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MANGROVE BLUE CARBON IN THE FACE OF DEFORESTATION, CLIMATE CHANGE, AND RESTORATION

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Abstract: Coastal wetlands have disproportionately high carbon densities, known as blue carbon, compared to most terrestrial ecosystems. Mangroves and their blue carbon stocks are at risk globally from land-use and land-cover change (LULCC) activities such as aquaculture, alongside biophysical disturbances such as sea-level rise and cyclones. Global estimates of carbon emissions from mangrove loss have been previously unable to differentiate between the variable impacts of different drivers of loss. This article discusses the impacts that different LULCC activities and biophysical disturbances have on carbon stocks (biomass and soil) and greenhouse gas fluxes (CO_2 and CH_4). The dynamics of carbon stocks

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and fluxes depends on the type of LULCC; aquaculture often results in biomass and soil carbon removal, and some forms of agriculture can substantially increase methane emissions. Natural disturbances have mixed impacts on mangrove carbon; sea-level rise will drown some mangroves and their carbon stocks but provide opportunities for new carbon accumulation, while cyclones can have immediate negative impacts on stocks but positive impacts on sequestration during recovery. Mangrove rehabilitation practices can actively restore carbon stocks and reduce greenhouse gas emissions from previous land uses. It is critical to consider the type of LULCC when estimating carbon emissions due to mangrove loss or rehabilitation. Mangrove blue carbon is now high on the international conservation policy agenda, and a better understanding of how carbon stocks and fluxes respond to anthropogenic and biophysical disturbance may provide better incentives for mangrove conservation and sustainable management.

Keywords: carbon sequestration, carbon stock, cyclone, degradation, land-use and land-cover change (LULCC), nature-based solution, payments for ecosystem services (PES), rehabilitation, sea-level rise,

1 Introduction

Mangrove forests are a threatened ecosystem found along tropical, subtropical and some warm temperate coastlines. Recent remote sensing studies suggest that the rate of global mangrove loss in the early twenty-first century was 0.16–0.39% per year (Hamilton and Casey, 2016), with hotspots of substantial mangrove loss across the tropics. Mangrove loss has been caused predominantly by land-use and land-cover change (LULCC), driven by the human use of the coastal zone for economic production. Proximate drivers of mangrove deforestation vary regionally (UNEP, 2014), but include conversion to aquaculture (Primavera, 2006), agricultural land uses such as rice and oil palm (Giri et al., 2008; Richards and Friess, 2016) and urban development (Branoff, 2017). Natural causes of mangrove loss also exist, particularly tropical cyclones and other climatic events (Krauss and Osland, 2020), though their scale of impact is substantially smaller than anthropogenic drivers of mangrove loss (Sippo et al., 2018). Regional-scale sea-level rise is also an increasing concern for mangrove forests (Lovelock et al., 2015).

Mangrove loss can have a real impact on the ecosystem services that mangroves provide to people (Estoque et al., 2018). Mangroves provide a broad range of ecosystem services, including provisioning services such as food production, timber, and pharmaceuticals, regulating services such as coastal protection and pollutant assimilation, and cultural services such as recreation, education, and spiritual value. A key ecosystem service provided by mangroves is their ability to regulate the global climate and mitigate anthropogenic climate change through carbon storage and sequestration. Mangroves are a ‘blue carbon’ ecosystem, alongside other coastal

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ecosystems such as salt marshes, seagrass meadows and tidal freshwater forested wetlands because they store and sequester carbon at greater densities than many terrestrial ecosystems (Donato et al., 2011). However, the carbon efficiency of mangroves means that their loss has a disproportionate impact on carbon emissions to the atmosphere compared to other ecosystems (Pendleton et al., 2012).

This article conceptualises the myriad links between mangrove area dynamics and blue carbon storage and sequestration, focusing on key anthropogenic (aquaculture, agriculture, rehabilitation) and biophysical (sea-level rise, tropical cyclones, temperature) drivers of mangrove area change, and understanding the differing impacts they have on mangrove blue carbon (Figure 1). Most models and estimates of carbon emissions due to mangrove loss assume that the same amount of carbon is emitted, regardless of the driver of mangrove loss. However, it is important to understand blue carbon dynamics under specific drivers of mangrove area change; LULCC and natural drivers have a gradient of impacts on carbon storage and sequestration (Sasmito et al., 2019), with implications on how emissions can be mitigated or managed. While mangrove blue carbon is currently being negatively impacted due to LULCC, this article also shows the different scales at which blue carbon can incentivise mangrove management, conservation and rehabilitation, to minimise blue carbon losses in the future.

2 Coarse Estimates of Mangrove Blue Carbon Dynamics

2.1 Global-scale Changes in Mangrove Blue Carbon

There are various estimates of global carbon emissions due to mangrove loss, ranging from 7 to 44 Mt CO_2e year^{-1} for mangrove soils, and 20–450 Mt CO_2e year^{-1} for mangrove soil carbon and biomass carbon combined (Table 1). For comparison, South Africa's total carbon emissions for 2013 were estimated at 330 Mt CO_2e (Olivier et al., 2014). The large variance in estimates of emissions from mangrove deforestation is due to differences in deforestation rates and the emission factors used, and the carbon pools and soil depths considered in the calculations (Table 1). Improvements in estimates of global mangrove extent and deforestation rate have particularly influenced the accuracy of emissions estimates; earlier emissions estimates relied on deforestation rates estimated through secondary data analysis (e.g. literature reviews), which suggested that mangroves were being lost at up to 3% per year (FAO, 2007). Later estimates of emissions have utilised deforestation rates that have been more accurately quantified through global-scale remote sensing, which put mangrove loss rates at almost an order of magnitude lower than previously estimated (Hamilton and Casey, 2016), with a similar effect on rates of carbon stock loss.

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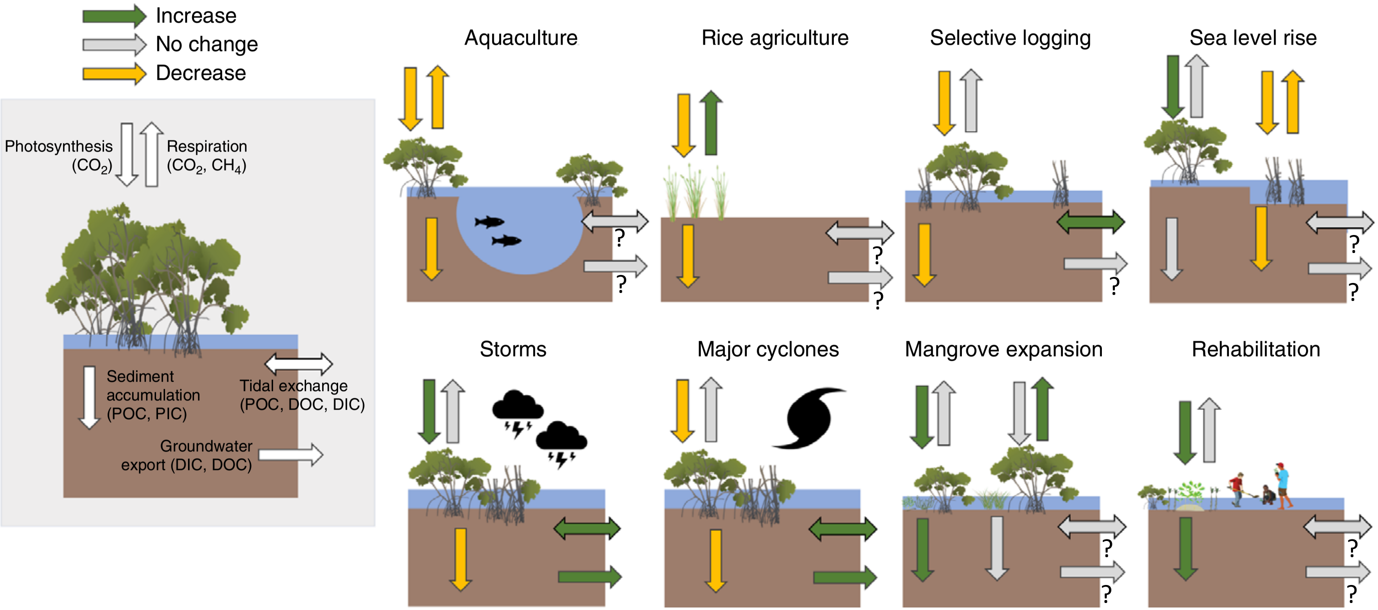

Figure 1 Conceptual diagram of how mangrove carbon stocks and fluxes respond to various land-use land-cover change (LULCC) activities and biophysical disturbances. DIC, dissolved inorganic carbon; DOC, dissolved organic carbon; PIC, particulate inorganic carbon; POC, particulate organic carbon. Source: Vector graphics courtesy of IAN Image Library, University of Maryland Center for Environmental Science.

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Table 1 Global annual emissions (Mt CO_2e year^{-1}) resulting from land-use and land-cover change (LULCC) in mangroves.

Annual emissions (Mt CO_2e yr^{-1})	**Carbon data description**	**Deforestation rate(s)**	**Emissions factor**	**Reference(s)**
Remote sensing/modelling estimates				
7–29	Global dataset Soil carbon (to 1 m deep) only	0.16–0.39% yr^{-1} (2000–2012)	0.43	Atwood et al. (2017)
111–447	Global dataset Soil carbon (to 1 m deep) only	0–0.42% yr^{-1} (2000–2015)	0.25–1.00	Sanderman et al. (2018)
79–317	Global dataset Biomass carbon and soil carbon to 1 m depth	0.17% yr^{-1} (2000–2012)	0.25–1.00	Hamilton and Friess (2018)
Coarse estimates				
20–120	Data from the Indo Pacific and global dataset of C density Biomass carbon and soil carbon (from 30 cm to >1 m depth)	1–2% yr^{-1} (1980–2005)	0.00–1.00	Donato et al. (2011)
90–450	Global dataset Biomass carbon and soil carbon to 1 m depth	0.7–3% yr^{-1} (1980–2003)	0.25–1.00	Pendleton et al. (2012)
84–159	Global dataset Aboveground biomass carbon and soil carbon to 1 m depth	0.7% (1990–2005)	0.75	Siikamäki et al. (2012)
24–39	Global dataset of soil C corrected for loss on ignition	0.17% (2000–2012)	1	Ouyang and Lee (2020)

Values are minimum and maximum estimates. Emissions factor = a representative value (from 0 to 1) of the proportion of GHG released from land-use conversion.

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2.2 Regional-scale Changes in Mangrove Blue Carbon

Carbon emissions from mangrove deforestation are particularly high in Southeast Asia, a region that has the largest mangrove area and some of the most carbon-dense mangrove systems (Atwood et al., 2017; Simard et al., 2019) but also some of the highest rates of mangrove loss globally (Hamilton and Casey, 2016). By combining datasets of national mangrove area change (Hamilton and Casey, 2016), national mangrove carbon stocks (Hamilton and Friess, 2018), and regional-scale information on mangrove LULCC (Richards and Friess, 2016), it is estimated that Southeast Asia lost 167.28 Mt CO_2e between 2000 and 2012, with carbon emissions almost equally split between aquaculture and agriculture (Table 2).

The magnitude and cause of mangrove carbon loss vary between countries in Southeast Asia, as rates and drivers of deforestation vary nationally (Figure 2). Myanmar has shown the highest twenty-first-century rate of mangrove deforestation globally (Hamilton and Casey, 2016), primarily due to mangrove conversion to rice agriculture (Richards and Friess, 2016). Between 2000 and 2012, conversion to rice generated emissions of 9.70 Mt of C (35.60 Mt CO_2e), equivalent to 8.1% of the remaining mangrove carbon stocks in Myanmar.

Malaysia and Indonesia have comparable mangrove carbon loss, when adjusted to their total carbon stocks (Figure 2). For the period 2000–2012, these two countries lost 2.1% and 2.7% of their 2012 mangrove carbon stocks, respectively. The main drivers of mangrove deforestation in Malaysia and Indonesia are agricultural land uses (particularly oil palm) and aquaculture (Richards and Friess, 2016). Aquaculture and agriculture are key for national economic and food security, with the two countries combined accounting for 84% and 8% of global palm oil and aquaculture production, respectively (FAO, 2018). Thailand and Cambodia showed lower rates of mangrove loss, with only 1.4% and 1.8% of their mangrove carbon stocks lost between 2000 and 2012. Vietnam, the Philippines, Brunei, Singapore, and Timor Leste showed the lowest carbon losses from mangrove deforestation in the region, with <0.7% of their national mangrove carbon stocks lost.

2.3 Issues with Current Global and Regional Estimates of Mangrove Carbon Emissions

While there are clear temporal and spatial patterns in mangrove blue carbon dynamics at global and regional scales, such large-scale viewpoints mask finer scale dynamics that differ between the various anthropogenic and natural drivers of mangrove area change. Current global and regional emissions estimates often make a key assumption that all LULCC drivers emit the same amount of mangrove carbon. In practice, this is not the case, as different LULCC drivers disturb mangrove carbon through different mechanisms.

Table 2 Southeast Asian mangrove carbon stocks and losses due to conversion to agriculture and aquaculture.

Country	Mangrove surface area in 2012[a] (km^2)	Mangrove surface area loss (2000–2012)[a] (km^2)	Annual rate of mangrove surface area loss (2000–2012) (%)	Mangrove C stock in 2012[b] (Mt C)	C loss from aquaculture (2000–2012)[a,b,c] (Mt C)	C loss from agriculture (2000–2012)[c] (Mt C)
Indonesia	23 324	749	0.26	1275	19.90	6.47
Myanmar	2557	235	0.70	119	0.17	9.70
Malaysia	4725	243	0.41	259	1.96	5.09
Thailand	1886	47	0.20	92	0.25	1.05
Philippines	2064	27	0.11	104	0.50	0.16
Cambodia	323	15	0.37	16	0.20	0.08
Viet Nam	708	8	0.10	34	0.08	0.04
Brunei	103	0.8	0.07	5	0.01	0.01
Timor-Leste	8.5	0.1	0.1	0.4	<0.01	<0.01
Singapore	1.7	0.0	0.0	0.1	0.0	0.0
Total	35 703	1325	2.32	1904.5	23.07	22.60

[a] Data from Hamilton and Casey (2016).
[b] Data from Hamilton and Friess (2018).
[c] Data adapted from Hamilton and Casey (2016), Hamilton and Friess (2018) and Richards and Friess (2016).

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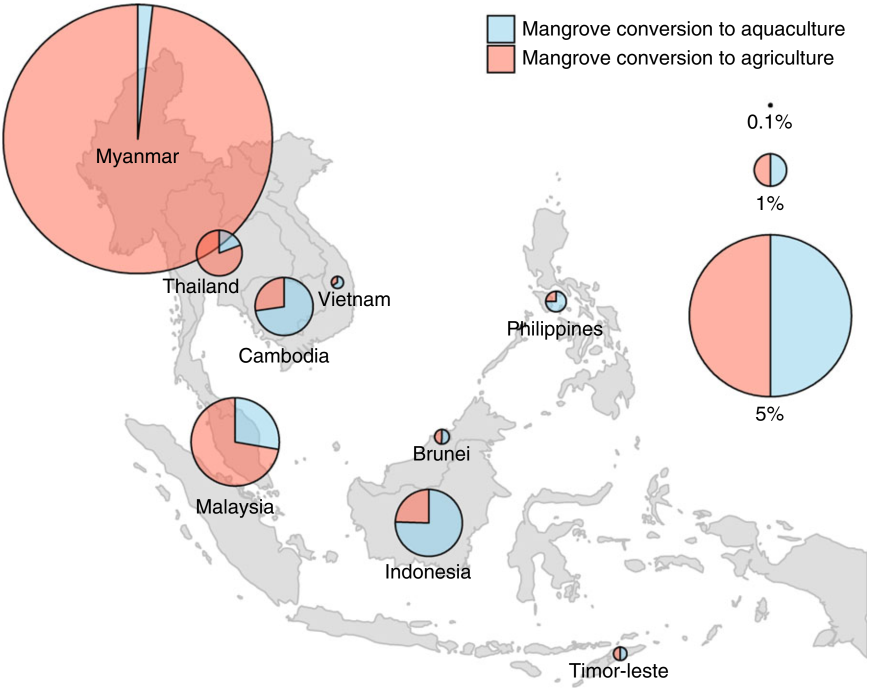

Figure 2 Spatial representation of carbon stock losses proportional to their total carbon stocks for the period 2000–2012 in Southeast Asia. The size of the pie chart is proportional to the quantity of mangrove carbon stock lost over the countries' remaining carbon stock (as of 2012).

Site-scale studies show that in general, highest emissions occur with LULCC drivers that cause full habitat conversion and disturbance of mangrove soils, while lower emissions result from practices such as small scale mangrove harvesting. For example, mangrove conversion to aquaculture can result in the emission of 83% of the carbon stock due to the removal of trees and substantial disturbance to the soil column (Kauffman et al., 2014), while mangrove harvesting causes the emission of less than 50% of the carbon stock, due to less soil disturbance (Adame et al., 2018a). Even where mangroves are killed entirely from hydrological alteration, as long as soils are not converted to alternate uses and remain wet, slowed decomposition and compaction of former mangrove soils can preserve soil carbon stocks for at least several decades (Krauss et al., 2018). To produce more robust global-scale estimates of mangrove carbon emissions due to LULCC, it is crucial to account for the specific emission factors of each land use (Sasmito et al., 2019).

To account for changes in carbon stocks and fluxes, international emissions reporting guidelines recommend two methodologies for quantifying carbon emissions and removals generated by LULCC (the 2019 Refinement to the

2006 IPCC Guidelines for National Greenhouse Gas (GHG) Inventories Volume 4 AFOLU; IPCC, 2019; see also IPCC, 2014). The stock change methodology allows carbon emissions and removals to be indirectly inferred from site chronosequences, which may or may not reflect flux change over space and time. In this approach, assumptions are often made for the soil carbon pool, for example that the top 1 m of the soil carbon pool is lost during all LULCC. This is appropriate for some LULCC activities such as aquaculture, where surface soils are excavated during pond development (Section 3.1), but not all LULCC activities.

The stock change approach also assumes that only in situ carbon stocks are lost, with no carbon exchange or flux between ecosystems, such as the transfer of dissolved organic, inorganic, and particulate carbon from disturbed mangroves to adjacent water bodies or seagrass beds (Pendleton et al., 2012). The fate of carbon in such exchanges, whether oxidised and released to the atmosphere, or remaining within marine environments for use in secondary production, is often not accounted for (Maher et al., 2018). However, carbon fluxes can add substantially to the emissions profile beyond that estimated from a stock change approach alone (Castillo et al., 2017; Maher et al., 2017). The flux change approach is less common in mangrove studies of LULCC (Sasmito et al., 2019) because it is technologically more difficult to estimate compared to the simpler stock change approach; carbon flux pathways show substantial complexity in mangroves, and their magnitude remains poorly quantified (Alongi, 2014). However, incorporating all carbon flux pathways will give a more complete picture of net carbon emissions and removals.

3 Blue Carbon Dynamics Under Human Land Use

When estimating carbon emissions from LULCC in mangroves it is important to understand why different LULCC drivers have different magnitudes of impact on carbon storage and sequestration, and how this leads to different emission profiles (Sasmito et al., 2019; Figure 1). Emissions profiles are influenced by the short-term loss of mangrove biomass and soil carbon immediately after conversion or disturbance, but longer-term mangrove carbon fluxes after immediate land-use change are driven by the soil and biomass characteristics of the final land use.

3.1 Emissions Profile of Aquaculture

Aquaculture is often considered to be the largest proximate driver of mangrove deforestation, particularly in Southeast Asia, Latin America, and parts of South America (Primavera, 2006; Quieroz et al., 2013). The 1970s and 1980s saw a boom in industrial onshore aquaculture with growing

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demand for seafood products. This led to substantial mangrove deforestation; 50% of the Philippines' mangroves were converted to aquaculture in ~35 years (Primavera, 1997), and more than 21 000 ha of mangroves were converted into aquaculture ponds within only 11 years in the Mahakam Delta, Indonesia (Rahman et al., 2013). Aquaculture continues to be a key driver of mangrove loss globally and remains the biggest driver of mangrove deforestation in Southeast Asia (Richards and Friess, 2016).

Aquaculture has substantial negative impacts on mangrove carbon stocks, leading to an average 83% reduction in biomass and a 52% reduction in soil carbon (Sasmito et al., 2019). Biomass carbon is lost with tree clearing prior to pond development (Rahman et al., 2013). Compared to LULCC drivers such as agriculture, soil carbon loss per unit area from aquaculture was 10% lower (Sasmito et al., 2019). This is presumably due to the heavy hydrological modification in agricultural land such as rice fields (Section 3.2), which allows soil organic matter to collapse due to decomposition. In contrast, aquaculture ponds commonly remain inundated throughout the year, limiting the decomposition process and resuspension of particulate organic matter. Soil carbon losses due to aquaculture are largely driven by the excavation of the top 1–2 meter of the soil column during initial development of ponds (Kauffman et al., 2018). Given the larger area of mangrove conversion into aquaculture compared to other land uses (Figure 2), carbon emissions from aquaculture expansion may be significant, particularly for aquaculture exporting countries such as Indonesia (Murdiyarso et al., 2015).

In addition to stock changes, aquaculture ponds emit GHGs, particularly CO_2 and CH_4 into the atmosphere as a result of soil respiration and methanogenic activity. However, compared to reference forests, the volume of emissions from active aquaculture is generally lower, particularly for CH_4 (Sasmito et al., 2019). However, our knowledge of aquaculture impacts on fluxes only applies to effluxes between the soil and air interface (sensu Cameron et al., 2019a; Sidik and Lovelock, 2013), while further carbon flux pathways must be accounted in order to improve the quantification of net carbon emissions. Further pathways include carbon emissions from the water–air interface, fluvial carbon export, carbon burial, and productivity. At the ecosystem level, GHG effluxes from aquaculture may be lower than reference mangrove forest sites; however, carbon inputs via burial and productivity here are also substantially lower, leading to aquaculture being a net GHG source (Arifanti, 2017). Ideally, assessment for net carbon emissions generated by aquaculture conversion will consider both stock and flux changes.

3.2 Emissions Profile of Agriculture

Agriculture has long been a driver of mangrove conversion, with mangroves in South America being cleared for coconut plantations since the nineteenth

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century (González et al., 2010). Coconut plantations continue to drive mangrove loss in South America (Santos et al., 2014), but other types of agriculture also impact mangroves. The potential for mangrove areas to be converted to rice has been known since the nineteenth century (Mouat, 1862), and is a major agricultural land use affecting mangroves in Southeast Asia, particularly Myanmar (Richards and Friess, 2016), as well as parts of East Africa (Giri and Muhlhausen, 2008; Beymer-Farris and Bassett, 2012) and West Africa. Oil palm has recently been identified as a key driver of mangrove loss in Indonesia, Malaysia (Richards and Friess, 2016) and Myanmar (de Alban et al., 2018).

Regardless of the type of agriculture, the first step in mangrove conversion to agriculture is biomass removal, resulting in direct carbon loss. With a mean aboveground biomass density of 129.1 MgC ha^{-1} (Simard et al., 2019), mangroves often have higher biomass than the cultivated vegetation that replaces them; above-ground biomass in oil palm plantations is estimated at 38–42 MgC ha^{-1} (Khasanah et al., 2015) and coconut plantation biomass at 51.1 MgC ha^{-1} (Bhagya et al., 2017). Soil carbon stocks are also lost during agricultural conversion, with a global meta-analysis of agricultural conversion compared to baseline mangrove forests showing high net soil carbon loss (Sasmito et al., 2019).

Mangrove conversion to agriculture requires hydrological manipulation such as soil dredging and drainage, leading to sediment remobilisation and exposure to aerobic conditions (Lovelock et al., 2017) and affecting soil carbon fluxes. Mangrove-derived carbon that was previously buried in water-saturated and anoxic sediment layers instead becomes oxidised via heterotrophic microorganism activity, releasing CO_2 to the atmosphere (Brodersen et al., 2019). Global meta-analyses suggest that on average there are no significant differences in CO_2 emissions before and after agricultural land-use change (Sasmito et al., 2019; O'Connor et al., 2020), though individual study locations do show changes in the emissions profile. For instance, CO_2 emissions on the coast of Honda Bay, Philippines, were up to seven times higher in undisturbed mangroves sites compared to a converted coconut plantation site (Castillo et al., 2017). Soil CO_2 emissions from undisturbed mangrove sites are the result of fresh organic matter decomposition and root respiration and are usually offset by CO_2 uptake from vegetation net primary productivity (Bouillon et al., 2008). Moreover, conventional agricultural systems require nutrient input, which could enhance soil microorganism metabolism and increase GHG emissions (sensu Whigham et al., 2009).

While impacts on CO_2 emissions are inconclusive, agricultural land uses such as rice can have substantial impacts on other GHGs such as CH_4 (Sasmito et al., 2019). Rice plantations are often water saturated, with low salinity values that favour methanogenesis over sulphate reduction (Poffenbarger et al., 2011). For instance, a study conducted in India reported an increase in CH_4 emissions by a factor of four between mangrove sediments and rice paddies (Chauhan et al., 2017). Ye et al. (2016) reported salinity and temperature to be the main factor controlling CH_4 emissions in coastal wetlands including rice fields, with greater emissions when salinity was <18 psi and soil temperature >18 °C, which covers large areas of mangrove extent. Increases in CH_4 emissions after agricultural conversion are concerning because CH_4 has a sustained global warming potential 45 times more important than CO_2 at the 100 year time scale (Neubauer and Megonigal, 2015), thus having a greater impact on the global radiative balance. This highlights the need to systematically characterise the land-use type when mangroves are converted to better estimate the effect of these changes and provide more accurate land management recommendations.

3.3 Emissions Profile of Forestry

Mangrove harvesting can lead to deforestation and concomitant carbon loss in areas where forest is clear-cut without planning for future management or regeneration and is common in regions where unmanaged and illegal logging occurs (Feka, 2015; Ilman et al., 2016). The total impact of forestry on carbon stocks depends on the scale of harvesting practice, whether large-scale clear felling in plantations, large-scale selective logging for industrial products such as wood chip or charcoal, or small-scale selective logging for community use. However, regardless of harvesting type, there will be biomass carbon (stems and branches) loss due to harvesting. There is also a decrease in soil organic carbon post-harvest due to loss of litterfall, soil acidification limiting decomposition (Kathiresan et al., 2014; Shaifullah et al., 2008) and mineralisation of organic matter (Kathiresan et al., 2014). Selective logging areas often follow the same trend, with smaller-scale logging having the least impact on soil carbon (Alongi and de Carvalho, 2008).

While mangrove carbon stocks are significantly reduced immediately after harvesting, carbon stocks change as the ecosystem recovers over time. Biomass carbon can recover to its pre-disturbance state after 30–50 years in the Indo-Pacific (Sillanpää et al., 2017). In certain sites, soil carbon shows a 25-year return period, with the first metre of the soil column showing an initial carbon loss post-harvest, followed by a rapid recovery (Sasmito et al., 2020). Post-harvest biomass carbon accumulation rates can range from 2.8 to 9.5 $\mathrm{Mg\,C\,ha^{-1}\,year^{-1}}$ (Adame et al., 2018a), while selective logging systems approximate $3.6 \pm 1.1\ \mathrm{Mg\,C\,ha^{-1}\,year^{-1}}$ (Sasmito et al., 2020). Biomass carbon stock recovery in harvest regimes can be attributed to rapid regrowth and reduced tree competition.

For carbon fluxes, harvesting immediately reduces carbon sequestration through photosynthesis (Gillis et al., 2017), and sediment CO_2 emissions decrease due to the loss of soil carbon to mineralisation and export to coastal waters, and decreases in root respiration (Grellier et al., 2017). In small-scale selective logging systems, CO_2 emissions may increase post-harvest with the decomposition of waste material and oxidisation of the disturbed soil by logging activity (Lang'at et al., 2014). However, the rates of CO_2 emissions in cleared mangroves are not always significantly different from undisturbed mangroves (Bulmer et al., 2015), with the emission rate depending on tidal regime and initial soil characteristics. The harvest methods employed also influence CO_2 emissions, as this determines whether remnant litter remains available for decomposition (Grellier et al., 2017).

Post-harvest carbon emissions often decrease with time as sites undergo mangrove regrowth and soil stabilisation. Carbon burial efficiency can increase with plantation stand age (Alongi et al., 2004), with selective logging systems following a similar trend (Alongi and de Carvalho, 2008). However, while carbon stocks are commonly measured, carbon fluxes other than atmospheric CO_2 emissions (such as dissolved organic carbon and dissolved inorganic carbon exchange) are rarely studied across mangrove harvest regimes. Further research on carbon pool fluxes in mangroves under different harvest regimes will allow further understanding of the impact of harvest management on blue carbon.

4 Blue Carbon and Climate Change

Climate change is expected to have both positive and negative impacts on mangrove extent and carbon stocks. Sea-level rise is projected to threaten mangroves across their entire range, and projected increases in cyclone activity will have region-specific impacts on mangrove structure. However, sea-level rise also promotes the landwards expansion of mangroves in some locations, and increased air and water temperatures at latitudinal limits allow mangroves to expand into neighbouring temperate intertidal ecosystems that have different carbon stock and sequestration characteristics.

4.1 Impacts of Sea-level Rise on Mangrove Blue Carbon

Sea-level rise is both a threat to carbon stocks and a potential driver of carbon accumulation. Mangroves are threatened when vertical soil accumulation lags behind sea-level rise over time, increasing periods of inundation beyond the physiological tolerance of mangrove species. Many mangrove forests experience such surface elevation deficits (Lovelock et al., 2015) often as a natural condition during rising seas, though submergence can take millennia to occur when deficits are low.

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It is assumed that mangrove mortality due to sea-level rise would result in the loss of biomass and soil carbon. As mangroves are submerged, carbon is rapidly lost as trees die, and since this necromass is mostly exposed to the atmosphere it is most likely decomposed with greater efficiency than soil carbon. Much less can be assumed about the fate of soil carbon under submergence. It is often assumed that carbon is mineralised and released to the atmosphere as CO_2 from soil and creek respiration, though recent evidence suggests that a large amount of particulate and dissolved organic carbon is exported after disturbance (Maher et al., 2018) and may become sequestered within deep oceanic sinks. Counting all standing mangrove carbon stocks as lost during submergence would overestimate total carbon losses. Soil submergence may also not lead to the loss of the entire submerged peat layers, as evident in cores taken through intact mangrove peats present meters below historical habitat transitions (Toscano et al., 2018). Finally, as mangrove trees are killed by submergence, live root turnover is stymied, leading to soil collapse just below the active root zone over time. This causes soil volume loss but not necessarily an immediate loss of total soil carbon (Krauss et al., 2018; Chambers et al., 2019).

Sea-level rise does not automatically lead to blue carbon loss but can encourage mangrove expansion and carbon accumulation. Low to moderate rates of sea-level rise (as occurred during the 'Big Swamp Phase' of coastal wetland development, 6800–5300 years BP; Woodroffe et al., 1993) allow mangroves to colonise suitable intertidal locations and build soil surface elevations vertically through sediment accretion, litter and woody debris accumulation, root growth, and microbial and algal development (Krauss et al., 2014). All of these processes encourage long-term soil carbon accumulation. Mangrove sediments from the Caribbean document continuous peat development throughout the Holocene at sea-level rise rates below 5 mm year^{-1} (Toscano and Macintyre, 2003), with peat development of up to 10 m on some carbonate islands (McKee et al., 2007). These peats were built by root growth stimulated by the slow emergence of accommodation space (i.e. vertical space available to be occupied by soil development, based on tidal range) created by gradually rising seas. Root growth occurs into vertical soil zones of low oxygen that reduce decomposition below that gained by new root growth, which over time, expand soil surfaces vertically (McKee et al., 2007) and facilitate carbon accumulation. Sea-level rise also drives shoreline transgression, causing mangroves to migrate landwards (if possible) and claim new blue carbon areas over time. For example, mangroves expanded inland in South Florida from 1927 to 2005 when sea-level rise rates were <3 mm year^{-1}, adding 35% to mangrove coverage (Krauss et al., 2011).

Maintenance of accommodation space with sea-level rise is an important driver of mangrove soil blue carbon accumulation. However, sea-level rise and accommodation space are not uniform globally. For example, eastern Australia experienced a period of sea-level fall just after a mid-Holocene

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period of sea-level stability, while the east coast of the Americas experienced a slow rise in sea level after the mid-Holocene that continues today (Woodroffe and Davies, 2009). Chronosequence studies in Australian mangroves suggest that soil carbon burial is among the highest in the world, but standing stocks do not register similarly high volumes of carbon in soil profiles (Rogers et al., 2019). Soil decomposition is still occurring, but without sea-level rise feeding a need for wetlands in Australia to build vertically over the last few thousand years through reduced accommodation space, carbon had enough time to exit the system through decomposition and respiratory pathways. Therefore, rising sea-levels in the mid-Holocene, versus falling sea levels, promoted 1.7–3.7 times greater soil carbon stocks in wetlands in the Americas versus Australia because accommodation space was sustained by sea-level rise in the Americas from the mid-Holocene to present (Rogers et al., 2019). Ongoing sea-level rise, in addition to multiple other factors, has resulted in the upslope migration of mangroves into previously saltmarsh dominated habitat in multiple sub-tropical and temperate areas resulting in the accumulation of and storage of carbon (Kelleway et al., 2016; Rogers et al., 2019).

4.2 Impacts of Increased Cyclone Activity on Mangrove Blue Carbon

Cyclones can have substantial impacts on mangroves in regions where they occur, accounting for 45% of all structural damage reported in mangrove ecosystems globally (Sippo et al., 2018). Cyclone impacts on mangroves include broken canopies, prolific downed woody debris production, anomalous surface sediment distributions, scour channels, and greater soil anaerobiosis in places (Krauss and Osland, 2020). Some mangrove species are particularly susceptible to wind damage, and not all species are able to regrow quickly after such disturbance (Villamayor et al., 2016). Climate change is expected to further increase the impact of tropical cyclones on mangrove forest structure and carbon in the future, by increasing the frequency of the strongest tropical cyclones, causing greater rainfall during cyclones and expanding cyclone impacts poleward (Patricola and Wehner, 2018; Krauss and Osland, 2020). Greater cyclone frequency may alter long-term mangrove recovery, and repetitive tropical cyclones have compounding impacts on mangrove forests and their carbon (Dutta et al., 2015).

Cyclone damage causes biomass carbon loss; a review of cyclone damage studies suggested that forest mortality averaged 40%, varying from an average low of 13% to an average high of 67% (Krauss and Osland, 2020). Complete removal of mangrove cover and concomitant loss of carbon has been documented in the Everglades region of Florida, USA, where tropical cyclone impact frequency has been high during parts of the Holocene (Jones

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et al., 2019). While mangroves are disturbance-adapted ecosystems (Lugo et al., 1981), tropical cyclones of even moderate intensity (i.e. maximum sustained wind speeds > 178 km h^{-1}) cause catastrophic damage to mangrove biomass carbon stocks. Over time, cyclones encourage the development of short-statured forests (Simard et al., 2019), which are more resilient to cyclone impacts but have reduced above ground biomass carbon stocks. The influence of cyclones on soil carbon stocks is expected to be less, as long as recovery ensues in a reasonable period of time to prevent peat collapse (sensu Chambers et al., 2019).

While cyclones have a rapid negative impact on carbon stocks, they can have a potentially positive impact on carbon sequestration. Mangroves often recover from cyclone events through prolific regeneration or re-sprouting in the years following the storm event, which may facilitate rapid biomass and root growth (including 'over yielding'; Lang'at et al., 2013) in some locations during recovery (Ward et al., 2006). Greater biomass carbon gain in the years following a cyclone must be balanced against structural losses caused by the cyclone, such that biomass carbon stocks are in a constant state of flux over decadal time scales along cyclone-prone coastlines.

4.3 Blue Carbon Implications of Latitudinal Mangrove Expansion

Mangroves are freeze intolerant, so are limited to tropical, sub-tropical and some warm temperate coastlines. Mangrove expansion at their latitudinal limit is controlled by regional changes in temperature and precipitation (Saintilan et al., 2014; Osland et al., 2017). Recent decreases in the frequency, duration, and severity of freezes have allowed for mangrove expansion to occur across multiple latitudinal range limits (Osland et al., 2013; Cavanaugh et al., 2014). Aridity is a further control of mangrove range limits (Osland et al., 2017; Sanders et al., 2016). Upslope mangrove expansion has also been studied heavily but is most often due to changes in sea level or hydrology (see Section 4.1).

With the expansion of mangroves into previously saltmarsh-dominated systems at their latitudinal limits, an increase in biomass carbon typically occurs. Increases in biomass carbon stocks are due to the greater biomass and the woody nature of mangroves compared to the replaced saltmarsh, even when mangroves are restricted to shrubby or stunted forms (Doughty et al., 2016; Yando et al., 2016; Kelleway et al., 2016; Simpson et al., 2019; Raw et al., 2019). Changes in soil carbon stocks are more complex and are often driven by freshwater availability, salt marsh species, and the subsequent productivity of saltmarshes, as well-developed soils already exist prior to mangrove expansion (Yando et al., 2016). Mangroves replacing productive saltmarshes (*Spartina* dominated) in moist areas often result in relatively little soil carbon change (Perry and Mendelssohn, 2009; Henry and Twilley, 2013; Doughty et al., 2016; Yando et al., 2016) although Simpson

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et al. (2019) saw large changes despite the already productive salt marshes. Mangroves replacing less productive, succulent-dominated saltmarshes in arid locations may conversely see large changes in carbon stocks under mangrove expansion (Comeaux et al., 2012; Bianchi et al., 2013; Kelleway et al., 2016; Yando et al., 2016), although Raw et al. (2019) did not see this development and hypothesised that mangroves had not yet had time to contribute meaningfully to soil carbon stocks. Soil carbon stocks with mangrove expansion are further controlled by soil moisture and hydrologic regime (Raw et al., 2019) as anoxic conditions from water-logging result in the accumulation of carbon-rich organic matter and low decomposition rates. Carbon accumulation rates, however, may occur on decadal lengths and soil carbon stocks, regardless of setting, may take longer periods of time to develop (Kelleway et al., 2016; Raw et al., 2019).

Carbon fluxes under mangrove expansion have been far less studied compared to carbon stocks. Soil CO_2 flux experiments using the natural saltmarsh-mangrove ecotone to predict future changes on the Atlantic Florida Coast saw little difference during the growing season in soil CO_2 flux, but significantly less flux in saltmarshes compared to mangroves outside of the growing season (Simpson et al., 2019). This highlights potential future implications of mangrove expansion and is likely due to the lack of activity and turnover by saltmarsh species outside of the growing season compared to mangroves (Hopkinson et al., 1978) in addition to the differences in soil temperature across the latitudinal gradient (Kirwan and Blum, 2011; Chen et al., 2012) with warmer temperatures. Paired biomass comparisons in a *Juncus* spp. marsh with nearby mangroves on the Florida Gulf Coast highlighted the role of both soil moisture and temperature in soil carbon flux (Lewis et al., 2014). This study also highlighted the highly recalcitrant *Juncus* spp. saltmarsh vegetation, going against the expectation that herbaceous saltmarshes are less recalcitrant than mangroves.

5 Rehabilitation of Mangrove Blue Carbon

The rehabilitation of degraded mangroves can potentially reinstate carbon lost during LULCC. However, most mangrove rehabilitation projects conducted to date have focused on low-diversity planting projects, with mixed success despite significant investments (Primavera and Esteban, 2008). Many attempts fail completely, principally through planting of inappropriate species in environmental settings not suitable for their establishment (Bayraktarov et al., 2016; Wodehouse and Rayment, 2019). This includes planting mangrove propagules in lower intertidal flats such as seagrass meadows, which are often too low in the tidal frame for mangroves to grow. Even where successful, such efforts replace another valuable and established intertidal ecosystem. Successful rehabilitation is instead contingent on the restoration of hydrological regimes to facilitate multispecies, unassisted recruitment of mangrove seedlings that establish at natural tidal inundation levels (Lewis, 2005). The large-scale hydrological restoration of mangroves has been successfully achieved in Vietnam's Mekong Delta (Nam et al., 2016) as well as sites in Florida and Latin America (Lewis and Gilmore, 2007), with similar success at smaller scales in Indonesia (Djamaluddin, 2007). Aside from greater species diversity, mangroves that establish at their natural tidal inundation levels tend to exhibit higher rates of growth and accumulate greater biomass over time. This results in more stable systems with greater degrees of ecological functionality than artificially replanted sites, as well as higher biomass carbon stocks (Alongi, 2012; Cameron et al., 2019b).

Maximising gains in carbon stocks through rehabilitation requires practitioners to consider factors such as geomorphic setting and biophysical conditions. More productive mangroves generally occur in deltaic systems with extensive riverine and hinterland zones typically traversed by a mosaic of rivers and streams (Lugo and Snedaker, 1974; Thom, 1984; Woodroffe, 1992). Such systems generally have greater biomass than restored sites on coastal fringes or oceanic coral atolls (Cintron and Novelli, 1984; Cameron et al., 2018). Biomass accumulation is driven by the interaction of biophysical factors conducive to high growth, such as nutrient availability, temperature, sediment supply, precipitation levels, freshwater influx and tidal amplitude (Alongi, 2009, 2014; Cameron et al., 2018; Ochoa-Gómez et al., 2019).

Rates of soil carbon stock change under mangrove rehabilitation are variable, given the heterogeneous environments in which mangroves exist, and the wide reported range in rates of sediment accumulation. Given the right circumstances, soil carbon burial rates post-rehabilitation may be substantial. Rapid increases in organic soil carbon of 50 Mg C ha^{-1} in just two years (Matsui et al., 2010), increasing to 83 Mg C ha^{-1} over 10 years (Matsui et al., 2012) was recorded following the rehabilitation of an abandoned aquaculture pond in Thailand. In this example, rehabilitation required the mechanical excavation of the site to 25 cm below the previous bed level, in order to restore hydrological flow. Similarly in Sulawesi, Indonesia, hydrological restoration through the deconstruction of pond walls led to the rapid infilling of sediments and subsequent soil carbon burial (Cameron et al., 2018). These results suggest that hydrological restoration through mechanical levelling, though expensive, or strategic breaching of barriers (less expensive) can promote high rates of allochthonous (ex situ) and autochthonous (from in situ primary productivity) soil carbon sequestration.

In addition to soil carbon accumulation, mangrove rehabilitation can be used to control GHG flux. Relative tidal inundation period is a key control on baseline GHG flux (Ahmed et al., 2017; Cameron et al., 2019a). This varies with the hydrological status of aquaculture ponds, which can exist in various states depending on their management status, including

(i) impounded ponds with intact walls and limited tidal mixing; (ii) dry and exposed with no tidal influx; and (iii) tidally inundated due to an open gate or breached wall. Typically, GHG emissions are lowest under impounded conditions (e.g. $1.1 \pm 0.2\,\mathrm{Mt\,CO_2e\,ha^{-1}\,year^{-1}}$ for ponds in South Sulawesi) and significantly higher when the pond is permanently dry and the soil exposed ($30.6 \pm 1.9\,\mathrm{Mt\,CO_2e\,ha^{-1}\,year^{-1}}$ for adjacent ponds in South Sulawesi), while partially inundated ponds fall between these two extremes ($17 \pm 0.8\,\mathrm{Mt\,CO_2e\,ha^{-1}\,year^{-1}}$ for ponds in North Sulawesi. Cameron et al., 2019b). Tidal inundation reduces the duration when soils are exposed and organic matter can be oxidised, and a significant consideration given gaseous exchange between soils and the atmosphere is greater than that from water to the atmosphere (Alongi, 2014). Additionally, rehabilitation can result in an overall net increase in soil CO_2 emissions due to respiration from lenticels and roots as the mangrove develops. Autotrophic respiration serves as a means of (*inter alia*) oxidising edaphic conditions to facilitate development of symbiotic mycorrhizae communities (Malhi et al., 2011; Cameron et al., 2019a), and when combined with heterotrophic respiration, can significantly increase net CO_2 flux. This effect, however, is outweighed by overall net carbon sequestration as the ecosystem develops.

6 Management and Policy Options for Blue Carbon Conservation

There is substantial management and policy interest in using blue carbon as a vehicle for mangrove conservation to protect remaining carbon stocks and their associated co-benefits. Blue carbon conservation can occur at the site-scale, through payments for ecosystem services (PES), where donors or investors pay custodians to change land-use practices or otherwise protect forest resources. Blue carbon conservation can also occur at the national scale, when incorporated into national-level carbon accounting systems, as part of a country's obligations to international climate change agreements.

6.1 Payments for Ecosystem Services

Blue carbon can incentivise mangrove conservation and restoration, with an ecosystem service buyer funding an ecosystem service provider to change land-use practices or otherwise avoid deforestation through PES. PES could provide a financial incentive to reduce many of the anthropogenic causes of mangrove loss described in this article. A range of potential financing mechanisms exist, including mandatory carbon credit schemes such as the Clean Development Mechanism under the Kyoto Protocol of the United

Nations Framework Convention on Climate Change, or voluntary carbon credit markets.

Despite strong interest from the international donor community over the last decade, mangrove PES remains at an embryonic stage. A handful of pilot studies exist in Kenya, Indonesia, Senegal, Myanmar, India (Wylie et al., 2016), and most recently in Madagascar (Rakotomahazo et al., 2019). Key lessons in PES implementation are being learned from these projects (Wylie et al., 2016), though substantial upscaling and investment are needed to have an appreciable impact on GHG emissions and local livelihoods.

PES faces many challenges generally, though some are specific to mangrove landscapes and are related to biophysical, economic and governance constraints. PES requires gains in carbon stocks to be permanent, but the dynamic biophysical setting where mangroves are found provides challenges to carbon permanence. Mangroves can be impacted by alongshore coastal management decisions that change local hydrodynamics and increase erosion. Anthropogenic changes in upstream catchments can affect mangrove health, such as increases in fluvial pollution or reductions in the sediment budget. Additionally, nonanthropogenic impacts such as tropical cyclones can cause carbon loss but can also not be controlled. These factors are external to the PES site, so are outside of the control of a PES site manager (Friess et al., 2015). PES projects, therefore, need to be framed against the degree of risk that can undo carbon gains. This risk must be evaluated, mitigated or accommodated, and requires management actions such as credit buffers (where more credits are generated than sold, in order to compensate for losses) and large-scale threat evaluations (Friess et al., 2015).

To be cost-effective, carbon credit revenue must be greater than the revenue produced by LULCC, known as the opportunity cost. Some LULCC activities in mangrove areas can be highly profitable, leading to high opportunity costs that reduce the financial viability of mangrove PES. For example, a carbon credit price of US$156 per tonne of carbon dioxide-equivalent is needed to compensate for highly productive aquaculture operations in Thailand (Thomas, 2014), though carbon markets generally sell credits at a price that is an order of magnitude lower than this. The financial competitiveness of mangrove PES projects reduces further when the substantial transaction costs of PES implementation are considered (Thompson et al., 2014).

In many coastal settings, it is unclear which government agency is ultimately responsible for mangrove management, as this intertidal ecosystem falls through the gap between terrestrial and marine management (Primavera, 2000). In some situations, mangrove PES has been used by government agencies as a tool to bring mangrove areas back under state control, to the exclusion of traditional mangrove users (Beymer-Farris and Bassett, 2012). Land tenure in many developing nations is often based on a range of formal and informal land claims and traditional ownership; unclear and competing land tenure claims in the coastal zone create uncertainty in who owns the blue carbon at a particular mangrove site, or to whom carbon credits should be shared with (Locatelli et al., 2014).

6.2 National/International Carbon Policy Mechanisms

Strategies to remove carbon dioxide from the atmosphere are considered essential to maintain global temperature increases below 2 °C by the end of the century. Among the different removal technologies, soil carbon sequestration and afforestation have been highlighted as important and cost-effective strategies (Griscom et al., 2017). Carbon sequestration by mangroves has an advantage compared to other natural climate solutions because while their global area is small, carbon stored per surface area is the highest among all terrestrial and coastal ecosystems (Taillardat et al., 2018), so the same amount of carbon can be stored in a smaller area for mangroves compared to other vegetated ecosystems.

Carbon accounting is being conducted at the national scale, as most countries are obligated to report their vegetated carbon sinks and sources to the United Nations Framework Convention on Climate Change. Every five years, countries must report their Nationally Determined Contributions (NDCs), that outlines how each country intends to meet their obligations to the Paris Agreement (UNFCCC, 2015). Numerous countries have discussed the use of mangrove blue carbon to meet their NDCs (Herr and Landis, 2016), though few countries have set quantitative targets. An exception is Belize, who's NDC calculates that 379 Gg CO_2 can be captured between 2015 and 2030 by sequestration and avoided deforestation (Herr and Landis, 2016). Instead, most NDCs describe the general role of mangroves or describe site-scale interventions that have resulted in carbon savings.

One reason that may explain a site-scale focus of many NDCs is the fact that mangroves may struggle to help achieve many NDC pledges compared to other ecosystems. Currently, many mangrove forests do not fulfil the definition of forests and are not included in their national inventory. For instance, in Mexico, forests are considered to have trees >4 m height, leaving out a large portion of mangroves of the country, and thus highly undervaluing their contribution to meet emissions targets (Adame et al., 2018b). Also, while mangroves store higher amounts of carbon per unit area, their areal coverage in many countries is small, so mangroves play a negligible role in offsetting national GHG emissions (Taillardat et al., 2018). However, for countries with extensive coastlines, limited deforestation and moderate GHG emissions, mangroves can offset a small portion of their national GHG emissions. This was the case for Nigeria, Bangladesh, and Colombia where mangrove carbon sequestration was estimated to offset 1.6%, 1.5%, and 1.1% of their 2012 national GHG emissions, respectively (Taillardat et al., 2018). When compared to emissions from the LULCC sector only, mangrove blue carbon can make a greater contribution, e.g. accounting for 6–10% of

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Mexico's target emissions for the Paris Agreement (Adame et al., 2018b). Although mangrove carbon sequestration will never become a large scale solution to mitigate global carbon emissions, it represents an affordable and readily available strategy, and secures additional co-benefits provided by the mangrove ecosystem. All carbon dioxide removal techniques must be considered and do not necessarily have to be significant at the global scale to be relevant for climate policies (Bellamy and Geden, 2019).

7 Conclusions

While mangrove forests are important stores of blue carbon, the long-term sustainability of these stocks is at risk from a variety of anthropogenic and biophysical threats. Though rates of global mangrove loss have decreased in recent decades, mangroves and their blue carbon stores are still heavily threatened in many countries, and carbon emissions to the atmosphere due to mangrove loss will continue to be significant across many landscapes.

While we have coarse estimates of carbon emissions at the global scale using a stock change approach, it is clear that the true extent of carbon emissions from mangrove deforestation is much more complex, and carbon emissions modelling needs to be conducted at a higher resolution, with a greater incorporation of the varied processes that affect carbon cycling in mangroves. Models must move beyond simple stock change approaches to also encompass changes in shorter-term carbon fluxes, and we must understand how the type of LULCC can influence carbon emissions. Carbon fluxes are generally underrepresented in emissions reporting, and more studies are required to directly measure fluxes to better reflect the full changes in the mangrove carbon cycle in response to LULCC activities.

It is encouraging that mangroves and their carbon are increasingly on the conservation and policy agendas of many countries and intergovernmental organisations. Mangrove conservation is being folded into international discussions on biodiversity conservation and carbon emissions reductions, as well as being a justification for rehabilitation projects. A deeper understanding of mangrove blue carbon and its dynamics under anthropogenic pressures will better enable us to conserve and manage this important coastal ecosystem, and reduce its contribution to climate change in the future.

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