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The Management of Natural Coastal Carbon Sinks

Edited by Dan Laffoley and Gabriel Grimsditch

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The Management of Natural Coastal Carbon Sinks

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Foreword

Climate change is arguably one of the biggest issues facing humanity. World leaders now recognise that urgent and significant reductions in our emissions of greenhouse gasses are needed if we are to avoid future dangerous climate change. Alongside such measures is an increasingly strong recognition that there is a need to properly manage particular habitats that act as critical natural carbon sinks. This is to ensure that they retain as much of the carbon trapped in the system as possible, and don't tend to become 'sources' to the atmosphere through poor management. Often the release of trapped carbon as carbon dioxide is accompanied by the release of other powerful greenhouse gases such as methane, and this situation exacerbates an already concerning global climate situation.

In recent decades there has been a significant focus, quite rightly, on major carbon sinks on land such as forests, particular soil types and peatland habitats. These are ecosystems that by their ecology inherently hold vast reservoirs of carbon, and where management can be put in place to attempt to retain such reserves within the natural systems. The challenge is to recognise other carbon sinks that could contribute and ensure that they too are subject to best practice management regimes.

Until now surprisingly little attention appears to have been paid to the ocean, despite the fact that this is a critical part of the carbon cycle and one of the largest sinks of carbon on the planet. This lack of attention may in part be due to a mistaken belief that quantification of discreet marine carbon sinks is not possible, and also in the mistaken belief that there is little management can do to sustain such marine carbon sinks.

The origin of this report lies within IUCN's World Commission on Protected Areas and Natural England in the UK, and a joint enthusiasm to address this issue. This initial enthusiasm sparked the interest of many global partners and scientists when it became apparent that evidence is available that could change the emphasis on the management of carbon sinks. There is an urgent need for the global debate and action now to encompass marine habitats, just as we already value and try to best protect more familiar forests and peatlands on land.

Over the past two years we have sought out and worked with leading scientists to document the carbon management potential of particular marine ecosystems. It turns out that not only are these habitats highly valuable sources of food and important for shoreline protection, but that all of them are amenable to management as on land when it comes to considering them as carbon sinks. In the ocean this management would be through tools such as Marine Protected Areas, Marine Spatial Planning and area-based fisheries management techniques. This report documents the latest evidence from leading scientists on these important coastal habitats.

Given the importance of examining all options for tacking climate change we hope the evidence in this report will help balance action across the land/sea divide so we don't just think about avoiding deforestation, but we also think about similarly critically important coastal marine habitats. We hope this report will, therefore, serve as a global stimulus to policy advisors and decision makers to encompass coastal ecosystems as key components of the wide spectrum of strategies needed to mitigate climate change impacts.

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Scale of Units used

Value	Symbol	Name
10^{3} g	kg	kilogram
10^{6} g	Mg	megagram (tonne)
10^{9} g	Gg	gigagram
10^{12} g	Tg	teragram
10^{15} g	Pg	petagram
10^{18} g	Eg	exagram
10^{21} g	Zg	zettagram

One Gigatonne = 1000 Teragrams
One hectare = 10,000 m^2

Executive Summary

This report focuses on the management of natural coastal carbon sinks. The production of the report has been stimulated by an apparent lack of recognition and focus on coastal marine ecosystems to complement activities already well advanced on land to address the best practice management of carbon sinks. The production of this report is timely as a number of Governments are now introducing legislation to tackle climate change. In the UK, for example, the Climate Change Act sets out a statutory responsibility to quantify natural carbon sink as part of the overall carbon accounting process. It is important that such quantifications and processes work with the latest science and evidence.

To construct this report we asked leading scientists for their views on the carbon management potential of a number of coastal ecosystems: tidal saltmarshes, mangroves, seagrass meadows, kelp forests and coral reefs. The resultant chapters written by these scientists form the core of this report and are their views on how well such habitats perform a carbon management role. These ecosystems were selected because the belief from the outset was that they are good at sequestering carbon, and are located in situations where management actions could secure the carbon sinks. There are of course other features of our ocean that are already established as good carbon sinks – the key focus for this initial work has, however, been on those ecosystems where management intervention can reasonably readily play a role in securing and improving the future state of the given carbon sinks. If proven this work could expand the range of global options for carbon management into coastal marine environments, unlocking many possibilities for action and possible financing of new management measures to protect the important carbon sinks.

The key findings of this report are:

- These key marine ecosystems are of high importance because of the significant goods and services they already provide as well as the carbon management potential recognised in this report, thus providing new convergent opportunities to achieve many political goals from few management actions.

- The carbon management potential of these selected marine ecosystems compares favourably with and, in some respects, may exceed the potential of carbon sinks on land. Coral reefs, rather than acting as 'carbon sinks' are found to be slight 'carbon sources' due to their effect on local ocean chemistry

- The table below highlights some of the key carbon sink data documented in this report for these coastal habitats. It provides summary data on the comparison of carbon stocks and long-term accumulation of carbon in the coastal marine ecosystems. Comparisons with information on terrestrial carbon sinks are provided in the body of this report.

- The chemistry of some specific marine sediments (for example salt marshes) suggests that whilst such habitats may be of limited geographical extent, the absolute comparative value of the carbon sequestered per unit area may well outweigh the importance of similar processes on land due to lower potential for the emission of other powerful greenhouse gases such as methane.

- Alongside the carbon management potential of these ecosystems, another key finding of this report is the lack of critical data for some habitat

Ecosystem type	**Standing carbon stock (gC m^{-2})**		**Total global area (*10^{12} m^{2})**	**Global carbon stocks (PgC)**		**Longterm rate of carbon accumulation in sediment (gC m^{-2} yr^{-1})**
	Plants	**Soil**		**Plants**	**Soil**	
Tidal Salt Marshes			Unknown (0.22 reported)			210
Mangroves	7990		0.157	1.2		139
Seagrass meadows	184	7000	0.3	0.06	2.1	83
Kelp Forests	120-720	na	0.02-0.4	0.009-0.02	na	na

types. Having comprehensive habitat inventories is critically important and this report highlights the urgent need, alongside recognising the carbon role of such ecosystems, to ensure that such inventories are completed for saltmarsh and kelp forests and then all such inventories are effectively maintained over time.

- These coastal marine ecosystems are also vital for the food security of coastal communities in developing countries, providing nurseries and fishing grounds for artisanal fisheries. Furthermore, they provide natural coastal defences that mitigate erosion and storm action. Therefore, better protection of these ecosystems will not only make carbon sense, but the co-benefits from ecosystem goods and services are clear.
- Significant losses are occurring in the global extent of these critical marine ecosystems due to poor management, climate change (especially rising sea levels), coupled to a lack of policy priority to address current and future threats.
- Certain human impacts – notably nutrient and sediment run-off from land, displacement of mangrove forests by urban development and aquaculture, and over-fishing - are degrading these ecosystems, threatening their sustainability and compromising their capacity to naturally sequester carbon. The good news is that such impacts can be mitigated by effective management regimes.
- Management approaches already exist that could secure the carbon storage potential of these ecosystems, and most governments have commitments to put such measures in place for other reasons. These include biodiversity protection or achieving sustainable development. Agreed management approaches that would be effective include Marine Protected Areas, Marine Spatial Planning, area-based fisheries management approaches, buffer zones to allow inland migration of coastal carbon sinks, regulated coastal development, and ecosystem rehabilitation.
- Greenhouse gas emissions that occur as a result of the management of coastal and marine habitats are not being accounted for in international climate change mechanisms (ie UNFCCC, Kyoto, CDM, etc) or in National Inventory Submissions. Not only does this mean that countries are under-estimating their anthropogenic emissions, but also that the carbon savings from measures to protect and restore coastal and marine habitats will not count towards meeting international and national climate change commitments.

This report provides the essential evidence needed to motivate discussions and initiatives on how such coastal ecosystems should be incorporated into international and national emission reduction strategies, national greenhouse gas inventories and, potentially, carbon revenues schemes. The latter could take the marine equivalent of the Reducing Emissions from Deforestation and Forest Degradation (REDD) scheme on land to safeguard these critical coastal carbon sinks. Don't just think REDD, think coastal too!

The evidence presented here makes clear why moving forward with effective Marine Protected Areas, Marine Spatial Planning and area-based fisheries management techniques is not only a political imperative for biodiversity conservation, food security, and shoreline protection, but also now for helping mitigate climate change.

Outlook on Gazi Bay (Kenya) from Kidogoweni creek, with Ceriops tagal *bearing propagules on the right front side.*
© Steven Bouillon, K.U.Leuven

Introduction

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As the evidence grows about the effects climate change is having on the environment, so too does the interest in and actions to address the underlying causes – regulation of anthropogenic emissions of greenhouse gases into the atmosphere, avoiding deforestation, management and protection of other natural terrestrial carbon sinks, and the development of fiscal measures that place a value on carbon and therefore provide an economic incentive to reduce emissions.

The ocean is the largest carbon sink on Earth but there has been scant attention paid to coastal and marine ecosystems when considering actions to address climate change concerns. Within that context the production of this report was stimulated by an interest in why coastal habitats were not being considered as important carbon sinks on a global scale – the focus other than in some popular books on the topic seems to be predominantly on terrestrial ecosystems, particularly forests, certain soil types and peatlands. This concern was brought into sharp focus in 2007 - 2008 when undertaking the research for a report by Natural England on Carbon Management by Land and Marine Managers (Thompson, 2008). It rapidly became evident that coastal and marine ecosystems are vital global carbon stores but that it was not easy to find the evidence base to substantiate this claim.

A clear robust rationale was required to progress efforts to include coastal carbon issues in broader climate discussions or heighten the need to manage better and protect these ecosystems. Alongside the Natural England work, in 2008 IUCN's World Commission on Protected Areas released their global Plan of Action (Laffoley, 2008). This set out the overall framework and direction for the work of the World Commission in marine environments. Within the framework it includes a strategic activity of bringing together work on Marine Protected Areas with actions to address climate change, food security and human health. The development of this report on coastal carbon management is a result of the Natural England and IUCN activities, and a particular contribution to the global Plan of Action for Marine Protected Areas. With ongoing support from the Lighthouse Foundation, the United Nations Environment Programme (UNEP) has also come on board to collaborate with IUCN and Natural England, further adding weight to this innovative report.

The logic behind this report is to attempt to quantify the greenhouse gas implications of the management of particular coastal ecosystems, being careful to choose those whose management can be influenced by application of existing policy agreements and well established area-based management tools and approaches. Only the management of natural carbon sinks can be included in a countries national inventory of greenhouse gas emissions and sequestration and therefore count towards their climate change mitigation commitments.

It follows that if management of such habitats delivers clear and quantifiable greenhouse gas benefits, and tools exist to secure their best management, then this opens up a new range of possibilities for better valuing them in terms of meeting international climate change objectives. If we want to maximize the potential for natural carbon sequestration, then it is imperative that we draw together the evidence base and protect these valuable coastal marine ecosystems as an additional option to add to our portfolio for mitigating climate change. The challenge, however, is that little concerted attention has previously been applied to this issue, thus hindering the development of national plans that might include recognition and improved protection of coastal carbon sinks.

The focus of this report is therefore on collating and publishing the science of carbon sinks for an initial set of five key coastal ecosystems. These are coastal ecosystems that not only meet the above potential carbon sink and management criteria, but that are already highly valued for their contribution to marine biodiversity and the goods and services that they provide: tidal saltmarshes, mangroves, seagrass meadows, kelp forests and coral reefs.

Through the goods and services they provide, these coastal ecosystems already play a major role in mitigating the effects of climate change on coastal communities, as well as providing them with livelihoods, food and income. Marine, coastal and terrestrial systems are interlinked, and often dependent on each other. For example, these coastal ecosystems act as filters for land-based nutrients and pollution and thus allow extremely precious coral reefs to exist. Some coastal ecosystems (e.g. mangroves) also act as natural defences, protecting vulnerable coastal communities from storm surges and waves, particularly tsunamis. The roots of mangrove and marsh plants stabilize soils and reduce coastal erosion. They also provide coastal communities with food from fisheries, nurseries for important fish stocks, and income through harvesting of commercially valuable resources. Thus there is an excellent basis of existing values to build on when considering their additional potential as carbon sinks.

We believe that this report is the first attempt to bring the in-depth carbon management role of such coastal ecosystems to international attention in one volume. In this report we also attempt to make a comparison with terrestrial carbon sinks. Future work will focus on the marine species dimension, deep sea ecosystems and broader coastal shelf processes. The timing of this report, in the run up to the UNFCCC COP-16 Copenhagen, is also particularly important. The report provides an evidence base on the carbon role of these critical coastal habitats and the contribution that their sustainable management can make to climate change mitigation which we hope policy advisors, decision makers and natural resource managers will use to include them in relevant debates, new management approaches and strategies and plans. We also hope that this report will stimulate further research into these important habitats, as we endeavour to increase our knowledge of which species, ecosystems or regions are most critical for carbon sequestration as well as co-benefits from food security and shoreline protection. In the same way that we are constantly increasing our understanding of the role their terrestrial counterparts play in the carbon cycle, we need to increase our understanding of these coastal carbon sinks too.

We hope that the evidence presented in this report will stimulate greater interest in the fate of these ecosystems, and a greater policy drive for their effective protection and management, using a diverse array of existing tools such as Marine Protected Areas. Unfortunately, as this report documents, these coastal ecosystems are disappearing at an alarming rate. Human activities such as deforestation, agricultural and industrial runoff, unsustainable coastal development, overfishing, oil spills, dredging, filling or drainage that cause sediment-loading, eutrophication and loss of biodiversity have all taken their toll. Now rising sea-levels are placing some of these ecosystems in a 'coastal squeeze', as their ability to expand inland to adapt to the rising water is severely restricted by urban developments and embankments. We hope the new evidence on their important roles as carbon sinks will strengthen the commitment to work already advancing on implementing the World Summit on Sustainable Development goal of building networks of MPAs by 2012.

We hope also that this work will stimulate a debate around the potential for the management, protection and restoration of coastal marine ecosystems to engage with the emerging carbon market. Fortunately, as this report has been developing, world's governments are beginning to realize the importance of addressing this situation and with the Manado Declaration agreed upon at the World Ocean Conference in 2009, they recognized that "healthy and productive coastal ecosystems… have a growing role in mitigating the effects of climate change on coastal communities and economies in the near term" and stressed the need "for national strategies for sustainable management of coastal and marine ecosystems, in particular mangrove, wetland, seagrass, estuary and coral reefs, as protective and productive buffer zones that deliver valuable ecosystem goods and services that have significant potential for addressing the adverse effects of climate change." In addressing the needs of these ecosystems additional costs may be incurred, but what are the hidden costs of not achieving carbon reduction goals?

In the following sections we set out the views of leading scientists on the carbon management potential of coastal ecosystems. The latest scientific information and perspectives on the role of these habitats have been used to develop each section, and the resultant chapters have all been subject to independent peer review. The report sequentially covers seagrass, mangroves, tidal salt marshes and kelp forests as carbon sinks, and then uses a different format to set out the ocean chemistry on the role of coral reefs in the carbon cycle (as research for this report shows them, perhaps counter intuitively in some peoples' minds, to be slight carbon sources and not sinks).

We also include a discussion of management requirements and interventions to maintain these coastal ecosystems as efficient carbon sinks. A further section focuses on a comparison of the carbon management role of these selected coastal marine ecosystems and how this relates to the existing body of knowledge on terrestrial carbon sinks. Finally a closing chapter examines the next steps to bring action, as well as improved recognition, to the role of these habitats as coastal marine carbon sinks.

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Shallow Thalassodendron ciliatum *bed mixed with corals, Zanzibar Tanzania. Photo: Mats Björk*

Tidal Salt Marshes

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Fast facts

- Intertidal ecosystems dominated by vascular plants.
- Occur on sheltered marine and estuarine coastlines from the sub-arctic to the tropics, but most extensive in temperate climates.
- Their soils store 210 g C $m^{-2}yr^{-1}$. This is a substantial rate and the carbon stored in tidal salt marsh soils of the USA comprises 1-2% of its total carbon sink.
- Each molecule of CO_2 sequestered in soils of tidal salt marshes and their tropical equivalents, mangrove swamps, probably has greater value than that stored in any other natural ecosystem due to the lack of production of other greenhouse gases. In contrast to freshwater wetland soils, marine wetlands produce little methane gas, which is a more potent greenhouse gas than CO_2. The presence of sulphates in salt marsh soils reduces the activity of microbes that produce methane.
- Extensive marsh areas have been lost from dredging, filling, draining, construction of roads and are now threatened by sea level rise.
- Restoration of tidal salt marshes can increase the world's natural carbon sinks. Returning the tides to drained agricultural marsh can also significantly increase this carbon sink.
- Sustainability of marshes with accelerating sea level rise requires that they be allowed to migrate inland. Development immediately inland to marshes should be regulated through establishment of buffer zones. Buffer zones also help to reduce nutrient enrichment of salt marshes, another threat to this carbon sink.

Definition and global occurrence

Tidal salt marshes are intertidal ecosystems vegetated by a variety of primary producers such as macroalgae, diatoms and cyanobacteria, but physically dominated by vascular plants. Vascular plants are absent from the tidal flats often found adjacent to the seaward edge of tidal salt marshes. In contrast to eelgrass communities which may be found on the edge of the lowermost intertidal zone, survival of the dominant vascular plants is dependent upon exposure to the atmosphere. During photosynthesis the marsh's vascular plants uptake carbon dioxide from the atmosphere, in contrast to eelgrass which uptakes carbon dioxide dissolved in seawater.

Chapman (1977) described the dominant plant forms of the marsh and how they vary geographically. Perennial grasses such as *Spartina alterniflora* and *Spartina patens* are dominant along much of the Atlantic coast of North and South America. In some other regions perennial broad-leaved herbaceous plants dominate, such as *Atriplex portuloides* along portions of Europe's coast. Perennial succulents such as the related *Salicornia, Sarcocornia* or *Arthrocnemum* species that grow to shrub size tend to dominate coastlines of Mediterranean climates where, dry, hot summers cause soils to develop hypersaline conditions.

Tidal salt marshes occur on sheltered marine and estuarine coastlines in a range of climatic conditions, from sub arctic to tropical, but are most extensive in temperature climates. Although it is often reported that mangrove trees replace salt marsh vegetation on tropical coasts salt marshes may exist above the higher elevation of the swamp.

Species		Below	Above	Region	Reference
		---g m^{-2} yr^{-1}---			
Chenopodieaceae					
Arthrocnemum	*macrostachyum*	1260	683	Po Delta	Ibañez et al. 2000
Arthrocnemum	*macrostachyum*	50	190	Ebre Delta	Ibañez et al. 2000
Arthrocnemum	*macrostachyum*	340	840	Ebre Delta	Ibañez et al. 2000
Salicornia	*fructosia*	950	580	Ebre Delta	Ibañez et al. 2000
Atriplex	*portulacoides*	1601	598	Guadiana River	Neves et al. 2007
Plantaginaceae					
Plantago	*maritima*	648	296	Bay of Fundy	Connor 1995
Poaceae					
Spartina	*patens*	1113	500	Bay of Fundy	Connor 1995
Spartina	*patens*	3300	785	Delaware Bay	Roman & Daiber 1984
Spartina	*alterniflora*	1575	718	Bay of Fundy	Connor 1995
Spartina	*alterniflora*	6500	1487	Delaware Bay	Roman & Daiber 1984

Table 1. Rates of above and below ground production of selected tidal salt marsh species from three different plant families in North America and Europe demonstrate the importance of below ground production with varied plant forms.

Value – goods and services provided

Tidal salt marshes provide valuable habitat for plants, birds and fish, many of which serve as food resources. Communities that provide services and supplies to recreational waterfowl hunters receive indirect economic benefits. In some regions marsh plants are harvested for subsistence consumption or commercial sale, like the glassworts of Europe. Native vegetation of salt marshes is also harvested as fodder or simply used as natural pastures. The salt tolerance of tidal salt marsh vegetation makes them potential candidates as alternative crops and forage in salinized soils (Gallagher 1985), which are likely to become more problematic as climate warms and sea level rises.

Marshes support direct, non-consumptive uses, as well. Their ponds and adjacent tidal flats attract wading birds and large flocks of migratory birds that provide recreational opportunities for bird watching. Marshes also provide opportunities to educate the public in natural history and ecology. Indirect benefits from marshes may be just as valuable. These indirect benefits include storm protection (Koch et al. 2009) and "filtering" of nutrients. By uptaking nutrients from ground water the salt marsh ecosystem helps to reduce nutrient enrichment that would endanger sea grass beds. However, gas flux studies have shown that enrichment of wetlands with nitrogen may enhance the release of nitrous oxide, a greenhouse gas with 298 times the global warming potential of carbon dioxide (Forster et al. 2007). Thus, the service provided by nutrient regulation may result in an increase in greenhouse gas emissions and loss in marsh sustainability as described below.

Productivity

Vascular plant production varies considerably (Figure 1). In North America above ground production ranges from 60 g C m^{-2} yr^{-1} in northern Canada and Alaska to averages as high as 812 g C m^{-2} yr^{-1} in the north central Gulf of Mexico (Mendelssohn and Morris 2000). Although estimates of productivity vary with methods used for calculation, some trends are evident. For instance, comparison of *Spartina alterniflora* marshes in North America reveals decreasing production with increasing latitude (Turner 1976). Most productivity studies have been limited to biomass produced by vascular plants aboveground, missing two critical components: below-ground vascular plant production and non-vascular plant production.

The microflora living on the marsh surface (cyanobacteria and eurkaryotic algae such as diatoms)

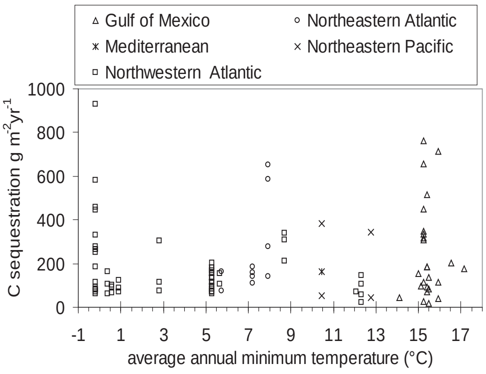

Figure 1. Rates of global carbon sequestration in the world's tidal salt marshes. Adapted from Chmura et al. (2003).

are an important source of marsh primary production. Sullivan and Currin (2000) compared the annual production of benthic microflora to vascular plants in salt marshes of the three U.S. coastlines. Microfloral production ranged from 8% of vascular plant production in Texas to 140% in a California salt marsh. The biomass of benthic microflora may comprise a significant portion of the diet of the invertebrate fauna (e.g., amphipods, gastropods, polychaetes) that form the base of the marsh food chain.

Another often overlooked portion of primary productivity is the belowground production of vascular plants. In many marshes more production is held below ground as roots and rhizomes (table 1) and salt marshes have significantly greater belowground production than their freshwater counterparts (Murphy 2009). This soil biomass is much less available for export to detrital food chains and stored in soil until organic matter is broken down through decomposition. Research has addressed how salinity and soil saturation affect aboveground growth, but we know less about their impact on belowground production – the more critical contribution to carbon storage. Hypersaline soils can limit vascular plant production and result in soil subsidence. However, the dominant plants of the intertidal zone can tolerate soil pore water salinity levels equal to sea water, but the presence of saline soil water still presents a physiological stress. This causes a greater nitrogen demand, thus the need for greater root production to obtain the limiting nutrient.

The value of tidal salt marshes in support of secondary production, particularly coastal fisheries is widely noted (e.g., Boesch and Turner 1984 and Deegan et al. 2000) and marsh area has been correlated to rates of fish and shrimp production in coastal waters. Marsh creeks, ponds and edges provide refuge to juvenile fish, many which feed on soil fauna when they access higher marsh surfaces during flooding tides (Laffaille et al. 2000). Exported primary production becomes part of a detrital food chain where the nutrient value of dead vascular plant tissue is enhanced by microbes.

Role as a carbon sink

A review of carbon stored in tidal salt marshes estimated that, globally, at least 430 Tg of carbon is stored in the upper 50 cm of tidal salt marsh soils (Chmura et al. 2003). The actual size of the sink is likely to be substantially greater, for two reasons. First, soils of many salt marshes obtain depths of meters and amounts of salt marsh carbon do not significantly decline with depth (Connor et al. 2001). Second, the aerial extent of salt marshes is not well documented for many regions of the world.

In considering feedbacks to climate the rate of carbon accumulation and storage is critical to know. Chmura et al. (2003) calculated that, on average, their soils store 210 g C $m^{-2}yr^{-1}$ or 770 g of carbon dioxide, one of the most important greenhouse gases. This is a substantial rate and the carbon stored in tidal salt marsh soils of the U.S. (which has a comprehensive inventory of salt marsh area) comprises 1-2% of the total yearly carbon sink estimated for the coterminous U.S.

When one considers feedbacks to climate, each molecule of carbon dioxide sequestered in soils of tidal salt marshes and their tropical equivalents, mangrove swamps, probably has greater value than that stored in any other natural ecosystem, due to the lack of production of other greenhouse gases. In contrast to freshwater wetland soils (Bridgham et al. 2006), marine wetlands produce little methane gas, which is 25 times more potent as a greenhouse gas (based upon a 100-yr time horizon) than carbon dioxide (Forster et al. 2007). The presence of sulphates in salt marsh soils reduces the activity of microbes that produce methane. In well-drained parts of salt marshes methane produced in lower depths is likely to oxidized as it moves through surface layers.

Tidal floodwaters contribute inorganic sediments to intertidal soils, but more importantly, they saturate the soil and reduce the potential for aerobic decomposition. Anaerobic decomposition is much less efficient, enabling accumulation of organic matter in the soil, and the effective carbon sink.

Another advantage of the soil carbon sink in tidal salt marshes and mangroves is that, unlike dry terrestrial systems, the content of soil carbon does not reach equilibrium. In dry terrestrial ecosystems soil surfaces that adsorb organic carbon eventually become saturated and carbon inputs become balanced by decomposition and release of carbon dioxide through respiration of decomposers. For instance, improved management of agricultural soils can increase rates of carbon storage, but gains may occur for only 50 year before equilibrium in carbon inputs and outputs occur (Canadell et al. 2007).

If there is adequate accumulation of organic matter and inorganic sediments in a marsh soil it will increase

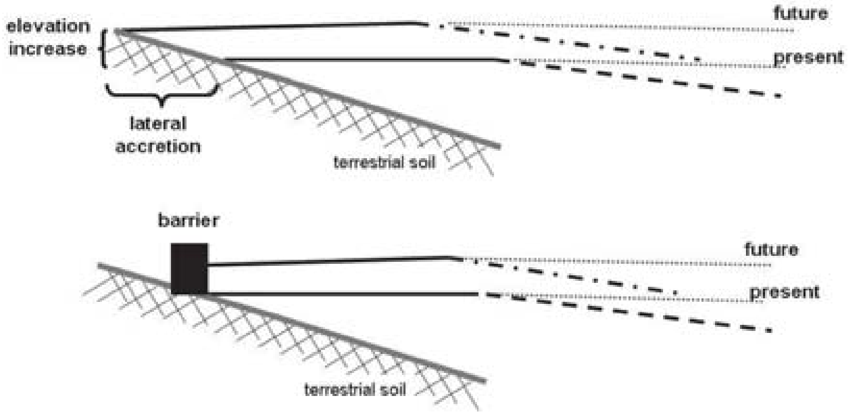

Figure 2. Two scenarios of tidal marsh response to rising sea level (dotted line). Elevation of the marsh surface (solid black line) increases as increased tidal flooding allows organic matter and mineral sediments to accumulate. Increasing elevation is accompanied by lateral accretion over inland terrestrial soils, as pictured in the upper diagram. Constructed barriers (e.g. wall, dykes) prevent lateral accretion on the inland edge of the marsh. At lower elevations (dashed-dotted lines), marsh vegetation does not survive increased submergence, resulting in loss of marsh on the seaward edge.

in elevation, tracking changes in sea level (figure 2). Paleoenvironmental studies of marsh soils (e.g., Shaw and Ceman, 1999) have documented both increase in surface elevation and lateral accretion of marsh soils as marsh plants colonize mudflats to the seaward side and adjacent terrestrial or wetlands environments to the landward side. In many estuaries the slow rate of sea level rise over the last 5,000 years has allowed development of carbon-rich deposits as much as 6 m thick.

Although the potential of wetland soils as a carbon sink has long been recognized, many studies had overlooked tidal salt marsh and mangrove swamp soils, perhaps due to the intensive research focus on carbon export and assumption that carbon concentration reflected carbon density. Conventionally, soil carbon content has been reported as the percent of the entire soil mass, but assessment of carbon storage potential requires calculation of mass of carbon per unit volume. In a soil that accretes vertically (i.e., wetland soils) the rate of accumulation of soil volume is also required. On many coasts tidal floodwaters contribute inorganic sediment to tidal wetland soils, diluting organic matter with material which is three orders of magnitude heavier than organic matter. Thus, a tidal salt marsh soil that contains 5% carbon but has a bulk density of 0.53 g cm^{-3} can hold the same amount of carbon as a bog soil that contains 46% C, but has a bulk density 0.06 g cm^{-3}.

Threats to ecosystem

On nearly every continent extensive areas of marsh already have been lost. Throughout history, marshes have been lost to dredging, filling, and drainage. In Europe, significant human impacts began thousands of years ago (Davy et al. 2009) and extensive marsh loss followed European colonization first of the Americas (e.g., Costa et al. 2009) and then of Australia and New Zealand (Thomsen et al. 2009). With the long history of intensive land use in China we can assume that there has been extensive loss of tidal salt marsh, and the report by Yang and Chen (1995) that the approximately 1,750,000 acres of land reclaimed from Chinese salt marshes exceeds the area of China's marshes today is probably quite conservative.

Tidal salt marshes are located on prime coastal real estate and in the last century extensive areas were lost to development of ports and residential complexes (e.g., Costa et al. 2009). Construction of roads and causeways through marshes and coastal bays has disrupted tidal flooding and marsh hydrology. Proposals to harness tidal power are one of the newest threats to marshes. Some schemes are based upon construction of barrages that alter tidal flooding patterns. These activities continue to threaten marshes, and in some countries marsh loss is permitted if equal or greater areas of marsh are created or restored elsewhere.

Marshes that remain face a suite of multiple stressors that include invasions of exotic species, climate change, and pollution with excessive nutrients, pesticides, herbicides, heavy metals and organic compounds released into coastal waters. Although these may disrupt components of the ecosystem, the potential for carbon storage depends on sustainability of marsh accretion, thus maintenance of vegetation cover.

Disruption of coastal food webs can have unanticipated cascade effects that result in increased populations of marsh herbivores whose grazing results in extensive denudation of marsh vegetation (Silliman et al. 2005; Holdredge et al. 2008). If vegetation cover does not return, marshes are subject to subsidence or erosion, thus cessation of soil carbon storage.

Worldwide, marshes now are threatened by increased rates of sea level rise associated mainly with climate change. Modelling studies show that rates of carbon accumulation will increase as vertical marsh accretion responds to rising sea levels – until sea-level rise reaches a critical rate that drowns the marsh vegetation and halts carbon accumulation (Mudd et al. 2009). The critical rate varies with inorganic sediment supply and hydrological conditions – both susceptible to anthropogenic modifications. Sustainability of tidal salt marshes is dependent upon their ability to vertically accrete through accumulation of organic matter and sediments. Anthropogenic activities that alter marsh hydrology, increase soil saturation, or reduce the supply of inorganic sediments are likely to reduce plant production and the potential for vertical accretion of marsh soil. Increased hydroperiods are expected within marshes around the world, lowering their threshold to withstand the added stresses from anthropogenic impacts. Examples of this problem already exist on coasts where subsurface subsidence results in exceptional levels of relative sea level rise, such as the Mississippi Delta in Louisiana (e.g., Turner 1997, Day et al. 2000). There, oil exploration led to extensive dredging of canals and deposition of spoil banks along their sides that altered marsh hydrology. Impounding of surface water exacerbated anoxic soil conditions causing physiological stress to plants, reducing the production of soil organic matter and marsh vertical accretion rates. Marsh surfaces degraded into ponds. The additional marsh edges created made marshes more susceptible to erosion during storms.

Increasing sea levels have already placed marshes on developed coastlines in what has been termed a "coastal squeeze." On these coasts the ability of marshes to expand inland is severely restricted by urban development or embankments associated with "reclamation" (fig 2). Walls, dikes, and paved surfaces present physical barriers to marsh expansion inland, and the seaward edge of salt marshes is expected to retreat. This situation will ultimately result in loss of tidal salt marshes. Increased rates of sea level rise will increase the duration of tidal flooding, limiting vegetation production at the lower elevations along the seaward edge of the marsh. If landward lateral accretion is not possible, these marshes will eventually disappear.

Management recommendations to maintain and enhance carbon storage potential

In many regions tidal salt marshes are now protected from direct impacts such as dredging and filling. However, sustainability of protected marshes also requires that they be protected from indirect impacts. Programs designed to protect marshes should encompass activities in the estuarine watershed that affect discharges of water and sediments. Loss of suspended sediments will decrease the ability of a marsh to maintain elevations with rising sea level. In arid regions, in particular, reduction of freshwater inflow can result in hypersaline conditions and loss of vegetation critical to marsh accretion and carbon storage. The impacts of nutrient-laden runoff from fertilized watersheds (through agriculture or even suburban landscapes) to many coastal ecosystems are widely recognized, but the negative impact of nutrient enrichment on marsh sustainability has only recently been recognized.

Fertilization experiments show that the two dominant grasses of western Atlantic salt marshes, *Spartina alterniflora* (Darby and Turner, 2008) and *Spartina patens* (Chmura, unpublished data) increase their above ground production, but decrease their below ground production (essential for vertical accretion) in response to nutrient additions. Turner et al. (2009) determined that long-term fertilization of a Massachusetts marsh resulted in a significant loss of marsh elevation, equivalent to about half the average rate of global sea level rise. Although tidal salt marshes are often recognized for their value as "nutrient filters," reducing the threat of eutrophication of coastal waters; provision of this service is made at the expense of all others performed by a salt marsh. "Filtering of nutrients" by tidal salt marshes must not be seen as an acceptable compromise to better management of non-point nutrient sources from watersheds or urban sewage.

Terrestrial buffer zones can help to reduce nutrient enrichment of salt marshes, a threat to the marsh carbon sink and the ecosystem's sustainability. Buffers distance marshes from sites where nutrients are applied and take up nutrients in vegetation and soils, thus reducing the level reaching the marsh. Terrestrial buffers can help ensure sustainability of marshes with accelerating sea level rise, allowing them to migrate inland. Development immediately inland to marshes should be discouraged and, if possible, regulated through establishment of buffer zones.

Restoration of tidal salt marshes is an excellent way to increase the world's natural carbon sinks. Returning the tides to drained agricultural marsh can make a significant increase in the salt marsh carbon sink. The U.K.'s managed realignment program, to shift embankments inland and restore flooding of agricultural marshes, is a progressive form of coastal management that not only deals with the threat of sea level rise, but promises to enhance carbon sequestration as tidal salt marshes recover. Such policies should be considered in other regions. For example, Connor et al. (2001) estimated that if all of Bay of Fundy marshes "reclaimed" for agriculture could be restored, the rate of carbon dioxide sequestered each year would be equivalent to 4-6% of Canada's targeted reduction of 1990-level emissions under the Kyoto Protocol.

Marsh near the mouth of the Bay of Fundy (at Dipper Harbour, New Brunswick, Canada). Approximately 3 m of carbon- rich soil has accumulated here over the last 2500 years. Paleoecological studies show that upper edges of the marsh prograded over adjacent terrestrial communities as sea level rose. As sea level rises more rapidly with global warming, marshes must migrate inland to survive, but urban development will prevent this on many coasts. © Olivia Yu

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Global distribution of Mangroves

UNEP WCMC

Mangroves

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Fast facts

- Salt-tolerant, mainly arboreal, flowering plants growing in the intertidal zone of tropical and sub-tropical shores.
- Global area of 157,000 km^2 to 160,000 km^2.
- Global carbon burial of approximately 18.4 Tg C yr^{-1} .
- Mangrove forests are estimated to have occupied 75% of the tropical coasts worldwide, but anthropogenic pressures have reduced the global range of these forests to less than 50% of the original total cover.
- These losses are largely due to over-harvesting for timber and fuel-wood production, reclamation for aquaculture and saltpond construction, mining, oil spills, pollution and damming of rivers that alter water salinity levels.
- Rehabilitation/restoration or plantation of mangrove forests are not only to be encouraged based on ecological or socio-economical considerations, but also have the potential of providing an efficient sink of CO_2.

Definition and global occurrence

Mangrove forests are a dominant feature of many tropical and subtropical coastlines, but are disappearing at an alarming rate. The main causes for the rapid destruction and clearing of mangrove forests include urbanization, population growth, water diversion, aquaculture and salt-pond construction (e.g. Farnsworth & Ellison 1997). On a global scale, mangrove plants are found throughout the tropical and subtropical regions of the world, and two species of *Avicennia* have penetrated into the warm temperate areas of both hemispheres. The global distribution of mangroves generally matches the winter 20°C isotherm. Mangroves are trees, shrubs, palms or ground ferns which normally grow above mean sea level in the intertidal zone of marine, coastal, or estuarine environments. Thus, mangrove plants do not form a phylogenetically related group of species but are rather species from very diverse plant groups sharing common morphological and physiological adaptations to life in the intertidal zone, which have evolved independently through convergence rather than common descent. The most recent global data compilation suggests a current global areal extent of about 152,000 km² (FAO 2007), with Indonesia and Australia together hosting about 30% of this area.

Mangrove goods & services

Besides the role mangroves play in the carbon cycle, mangrove ecosystems have a wide range of ecological and socio-economical functions.

For many communities living within or near to mangrove forests in developing countries, mangroves constitute a vital source of income and resources, providing a range of natural products such as wood (for firewood, construction, fodder, etc), medicines, and as fishing grounds. They are known to provide essential support for a wide range of intertidal and aquatic fauna, and act as nursery habitats for many commercial (and non-commercial) aquatic species such as crabs, prawns and fish (Nagelkerken et al., 2008). Whether this link is due to the provision of habitat, protection or predation, or via a direct trophic link is still under debate, but the value of mangroves in supporting coastal fisheries is unquestionable (see e.g., Mumby et al. 2004).

Furthermore, the presence of mangroves has been demonstrated to provide an efficient buffer for coastal protection: their complex structure attenuates wave action, causing reduction of flow and sedimentation of suspended material. This topic has received a great deal of attention following the 2004 Tsunami which hit SE Asia (e.g., Dahdouh-Guebas et al., 2005; Alongi, 2008; Yanagisawa et al., 2009; Das & Vincent, 2009), although demonstrating the causal link between mangroves and coastal protection is not always straightforward (e.g., see Vermaat & Thampanya 2005). This function of mangrove forests is also likely to act as an important buffer against sea level rise.

Finally, mangrove ecosystems have been shown to be effective as nutrient traps and 'reactors', thereby mitigating or decreasing coastal pollution. The feasibility of using (constructed rather than natural) mangrove wetlands for sewage or shrimp pond effluents has recently been demonstrated (e.g., Boonsong et al., 2003; Wu et al. 2008) and could offer a low-cost, feasible option for wastewater treatment in tropical coastal settings.

Productivity of mangroves

Mangrove forests are considered as highly productive ecosystems. Most data on their productivity are in the form of litter fall estimates, obtained by regularly collecting all litter in litter traps suspended below the canopy. Unfortunately, much less information is available on their biomass production in terms of wood and belowground production. When estimating overall global net primary production for mangroves, we therefore need to rely on relationships between litter fall and wood or belowground production to upscale the data on litter fall. Using a global area of mangroves of 160,000 km², the net primary production was recently estimated at 218 ± 72 Tg C yr^{-1} (Bouillon et al. 2008), with root production responsible for ~38% of this productivity, and litter fall and wood production both ~31%. There is a general latitudinal gradient in the productivity of mangroves, being significantly higher in the equatorial zone compared to higher-latitude forests – a pattern recognized for a number of decades (Twilley et al. 1992, Saenger & Snedaker 1993) and confirmed by new data compilations (Bouillon et al. 2008).

Carbon sinks in mangrove systems

Biomass produced by mangrove forests can ultimately have a number of different destinations (i) part of the biomass produced can be consumed by fauna, either directly or after export to the aquatic system, (ii) carbon can be incorporated into the sediment, where it is stored for longer periods of time, (iii) carbon can be remineralized and either emitted back to the atmosphere as CO_2, or exported as dissolved inorganic carbon (DIC), (iv) carbon can be exported to adjacent ecosystems in organic form (dissolved or particulate) where it can either be deposited in sediments, mineralized, or used as a food source by faunal communities.

In the context of CO_2 sequestration, the relevant carbon (C) sinks to consider are:

- the burial of mangrove C in sediments – locally or in adjacent systems,
- net growth of forest biomass during development, e.g. after (re)plantations.

The first process represents a long-term C sink, while the second should be considered relevant only on the shorter (decennial) term.

Three different global estimates for **carbon burial** within mangrove systems all converge to a value equivalent to ~18.4 Tg C yr^{-1} (when applying a global area of 160,000 km²). These estimates are derived either from sedimentation estimates combined with typical organic carbon concentrations in mangroves (Chmura et al. 2003), or from mass-balance considerations – despite a number of uncertainties in these estimates there are insufficient data available to better constrain these values.

The amount of carbon stored within sediments of individual mangrove ecosystems varies widely, from less than 0.5% (on a dry weight basis) to <40%, with a global median value of 2.2 % (Kristensen et al. 2008 – see Figure 1) – extrapolations to carbon stocks on an areal basis are difficult to make due to varying depths of sediments and the paucity of concurrent data on sediment densities (i.e. volumetric weight of the sediment). Furthermore, carbon accumulating is not necessarily all derived from the local production by mangroves – organic matter can be brought in during high tide and can originate from rivers, or from adjacent coastal environments. Both the quantity and origin of carbon in mangrove sediments appear to be determined to a large extent by the degree of 'openness' of mangroves in relation to adjacent aquatic systems: mangroves with low tidal amplitude or high on the shoreline have little opportunity to export organic matter produced, and also little other material is brought in: such systems or sites typically have high carbon contents, and the organic matter accumulating is locally produced. In contrast, in low intertidal sites or systems with high tidal amplitude, a larger fraction of the organic matter produced can be washed away, and sediment with associated organic matter from adjacent systems is imported during high tide and is deposited within the system (Twilley 1995). These patterns are observed not only in mangroves (Bouillon et al. 2003) but also in salt marshes (Middelburg et al. 1997).

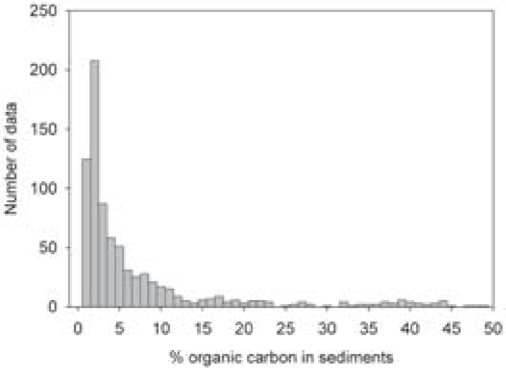


Figure 1: Compilation of literature data on sediment organic carbon concentrations in mangrove sediments (from Kristensen et al. 2008).

Irrespective of the origin of carbon in mangrove sediments, the presence of mangroves clearly has an impact on sediment carbon storage, by (i) direct inputs of mangrove production to the sediment pool, and (ii) by increasing sedimentation rates (e.g., Perry & Berkeley 2009). Conversely, clearing of mangroves can rapidly result in significantly reduced C stores in sediments (e.g., from up to ~50% over an 8 yr period in the study by Granek & Ruttenberg 2008), indicating that the carbon pool lost through deforestation substantially exceeds that of simple removal of standing biomass.

An overview of current quantitative estimates of carbon flow in mangrove systems is presented in Table 1.

Two important aspects emerge: (i) carbon burial in mangrove sediments represents a relatively small

Net primary production	**218 ± 72**
Litter fall	68 ± 20
Wood production	67 ± 40
Root production	82 ± 57
Fate of mangrove production	
CO_2 efflux	42 ± 31
Export as POC and DOC	45 ± 31
Burial	18.4
Unaccounted	**112 ± 85**

Table 1: Overview of current global estimates of net primary production and carbon sinks in mangrove systems (from Bouillon et al. 2008). All rates reported are in Tg C yr^{-1}.

fraction (<10%) of the overall net primary production, and (ii) current literature estimates of CO_2 efflux from sediments and water, export as organic carbon and burial in sediments together only explain <50% of the primary production estimate. This large discrepancy may in part be solved by a large and previously unaccounted flux of dissolved inorganic carbon towards adjacent systems (see Bouillon et al. 2008).

Woody debris and carbon accumulation in mangrove forests

Mangrove wetlands support less woody debris than upland forests (Allen et al. 2000, Krauss et al. 2005). Hydrological conditions of mangrove wetlands, which include a diversity of tide, precipitation, and river-flow regimes, can complicate direct comparisons with upland forests. Polit and Brown (1996) showed that lowered stocks of woody debris could be partially explained by the higher decomposition rates of woody debris in wetlands. Also, decay of fallen mangrove wood may be quick at first, relative to most temperate systems, due in part to consistently higher temperatures, a prolonged wet season, and a combined terrestrial and marine fungal community in mangroves (e.g., Kathiresan & Bingham 2001).

Woody debris values in mangrove forest after major disturbances (i.e., massive mortalities due to changes in hydrology, hurricanes) are scarce, making it difficult to determine their role in carbon storage in the long term. However, some studies indicate the potential role of wood components in nutrient cycling and carbon flux. For example, Rivera-Monroy et al. estimated a range of 16.5-22.3 Mg ha^{-1} of woody debris in a mangrove forest affected by hypersalinity conditions in a deltaic environmental setting in the Caribbean Sea (Cienaga Grande de Santa Marta, Twilley et al. 1998, Rivera-Monroy et al. 2006). As result of increasing salinity of up to 90 ppt, 271 km^2 of mangrove area were lost in a period of 40 years (Simard et al. 2008). A current estimate of live above ground biomass for this forest (using radar interferometry and Lidar data) ranges from 1.2 to 1.7 (±0.1) Tg over the total area, whereas estimated dead biomass was 1.6 Tg, which represent 0.72 Tg of carbon (assuming a 48% carbon content) input for decomposition and export to adjacent ecosystems. This carbon value is a conservative estimate since no information of belowground biomass (coarse roots) is available for this site and in mangrove forests overall (Bouillon et al. 2008).

Krauss et al. (2005) estimated woody debris in subtropical mangrove forest 9-10 yr after the impact of hurricane Andrew in South Florida. The total volume of woody debris for all sites sampled in this study was estimated at 67 m^3/ha and varied from 13 to 181 m^3/ha depending upon differences in forest height, proximity to the storm, and maximum estimated wind velocities. Large volumes of woody debris were found in the eye wall region of the hurricane, with a volume of 132 m^3/ha and a projected woody debris biomass of approximately 36 Mg ha^{-1}; this value is lower that the 59 Mg ha^{-1} dead biomass estimated in the CGSM, Colombia (Simard et al 2008). Smith et al. (1994) in a large spatial survey study immediate to hurricane Andrew, estimated a total woody debris of up 280 Mg ha^{-1} (135 Mg carbon) including 0.6 and 0.18 Mg of nitrogen and phosphorous.

Rehabilitation and Restoration: biomass production in planted/replanted mangrove forests

As result of the extensive loss of mangrove area and the recognized ecological and economic values of mangrove-dominated ecosystems, there has been an increasing effort to rehabilitate and restore disturbed forests. Unfortunately, the success has frequently been limited due to the lack of a conceptual framework guiding such efforts, particularly given the absence of clear objectives and performance measures to gauge the success of such management strategies (Field 1999, Kairo et al. 2001, Twilley & Rivera-Monroy 2005, Samson & Rollon 2008). Understanding if nutrient and carbon cycling could be rehabilitated in perturbed mangrove forests on a long term basis requires a clear definition of terms. Field (1999) proposed that rehabilitation of an ecosystem is the act of *"partially or, more rarely, fully replacing structural or functional characteristics of an ecosystem that have been diminished or lost, or the substitution of alternative qualities or characteristics than those originally present with proviso that they have more social, economic or ecological value than existed in the disturbed or degraded state"*. In contrast, restoration of an ecosystem *is "the act of bringing an ecosystem back into, as nearly as possible, its original condition"*. In this conceptual framework, restoration is seen as a special case of rehabilitation. Field (1999) stressed *"land use managers are concerned primarily with rehabilitation and are not much concerned with ecological restoration. This is because they require the flexibility to respond to immediate pressures and are wary of being obsessed with recapturing the past"*. Because this definition has not been clearly included in mangrove management plans, it is not surprising that despite the recognized ecological role of mangrove forest there are no long-term studies assessing whether the functional properties (including carbon sequestration and primary productivity) have been restored through management in regions where restoration/rehabilitation projects have been implemented (e.g., Twilley et al. 1998, Samson & Rollon 2008). Recent reviews indicate that newly created mangrove ecosystems may or may not resemble the structure and function of undisturbed mangrove ecosystems and that objectives should be clearly established before any major small or landscape level rehabilitation is implemented (Kairo et al. 2001, Lewis 2005, Twilley & Rivera-Monroy 2005).

To our knowledge, there is no published information describing projects specifically aiming to enhance carbon sequestration through restoration or rehabilitation. However, a good indicator of potential magnitude of this sink is information reported for mangrove plantations or sites undergoing rehabilitation. Aboveground biomass estimates in replanted mangroves stand have varied from 5.1 Mg ha^{-1} in a 80 year plantation (Putz & Chan 1986) to 12 Mg ha^{-1} in a 12 year-old stand (Kairo et al. 2008), with part of the variation attributed to the age of plantations, management systems, species and climatic conditions (Bosire et al. 2008). Species variation in root biomass allocation was observed in a 12-year old replanted mangroves where *S. alba* allocated higher biomass to the root components (75.5 ± 2.0 Mg ha^{-1}) followed by *A. marina* (43.7 ± 1.7 Mg ha^{-1}) and *R. mucronata* 24.9 ± 11.4 Mg ha^{-1} (Tamooh et al. 2008). From the few data available, it would appear that productivity of replanted sites is in the same range as expected for natural forests, e.g. litter production in 7-year old *R. mucronata* plantation in Vietnam ranged between 7.1 and 10.4 Mg DW ha^{-1} yr^{-1}, and 8.9 to 14.2 Mg DW ha^{-1} yr^{-1} for *R. apiculata* monocultures (Nga et al. 2005). Overall, young mangrove forest can store from 2.4 to 5.8 Mg C ha^{-1} in aboveground biomass while C in root biomass ranges from 21 to 36 Mg C ha^{-1}. These values are first- order approximations based on average carbon content of plant material (48%). The study of McKee & Faulkner (2000) also suggested that productivity of restored mangrove stands (both above- and belowground) were similar to those of natural stands, and any variability more likely to be related to environmental conditions rather than to the natural or replanted status. Thus, site selection and a critical assessment of environmental conditions appears a critical factor to ensure that the natural productivity of replanted mangrove stands is ensured.

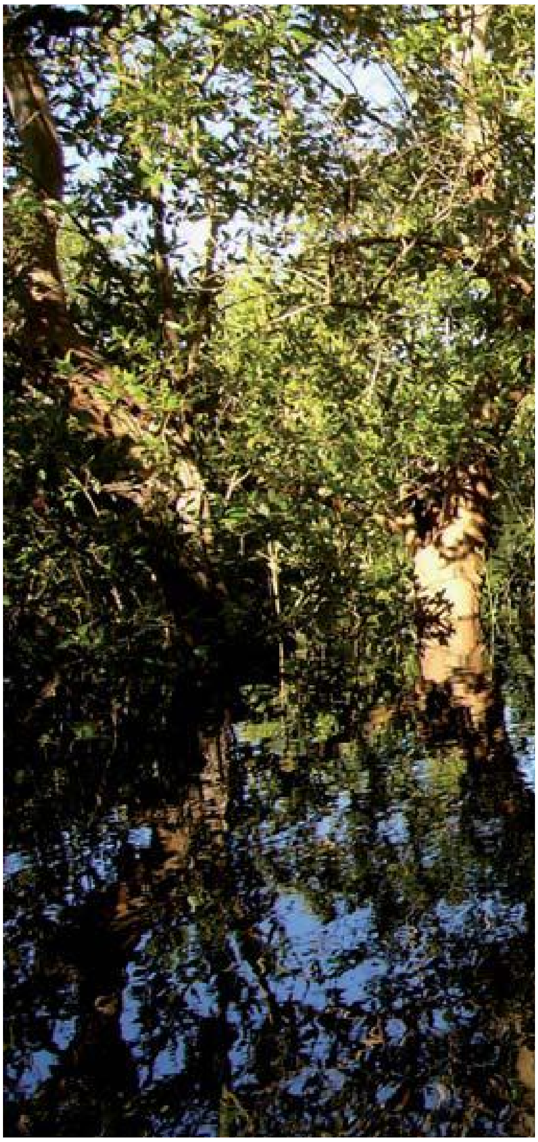
Threats to mangrove ecosystems

Mangrove forests are estimated to have occupied 75% of the tropical coasts worldwide (Chapman 1976), but anthropogenic pressures have reduced the global range of these forests to less than 50% of the original total cover (Spalding et al.1997, Valiela et al. 2001). These losses have largely been attributed to anthropogenic pressures such as over-harvesting for timber and fuel-wood production, reclamation for aquaculture and saltpond construction (Spalding et al., 1997, Farnsworth & Ellison (1997), mining, pollution and damming of rivers that alter water salinity levels. Oil spills have impacted mangroves dramatically in the Caribbean (Ellison & Farnsworth 1996), but little documentation exists for other parts of the world (Burns et al. 1994). Similarly, information (if any) about carbon losses associated to clear-falling are difficult to obtain since this activity is illegal in most countries; actual records of total biomass extracted to use mangrove area for other purposes (e.g., roads, urban development) is also rare making it difficult to determine this component in global estimates of carbon sequestration. Field (1999) underlined how, historically, information about mangrove use and rehabilitation projects usually remains in the grey literature in government agencies where it is difficult to obtain it for evaluation of management strategies and develop research priorities. Perhaps the major cause of mangrove decline has been conversion of the area to aquaculture. In the Indo-Western Pacific region alone, 1.2 million hectares of mangroves had been converted to aquaculture ponds by 1991 (Primavera 1995). These numbers, given their large magnitude, make it evident that conservation, rehabilitation and replantation efforts are critically needed to ensure the sustainability of these unique habitats for the future (Duke et al. 2008). There are, however, also positive signs emerging: (i) the latest FAO assessments suggests that although the rate of mangrove loss is still high, it has decreased significantly and was estimated at an annual relative loss of ~0.7% the period 2000-2005, (ii) replantation or rehabilitation initiatives are increasing, (iii) an increasing number of coastal mangrove wetlands have been designated as Ramsar sites during the past decade.

Management recommendations to enhance the potential of mangroves as a carbon sink

The data presented above make it clear that rehabilitation/restoration or plantation of mangrove forests are not only to be encouraged based on ecological or socio-economical considerations, but also have the potential of providing an efficient sink of CO_2, both on short and longer time-scales (i.e. biomass production during forest establishment and growth, accretion of carbon in mangrove sediments). The magnitude of this carbon sink, however, can be expected to be highly variable, and depends both on factors related to the primary production side (i.e. productivity will depend in part on the species or species assembly, latitude, and site conditions such as nutrient status, hydrology etc.) and on factors influencing the degree of longer-term sequestration of biomass in sediments, such as the rate of sediment deposition and exchange of carbon with adjacent systems. Indeed, there is a diversity of geomorphological settings where mangrove forest growth and develop, and that can be subdivided into a continuum of landforms based on the relative processes of river input, tides, and waves (Woodroffe, 2002). There is some indication that these diverse geomorphological habitats, each with different vegetation types, results in specific mangrove structural and productivity characteristics. This correlation between coastal landform and ecological function has particularly been documented relative to the net primary productivity (NPP) and detritus exchange across a variety of mangrove locations (Twilley & Rivera-Monroy, 2009). Thus, given the paucity of documented case studies, proposing specific guidelines for mangrove management/ rehabilitation in the face of their carbon sink potential would be premature. Particularly since mangrove rehabilitation efforts have had mixed success (Field et al. 1998, Kairo et al. 2001 and references therein) and inadequate planting strategies can lead to large-scale failures (Samson & Rollon 2008). These ecological and management aspects need to be considered for all mangrove rehabilitation or restoration initiatives where adequate selection of the right combination of both species and sites is critical in enabling a successful establishment of mangroves.

One proposed strategy to improve our capability to estimate and forecast mangrove carbon and nutrient cycling patterns with limited, but robust information, is the use of simulation models. This approach, in association with field studies, shows some promises to develop tools for improving and enhancing management plans for mangrove protection, rehabilitation and restoration; including optimal scenarios for carbon allocation and CO_2 uptake, not only due to landscape-level natural variations, but also under the influence of human disturbances (e.g climate change). Current available models have been useful in synthesizing current knowledge about mangrove forest dynamics (see Berger et al 2008 and references therein). The modeling approach is suitable for simultaneously evaluating the effects of environmental changes and disturbances on ecological processes such as tree recruitment, establishment, growth, productivity, and mortality (Berger et al. 2008). Such estimates on the sustainability of mangrove resources may contribute not only to evaluating impacts of mangrove degradation to socio-economic systems but also help assessing the role of mangrove forest in the global carbon cycle.



A mature Avicennia marina *stand during high tide (i.e. flooded) conditions, Gazi Bay (Kenya) © Steven Bouillon, K.U.Leuven*

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Global distribution of Seagrasses

UNEP WCMC

Seagrass Meadows

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Fast facts

- Flowering marine plants that form extensive meadows and are globally distributed. Found in shallow waters of all continents except the Antarctic.
- Responsible for about 15% of total carbon storage in the ocean.
- Global extent of seagrass now estimated to be about 0.3 million km^2.
- Turnover time of seagrass biomass is long (2 weeks to 5 years for leaves and roots, while rhizomes can sometimes persist for millennia), relative to that of phytoplankton, making the role of seagrasses in the oceanic carbon budget proportionally more significant than expected from their areal cover.
- Long-term carbon burial of 83 g C m^{-2}yr^{-1}. This translates to global storage rates of between 27 and 40 Tg C yr^{-1}.
- The seagrass *Posidonia oceanica* is currently thought to be the most effective species in terms of long-term carbon storage. It is endemic to the Mediterranean and locally widespread. The capacity of it's meadows to accumulate carbon exceeds that of many terrestrial ecosystems such as boreal forests and show values commensurate with wetlands.
- However, two-thirds of the world's seagrass meadows within inhabited areas have been lost through human activities that lead to eutrophication and siltation.
- Management plans aimed at reducing the nutrient loads and preserving water clarity of coastal waters are a priority.

Seagrasses, a globally distributed group of marine flowering plants, form extensive meadows that are one of the world's most productive ecosystems. The seagrass leaves degrade slowly and, through their roots and rhizomes, seagrasses deposit large amounts of underground, partly mineralised, carbon. Thus, they constitute an important CO$_2$ sink, responsible for about 15% of the total carbon storage in the ocean. In this chapter, we present a brief overview of what seagrasses are; their ecological functions and their importance as carbon sinks; the threats to them; as well as recommendations on how to manage them to preserve/restore their capacity.

1. Ecosystem definition and global occurrence

The seagrasses, a functional group of about 60 different species of underwater flowering plants, have a near global distribution, and form extensive seagrass meadows in the coastal waters of all the world's continents except the Antarctic (Figure 1). In temperate areas, the meadows are usually dominated by one or a few species (e.g. *Zostera marina* in the North Atlantic), whereas tropical meadows are usually more diverse, often with up to ten or even thirteen species.

Extent and the functions they provide in the marine ecosystem: The global extent of seagrass meadows

Figure 1 Syringodium sp *growing in dense meadows off the coast of Tanga, Tanzania. (photo Jerker Tamelander/IUCN)*

has been estimated to between 0.6 million km^2 (Charpy-Roubaud & Sournia, 1990) and 0.3 million km^2 (Green & Short, 2003; Duarte et al., 2005), with the latter estimate taking into account reports of long-term decline rates in seagrass coverage. Although seagrass meadows cover a relatively small portion of the ocean (approx 1%), they play an important role in the coastal zone and provide ecosystem goods and services that have been estimated to be of high value compared with other marine and terrestrial habitats (Costanza et al., 1997). Furthermore, the presence of seagrass meadows is global, unlike mangroves, corals or salt marsh plants, which have a more limited spatial distribution (Short et al., 2007)

Seagrass meadows provide important services supporting the overall functioning of the coastal zone. Some larger animals (like dugongs, turtles, geese, and some fish) are able to digest seagrass leaves. However, more important is the fact that the leaves of seagrasses often become covered by a wide range of algae and invertebrates, which are eaten by a variety of fauna, which both live within the seagrass meadow and migrate from outside the meadows. The dense cover of seagrass shoots also constitutes a sheltering structure, attracting several species of animals. Fish use the seagrass meadow as a nursery where their fry can hide, prawns settle in the seagrass meadows at their post-larval stage and remain there until they become adults (Watson et al. 1993). Moreover, several animals migrate from other habitats, like coral reefs and mangrove areas, to the seagrass meadows to feed (Unsworth et al. 2008), suggesting that seagrass meadows provide an important link between terrestrial and other marine habitats (Heck et al., 2008).

Other important functions of seagrass meadows are their sediment trapping and sediment binding capacities. They trap suspended materials from the currents, and thereby clear the water. The rhizome and root systems stabilise the sediments and help prevent coastal erosion during storms, rains and floods. Seagrass detritus is also important, through export and maintenance of food-webs in deeper waters and as an important carbon sink, due to its slow rate of decomposition.

Biomass and production: The average standing biomass built up within seagrass meadows of 184 gC m^{-2} (Duarte & Chiscano, 1999) is relatively low compared with terrestrial ecosystems as it represents les than 1% of the average standing biomass in tropical, temperate and boreal forests combined (Table 1,). In contrast, the absolute rate of seagrass net production (400-817 gC m^{-2} yr^{-1}), which is of comparable magnitude to other coastal plants, is in fact higher than most terrestrial ecosystems (Table 1, Duarte & Chiscano, 1999; Mateo et al., 2006). Added to this, in the marine environment, the turnover time of seagrass leaves is long (15-1827 days), relative to the few days of phytoplankton,

Ecosystem	NPP ($gC\ m^{-2}\ a^{-1}$)	Standing stock ($gC\ m^{-2}$)		Area ($*10^{12}\ m^{2}$)	NPP ($PgC\ a^{-1}$)	Global carbon stocks (PgC)	
		Plants	Soil			Plants	Soil
Tropical forests	778	12045	12,273	17.6	13.7	212	216
Temperate forests	625	5673	9,615	10.4	6.5	59	100
Boreal forests	234	6423	34,380	13.7	3.2	88	471
Tropical savannas and grasslands	787	2933	11,733	22.5	17.7	66	264
Temperate grasslands and shrublands	424	720	23,600	12.5	5.3	9	295
Deserts and semi-deserts	31	176	4,198	45.5	1.4	8	191
Tundra	105	632	12,737	9.5	1	6	121
Croplands	425	188	8,000	16	6.8	3	128
Wetlands	1229	4286	72,857	3.5	4.3	15	225
Seagrass meadows (*Posidonia oceanica*)	400-817 (60-184^{a})	184 (124^{b})	7,000^{c} (40,000-160,000^{d})	0.3 (0.035)	0.49 (0.002-0.0064)	0.06 (0.004)	2.1 (1.4-5.6)

Table 1 Comparison of carbon stocks on a unit area basis in terrestrial ecosystems (Janzen et al., 2004) and seagrass meadows (Duarte & Cebrian 1996, Duarte & Chiscano 1999,Duarte et al., 2005), with global pools determined by using the reported surface areas covered by each ecosystems.

a. Pergent et al., 1994, b. Romero et al., 1992, c. Calculated using organic carbon concentration of 0.7 wt%, porosity of 80% and dry solid density 2.5 g cm^{-2}, d. Mateo et al., 1997

making the role of seagrasses in the oceanic carbon budget proportionally more significant than expected from their cover or primary production alone (Smith, 1981).

Many seagrasses also deposit considerable amounts of carbon in their below-ground tissues with ratios of below-ground to above-ground biomass ranging from 0.005 to 8.56 (Duarte & Chiscano, 1999). Larger seagrass species tend to develop high below-ground biomass and hence have a greater capacity for accumulation of carbon due to the relatively slow turnover of the roots and rhizomes (40 days to 19 years). The seagrass species *Posidonia oceanica* can bury large amounts of the carbon it produces, resulting in partly mineralised, several metres thick, underground *mattes* with an organic carbon content of as much as 40 %. These mattes can persist for millennia, thus representing a long-term carbon sink (Pergent et al., 1994; Romero et al., 1994; Mateo et al., 1997, 2006).

2. Carbon cycling in the ecosystem and its importance as a carbon sink

Fate of carbon: The proportion of biomass produced by seagrasses that is directed into carbon storage is dependent on the extent to which carbon is channelled through herbivory, export and decomposition. Estimates of herbivory, decomposition and export all vary greatly due to the intrinsic properties of individual species and although carbon fluxes in different species may follow the same general routes, the relative importance of the different carbon pathways among seagrass species may differ widely.

Overall, herbivory generally has a limited role in constraining seagrass biomass with most estimates being <10% of plant production. In coastal vegetation, only mangroves have a lower proportion of their net biomass production lost through herbivory (Duarte & Cebrian, 1996; Cebrian, 2002). The extent of herbivory in seagrasses reflects specific intra and inter-species differences and ranges from negligible values up to 50% of net production (Cebrian & Duarte 1998; Mateo et al., 2006). An important fact to note is that few herbivores consume below-ground production and yet these tissues (roots and rhizomes) can constitute 15-50% of the net production. Thus, in some species the below-ground deposits can accumulate and contribute strongly to the carbon stored in the sediment.

Thus, only a small proportion of seagrass biomass is lost to herbivory and the major pathway for leaf production is to enter the detritus, some fraction of which is exported, while most is decomposed locally.

Compared to other coastal plants, the decomposition rate of seagrass tissues is low, averaging 0.0107 d^{-1}, compared to 0.0526 d^{-1} for phytoplankton and 0.082 d^{-1} for mangroves (Enriquez et al., 1993, Duarte & Chiscano, 1999). The difference among ecosystems in the fraction of net production that is decomposed has been suggested to be related to tissue nutrient concentrations, with the slower growing seagrass species generally having lower nutrient concentrations (Duarte, 1990) and hence the lowest decomposition rates. In addition, roots and rhizomes generally tend to have a lower nutrient concentration than leaves, leading to the slower decomposition of below-ground than above-ground biomass.

Based on a compilation of available data, Duarte & Cebrian (1996) estimated that the largest part of the biomass produced is decomposed (50%), with export and herbivory amounting to 24. and 19% respectively, and the remaining 16% is stored (Figure 1). However, it must be noted that these are average values and therefore do not identify the relative importance of different seagrass species, in that some species may route more carbon to storage than others. The data compilation also showed an imbalance in the number of observations that constitute each average value, with observations for seagrass above ground biomass and net production far outweighing the number of observations of below-ground biomass, herbivory, export, decomposition and storage, with the latter probably being the least reported. For example, in a recent assessment of long-term (i.e. decades) carbon burial in vegetated sediments, the measured rates of carbon burial comprised of only five estimates and in a compilation of short-term (i.e. years) accumulation of seagrass tissues, the data set for carbon storage comprised of only about ten estimates. The accuracy of some estimates can also be called into question as they have not been measured directly, but have been calculated by measuring the other carbon routes and simply assuming that storage represents the missing term in the carbon budget.

Carbon storage: Estimates of the short-term (years) carbon storage in sediments average 133 gC m^{-2} yr^{-1} (range 10-350 gC m^{-2} yr^{-1}, n=10, Cebrian, 2002). This value compares well with direct estimates of longer term carbon burial, averaging 83 gC m^{-2} y^{-1} (range 45-190 gC m^{-2} yr^{-1}, n=5, Duarte et al., 2005). These values translate to global storage rates of between 27 TgC yr^{-1} and 40 TgC yr^{-1} assuming the areal coverage of seagrass as $0.3*10^{12}$ m^2. The data sets used in calculations of longer term carbon burial are obtained from a restricted geographical area (the Mediterranean) and are representative of meadows consisting of only two seagrass species (*Cymodocea nodosa* and *Posidonia oceanica*). It is not yet clear whether other seagrass species (e.g. *Thalassodendron ciliatum*, *Posidonia australis* and *P. oceanica*), which are known to produce organic matter that generates refractory deposits, also make a significant contribution to long term carbon burial (Mateo et al., 2006). The estimates of short term carbon storage, although more geographically spread, only include four additional geographic areas (in America and Europe) and two additional types of seagrass (*Zostera sp.* and *Thalassia testudinium*).

Based on the data from Duarte et al (2005), most

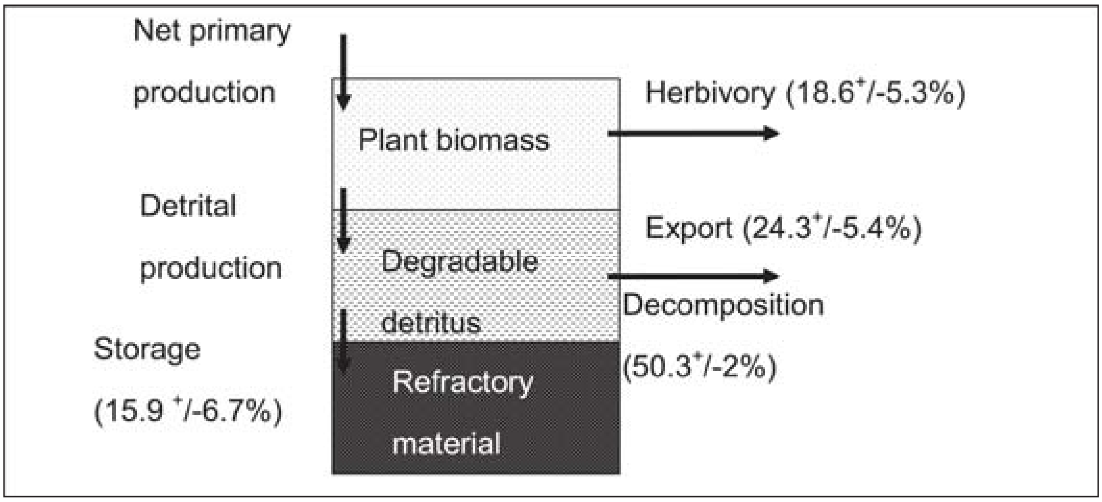

Figure 2 Fate of primary production, values in brackets represent% of net primary production. The data were derived by averagingindependent estimates from a range of seagrass species (adapted from Cebrian 1999 & Duarte & Cebrian 1996)

Ecosystem type	Long-term rate of accumulation (gC m^{-2} a^{-1})
Tropical forests	2.3-2.5
Temperate forests	1.4 – 12.0
Boreal forests	0.8 – 2.2
Temperate grasslands	2.2
Temperate deserts	0.8
Tundra	0.2 – 5.7
Wetlands	20
Posidonia oceanica meadows	9 – 112

Table 2 Long-term carbon accumulation rates in Holocene (<10,000yr old) soils and wetlands (Schlesinger 1990, Armentano & Menges1986) and Posidonia oceanica *(6,000 yr old) as one of the few species of seagrass that accumulate refractory organic matter in below-ground deposits termed mattes (Romero et al., 1994, Mateo et al., 1997, 2006).*

seagrass burial rates are about half as high as those for mangroves and salt marshes on an areal basis, and account for 12%, 9% and 25% respectively of the total carbon burial in coastal sediments. However, the rates of long-term carbon accumulation by *Posidonia oceanica* exceed those of terrestrial ecosystems and show values commensurate with wetlands (Table 2). Reported cases of seagrass loss have increased by an order of magnitude over the last 40 years (Orth et al., 2006) and the areal extent of seagrasses has been estimated to have reduced by 50% over a period of about 15 years (Duarte et al 2005), thus it is possible that the proportion of the global carbon sink that is represented by seagrass meadows has been depleted by a similar amount. Hence, a better quantification of the magnitude of carbon sinks in seagrass meadows and a better understanding of future trends for seagrass meadows will be necessary if we are to fully realise how the capacity of the coastal zone to act as a carbon sink will change in the future.

3. Threats to the ecosystem

Human interventions have altered coastal habitats severely, causing extensive losses in seagrass habitats (Orth et al., 2006; Björk et al., 2008). For example, in the northwest Atlantic 65% of eelgrass has been lost south of Cape Cod, an industrialized area, as compared to only 20% lost north of Cape Cod, a less populated and industrialized area, since European settlement (Short & Short, 2003). It has even been proposed that two thirds of the seagrass meadows of the world within inhabited areas have been lost (Lotze et al., 2006). The major causes of decline are anthropogenic disturbances that lead to eutrophication and siltation; drivers that may be accelerated by global climate change now and in the near future. Sedimentation from watershed, terrestrial deforestation and mangrove clearing leads to transport of sediments through rivers and streams to estuaries and coastal waters, reducing water clarity and negatively affecting seagrass growth and development. Similarly, eutrophication caused by waste discharges into coastal areas have major impacts on water clarity, thus reducing the light available to seagrass (Terrados et al., 1998). Seagrasses are particularly susceptible to alterations in water clarity as they normally require higher light levels compared to other aquatic vegetation.

Global change impacts, such as increases in the concentration of CO_2 in seawater and consequently decreasing pH values (ocean acidification), will probably directly affect seagrass photosynthesis and growth. However, as of today, too little data is available to predict the effects of such changes on the productivity of seagrass meadows. The main effect of global change will probably be synergistic, where already stressed seagrass systems will be exposed to additional stressors, further reducing productivity and eventually resulting in increased losses of seagrass meadows (Björk et al., 2008).

4. Management recommendations to enhance the potential of the ecosystem as a carbon sink

The carbon sink service that seagrass meadows provide can only be sustained by preserving the health and extent of the world's seagrass meadows. Evidence shows that it is difficult to reverse seagrass loss at the meadow scale (Ralph et al., 2005; Orth et al., 2006), but conservation and restoration of less extensively affected meadows could be possible. Thus, management should be directed towards maintaining general environmental conditions that favour seagrass growth. However, since different seagrass species appear to have different capacities for carbon storage, it is also possible to specifically target those to maximise their carbon storage function.

Management aimed at preserving general health of seagrass meadows: Because the observed global decline in seagrass distribution is mainly caused by human activities such as sediment loading and eutrophication, management plans aimed at reducing the nutrient loads and preserving water clarity of coastal waters becomes a priority (Björk et al., 2008).
Controlling anthropogenic activities is one of the main ways that good seagrass health and hence productivity

can be preserved (see Short et al., 2002, Short & Burdick, 2005, Björk et al., 2008). To provide the most favourable conditions a number of requirements must be met. 1. A high water quality, This mean low turbidity waters, low concentrations of coloured dissolved organic matter and low levels of eutrophication. All of these will ensure that the waters support sufficient light penetration for seagrasses to thrive. 2. Good sediment conditions. The sediments should experience only low levels of disturbance/mechanical perturbations, low carbon accumulation rates and low concentrations of sulphide. 3. Maintenance of genetic variability and connectivity with other biological systems, and 4. Favourable water movement

In recent years it has become evident that these requirements cannot be met without creating a public awareness of the purpose of the management plans, and ensuring the participation by stakeholders, both in planning and implementation of management strategies.

Management aimed at preserving especially high carbon storage capacity: There are certain features of seagrasses that can enhance their potential to act as important sites for carbon storage. The low nutrient concentrations and high proportion of structural carbon in seagrass tissues, enhance carbon accumulation in the meadow by slowing down the destruction of organic carbon, and the large proportion of below ground biomass enhance carbon accumulation in the meadow by burying organic carbon quickly, before it can be exported from the meadow. It follows that the greatest proportion of organic carbon preserved in the sediments will be found in meadows consisting of slow growing species with a high allocation of biomass to the growth of below ground organs.

Of all the seagrasses studied, *Posidonia oceanica* probably represents one of the best species for carbon storage; it is also the best studied species in terms of carbon burial and probably provides the best estimate of the size of the carbon sink in at least one area of our coastal oceans. *Posidonia oceanica* is widespread and endemic to the Mediterranean and sustains carbon burial rates of 17-191 g Cm^{-2} yr^{-1}, forming a *matte* that can be thousands of years old. The thickness of the *matte* in one bay of the NW Mediterranean has been recently estimated using high-resolution seismo-acoustic imaging (Iacono et al., 2008), allowing the carbon accumulation to be calculated at 0.18 Mg m^{-2}. Given that *Posidonia oceanica* is thought to cover 0.035 million km^2 of the Mediterranean, the sediments below *Posidonia oceanica* meadows could represent a store of ~6 x 10^{15} tonnes of carbon, with a carbon accumulation rate of between 0.6-7 MgC yr^{-1} or 2-24% of global seagrass burial.

Although *Posidonia oceanica* may appear to make the Mediterranean a hot spot in terms of carbon burial, other seagrass species may, although today undiscovered, have similar potential for carbon burial. Even species with a lower carbon burial but a more widespread distribution may actually make a larger overall contribution to global carbon storage. Thus to make accurate predictions concerning the fate of seagrass production on a global scale, reliable estimates of the distribution and density of the dominating seagrass species in all different biogeographical regions and the potential of each species for carbon burial would be needed. These figures for seagrasses are not currently available as shown in a review of the literature on seagrass ecology (Duarte 1999). Of the papers reviewed in this study, 25% related to the ecology of just two of the seagrass species (*Thalassia testudinium* + *Posidonia oceanica*) and there was a geographic bias in published results, with 50% of the studies being undertaken in Caribbean and Mediterranean seagrass meadows.

Thus today, although we can only approximate the current importance of seagrass meadows as a carbon sink, the recent focus within the scientific community on global change and the importance of natural carbon sinks has resulted in a large number of research projects aiming at making it possible to incorporate the biogeography of seagrass species and their propensity for carbon storage into an accurate global carbon budget.

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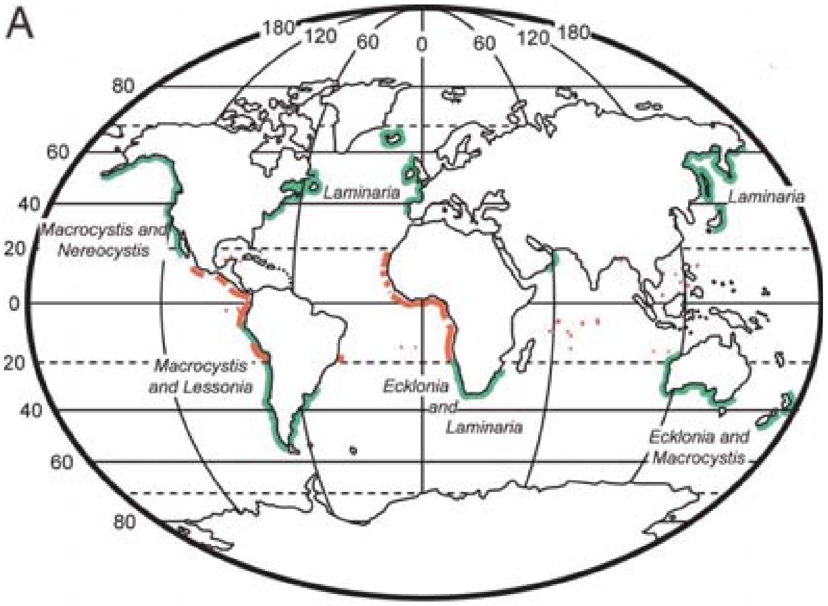

Geographic distribution of kelp forests in surface (green line) and deep (red line) waters, reproduced from Santelices - Santelices, B., 2007. The discovery of kelp forests in deep-water habitats of tropical regions, PNAS, 104 (49), 19163 – 19164 by kind permission of Proceedings of the National Academy of Sciences (PNAS).

Kelp Forests

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Fast facts

- Assemblages of large brown algae in the Order Laminariales.
- Kelps dominate the autotrophic biomass and production of shallow rocky substrates in temperate and arctic regions of the world but a complete survey of the world's kelp forest has never been done.
- Carbon cycling within kelp forests is characterized by rapid biomass turnover that can be as high as 10 times per year. There are few data on the fraction of kelp carbon that is incorporated into long-term carbon reservoirs such as marine sediments.
- It is likely that carbon storage by kelp dominated ecosystems will mainly be a function of the size of the standing biomass of kelp and associated understory algae. This means that the limit of carbon storage in these systems will be a direct function of the amount and condition of suitable habitat.
- The global kelp standing crop can be estimated to be from ~7.5 Tg C and (if modelled predictions of distribution are accurate) could be as much as 20 Tg C.
- Applying a conservative estimate for kelp forest net primary production of 1000 g C m-2 yr-1 to the area of shallow coastline with significant kelp yields a global kelp production of 15 Tg C yr-1. If deep tropical areas of potential kelp are accounted for, then global kelp production approaches 39 Tg C yr-1.
- Land use practices that alter the amount and constituents of runoff and the coastal discharge of municipal, agricultural and industrial wastes negatively impact kelp forests by increasing turbidity, sedimentation and nutrient loads. Human harvests of top predators such as lobster, fish, and sea otters have been implicated as a cause of kelp forest degradation world wide.
- To protect kelps, it is necessary to implement policies that restrict the chronic discharge of municipal and industrial waste waters into the nearshore, and land use practices that elevate the concentrations of sediments, nutrients and pollutants in runoff delivered to the ocean. Fishing damage is best managed by restricting the harvest of kelp and associated biota, which can be done using traditional fishery management practices in combination with the establishment of marine protected areas.

Definition and global occurrence

Kelp forests consist of conspicuous assemblages of large brown algae in the Order Laminariales. They largely occur in shallow subtidal (< 30 m) rocky habitats in most temperate and high latitude coastal areas of the world (Dayton 1985, Schiel and Foster 1986). Some species of kelp may occur at much greater depths (i.e., 60 - 200 m) in areas of high water clarity (Kain 1979), including tropical regions where they are known to form extensive deep-water forests (Graham et al. 2007a). While kelps are found worldwide, their present day distribution suggests a Pacific origin, with all but one of the 27 genera occurring in the North Pacific, 19 of these exclusively (Estes and Steinberg 1988).

Kelps are taxonomically diverse and kelp species differ greatly in morphology. The giant kelps produce floating canopies that extend throughout the water column making them the largest algae in the world. Non-floating stipitate forms produce a subsurface canopy up to 3 m in height, while low-lying prostrate species form a canopy covering the bottom. Mann (2000) characterized kelp forests by their dominant genera and recognized three general types, those dominated by *Laminaria*, *Ecklonia*, and *Macrocystis. Laminaria* is the dominant genera in the eastern and western Atlantic and western Pacific, *Ecklonia* is prevalent in Austral Asia and South Africa, and the giant kelp *Macrocystis* dominates in the eastern Pacific off the coasts of North and South America.

Although kelps are technically restricted to the order Laminariales, large brown algae in the Order Fucales are occasionally referred to as kelps. Much like kelps, these fucalean algae (commonly referred to as rockweeds) occur world wide, but unlike kelps they are most diverse in the Southern Hemisphere where they form dense forests subtidally (Schiel and Foster 1986).

Goods and services

Economically, kelp forests are one the most important marine ecosystems in temperate regions. They are the primary habitat for many commercial and recreational fisheries that include a wide diversity of mollusks, crustaceans, and finfish (Foster and Schiel 1985, Mann 2000, Graham et al. 2007b). Kelp itself is harvested for a wide range of uses such as food, food additives, pharmaceutical and cosmetic applications, animal fodder, and biofuel (Neushul 1987, Leet et al. 1992). In addition, vast amounts of kelp are grown commercially in marine farms in many parts of the world where it is harvested for human and animal consumption (Tseng 1981, Gutierrez et al. 2006).

In addition to provisioning services, kelp forests provide many regulating and cultural services as well. Importantly, they constitute one of the most diverse marine systems in temperate regions. As foundation species (*sensu* Dayton 1975) kelps provide the main source of food and shelter for many forest inhabitants (Schiel and Foster 1986), and they exert a profound influence on the abundance and distribution of the vast number of species that associate with them (Eckman and Duggins 1991, Graham 2004, Arkema et al. 2009). As such kelp forests play a critically important role in the conservation of biodiversity; a ecological service that has long been recognized (Darwin 1839). The trophic importance of kelp, however, is not limited to the area within kelp forests as the majority of kelp biomass can be exported out of the forest to adjacent habitats where it has been shown to be an important dietary component of terrestrial, intertidal and deep sea food webs (Polis and Hurd 1996, Harrold et al. 1998, Dugan et al. 2003).

Kelp forests also have high recreational value for fishing, diving, and boating, and they are a favorite area for sightseeing and photographing marine birds and mammals. Importantly, kelp forests provide many opportunities for education. They are a popular exhibit at most public aquaria, and they serve as a natural laboratory and classroom for training marine scientists and the general public at large, which enhances stewardship of the ocean and its resources.

Biomass and production

Kelps dominate the autotrophic biomass and production of shallow rocky substrates in temperate and artic regions of the world (Mann 2000). A complete survey of the world's kelp forest has never been done. The length of all coastlines where kelp forests are expected to occur has been estimated at 58,774 km of which about 30,000 km are believed to have significant kelp forests (de Vouys 1979). Deriving estimates of the global standing crop of kelp on these coastlines is challenging because the biomass density and cross-shore width of kelp forests vary greatly with species, time (both seasonally and inter-annually), and location (both within and among sites). If one was to assume that kelp forests were restricted to coastlines with significant kelp and had an average biomass density of 500 g C m^{-2} (Table 1), and an average forest width of 500 m, then the global kelp standing crop would be ~7.5 Tg C. Understory algae associated with kelp forests may increase the standing crop of the ecosystem by 20%

	Wet g / m^2	g C / m^2	Reference
Laminaria	4,800 – 16,000	220 - 720	Mann 1972a
Ecklonia	6,000 – 18,000	270 - 610	Mann 2000
Macrocystis	70 - 22,000	21- 660	Foster & Schiel 1985
Understory algae within Macrocystis forests	2 – 4,800	0.6 - 144	Foster & Schiel 1985

Table 1. Estimates of standing biomass for three common kelp genera and for understory algae within Macrocystis *forests (other than* Macrocystis*). Dry wt was assumed to be 15% of wet wt for* Laminaria *and* Ecklonia *and 10% for* Macrocystis *and its associated understory; carbon wt was assumed to be 30% of dry wt for all species (Mann 1972, Rassweiler et al. 2008).*

Forest type	Standing crop (g C m^{-2})	NPP (g C m^{-2} yr^{-1})	Litter mass (dry kg m^{-2})	Reference
*Boreal forest	1000	400	4.0	Whittaker 1975
*Temperate deciduous forest	1500	600	2.0	Whittaker 1975
*Temperate evergreen forest	1,750	650	3.0	Whittaker 1975
*Tropical seasonal forest	1,750	800	0.5	Whittaker 1975
*Tropical rain forest	2,250	1100	0.2	Whittaker 1975
Laminaria	220 - 720	1750		Mann 1972a, 1972b
Ecklonia	270-610	1100 - 1600		Mann 2000
Macrocystis	120 - 273	670 - 1300	0.015	Gerard 1976 Wheeler & Druehl 1986 Reed et al. 2008

*Table 2. Comparison of standing crop, production and liter mass of terrestrial forests and three general types of kelp forests. *(based on dry weight and the assumption that dry weight of forest trees is 50% carbon; Hamburg et al. 1997).*

or more (Table 1). This estimate does not account for deep (30 m – 200 m) kelp in unexplored tropical waters, which Graham et al. (2007) estimated at > 23,500 km^2 using an oceanographic-ecophysiological model that accurately identified known kelp populations. If their model predictions are accurate, then the global standing crop of kelp could be as much as 20 Tg C.

Kelps are among the fastest growing autotrophs in the world with growth rates averaging up to 2 to 4% of the standing biomass per day (Wheeler and Druehl 1986; Reed et al. 2008). The high growth rate of kelps is principally responsible for the high rates of primary production recorded for kelp forests, which rank as one of the most productive ecosystems on earth (Table 2). The methods used to measure primary production in kelps have varied greatly among species and among studies of the same species, which have led to a wide range in estimates of net primary production (NPP) that are difficult to compare. Nonetheless, high rates of NPP have been measured for kelp forests regardless of the method used. This is despite the fact that significant production by kelp lost as dissolved exudates and production by autotrophs in the forest other than the dominant species of kelp were not accounted for in the vast majority of studies, even though both can be substantial (Mann 2000, Miller et al. 2009). Applying a conservative estimate for kelp forest NPP of 1000 g C m^{-2} yr^{-1} (Table 2) to the area of shallow coastline with significant kelp yields a global kelp production of 15 Tg C yr^{-1} (~ 30% less than that estimated by de Vouys 1979). If deep tropical areas of potential kelp are accounted for, then global kelp production approaches 39 Tg C yr^{-1}.

Carbon cycling and its importance as a carbon sink

Our knowledge of carbon cycling in kelp forests comes almost exclusively from shallow temperate reefs. Much less is known about carbon cycling in deep tropical kelp forests due to their relatively recent discovery (Graham et al. 2007a) and the difficulty of conducting studies at greater depths. Current estimates of the biomass and productivity of the deep tropical kelp forests are comparable to their shallow water counterparts. Until we gain detailed knowledge of the distribution, food web dynamics and productivity of deep reefs global estimates of carbon cycling and storage in kelp forests will have high uncertainty. The more extensive knowledge of kelp forests on shallow reefs allows for a more complete analysis of carbon cycling and storage in those systems.

In contrast to terrestrial forests, a relatively small amount of standing biomass contributes to the high rates of NPP by kelp forests in shallow water (Table 2). Thus carbon cycling within kelp forests is characterized by rapid biomass turnover that can be as high as 10 times per year (Mann 1972b). Small amounts of litter mass typically accumulate on the floor of kelp forests compared to terrestrial forests (Table 2) as most kelp detritus is either is quickly consumed, decomposed and recycled, or exported out of the system (Gerard 1976, Spalding et al. 2003).

Excessive grazing by sea urchins can denude entire kelp forests (reviewed in Harrold and Pearse 1987). However, in more persistent stands grazers typically consume only a small fraction of the kelp that is produced (Gerard 1976). Despite the seemingly low proportion of kelp consumed in actively growing kelp forests, food web analyses using stable isotopes show that kelp-derived carbon provides a significant source of nutrition to a wide diversity of kelp forest consumers (Dunton & Schell 1987, Duggins et al. 1989, Kaehler et al. 2000, Page et al. 2008). The trophic importance of kelp is not limited to the kelp forest as the majority of kelp carbon can be exported out of the system to adjacent habitats where it has been shown to contribute significantly to terrestrial, intertidal and deep sea food webs (Polis and Hurd 1996, Harrold et al. 1998, Dugan et al. 2003). Less obvious is the fate of kelp production that is excreted as dissolved organic matter, which has been estimated to be on order of 30% to 40 % of NPP (Mann 2000).

The high turnover of kelp biomass within kelp beds means that carbon storage in kelp dominated ecosystems will be mainly a function of kelp standing crop and the efficiency with which detached particulate and dissolved kelp carbon is sequestered into long-term reservoirs. There are few data on the fraction of kelp carbon that is incorporated into long-term carbon reservoirs such as marine sediments, however decomposition studies on beaches and in the ocean suggest that most of this material is rapidly consumed or decomposed (Dugan et al. 2003, Spalding et al. 2003). The amount of kelp incorporated into offshore sediments is unknown although the flux of kelp carbon to deep waters can be substantial (Harrold et al. 1998) Despite these uncertainties it is likely that carbon storage by kelp dominated ecosystems will mainly be a function of the size of the standing biomass of kelp and associated understory algae. This means that the limit of carbon storage in these systems will be a direct function of the amount and condition of suitable habitat.

Threats to the ecosystem

Kelp forests are potentially at risk to a number of human activities that occur in the nearshore coastal zone. Land use practices that alter the amount and constituents of runoff and the coastal discharge of municipal, agricultural and industrial wastes are known to negatively impact kelp forests by degrading the physical, chemical and biological environment in which they occur (North et. al. 1964, Meistrell and Montagne 1983, Ambrose et al. 1996, Airoldi & Beck 2007, Connell et al. 2009). The most severe effects appear to result from increased turbidity and sedimentation, which cause a reduction in bottom irradiance and loss of suitable rocky substrata needed for kelp attachment (Foster and Schiel 1992). Elevated nutrients associated with agricultural runoff and sewage may reduce the competitive ability of kelp and lead to its demise in some areas (Gorman et al. 2009), but have positive effects on kelp recruitment and growth in other areas where oceanic conditions are more oligotrophic (Tegner et al.1995). While kelps appear to be relatively resistant to the direct effects of oil pollution (Foster 1971), many kelp forest consumers are not, and oil spills may have both positive or negative effects on kelp depending on the consumers affected and their position in the food web (North et al. 1964, Dean et al. 2000).

Perhaps the most publicized human impacts to kelp forests are those related to fishing. Human harvests of top predators such as lobster, fish, and sea otters have been implicated as the cause of kelp forest degradation world wide (Jackson et al. 2001, Steneck et al. 2002). Detailed data documenting the time course and extent of kelp forest loss, however, are lacking for most areas. While there is sufficient evidence to implicate fishing and hunting as a cause for kelp forest decline, the magnitude and extent of degradation caused by these activities has been the subject of considerable debate (reviewed in Schiel and Foster 1992, Graham et al. 2007b). By nature, kelp forests are extremely dynamic systems that vary greatly in space and time in response to naturally occurring changes in oceanographic conditions. Such dynamics, coupled with insufficient long-term data make it difficult to distinguish the effects of fishing from natural occurring events in many areas.

Changes in global climate are also likely to impact kelp forests, but the nature of these changes are difficult to predict. Increases in sea surface temperatures will likely be accompanied by increased water column stratification resulting in decreases in the supply of nutrients to reefs, which should have an adverse effect on kelp growth. In contrast, to the extent that kelps are limited by the availability of carbon, predicted increases in the concentration of atmospheric CO_2 could have a positive effect on kelp photosynthesis and growth. The increase in bicarbonate ion concentration with enhanced CO_2 dissolution in seawater would benefit all kelp species. Kelp species that produce floating surface canopies may further benefit by directly exploiting increased atmospheric CO_2 at the air-water interface. The accompanying consequences of ocean acidification on kelp forests are more difficult to predict due to the limited information on the effects of reduced pH on kelp physiology.

Climate driven changes in the frequency and intensity of storms are likely to have one of the largest impacts on the production and storage of kelp carbon as disturbance from waves is one of the main factors affecting the standing crop of kelps (Dayton 1985, Schiel and Foster 1986). It has been suggested that the deep water forests in tropical regions may serve as a spatial refuge for kelp during extended periods of climate change (Santelices 2007). In any case, climate related changes will undoubtedly affect the entire forest community of kelp, algal competitors, invertebrate grazers, and vertebrate predators. The impacts of climate change on kelp will undoubtedly be influenced by direct and indirect interactions involving a suite of forest inhabitants.

Management

The most prudent approach to managing the world's kelp forests is to avoid, prevent, or limit habitat degradation and loss caused by humans. Kelp forests require good water quality and suitable hard substrate for attachment. Consequently, management practices aimed at protection should focus on policies that preserve water quality and rocky habitats in areas where kelp forests are found. Chief among these should be policies that restrict the chronic discharge of municipal and industrial waste waters into the nearshore, and land use practices that elevate the concentrations of sediments, nutrients and pollutants in runoff delivered to the ocean. Degradation of kelp forests caused by the direct and indirect effects of fishing are best managed by restricting the harvest of kelp and associated biota, which can be done using traditional fishery management practices in combination with the establishment of marine protected areas (Allison et al. 1991). One advantage that marine protected areas have over traditional fishery management is that they are better able to guard against the indirect effects of fishing as they protect the entire kelp forest ecosystem rather than selected species.

The natural recovery of kelp often ensues rapidly once human induced stressors have been removed and the nearshore environment has been returned to its natural state. Without nearby kelp populations, however, kelp recovery may be slow and highly variable, depending in part on dispersal to the area, the size of the habitat to be restored, and the presence and activity of kelp grazers. Attempts to restore degraded kelp forests using a variety of techniques have been met with mixed success and the effects of human intervention have often been obscured by inadequate study designs and natural events (Schiel and Foster 1992).

The construction of an artificial reef has been used to mitigate for the loss of kelp forest habitat in the case where the stressors causing degradation (i.e., the cooling water effluent from a coastal power plant) were allowed to continue (Reed et al. 2006). Artificial reefs have also been used to transform soft bottom habitats into hard bottom areas in efforts to expand kelp habitat beyond its natural occurrence (Turner et al. 1969, Lewis and McKee 1989, Terawaki et al. 2003). The depth, topography, and bottom coverage of an artificial reef as well as its proximity to existing kelp forests are important in determining the timing and extent of colonization by kelp and its ability to persist (Reed et al. 2004, 2006). If designed properly, artificial reefs could be used to expand kelp habitat and thereby increase the storage of kelp carbon. Although the technology for building artificial reefs that support kelp is largely developed, the large-scale transformation of soft bottom habitats into hard bottom kelp forests is likely to be expensive and will involve trading resources associated with one habitat type for those associated with another. For these reasons, the pros and cons of using artificial reefs for habitat transformation should be carefully weighed and considered.

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Global distribution of Coral Reefs

UNEP WCMC

Coral Reefs

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Smith and Gattuso show from ocean chemistry that coral reefs are not a sink for the greenhouse gas carbon dioxide. The point is we cannot count on reefs to clean the atmosphere of our carbon dioxide emissions. We have to act decisively and do it right now, before it is too late.» – Richard B. Aronson, Florida Institute of Technology and President of the International Society for Reef Studies.

Coral reefs support the highest marine biodiversity in the world, containing an estimated 25% of all marine species (Roberts, 2003). More than 500 million people worldwide depend on them for food, storm protection, jobs, and recreation. Unfortunately, many of the world's coral reefs have been degraded, mainly due to human activities. According to the Status of Coral Reefs of the World: 2008, we have lost 19% of the original area of coral reefs since 1950, 15% of coral reefs are in a critical state with loss possible within the next 10 to 20 years, and a further 20% are seriously threatened with loss predicted within 20 to 40 years.

Introduction

Coral reefs are highly visible, "charismatic," and metabolically active benthic ecosystems that contain the two primary carbon (C) reservoirs found in other marine ecosystems: organic matter and calcium carbonate. They are recognized to be among the most rapid producers, per unit of area, of both organic carbon (the transformation of inorganic carbon to organic carbon by photosynthetic organisms is called "primary production") and skeletal $CaCO_3$ (calcium carbonate – precipitated through the calcification process).

Aqueous inorganic carbon chemistry and calcification

Consider that carbon dioxide (CO_2) at a partial pressure (pCO_2) in the atmosphere is in equilibrium with water (that is, $pCO_{2(water)} = pCO_{2(air)} \equiv pCO_2$) ; the ratio of the molar concentration of CO_2 dissolved in the water, $CO_{2(water)}$, to pCO_2 is given by the CO_2 solubility (the Henry's Law coefficient, K_0). The $CO_{2(water)}$ reacts with water by hydration, to form carbonic acid (H_2CO_3). That acid dissociates into bicarbonate (HCO_3^-), carbonate (CO_3^{-2}) anions.

$$CO_{2(water)} \div pCO_2 = K_0 \qquad (1)$$

The value of K_0 is dependent upon salinity, temperature and hydrostatic pressure.

$$CO_{2(water)} + H_2O \rightarrow H_2CO_3 \qquad (2)$$

$$H_2CO_3 \rightarrow HCO_3^- + H^+ \qquad (3)$$

$$HCO_3^- \rightarrow CO_3^{-2} + H^+ \qquad (4)$$

CO_2 partial pressure equilibration between the water and overlying atmosphere is relatively slow, while the hydration and dissociation steps are effectively

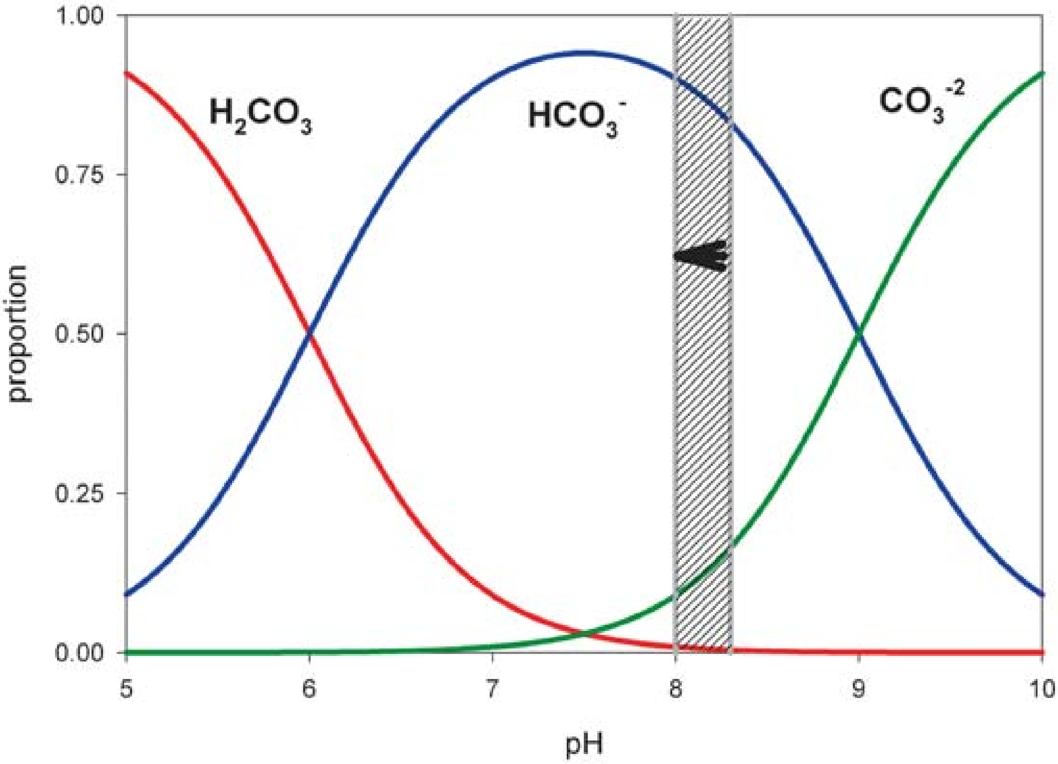

Figure 1. Proportional concentrations of aqueous CO_2 (H_2CO_3), HCO_3^-, and CO_3^{2-} in seawater as a function of pH. The vertical shaded area indicates the pH range of most tropical surface seawater, while the horizontal black arrow indicates the direction of pH shift in response to anthropogenic CO_2 emissions and acidification. With respect to anthropogenic effects on Ω, note the proportionally large decrease in CO_3^{-2}, relative to HCO_3^-, with falling pH in the range 8.3 to 8.0.

instantaneous. The distribution of the various forms of dissolved inorganic carbon among H_2CO_3, HCO_3^- and CO_3^{2-} is dependent upon hydrogen ion activity (defined by pH, -log[H^+], as well as water composition, temperature, and pressure). At the typical pH of surface seawater (~8.0 to 8.3) these three forms of dissolved inorganic carbon make up about 1% (H_2CO_3), 90% (HCO_3^-), and 9% (CO_3^{2-}) of the total DIC (Figure 1).

Primary production is the reaction by which organisms use solar energy to form organic carbon (chemically represented as CH_2O) from CO_2 plus H_2O. Leaving out essential nutrients, the chemical reaction describing conversion of CO_2 to CH_2O can be represented as follows:

$$CO_2 + H_2O \rightarrow CH_2O + O_2 \qquad (5)$$

Submerged aquatic organisms derive the CO_2 used in primary production from the various forms of dissolved inorganic carbon. CO_2 that is removed from the water is replaced from the atmosphere via equation 1.

Respiration is the process by which organisms (both photosynthetic and non-photosynthetic) oxidize organic matter and return organic carbon to dissolved inorganic carbon. Aerobic respiration (that is, respiration in the presence of O_2) is of direct relevance and is represented by:

$$CH_2O + O_2 \rightarrow CO_2 + H_2O \qquad (6)$$

As dissolved inorganic carbon is taken up by primary production or released by respiration, equations 2-4 quickly partition dissolved inorganic carbon among the various ionic forms within the water, and equation 1 slowly returns pCO_2 to equilibrium between air and water.

Calcification is a somewhat more complicated process. A simplified version of the reaction is given by:

$$Ca^{2+} + HCO_3^- \rightarrow CaCO_3 + H^+ \qquad (7)$$

A complication not represented by this equation involves the pH shift that accompanies the consumption of CO_3^{2-} (see equations 3, 4 and 7; also Figure 1). As a result of this pH shift, equation 7 is modified to the general form:

$$Ca^{2+} + 2HCO_3^- \rightarrow CaCO_3 + CO_2\uparrow + H_2O \qquad (8)$$

So a curious characteristic of $CaCO_3$ precipitation from water (whether by inorganic precipitation or calcification) is that the inorganic carbon used in the reaction is the HCO_3^- in the water, not CO_2 in the atmosphere. The calcification process thus actually releases CO_2 from the water back to the atmosphere, rather than removing it from the atmosphere. It will be pointed out below that Eq. 8 does not quite explain the real world quantitatively.

First we wish to point out the reason for the counter intuitive result represented by Eq. 8 (Gattuso et al., 1999a). The long-term (geological time scale; millions of years) CO_2 cycle involves release of CO_2 from the Earth interior into the atmosphere. This delivery is geochemically significant, but is a small fraction of the fluxes among the Earth Surface System reservoirs. As the volcanic CO_2 emissions are introduced into the atmosphere, they induce weathering of volcanically derived silicate minerals also emanating from the Earth interior. The igneous rocks are chemically unstable and react (by chemical weathering) with CO_2 and water. Igneous rocks are diverse in chemical composition; but to relate the carbonate and silicate cycles, we use $CaSiO_3$ (wollastonite) as an example of reacting silicate minerals:

$$CaSiO_3 + 2CO_2 + H_2O \rightarrow Ca^{2+} + 2HCO_3^- + SiO_2 \qquad (9)$$

This and similar reactions account for both the DIC and the dissolved positive ions (cations) in seawater. The HCO_3^- — rich water reacts with Ca^{2+} according to Eq. 8 to form $CaCO_3$. So the summed effect of Eqs. 9 and 8 is:

$$CaSiO_3 + CO_2 \rightarrow CaCO_3 + SiO_2 \qquad (10)$$

The chemically igneous silicate minerals are chemically unstable at Earth Surface temperature and pressure. These minerals react with CO_2 in the presence of water to form the more stable sedimentary minerals $CaCO_3$ and SiO_2. The atmosphere is the source of CO_2 that dissolves in the water during weathering. That dissolved CO_2 hydrates and dissociates (primarily to HCO_3^-, at oceanic pH: Figure 2) and is the source of the C that enters $CaCO_3$; the process of forming that $CaCO_3$ also delivers CO_2 from the oceanic DIC back to the atmosphere. The important points to this analysis, are (a) the demonstration that $CaCO_3$ precipitation taken alone (Eq. 8) is an atmospheric CO_2 source, not a sink, and (b) the geochemical explanation for this result.

As was the case for primary production (Eq. 5) and respiration (Eq. 6), the back reaction of Eq. 8 occurs ($CaCO_3$ dissolution), is a sink for atmospheric CO_2, and draws CO_2 out of the atmosphere:

$$CaCO_3 + CO_2 + H_2O \rightarrow Ca^{2+} + 2HCO_3 \qquad (11)$$

However, equation 8 does not quantitatively describe what happens when $CaCO_3$ is precipitated from seawater (Smith, 1985; Ware et al, 1991 and Frankignoulle et al, 1994). Consider seawater with pCO_2 in equilibrium with the overlying atmosphere (equation 1). Precipitating $CaCO_3$ quantitatively according to equation 8, that is, one mole of gaseous CO_2 release for each mole of $CaCO_3$ precipitation does not apply to seawater due to its buffering effect. Put simply, some of the CO_2 generated by calcification is scavenged by the CO_3^{2-} ions according to:

$$CO_2 + CO_3^{2-} + H_2O \text{ -> } 2HCO_3^- \qquad (12)$$

Removing equimolar amounts of C as CO_2 and $CaCO_3$ from seawater open to the atmosphere would cause pCO_2(water) to drop below pCO_2(air). Yet the physical process that drives the CO_2 gas out of seawater is the pCO_2 differential between the water and air; gas moves from the higher-pressure to the lower-pressure reservoir, so gas evasion occurs only if there is a positive gradient from water to air. This constraint places an upper limit on the ratio of CO_2 evasion as gas to C precipitation in $CaCO_3$.

At an atmospheric pCO_2 of about 350 ppmv, it was found that the molar ratio of CO_2 evasion: $CaCO_3$ precipitation was ~0.6, generally known as the “0.6 rule”. Frankignoulle et al. (1994) undertook a more detailed analysis and demonstrated that the ratio rises towards 1.0 in seawater as pCO_2 rises.

The surface ocean pCO_2 is not constant, largely because of the details of oceanic vertical circulation; and coral reefs are found in waters of variable pCO_2. The calculation of reef performance is scaled to average conditions, near equilibrium, between the air and gas phases while recognizing that details of local flux depend upon local water composition.

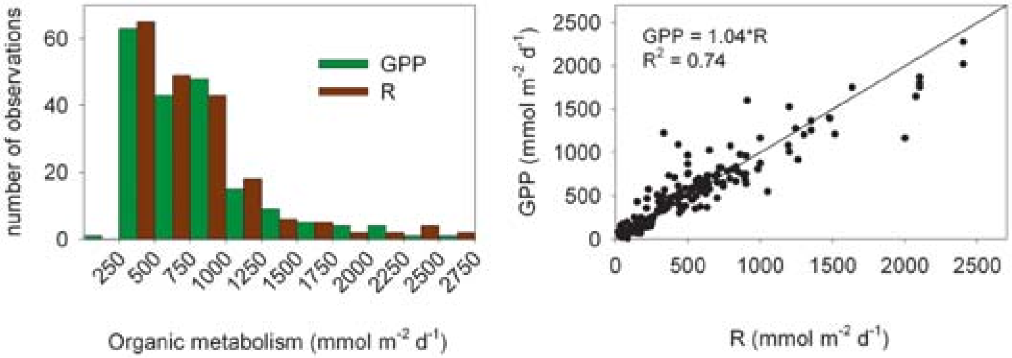

Figure 2. Gross Primary Production (GPP) and Respiration (R) for coral reef systems. The histogram demonstrates the general spread of data from individual studies, while the scatter diagram demonstrates the high correlation and near-unity regression coefficient between GPP and R. The mean ± standard deviation (median) are as follows:
GPP = 580 ± 602 (471); R = 548 ± 524 (433); NEP = 41 ± 307 (2).

Reef Area and Metabolism Area

We use a nominal area of 600,000 km^2, as being a round-number intermediate in estimates used for coral reef area studies (Smith, 1978; Kleypas, 1997; Spalding and Grenfell, 1997; Spalding et al., 2001) and recognize that inclusion of other shallow to intermediate depth tropical to high-latitude benthic communities that show evidence of primary production and calcification might double this area, or more (e.g., Andersson et al., 2008).

Organic and inorganic metabolism

A database compiled through 2004 is available for us to assess reef metabolism. Organic metabolism for individual sets of reef system measurements spans a range of ~0 to 3,000 mmol C m^{-2} d^{-1}, for both Gross Primary Production (GPP) and Respiration (R) (Figure 2). When the two measurements are plotted against one another, it can be seen that they are highly correlated with a P:R slope of about 1.04. In general, reefs appear marginally autotrophic.

There has been discussion of the likelihood of "modes" of typical reef performance. For example, Kinsey (1985) summarized various classes of data and found a range from whole systems and reef sediments near 300 mmol m^{-2} d^{-1} up to high-activity algal-dominated flats in excess of about 900 mmol m^{-2} d^{-1}. It appears that reefs typically exhibit both GPP and R values near 450 mmol m^{-2} d^{-1}, with NEP near (but probably slightly above) 0. Considering estimates of reef calcification, as derived from total alkalinity measurements (Figure 3), we see a range from slightly negative calcification estimates (representing $CaCO_3$ dissolution, largely in sediment-dominated portions of reefs) to values in excess of 300 mmol m^{-2} d^{-1}. In comparison with the relatively smooth change in organic metabolism, there are stronger suggestions of "modes" of calcification, ~25, 100, and 250 mmol m^{-2} d^{-1} (Smith and Kinsey, 1976; Kinsey, 1985), generally representing largely sedimentary areas, active reef flats and other shallow reef areas, and coral thickets, respectively.

Smith (1983) argued that modes approximating the three seen in the Gattuso database cover 90-95%, 5-10%, and 1-2%, respectively of reef area. If the proportional areas of the modes are multiplied by the estimated modal calcification rates, they yield a mean reef ecosystem calcification rate of about 35 mmol m^{-2} d^{-1} (or about 1,200 g $CaCO_3$ m^{-2} yr^{-1}).

Coral reefs as a carbon source or sink

The metabolic data presented above indicate that the preponderance of reef carbon sequestration is inorganic carbon. Effectively, organic carbon production can be ignored. This is consistent with the observation that the $CaCO_3$ percentage in reef sediments is close to 100% (so inorganic carbon percentage is close to 12% by mass), while the organic carbon percentage is typically ~0.5%. These proportions imply that inorganic carbon accounts for about 95% of carbon burial in reef sediments. If we take the reef area to be 0.6 x 10^{12} m^2 and the $CaCO_3$ burial to be ~1,200 g m^{-2} yr^{-1}, then the contemporary accumulation of $CaCO_3$ in coral reefs is ~ 700 Tg yr^{-1}. Based on mass contribution of C to the molecular weight of $CaCO_3$ (12/100), inorganic carbon burial is about 80 Tg yr^{-1}. Recalling that $CaCO_3$ precipitation causes gas evasion and applying the "0.6

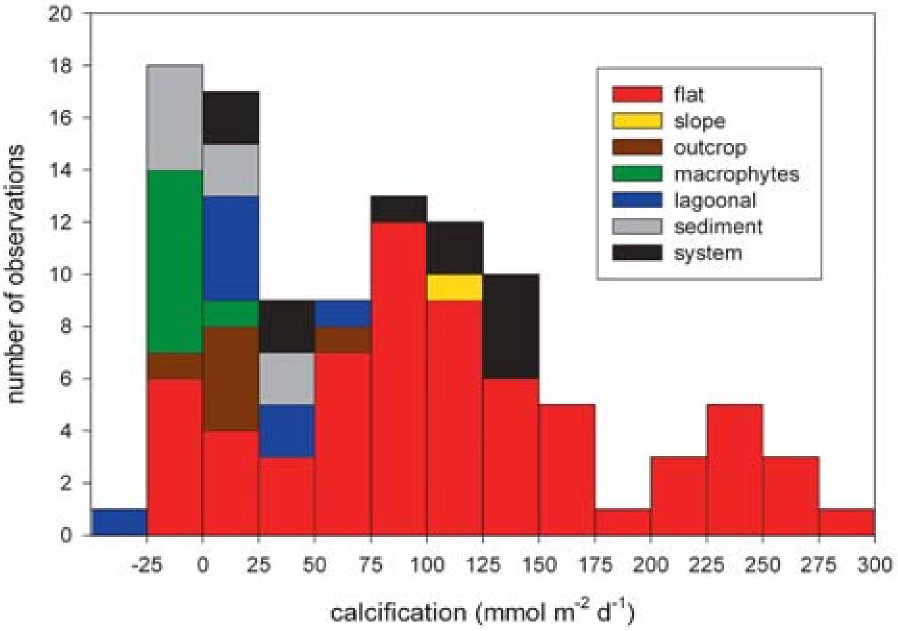

Figure 3. Calcification in coral reef ecosystems. Note the three apparent modes in the data.

rule” for the ratio of precipitated inorganic carbon to evaded CO_2 implies that coral reefs are a source for approximately 50 Tg C yr^{-1}. This is close to the model estimate derived by Kleypas (1997) (43 Tg yr^{-1}).

Reef Metabolic Responses to Anthropogenic Effects

Lowered carbonate saturation state

It has long been recognized that the process of $CaCO_3$ precipitation is related to the $CaCO_3$ saturation state of the water from which the precipitation, where the saturation state Ω is related to concentrations (or, more properly, activities) of Ca^{2+} and CO_3^{2-} in the solution:

$$\Omega = \frac{[Ca^{2+}][CO_3^{2-}]}{K_{sp}} \qquad (13)$$

where K_{sp} is the solubility product for the carbonate mineral in question (Gattuso et al, 1999a and Kleypas et al, 1999).

Coral reef calcifying organisms and sediments are dominantly aragonite, with high-Mg calcites being next in dominance, followed by low-Mg calcite. K_{sp} for aragonite is thus typically used for defining Ω. Ω variation in open seawater is operationally defined by $[CO_3^{2-}]$ and Ω for aragonite in tropical surface ocean water is typically 2-3.

The concept that Ω might limit $CaCO_3$ precipitation is not new and has been recognized since at least Rodgers (1957). Smith and Buddemeier (1992) were apparently the first authors to put global observations into the context of possible effects of anthropogenic changes in atmospheric CO_2 content on coral reef calcification. Subsequent authors cited by Kleypas et al. (1999) found evidence for saturation-state limitation by individual groups of calcifying organisms as well as in experimentally manipulated coral reef communities. Rising atmospheric CO_2 is causing Ω to fall.

The paper by Kleypas et al. brought this issue to scientific prominence both for coral reefs and other marine calcifying communities by their observation:

“By the end of middle of the next century (i.e., 21st), an increased concentration of carbon dioxide will decrease the aragonite saturation state in the tropics by 30 percent and biogenic aragonite precipitation by 14 to 30 percent. Coral reefs are particularly threatened…”

That paper has had substantial impact on scientific thinking, with many papers describing experimental or observational results consistent with the findings of diminished calcification in response to diminished Ω, as well as secondary functional and structural responses in coral reef systems. Examples of calcifying organisms (both reef and non-reef) showing evidence of slowed calcification in response to changed Ω include corals, several phyla of calcifying algae, foraminifera, and arthropods. A particularly noteworthy paper by D'Eath et al. (2009) presents evidence that coral calcification on the Great Barrier Reef may have diminished by about 14% since 1990.

There are some studies that challenge this "simplistic" explanation (e.g., Atkinson et al. 1995; Atkinson and Cuet, 2008) of Ω as the sole control on calcification, arguing that additional factors are likely to be significant as well. A paper by Iglesias-Rodriguez et al. (2008) argued that both calcification and primary production in a coccolithophorid species (non-reefal) actually increases with elevated pCO_2 but this conclusion is hotly debated (Riebesell et al., 2008). Nevertheless, the preponderance of evidence seems to point towards a link between Ω and calcification (Ilyina et al., 2009).

Moving beyond calcification itself, elevated $CaCO_3$ dissolution, including reef structures and sediments could be a widespread consequence of elevated atmospheric CO_2 and diminished Ω (Manzello et al., 2008). Any process which either lowers calcification rate or elevates dissolution rate of calcareous sediments or limestone constitutes a sink for anthropogenic CO_2 in the atmosphere (equation 9). The magnitude of this potential sink is difficult to assess, but could be larger than diminished reef calcification.

Eutrophication

Nutrient enrichment undoubtedly has affected many reefs around the world (Szmant, 2002). An important prediction by Kleypas et al. (1999) is a shift of community structure towards more non-calcifying species (e.g., more fleshy algae) in response to diminished Ω and calcification. In itself, a shift in GPP should have little impact of reefs (or the sites presently occupied by reefs as we know them) on the global carbon balance. Based on Figures 4 and 5, we assume that NEP would remain approximately constant. That is, any shift in GPP would be approximately balanced by a concomitant shift in R.

Management Implications

Unmanaged reef metabolism is a CO_2 source, because of the effect of $CaCO_3$ precipitation as discussed above. Organic metabolism appears not to be significant. The estimated source strength is presently ~60% of reef calcification. This is a rate of about 50 Tg C yr^{-1}, larger, if extended to include non-reef calcifying benthic ecosystems. There is some evidence that diminished calcification will reduce this source term for unmanaged reefs and perhaps reverse it if reefs and other calcareous materials move from a state of net calcification to net dissolution. The fact of reef calcification as a CO_2 source is not really a matter of informed debate, and arguments to the contrary reflect a misunderstanding of the underlying chemistry. Nor is the issue of dissolution being a CO_2 sink.

Intermediate between calcification at unchanging rates and dissolution of calcareous materials as Ω falls, there *is* legitimate scientific disagreement. The strong prevailing view is that reef calcification decreases as atmospheric CO_2 rises. It seems likely that, whatever the resolution of this discussion, reef communities of the future will look different from present coral reefs. There is reason to think that calcification will not cease, but details of community structure seem likely to shift. So reef appearance will probably change from what we presently recognize as coral reefs.

These changes could lead to loss of storm protection or loss of recreational value (the two major goods and services singled out for reefs by Costanza et al., 1997) or loss of fisheries, these changes are deleterious.

It therefore seems obvious that the role of coral reefs in discussion of management of anthropogenic CO_2 emissions is much more one of reefs being likely beneficiaries of CO_2 management, rather than being a useful management option in their own right.

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Carbon Sequestration by Coastal Marine Habitats: Important Missing Sinks

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Over the last two hundred years, the concentration of carbon dioxide (CO_2) in the Earth's atmosphere has increased by more than 30% (IPCC 2007). This increase has been driven by the combustion of fossil fuels, deforestation, destruction of other biological carbon reserves, cement production and other human sources of CO_2. The current rate of CO_2 increase in the atmosphere is at least an order of magnitude faster than has occurred for millions of years (Doney & Schimel 2007), and the current atmospheric CO_2 concentration is greater than the Earth has experienced in at least 800,000 years (Luthi et al. 2008). These changes have dramatic and longterm consequences for the Earth's climate – both atmospheric and oceanic – and for all life on Earth. Resulting shifts in the distribution and population of species and impacts on human communities from the Equator to the poles have already being observed (Parmesan 2006).

The Oceans and CO_2 Sequestration

Nearly a third of the anthropogenic CO_2 added to the atmosphere has been absorbed by the oceans (Sabine et al. 2004). Currently, the ocean and land absorb similar amounts of CO_2 from the atmosphere (Bender et al. 2005). However, projections suggest that CO_2 absorption by land sinks may decrease during this century (Friedlingstein et al., 2006), while the oceanic absorption of atmospheric CO_2 will continue to grow (Orr et al. 2001). The oceans are therefore critical as the ultimate sink for anthropogenic CO_2.

The long term implications of climate change for both terrestrial and marine systems have lead to strong international recognition of the need to stabilize the concentration of atmospheric CO_2 and other greenhouse gases. To achieve this, both dramatic decreases in the rate of greenhouse gas emissions and increases in the sequestration of atmospheric CO_2 must be rapidly implemented. Ongoing development of artificial and geo-engineering methods of carbon sequestration include techniques for CO_2 injection into the deep ocean, geological strata, old coal mines and oil wells, and aquifers along with mineral carbonation of CO_2. These techniques have potential for sequestering vast quantities of CO_2. However, these techniques are expensive, have leakage risks, significant potential environmental risk and will likely not be available for routine use until 2025 or beyond (Lal 2008). In contrast, preservation and restoration of naturally occurring biological carbon reservoirs represent CO_2 sequestration options that are immediately applicable, cost-effective, have numerous ancillary benefits, and are publicly acceptable. Biological reservoirs of carbon are, however, finite in capacity, making it likely that a combination of biological and artificial mechanisms of carbon sequestration will be required.

Currently approximately 8.5×10^{15} g C yr^{-1} is emitted by fossil fuel combustion and 1.6×10^{15} g C yr^{-1} by changes in biological systems resulting from the anthropogenic degradation or destruction of naturally occurring terrestrial biological carbon reservoirs. To date accounting for the CO_2 emissions resulting from land use, land use change and forestry has almost exclusively focused on degradation of terrestrial carbon sinks particularly forests, wetlands, and soils. However ocean habitats and especially coastal ocean habitats, have high densities of sequestered carbon and so their loss can be a significant contributor to CO_2 emissions.

Coastal Ocean Habitats and Carbon

Coastal oceans receive large inputs of organic matter and nutrients from land through riverine flows, sediment runoff and from the ocean from upwelling and current

Ecosystem type	**Standing carbon stock (gC m^{-2})**		**Total global area ($*10^{12}$ m^2)**	**Global carbon stocks ($*10^{15}$ gC)**		**Longterm rate of carbon accumulation in sediment (gC m^{-2} y^{-1})**
	Plants	**Soil**		**Plants**	**Soil**	
Tropical forests	12045	12273	17.6	212	216	2.3-2.5
Temperate forests	5673	9615	10.4	59	100	1.4 – 12.0
Boreal forests	6423	34380	13.7	88	471	0.8 – 2.2
Tropical savannas and grasslands	2933	11733	22.5	66	264	
Temperate grasslands and shrublands	720	23600	12.5	9	295	2.2
Deserts and semi-deserts	176	4198	45.5	8	191	0.8
Tundra	632	12737	9.5	6	121	0.2 – 5.7
Croplands	188	8000	16	3	128	
Wetlands	4286	72857	3.5	15	225	20
Tidal Salt Marshes			Unknown (0.22 reported)			210
Mangroves	7990		0.152	1.2		139
Seagrass meadows	184	7000	0.3	0.06	2.1	83
Kelp Forests	120-720	na	0.02- 0.4	0.009-0.02	na	na

Table 1 Comparison of carbon stocks and longterm accumulation of carbon in soils in key terrestrial and coastal marine ecosystems. (Terrestrial ecosystems (Kennedy & Bjork 2009), seagrass meadows (Duarte & Cebrian 1996, Duarte & Chiscano 1999, Duarte et al. 2005, Kennedy & Bjork 2009), Tidal Salt marshes (Chmura et al. 2003), Mangroves (Alongi 2002, Cebrian 2002, Duarte et al. 2005, FAO 2007), Kelp (Reed & Brzezinski 2009))

systems. These high levels of nutrient input make the coastal ocean one of the most biologically productive areas of the biosphere (Chen and Borges 2009). The high productivity of the coastal ocean environment supports a diverse spectrum of ecosystems and, like terrestrial ecosystems, sequesters significant carbon stocks in plants, animals and in the sediment. Table 1 summarises the carbon sequestration of key terrestrial and coastal marine ecosystems.

Coastal marine habitats such as mangroves, seagrass meadows, kelp forests and tidal salt marshes each account for areas 1% or less of the dominant terrestrial habitats of forests, grasslands and deserts. The carbon stocks in these marine systems, however, is similar to that observed in many of these terrestrial systems. Mangroves are one of the most productive ecosystems globally; the standing stock of carbon in above ground mangrove biomass is estimated to be 7990 gC m^{-2}. The belowground biomass of these trees has not been extensively surveyed but may store similar amounts of carbon (Cebrian 2002). The carbon stock in mangroves is therefore likely to be similar in magnitude to the highly productive terrestrial forest habitats: Tropical forests 12045 gC m^{-2}; Temperate forests 5673 gC m^{-2}; and Boreal forests 6423 gC m^{-2}. Similarly, the carbon contained in seagrasses and kelp forests (184 gC $m^{-2}$and 120-720 gC m^{-2} respectively) is similar to many terrestrial ecosystems, including croplands. Degrading or removing these coastal habitats therefore has comparable immediate carbon emissions as degrading or removing similar sized areas of terrestrial habitat.

The dramatic difference between the coastal marine and terrestrial habitats is the capacity of marine habitats for longterm carbon sequestration in sediments. Carbon burial in coastal ocean sediments by mangroves, seagrasses and other vegetation has been largely ignored in most accounts of the global carbon cycle – likely a result of the small areal extent of these habitats and a reflection of the fact that only human-induced sequestration (ie afforestation and reforestation) is accounted for in national greenhouse gas inventories. However, vegetated coastal habitats transfer large amounts of carbon to the sediments, contributing about half of the total carbon sequestration in ocean sediments even though they account for less than 2% of the ocean surface. Moreover, these high burial rates can be sustained over millennia (Duarte et al. 2005).

The large carbon sequestration capacity of coastal habitats arises in part from the extensive belowground biomass of the dominant vegetation. For instance, the ratio of root to shoot biomass in salt marsh plants ranges from 1.4 to 50. Thus, much of the primary production is belowground, generating extensive carbon deposits in the sediment that can be as deep as 8 m deep (Chmura et al. 2003). Similarly, the biomass of living mangrove roots in the sediments can equal that above (Dittmar & Lara 2001). However, in all these habitats the percentage of buried carbon strongly depends on environmental conditions.

The high rate of carbon transfer to sediments by mangroves occurs over the global area covered by mangrove forests. Mangroves therefore generate approximately 15% of the total carbon accumulation in ocean sediments (Jennerjahn & Ittekkot 2002). The efficiency of this carbon sequestration in sediments by mangroves increases with the age of the mangrove forest, from 16% for a 5-year-old forest to 27% for an 85-year-old forest. Further, carbon sequestration by mangroves is longterm: organic carbon in the upper 1.5 m of the sediment of Brazil's Furo do Meio mangrove forest has been dated to be between 400 and 770 years old (Dittmar & Lara 2001).

The importance of seagrass meadows, mangrove forests and salt marshes for longterm carbon sequestration through burial in the sediment is particularly apparent when compared to terrestrial burial rates (Table 1). The rate of carbon storage in the sediment by salt marshes, mangroves and seagrasses is approximately 10 times the rate observed in temperate forests and 50 times the rate observed in tropical forests. (For instance, 139 gC m^{-2} y^{-1} mangroves compared to 2.5 gC m^{-2} y^{-1} for tropical forests.) The simple implication of this is that the longterm sequestration of carbon by 1 km^2 of mangrove area is equivalent to that occurring in 50km^2 of tropical forest. Hence, while relatively small in area, coastal habitats are extremely valuable for their longterm carbon sequestration capacity.

These estimates of longterm carbon sequestration by coastal vegetated habitats are likely underestimates. In some cases, important organic carbon sources such as detritus from the marine plants and terrestrial material are not considered in the calculation of the accumulation rates. The accumulation rates also do not account for the tidal pumping of CO_2 from these habitats onto the outershelf and into the open ocean. Coastal marine plants such as mangroves and salt marsh vegetation absorb CO_2 directly from the atmosphere and then release carbon into the coastal waters through losses from roots and the degradation of leaf litter and other plant debris. Tides then transport this carbon away from the coast. Accurate global estimates of the tidal carbon pump in coastal habitats are not yet achievable, but several studies suggest that this is a significant mechanism for removing CO_2 from the atmosphere (Jahnke 2008).

Note that other coastal marine habitats such kelp forests and coral reefs have comparatively negligible carbon sediment burial rates. As noted above, however, kelp plants contain carbon stocks comparable with terrestrial habitats and debris from all coastal ecosystems is taken up by the tidal pump for long term sequestration.

Greenhouse Gas Impact of Coastal Habitat Loss

The capacity of marine habitats as carbon reservoirs and in the longterm sequestration of carbon in sediments has been dramatically eroded by the extensive areal losses of these systems. Almost 20% or 36,000 km^2 of mangroves have been lost since 1980. From 2000-2005 mangroves were lost at approximately 118 km^2 per year (FAO 2007). Seagrasses have been disappearing at a rate of 110 km^2 per year since 1980 and this rate is accelerating. In total, 29% (51,000 km^2) of the total known areal extent of seagrasses have disappeared (Waycott et al. 2009). Similar rates of salt marsh loss have been estimated. The most significant causes of these coastal habitat losses are conversion of coastal and nearshore areas for aquaculture, agriculture, infrastructure and tourism. The extent of coastal marine habitat loss is a large fraction of the global total areas of these systems. The potential loss of these carbon reservoirs should be accounted for by coastal protection and management.

The size of coastal marine habitats, however, are small when compared to terrestrial deforestation and so the associated global loss of carbon reservoirs is similarly relatively modest. For instance, the annual deforestation of the tropical Amazon forest (total area 600,000km^2) is estimated to have been 13,000 km^2 in 2006. The carbon reservoirs lost through tropical or temperate deforestation are therefore very significantly greater than in coastal marine areas (approximately 150 x 10^{12} gC for the Amazon annual deforestation rate, compared to 0.9 x 10^{12} gC and 0.02 x 10^{12} gC for the global annual loss of mangroves and seagrasses respectively).

However, despite the relatively small areas, the capacity of coastal vegetated habitats for longterm carbon sequestration is comparable to terrestrial forests. This has not been accounted for in assessments of the cost of degradation and loss of coastal marine habitats. This very significant global impact of the coastal habitat loss is demonstrated by calculating the areas of terrestrial forest with equivalent sediment carbon sequestration capacity (see Table 2). For example, the total annual loss of mangroves and seagrasses has the longterm carbon sequestration capacity of a tropical forest area similar to the annual deforestation rate in the Amazon. The total carbon sequestration capacity lost through mangrove and seagrass clearing is equivalent to the sediment sequestration capacity of a tropical forest area greater than the Amazon forest. Since reducing carbon emissions will be a global concern for centuries, longterm carbon sequestration capacity must now also be accounted for in the benefits associated with coastal marine habitat restoration and protection.

Multiple Benefits of Coastal Habitat Protection and Restoration

In addition to providing extensive longterm carbon sequestration benefits, coastal habitats are the source of numerous valuable ecosystem services. Mangroves are extensively used traditionally and commercially worldwide, particularly in developing countries, and have been valued at 200,000-900,000 USD ha-1 (UNEP-WCMC, 2006). Seagrasses provide important ecosystem services including nutrient cycling, enhancement of coral reef fish productivity, and habitat for fish, mammal, bird and invertebrate species. In addition, seagrasses support subsistence and commercial fisheries worth as much as $3500 ha-1 yr-1 (Waycott et al. 2009). Tidal salt marshes are important for their nutrient cycling and sediment stabilization of near coastal areas.

Corals and kelp habitats are essential components of the coastal environment, providing their own extensive range of ecosystem services (Moberg & Folke 1999, Steneck et al. 2002). These habitats are also critical to the longterm survival of mangroves and seagrasses by providing habitat and food sources for species common to numerous coastal ecosystems. All coastal habitats are therefore critical either directly or indirectly for the high rates of carbon sequestration in coastal areas.

Increasing emphasis is now also being placed on the role of coastal habitats in climate change adaptation, both for human communities and marine species. Increased coastal protection and stability will be needed in response to sea level rise and the changing storm conditions expected as a result of climate change. Under appropriate conditions, tidal salt marshes, mangroves and coral reefs provide protection from waves, storm events, can reduce shoreline erosion and provide sediment stabilization along many coasts. The food resources provided by coastal marine ecosystems will essential to maintaining human adaptive capacity to changing resource availability.

Protecting and restoring coastal marine ecosystems therefore has significant multiple benefits that are global (longterm carbon sequestration) to local (community fisheries) in scale. Coastal protection, management and development decisions therefore should account for all of these services provided by coastal habitats specifically including the longterm carbon sequestration.

	Mangroves	**Seagrasses**
Annual average global loss (km^2/year)	118	110
Equivalent tropical forest loss (km^2/year)	6600	3600
Equivalent temperate forest loss (km^2/year)	1400	770
Total estimated global loss (km^2)	36,000	51,000
Equivalent tropical forest loss (km^2)	2,000,000	1,700,000
Equivalent temperate forest loss (km^2)	430,000	350,000

Table 2: Annual and total loss of mangrove and Seagrass habitat (FAO 2007, Waycott et al. 2009) and the equivalent areas of tropical and temperate terrestrial forest needed for longterm carbon sequestration in sediments (calculated from longterm rate of carbon accumulation in soils in Table 1).

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Next steps for the Management of Coastal Carbon Sinks

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This report provides a strong new evidence base on the role of selected coastal marine habitats as carbon sinks. There is now an urgent need to take the next step - to turn such knowledge into action – by ensuring that such coastal marine sinks are included in National Inventory Submissions.

Those countries who have signed the United Nations Framework Convention on Climate Change (UNFCCC) have to make annual National Inventory Submissions (NIS) which records their Green House Gas (GHG) emissions from energy use, industrial processes, agriculture, land use and waste as well as any sequestration from land use and forestry. These national inventories have to be submitted annually to the UNFCCC and be based on guidance from the Intergovernmental Panel on Climate Change (IPCC). They are used to assess compliance with international treaties to reduce emissions (ie Kyoto, EU) and for any national commitments (ie Climate Change Act for UK). Land Use, Land Use Change & Forestry (LULUCF) is the section in the national inventories that accounts for emissions and sequestration from the management of *terrestrial* carbon sinks. The types of activities covered by LULUCF include afforestation, reforestation & deforestation, changes to soil carbon stocks from land use and land use change, peatland extraction and drainage, liming of soils, etc.

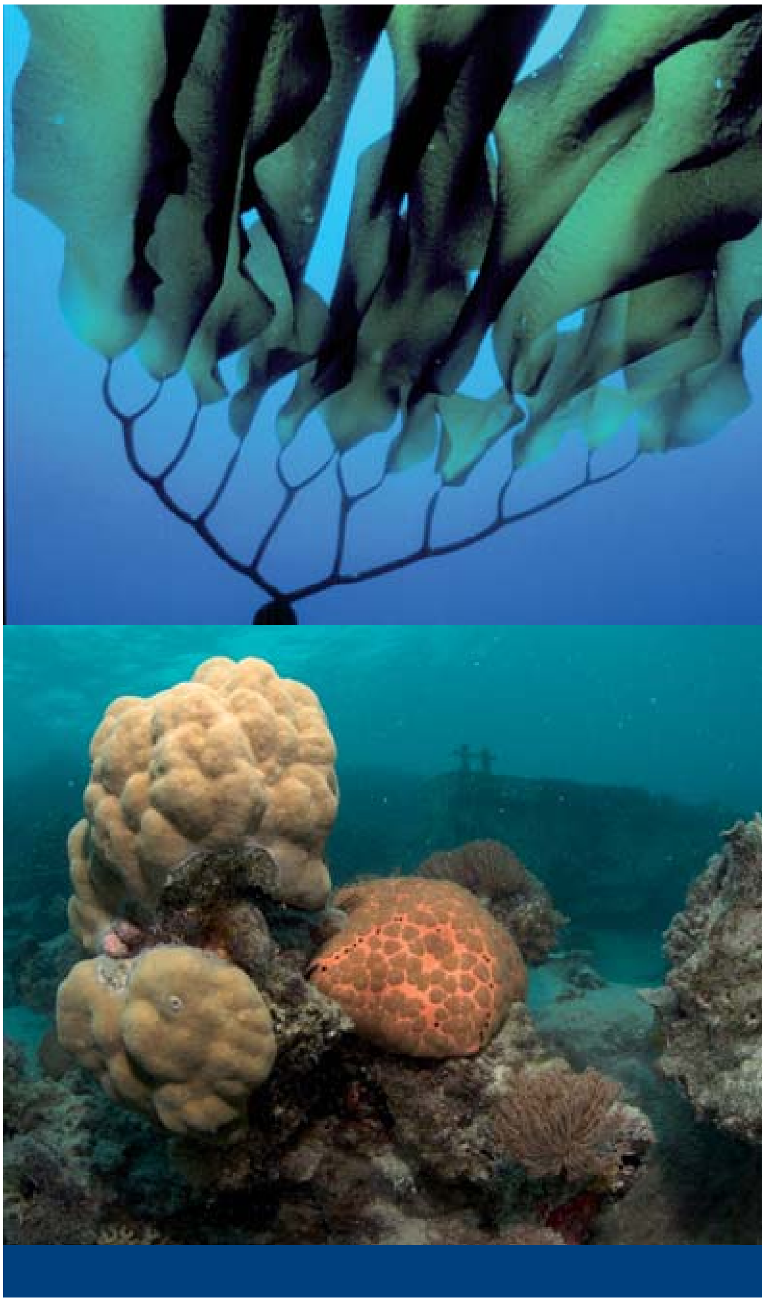
For the LULUCF section of NIS, only GHG emissions and sequestration that occur *as a direct result of human activity can be counted.* Any natural sequestration (or emissions) from unmanaged/pristine habitats cannot count towards a countries' GHG commitments. Carbon credits cannot be earned for sequestration from unmanaged habitats. GHG emissions and sequestration that occur as a result of the management of coastal and marine habitats are currently *not accounted for by LULUCF* and for that reason are *not included in international climate change mechanisms* (ie UNFCCC, Kyoto, CDM, etc) and are *not included, for example, in the UK's carbon budgets.*

To get coastal/marine habitats included in LULUCF would require the IPCC to update their guidance and possibly even need the agreement of the UNFCCC. The IPCC would need to be convinced that there is enough of a robust evidence base to demonstrate that the degradation of coastal and marine habitats due to direct human activity results in GHG emissions. They would also need to be confident that restoration (or creation) of coastal habitats will reduce those emissions and deliver sequestration.

An essential step to including coastal marine sinks in NISs will be to build on the evidence base provided in this report. In particular we need to know that coastal marine habitats are not just important as global carbon sinks but what happens (from a GHG perspective) when any of these habitats are damaged, developed or lost? The logical conclusion is that anthropogenic activities cause the carbon to be lost back to the atmosphere, but do they lose their stored carbon and if so where to? Does it result in other GHG emissions and if so what type and on what scale?

Processes are slightly ahead when considering some of the terrestrial sinks. For example, with peatlands it is known that they are an important carbon store and that they sequester carbon when in a pristine state. There is fairly good evidence that drainage, cultivation and over-grazing/burning results in carbon losses and that restoration stops those losses (although may increase methane) and, possibly, re-starts sequestration. Unfortunately the UK LULUCF inventory does not fully record these carbon losses and so does not recognise the carbon savings delivered by restoration. There is therefore now common cause across existing terrestrial carbon sinks and coastal marine carbon sinks to both improve the accuracy of the inventory process, so that the full carbon emissions from degraded habitats are recorded in countries' NISs, and to improve the evidence base to quantify the carbon savings from restoration.

This report already enables us to better recognise and acknowledge the role of coastal marine carbon sinks as a critical missing part of climate change mitigation activities. If an ultimate aim, however, is to explore the scope for engaging with the carbon market, then international actions to include such habitats in national inventories, improving their protection and management, and improving the evidence base on quantifying the savings from restoration is where efforts should now be focussed.



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