

12.10 Climate Regulation as a Service from Estuarine and Coastal Ecosystems

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Abstract

Coastal regions, at the interface between terrestrial and oceanic ecosystems, play an important role in global biogeochemical cycles. This chapter reviews the climate regulation services of estuarine and coastal ecosystems (ECEs) including tidal salt marshes, mangroves, seagrass beds, macroalgal forests, coral reefs, and coastal shelf ecosystems. ECEs regulate global and regional climates by sequestering or releasing carbon dioxide and other greenhouse gases (GHGs). ECEs are extremely productive biologically, with net primary production rates per unit area among the highest of any ecosystem. Consequently, ECEs play a globally significant role as carbon sinks, with carbon storage rates per unit area of many habitats far exceeding that of land habitats at the rate of about 10 times that of temperate terrestrial forests and 50 times that of tropical forests. Furthermore, sedimentation does not reach an equilibrium carbon balance as occurs in terrestrial systems, whose sequestration capacity is forecasted to decrease this century. Conversely, they are large potential sources of GHG's if disturbed or mismanaged. Critically, carbon sequestration in many coastal habitats is superior to that of terrestrial habitats, as carbon is generally stored over long time frames (thousands of years) as a consequence of the large belowground biomass and the absence of fire threat. Furthermore, carbon is generally broken down anaerobically; hence, emissions of other potent GHGs such as methane and nitrous oxide are negligible. A review of literature provided sequestration rates for various coastal habitats. Using these in combination with global extent of selected habitats, this chapter finds that GHGs worldwide, mangroves, seagrass beds, and salt marshes combine to sequester a minimum of 136 000 tonnes C annually into long-term carbon storage. Assuming prices of CO₂e from \$10 to \$90 per tonne, the value of the annual sequestration is \$5–45 billion. This is an underestimate due to data gaps and, the limited assessment of the area of these three coastal ecosystem habitats, and relates only to long-term storage. The figures do not include short-term carbon storage in biomass, and further unaccounted for carbon sequestration occurs in kelp forests, estuaries, and coastal shelf seas. Many ECEs are under threat globally from sea-level rise, coastal development, pollution, and other anthropogenic stressors, and protection and restoration of ECEs may be an important tool for mitigating climate change. Currently, these habitats are not included in the United Nations Framework Convention on Climate Change (UNFCCC) carbon accounting frameworks, and therefore are excluded from incentive schemes such as carbon markets and other incentive programs, but their inclusion deserves consideration due to their potential for mitigating global climate change. The global distribution of C sequestration in ECEs reveals that large areas of the tropics are home to the highest sequestration rates and occur in developing countries, which also have the highest rates of coastal habitat degradation. Schemes such as Reducing Emissions from Deforestation and Forest Degradation (REDD) may bring revenues and

added benefits to developing countries for instigating projects and marine protected areas for conservation. Many small island nations and developing countries in the tropics are particularly vulnerable to climate change and contain large swathes of seagrasses and mangroves compared to overall land area, but do not contain large areas of forests or grasslands, which would apply to REDD in its current form. Coastal habitats thus not only present an untapped potential for inclusion in climate change mitigation schemes, but also present a little-recognized risk of loss of large carbon stocks if their degradation and destruction are not reduced or halted.

12.10.1 Introduction

Coastal regions, at the interface of terrestrial and oceanic ecosystems, play an important role in global biogeochemical cycles of the biosphere. Estuarine and coastal ecosystems (ECEs) are capable of regulating both global and regional climates by sequestering or releasing carbon and other greenhouse gases (GHGs), and by modifying local temperature and evapotranspiration regimes. They are extremely productive biologically, with net primary production (NPP) rates per unit area among the highest of any ecosystem. They are generally heterotrophic systems, largely due to the inputs of nutrients and sediments from terrestrial systems via rivers and tidal flows, and thus release carbon dioxide and methane through biotic respiration. Sediments in seagrass and tidal wetland habitats tend to be oxygen deficient; hence, soil respiration and the release of carbon from decaying matter by microbes and fungi are repressed, and thus bioaccumulation of organic matter is larger than in terrestrial ecosystems. In addition, mangroves, salt-marsh plants, and seagrasses have considerable production below ground in roots and rhizomes, which enhances belowground accumulation of carbon. This coastal vegetation is estimated to sequester carbon at the rate of about 10 times that of temperate terrestrial forests and 50 times that of tropical forests (Pidgeon, 2009). Furthermore, sequestration in coastal systems through sedimentation does not reach an equilibrium carbon balance as occurs in terrestrial systems, whose sequestration capacity is forecasted to decrease this century (Friedlingstein et al., 2006; Phillips et al., 2008; Piao et al., 2009). ECEs therefore represent large potential carbon sinks, yet they are also large potential sources of GHGs, if disturbed or mismanaged. This chapter reviews the climate regulation services of key components of ECEs, including tidal marshes, mangroves, seagrass beds, macroalgal forests, coral reefs, and coastal shelf seas.

Impacts on ECEs include conversion or degradation from anthropogenic activities and also climate change. Anthropogenic activities include deforestation of mangroves, and destruction of seagrass meadows through sedimentation from land catchment runoff, fishing practices, and coastal development. Water quality from river runoff and coastal discharge has resulted in eutrophication and hypoxia, and disruption of food webs. Climate change impacts such as sea-level rise will require ECEs to migrate inland, and coastal development will hinder this. Warming seas and ocean acidification may also affect ECEs. The outcome of climate change and direct anthropogenic impacts is disturbance in ecosystem functioning and a reduction in C sequestration potential, and existing C stocks may also be lost or diminished.

Given the importance of ECE in global C cycles, and considering the high levels of impacts they face, possibilities for protecting or reclaiming carbon sequestration potential can contribute significantly to global climate change mitigation.

Mitigation possibilities include protecting ECEs through conservation initiatives, thereby maintaining existing stocks and C sequestration rates. Restoration projects for degraded ECEs can renew their ability to sequester C (Erwin, 2009).

However, regulations and market-based instruments are not yet present to incentivize climate change mitigation projects in ECEs. For terrestrial ecosystems, various mechanisms exist to financially support GHG mitigation projects, such as carbon markets, statutory incentive schemes of national governments, and international initiatives such as Reducing Emissions from Deforestation and Forest Degradation (REDD). Terrestrial ecosystems have accepted GHG accounting frameworks and UNFCCC guidelines for project-level accounting, and require assurances on the additionality and permanence of C sequestered. However, accounting frameworks for ECEs are not yet robust compared to terrestrial ecosystems and, therefore, measuring the change in carbon stocks resulting from coastal mitigation projects is insufficient for assuring financial incentives. Extending existing protection measures such as marine protected areas can conserve ecosystem functions. We thus find that coastal habitats present an untapped potential for inclusion in climate change mitigation schemes such as REDD, particularly for developing countries of the tropics where vast extents of these coastal habitats are found. However, disturbance of ECEs also presents a little-recognized risk of releasing existing C stocks, if their degradation and destruction are not reduced or halted.

12.10.1.1 Definition and Global Occurrence

Coastlines vary greatly around the world from frozen polar shorelines to tropical mangroves and beaches. There is estimated to be >1.6 million km of coastline globally (UNEP, 2006). Coastal populations are growing rapidly, with over a third of the world's population living on or near the coast, and more than half of the coastal countries presently having >80% of their population living in coastal areas (Burke et al., 2001; UNEP, 2006; Martinez et al., 2007). Here, we define the coastal region to include coastal seas on the continental shelf (to a depth of 200 m), the coastline, and the adjacent land that is routinely inundated with seawater. This area covers multiple ecosystems across open coastlines, embayments, and estuaries. In general, tropical and subtropical coastlines are dominated by mangroves, sandy beaches, coral reefs, and seagrass beds, whereas tidal marshes, macroalgal forests, and seagrass beds abound on higher latitude coastlines.

Coastal shelf seas cover 26×10^6 km² of the global ocean. Within or adjoining these, there are estimated to be 600×10^3 km² of coral reefs, 300×10^3 km² of seagrass beds, 160×10^3 km² of mangroves, 22×10^3 km² of salt marsh, and 15×10^3 km² of kelp forests (Gattuso et al., 1998; Chmura et al., 2003; Bouillon et al., 2009; Kennedy and Bjork, 2009; Reed and Brzezinski, 2009). These figures are likely to be

grossly underestimated for seagrasses, salt marsh, and kelp forests, as many country inventories are incomplete or missing, particularly for developing nations in Africa, Asia, Oceania, and South America.

12.10.1.2 Climate Regulation Services

ECEs have been overlooked as globally significant and easily manageable carbon sinks. Despite their relatively small surface area (~7% of global ocean surface area), they are significant in the carbon cycle (Gattuso et al., 1998). They receive massive inputs of terrestrial nutrients and sediment and organic matter from rivers and groundwater discharge, exchange nutrients and carbon with the open ocean, and sustain a disproportionately large geochemical and biological activity compared to their relative surface area (Gattuso et al., 1998; Giraud et al., 2008; Chen and Borges, 2009). Almost half of the world's primary production occurs in the oceans (Field et al., 1998) and ECEs account for 14–30% of the oceanic primary production and >40% of oceanic carbon sequestration (Gattuso et al., 1998; Muller-Karger et al., 2005; Giraud et al., 2008).

Above ground, ECEs are generally heterotrophic systems, with respiration and decomposition contributing carbon dioxide to the atmosphere. However, for seagrasses, mangroves, and salt marshes, organic carbon is accumulated in plant biomass, including in the extensive belowground biomass, and deposited in sediments, resulting in these ECEs being a net sink of GHGs. Chen and Borges (2009) reviewed the literature on carbon balances of coastal systems, and concluded that most open-shelf areas are sinks for atmospheric CO₂, although many inner estuaries, nearshore coastal waters, and intensive upwelling areas are oversaturated in CO₂. On the whole, continental shelves are significant sinks for atmospheric CO₂, which corresponds to 27–30% of the CO₂ uptake by the open oceans (Gattuso et al., 1998; Chen and Borges, 2009).

ECEs have the potential to rapidly sequester carbon and remain long-term sinks (Brevik and Homburg, 2004). Mangroves, salt marshes, and seagrasses account for 9%, 25%, and 12% respectively of long-term carbon sequestration in sediments (Duarte et al., 2005). Fast carbon sequestration rates mean that on a country scale, coastal ecosystems may sequester as much carbon as terrestrial forests, despite their smaller areas (Laffoley and Grimsditch, 2009). Furthermore, much of this sequestration may be long-term (+1000 years) and, therefore, is inherently more valuable than carbon sequestered by terrestrial forests, which is stored over decades before being returned to the atmosphere through biological decomposition or fire.

12.10.1.3 Anthropogenic Impacts on Climate Regulation Services from Estuaries and Coasts

Coastal regions have been modified by humans, with impacts separated here into recent and historical direct impacts, those induced by recent climate change, and those projected for the future. Anthropogenic impacts can alter the structure and ecosystem processes of ECEs, reducing their ability to sequester carbon, and threaten the stocks of carbon that have built up over time. Direct anthropogenic disturbances include the clearing, draining, or filling of coastal habitats for agricultural, aquaculture, forestry, industrial, and residential use. Land use

in coastal catchments can result in altered sediment and nutrient delivery to coastal habitats through changes in runoff, river flow, and groundwater flows. Broadly, ECEs are threatened by climate change through rising sea levels, which will drown coastal fringes where sedimentation rates are too low to allow comparable accretion rates or where the landward migration of coastal habitats is blocked by coastal development or natural features. Microtidal areas are predicted to be more vulnerable to sea-level rise than macrotidal areas, although enhanced storm activity and alteration to rainfall regimes may increase physical disturbance and erosion and, therefore, vulnerability to sea-level rise. Rising temperatures will alter community structure and lead to local expansions or contractions of vegetated habitat. Ocean acidification is expected to impact marine species that have calcareous structures, such as corals (Hoegh-Guldberg et al., 2007), and may alter food webs.

Anthropogenic disturbance or climate change impacts can lead to degradation of ecosystem functions and processes of ECEs. At local and regional scales, habitats are currently being degraded and substantially altered. Eutrophication and hypoxia result in pulses of enhanced productivity of coastal waters, which increases GHG emissions, particularly N₂O. At regional to global scales, damage to ecosystem functioning through pollution and overfishing leads to declines in fauna and flora, and climate change impacts alter the physical and chemical processes underlying ecosystem processes. Sea-level rise presents a threat to coastal ecosystems, and responses of ECEs will depend on the local and regional 'foundation species' that provide the basis of habitat structure and control ecosystem dynamics, sediment budgets, and local hydrodynamics. Coastal erosion and associated habitat loss, altered tidal ranges, changed sediment and nutrient transport, and increased coastal flooding present adverse conditions to coastal ecosystems, particularly where responses are restrained by other anthropogenic stresses.

Coastal developments can act to restrict landward migration of coastal habitats with rising sea levels, termed the 'coastal squeeze', leading to losses of intertidal vegetation and a reduction of coastal carbon sequestration ability. Even where there may be limited opportunity for migration inland of an intertidal ecosystem such as mangroves, this is likely to be at the expense of other higher terrain ecosystems, such as salt-marsh habitat, thus altering carbon budgets.

ECEs have already been altered by anthropogenic and climate change impacts, which combine to degrade system health and resilience. Globally, 29% of seagrass area has disappeared (Waycott et al., 2009), and it is estimated that 35% (see Chapter 12.06) to 50% (Bouillon et al., 2009) of the area of mangroves has been lost, with 20% of the pristine area removed since the 1980s (FAO, 2007). Estimates of salt-marsh loss are also high (Adams, 2002), with current loss or degradation estimated at 50% (Barbier et al. in this volume), and with estimates of future loss of up to 22% of the world's coastal wetlands by 2080 (Nicholls et al., 1999). Kelp forests are threatened by declining water quality, disturbance of kelp food webs through overfishing (Steneck et al., 2002; Ling et al., 2009), warming of coastal seas, and species introductions or distributional expansions (e.g., in Tasmania (Ling, 2008)). Widespread loss of vegetated coastal habitats is estimated to have reduced carbon burial in the ocean by around 30 Tg-C yr⁻¹, representing 13% of global ocean burial (Duarte et al., 2005). Corals reefs are susceptible to

pollution, eutrophication, and sedimentation resulting from land-use change as well as destructive fishing practices. A total of 19% of coral reefs have been lost over the past 60 years, while 20% are under threat of loss over the next 20–40 years (Wilkinson, 2004). The widespread bleaching on coral reefs in response to prolonged periods of elevated temperatures, not recorded anywhere before 1979, is probably one of the most widely recognized climate change threats. Ocean warming coupled with ocean acidification is predicted to substantially reduce coral cover by the end of this century (Hoegh-Guldberg et al., 2007).

12.10.2 Climate Regulation Services of Coastal Habitats

Ecosystem services are defined as the benefits people obtain from ecosystems (UNEP, 2006) through functions or processes contributing to human well-being or having the potential to do so in the future (EPA, 2009). These benefits accrue according to different types of values such as direct and indirect use values, as well as existence and bequest nonuse values. Thus, ecosystems are assets that produce a flow of beneficial goods and services over time (Chapter 12.6; Barbier (2009)), and contribute to human welfare, both directly and indirectly, and therefore represent part of the total economic value of the planet (Costanza et al., 1997).

Coastal regions provide ecosystem services for all of the 17 services listed in Costanza et al. (1997), and account for the highest per hectare value for any ecosystem determined in Costanza et al. (1997). The goods from ECE include food for humans and animals (including fish, shellfish, krill, and seaweed); salt; minerals and oil resources; construction materials (sand, rock, coral, lime, and wood); and biodiversity, including the genetic stock that has potential for various biotechnology and medicinal applications (Burke et al., 2001; UNEP, 2006). Here, we outline the contributions of ECE to regulating climate through carbon dynamics of specific habitats.

Carbon dioxide released through anthropogenic sources is taken up by terrestrial and oceanic ecosystems. Although highly variable each year, on average 43% of anthropogenic emissions each year remains, in the atmosphere (Le Quere et al., 2009). Around 30% of the CO₂ produced (about 2 Gt C yr⁻¹) since the Industrial Revolution has been absorbed by the oceans (Orr et al., 2001; Sabine et al., 2004; Denman et al., 2007). Current estimates of carbon sequestration are about 2.2 GtC yr⁻¹ for the oceans over the 1980s through to 2005, with the land sink fluctuating from 0.3 GtC yr⁻¹ in the 1980s to 1.0 GtC yr⁻¹ in the 1990s and 0.9 GtC yr⁻¹ over 2000–05 (Denman et al., 2007). However, model projections suggest that CO₂ absorbed by land sinks may decrease during this century, as soils saturate and vegetation systems mature (Friedlingstein et al., 2006; Phillips et al., 2008; Piao et al., 2009). Although carbon is generally stored in plants over decadal timescales, C stocks equilibrate over time to a no-net-gain-or-loss balance as C is released back from terrestrial habitats as CO₂ via death and subsequent decomposition, burning, and herbivory/grazing. By contrast, carbon sequestration rates by coastal vegetation are maintained over thousands of years because sediments do not become carbon saturated

and habitats are not threatened by fire (Wanless and Tagett, 1989; Lo Iacono et al., 2008).

The coastal zone plays a significant role in biogeochemical cycles because it receives inputs of terrestrial organic matter and nutrient through runoff and groundwater discharge, and also exchanges large amounts of matter and energy with the open ocean (Gattuso et al., 1998). Estimates of the contribution of ECEs to net global air–sea CO₂ exchange vary, as it is uncertain how much organic carbon arrives from rivers and runoff, and also how much primary productivity is permanently sequestered in coastal habitats, on the shelf, or exported to the deeper ocean (Chen and Borges, 2009). Global estimates, extrapolated from regional observations, suggest that the coastal ocean may be a significant net sink for CO₂ (Tsunogai et al., 1999), although there are uncertainties in underlying mechanisms and temporal and spatial patterns in fluxes (Ianson et al., 2009). The surface aquatic component of tidal wetlands and estuaries, excluding sedimentation, may be net emitters of CO₂, while offshore continental shelves act as sinks (Borges et al., 2005; Chen and Borges, 2009). An analysis of the literature scaling up across ecosystem components and latitudes suggests that the oligotrophic tropical and subtropical coastal oceans may act as sources of CO₂ to the atmosphere, whereas coastal seas at high and temperate latitudes are sinks of CO₂ from the atmosphere (Borges et al., 2005; Chen and Borges, 2009). This, however, does not consider the net balance of GHG, which also includes the sedimentation process. When including a full net GHG balance, coastal regions are a significant sink of GHGs. Overall, coastal oceans may represent an additional sink corresponding to 27–30% of the CO₂ uptake by the open ocean (Chen and Borges, 2009).

Mangroves, salt marshes, seagrass meadows, macroalgal forests, and coastal shelves sequester carbon from the atmosphere and, in some cases, from dissolved aquatic CO₂. Estuaries, reefs, and coastal soils (notably agricultural land) may act as sources of CO₂, and biodegradation in coastal areas involves methane and nitrogen cycling. Coasts also regulate local climate through evapo-transpiration, dimethyl sulfide production that causes cloud formation, and gas exchange.

Vegetated coastal ecosystems are significant carbon sinks, with the exception of macroalgal forests, which are not ‘rooted’ in sediments. The large carbon sink capacity of ECEs is a consequence of their slow decomposition rates in anaerobic conditions, which build organic soil stocks. ECEs also trap large quantities of sediments from anthropogenic and natural water sources. Carbon fluxes and stock size vary widely, with tidal marshes having NPP rates matching that of tropical forests (Mitra et al., 2005). It has been estimated that detritus burial from vegetated coastal habitats contributes about half of the total carbon burial in the ocean (Duarte et al., 2005). Each unit of carbon sequestered in mangroves, salt marshes, and seagrasses is considered to be of greater value than that stored in any other natural ecosystem due to the lack of production of other GHGs, as tidal wetlands produce little methane gas compared to freshwater wetlands, which emit a large fraction of the annual global flux of methane (Chmura et al., 2003; Bridgman et al., 2006; Bricker et al., 2008; Chmura, 2009).

Here, we describe the major coastal habitats, focusing on their role in climate regulation and, in particular, carbon sequestration. A summary of the major coastal habitats

Table 1 Estimated long-term carbon sequestration and value for coastal ecosystems including mangroves, seagrass and salt marsh, reporting global area based on Martinez et al. (2007) for mangroves and Duarte et al. (2005) for salt marsh and seagrass, long-term carbon burial rates from Laffoley and Grimsditch (2009), and converted into values for CO₂-e assuming a \$10–\$90 range.

Coastal ecosystem	Global area (km ²)	Long-term carbon burial rates (Gg-C yr ⁻¹)	Value of annual sequestration service (at \$10–\$90 tCO ₂ -e ⁻¹)
Mangroves	196 816	27 357	\$1–\$9B
Seagrass	300 000	24 900	\$0.9–\$8.2B
Salt-marsh	400 000	84 000	\$3–\$27.7B
Total	896 816	136 257	\$5–\$45B

described, their area, long-term carbon burial rates, and value of their carbon sequestration is given in Table 1. Figures of long-term carbon sequestration are likely to be underestimates as they do not account for tidal pumping of CO₂ from these habitats onto the outer shelf and into the open ocean (Pidgeon, 2009), and are limited by inadequate estimates of the area of many of these habitats and the limited research in carbon cycling in many of these habitats. There is also uncertainty about how much carbon is sequestered into shelf soft sediments, which is excluded from the figures in Table 1, and discussed in detail in the following sections.

12.10.2.1 Salt Marshes

Salt marshes are found on coastlines globally, and are particularly prevalent at temperate latitudes. In the tropics and subtropics, salt marshes tend to form the landward fringe of mangroves. They are formed principally of salt-tolerant herbaceous plants. No single global inventory of salt-marsh area has been published, but regional assessments for Canada, Europe, the USA, and South Africa estimate an area of 22 000 km² (Chmura et al., 2003). There is also an estimated 13 600 km² of estuarine salt marsh in Australia and 2133 km² of salt marsh

on the Atlantic coast of South America below southern Brazil. The global extent is considerably greater though, as temperate coasts of Asia, South America, and Oceania likely to hold substantial extents of salt marsh are not currently inventoried (Chmura et al., 2003). Salt marshes are important carbon sinks and long-term carbon sequestration rates (Gg-C yr⁻¹) globally are shown in Figure 1.

Salt marshes are one of the most productive ecosystems in the world (able to sequester up to 3900 g-C m⁻² yr⁻¹) and therefore sequester millions of tons of carbon annually (Cebrian, 2002). Salt-marsh waters are net emitters of CO₂ to the atmosphere, but above- and belowground biomass represent a sink that is higher than the emission from the aquatic compartment (Chen and Borges, 2009). Primary production is stored not only as aboveground biomass, but also as belowground vascular biomass and nonvascular plant biomass (Chmura, 2009). These components vary considerably among sites, but in many cases the proportion of production held below ground far outweighs that above ground (Connor et al., 2001). Soils of salt marshes store, on average, 210 g-C m⁻² yr⁻¹, driven by rapid burial rates, which converts to 770 g of CO₂ m⁻² yr⁻¹ (Chmura et al., 2003; Chmura, 2009). Globally, 430 Tg of carbon is stored in the upper 50 cm of sediment in tidal salt marshes, and this is a conservative estimate because there can be carbon stored deeper in the sediment and the amount does not necessarily decline significantly with depth (Chmura, 2009). Because of the anoxic nature of the marsh soils (as in most wetlands), carbon sequestered by salt-marsh plants during photosynthesis is often lost from the short-term carbon cycle (10–100 years) to the long-term carbon cycle (1000 years) as buried, slowly decaying root material.

Importantly, tidal marsh sediments continue to accumulate carbon over long time periods through accretion, and therefore do not reach carbon equilibrium as terrestrial soils do, where carbon inputs are balanced by decomposition and release of CO₂ (Connor et al., 2001). Sediment levels in salt marshes can rise continually, provided favorable conditions of plant growth persist, through the accumulation of peat and organic-rich sediments (Cahoon et al., 2006). This

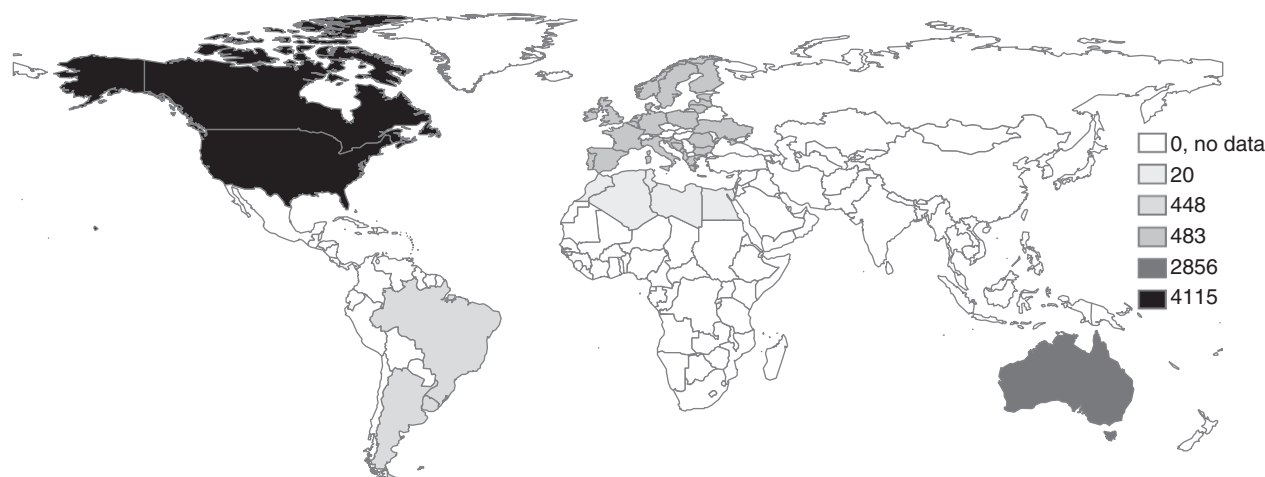


Figure 1 Estimated long-term carbon sequestration rates (Gg-C yr⁻¹) of salt marshes within global regions, calculated from salt marsh area reported in Martinez et al. (2007) and long-term carbon sequestration rates (210 g-C m⁻² yr⁻¹) from Laffoley and Grimsditch (2009). Note that salt-marsh inventories are incomplete for many countries reporting data. White countries are either landlocked, or do not contain salt marshes or inventory data is not available.

capability is unique compared to many ecosystems, where carbon is mostly turned over quickly and does not often move into the long-term carbon cycle. Thus, salt marshes are as effective as Arctic tundra and boreal forests at sequestering carbon into the long-term carbon cycle (Mayor and Hicks, 2009).

Climate change and direct anthropogenic impacts threaten the ability of salt marshes to sequester carbon and adapt to climate change. In particular, rapid sea-level rise threatens tidal wetlands. Salt marshes have the ability to track sea-level rise, increasing surface elevation through accretion depending on sediment budgets, vegetation type, and local hydrodynamics (Pye, 1995; Connor et al., 2001; Mudd et al., 2009). However, once sea-level rise reaches a critical rate, the marsh drowns or erodes, and carbon accumulation is halted (van der Wal and Pye, 2004; Mudd et al., 2009). On developed coastlines, the 'coastal squeeze', where hard built structures act as physical barriers, can restrict the ability of tidal marshes to retreat inland with rising sea levels and lead to losses of salt-marsh area (Hughes and Paramor, 2004). Erosion or disturbance of salt marshes or drowned marshes may result in the release of stored carbon from sediments. The encroachment of mangroves in salt-marsh areas is also being recorded, linked to warming temperatures and/or changes in rainfall regimes (Perry and Mendelssohn, 2009). Initial carbon assimilation rates appear to be similar but the long-term effects on ecosystem processes are undetermined.

Other threats to salt marshes include disturbance of hydrology and sediment regimes by dredging, groundwater extraction and alteration of runoff and river flows, and input of nutrients from fertilized watersheds (Chmura, 2009). Although salt marshes are valued for their role in filtering nutrients, nutrient enrichment can have negative impacts, decreasing belowground production and carbon accumulation rates, and soil elevation (Darby and Turner, 2008a, 2008b; Turner et al., 2009).

12.10.2.2 Mangroves

Mangroves are diverse assemblages of trees, shrubs, palms, and ferns that are adapted to the intertidal zone of flat, sheltered coastlines in the tropics and subtropics

(FAO, 2007). Global mangrove area is estimated at 152 000–160 000 km², with 43% found in Indonesia, Australia, Brazil, and Nigeria (Alongi, 2002). Mangroves provide a wide range of ecosystem services and are particularly important for communities in developing, tropical nations with subsistence economies (Alongi, 2002). Mangroves are highly productive systems and extremely efficient carbon sinks (Komiya et al., 2008). The estimated carbon stock in mangroves (7990 g-C m⁻²) is of similar magnitude to estimated stocks in tropical forests (12 045 g-C m⁻²), temperate forests (5673 g-C m⁻²), and boreal forests (6423 g-C m⁻²) (Pidgeon, 2009). The proportional allocation of carbon below ground is similar to that of terrestrial forests, but soil respiration (CO₂ efflux) is lower (Lovelock, 2008; Pongpam et al., 2009). In contrast to terrestrial forests, however, mangroves (together with salt marshes and seagrasses) can sustain high carbon sequestration rates even as the forest matures (Jennerjahn and Ittekkot, 2002; Tamoo et al., 2008). Mangroves are important carbon sinks, and long-term carbon sequestration rates (Gg-C yr⁻¹) globally are shown in Figure 2.

Global NPP of mangroves is estimated at 218 ± 72 Tg-C yr⁻¹, with root production constituting 38% and litter fall and wood production around 31% each (Bouillon et al., 2008). Mangroves have up to 60% of their total dry weight below ground (Comley and McGuinness, 2005; Komiya et al., 2008) in extensive root systems, often in anoxic sediments, which accounts for the rapid carbon sequestration rates (Tamoo et al., 2008). On an areal basis, mangroves are generally more productive than salt marshes and seagrasses (Alongi, 2002). Productivity is higher in the tropics than in the subtropics, with a decline in aboveground biomass with increasing latitude (Alongi, 2002; Komiya et al., 2008).

Overall, mangroves are a net sink of CO₂. However, estuarine mangrove creek waters are significant sources of CO₂ and methane, particularly after heavy or monsoonal rainfall when phytoplankton blooms are stimulated (Borges, 2005; Biswas et al., 2007; Chen and Borges, 2009), although these levels are low in contrast to freshwater peatlands (Chmura et al., 2003). Emissions for the aquatic component of mangroves is outweighed by the greater carbon sink in aboveground biomass



Figure 2 Estimated long-term carbon sequestration rates (Gg-C yr⁻¹) of mangrove habitats by global regions, calculated from long-term sequestration rates reported in Laffoley and Grimsditch (2009) (139 g-C m⁻² yr⁻¹) and mangrove area from Martinez et al. (2007). White countries are either landlocked or do not contain mangroves.

and sediments (Chen and Borges, 2009). Carbon sinks in mangroves operate over both short timescales (decadal) through net growth of forest biomass and long timescales through burial of organic matter in sediments (Bouillon et al., 2008). Mangrove systems are estimated to bury $18.4 \text{ Tg-C yr}^{-1}$ in sediments globally, although the rate of carbon storage in sediments can vary considerably among mangrove systems (Bouillon et al., 2008, 2009). Due to the fast rate of carbon transfer to sediments and the fact that efficiency of carbon sequestration in sediments does not decrease with the age of the mangrove forest (Jennerjahn and Ittekkot, 2002), mangroves generate ~15% of the total carbon accumulation in ocean sediments. Mangroves can raise sediment levels through the accumulation of organic-rich sediments (Cahoon et al., 2006). Mangroves are also expanding into new areas and accumulating carbon, such as on damaged coral reefs (Alongi et al., 2008) and on rapidly accreting coastlines caused by land-use change (Lovelock, 2008).

The presence of mangroves increases sedimentation rates, and, conversely, the clearing of mangroves can rapidly result in reduced C stores in sediments, up to 50% in 8 years (Granek and Ruttenberg, 2008). The loss of the carbon sink thus goes beyond the loss of biomass and can erode past stores.

Mangroves are highly sensitive to changes in soil salinity and humidity; hence, rainfall patterns are a key regulating factor (Eslami-Andargoli et al., 2009). Photosynthesis and reproductive success are limited by cool temperatures toward higher latitudes; therefore warming may result in an expansion of mangrove extent at these latitudes. Certainly, temperature may be a contributing factor to observations of mangrove expansion at high-latitude range edges (Perry and Mendelssohn, 2009), although changes in land use may be the dominant factor (McKee and Rooth, 2008; Harty, 2009).

Mangroves, in common with many terrestrial trees, have also been shown to emit hydrocarbons (Barr et al., 2003). After release, their conversion contributes to aerosol formation and haze, thus regulating local climate. However, for red mangroves at least, emissions have been shown to be low compared to terrestrial forests, although compounds may be released during flowering in sufficient amounts to contribute to biogenic aerosol formation (Barr et al., 2003).

Mangrove forests are estimated to have covered 75% of tropical coasts, but have been reduced to <50% of original total cover by over-harvesting for wood, agriculture conversion, mining, oils spills, pollution, and damming (Alongi, 2002). Mangrove clearing in the 1980s and 1990s has led to the loss of at least $3.8 \times 10^{14} \text{ g C}$ in mangrove biomass, which is a gross underestimate as belowground biomass and detrital mass is not included (Cebrian, 2002). Mangroves are heavily used traditionally, as sources of wood for building materials and for food provision in terms of gathering and cultivation of shellfish, fish, and crustaceans, and commercially for wood products (Alongi, 2002; see also Chapter 12.08). The greatest threats to mangroves are deforestation, overexploitation of wood and fisheries resources, and pond aquaculture, particularly prawn aquaculture, which requires clearing of mangroves, alters groundwater tables, and releases toxic wastes (Farnsworth and Ellison, 1997; Alongi, 2002).

Mangroves are extremely sensitive to impacts of climate change, particularly rising sea level and alteration of rainfall. A number of factors interact to determine the vulnerability of

mangroves to sea-level rise, including coastal sediment budgets, tidal range, growth rates of trees, and species composition. Reduction of sediment supply slows accretion rates in mangroves, reducing their ability to track rising seas, while excessive sediment input can lead to reduced productivity and even mortality (Ellison and Stoddart, 1991; Ellison, 1998). Coastal developments also threaten the ability of mangrove systems to respond to rising sea levels through coastal squeeze (Gilman et al., 2007). However, elevated CO_2 may accelerate mangrove growth, hence enhance allocation to roots, and therefore sequester carbon in sediments (McKee and Rooth, 2008; Langley et al., 2009).

The restoration (reforestation) of mangrove stands can greatly increase the carbon sequestration ability of the degraded landscapes (Ren et al., 2010). The belowground biomass in replanted mangrove stands may be similar to that in natural stands after only a decade (Tamooch et al., 2008). The fast growth rates and rapid carbon accumulation rates of young plants, coupled with the maintenance of high carbon sequestration rates in mature forests, warrants the consideration of mangrove reforestation for carbon management, as well as land restoration.

12.10.2.3 Seagrass Meadows

Seagrasses are marine flowering plants found in intertidal and shallow waters on most coastlines globally. They can form extensive meadows, with temperate meadows tending to be dominated by one or two larger species and tropical meadows having a greater diversity of plants. Seagrass global extent is estimated at 0.3 million km^2 (Kennedy and Bjork, 2009), but this is likely to be grossly underestimated as inventories have not been carried out or are incomplete for many countries. Seagrasses are important carbon sinks, and long-term carbon sequestration rates (Gg-C yr^{-1}) globally are shown in Figure 3.

Seagrasses form an important carbon sink, responsible for 15% of carbon storage in the oceans (Duarte and Cebrian, 1996; Suzuki et al., 2003). They are extremely productive systems comparable to tropical and temperate forests and grasslands (Rasheed et al., 2008; Kennedy and Bjork, 2009). Seagrass carbon burial rates are about half as fast on a per area basis as mangroves and salt marshes (Duarte et al., 2005).

Seagrass meadows can form long-term carbon sinks because much of their biomass (15–50%) is below ground, depending on the species (Duarte et al., 1998; Duarte and Chiscano, 1999). Seagrasses tend to have an extensive underground system of roots and rhizomes (horizontal underground stems that can form extensive networks) and spread through vegetative budding of rhizomes as well as by flowering and seed production. Larger species such as *Posidonia oceanica*, which is endemic to the Mediterranean Sea, have large rhizomes and can raise the seafloor by several meters over thousands of years (Wanless and Tagett, 1989; Romero et al., 1994; Lo Iacono et al., 2008). More than a third of the primary production (rhizomes and leaf sheaths) may be stored in the matte and organic carbon content can be as high as 40% where decomposition rates are slow (Pergent et al., 1994, 1997; Romero et al., 1994). Mattes can persist for millennia, providing a long-term carbon sink. Vertical accretion rates of *P. oceanica* have been estimated at $0.13\text{--}1.1 \text{ m kyr}^{-1}$, with the base of some mattes radiocarbon dated to 6000 years old



Figure 3 Estimated long-term carbon sequestration rates (Gg-C yr^{-1}) of seagrass habitats by global regions, calculated from long-term sequestration rates from Laffoley and Grimsditch (2009) ($83 \text{ g-Cm}^{-2} \text{ yr}^{-1}$) and area of seagrass from Martinez et al. (2007). White countries are either landlocked, do not contain seagrasses or inventory data is unavailable (limited data for SE Asia, Oceania, Caribbean, Africa).

(Lo Iacono et al., 2008; Lopez-Saez et al., 2009). *P. oceanica* meadows in the Mediterranean Sea are estimated to store between 1.2×10^6 tonnes (Pergent et al., 1997) and 6×10^{15} tonnes of carbon per year (Kennedy and Bjork, 2009). *P. oceanica* is currently thought to be the most effective seagrass species for long-term carbon storage, although the extent to which other species generate refractory carbon deposits is currently unclear and the species-specific carbon storage rates are unknown.

Generally, only an average of 19% of seagrass production is lost to herbivory, with around half of the biomass decomposed, and the rest exported to deeper water or stored (Pergent et al., 1994, 1997; Mateo and Romero, 1997; Cebrian, 2002). Modeling of CO_2 sequestration from *Zostera marina* seagrass beds to the deep ocean over 100 years suggests that 4–5% of gross production accumulates in deep water (Suzuki et al., 2003). Once in the deep ocean, CO_2 is effectively part of the long-term carbon cycle as the residence time of water in the deep ocean is about 1600 years (Garrison, 2007). Short-term global carbon storage rates are estimated at 27 Tg-C yr^{-1} , with longer-term rates around 40 Tg-C yr^{-1} (Kennedy and Bjork, 2009).

Around 29% of seagrass beds have disappeared in the past 130 years, and rates of decline have accelerated since 1990 (Waycott et al., 2009). These losses are attributed to reduced water quality from changes in land use resulting in eutrophication and sedimentation, coastal development, invasive species, and climate change (Walker and McComb, 1992; Ruiz and Romero, 2003; Orth et al., 2006; Short et al., 2006; Bricker et al., 2008; Freeman et al., 2008; Waycott et al., 2009).

Sea-level rise exacerbated by increases in intense storms regimes is likely to reduce seagrass area, although coastal inundation could lead to the formation of additional seagrass habitat. A 50-cm rise in sea level could result in a 30–40% reduction in the growth of the widespread Northern Hemisphere seagrass *Z. marina* (Short and Neckles, 1999). Both direct and indirect impacts of recent climate change on seagrass beds have been observed (Short et al., 2006). For example, stronger and more frequent storms in Tamandré, Brazil, have led to increased sediment movement and a decline in seagrass. Warmer winters in northeast USA have caused Canada geese to overwinter there rather than further south,

increasing grazing pressure on intertidal seagrass beds (Short et al., 2006).

Increasing atmospheric CO_2 concentrations are expected to result not only in higher dissolved CO_2 concentrations in coastal waters, but also in an increase in the relative proportion of dissolved CO_2 to HCO_3^- (Short and Neckles, 1999). The effect of this on seagrasses may be positive, with increases in productivity and biomass (Invers et al., 2002; Palacios and Zimmerman, 2007), but it is likely that this will be counteracted by warmer waters reducing CO_2 solubility. Furthermore, benefits of increased productivity from elevated CO_2 levels cannot be attained when other factors such as nutrients, light, or temperature are limiting.

12.10.2.4 Kelp Forests

Kelps are large, brown algae found along rocky intertidal and shallow coastal areas globally. They dominate autotrophic biomass in temperate and Arctic regions, where they form extensive stands or forests (Velimirov et al., 1977; Attwood et al., 1991; Steneck et al., 2002). Giant canopy-forming kelp, *Macrocystis* spp., can grow to 45 m long within a couple of years (Steneck et al., 2002). A total of 58 774 km of coastline globally is believed to support kelp, although a complete world survey has not been conducted (Steneck et al., 2002; Reed and Brzezinski, 2009). Little is known of tropical kelps, but deepwater kelp forests do occur in the tropics, where cool oceanic currents and upwelling exist below uninhabitable warm surface waters (Graham et al., 2007; Santelices, 2007). These tropical kelp forests are estimated to cover $23\,504 \text{ km}^2$ (Graham et al., 2007).

The most conspicuous kelps are the giant kelps *Laminaria*, *Ecklonia*, and *Macrocystis*. They are found along temperate, nutrient-rich coastlines where they are foundation ecosystem species supporting a wide variety of plants and animals, including commercial species of fish, lobster, abalone, and urchins (Velimirov et al., 1977; Bustamante and Branch, 1996; Steneck et al., 2002; Graham, 2004). Kelp forests are one of the most productive ecosystems on Earth, with fast growth rates averaging 2–4% of standing biomass per day (Mann, 1973; Hatcher et al., 1977). NPP of kelp forests is in the range of

670–1750 g-C m⁻² yr⁻¹, depending on species, greater than the productivity of terrestrial forests (Reed and Brzezinski, 2009).

Considering temperate systems, the global kelp standing crop has been roughly estimated at 7.5 Tg C. Understory algae may increase this by a further 20% and deeper-water tropical kelp may increase estimates to 20 Tg C. Conservatively, global kelp production may be 15 Tg-C yr⁻¹ and may approach 39 Tg-C yr⁻¹ if deeper-water tropical kelp species are considered (Reed and Brzezinski, 2009).

Unlike salt-marsh plants, mangroves, and seagrasses, kelp species have no belowground biomass, as kelp plants are anchored to the surface of hard substrate by holdfasts. Carbon cycling in kelp is characterized by rapid biomass turnover and kelp detritus is consumed, decomposed, or exported out of the system. Up to 30–40% of kelp NPP may be exuded annually as dissolved organic matter (Abdullah and Fredriksen, 2004; Wada et al., 2007). Along higher latitude coasts, where kelp growth is limited seasonally, dissolved organic matter can continue to be exuded and play a significant role in coastal ecosystems (Abdullah and Fredriksen, 2004). Kelp detritus (as wracks and kelp-derived particulate organic matter) is also exported to adjacent ecosystems (Velimirov et al., 1977; Attwood et al., 1991; Wernberg et al., 2006; Crawley et al., 2009). The majority of kelp carbon can be exported many kilometers where it supports benthic and, possibly, planktonic food webs (Harrold et al., 1998; Vetter and Dayton, 1999; Rodriguez, 2003; Kaehler et al., 2006; Vanderklift and Wernberg, 2008).

The potential for long-term carbon sequestration from kelp forests is reliant on detritus reaching the deep ocean and/or being buried in marine sediments. It is likely that the carbon storage by kelp will be a function of the kelp standing crop and understory algae, and most importantly, its eventual export to deeper waters, which is unknown and varies among systems, but may be substantial in many cases (Attwood et al., 1991). Given the fast growth rates of kelp and other microalgae, there is potential for CO₂ mitigation through harvesting and appropriate processing of kelp (Ritschard, 1992; Chung et al., 2009).

Kelp forests are sensitive to environmental conditions, and are bioindicators of the integrity of temperate reef ecosystems (Steneck et al., 2002). As growth depends on interactions among temperature, nutrient availability, and light, kelp forests are threatened by declining water quality and climate change (Steneck et al., 2002). Furthermore, overfishing causes cascading disturbance of kelp food webs, as removal of lobsters, fish, and sea otters has led to increases of grazer populations such as urchins that can deplete kelp forests (Mann and Breen, 1972; Steneck et al., 2002). The transformation of southeast Australian kelp forests to urchin barrens has been linked to warmer temperatures exacerbated by strengthening of the warm, poleward-flowing, oligotrophic East Australian Current (Ridgway, 2007), enabling a range expansion of the grazing sea urchin *Centrostephanus rodgersii* (Poloczanska et al., 2007). The resilience of these kelp forests was further reduced by lobster fishing, which increased the risk of a catastrophic shift to urchin barrens as lobsters are the major predators of sea urchins (Ling, 2008; Ling et al., 2009).

12.10.2.5 Coral Reefs

Coral reefs are complex structures made from calcium carbonate secreted by colonies of small cnidarians known as polyps.

Coral reefs are distributed in warm, shallow tropical and subtropical seas covering some 600 000 km² (Smith and Gattuso, 2009), although inclusion of other shallow tropical to high-latitude benthic communities that exhibit primary production and calcification might double this area (Andersson et al., 2005). Coral reefs are crucibles of biodiversity known as ‘rain-forests of the sea’, occupying <1% of the area of the ocean but harboring 25% of all marine species (Knowlton, 2001; Hughes et al., 2002; Briggs, 2005). Services provided by coral reefs are critical for the livelihoods of millions of people and include fisheries, coastal protection, building materials, new biochemical compounds, and tourism (Hoegh-Guldberg et al., 2007).

Most of the carbon (~95%) sequestered by coral reefs is inorganic carbon in reef calcification (Smith and Gattuso, 2009). They are rapid producers of organic carbon and skeletal calcium carbonate. Rising atmospheric CO₂ concentrations have led to enhanced oceanic CO₂ and altered the chemical speciation of the oceanic carbon system (McNeil et al., 2003). Put simply, when CO₂ dissolves in seawater, it forms a weak acid (H₂CO₃) that dissociates to bicarbonate (HCO₃⁻). Precipitation of calcium carbonate (CaCO₃) from seawater uses HCO₃⁻ and not CO₂, and because of an associated pH shift, actually releases CO₂ back into the atmosphere (see Smith and Gattuso, 2009 for further details). It is estimated that coral reefs sequester around 80 Tg-C yr⁻¹ globally (this carbon coming from HCO₃⁻), but reef calcification processes emit around 50 Tg-C yr⁻¹ in the form of dissolved CO₂, and this is larger if nonreef calcifying benthic ecosystems are considered (Smith and Gattuso, 2009). Therefore, coral reefs, perhaps counterintuitively, act as slight ‘carbon sources’ rather than ‘carbon sinks’ due to their effect on local ocean chemistry (Borges, 2005; Smith and Gattuso, 2009). Coral reef systems, thus, do not have the potential to play a significant role in the long-term management of GHGs as in other coastal habitats (Smith and Gattuso, 2009).

A total of 19% of coral reefs have been lost over the past 60 years, while 20% are under threat of loss over the next 20–40 years (Wilkinson, 2004). Much of this is because of eutrophication and removal of grazing fish, but sea surface temperature rise and ocean acidification are considered as increasingly important threats to the biological performance and survival of corals and calcareous algae (Hoegh-Guldberg et al., 2007). Increasing CO₂ concentration in seawater will reduce calcification (Kleypas et al., 1999; Feely et al., 2004). In fact, De’ath et al. (2009) present evidence that coral calcification on the Great Barrier Reef has already declined by 14% since 1990. There is growing evidence that calcification on coral reefs will diminish in the future, as the aragonite saturation state declines, and this will reduce coral reefs as a source of CO₂ in the future. A major unknown is whether ocean acidification will tip coral reefs (and carbonate sediments in general) from a state of net calcification to one of net dissolution. This would mean that coral reefs and carbonate sediments could thus constitute a potential sink for anthropogenic CO₂ due to alteration of the chemical speciation of oceanic carbon system (Smith and Gattuso, 2009).

12.10.2.6 Estuaries

Estuaries are highly dynamic interfaces between rivers and oceans, with strong physical, chemical, and biological

gradients. Tidal salt marshes, mangroves, and seagrasses all form important habitats within estuaries and these have already been discussed in detail; hence, here we will restrict our discussion to the implications of sediment transport for carbon fluxes. Processes within estuaries drive the flux of carbon from continents to the open ocean and influence carbon metabolism of the coastal zone (Gattuso et al., 1998). Large changes in the fluxes and speciation of riverine-dissolved and particulate material occur within estuaries.

A large fraction of the particulate load of rivers can accumulate within estuaries and not reach the continental shelf (Gattuso et al., 1998). In many estuaries, the slow rate of sea-level rise over the past 5000 years has allowed development of carbon-rich deposits as much as 6 m thick (Chmura, 2009). Long-term sediments that accumulate in outer estuaries, near river mouths and in large river deltas, may account for 80% of organic carbon in marine sediments (Bianchi and Allison, 2009). A decrease in sediment loading of rivers from human activities may result in a reduction of the carbon storage efficiency of sediments in deltas, and lead to carbon being dispersed over a greater area (Bianchi and Allison, 2009).

Human disturbance has altered riverine sediment input into the coastal ocean and the composition of organic and inorganic material transported to estuaries, thus modifying the sequestration and emission of anthropogenic CO₂ (Gattuso et al., 1998; Borges, 2005; Bianchi and Allison, 2009). Increased fluxes of nutrients and organic carbon have led to eutrophication and anoxia in disturbed areas, which affect other biogeochemical processes (Gattuso et al., 1998; Bianchi and Allison, 2009), and can promote jellyfish blooms (Richardson et al., 2009). The balance between autotrophy and heterotrophy has been modified by human disturbance, but is difficult to quantify because of antagonistic responses (Gattuso et al., 1998). Generally, estuaries are net sinks for organic matter and of CO₂ from the surrounding water, but a source of CO₂ to the atmosphere (Gattuso et al., 1998; Borges, 2005; Chen and Borges, 2009). Estuaries also produce other gases important in climate regulation, in particular the GHGs nitrous oxide, methane, and carbonyl sulfide, and dimethyl sulfide, which can be a precursor for cloud formation, particularly where nutrient loading is high or anoxic sediments occur (Gattuso et al., 1998; Biswas et al., 2007; Bianchi and Allison, 2009).

12.10.2.7 Continental Shelf Seas

The coastal ocean plays a key role in global climate regulation services, as it receives massive inputs of terrestrial organic matter and nutrients, exchanges large amounts of matter and energy with the open ocean, and is extremely active both geochemically and biologically (Gattuso et al., 1998). Continental shelf areas represent ~7% (26×10^6 km²) of the global ocean surface, but account for 14–30% of oceanic primary production (primarily by phytoplankton) and 80% of total organic matter burial and 75–90% of the oceanic sink of suspended river loads (Gattuso et al., 1998).

Evidence is mounting that open continental shelves act as a sink for CO₂, exporting carbon to their sediments and adjacent deep oceans (Borges, 2005; Chen and Borges, 2009). Gradients in the physical, chemical, and biological properties of shelf

systems generate fluxes of matter and energy between the shelf and the open ocean at shelf margins (Gattuso et al., 1998). There are three key ‘pumps’ that interact and can deliver CO₂ into the deep ocean, and these can contribute to a positive feedback between climate change and the ocean carbon cycle. Such positive feedback cycles can lead to rapid and potentially unstable climate shifts.

The first major pump operating on continental shelves that can remove CO₂ absorbed in surface waters and transport it to the deep ocean (at least subthermocline) is the ‘continental shelf pump’ (Tsunogai et al., 1999). There is evidence from a variety of systems that the continental shelf pump may operate in many areas globally (Yool and Fasham, 2001). In polar areas, the continental shelf pump may work through the formation of sea ice in shelf areas, which leaves behind hypersaline and dense waters that sink into the deep ocean. In temperate regions, the continental shelf pump could work through winter cooling of surface waters, which leads to this denser water sinking off the shelf. In tropical regions, evaporation of water on shelves can cause increased salinity and density of shelf waters, which could then sink into the deep ocean.

The other major pump on the shelf that can remove CO₂ from surface waters is the ‘biological pump’. Phytoplankton fix dissolved CO₂ during photosynthesis, reducing the concentration of CO₂ at the ocean surface. This maintains a diffusion gradient between the atmosphere and ocean, continually drawing CO₂ from the atmosphere into the oceans. Plankton play a further role in the biological pump because much of the CO₂ that is fixed by phytoplankton and then eaten by zooplankton sinks to the ocean floor in the bodies of uneaten and dead phytoplankton, and zooplankton fecal pellets. This carbon can then be locked up in sediments and forms part of the long-term carbon cycle. Climate change could negatively impact the biological pump. As the oceans heat up, the increased stratification could result in reduced nutrients from colder, deeper waters mixing into surface waters. This is likely to lead to a lower phytoplankton abundance, dominated by smaller phytoplankton cells (Bopp et al., 2005). This reduces the efficiency of the biological pump.

Shelf ecosystems, and the phytoplankton that dominate their primary productivity, also help to shape global climate by influencing the amount of solar radiation reflected back to space. Many phytoplanktons, including diatoms and coccolithophores, produce dimethylsulfonium propionate, a precursor of dimethylsulfide (DMS). DMS evaporates from the ocean, is oxidized into sulfate in the atmosphere, and then forms cloud condensation nuclei. This leads to more clouds, increasing the Earth’s albedo, and cooling the climate.

Processes such as seasonal stratification and productivity cycles will influence carbon budgets and flows on open continental shelves (Borges, 2005). Temperate shelf systems can be net heterotrophic in winter and net autotrophic in summer, reflecting seasonal cycles in photosynthesis and temperature (Gattuso et al., 1998). Coastal seas that are stratified seasonally may be more efficient exporters of carbon to the deep ocean than well-mixed regions (Borges, 2005). Continental shelves in high latitudes are significant sinks for CO₂, whereas tropical and subtropical coastal shelves are considered as sources of CO₂ due to oversaturation with respect to atmospheric CO₂ (Borges, 2005; Borges et al., 2005).

12.10.3 Coastal Anthropogenic Impacts and Changes in GHG Dynamics

Anthropogenic impacts alter the structure and function of coastal ecosystems and, thus, the provision of ecosystem services (ES). A total of 72% of the coastal zone globally (defined by the 100 km inland limit) is still covered by 'natural' ecosystems, with 28% altered by human activities such as urban development and agriculture (Martinez et al., 2007). All face climate change impacts through processes such as rising sea level, warming sea temperatures, ocean acidification, and altered storm intensity and rainfall regimes. Climate change will directly and indirectly impact the physical and biological processes that underpin the ES from coastal habitats. Although many impacts are thought to be negative, there is the potential for enhancement of some coastal ecosystems and their ES through the formation of new habitats, increased availability of CO₂ for plants that are CO₂ limited, and longer-term burial of sequestered carbon, should other climate change impacts not cause adverse conditions for functional ecosystem processes.

The ability of coastal ecosystems to minimize emissions and sequester carbon is compromised by numerous human impacts other than climate change. Coastal habitats are cleared, drained, overfished, or filled for agricultural, forestry, aquacultural, industrial, and residential use. Inappropriate urban development is a rapidly escalating threat given the burgeoning coastal populations in many regions of the world. Apart from urban areas, changes in land use threaten coastal habitats through alteration of runoff, river flow and groundwater reserves, sediment loading, and inputs of agricultural and waste nutrients and pollutants.

The literature contains many empirical studies with site-specific conclusions, but extrapolating globally involves high uncertainty. Global climate change models are poorly resolved for the coastal zone and capture few of the underlying processes in this region. However, understanding changes to coastal ecosystems and their GHG budgets under climate change can be inferred through repeated examples in the literature of ecosystems losing critical functions and altering their structure, and thus resilience to disturbance. Because impacts are numerous and can be synergistic, threats facing coastal ecosystems require a perspective of cumulative impacts.

12.10.3.1 Coastal Modification and Use

Humans have modified coastal ecosystems, ever since man started exploiting coastal resources for food and materials (Jackson, 2001; Chmura, 2009; Gedan et al., 2009). Modeling studies suggest that the global coastal ocean has changed from a CO₂ source to a CO₂ sink over the past century, as a consequence of increasing input of organic matter and nutrients from human activities, and rising atmospheric CO₂ levels, and is likely to continue as a CO₂ sink in the near future (Andersson et al., 2005).

During the past century, human modification of coastal systems has greatly accelerated. The number of hypoxic (low oxygen) zones is doubling in the coastal margin every decade since the 1960s due to human activities,

particularly in estuaries and deltas (Diaz and Rosenberg, 2008; Bianchi and Allison, 2009). Hypoxia has detrimental effects on the growth and survival of organisms as well as on sedimentary processes such as nitrogen fluxes (Diaz and Rosenberg, 2008). About 20% of global mangrove area has been destroyed since the 1980s (FAO, 2007), with the greatest threat being the conversion of mangroves to agricultural or infrastructure land or aquaculture production, particularly in Southeast Asia (Farnsworth and Ellison, 1997; Alongi, 2002). A total of 29% of seagrass area globally has disappeared since 1880 due to anthropogenic impacts, in particular unsustainable and destructive coastal development, destructive fishing practices, and declining coastal water quality (Waycott et al., 2009). The conversion of coastal habitats for human use and issues associated with increasing human populations such as declining water quality (sediment and nutrient loading, pollutants), alteration of hydrological patterns, and heavy exploitation are threats that resonate across coastal ecosystems globally (e.g., Kennish, 2001; Steneck et al., 2002; Laffoley and Grimsditch, 2009; Bulleri and Chapman, 2010).

Indirectly, ECEs are impacted by disruption of coastal food webs and other components of ecosystem interactions. Removal or increases of key predators or grazers can have a cascading effect through ecosystems (Jackson et al., 2001). For example, historical hunting of large herbivores found on seagrass beds (turtles, dugongs, and manatees) has altered the ecology of these habitats in ways that have increased their sensitivity to disturbance (Jackson et al., 2001). The balance of interactions between competitors, predators, and grazers can be shifted by disturbance (Daleo and Iribarne, 2009; Holdredge et al., 2009) and lead to a reduction of carbon storage rates.

Anthropogenic modification of ECEs affects GHG emission regimes and carbon sequestration rates. The potential for carbon sequestration in mangroves, salt marshes, and seagrasses depends on the maintenance of vegetation cover, the species in the community, and sedimentation rates (Chmura et al., 2003; Bouillon et al., 2008; Chmura, 2009). Nutrient enrichment of ECEs can lead to enhanced productivity but this may be at the expense of carbon sequestration rates (Darby and Turner, 2008a, 2008b; Turner et al., 2009). Eutrophication and hypoxia also lead to depletions of associated fauna and flora in coastal habitat regions, or pulses of enhanced primary productivity and increases in GHG emissions (in particular N₂O) from decomposition of organic matter (Naqvi et al., 2000; Diaz and Rosenberg, 2008).

Clearing or alteration of ECEs can result in large emissions of GHGs releasing carbon stored from their deposits. Although estimates are imprecise, the carbon stock in ECEs is likely to be similar in magnitude to the stores found in terrestrial forests and croplands. Clearing of mangroves results in significantly reduced carbon content in sediments (Granek and Ruttenberg, 2008; Kristensen et al., 2008), indicating that deforestation not only removes more carbon than expected through removal of biomass (Bouillon et al., 2009), but also liberates some existing carbon in sediments. Removing or degrading these habitats may release immediate carbon emissions comparable to those from cleared or degraded terrestrial systems (Pidgeon, 2009).

12.10.3.2 Impacts of Climate Change

NPP of terrestrial systems has likely increased over recent decades, and is linked to CO₂ fertilization, global warming, and, in some places, increased precipitation (Lewis et al., 2009; Piao et al., 2009), with an estimated global increase of 0.4% yr⁻¹ from 1980 to 2002 (Piao et al., 2009). Coastal plants (mangroves, seagrasses, and salt-marsh vegetation) are also expected to show increased NPP with CO₂ fertilization and may even increase carbon storage capability, as long as they are not adversely affected by other impacts of climate change such as increased inundation, sea-level rise (and coastal squeeze), and altered rainfall and nutrient regimes (Farnsworth et al., 1996; McKee and Rooth, 2008; Charles and Dukes, 2009).

Another anticipated impact of climate change is the poleward shift in the ranges of many species, including commercial fish, as temperatures warm, with secondary effects on their predators and prey (Scavia et al., 2002). Differential species' responses will result in shifts in community structures of ECEs and thus alteration of GHG balances. The expansion of mangroves into salt-marsh areas, a likely response to warming, precipitation changes, and sea-level rise where coastal infrastructure prevents salt marshes from moving inland, is expected to alter the ecosystem processes of salt marshes (Perry and Mendelssohn, 2009). Warming may directly increase GHG emissions (Chmura et al., 2003) as decay processes increase. CO₂ emissions from plants and microbial communities in soils approximately double with every 10 °C increase in temperature (Lovelock, 2008).

Ocean acidification as a result of increased CO₂ concentrations may broadly damage marine habitats, selectively impact species with calcium carbonate structures, alter marine resource availability, and disrupt other ecosystem services. Coral reef bleaching is expected to become more frequent with warming, while ocean acidification may reduce coral calcification, making it more difficult for corals to recover from other disturbances (Scavia et al., 2002; Baker, 2008; Lough, 2008). Although the impact on reef systems is not expected to be accompanied by significant changes in emissions or sequestration dynamics, the loss of coral could result in reduction of shoreline protection and losses of mangroves, seagrasses, and coastal marshes.

Sea-level rise is a particular threat to coastal ecosystems, and responses of ECEs will depend on vegetation, sediment budgets, and local hydrodynamics. Evidence suggests that low-lying coastal habitats are degrading in response to recent sea-level rise (Kirwan and Temmerman, 2009). Sea-level rise may accelerate coastal erosion and associated habitat loss, increase salinity in estuaries and groundwater, alter tidal ranges, change sediment and nutrient transport, and increase coastal flooding. The productivity and species composition of ECEs are strongly regulated by soil salinity, coastal water quality, and, in the case of mangroves and salt marshes, by humidity. They are, therefore, also sensitive to changes in rainfall. Landward migration of salt marshes (Kennish, 2001; Stevenson et al., 2002) and mangroves (Jagtap and Nagle, 2007) is not expected to be adequate to replace existing communities because the sea level is rising faster than the plants can colonize new areas and generate functional ecosystem processes. Wetland losses will occur under sea-level rise, with estimates of sea-level rise causing the loss of up to 22% of the world's coastal wetlands by

2080. The largest losses are anticipated for the Mediterranean and Baltic Seas and to a lesser extent for the Atlantic coast of Central and North America and smaller islands of the Caribbean (Nicholls et al., 1999).

Degradation or losses of ECEs will result in decreased carbon sequestration and increased GHG emissions, and may contribute to positive feedback to the global carbon cycle (Henman and Poulter, 2008). In many areas, intensive human alteration and use of coastal environments have already reduced the capacity of ECEs to respond to climate change.

12.10.4 Maintaining ES and Mitigating GHGs in Coastal Zones

Climate change will exacerbate many of the current stressors on ECEs, and management and governance practices may need to be adjusted to accommodate the effects of climate change. Adaptation, which can be autonomous or planned intervention, will increase the coping capacity of ECEs and so maintain their ecosystem services. ECEs also have a role to play in mitigation, either by reducing their role as a source of carbon or by increasing their storage capability as a sink. Coastal areas offer opportunities for GHG mitigation through various anthropogenic interventions.

Significant losses of ECEs are occurring globally due to inadequate management, which is exacerbated by climate change, and a lack of policy priority to address present and future threats (Laffoley and Grimsditch, 2009). There is a need to invoke adequate management to conserve the ES from these habitats under a multitude of stressors, including climate change, and to ensure that they remain sinks and not sources of GHGs.

12.10.4.1 Adaptation and Mitigation Potential

Rising awareness of the economic and ecological values of many ECEs has resulted in increased efforts to facilitate the adaptation of ECEs and increase the adaptive capacity of industries and communities dependent upon them. Such policies will ultimately protect or enhance the climate regulation services of ECEs. Such policies are often implemented at local and regional scales, and improved governance and adaptation of management approaches are necessary. Policies aimed at reducing other stressors on ECEs (such as coastal protection, water quality management, and nonimpacting coastal development) are critical for enhancing the resilience of coastal systems to climate change.

A key adaptation strategy that needs to be enhanced to maintain climate services provided by coastal ecosystems is to limit coastal squeeze. If land is uninhabited or relatively undeveloped, there is an opportunity for the landward migration of many habitats such as mangroves and salt marshes as sea levels rise. However, if there are built structures that prevent this migration landward, these ecosystems are likely to be reduced by coastal squeeze (Figure 4). Even where there may be opportunity for the migration inland of an intertidal ecosystem such as mangroves, this can be at the expense of other higher terrain ecosystems such as salt marshes.

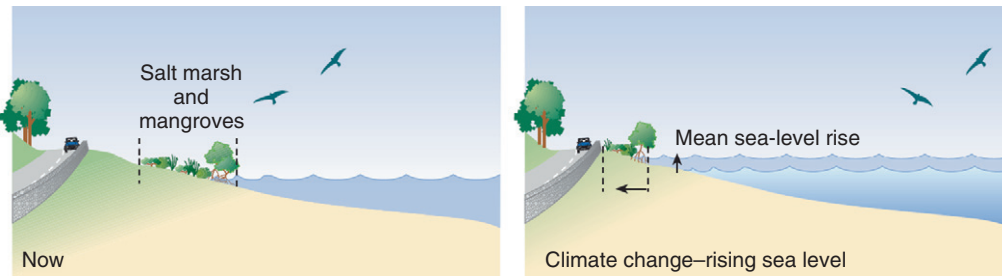


Figure 4 Coastal habitats caught in a coastal squeeze between a rising sea level and hard infrastructure. From AG (Australian Government Department of Climate Change), 2009. *Climate Change Risks to Australia's Coasts: A First Pass National Assessment*. Chapter 4: ISBN 9781-1-921298-71-4.

ECEs can also contribute to mitigation of climate change through mechanisms such as regulations and market-based instruments for environmental management. The mitigation activities of natural carbon sinks have progressed for terrestrial ecosystems, but ECEs as carbon sinks have received little attention though they have the potential to complement land-based measures and enhance global options for carbon management (Laffoley and Grimsditch, 2009). There are benefits in conserving existing ECEs, given their high carbon sequestration efficiencies and the size of carbon sinks.

Restoration of tidal wetlands can enhance global carbon sinks given their high productivity, high rates of carbon storage, potential for long-term sequestering as the volume of soil continues to increase over long time periods, and small or negligible emissions of methane and N_2O (Connor et al., 2001). In the Bay of Fundy area (Canada), it has been calculated that restoring all the tidal wetlands drained for agriculture during the seventeenth to nineteenth centuries would sequester an additional $0.36 \text{ Tg-C yr}^{-1}$, equivalent to 4–6% of Canada's targeted reduction of 1990-level emissions of CO_2 under the Kyoto Protocol (Connor et al., 2001). Constructed salt marshes along the North Carolina coast, USA, were found to achieve equivalence to natural marshes for most ecological attributes after 5–15 years, thus providing sustainable ecosystem services comparable to services provided by natural marshes (Craft et al., 2003). Furthermore, the organic carbon pool in salt-marsh soils improved with marsh age, although carbon sequestration rates were more rapid in young, constructed marshes (Craft et al., 2003). Over the long term, however, the constructed marshes may be less effective at sequestering carbon than natural marshes (Craft et al., 2003). Protective measures for conserving natural ECEs or those proposed to allow adaptation to climate change will also serve to enhance carbon storage facilities. Mangroves can successfully colonize intertidal muddy sediments and grow rapidly and, hence, are ideal for reforestation projects (Cann et al., 2009), while seagrass replanting is a viable option for degraded coastal waters (Rodriguez-Salinas et al., 2010).

Because coral reefs are insignificant carbon sources, they are not a useful management option in their own right for management of anthropogenic CO_2 emissions, but coral reefs, and the ecosystem goods and services they produce, are likely to benefit from other carbon sequestration initiatives that will slow and reduce ocean acidification (Smith and Gattuso, 2009).

Overall, mangroves, salt marshes, and seagrasses have a far greater capacity (per unit of surface area) than vegetation on land or kelp forests to achieve long-term carbon sequestration in sediments, arising in part from the extensive belowground biomass of the dominant vegetation and their capacity for continual accretion of sediments. The rate of carbon storage in the sediment by these ECEs is approximately 10 times the rate observed in temperate forests and 50 times the rate observed in tropical forests per unit area. Combined with the particular ways in which such habitats trap carbon, this means that they are critical components to include in future carbon management discussions and strategies (Connor et al., 2001; Laffoley and Grimsditch, 2009).

Project-based mitigation options also exist for coastal ecosystems, similar to the many terrestrial projects that are increasingly finding value in carbon markets and are contributing to national GHG accounts. One example is the production of biomass from algae, which can be done using aquaculture-style production systems and wild oceanic algae (Skjanes et al., 2007; Packer, 2009). Biomass produced from this high productivity system might be used to generate offset permits, provide human and nonhuman food sources, and be used in the production of biofuels. However, the first step in achieving such mitigation projects is a GHG accounting framework inclusive of aquatic ecosystems.

Despite the importance of GHG fluxes from coastal ecosystems, these are neither accounted for in UNFCCC and Kyoto guidelines, nor in National Inventories. Not only does this mean that countries are underestimating their anthropogenic emissions, but also that GHG mitigation does not count toward national GHG emissions targets (Pidgeon, 2009). This accounting gap is due to the requirement that accountable emissions are directly sourced from anthropogenic activities. This allows terrestrial land use to be accountable and considered as a mitigation option. The 2006 Intergovernmental Panel on Climate Change (IPCC) Good Practice Guidelines for agriculture, forestry, and other land uses recognizes direct human-induced effects on GHG emissions and removals that occur on six categories of management lands including forest land, cropland, grassland, wetland, settlements, and other land, which together incorporate all 'managed lands' within a country.

The lack of inclusion of ECE in GHG inventories is due to a number of factors. First, we have a smaller knowledge base concerning GHG dynamics in coastal and shelf zones compared to that of terrestrial systems. Second, there is also a perspective that marine resources are used but perhaps not 'managed' directly in the same way as terrestrial systems.

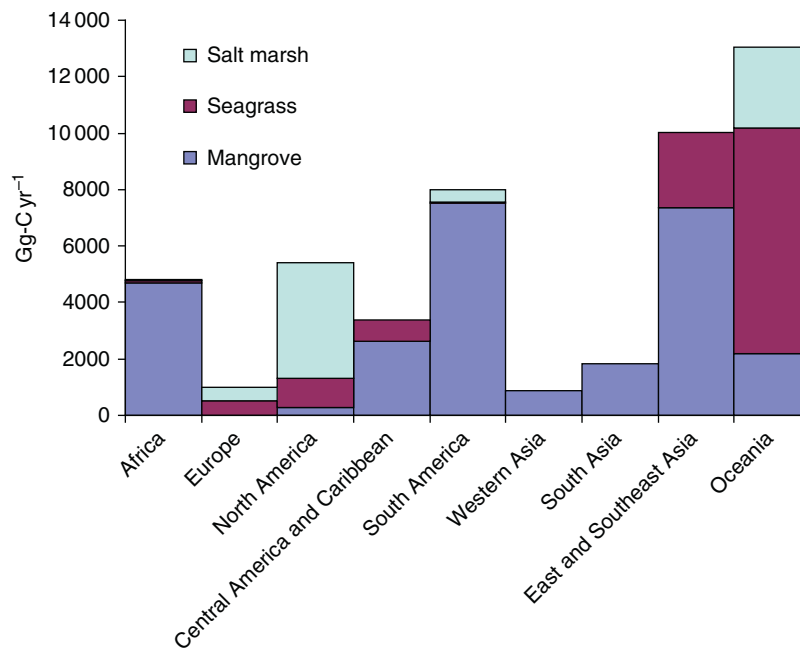


Figure 5 Long-term carbon sequestration rates (Gg-C yr^{-1}) of coastal habitats within world regions. Rates from Laffoley and Grimsditch (2009) for mangroves ($139 \text{ g-Cm}^{-2} \text{ yr}^{-1}$), seagrass ($83 \text{ g-Cm}^{-2} \text{ yr}^{-1}$), and salt-marsh ($210 \text{ g-Cm}^{-2} \text{ yr}^{-1}$). Regional extent of habitats calculated from Martinez et al. (2007), with additional estimates of salt-marsh extent from Adams (2002), Chmura et al. (2003), and Isacch et al. (2006). Note that salt-marsh inventories are not available for Central America, the Pacific coast of South America, Asia, and is limited for Africa. Sequestration is therefore underestimated, especially given the potential salt-marsh expanse along the temperate coastlines of Asia.

However, for the case of mangroves and salt marshes in particular, use and conversion of these ecosystems is a direct human impact of management or lack of management upstream. Last, accounting frameworks do not consider a whole-of-system perspective on the ultimate source of impacts. For example, the decline of seagrass beds is directly related to land use and pollution from sedimentation, a second-order impact from upstream activities that are accountable. These reasons serve to omit GHG emissions and sinks from ECE in IPCC accounting frameworks.

International schemes such as REDD could be amended to include ECEs, once accounting frameworks are in place. Figure 5 presents global ECE sequestration by region, highlighting that much of the annual sequestration occurs in regions in developing countries. These countries are also likely to have higher rates of ECE loss due to overuse and lack of protection mechanisms.

12.10.5 Conclusions

Coastal ecosystems play a critical role in global climate and the carbon cycle by sequestering carbon. Carbon storage rates per unit area exceed those of terrestrial ecosystems. For the coastal ecosystems examined here, mangroves, salt marshes, and seagrasses combine to sequester a minimum of 136 000 tonnes C into long-term storage annually. Assuming prices of CO_2e from \$10 to \$90 per tonne, the value of the annual sequestration is \$5–45 billion. This is an underestimate due to data gaps and the limited assessment of these three coastal ecosystem areas.

ECEs are large potential sources of GHGs if disturbed or mismanaged. Impacts on ECEs include conversion or degradation from anthropogenic activities, and also climate change. Anthropogenic activities include conversion and draining of salt marshes, deforestation of mangroves, and destruction of seagrass meadows through fishing and coastal development. Climate change-induced impacts include sea-level rise and ensuing coastal squeeze, and warming seas and ocean acidification.

Possibilities for conserving and reclaiming ECEs can contribute significantly to global climate change mitigation. Mitigation possibilities include protecting ECEs through conservation initiatives, thereby maintaining existing stocks and carbon sequestration rates. Restoration projects for degraded ECEs can renew their ability to sequester. Unlike terrestrial ecosystems, coastal carbon stocks are not accounted under UNFCCC guidelines and national GHG inventories. Developing accounting frameworks is the first step in enabling incentive schemes, which can assist in mitigating climate change through protecting and restoring coastal carbon sinks. For terrestrial ecosystems, various mechanisms, such as carbon markets, statutory incentive schemes of national governments, and international initiatives such as REDD, exist to financially support GHG mitigation projects. The distribution of global ECE carbon sequestration capacity highlights that much of the annual carbon burial occurs in developing countries in tropical regions. These regions likely face the highest environmental impacts and lack secure marine conservation institutions. Similar to tropical forests, tropical ECEs seem well suited for international financial transfers for the protection of ecosystem services in developing countries.

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