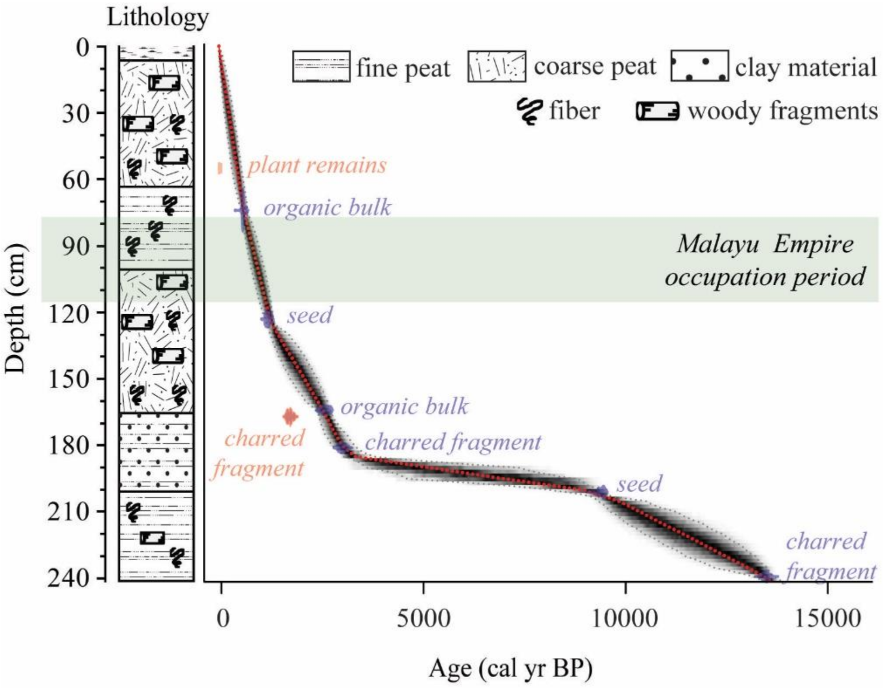

Fig. 3.2. Age-depth model of Sungai Buluh peatland (after Hapsari et al., 2017). Red points and font represent the excluded samples. The boundaries of Malayu Empire occupation period range follow the best estimated date (red dotted line).

The exact starting date of Malayu Empire occupation in Muara Jambi is, however, not available. According to Boechari (1985), the Muara Jambi temple complex was built somewhere "between the middle of the 9th and the early 10th century". This date is circumstantial, as it was proposed based on the similarity of the character found in the inscription of Muara Jambi with the "Kawi" script that was used during the same range of time (Boechari, 1985; J. Miksic, pers. comm. 24 April 2017). We converted this 'century' date into yr BP and divided the 'early', 'middle' and 'late' century periods into '(x)00-(x)35', '(x)35-(x)65' and '(x)65-(x+1)00'. The estimated starting date of Malayu Empire occupation therefore ranges from 835 to 935 AD (1115 to 1015 yr BP) which, according to the age-depth model of the SB-B core, occurs at a depth of 109-117 cm (Fig. 3.2). Conversely, the exact date when the Empire was moved to the hinterland was previously reported to be 1347 AD (603 yr BP; Witrianto, 2014) which falls around a depth of 77.5 cm in the core (Fig. 3.2). We therefore set the boundaries of zone II (Malayu Empire) based on the occurrence of human activity indicators within the occupation period range of the Malayu Empire and the site abandonment date.

To assess the impact of human activities on the C sequestration ability during the Malayu Empire, the pre-, during and post- Malayu Empire, PAR and CAR of Sungai Buluh peatland were statistically compared by calculating the average rates during each period in R (R Core Team, 2014). The average PAR and CAR during the topogenous and ombrogenous phases in the pre-Malayu Empire zone were also differentiated.

The trends in floristic composition of Sungai Buluh were analysed using a principle component analysis (PCA) of pollen percentage data in R. Square root data transformation was applied prior to ordination to minimise biases arising due to differential pollen productivity, dispersal and preservation.

3.4. Results

3.4.1. *Palaeoenvironmental data*

Zone I. Pre-Malayu Empire (240-112 cm; ca. 13,400-1050 cal yr BP) covers the entire topogenous phase and part of the ombrogenous phase of the Sungai Buluh peatland. This zone was characterised by domination of trees/shrubs pollen (average 96 %; the following values are also averages), with a low percentage of herbs/subshrubs pollen (4 %; Fig. 3.3). In this zone, the average charcoal concentration was 86 particles cm^{-3} while the average charcoal accumulation rate (CHAR) was 1 particle cm^{-2} yr^{-1}. A total of 20 charcoal peaks were identified in this zone, with a low average fire frequency (2 fire events in 1000 years; Fig. 3.4).

Zone II. Malayu Empire (112-77.5 cm; ca. 1050-600 cal yr BP) was characterised by a decrease in the total average of trees/shrubs pollen from 96 to 74 %, accompanied by an increasing proportion of herbs/subshrubs pollen from 4 to 26 % (Fig. 3.3). In this zone, pollen of Piperaceae, Poaceae, *Ardisia* and Grewioideae increased significantly. The average charcoal concentration slightly decreased to 63 particles cm^{-3} and the average CHAR increased to 5 particles cm^{-2} yr^{-1}. Only one charcoal peak was identified, with no change observed in the average fire frequency (2 fire events in 1000 years; Fig. 3.4).

In the beginning of zone III. Post-Malayu Empire (77.5-0 cm; ca. 600 cal yr BP to present), the pollen percentages of Poaceae and Grewioideae reached their highest values (16 and 7 %), whereas other herbs/subshrubs pollen decreased to 4 % (Fig. 3.3). At around 450 cal yr BP, the proportion of Poaceae and Grewioideae pollen decreased to 2 %, followed by an increase of trees/shrubs pollen to 93 %, while other herbs/subshrubs made up 5 %. In this zone, both averages of charcoal concentration and CHAR increased to 137 particles cm^{-3} and 17 particles cm^{-2} yr^{-1}, respectively. Two charcoal peaks were detected and the average fire frequency slightly increased to 3 fire events in 1000 years (Fig. 3.4).

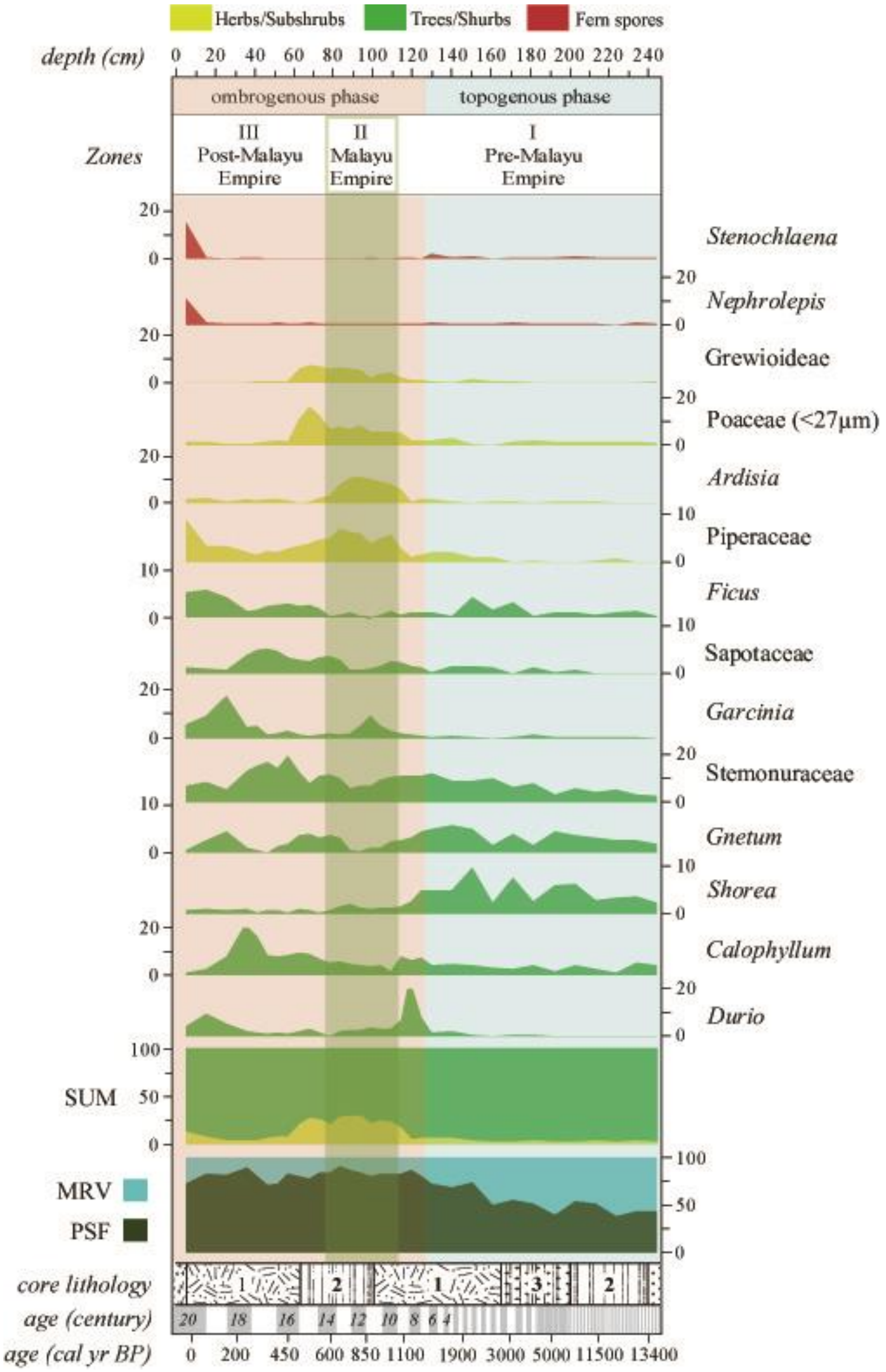

Fig. 3.3. Pollen diagram of the Sungai Buluh. Selected taxa are displayed. All values are in percentages (%). Numbers in the lithology column represent the following peat characteristics (1) coarse peat; (2) fine peat; (3) fine peat with clay. Each of the grey and white bar on age(century) represents 100 years. The proportion of mixed-riverine (MRV) and peat-swamp forest (PSF) of Sungai Buluh is calculated based on the total MRV and PSF pollen (after Hapsari et al., 2017).

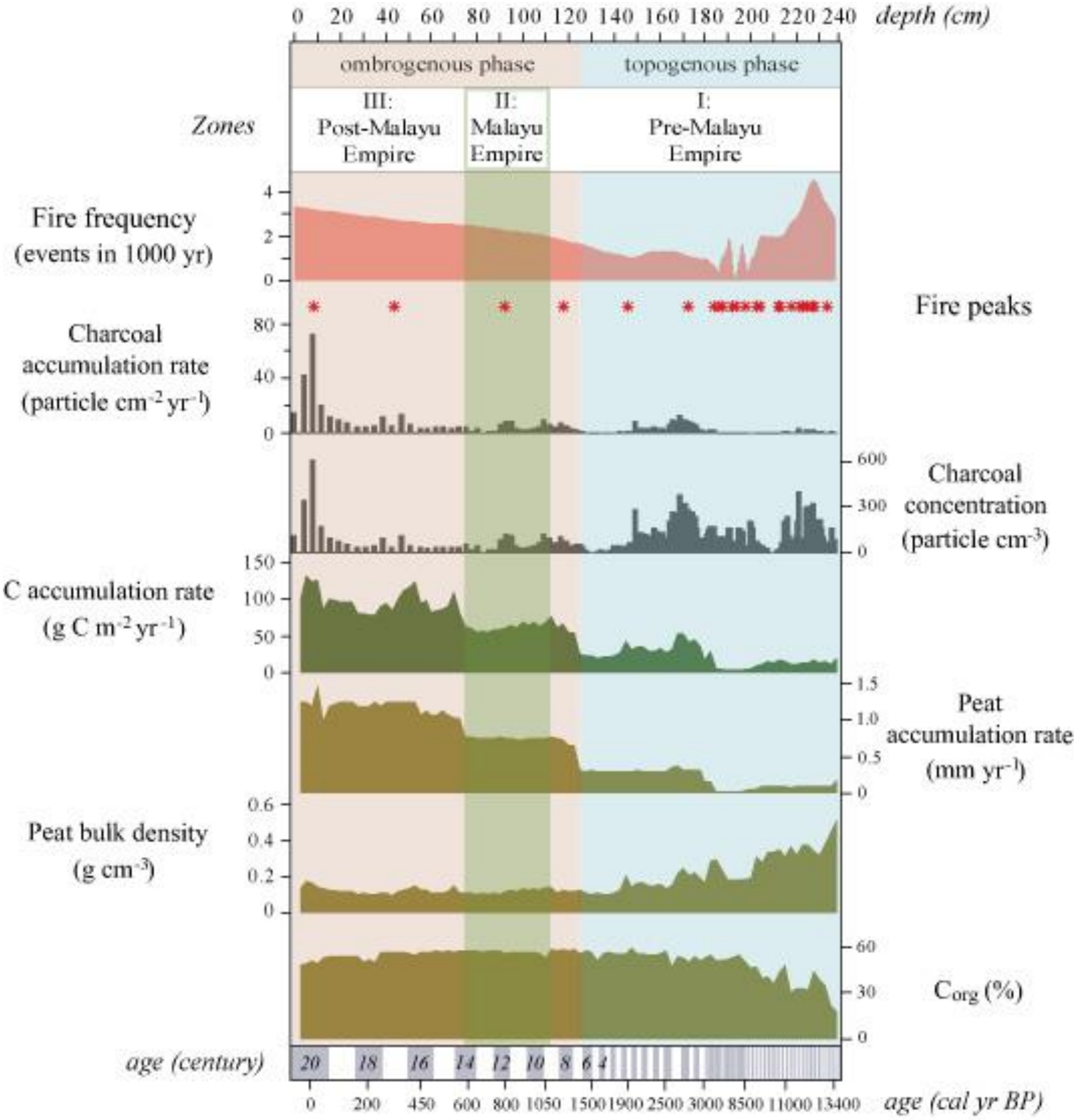

Fig. 3.4. C_{org}, peat bulk density, peat accumulation rate (PAR), C accumulation rate (CAR) and charcoal analysis results for the Sungai Buluh record (after Hapsari et al., 2017). Each grey and white bar on age-century represents one hundred years.

3.1. Carbon accumulation rate (CAR)

Initially, the CAR of Sungai Buluh peatland can be differentiated according to its development into a topogenous and an ombrogenous phase (Hapsari et al., 2017). Peatlands accumulate peat and C slowly during the topogenous phase and then at much higher rates during the ombrogenous phase (Page et al., 2011; Hapsari et al., 2017).

Before the occupation period of the Malayu Empire and during its topogenous phase, the PAR and CAR of Sungai Buluh peatland were low (average 0.2 mm yr^{-1} and 25 g C m^{-2} yr^{-1}; Fig. 3.4 and 3.5). The rates then increased to 0.8 mm yr^{-1} and 69 g C m^{-2} yr^{-1}, respectively, during the ombrogenous phase at around 1200 cal yr BP. The CAR decreased slightly around the occupation period of the Malayu Empire to 62 g C m^{-2} yr^{-1} (Fig. 3.4 and 3.5). Following site abonnement after the demise of the Malayu Empire, the PAR and CAR of Sungai Buluh increased to 1.2 mm yr^{-1} and 96 g C m^{-2} yr^{-1}, respectively (Fig. 3.4 and 3.5).

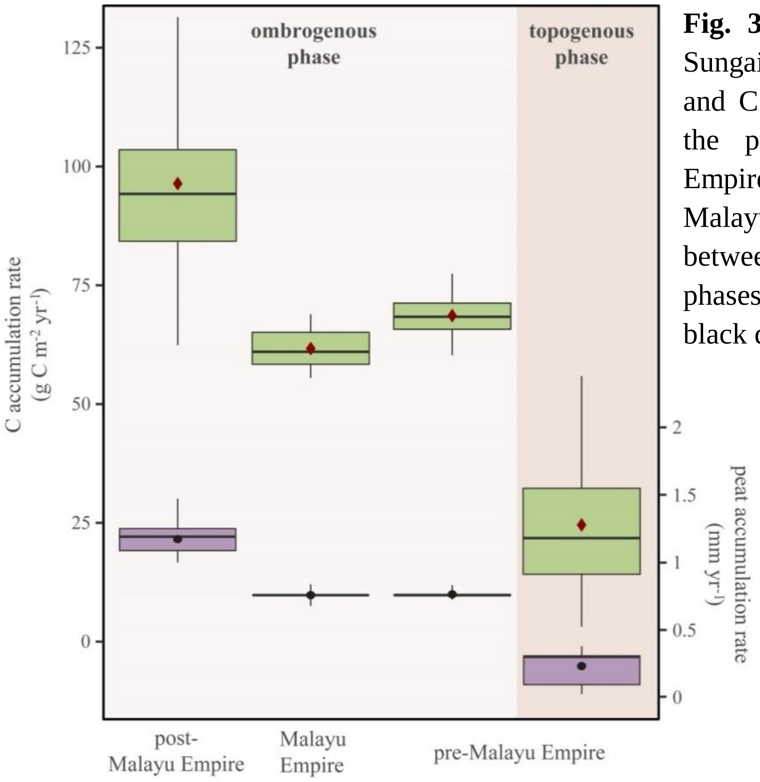

Fig. 3.5. Boxplot comparison of the Sungai Buluh peatland peat (purple) and C (green) accumulation rates for the pre-, during and post-Malayu Empire periods. The rates ot the pre-Malayu Empire are differentiated between topo- and ombrogenous phases. Red diamond symbols and black dots represent the average values

3.2. *Trends in floristic composition of Sungai Buluh*

PCA results based on pollen and spore data demonstrate the characteristics of taxa assemblage and reveal changes in vegetation over time (Fig. 6). The first and second axes of the PCA (PC1 and PC2) represent the two main factors of vegetation changes that explain 41 % and 10 % of the total variance in the data. PC1 can be interpreted as the development stage of Sungai Buluh peatland, whereas PC2 likely represents the disturbance phase. The PCA plot for the Sungai Buluh record samples shows a clear separation between three phases of vegetation composition: mixed-riverine forest, peat-swamp forest and forest opening. The pattern indicates a change of vegetation from mixed-riverine forest to peat-swamp forest around 1200 cal yr BP and later an alteration towards Poaceae, *Ardisia* and Grewioideae at 1050 cal yr BP. The PCA pattern subsequently shifted back to peat-swamp forest at 430 cal yr BP.

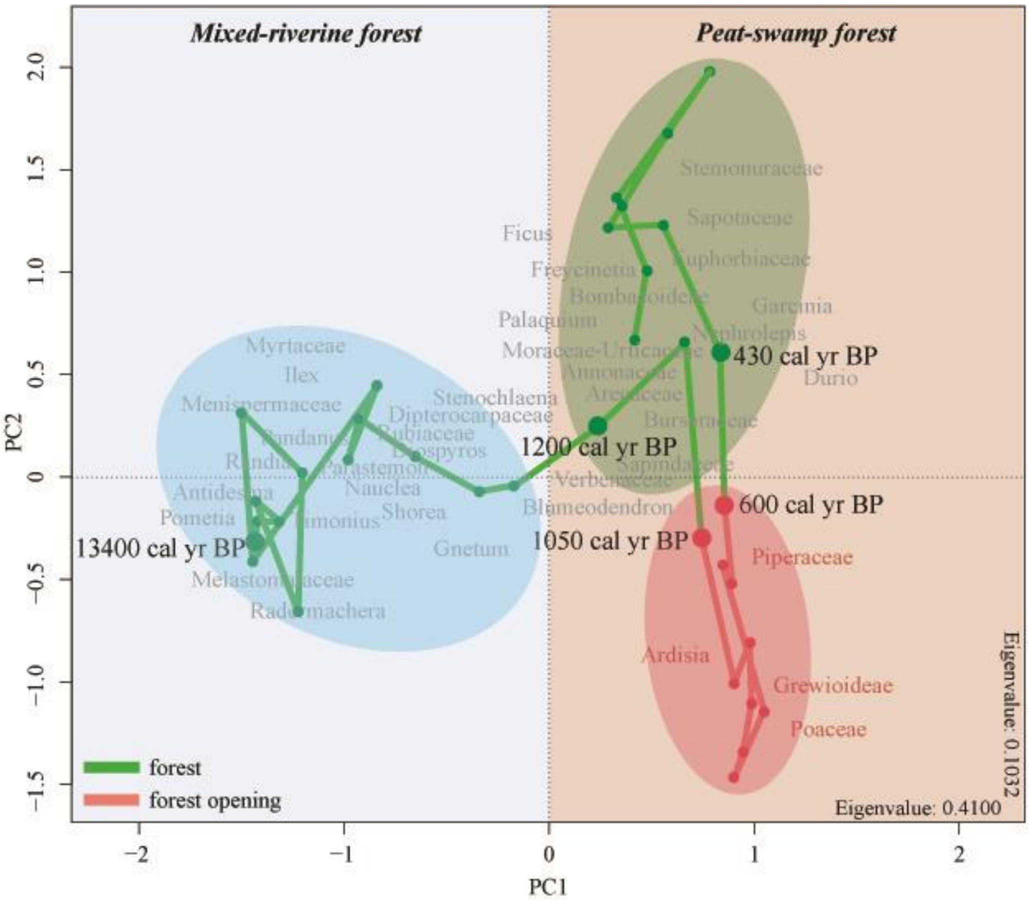

Fig. 3.6. PCA plot for Sungai Buluh display the taxa assemblage and the trend in vegetation change. Green dots represent the forest cover while red dots depict the forest opening. The circles represent three different phases of Sungai Buluh vegetation: mixed-riverine forest (blue), peat-swamp forest (green) and forest opening (red). PC1 can be interpreted as the development stage of the Sungai Buluh peatland, whereas PC2 likely represents the disturbance phase.

3.4. Discussion

3.4.1. Human activities and their impacts on peatland

Before the Malayu Empire occupation period, the forest cover of the Sungai Buluh peatland was dense, as indicated by the domination of trees and shrubs pollen in the diagram (Fig. 3.3). The Sungai Buluh forest was initially dominated by mixed-riverine vegetation. Due to fewer river floods following the late Holocene sea level regression, Sungai Buluh forest cover gradually shifted towards peat-swamp forest, which was fully established at 1200 cal yr BP (Fig. 3.3; see **Section 2.2**).

Sungai Buluh forest cover declined after the establishment of Muara Jambi temple, as shown by the decrease in the proportion of pollen from peat-swamp taxa, such as *Shorea*, *Durio*, *Gnetum*, *Calophyllum* and *Garcinia* (Fig. 3.3), and confirmed by a similar decline in their absolute terms (i.e. pollen concentration; **Appendix S2**). The proportion of Stemonuraceae also decreased slightly (Fig. 3.3). This decline was accompanied by an increase of Piperaceae, Poaceae, *Ardisia* and Grewioideae which usually grow in disturbed areas due to logging, cattle grazing, road building and/or forest clearing (e.g. Chung, 2006; van Eijk et al., 2009; Lim, 2012; Padmanaba & Sheil, 2014).

Shorea, *Durio* and *Calophyllum* trees are well-known timber plants commonly used for construction, boats and furniture (Soerianegara & Lemmens, 1994; Lemmens et al., 1995). Their wood is easy to work with and highly durable (Soerianegara & Lemmens, 1994; Lemmens et al., 1995). *Garcinia* is used as an alternative timber plant for temporary constructions, furniture and decoration (Sosef et al., 1998). The wood of *Garcinia* is easy to work with and durable, but it often shrinks with age (Sosef et al., 1998). On the other hand, Stemonuraceae are preferably used as charcoal wood and are less popular as timber trees, likely due to their low durability and because their wood is very hard to work with (Sosef et al., 1998).

Considering that the tree taxa which exhibited a pronounce decline in absolute and proportional abundance following the establishment of Muara Jambi temple are commonly used for timber (Soerianegara & Lemmens, 1994; Lemmens et al., 1995; Sosef et al., 1998), it is conceivable that the opening in forest vegetation resulted from logging and timber harvesting. According to an ancient text written by Chau Ju-Kua in 1225 AD, the inhabitants of Muara Jambi utilised the available natural resources such as woods, rattan or palm leaves to construct their houses (Hirth & Rockhill, 1966; Tjoa-Bonatz et al., 2009). Moreover, the people of the Malayu Empire were not an egalitarian society, which means that there was a need to show their social status by constructing elite houses with massive and detailed ornamentation (Tjoa-Bonatz et al., 2009; Alam, 2014). This may have also led to the use of timber for decorative purposes. Also, taking into account that connections and trade between the capital Muara Jambi and the surrounding settlements and communities were accommodated through the river (Tjoa-Bonatz et al., 2009; Alam, 2014), it is also conceivable that the wood was used in construction of boats and trading ships.

Ardisia is a plant that commonly grows rapidly on grazing grounds (Hoogerwerf, 1970). Unlike most species of wild Poaceae and herbaceous Grewioideae such as *Corchorus* that are usually grazed by animals or cut as livestock fodder (Humphreys, 1991; Quattrocchi, 2012; Ali et al., 2014), *Ardisia* is unpalatable due to the high alkaloid content in the leaves that results in a bitter taste and potential toxicity at higher intakes (Johnsingh, 2004; Ong, 2004; Dolan et al., 2010). Thus, whenever grasses and other herbs are grazed or cut for fodder, the reduced competition for light, nutrients and other resources favours the growth of *Ardisia* (Koop, 2004). The abundance of *Ardisia* pollen in the record thus indicates grazing or cut-and-carry (a fodder system in which the grasses and herbaceous plants are cut from the field and delivered to confined livestock) activities conducted by people of the Malayu Empire in the Sungai Buluh peatland.

Although the history of livestock in Indonesia was already documented since the 4th century in ancient inscriptions and texts, historical records of livestock raising in the Malayu Empire is not available. In ancient inscriptions of the Kutai Kingdom in Borneo (4th-14th century), for instance, it was described that King Mulawarman of Kutai donated thousands of cows to the Brahmin priests (Poerbatjaraka, 1952). A Chinese ancient text also described that people of the Nagur Kingdom, North Sumatra (5th-12th century), bred and raised livestock for personal consumption (Damanik, 1974). Currently, goat, buffalo and cow grazing are common practices in SE Asian peatlands, with Palangkaraya, Kalmantan (Setianah et al., 2004) and in Phru Kuan Kreng, Thailand (Sundari, 2005) being prime examples.

According to the historical record, the Malayu Empire economy was primarily based on fishing, collection of forest and ocean products, and trading of goods through a regional trading centre such as Muara Jambi (Andaya, 1993). This empire was also very well known as the largest pepper (*Piper* sp. Family: Piperaceae) centre of Sumatra (Witrianto, 2014). Even though no clear documentation remains regarding the type of forest goods and what kind of forest they were taken from, or whether pepper was amongst the non-timber forests products (NTFP), it is feasible that the Malayu Empire population benefitted from the increase of Piperaceae by conducting wild-harvesting activities in the Sungai Buluh peatland.

Records of human activities in SE Asian peatlands such as Danau Sentarum, Borneo (Anshari et al., 2001), Sikijang swamp, Sumatra (Flenley & Butler, 2001), Kutai peatland, Borneo (Hope et al., 2005) and Loagan Bunut, Sarawak-Borneo (Hunt & Premathilake, 2012) are mainly based on the abundance of charcoal. Increased charcoal deposition in those records were interpreted as resulting from arboriculture activities (Hunt & Premathilake, 2012), slash-and-burn agriculture (Flenley & Butler, 2001) and fishing (Hope et al., 2005). However, charcoal increases in those records could have also resulted from repeated natural fires (Anshari et al., 2001; Flenley & Butler, 2001).

In contrast to the above human activity records, the charcoal concentration and fire frequency in the Sungai Buluh record were low during the Malayu Empire period (Fig. 3.4). The proportion of *Nephrolepis* and *Stenochlaena*, which vigorously grow on peatlands following fire disturbances, were also very low during this period (Fig. 3.3; Wösten et al, 2008). This suggests that the Malayu Empire population did not use fire extensively for ground clearance or slash-and-burn activities.

In the Sungai Buluh record, pollen of cultivated Poaceae, e.g., rice (*Oryza sativa*-complex; >42 μm; Chaturvedi et al., 1998), corn (*Zea mays*; >58 μm; Skvarla & Larson, 1966), sugar cane (*Saccharum* sp.; >36 μm; Chaturvedi & Datta, 2001) and wheat (*Triticum* sp.; >49 μm; Cetl, 1960) are lacking. All Poaceae pollen grains found in our samples are <27 μm in size, indicating they are from wild grasses (Page, 1978; Köhler & Lange, 1979; Dickson, 1988). This suggests that inhabitants of the Malayu Empire did not utilise the peatland for agricultural purposes.

During the Malayu Empire occupation period, there was approximately 10 % lower CAR compared to the previous ombrogenous phase (Fig. 3.4 and 3.5). Initially, during the topogenous phase, CAR of Sungai Buluh was low due to a strong water table fluctuation (see **Section 2.2**). However, as the water table stabilised, OM accumulation in Sungai Buluh accelerated, resulting in the establishment of an ombrogenous peatland with rapid C accumulation around 1200 cal yr BP (see **Section 2.2**).

The activities conducted by inhabitants of the Malayu Empire caused a reduction in Sungai Buluh forest cover (Fig. 3.3). A decline in forest cover can negatively affect an ecosystem's net primary productivity (NPP; Clark et al., 2001; Gautam & Mandal, 2016). Also, herbs and grasses that grow on logged or disturbed forest floors, have slower turnover rates of fine roots (Cheplick, 1998), which are a major contributor to the CAR in tropical peatlands (Chimner & Ewel, 2005). Moreover, the shift in composition from woody plants to herbs and grasses can reduce the input of recalcitrant litter, which increases OM decomposability (Silver & Miya, 2001; Chimner & Ewel, 2005; Hapsari et al., 2017). Although a harvested or logged forest can receive high pulses of lignin-rich OM through the accumulation of on-site woody wastes such as sticks and wood chips (Liu et al., 2006), their contribution is still lower than the total biomass input from living vegetation (Clark et al., 2002). Therefore, the slight decrease in CAR observed in the ombrogenous Sungai Buluh peatland during the Malayu Empire occupation period likely resulted from a lower NPP and litter quality concomitant with forest opening (Hapsari et al., 2017).

Despite the reduction of NPP and lignin-rich OM input, the decrease in Sungai Buluh's CAR during the Malayu Empire occupation period can be considered minor. Considering that the CAR in tropical peatland is a result of the interplay between NPP, litter quality and slow anaerobic decay (Sjögersten et al., 2014), the water-saturated conditions of the Sungai Buluh peatland must have been maintained to ensure OM preservation during the Malayu Empire occupation. This, in turn, suggests that logging/timber harvesting, grazing/cut-and-carry and wild-harvesting activities conducted by the Malayu Empire did not significantly alter Sungai Buluh peatland hydrology. The persistence of Sungai Buluh peatland C sequestration, despite a slight decrease during the Malayu Empire occupation, also demonstrates that human activities do not always abolish all ecosystem functions.

3.4.2. Peatland response to human disturbance

After the conquest by Majapahit in the 14th century, the Malayu Empire was ruled by an official of the Majapahit court, namely Adityawarman. The empire then became a vassal of Majapahit until Adityawarman declared his independence in 1347 AD and subsequently moved the Empire to Saruaso, West Sumatra (Witrianto, 2014; Kozok & van Reijn, 2010). Following the abandonment of the site, the forest of the Sungai Buluh peatland regenerated as indicated by an increase of trees/shrubs pollen (Fig. 3.3).

Human activities might impair the forest’s ability to regenerate or recover (Newton & Tejedor, 2011). For instance, logging decreases the abundance of mature trees, increases pollination distance and enhances the risk of inbreeding (Ghazoul, 2005; Carneiro et al., 2009; Tambarussi et al., 2017). Grazing hinders forest regeneration by reducing seedling growth and survival, as young plants are consumed or trampled (Navroodi, 2015). Collection of non-timber forest products such as fruits or nuts can also increase “seed-predation” and subsequently reduce seed availability (Murali et al., 1996).

Thus, the site abandonment with subsequent cessation of human activities in Sungai Buluh might explain the high proportions of Poaceae and Grewioideae which accompanied the decrease of *Ardisia* between 600 and 450 cal yr BP (Fig. 3.3). According to Bruijnzeel et al. (2010), grasses and other palatable plants can grow vigorously once grazing is discontinued. The cessation of grazing/cut-and-carry activities in Sungai Buluh peatland allowed Poaceae and Grewioideae to expand, resulting in increasing competition for *Ardisia* which suppressed its growth (Koop, 2004).

Additionally, the cessation of human activities in Sungai Buluh allowed the forest to regenerate/recover which was initiated by the growth of rapidly regenerating trees such as *Gnetum*, *Calophyllum*, Sapotaceae and *Ficus* (Fig. 3.3; Soerianegara & Lemmens, 1994; Manner & Elevitch, 2006; Gunawan et al., 2012; Cottee-Jones et al., 2016). *Gnetum* is a taxon which is able to resprout (Bullecer & Bullecer, 2011). This taxon and the taxon *Calophyllum* can reproduce more than once per year and produce around 100 kg seeds annually, with a high seed viability and relatively fast germination (3-12 months; Cadiz & Florido 2001; Friday & Okano, 2006; Manner Elevitch, 2006; Kumar et al., 2015). Moreover, *Calophyllum* exhibits a high rate of seed germination and seedling survival (>90 % and ~100 %; Friday & Okano, 2006). Compared to *Calophyllum*, the seed germination and seedling survival rates of *Ficus* (50-75 % and ~10 %, respectively) are lower (Kuaraksa & Elliott, 2013). However, it seems that the reproduction of *Ficus* is compensated for by the enormous numbers of seeds produced (>100 seeds/fruit) in addition to its resprouting ability (Sosef et al., 1998; Howe, 2016). The fleshy-fruits of *Ficus* are also attractive to frugivorous animals such as birds or bats, which enhances the effectiveness of its seed dispersal (Cottee-Jones et al., 2016). Sapotaceae also produce fleshy-fruits that attract frugivorous animals and their rates of seed germination and seedling survival are high (75-85 % and 92 %, respectively; Soerianegara & Lemmens, 1994; Mizuno et al., 2016).

Forest regeneration/recovery processes begin with colonisation (Chazdon, 2014). The success of this phase relies upon several factors, such as seed availability, the role of remnant vegetation, dispersal effectiveness and post-dispersal seed fate (Guariguata & Ostertag, 2001). Considering the regrowth of Sungai Buluh peat-swamp forest, it is likely that the pre-requisites for successful colonisation of forest gaps/openings in this peatland were met. The resprouting ability and seed production, effective seed dispersal by animals as well as excellent seed germination and seedling survival of *Gnetum*, *Calophyllum,* Sapotaceae and *Ficus* likely favoured the colonisation of gaps/openings in the Sungai Buluh peat-swamp forest during its regeneration/recovery process.

The next stage of the forest regeneration/recovery process is species accumulation (Guariguata & Ostertag, 2001; Chazdon et al., 2007). During a natural recruitment, species will accumulate gradually (Chazdon et al., 2007). It may take decades for the forest to reach a floristic composition that is similar to that of an old-growth forest, although it is often questioned whether the forest regrowth will ever converge with the initial floristic composition (Finegan, 1996; Guariguata & Ostertag, 2001; Chazdon, 2003; Chazdon et al., 2007). Based on the PCA analysis of the Sungai Buluh pollen data, the regrowth of the peat-swamp forest returned to its initial floristic composition at 430 cal yr BP (Fig. 3.6). This suggests that the natural recruitment in Sungai Buluh was successful, likely due to sufficient seed sources and the sufficient remaining vegetation from the old-growth forest following the disturbance (Saldarriaga et al., 1988; Finegan, 1996).

Some species, however, may fail to recruit (Chazdon et al., 2007; Clark, 2009). In Sungai Buluh’s case, *Shorea* was not able to regrow (Fig. 3.3). This result is in agreement with other studies showing limited regeneration of *Shorea*, such as in the

Pelalawan peatland, Riau, Sumatra after wind disturbance (Gunawan et al., 2007) and in Sungai Damit peatland, Darussalam, Brunei after logging operations (Kobayashi, 2000). Although the seedling survival rate of *Shorea* is reportedly high (80-90 %; Wibisono & Gandrung, 2008), *Shorea* is an insect-pollinated species that suffers from limited pollination (Ghazoul, 1998; Masuda et al., 2013). *Trigona* bees, the main pollinator of *Shorea*, are weak fliers (Masuda et al., 2013). The increasing distance between flowering *Shorea* trees due to logging or tree fall thus limits the movement and hence the pollination success of *Trigona* (Ghazoul, 1998). Moreover, as the seeds of *Shorea* are edible and energy rich, their reproductive success is probably low owing to the high risk of seed predation by vertebrates and invertebrates in both pre- and post-seed dispersal phases (Ghazoul, 2016). Thus, the restricted regeneration of *Shorea* in the Sungai Buluh, the Pelalawan and the Sungai Damit peatlands are likely related to the above limitations during their early recruitment stage (i.e. seed production and survival). The recovery of Sungai Buluh thus raises a question: how long did it take for the forest to return to its previous floristic composition? Sungai Buluh forest recovered at around 430 cal yr BP (Fig. 3.6). Considering that the site of Muara Jambi temple complex was abandoned at 603 cal yr BP, it is estimated that the recovery process of Sungai Buluh forest took around 170 years. Taking dating uncertainties into account, the estimated forest recovery time ranges from 75 to 300 years.

The recovery time of the Sungai Buluh peat-swamp forest is in the same range as the 150-year recovery time of a lowland forest in Guatemala after Mayan polities dissolved (Islebe et al., 1996) and the 200-year average recovery time of global tropical forests derived from 71 palaeoecological records (Cole et al., 2014). However, the context of ‘recovery’ in those studies is only defined as “increased abundance of forest taxa”

(Islebe et al., 1996) and “the maximum percentage of forest pollen in the displayed pollen sum after a decline, before a stabilising point or further decline” (Cole et al., 2014), without clear annotations indicating whether or not the return of the previous floristic composition was achieved. Moreover, the recovery time in Cole et al. (2014) was not restricted to forests subjected to anthropogenic disturbance, but also included those subjected to natural disturbances such as hurricane and drought.

Forest recovery from human disturbances might take longer than from natural causes (Cole et al., 2014). Compared to the natural factors, human activities can cause more severe disruption in forest structure and composition that might also limit forest recovery, such as a reduction in seed availability, an increase in the distance of seed sources, or changes to the community of seed-dispersing fauna (Chazdon, 2003). In some cases, human activities can also impact soil properties by causing compaction and nutrient leaching (Chazdon et al., 2007). Following severe human disturbances, it may take centuries for forests to fully recover (Chazdon et al., 2007). For instance, neotropical forests in Colombia, Venezuela and Panama still do not converge with their previous floristic composition even after 80-100 years of succession following grazing and slash-and-burn agriculture (Saldarriaga et al., 1988; Denslow & Guzman, 2000).

The recovery time of Sungai Buluh, however, cannot be compared specifically to those of other anthropogenically disturbed tropical peat-swamp forests due to a lack of data. The study from Cole et al. (2015) that aimed to assess the resilience of coastal peatlands in Malaysia did not find any evidence of forest regeneration since the sites were first subjected to human disturbance at 500 cal yr BP.

In contrast to the floristic composition, the Sungai Buluh CAR already began increasing around 540 cal yr BP (Fig. 3.4), suggesting that the peatland capacity to sequester C had returned to its pre-disturbance state. The recovery time of this ecosystem function in Sungai Buluh is estimated to be on the order of 60 years, or ranging from immediate recovery to 130 years considering the dating uncertainties. Ecosystems are able to recover some of their functions before the recovery of their former floristic composition occurs, which may not happen, i.e. a different species composition may result post-disturbance (Guariguata & Ostertag, 2001). For instance, the regeneration of any plants, regardless of the species, can enable important function to resume in the ecosystem such as soil stabilisation due to root growth (Guariguata & Ostertag, 2001).

During the forest recovery process, biomass accumulation, particularly of leaves and fine roots, is brisk due to rapid seedling establishment and fast growth of saplings (Denslow & Guzman, 2000; Guariguata & Ostertag, 2001; Feshe et al., 2002; Letcher & Chazdon, 2009). In tropical peatlands, fine root biomass largely contributes to the CAR due to their slow decomposition under waterlogged conditions, in addition to their high lignin content (Brady, 1997; Chimner & Ewel, 2005). Thus, the rapid recovery of the C sink function of Sungai Buluh peatland is likely related to rapid fine-root biomass accumulation of juvenile forest vegetation during the forest recovery process. It is also inferred that the recovery time of individual ecosystem functions may vary, with recovery of the C sequestration function being faster than recovery of the floristic composition in the case of the Sungai Buluh peatland.

3.5. Implication for tropical peatland conservation and restoration

Understanding peatland resilience to human disturbances can assist in integrating local people and their socioeconomic needs into conservation and management strategies. This insight, in combination with adequate ecological knowledge, is essential for successful conservation programs (Muhumuza & Balkwill, 2013). However, the records of human disturbance in tropical peatlands are sparse and evidence of peatland resilience to human disturbance is as yet not available. Thus, the palaeoecological record from Sungai Buluh provides the first evidence of tropical peatland resilience to anthropogenic disturbance by revealing recovery following the cessation of human activities (visualised in Fig. 3.7).

In Indonesia, conservation efforts often attempt to exclude the people and their activities from the targeted ecosystems (Mulyana et al., 2010). As the local people's livelihoods heavily depend on natural resources, such attempts caused conflicts between local people and practitioners. To overcome these conflicts, the interests of local people need to be accommodated by allocating some conservation areas to support the livelihood and interest of the local community (Mulyana et al., 2010). However, proper constraints on the use of natural resources need to be determined to ensure the sustainability of the ecosystems (Mulyana et al., 2010).

Determination of an 'acceptable' utilisation of peatland natural resources thus can imitate the type and magnitude of past human activities that have proven to be 'resilience-friendly,' meaning that the activities did not undermine ecosystem resilience. In the case of Sungai Buluh, the vegetation and C sequestration ability were able to recover from past logging/timber harvesting, grazing/cut-and-carry and wild-harvesting because the hydrological conditions of the peatland were maintained. Therefore, similar or other activities that do not significantly alter peatland hydrology (i.e. high water table) could potentially be permitted on targeted peatland.

The intensity of the disturbance, on the other hand, is difficult to assess via palaeoecological records. Based on the changes in pollen proportion alone, Sungai

Buluh was able to maintain its regenerative ability after a 30 % decrease in forest cover. However, due to the differences in pollen production, dispersibility and preservation, this data cannot be used quantitatively without supporting information on pollen productivity and dispersal (Williams & Jackson, 2003; Cole et al., 2015; Hjelle et al., 2015). Moreover, limits on wild-harvesting or NTFP collection to prevent seed predation cannot be determined from fossil pollen data.

Insights into ecosystem resilience to disturbances are also crucial for the success of restoration projects and can improve their efficiency and effectiveness in terms of cost and effort (Lake, 2013; Holl, 2017). Restoration is defined as 'the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed’ (SER, 2004) with the major goal being to enhance ecosystem resilience (Lake, 2013). Understanding how ecosystems resist and/or recover from disturbances can thus help restoration projects select suitable techniques/approaches, develop plans, allocate resources and create realistic goals (Holl, 2017).

In Indonesia, peatland restoration is commonly carried out through rewetting and reforestation (BRG, 2016), which require the construction of canal blocks, water ponds, deep wells, nurseries and artificial mounds followed by seedling transportation (Giesen, 2004; BRG, 2016). Such efforts undoubtedly require a large amount of money, which is often not available (Giesen, 2004; BRG, 2016).

The evidence from Sungai Buluh suggests that a tropical peat-swamp forest can passively recover (without assistance), as long as certain prerequisites are met, including maintained hydrological conditions, the availability of seed sources (seed banks or remnant vegetation) and seed dispersers. Therefore, considering the high cost of tree planting on peatlands (US$1100/ha; Graham et al., 2013), passive restoration can be an excellent solution if the current condition of the targeted peatland still fulfills the requirements for natural recovery. Tree planting thus can be restricted to poor-regenerating taxa, such as *Shorea* in the Sungai Buluh peatland, or taxa with high economic value (Giesen & van der Meer, 2009) if desired. Moreover, active restoration is defined to be less efficient than passive restoration, as it is costly, results in a less diverse forest and may negatively affect natural recovery by suppressing the growth of pioneer species (Rey Benayas et al., 2008; Meli et al., 2017).

In some cases, such as forest clearing or when the seed bank has been annihilated by fire, assisted regeneration or recovery is required. In Indonesia, this is commonly carried out by planting tree seedlings mainly of *Shorea*, *Gonystylus* and *Dyera*, in addition to hydrological recovery through rewetting (Giesen & van der Meer, 2009; van Eijk et al., 2009; Graham et al., 2013; Nurjanah et al., 2013). The above species are selected due to their high economic value and high seedling survival rates (Giesen, 2004; Graham, 2009). *Dyera* and *Shorea* are also described as fast-growing taxa (Giesen, 2004; Graham et al., 2013). However, the expectations for success of these attempts are low, supposedly due to a lack of ecological knowledge and attention to socioeconomic factors (Page et al., 2009; van Eijk et al., 2009; Graham et al., 2017).

The success of restoration efforts, particularly broad planting, are thwarted not only by inadequate ecological knowledge and poor understanding of socioeconomic condition, but also by the poor reproductive success of restored populations (Bozzano et al., 2014; Prill, 2016). Therefore, to ensure the long-term success of assisted regeneration, the reproductive patterns of selected taxa must also be considered. For example, *Shorea* experiences limitations in its early reproductive stages (**Section 5.2**). *Gonystylus* does not flower annually and only 20 % of their flowers develop into fruits (Ismail et al., 2011). It also suffers from high seed predation by squirrels and bats and its seed dispersal and germination rates are low (Ibrahim, 1996). Although *Dyera* produces enormous amounts of seeds with high germination rates (Sosef et al., 1998), it flowers only once every 4-5 years (Giesen, 2004). Thus, using *Shorea*, *Gonystylus* and *Dyera* for initial planting might not be optimal to accommodate forest regrowth, as such ecosystems will likely continue to require seedling supplies and active plantings.

As the resources for seedling transplantation are often limited, selecting rapidly regenerating taxa for reforestation might improve the efficiency and reduce the cost of restoration efforts. In the Sungai Buluh record, *Gnetum, Calophyllum, Ficus* and Sapotaceae displayed rapid regeneration, likely due to their high seed productivities, seed viabilities, seed germination rates and seedling survival rates. Those taxa, and/or other peat-swamp taxa that have similar reproductive abilities, are thus more suitable for tree planting on targeted peatlands, particularly as colonisers. The desired taxa, either of conservation or economic interests (e.g. *Shorea*, *Gonystylus* and *Dyera*), can subsequently be transplanted once the forest cover is re-established. Moreover, taxa with high regenerative power can possibly be used in lower-cost alternative restoration efforts such as direct seed sowing (Graham et al., 2013; Saito et al., 2016) or for creating “woodland islets” by planting many small and dense “islands” of trees which later act as seed sources for the surrounding area (Rey Benayas et al., 2008). The savings obtained from these lower-cost alternatives can be allocated towards costly long-term monitoring, aftercare and maintenance, important aspects in restoration that are often neglected (Graham et al., 2017).

In Indonesia, peatland restoration efforts are mostly designated for 60 years (e.g. http://katinganproject.com/explore-katingan; http://www.rekoforest.org/). This

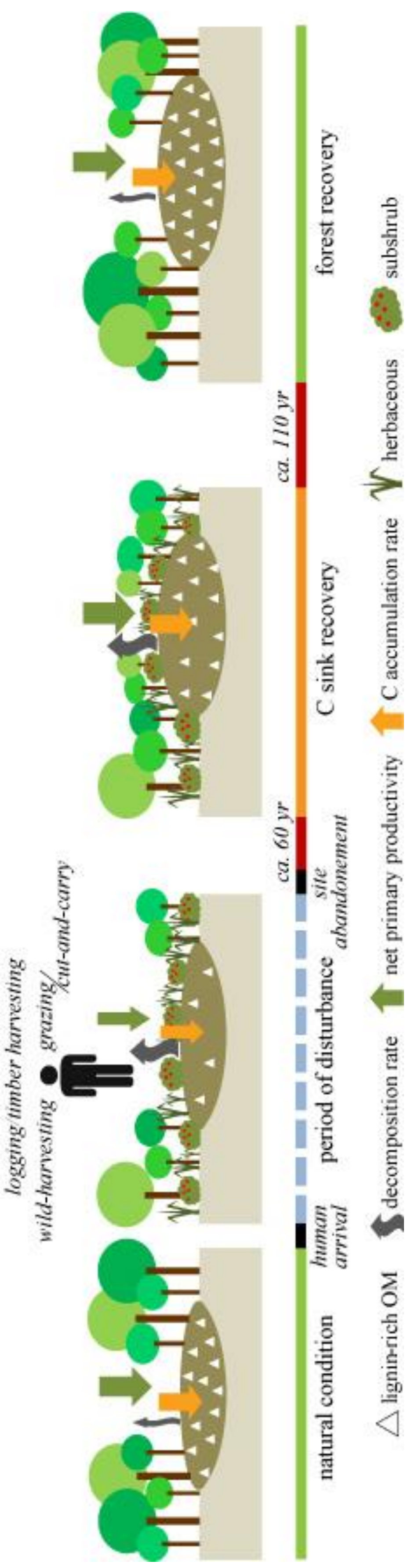

Fig. 7. Visualization of past anthropogenic impact on the Sungai Buluh peatland and the ecosystem's response. Modified from visual summary in Hapsari et al. (2017).

framework follows the 60-year concession license granted by the Ministry of Forestry, Republic of Indonesia. Each permit granted has the possibility of a one-time extension of 35 years (http://kph.menlhk.go.id/). In Sungai Buluh, it took 60 years for the C sequestration function and 170 years for the floristic composition to recover from past human disturbance. Therefore, the current 95-year restoration frames, assuming that the extension is given, are likely insufficient. While this period may be adequate for restoring the C sink potential, it is insufficient for the restoration of biodiversity values, suggesting that policies aimed at granting longer concession licenses should be considered. Alternatively, concessions could be granted for different validity periods according to the aims and goals of the project, with shorter licenses for projects targeting only C sequestration enhancement and longer for those targeting full biodiversity recovery.

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Data accessibility

The data used in the analyses are available at PANGAEA https://pangaea.de/ (Hapsari et al., 2017)

Authors' contribution

KAH conceived the ideas, interpreted the data and led the manuscript writing; TCJ substantially contributed to the intellectual content; KAH, SB and HB analysed the data; KAH, SB and PR collected the data; SB and HB obtained the funding. All authors critically contributed to the manuscript draft and gave final approval for publication.

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Suplementary 1

Detailed methodology

Detailed methodology provided is a reproduction from **Hapsari et al., 2017.**

3.1. Core sampling

Using a Russian Corer (Jowsey, 1966), the SB-B core (1°14’10” S, 103°37’12” E; 18 m asl) was recovered from the Sungai Buluh peat restoration area in 2013. The SB-B core consists of layers of organic matter (OM; from 240 cm to the top of the core) and underlying mineral clay (350 to 240 cm). Following the SE-Asian peat classification system (Esterle & Ferm, 1994; Wüst et al., 2003), the SB-B peat core was also classified into coarse and fine peat in accordance to the size of the fragments and the availability of woody material. An inserted clayey-peat layer was observed in the depth of 200 to 165 cm (see main text **Fig. 3.3. Pollen diagram-core chronology**).

3.2. Radiocarbon chronology

Eight samples of the peat core were selected for radiocarbon dating. The age-depth model of the SB-B core (see main text **Fig. 3.2. Age depth model**) was constructed using the Bayesian accumulation model, Bacon script (Blaauw & Christen, 2011) in R (R Core Team, 2014). The SHCal13 calibration curve was used here. Two samples (at depth 55 and 167 cm) were identified as outliers using a robust Student’s T method (Christen & Pérez, 2009) which may have been part of a root that intruded into the deeper layers.

3.3. Palynological analysis

A total of 35 subsamples (subsampling depth is presented in Table S1.1 below) were processed following standard pollen extraction methods (Faegri & Iversen, 1989). Two tablets of exotic marker *Lycopodium* spores were added to each subsample prior to the extraction. Pollen and spores were identified using the reference collection of pollen and spores available at the Department of Palynology and Climate Dynamics, University of Göttingen, and other available literatures (see main text **Section 3. Material and methods**). The pollen and spore grains were counted up to a sum of 300 pollen grains.

Table S1.1. Subsampling depth for pollen analysis

Sample no.	Depth (cm)	Age (cal yr BP)
1	5	-25
2	15	58
3	25	140
4	35	223
5	40	264
6	45	304
7	50	344
8	55	384
9	60	429
10	65	475
11	70	520
12	75	567
13	80	632
14	85	698
15	90	764
16	95	829
17	100	896
18	105	963
19	110	1029
20	115	1094
21	120	1162
22	125	1237
23	135	1560
24	145	1892
25	155	2216
26	165	2541
27	175	2827
28	185	3309
29	190	5165
30	195	6991
31	205	9821
32	210	10364
33	215	10882
34	225	11959
35	235	13012

3.4. Carbon accumulation rate

Peat bulk density (g m^{-2}) was measured by taking 92 subsamples along the core at 2.5 cm intervals (exception: a 5 cm interval was applied between depths of 185 cm and 200 cm due to insufficient amount of material) and dried overnight at 105ºC (Chambers et al., 2011). Duplications for peat bulk density quantification were performed on 16 of the 92 samples. Organic carbon (C_{org}) content was measured by taking 92 additional subsamples at the same depths that were subsequently dried at 60ºC for 48 h, finely ground, and weighed (~1-1.5 mg). Subsamples were then treated with 1N HCl prior to C_{org} determination in order to remove carbonates. C_{org} content was determined using high temperature oxidation in a Euro EA3000 elemental analyser. The C accumulation rate (g C m^{-2} yr^{-1}) was calculated following the equation used by Tolonen & Turunen (1996; see main text **Section 3. Material and methods**).

3.5. Macro-charcoal analysis

Subsamples were taken continuously along the core at 0.5 cm intervals from the depths of 240-180 cm and at 1 cm intervals from 180-0 cm. A higher subsample resolution was applied to the lower part of the core due to its lower accumulation rate. Each subsample was washed with weak hydrogen peroxide (6 % H_2O_2) to remove organic matter in the sediment and gently wet-sieved (Rhodes, 1998; Stevenson & Haberle, 2005)**.** Particles >125 µm were retained and charcoal particles >150 µm were counted under a stereomicroscope. Charcoal concentrations were calculated as particles cm^{-3}.

The CharAnalysis program (Higuera et al., 2009) was used to identify the fire regime of the Sungai Buluh peatland. The raw charcoal data were interpolated into 32-yr median temporal resolution and transformed into charcoal accumulation rates (particle cm^{-2} yr^{-1}).

The charcoal accumulation rates were subsequently separated into a background (low frequency; related to regional charcoal production) and peak (high frequency; related to local fire episodes) components by applying a 1,000-yr window locally weighted regression that is robust to outliers. Using a locally defined Gaussian mixture model (Gavin et al., 2006; Higuera et al., 2008), the noise results from distant fires, redeposition, or analytical error were removed and the peak component (C_{peak}) was defined as residual. Only the peaks exceeding the 99th percentile threshold of the noise distribution were identified as fire peaks which were subsequently subjected to a "Poisson minimum count" to reduce the counting variance bias. Fire frequencies were calculated based on the frequency of fire peaks per 1,000 yr.

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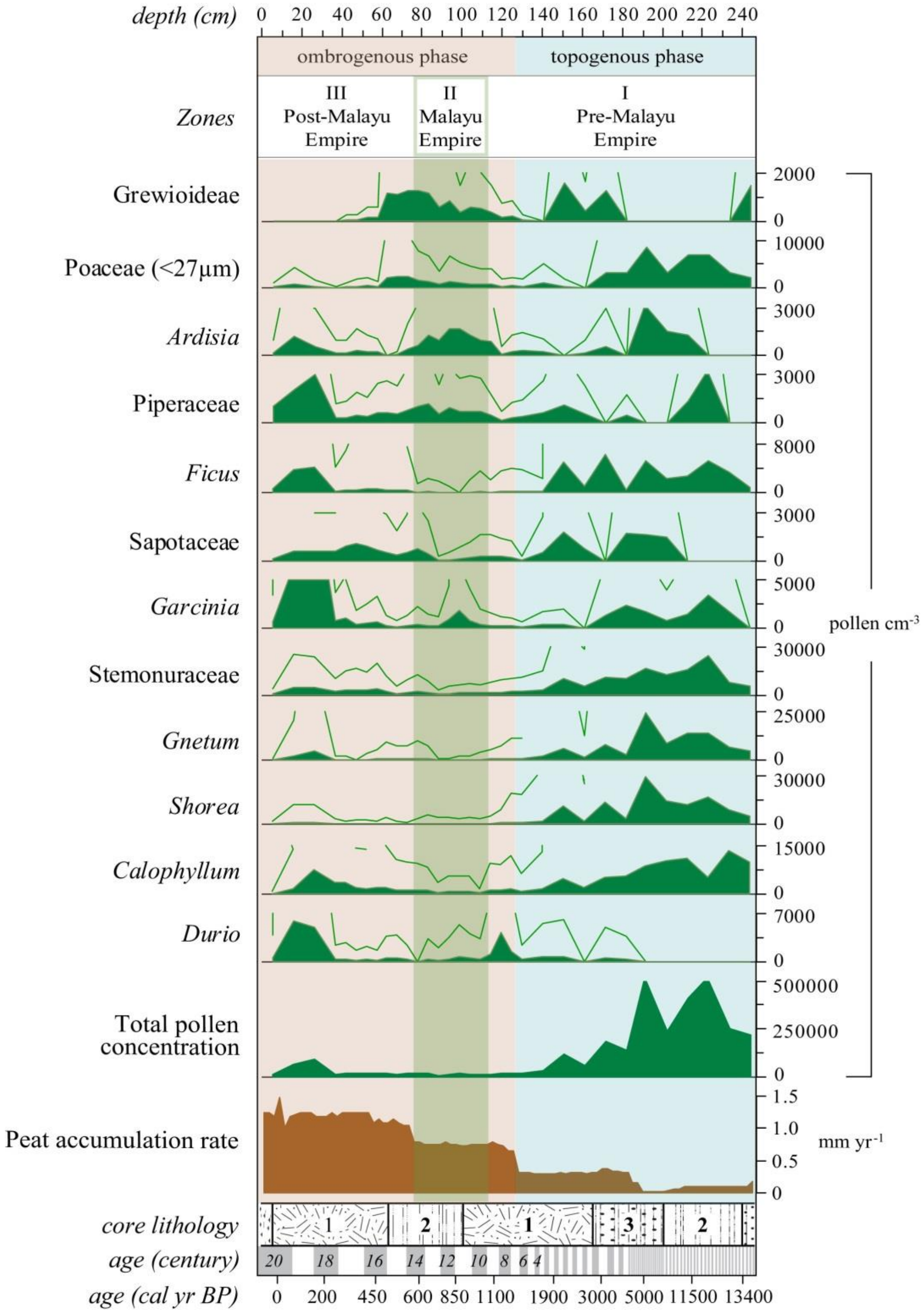
Supplementary 2

Diagram of pollen concentration

The pollen concentrations were calculated using the following equation:

$$(pollen\ grains/lycopodium\ counted) \times$$

$$total\ lycopodium\ added/sample\ volume$$



Green lines without filling represent the exaggerated values of pollen concentration. Only the concentrations of taxa presented in the main text (**Fig. 3.3)** are shown. As pollen concentration appears to be inversely related to the peat accumulation rate (Dickinson, 1975), the enormous pollen concentration in the SB-B core during the topogenous phase is very likely related to its low peat accumulation rate.

Reference

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Chapter 4

Climate and land use change governed environmental dynamics and carbon accumulation in the mangrove-fringed Segara Anakan lagoon, Java, Indonesia in the Anthropocene and beyond

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under review

Abstract

Carbon (C) sequestration in natural sinks is currently urged with regard to global climate change mitigation. Tropical coastal wetlands are considered important in the global C cycle, yet knowledge on their dynamics and quantitative data are still scarce. Moreover, these ecosystems are vulnerable to land use and climate change. In order to quantify the C accumulation rate and to understand the relevance of land use and climate for it in the Anthropocene and beyond, a palaeoecological study was conducted in the mangrove-fringed Segara Anakan Lagoon (SAL) in Java, Indonesia. A sediment core was age dated and analyzed for its pollen and spore, elemental and biogeochemical (organic carbon, nitrogen, stable carbon and nitrogen isotope) compositions. The results indicate that the dynamics of SAL and its C accumulation in the past 400 years were controlled mainly by climate oscillations and anthropogenic activities. Both factors changed the lagoon's sediment supply and salinity that consequently altered the C burial and organic matter (OM) deposition in the lagoon. In the future, the projected precipitation reduction over Java, as well as the increasing trend in lagoon sedimentation due to land use change in the catchment area, will likely favor the SAL's capacity to sequester C. However, widespread logging of the surrounding mangrove forest being the main organic C source in SAL, may diminish the lagoon's C accumulation capacity. Therefore, the forest in SAL needs to be maintained and improved to retain its C sequestration capacity, as it is currently among the highest compared to other mangrove-dominated ecosystems. Considering the urgent need for the removal of atmospheric CO_2, ecosystems with a high C sequestration capacity, such as the mangrove-dominated Segara Anakan Lagoon, thus need to be given priority for conservation and restoration.

Keywords: *Indonesia, palaeoecology, mangrove, estuary, carbon accumulation rate, land use change, climate change*

4.1. Introduction

Coastal lagoons often comprise very dynamic and highly productive wetland ecosystems that provide a wide range of ecosystem services, including, for example, habitats for mangrove forests, salt marshes, and aquatic biota, nutrient cycling, storm protection, fisheries, salt production, and tourism (Anthony et al., 2009; Alongi, 2012). Coastal lagoons are also important for the global carbon (C) cycle because of their high productivity and accretion rate (Twilley et al., 1992; Eong, 1993; Jennerjahn and Ittekkot, 2002; Brevik and Homburg, 2004; Chmura et al., 2003). Proportionally, coastal wetland ecosystems allocate more C below- than aboveground, in particular in their sediments (Lovelock, 2008; Alongi, 2012; Ouyang et al., 2017), highlighting the significance of belowground C storage in these ecosystems. However, as yet the data base is small (Donato et al., 2011; Ezcurra et al., 2016) and the understanding of the underlying dynamics is limited (Marchio et al., 2016).

Coastal lagoons and the ecosystem services they provide are threatened by the effects of global environmental change, such as sea level rise, which can cause coastal erosion, extreme floods, destruction of coastal wetlands, and habitat loss for aquatic biota (Anthony et al., 2009). This will simultaneously affect the social and economic conditions of human societies (Silva et al., 2013). Extensive anthropogenic pressures in coastal areas, such as land use change and natural resource exploitation, also threaten these ecosystems (Anthony et al., 2009; Silva et al., 2013). In this context mangrove ecosystems, which are lining major parts of tropical coasts, are under threat mainly from human interventions like, for example, land use conversion to agriculture and aquaculture, alterations to river catchment hydrology, timber harvesting, overexploitation of natural resources and infrastructure construction (Chowdhury et al., 2017). However, outcomes of climate change may also negatively affect them and their carbon storage potential (Alongi, 2015; Jennerjahn et al., 2017). Mangrove forests have been lost at rates of 1-2 % per year for a long time (Valiela et al., 2001). These rates have slowed down, nevertheless, the loss is still alarmingly high in particular in Southeast Asia (Richards and Friess, 2016).

Humans have been considered as major environmental force, as they have significantly modified biodiversity, climate, and natural cycles of e.g. nutrients, water and carbon (Raupach and Canadell, 2010; Corlett, 2015). The term Anthropocene is used to refer to the current period where humans are seen as important as natural processes in shaping the planet (Corlett, 2015). The Anthropocene is characterized by a substantial increase of anthropogenic CO_2 emissions since the onset of industrialization (Raupach and Canadell, 2010), leading to climate warming, which has devastating consequences, such as sea level rise, extreme weather events, and food shortage (Huber and Gulledge, 2011; Wheeler and von Braun, 2013; Marzeion et al., 2014).

As an effort to mitigate climate warming, a global political action plan was set in place to limit the global temperature increase below 2°C by reducing C emissions (UNFCCC, 2015). However, emission-cuts alone will not be sufficient to keep global warming below 2°C (Rau and Greene, 2015). Efforts to reduce atmospheric CO_2 through natural sequestration, e.g. conservation, restoration and improved land management, need to be undertaken (Griscom et al., 2017).

The Segara Anakan Lagoon (SAL), the last large mangrove-fringed ecosystem on the Indonesian island of Java, is a coastal wetland system that is important for C sequestration, but that has been stressed from intense human activities for centuries (Yuwono et al., 2007). SAL's biodiversity and ecosystem services, e.g. habitat and nursery ground provisions for aquatic biota, are threatened by land conversions to aquaculture/agriculture, mangrove exploitation, and high levels of sedimentation and pollution (White et al., 1989; Yuwono et al., 2007; Jennerjahn and Yuwono, 2009).

Despite being considered a "degraded" mangrove-fringed estuarine ecosystem, soil organic carbon stocks in SAL are similar to those of some "undegraded" mangrove forests in Indonesia (Weiss et al., 2016). However, carbon accumulation and its dynamics are not known, and the knowledge of longer-term environmental dynamics is generally limited. Sedimentation, which has rapidly reduced the size of the lagoon, has been analyzed through shoreline change reconstruction going back to the mid-19th century (Lukas, 2014a; Lukas, 2015; Lukas, 2017), yet the longer-term sedimentation dynamics are not known.

In this paper, we provide information on the long-term environmental dynamics of SAL. This study aims (i) to quantify the C accumulation in lagoon sediment, (ii) to explain how climate and land use change affect lagoon dynamics and its C sequestration capacity, and (iii) to assess the significance of the C accumulation capacity of SAL in comparison to other mangrove-dominated ecosystems.

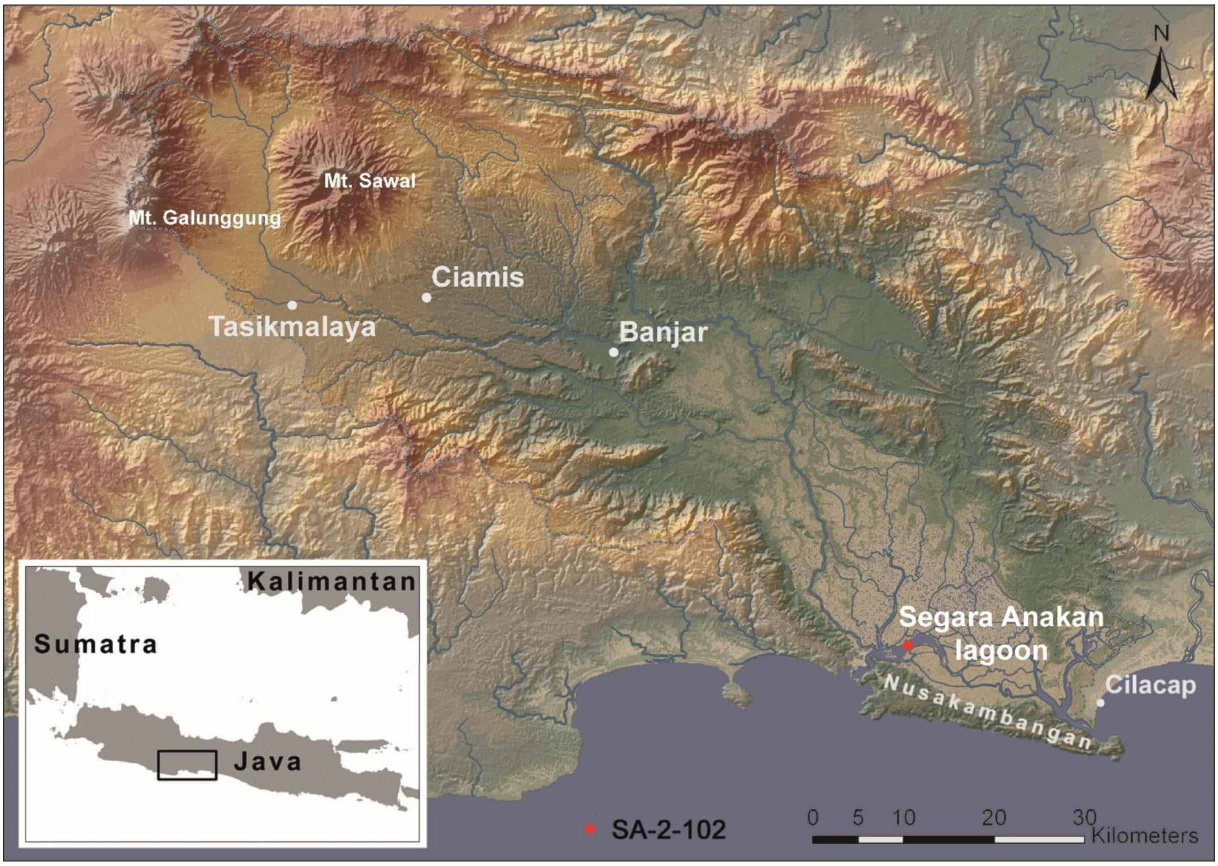


Fig. 4.1. Map of the Segara Anakan Lagoon and its catchment area.

4.2. Study site

A. Ecological setting

The Segara Anakan Lagoon (SAL) is located on the southern coast of Java Island, west of the city of Cilacap (Fig. 4.1). It is surrounded by around 9,200 ha mangrove forest (Ardli and Wolff, 2009) which mainly consists of *Rhizophora apiculata*, *Avicennia corniculatum*, *A. alba*, *Nypa fruticans*, *Sonneratia caseolaris* and *Bruguiera gymnorrhiza* (Hinrichs et al., 2009). The catchment of the SAL comprises an area of around 450,000 ha, with a plain to hilly central river basin, volcanic mountains in the north-western part, and sedimentary mountains in the north-eastern and southern parts (Lukas, 2017).

SAL is separated from the Indian Ocean by the rocky mountainous Nusakambangan Island and is connected to the ocean through two channels in the West and in the East. Lagoon salinity (reported in practical salinity units (psu)) is ranging from 1 to 28 depending on freshwater input and rainfall seasonality (White et al., 1989; Noegrahati and Narsito, 2007; Holtermann et al., 2009). The salinity fluctuation is strongly influenced by the freshwater input from the Citanduy River, which contributes about 80% of the freshwater discharge into SAL with a range of 78-300 $m^3 s^{-1}$ (average 140 $m^3 s^{-1}$; Holtermann et al., 2009). A minor contribution of freshwater input comes from several smaller rivers, such as the Cibereum and Cikonde.

The tides within SAL are semidiurnal with an amplitude ranging from 0.2 to 2.6 m (White et al., 1989). The climate in the lagoon area is tropical humid with a mean annual temperature of 27.2°C and little variation throughout the year. The mean annual precipitation is 3400 mm with drier conditions from July to September and wetter conditions from October to May, and peak rainfall in November (Weatherbase, 2016; www.weatherbase.com). The interannual rainfall variability in this area is influenced by the El Niño Southern Oscillation (ENSO; Qian et al., 2010).

B. Land use change in the Segara Anakan Lagoon and its hinterland

“Segara Anakan” is loosely translated as child of the sea. The lagoon is inhabited by ca. 17,300 “Orang Laut” (people of the sea) who live in the four villages of “Kampung Laut” (Sea Village) spreading across the area of Ujungalang, Ujunggagak, Klaces and Penikel (BPS Cilacap, 2017). Most residents of Kampung Laut make their living as fishers and farmers, whereby farming and aquaculture have gained importance over fishing, as sedimentation has turned substantial parts of the lagoon into land (Olive, 1998; Ardli and Wolff, 2009; Heyde, 2016; BPS Cilacap, 2017).

SAL was already populated by the descendants of guards from the Mataram kingdom, the predecessors of Orang Laut, since at least 1705 (Waryono, 2002; *Schwerdtner* Máñez, 2010; Mulyadi, 2013). In 1724, nine villages were documented in the SAL by Francois Valentijn, a Dutch minister and naturalist (Schwerdtner Máñez, 2010; Mulyadi, 2013). The population of Kampung Laut has risen over time, with some phases of particularly rapid growth, such as between 1924 and 1939, when the number of residents doubled from 680 to 1300 (Waryono, 2002).

According to historical records, SAL was already subjected to pearl exploitation by the Dutch in 1699 (Schwerdtner Máñez, 2010). Documentations of mangrove deforestation go back to the second half of the 19th century. Between the 1870s and 1930s, rising fuelwood and charcoal demands of sugar factories and railway companies by far exceeded local lagoon residents’ timber demands for houses and fishing stakes. This triggered extensive wood extraction, rendering management attempts of the colonial forest administration ineffective (de Haan, 1931).

In the course of the 20th century, substantial mangrove and swamp forest areas surrounding the SAL and in the lower river basin further upstream have been converted into settlements, rice fields, and aquaculture ponds – in the frame of government-initiated resettlements and agricultural development projects, as well as by the growing population and individual settlers moving in from other parts of Java (Olive, 1998; Ardli & Wolff 2009; Irwansyah, 2010; Lukas, 2014b, 2017). By the 1970s, the mangrove area converted to settlements and agricultural land was estimated 8-times (ca. 1,100 ha) larger than in the 1940s (ca. 140 ha; Irwansyah, 2010). Between 1968 and 1987, an additional 10,000 ha of mangrove forest was converted into rice fields (Olive, 1998; Irwansyah, 2010). Much of the remaining mangrove forest area is degraded. Tenure conflicts and mismatches between government planning and the perceptions, claims and practices of local residents on the ground impair mangrove reforestation efforts (Heyde, 2016).

The catchment/hinterland of the SAL has also experienced a long history of land use change. In the early 18th century, the peasants of the former Priangan Regency, to which considerable sections of the lagoon's catchment belonged to, started to shift from swidden agriculture (locally called *ngahuma*), to permanent irrigated rice farming (*sawah*; Herayati et al., 1993). This agricultural system was already introduced by people from the Mataram kingdom during their reign in Priangan from 1620 to 1705 (Herayati et al., 1993; Muhzin, 2008). Compulsory coffee cultivation under the colonial regime, excessive timber extraction, plantation development, and in-migration considerably reduced forest cover in the catchment of the SAL in the 19th and early 20th century (Lukas, 2017). As in the whole of Java, rainfed and irrigated agriculture were considerably expanded particularly in the late 19th and early 20th century (Boomgaard and van Zanden, 1990; Smiet, 1990; van der Eng, 2008). The arable land area in Java increased from about 2.1 million hectares in 1880 to about 4.2 million hectares in 1915 (Boomgaard and van Zanden, 1990). In the catchment of the SAL, the expansion of both irrigated and rainfed agricultural land continued until the 1990s (Lukas, 2017).

Most of the remaining forests were converted into monocultural production forests under the colonial Forest Service and its successor organizations (Peluso, 1992; Lukas, 2015). Today, more than one fifth of the total land area of Java is formally under the control of the state forest corporation. About 50% of these lands are planted with teak (*Tectona grandis*), and 36% are planted with pine (Perum Perhutani, 2016). These two species also dominate most state forest lands in the catchment of the SAL (Lukas, 2015). The pine species *(Pinus merkusii)* is not native to the island of Java, but was introduced from Sumatra and planted for the purpose of resin production starting in the 1920s/30s (Becking, 1935; Fitriani, 2012).

Along with the land use changes described above, forest management practices, conflicts over state forest and plantation lands, as well as erosion and mass-movements from roads, trails, and settlements raised river sediment loads and contributed to lagoon sedimentation from the late 19th and throughout the 20th century (Lukas, 2017). *Ngaguguntur* (the digging back of hill slopes to enlarge agricultural land and shoveling the excavated soil into streams and rivers), agriculture in riparian zones, and river bank erosion have also contributed to high river sediment loads (Diemont et al., 1991; Lukas, 2017). Furthermore, the reclamation of the swamp forests in the lower river basin upstream of the SAL for agriculture and settlements and extensive river straightening and embankments, which started in the first three decades of the 20th century and which were considerably pushed forward in the frame of river and agricultural development projects in the 1970s and 80s, have contributed to sedimentation of the SAL through enhancing sediment transport (Lukas, 2015). A floodway and a river diversion constructed in the frame of these projects in the 1970s/80s have had similar effects (Lukas, 2015).

4.3. Materials and methods

A. Sediment core

In 2014, a 500 cm long sediment core (SA-2-102; 7°40’S 108°49’E; Fig. 4.1) was recovered from SAL using a 5 cm diameter Livingstone piston corer (Wright, 1967). A short replicate core (100 cm) was taken ca. 20 cm away from the SA-2-102 borehole, because the lower half of the first meter (67-100 cm) of the initial core was lost.

B. Sediment dating (^{210}Pb and AMS radiocarbon dating)

The top 270 cm of the SA-2-102 core (samples from depths 60-100 cm were taken from the replicate core) were analyzed for ^{137}Cs, ^{210}Pb and ^{214}Pb at the Laboratory for Radioisotopes (ISOLAB) in Goettingen, Germany. The samples were sliced into increments of 10 cm, dried and ground. Samples were then packed in sealed plastic tins (ca. 35 cm^3) and let to rest for at least 3 weeks prior to the measurement to reach equilibrium between ^{226}Ra and ^{214}Bi (Goodbred and Kuehl, 1998). The gamma-ray measurements on ^{137}Cs (661.7 keV), ^{210}Pb (46.6 keV) and ^{214}Pb (295.2 and 351.9 keV) were performed on each sample using three low background Ge(Li) detectors for 250,000 s (2.9 days). Three bulk samples were selected from depths 270-500 cm of SA-2-102 core and measured for *Accelerator Mass Spectrometry (AMS) radiocarbon analysis* at the National Taiwan University (NTU) AMS-Laboratory, Taiwan (Table 4.1).

Table 4.1. AMS-^{14}C results of core SA-2-102. Outlier is in italic.

depth (cm)	Sample code	Material	^{14}C age	Calibrated age (2σ)	Posterior probability (outlier analysis*)
228	*SA102-3*	*Plant remains*	*F14C 1.005 ± 0.0001*		*100*
437	SA102-2	Plant remains	112 ± 1	1816-1922 AD	10
499	SA102-1	Bulk sediment (~2 cm^3)	385 ± 2	1480-1624 AD	15

* using SSimple model (Christen, 1994) in OxCal (Ramsey, 2009)

C. *Pollen analysis*

Plant communities are largely shaped by climate and environmental condition (Franklin et al., 2016). Thus, identifying past vegetation and how it shifted through time could reveal the changes of climatic and/or environmental conditions in the past (Nolan et al., 2018). Palynology or the analysis of pollen and spore analysis is widely used to reconstruct the past vegetation (Nolan et al., 2018).

To assess past vegetation composition in the SAL area and its hinterland, 30 subsamples of 2 cm^3 each were processed for pollen analysis following standard extraction methods (Faegri and Iversen, 1989). One tablet of *Lycopodium* spores was added to each subsample prior to the pollen extraction process. Pollen and spores were identified using the reference collection of pollen and spores of the Department of Palynology and Climate Dynamics, University of Goettingen, and other available literatures (e.g. Pollen and Spore Image Database of the University of Goettingen-available at http://gdvh.uni-goettingen.de/; Mildenhall and Brown, 1987; Mao et al., 2012; Li et al., 2012; Poliakova and Behling, 2016). Due to the poor pollen preservation, pollen and spores were counted up to a sum of 200 pollen grains for each sample. Pollen concentration was calculated as grains cm^{-3} based on the total grains of pollen counted, whereas spore concentration was calculated as grains cm^{-3} based on the total grains of pollen and spores counted. The pollen taxa were then classified into five groups according to the ecological characteristic or the function of the source plants (e.g. Sosef et al., 1998; Giesen et al., 2007; Ni et al., 2010; Wang et al., 2011; Mao et al., 2012). These groups are: (i) mangrove and mangrove associate (MMA) that represents the group of taxa that grow exclusively in mangrove areas and the associated plants; (ii) rainforest (RF) that represents the vegetation making up coastal, riparian, lowland and montane forest; (iii) open vegetation (OV) that represents the taxa that commonly grow in non-forested areas that never supported forest or resulted from disturbance such as forest clearing or conversion; (iv) staple agriculture (SA) that represents the taxa cultivated for staple food and the plants that grow on cultivated land; and (v) exotic taxa (EX) that represents introduced or non-native taxa.

D. XRF scanning and biogeochemical analysis

In order to trace changes in environmental conditions of the SAL, XRF scanning and analysis of total organic carbon (C_{org}), nitrogen (N) and the stable isotope composition of organic carbon ($\delta^{13}C_{org}$) were conducted. The XRF scanning of SA-2-102 core and the replicate core were performed at the Geomorphological-Sedimentological Laboratory of the Geomorphology and Polar Research (GEOPOLAR), University of Bremen. The sediment cores were transferred to the U-channel and scanned on the ITRAX (CS-8)-XRF scanner with Molybdenum-(Mo)-tube (Croudace et al., 2006). XRF scanning was conducted at 30 kV and 10 mA in 1 mm resolution with 10 s exposure time. XRF counts were normalized against the scattering coherent (coh) peaks of Mo (Hahn et al., 2014). The data were reduced by calculating average values for 1 cm intervals.

C_{org}, N and $\delta^{13}C_{org}$ of SA-2-102 core were determined by analyzing 45 subsamples of 2 cm^3 that were dried at 60°C and finely ground before analysis. C_{org}, and N were determined by high temperature oxidation in a Euro EA3000 elemental analyzer. The analytical precisions for C_{org} and N were ±0.03% and ±0.01%, respectively.

Samples for C_{org} determination were treated using 1N HCl prior to analysis to remove carbonates. A similarly treated sample was used for determination of $\delta^{13}C_{org}$ in a Thermo Finnigan Delta Plus gas isotope ratio mass spectrometer after high temperature combustion in a Flash 1112 EA elemental analyzer. The $\delta^{13}C_{org}$ is reported as per mil (‰) deviation from the carbon isotope composition of the Vienna Pee Dee Belemnite (VPDB) and the nitrogen isotope composition of atmospheric air, respectively, with ±0.1‰ uncertainty.

E. C accumulation rate

The C accumulation rate of SA-2-102 was calculated by multiplying BD with total organic carbon (C_{org}) and sediment accumulation rates produced by the age-depth model (Section IV.A). The C accumulation rate is displayed as g C m^{-2} yr^{-1}. The sediment bulk density (BD) itself was calculated as the dry weight (g) divided by the wet sample volume (cm^3). Subsamples of 2 cm^3 were taken along the core at 5 cm intervals and subsequently dried at 105°C for 24 h to calculate the sediment dry weight.

4.4. Results

A. Lithology and age control

Both SA-2-102 and the replicate core consist mainly of silt and clay material without a clear indication of changing sediment composition (Fig. 4.2). The unsupported ^{210}Pb activity of the core shows an irregular decrease that suggests a non-constant sedimentation rate over time (Fig. 4.2). Considering such circumstances, the use of a constant initial concentration (CIC) model for age calculation is not appropriate. The CIC model can only be applied when the activity of unsupported ^{210}Pb declines exponentially with depth (Applebly and Oldfield, 1983;

Appleby, 2008). Therefore, a constant rate of supply (CRS) model was chosen to calculate the age (Applebly and Oldfield, 1978; 1983).

The CRS model assumes that the atmospheric deposition of excess ^{210}Pb is constant over time and applicable for a sediment core with a non-monotonic ^{210}Pb profile (Applebly and Oldfield, 1978; 1983; Appleby, 2008). Moreover, CRS is widely used and considered to be more suitable to calculate ^{210}Pb dates in lakes, coastal zones or estuaries where sedimentation processes are highly influenced by anthropogenic activities (Appleby and Oldfield, 1978; Lubis, 2006). An independent tracer such as ^{137}Cs is often used to confirm the ^{210}Pb dates of the CRS model. However, in our case the ^{137}Cs values from both SA-2-102 and the replicate core were constantly below the detection limit.

The calendar dates produced from the CRS model (Table 4.2) are subsequently integrated with AMS ^{14}C dates to construct an age-depth model for the whole SA-2-102 core. The age-depth model was constructed using the P-sequence depositional model in OxCal v.4.3.1 (Ramsey, 2009) with a k value of 1. Prior to the construction of the age-depth model, an outlier detection was conducted using the SSimple model (Christen, 1994) in OxCal. Sample SA102-3 is flagged as an outlier by its high posterior probability and is therefore excluded from the age-depth model (Table 4.1). The discrepancy of sample SA102-3 might be attributed to bioturbation which possibly moved the sample from its correct stratigraphic position. The dates produced from this analysis are expressed as anno Domini/before Christ (AD/BC) and applied throughout the paper. The age-depth model indicates that core SA-2-102 spans a time period of around 400 years from 1620 to 2014 (Fig. 4.2).

Table 4.2. ^{210}Pb dates of Segara Anakan core based on CRS model.

depth (cm)	unsupported ^{210}Pb activity (Bq kg^{-1})	uncertainty (Bq kg^{-1})	Calendar yr (AD)	Posterior probability (outlier analysis*)
0	13.7	3.0	2014 ± 3	10
10	9.0	3.0	2011 ± 3	10
20	7.6	2.9	2009 ± 3	10
30	5.5	2.9	2007 ± 3	10
40	13.2	3.0	2006 ± 3	10
50	11.5	2.9	2002 ± 3	10
60	7.4	2.9	1999 ± 4	10
70	8.7	3.0	1996 ± 4	10
80	9.1	2.8	1994 ± 4	10
90	18.4	3.0	1991 ± 4	10
100	12.8	2.9	1984 ± 4	10
120	7.4	2.9	1977 ± 5	10
110	8.8	2.9	1970 ± 5	10
130	13.0	2.9	1964 ± 5	10
140	9.5	2.9	1948 ± 7	10
150	10.9	2.9	1929 ± 9	10
160	0.7	2.6		
170	4.5	2.8		
180	2.6	2.7		
190	4.0	2.7		
200	0.4	2.5		
210	-0.9	2.5		
220	1.5	2.5		
230	2.0	2.5		
240	1.2	2.5		
250	1.5	2.6		
260	2.8	2.5		

* using SSimple model (Christen, 1994) in OxCal (Ramsey, 2009)

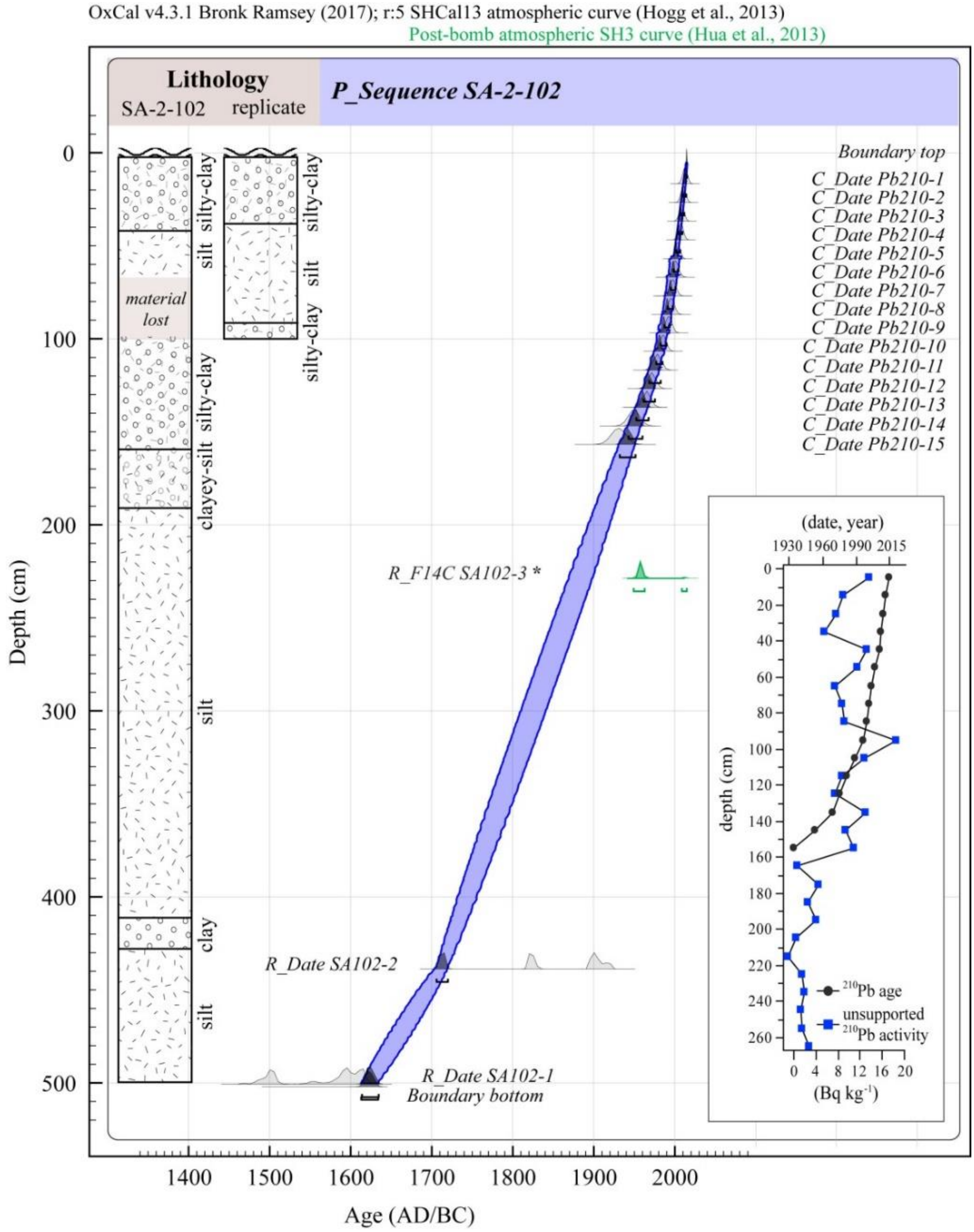


Fig. 4.2. Lithology and age-depth model of cores taken from the Segara Anakan Lagoon. The age-depth model was constructed using the P-sequence in OxCal v 4.3.1 (Ramsey, 2009) and the initial ShCal.13 and PostBomb calibration curves (Hogg et al., 2013; Hua et al., 2013). The unsupported ^{210}Pb activity and ^{210}Pb dates are presented in the insert. An outlier date is marked with an asterisk (*).

B. Pollen analysis, XRF and biogeochemical analysis

The detailed results of pollen, XRF and biogeochemical analyses are comprised in Table 4.3. The pollen record of SAL is divided into five palynological zones with two subdivisions in zone II (Fig. 4.3) based on the agglomeration in a constrained cluster analysis (CONISS; Supplementary S2; Grimm, 1987). The same zonation is used to divide the results of XRF scanning and biogeochemical analyses, and also the calculation of C accumulation rates.

Zones I, II and IV are dominated by the MMA (mangroves and mangrove associates) pollen group, while zones III and V are dominated by SA (staple agriculture). The proportion of pollen group RF (rainforest) and OV (open vegetation) are relatively stable throughout the record with a higher RF proportion in zone IV, and a lower OV proportion in zones I and IV. The pollen group EX (exotic species) contributes only a small proportion throughout the record.

The results of XRF scanning are shown as element count per second (cps) at each measuring point. Selected elements, sulfur (S), chlorine (Cl), bromine (Br), titanium (Ti) and calcium (Ca) are reported in the XRF profile (Fig. 4.4). Those elements show notable increase in Zone IV, except for Ti that shows a slight decrease.

The values of N are relatively stable throughout the record (0.11%), while the C_{org} values fluctuate from 0.9 to 2.9% with higher values in zone IV (Fig. 4.4). Consequently, the C_{org}/N ratio is also higher in zone IV. The $\delta^{13}C_{org}$ values fluctuate from -22 to -28‰ (average -26‰) throughout the record, with lower values in zones IIa and IV (Fig. 4.4).

Table 4.3. Results of pollen analysis, XRF scanning and biogeochemical analysis. Values presented are in average.

Zonation (depth and age)	*Pollen analyses* (Fig. 4.3)*	*XRF profile (Fig. 4.4)*	*Biogeochemical analysis (Fig. 4.4)*
SA-I 500-486 cm 1620-1640	56% MMA (e.g. *Rhizophora*, *Nypa*, *Bruguiera* and *Avicennia*); 5% SA (e.g. *Oryza*-type, Colocasia and Solanaceae); 6 % OV (e.g. wild Poaceae and Asteraceae); 33% RF (e.g. Moraceae-Urticaceae, Arecaceae, *Nauclea*, *Pometia* and *Podocarpus*); 0% EX (*Pinus*)	S, Cl and Br (exponentially decrease towards present time) Ti/Ca (exponentially increases towards present time)	C_{org} (1.6%); N (0.11%); C_{org}N (12); $\delta^{13}C_{org}$ (-26.7‰)
SA-IIa 435-486 cm 1640-1700	50% MMA (▼); 7% SA (▲); 18 % OV (▲); 25% RF (▼); 0% EX (↔)		
SA-IIb 335-435 cm 1700-1790	45% MMA (▼); 12% SA (▲); 18 % OV (↔); 25% RF (↔); 0% EX (↔)		C_{org} (1.5%;▼); N (0.11%; ↔); C_{org}N (10; ▼); $\delta^{13}C_{org}$ (-25.3‰; ▲)
SA-III 295-335 cm 1790-1830	18% MMA (▼); 35% SA (▲); 24 % OV (▲); 19% RF (▼); 4% EX (▲)		C_{org} (1%;▼); N (0.11%; ↔); C_{org}N (7; ▼); $\delta^{13}C_{org}$ (-25.7‰; ▼)
SA-IV 295-190 cm 1830-1910	49% MMA (▲); 5% SA (▼); 11 % OV (▼); 35% RF (▲); 0% EX (▼)	S, Cl and Br (▲) Ti/Ca (▼)	C_{org} (2.1%;▲); N (0.11%; ↔); C_{org}N (16; ▲); $\delta^{13}C_{org}$ (-27.7‰; ▼)
SA-V 190-0 cm 1910-2014	24% MMA (▼); 27% SA (▲); 20 % OV (▲); 23% RF (▼); 6% EX (▲)	S, Cl and Br (▼) Ti/Ca (▲)	C_{org} (1.2%;▼); N (0.11%; ↔); C_{org}N (9; ▼); $\delta^{13}C_{org}$ (-25.7‰; ▲)

▲: value increase ▼: value decrease ↔: value stable

*: pollen group abbreviation (see Section III.C.)

C. C accumulation rate

The BD values are relatively stable throughout the SA-2-102 core ranging from 0.5 to 0.9 g cm^{-3} (average 0.7 g cm^{-3}; Fig. 4.5). The average rate of sediment accumulation over the entire period is 1.6 cm yr^{-1} (range 0.7 to 5 cm yr^{-1}) with a minimum in Zone I and IIa (0.7 cm yr^{-1}) and a maximum in Zone V (2.2 cm yr^{-1}; Fig. 4.5). The temporal variation of the C accumulation rate generally mirrors the trend of the sediment accumulation rate (average 126 g C m^{-2} yr^{-1}; range 55-370 g C m^{-2} yr^{-1}), except for Zone IV (Fig. 4.5).

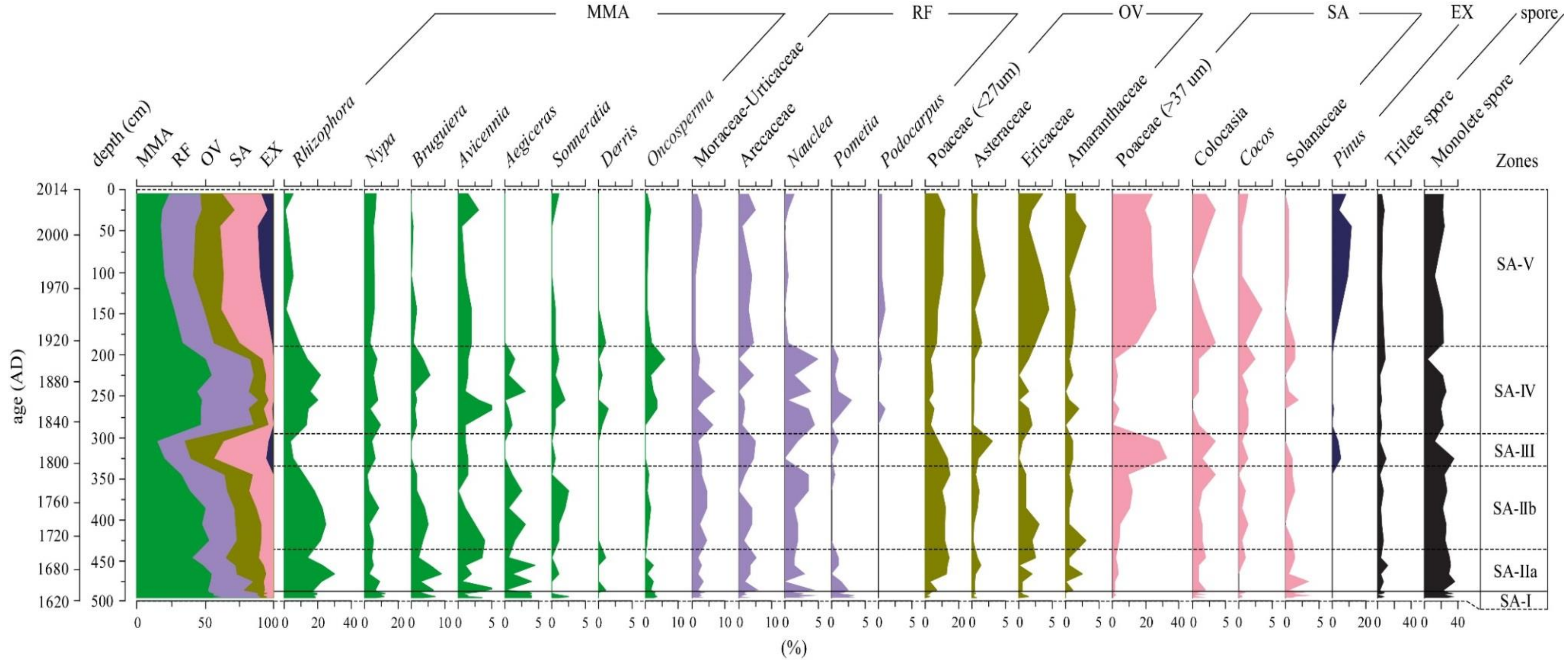


Fig. 4.3. Pollen diagram of the Segara Anakan Lagoon. The abbreviations of the group stand for MMA: mangrove and mangrove associate; RF: rainforest; OV: open vegetation; SA: staple agriculture; and EX: exotic species. Selected taxa are shown. Complete list of taxa is provided in Supplementary 1.

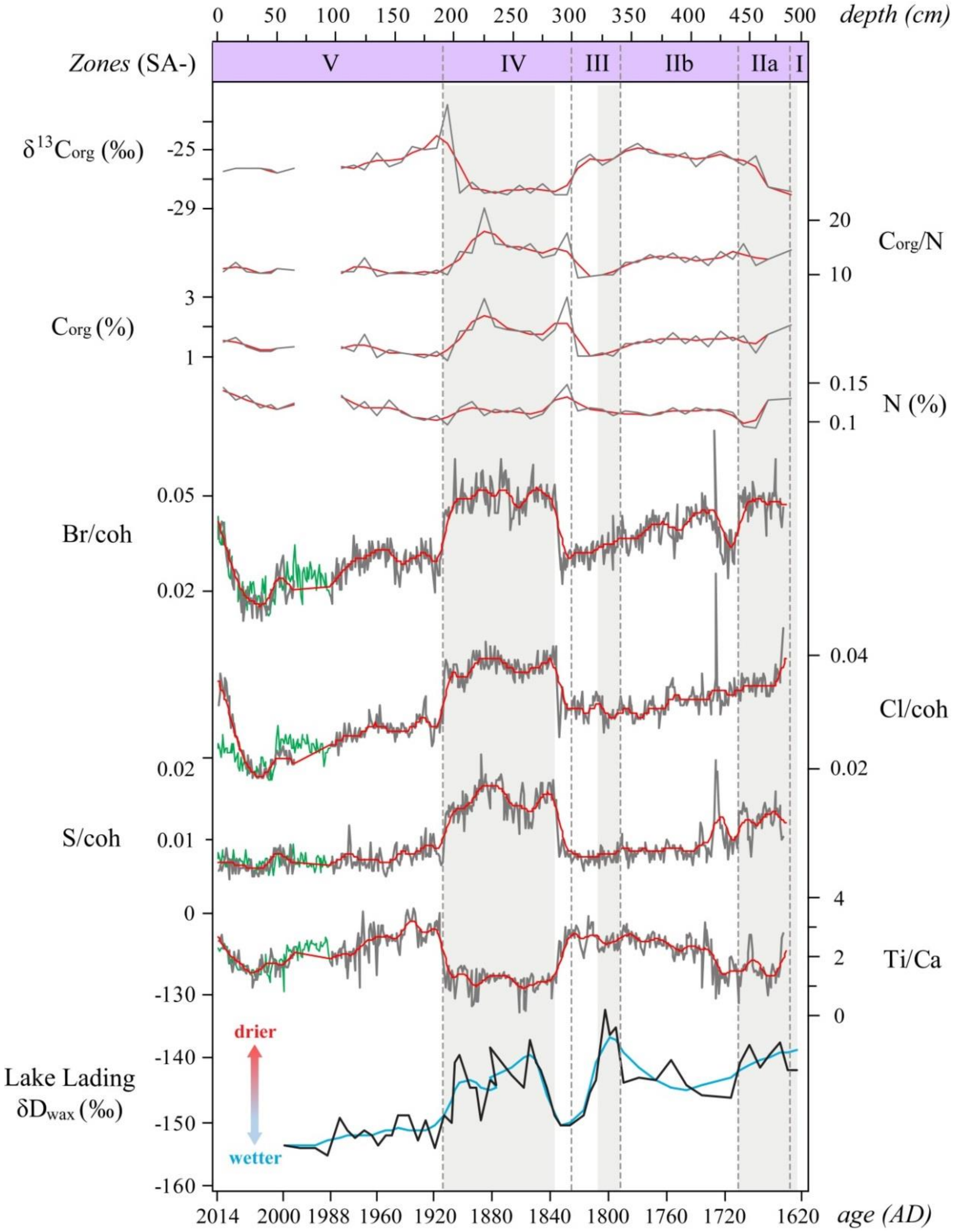


Fig. 4.4. Combined diagram of selected elements (Ti/Ca, Br, S, Cl; green lines represent results from the replicate core) and biogeochemistry (C_{org}, N, C_{org}/N and $\delta^{13}C_{org}$), in comparison with precipitation of Lake Lading (Java) derived from δD_{wax} (Konecky et al., 2003; the raw δD_{wax} data are denoted by the black lines while the low-pass filter data are presented in blue). The reported XRF elements are normalized with coherent (coh) peaks. The periods of lower precipitation are highlighted by the

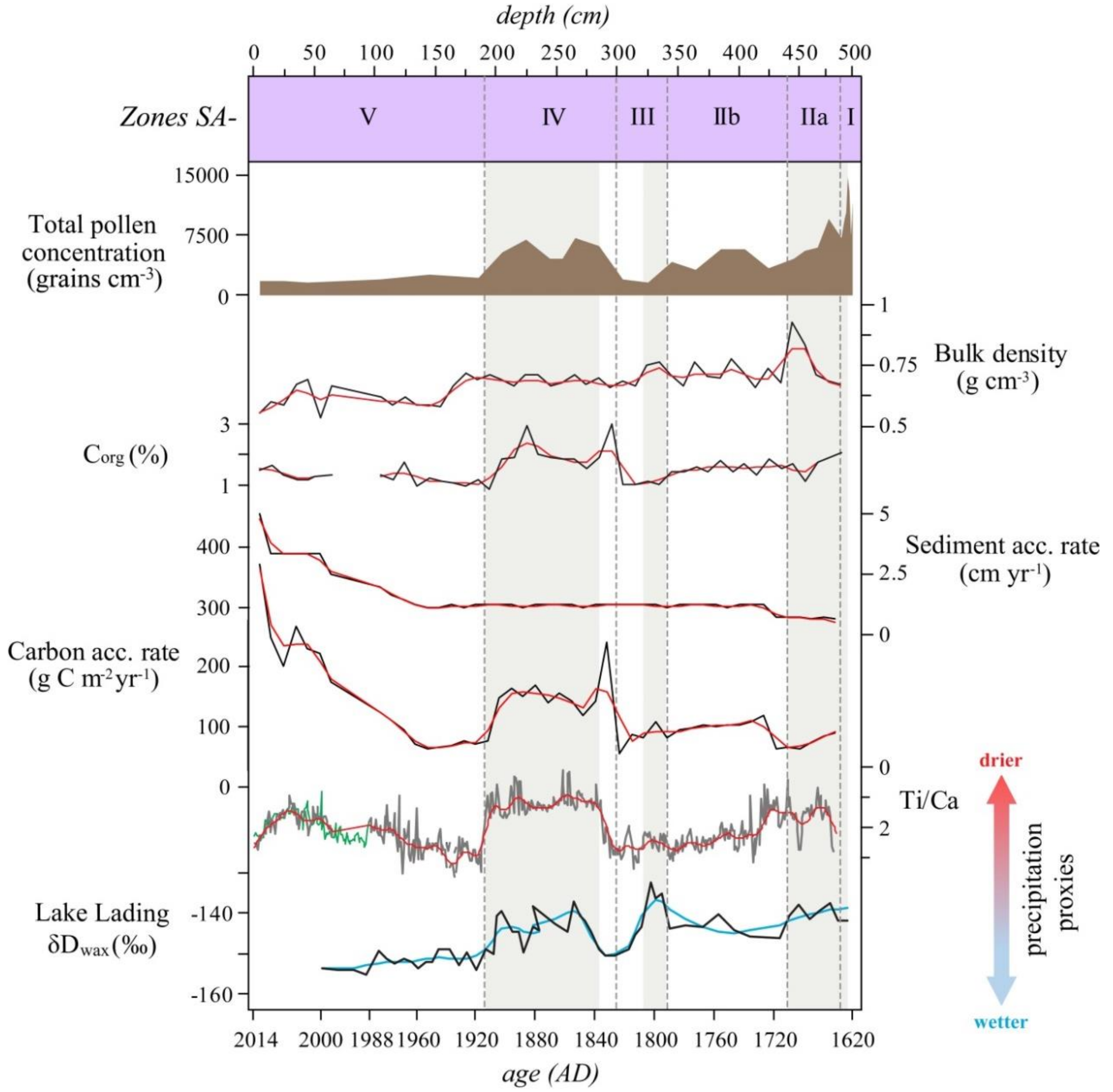


Fig. 4.5. Sediment accumulation rate, bulk density, C$_{org}$ content and C accumulation rate of SAL in comparison with precipitation proxies of Ti/Ca (SAL) and δD$_{wax}$ (Lake Lading, Java; Konecky et al., 2003). Raw δD$_{wax}$ data are denoted by the black lines while the low-pass filter data are presented in blue. The periods of lower precipitation are highlighted by the grey bars.

4.5. Discussion

4.5.1. Segara Anakan lagoon dynamics

1. The early period 1620-1830 AD

The sediment accumulation rate and the Ti/Ca ratio were relatively low from 1620 to 1700 (Fig. 4.4). During the same period, the values of Br, S and Cl were considerably higher than in the following. Concomitantly, around the same time interval, between 1600 and 1690, a period of decreasing precipitation was reported in Java (Konecky et al., 2013).

A low Ti/Ca ratio indicates a low detrital input from rivers or runoff (Mohtadi et al., 2011; Steinke et al., 2014). Meanwhile, the higher values of Br, S and Cl suggest a higher marine input into the lagoon (Burton et al, 2000; Haenssler et al., 2014). The lower precipitation from 1600 to 1690 likely played a role in reducing sediment transport from the catchment area by limiting surface runoff and freshwater discharge from the river. Consequently, the sediment accumulation rate was low during this period and lagoon salinity was higher.

According to the pollen abundance and composition, from 1620 to around 1700 the vegetation surrounding SAL was dominated by mangrove forest, with the dominant species being *Rhizophora, Bruguiera, Avicennia, Aegiceras* and *Nypa*, with some rainforest vegetation (Fig. 4.3, Supplementary 3). The domination of mangrove ($\delta^{13}C_{org}$ range -28 to -25‰; Rodelli et al., 1984) and rainforest ($\delta^{13}C_{org}$ range -38 to -26‰; Ometto et al., 2006) vegetation is also confirmed by an average $\delta^{13}C_{org}$ of -26.7‰. A rather 'natural' environmental condition as suggested by the domination of mangrove and rainforest vegetation, likely minimized erosion in the hinterland and hence limited the sediment input into SAL during this period.

The sedimentation rate doubled after 1700. This was accompanied by an increase of the Ti/Ca ratio and decreasing values of Br, S, and Cl. It indicates a higher sediment input from the hinterland as a result of a generally increasing precipitation in Java after 1690 (Konecky et al., 2013). This may also have reduced the intrusion of sea water into the lagoon and kept lagoon salinity low. Rainfall intensification that decreased salinity is also mentioned in the record of pearl fishing in SAL. It was reported that an ‘unseasonal rainfall killed the young oyster’ in 1722 (Schwerdtner Máñez, 2010 and references therein).

Around 1700 and thereafter, the proportion of the wild Poaceae, Ericaceae and Amaranthaceae as well as the cultivated Poaceae pollen increased (e.g. *Oryza, Sorghum, Saccharum, Panicum, Setaria*) followed by a decreasing proportion of mangrove pollen (Fig. 4.3; Table 4.3). These changes are corroborated by similar trends in the pollen concentration of the aforementioned taxa (Supplementary 3) and a higher $\delta^{13}C_{org}$ value, which indicates an admixture of C4 plants to the deposited organic matter (OM; Meyers, 1994; França et al., 2013; Fig. 4.4).

Considering that SAL was already inhabited by "Orang Laut" at that time (Section II.B), the decline in mangrove forest in this period was likely related to human mangrove forest utilization. The residents of Kampung Laut utilized mangrove wood mainly for house construction (Waryono, 2002; Mulyadi, 2013). The increase of cultivated Poaceae in the pollen record throughout the 18th century (Fig. 4.3; Table 4.3) likely reflects the onset of larger-scale anthropogenic land use changes in the lagoon’s catchment area (Section II.B).

The first occurrence of pine pollen (*Pinus* sp.) in the sediment record in the early 19th century (Fig. 4.3) suggests that it was introduced to Java earlier than previously thought. It was reported that pine was introduced from Sumatra in the frame of small scale trials in the 1920s and was then planted on a larger scale in the 1930s (Becking, 1935; Fitriani, 2012). However, pine could have been introduced earlier to Java as forest species or the pine pollen are transported by wind into the Segara Anakan catchment area and subsequently transported along the river into the lagoon, considering that the bisaccate morphology of *Pinus* pollen allows for long-distance wind-transport (ca. 300 km; Reitz and Shackley, 2012; Twiddle et al., 2012).

2. The 1830-1910 drought period

A period of drier climatic conditions from 1830 to 1910 (Konecky et al., 2013) is reflected in several records of drought-related crop failures in Java in 1832-1836, 1844-1848, 1849-51, 1858-62, 1877-88 and 1900-02 (Creutzberg and van Laanen, 1987; Boomgaard, 2002; Fernando, 2010; van der Eng, 2010). As suggested by a lower Ti/Ca ratio and higher values of Br, Cl and S (Fig. 4.4), the 1830-1910 droughts in Java reduced freshwater input into the SAL, hence lagoon salinity increased. During this period, the $\delta^{13}C_{org}$ value decreased to an average of -27.7‰ and the C_{org}/N ratio increased to 16, suggesting less admixture of OM from non-woody vegetation (Meyers, 1994). This is corroborated by a higher proportion of mangrove (e.g. *Rhizophora*, *Bruguiera* and *Avicennia*) and rainforest (e.g. *Nauclea*, Moraceae-Urticaceae and *Pometia*) pollen and a lower proportion of cultivated Poaceae and *Pinus* pollen (Fig. 4.3). Similar trends in their absolute terms, i.e. pollen concentration, were observed (Supplementary 3). In fact, during the 1830-1910 drought period, the total pollen and almost all pollen displayed a higher concentration, except for wild-grass Poaceae and cultivated Poaceae (>37 um), suggesting they were likely transported by a river (Fig. 4.5 and Supplementary 3).

Under the common assumption that the pollen-spore assemblage in the sediment reflects the surrounding vegetation at the time of their deposition (Bradley, 1999), such changes in the pollen record of Segara Anakan indicate the expansion of mangrove and rainforest surrounding the lagoon. However, in interpreting pollen assemblages attention needs to be paid to taphonomy, the process transforming plant communities into pollen assemblages, for example by pollen deposition (Goring et al., 2013). In a water body, the deposition of pollen grains commonly relies upon two factors: its settling velocity and water movement (Brush and Brush, 1972). Those factors strongly control the pollen suspension time in the water column (Brush and Brush, 1972).

Lower river discharge during the 1830-1910 drought period likely increased pollen deposition by reducing water flow velocity, hence allowing the suspended pollen grains to settle to the lagoon floor. In an estuarine environment, the pollen deposition rate is also influenced by encapsulation in floccules of clay, silt and organic particles (e.g. amorphous OM, phytoclasts and palynomorphs), the formation of which (=flocculation) largely depends on the salt concentration (Chmura and Eisma, 1995; Gastaldo and Feng, 1996; Gastaldo, 2012). Higher salt contents cause fine particles to stick together and to form larger and heavier aggregates, which thus settle more rapidly (Sutherland et al., 2014).

In addition to that, lower river discharge resulted in lower sediment input into the lagoon hence reduced the effect of sediment dilution (Fig. 4.5). Commonly, in an ecosystem where pollen mainly originate from local vegetation, which is the case in the SAL, sediment accumulation is inversely reflected in total pollen concentration (Brush, 1989). This means that pollen concentration will increase whenever mineral input decreases, and vice versa (Brush, 1989). The higher proportion and concentration of mangrove and rainforest pollen during the 1830-1910 drought period are therefore likely influenced by the changes in the pollen deposition mechanisms and the effect of sediment dilution, rather than by the expansion of the plant source itself.

3. The last century (1910 until present)

After 1910, the pollen record indicates a decline of mangrove forest and most of the rainforest taxa (Fig. 4.3). The pollen record also indicates an increase of herbaceous plants, such as wild Poaceae and Ericaceae, and of coconut trees (*Cocos*; Fig. 4.3) which are commonly cultivated throughout the lagoon's catchment area. These changes are confirmed by similar trends in their absolute terms (Supplementary 3), a higher $\delta^{13}C_{org}$ indicating the addition of C4 plant-derived OM, and a lower CN ratio that suggests the decline of vascular plant OM input (Meyers, 1994; França et al., 2013; Fig. 4.4).

Despite being likely exaggerated by the dilution effect following a higher sediment input, the decline of mangrove taxa in the pollen record conforms to documentation of mangrove uses and land conversions. The drastic decline of *Bruguiera* in the pollen record around the turn to the 20th century is in line with a forest inventory by de Haan (1931), who found that the *Bruguiera* stands were particularly exploited and degraded, with only very limited regrowth. While exploitation of *Rhizophora* first remained limited to 100-200 meter corridors along the creek shores (de Haan, 1931), our pollen record indicates a massive decline of the *Rhizophora* stands until the 1940s. In addition to the wood demand of the colonial industry, local uses of mangrove wood and the expansion of agricultural land by the growing population also contributed to the decline of mangrove cover.

The pollen record also suggests an increase of cultivated Poaceae and the exotic taxon *Pinus* after 1910 (Fig. 4.3). The increase of cultivated Poaceae in our pollen record likely reflects the increase in rice cultivation in the catchment area of the SAL since the early 20th century. Meanwhile, the increase of *Pinus* pollen reflects the expansion of pine plantations in the hinterland. Although teak (*Tectona grandis*) was and is also widely planted in Java, its pollen is not captured by the pollen record. In contrast to *Pinus* spp. that produces a high amount of pollen, which can be transported over long distances (Twiddle et al., 2012), *Tectona* spp. is known to produce a high amount of pollen, but its pollen are poorly preserved and have a low dispersal efficiency (Quamar and Bera, 2014).

The sediment core indicates rapidly increasing sedimentation after 1960 (Fig. 4.2 and 4.5). This is roughly in line with the results of a historical-cartographic and remote sensing analysis of lagoon shore line changes (Lukas, 2014a; 2017), which shows continuously increasing sedimentation between the first half of the 20th century and the 1990s – with aggradation progressing from the northern to the southern parts of the lagoon, where the sediment core was taken. The causes of increasing sedimentation in the second half of the 20th century include the expansion of rainfed agriculture, erosion on contested state forest and plantation lands, erosion on roads trails and in expanding settlement areas, *ngaguguntur* (the digging back of hill slopes), agriculture in riparian zones, and river and floodplain modifications, including the removal of river meanders (Lukas, 2017). Meander removal and river straightening artificially shortens the water flow distance and steepens the slope (Sapkota, 2017). This increases water velocity and enhances streambank and bed erosion, thus contributing to increased sediment input into the lagoon. Furthermore, a floodway was constructed to divert the peak floods of the Citanduy into the Cibeureum River and to redirect part of the high sediment load into the northern part of the lagoon, where much of it is deposited, instead of being transported to the ocean (Lukas, 2017).

4.5.2. Carbon sequestration in the Segara Anakan Lagoon

1. Temporal variation in C accumulation rates

The temporal variation in the C accumulation greatly resembles the variation in the sediment accumulation rates (Fig. 4.5). An exception occurs during the 1830-1910 drought period when the C_{org} content was higher (Fig. 4.5). Sediment input indeed contributes to the preservation and burial of C_{org} (Schwarzbauer and Jovančićević, 2015). However, in case of a very high sediment accumulation, as in the SAL, C_{org} can be diluted by the inorganic clastic material, i.e. the C_{org} concentration may decrease with an increasing sediment supply (Tyson, 2001). Such "dilution" only reduces the C_{org} concentration in the sediment and not its total amount (Schwarzbauer and Jovančićević, 2015). Thus, in turn, a lower sediment input would reduce the dilution effect and result in higher C_{org} concentrations as observed in the SAL during the 1830-1910 drought period (Fig. 4.5).

Salinity-induced flocculation could also play a role in increasing the C_{org} content in the SAL during the 1830-1910 drought period. Organic particles, like silt and clay, possess a negative charge and consequently repel each other, which prevents them from aggregating (Roberts et al., 2006). Increased lagoon salinity adds to the cationic charge that limits or eliminates the repulsion effect (Shamlou, 1993; Somasundaran, 2006). This allows the organic particles in SAL, which based on the $\delta^{13}C_{org}$ values sourced mainly from mangrove (Fig. 4.4; Meyers, 1994; França et al., 2013), to undergo flocculation along with silt and clay particles.

This, in turn, suggests that changes in salinity might play an important role in determining variations in the CAR of the SAL. Salinity-induced flocculation was also suggested to be an important mechanism in soil C accumulation in mangrove ecosystems by a recent experimental study (Kida et al., 2017). It was observed that humic substances exist in floccules that were formed due to the supply of sea water into the ecosystem. The reduction of salinity in a mangrove ecosystem would then lead to a breakdown of floccules, hence the release of humic substances from flocculation (Kida et al., 2017).

2. Significance of long-term C sequestration rates of Segara Anakan lagoon

Estimating the ecosystem's C accumulation capacity by measuring short-term C sequestration using surface sediment (≤10 years; ±5-cm depth) would lead to an overestimation (Donato et al., 2011; Breithaupt et al., 2014). This is primarily due to the fact that the surface sediment is vulnerable to remineralization and erosion (Breithaupt et al., 2014). On the other hand, using longer-term CAR measurements, e.g. a 50- to 100-year timeframes, would suppress the bias of temporal scale and reduce the spatial variability (Breithaupt et al., 2014). Also, a 100-year timespan is considered to "provide the most conservative forecast of the regional long-term rates" for a longer-term C sequestration assessment in coastal wetlands (Breithaupt et al., 2014).

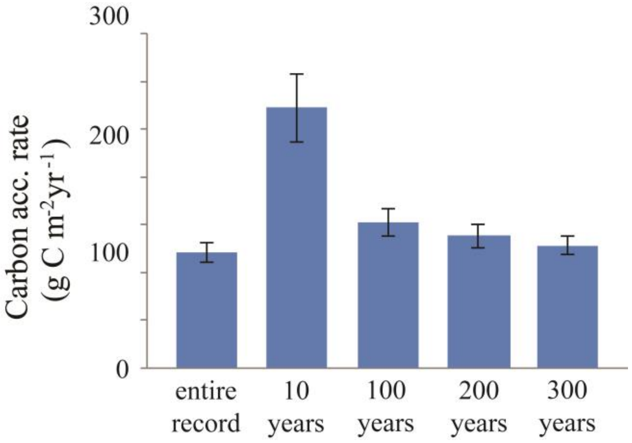


Fig. 4.6. Average C accumulation for SAL in different time slices.

In SAL, the 100-year average CAR is similar to its average CAR for the entire measurement period (400 years; Fig. 4.6). It indicates that a 100-year timeframe is appropriate to represent the CAR, particularly of lagoon ecosystems, on a longer time scale. Here, the 100-year CAR sequence is used to compare the SAL's C sequestration capacity to records from other mangrove-dominated lagoons that mostly cover the last 100 years (Fig. 4.7a).

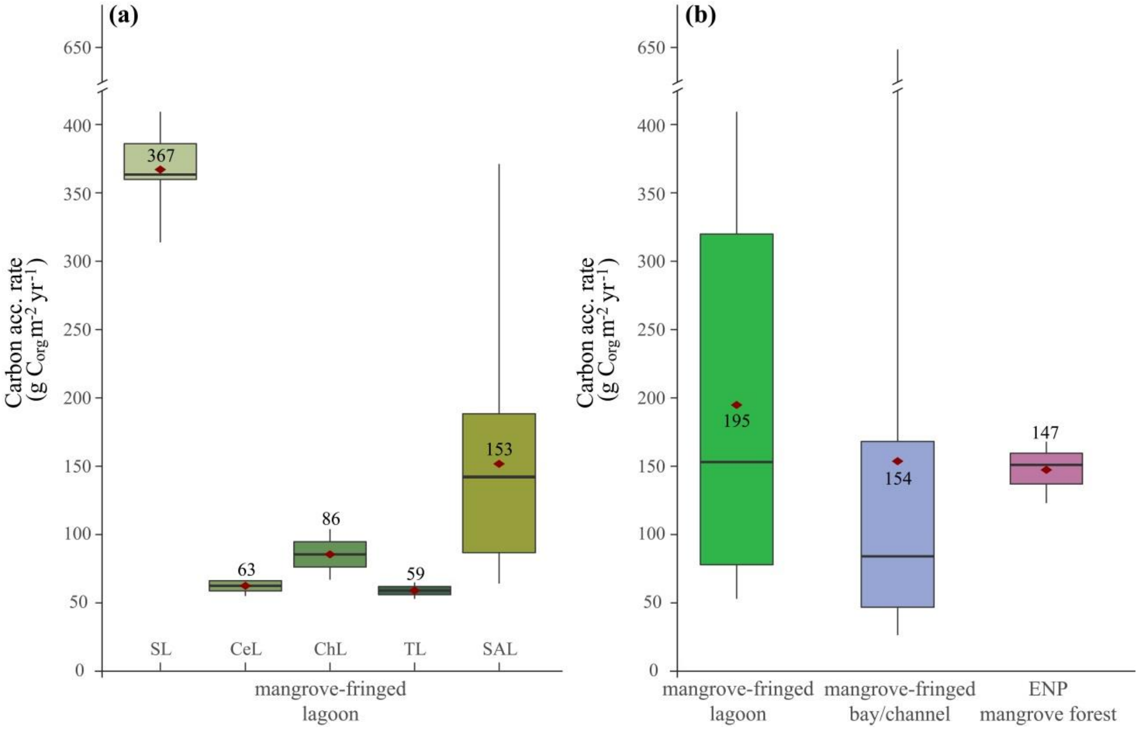


Fig. 4.7. Boxplot comparison of C accumulation rates during the last ca. 100 years of: a. mangrove-dominated lagoons; and b. mangrove-dominated coastal ecosystems (datasets included in the analysis are provided in Table 4.4). Red diamond symbols represent the average values. Site abbreviations: SL-Soledad Lagoon (Ruiz-Fernández et al., 2011), CeL-Celestun Lagoon, ChL-Chelem Lagoon, TL-Términos Lagoon (Gonneea et al., 2004), SAL-Segara Anakan Lagoon, ENP-Everglades National Park (Smoak et al., 2013; Breithaupt et al., 2014).

Compared to other mangrove-fringed lagoons such as the Celestun (CeL), Chelem (ChL) and Términos (TL) on the Yucatan Peninsula in Mexico, SAL accumulated 2-3 times more carbon during the last century (Fig. 4.7a). Owing to the karstic geological condition, lagoons on the Yucatan Peninsula receive little sediment through surface runoff or river input (Gonneea et al., 2004) and hence sedimentation rates and C accumulation rates are fairly low. Soledad Lagoon (SL) in Colombia, on the other hand, accumulated over two times more carbon than SAL during the last century (Fig. 4.7a). Based on its geomorphology, SL can be defined as a choked lagoon (Kjerfve, 1986) and it received river water input only occasionally (Ruiz-Fernández et al., 2011). SL has a long water residence time and restricted tidal or river energy influence. This can result in rapid biomass accumulation as well as in limited sediment resuspension and C export through tidal or river flushing (Kennish and Paerl, 2010).

The comparison also suggests that, in general, river-fed mangrove-fringed lagoons accumulate C more effectively than, for example, the mangrove forest of the Everglades National Park (ENP; Supplementary 4; Smoak et al., 2013; Breithaupt et al., 2014) and other mangrove-fringed ecosystems with little or no river input such as bays or channels during the last century (Fig. 4.7b). Sediment is transported to the coastal area mainly by river discharge (Wolanski, 1994). In a mangrove forest that is not under the influence of river input, sediment particles are distributed at high tide when the water spills over and inundates the forest (Kathiresan, 2003), in contrast to direct sediment delivery into lagoons, bays or channels. This explains the lower sediment supply and hence lower C accumulation in such a mangrove forest compared to other coastal ecosystems. Meanwhile, although channels and bays also receive high allochthonous sediment input from rivers and runoff, their geomorphological settings tend to be more exposed and less protected than lagoons (Brunskill et al., 2002). This makes them more prone to sediment redeposition due to wind/tidal energy and/or C export through tidal flushing (Brunskill et al., 2002), thus they accumulate C less effectively than lagoon ecosystems.

Table 4.4. Datasets included in the comparison (Fig. 4.7b)

No.	Sites			CAR (g C m^{-2} yr^{-1})	Data source
1	Indonesia	mangrove-fringed lagoons	Segara Anakan lagoon, Java	153	*this study*
2	Mexico		Celestun lagoon	55 70	Gonneea et al., 2004
3			Chelem lagoon	67 104	
4			Terminos lagoon	53 65	
5	Colombia		Soledad lagoon	367	Ruiz-Fernández et al., 2011
6	Australia	mangrove-fringed bays/channels	Missionary Bay	70.9 97.3 32.4 26.4	Brunksill et al., 2002
7			Hinchinbrook Channel	26.4 168.1 84.1 336.3 300.3 99.7 648.6 62.5 46.8	
8	Florida, US	mangrove forest	Everglades National Park	151 168	Smoak et al., 2013
				123	Breithaupt et al., 2014

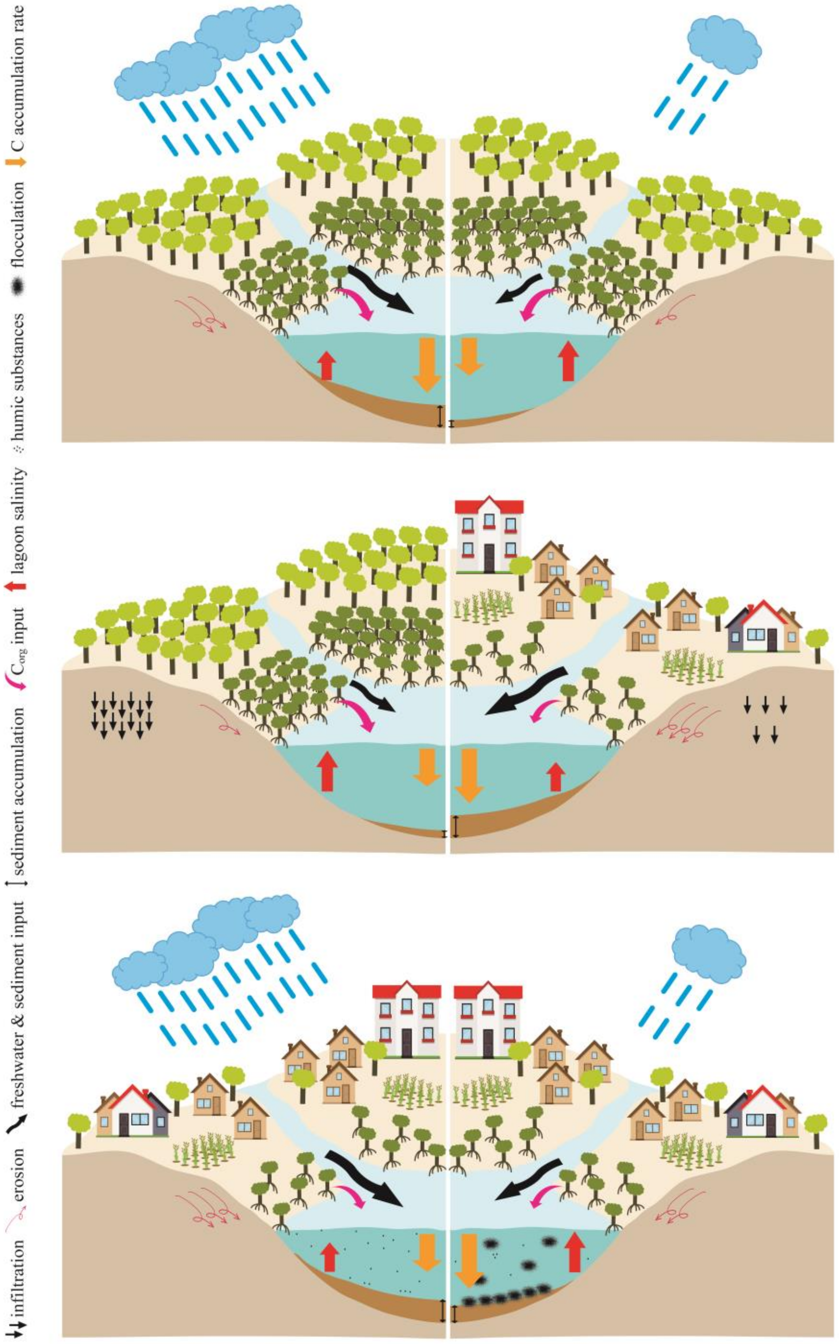
4.6. Conclusion

The Segara Anakan Lagoon's capacity to accumulate C, as its environmental condition, is highly dynamic and mainly governed by climatic conditions and human-induced landscape modification (Fig. 4.8). According to the palaeoecological information, during a drier climate in the past the CAR of SAL increased possibly due to reduced sediment dilution and accelerated OM accumulation through salt-induced flocculation. As a consequence, the predicted decrease of precipitation over Java in the future (Cruz et al., 2007; Qalbi et al., 2017) will likely increase the CAR of SAL as happened in the past.

Land use changes in the catchment and river modifications contributed to the increasing CAR in SAL through higher preservation rates due to increased sediment input into the ecosystem, although it also reduces the lagoon's size. If the trend of sediment input into the lagoon continues, the CAR of SAL will likely increase in the future. However, continuous logging of the surrounding mangrove forest, which is a major source of the sequestered C, could result in a reduced CAR.

If deforestation and degradation of the mangrove forest in SAL continue, its C sink function will diminish. Thus, the coverage area and condition of the surrounding mangrove forest needs to be maintained or improved in order to sustain the C sequestration function of SAL. This is especially important due to the fact that, on a global scale, SAL is among the most effective mangrove-dominated lagoons in terms of C sequestration (Fig. 4.7a).

Meanwhile, among mangrove-dominated ecosystems, lagoons sequester C most effectively because they tend to receive a higher sediment input than other coastal ecosystems and are geomorphologically more protected (Fig. 4.7b). Following the urgency for the removal of atmospheric CO_2 through natural sequestration, mangrove-dominated lagoons are thus quantitatively important natural carbon sinks that are relevant in terms of global climate change mitigation. These ecosystems need to be prioritized for conservation and restoration.



8a. Climate (precipitation)

Under natural condition, reduction of precipitation reduced C accumulation rate of SAL

8b. Land use change

Land use change increased C accumulation rate of SAL due to higher preservation rate following enhanced sediment input

8c. Climate and land use change

Reduced precipitation on converted SAL accelerated its C accumulation rate through salt-induced OM flocculation and reduced sediment dilution effect

Fig. 4.8. Visualization of how climate and land use change governed the dynamics of environment and C accumulation in SAL during the last 400 years. The house vector arts were downloaded from www.freepik.com

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Supplementary 1

Complete list of plant taxa identified from pollen and spore analyses

Mangrove and mangrove associate (MMA)

Plant family	***Pollen taxon***	**Plant family**	***Pollen taxon***
Acanthaceae	*Acanthus*	Lythraceae	*Sonneratia alba*
	Avicennia		*S. caseolaris*
Apocynaceae	*Cerbera*		*S. apetala*
Arecaceae	*Nypa*		*S. ovata*
	Oncospermae	Meliaceae	*Xylocarpus*
Aquifoliaceae	*Ilex*	Mimosaceae	
Casuarinaceae	*Casuarina*	Moraceae-Urticaceae	*Ficus*
Combretaceae	Combretaceae-*type*	Pandanaceae	*Pandanus*
	Lumnitzera	Phyllanthaceae	*Glochidion*
Euphorbiaceae	*Blumeodendron*	Primulaceae	*Aegiceras*
	Excoecaria	Rhizophoraceae	*Bruguiera*
Lecythidaceae	*Barringtonia*		*Ceriops*
Leguminosae	*Derris*		*Rhizophora*
		Rubiaceae	*Ixora*

Rainforest (RF)

Plant family	***Pollen taxon***	**Plant family**	***Pollen taxon***
Aizoaceae	*Sesuvium*	Leguminosae	*Acacia*-type
Araceae		Liliaceae	
Arecaceae	*Areca*	Melastomataceae	
	Arecaceae type	Meliaceae	*Aglaia*
	Arenga		Meliaceae-type
	Calamus	Moraceae-Urticaceae	
	Nenga	Myrsinaceae Myrtaceae	
	Pinanga	Musaceae	
Amaryllidoideae	*Crinum*	Nymphaeaceae	*Nuphar*
Anacardiaceae		Oleaceae	
Apocynaceae	*Alstonia*	Paracryphiaceae	*Quintinia*
Begoniaceae		Pinaceae	*Pinus*
Campanulaceae		Podocarpaceae	*Dacrycarpus*
Cannabaceae			*Dacrydium*
Capparaceae	*Capparis*		*Podocarpus*
Convolvulaceae		Rutaceae	
Cyperaceae		Rubiaceae	*Nauclea*
Escalloniaceae	*Polyosma*		*Psychotria Randia*
Fabaceae		Sapindaceae	*Acer*-type
Fagaceae	*Castanopsis*		*Pometia*

	Fagaceae-type	Sapotaceae	
	Lithocarpus	Stemonuraceae	*Stemonurus*
Gnetaceae	*Gnetum*	Vitaceae	

Open vegetation (OV)

Plant family	***Pollen taxon***	**Plant family**	***Pollen taxon***
Amaranthaceae		Lamiaceae	
Asteraceae		Leguminosae	*Bauhinia*
Bignoniaceae		Malvaceae	
Cornaceae		Poaceae	
Dasticaceae		Rubiaceae	*Spermacoce*
Ericaceae		Tiliaceae	

Staple Agriculture (SA)

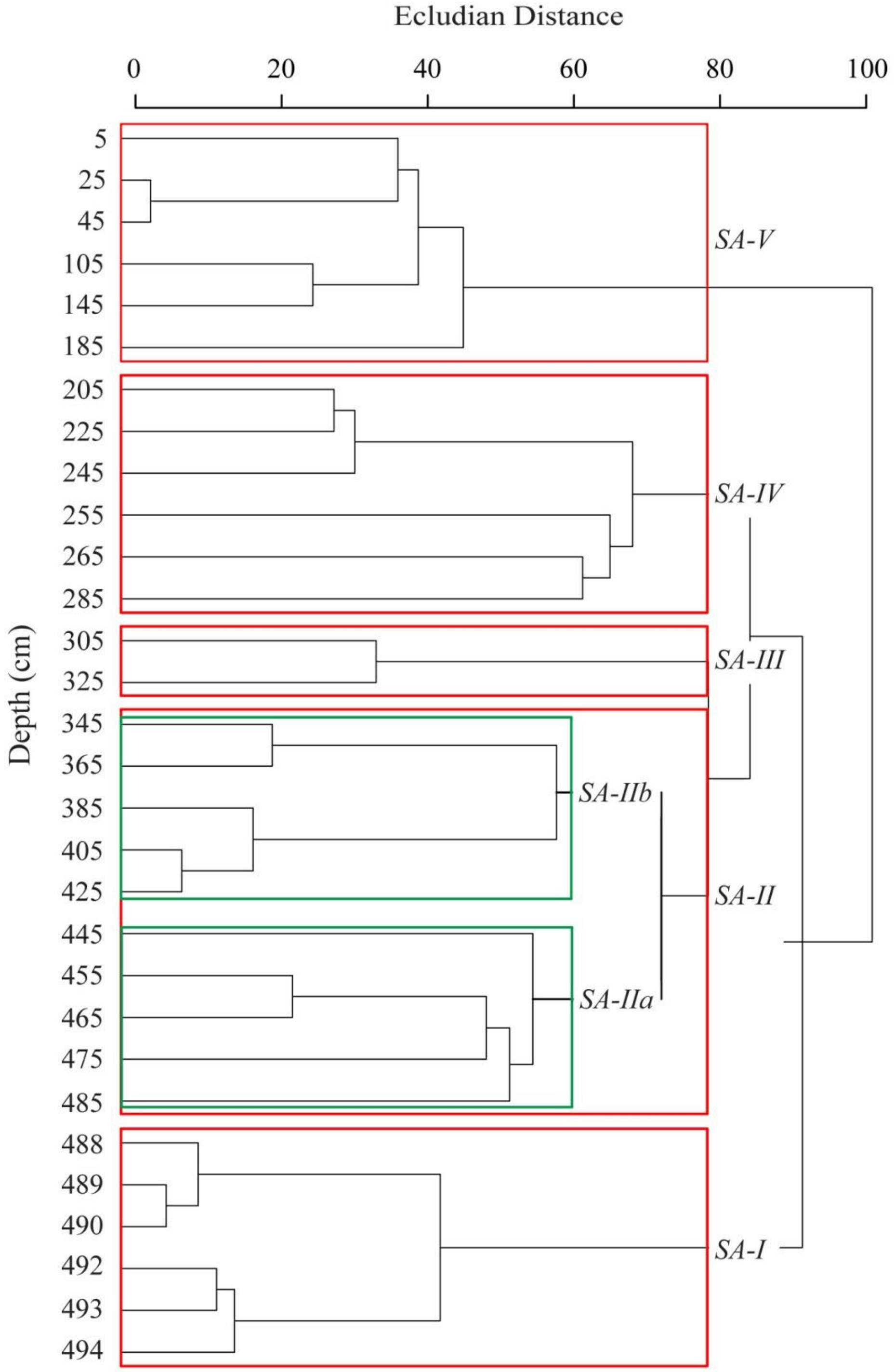
Plant family	***Pollen taxon***	**Plant family**	***Pollen taxon***
Araceae	*Colocasia*	Oxalidaceae	*Oxalis*
Arecaceae	*Cocos*	Poaceae	*Oryza*-type
Droseraceae		Rutaceae	*Citrus*
Nymphaeaceae	*Nymphaea*		*Zanthoxylum*
Onagraceae	*Ludwigia*	Solanaceae	

Spores

Plant family	***Spore taxon***
Monolete spores	
Trilete spores	

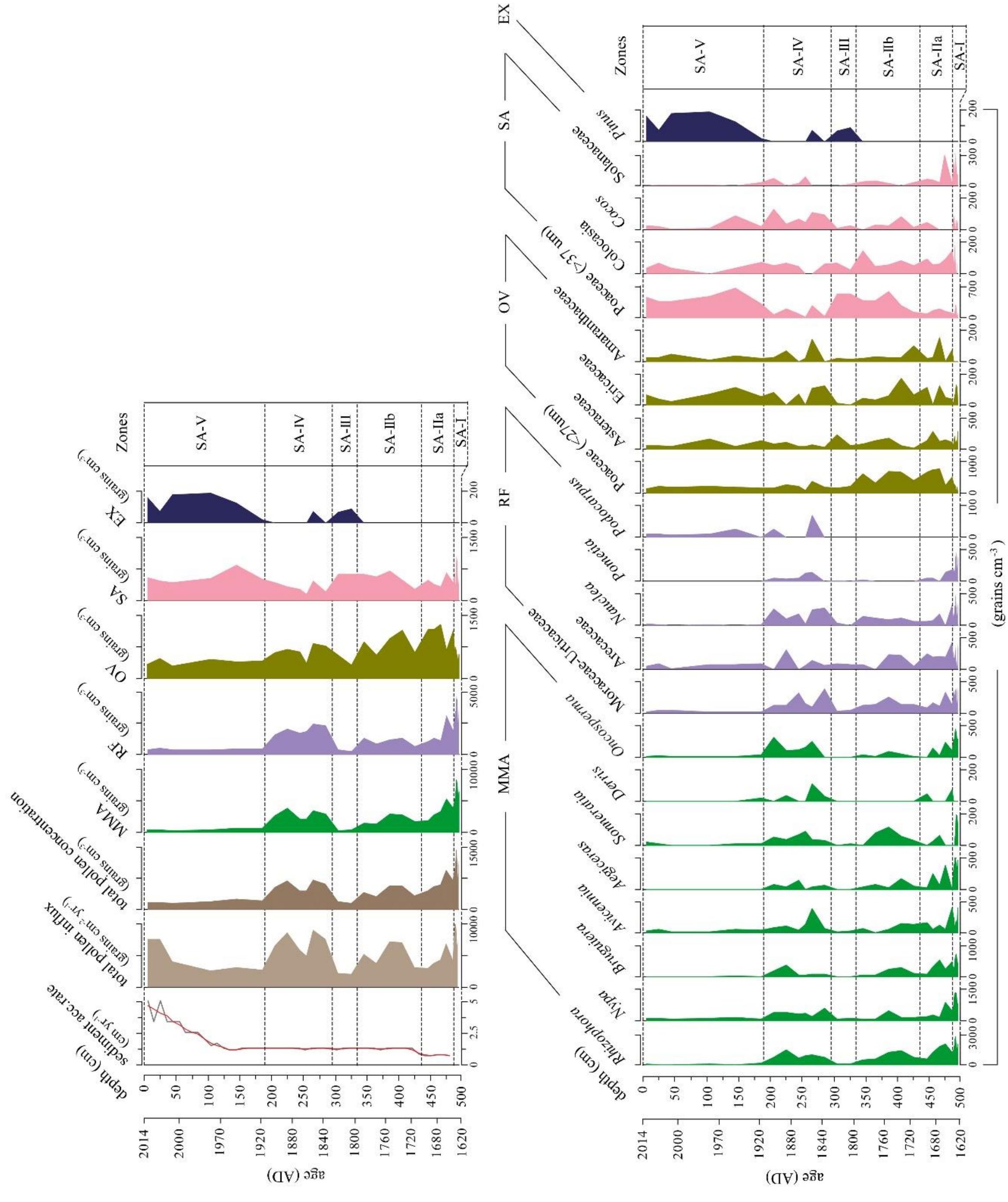
Supplementary 2

Cluster analysis result of pollen and spore data



Supplementary 3

Diagram of total pollen influx and pollen concentrations of Segara Anakan Lagoon. Only the concentration of taxa that are shown in main text Fig. 4.3 are presented here.



Chapter 5

Synthesize

This research contributes to an enhanced ecological understanding of Indonesian coastal wetlands by providing insights on the environmental processes and controlling factors of the dynamics of the ecosystems and their C sequestration capacities. The significance of the ecosystems' C sequestration capacities and their potential for global climate change mitigation is also underlined here. The resilience of the ecosystems to both natural and anthropogenic disturbances is also assessed. Altogether, the outcomes of this research help improving the effectiveness and efficiency of Indonesian coastal wetlands conservation, management and restoration.

5.1. Environmental and C accumulation dynamics of coastal wetlands in Indonesia

Indonesian coastal wetlands are very dynamics ecosystems that are mainly influenced by sea level changes, climatic variabilities and human activities. These factors affect the ecosystems by changing their hydrological conditions and vegetation composition. The impacts of these factors are, however, can be site-specific. For instance, the occurrence of ENSO-related drought in the late Holocene that caused little impact in vegetation and hydrological condition of Sungai Buluh peatland resulted in forest burning and vegetation change in Kalampangan peatland, Borneo (Yulianto and Hirakawa, 2006). Meanwhile the 19th-century dryer climatic condition of Java has increased salinity of Segara Anakn Lagoon (SAL) that changed the taphonomic process of the lagoon.

Similar to their environmental conditions, the ecosystem capacities of Indonesian coastal wetlands to store C are also dynamics. The variation in C accumulation of these ecosystems is closely related to the changes in environments hence controlled by similar driving factors. Consistent with their site-specific impacts on the environments, the influence of these factors to the C accumulation capacities of the coastal wetlands can also be unique and distinct in different sites. Human activities, for example, decreased the C accumulation rate in Sungai Buluh by changing the vegetation composition that negatively affect both ecosystem productivity and litter quality. Contrarily, it enhanced the C accumulation in SAL by increasing sediment input into the lagoon that fuels C burial.

Generalization on the factors and ecosystem processes of coastal wetlands in Indonesia thus can be misleading. Identification of the site-specific processes and driving factors in addition to their potential unique consequences on the coastal wetland environments and their C sequestration functions is therefore important. This is particularly necessary in determining the future fate of the ecosystems in response to the global environmental changes such as warming climate or SLR (Webb et al., 2013).

5.2. Potential of Indonesian coastal wetlands for climate change mitigation

In term of belowground C sequestration, the potential of Indonesian coastal wetlands in the global C storage in comparison with other tropical coastal wetlands seems modest (Fig. 5.1). However, this comparison should be interpreted cautiously for two reasons. First, the global data can only include mangrove dominated ecosystems owing to the lack of CAR data from coastal peatlands in other tropical regions (Appendix I.a). Second, it cannot include Africa due to a remarkable lack of CAR data from African tropical coastal wetlands.

Compared to other similar ecosystems globally, Indonesian mangrove dominated ecosystem has high C accumulation capacity (Fig. 5.1). However, as it is only represented by a single site in this comparison (Appendix I.a), the underlying cause of its higher C accumulation rate amongst other global mangrove dominated ecosystems is difficult to define. We hypothesize that it is due to a higher input of sediment into the ecosystem.

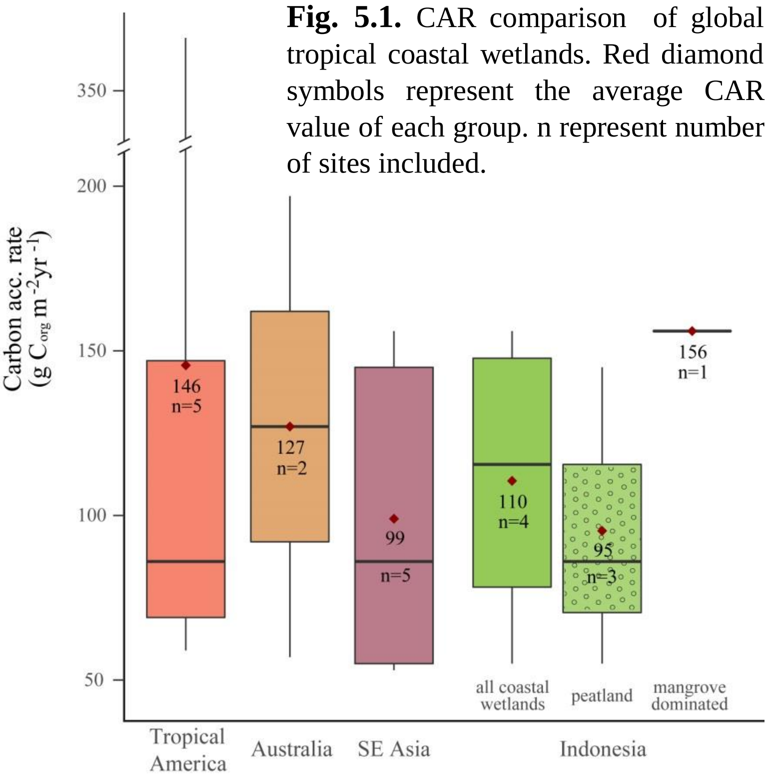


Fig. 5.1. CAR comparison of global tropical coastal wetlands. Red diamond symbols represent the average CAR value of each group. n represent number of sites included.

In Indonesia, mangrove dominated ecosystem also sequesters C faster than the coastal peatlands. This is conceivably due to higher allochtonous sediment input into mangrove dominated ecosystems that promotes faster sediment accretion and OM hence C burial (Sanders et al., 2014). The peat and C accretions on peatlands, on the other hand, rely almost exclusively on the autochthonous OM production (Wüst et al., 2003).

This comparison however should also be interpreted cautiously because of the difference temporal length of the datasets between mangrove dominated ecosystems and peatlands (Appendix I.a). The shorter dataset might show a higher CAR of a coastal wetland due to the partial cycle of the ongoing OM degradation (Breithaupt et al., 2014). Although the dataset from SAL suggests that 100 years dataset is suitable to represent the stabilized long-term CAR of the lagoon (Chapter 4.5.2), the length of “suitable dataset representation” can vary between sites. Differentiation between similar ecosystems, in this case between mangrove dominated ecosystems and peatlands, in comparing the CAR of tropical coastal wetlands is therefore necessary.

Due to lack of data in coastal tropical region particularly outside SE Asia, it is as yet not possible to compare the CAR of Indonesian costal peatland with the other tropical coastal peatlands. However, in comparison with the CAR of other inland tropical peatlands, Indonesian coastal peatlands have a high potential in sequester C (Fig. 5.2). This is possibly due to the generally wetter climates in the coastal areas than the inlands (Mongillo and Zierdt-Warshaw, 2000) hence the water table of the coastal peatlands are better maintained and the OM are better preserved.

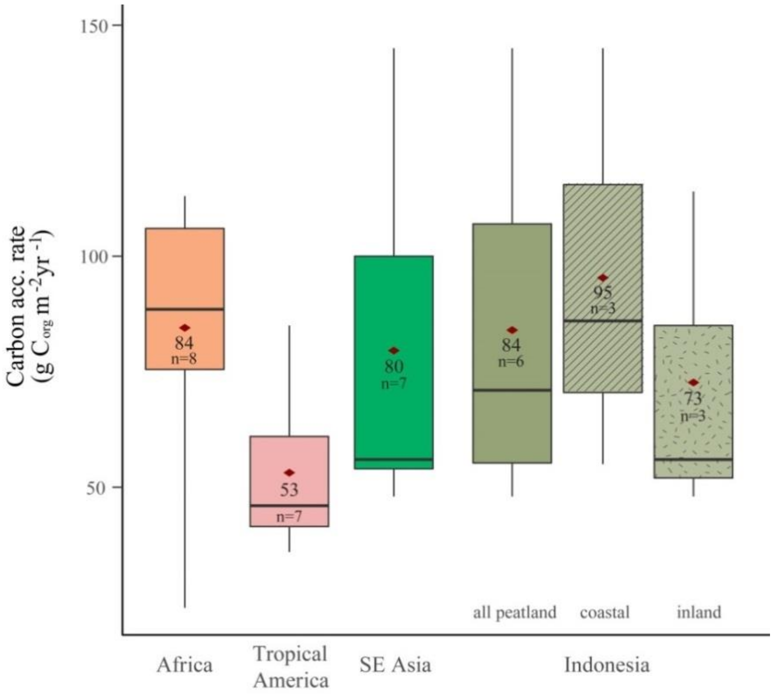


Fig. 5.2. CAR comparison of global tropical peatlands. Red diamond symbols represent the average CAR value of each group. n represent number of sites included. Datasets used are listed in Appendix I.b

5.3. Resilience of Indonesian coastal wetlands to disturbances

Indonesian coastal wetlands have shown resilience to natural disturbances in the past. Under their natural condition, coastal peatlands in Indonesia tend to be resistant to drought due to their high humidity that acts as fire barriers (Chapter 2.5.2). Hence, these ecosystems were able to maintain their ecosystem functions to sequester carbon during the drier climatic conditions. Similarly, drier climatic condition did not negatively impact the C accumulation capacity of mangrove dominated ecosystem in Indonesia (Chapter 4.5.2).

As to the natural disturbances, Indonesian coastal wetland also displayed resilience to anthropogenic disturbances. Indonesian coastal peatland was able to recover its vegetation composition and C sequestration function from the impacts of logging/timber harvesting, grazing/cut-and-carry, and wild-harvesting activities (Chapter 3.6.). On the other hand, recovery of mangrove dominated ecosystem in Indonesia after human disturbance such as mangrove exploitation, river modification and land use conversion, currently cannot be assessed as the disturbance is still ongoing. Regardless, mangrove dominated ecosystem in Indonesia can still maintain its C sink capacity despite the severe human disturbances in the ecosystem (Chapter 4.5.2).

5.4. Implications for conservation, management and restoration of coastal wetlands in Indonesia

5.4.1. Priority setting

Recently, in order to make the best use of limited resources, conservation and restoration efforts recognized the significance of "*priority setting*" (Noss et al., 2009; Hagen et al., 2016). Priority setting helps the conservation and restoration practitioners to solve the most critical problems, optimize the efforts and allocate the resources by identifying and classifying one matter in terms of species, ecosystems, areas or actions as more important than the other (Hagen et al., 2016). A thorough understanding of ecological values and process and the future fate of the ecosystem is required to set priorities in conservation and restoration planning (Noss et al., 2009).

a. Priority sites

This research shows that both study sites have considerable C sequestration capacities. Therefore, these two wetlands can be prioritized for conservation or restoration efforts particularly those that are designated for climate change mitigation (e.g. REDD+). Decisions taken to target these two wetlands for conservation and restoration efforts are therefore appropriate and their statuses need to persevere.

b. Priority actions

In the face of SLR, this research suggests that the C storage capacity of Sungai Buluh might be jeopardized by the effect of river flooding (Chapter 2.6). To preserve its noteworthy role in global C sequestration in the future, SLR mitigation action such as the construction of higher levee thus can be prioritized in the Sungai Buluh. On the other hand, SRL will likely favor SAL's capacity to accumulate C by enhancing C_{org} deposition through salinity-induced flocculation (Chapter 4.6.). However, as mangrove acts as the main C_{org} source in SAL, the lagoon's capacity to accumulate C would heavily depend on the existence of the surrounding mangrove forest. Mangrove forest in surrounding SAL is currently threatened by illegal logging and conversion (Setyoko and Rosyadi, 2009). Thus, in SAL's case, it is crucial to take immediate action to prevent further mangrove forest degradation. Better yet, to also minimize the barriers and provide adequate space for future landward migration.

c. Priority taxa

Often, wetland restoration frameworks target the recovery of biodiversity (Meli et al., 2014). However, due to lack of resources and ecological deterioration severity, targeting full biodiversity recovery can give unrealistic expectation (Hobbs et al., 2011). In that case, prioritizing some alternative reasonable targets such as an enhanced C sequestration capacity is therefore suggested. Enhancing the sequestration capacity of the targeted ecosystem might be conducted by planting the keystone taxa for rapid C accumulation. In Sungai Buluh's case, for instance, *Freycinetia,* Myrtaceae, *Calophyllum,* Stemonuraceae, *Ficus* and Euphorbiaceae can be prioritized for peatland C management and conservation (Chapter 2.5.2).

5.4.2. Conservation and sustainable use

To supports the success of conservation efforts in coastal wetlands, the issues resulted from the tensions and conflicts between the local stakeholders and conservation practitioners need to be solved (Mulyana et al., 2010). This can be done by accommodating the needs of local people through the allocation of conservation area for sustainable use with some pre-set constrains that are settled to ensure the success of the conservation efforts (Mulyana et al., 2010). The limits of the acceptable natural resource utilization can then follow the past type and magnitude "resilience-friendly" human activities as blueprints. Taking example from the case of Sungai Buluh, grazing or cut-and-carry and the collection of timber and NTFP thus can be allowed given that their impacts to the ecosystem were reversible (Chapter 3.6).

5.4.3. Restoration frameworks

Restoration efforts can be very costly (Rey Benayas et al., 2009) while the available funding is often limited (Groves et al., 2002). Thus, to optimize the ecological benefit from the often limited resource, restoration efforts need to be effective and efficient (Rey Benayas et al., 2008; Graham et al., 2013). Developing an accurate restoration framework by appropriately selecting the approaches and allocating the resource can enhance the effectiveness and efficiency of restoration efforts (Holl, 2017).

Restoration option spreads from passive, which is cheaper, to active that is more expensive (Noss et al., 2009). Knowing whether the target ecosystem still have the requirements to passively recover can helps to decide whether the application of an expensive restoration technique is really necessary. Moreover, cost-efficient restoration technique will enable the allocation of resources for the aftercare monitoring and maintenance which is often disregarded (Graham et al., 2017).

Active restoration, which is commonly carried out by planting tree seedlings, is very expensive and often provides unsatisfying outcomes (Meli et al 2017). This is due to the selection of planted taxa that more of economic and conservation interest rather than of their regeneration abilities. The taxa with high regenerative power such as *Gnetum, Calophyllum, Ficus* and Sapotaceae from Sungai Buluh's case is therefore not only more appropriate to be selected for the regeneration of forest cover through tree planting, but also suitable to be used in lower cost techniques (Chapter 3.6.).

5.4.4. Conservation and restoration policies

The current peatland restoration frameworks in Indonesia, which mostly aims for full biodiversity recovery, are designated for 60 years following the concession license granted by the local government. Considering the 170-years recovery time of Sungai

Buluh, the 60-years restoration frameworks is undoubtedly insufficient to meet their targets. This suggests that the current restoration policy in Indonesia is not supportive enough. Legal regulation is supposed to support and help the restoration efforts, rather be an obstacle, to accomplish their goals (Aronson et al., 2011). The local policy therefore needs to be evaluated or reformulated more flexibly in accordance to the underlying intention of the restoration project.

More attention also needs to be given in regards to the conservation policy in Indonesia. Taking SAL as an example, although the area is conserved and protected under the Decree No. 6/2001 issued by local government, illegal logging and unauthorized conversion still often occur in the surrounding mangrove forests (Setyoko and Rosyadi, 2009). It appears that the implementation of the decree has been weak and ineffective due to (1) lack of resources, (2) poor communication between the government and local people, (3) unawareness from local people of the purpose and content of the decree, (4) unreliable law enforcement, and (5) overlap of authority amongst different government/non-government organizations (Setyoko and Rosyadi, 2009). As the existence of surrounding mangrove is crucial to maintain SAL's capacity to accumulate C, a strengthened implementation of conservation policy is required and the identified problems need to be solved urgently.

5.5. Future challenges

Despite of the implications provided by this research to enhance the conservation, management and restoration efforts of the coastal wetlands particularly in Indonesia, there are still some challenges and further works identified that need to be solved and undertaken in the future to help ensuring the sustainability of the values and functions these important ecosystems.

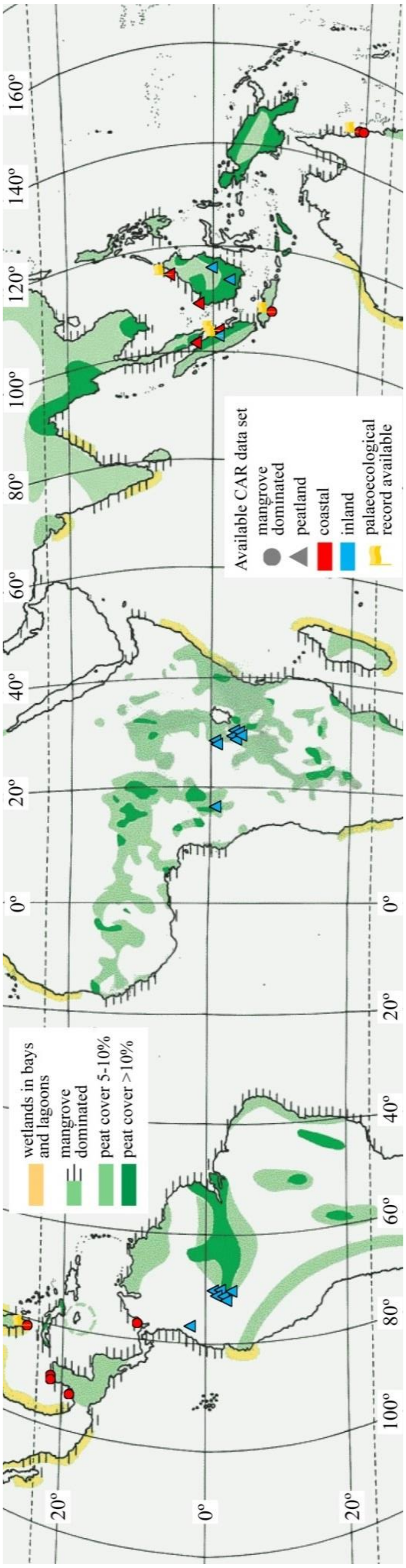


Fig. 5.3. Map of available CAR datasets from tropical coastal wetlands containing mangrove dominated ecosystems and peatlands (after Lappalainen, 1996).

5.5.1. Similar studies

Coastal wetlands are currently of main interest in relation to global sea level rise and climate change mitigation for they have high C sequestration capacity (Grimsditch et al., 2012) with long-term CAR considered to best reflect the future trend (Breithaupt et al., 2014). Yet the available information regarding the long-term belowground C accumulation of global coastal wetlands remains surprisingly limited with lack representation of the different coastal ecosystems or geographical regions (Fig. 5.3). Particularly for mangrove dominated ecosystems, the available long-term C accumulation data comprise only around 100 years. More studies focusing on long-term C sequestration in coastal wetlands are therefore required to conduct.

The available palaeoecological information complementing the available long-term C dataset is also very scarce (Fig. 5.3). Such information is crucial to thoroughly understand the dynamics of C sequestration in coastal wetlands as well as the controlling factors. This is particularly important as the different ecosystems, as well as their C sequestration capacities, often respond differently to the same environmental factor (Webb et al., 2013; Hapsari et al., 2017). More palaeoecological studies that complement the available long-term C sequestration data therefore need to be carried out.

5.5.2. Pollen and vegetation relationship

The assemblages of pollen and spores in the core generally reflect the vegetation of the surrounding area at the time of their deposition (Bradley, 1999). However, there are still biases in the data due to the differences in pollen production, dispersibility and preservation. Unless the biases are fully addressed, the pollen data cannot be used quantitatively (Williams and Jackson, 2003).

Taking the case of Sungai Buluh peatland as an example, the degrees of acceptable timber harvesting in Sungai Buluh peatland cannot be defined using the available pollen data (Chapter 3.6.). The pollen-vegetation relationship in Sungai Buluh peatland needs to be previously determined in order to “transpose” the 30% reduction of pollen of forest trees into a coverage area. Unlike in Europe and North-South America where the relationships between pollen rain and vegetation cover are very well explored (e.g. Rull, 2001; Sugita, 2007; Dawson et al., 2016), the pollen-vegetation relationship in Southeast Asia especially Indonesia has never been studied. Thus, studies assessing the pollen-vegetation relationship are needed and encouraged in South East Asian regions, especially Indonesia with its vast coastal wetlands.

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Appendix I

Available dataset on C accumulation rates of tropical coastal wetlands and tropical peatlands

Appendix I
Available dataset on CAR of tropical coastal wetlands and peatlands

Appendix I.a. Available dataset on C accumulation rate of tropical coastal wetlands

No.	Sites				Record length (years)	C accumulation rate ($g\ C\ m^{-2}\ yr^{-1}$)	Reference
1	South-East Asia	Brunei	peatlands	Mendaram, Borneo	2776	53	Dommain et al., 2015
2		Indonesia		Teluk Keramat, Borneo	4521	86	Neuzil, 1997; Dommain et al., 2011
3				Siak Kanan, Sumatra	5986	145	
4				Sungai Buluh, Sumatra	13419	55	Hapsari et al., 2017
5			mangrove dominated ecosystems	Segara Anakan lagoon, Java	400	126	*This study (Chapter 4)*
					100	153	
6	Tropical America	Mexico		Celestun lagoon	100	69	Gonneea et al., 2004
7				Chelem lagoon	100	86	
8				Terminos lagoon	100	59	
9		Colombia		Soledad lagoon	100	367	Ruiz-Fernández et al., 2011
10		Florida, US		Everglades National Park	100	147	Smoak et al., 2013
11	Australia			Missionary Bay	100	197	Brunksill et al., 2002
12				Hinchinbrook Channel	100	57	

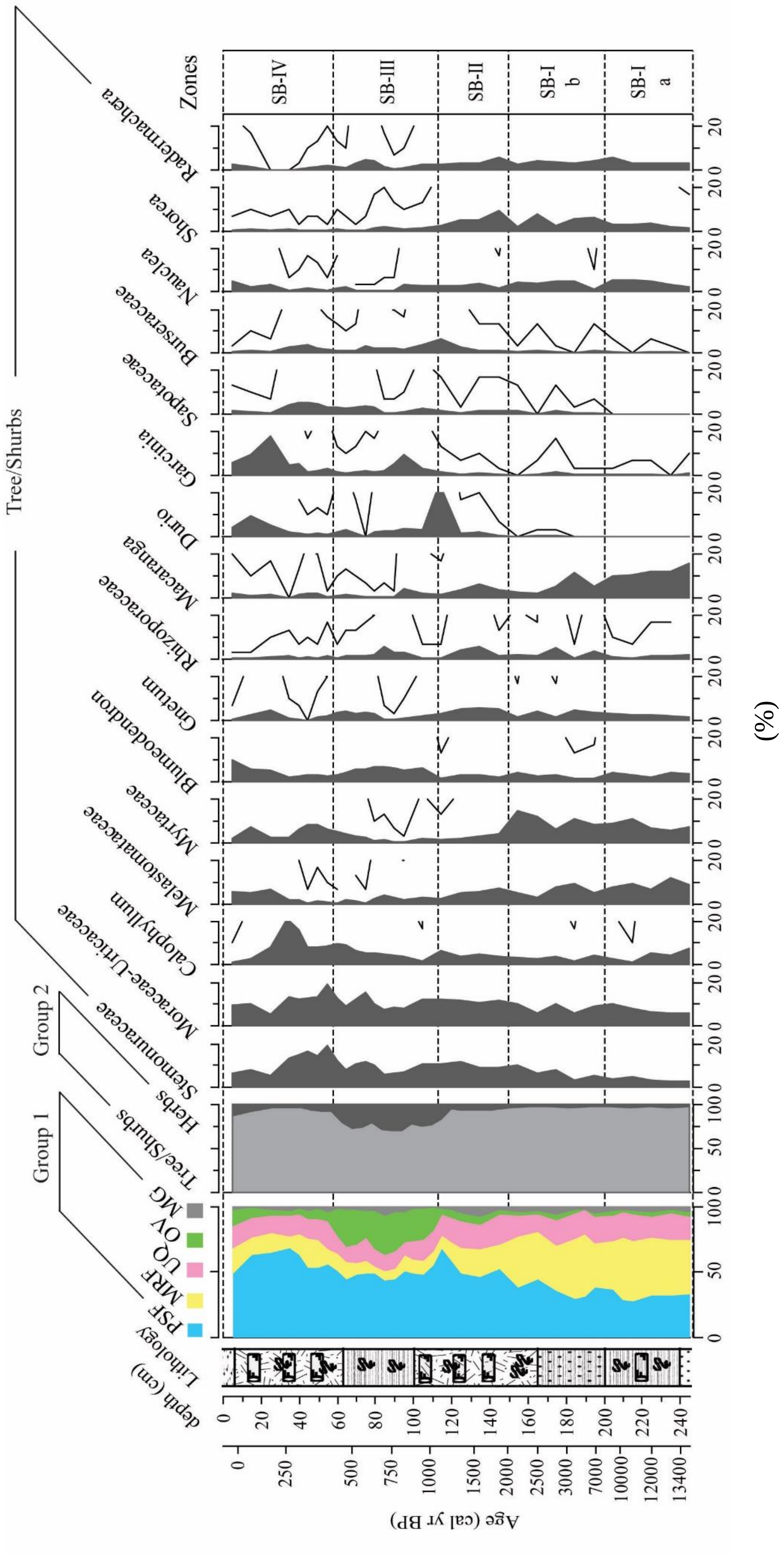
Appendix I.b. Available dataset on C accumulation rate of tropical peatlands

No.	Sites				Peat basal age (cal yr BP)	C accumulation rate (g C m^{-2} yr^{-1})	Reference
1	South-East Asia	Brunei	coastal	Mendaram, Borneo	2776	53	Dommain et al., 2015
2		Indonesia		Teluk Keramat, Borneo	4521	86	Neuzil, 1997; Dommain et al., 2011
3				Siak Kanan, Sumatra	5986	145	
4				Sungai Buluh, Sumatra	13419	55	Hapsari et al., 2017
5			inland	Kutai, Borneo	8285	114	Hope et al., 2005; Dommain et al., 2014
6				Air Hitam, Sumatra	8562	48	Biagioni et al., 2015
7				Sebangau, Borneo	26000	56	Page et al., 2004
8	South America	Peru		Riñón	1615	39	Lähteenoja et al., 2009
9				Quistococha	2335	74	
10				San Jorge	2945	85	
11				Maquia	1975	48	Lähteenoja et al., 2012
12				Roca Fuerte	5170	36	
13				Aucayacu	8870	44	
14		Equador		Cayembre-Coca	2,770	46	Chimner and Karberg, 2008
15	Africa	Rwanda/Burundi		Gishoma	1510	86	Panujen, 1996
16				Mashya	1800	91	
17				Cyili	2610	113	
18				Ndurumu	8130	78	
19				Rwamiko	8440	112	
20				Kinamba	11710	68	
21				Buyongwe	13730	104	
22		Congo		Cuvette Centrale	8835	24	Dargie et al., 2017

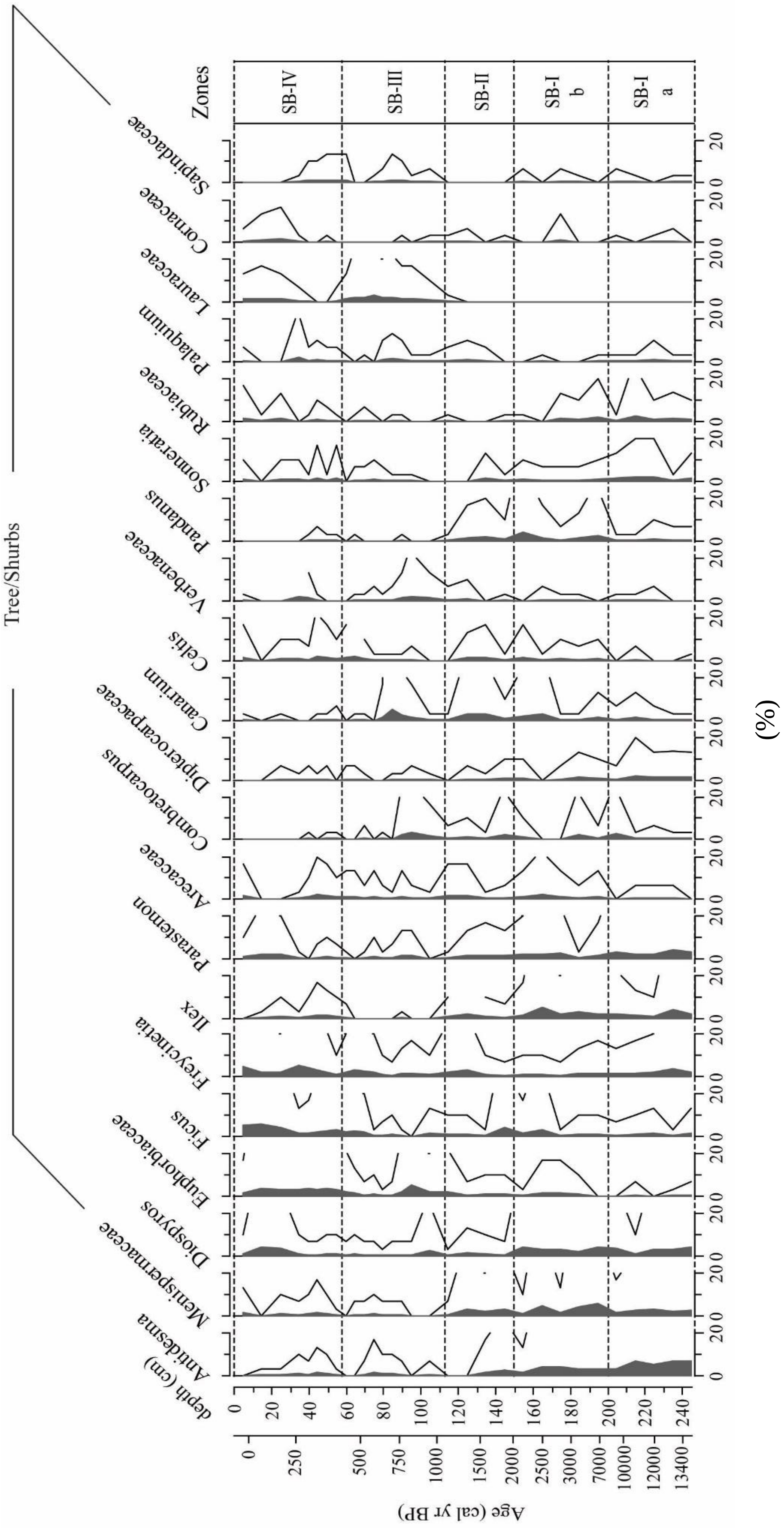
Appendix II

Complete pollen diagram of Sungai Buluh peatland

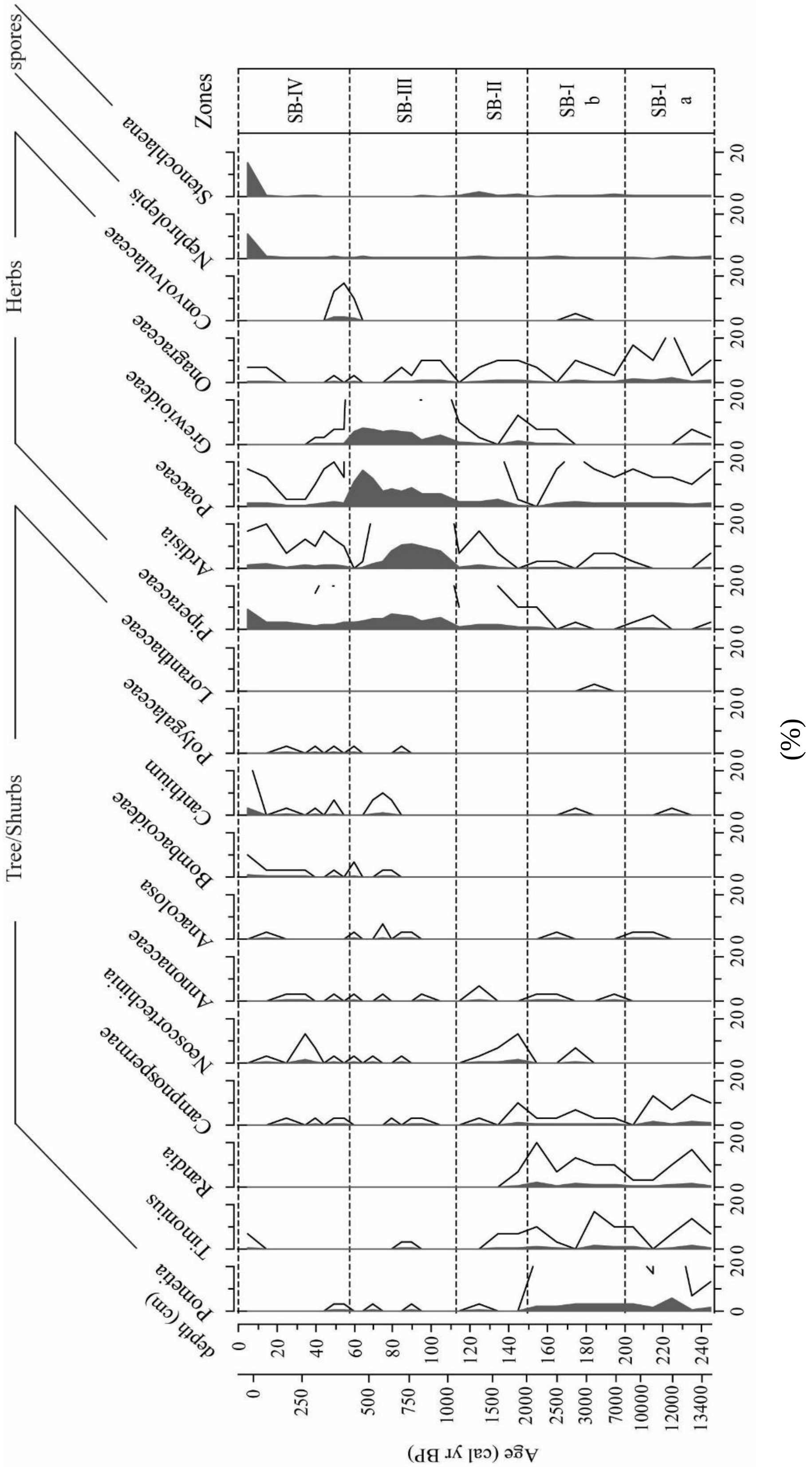
Appendix II
Complete pollen diagram of SB peatland



Complete pollen diagram of SB peatland



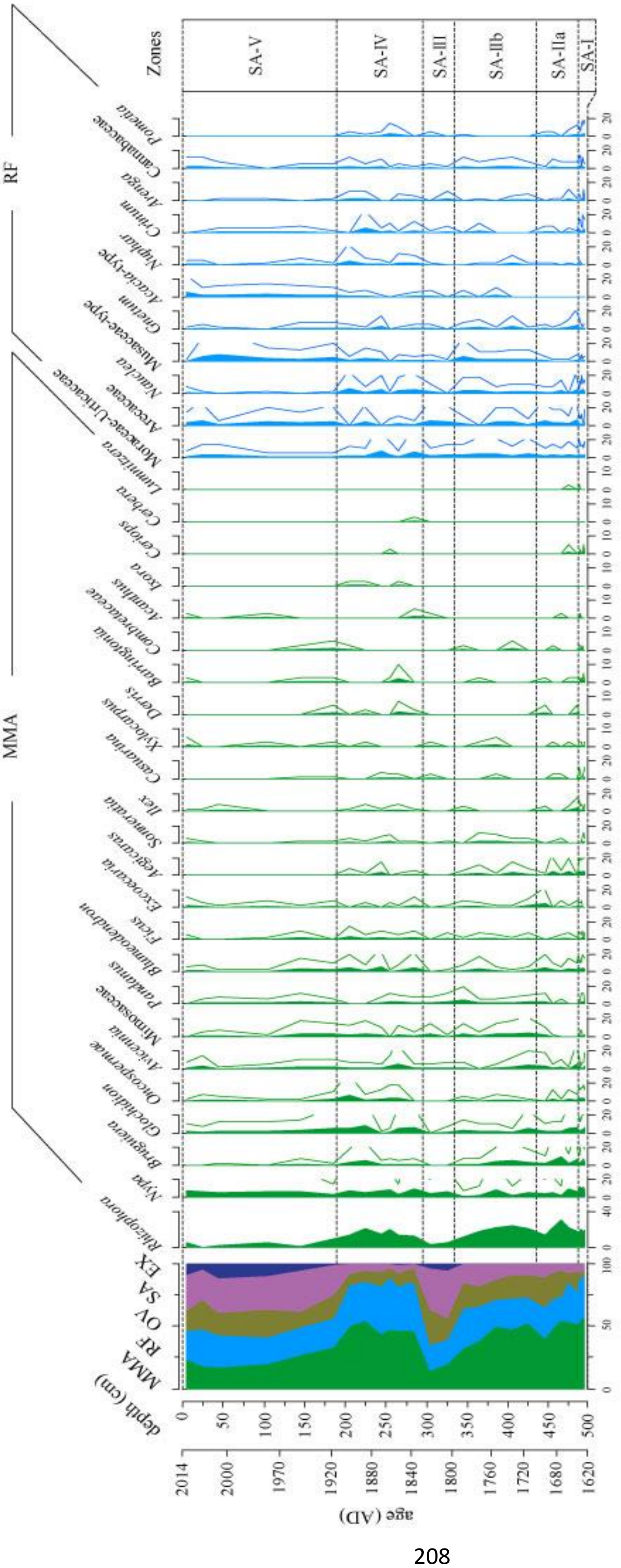
Appendix II
Complete pollen diagram of SB peatland



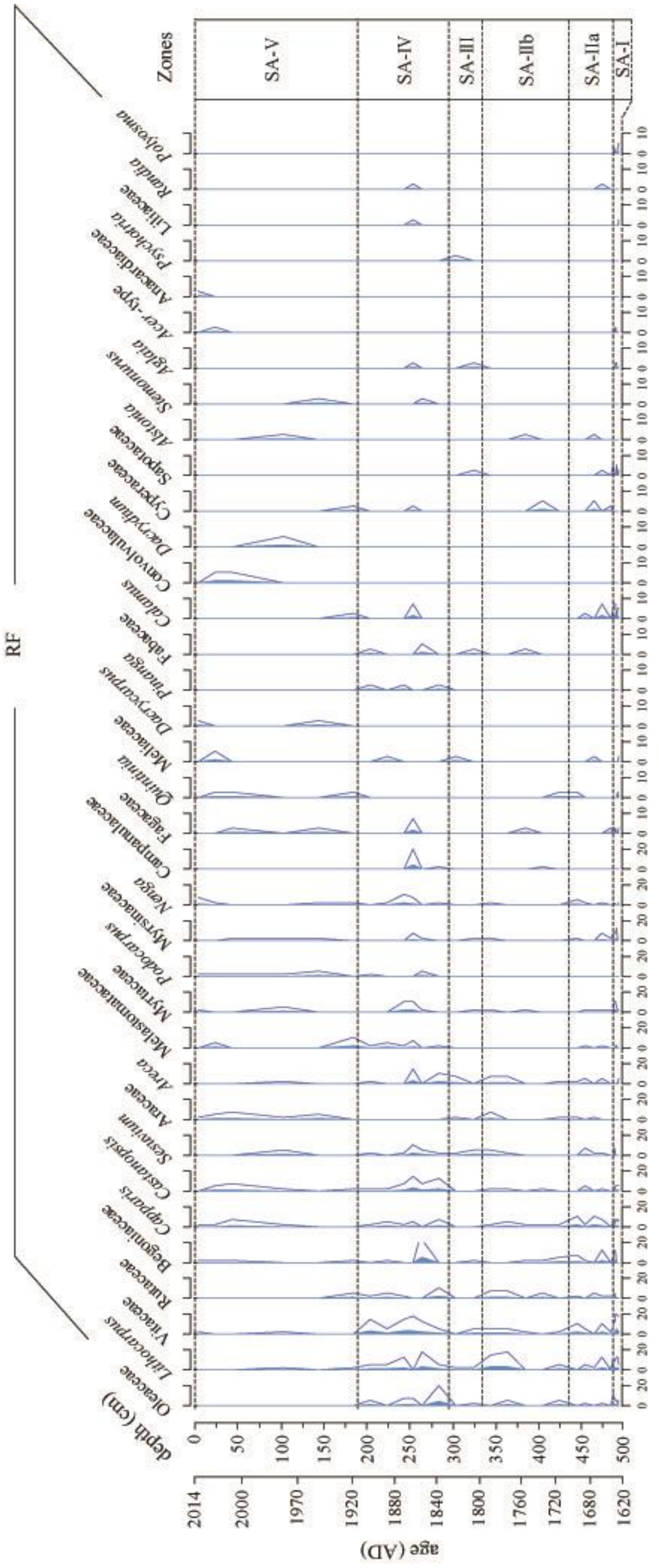
Appendix III

Complete pollen diagram of Segara Anakan lagoon

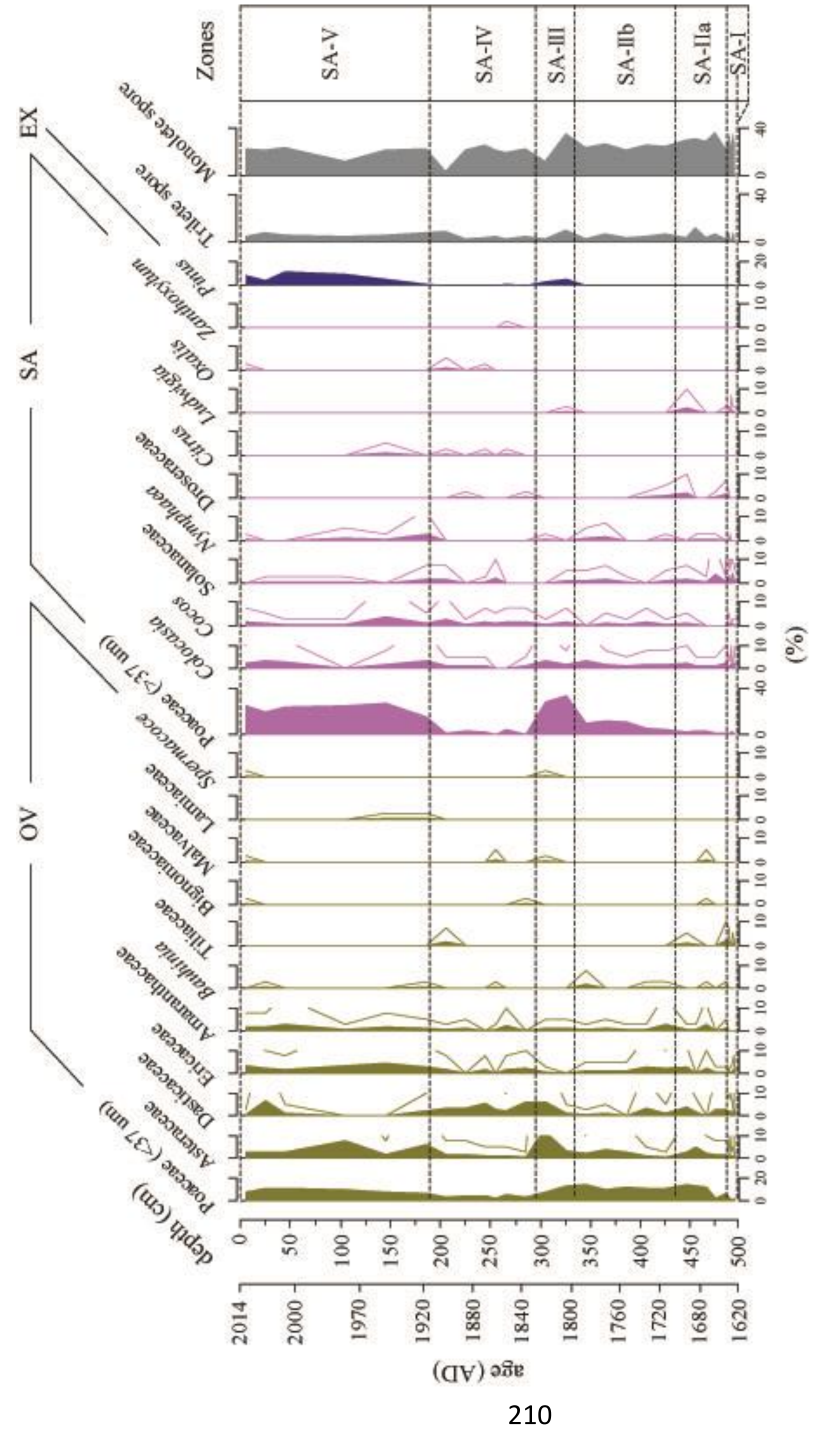
Appendix III
Complete pollen diagram of SAL



Appendix III
Complete pollen diagram of SAL



Complete pollen diagram of SAL



Appendix IV

List of identified pollen taxa

Appendix IV
List of identified pollen taxa

Pollen

Picture no.	Plant family	Pollen taxon	Reference site
1	Acanthaceae	*Acanthus*	b)
2		*Avicennia*	b)
	Aizoaceae	*Sesuvium*	b)
3	Amaranthaceae		b)
	Amaryllidoideae	*Crinum*	b)
	Anacardiaceae	Anacardiaceae type	b)
		Campnosperma	a)
	Anisophylleaceae	*Combretocarpus*	a)
	Annonaceae		a)
	Apocynaceae	*Alstonia*	b)
		Cerbera	b)
4	Aquifoliaceae	*Ilex*	a) b)
	Araceae	Araceae type	b)
		Colocasia	b)
5	Arecaceae	*Areca*	b)
		Arecaceae type	a) b)
		Calamus	b)
6		*Cocos*	b)
7		*Nenga*	b)
8		*Nypa*	b)
9		*Oncosperma*	b)
		Pinanga	b)
10	Asteraceae		b)
	Begoniaceae		b)
11	Bignoniaceae	Bignoniaceae type	b)
		Radermachera	a)
12	Burseraceae	*Canarium*	a)
		Burseraeae type	a)
	Campanulaceae		b)
	Cannabaceae	Cannabaceae type	b)
13		*Celtis*	a)
14	Capparaceae	*Capparis*	b)
15	Casuarinaceae	*Casuarina*	b)
	Chrysobalanaceae	*Parastemon*	a)
16	Clusiaceae	*Calophyllum*	a)
17		*Garcinia*	a)
18	Combretaceae	Combretaceae	Ist here not in a?
19		*Lumnitzera*	b)

20	Cornaceae		a) b)
	Convolvulaceae		a) b)
	Cyperaceae		b)
21	Dasticaceae		b)
	Dipterocarpaceae	Dipterocarpaceae type	a)
22		*Shorea*	a)
23	Droseraceae		b)
	Ebenaceae	*Diospyros*	a)
24	Ericaceae		b)
25	Escalloniaceae	*Polyosma*	b)
26	Euphorbiaceae	*Blumeodendron*	a) b)
		Euphorbiaceae type	a)
27		*Excoecaria*	b)
28		*Macaranga*	a)
		Neoscortechinia	a)
	Fabaceae		b)
29	Fagaceae	*Castanopsis*	b)
		Fagaceae-type	b)
30		*Lithocarpus*	b)
31	Gnetaceae	*Gnetum*	a) b)
	Lamiaceae		b)
32	Lecythidaceae	*Barringtonia*	b)
33	Leguminosae	*Acacia*-type	b)
		Bauhinia	b)
34		*Derris*	b)
	Liliaceae		b)
35	Loranthaceae		a)
	Lythraceae	*Sonneratia* type	a) b)
36		*Sonneratia* cf. *alba*	b)
		Sonneratia cf. *apetala*	b)
37		*Sonneratia* cf. *caseolaris*	b)
38		*Sonneratia* cf. *ovata*	b)
	Malvaceae	Bombacoideae	a)
39		*Durio*	a)
40		Grewioideae	a)
		Malvaceae type	b)
41	Meliaceae	*Aglaia*	b)
		Meliaceae type	b)
		Xylocarpus	b)
	Melastomataceae		a) b)
	Menispermaceae		a)

	Mimosaceae		b)
42	Moraceae-Urticaceae	Moraceae-Urticaceae type	a) b)
43		*Ficus*	a) b)
	Musaceae		b)
	Myrsinaceae		b)
44	Myrtaceae		a) b)
45	Nymphaeaceae	*Nuphar*	b)
		Nymphaea	b)
	Olacaceae	*Anacolosa*	a)
	Oleaceae		b)
	Onagraceae	Onagraceae type	a)
46		*Ludwigia*	b)
	Oxalidaceae	*Oxalis*	b)
	Pandanaceae	*Freycinetia*	a)
47		*Pandanus*	a) b)
	Paracryphiaceae	*Quintinia*	b)
48	Phyllanthacaeae	*Antidesma*	a)
		Glochidion	b)
49	Pinaceae	*Pinus*	b)
50	Piperaceae	*Piper*	a)
51	Poaceae	wild grass (<27 µm)	a) b)
52		cultivated Poaceae (>27 µm; e.g. *Oryza, Zea*)	b)
53	Podocarpaceae	*Dacrycarpus*	b)
		Dacrydium	b)
54		*Podocarpus*	b)
	Polygalaceae		a)
55	Primulaceae	*Aegiceras*	b)
56		*Ardisia*	a)
	Rhizophoraceae	Rhizophoraceae type	a) b)
57		*Bruguiera*	b)
58		*Ceriops*	b)
59		*Rhizophora*	b)
	Rubiaceae	*Canthium*	a)
60		*Ixora*	b)
61		*Nauclea*	a) b)
		Psychotria	b)
62		*Randia*	a) b)
		Rubiaceae type	a)
		Spermacoce	b)
63		*Timonius*	a)
	Rutaceae	Rutaceae type	b)
64		*Citrus*	b)
		cf. *Zanthoxylum*	b)

65	Sapindaceae	*Acer*-type *Pometia* Sapindaceae type	b) a) b) a)
66	Sapotaceae	*Palaquium* Sapotaceae type	a) a) b)
	Solanaceae		b)
67	Stemonuraceae	Stemonuraceae type *Stemonurus*	a) a) b)
68	Tiliaceae		b)
	Verbenaceae		a)
	Vitaceae		b)

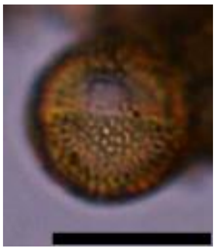
Spores

Picture no.	**Plant family**	**Pollen taxon**	**Reference site**
69	Blechnaceae	*Stenochlaena*	a)
70	Davalliaceae	*Davallia*	a) b)
71	Monolete spore		a) b)
	Nephrolepidaceae	*Nephrolepis*	a)
72	Trilete spore		a) b)

Reference site : a) Sungai Buluh peatland, Sumatra
b) Segara Anakan lagoon, Java

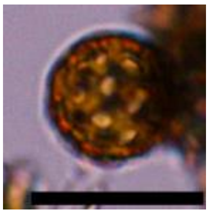
Acanthaceae

1) *Acanthus*



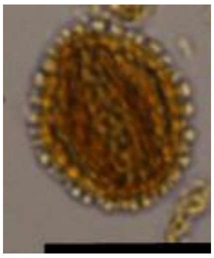
2) *Avicennia*

Amaranthaceae



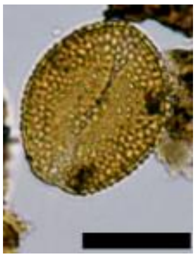
3) Amaranthaceae

Aquifoliaceae

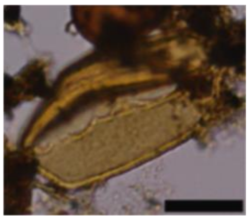


4) *Ilex*

Arecaceae



5) *Areca*



6) *Cocos*

7) *Nenga*

8) *Nypa*

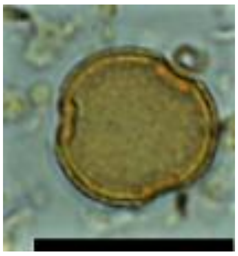
Arecaceae

9) *Oncosperma*

Asteraceae

10) Asteraceae

Bignoniaceae

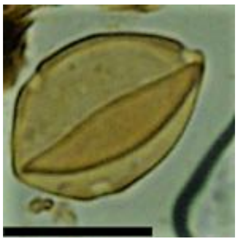


11) *Radermachera*

Burseraceae

12) *Canarium*

Cannabaceae

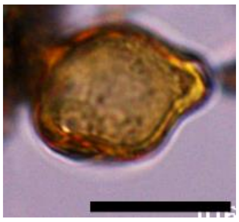


13) *Celtis*

Capparaceae

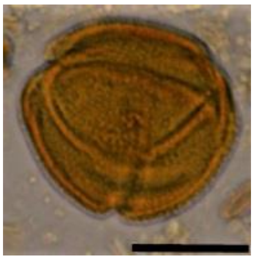
14) *Capparis*

Casuarinaceae

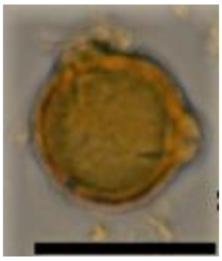
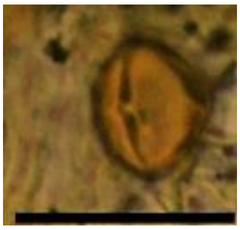
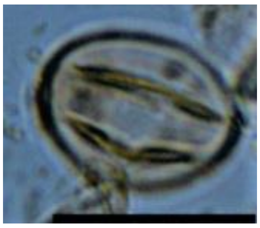
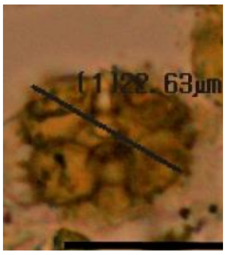
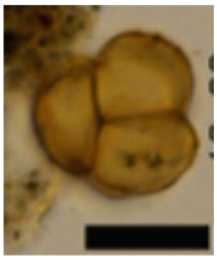
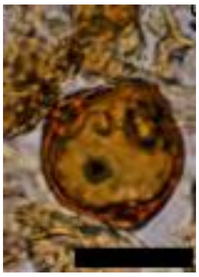
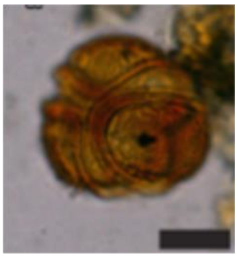


15) *Casuarina*

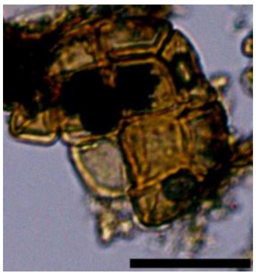
Clusiaceae



16) *Calophyllum*

Clusiaceae  *17) Garcinia*	**Combretaceae**  18) Combretaceae	 19) *Lumnitzera*	**Cornaceae**  20) Cornaceae
Dasticaceae  21) Dasticaceae	**Dipterocarpaceae**  *22) Shorea*	**Droceraceae**  23) Droceraceae	**Ericaceae**  24) Ericaceae
Escalloniaceae  *26) Polyosma*	**Euphorbiaceae**  *26) Blumeodendron*	 *27) Excoecaria*	 *28) Macaranga*
Fagaceae  *29) Castanopsis*	 *30) Lithocarpus*	**Gnetaceae**  *31) Gnetum*	**Lecythidaceae**  *32) Barringtonia*

Leguminosae



33) *Acacia*-type

34) Derris

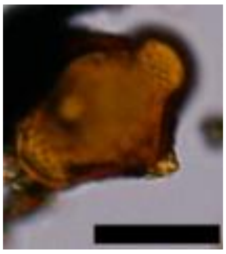
Loranthaceae

35) Loranthaceae

Lythraceae

36) Sonneratia cf. *alba*

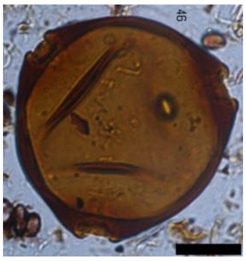
Lythraceae



37) S. cf. *caseolaris*

38) S. cf. *ovata*

Malvaceae



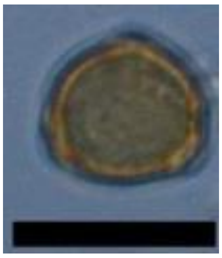
39) *Durio*

40) Grewioideae

Meliaceae

41) Aglaia

Moraceae-Urticaceae



42) Moraceae-Urticaeae

43) Ficus

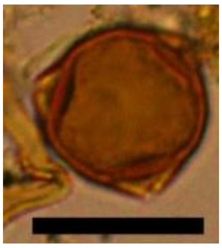
Myrtaceae

44) Myrtaceae

Nymphaeaceae

45) Nuphar

Onagraceae



46) Ludwigia

Pandanaceae

47) Pandanus

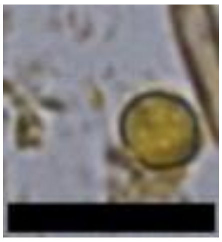
Phyllanthaceae

48) Antidesma

Pinaceae

49) Pinus

Piperaceae



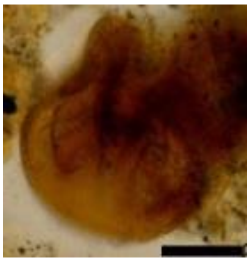
50) Piper

Poaceae

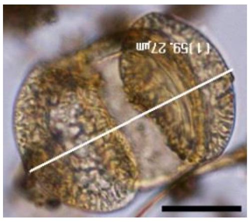
51) wild grass

52) cultivated Poaceae

Podocarpaceae



53) Dacrycarpus



54) Podocarpus

Primulaceae

55) Aegiceras

56) Ardisia

Rhizophoraceae

57) Bruguiera

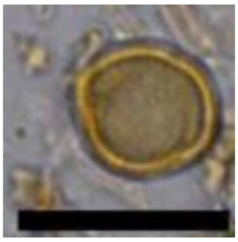
58) Ceriops

59) Rhizophora

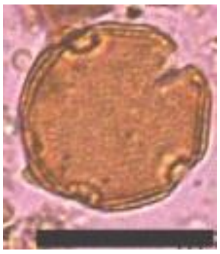
Rubiaceae

60) Ixora

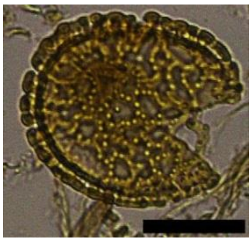
Rubiaceae



61) Nauclea



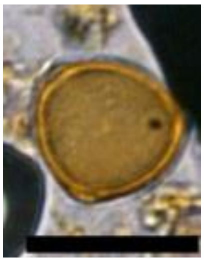
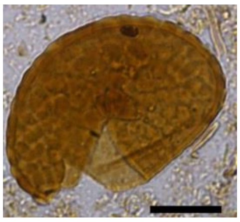
62) Randia



63) Timonius

Rutaceae

64) Citrus

Sapindaceae  65) *Pometia*	**Sapotaceae**  66) Sapotaceae-type	**Stemonuraceae**  67) *Stemonurus*	**Tiliaceae**  68) Tiliaceae
Blechnaceae  69) *Stenochlaena*	**Davalliaceae**  70) *Davallia*	**Monolete spore**  71) Monolete	**Trilete spore**  72) Trilete

Each black bar represents 20 μm.