Marine Pollution Bulletin 160 (2020) 111168

Contents lists available at ScienceDirect

Marine Pollution Bulletin

journal homepage: www.elsevier.com/locate/marpolbul

Review

Coastal and estuarine blue carbon stocks in the greater Southeast Asia region: Seagrasses and mangroves per nation and sum of total

A. Thorhaug[a,*], John Barry Gallagher[b], W. Kiswara[c], Anchana Prathep[d], Xiaoping Huang[e], Tzuen-Kiat Yap[f], Sue Dorward (M.Sc)[g], Graeme Berlyn[a]

[a] Yale University School of Forestry & Environmental Studies, New Haven, CT 00651, USA
[b] Institute of Marine and Antarctic Studies, University of Tasmania, Hobart, 7000, Australia
[c] Division of Earth Sciences, Indonesian Institute of Sciences, Jakarta, Indonesia
[d] Seaweed and Seagrass Research Unit, Prince of Songkla University, HatYai, Songkhla, 90112 Thailand
[e] South China Sea Institute of Oceanology, Chinese Academy of Sciences,Guangzhou, China
[f] Borneo Marine Research Institute, University Malaysia Sabah, Kota Kinabalu, Malaysia
[g] Raritan Valley Community College, NJ, USA

ARTICLE INFO

Keywords:
Blue carbon seagrass
Blue carbon mangroves
Tropical Southeast Asian blue carbon
Black carbon
Southeast Asia tropical estuarine carbon
Blue Carbon

ABSTRACT

Climate Change solutions include CO_2 extraction from atmosphere and water with burial by living habitats in sediment/soil. Nowhere on the planet are blue carbon plants which carry out massive carbon extraction and permanent burial more intensely concentrated than in SE Asia. For the first time we make a national and total inventory of data to date for "blue carbon" buried from mangroves and seagrass and delineate the constraints. For an area across Southeast Asia of approximately 12,000,000 km^2, supporting mangrove forests (5,116,032 ha) and seagrass meadows (6,744,529 ha), we analyzed the region's current blue carbon stocks. This estimate was achieved by integrating the sum of estuarine *in situ* carbon stock measurements with the extent of mangroves and seagrass across each nation, then summed for the region. We found that mangroves ecosystems regionally supported the greater amount of organic carbon (3095.19Tg C_{org} in 1st meter) over that of seagrass (1683.97 Tg C_{org} in 1st meter), with corresponding stock densities ranging from 15 to 2205 Mg ha^{-1} and 31.3 to 2450 Mg ha^{-1} respectively, a likely underestimate for entire carbon including sediment depths. The largest carbon stocks are found within Indonesia, followed by the Philippines, Papua New Guinea, Myanmar, Malaysia, Thailand, Tropical China, Viet-Nam, and Cambodia. Compared to the blue carbon hotspot of tropical/subtropical Gulf of Mexico's total carbon stock (480.48 Tg Corg), Southeast Asia's greater mangrove–seagrass stock density appears a more intense Blue Carbon hotspot (4778.66 Tg Corg). All regional Southeast Asian nation states should assist in superior preservation and habitat restoration plus similar measures in the USA & Mexico for the Gulf of Mexico, as apparently these form two of the largest tropical carbon sinks within coastal waters. We hypothesize it is SE Asia's regionally unique oceanic–geologic conditions, placed squarely within the tropics, which are largely responsible for this blue carbon hotspot, that is, consistently high ambient light levels and year-long warm temperatures, together with consistently strong inflow of dissolved carbon dioxide and upwelling of nutrients across the shallow geological plates.

1. Introduction

There is a general consensus that advent of anthropogenic greenhouse gas emission has led to the heating of both the ocean and atmosphere. In response, the International Panel for Climate Change of the United Nations (IPCC) in 2014 has recommended that action is required to reduce anthropogenic heating to mitigate emission's to a projected effect on pre-industrial heating to < 1.5 °C between 2030 and 2052. Such an undertaking requires a number of strategies, both active and passive. Passive approaches can be best epitomized as the conserving the earth's self-sustaining natural biological carbon sinks, or an approach which reduces current emissions by restoring post-industrial losses of sinks. Of those carbon sinks, the important "blue carbon" coastal fundamental ecosystems, namely, seagrass, mangroves and saltmarsh receive increasing attention. First order assessments indicate that around half of the oceans total carbon storage securely buried within blue carbon habitat's sediments and biomass (Duarte et al., 2005; Thorhaug et al., 2009; Fourqurean et al., 2012; Alongi et al., 2016) in < 2% of the area of the ocean. The systems then are both valuable plus

* Corresponding author.
E-mail addresses: athorhaug@msn.com (A. Thorhaug), JohnBarry.Gallagher@utas.edu.au (J.B. Gallagher), susandorward@gmail.com (S. Dorward).

https://doi.org/10.1016/j.marpolbul.2020.111168
Received 27 November 2019; Received in revised form 16 March 2020; Accepted 11 April 2020
Available online 10 October 2020

manageable, but also clearly vulnerable and fragile. Global "Blue carbon" storage losses are at an estimated cost to society from environmental with economic damage, plus a direct loss of their ecosystem services, between $US 6–42 billion annually (Pendleton et al., 2012). Our review is the first integration of the tropical seagrass organic carbon nation by nation compared with national mangrove carbon in SE Asia. The global oceans distribution of "blue carbon" tend to be extensive within estuaries, which in the tropics contains two regions of greatest potential and enclosure to store carbon: the Pacific's Southeast Asia, and the Greater Caribbean Sea (including Gulf of Mexico). Other continents containing tropics, tend to have open oceans juxtaposed next to their tropical estuaries such as South America, Africa, or the Indian subcontinent.

Nowhere are tropical blue carbon stocks as dense, extensive or under greater anthropogenic pressure than within the bounds of the Southeast Asia region for mangroves (Giri et al., 2011; Spalding, 2010) and seagrasses (Fourqurean et al., 2012; Alongi et al., 2016; Kaufmann and Bhomia, 2017; Fortes et al., 2018). However, traditional boundaries of Southeast Asian take into account only parts of the socio-ecological connections that effect change throughout such a regional ecosystem. For the purpose of this review, the regions' blue carbon coastlines are gathered through both overlapping and common climatic, geological, oceanographic (Fig. 5), plus socioeconomic drivers. For this blue stock assessment our operative definition of Southeast Asia is discussed in Section 1.1 which is from the Philippines to Myanmar in the north while Papua New Guinea to tropical northern Australia, in the south function as boundaries (Fig. 5). Northwestern coastline of tropical Australia is included by virtue of its proximity to Indonesia and through the common influence of the Indonesian "through flow" (ITF) (Wyrtki, 1961, 1987). This "Through flow" current redistributes heat, salts, nutrients and gases, while a great deal of the surface waters of the Sunda plate is effected as the "through flow", as the Pacific equatorial current circulates around the islands of the Southeast Asian archipelagos then exits into the straights immediately north of Australia (Ayers et al., 2014). Additional inflows, are reinforced by the large catchment sedimentary inputs from the highlands of the Asian continent that overlap inputs radiated throughout the terrestrial regions of the central island archipelagos of Borneo, Indonesia, Philippines, the Island of New Guinea, and the Malaysian Peninsula. Finally, much of this coastal area is influenced by the regions' common development goals as nations are members of the Association of Southeast Asian Nations (ASEAN) (inclusive of Australia), which attends to regional scale management, including environmental issues and policies.

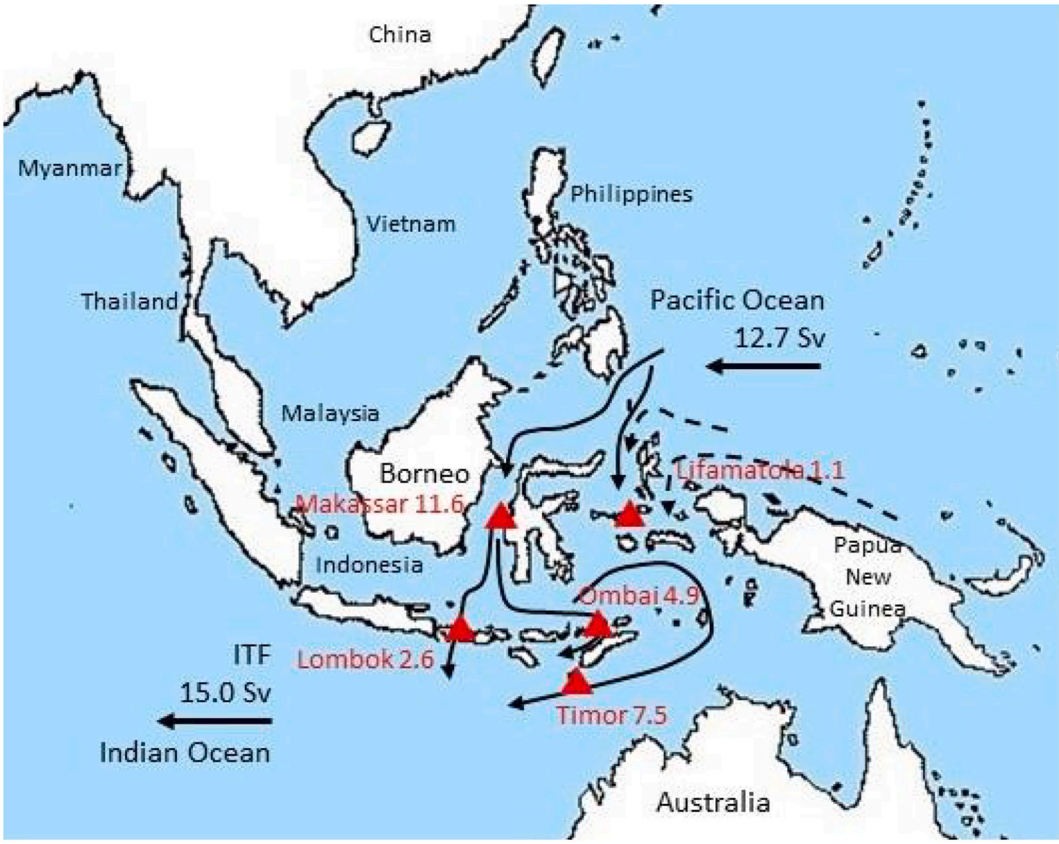
With great power in the ability to store carbon, comes great responsibility in the face of vulnerability. Southeast Asia has regionally experienced both a rapidly expanding population and a large economic development. As a result, coastal development of infrastructure, pollution, catchment eutrophication, and soil erosion has accelerated decimation of seagrass and mangroves. Estimates put mangrove losses at 250,000 ha/yr in Indonesia alone (Giri et al., 2011; Hamilton and Friess, 2016, Spalding, 2010), and seagrass losses at around 7% y^{-1} or more or 20,000 ha/yr (Waycott et al., 2009, Fortes et al., 2018). In total SE Asian mangrove loss is stated to be the greatest reduction in any blue carbon ecosystem across the globe (Giri et al., 2011), which amounts to > 2.5 Million ha from 2002 to 2011. The loss effect remains indeterminate on the current carbon stock regionally mitigation services. The region has not been well served with seagrass or seagrass physiological studies (Duarte et al., 2013). The focus has been mainly at a few sites in the northern Philippines, South Sulawesi, Java, Kalimantan, Hainan Island, China and West Thailand (Ooi et al., 2011; Duarte et al., 2013). Consequently, recent compilations of seagrass carbon stock densities may suffer from under sampling and without context for total blue carbon stocks when used to calculate the total stock across the whole region (Fourqurean et al., 2012; Alongi, 2014). An example are recent compilations across Indonesia's Kalimantan and Sulawesi (Alongi et al., 2016), although the only inclusive for both seagrass and mangrove stocks accounting. For mangroves alone, carbon stock density assessments have been generally far more detailed and ubiquitous across the region (Hamilton and Friess, 2018, Spalding, 2010, Giri et al., 2011). These total extent mangroves carbon stocks accountings, while based on variance across nations, only includes parts of this Southeast Asian socioecological/geographic/oceanographic space, omitting Northern Australia and the nation of Papua New Guinea (Giri et al., 2011). Despite drawbacks, studies to date, clearly demonstrate the tropical Southeast Asian region is by far the richest of the blue carbon regions. Furthermore, the region likely supports the largest total carbon stock connected across global tropical regions, if not in the value of their stock densities (Kaufmann and Bhomia, 2017).

We suggest herein, a need for up-to-date assessment of integrated seagrass and mangrove carbon within a more clearly defined socioeconomically and oceanic-geographically connected Southeast Asian region. Such a compilation would function as a useful rhetorical tool for long term Tier II greenhouse gas emission assessments (Kennedy et al., 2013) for both the regional and the national uses. This integration of seagrass and mangroves forms an important first step in preservation and restoration of their stocks in a format that can be incorporated relatively easily into traditional economic models (Gallagher, 2017), as well as an important component of an integrated marine Southeast Asian emission models (Liu et al., 2018). Importantly, other less tangibly costed natural capital services are returned rapidly as the ecosystems are restored (McLaughlin et al., 1983; Bell et al., 1993; Potouroglou et al., 2017; Thorhaug et al., 2017) not seen using artificial means. That is to say, services directly affecting society: fisheries nurseries (sports, subsistence and commercial), shoreline resilience, biodiversity habitat and endangered species, mineral recycling (Costanza et al., 2017), and water clarity. Indeed, such service inventories have been updated recently for the East Coast of North America and its Gulf of Mexico (GoM), as temperate and tropical/subtropical examples (Herrmann et al., 2015; Thorhaug et al., 2017; Najjar et al., 2018; Thorhaug et al., 2019). Global analyses of restoration show far fewer investigations to tropical seagrasses than temperate (van Katwjik et al., 2016), with Indo-Pacific studies fewer than in the Atlantic.

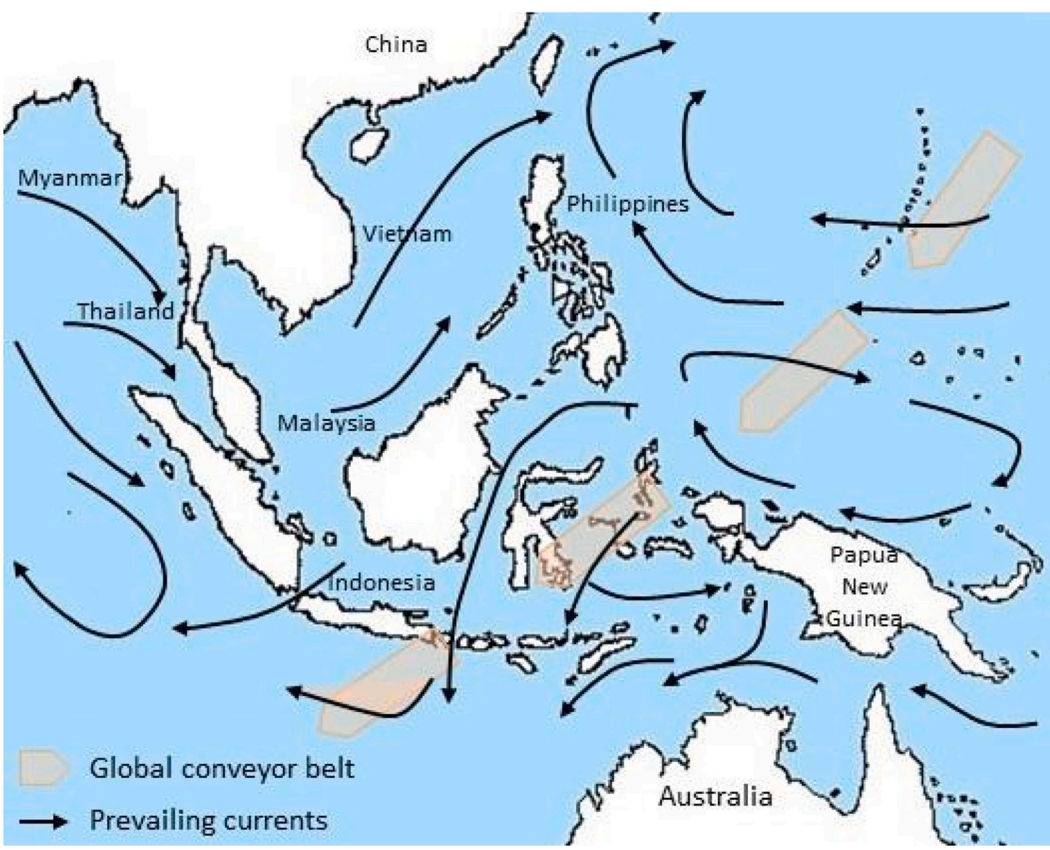
Effective long term Tier II management also requires a knowledge of how regional stock variability has been affected by stochastic and pressed drivers on both the aerial stock density and the loss of aerial coverage. Stochastic factors are by their nature relatively unpredictable one-off events that can have short-term detrimental impact from intense storms, sedimentation from local floods plus periodic volcanic eruptions, earthquakes and tsunamis. Pressed natural drivers of stock density and total extent are the following: 1) Natural drivers (edaphic climatic, geological and oceanography regional parameters); and 2) Anthropogenic drivers (human-induced changes or human introduction of new pressures not as yet fully experienced by these ecosystems).

1.1. Climate, hydrology/oceanography and lithological parameters across the Southeast Asian marine and estuarine region

Southeast Asian region lies squarely with the tropics (refer to Fig. 1a & b), enhancing 12 month productivity by greater exposure to ambient light plus relatively warm constant water temperatures throughout the year, creating greater carbon sequestration. The Southeast Asia region has other notable drivers potentially adding to carbon stocks: increased primary productivity and thus net sequestration from additional supplies of CO_2 and inorganic nutrients from the warmed waters of Pacific equator (Wyrtki, 1961, 1987). As the "Flow through" flows westwards, it is warmed before splitting across the Southeast Asian archipelagos through to the Indian Ocean at a rate of 12.7 Sverdrups y-1 (Wyrtki, 1961, 1987). Its influence is high throughout the east portion of the Sunda Plate (Fig. 2), and through the South China Sea to the Java/Sumatra Straits (refer to Fig. 1a & b), while being less along the Asian mainland. In this transit, vigorous tidal exchanges and current shear over the rough bathymetry of the archipelago entraining nutrients to the clear surface waters in what appears in proportion to the ITF influence across the more eastern sectors (Ayers et al., 2014). Other sources of seasonal upwelling are evident at the borders of the narrower parts of the Sunda shelf off Northwest Borneo (Abdul-Hadi et al., 2013) and the northern parts of the Malacca straights (Siswanto and Tanaka, 2014). While notable, the extent of each upwelling area is relatively small, whereas



a



b

(caption on next page)

Fig. 1. a) The Southeast Asian 'Flow through" (first defined by Wyrtki, 1957) from Pacific Equatorial Current to Indian Ocean showing Island Archipelagos and mainland tropical SE Asia.(drawing by Yap). b) SE Asian "Flow Through" currents (Wyrtki, 1987) including monsoons changes of currents and more details of currents including global conveyor belt and South Pacific equatorial current (drawn by Yap).

in the case of the Malacca Straights, productivity maybe be restricted by its relatively turbid waters from a high density of riverine inputs (Siswanto and Tanaka, 2014). Nevertheless, the sum of the overall combination of a fortunate set of circumstances of a complex landscape, plus local upwelling, along with a moderate wind regime results in a productive landscape born of extensive archipelagos. Local coastal upwelling is fed by a rich oceanic current (ITF), accelerated as it is constricted and turned through the complex landscape while maintaining a less than well-wind-mixed thermocline, in a tropical environ replete with light and upwelled carbon-dioxide. These are the counter factual conditions of stability and upwelling set for a potential water column productivity optimum (Pingree et al., 1975). Taking geography and coastal processes together: it is an area that can potentially support a large mangrove, seagrass, and coral reef coastal niche, to both establish and self-sustain this niche (Guannel et al., 2016); and an area where nutrient supply directly or indirectly significantly contributes to productivity of seagrass and often nutrient-limited mangrove growth within both their adjacent coastal zones, sheltered embayments, lagoons, and estuaries (Reef et al., 2010; Cloern et al., 2014). The relatively larger residence times in these semi-enclosed systems can be expected to retain demineralized nutrients from the inputs of the coastal phytoplankton, which have previously utilized a fraction of coastal nutrient supply (Cloern et al., 2014). With an increased productivity and extent comes an increase in the supply of litter and denser canopy and more efficient trapping of both allochthonous and autochthonous detritus in the more sheltered systems. Furthermore, where light is plentiful and nutrients drive productivity but do not saturate the response, the fraction of net plant production diverted to the production of excess metabolic carbon can be enhanced the consequence of the plant's elevated C/N,P ratio (Sterner and Elsner, 2002). This can direct a reduction in either herbivory that may augment sedimentary litter supply or indirectly through an increase in egestion and in general a relatively high C/N,P ratios synonymous with smaller rates of mineralization. Interestingly, high nitrate levels in soils are also synonymous with low rates of sedimentary decomposition of the more recalcitrant organic matter (Fog, 1988), marked in part by its high C/N,P ratios (e.g., lignocelluloses and soils characteristic and captured by canopy ecosystems). Although, this has not be tested within marine systems, the result would conceivably further add to the litter sedimentary carbon stocks, not forgetting material trapped after the erosion of abundant amounts of catchment sand, clay and volcanic soils (Gupta, 1996; Green and Short, 2003; Spalding, 2010; Giri et al., 2011; Alongi et al., 2016; Fortes et al., 2018), with possible contributions from highly organic washout from freshwater coastal peatlands (Braun et al., 2019). (Extensive peatland in this Southeast Asian region are concentrated around Sumatra, Malay Peninsula, Borneo, and Western Papua New Guinea.) Within these boundaries lie the nation states of Myanmar, Thailand, Malaysia, Singapore, Indonesia, Brunei, Papua New Guinea, Cambodia, Laos, Viet-Nam, plus the southern parts of the tropical Chinese Peninsula including Hainan Island. Of course, in the south and east the archipelagos of Indonesia and Philippines with the New Guinea island halves (i.e. East Timor, and Papua New Guinea) are included. All these nations (other than land-locked Laos) are presently attempting to steward large extents of blue carbon mangrove and seagrass resources at various levels of sustainability.

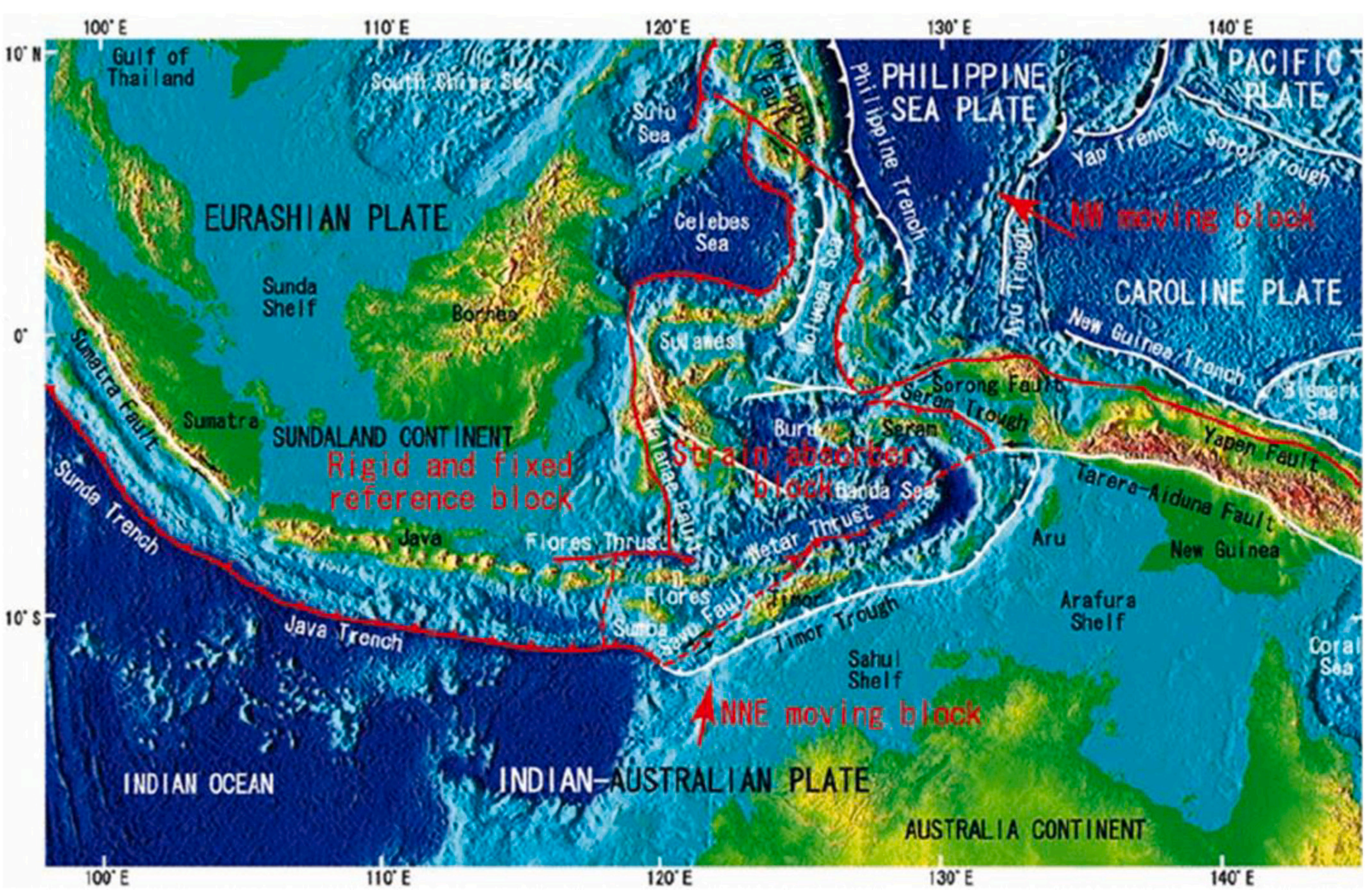


Fig. 2. The Eurasian Plate with Sunda Shelf on which many of the SE Asian nations and archipelagos lie. Surrounding Australian Philippine and Caroline plates emerge. The area also includes the Philippine Sea plate (from Semantic Scholar).

Along with natural drivers of gross productivity and stock accumulation, anthropogenic impacts have likely been both sufficiently persistent and extensive over time as to effect both change to the amount of carbon stock accumulated or in some cases its near complete loss (Pendleton et al., 2012). The impacts come from a region that is a complex mix of individual nation histories and differing stewardship approaches, many sprung from the 1972 Stockholm Conference on the Environment. The Stockholm Conference pointed out overlaid regional industrial, extraction mining and drilling, agricultural, fisheries, and aqua-cultural development, which contributes to the gross economic benefits to various nations' use of the estuaries and coasts (Costanza et al., 2017). Shipping is a key Southeast Asian regional industry, and further cements the connectivity across the region, which includes intensive movement of petroleum from the nearby middle-east and regional Asian oil fields through the seas to Japan, South Korea and eastern Chinese Ports. Urbanization is nowhere more intense than in the Chinese Megalopolises', in Singapore and increasingly in Penang Island. Here the impact on seagrass and mangroves is composed of shoreline port and industrial, urban development, with substantial dredge and fill practices, plus plastics and garbage in confined areas such as the Malacca Straits (not regulated until a decade ago as noted by Chee et al., 2017). This resulted in little natural seagrass remaining in urban and industrially developed areas, despite being surrounded by the densest and most productive seagrass beds in the world. It is also important to understand that losses of mangroves and seagrasses are not just confined to industrialized and urban hotspots. Fishermen's disruptive practices (e.g. bottom trawling for shellfish) also decimates nearshore seagrass. Agriculture, ubiquitous across the region, has led to increasing soil erosion causing turbidity and loss of seagrass in and around riverine mouths, especially from the extensive rivers that direct erosion of soils due to poor stewardship (Van Katwijk et al., 2011). The removal of mangroves for aquaculture developments continue to be implemented without consideration of the benefits of natural mangrove ecosystem services (Sidik and Lovelock, 2013; Fortes et al., 2018). Indeed, anthropogenic events have led to a patchwork of seagrass and mangrove habitats, coupled with mining runoff, war bombing, and other collateral damage, plus substantial effluent dumping, herein outlined regionally in terms of loss per type of impact in Tables S-6.

Along with more proximate causes of loss, distal causes from the effects of climatic change are not as clear. Nevertheless, all things being equal, it can be argued that as long as light remains non-limiting to growth (i.e., no increase in coastal turbidity), and water temperatures do not exceed physiological limits, then increases in the $p\mathrm{CO_2}$ from atmospheric flux and acidification will continue to enhance the production of excess plant carbon, synonymous with increases in sedimentary stocks (see Tables 1, 2, 3 below). A significant sea-level rise is one of the major anticipated consequences of climate change. Estimates of global average sea-level rise are around 1–1.5 mm $\mathrm{year^{-1}}$ during the twentieth century. It is estimated that there has been an increase in this rate in more recent times—the average rate for the past 25 years being ~2.1 mm $\mathrm{year^{-1}}$ (Church et al., 2001; Pugh et al., 2002; Church and White, 2016). Global mean sea level is expected to rise by 18–76 cm by 2100 (IPCC, 2007). It is expected that any sea level rise effects intertidal species such as mangroves more than the subtidal seagrass. Seagrasses extent will change less at the intertidal edge, growing shoreward, but may lose extent at the deeper edge due to required light. In total, Southeast Asia is experiencing a significant loss in blue carbon habitats which needs addressing to estimate total carbon loss from mangroves and seagrasses combined.

1.2. Objectives

This review attempts to examine Southeast Asian regional patterns in both total national blue carbon stock density and areal extent data of seagrass plus mangroves that may show consistency with previous stated postulates on the control of stock estimates, natural and anthropogenic. The tropics' blue carbon is studied less than the temperate zone, and needs delineation in Southeast Asia particularly. We then attempt to delineate the potential for net sequestration and storage of organic carbon per nation by the combination of both seagrasses and mangroves presently in the Southeast Asian region. We view the future scenarios by quoting our finding (Thorhaug et al., 2020a, 2020b) of the potential for seagrass restoration along with others' estimates for potential mangroves' restoration. We caution the results of this study and others in that this concept may ultimately confound mitigation services should the gradient across the region be counter to stock densities and extents. For a global context, we compare our blue carbon estimates to other tropical/subtropical regions the Atlantic Gulf of Mexico (GoM). Our hypothesis is that both mangroves and seagrasses add substantially to carbon sedimentary deposits across the greater Southeast Asia region, while blue carbon stocks differ widely among nations.

Table 1
Southeast Asian Blue Carbon Stock Estimates from Seagrass and Mangroves.

Mangroves	Extent in hectares	$C_{organic}$ in Mgha-1	Total C_{org} Tg
Seagrass	5,116,032	950.5	3095.19
	6,744,529	251 (range 119-2450) 1371.86	1683.97 This is with top value for Philippines.
Total	**11,860, 561**		**4779.16**

2. Methods

2.1. Blue carbon sequestration and organic carbon stock densities

This paper integrates a series of previously published and some unpublished results about seagrass and mangrove blue carbon values within Southeast Asian nations and their national areal extents to **come to** both national and regional sums. All authors were involved in obtaining the integrated data and charts (see "Acknowledgements" for appreciation of their work) from national, and regional sources including **gray** literature in multiple languages. The present document's authors were also involved in this integration of carbon per habitat type, while many investigators were involved in obtaining new carbon stock and areal extent estimates, especially for the previously sparse seagrass data. It should also be noted that only some of the aforementioned studies elucidated the blue carbon stock as an ecosystem service or used as a control the difference with either proximate currently-barren sediments, but with previous vegetation, as well as always barren sediments (75–100 yr). For more details on the use of control types across see Thorhaug et al. (2017). The methods used for estimating blue carbon components within sediments and (C_{org}), vegetation (above and below ground), water and air are described in detail in the original publications. These field methods range in scope from the following: carbon and nitrogen contents in sediment and habitats; their isotopic signatures for biomass, water and sediments used for partitioning of carbon sources and flux rates; the use carbon-dioxide flux towers and radio-isotopic geochronologies for sequestration rates (Donato et al., 2011; Alongi, 2014; Asmoro and Setiadi, 2010; Alongi et al., 2016; Kiswara et al., 2010; Rattanachot et al., 2018; Poovachiranon et al., 2006; Prathup, 2012; Gao et al., 2013; Miyajima et al., 2015a; Husodo et al., 2017; Gallagher et al., 2020, UNESCO, 2012; Jiang et al., 2019; Kiswara, 2018; Nienhuis et al., 1989; Liu et al., 2013; Liu et al., 2018; Ha et al., 2018; Gacia et al., 2003). In order to fully integrate total carbon stock of all regional nations, we have tailored the extent measurements (discussed below) in Tables 2 and 3 mean levels and ranges, and vegetation densities with sedimentary C_{org} estimates derived from neighboring nations' measurements only for Cambodia, Myanmar, PNG, and Brunei, where carbon metrics have not yet been established.

2.2. Extent of mangroves and seagrass

The aerial coverage for both mangroves and seagrasses per nation contain various degrees of inaccuracy. Problems arise from the age of the data and area perimeters that may not overlap or include all zones,

Table 2
SE Asian mangrove stock per nation. Extent data is from Giri et al. (2011), Hamilton & Friess (2018), Spalding (2010), PEMSEA (2016), Breithaupt et al. (2012), Lamit et al. (2017) for mangroves. A series of investigators have contributed to sediment carbon data for stock at 1 meter including Alongi et al. (2016) (for mangrove have 774.7 Mg C ha-1, although others state the median carbon storage in an Indonesian mangrove forest is 950.5 Mg C ha-1), Breithaupt et al. (2012), Hamilton and Freiss (2018), PEMSEA, (2016), BOBLME, 2015, Giardino et al. (2016), Chew and Gallagher (2018), Liu et al. (2013), Li (2017), Jiang et al. (2017), Kindbohom (2016), Lamit et al. (2017), Pulhin and Gevana (2010), Yee et al. (2010), Shearman (2010), FAO (2003). Extrapolated to the total estimated mangrove area of 31,894 km2.

Mangroves Nation	Extent in hectares	Organic Carbon in Mgha-1	Sedimentary Carbon Stock Corg Tg	Citations for extent and buried organic carbon (1st m)
Indonesia	2,707,572	950.5 Range:174.1-1169	2573	Giri et al., 2011; Alongi, 2014; Alongi et al., 2016; Hamilton & Friess, 2018; Breithaupt et al., 2012
Malaysia	558,58	246.21 non-degraded Range: 176.32 degraded-151.40	137.5	Giri et al., 2011; Alongi et al., 2016; Hamilton & Friess, 2018; Liu et al., 2013; Chew and Gallagher, 2018; Breithaupt et al., 2012
Thailand	245,121	141.53 Range 90-808	34.69	Giri et al., 2011; UNEP, 2010; Breithaupt et al., 2012
Brunei	11.1	176.32	1.96	Spalding, 2010; Giri et al., 2011; Lamit et al., 2017;
Myanmar	494,584	176.32	87.21	Giri et al., 2011; BOBLME, 2015; Kindbohom, 2016; Giardino et al., 2016
Cambodia	47,572	176.32	8.39	Spalding, 2010; Giri et al., 2011; Hamilton & Casey, 2016; Hamilton & Friess, 2018; PEMSEA, 2016
Viet-nam	215,529	159.45	34.37	Spalding, 2010; Giri et al., 2011; Breithaupt et al., 2012; PEMSEA, 2016; Huong et al., 2017; Hamilton & Friess, 2016, 2018;
Tropical China	95,100	355.25	33.78	Zhou et al., 2010; China Forestry, Wang et al., 2013; Breithaupt et al., 2012; Liu et al., 2013; Alongi, 2014; Li, 2017; Jiang et al., 2017
Philippines	259,037	184 Range 102-576	47.66	Philippine Forestry Dept; Giri et al., 2011; Pulhin and Gevana, 2010; Hamilton, 2016
Singapore	659	176.32	0.116	Yee et al 2010; Shearman, 2010; Giri et al., 2011
Papua New Guinea & Timor Leste	481.2	283.7	136.51	FAO, 2003; Spalding, 2010; Shearman, 2010; Giri et al., 2011; Breithaupt et al., 2012; Hamilton & Friess, 2016, 2018; Alongi et al., 2016; Alongi, 2014
Total	**5,116,032**		**3095.19**	

Table 3
Sedimentary Carbon Stock for SE Asian Seagrasses. Extent data for SE Asian seagrass is from Green and Short (2003), Fortes et al. (2018), Alongi et al. (2016), Lamit et al. (2017), Yaacub et al. (2013), and Neihaus (1993). Carbon data is primarily from Alongi et al. (2016), Kiswara (2018), Miyajima et al. (2015b), Yap and Gallagher (2018), Chew and Gallagher (2019), Prathap (2010), Poovachiranon et al. (2006), Raatanchoot et al. (2010), Gacia et al. (2003), Hutchinson and Freiss (2016), Luong et al. (2012), Ha et al. (2018), Liu et al. (2018), and Ren (2017), Pulhin and Gevana (2010), Patty (2016), Kawaroe et al. (2016), Fitrian et al. (2017), Wawo, (2017), Jiang et al. (2017), Huang et al. (2016), Shu et al. (2017), and Li et al. (2016).

Seagrass	Extent in Ha	Organic Carbon Mg ha-1	Sedimentary organic carbon Stock Tg	Citations for extent and buried organic carbon (1st m)
Nation				
Indonesia	3,000,000	251 Range 62-2450	753	Neihaus, 1993; Green & Short, 2003; Alongi et al., 2016; Kiswara, 2018; Patty, 2016; Kawaroe et al., 2016; Fitrian et al., 2017; Wawo, 2017
Malaysia	16,300	251	4.1	Green & Short, 2003; Yap et al., 2018; Chew and Gallagher, 2018; Jiang et al., 2020; Patty, 2016; Kawaroe et al., 2016; Fitrian et al., 2017, Wawo, 2017; Liu et al., 2018; Spalding et al., 2001.
Thailand	14,800	155	2.29	Poovachiranon et al., 2006; UNEP, 2008; Raatanchoot, 2010;
Brunei	1,500	96.79	0.145	Lamit et al., 2017
Myanmar	430	96.79	0.042	BOBLME, 2015; Raatanchoot, 2010 Giardino et al., 2016
Singapore	300	96.79	0.029	Yaakub et al., 2013; Fortes et al., 2018
Cambodia	32,490	96.32	3,.144	Bouillon et al., 2008; UNEP, 2008; Poovachiranon et al., 2006; Raatanchoot 2010; Giardino et al., 2016
Viet-nam	15,740	159.45	2.50	Luong et al., 2012; Ha et al., 2018; Huong et al., 2017; Spalding et al., 2001
Tropical China	5,218	159.45	0.62	
Philippines	2,726,200(Fortes) Or 1,500,015(Green& Short)	251 Range 101-650	684.3 or 376.5	World Bank, 2005; Giri et al., 2011; Gacia et al., 2003; Green and Short, 2003; Fortes et al., 2018;
PNG/OETimor	931,551	251	233.8	
Total	**6,744,529**	**601.9**	**1682.3 or1371.86* (*smaller Philippine extent)**	

especially extents of deeper seagrass habitat. Satellite imagery extents of mangroves are more exact than seagrass, mangrove images being captured far more easily. Nevertheless, we utilized our own and other investigators' blue carbon habitat areal extent data cited in Tables 1, 2 and 3 (FAO, 2003; Green and Short, 2003; Asmoro and Setiadi, 2010; Spalding, 2010; UNEP, 2010; Shearman, 2010; Giri et al., 2011, 2015; Hutchinson et al., 2014; Alongi et al., 2016; Hamilton and Casey, 2016; Hamilton and Friess, 2016; Lamit et al., 2017; Huong et al., 2017; Fortes et al., 2018; Jiang et al., 2018). The areal extent of blue carbon habitat type has been mapped excellently in some locations by the national governments utilizing remote image capture with ground truth including recent areal measurements of mangroves (Tables 2, 3). However, there remain a discrepancy in numbers of national aerial estimates for total mangroves across Southeast Asia which create estimates ranging from 4 million to 6.4 million hectares (Spalding, 2010; Shearman et al., 2009; Giri et al., 2011; Giri, 2016; Hamilton & Friess, 2018; Hamilton and Casey, 2016). For this study the regional extent of mangrove forests were taken largely from Giri et al. (2011), and, in part, constrained by these authors' projected annual losses. The Giri et al. study used over 1000 Landsat images at 30 m resolution. This was augmented with less extensive studies but with more careful satellite metrics. This is in the recognition that some of newly-developing nations treated ground truth differently, so that integrations are difficult to access. The extent of seagrasses was more problematical. We recognize that aerial assessments may contain a great deal of error (Huong et al., 2017). This derives from: 1) difficulty of depicting submerged seagrass in often turbid waters (Hedley et al., 2017); 2) the varying decisions of estimates by investigators concerning inclusion of portions of hectares covered in degraded, and/or spotty meadows; and 3) lack of the correct satellite imagery for some areas (more correctly found in Hedley et al., 2017) which generally is far less accurate than the above sea level mangrove estimates, easily captured remotely. Nevertheless, for seagrasses, we used extent data derived from a number of sources (Green and Short, 2003; UNEP, 2010; Spalding, 2010; Alongi, 2014; Alongi et al., 1998; Giardino et al., 2016; Fortes et al., 2018; Huong et al., 2017; Jiang, 2018) to an estimate aerial coverage across the region's nations. Where we found large discrepancies in the extent of seagrass coverage between studies, [note particularly between Green and Short, 2003 versus Fortes et al., 2018 for the Philippines]. We chose to list in Tables 1 and 3 both values and then use in summaries use the higher value of Fortes et al. (2018). How do extents vary as to position in estuary? In general the lower energy areas of sites with high nutrient content have higher densities of seagrass and higher sequestered organic carbon unless anthropogenic disturbance is present. have shown this result measuring within a double-branched Malay estuary containing both mangrove and seagrass.

2.3. Carbon stock measurements

The amount of carbon stock data for mangroves both historic and more recent, is reviewed by several, however, no total SE Asian regional with subcalculations of nationally-tailored carbon stock of seagrass has been made. The attempt to synthesize Southeast Asian Blue Carbon is urgent and it would serve no purpose to wait for complete inventories of national estimates, especially for seagrasses. For this study, data for national carbon accounting were taken from a number of differing investigators, and necessarily from studies that focused on different questions with differing metrics for the depths of carbon, although most were 1 m. The carbon data used for calculation per nation was take for mangroves from the following sources: Alongi (2014), Alongi et al. (2016), Hamilton and Friess (2018), Breithaupt et al. (2012), Chew and Gallagher (2018), Lamit et al. (2017), Hamilton and Casey (2016), PEMSEA (2016), Wang et al. (2013), Liu et al. (2013), Li (2017), Jiang et al. (2017), Pulhin and Gevana (2010), Spalding (2010). The carbon sources for seagrasses were taken from these sources: Alongi et al. (2016), Neihaus (1993), Kiswara (2018), Patty (2016), Kawaroe et al. (2016), Fitrian et al. (2017), Wawo (2017), Yap et al. (2018), Chew and Gallagher (2018), Jiang et al. (2020), Liu et al. (2018), UNEP (2008), Rattanachot (2010; Giardino et al. (2016), Ha et al. (2018), Huong et al. (2017), Gacia et al. (2003), Jiang et al. (2017), Huang et al. (2016), Shu et al. (2017). Therefore, this report utilizes for mangrove and for seagrass the mean carbon stock density at 1 m of all national samples times the aerial national extent summed across each nation, realizing that in many areas much more carbon at depths greater than 1 m is sequestered. In all cases, including the tables, we reference the size of the sedimentary stock to a depth of 1 m. This is a conservative measure. All things being equal, this is considered to be the depth to which canopy and root system of seagrass protects the carbon stock from remineralization (McLeod et al., 2011; Serrano et al., 2016). For mangroves, it is highly possible with some species that they possess a deeper root system necessitating a differing deeper metric to estimate the total organic carbon (Fujimoto et al., 1999, López-Portillo et al., 2018). Of course not all things are equal in the world of disturbance and depth of sediments, and shallow areas built on a layer of rock just inches under the seagrass or mangroves (e.g. northwestern Mexico) are not accounted for in the 1 m depth. Indeed, variability in how much of that disturbed organic carbon would have been mineralized is a source of local bias. To approximate that assessment, Thorhaug et al., 2017 made use of long-term available, nearby non-vegetated sediments (Section 2.1). In these base organic carbon measurements taken from the literature the discussions of "Black carbon" had not been incorporated into the resultant numbers. Therefore, our resulting compilation of blue organic carbon does not handle black carbon, although we discuss what proper measurements should in the future be carried out and should be compiled. Only in some studies such as Gallagher et al. (2019) is this carried out, which are small compared to overall, and not in Indonesia, the largest compilation of organic carbon. Other conceptual errors may arise from the amount, the origin and the role of particulate inorganic carbon. For the moment, the science on inorganic carbon stock and sequestration services is not clear (Gallagher, 2017), and beyond the scope of this and other past regional estimates. To that we add a comment based on limited availability of its sedimentary contribution data in context with well-known emissions factors across the region. Exclusion or inclusion of other carbon forms such as the inorganic calcareous components are not as clear. Disturbance of blue carbon sediments, the arbiter of how stock mitigation services are assessed, leads to carbonate oxidation and dissolution (Ware et al., 1992; Howard et al., 2017). This results in an increase in dissolved inorganic carbon (DIC) considered as a long term atmospheric sink, which for mangroves are known can be advected laterally from sediments into adjacent waters (Maher et al., 2018). However, black carbon mangrove and seagrass data is presently confined to parts of North Borneo, with one mangrove study reported from the subtropical Chinese coastline, and is only discussed but not included in the final accounting. Other errors in analysis arise from the investigator, from collection and sample representation (as seen in the collection site gradient in back reef in Tanzania as explicated by Belshe et al., 2018, and for example a series of samples across Queensland estuaries by Serrano et al., 2016), and in many cases, from the source of the calibration used from converting organic matter to organic carbon.

3. Results and discussion

3.1. Mangrove and seagrass total organic carbon stocks for Southeast Asia

Mangroves (5,116,032 ha) and seagrass (6744529 ha) appear to have roughly similar aerial coverage across this Southeast Asian region (Tables 1, 2, 3), when account is taken of the paucity of well-considered surveys and the greater errors and decreased accuracy in remotely measuring seagrass in turbid conditions in this region (Hossain et al., 2016). Furthermore, total carbon stock densities were, in general, greater across mangroves (mean 950.5 in Indonesia, range 174.1 to 1169 Mg ha^{-1}) than that of seagrass (mean 251 in Indonesia, range 62 to 2450 Mg ha^{-1}) (Tables 2, 3 and Supplementary Table S-4). The effect was to increase the difference between the means and their regional total organic stock capacity to 3095.19 TgC for mangroves and 1683.97 TgC for seagrass (Tables 2 & 3). In other words, on an average around 65% of SE Asian total blue carbon comes from mangroves, when constrained to the first meter of sediment deposition. This results, as an average, show a 4779.16 TgC total blue carbon regional capacity. The first meter of sediment has been assigned, for now, as the disturbance depth parameter and mineralization for an agreed stock valuation assessments, of which Frank Golley et al. (1962) discovered wherein mangrove peat fine roots biomass (< 0.5 cm diameter) exceeded by a ratio of 5:1 all other tree biomass components combined. Although our metrics use 1 m comparing among sites, we emphasize there are no total carbon sedimentary values for total depths at all sites thus 1 m is a conservative estimate. It should be noted Fujimoto et al. (1999) and Macintyre et al. (2004) reported depth of Southeast Asia state mangrove peat varying from 2 m to over 9 m depth. This supports the notion that region's mangrove carbon stores are highly productive with sufficient persistence to maintain high accretion rates with sea level rise over geological periods. A favorable accretion rate constrains the alternative explanation for deep sediments as more ancient, relatively low productive systems with rates of low accretion.

3.2. National total organic carbon stock density and the importance of sediments

Our Tables 1, 2, 3 delineate for the first time the importance of both mangroves and seagrass organic carbon to the regional larger Southeast Asia and to its individual nations in their dual roles as both sequestered carbon's stewards and beneficiaries (Tables 1, 2, 3). The sediments' importance to the stability and contribution of these national Blue Carbon estimates cannot be over emphasized. Firstly, organic sediment contains the majority of organic stocks. Alongi and Mukhopadhyay (2015) have complied in Indonesian selected sites a large organic carbon stock list. The above and below ground organic carbon for seagrasses in Thailand measured by Raatanchoot et al., (2010) shows twice the below ground carbon as above ground (Supplementary Table S-1). The loss of seagrass extent is reported frequently in antidotal form or generalized qualitative estimates (Alongi et al., 2016; Ooi et al., 2011; Fortes et al., 2018) except from aerial imagery investigators. Giri et al. (2011) for mangroves in most of the Southeast Asian nations and Huong et al. (2017) for seagrass in 4 nations attempted to quantify the extent of losses over varying timescales (Supplementary Tables S-5 from Huong et al., 2017 seagrass loss estimates, Table S-6 for mangrove driving force loss estimates data and Giri et al., 2011 for many Southeast Asian nations quantitatively). Secondly, sedimentary organic mineralization is in general more temporally resilient over inter-decadal periods (Pendleton et al., 2012). Note that loss of biomass, generally required for sedimentary organic mineralization, is effectively immediate over a number of months (e.g. for seagrass detritus) (Fourqurean and Schrlau, 2003). Reasons behind mangroves losses are more accessible and visible (Hutchinson et al., 2014). In the most detailed loss estimate, Hamilton and Friess (2016) have quantified specific mangrove losses from various anthropogenic activity over two years for various Southeast Asian nations (Supplementary Table S-6). Oil-palm plantation land and deforestation accounted for greatest losses and the major reason for large loss variability among this region's nations. Loss due to mangrove carbon used as wood, the rate will depend on the use of the product between burning and building materials (Eong, 1993). No such quantitative specific cause loss compilation for seagrass exists.

Due to the large extents in Indonesia, it was found that Indonesia possesses the stewardship to the predominant total blue carbon stock for the Southeast Asian region (Tables 2, 3), supporting on average 53% of the mangrove forests and 45% of the regional seagrasses meadows. The extent of Indonesian estuaries and coastal waters is roughly equivalent proportionately to the extent of total blue carbon stock, but among nations tempered with local very large differences in carbon sequestration between seagrass and mangroves as well as watershed soil deposits enhancing sediments' stocks differentially. On the mainland side large watersheds occur from the Himalayas, or alternatively the Southern Chinese Mountains, with the exception of the Malaya Peninsula where watershed sediment supply is confounded by shorter watersheds. Rapid erosion of volcanic soils and associated soils increases the density of the sediment and with it the carbon capacity with depth. The island archipelagos of Indonesia, the Philippines, and the Island of New Guinea appear to support a large carbon stock density and with it a large fraction of Southeast Asia regions' blue carbon stocks (Tables 2, 3). FAO (1991) found, "Soil erosion is the most important threat to soil in Asia". "Southeast Asia is potentially a region of high erosion and sediment transfer due to several environmental factors, including tectonic movements and volcanism associated with skirting subduction trenches, steep local relief, the mountainous interior of the islands and peninsulas and a very high annual rainfall" (Gupta, 1996).

3.2.1. The importance of biomass to blue carbon

The mangroves throughout the region have both substantial above-ground and below-ground biomass (Supplementary Table S-3). Given this vegetated coverage, seemingly with continuous coverages together with similar edaphic soils across the region, extensive sampling of carbon stocks around the Island archipelagos which experienced the Wyrtki "Through flow", (Indonesia, Philippines, PNG, Brunei), or the mainland Asia (Malaysia, Thailand, Myanmar, Viet-nam, China's Gulf of Tonkin Peninsula) with far less oceanic current showed little variability in sedimentary carbon stock densities ha-1 in those two types of sites (Table 2). This realization is supported across the remaining parts of the region from less extensive sampling. For seagrass, both above and below biomass has been measured (Supplementary Table S-1) wherein above-ground generally is seen to contribute a minor fraction of the total organic carbon stock despite large blade size of a few species. The most-frequently present genera and species are medium sized, throughout the region supporting blades to approximately 20 cm height (*Cymodocea, Thalassia, Halodule, Syringodium*), with proportionally medium-sized rhizome and root densities. Another ubiquitous group has a small biomass (< 5 cm height of multiple species of *Halophila*) (Supplementary Table S-4). While substantial variability occurs among species and within each nation, what controls their carbon stocks is not yet clear. What controls pre- or post-disturbance baseline carbon values has not been estimated by most investigators with exceptions of Thailand, by Prathep (2010), in Borneo by Gallagher et al. (2019), and China, by Jiang et al. (2019). As a consequence, extrapolating examples of carbon for estimates from estuaries outside the region is fraught with danger because different regions will have different edaphic soils and physico-chemical constraints and very different sources of organic carbon such from adjacent riverine and inlet and plankton sources, as well as for the adjacent mangroves (as seen in seagrass/mangrove interaction by Poulos et al., 2018).

Other possible sources of conceptual bias in this and other regional approaches for blue carbon stock estimations is the failure to account for the loss of carbon from the forest or meadow's deposited detritus to the shelf waters' food webs (Gallagher, 2014, Gallagher, 2015), by assuming this carbon has been all been respired (Duarte and Agusti, 1998; Cebrian, 2002). Total riverine carbon flux through estuaries to the China Seas alone is estimated by Liu et al. (2018) as 59.6 ± 6.4 Tg C yr^{-1} and they estimate the total export flux of particulate organic carbon from the upper ocean of the China Seas is 240 ± 80 Tg C yr^{-1}. Additionally, riverine flux estimates are measured by Milliman (1995), and Milliman et al. (2015). More synoptic riverine carbon flux measurements are clearly needed for Southeast Asia as carried out for North America by Butman et al. (2016).

3.2.2. Variability within seagrass species and organic carbon stock densities

It is apparent that it is the seagrass climax species which are larger and relatively slower-growing laterally, dominant *Enhalus acoroides* plus *Thalassia hemprichii, Cymodocea serrulata, C. rotundata,* and *Syringodium isoletiforme,* are associated with the bulk of organic carbon in most Southeast Asian estuaries, not the smaller pioneer species (*Halophila sp.*) [demonstrated in Supplementary Table S-5, "Comparison of the carbon stock densities from 9 seagrass species compared among 5 sites in Indonesia" (Kiswara, 2018)]. A similar pattern occurs in the Indian Ocean west coast of Thailand wherein dominating sedimentary carbon stock densities of *Cymodocea* sp. and *Syringodium* sp. are compared to a lesser biomass of smaller species such as *Halophila sp.* (Supplementary Table S-4; Raatanchot, 2016; UNESCO, 2012). This Thailand biomass study also demonstrates seagrass growth rates on sand vs. mud substrates of meadow expansion (Rataanchot et al., 2016) (Supplementary Table S-2.). Some of the carbon difference is undoubtedly due to greater rhizome/root development in the large dominants (which enhances carbon exudates into sediment from which carbon is directly buried) vs. the more ephemeral above-ground blade growth which is in large part exported by water movements and/or consumed in the food web. Importantly, carbon stock densities of these Indo-Pacific species across back reef to shoreline transects in the Western Indian Ocean (Tanzanian) (Belshe et al., 2018) show large differences in density, dependent on seagrass position to shoreline vs. back reef. How much these species' lateral growth dependence is the cause of carbon stock variability or is simply covariant with the environments' seasonal exposure to fluid dynamic's and other environmental factors is yet undetermined. Faster growing alpha-colonizing *Halophila* sp. and *Halodule* sp. pioneer species occupy more exposed areas over more sheltered environments, where periodic large storm events occur, not supporting maximal carbon burial. Sheltered areas anecdotally appear to support slower growing relatively more stable climax genera such as *Enhalus* and *Thalassia*

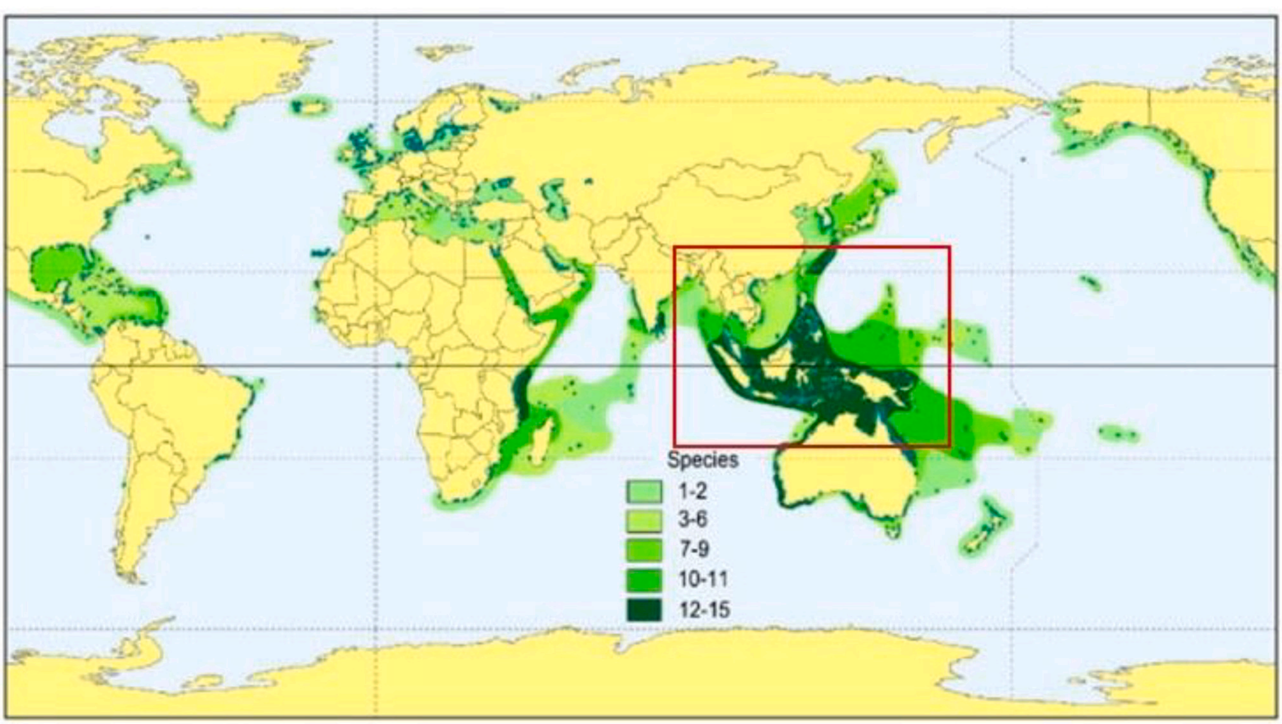


Fig. 3. Seagrass extent map including SE Asia. Dense green means more seagrass species. (from Green and Short, 2003). Red box shows the SE Asian Region. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

Fig. 4. SE Asian Mangroves extent (from Ellison and Strickland 2013).

(Thorhaug and Cruz, 1987; Kilminster et al., 2015; Kiswara, 2018). However, this may be confounded within more sheltered estuarine systems where turbidity, with associated chemico-physical factors, apparently inhibit the presence of all but *Enhalus sp* Gallagher et al. (2020). In summation, a large organic carbon variability occurs regionally among nations (Fig. 5), among seagrasses' (Fig. 3) and mangroves' distributions (Fig. 4), and among genera (Supplementary Tables S-3, S-4). The "Through-flow" (Wyrtki, 1949) nations experiencing the peak effects are Philippines, Indonesia, Malaysia (especially Borneo) and PNG island. For seagrasses "flow through" shows 1674.25 Tg C seagrass and 2942.61 TgC for mangroves. The mainland nations with far less "flow" show seagrass 8.74 TgC and mangroves 464.19 Tg C.

3.3. The exclusion of inorganic and black carbon in blue carbon estimates

More calcium carbonate creation occurs from seagrass metabolism and its environment as measured in the sediment than solely organic Carbon, although herein and most reports of Blue carbon present only the organic portion (Serrano et al., 2016). The role of all calcium carbonate stocks as a mitigation service is not clear when the consequences of sediment disturbance are factored in. During calcium carbonate precipitation carbon-dioxide is released. Consequently, it has been recommended that a fraction (0.63) of biotic calcareous carbon stock should be subtracted from that total organic stock as a mitigation service (Howard et al., 2017). However, the effects of disturbance, the measure of mitigation services (Gallagher, 2017), and resultant oxidation may result in an increase in bicarbonate alkalinity and a reduction in the pCO2 of its water body (Ware et al., 1992; Howard et al., 2017). The result is a water body that continues to act as an atmospheric sink, and compensate to some unknown degree the calcareous stock's previous role as an atmospheric source. On the other hand, in warm tropical/subtropical waters, observations indicate that sediments vulnerable to disturbance may also lead to "whiting" events. These events are the expression of water column calcareous carbonate precipitation (Broecker et al., 2000; Sondi and Juračić, 2010), and presumably, would confound any increase in the partial pressure of carbon dioxide (pCO2) of its water body. Furthermore, biotic calcareous carbonates as found in *Thalassia* sp. need not have resulted in an increased pCO2. Instead, CO2 produced within the lacunae of the blade is fixed as new plant material, as part of a carbon photosynthetic concentrating mechanism (Enriquez and Schubert, 2014).

Black carbon has the potential to contribute a uniquely large sediment component in Southeast Asia as the result of forest and peat fires throughout the region (Field et al., 2016). These stable forms produced outside of blue carbon ecosystems should be subtracted from sedimentary stocks as they do not require the protection afforded by blue carbon sediment deposition (Chew and Gallagher, 2018; Gallagher et al., 2018, Chen et al 2017). The full extent of their contribution remains under sampled. Nevertheless, mangrove data from the regions polluted and unpolluted environs indicate a relatively small contribution to the TOC contents [4.3 ± 1.1 (± 95% confidence limits)] (Chew and Gallagher, 2018). However, this is in contrast to relatively unpolluted seagrass meadows of coastal bays where contributions can rise between 26% ± 4.9 to 36% ± 1.5 (± 95% confidence intervals) (Gallagher et al., 2019). Until the science is convergent, we suggest that carbon stock assessments subtract PICequiv, and measure Black C particularly for seagrass meadows across the more polluted areas of the region. In this way, perverse environmental outcomes may be avoided when traders have effectively been given permission to emit beyond the capacity of the ecosystems' carbon sink (Sophia and Robie, 2016).

3.4. Disruptions and disturbances: anthropogenic and natural

It is clear that the blue carbon of seagrasses and mangroves in Southeast Asia has been and continues to be lost both with resulting habitat being decimated destabilizes sedimentary carbon underneath it and an effect on further production of blue carbon. Fortes et al. (2018) estimated that 50% of the seagrass habitat has been lost in Southeast Asian seagrasses. Giri et al. (2011) state that their investigation with remote imagery indicated a mangrove loss of 250,000 ha per year since 2000 (or 2.5 million ha over a decade) in SE Asia, globally the greatest regional mangrove loss. Alongi et al.'s (2016) estimated blue carbon losses for Indonesia mangrove and seagrass habitat are similar in magnitude to Giri et al.'s habitat loss values. Importantly, Indonesia has by far the greatest mangrove plus seagrass stock regionally. These above authors all attribute most of their loss metrics to anthropogenic disturbances. There is a useful supplementary chart (Supplementary Table S-6) cited from Hamilton and Friess (2016), which quantifies the loss to mangroves' habitat extent from nations Southeast Asia of each major various anthropogenic actions. The mangrove fractions lost are skewed toward direct removal rather than natural disruptions such as typhoons, or collateral disruptions such as soil erosion. When mangrove disruptions occur, then substantial blue carbon sequestration, burial and stock of mangroves above-ground are lost (Pendleton, 2013; Murdiyarso et al., 2015) as shown in several Southeast Asian nations by cutting down mangroves to create shrimp ponds (e.g. 29,000 ha mangroves converted into rearing shrimp in Indonesia (Hamilton and Friess, 2016)). Further mangroves' loss results (death or degradation) in sedimentary back-gassing of methane and other harmful greenhouse gases which are estimated to reach 1135 Mg ha^{-1} (Pendleton et al., 2012; Kauffman et al., 2016, 2014) from sediments below mangrove plus loss of mangrove buried sedimentary carbon itself, and the future mangrove stock and flux. The seagrass organic carbon loss from sediment leaking back into seawater has been shown (Thorhaug et al., 2017) to occur relatively slowly over many years. This loss is being reversed by mangrove and seagrass restoration in many other parts of the world (van Katwijk et al., 2016). This is best shown for the Gulf of Mexico (Thorhaug et al., 2017) who calculated seagrass organic carbon sedimentary loss for the USA Gulf of Mexico as totaling 21.69 Tg C_{org}. Thorhaug et al. (2017) showed seagrass Carbon gain and loss by using a unique method of measuring naturally-occurring (NS), sedimentary carbon beneath restored seagrass (RS), barren-once polluted (PB) sediment carbon, and always barren sediment carbon (AB). Then, NS − (PB − AB) = loss of seagrass organic carbon in sediment at a series of sites. Measurements of continually-low-level impacted natural and restored mangroves and seagrass vs non-impacted areas shows less Carbon sequestration (Thorhaug et al., 2017; Poulos et al., 2018). In preserved areas mangroves' blue carbon stock has been shown to be 246.21 Mg ha^{-1} in Malaysia whereas in disturbed areas it was 151.40 $Mgha^{-1}$ (Liu et al., 2018). Rattanachot and Prathep (2015) found far more carbon sequestration in sediments of healthy seagrass in Thailand than in polluted seagrass. Gao et al. (2018) and Jiang et al. (2019) in China's Hainan Island, where fish mariculture and urban wastes were prevalent, found depressed organic carbon production for polluted seagrass sedimentary organic carbon. Thorhaug et al. (2017) found similarly depressed organic carbon sequestration for continually impacted seagrass around the Gulf of Mexico at 8 sites as Poulos et al. (2018) found for the mangrove *Rhizophora mangle* at 4 restored vs. 4 natural sites in the GoM. More conservation areas may be effective tools for sustaining high blue carbon stock production.

Anthropogenic disruptions to mangroves and seagrass received appreciable investigations, but far fewer measurements are carried out for naturally-occurring event disruptions. Southeast Asia is replete with natural disruptive events: Typhoons, tornadoes, tsunamis, earthquakes, and volcanic eruptions are documented to disrupt the blue-carbon habitats (Thorhaug, 2001; Thorhaug and Wanless, 2001; Nakaoka et al., 2007; Chuan et al., 2019). Both lava flow into coastal habitats, and atmospherically-dispersed volcanic material settling on seawater are all frequent in this region. Where studies exist the focus has been on losses and changes to the seagrass canopy species from a tsunami (Nakaoka et al., 2007) or more recently noting the evidence of previous additional depositional facies within an existing seagrass sedimentary record from a rare storm surge (Chuan et al., 2019). Additionally, there are natural petroleum seeps, in petroleum rich areas such as Borneo, off mainland east Malaysia, and Indonesia. Sufficient quantitative data is not yet available to make a quantitative estimate as to what portion of the blue carbon habitat degradation noted by Giri et al. (2011), Waycott et al. (2006) and Fortes et al. (2018) can be attributed to these natural events, but the above authors indicate far less than anthropogenic disturbances. However, there is doubtless interaction between anthropogenic impacts on environmentally-induced-impacted extents of mangroves and seagrasses such as hurricanes eroding propeller scars (Durako et al., 1992).

3.5. Partitioning of various mangroves and seagrasses in above and below ground organic carbon

Mangroves live in mixed species communities. Supplementary Table S-3 shows the above and below ground portions clearly of various mangroves genera and species in a variety of nations in SE Asia (Alongi et al., 2016). Mixed communities of seagrass are also normal in Southeast Asia. The question, "In which mixture of seagrasses species and variables is the highest sedimentary carbon found?" has a beginning in the studies of Prathup (2016) and Kiswara (2018) and accompanies the question of above-ground and below carbon. Alongi et al.'s analysis points to far more carbon below ground. Raatanchot (2010) shows (Table S-1) twice or more seagrass living tissue below ground, although there are large variabilities among species. The above-ground seagrass blades are continually shed into the water column as are shed mangrove tree leaves, while the below ground appears stable. Stankovic et al. (2017) reviewing carbon among species states, "The highest above-ground biomass found in healthy *Enhalus acoroides* stands at 200 g/m^2. The smallest species, *Halophila ovalis*, provided the smallest biomass. A similar pattern was also observed in the below ground plant parts: the larger plants had a larger below ground biomass reflecting their coarse, tough rhizomes. The highest average below ground biomass was 419 g m^{-2}, while the smallest average below ground biomass was only 1.0 g/m^2 found in the *C. serrulata* stand with 20% coverage." There were significant differences in carbon production between above and below ground among sites, species and coverage. The below ground production, however, was over 2 times greater at 151.16 g/m^2 (Supplementary Table S-1), representing a more significant amount of carbon storage. This introduces the question, "What is the function of the large root and rhizome masses below ground?" Knowing from radiotracer studies that seagrasses can extract nutrients from with seawater pool or sediment-water pools (Schroeder and Thorhaug, 1980) the roots do not totally function as they do in higher plants as the sole pathway of absorption of nutrients and water into the above ground portion. Is part of the root/rhizome chief function a stability factor, especially functionally based on the massive rhizomes and their root structures' bonding with sediment particles?

3.6. Monetary valuing of services of mangroves and seagrasses

In a study of burial of mangrove carbon Lui et al., (2017) in Malaysia have stated, "It was estimated that the minimum carbon credit value for the mangrove forest in the SHD and KSNP was USD \$3,314.23 ha^{-1} and USD \$5,089.83 ha^{-1} respectively, based on the market price in the voluntary market. The undisturbed mangrove forests have a higher potential for economic return in carbon credits". Estrada et al., 2015 have set prices from \$19 to \$68 ha^{-1} for mangrove blue carbon value in Brazil. These values would indicate that the Southeast Asian mangroves multiplied by their extent are worth billions of dollars over the long term to the region and restoring more creates more value. Estrada et al. states, "Considering the area occupied by each physiographic type, the service of carbon sequestration may be worth up to \$455,827 USD yr^{-1}. In regard to carbon storage, \$3,477,041 USD are stored in these forests, and values between \$104,311 and \$208,622 USD ha^{-1} yr^{-1} can be considered as the annual maintenance cost of this service. The income generated by future projects for the maintenance of carbon-related functions may represent a major advance for the conservation of this ecosystem."

4. General discussion

4.1. Discussion of tropical ecosystems' blue carbon with comparisons of areas within Southeast Asia: sources and constraints

Tropical forests, compared to other climatic zones, present the largest area and rates of carbon sequestration across terrestrial ecosystems. This same question of dominance of carbon stocks between temperate vs. tropical regions has not yet been clearly established for the estuarine and marine blue carbon oceanic ecosystems. Although as a first order approximation the annual net productivity of these marine tropical ecosystems is generally regarded as larger, the result of carbon productivity not being interrupted by temperate winters (Thorhaug and Roessler, 1977; Teas, 1977) and producing carbon near optimum much of the year. One clear difference between tropical and temperate is that mangroves are a dominant form in tropical blue carbon. For mangroves, strictly limited to tropical or subtropical climates, their subtropical species can experience occasional decadal winter freezes but tolerate below 0 °C temperatures for only several

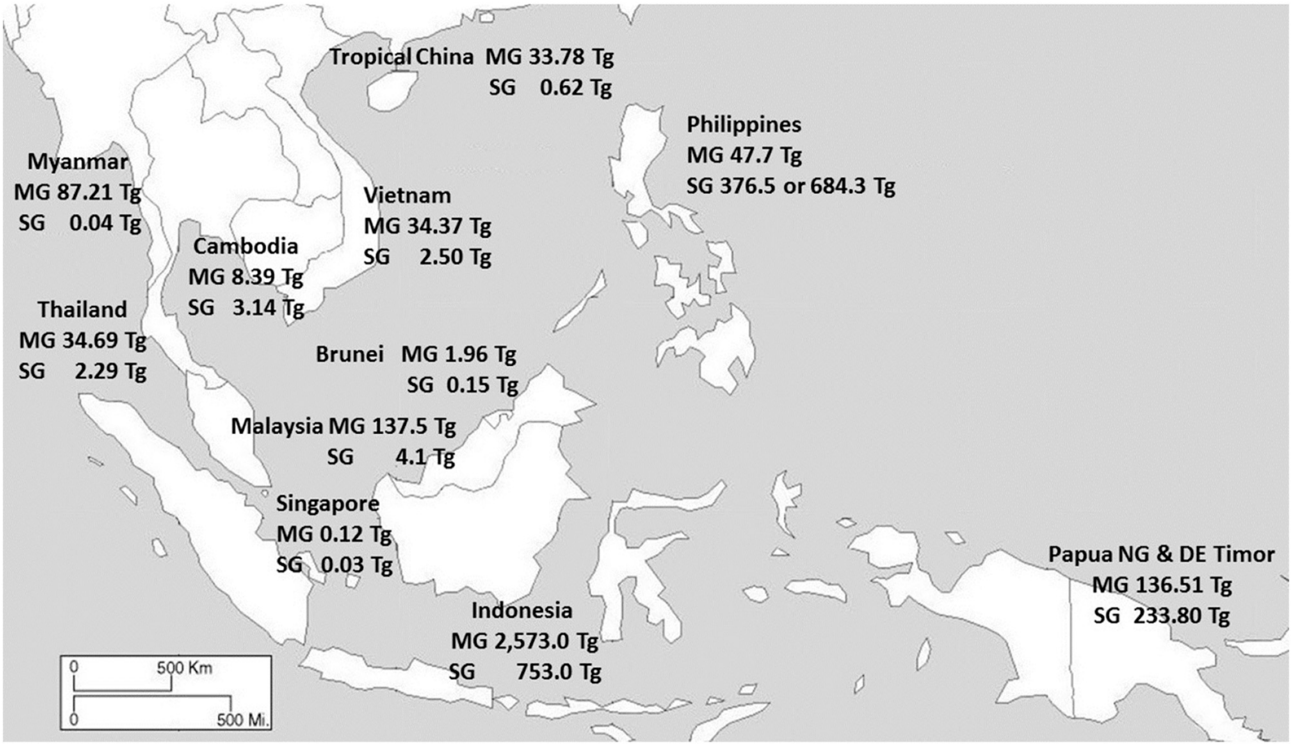


Fig. 5. South East Asia seagrass and mangrove sedimentary carbon per nation in TgCarbon including tropical marine China and the Island of New Guinea.

days before dying (Teas, 1977; Perry and Mendelssohn, 2009; Armitage et al., 2015; Comineax et al., 2012). Furthermore, mangroves' high rates of annual net productivity appear to be accompanied by high carbon storage, irrespective of warmer tropical vs. subtropical winter temperatures or greater rates of organic decomposition (Donato et al., 2011). Seagrasses occupy a far greater climatic range than mangroves, from the tropics to the northern Hemisphere's Polar Regions (e.g. northwestern Alaska, and Hudson Bay, Canada) as would be expected to amplify differences in their sum of contributions to temperate and tropical productivity in the sublittoral zone. The temperate to arctic zone blue carbon replacement for mangroves is by the presence of saltmarsh in the intertidal zone, usually containing less than a third of the C stock density of mangroves (Siikamäki et al., 2012a, 2012b, Thorhaug et al., 2019), most of which is the below sediment.

Nevertheless, high rates of productivity when driven by light exposure, as prevalent across the tropics, results in increases in C/N ratios of plant material. This can theoretically reduce C-usage of plant material across the food web and with it higher rates of organic matter accumulation (Gallagher, 2015). This appears for similar Southeast Asia regional coverage to have resulted in a mangrove mean carbon stock (3095.19 Tg C) considerably larger than our estimated seagrass mean carbon stock (1372 to 1683.19 Tg C), despite mangroves being 24% less extensive. It is hypothetically possible since mangrove canopies photosynthesize with primarily red light with Chlorophylls A and B (Photosystem I) and directly absorb CO2 from the atmosphere, vs. seagrasses, which capture light with a variety of pigments over a wide spectral range (Photosystems I and II) and obtain C from dissolved CO2 carbonate cycle, these physiological differences may affect productivity (Thorhaug et al., 2015, Thorhaug et al., 2016). Clearly, along with the large extent of South East Asia's Blue carbon, compared to other tropical global estuarine areas, some other factors unique to Southeast Asia occurred other than differences in ambient light exposure.

As previously discussed (Section 1.1) the regions' surface waters are continuously fed by Pacific Equatorial ocean currents' extensive "through flow" induced entrainment of waters laden with carbon dioxide and nutrients, and monsoonal upwelling (Section 1.1) with the island Archipelagos and major passes receiving the predominant currents. The "Through-flow" (Wyrtki, 1949) nations experiencing the peak effects are Philippines, Indonesia, Malaysia (especially Borneo) and New Guinea island. For seagrasses "flow through" nations show 1674.25 TgC seagrass and 2942.61 TgC for mangroves. The mainland nations with far less "through flow" show seagrass 8.74 TgC and mangroves 464.19 TgC (Fig. 5). The carbon stock densities for both seagrasses and mangroves are apparently greater within the eastern quarter of the region including Indonesia near "through flow" such as Borneo (Alongi et al., 2016), as PNG, and Philippines, where the Wyrtki "through flow" is the strongest and water column eutrophication is the greatest (Ayers et al., 2014). This appears to differ from nations on main land Asia with lower carbon. If normalized for catchment inputs (as essentially similar across the archipelagos), this is consistent with systems that are nutrient limited and driven by ocean supply and local upwelling. Additionally, the specific watersheds of Southeast Asia are extensive transporting carbon to the estuaries from the Himalayas of Myanmar, Cambodia and parts of Thailand, and Southeast China mountains to Viet-nam. This stimulating input of carbon is opposed to the anthropogenic impact, which is far more intense for the precarious shoreline intertidal existence of mangroves than for seagrasses, although both impacts are extensive. There is decimation by intensive use of shoreline throughout Southeast Asia plus massive pollution from large urban centers (e.g. Jakarta, Manila, Kuala Lumpur, Singapore, Port Moresby), much shrimp farming utilizing former mangrove areas, plus intensive soil erosion from agriculture and mining sources enter into the estuaries (see Supplementary Table S-6 for amounts of impact/nation). The total riverine carbon flux through estuaries to the China Seas is estimated as 59.6 ± 6.4 Tg C yr^{-1} (Liu et al., 2018). There is a very large net carbon influx from the Western Pacific to the South China Sea, amounting to ~2.5 Pg C yr^{-1} (Liu et al., 2018). This is a considerable vector to account for, even if a small fraction is caught by the mangrove and seagrass canopy. The effect is expected to be two-fold. Firstly, the more proximate sentinels of soil erosion, namely the mangrove forests, are in a better position than seagrass to filter out much of the soil loadings and incorporate into their stabilized sedimentary carbon stocks. Secondly, seagrasses are more sensitive to water quality than mangroves, especially turbidity which blocks light (Dunton & Tomasko, 1994) due to most major species existing chiefly immersed in seawater. Thus lack of water clarity is a major factor in the loss of seagrasses (Dunton & Tomasko, 1994, Onuf, 1996, Freeman et al., 2008). Degradation of water quality is not confined to soil erosion. While, petroleum production is more ubiquitous in the western Gulf of Mexico, both Indonesia and Brunei have petroleum drilling and natural oil seepage. This can inadvertently accumulate organic carbon in the sediment (Tedesco and Wanless, 1991; Thorhaug and Wanless, 2019). A final constraint in Southeast Asian sedimentary Blue Carbon has already been discussed above, which is Black Carbon from burning forests, and combustion of vehicles. The problem in restoring mangroves to regain carbon is that much of the mangrove-decimated shoreline is under intensive use, whereas the decimated seagrass bottoms not under as intensive use.

4.1.1. Similarities and differences in tropical stocks of blue carbon in Southeast Asia and Gulf of Mexico

The total area appears less important in comparing organic carbon than the coastline extent for these two regional bodies of water similar in total area, but differing in coastline (108,292 km, SE Asia; 34,276 km, GoM) (Table 3). Consequent to a much shorter coastline among other factors, mangroves sequester only 38% of the organic carbon per ha in GoM than in SE Asia. The seagrass buried carbon per hectare in Southeast Asia is about three times greater than in the Gulf of Mexico (GoM) potentially due to a greater extent of sublittoral shallows and to several larger seagrasses plus dense mixed-species meadows (Table 4). We note the SE Asian seagrass estimates are far less accurate than the precise extent measurements of the GoM where all is aerially plus ground truth mapped in USA portion.

A. *Similarities:* The tropical Western Atlantic central coast contains the tropical Greater Caribbean Sea where the upper basin is the Gulf of Mexico (GoM). This sea has many features in common with the western central Pacific areas' tropical basins containing SE Asia. We compare them primarily for factors effecting sequestration of sedimentary carbon of mangroves and seagrass and their constraints. First both have warm temperatures, and clear waters. Both regions are bathed in steady, large flows of dissolved carbon and nutrients from equatorial oceanic currents. In general the spatial extent is around 4.6 million ha for Gulf of Mexico and 4.5 million ha for SE Asia (without PNG). Due to the island archipelagos throughout SE Asian underlying geo-tectonic plate, the SE Asian coastline is far more extensive as are shallows for seagrass within relatively similar regional spaces. Both regions have 12 months production of blue carbon material. Both regions have massive mangrove shorelines in estuaries with seagrass subtidally, resulting in complex food webs used for commercial, artesanal, and sports fisheries with minimally 1500 Gulf of Mexico fish species (Chen, 2017) and 8000 invertebrate species (Fautin et al., 2010). The Indo Pacific coastal fish species number over 6000 (Sanciangco et al., 2013, Allen, 2008) and over 1000 Indo Pacific coral species (Allen, 2008) with > 1200 Indo-Pacific crustaceans and molluscs (Sanciangco et al., 2013) plus many thousands of other types of invertebrate species (in Western Pacific and southeast Asia). Much of the both coastal areas is punctuated by estuaries or bays which are chiefly separated by barrier islands between open waters and estuaries. B. *Differences:* The Southeast Asian area's coastlines for estuaries and shelves (108,292 km) are far larger than that of the Gulf of Mexico (34,276 km) (Mendelssohn et al., 2017). In Southeast Asia the mangroves' carbon stock measured from this larger area with longer shorelines appears much higher (3095.19 TgC) than seagrass carbon stock (estimated from 1371.17 to 1683.97 TgC) although their total extents (5.1 million ha of mangroves vs 6.7 Million ha of seagrass) are similar. GOM possesses 15% of Southeast Asian seagrass extent and about 13% of mangroves. In the GoM mangroves are higher slightly higher in blue carbon (197 TgC) than seagrass (184 TgC) although seagrass are a third greater in extent throughout the GoM region. Both regions have massive anthropogenic pollution. Soil erosion from agriculture and mining sources enter into the estuaries creating degradation of blue carbon habitat (Van Katwijk et al., 2011). More ubiquitous coastal petroleum production occurs in western GoM, although Indonesia and Brunei have petroleum drilling and natural oil seepage (which may accumulate organic Carbon in the sediment).

Estuaries' total baseline sedimentary carbon differ up to ten times between the two regions (4779.16 TgC vs. 480.48 TgC). These differ from one another due partly to differing extents of mangroves and seagrasses. The underlying lithography as well as the energy regime in GoM is critical in retaining and storing the sequestered carbon differences (Thorhaug et al., 2019). The GoM is a cul-de-sac, whereas much of the Southeast Asian region is a "through flow" between two oceans, where areas near the "through Flow" are more productive than mainland areas. Together with lower Caribbean basin, they form a predominance of tropical blue carbon.

4.2. Future blue carbon investigation considerations

It is clear that both refined scientific estimates and national plans to return a portion of the large lost organic Blue Carbon to Southeast Asia are of great importance. This requires coupling with government at local, national and international levels to aid government awareness in the services the seagrasses and mangroves provide, including IPCC carbon buried vs. their carbon loss (e.g. shrimp ponds from mangroves creating degassing). When one compares the future of the Southeast Asian region Blue Carbon to Gulf of Mexico regional blue carbon (Thorhaug et al., 2019) one sees needs for the following: More detailed measurements of seagrass and mangrove blue carbon throughout the Southeast Asian nations are required for refining to the next level of estimation especially carried out as transects from mangroves at riverine mouth through the seagrasses meadows to inlet pass per estuary in selected typical estuaries throughout (e.g. Chew and Gallagher, 2019) including the net flux. The Black Carbon needs to be separated from Blue Carbon especially in Southeast Asia, where forest fires and vehicular black carbon are appreciable. Baseline controls, for long-term barren areas, vs. blue carbon habitat areas are needed to better define blue carbon ecosystem contributions vs. physical/chemical/geological contributions to buried organic carbon such as riverine and shelf carbon flux. Some estimates of the full carbon content to bedrock are needed per nation. Further definitions of riverine contributions to estuarine carbon and estuaries contributions to shelves as Butman et al. (2016) have done for North America.is lacking in many southeast Asian areas (Milliman, 1995; Milliman et al., 2015; and Liu et al., 2018). Plankton organic carbon contributions to buried organic carbon in SE Asian tropical especially in estuaries, also in bays and shelves, are poorly defined at present.

4.2.1. Overview

Constraints of the lithosphere of the geotectonic Sunda Plate forming much of central Southeast Asia include variously derived estuaries and shelves as well as various types of watersheds (Fig. 2). Sedimentary Blue Carbon regionally is usually found accompanied by chiefly sand or muddy sand. Regional Black Carbon found chiefly in estuaries is a constraint yet to be examined widely for comprehending the total carbon cycled within these ecosystems. Although past geological histories differ among nations,

Table 4
The comparison of two tropical hotspots of sedimentary organic blue carbon stock from mangroves and seagrasses. Southeast Asian metrics from this report. Gulf of Mexico metrics from Thorhaug et al. (2019).

Region & Blue carbon habitat	Extent in ha	Organic MgCorg ha-1	Regional Stock In Tg	Global extent km2
Mangroves				165,000 to 198,800
Southeast Asian	5,116,032	877	2,584.29	
Gulf of Mexico	650,431	98-378	196.88	
SEAGRASSES				171,000
Southeast Asian	6,744,529	491	1,683.37	
Gulf of Mexico	972,327	170	184.1	
MARSH GoM	538,637	170-235	99.5	
Total Seagrass + mangroves	**14,021,956**		**4,748.14** With GoM marsh	**369,800** Global Without Marshes

particularly in the geological formation of the Southeast Asian landmass vs. the volcanic island archipelagos, the mangrove and seagrass estuarine blue carbon habitats appear ubiquitous regionally. In the mainland Southeast Asian coastlines, carbon has accumulated from terrestrial watersheds emanating in the Himalayas and Chinese Southern highlands coupled with a long-shore oceanic particle transport. The Island archipelagos (which contain a predominate portion of the extent of blue carbon) gain materials for vegetative growth from dissolved Equatorial Pacific carbon dioxide and upwelled nutrients from deeper layers at the edge of the Sunda Plate. The one common element in the archipelagos are the volcanic mountains from which relatively short rivers bring watershed soils into the estuaries and shelves. A second constraint are the physical-chemical effects on the seagrasses' and mangroves' nutrient physiology and productivity from intense equatorial Pacific and Indian Oceans oceanic currents pouring through the maze-like passage of the islands, shelves and estuaries creating the largest single extent of mangroves plus seagrass meadows globally. This Pacific Equatorial water rich in nutrients and dissolved carbon dioxide, is accompanied by upwellings due to depth differences. The third very large constraint involves the anthropogenic effects on the seagrasses' and mangroves' extent and health. Over the past hundred years, a huge human population increase occurred which has become more dependent on terrestrial crops and nearshore fisheries for nutrition. Increase in life style in a few of the nations to equal that of Europe has resulted in dredging and filling for channels and Ports, land fill along coasts, effluents, accidental industrial spills, soil erosion from agriculture and mining without strict environmental resource policy and regulation enforcements until recently. All these activities are accelerating resulting in the degradation of existing mangrove and seagrass extents (Giri et al., 2011, Giri, 2016; Alongi et al., 2016; Fortes et al., 2018; Waycott et al., 2009; Unsworth et al., 2018, Hamilton and Friess, 2016). Degraded habitats in turn degrades their services (such as carbon buried, shoreline resilience, or their fisheries food web). The Government stewardship, particularly enforcement of regulations, to protect the remaining mangroves and seagrass appears frequently inadequate to prevent their loss. The bulk of the governments' priorities' value preferences appear to be large scale infrastructure and industrial/commercial projects especially around Ports and mining above living resource sustainability. This inadequate management has been occurring for at least two centuries certainly during the colonial periods. There is generally poorer resource management of seagrass than mangroves (perhaps due to the “invisibility factor” noted by Pavan Sukhdev, 2012). The solution of preserving mangrove and seagrass for their carbon inventory values does not appear to have permeated into the general awareness of citizens or legislation in many of these nations. However, most SE Asian nations have signed United Nations IPCC and regional treaties and agreements such as RAMSAR, and Biodiversity treaties to preserve and sustain critical habitat. As an example, only Philippines has a master plan for seagrasses.

Specifically, when accurate metrics for historical extents of mangroves and seagrass are available, these two important blue carbon habitats appear ubiquitously to have decreased in the region (Giri et al., 2011; Alongi et al., 2016; Huong et al., 2017; Fortes et al., 2018). Defoliation of mangroves during the Viet-nam war was such an example. Creating dredge and fill land, and large-scale ports in Singapore is another example. The percentages of decreases differ widely per nation, although Indonesia, Philippines, and others with massive stocks appear rapidly accelerating in degradation. However, especially in the case of mangroves projects have destroyed at minimum 2.5 Million ha of mangroves. Destroyed or removed mangroves create severe back-gassing which adds to the national negative carbon foot print. The seagrasses sediment also loses carbon once seagrasses are killed (Thorhaug et al., 2017), but no evidence for methane is back-gassed from killed seagrass meadows is present. Burning forests contribute Black carbon to the sediment, particularly a constraint on Blue carbon calculation in this region of significant burning forests.

The differences in Indonesia with the highest values of sedimentary blue carbon and the highest mangrove and seagrass extents globally (Giri et al., 2011; Green and Short, 2003; Alongi et al., 2016; Fortes et al., 2018) show rapid decline. The differences in blue carbon seen throughout the area apparently have much to do with the extents of mangroves and seagrasses. There are finer scale large carbon differences within an estuary measured from river mouth to ocean inlet (Chew, Gallagher et al. 2019), as are carbon differences apparent on shelves from shore to reef (Belshe et al., 2018).

5. Conclusions

A meta-analysis of detailed blue carbon stock in Southeast Asia per nation estimates a globally very high organic carbon sequestration (4779.16 Tg C in 1st meter) from combined blue carbon habitats snapshot of seagrass plus mangroves detailed per nation synoptically for the first time. The Southeast Asian estuaries vary substantially in depth and underlying lithospheric structure. Lithospheric structure is important in determining the total carbon depth. The lithosphere originally created and now continues to add much terrigenous sediment derived from Himalayan and south-central Chinese mountainous watersheds to Southeast Asian estuaries in the mainland portion. Unfortunately, carbon content data of these watersheds is not yet available in most areas (Milliman et al., 2015; Milliman, 1995, and Liu et al., 2018) as found for the Western Hemisphere (Butman et al., 2016). This constraint differs from the Island archipelagos (Indonesia, Philippines, and the Island New Guinea) wherein volcanic erosion dominates the relatively short watersheds, but a massive current of Equatorial tropical Pacific “flow through” continually brings nutrients through the archipelagos' estuaries.

Constraints of the unique Pacific Equatorial Oceanic current which then partially becomes the Wyrtki “flow through” (volume = 12.7 Svedrups per yr) (Wyrtki, 1961) with a 16 to 40 cm water head from the Pacific through the maze of passages among terrestrial features to the eastern Indian Ocean (primarily on the geotectonic Sunda Plate and secondarily the Philippine Plate) demonstrates the importance of geological structure to blue carbon habitat's rich productivity and sequestration. Intense continual streams of high levels of nutrients including dissolved carbon dioxide from the Pacific Equatorial upwelling combined with site specific convergences creating additional nutrients from rich deeper water upwellings penetrate clear upper layer waters and pour across shelves and into estuaries. Additionally, predominant atmospheric patterns for mangrove carbon uptake occur from the Equatorial Pacific and in the Indian Ocean also provide continually excellent CO_2 levels for mangroves. The “Through-flow” (Wyrtki, 1961) nations experiencing the peak effects are Philippines, Indonesia, Malaysia (especially Borneo) and New Guinea Island. For seagrasses “flow through” shows 1674.25 Tg C seagrass and 2942.61 TgC for mangroves. The mainland nations with far less “flow” show seagrass 8.74 TgC and mangroves 464.19 Tg C. This is the first time that the island archipelago nations clearly have demonstrated higher sequestered mangrove and seagrass carbon than the mainland nations (Fig. 5).

The largest global extent of seagrasses plus mangroves lie in Southeast Asian estuaries. Once destroyed, there is no resource to replace the blue carbon from this region unless intensive restoration occurs. Unfortunately there is strong anthropogenic pressure for “taking” of these habitats to alter the coastlines to man's values for more “built” structures. This occurs in governmental and commercial/industrial priorities with little understanding of the uniqueness of this region vs. further coastal development impacts in terms of local and global living-resource sustainability. An understanding of the services of blue carbon habitats provided to the cultures' social fabric in these areas such as fisheries nursery, shoreline resilience and stability, and the biodiversity role are highly needed to inform citizens and resource managers alike. The underlying accelerating loss of these living resources' services has not in the past and presently is not being valued properly by the governmental stewardship. The economic pressures of “taking” living blue carbon resources is not adequately charged to the “taker” to rehabilitate the resource elsewhere. Anthropogenic activities which are reported to be degrading both mangroves and seagrass more rapidly than any other global site (Giri et al., 2011; Fortes et al., 2018; Alongi et al., 2016) are multiple in origin: lack of citizen awareness, lack of government stewardship, lack of funding to preserve or protect habitats;

lack of constructive solutions being carried in large scale such as habitat restoration and preservation at critical sites. Below the surface of this issue are anthropogenic driving forces is high population growth (e.g. Myanmar in 1960, population was 20 million, now is 40 million; Thailand's population was 22 million in 1960, now is 57.7 million). These population pressures create more nutritional requirement pressures to feed these growing populations from continually eroding agricultural soils terrestrially and from degrading fisheries nurseries in the sea. A cycle of degradation of resources spirals downward without citizens being fully aware of tradeoffs of large scale coastal projects for their blue carbon, fisheries, and shoreline resilience resources.

Authors' contributions

The authors Gallagher and Yap (Malaysia), Kiswara (Indonesia), Prathap (Thailand), Huang (Southeastern China) sent data, tables, figures, photos and various data syntheses for their respective nations. Thorhaug, Gallagher, Yap, Prathep, Huang created partial literature surveys. The authors Thorhaug and Gallagher with numerous consultations with Berlyn wrote the text first few drafts, while Berlyn, Yap, Dorward, Prathep, Huang, Kiswara edited, including fact-checking. Thorhaug, Gallagher, and Berlyn carried out comparative Atlantic vs. SE Asian carbon analysis. Yap and Dorward edited and created figures. All authors contributed editing to final draft.

Declaration of competing interest

None of the authors has financial or other relationships which will mean that there is conflict of interest with the results. No author has financial interests in any part of this.

Acknowledgements

We thank the International Seagrass Workshops in Hainan China and in Singapore for bringing the group together. We thank all those whose extent data was utilized, as well as those whose carbon sequestered data was utilized. In particular we thank Giri et al., Spalding, Lamit, Hutchinson, Freiss, Hamilton, Alongi et al., Rattaoni, Miyajima et al., Jiang et al., Ha et al., Li, Ren, Liu et al., Huong et al., Silverstrum, Shearman, Yucaob, et al. and others' data we cited. Funding sources to create this document include Kempner Foundation.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.marpolbul.2020.111168.

References

Abdul-Hadi, A., et al., 2013. Seasonal variability of chlorophyll-a and oceanographic conditions in Sabah waters in relation to Asian monsoon - a remote sensing study. Environ. Monit. Assess. 185 (5), 3977–3991.

Allen, G.R., 2008. Conservation hotspots of biodiversity and endemism for Indo-Pacific coral reef fishes. Aquat. Conserv. 18 (5), 541–556.

Alongi, D.M., 2014. Carbon cycling and storage in mangrove forests. Annu. Rev. Mar. Sci. 6, 195–219.

Alongi, D.M., Mukhopadhyay, S.K., 2015. Contribution of mangroves to coastal carbon cycling in low latitude seas. Agric. For. Meteorol. 213, 266–272. https://doi.org/10.1016/j.agrformet.2014.10.005.

Alongi, D.M., Sasekumar, A., Tirendi, F., Dixon, P., 1998. The influence of stand age on benthic decomposition and recycling of organic matter in managed mangrove forests of Malaysia. J. Exp. Mar. Biol. Ecol. 225, 197–218.

Alongi, D., et al., 2016. Indonesia's blue carbon: a globally significant and vulnerable sink for seagrass and mangrove carbon. Wetl. Ecol. Manag. 24 (1), 3–13.

Armitage, A.R., Highfield, W.E., Brody, S.D., Louchouarn, P., 2015. The contribution of mangrove expansion to salt marsh loss on the Texas Gulf Coast. PLoS One 10 (5), e0125404. https://doi.org/10.1371/journal.pone.0125404.

Asmoro, P.J.P., Setiadi, A., 2010. Carbon and financial potency of Pometia pinnata, an endemic Papuan species. Int. For. Rev. 12 (5), 41 (Forestry Research Institute of Manokwari, Indonesia; pandumail@gmail.com.

Ayers, J.M., et al., 2014. Indonesian throughflow nutrient fluxes and their potential impact on Indian Ocean productivity. Geophys. Res. Lett. 41 (14), 5060–5067.

Bell, S.S., Clements, L.A.J., Kurdziel, J., 1993. Production in natural and restored seagrasses: a case study of a macrobenthic polychaete. Ecol. Appl. 3, 610–621.

Belshe, F., Hoejimakers, Moltera, M., Herran, F., et al., 2018. Seagrass community-level controls over organic carbon storage are constrained by geophysical attributes within meadows of Zanzibar, Tanzania. Biogeosciences 15 (14), 4609–4626.

BOBLME, 2015. Assessment of Transboundary Governance Architecture for the Bay of Bengal LME. BOBLME-2015-Governance-07 by R Mahon, L Fanning.

Bouillon, S., Borges, A.V., Castañeda-Moya, E., Diele, K., Dittmar, T., Duke, N.C., Kristensen, E., Lee, S.Y., Marchand, C., Middelburg, J.J., Rivera-Monroy, V.H., Smith III, T.J., Twilley, R.R., 2008. Mangrove production and carbon sinks: a revision of global budget estimates. Glob. Biogeochem. Cycles 22 (2), 1–12.

Braun, K.N., et al., 2019. Modeling organic carbon loss from a rapidly eroding freshwater coastal wetland. Sci. Rep. 9 (1), 4204.

Breithaupt, J.L., Smoak, J.M., Smith, T.J., Sanders, C.J., Hoare, A., 2012. Organic carbon burial rates in mangrove sediments: strengthening the global budget. Glob. Biogeochem. Cycles 26 (3), GB3011 26(3): GB3011.

Broecker, W.S., et al., 2000. The origin of Bahamian Whitings revisited. Geophys. Res. Lett. 27 (22), 3759–3760.

Butman, D., Stackpoole, S., Stets, E., McDonald, C.P., Clow, D.W., Striegl, R.G., 2016. Aquatic carbon cycling in the conterminous United States and implications for terrestrial carbon accounting. Proc. Natl. Acad. Sci. 113, 58–63.

Cebrian, J., 2002. Variability and control of carbon consumption, export, and accumulation in marine communities. Limnol. Oceanogr. 47 (1), 11–22.

Chee, S.Y., et al., 2017. Land reclamation and artificial islands: walking the tightrope between development and conservation. Glob. Ecol. Conserv. 12, 80–95.

Chen, Y., 2017. Fish resources of the Gulf of Mexico. In: Ward, C. (Ed.), Habitats and Biota of the Gulf of Mexico: Before the Deepwater Horizon Oil Spill. Springer, New York, NY..

Chew, S.T., Gallagher, J.B., 2018. Accounting for Black carbon lowers the estimate for Blue Carbon storage services. Sci. Report 8, 2553–2558. https://doi.org/10.1038/s41598-018-20644-2.

Chuan, C.H., et al., 2019. Blue carbon sequestration dynamics within tropical seagrass sediments: long-term incubations for changes over climatic scales. bioRxiv: 604587. Glob. Ecol. Conserv. 12, 80–95.

Church, J.A., White, N.J., 2016. A 20th century acceleration in global sea-level rise. Geophys. Res. Lett. 33, 1–4. https://doi.org/10.1029/2005GL024826.

Church, J.A., Gregory, J.M., Huybrechts, P., et al., 2001. Climate change 2001: the scientific basis. In: Contribution of Working Group 1 to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge (639 pp).

Cloern, J.E., et al., 2014. Phytoplankton primary production in the world's estuarine-coastal ecosystems. Biogeosciences 11 (9), 2477–2501.

Comineax, R.S., Allison, M.A., Bianchi, T., 2012. Mangrove expansion in the Gulf of Mexico with climate change: implications for wetland health and resistance to rising sea levels. Estuar. Coast. Shelf Sci. 96, 81–95.

Costanza, et al., 2017. 20 years of ecosystems services: how far have we come and how far do we need to go. Ecosyst. Services 28, 1–16.

Donato, D.C., et al., 2011. Mangroves among the most carbon-rich forests in the tropics. Nat. Geosci. 4 (5), 293–297.

Duarte, C.M., Agusti, S., 1998. The CO2 balance of unproductive aquatic ecosystems. Science 281 (5374), 234–236.

Duarte, C.M., Middelburg, J.J., Caraco, N., 2005. Major role of marine vegetation on the oceanic carbon cycle. Biogeosciences 2, 1–8.

Duarte, C.M., et al., 2013. Assessing the capacity of seagrass meadows for carbon burial: current limitations and future strategies. Ocean Coast. Manag. 83, 32–38.

Dunton, K.H., Tomasko, D.A., 1994. In situ photosynthesis in the seagrass Halodule wrightii in a hypersaline subtropical lagoon Marine Ecology Progress Series,: 994.

Durako, M.J., Hall, M.O., Sargent, F., Peck, S., 1992. Propeller scars in seagrass beds: an assessment and experimental study of recolonization. Mar. Ecol. Prog. Ser. 110, 59–66.

Ellison, J.C., Strickland, P., 2013. Establishing relative sea level trends where a coast lacks a long term tide gauge. Mitig. Adapt. Strateg. Glob. Change. https://doi.org/10.1007/s11027-013-9534-3.

Enriquez, S., Schubert, N., 2014. Direct contribution of the seagrass *Thalassia testudinum* to lime mud production. Nat. Commun. 5, 3835.

Eong, O.J., 1993. Mangroves - a carbon source and sink. Chemosphere 27 (6), 1097–1107.

Estrada, G.C.D., Soares, M.L., Cavalcanti, V.F., Almeida, P.M.M., 2015. The economic evaluation of carbon storage and sequestration as ecosystem services of mangroves: a case study from southeastern Brazil. Int. J. Biodivers. Sci., Ecosys. Services Manag. 11 (1), 29–35. https://doi.org/10.1080/21513732.2014.963676.

FAO, 1991. Nation Sustainable Agriculture and Development in Asia and the Pacific. Regional document No. 2 FAO/Netherlands Conference on Agriculture and Environment, S-Hertogenbosch, the Netherlands 15–19 April 1991.

FAO, 2003. FAO's database on mangrove area estimates. In: Wilkie, M.L., Fortuna, S., Souksavat, O. (Eds.), Forest Resources Assessment Working Paper No. 62. Forest Resources Division, FAO, Rome.

Fautin, D., Dalton, P., Incze, L.S., Leong, J.-A.C., Pautzke, C., Rosenberg, A., et al., 2010. An overview of marine biodiversity in United States waters. PLoS One 5 (8), e11914. https://doi.org/10.1371/journal.pone.0011914.

Field, R.D., et al., 2016. Indonesian fire activity and smoke pollution in 2015 show persistent nonlinear sensitivity to El Niño-induced drought. Proc. Natl. Acad. Sci. USA **113**, 9204–9209.

Fitrian, T., Kusnadi, R., Peresllete, R., 2017. Seagrass community structure of Tayando-Tam Island, Southeast Moluccas, Indonesia. Biodiversitas 18 (2). https://doi.org/10.13057/biodiv/d180246.

Fog, K., 1988. The effect of added nitrogen on the rate of decomposition of organic matter. Biol. Rev. 63 (3), 433–462.

Forqurean, J., Schrlau, S., 2003. The measurement of nutrient chemistry along a gradient

for seagrass and mangroves. Environ. Chem. 19 (5), 673–676.
Fortes, M.S., Ooi, J.L.S., Tan, Y.M., Prathep, A., Bujan, J.S., Yaakub, S.M., 2018. Seagrass in Southeast Asia: a review of status and knowledge gaps, and a road map for conservation. Bot. Mar. 61 (3), 269 20 p.
Fourqurean, J.W., Duarte, C.M., Kennedy, H., Marba, N., Holmer, M., Mateo, M.A., Apostolaki, E.T., Kendrick, G.A., Krausse-Jensen, D., McGlathery, S.O., 2012. Seagrass ecosystems as a globally significant carbon stock. Nat. Geosci. https://doi.org/10.1038/NGEO1477.
Freeman, A.S., et al., 2008. Seagrass on the edge: land-use practices threaten coastal seagrass communities in Sabah, Malaysia. Biol. Conserv. 141 (12), 293–300.
Fujimoto, K., Imaya, A., Tabuchi, R., Kuramoto, S., Utsugi, H., Murofushi, T., 1999. Belowground carbon storage of Micronesian mangrove forests. Ecol. Res. 14, 409–413.
Gacia, E., Duarte, C.M., Marba, N., Terrados, J., Kennedy, H., Fortes, M.D., Tri, N.H., 2003. Sediment deposition and production in SE Asia seagrass meadows. Estuar. Coast. Shelf Sci. 56, 909–919.
Gallagher, J.B., 2014. Explicit and implicit assumptions within the blue carbon conceptual model: a critique, Proceedings of the International Conference on Marine Science and Aquaculture. In: Ecosystem Perspectives in Sustainable Development, 18-20 March 2014. Kota Kinabalu, Sabah Malaysia, 9789834015640, pp. 26–40.
Gallagher, J.B., The implications of global climate change and aquaculture on blue carbon sequestration and storage within submerged aquatic ecosystems, in Aquaculture Ecosystems, S.S. Mustafa, R, Editor. 2015, Wiley Blackwell: Oxford. p. 243–280.
Gallagher, J.B., 2017. Taking stock of mangrove and seagrass blue carbon ecosystems: a perspective for future carbon trading. Borneo J. Mar. Sci. AquaC. 1 (1), 71–74.
Gallagher, J.B., Chuan, C.H., Yap, K.T., Farahain, W., 2018. Carbon Sink Services for Tropical Coastal Seagrass are Far Lower than Anticipated When Accounting for Black Carbon. BioRxiv 493,262 [Preprint] [cited 2018 Dec 24]. Available from. https://doi.org/10.1101/493262.
Gallagher, J.B., Yap, T.K., Chew, S.T., Chuan, C., Dona, W., 2019. Carbon stocks of coastal seagrass in Southeast Asia may be far lower than anticipated when accounting for black carbon. Biol. Lett. 15. https://doi.org/10.1098/rsbl.2018.0745.
Gallagher, J.B., Chew, S.-T., Madin, J., Thorhaug, A., 2020. Valuing Carbon Stocks across a Tropical Lagoon after Accounting for Black and Inorganic Carbon: Bulk Density Proxies for monitoring. J. Coast. Res. https://doi.org/10.2112/JCOASTRES-D-19-00127.1. (in press).
Gao, Y., Fang, J.G., Tang, W., Zhang, J., Ren, L., Du, M., 2013. Seagrass meadow carbon sink and amplification of the carbon sink for eelgrass bed in Sanggou Bay. Progress Fish. Sci. 34 (1), 1000.
Gao, T., Ding, D., Guan, W., Liao, B., 2018. Carbon stocks of coastal wetland ecosystems on Hainan Island, China. Pol. J. Environ. Stud. 27 (3), 1061–1069.
Giardino, C., et al., 2016. Mapping submerged habitats and mangroves of Lampi Island Marine National Park (Myanmar) from in situ and satellite observations. Remote Sens. 8 (2), 1–13.
Giri, C., 2016. Observation and Monitoring of Mangrove Forests Using Remote Sensing: Opportunities and Challenges. Remote Sens. 8, 783.
Giri, C., et al., 2011. Status and distribution of mangrove forests of the world using earth observation satellite data. Glob. Ecol. Biogeochemistry 20, 154–159.
Giri, C., Long, J., Abbas, S., et al., 2015. Distribution and dynamics of mangrove forests of South Asia. *J. Environ. Manage.* 148, 101–111. https://doi.org/10.1016/j.jenvman.2014.01.020.
Golley, F.B., Odum, H.T., Wilson, W.F., 1962. The structure and metabolism of a Puerto Rico mangrove forest in May. Ecology 43, 9–19.
Green, E.P., Short, F.T., 2003. World Atlas of Seagrasses. UNEP World Conservation Monitoring Centre. University of California Press, Berkeley, USA.
Guannel, G., et al., 2016. The power of three: coral reefs, seagrasses and mangroves protect coastal regions and increase their resilience. PLoS One 11 (7), e0158094.
Gupta, A., 1996. Erosion and sediment yield in Southeast Asia: a regional perspective. In: Erosion and Sediment Yield: Global and Regional Perspectives (Proceedings of the Exeter Symposium, July 1996). IAHS Publ. no. 236, pp. 215.
Ha, T.H., Marchand, C., Aimé, J., Dang, H.N., Phan, N.H., Nguyen, X.T., Nguyen, T.K.C., 2018. Belowground carbon sequestration in a mature planted mangroves (Northern Viet Nam). For. Ecol. Manag. 407 (1), 191–199.
Hamilton, Casey, 2016. Terrestrial Ecosystems of the World dataset (Hamilton and Casey 2016); World Atlas of Mangrove dataset (imagery between 1999 & 2003).Terrestrial Ecosystems of the World dataset (Hamilton and Casey 2016); World Atlas of Mangrove dataset (imagery between 1999 2003).
Hamilton, S. E. and D. Friess. 2016. “High-resolution annual maps of whole system mangrove carbon stocks from 2000 to 2012.” (arXiv preprint arXiv: 1611.00307. 26).
Hamilton, S.E., Friess, D.A., 2018. Global carbon stocks and potential emissions due to mangrove deforestation from 2000 to 2012. Nat. Clim. Chang. 8, 240–244. https://doi.org/10.1038/s41558-018-0090-4.
Hedley, J.D., Russell, B.J., Randolph, K., Pérez-Castro, M.Á., Vásquez-Elizondo, R.M., Enriquez, S., Dierssen, H.M., 2017. Remote sensing of seagrass leaf area index and species: the capability of a model inversion method assessed by sensitivity analysis and hyperspectral data of Florida Bay. Front. Mar. Sci. https://doi.org/10.3389/fmars.2017.00362.
Herrmann, M., Najjar, R.G., Kemp, W.M., Alexander, R.B., Boyer, E.W., Cai, W.J., Griffith, P.C., Kroeger, K.D., McCallister, S.L., Smith, R.A., 2015. Net ecosystem production and organic carbon balance of US East Coast estuaries: a synthesis approach. Glob. Biogeochem. Cycles 29, 96–111.
Hossain, M.S., et al., 2016. Marine and human habitat mapping for the Coral Triangle Initiative region of Sabah using Landsat and Google Earth imagery. Mar. Policy 72, 176–191.
Howard, J.L., et al., 2017. CO2 released by carbonate sediment production in some coastal areas may offset the benefits of seagrass “Blue Carbon” storage. Limnol. Oceanogr. 63 (1), 160–172.
Huang et al., 2016.
Huong, N.T.T., Tuan, T.A., Thach, V.T., Tin, H.C., 2017. A review of seagrass studies by using satellite remote sensing data in the Southeast Asia: status and potential. Vietnam J. Sci. Technol. 55 (4C), 148–154.
Husodo, T., Palabbi, S.D.G., Abdoellah, O.S., Nurzaman, M., Fitriani, N., Partasasmita, P., 2017. Seagrass diversity and carbon sequestration: case study on Pari Island, Jakarta Bay, Indonesia. Biodiversitas 18, 1596–1601.
Hutchinson, J., Manica, A., Swetnam, R., Balmford, A., Spalding, M., 2014. Predicting global patterns in mangrove forest biomass. Conserv. Lett. 7, 233–240.
IPCC, 2007. In: Solomon, S. (Ed.), Climate change 2007: The Physical Science Basis. Cambridge University Press, Cambridge.
Jiang, Z., Liu, S., Zhang, J., Zhao, C., Wu, Y., Yu, S., Zhang, X., Huang, C., Huang, X., Kumar, M., 2017. Newly discovered seagrass beds and their potential for blue carbon in the coastal seas of Hainan Island, South China Sea. Mar. Pollut. Bull. 125 (1–2), 513–521.
Jiang, Z., Liu, S., Zhang, J.P., Wu, Y.C., Zhao, C., Lian, Z., Huang, X.P., 2018. Eutrophication indirectly reduced carbon sequestration in a tropical seagrass bed. Plant Soil 426, 135–142.
Jiang, A., et al., 2019. Contrasting root length, nutrient content,c sequestration of seagrass in the South China Sea. Sci. Total Environ. 662, 151–159.
Jiang, Z., Cui, L., Liu, S., Zhao, C., Wu, Y., Chen, Q., Yu, S., Li, J., He, J., Fang, Y., Ranvilage, Chanaka IsurangaPremarathne Maha, Huang, X., 2020. Historical changes in seagrass beds in a rapidly urbanizing area of Guangdong Province: implications for conservation and management. Global Ecology and Conservation 22. https://doi.org/10.1016/j.gecco.2020.e01035.
Kauffman, J.B., Heider, C., Norfolk, J., Payton, F., 2016. Carbon stocks of intact mangroves and carbon emissions arising from their conservation in the Dominican Republic. Ecol. Appl. 24 (3), 518–527.
Kaufmann, J.B., Bhomia, R.K., 2017. Ecosystem carbon stocks of mangroves across broad environmental gradients in West-Central Africa: global and regional comparisons. PLoS One 12 (11), e0187749. https://doi.org/10.1371/journal.pone.0187749.
Kawaroe, M., Nugraha, A.H., Juraij, J., 2016. Seagrass biodiversity at three marine ecoregions of Indonesia: Sunda Shelf, Sulawesi Sea, and Banda Sea. Bidiversitas 17 (2), 585–591. https://doi.org/10.13057/biodiv/d170228.
Kennedy, H.A., Fourqurean, J., Johnson, B., Kauffman, J.B., Lovelock, C., Saintilan, N., et al., 2013. Field sampling of vegetated carbon pools in coastal ecosystems. In: Howard, J., Hoyt, S., Isensee, K., Telszewski, M., Pidgeon, E. (Eds.), Coastal Blue Carbon: Methods for Assessing Carbon Stocks and Emissions Factors in Mangroves, Tidal Salt Marshes, and Seagrasses. Conservation International, UNSESCO, IUCN, Arlington, Virginia, USA, pp. 67–108.
Kilminster, K., et al., 2015. Unravelling complexity in seagrass systems for management: Australia as a microcosm. Sci. Total Environ. 534, 97–109.
Kiswara, W., 2018. Seagrass and carbon studies in Indonesian waters. In: Presentation to Blue Carbon Summit 17–18. July 2018. Jakarta, Indonesia.
Kiswara, W., Erlangga, D.K., Kawaroe, M., Radnian, N., 2010. Transplanting enhalus acoroides (l.f) royle with different length rhizome on the muddy substrate and high water dynamic at banten bay, indonesia transplanting enhalus acoroides (l.f) royle with different length rhizome on the muddy substrate and high water dynamic at banten bay, indonesia. Indonesian Institute of Sciences 35 (1), 1–33.
Lamit, N., Tanaka, Y., Majid, H.M.B.A.H., 2017. Seagrass biodiversity in Brunei Darrasalem. Scientia Bruneiana 16 (2).
Liu, C.H., Ren, H., Hui, D., Wang, W., Liao, B., Cao, Q., 2013. Carbon stocks and potential carbon storage in the mangrove forests of China. J. Environ. Manag. 133C, 86–93.
Liu, Q., et al., 2018. Carbon fluxes in the China Seas: an overview and perspective. Sci. China Earth Sci. 61 (11).
López-Portillo, J., Lara-Domínguez, A., Bravo-Mendoza, M., 2018. Carbon Stocks in fringe and basin mangroves along the State of Veracruz, Mexico. INECOL, Xalapa, Veracruz, México, pp. 45.
Luong, C.V., Thao, N.V., Komatsu, T., Ve, N.D., Tien, D.D., 2012. Status and threats on seagrass beds using GIS in Vietnam. Proceedings volume 8525, remote sensing of the marine environment II. Proc. SPIE 8525, 852512 13 pp. https://doi.org/10.1117/12.977277.
Macintyre, I.G., Toscano, M.A., Lighty, R.G., Bond, G.B., 2004. Holocene history of the mangrove islands of Twin Cays, Belize, Central America. Atoll Res. Bull. 510, 1–16.
Maher, D.T., et al., 2018. Beyond burial: lateral exchange is a significant atmospheric carbon sink in mangrove forests. Biol. Lett. 14 (7).
McLaughlin, P.A., Treat, S.A.F., Thorhaug, A., Lemaitre, R., 1983. A Restored Seagrass (Thalassia) Bed and Its Animal Community Environmental Conservation. 10 (3). pp. 247–254.
McLeod, E., Chmura, G.L., Bouillon, S., Salm, R., Björk, D.C., Lovelock, C.E., Schlesinger, W.H., Silliman, B.R., 2011. A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO2. Front. Ecol. Environ. 9, 552–560.
Mendelssohn, I.A., Byrnes, M.R., Kneib, R.T., Vittor, B.A., 2017. Coastal habitats of the Gulf of Mexico. In: Ward, C. (Ed.), Habitats and Biota of the Gulf of Mexico: Before the Deepwater Horizon Oil Spill. Springer, New York, NY.
Milliman, J.D., 1995. Sediment discharge to the ocean from small mountainous rivers: the New Guinean example. Geo-Mar. Lett. 1995 (15), 127–133. https://doi.org/10.1007/BF01204453.
Milliman et al., 2015.
Miyajima, T., Hori, M., Hamaguchi, M., Shimabukuro, H., Adachi, H., Yamano, H., Nakaoka, M., 2015a. Geographic variability in organic carbon stock and accumulation rate in sediments of East and Southeast Asian seagrass meadows. Global Biocheochemical Cycles 29 (10), 397–415. https://doi.org/10.1002/2014GB004979.
Miyajima, et al., 2015b. Global variability in carbon stock in seagrass meadows in East and Southeast Asia. Global Geochemistry 29, 397–402.
Murdiyarso, D., Purbopuspito, J., Kauffman, J.B., Warren, M.W., Sasmito, S.D., 2015. The potential of Indonesian mangrove forests for global climate change mitigation. Nat. Clim. Chang. 5 (12), 1089–1109.
Najjar, R.G., Herrmann, M., Alexander, R., Boyer, E.W., Burdige, D.J., Butman, D., Cai, W.-J., Canuel, E.A., Chen, R.F., Friedrichs, M.A.M., Feagin, R.A., Griffith, P.C., Hinson, A.L., Holmquist, J.R., Hu, X., Kemp, W.M., Kroeger, K.D., Mannino, A., McCallister, S.L., McGillis, W.R., Mulholland, M.R., Pilskaln, J, C.H., 2018. Carbon

budget of tidal wetlands, estuaries, and shelf waters of eastern North America. Glob. Biogeochem. Cycles 32, 389–416. https://doi.org/10.1002/2017GB005790.
Nakaoka, M., et al., 2007. Tsunami Impacts on Biodiversity of Seagrass Communities in the Andaman Sea, Thailand: (1) Seagrass Abundance and Diversity. Seto Marine Biological Laboratory, Field Science Education and Research Center, Kyoto University.
Neihaus, 1993.
Nienhuis, P.H., Coosen, J., Kiswara, W., 1989. Community structure and biomass distribution of seagrasses and macrofauna in the Flores Sea, Indonesia. Neth. J. Sea Res. 23, 197–214.
Onuf, C., 1996. Seagrass responses to long-term light reduction by brown tide in upper Laguna Madre, Texas: distribution and biomass patterns. Mar. Ecol. Prog. Ser. 138, 219–231.
Ooi, J.L.S., et al., 2011. Knowledge gaps in tropical Southeast Asian seagrass systems. Estuar. Coast. Shelf Sci. 92 (1), 118–131.
Patty 2016.
PEMSEA, 2016. Authored by Silverstrum Climate Associates. Understanding Strategic Coastal Blue Carbon Opportunities in the Seas of East Asia.
Pendleton, L., et al., 2012. Estimating global 'blue carbon' emissions from conversion and degradation of vegetated coastal ecosystems. PLoS One 7 (9), e43542.
Pendleton, L.H., Sutton-Grier, A.E., Gordon, D.R., Murray, B.C., Victor, B.E., 2013. Considering "coastal carbon" in existing US federal statutes and policies. Coast. Manag. 41 (5), 439–456.
Perry, C.L., Mendelssohn, I.A., 2009. Ecosystem effects of expanding populations of *Avicennia germinans* in a Louisiana salt marsh. Wetlands 29, 396–406.
Philippine Forestry Dept. n.d.
Pingree, R.D., et al., 1975. Summer phytoplankton blooms and red tides along tidal fronts in the approaches to the English Channel. Nature 258 (5537), 672–677.
Poovachiranon, A., et al., 2006. Temporal variation in growth and reproduction of *Enhalus acoroides* (L.f.) Royle in a monospecific meadow in Haad Chao Mai National Park, Trang Province, Thailand. Bot. Mar. 54 (2), 201–207.
Potouroglou, M., Bull, J.C., Krauss, K.W., Kennedy, H.A., Fusi, M., Daffonchio, D., Mangora, M.M., Githaiga, M.N., Diele, K., Huxham, M., 2017. Measuring the role of seagrasses in regulating sediment surface elevation. Sci. Rep. 7 (1), 11917. https://doi.org/10.1038/s41598-017-12354-y.
Poulos, H.M., Thorhaug, A., Wiscynsky, D., Ku, T.C. 2018. Mangrove carbon in restored and natural populations in Florida and Mexico. pp. 26. Abstract. Botany 2018.
Prathup, A., 2012. Seagrass bed as a carbon sink in Ranong biosphere reserve and Trang-Haad Chao Mai Marine National Park; an important role of seagrass. In: Report to Man in the Biosphere. UNESCO.
Pugh, D., Hunter, J., Coleman, R., Watson, R., 2002. A comparison of historical and recent sea level measurements at Port Arthur, Tasmania. Int. Hydrogr. Rev. 3, 3–23.
Pulhin, F.B., Gevana, D.T., 2010. Climate change mitigation of the three mangrove communities in Quezon, Philippines. Int. For. Rev. 12 (5), 31(University of the Philippines Los Banos, Philippines; yaybpulhin@yahoo.com; wuweidix@yahoo.com), Lasco, R.D. (World Agroforestry Centre, r.lasco@cgiar.org).
Rattanachot, E., Prathep, A., 2015. Species specific effects of three morphologically different belowground seagrasses on sediment properties-Estuarine. Coastal and Shelf Science 167 (Part B), 427–435.
Rattanachot, E., Stankovic, M., Aongsara, S., Prathep, Anchana, 2018. Ten years of conservation efforts enhance seagrass cover and carbon storage in Thailand. Botanica Marina 2018 61 (5), 441–451.
Reef, R., et al., 2010. Nutrition of mangroves. Tree Physiol. 30 (9), 1148–1160.
Sanciangco, J.C., Carpenter, K.E., Etnoyer, P.J., Moretzsohn, F., 2013. Habitat availability and heterogeneity and the Indo-Pacific warm pool as predictors of marine species richness in the tropical Indo-Pacific. PLoS One 8 (2), e56245. https://doi.org/10.1371/journal.pone.0056245.
Schroeder, P.A., Thorhaug, A., 1980. Trace metal cycling in tropical-subtropical estuaries dominated by the seagrass Thalassia testudinum. Am. J. Bot. 67 (3), 118–129.
Serrano, O., Ricart, A.M., Lavery, P.S., Mateo, M., Arias-Ortiz, A., Masque, P., Rozaimi, M., Steven, A.D., Duarte, C., 2016. Key biogeochemical factors affecting soil carbon storage in Posidonia meadows. https://doi.org/10.5194/bg-13-4581-2016.
Shearman, P.L., 2010. Recent change in the extent of mangroves in the Northern Gulf of Papua, Papua New Guinea. Ambio 39 (2), 181–189. https://doi.org/10.1007/s13280-010-0025-4.
Shearman, P.L., Bryan, J.E., Ash, J., Mackey, B., Lokes, B., 2009. Forest conversion and degradation in Papua New Guinea 1972–2002. Biotropica. 41 (3), 379–390. https://doi.org/10.1111/j.1744-7429.2009.00495.x.
Shu et al. 2017.
Sidik, F., Lovelock, C.E., 2013. CO_2 efflux from shrimp ponds in Indonesia. PLoS One 8 (6), e66329. https://doi.org/10.1371/journal.pone.0066329.
Siikamäki, J., et al., 2012a. Blue carbon: coastal ecosystems, their carbon storage, and potential for reducing emissions. Environ. Sci. Policy Sustain. Dev. 55 (6), 14–29.
Siikamäki, J., Sanchirico, J.N., Jardine, S., McLaughlin, D., Morris, D.F., 2012b. Blue carbon global options for reducing emissions from the degradation and development of coastal ecosystems. 2012. In: Resources for the Future, (Washington DC).
Siswanto, E., Tanaka, K., 2014. Phytoplankton biomass dynamics in the Strait of Malacca within the period of the SeaWiFS full mission: seasonal cycles, interannual variations and decadal-scale trends. Remote Sens. 6 (4), 2718–2742.
Sondi, I., Juračić, M., 2010. Whiting events and the formation of aragonite in Mediterranean Karstic Marine Lakes: new evidence on its biologically induced inorganic origin. Sedimentology 57 (1), 85–95.
Sophia, C.J., Robie, W.M., 2016. Geoengineering with seagrasses: is credit due where credit is given? Environ. Res. Lett. 11 (11), 113001.
Spalding, M., 2010. World Atlas of Mangroves. Routledge.
Spalding, M.D., Taylor, M., Martins, S., Green, E., Edwards, M., 2001. The Global Distribution and Status of Seagrass Ecosystems. World Conservation Monitoring Centre.
Stankovic, M., Panyawai, J., Jansanit, K., Upanoi, T., 2017. Carbon content in different seagrass species in Andaman Coast of Thailand. Sains Malaysiana 46 (9), 1441–1447.
Sterner, R.W., Elsner, J.J., 2002. Ecological Stoichiometry: The Biology of Elements from Molecules to the Biosphere. Princeton University, Princeton.
Sukhdev, P., 2012. The economics of ecosystems and biodiversity in business and enterprise. In: Earthscan. James & James, London, UK..
Teas, H.J., 1977. Ecology and restoration of mangrove shorelines in Florida. Environ. Conserv. 4, 51–58.
Tedesco, L.P., Wanless, H.R., 1991. Generation of sedimentary fabrics and facies by repetitive excavation and storm infilling of burrow networks, Holocene of South Florida and Caicos Platform, B.W.I. PALAIOS 6 (3), 326–343. JSTOR. www.jstor.org/stable/3514912.
Thorhaug, A., 2001. Conf on Oil Spill Conference APT/EPA/USCG, The use of seagrass restoration after petroleum accidents. pp. 386–391.
Thorhaug, A., Cruz, R., 1987. "Seagrass transplantation trials. Philippines." FAO Rome, Italy. FAO-FI-TCP/PHI/4511. In: Coastal Rehabilitation through Seagrass Transplantation. Philippines, pp. 52.
Thorhaug, Roessler, M.A., 1977. Seagrass community dynamics in a subtropical estuarine lagoon. Aquaculture 12, 253–277.
Thorhaug, A., Wanless, H., 2001. The role of hurricanes, tornadoes, and gale force winds on Seagrasses. Am. J. Bot. 67, 73–76 abstracts.
Thorhaug, Wanless, 2019. Sediment and seagrass accretion using artificial sediment tubes in field. American J. Bot.
Thorhaug, A., Raven, J., Young, E., Franklin, L., 2009. Carbon cycling and sequestration in the Sea by phytoplankton and macrophytes. Plant Science Bulletin 55 (4), 156–162.
Thorhaug, A., Berlyn, G.P., Poulos, H.M., Goodale, U.M., 2015. Pollutant tracking for 3 Western North Atlantic sea grasses by remote sensing: preliminary diminishing white light responses of Thalassia testudinum, Halodule wrightii, and Zostera marina. Mar. Pollut. Bull. 97 (1-2), 460–469.
Thorhaug, A., Poulos, H.M., Lopez-Portillo, J., Berlyn, G.P., Ku, T.W., 2016. Pollutant tracking for 3 Western North Atlantic sea grasses by remote sensing: Preliminary diminishing white light responses of Thalassia testudinum, Halodule wrightii, and. Mar. Pollut. Bull. 97 (1-2), 460–469.
Thorhaug, A., Poulos, H.M., Lopez-Portillo, J., Ku, T.W., Berlyn, G.P., 2017. Seagrass blue carbon dynamics in the Gulf of Mexico: Stocks, losses from anthropogenic disturbance, and gains through seagrass restoration. Sci. Total Environ. 605–606, 626–636 (ISSN: 1879–1026).
Thorhaug, A., Poulos, H.M., Lopez-Portillo, J., Barr, J., Laura-Dominguez, A.L., Berlyn, G.P., Ku, T.W., 2019. Blue carbon stock and flux. In the Gulf of Mexico. Sci. Total Environ. 606, 1253–1261. https://doi.org/10.1016/j.scitotenv.2018.10.011.
Thorhaug, A., Belaire, C., Verduin, J.J., et al., 2020a. Seagrass restoration in 3 ocean basins: longevity, techniques, success. Mar. Pollut. Bull (in review).
Thorhaug, A., Verduin, J.J., Kismura, J.B., et al., 2020b. Southeast Asia Seagrass Restoration. Ambio (in review).
UNEP, 2008. National Reports on Seagrass in the South China Sea. UNEP/GEF/SCS Technical Publication No. 12.
UNEP, 2010. National Reports on Seagrass in the South China Sea. UNEP/GEF/SCS. Technical Publication.
UNESCO, 2012. Report from Anchara Prathep to UNESCO. Man in the Biosphere Program. Seagrass Bed as a Carbon Sink in Ranong Biosphere Reserve and Trang-Haad Chao Mai Marine National Park; An Important Role of Seagrass. Bangkok, Thailand.
Unsworth, R.K.F., McKenzie, L.J., Collier, C.J., et al., 2018. Ambio. https://doi.org/10.1007/s13280-018-1115-y.
Van Katwijk, M.M., van der Welle, M.E.W., Lucassen, E., Vonk, J.A., Christianen, M.J.A., Kiswara, W., al Hakim II, I., Arifin, A., Bouma, T.J., Roelofs, J.G.M., Lamers, L.P.M., 2011. Early warning indicators for river nutrient and sediment loads in tropical seagrass beds: a benchmark from a near-pristine archipelago in Indonesia. Mar. Pollut. Bull. 62, 1512–1520.
van Katwijk, M.M., Thorhaug, A., Marbà, N., Orth, R.J., Duarte, C.M., Kendrick, G.A., et al., 2016. Global analysis of seagrass restoration: the importance of large-scale planting. J. Appl. Ecol. 53 (2), 567–578.
Wang, G., Guan, D.S., Peart, M.R., Chen, Y.J., Peng, Y.S., 2013. Ecosystem carbon stocks of mangrove forest in Yingluo Bay, Guangdong Province of South China. For. Ecol. Manag. 310, 539–546.
Ware, J.R., et al., 1992. Coral reefs: sources or sinks of atmospheric CO2? Coral Reefs 11 (3), 127–130.
Wawo, M., 2017. Social-ecological system in seagrass ecosystem management at Kotania Bay Waters, Western Seram, Indonesia. IOP Conf. Ser.: Earth Environ. Sci. 89, 12023.
Waycott, M., Duarte, C.M., Carruthers, T.J.B., Orth, R.J., Dennison, W.C., Olyarnik, S., 2006. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proc. Natl. Acad. Sci. 106 (30), 12377–12381.
Waycott, A., Duarte, M.C.M., Carruthers, T.J.B., Orth, R.J., Dennison, W.C., Olyarnik, S., Calladine, A., Fourqurean, J.W., Heck Jr., K.L., Hughes, A.R., Kendrick, G.A., Kenworthy, W.J., Fk, T. Short, Williams, S.L., 2009. Accelerating Loss of Seagrasses Across the Globe Threatens Coastal Ecosystems.
Wyrtki, K., 1957. Die Zirkulation an der Oberfläche der südostasiatischen Gewässer. In: Deutsche Hydrografische Zeitschrift. 10. Springer, pp. 1–13.
Wyrtki, K., 1961. Physical Oceanography of the Southeast Asian waters. Scripps Institution of Oceanography, UC San Diego Retrieved from. https://escholarship.org/uc/item/49n9x3t4.
Wyrtki, K., 1987. Indonesian through flow and the associated pressure gradient. J. Geophys. Res. Oceans 92 (C12). https://doi.org/10.1029/JC092iC12p12941.
Yap, K.T., Gallagher, J.B., et al., 2018. Seagrass Meadow Impacts on Universiti Malaysia Sabah (UMS) Beach, Kota Kinabalu Sabah (Malaysia). Borneo J. Mar. Sci. Aquac. 2, 48–53.
Zhou, Y., Zhao, B., Peng, Y., Chen, G., 2010. Influence of mangrove reforestation on heavy metal accumulation and speciation in intertidal sediments. Mar. Pollut. Bull.