

Conservation of Blue Carbon Ecosystems for Climate Change Mitigation and Adaptation

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1. WHAT IS BLUE CARBON?

Vegetated coastal ecosystems, specifically tidal marshes, mangroves, and seagrass meadows, have exceptional capacities to sequester carbon dioxide (CO₂). They rank among the most productive habitats on earth, and through photosynthesis they fix CO₂ as organic matter, thus removing CO₂ from the atmosphere (Duarte et al., 2005). The capacity of vegetated coastal ecosystems to store organic carbon (C) in their canopies and soils over centennial to millennial time scales has been termed blue carbon (BC; Nellemann and Corcoran, 2009; Duarte et al., 2013). Recently, marine macroalgae (seaweed), that are the dominant primary producers in the coastal zone, have also been considered to contribute toward global BC sequestration, acting as C donors to other ecosystems where their organic material accumulates (Hill et al., 2015; Krause-Jensen and Duarte, 2016).

The burning of fossil fuels (e.g., coal, gas, and petroleum) is increasing the atmospheric levels of greenhouse gases (GHGs) including CO₂, methane (CH₄), and nitrous oxides (NO_x), which contributes to climate change and global warming. Land use change, and deforestation in particular, is also a major source of GHG emissions, accounting for 8%–20% of all

global emissions (van der Werf et al., 2009). Oceans play a key role in the capture and recycling of atmospheric CO₂ globally, due to the gaseous exchange at the ocean–atmosphere interface (Siegenthaler and Sarmiento, 1993). The oceans have absorbed over one-third of anthropogenic CO₂ emissions, through biological, physical, and chemical processes. In the open ocean, marine living organisms capture C, converting CO₂ into biomass through photosynthesis. However, most of the CO₂ assimilated by photosynthetic organisms in the ocean is recycled near the ocean surface and converted back into CO₂ by marine bacteria, which has been termed the “biological pump” (Longhurst and Harrison, 1989). Yet a significant portion of this biogenic C reaches the seafloor, where it can be buried and effectively locked away from the atmosphere over long time scales (centuries to millennium), constituting a sink of CO₂ and contributing to mitigate climate change (Bowler et al., 2009).

The CO₂ sequestration process in BC ecosystems is similar to that occurring in the open ocean (i.e., conversion of CO₂ into biomass through photosynthesis), but their C storage capacity is exceptional compared with other natural C sinks such as the open ocean and terrestrial forests (Nellemann and Corcoran, 2009, Fig. 28.1). Despite tidal marsh, mangrove, and

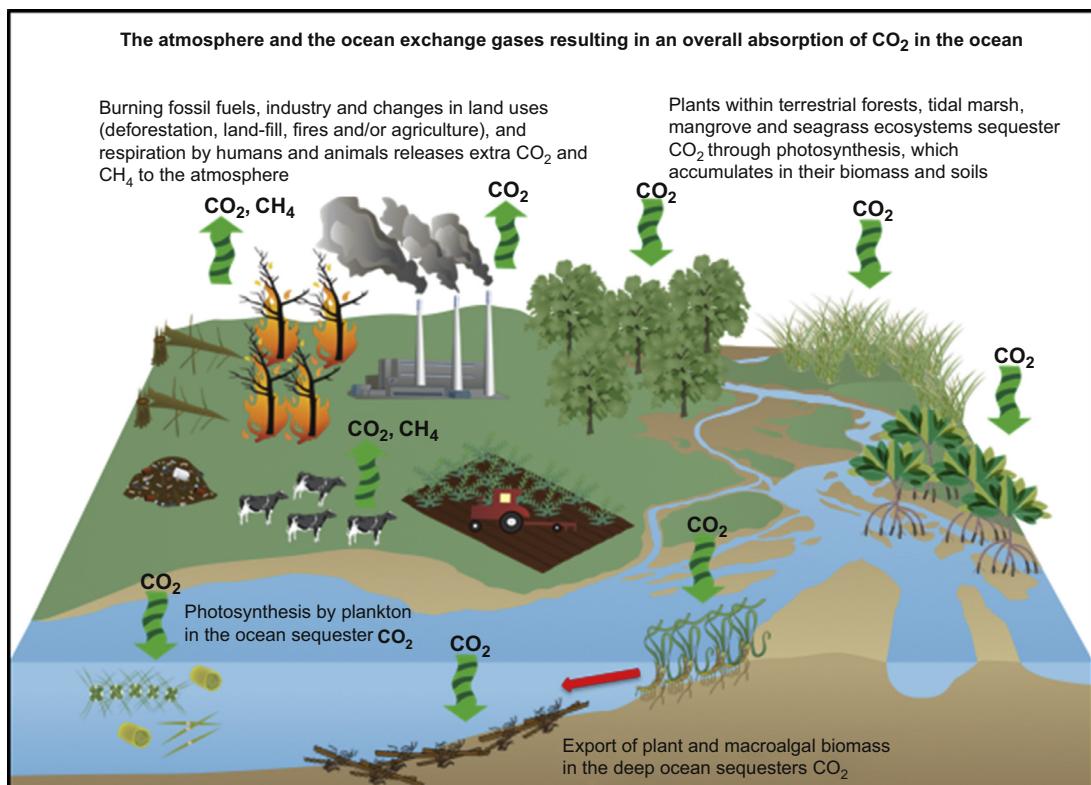


FIGURE 28.1 Conceptual diagram of carbon sequestration by blue carbon ecosystems and some of the activities that influence CO₂ exchange among the atmosphere, soil, and ocean in coastal areas and the open ocean. The major global C pools include the atmosphere, oceans, fossil fuels, vegetation, soils, and detritus. Landfill, smokestacks, cattle farming, and other human activities result in additional methane (CH₄) emissions.

seagrass ecosystems occupying less than 0.2% of the seabed area, they contribute nearly 50% of the CO₂ sequestration in marine sediments, and their C sequestration rates exceed those in the soils of many terrestrial ecosystems by 30- to 50-fold (Chmura et al., 2003; Duarte et al. 2005, 2013; Mcleod et al., 2011). Most macroalgal communities grow on rocky substrate and do not form significant *in situ* sedimentary C deposits, but the initial estimates of the amount of macroalgae C sequestered in sediments and deep-sea waters suggest that it is comparable to the C sequestered by all other BC ecosystems combined (Krause-Jensen and Duarte, 2016). Furthermore, the C captured by BC ecosystems is stored in marine soils for millennia, rather than the decades or centuries typical of terrestrial forests. This is due in part to the high rates of vertical accretion in tidal marsh, mangrove, and seagrass ecosystems, ranging from 0.4 to 21 mm year⁻¹ (Mateo et al., 1997; McKee et al., 2007; Duarte et al., 2013). This process of raising the seafloor is driven partly through the trapping and settling of particles from the water column and partly through organic matter production. This acts to bury the C in anoxic conditions, thereby slowing down its remineralization by microbes (Krauss et al., 2014; Mateo et al., 2006; Pedersen et al., 2011). Globally, tidal marsh, mangrove, and seagrass ecosystems sequester annually a similar amount of C to terrestrial forests, despite their extent being less than 3% of that of forests (Duarte et al., 2013). Unlike terrestrial forests, mangroves and tidal marshes rarely burn in wildfires, although they are exposed to other disturbances (e.g., tropical storms).

BC ecosystems provide important and valuable ecosystem services critical for climate change mitigation and adaptation, including coastal protection from storms and shoreline erosion, regulation of water quality, provision of habitat for commercially important fisheries and enhancing biodiversity, and being globally significant C sinks (Costanza et al., 1997; Duarte et al., 2013; Smale et al., 2013). Despite these recognized values, tidal marshes, mangroves, and seagrasses rank among the most threatened habitats on Earth (Valiela et al., 2001; Waycott et al., 2009; Gedan et al., 2009; Pendleton et al., 2012; Hamilton and Casey, 2016). The degradation, conversion to alternative land uses, and loss of BC ecosystems can result in the release of GHG to the atmosphere from the microbial decomposition of C stocks in their biomass and soil, fueling climate change (Pedersen et al., 2011; Pendleton et al., 2012; Murdiyarno et al., 2015; Kroeger et al., 2017; Lovelock et al., 2017a, Fig. 28.1). Despite the relatively small global decline of macroalgal communities (Byrnes et al., 2016), aquaculture of macroalgae has been identified as a prospective BC resource with great potential (Duarte, 2017).

Knowledge of the role of natural ecosystems in capturing and storing CO₂ is an increasingly important component in developing strategies to mitigate climate change associated with anthropogenic inputs of CO₂ to the earth's atmosphere (Raupach and Canadell, 2008). The loss and degradation of natural ecosystems comprise at least 20%–30% of global CO₂ emissions (UNEP, 2012; Pendleton et al., 2012). Although overall emissions from the burning of fossil fuels need to be severely reduced, mitigating climate change can also be achieved by protecting, restoring, or creating natural ecosystems, which has the simultaneous advantage of returning multiple other benefits (Canadell and Raupach, 2008; Trumper, 2009).

The term BC involves different global initiatives, leaded by United Nations, International Union for Conservation of Nature, and other nongovernmental organizations, with the objective of developing conservation, mitigation, and adaptation strategies for coastal ecosystems (Laffoley and Grimsditch, 2009; Nellemann and Corcoran, 2009). BC strategies comprise a

range of management activities for preventing or mitigating CO₂ emissions and/or enhancing CO₂ sequestration through their conservation and restoration (Duarte et al., 2013), similar to those already available for terrestrial forests (i.e., REDD+; Angelsen, 2009).

The potential of managing BC ecosystems as a strategy to mitigate climate change throughout the capture of CO₂ and to adapt to climate change throughout coastal protection is increasingly being recognized by policy-makers (Duarte et al., 2013; Macreadie et al., 2017a). In 2013, the Intergovernmental Panel on Climate Change released a Supplement to its guidelines for national GHG accounting (IPCC, 2014) acknowledging the role of coastal wetlands, including mangroves and tidal marshes in sequestering CO₂. In the interim, many nations, including Australia, India, and Indonesia, have made steps toward including BC strategies in their portfolios of initiatives to abate climate change impacts (Alongi et al., 2016; Kelleway et al., 2017a). At a global level, the International Partnership for BC—comprising nation states, research institutions, and nongovernment organizations—has been established to build awareness, share knowledge, and accelerate action to protect and restore coastal BC ecosystems for climate change mitigation and adaptation.

2. ECOLOGY OF BLUE CARBON ECOSYSTEMS

Tidal marshes, mangroves, and seagrass meadows are widely distributed along low-energy shorelines, occupying intertidal and subtidal coastal areas of every continent except Antarctica (Duarte et al., 2013, Fig. 28.2). The extent of mangroves has been estimated at 152,000 km², whereas the extent of seagrasses (ranging from 177,000 to 600,000 km²) and tidal marsh (22,000 to 400,000 km²) is poorly constrained (Table 28.1 and references therein). For seagrasses, this is partly due to the difficulties associated with mapping underwater vegetation.

Tidal marsh and mangrove ecosystems are coastal, saline ecosystems typically restricted to the upper–intertidal zones inundated from tidal flows (Hutchings and Saenger, 1987; Adam, 1993, Fig. 28.3). Tidal marshes, vegetated by salt-tolerant plants comprising a diversity of grasses, rushes, and herbs often referred to as salt marshes (Adam, 1993), are currently considered BC ecosystems, though freshwater tidal marshes are not. Mangroves are trees and shrubs adapted to live in coastal saline or brackish water within estuaries and marine shorelines (Duke et al., 1998). Seagrasses are also flowering plants (ranging from large forms with straplike leaves through to small forms with oval- to strap-shaped leaves) that grow in marine and estuarine areas and are common in intertidal and shallow waters to depths of about 30 m, where there is sufficient light for them to grow (Hemminga and Duarte, 2000). Macroalgae or seaweeds are a polyphyletic group of multicellular algae: red, green, and brown algae inhabiting the littoral zone to a depth with sufficient light to drive photosynthesis (Hurd et al., 2014).

BC ecosystems are experiencing a steep global decline at rates four times faster than rain forests (Duarte, 2009). Though quantifications can be problematic, it has been estimated that between 25% and 40% of tidal marsh, mangrove, and seagrass ecosystems extent has been lost since the 1940s, with the rate of loss accelerating in many countries (Valiela et al., 2001; Duarte et al., 2008; Duarte, 2009; Waycott et al., 2009; Nellemann and Corcoran, 2009; Atwood et al., 2017, Fig. 28.4). Major threats to BC ecosystems include pressures

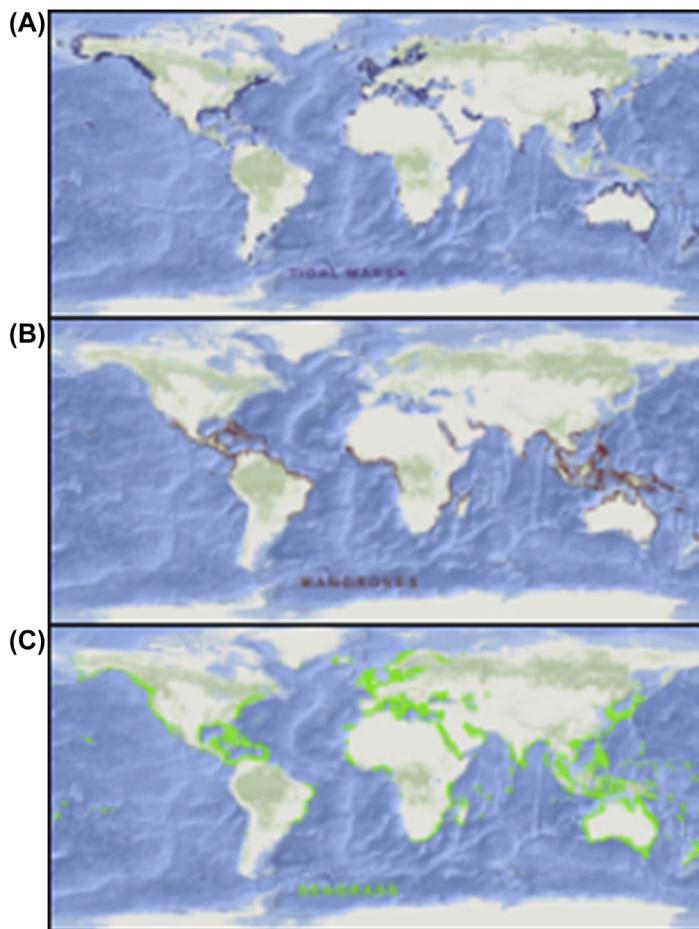


FIGURE 28.2 Global distribution of tidal marsh (A), mangrove (B), and seagrass (C) ecosystems. Data sources: United Nations Environment Programme World Conservation Monitoring Centre (UNEP-WCMC). References: Hutchison, J., Manica, A., Swetnam, R., Balmford, A., Spalding, M., 2014. Predicting global patterns in mangrove forest biomass. *Conservation Letters* 7 (3), 233–240; UNEP-WCMC, Short F.T., 2017. Global distribution of seagrasses (version 5.0). In: Fourth Update to the Data Layer Used in Green and Short (2003). Cambridge (UK). UNEP World Conservation Monitoring Centre. <http://data.unepwcmc.org/datasets/7>; McOwen C., Weatherdon L.V., Bochovre J., Sullivan E., Blyth S., Zockler C., Stanwell-Smith D., Kingston N., Martin C.S., Spalding M., Fletcher S., 2017. A global map of saltmarshes. *Biodiversity Data Journal* 5, e11764. Paper DOI: <https://doi.org/10.3897/BDJ.5.e11764>. <http://data.unepwcmc.org/datasets/43> (v.5).

associated with coastal development (e.g., conversion of coastal vegetated areas due to land clearance, urbanization, aquaculture, or restriction of tidal flow), deterioration of water quality (e.g., eutrophication), global warming, and sea level rise (SLR) (Nelleman and Corcoran, 2009; Mcleod et al., 2011; Atwood et al., 2017; Arias-Ortiz et al., 2018). Since the 1950s, there has been a global decline in the extent of macroalgal kelp forests of 0.018 year^{-1} , attributed to harvesting, pollution, invasive species, and/or temperature (Byrnes et al., 2016). Though the risks facing rain forests and their socioeconomic and ecological benefits are relatively well

TABLE 28.1 Extension and C Stocks and Burial Rates Within the Top 1 m of Soil of Tidal Marsh, Mangrove and Seagrass Ecosystems, and Macroalgae C Buried in the Ocean

Ecosystem	Global Extension (km^2)	Global C Burial Rate (Tg C year^{-1})	Global C Stock in Soil (Pg C)
Tidal marshes	22,000–400,000 ^{a,b} 200,000 ^c	4.8–87.3 ^a	0.4–6.5 ^d
Mangroves	137,760–152,361 ^a 152,308 ^g	22.5–24.9 ^e	5 ^f –10.4 ^d
Seagrasses	177,000–600,000 ^a	48–112 ^a	4.2–8.4 ^j
Macroalgae	1,400,000 ^h –5,700,000 ⁱ 3,540,000 ^h	61–268 ^h 173	n/a

Mean, maximum and minimum (range) estimates. Superscript numbers indicate the sources of data. n/a, not applicable. References indicated with a number: ^a, Mcleod et al., 2011; ^b, McOwen et al., 2017; ^c, Cai 2010; ^d, Duarte et al., 2013; ^e, Breithaupt et al., 2012; ^f, Jardine and Siikamäki 2014; ^g, Spalding et al., 2010; ^h, Krause-Jensen and Duarte 2016; ⁱ, Gattuso et al., 2006; ^j, Fourqurean et al., 2012.

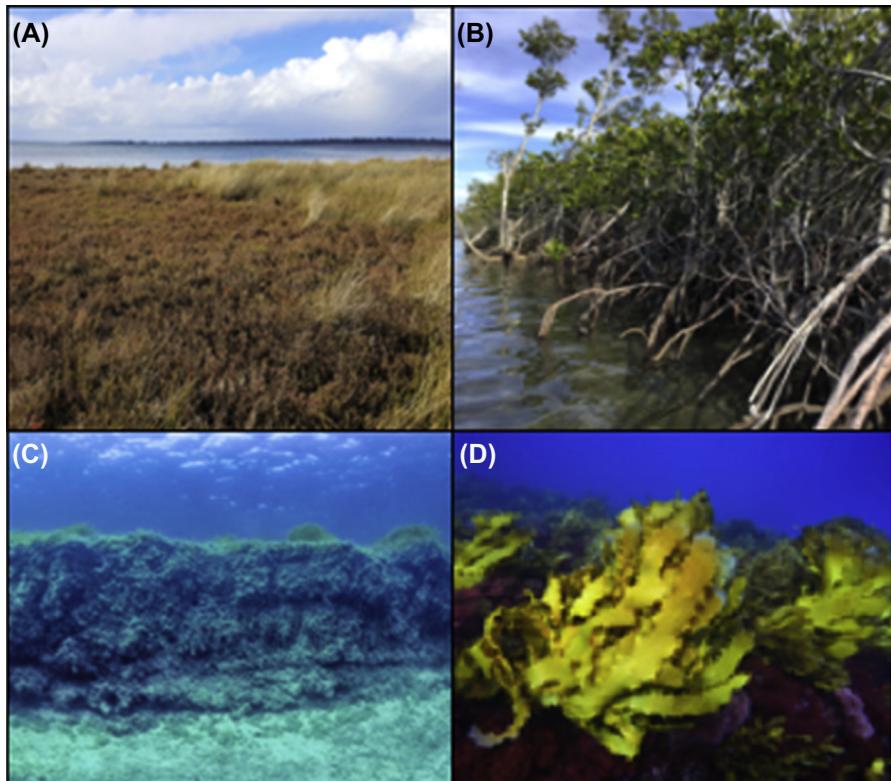


FIGURE 28.3 Tidal marshes, mangroves, seagrass meadows, and macroalgal communities. (A) *Sarcocornia* and *Juncus* tidal marsh species along the Peel-Harvey estuary (Western Australia). (B) *Avicennia marina* mangrove forest at Lake Doonella (Australia). (C) Erosional escarpment showing *Posidonia oceanica* C-rich sedimentary deposits at Es Pujols (Spain). (D) *Ecklonia radiata* macroalgal communities at Abrolhos Island. (D) Australia; Credit: Joan Costa.

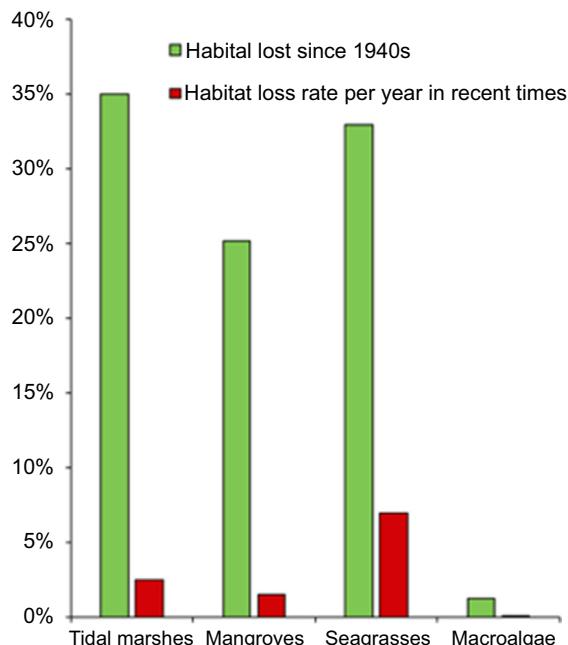


FIGURE 28.4 Loss of the aerial extent of tidal marsh, mangrove, and seagrass ecosystems, and macroalgal communities since the 1940s, and recent loss rates per year. Coastal eutrophication, reclamation, engineering, urbanization, harvesting, and climate change are the main threats to blue carbon ecosystems (Duarte et al., 2008; Duarte, 2009; Nelleman and Corcoran, 2009; UNEP, 2012; Byrnes et al., 2016).

understood, there is a comparatively poor understanding of the status and benefits associated with BC ecosystems (Chmura, 2013; Nelleman and Corcoran, 2009).

3. CARBON CYCLING IN BLUE CARBON ECOSYSTEMS

Tidal marshes, mangroves, seagrass meadows, and macroalgal communities are net auto-trophic ecosystems (i.e., fix CO₂ as organic matter photosynthetically in excess of the CO₂ respired back by biota; Duarte and Cebrian, 1996). They constitute hot spots of C cycling, ranking among the most intense C sinks in the biosphere, often with rates of production comparable to the most productive agricultural crops (Nixon, 1980; Gattuso et al., 1998; Duarte et al., 2013). There are several reasons why BC ecosystems are hot spots for C sequestration. First, they are highly productive ecosystems converting CO₂ into plant biomass. Second, they are mainly found in (and/or export C to) depositional environments, which accumulate both autochthonous and allochthonous particulate C (Kennedy et al., 2010; Saintilan et al., 2013). Third, soils within BC ecosystems have high accretion rates resulting in the rapid burial of organic matter in anoxic conditions that slow down microbial decomposition thereby contributing to the formation of organic-rich soils that can exceed 10 m in depth (Donato et al., 2011; Lo Iacono et al., 2008; Serrano et al., 2016a). In addition, their vegetation canopies and/or

aerial root networks are complex three-dimensional structures that slow water flow and facilitate the trapping and settling of particles, including C, while protecting the soil C deposits from erosion (Gacia and Duarte, 2001).

Tidal marshes, mangroves, and seagrass meadows store C in two main pools: (1) an above-ground pool in the form of standing biomass (leaves, stems, branches, and trunks), in situ dead biomass such as trees and plant litter, and epiphytes that grow on the surface of these materials and (2) a belowground pool comprising living and dead belowground biomass (roots and rhizomes) and C within the soils. Typically, the majority of the C stocks are found in the soils, with ~90% of total C stocks found in the soils of tidal marshes and seagrasses, and 75% in the soils of mangroves (Nellemann and Corcoran, 2009). A proportion of both the autochthonous and allochthonous C is buried in their soils where it can be preserved thousands of years (Mateo et al., 1997; Filho et al., 2006; Ward et al., 2008), thereby constituting a relevant C sink for climate change mitigation.

In contrast, most macroalgal communities grow on rocky shores and, therefore, do not accumulate sediments. Consequently, within macroalgal habitats, such as kelp forests, the C stock is in the form of standing biomass. However, as with the other BC ecosystems, a large portion of the organic matter and C produced within macroalgal ecosystems is exported as dissolved or particulate organic C to adjacent ecosystems, including sandy shores and the open ocean, where it can also accumulate over time scales relevant for climate change mitigation (Bouillon et al., 2008; Krause-Jensen and Duarte, 2016).

At a global scale, BC ecosystems combined (tidal marsh, mangrove, seagrass, and macroalgae) sequester 130–490 Tg year⁻¹ of C (equivalent to 500–1800 Tg of CO₂ per year; Table 28.1 and references therein), whereas current fossil fuel emissions total some 9900 Tg C year⁻¹ (36,200 Tg CO₂ per year; CDIAC, 2018). Their annual C sequestration, therefore, is equivalent to 1%–5% of current CO₂ emissions from fossil fuel combustion, making them important and efficient (sequestration rate/area) C sinks. Indeed, over the last millennia and owing to their capacity to raise the seafloor, tidal marsh, mangrove, and seagrass ecosystems accumulated 10–25 Pg C in 1-m thick soils (Table 28.1), equivalent to 2.4%–6.3% of global fossil fuel CO₂ emissions between 1751 and 2014 (estimated at 1474 Pg CO₂; CDIAC, 2018). However, this is likely an underestimate because the long-term preservation and continuous accretion of C in tidal marsh, mangrove, and seagrass soils results in the formation of organic-rich deposits several meters in thickness (Mateo et al., 1997; Donato et al., 2011).

4. FACTORS INFLUENCING CARBON STORAGE IN BLUE CARBON ECOSYSTEMS

BC ecosystems encompass a wide variety of species across a range of depositional environments and water depths, and the variability in the sedimentary C stocks has been found to be up to 18-fold among seagrasses (Lavery et al., 2013) and up to 4-fold in mangroves and tidal marshes (Pendleton et al., 2012). Based on terrestrial analogues and research undertaken on BC ecosystems, it is likely that multiple factors influence C storage, including biotic and abiotic factors acting in the water column, canopy, and the soils, as well as the history of the landscape and past variation in sea level (Fig. 28.5).

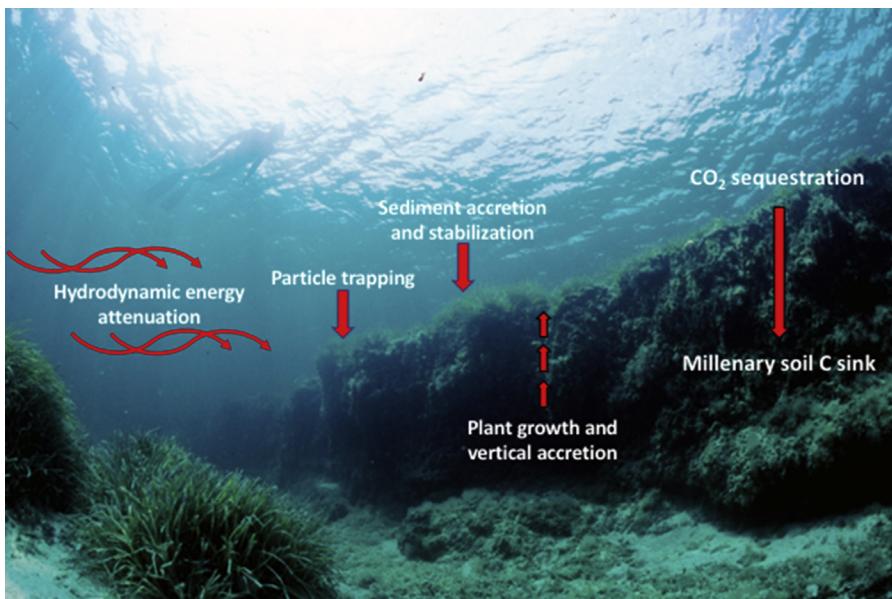


FIGURE 28.5 Processes explaining the capacity of seagrass ecosystems for climate change mitigation (CO_2 sequestration) and adaptation (coastal protection from erosion and sea level rise). The processes occurring in tidal marsh and mangrove ecosystems are similar.

The plants themselves exert a primary control on C storage through production of biomass and nutrient cycling (Lavery et al., 2013; Serrano et al. 2014, 2016a; Miyajima et al., 2015; Kelleway et al., 2016a; Krause-Jensen and Duarte, 2016; Atwood et al., 2017), both of which are highly variable depending on plant species and ecosystem characteristics (Pianka, 1970; Smith, 1981; Lovelock et al., 2014; Saintilan et al., 2013). Plant density, biomass, and productivity are strongly related to the underwater light penetration and soil type in seagrasses (Dennison, 1987; Duarte, 1991), whereas salinity, nutrient supply, and precipitation are important factors for mangroves and tidal marshes (Reef et al., 2010; Wigand et al., 2015; Lovelock et al., 2006). Geomorphological settings (i.e., encompassing variation in landscape and hydrology), soil characteristics (e.g., mineralogy and texture), and biological features (e.g., primary production and remineralization rates) control soil C storage in BC ecosystems (Donato et al., 2011; Adame et al., 2013; Ouyang and Lee, 2014; Kelleway et al., 2016b; Hayes et al., 2017). Both soil C stocks and accumulation rates for BC ecosystems vary with plant community types due to differences in their abilities to trap and retain sediments (Mudd et al., 2009; Lavery et al., 2013; Lovelock et al., 2014; Kelleway et al., 2017b). In addition, tidal transport, fluvial inputs, local hydrodynamics, and seagrass canopy density can all affect how much allochthonous C that is delivered and trapped (Kennedy et al., 2010; Saintilan et al., 2013; Kelleway et al., 2016b).

Once the C is buried in the soil, interactions between biotic and abiotic factors control its accumulation and preservation (Burdige, 2007). The rates of soil accretion, the soil texture, and the biochemical composition of the organic matter buried may strongly influence C accumulation and preservation, parameters that are highly variable among BC ecosystems

(De Falco et al., 2000; Marchand et al., 2003; Kennedy et al., 2010; Saintilan et al., 2013; Ouyang and Lee, 2014; Serrano et al., 2016a). If the accumulated soils are fine-grained, then they are likely to enhance the preservation of C by reducing oxygen exchange thereby reducing remineralization rates (Keil and Hedges, 1993; Serrano et al., 2016a). Finally, while both autochthonous (e.g., plant detritus and epiphytes) and allochthonous (e.g., seston and terrestrial matter) sources contribute to the soil C pool in BC ecosystems (Kennedy et al., 2010; Kelleway et al., 2016b), the proportion of plant-derived C is an important factor controlling their C storage capacity (Serrano et al., 2016a). Plant tissues contain relatively high amounts of degradation-resistant organic compounds (e.g., lignin; Harrison, 1989, Klap et al., 2000, Burdige, 2007, Trevathan-Tackett et al., 2015; Kaal et al., 2016) compared with seston and algal detritus (Laursen et al., 1996), which are more prone to remineralization during early diagenesis (Henrichs, 1992).

The group of macroalgae comprises species with large diversity in forms and size, from small algae with short life spans and high productivity rates (r-selection) to larger algae with slower turnover rates (K-selection; Pianka, 1970). Typically, r-selected species such as the sea lettuce (*Ulva* spp.) hold less structural tissues and are nutrient-enriched compared with K-selected species such as kelp, which are less grazed and more susceptible to contribute to C sequestration (Littler et al., 1983; Steneck and Dethier, 1994; Trevathan-Tackett et al., 2015; Krause-Jensen and Duarte, 2016). A comparison of vascular marine plants (mangrove, tidal marsh, and seagrass species) and macroalgae revealed greater stability in vascular plants related to their lignocellulose matrix (Trevathan-Tackett et al., 2015). In contrast, the presence of specific compounds (e.g., carbonates, long-chain lipids, alginates, xylans, and sulfated polysaccharides) in macroalgae means they may also contribute to long-term sedimentary C storage.

5. ROLE OF BLUE CARBON ECOSYSTEMS FOR CLIMATE CHANGE ADAPTATION

The conservation and creation of BC ecosystems also provide numerous benefits and services that are essential for climate change adaptation along coasts, including protection from storms, prevention of coastal flooding and shoreline erosion, regulation of water quality, provision of habitat for commercially important fisheries and endangered marine species, and food security for coastal communities (Beck et al., 2001; Nelleman and Corcoran, 2009; Barbier et al., 2011; Jones et al., 2012; Duarte et al., 2013; Smale et al., 2013). BC ecosystems function as “ecosystem engineers” by altering light levels, water flow, and near-bed velocity, sedimentation rates, modifying the local environment, and providing shelter from predation for other organisms of significant ecological or socioeconomic importance (Duarte et al., 2013). Through direct provision of food and structural habitat, BC ecosystems support higher levels of biodiversity and biomass than simple, unstructured habitats (Beck et al., 2001; Steneck et al., 2002; Barbier et al., 2011).

The increase in atmospheric CO₂ concentrations not only results in global warming but also entails SLR, and a higher intensity and frequency of severe storms, and extreme sea level events increasing the risks of coastal flooding and erosion, posing the well-being of vulnerable coastal communities under threat (Menéndez and Woodworth, 2010; Hoegh-Guldberg

and Bruno, 2010; Young et al., 2011; Church and White, 2011). The melting of ice has doubled SLR globally between 1900s until present ($0.17\text{--}0.32\text{ cm year}^{-1}$), whereas SLR projections point to peaks of 0.45 cm year^{-1} in 2050 and 1.1 cm year^{-1} by 2100 (Church et al., 2013). The global population under flooding threat increases comparably, from 190 million people in 2000 to 400 million by 2060 (Neumann et al., 2015). The socioeconomic impacts of SLR have been valued in millions of dollars, in particular in tropical and subtropical areas which are exposed to major risks (Hallegatte et al., 2013; Neumann et al., 2015). Tidal marshes, mangroves, seagrass meadows, and macroalgal communities constitute a natural protection against SLR, thereby their conservation and restoration support adaptation to climate change (Duarte et al., 2013). BC ecosystems dissipate hydrodynamic energy and favor particle deposition, and the rate of soil accretion in BC ecosystems is comparable to that of SLR (global estimates ranging from 0.2 to 0.7 cm year^{-1} globally; Duarte et al., 2013 and reference therein), which suggest that tidal marsh, mangrove, and seagrass ecosystems have some capacity to tolerate SLR (Krauss et al., 2014), though there remain uncertainties as to their likely response to future accelerations in SLR.

Over the last 200 years, significant amounts of anthropogenic CO₂ have been released into the atmosphere, reaching a concentration of >400 ppm in the atmosphere, the greater mark since the scientists began to measure it in Mauna Loa Observatory, Hawaii (Showstack, 2013). CO₂ has been continuously absorbed by oceans within a rate of 30% (Millero, 1995), causing a reduction of seawater pH. Carbon dioxide dissolves in seawater and forms carbonic acid (H₂CO₃), which is in turn dissociated in bicarbonate (HCO₃⁻) and can also dissociate in carbonate (CO₃²⁻). The last two reactions also form hydrogen ions (H⁺), that lower the seawater pH and decrease carbonate concentration. This process, known as ocean acidification (Caldeira and Wickett, 2003), affects directly the capacity of calcifying organisms to build their own calcium carbonate (CaCO₃) shells (Orr et al., 2005; Doney et al., 2009). BC ecosystems, and seagrass meadows and macroalgal communities in particular, modify carbonate chemistry and raise pH in the surrounding water by intense photosynthesis, providing refuge for calcifying organisms from ocean acidification (Middelboe and Hansen, 2007; Fabry et al., 2008; Hendriks et al., 2014). Hence, the conservation and restoration of BC ecosystems contributes to climate change adaptation at local scales, providing direct benefits (i.e., protection against coastal, flooding, erosion, ocean acidification, and food security) for communities inhabiting coastal areas (Duarte et al., 2013).

6. DISTURBANCE OF BLUE CARBON ECOSYSTEMS AND ASSOCIATED EMISSIONS

Terrestrial ecosystems can shift from net sinks of CO₂ to sources of CO₂ as a result of changes in land use and climate effects (e.g., deforestation, afforestation, agricultural practices, changes in precipitation and temperature, fires, and damage by pollution; Solomon et al., 2007). BC ecosystems are no different, and all are currently experiencing global declines in area (Gedan et al., 2009; Waycott et al., 2009; Byrnes et al., 2016; Atwood et al., 2017). The major drivers of these declines include land use conversion (clearing for development, agri(aqua)culture, salt production), altered hydrology and tidal flow, eutrophication, dredging, and other light-reducing activities (Pendleton et al., 2012, Fig. 28.6).

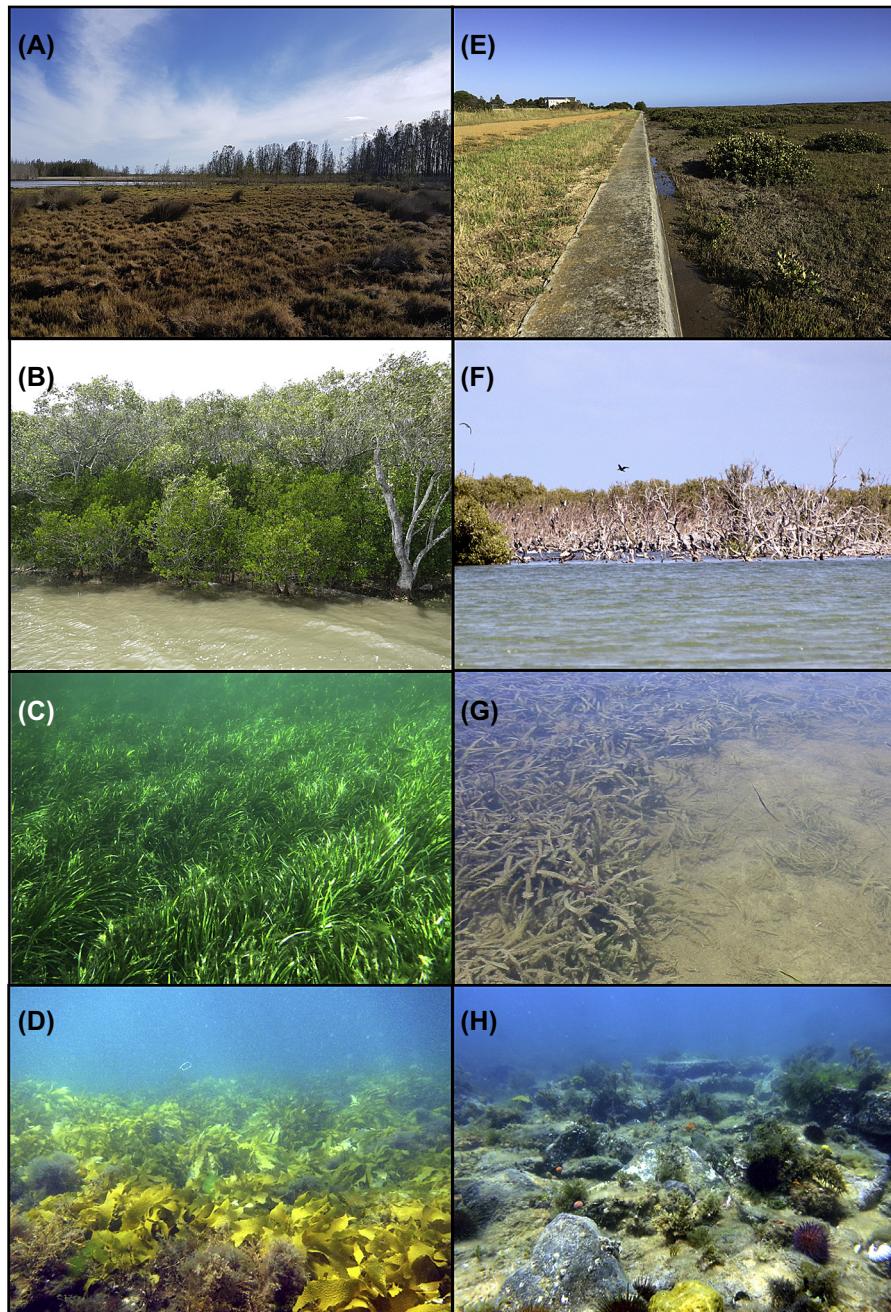


FIGURE 28.6 Examples of healthy (A–D) and damaged (E–H) blue carbon ecosystems that entail CO₂ emissions to the atmosphere. Images are of tidal marshes (A and E), mangroves (B and F), seagrass meadows (C and G), and macroalgal communities (D and H). Tidal marsh ecosystems at Madagascar (A and E) were impacted by land conversion and impounding, restricting landward migration. Mangrove dieback at Shark Bay (Australia; B and F) during fluctuating sea levels (Lovelock et al., 2017b). Eutrophication of coastal water resulted in the dieback of *Posidonia australis* meadows (C and G) at Cockburn Sound (Australia). Kelp communities cleared by sea urchins at Port Phillip Bay (D and H; Australia; Carnell and Keough, 2016). Credits: (B) Sharyn Hickey; (D and H) Paul Carnell.

Long-term sea temperature increases, shorter-term heat wave events, and SLR associated with climate change poses additional threats (Warren and Niering, 1993; Lovelock et al., 2017b; Arias-Ortiz et al., 2018).

These pressures have the potential to affect the C storage of BC ecosystems in two ways: through direct impacts on the primary producer and through direct or indirect disturbance of the soils (Fig. 28.7). A particularly unique threat to BC ecosystems is that of coastal squeeze, the phenomenon by which SLR causes the coastal ecosystem to retreat landward, to stay within a suitable water depth, but where this is prevented by the presence of coastal development at the landward margin (Doody, 2004; Torio and Chmura, 2013).

Threats that result in a loss or reduction of primary producer biomass and productivity include light reduction (e.g., through dredging, eutrophication, or increased turbid river flows), altered hydrology and salinity regimes, increased grazing, fire, and heat waves (Lovelock et al., 2017a). These pressures have the initial effect of reducing the production of autochthonous C that can be potentially sequestered and leading to reduced canopy density which may affect the trapping of allochthonous C. Even if the ecosystem suffers no further damage, these changes will likely lead to much-reduced C sequestration. Furthermore, there is likely to be an enhanced GHG emission through the decomposition of the dead biomass in the system.

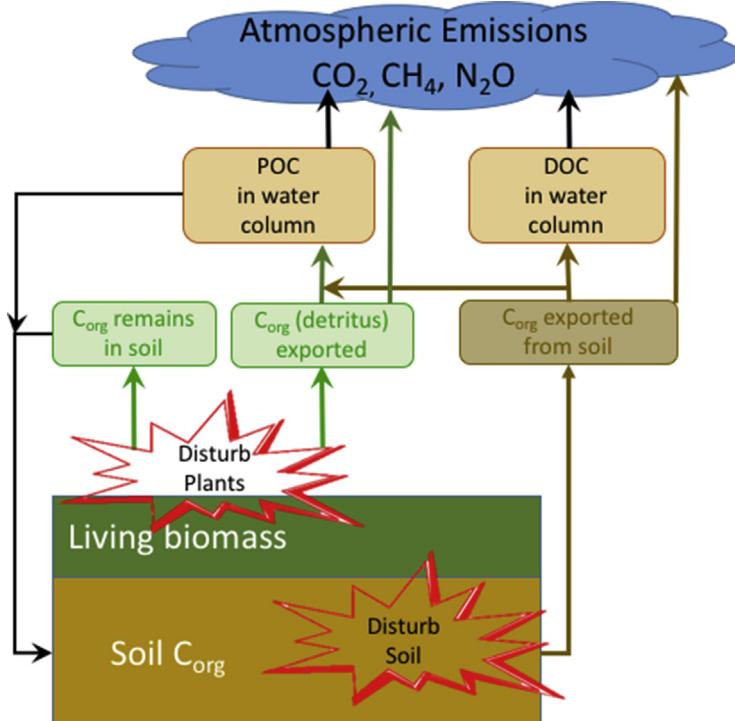


FIGURE 28.7 Conceptual model for the disturbance mechanisms and potential remineralization of organic carbon (C_{org}) following disturbance of blue carbon ecosystems, which could result in carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) emissions. DOC, dissolved organic carbon; POC, particulate organic carbon.

Other threats act on the soil of the BC ecosystems. These physical disturbances include clearing or reclamation (e.g., for housing, timber, or aquaculture development), dredging for mining or infrastructure development, boat moorings, anchor damage, impounding, and erosion driven by altered hydrodynamics. Threats that do not initially lead to physical disturbance of the soils may ultimately do so once the loss of biomass results in an enhanced susceptibility to soil erosion through hydrodynamic forces. These disturbances can lead to previously buried C stocks being resuspended and exposed to oxygenated conditions, enhancing remineralization and CO₂ emissions (Burdige, 2007; Pendleton et al., 2012). Therefore, in these situations, the threats may result not only in reduced sequestration but also enhanced GHG emissions (Hicks et al., 2003; Serrano et al., 2016b; Boxes 28.1 and 28.2).

BOX 28.1

GREENHOUSE GAS EMISSIONS FROM IMPOUNDING COASTAL WETLANDS

In 1976, the mangrove forests and tidal marshes surrounding the East Trinity Inlet (Cairns, Australia) were drained for sugar cane production (Fig. 28.8). The soils within 110 ha of coastal wetland habitat were drained and exposed to erosion (Hicks et al., 2003). This

resulted in the loss of 1.3 m of soil elevation and the subsequent loss of 74,800 Mg of soil C per hectare over 23 years after the disturbance occurred. The CO₂ emissions associated with soil organic carbon remineralization were estimated at 0.27 Tg of CO₂, significantly



FIGURE 28.8 Impounded coastal wetlands at Trinity Inlet (Australia). Degradation of tidal marsh and mangrove forest can be observed at both sides of the levee used to drain the wetland (Hicks et al., 2003).

BOX 28.1 (*cont'd*)

contributing to Australia's CO₂ emissions despite the small area of the site. In addition to CO₂ emissions, the exposure of soil C to oxygen resulted in the production of sulfuric acid and highly acidic water (pH of 3.2) that reached the adjacent estuary causing

unquantified damage to adjacent ecosystems (Hicks et al., 2003). In 2000, the tidal flow was reintroduced to restore this site, yet the avoided CO₂ emissions and enhanced soil C burial after restoration remains unassessed.

BOX 28.2

CO₂ EMISSIONS FROM MOORING ACTIVITIES IN SEAGRASS MEADOWS

Rottnest Island is a popular holiday destination in Perth (Australia). The first moorings to support boating activities were installed in 1930s, expanding to nowadays

reach 893 moorings distributed within Rottnest embayment. As moored boats drift with the currents, they drag a heavy chain across the seafloor removing the seagrass.

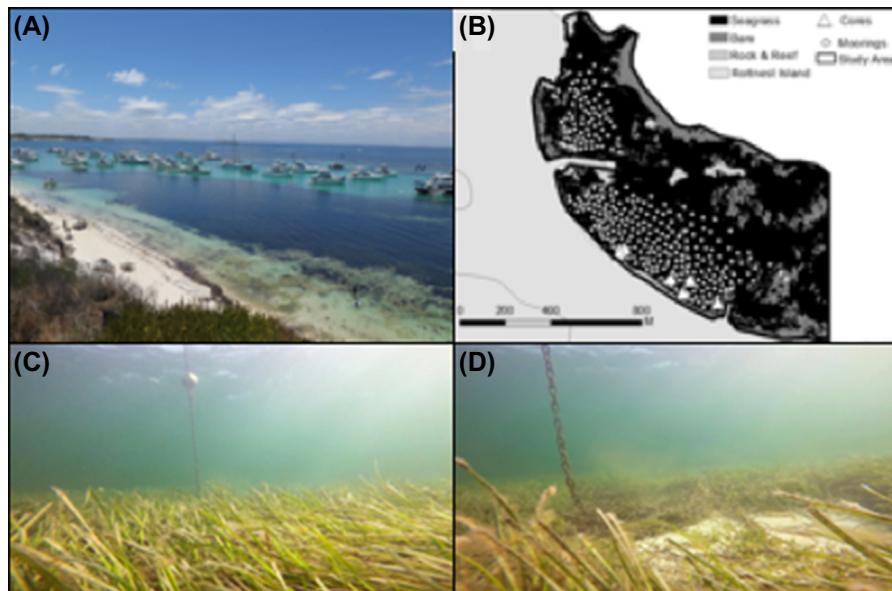


FIGURE 28.9 Rottnest Island is located 18 km off the coast of Perth (Australia). (A) Scouring of seagrass meadows by boating activities can be observed around the moorings; (B) Boating activities triggered the installation of 316 moorings at Thomson Bay; (C and D) Effects of moorings on seagrass meadows exposing soils to erosion (Serrano et al., 2016b).

BOX 28.2 (cont'd)

The soil C stocks underneath seagrass meadows have been compromised by the mooring deployment, which involved both the erosion of existing soil C stocks and the lack of further accumulation of C (Serrano et al., 2016b). The CO₂ emissions associated with the soil C remineralization within the top 50-cm soil accumulated over the last 200 years were estimated at 845 metric tons CO₂. Over the last 25 years, the Rottnest

Island Authority has replaced the swing-chain moorings with either pin moorings or secured chain moorings, which reduces the disturbance of seagrasses. However, the presence of barren patches within the meadows at mooring sites can still be seen (Fig. 28.9), indicating that some physical impacts of boating disturbances continues despite conservation measures were undertaken.

Changes in C storage from BC ecosystems has been reported due to land reclamation, chemical and physical disturbances, eutrophication, poleward expansion of mangroves, loss of top-down control by the removal of associated fauna, and marine heat waves (Bu et al., 2015; Deegan et al., 2012; Atwood et al., 2015; Marbà et al., 2015; Kelleway et al., 2016a; Lovelock et al., 2017b; Arias-Ortiz et al., 2018). Given that the soils are typically the greatest sink of C within BC ecosystems and, conversely, present the greatest risk of substantial CO₂ emissions after ecosystem disturbance, the belowground C stocks are the primary interest in BC initiatives (Sutton-Grier et al., 2014) and often dominate potential GHG emissions assessments (Lovelock et al., 2017a).

Estimating the consequences of ecosystem disturbance for GHG emissions can be difficult due to uncertainty around some key factors. These uncertainties can include the size of the C stock, the baseline conditions that postdisturbance conditions need to be compared to, and the fate of the disturbed C sink, the latter of these often the most difficult to resolve (Lovelock et al., 2017a; Duarte, 2017). Although rigorous but fairly routine assessments can determine how much C has been lost from a site following disturbance (e.g., Marbà et al., 2015; Hicks et al., 2003; Coverdale et al., 2014), it is much less clear how much of that stock will be remineralized, and whether the emissions comprise either CO₂, CH₄, and/or NO_x, and whether fluxes will be via the water column or directly across the soil-atmosphere interface (Fig. 28.7). BC ecosystems differ biogeochemically from freshwater wetlands, which may act as substantial natural sources of methane (CH₄) to the atmosphere (Whalen, 2005). This is due to the supply of sulfates from marine waters to BC soils, which acts to suppress CH₄ production by providing a lower energy decomposition pathway (i.e., sulfate reduction; Lovley et al., 1982). In general, it is expected that there is little CH₄ emission from marine ecosystems because the high levels of sulfate generally restrict methanogenesis occurring. However, mangrove and tidal marsh ecosystems which lose the supply of sulfate (e.g., through disconnection of tidal flow) and are converted into freshwater habitat may become major sources of methane emission (Whalen, 2005; Gatland et al., 2014).

Boxes 28.1 and 28.2 show examples of assessments of C stock loss and potential emissions following disturbances, for a mixed mangrove/tidal marsh ecosystem and a seagrass ecosystem, respectively. In both cases, there was a relatively high level of certainty about the stocks that were lost from the site but, in both cases, significant assumptions needed to be made regarding the emission that this represented. Generally, emissions are estimated by assuming a proportion of the lost stock is remineralized under oxic conditions (Lovelock et al., 2017c). The IPCC (IPCC, 2003, 2006 and 2014) provides default values for the size of C stocks in some coastal wetland ecosystems as well as emission factors, using a tiered approach. Progressing from Tier 1 to Tier 3 estimates (e.g., from global values to site-specific estimates) typically reduces the level of uncertainty around the potential GHG emission, though this usually comes at the cost of increased investment in measurements, modeling, and other studies that underpin the improved estimates. Unfortunately, for coastal wetlands, the range of activities (e.g., land use changes) that emission factors are provided for is currently limited and the uncertainties around them quite substantial, reflecting the limited empirical data on GHG emissions from coastal wetlands (Table 28.2).

For activities not covered by the IPCC Wetlands Supplement (IPCC, 2014), other approaches are required to estimate emissions. Lovelock et al. (2017a) proposed a risk assessment approach that combines the size of soil C stocks with assessments of the likelihood of remineralization of soil C. Lovelock et al. (2017c) used a simple model to estimate CO₂ emissions from tidal marsh, mangrove, and seagrass ecosystems based on decomposition rates for organic matter in these ecosystems under either oxic or anoxic conditions combined with assumptions of the proportion of C being exposed to either oxic or anoxic environments following disturbance. Recently, Kelleway et al. (2017a) assessed the potential emission reductions or enhanced C sequestration associated with a range of activities designed to increase C capture and storage by management of BC ecosystems (see Box 28.3). They estimated the size of C stocks from previously reported estimates for BC ecosystems and estimated the potential change in emissions by using the IPCC default values or by using

TABLE 28.2 A Partial Summary of the Possible Avoided Greenhouse Gas (GHG) Emissions and Enhanced GHG Sequestration Associated With Management Activities in Different Blue Carbon Ecosystems Reported by Kelleway et al. (2017a)

Ecosystem Activities		Avoided Emissions (Mg CO ₂ ha ⁻¹ year ⁻¹)	Enhanced Sequestration (Mg CO ₂ ha ⁻¹ year ⁻¹)
Tidal marsh	Tidal introduction to land without mangroves; enhanced sediment supply; land use planning for future sea level rise (SLR).	0.11–8	0.3–9.2
Mangrove	Tidal introduction to land without mangroves; enhanced sediment supply; land use planning for future SLR.	3.7–161	1.8–73
Seagrass	Avoiding loss; reestablishment of habitat; creation of new habitat; treatment of wrack.	1–1678 ^a	0.07–5.2

^aOne-off emissions estimated assuming that 25%–75% soil C stocks in 1 m soil deposits are Remineralized after disturbance and emitted as CO₂.

BOX 28.3**BLUE CARBON SCIENCE AND POLICY DEVELOPMENT IN AUSTRALIA**

Australia is home to vast areas of mangrove, tidal marsh, and seagrass and has a long history of scientific investigation into the function and management of these ecosystems. Consequently, Australian research has been at the forefront as the concept of BC has developed globally over the past decade. In 2013, Australia's Commonwealth Scientific and Industrial Research Organisation (CSIRO) commenced a 3-year research partnership with multiple local research institutes through the Coastal Carbon Biogeochemistry Cluster. The purpose of this partnership was to build the knowledge base of C cycling in Australian coastal waters and BC ecosystems (mangrove, tidal marsh, and seagrass). Key outcomes included the production of national estimates of BC stocks, accumulation rates, and patterns of BC losses (e.g., Atwood et al., 2017; Hayes et al., 2017; Macreadie et al., 2017b; Arias-Ortiz et al., 2018). At the same time, there has been improved understanding and quantification of key coastal biogeochemical processes relevant to burial and export of BC (e.g., Lavery et al., 2013; Sippo et al., 2016; Krause-Jensen and Duarte, 2016; Kelleway et al., 2017b; Maher et al., 2017).

These new data and insights have underpinned recent considerations of policy mechanisms which might incorporate BC into coastal management and environmental accounts in Australia. In 2017, Australia was also one of the first countries to voluntarily report human-induced GHG emissions and removals associated with coastal wetlands as part of its National Greenhouse Accounts.

The Department of the Environment and Energy compiles these accounts annually and reports to the United Nations' Framework Convention on Climate Change (UNFCCC). The inclusion of wetlands in the national inventory enables quantification of how mitigation initiatives (e.g., avoiding loss or degradation of wetlands and/or the restoration or creation of wetland habitat) may contribute to Australia meeting its international GHG commitments.

Although a growing number of countries aspire to report GHG emission and C sequestration changes from these ecosystems under voluntary international reporting requirements, few countries have domestic policy frameworks that specifically support the quantification and financing of C abatement through coastal wetland management. In 2016, the Australian Government also commissioned a technical review of opportunities for including BC activities within the nation's main domestic C accounting mechanism, the Emissions Reduction Fund (ERF) (Kelleway et al., 2017a). This review used a participatory workshop of scientific experts and C industry stakeholders to identify BC management actions that would meet the offsets integrity standards of the ERF ([Table 28.3](#)).

The review identified a number of activities which might be suitable for inclusion under the ERF. These include actions which (1) avoid the loss or disturbance of existing mangroves, tidal marshes, and seagrasses (e.g., through clearing and dredging); (2) restore biogeochemistry of BC ecosystems

BOX 28.3 (cont'd)

(e.g., restore tidal flow to coastal wetlands; improve water quality for seagrasses); and/or (3) plan for the creation of new habitat, including in the context of SLR (Kelleway et al., 2017a). Although significant information gaps remain in regard to both BC

science and policy-making, the Australian experience to date provides a broad template which other nations may follow as they identify options for C abatement through management of coastal landscapes.

TABLE 28.3 Factors to Consider in the Identification of Potential C Abatement Activities

Integrity Requirement	Description of Requirement
Additional abatement	Undertaking the BC enhancement activity must result in C abatement that is unlikely to occur in the ordinary course of events.
Measurement and verification	Estimating the activity's C removals, reductions or emissions must be achieved using an approach that is measurable and capable of being verified.
Eligible abatement	Carbon abatement using in ascertaining the CO ₂ net abatement amount for the activity must be eligible C abatement in accordance with Australia's nationally reported GHG emissions.
Evidence base	The approaches used for the activity must be supported by clear and convincing evidence.
Material amounts considered	Material amounts of greenhouse gases that are emitted as a direct consequence of the activity must be considered.
Conservative estimates	Estimates, projections, or assumptions regarding activity abatement are conservative

These abatement integrity requirements formed part of the assessment of potential Blue Carbon (BC) activities under Australia's Emissions Reductions Fund (Kelleway et al., 2017a).

assumed rates of remineralization based on expert knowledge. This applied example of estimating potential changes in GHG emissions reveals the complexity of the exercise and the large amount of assumptions and mixed methods that may be required to produce estimates.

7. CONSERVATION AND RESTORATION OF BLUE CARBON ECOSYSTEMS AS A STRATEGY FOR CLIMATE CHANGE MITIGATION AND ADAPTATION

Removal of atmospheric CO₂ through sequestration in natural ecosystems is necessary to keep global warming under 2°C by 2050, consistent with the Paris agreement, as reduction of emissions alone may not suffice to achieve this target. Whereas technological approaches to

remove atmospheric CO₂ are being sought, biosequestration of CO₂ is already a proven process that can be conserved, strengthened, and incorporated into emission accounting frameworks (IPCC, 2014) and has additional benefits for society.

Actions to prevent these emissions and mitigate climate change have concentrated on the preservation and restoration of terrestrial ecosystems, mainly tropical forests (Agrawal et al., 2011). Determination to reduce CO₂ emissions caused by forest clearance and land degradation led to the elaboration of global climate change mitigation solutions such as the Reducing Emissions from Deforestation and Forest Degradation program (REDD+; IPCC, 2003). The basis of this strategy is to economically compensate nations to manage terrestrial vegetation to enhance C sequestration and avoid CO₂ emissions. With an increased focus on C-trading schemes worldwide, it is important to better understand the C cycle and clearly identify both C sinks and sources of emissions (Ullman et al., 2013).

Coastal regions have constituted strategic points of human settlement through history, causing persistent and severe impacts on BC ecosystems by a wide variety of human activities (Lotze et al., 2006). The loss of BC ecosystems entails the loss of all the ecological services they provide, including C sequestration (Barbier et al., 2011; Kirwan and Megonigal, 2013) and the emission of GHG (Pendleton et al., 2012). Conservation and restoration of BC ecosystems will facilitate the maintenance of the benefits they provide, including fisheries, coastal protection, and related ecosystem services that support coastal communities and their livelihoods (Barbier et al., 2011; Duarte et al., 2013), while contributing to achieve the goal of keeping global warming under 2°C by 2050 (UNFCCC, 2016). As such, the need to identify, accurately value and quantify, and incorporate systems which naturally sequester atmospheric CO₂ into future C inventories and emission frameworks is becoming increasingly paramount. With growth in C markets around nature-based climate mitigation (via biosequestration), there are opportunities to finance the restoration of BC ecosystems using C offset schemes (Thomas, 2014).

As a result, anthropogenic activities that enhance C sequestration and/or avoid emissions of GHG (CO₂, CH₄, and NO_x) form the basis for BC strategies to reduce emissions and mitigate climate change. An important point here, however, is that often it is only activities or interventions that fall outside business-as-usual scenarios (i.e., entailing “additionality”) that may be eligible for crediting. These activities may include creating, restoring, and/or managing hydrological conditions, sediments supply, salinity characteristics, water quality, and/or native plant communities (VCS, 2015; Kelleway et al., 2017a). Quantifying the C potential of habitat restoration has been a focal point of BC research. Areas of demonstration include the generation of BC credits through the sustainable management of mangrove ecosystems in Madagascar (Benson et al., 2017; Box 28.4), restoration of aquaculture ponds to mangrove habitat (Matsui et al., 2012), reforestation of logged and degraded mangrove forests in Kenya (Huxham et al., 2015; Box 28.5), the restoration of hydrology to degraded coastal floodplains (Howe et al., 2009), and the revegetation of lost seagrass ecosystems (Marbà et al., 2015; Box 28.6). Managed realignment is an option for recreating or restoring tidal marshes aimed to a more sustainable coastal flood defense together with the provision of other services, including C benefits (Luisetti et al., 2011). The creation of macroalgal belts in southern Korea promotes CO₂ removal, and the management of macroalgal ecosystems could also be considered within BC projects (Chung et al., 2013; Box 28.7).

To quantify net GHG avoided emissions and enhanced sequestration from management activities implemented, there is a need to use approved methodologies for this purpose.

BOX 28.4**BLUE CARBON IN ACTION IN MADAGASCAR**

The African island nation of Madagascar contains 2% of the global mangrove distribution, but more than one-fifth of its extent has been deforested since 1990 (Jones et al., 2014). The main drivers of this deforestation have been increased extraction for the production of charcoal, timber, and conversion to agriculture and aquaculture. Under their Blue Forests initiative, the nongovernment organization Blue Ventures is supporting the generation of BC credits through the sustainable management of mangrove ecosystems, to help alleviate poverty and support biodiversity conservation in Madagascar's coastal areas.

BC projects currently under development across Madagascar incorporate multiple C accounting standards, including the Verified Carbon Standard, Climate Community and Biodiversity Alliance, and Plan Vivo. The

accurate accounting of BC outcomes and alignment with standards requires significant research and monitoring activity. To undertake this, Blue Ventures works in partnership with local community guides, Malagasy scientists, and international researchers.

Nationwide mapping of historical changes of mangrove forest cover, incorporating remote sensing and field data (Jones et al., 2016), has been central to identifying targeted project areas and defining baseline conditions. Other research initiatives have included the measurement of C stocks (biomass and soil C pools) in healthy, degraded, and restored mangroves, analysis of the drivers of mangrove loss (Scales et al., 2017) and the modeling of future mangrove forest and wetland changes (Benson et al., 2017) (Fig. 28.10).

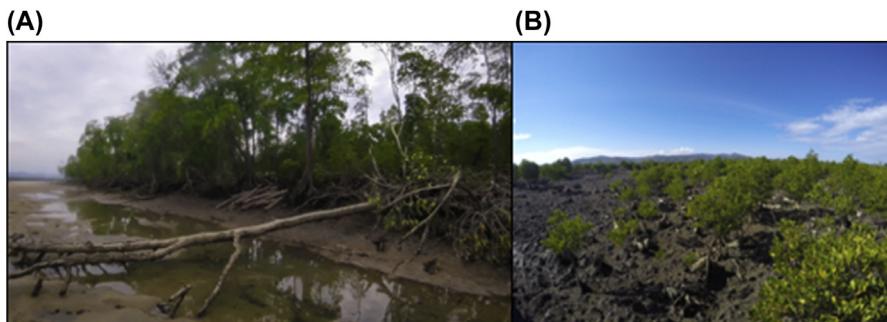


FIGURE 28.10 (A) Deforestation of mangrove forests in Madagascar for the production of timber and charcoal. (B) Revegetation of mangroves in previously vegetated habitats to enhance carbon sequestration and avoid emissions from soil carbon deposits.

BOX 28.5**NEW POLICY CONCEPT OF CLIMATE COMPATIBLE DEVELOPMENT: A CASE STUDY OF KENYA'S MANGROVE FORESTS**

Mangrove forests in Kenya are being degraded for timber, fuelwood, and coastal development. However, the benefits associated with mangrove conservation and restoration (e.g., fisheries, tourism, C sequestration, and protection against coastal erosion and extreme weather events) remain undervalued and are being dismissed (Tamooch et al., 2008). Huxham et al. (2015) developed a new policy concept of Climate Compatible Development (CCD) to predict the economic consequences of plausible alternative future scenarios for Kenya's mangrove forests. The projection of recent mangrove loss trends over the next 20 years

suggests a 43% loss of total forest extent over that time, whereas the restoration of the mangrove extent loss between 1992 and 2012 is still plausible. The combination of these scenarios allowed modeling economic outcomes, suggesting that the conservation of Kenya's mangroves entails a net present value of US\$20 million. Huxham et al. (2015) provides economic evidence for mangrove conservation and demonstrates a policy tool (CCD) that can be used to convince stakeholders and policy-makers to implement actions for mangrove conservation and restoration (Fig. 28.11).



FIGURE 28.11 Studying the carbon storage potential of mangrove forests in Kenya. Credit: Gloria Salgado.

BOX 28.6

IMPACT OF SEAGRASS LOSS AND SUCCEEDING REVEGETATION ON CARBON SEQUESTRATION

The loss of seagrass meadows is a common problem around the world, but thanks to a citizen project, seagrass meadows in Oyster Harbour (Australia) are recovering. Around 90% of the seagrass disappeared in the early 1980's in Oyster Harbour due to extensive land use changes in the catchment area and subsequent eutrophication of the Harbour; however, Geoff Bastyan began transplanting seagrass in 1994. Transplanting seagrass was proved successful, and a recent study by [Marbà et al. \(2015\)](#) demonstrated

that soil C stocks erode following seagrass loss but revegetation effectively restored seagrass C sequestration capacity, resulting in avoided CO₂ emissions and enhanced C sequestration. The C burial increased with the age of the restored sites, reaching similar C sequestration capacity than continuously vegetated meadows after 18 years of planting, demonstrating the conservation and restoration of seagrass meadows are effective strategies for climate change mitigation (Fig. 28.12).

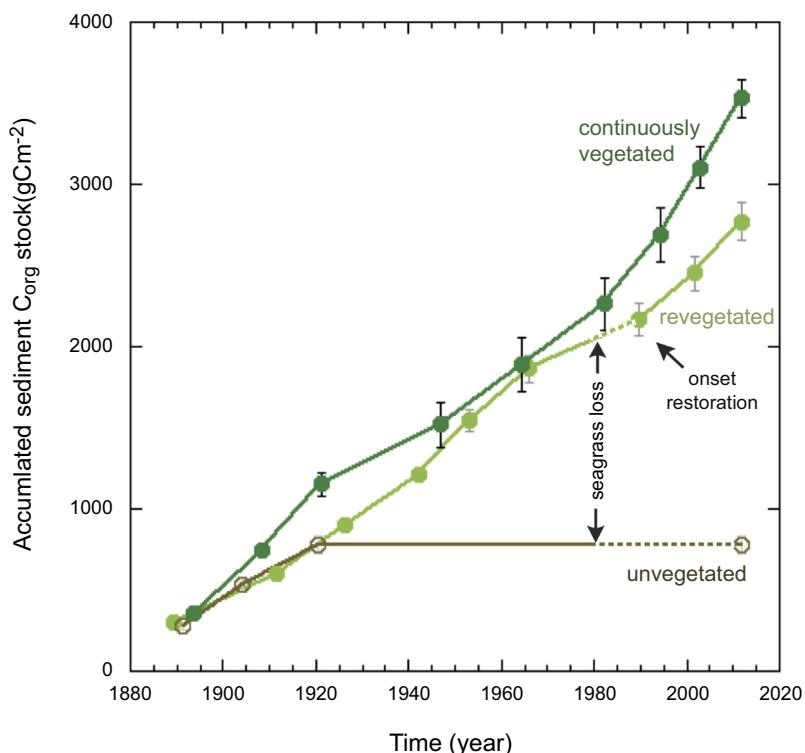


FIGURE 28.12 Reconstructed trajectories of carbon accumulation in resilient seagrass meadows (black line), restored meadows (dark gray line), and bare sediments (light gray line) at Oyster Harbour (Australia) since year 1890 ([Marbà et al., 2015](#)).

BOX 28.7**INSTALLATION OF MACROALGAE ECOSYSTEMS FOR MITIGATION AND ADAPTATION AGAINST CLIMATE CHANGE: KOREAN PROJECT OVERVIEW**

Macroalgae beds can sequester CO₂ in their biomass while providing food, fuel, pharmaceutical products, and other goods and services for climate change mitigation and adaptation. The Korean Project has established the Coastal CO₂ Removal Belt (CCRB), which encompasses both natural and farmed seaweed communities in southern Korea (Chung et al., 2013). This scheme promotes the CO₂ removal via macroalgae production and estimated that, e.g., *Ecklonia* kelp forests can draw down ~10 tones of CO₂ per ha per year, resulting in a decrease in dissolved organic carbon concentrations in the water column. The conservation and creation of macroalgae ecosystems has also the potential to, e.g., enhance coastal

protection from storms and shoreline erosion and food security for coastal communities (Smale et al., 2013). Despite recent evidence that highlights the major direct role of macroalgal C in C sequestration through carbon export and burial beyond the macroalgal habitat (e.g., Krause-Jensen and Duarte, 2016), there is substantial discussion as to whether macroalgae meet the principles to be contemplated within the BC model (Howard et al., 2017; Smale et al., 2018). A research priority is to clearly establish the carbon benefits of macroalgal culture and set a foundation for developing the necessary crediting and regulatory frameworks by which those benefits can be recognized (Chung et al., 2013).

Currently, methodologies available for BC accounting include the (1) Clear Air Development (CDM) methodology for mangroves (AR-AM0014: Afforestation and reforestation of degraded mangrove habitats), which can be used in developing nations; (2) a mangrove methodology for the Mississippi Delta that is available in the American Carbon Registry (ACR) (<https://americancarbonregistry.org/carbon-accounting/standards-methodologies/restoration-of-degraded-deltaic-wetlands-of-the-mississippi-delta>) (Mack et al., 2012), which is linked to a methodology for accounting for C sequestration with the restoration of tidal marshes in California (<https://americancarbonregistry.org/carbon-accounting/standards-methodologies/restoration-of-california-deltaic-and-coastal-wetlands>); and (3) the Verified Carbon Standard (VCS), which has an approved methodology for restoration and conservation of tidal marshes, mangroves, and seagrasses (VM0033; <http://database.verra.org/methodologies/methodology-tidal-wetland-and-seagrass-restoration-v10>), as well as a methodology in development for avoided deforestation for mangroves and other flooded forested wetlands (VM0007, REDD + Methodology Framework (REDD + MF), v1.6, <http://database.verra.org/methodologies/redd-methodology-framework-reddmf-v16>).

All existing methodologies share common features. First, there is a need to establish the project boundaries, defined by the extent of area potentially affected by the management activity. When the project area is not homogeneous in terms of C storage or habitat characteristics, stratification of the project area is required to enhance the accuracy of avoided GHG

emissions or enhanced C sequestration estimates. Prior to the activity, the most plausible baseline scenario in the absence of the project activity must be identified (enhanced C sequestration and/or avoided emissions). An important point here, however, is that often it is only activities or interventions that fall outside business-as-usual scenarios (i.e., entailing “additionality”) that may be eligible for crediting. Therefore, the natural C sink capacity of an ecosystem will not mitigate anthropogenic CO₂ emissions except when human-induced changes increase their C sequestration rate over time, such as restoration resulting in increased areal extent or primary productivity. After the activity, the project proponent must reassess the change in C storage every 5–10 years and to quantify enhanced C sequestration and/or avoided emissions.

8. FUTURE OF BLUE CARBON

Despite advances in BC science and governance over the past decade, there remains technical, financial, and policy barriers which limit the upscaling of local initiatives to have impacts that are globally relevant to climate change mitigation. Significant barriers include biases in the geographic coverage of research and data, poor prediction capacity for some C pools (e.g., atmospheric emissions remain poorly quantified outside of near-pristine ecosystems), high transaction costs, and ensuring that equity and justice are achieved. In addition, most demonstrated BC projects are relatively recent actions with little quantification of C mitigation benefits (or societal outcomes) beyond the scale of a few years, thereby the feasibility of BC project (e.g., cost/benefit and success of restoration) remains poorly understood ([Bayraktarov et al., 2016](#)).

Importantly, we now have the fundamental knowledge to justify the inclusion of BC ecosystem protection, restoration, and creation in coastal management directions and climate mitigation and adaptation mechanisms. The effective demonstration, monitoring, and reporting of existing and new BC projects will also address many of the outstanding issues. Adoption of standardized methods will also facilitate linking BC to National Determined Contributions (NDCs) and the transfer of information from early adopters of BC projects to those who follow. Currently, successful BC projects involve local stakeholders and consider alternative livelihoods for coastal communities ([Wylie et al., 2016](#)). BC projects have been successfully implemented within small scale in Kenya, Madagascar, India, and Vietnam, largely due to simpler requirements for project development, and funding has come from either the voluntary carbon market or alternative financing mechanisms. As more countries seek to respond and adapt to climate change impacts by reducing their carbon footprint, and with BC becoming an emerging issue in the United Nations Framework Convention on Climate Change (UNFCCC), it is likely that more BC projects will be implemented around the world in the near future.

References

- Adam, P., 1993. Saltmarsh Ecology. Cambridge University Press.
Adame, M.F., Kauffman, J.B., Medina, I., Gamboa, J.N., Torres, O., Caamal, J.P., et al., 2013. Carbon stocks of tropical coastal wetlands within the karstic landscape of the Mexican Caribbean. *PLoS One* 8 (2), e56569.

- Agrawal, A., Nepstad, D., Chhatre, A., 2011. Reducing emissions from deforestation and forest degradation. *Annual Review of Environment and Resources* 36, 373–396.
- Arias-Ortiz, A., Serrano, O., Masqué, P., Lavery, P.S., Mueller, U., Kendrick, G.A., 2018. A marine heatwave drives massive losses from the world's largest seagrass carbon stocks. *Nature Climate Change* 1.
- Alongi, D.M., Murdiyarso, D., Fourqurean, J.W., Kauffman, J.B., Hutahaean, A., Crooks, S., et al., 2016. Indonesia's blue carbon: a globally significant and vulnerable sink for seagrass and mangrove carbon. *Wetlands Ecology and Management* 24 (1), 3–13.
- Angelsen, A., 2009. Realising REDD+: National Strategy and Policy Options. Cifor.
- Atwood, T.B., Connolly, R.M., Almahaasheer, H., Carnell, P.E., Duarte, C.M., Lewis, C.J.E., et al., 2017. Global patterns in mangrove soil carbon stocks and losses. *Nature Climate Change* 7 (7), 523.
- Atwood, T.B., Connolly, R.M., Ritchie, E.G., Lovelock, C.E., Heithaus, M.R., Hays, G.C., et al., 2015. Predators help protect carbon stocks in blue carbon ecosystems. *Nature Climate Change* 5 (12), 1038.
- Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. *Ecological Monographs* 81 (2), 169–193.
- Bayraktarov, E., Saunders, M.I., Abdullah, S., Mills, M., Beher, J., Possingham, H.P., et al., 2016. The cost and feasibility of marine coastal restoration. *Ecological Applications* 26 (4), 1055–1074.
- Beck, M.W., Heck Jr., K.L., Able, K.W., Childers, D.L., Eggleston, D.B., Gillanders, B.M., et al., 2001. The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates: a better understanding of the habitats that serve as nurseries for marine species and the factors that create site-specific variability in nursery quality will improve conservation and management of these areas. *BioScience* 51 (8), 633–641.
- Benson, L., Glass, L., Jones, T.G., Ravaoarinorotsiharoana, L., Rakotomahazo, C., 2017. Mangrove carbon stocks and ecosystem cover dynamics in Southwest Madagascar and the implications for local management. *Forests* 8 (6), 190.
- Bouillon, S., Connolly, R.M., Lee, S.Y., 2008. Organic matter exchange and cycling in mangrove ecosystems: recent insights from stable isotope studies. *Journal of Sea Research* 59 (1–2), 44–58.
- Bowler, C., Karl, D.M., Colwell, R.R., 2009. Microbial oceanography in a sea of opportunity. *Nature* 459 (7244), 180.
- 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. *Global Biogeochemical Cycles* 26 (3).
- Bu, N.S., Qu, J.F., Li, G., Zhao, B., Zhang, R.J., Fang, C.M., 2015. Reclamation of coastal salt marshes promoted carbon loss from previously-sequestered soil carbon pool. *Ecological Engineering* 81, 335–339.
- Burdige, D.J., 2007. Preservation of organic matter in marine sediments: controls, mechanisms, and an imbalance in sediment organic carbon budgets? *Chemical Reviews* 107 (2), 467–485.
- Byrnes, J.E.K., Krumhansl, K.A., Okamoto, D., Rassweiler, A., Novak, M., Bolton, J.J., et al., 2016. Linking global patterns of kelp forest change and variation in climate over the past half-century. In: 11th International Temperate Reefs Symposium.
- Cai, W.J., 2010. Estuarine and coastal ocean carbon paradox: CO₂ sinks or sites of terrestrial carbon incineration? *Annual Review of Marine Science* 3, 123–145.
- Caldeira, K., Wickett, M.E., 2003. Oceanography: anthropogenic carbon and ocean pH. *Nature* 425 (6956), 365.
- Canadell, J.G., Raupach, M.R., 2008. Managing forests for climate change mitigation. *Science* 320 (5882), 1456–1457.
- Carnell, P.E., Keough, M.J., 2016. The influence of herbivores on primary producers can vary spatially and interact with disturbance. *Oikos* 125 (9), 1273–1283.
- CDIAC, 2018. Carbon Dioxide Information Analysis Center. <http://cdiac.ess-dive.lbl.gov>.
- Chmura, G.L., 2013. What do we need to assess the sustainability of the tidal salt marsh carbon sink? *Ocean and Coastal Management* 83, 25–31.
- Chmura, G.L., Anisfeld, S.C., Cahoon, D.R., Lynch, J.C., 2003. Global carbon sequestration in tidal, saline wetland soils. *Global Biogeochemical Cycles* 1 (4).
- Chung, I.K., Oak, J.H., Lee, J.A., Shin, J.A., Kim, J.G., Park, K.S., 2013. Installing kelp forests/seaweed beds for mitigation and adaptation against global warming: Korean project overview. *ICES Journal of Marine Science* 70 (5), 1038–1044.
- Church, J.A., White, N.J., 2011. Sea-level rise from the late 19th to the early 21st century. *Surveys in Geophysics* 32 (4–5), 585–602.

- Church, J.A., Clark, P.U., Cazenave, A., Gregory, J.M., Jevrejeva, S., Levermann, A., et al., 2013. Sea Level Change. PM Cambridge University Press.
- Costanza, R., d'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., et al., 1997. The value of the world's ecosystem services and natural capital. *Nature* 387 (6630), 253.
- Coverdale, T.C., Brisson, C.P., Young, E.W., Yin, S.F., Donnelly, J.P., Bertness, M.D., 2014. Indirect human impacts reverse centuries of carbon sequestration and salt marsh accretion. *PLoS One* 9 (3), e93296.
- da Silva Copertino, M., 2011. Add coastal vegetation to the climate critical list. *Nature* 473 (7347), 255–256.
- De Falco, G., Ferrari, S., Cancemi, G., Baroli, M., 2000. Relationship between sediment distribution and Posidonia oceanica seagrass. *Geo-Marine Letters* 20 (1), 50–57.
- Deegan, L.A., Johnson, D.S., Warren, R.S., Peterson, B.J., Fleeger, J.W., Fagherazzi, S., Wollheim, W.M., 2012. Coastal eutrophication as a driver of salt marsh loss. *Nature* 490 (7420), 388.
- Dennison, W.C., 1987. Effects of light on seagrass photosynthesis, growth and depth distribution. *Aquatic Botany* 27 (1), 15–26.
- Donato, D.C., Kauffman, J.B., Murdiyarsa, D., Kurnianto, S., Stidham, M., Kanninen, M., 2011. Mangroves among the most carbon-rich forests in the tropics. *Nature Geoscience* 4 (5), 293.
- Doney, S.C., Fabry, V.J., Feely, R.A., Kleypas, J.A., 2009. Ocean Acidification: The Other CO₂ Problem.
- Doody, J.P., 2004. 'Coastal squeeze'—an historical perspective. *Journal of Coastal Conservation* 10 (1), 129–138.
- Duarte, C.M., 1991. Seagrass depth limits. *Aquatic Botany* 40 (4), 363–377.
- Duarte, C.M., Cebrian, J., 1996. The fate of marine autotrophic production. *Limnology and Oceanography* 41 (8), 1758–1766.
- Duarte, C.M., Middelburg, J.J., Caraco, N., 2005. Major role of marine vegetation on the oceanic carbon cycle. *Biogeosciences Discussions* 1 (1), 659–679.
- Duarte, C.M., Dennison, W.C., Orth, R.J., Carruthers, T.J., 2008. The charisma of coastal ecosystems: addressing the imbalance. *Estuaries and Coasts* 31 (2), 233–238.
- Duarte, C.M., 2009. Global Loss of Coastal Habitats: Rates, Causes and Consequences. FBBVA, Madrid, 181 p. ISBN: 978-84-96515-84-0 available as pdf's at: <http://www.fbbva.es/TLFU/tlfu/esp/publicaciones/libros/fichalibro/index.jsp?codigo=411>.
- Duarte, C.M., Losada, I.J., Hendriks, I.E., Mazarrasa, I., Marbà, N., 2013. The role of coastal plant communities for climate change mitigation and adaptation. *Nature Climate Change* 3 (11), 961–968.
- Duarte, C.M., 2017. Reviews and syntheses: hidden forests, the role of vegetated coastal habitats in the ocean carbon budget. *Biogeosciences* 14 (2), 301.
- Duke, N., Ball, M., Ellison, J., 1998. Factors influencing biodiversity and distributional gradients in mangroves. *Global Ecology and Biogeography Letters* 7 (1), 27–47.
- Fabry, V.J., Seibel, B.A., Feely, R.A., Orr, J.C., 2008. Impacts of ocean acidification on marine fauna and ecosystem processes. *ICES Journal of Marine Science* 65 (3), 414–432.
- Filho, P.S., Cohen, M.C.L., Lara, R.J., Lessa, G.C., Koch, B., Behling, H., 2006. Holocene coastal evolution and facies model of the Bragança macrotidal flat on the Amazon Mangrove Coast, Northern Brazil. *Journal of Coastal Research* 306–310.
- Fourqurean, J.W., Duarte, C.M., Kennedy, H., Marbà, N., Holmer, M., Mateo, M.A., et al., 2012. Seagrass ecosystems as a globally significant carbon stock. *Nature Geoscience* 5 (7), 505–509.
- Gacia, E., Duarte, C.M., 2001. Sediment retention by a Mediterranean Posidonia oceanica meadow: the balance between deposition and resuspension. *Estuarine, Coastal and Shelf Science* 52 (4), 505–514.
- Gatland, J.R., Santos, I.R., Maher, D.T., Duncan, T.M., Erler, D.V., 2014. Carbon dioxide and methane emissions from an artificially drained coastal wetland during a flood: implications for wetland global warming potential. *Journal of Geophysical Research: Biogeosciences* 119 (8), 1698–1716.
- Gattuso, J.P., Frankignoulle, M., Wollast, R., 1998. Carbon and carbonate metabolism in coastal aquatic ecosystems. *Annual Review of Ecology and Systematics* 29 (1), 405–434.
- Gattuso, J.P., Gentili, B., Duarte, C.M., Kleypas, J.A., Middelburg, J.J., Antoine, D., 2006. Light availability in the coastal ocean: impact on the distribution of benthic photosynthetic organisms and contribution to primary production. *Biogeosciences* 3 (4), 489–513.
- Gedan, K.B., Silliman, B.R., Bertness, M.D., 2009. Centuries of human-driven change in salt marsh ecosystems. *Annual Reviews in Marine Science* 1, 117–141.

- Hallegatte, S., Green, C., Nicholls, R.J., Corfee-Morlot, J., 2013. Future flood losses in major coastal cities. *Nature Climate Change* 3 (9), 802.
- Hamilton, S.E., Casey, D., 2016. Creation of a high spatio-temporal resolution global database of continuous mangrove forest cover for the 21st century (CGMFC-21). *Global Ecology and Biogeography* 25 (6), 729–738.
- Harrison, P.G., 1989. Detrital processing in seagrass systems: a review of factors affecting decay rates, remineralization and detritivory. *Aquatic Botany* 35 (3–4), 263–288.
- Hayes, M.A., Jesse, A., Hawke, B., Baldock, J., Tabet, B., Lockington, D., Lovelock, C.E., 2017. Dynamics of sediment carbon stocks across intertidal wetland habitats of Moreton Bay, Australia. *Global Change Biology* 23 (10), 4222–4234.
- Hemminga, M.A., Duarte, C.M., 2000. *Seagrass Ecology*. Cambridge University Press.
- Henrichs, S.M., 1992. Early diagenesis of organic matter in marine sediments: progress and perplexity. *Marine Chemistry* 39 (1–3), 119–149.
- Hendriks, I.E., Olsen, Y.S., Ramajo, L., Basso, L., Steckbauer, A., Moore, T.S., et al., 2014. Photosynthetic activity buffers ocean acidification in seagrass meadows. *Biogeosciences* 11 (2), 333.
- Hicks, W., Fitzpatrick, R., Bowman, G., 2003. Managing coastal acid sulfate soils: the East Trinity example. In: *Advances in Regolith: Proceedings of the CRC LEME Regional Regolith Symposia*. CRC LEME, Bentley, pp. 174–177.
- Hill, R., Bellgrove, A., Macreadie, P.I., Petrou, K., Beardall, J., Steven, A., Ralph, P.J., 2015. Can macroalgae contribute to blue carbon? An Australian perspective. *Limnology and Oceanography* 60 (5), 1689–1706.
- Hoegh-Guldberg, O., Bruno, J.F., 2010. The impact of climate change on the world's marine ecosystems. *Science* 328 (5985), 1523–1528.
- Howard, J., Sutton-Grier, A., Herr, D., Kleypas, J., Landis, E., Mcleod, E., et al., 2017. Clarifying the role of coastal and marine systems in climate mitigation. *Frontiers in Ecology and the Environment* 15 (1), 42–50.
- Howe, A.J., Rodríguez, J.F., Saco, P.M., 2009. Surface evolution and carbon sequestration in disturbed and undisturbed wetland soils of the Hunter estuary, southeast Australia. *Estuarine, Coastal and Shelf Science* 84 (1), 75–83.
- Hurd, C.L., Harrison, P.J., Bischof, K., Lobban, C.S., 2014. *Seaweed Ecology and Physiology*. Cambridge University Press.
- Hutchings, P.A., Saenger, P., 1987. *Ecology of Mangroves*. University of Queensland Press, 388 pages. Maher 44 (10), 4889–4896.
- Hutchison, J., Manica, A., Swetnam, R., Balmford, A., Spalding, M., 2014. Predicting global patterns in mangrove forest biomass. *Conservation Letters* 7 (3), 233–240.
- Huxham, M., Emerton, L., Kairo, J., Munyi, F., Abdirizak, H., Muriuki, T., et al., 2015. Applying climate compatible development and economic valuation to coastal management: a case study of Kenya's mangrove forests. *Journal of Environmental Management* 157, 168–181.
- IPCC, 2003. 2003 IPCC Good practice guidance for land use, land-use change and forestry. In: Penman, J., Gyartsoky, M., Hiraishi, T., Krug, T., Kruger, D., Pipatti, R., Buendia, L., Miwa, K., Ngara, T., Tanabe, K., Wagner, F. (Eds.). IPCC, Switzerland.
- IPCC, 2006. 2006 IPCC guidelines for national greenhouse gas inventories. In: Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), Prepared by the National Greenhouse Gas Inventories Programme. IGES, Japan.
- IPCC, 2014. In: Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M., Troxler, T.G. (Eds.), 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands. IPCC, Switzerland.
- Jardine, S.L., Siikamäki, J.V., 2014. A global predictive model of carbon in mangrove soils. *Environmental Research Letters* 9 (10), 104013.
- Jones, H.P., Hole, D.G., Zavaleta, E.S., 2012. Harnessing nature to help people adapt to climate change. *Nature Climate Change* 2 (7), 504–509.
- Jones, T.G., Ratsimba, H.R., Ravaoarinorotsihorana, L., Cripps, G., Bey, A., 2014. Ecological variability and carbon stock estimates of mangrove ecosystems in northwestern Madagascar. *Forests* 5 (1), 177–205.
- Jones, T.G., Glass, L., Gandhi, S., Ravaoarinorotsihorana, L., Carro, A., Benson, L., et al., 2016. Madagascar's mangroves: quantifying nation-wide and ecosystem specific dynamics, and detailed contemporary mapping of distinct ecosystems. *Remote Sensing* 8 (2), 106.

- Kaal, J., Serrano, O., Nierop, K.G., Schellekens, J., Cortizas, A.M., Mateo, M.Á., 2016. Molecular composition of plant parts and sediment organic matter in a Mediterranean seagrass (*Posidonia oceanica*) mat. *Aquatic Botany* 133, 50–61.
- Keil, R.G., Hedges, J.I., 1993. Sorption of organic matter to mineral surfaces and the preservation of organic matter in coastal marine sediments. *Chemical Geology* 107 (3–4), 385–388.
- Kelleway, J.J., Saintilan, N., Macreadie, P.I., Skilbeck, C.G., Zawadzki, A., Ralph, P.J., 2016a. Seventy years of continuous encroachment substantially increases 'blue carbon' capacity as mangroves replace intertidal salt marshes. *Global Change Biology* 22 (3), 1097–1109.
- Kelleway, J.J., Saintilan, N., Macreadie, P.I., Ralph, P.J., 2016b. Sedimentary factors are key predictors of carbon storage in SE Australian saltmarshes. *Ecosystems* 19 (5), 865–880.
- Kelleway, J., Serrano, O., Baldock, J., Cannard, T., Lavery, P., Lovelock, C.E., et al., 2017a. Technical Review of Opportunities for Including Blue Carbon in the Australian Government's Emissions Reduction Fund. Department of Environment and Energy, Canberra, p. 287.
- Kelleway, J.J., Saintilan, N., Macreadie, P.I., Baldock, J.A., Ralph, P.J., 2017b. Sediment and carbon deposition vary among vegetation assemblages in a coastal salt marsh. *Biogeosciences* 14 (16), 3763.
- Kennedy, H., Beggins, J., Duarte, C.M., Fourqurean, J.W., Holmer, M., Marbà, N., Middelburg, J.J., 2010. Seagrass sediments as a global carbon sink: isotopic constraints. *Global Biogeochemical Cycles* 24 (4), 1–8.
- Kirwan, M.L., Megomigal, J.P., 2013. Tidal wetland stability in the face of human impacts and sea-level rise. *Nature* 504 (7478), 53.
- Klap, V.A., Hemminga, M.A., Boon, J.J., 2000. Retention of lignin in seagrasses: angiosperms that returned to the sea. *Marine Ecology Progress Series* 1–11.
- Krause-Jensen, D., Duarte, C.M., 2016. Substantial role of macroalgae in marine carbon sequestration. *Nature Geoscience* 9 (10), 737.
- Krauss, K.W., McKee, K.L., Lovelock, C.E., Cahoon, D.R., Saintilan, N., Reef, R., Chen, L., 2014. How mangrove forests adjust to rising sea level. *New Phytologist* 202 (1), 19–34.
- Kroeger, K.D., Crooks, S., Moseman-Valtierra, S., Tang, J., 2017. Restoring tides to reduce methane emissions in impounded wetlands: a new and potent Blue Carbon climate change intervention. *Scientific Reports* 7 (1), 11914.
- Laffoley, D., Grimsditch, G.D. (Eds.), 2009. The Management of Natural Coastal Carbon Sinks. Iucn.
- Laursen, A.K., Mayer, L.M., Townsend, D.W., 1996. Lability of proteinaceous material in estuarine seston and subcellular fractions of phytoplankton. *Marine Ecology Progress Series* 227–234.
- Lavery, P.S., Mateo, M.Á., Serrano, O., Rozaimi, M., 2013. Variability in the carbon storage of seagrass habitats and its implications for global estimates of blue carbon ecosystem service. *PLoS One* 8 (9), e73748.
- Littler, M.M., Littler, D.S., Taylor, P.R., 1983. Evolutionary strategies in a tropical barrier reef system: functional-form groups of marine macroalgae. *Journal of Phycology* 19 (2), 229–237.
- Lo Iacono, C., Mateo, M.A., Gracia, E., Guasch, L., Carbonell, R., Serrano, L., et al., 2008. Very high-resolution seismic-acoustic imaging of seagrass meadows (Mediterranean Sea): implications for carbon sink estimates. *Geophysical Research Letters* 35 (18).
- Longhurst, A.R., Harrison, W.G., 1989. The biological pump: profiles of plankton production and consumption in the upper ocean. *Progress in Oceanography* 22 (1), 47–123.
- Lotze, H.K., Lenihan, H.S., Bourque, B.J., Bradbury, R.H., Cooke, R.G., Kay, M.C., et al., 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science* 312 (5781), 1806–1809.
- Lovelock, C.E., Ball, M.C., Feller, I.C., Engelbrecht, B.M., Ling Ewe, M., 2006. Variation in hydraulic conductivity of mangroves: influence of species, salinity, and nitrogen and phosphorus availability. *Physiologia Plantarum* 127 (3), 457–464.
- Lovelock, C.E., Adame, M.F., Bennion, V., Hayes, M., O'Mara, J., Reef, R., Santini, N.S., 2014. Contemporary rates of carbon sequestration through vertical accretion of sediments in mangrove forests and saltmarshes of South East Queensland, Australia. *Estuaries and Coasts* 37 (3), 763–771.
- Lovelock, C.E., Atwood, T., Baldock, J., Duarte, C.M., Hickey, S., Lavery, P.S., et al., 2017a. Assessing the risk of carbon dioxide emissions from blue carbon ecosystems. *Frontiers in Ecology and the Environment* 15 (5), 257–265.
- Lovelock, C.E., Feller, I.C., Reef, R., Hickey, S., Ball, M.C., 2017b. Mangrove dieback during fluctuating sea levels. *Scientific Reports* 7 (1), 1680.
- Lovelock, C.E., Fourqurean, J.W., Morris, J.T., 2017c. Modeled CO₂ emissions from coastal wetland transitions to other land uses: tidal marshes, mangrove forests, and seagrass beds. *Frontiers in Marine Science* 4, 143.

- Lovley, D.R., Dwyer, D.F., Klug, M.J., 1982. Kinetic analysis of competition between sulfate reducers and methanogens for hydrogen in sediments. *Applied and Environmental Microbiology* 43 (6), 1373–1379.
- Luisetti, T., Turner, R.K., Bateman, I.J., Morse-Jones, S., Adams, C., Fonseca, L., 2011. Coastal and marine ecosystem services valuation for policy and management: Managed realignment case studies in England. *Ocean and Coastal Management* 54 (3), 212–224.
- Mack, S.K., Lane, R.R., Day, J.W., 2012. Restoration of Degraded Deltaic Wetlands of the Mississippi Delta v2.0. American carbon Registry (ACR). Winrock international.
- Macreadie, P.I., Nielsen, D.A., Kelleway, J.J., Atwood, T.B., Seymour, J.R., Petrou, K., et al., 2017a. Can we manage coastal ecosystems to sequester more blue carbon? *Frontiers in Ecology and the Environment* 15 (4), 206–213.
- Macreadie, P.I., Ollivier, Q.R., Kelleway, J.J., Serrano, O., Carnell, P.E., Lewis, C.E., et al., 2017b. Carbon sequestration by Australian tidal marshes. *Scientific Reports* 7, 44071.
- Maher, D.T., Santos, I.R., Schulz, K.G., Call, M., Jacobsen, G.E., Sanders, C.J., 2017. Blue carbon oxidation revealed by radiogenic and stable isotopes in a mangrove system. *Geophysical Research Letters*.
- Marbà, N., Arias-Ortiz, A., Masqué, P., Kendrick, G.A., Mazarrasa, I., Bastyan, G.R., et al., 2015. Impact of seagrass loss and subsequent revegetation on carbon sequestration and stocks. *Journal of Ecology* 103 (2), 296–302.
- Marchand, C., Lallier-Vergès, E., Baltzer, F., 2003. The composition of sedimentary organic matter in relation to the dynamic features of a mangrove-fringed coast in French Guiana. *Estuarine, Coastal and Shelf Science* 56 (1), 119–130.
- Mateo, M.A., Romero, J., Pérez, M., Littler, M.M., Littler, D.S., 1997. Dynamics of millenary organic deposits resulting from the growth of the Mediterranean seagrass *Posidonia oceanica*. *Estuarine, Coastal and Shelf Science* 44 (1), 103–110.
- Mateo, M., Cebrián, J., Dunton, K., Mutchler, T., 2006. Carbon flux in seagrass ecosystems. *Seagrasses: Biology, Ecology and Conservation* 159–192.
- Matsui, N., Morimune, K., Meepol, W., Chukwamdee, J., 2012. Ten year evaluation of carbon stock in mangrove plantation reforested from an abandoned shrimp pond. *Forests* 3 (2), 431–444.
- McKee, K.L., Cahoon, D.R., Feller, I.C., 2007. Caribbean mangroves adjust to rising sea level through biotic controls on change in soil elevation. *Global Ecology and Biogeography* 16 (5), 545–556.
- McLeod, E., Chmura, G.L., Bouillon, S., Salm, R., Björk, M., Duarte, C.M., et al., 2011. A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO₂. *Frontiers in Ecology and the Environment* 9 (10), 552–560.
- McOwen, C., Weatherdon, L.V., Bochove, J., Sullivan, E., Blyth, S., Zockler, C., Stanwell-Smith, D., Kingston, N., Martin, C.S., Spalding, M., Fletcher, S., 2017. A global map of saltmarshes. *Biodiversity Data Journal* 5, e11764. <https://doi.org/10.3897/BDJ.5.e11764>. <http://data.unepwcmc.org/datasets/43> (v.5).
- Menéndez, M., Woodworth, P.L., 2010. Changes in extreme high water levels based on a quasi-global tide-gauge data set. *Journal of Geophysical Research: Oceans* 115 (C10).
- Middelboe, A.L., Hansen, P.J., 2007. High pH in shallow-water macroalgal habitats. *Marine Ecology Progress Series* 338, 107–117.
- Millero, F.J., 1995. Thermodynamics of the carbon dioxide system in the oceans. *Geochimica et Cosmochimica Acta* 59 (4), 661–677.
- Miyajima, T., Hori, M., Hamaguchi, M., Shimabukuro, H., Adachi, H., Yamano, H., Nakaoka, M., 2015. Geographic variability in organic carbon stock and accumulation rate in sediments of East and Southeast Asian seagrass meadows. *Global Biogeochemical Cycles* 29 (4), 397–415.
- Mudd, S.M., Howell, S.M., Morris, J.T., 2009. Impact of dynamic feedbacks between sedimentation, sea-level rise, and biomass production on near-surface marsh stratigraphy and carbon accumulation. *Estuarine, Coastal and Shelf Science* 82 (3), 377–389.
- Murdiyarsa, D., Purbopuspito, J., Kauffman, J.B., Warren, M.W., Sasmito, S.D., Donato, D.C., et al., 2015. The potential of Indonesian mangrove forests for global climate change mitigation. *Nature Climate Change* 5 (12), 1089.
- Nellemann, C., Corcoran, E. (Eds.), 2009. Blue Carbon: The Role of Healthy Oceans in Binding Carbon: a Rapid Response Assessment. UNEP/Earthprint.
- Neumann, B., Vafeidis, A.T., Zimmermann, J., Nicholls, R.J., 2015. Future coastal population growth and exposure to sea-level rise and coastal flooding-a global assessment. *PLoS One* 10 (3), e0118571.

- Nixon, S.W., 1980. Between coastal marshes and coastal waters—a review of twenty years of speculation and research on the role of salt marshes in estuarine productivity and water chemistry. In: *Estuarine and Wetland Processes*. Springer US, pp. 437–525.
- Orr, J.C., Fabry, V.J., Aumont, O., Bopp, L., Doney, S.C., Feely, R.A., et al., 2005. Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. *Nature* 437 (7059), 681.
- Ouyang, X., Lee, S.Y., 2014. Updated estimates of carbon accumulation rates in coastal marsh sediments. *Bio-geosciences* 5057.
- Pedersen, M.Ø., Serrano, O., Mateo, M.Á., Holmer, M., 2011. Temperature effects on decomposition of a *Posidonia oceanica* mat. *Aquatic Microbial Ecology* 65 (2), 169–182.
- Pendleton, L., Donato, D.C., Murray, B.C., Crooks, S., Jenkins, W.A., Sifleet, S., et al., 2012. Estimating global “blue carbon” emissions from conversion and degradation of vegetated coastal ecosystems. *PLoS One* 7 (9), e43542.
- Pianka, E.R., 1970. On r-and K-selection. *The American Naturalist* 104 (940), 592–597.
- Raupach, M.R., Canadell, J.G., 2008. Observing a vulnerable carbon cycle. In: *The Continental-Scale Greenhouse Gas Balance of Europe*. Springer, New York, NY, pp. 5–32.
- Reef, R., Feller, I.C., Lovelock, C.E., 2010. Nutrition of mangroves. *Tree Physiology* 30 (9), 1148–1160.
- Saintilan, N., Rogers, K., Mazumder, D., Woodroffe, C., 2013. Allochthonous and autochthonous contributions to carbon accumulation and carbon store in southeastern Australian coastal wetlands. *Estuarine, Coastal and Shelf Science* 128, 84–92.
- Scales, I.R., Friess, D.A., Glass, L., Ravaoarinorotsihoaiana, L., 2017. Rural livelihoods and mangrove degradation in south-west Madagascar: lime production as an emerging threat. *Oryx* 1–5.
- Serrano, O., Lavery, P.S., Rozaimi, M., Mateo, M.Á., 2014. Influence of water depth on the carbon sequestration capacity of seagrasses. *Global Biogeochemical Cycles* 28 (9), 950–961.
- Serrano, O., Ricart, A.M., Lavery, P.S., Mateo, M.A., Arias-Ortiz, A., Masqué, P., et al., 2016a. Key biogeochemical factors affecting soil carbon storage in *Posidonia* meadows. *Biogeosciences* 13 (15), 4581.
- Serrano, O., Ruhon, R., Lavery, P.S., Kendrick, G.A., Hickey, S., Masqué, P., et al., 2016b. Impact of mooring activities on carbon stocks in seagrass meadows. *Scientific Reports* 6, 23193.
- Showstack, R., 2013. Carbon dioxide tops 400 ppm at Mauna Loa, Hawaii. *Eos, Transactions American Geophysical Union* 94 (21), 192.
- Siegenthaler, U., Sarmiento, J.L., 1993. Atmospheric carbon dioxide and the ocean. *Nature* 365 (6442), 119.
- Sippo, J.Z., Maher, D.T., Tait, D.R., Holloway, C., Santos, I.R., 2016. Are mangroves drivers or buffers of coastal acidification? Insights from alkalinity and dissolved inorganic carbon export estimates across a latitudinal transect. *Global Biogeochemical Cycles* 30 (5), 753–766.
- Smale, D.A., Burrows, M.T., Moore, P., O'Connor, N., Hawkins, S.J., 2013. Threats and knowledge gaps for ecosystem services provided by kelp forests: a northeast Atlantic perspective. *Ecology and Evolution* 3 (11), 4016–4038.
- Smale, D.A., Moore, P.J., Queirós, A.M., Higgs, N.D., Burrows, M.T., 2018. Appreciating interconnectivity between habitats is key to blue carbon management. *Frontiers in Ecology and the Environment* 16 (2), 71–73.
- Smith, S.V., 1981. Marine macrophytes as a global carbon sink. *Science* 211 (4484), 838–840.
- Solomon, S., Qin, D., Manning, M., Averyt, K., Marquis, M. (Eds.), 2007. Climate change 2007—the physical science basis: Working group I contribution to the fourth assessment report of the IPCC (Vol. 4). Cambridge university press.
- Spalding, M., Kainuma, M., Collins, L., 2010. *World Atlas of Mangroves*. A Collaborative Project of ITTO, ISME, FAO, UNEP-WCMC. Earthscan, London, UK.
- Steneck, R.S., Dethier, M.N., 1994. A functional group approach to the structure of algal-dominated communities. *Oikos* 476–498.
- Steneck, R.S., Graham, M.H., Bourque, B.J., Corbett, D., Erlandson, J.M., Estes, J.A., Tegner, M.J., 2002. Kelp forest ecosystems: biodiversity, stability, resilience and future. *Environmental Conservation* 29 (4), 436–459.
- Sutton-Grier, A.E., Moore, A.K., Wiley, P.C., Edwards, P.E., 2014. Incorporating ecosystem services into the implementation of existing US natural resource management regulations: operationalizing carbon sequestration and storage. *Marine Policy* 43, 246–253.
- Tambooh, F., Huxham, M., Karachi, M., Mencuccini, M., Kairo, J.G., Kirui, B., 2008. Below-ground root yield and distribution in natural and replanted mangrove forests at Gazi bay, Kenya. *Forest Ecology and Management* 256 (6), 1290–1297.
- Thomas, S., 2014. Blue carbon: knowledge gaps, critical issues, and novel approaches. *Ecological Economics* 107, 22–38.

- Torio, D.D., Chmura, G.L., 2013. Assessing coastal squeeze of tidal wetlands. *Journal of Coastal Research* 29 (5), 1049–1061.
- Trevathan-Tackett, S.M., Kelleway, J., Macreadie, P.I., Beardall, J., Ralph, P., Bellgrove, A., 2015. Comparison of marine macrophytes for their contributions to blue carbon sequestration. *Ecology* 96 (11), 3043–3057.
- Trumper, K., 2009. The Natural Fix?: The Role of Ecosystems in Climate Mitigation: a UNEP Rapid Response Assessment. UNEP/Earthprint.
- Ullman, R., Bilbao-Bastida, V., Grimsditch, G., 2013. Including blue carbon in climate market mechanisms. *Ocean and Coastal Management* 83, 15–18.
- UNEP, 2012. The Emissions Gap Report 2012. United Nations Environment Programme (UNEP), Nairobi.
- UNEP-WCMC, Short F. T., 2017. Global distribution of seagrasses (version 5.0). In: Fourth Update to the Data Layer Used in Green and Short (2003). UNEP World Conservation Monitoring Centre, Cambridge (UK). <http://data.unepwcmc.org/datasets/7>.
- UNFCCC (United Nations Framework Convention on Climate Change), 2016. Report of the Conference of the Parties on its Twenty-first Session, Held in Paris from 30 November to 13 December 2015. FCCC/CP/2015/10/Add.1. Bonn. UNFCCC. <http://unfccc.int/resource/docs/2015/cop21/eng/10a01.pdf>.
- Valiela, I., Bowen, J.L., York, J.K., 2001. Mangrove forests: one of the world's threatened major tropical environments. *BioScience* 51 (10), 807–815.
- Van der Werf, G.R., Morton, D.C., DeFries, R.S., Olivier, J.G., Kasibhatla, P.S., Jackson, R.B., et al., 2009. CO₂ emissions from forest loss. *Nature Geoscience* 2 (11), 737–738.
- VCS (Verified Carbon Standard), 2015. Methodology for Tidal Wetland and Seagrass Restoration (VM003). http://database.v-c-s.org/sites/vcs.benfredaconsulting.com/files/VM0033%20Tidal%20Wetland%20and%20Seagrass%20Restoration%20v1.0%202020%20NOV%202015_0.pdf.
- Ward, L.G., Zaprowski, B.J., Trainer, K.D., Davis, P.T., 2008. Stratigraphy, pollen history and geochronology of tidal marshes in a Gulf of Maine estuarine system: climatic and relative sea level impacts. *Marine Geology* 256 (1–4), 1–17.
- Warren, R.S., Niering, W.A., 1993. Vegetation change on a northeast tidal marsh: interaction of sea-level rise and marsh accretion. *Ecology* 74 (1), 96–103.
- Waycott, M., Duarte, C.M., Carruthers, T.J., Orth, R.J., Dennison, W.C., Olyarnik, S., et al., 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of the National Academy of Sciences* 106 (30), 12377–12381.
- Whalen, S.C., 2005. Biogeochemistry of methane exchange between natural wetlands and the atmosphere. *Environmental Engineering Science* 22 (1), 73–94.
- Wigand, C., Davey, E., Johnson, R., Sundberg, K., Morris, J., Kenny, P., et al., 2015. Nutrient effects on belowground organic matter in a minerogenic salt marsh, North Inlet, SC. *Estuaries and Coasts* 38 (6), 1838–1853.
- Wylie, L., Sutton-Grier, A.E., Moore, A., 2016. Keys to successful blue carbon projects: lessons learned from global case studies. *Marine Policy* 65, 76–84.
- Young, I.R., Zieger, S., Babanin, A.V., 2011. Global trends in wind speed and wave height. *Science* 332 (6028), 451–455.