CHAPTER 3

Human Impacts on Blue Carbon Ecosystems

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HIGHLIGHTS

1. Coastal blue carbon ecosystems are often associated with deltas and estuaries; they are highly productive and thus have been sites of intense human activities for thousands of years.
2. Degradation of blue carbon ecosystems has been caused by impoundment, conversion to alternative land uses, over exploitation, and pollution, all of which contribute to CO_2 emissions and reduce carbon sequestration and other ecosystem services offered by blue carbon ecosystems.
3. Emerging threats include damming rivers, reducing sediment supply and increasing nutrient availability, which are likely to exacerbate the effects of climate change and sea-level rise on blue carbon ecosystems.
4. Land-use planning for sustaining blue carbon ecosystems with sea level rise has the potential to increase the area of blue carbon ecosystems and global carbon sequestration.

3.1 INTRODUCTION

Coastal blue carbon ecosystems occur mainly in estuaries and deltas, which is where 20% of the human population of the planet live at densities that are three-fold that in inland areas (Small and Nicholls 2003). This confluence of human settlements and estuaries, deltas and embayments in which blue carbon ecosystems are abundant is no coincidence, as blue carbon ecosystems provide a wide range of services including the provision of food, fuel, fisheries, nutrient processing, and coastal protection (UNEP 2014). However, the colocation of human settlements and blue carbon ecosystems has resulted in losses and degradation of these ecosystems as human settlements have expanded, watersheds have been transformed, economies have become increasingly complex and blue carbon ecosystems have been over-exploited or converted to alternative land-uses (Pendleton et al. 2012).

Human activities on coasts and coastal watersheds have increased with population growth and associated increasing food demand, demands for fuel and timber, increasing urbanization, the extensive use of fertilizers, and extraction of water resources; all of which have led to loss, conversion, and degradation of mangroves, seagrass, and tidal marsh habitats and associated loss of ecosystem services. Emissions of CO_2 after human disturbances of blue carbon ecosystems occur as biomass is removed and often burned and organic matter in soils is oxidized (Table 3.1, Pendleton et al. 2012). Many disturbances enhance the decomposition of the large organic matter stocks held within soils of blue carbon ecosystems, leading to high levels of CO_2 emissions (Figure 3.1), (Mount and Twiss 2005; Marba et al. 2015; Kauffman et al. 2014, 2017; Lovelock et al. 2017). Human activities that result in the loss and degradation of blue carbon ecosystems also result in the loss of carbon sequestration potential as they are often replaced by systems that do not sequester as much carbon as blue carbon ecosystems (Duarte et al. 2013), and are often net sources of CO_2, methane, and nitrous oxide to the atmosphere (Figure 3.2) (Hu et al. 2013; Webb et al. 2016).

The major impacts and timing of human activities on coastal blue carbon ecosystems differ among blue carbon ecosystems and regionally, which has implications for regional strategies for the conservation and restoration of blue carbon stocks. Human activities can have direct effects on blue carbon ecosystems, for example, clearing and conversion of habitat for alternative uses (e.g., conversion to aquaculture, Kauffman et al. 2014, 2017), or indirect, where activities on the land (e.g., fertilizer use, changing hydrology) or in the sea (e.g., overfishing) can have negative effects on blue carbon ecosystems, their carbon stocks and carbon sequestration potential (Deegan et al. 2012; Atwood et al. 2015). While the effects of human impacts are evident on all coastal wetland types, in this chapter we focus on mangrove, seagrass, and tidal marshes (see Chapter 1).

Table 3.1 Global Cover of Blue Carbon Ecosystems, Their Rates of Conversion to Alternative Land-Uses, Estimated CO_2 Emissions due to Human Activities (see Figures 3.1 and 3.2) and Their Estimated Cost

Ecosystem	Global Extent (Mha)	Conversion Rate (% year^{-1})	Organic Carbon in Biomass and the Top Meter of Sediment (Mg CO_2 ha^{-1})	Carbon Emissions (Pg CO_2 year^{-1})	Estimated Cost (Billion US$ year^{-1})
Tidal marsh	2.2–40 (5.1)	1.0–2.0 (1.5)	237–949 (593)	0.02–0.24 (0.06)	0.64–9.7 (2.6)
Mangrove	13.8–15.2 (14.5)	0.7–3.0 (1.9)	373–1,492 (933)	0.09–0.45 (0.24)	3.6–18.5 (9.8)
Seagrass	17.7–60 (30.0)	0.4–2.6 (2.5)	131–522 (326)	0.05–0.33 (0.15)	1.9–13.7 (6.1)
TOTAL	33.7–115.2 (48.9)			0.15–1.02 (0.45)	6.1–41.9 (18.5)

Source: Data are from Pendleton et al. (2012).
Values are minimums and maximums with the central estimate in parentheses.

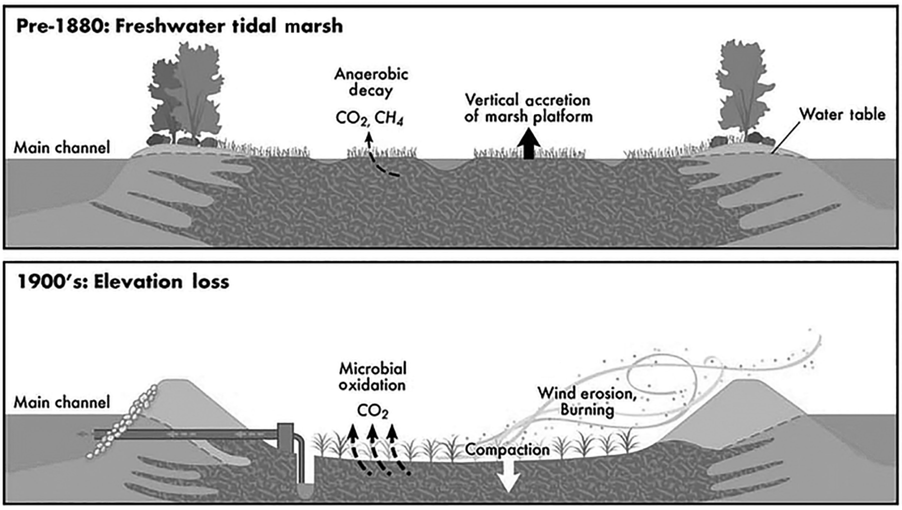

Figure 3.1 Consequences for the soil carbon stock (original condition, upper panel; altered condition, lower panel) when blue carbon ecosystems are impounded, drained, and converted to agricultural lands. (Source: Mount and Twiss (2005) http://escholarship.org/uc/item/4k44725p, originally from Ingebritsen et al. (2000).)

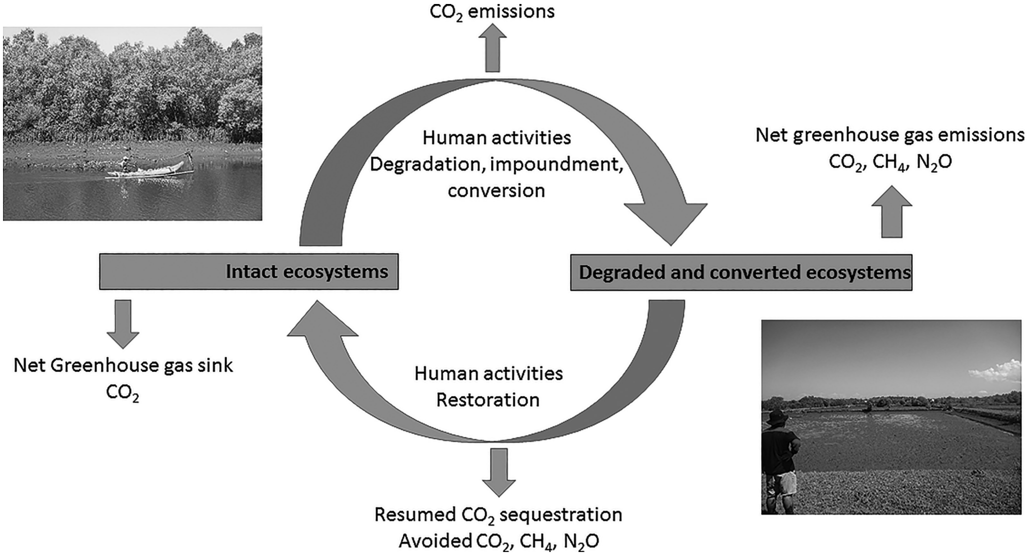

Figure 3.2 **(See color insert following page 266.)** Representation of the effects of human activities on greenhouse gas emissions with degradation, impoundment, and conversion of blue carbon ecosystems, and their restoration.

3.2 MANGROVES

Mangrove ecosystems are distributed over 121 countries along tropical to subtropical coastal zones across the world (Giri et al. 2011; Adame et al. Chapter 27; Giri et al. Chapter 13; Kairo et al. Chapter 24; Murdiyarso Chapter 21). Large-scale human impacts on mangrove forests have been relatively recent, though mangroves were overexploited historically such as the use of mangrove timber to construct fleets of boats and for the construction of buildings on the east coast of Africa (Curtin 1981). Numerous other colonial accounts of mangrove resource extraction also exist (Friess 2016).

Beginning in the 1970s and continuing until today, the expansion of shrimp aquaculture in response to demand for seafood led to widespread losses of mangrove forests throughout central America and Asia as ponds were constructed within the intertidal zone (Valiela et al. 2001; Alongi et al. 2002; Hamilton and Lovett 2015). Many ponds were unsustainable due to low-water quality and disease within the ponds, leading to the need to clear more area to maintain production. In some nations, 80% of mangrove forests have been converted to other uses (e.g., Philippines, Primavera 2005). In addition to aquaculture, agriculture, charcoal production, urbanization, construction of infrastructure, conversion to salt ponds, and exploitation of firewood have been associated with losses in some regions (Valiela et al. 2001; Alongi et al. 2002; Richards and Friess 2016). Diversion and modification of upstream river flows due to dams associated with hydroelectric development and irrigation projects have had global influences on coastal ecosystems (Cloern et al. 2016). For example, in the Chao Phraya Delta in the Gulf of Thailand, reduced sediment accretion within mangrove forests with upstream damming of rivers and low-sediment availability has resulted in losses of forest area as coastlines have retreated under the influence of storms and sea level rise (Goisan et al. 2014).

Indirect negative effects on mangroves due to modifications in hydrology (Jimenez et al. 1985), eutrophication (Lovelock et al. 2009), and pollution (Bayen 2012) have been documented or anticipated. Often indirect effects contribute to vulnerability to extreme events, like intense storms. For example, Feller et al. (2016) showed that fertilized trees had greater susceptibility to hurricanes in Florida, and Lovelock et al. (2009) found that fertilized trees had higher mortality with drought. Human activities that lower the productivity of blue carbon ecosystems are likely to enhanced vulnerability to climate change. For example, reduced root production is likely to lead to vulnerability to sea level rise in forests where roots contribute more than sediment to soil elevation gains (McKee et al. 2011).

Human activities have driven the loss of an estimated 30% of the original global mangrove extent (Alongi 2002), although there is a high level of uncertainty around regional trends (Friess and Webb 2014) and the extent of original mangrove cover. Whatever the exact rate of mangrove loss in the second half of the 20th century, it is clear that global annual losses in mangrove area have slowed in the first decade of the 21st century, averaging at approximately 0.2% per year (Hamilton and Casey 2016). However, while the overall global trend may be one of a reduction in deforestation rates, some countries such as Myanmar still show rapid rates of loss between 2000 and 2012 (Richards and Friess 2016). While in more recent decades aquaculture remains an important threat to mangroves, impoundment for conversion to rice agriculture, cattle pastures, and oil palm have been reported as increasing in impact (Kauffman et al. 2015; Richards and Friess 2016), with losses of soil carbon and other ecosystem services observed (Kauffman et al. 2015; Figure 1).

The drivers of mangrove deforestation vary regionally and often are associated with complex social and economic factors (e.g., Primavera 2006). A focus on enhancing ecosystem services from mangroves and the livelihoods of communities associated with mangrove forests is a recent development (Bosire et al. 2014; Brown et al. 2014) in which blue carbon has a role (e.g., Thompson et al. 2014; Wylie et al. 2016). However, analysis of the CO_2 emissions of shrimp and beef from deforested mangrove lands suggests that a focus on consumers of food sourced from deforested mangrove lands is also an important component of reducing human impacts on mangrove forests (Kauffman et al. 2017). They report that production of one kilogram of beef produces 1,440 kg of CO_2e while a kilogram of shrimp produced 1,603 kg of CO_2e when grown on converted mangrove habitats, highlighting the significant CO_2 emissions associated with these land-use conversions.

3.3 SEAGRASS MEADOWS

Seagrass meadows are suffering huge losses of area largely due to reductions in water quality (Short and Wyllie-Echeverria 1986; Waycott et al. 2009), although local direct effects of boating, dredging, altered hydrology and disease have been important in some locations (Robblee et al. 1991; Erftemelier and Lewis 2006; Serrano et al. 2016). Low-water quality affects: (i) the light available to plants growing on the seafloor, which reduces productivity and eventually results in mortality and (ii) enhances algal growth, which competes with seagrass for light (Duarte et al. 1995; Sherwood et al. Chapter 26; Oreska et al. Chapter 12). In some cases, local extinctions and complete loss of habitat have occurred (van Katwijk et al. 2009). Reductions in water quality have been due to enhanced levels of fine sediment and nutrients delivered to the coast associated with clearing of land in the watershed for agriculture and grazing, which has occurred in the last century in many regions (Morelli et al. 2012) but earlier in Europe and elsewhere (Lopez-Merino et al. 2017).

More recently the intensification and expansion of agriculture has also been an important driver of declining water quality, particularly after the 1940s, when nitrogen fertilizer (from the industrialized Haber Bosch process) became abundant and high loads of nitrogen (urea) were applied to agricultural fields subsequently enriching the coastal oceans (Erisman et al. 2012; Cloern et al. 2001) which has resulted in losses of seagrass globally (Waycott et al. 2009). Losses of tidal marshes and mangroves in these areas are also likely to have increased the negative impacts of nutrient enrichment on seagrass (Valiela and Cole 2002; Jickells et al. 2016). Although reducing the flow of nutrients and other pollutants into the coastal oceans is the target of policies of many countries (e.g., European Union Foden and Brazier 2007; Chesapeake Bay Dennison et al. 1993), many of which have been successful (e.g., Tampa Bay, Greening and Janicke 2006; Greening et al. 2014), much of the fine sediment delivered to the coasts in the past centuries is re-suspended with high levels of wave energy leading to chronically low-water quality in some sites (e.g., Fabricius et al. 2013) which continues to have negative impacts on seagrass meadows (Collier et al. 2016).

Direct conversion of seagrass beds through dredging and filling is rarer than loss of seagrasses as a result of decreased water clarity, but port dredging and filling for development (land reclamaition) locally affects seagrasses. The cumulative impacts of unintentional disturbance of seagrasses through accidental groundings or damage from fishing gear can be locally important drivers of seagrass loss as well (Dawes et al. 1997). Further, beach resort developments in areas that support seagrasses have been known to actively remove seagrasses in an attempt to improve the experience of beachgoers (Daby 2003). Of the three blue carbon ecosystems seagrass meadows have the most uncertainty in their current distribution and extent as they are poorly mapped because they occur underwater. As such, they are difficult to map with remote-sensing techniques which have difficulty penetrating water or differentiating seagrass from micro- and macroalgal communities, and mapping by direct observation by divers is expensive and logistically impractical in many regions. As reviewed in Chapter 12, compilations of mapping data worldwide document an extent of mapped area of ca. 175,000 km^2 (Green and Short 2003); however, that extent is surely an underestimate and the true global extent of seagrasses is likely between 300,000 and 600,000 km^2 (Duarte et al. 2010). As a result, it is likely that the true extent of seagrasses is underestimated. There are many unknowns about the impact of human activities on the carbon stores of seagrass meadows because besides a few well-studied sites (e.g., Moreton Bay; Shark Bay, Florida Bay; North Sea) there is a lack of data of the distribution and carbon stocks and how these have changed through time with human activities.

3.4 TIDAL MARSHES

Tidal marshes are the dominant blue carbon ecosystem over much of the temperate zone and polar coastal regions of the world but also occur in the high intertidal zone in the tropics. They have been widely used and converted to alternative land-uses for hundreds to thousands of years in the temperate zone (Gedan et al. 2009; Kroeger et al. Chapter 25). Because conversion of tidal marshes occurred so early in the history of human exploitation of the coastal zone, before global mapping, it is difficult to assess the pre-cleared area of tidal marshes and thus the total global amount of marsh converted by human activities is difficult to determine (Gedan et al. 2009). However, in a study of 12 temperate and subtropical estuaries, Lotze et al. (2006) found a 67% loss of coastal wetlands during human history.

Alternative land-uses for tidal marsh lands includes agriculture, grazing lands, urban and industrial development, and mining soils for peat and salt (Gedan et al. 2009; Drexler et al. Chapter 23). Conversion to agriculture, where marshes have been drained and isolated from tidal flows, utilized their often deep, organic matter-rich soils because they supported high levels of plant production with few inputs. This activity stimulates subsidence of the land as organic matter in soils is oxidized (Connor et al. 2001; Turner 2004, Byrd et al. this book, Figure 1). They have been widely filled and reclaimed for development associated with human settlements and have been isolated from tidal flows for conversion to grazing lands (e.g., Ganong 1903). Many of the alternative land-uses for tidal marshes result in losses of carbon sequestration potential, nitrogen removal (Jickells et al. 2016), as well as giving rise to enhanced emissions of CO_2, methane, and nitrous oxides (Deng et al. 2016).

In addition to conversions of tidal marshes to other uses, nutrient enrichment from agricultural pollutants from upstream sources has been significant factor in tidal marsh degradation (Verhoeven et al. 2006; Tuner et al. 2009; Deegan et al. 2012). High levels of nutrients reduce allocation of biomass to roots, making soils less stable and thus more vulnerable to disturbance, erosion, and export of particulate and dissolved carbon (Deegan et al. 2012). The losses of predators of bioturbating crabs, from overfishing, have also been suggested to lead to loss of tidal marsh and significant CO_2 emissions as high levels of bioturbation has facilitated erosion of the marsh edges (Coverdale et al. 2013). Similar to mangrove forests, losses of tidal marshes, for example, within the Mississippi delta, has occurred as a result of reduced sediment inputs from dammed and modified river systems thus diminishing rate of sediment accretion. This makes them more susceptible to submergence and erosion during storms as well as increasing their vulnerability to sea level rise (Day et al. 2011; Mudd 2011). Over many continents, sea level rise in addition to other factors including human activities are contributing to encroachment of mangroves into tidal marshes (Saintilan et al. 2014; Amitage et al. 2015) with consequences for carbon storage (Doughty et al. 2016).

3.5 EMERGING THREATS

While the clearing and conversion of mangroves has slowed in the first part of the 21st century in some areas (e.g., Hamilton and Casey 2016), other human activities are emerging as threats. For mangroves, clearing for oil palm cultivation in Southeast Asia has recently emerged on the research and conservation agenda (Richards and Friess 2016), and is expected to increase substantially in the future as governments set ambitious targets for economic growth and food security. For example, Indonesia has already designated 118,000 ha of mangrove for oil palm concessions (Ilman et al. 2016), many of which will be converted in the future. While the oil palm threat to mangroves has not been systematically assessed in other regions, we would assume that oil palm and the production of other commodities, such as lime, are emerging (Scales et al. 2017).

Blue carbon habitats are affected by human impacts such as land conversion, interactions between human activities and climate change factors. Given the sub- and intertidal nature of blue carbon ecosystems, they are strongly influenced by climate change, experiencing the effects of changes in the oceans, e.g., sea level rise, rising sea surface temperatures as well those on the land, e.g., changes in rainfall, river flows, sediment delivery, air temperature. While the productivity of some species is likely to have been directly enhanced by elevated levels of atmospheric CO_2 (Campbell and Fourqurean 2013; Reef et al. 2015) many climate change factors are likely to interact with human modifications of the land and seascape to have adverse effects on blue carbon ecosystems.

The need for freshwater to support agriculture and human settlements has led to damming of rivers that reduce freshwater and sediment supply to the coast and thus to coastal wetlands (Milliman and Farnsworth 2011). Globally, the building of dams is increasing (Zarfi et al. 2015). While reductions in river flows resulted in salinization of soils and changes in species distributions (Colonnello and Medina 1998), few adverse effects of low-sediment supply were observed. However, evidence for negative effects of these catchment level modifications are becoming more evident for blue carbon ecosystems as rising sea levels and enhanced storm activity result in losses of habitat and erosion of soils (e.g., Leonardi et al. 2016). For seagrass, catchment modifications can lead to hyper-saline conditions which can be devastating (e.g., Florida Bay, Hall et al. 2016). Reduced rainfall with climate change, which is predicted for many regions of the globe, may exacerbate the effects of reduced freshwater riverine inputs, as well as sediment supply, by further suppressing productivity of plants.

A number of stressors on blue carbon also interact with each other, with potentially additive or synergistic effects. For seagrass ecosystems, thermal anomalies are another emerging threat (Diaz-Almela et al. 2007; Nowicki et al. 2017) for which negative consequences maybe increased with human activities such as eutrophication (Unsworth et al. 2015). Similarly, mangroves in the Indo-Pacific are threatened by sea level rise, though their resilience is reduced further due to declining sediment inputs associated with damming of rivers (Lovelock et al. 2015). Low-water quality from eutrophication and legacy sediments remains a major persistent threat to seagrass ecosystems and a challenge to restoration in many locations (van Katwijk et al. 2016).

3.6 SOLUTIONS AND OPPORTUNITIES

While policies based on climate change mitigation are covered elsewhere in this volume (Chapters 15–19), here we focus on human activities that contribute to conserving blue carbon ecosystems as well as contribute to their restoration and maintenance.

Robust and long-term monitoring is urgently required if we are to accurately assess the contribution of habitat loss to blue carbon-related emissions. Currently, we do not have a clear idea of the global distribution of tidal marshes and seagrass. While recent efforts have been made to estimate global tidal marsh extent (McOwen et al. 2017), it is based on a literature review as opposed to a standardized remote-sensing survey, and most probably underestimates the extent of tropical marsh. Additionally, we still lack robust estimates of habitat loss for many blue carbon ecosystems, or use out of date information and sources. For example, previous studies of carbon emissions from mangrove deforestation utilized mangrove loss estimates of 1%–3% per year (e.g., Murdiyarso et al. 2015). More recent studies based on new data suggest substantially lower deforestation rates (Hamilton and Casey 2016).

The targeted protection of blue carbon ecosystems within marine protected areas could reduce losses of habitat and the ecosystem services they provide (Spalding et al. 2014; Thompson et al. 2017). More generally, land-use planning for the persistence of blue carbon ecosystems is critical (Enwright et al. 2016) and highly cost effective (Mills et al. 2016; Runting et al. 2016). Models indicate that landward migration of blue carbon ecosystems with sea level rise could see large increases in the area of blue carbon ecosystems (e.g., Mills et al. 2016), however, changes in habitats, e.g., tidal marsh habitats, converted to mangrove habitats, may also occur (Saintilan et al. 2014).

Restoration (Figure 3.2) provides a clear pathway to redressing many of the losses that have occurred in the past (Lewis 2005; van Katwijk et al. 2016) and may become even more desirable over time as sea level rise progresses and older infrastructure that currently limits tidal flooding is decommissioned (Temmerman et al. 2013). There is a growing knowledge of the role of blue carbon ecosystems in coastal protection (Duarte et al. 2013), which continues to stimulate justifications for restoration (Barbier 2015). The knowledge of processes that can improve the condition of degraded ecosystems is also growing (Lewis et al. 2016). For example, restoration of hydrology has been successful on a large scale in Florida (Howard et al. 2016) and experimental addition of sediments has enhanced marsh condition (Stagg and Mendelssohn 2011). There are developing projects that explore how to encourage restoration within modified aquaculture and agricultural landscapes, such that human livelihoods are maintained and enhanced as blue carbon ecosystems are restored (Lewis et al. 2016).

Reducing pollutants and thereby improving water quality is an opportunity for increasing the area and quality of blue carbon ecosystems. Economic incentives and technological developments have been used to reduce nutrient loads reaching the coastal oceans (Conley et al. 2009). Models have indicated that improving water quality can offset the negative effects of sea-level rise on seagrass ecosystems (Saunders et al. 2015) and is also likely to increase carbon sequestration in many blue carbon ecosystems (Macreadie et al. 2017).

3.7 CONCLUSIONS

Human activities have had enormous impacts on blue carbon ecosystems, but many are reversible and opportunities for increasing the area and quality of blue carbon ecosystems and avoiding greenhouse gas emissions from alternative land uses is available. Currently, the ecosystem services that are derived from blue carbon ecosystems are greatly undervalued (UNEP 2014). Climate change poses an overlay on past and current human activities that adds challenges for sustaining and restoring blue carbon ecosystems. Human activities of damming rivers, reducing sediment supply, and increasing nutrient availability are likely to exacerbate the effects of sea level rise on blue carbon ecosystems. However, land-use planning for sustaining blue carbon ecosystems with sea level rise has the potential to increase the area of blue carbon ecosystems and global carbon sequestration.