NICHOLAS INSTITUTE REPORT

Green Payments for Blue Carbon
Economic Incentives for Protecting Threatened Coastal Habitats

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Executive Summary

Coastal habitats worldwide are under increasing threat of destruction through human activities such as farming, aquaculture, wood harvest, fishing, tourism, marine operations, and real estate development. This loss of habitat carries with it the loss of critical functions that coastal ecosystems provide: support of marine and terrestrial species, retention of shorelines, water quality, and scenic beauty, to name a few. These losses are large from an ecological standpoint, but they are economically significant as well. Because markets do not easily capture the values of ecosystem services, those who control coastal resources often do not consider these values when choosing whether to clear habitat to produce goods that can be sold in the marketplace. This market failure leads to excessive habitat destruction. As a result, scientists, policymakers, and other concerned parties are seeking ways to change economic incentives to correct the problem.

One possibility for changing the economic calculus is to connect coastal ecosystems monetarily to the role they play in the global carbon cycle and the climate system. In many cases, coastal habitats store substantial amounts of carbon that can be released as carbon dioxide upon disturbance, thereby becoming a source of greenhouse gas (GHG) emissions. Global efforts to reduce GHG emissions, principally emission trading systems or "carbon markets," create a potentially large economic incentive to convince the holders of coastal ecosystems to avoid habitat conversion and thus lessen the likelihood that ecosystems will change from GHG sinks to sources.

A critical question is whether monetary payments for *blue carbon*—carbon captured and stored by coastal marine and wetland ecosystems—can alter economic incentives to favor protection of coastal habitats such as mangroves, seagrass meadows, and salt marshes. This idea is analogous to payments for REDD+ (reduced emissions from deforestation and degradation), an instrument of global climate policy that aims to curtail forest clearing, especially in the tropics. Like payments for REDD+, incentives to retain rather than emit blue carbon would preserve biodiversity as well as a variety of other ecosystem services at local and regional scales.

This report produces a first-order assessment of whether payments for blue carbon protection—money received for carbon emissions avoided by *not* converting coastal ecosystems—can provide economic incentives strong enough to substantially curtail existing rates of habitat loss. This report answers these questions first at a broad level, with a global view of current conditions, threats, and opportunities. It then focuses on the economic prospects for mangrove protection, which our initial assessment suggests may have the highest potential from biophysical, economic, and policy-readiness perspectives.

Coastal habitats as carbon sinks

In this study, we focus on the three coastal habitats that appear to possess the greatest GHG mitigation potential. Seagrass meadows are submerged ecosystems found from cold polar waters to the tropics. The other two habitats, salt marshes and mangroves, are intertidal systems. The former are most abundant in the temperate zone; the latter are confined to tropical and sub-tropical areas. Combined, these habitats are thought to cover approximately 50–80 million hectares. With soils that range in depth from less than one meter to over ten meters, intact coastal habitats store hundreds to thousands of tonnes (one tonne equals one metric ton)[1] of carbon beneath each hectare.

Soil organic carbon is by far the biggest carbon pool for the focal coastal habitats. In the first meter of sediments alone, soil organic carbon averages 500 t CO_2e/ha (tonnes of carbon dioxide equivalent per hectare) for seagrasses, 917 t CO_2e/ha for salt marshes, 1,060 t CO_2e/ha for estuarine mangroves, and nearly 1,800 t CO_2e/ha for oceanic mangroves.[2] In relative terms, about 95% to 99% of total carbon stocks of salt marshes and seagrasses are stored in the soils beneath them. In mangrove systems, 50% to 90% of the total carbon stock is in the soil carbon pool. The rest is in living biomass, such as woody vegetation.

Even though the total land area of mangroves, coastal marshes, and seagrasses is small compared with land in agriculture or forests, the carbon beneath these habitats is substantial. If released to the atmosphere, the carbon stored in a typical hectare of mangroves could contribute as much to GHG emissions as three to five hectares of tropical forest. A hectare of intact coastal marsh may contain carbon with a climate impact equivalent to 488 cars on U.S. roads each year. Even a hectare of seagrass meadow, with its small living biomass, may hold as much carbon as one to two hectares of typical temperate forest.

1. 1 tonne (t) = 1 metric ton = 1 megagram (Mg) = 1,000 kg. The abbreviations *Mt* and *Gt* refer to the megatonne (1 million tonnes) and the gigatonne (1 billion tonnes), respectively.
2. These values are global mean estimates for seagrass and salt marsh ecosystems, whereas they are weighted averages for each type of mangrove across four global regions (tropical Americas, tropical Asia, tropical Africa, and the subtropics).

Blue carbon at risk

Coastal habitats are being destroyed steadily; estimated annual loss rates are roughly 0.7% to 2%. On the basis of these rates, we estimate that between 340,000 and 980,000 hectares of these endangered habitats may be lost each year. Although the original extent of the focal habitats is unknown, cumulative losses of seagrasses, salt marshes, and mangroves have been estimated at 29%, at least 35%, and up to 67%, respectively, of their historic area. The main causes of habitat conversion vary around the world and include aquaculture, agriculture, forest exploitation, and industrial and urban development.

For centuries, salt and other tidal marshes have been diked and drained, primarily for agricultural or salt production. In the process, carbon from the previously inundated soils is directly exposed to oxygen and subsequently emitted as carbon dioxide into the atmosphere, where it can persist for decades. Although significantly slowed by environmental laws in developed countries, salt marsh conversion continues unabated in other settings.

The conversion of mangrove forests to agriculture or aquaculture has been widespread. In the case of shrimp farming, substantial carbon emissions are released through the excavation of mangrove soils to depths of about one meter. As with drained marshes, the carbon in these mangrove soils oxidizes following disturbance and continues to do so for decades following habitat conversion. Even when mangroves are exploited for forestry, and not excavated, the death of living mangrove biomass can lead to soil erosion, subsequent releases of carbon into the water column, and ultimately releases of carbon into the atmosphere.

Water quality impairment, generally from excess nutrients or sediments from terrestrial sources, is a leading cause of seagrass habitat destruction. Direct impacts, such as dredging (for example, for harbor creation and channel deepening), trawling, and anchoring also take their toll on seagrass beds. Seagrass death can lead to the erosion of submerged carbon-rich soils into the water column, which in turns allows the oxidation and release of carbon dioxide into the atmosphere.

With regard to carbon stocks at risk, we make a first-order assumption that the first meter of soil is disturbed when coastal habitats are converted or damaged. On the basis of the rough estimates of global conversion rates and those of carbon loss, we find that the annual mitigation potential across the three habitat types is roughly between 300 and 900 million t CO_2e, approximately equal to the annual CO_2 emissions from energy and industry for Poland and for Germany, respectively. Potential emissions from mangrove habitat loss comprise over half of that total mitigation.

The estimates found in this analysis are far larger than the few published studies that have assessed the climate impact of the destruction of coastal habitats. These studies have put this impact at 76 million t CO_2e per year for mangroves and salt marshes and at 3.4 million t CO_2e/year for mangroves and seagrasses. This research focused only on the very small sequestration flux that is lost when the ecosystem is destroyed. We estimate, for the first time, the substantially greater emissions released from the pools of previously sequestered carbon stored in the biomass and soil of seagrass, salt marsh, and mangrove ecosystems.

According to carbon-loss analyses, the majority of biomass and soil carbon in the top meter of soil is emitted in the years and decades following conversion of the focal ecosystems. As a marine ecosystem, the carbon release proceeds most quickly for disturbed seagrass meadows. The time path release of soil carbon is similar for salt marshes and mangroves. Typically, arboreal systems, mangrove biomass is also considerable, and the bulk of it is emitted immediately when burned.

Policy incentives to keep blue carbon out of the atmosphere

As is the case with many types of carbon emissions, the costs of keeping carbon out of the atmosphere would be shouldered by a limited number of parties who would pay the cost of preventing habitat loss (the opportunity cost of conversion to marketable uses and the costs of habitat protection). Around the globe, coastal habitats are lost because of market forces that give landowners an incentive to convert habitat to other uses. Elsewhere, habitats are lost because governments have been unwilling or unable to enforce environmental regulations and other measures that would help guarantee the continued ecological sustainability of habitats.

The absence of mechanisms to pay landowners, managers, or governments to protect the carbon stored in coastal habitats greatly undermines incentives to protect the habitats. The cost of habitat protection can be high and would include the costs of creating and managing protected areas and of improving water quality and, in particular, the opportunity costs of forgone alternative uses (for example, aquaculture and development). With increasing world population and per capita consumption and without a mechanism to pay land managers for the carbon value of habitat protection, the pace of habitat destruction is likely to continue.

Economic incentive mechanisms for GHG emission reductions could help remedy this situation. One such mechanism, emissions trading and the creation of "carbon" markets, has been operating throughout the world since the adoption of the Kyoto Protocol by the United Nations Framework Convention on Climate Change, but the role for terrestrial (for example, forest) carbon reductions is very limited and that for carbon in coastal habitats is nonexistent. Recent efforts appear to be creating a global market opportunity for reduced emissions from deforestation and degradation (called "REDD+" in the global climate policy discussions). Although mangrove protection could conceivably be included in REDD+, it is unclear whether REDD+ or other proposed protocols would include carbon in coastal habitats, especially in soils, an omission that could result in failure to protect some of the largest and most vulnerable carbon stocks on Earth.

Carbon revenue potential and costs

Mechanisms that establish payments for blue carbon protection could value avoidance of carbon emissions from habitat conversion, potentially altering economic incentives and inducing those with *de facto* control of resources to forgo conversion. Assuming that carbon prices of \$0 to \$30 t CO_2e could be applicable to blue carbon in the future, we find that gross financial returns to avoided habitat-conversion projects fall anywhere between \$0 and \$37,000 per hectare of protected habitat. Mangroves are by far the coastal ecosystem with the greatest blue carbon value; at a carbon price of \$15/t CO_2e, the average gross returns are over \$18,000/ha for oceanic ecosystems and over \$13,000/ha for estuarine ecosystems. Average gross returns for salt marshes are nearly \$8,000/ha, slightly higher than those for seagrass meadows. Oceanic mangroves have greater blue carbon values than estuarine mangroves in all regions due to greater carbon density in the top meter of soil. Oceanic mangroves in tropical Africa and tropical Asia have the highest gross returns to avoided habitat-conversion projects.

The costs of avoiding habitat conversion need to be considered to fully evaluate the economic potential of blue carbon payments to protect habitat. They may include the costs of establishing and managing a protected area, whether it be terrestrial or marine, as well as the opportunity costs of protection, which represent the most profitable alternative (converted) use of a hectare of habitat and which are generally the largest component of protection costs. These costs of protection vary greatly across countries and regions. On average, costs are highest for protection of salt marshes, because they tend to be found in temperate, developed countries. The study finds illustrative cases of when carbon revenue potential would appear to outweigh protection costs.

Economic potential for mangrove protection: Key regions and countries

Due to data limitations, country-level analyses of the net economic returns to blue carbon investments could only be performed for mangroves. Assuming the upper bound of costs, these returns are negative in about 40% to 50% of mangrove countries at a \$5/t CO_2e, but about 15% or less at \$15/t CO_2e. Although a blue carbon value potential of less than \$10,000/ha is common among countries at low carbon prices, the majority of countries have returns that are above \$10,000/ha; these returns start at \$15/t CO_2e for oceanic mangroves and at \$20/t CO_2e for estuarine mangroves. Net returns between \$20,000/ha and \$30,000/ha are found for oceanic mangroves in 82% and 94% of countries at prices of \$25/t CO_2e and \$30/t CO_2e, respectively.

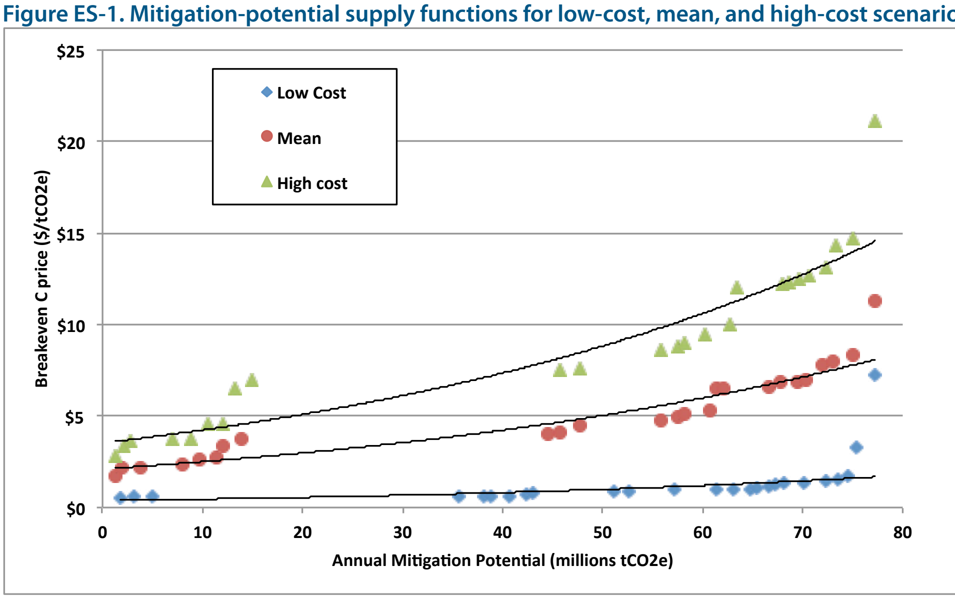
Investors in blue carbon emission reductions will likely focus their efforts on countries that appear to have the least expensive reductions, though countries with large-scale mitigation potential may have lower transaction costs. Of the top 25 countries in terms of mitigation potential, Senegal is found to have the cheapest reductions (break-even cost of less than \$2 per tonne) and Columbia, the most expensive (greater than \$11 per tonne). Five of the top seven countries with the lowest break-even prices are in tropical Africa. In addition to having the highest gross returns for blue carbon, Africa enjoys the lowest costs of protection worldwide. At \$15/t CO_2e, every tropical African country with mangroves, except one, demonstrates net returns of over \$10,000 per hectare.

With respect to mitigation potential, tropical Asia is clearly the most prominent region for mangrove protection. Four of the top five countries with the highest biophysical mitigation potential—Indonesia, Malaysia, Papua New Guinea, and Vietnam—are in southeast Asia, and together they constitute about half of global potential. That potential is driven mainly by Indonesia, whose annual mitigation potential is about one-third the worldwide total. The average break-even price for Indonesia is about $4 per tonne of emissions avoided, which is eighth lowest of all country-specific estimates.

We combine the break-even prices and mitigation quantities to construct an emission reduction supply curve for avoided mangrove destruction (Figure ES-1), which roughly captures the quantity of mitigation that countries could provide if they were compensated at levels sufficient to cover their habitat protection costs. Assuming mean cost levels, we see that at a carbon price of $4/t CO_2e, about 40 million tonnes of mitigation might be supplied. At $6/t CO_2e, another 20 million tonnes might be achieved, pushing the total to 60 million tonnes. At $8/t CO_2e, the total supplied could approach 80 million tonnes. Under the high-cost scenario, only 10 million tonnes and 45 million tonnes t CO_2e are supplied at carbon prices of $4 and $8, respectively. However, under the low-cost scenario, a carbon price of merely $2/t CO_2e is necessary to secure 80 million tonnes of mitigation.

In many ways, payments for blue carbon are simply a means to reward habitat protection, particularly for fragile and threatened coastal ecosystems such as mangroves. At $4–$8/t CO_2e, the mean case would supply 400,000 to 800,000 hectares of protected mangrove habitat per year, whereas the high-cost case would yield 100,000 to 450,000 hectares. Overall, if blue carbon offsets were to gain value, a carbon price of $15/t CO_2e might not be unreasonable and would be sufficient to protect all mangrove habitat considered here, even assuming high protection costs. However, implementing protection activities involves transaction costs (measurement, monitoring, accounting, and distribution) that, although not monetized in this study, must eventually be factored in.

Figure ES-1. Mitigation-potential supply functions for low-cost, mean, and high-cost scenarios.



Green Payments for Blue Carbon

Economic Incentives for Protecting Threatened Coastal Habitats

1. Introduction

Coastal habitats worldwide are under increasing threat of destruction through human activities such as farming, aquaculture, wood harvest, fishing, tourism, marine operations, and real estate development. This loss of habitat carries with it the loss of critical functions that coastal ecosystems provide: support of marine and terrestrial species, retention of shorelines, water quality, and scenic beauty, to name a few. These losses are large from an ecological standpoint, but they are economically significant as well (Barbier 2007). Because markets do not easily capture the values of ecosystem services, those who control coastal resources often do not consider these values when choosing whether to clear habitat to produce goods that can be sold in the marketplace. This market failure leads to excessive habitat destruction. As a result, scientists, policymakers, and other concerned parties are seeking ways to change economic incentives to correct the problem.

One possibility for changing the economic calculus is to connect coastal ecosystems monetarily to the role they play in the global carbon cycle and the climate system. In many cases, coastal habitats store substantial amounts of carbon that can be released as carbon dioxide upon disturbance, thereby becoming a source of greenhouse gas (GHG) emissions.[3] Global efforts to reduce GHG emissions, principally emission trading systems or "carbon markets," create a potentially large economic incentive to convince the holders of coastal ecosystems to avoid habitat conversion and thus lessen the likelihood that ecosystems will change from GHG sinks to sources.

A critical question is whether monetary payments for *blue carbon*—carbon captured and stored by coastal marine and wetland ecosystems—can alter economic incentives to favor protection of coastal habitats such as mangroves, seagrass meadows, and salt marshes. This idea is analogous to payments for REDD+ (reduced emissions from deforestation and degradation), an instrument of global climate policy that aims to curtail forest clearing, especially in the tropics. Like payments for REDD+, incentives to retain rather than emit blue carbon would preserve biodiversity as well as a variety of other ecosystem services at local and regional scales. Success will hinge on structuring the incentives to avoid negative impacts on the well-being of local populations who depend on these resources for their livelihoods.

This report produces a first-order assessment of whether payments for blue carbon protection—money received for carbon emissions avoided by *not* converting coastal ecosystems—can provide economic incentives strong enough to substantially curtail existing rates of habitat loss. We pursue this task by asking and answering several questions:

- What coastal habitats are being lost, and where?
- How much carbon is stored in these ecosystems, and how much is at risk?
- What is the biophysical potential for mitigating these losses?
- What is the nature and range of payments that might be available for avoided blue carbon emissions?
- What are the costs associated with measures needed to avoid coastal habitat conversions?
- Given the benefits and costs of these measures, what is the economic potential to avoid these carbon emissions and corresponding loss of habitat?
- What types of habitats are most economically suited for protection through blue carbon payments, and where are they located?

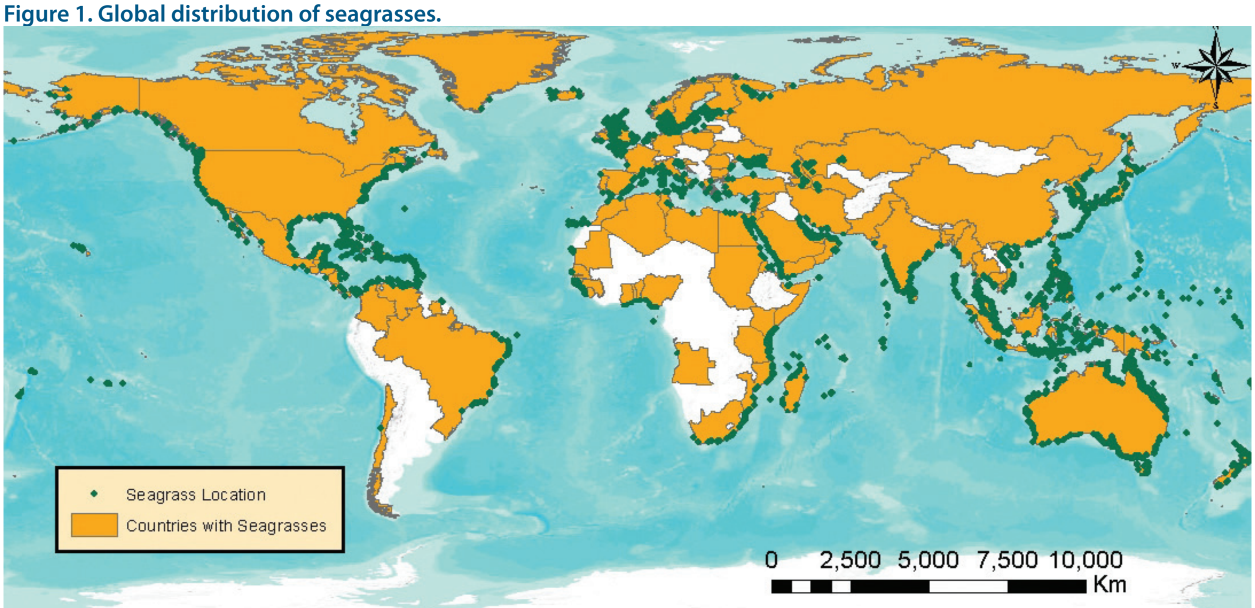
This report answers these questions first at a broad level, with a global view of current conditions, threats, and opportunities. It then focuses on the economic prospects for mangrove protection, which our initial assessment suggests may have the highest potential from biophysical, economic, and policy-readiness perspectives.

3. The focus here is on carbon dioxide because methane, a more potent GHG with 25 times the global warming potential of carbon dioxide, is emitted in relatively small quantities in these saline habitats due to the presence of sulfates. Moving down the salinity gradient from saltwater to freshwater, methane emissions gradually increase and can be substantial in freshwater wetland systems.

2. What and Where Is Blue Carbon?

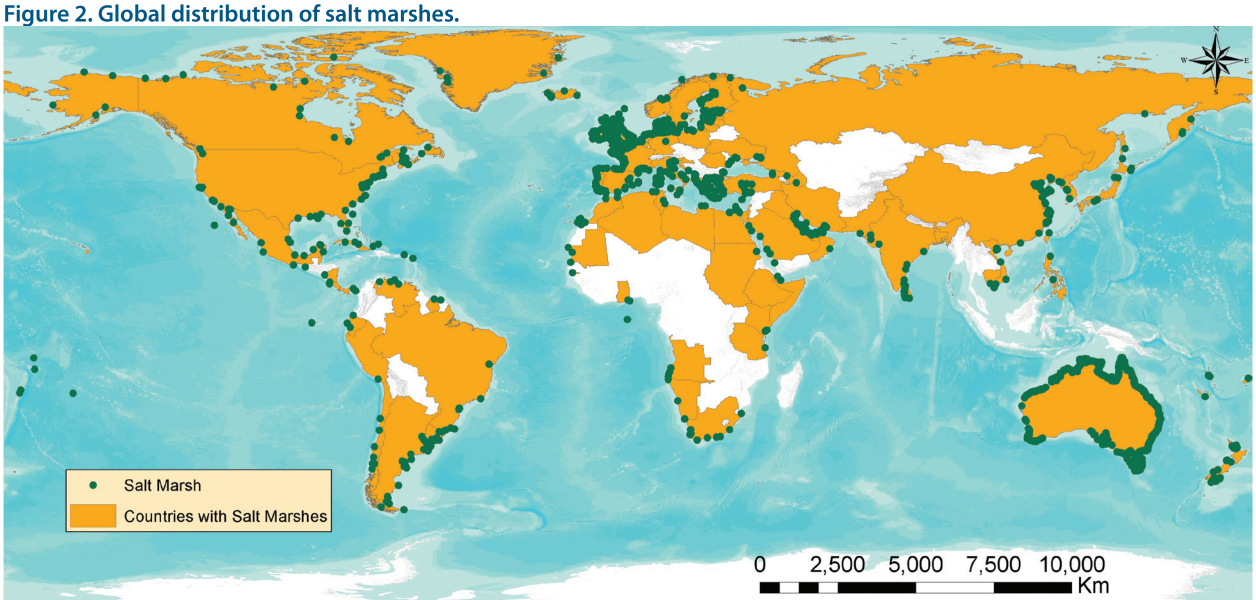
This report examines blue carbon stored in three coastal habitats: seagrass meadows, salt marshes, and mangroves, which are thought to be the largest repositories of carbon in coastal ecosystems. Seagrass meadows are communities of underwater-flowering plants found in coastal waters of all continents except Antarctica (Figure 1). More than 60 seagrass species are known to exist, and as many as 10 to 13 of them may co-occur in tropical sites. Salt marshes are intertidal ecosystems occurring on sheltered coastlines ranging from the sub-arctic to the tropics, though most extensively in temperate zones (Figure 2). They are dominated by vascular flowering plants, such as perennial grasses, but are also vegetated by primary producers such as macroalgae, diatoms, and cyanobacteria. Mangroves are salt-tolerant flowering plants, predominantly arboreal, that grow in the intertidal zone of tropical and subtropical shores (Figure 3). More than 50 species are known, and they are divided into two groups: the Old World and the New World and West African mangrove swamps. The greatest species diversity is found in the Indo-West Pacific (Old World).

Figure 1. Global distribution of seagrasses.



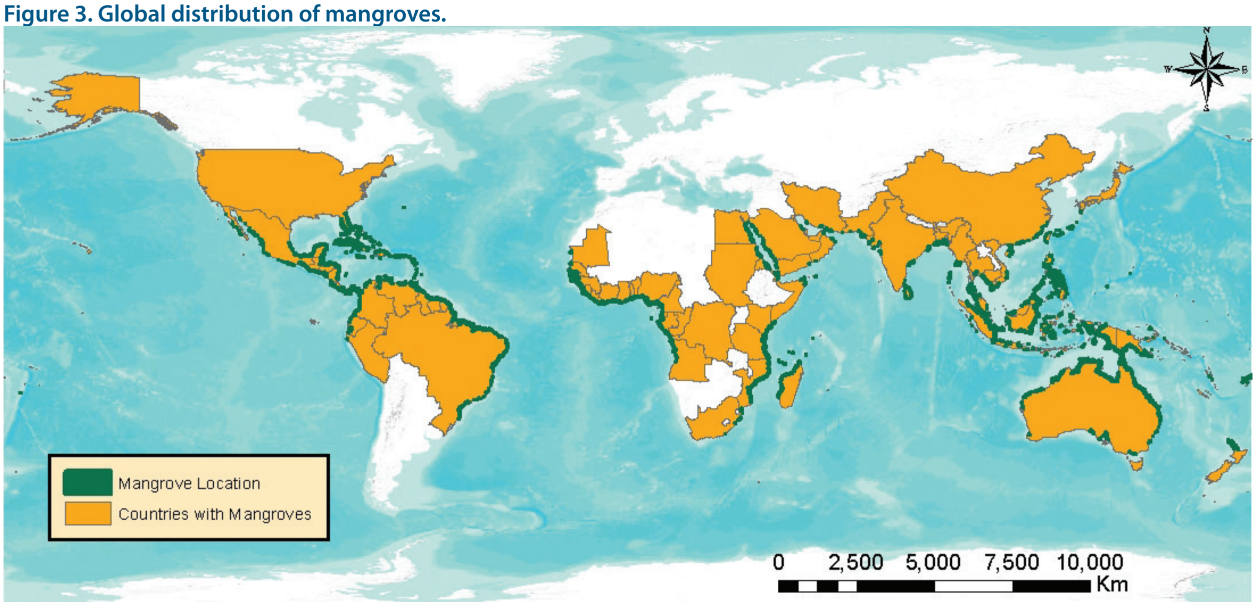
Source: Seagrasses (version 2.0) of the global polygon and point dataset compiled by UNEP World Conservation Monitoring Centre (UNEP-WCMC), 2005. For further information, e-mail spatialanalysis@unep-wcmc.org.

Figure 2. Global distribution of salt marshes.



Source: Saltmarsh (version 1.0) of the provisional global point dataset developed jointly by UNEP-WCMC and TNC. This dataset is incomplete.

Figure 3. Global distribution of mangroves.



Source: Mangroves (version 3.0) of the global polygon dataset compiled by UNEP World Conservation Monitoring Centre (UNEP-WCMC) in collaboration with the International Society for Mangrove Ecosystems (ISME), 1997. For further information, e-mail spatialanalysis@unep-wcmc.org. Mangroves of Western Central Africa raster dataset processed from Landsat imagery, circa 2000. Compiled by UNEP World Conservation Monitoring Centre (UNEP-WCMC), 2006. For further information, e-mail spatialanalysis@unep-wcmc.org. East African mangroves extracted from version 4.0 of the polygon dataset compiled by UNEP World Conservation Monitoring Centre (UNEP-WCMC), 2006. For further information, e-mail spatialanalysis@unep-wcmc.org.

3. Coastal Habitats: Extent, Location, and Rate of Loss

Seagrass meadows can be found in the shallow waters of all continents, whereas salt marshes, though also globally distributed, occur most extensively in temperate areas. Mangroves are for the most part limited to tropical and subtropical regions of the world. Salt marshes and mangroves exist in the intertidal zone between land and sea, whereas seagrasses grow in shallow water (0–45 m) on the continental shelf and may be near or far from land. Approximate at best, current estimates of the global extent of these habitats range from 13.8 million hectares (Mha) to 17 Mha for mangroves, up to 60 Mha for seagrasses, and 5.1 Mha for salt marshes[4] (see Table 1). Together these habitats cover a relatively small area, somewhere between 49 Mha and 82 Mha. Although this area amounts to only 1% to 2% of the global coverage of forest (3.95 billion hectares) (FAO 2005), these ecosystems are some of the most threatened in the world. Over the 1980–2000 period, mangroves experienced annual loss rates of 0.7% to 2.1% per year. These rates were driven mainly by agriculture, aquaculture, and wood harvests. Salt marshes have historically been reclaimed for agricultural use and salt ponds and, although loss rates in developed countries have slowed considerably (Dahl 2008), they continue to be high in the developing world due to agricultural use, industrial or urban use, and reduced sediment supply (Coleman et al. 2008; Yang et al. 2006). Overall, salt marsh loss rates may be 1% to 2% annually, though estimates are uncertain given the lack of data on areal extent. Globally, seagrass meadows are disappearing at a similarly rapid rate, at about 1.2% to 2% per year since 1980, due mainly to water quality degradation and mechanical damage, such as dredging, trawling, and anchoring.

Table 1. Coastal ecosystems: Global area and conversion rates by type.

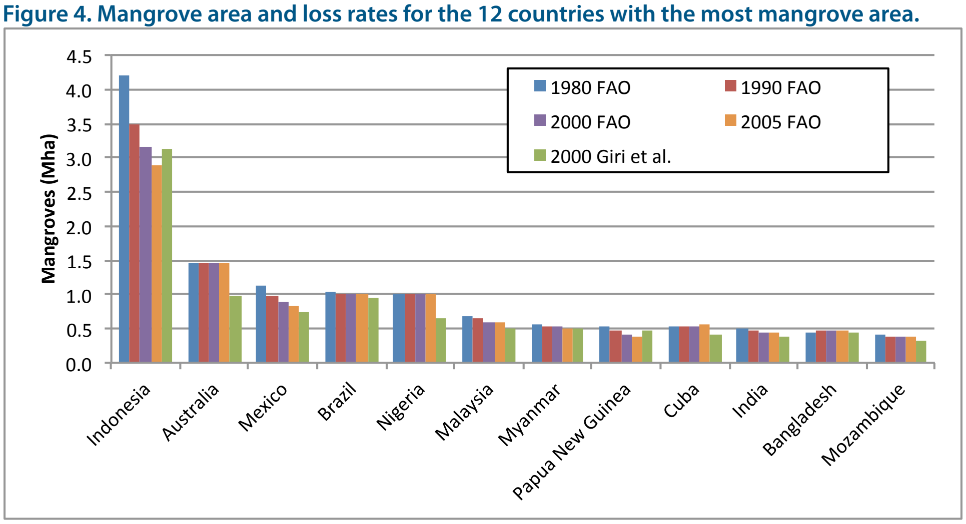
Habitat type	Global extent (Mha)	Loss drivers	Annual loss rate (~1980–2000)	Total historical loss (%)
Seagrass	30–60[a]	Water quality degradation, mechanical damage	1.2%–2%[b]	29[c]
Salt marsh	5.1[d]	Agriculture, urban and industrial development, sediment starvation	1%–2%[e]	67[f]
Mangroves	13.8–17[g]	Agriculture, aquaculture, wood harvests	0.7%–2.1%[h]	35[i]

Sources: (a) Charpy-Roubaud and Sournia 1990; Duarte et al. 2005; (b) Short and Wyllie-Echeverria 1996; adapted from Waycott et al. 2009; (c) Waycott et al. 2009; (d) Chmura et al. 2003; UNEP-WCMC and TNC 2010; (e) Adam 2002; Duarte et al. 2008; (f) Lotze et al. 2006; Gedan et al. 2009; (g) Valiela et al. 2001; FAO 2007; Giri et al. 2011; (h) Valiela et al. 2001; FAO 2007; (i) Valiela et al. 2001; Duke et al. 2007.

4. Due to the limitations of geospatial data sets, our estimates of mangrove areas include brackish estuarine environments, but our estimates of salt marsh area do not. Mangroves have a different classification system than that for marshes, so that not all estuarine mangroves are in brackish environments.

To date, the best geospatial data are for mangroves. Whereas country-level estimates are unavailable for seagrasses and salt marshes, a 2007 report from the FAO provides area estimates for all mangrove countries in the world from 1980 to 2005. The report's estimates of total global mangrove area are 18.8 Mha, 16.9 Mha, 15.74 Mha, and 15.2 Mha for 1980, 1990, 2000, and 2005, respectively. These estimates equate to annual mangrove loss rates of 0.99%, 0.70%, and 0.65% for the 1980–1990, 1990–2000, and 2000–2005 periods, respectively. In comparison, a study using satellite data has estimated that the total extent of mangroves was only 13.8 Mha in 2000 (Giri et al. 2011). Wide discrepancies between some of this study's data and the FAO data on mangrove coverage in the 12 countries that constitute about 70% of world's mangrove extent (Figure 4) indicate that additional mangrove mapping is needed. Nevertheless, all datasets indicate substantial declines in mangrove habitats worldwide.

Figure 4. Mangrove area and loss rates for the 12 countries with the most mangrove area.

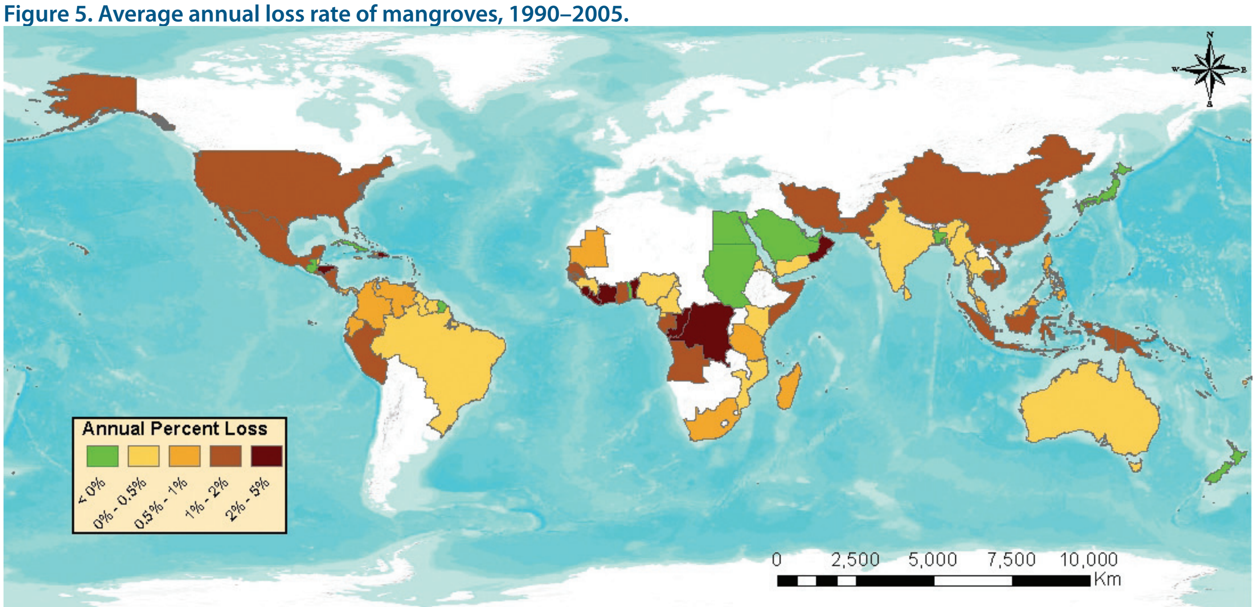


Source: UN Food and Agricultural Organization (FAO) reports, assembled by authors.

Using the mangrove data from FAO, we calculated average annual loss rates for the 1990–2005 period for all the mangrove countries of the world. The results are mapped in Figure 5. Of the 104 mangrove countries, 15 experienced habitat loss at a rate above 2% per year. These countries included Honduras, the Dominican Republic, and several countries in central and west Africa. Twenty-three countries, including mangrove-rich Indonesia, Papua New Guinea, Vietnam, and Mexico, fell into the 1% to 2% loss-rate category. Overall, the highest number of countries showed rates between 0% and 1%. During the period, mangrove area actually increased in a few countries, most importantly Cuba and Bangladesh, the ninth and eleventh most mangrove-rich countries, according to Figure 4.

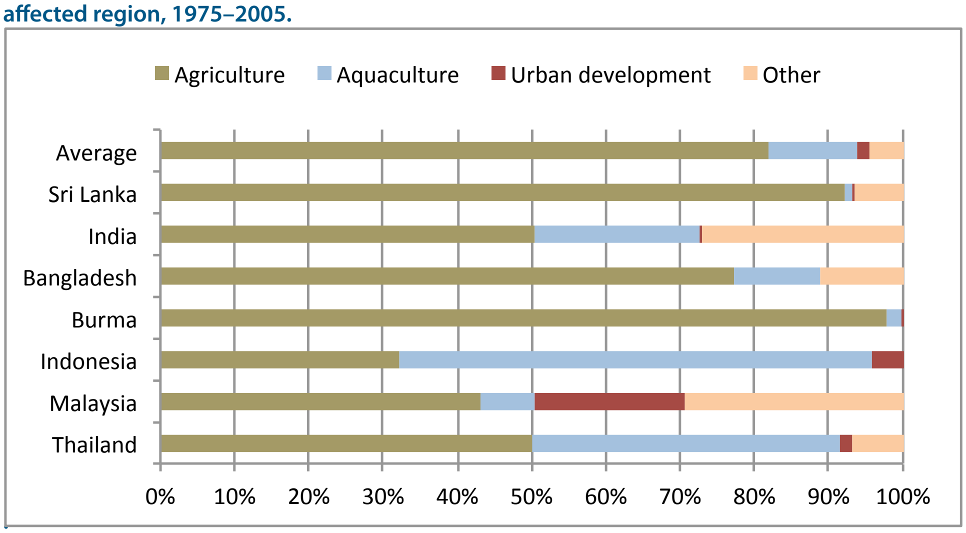
The main anthropogenic drivers of mangrove conversion are fairly well known, namely agriculture, aquaculture, wood harvests, and urban and tourism development (FAO 2007; Spaulding et al. 2010). What is not as well understood is the proportion of mangrove losses that can be attributed to each of those drivers in different parts of the world. Using data sources from the 1980s and 1990s, one study determined that global mangrove area losses are due principally to shrimp culture (38%), wood harvests (26%), fish culture (14%), diversion of freshwater (11%), land reclamation (5%), herbicides (3%), and agriculture (1%) (Valiela et al. 2001). Analyzing satellite data for the 1975–2005 period for countries and portions of countries in the Indian Ocean region affected by the 2004 tsunami, other researchers came to starkly different conclusions (see Figure 6) (Giri et al. 2008). In their study region, on average, 82% of mangrove deforestation was found to be caused by agriculture; 12% was due to aquaculture, and 2%, to urban development. Although agricultural expansion was the key driver, causes of mangrove loss varied considerably by country. For instance, 63% and 41% of habitat loss in Indonesia and Thailand, respectively, were attributed to aquaculture, whereas 20% of mangrove deforestation in Malaysia was related to urban development. This study does not elucidate whether key drivers have changed over time, so whether aquaculture has become more important than agriculture in recent years is unclear. Moreover, loss drivers in regions of the world outside the study area may have different relative contributions. Overall, drivers often vary geographically and temporally, and more research is needed to discern their relative impacts.

Figure 5. Average annual loss rate of mangroves, 1990–2005.



Source: FAO 2007.

Figure 6. Percentage of mangrove loss attributed to drivers for countries and portions of countries in the 2004 tsunami-affected region, 1975–2005.



Source: Giri et al. 2008.

4. Carbon Storage in Coastal Ecosystems

Figure 7 and Table 2 provide mean and ranges of estimates of carbon stocks and sequestration rates across the different pools and regions of the world for each of the focal habitats. Coastal ecosystems remove carbon dioxide from the atmosphere via photosynthesis, return some to the atmosphere through respiration and oxidation, and store the remaining carbon in two pools: living biomass (both aboveground and belowground vegetation) and soil organic carbon. The carbon sequestration rate quantifies how much carbon is added to the biomass and soil carbon pools annually. Because these intact ecosystems typically have mature vegetation that maintains a steady biomass, virtually all the sequestration ends up buried in the soil carbon pool.[5] This sequestration rate is assumed to be constant over time for the purposes of this paper.[6]

5. These systems store carbon, which if disturbed is returned to the atmosphere in the form of carbon dioxide (CO_2). We use CO_2 equivalent as units of measure here because of the emphasis on carbon in the climate system and the fact that the potential payment schemes discussed in the report are based on CO_2 equivalent units.
6. In the case of mangroves, carbon burial rates seem to be keeping pace with sea level rise, but if this rise were to outstrip the elevation of the sediment surface, the seaward and landward margins of the mangrove forest would retreat landward because the mangrove

Mangrove ecosystems are classified on the basis of geomorphological differences as oceanic—fringing, or occurring along the sea—or as estuarine—deltaic, or growing where river deltas meet saltwater bodies. Two other classes of mangrove—basin and dwarf—are not very abundant and therefore are not included in this analysis.

For marshes, we present estimates for *salt marshes only because of limitations in geospatial data for brackish marshes, which occur in mixed freshwater-saltwater environments and which* also are likely to be important stores of blue carbon. Given this study's focus on marine and coastal systems, *freshwater marshes* are likewise not included in the analysis.

Seagrass meadows are ecological communities whose species compositions vary within and across regions. They are reported here as one system type across their range because knowledge about the carbon pools for various seagrass assemblages is insufficient to differentiate them.

Annual carbon sequestration rates vary little across the three coastal habitats but vary greatly within each habitat type. Both marshes and mangroves average between 6 tonnes and 8 tonnes of carbon dioxide equivalent (CO_2e) per hectare per year, whereas seagrasses tend to sequester carbon at a somewhat lower rate of approximately 4 t CO_2e/ha/yr. These rates are about two to four times greater than global rates observed in mature tropical forests (1.8–2.7 t CO_2e/ha/yr) (Lewis et al. 2009). The amount of carbon held in living biomass is much more variable among the habitat types; seagrasses contain 0.4–18.3 t CO_2e per hectare, and salt marshes, on average, a few times higher than that at 12–60 t CO_2e/ha. Mangrove forests, which can grow up to 40 meters tall (Spaulding 2010), clearly lead in this area and maintain 237–563 t CO_2e per hectare in living biomass.

Table 2. Global averages and standard deviations of carbon sequestration rates and global ranges for the main carbon pools, by habitat type. Only the top meter of soil is included in the soil carbon estimates.

Habitat type	Annual carbon sequestration rate (t CO_2e/ha/yr)	Living biomass (t CO_2e/ha)	Soil organic carbon (t CO_2e/ha)
Seagrass	4.4 ± 0.95[a]	0.4–18.3[b]	66–1,467[c]
Salt marsh	8.0 ± 8.5[d]	12–60[e]	330–1,980[f]
Estuarine mangroves	6.3 ± 4.8[g]	237–563[h]	1,060[h]
Oceanic mangroves	6.3 ± 4.8[g]	237–563[h]	1,690–2,020[h]

Sources: (a) Duarte et al., in press; (b) Duarte and Chiscano 1999; N. Marba and J.W. Fourqurean, pers. comm.; (c) Duarte and Chiscano 1999; N. Marba and J.W. Fourqurean, pers. comm.; (d) Morgan and Short 2002; PWA and SAIC 2009; Yu and Chmura 2009; Brevik and Homburg 2004; Bridgham et al. 2006; Chmura et al. 2003; Choi and Wang 2001; Choi and Wang 2004; Connor et al. 2001; Craft and Richardson 1998; Duarte et al. 2005; Giani et al. 1996; Hussein et al. 2004; Johnson et al. 2007; Mudd et al. 2009; Nellemann et al. 2009; PWA and SAIC 2009; (e) Morgan and Short 2002; Bridgham et al. 2006; Yu and Chmura 2009; (f) Bridgham et al. 2006; PWA and SAIC 2009; adapted from Chmura et al. 2003; (g) Bouillion et al. 2009; Bridgham et al. 2006; Chmura et al. 2003; Duarte et al. 2005; Fujimoto et al. 1999; Jennerjahn and Ittekkot 2002; Nellemann et al. 2009; PWA and SAIC 2009; Twilley et al. 1992; (h) D.C. Donato and J.B. Kauffman, pers. comm.

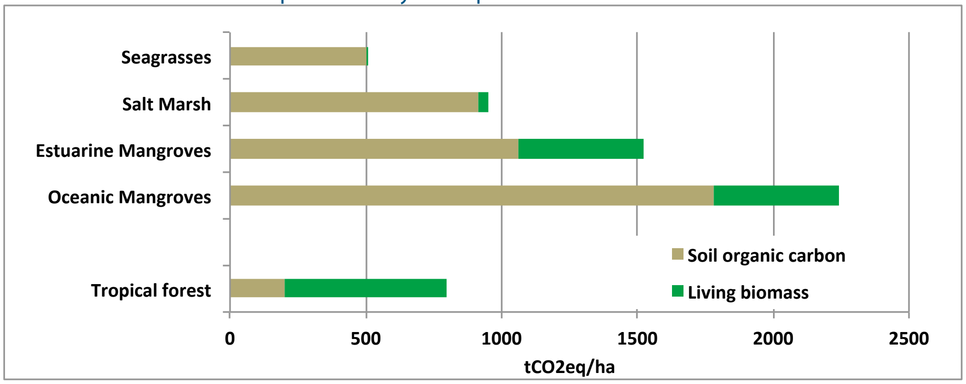
Soil organic carbon is by far the biggest carbon pool for all the focal coastal habitats. In the first meter of sediments alone, soil organic carbon averages 500 t CO_2e/ha for seagrasses, 917 t CO_2e/ha for salt marshes, 1060 t CO_2e/ha for estuarine mangroves, and nearly 1800 t CO_2e/ha for oceanic mangroves. In relative terms, about 95% to 99% of total carbon stocks of salt marshes and seagrasses are stored in the soils beneath them, while in mangrove systems, 50% to 90% of the total carbon stock is in the soil carbon pool; the rest is in living biomass. These numbers represent the carbon storage for only the first meter of soil depth in order to facilitate consistent comparisons among habitat types and in recognition of the fact that the top meter of carbon is most at risk after conversion. There can be great variability in the depth of the organic rich sediments underlying these habitats (some reach depths of several meters), yet most have at least a meter. Salt marshes can harbor up to six meters of carbon-rich deposits (Chmura 2009); the common soil depths for estuarine and oceanic mangroves are three meters or more and one to two meters, respectively. Seagrass meadows sit on top of about one meter of organic soil, on average.

Even though the total land area of mangroves, coastal marshes, and seagrasses is small compared with land in agriculture or forests, the carbon beneath these habitats is substantial. If released to the atmosphere, the carbon stored in a typical hectare of mangroves could contribute as much to GHG emissions as three to five hectares of tropical forest with non-peat soils, which are common in the Amazon and upland areas of other tropical regions (IPCC 2007; Malhi 2009). Tropical forests with peat soils, like those found in coastal lowlands in Indonesia and Malaysia, may range from 2,000–4,500 t CO_2 per hectare (with about 60% to 80% of the total stock contained in the soil)[7] and may be roughly equal to

species need to maintain their preferred hydroperiod (Alongi 2008; Gilman et al. 2008).
7. Only includes top meter of soil for consistent comparison (Murdiyarso et al. 2010).

mangroves in terms of emission potential per hectare. A hectare of intact coastal marsh may contain carbon with a climate impact equivalent to 488 cars on U.S. roads each year (U.S. EPA 2005). Even a hectare of seagrass meadow, with its small living biomass, may hold as much carbon as one to two hectares of typical temperate forest (Smith et al. 2006).

Figure 7. Global averages for carbon pools (soil organic carbon and living biomass) of focal coastal habitats. Tropical forests are included for comparison. Only the top meter of soil is included in the soil carbon estimates.



Source: Authors.

The carbon pool estimates for estuarine and oceanic mangroves presented above are global averages from four regions—tropical Americas, tropical Africa, tropical Asia, and the subtropics—and are weighted by areal extent. On average, biomass carbon is greatest in tropical Asian mangroves (563 t CO_2e/ha) and lowest in subtropical stands (237 t CO_2e/ha). Data on estuarine mangroves is limited and thus there is no estimated variation in soil organic carbon across the four regions. In the top meter of soil, oceanic mangroves in tropical Africa and the subtropics have, on average, about 20% more carbon than those in the tropical Americas and tropical Asia.

5. Biophysical Mitigation Potential for Coastal Habitats

Two phenomena occur after coastal ecosystems are disturbed or converted to an alternative use: the sequestration process that removes carbon dioxide from the atmosphere terminates (to the extent that vegetation is killed), and the carbon stored on site begins to be released back into the atmosphere as carbon dioxide. In this section, we focus on the amount of stored carbon that is at risk of being released, without considering the timing of its release. We examine the timing of the release of carbon stocks as well as the termination of the sequestration process in Section 6.

As noted above, the coastal ecosystems store substantial amounts of carbon, primarily in their soils. But simply summing up all the carbon stored in these systems would overestimate the stock of carbon at risk of release. To estimate just that portion of the stock, we must determine what portion of the habitats is at risk for conversion and, if converted, what portion of the carbon stock in the converted area is at risk of release. This calculation provides an estimate of the biophysical mitigation potential, which is the tonnes of carbon dioxide equivalents whose release could be avoided through interventions such as payments for blue carbon.

With regard to carbon stocks at risk, we make a first-order default assumption that the first meter of soil is disturbed when coastal habitats are converted or damaged. Although salt marshes and mangroves often have more than one meter of organic soil beneath them and may be affected when the habitat is converted to another land use, we assume that all carbon stored below one meter remains unreleased. Our one-meter assumption may seem conservative. But other forms of conversion—for example, agriculture that does not require deep cultivation and development (roads, buildings) that encases below-ground carbon with impermeable or semi-permeable surfaces—may put less than one meter's worth of carbon at risk of release. Thus, the estimate of the depth of the carbon at risk, which is activity- and site-specific, is highly uncertain. Local assessment should use more specific data to refine our assumption.

Regardless of the exact depth of carbon at risk, habitat conversion releases to the atmosphere previously stored carbon as carbon dioxide, from both biomass and soil, and, because the living matter is no longer active, abruptly stops annual carbon sequestration. If planted with annual crops, the soil could continue to accumulate carbon, albeit at a much slower rate. If planted with perennial crops, both soil carbon and aboveground carbon could continue to accumulate.

In Table 3, the total carbon at risk is the average amount of soil carbon held in the top meter plus the biomass. It ranges from 949 t CO_2e for salt marshes to 1762 t CO_2e for mangroves. Note that, for mangroves, we use a weighted average of carbon stocks for oceanic and estuarine mangroves (estuarine mangroves cover more land area) across the four mangrove regions. To be conservative, we use the low end of the habitat extent for seagrasses and mangroves. By multiplying the current habitat extent, we arrive at the total carbon stock at risk, which adds up to nearly 45 billion tonnes of CO_2e across all three systems of interest. For context, this total amount of carbon at risk in these habitats is approximately one and a half times the annual global emissions of carbon dioxide from all industrial emissions.[8] Annual estimated blue carbon losses, of course, are much smaller, as shown in Table 3.

Table 3. Estimates of total carbon at risk (both biomass carbon and soil organic carbon) in top meter of sediments beneath coastal habitats. Mha = million of hectares; Gt CO_2e = gigatonnes (billions of tonnes) of carbon dioxide equivalents.

Habitats	Soil organic carbon per unit area (t CO_2e/ha)	Total carbon per unit area (t CO_2e/ha)	Current habitat extent (Mha)	Total carbon stock at risk (Gt CO_2e)	LOWER annual C loss (biophysical mitigation potential) @ 0.7% rate (Gt CO_2e)	HIGHER annual C loss (biophysical mitigation potential) @ 2% rate(Gt CO_2e)
Seagrass	500	511	30	15.3	0.11	0.31
Salt marsh	917	949	5.1	4.8	0.03	0.10
Mangroves	1,298	1,762	13.8	24.3	0.17	0.49
Total			**48.9**	**44.5**	**0.31**	**0.89**

To account for the fraction of habitat currently at risk for conversion, we draw from the loss rates in the previously noted literature (see Table 1). These rates are expressed on an annual basis as it is most intuitive to focus on risk of conversion in one year. Moreover, this basis aligns with how carbon crediting would work—getting credit for a mitigating action taken at a given point in time and properly accounting for the time profile of emissions to follow (see below). We calculate lower- and upper-end annual mitigation potential by multiplying total habitat mitigation by the loss rates of 0.7% and 2% per year, respectively. The resulting values are estimates of lifetime carbon losses for each hectare of converted habitat in a year.[9] Thus, annual mitigation potential across the three habitat types is found to lie roughly between 300 million t CO_2e and 900 million t CO_2e, approximately equal to the annual CO_2 emissions due to human activity (except for land use change) for Poland and for Germany, respectively.[10] Potential emissions from mangrove loss comprise about 55% of the total.

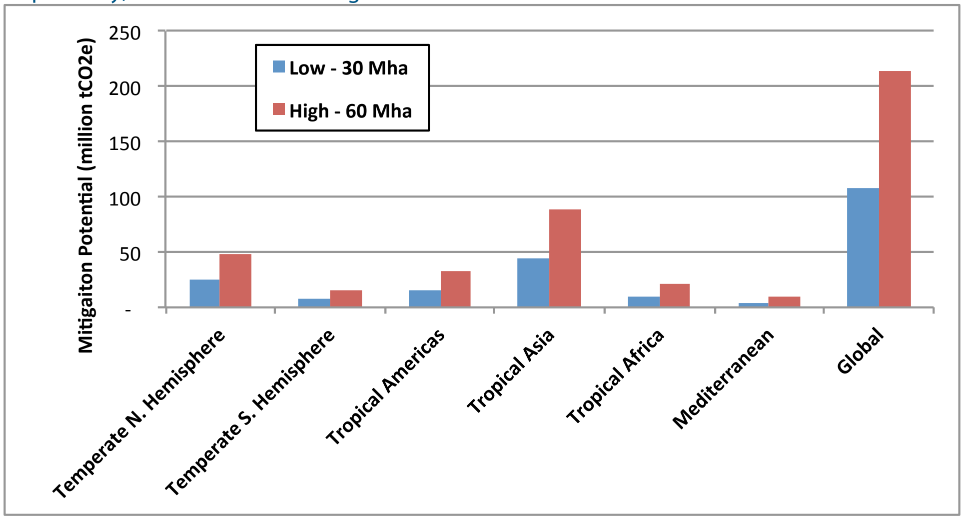
Our mitigation potential estimates are far larger than the few published estimates of the climate impact of coastal habitat destruction. Previous studies have put this impact at 76 million t CO_2e per year for mangroves and salt marshes and at 3.4 million t CO_2e/year for mangroves and seagrasses (Bridgham et al. 2006; Pidgeon 2009). These studies focused only on the small sequestration flux that is lost when the ecosystem is destroyed, whereas we estimate the substantially greater emissions released from the pools of previously sequestered carbon stored in the biomass and soil of seagrass, salt marsh, and mangrove ecosystems.

The current state-of-the-science estimate for global anthropogenic CO_2 emissions (including land use change) is about 34 billion t CO_2 for 2009 (Friedlingstein et al. 2010). The annual mitigation potential for the focal coastal habitats, at current conversion rates, is approximately 0.9% to 2.6% of that global figure. Our annual estimates for blue carbon losses are 8% to 22% of the estimated average of land-use change emissions for the 2000–2009 period (4.0 ± 2.6 billion t CO_2e/yr).

Limitations of geospatial data for coastal habitats dictate what we can say about mitigation potential for the focal coastal habitats. Salt marsh habitat is not well mapped, and precise global and regional area estimates do not yet exist. Estimates for the six global regions that seagrasses inhabit can be bracketed by the parameters in Gattuso et al. (2006) and the lower and upper bounds on total global seagrass area (30 Mha and 60 Mha, respectively). In Figure 8, we see that tropical Asia contains the greatest share (about 41%) of the biophysical mitigation potential, followed by the temperate northern hemisphere (nearly 23%).

8. For 2009, CO_2 emissions associated with fossil fuel use and cement production were 30.8 billion t CO_2e (Friedlingstein et al. 2010).
9. If conversion rates are in some sort of steady state, "annual" estimates not only capture future lifetime losses for what is converted this year, but also roughly capture what is actually going into the atmosphere from all conversions, past and present.
10. These emissions are for 2007 (Boden et al. 2010).

Figure 8. Biophysical mitigation potential (carbon stock at risk of release at a conversion rate of 0.7%) of seagrass ecosystems. Two estimates, derived from lower- and upper-bound estimates of total global seagrass area (30 Mha and 60 Mha, respectively) are shown for each region.



Source: Authors.

In contrast with the salt marsh and seagrass habitat areas, mangrove habitat area has been quantified by the UN Food and Agriculture Organization (FAO) at the country scale, allowing us to explore mitigation potential and other questions relevant to this analysis at that level of resolution. A further adjustment is made to account for different drivers of mangrove habitat loss. As reported in Figure 6, over 90% of mangrove destruction is attributable to either agriculture or aquaculture (Giri et al. 2008). In a similar vein, another analysis (Valiela et al. 2001) indicates that approximately 90% of habitat loss is due to aquaculture, wood harvests, or agriculture-related factors. Although urban development and other activities also contribute to habitat conversion, we focus on lands at risk for conversion to agriculture or aquaculture and lands overexploited for wood. Real estate development and other urban uses may imply very high land values that could make habitat protection uneconomical relative to other lands (see Section 10). Therefore, we reduce the mitigation potential of all mangrove countries by 10% to reflect the assumption that agriculture, aquaculture, or forestry cause about 90% of mangrove loss. In Figure 9, the blue bars represent the total mitigation potential, and the green bars indicate estimates that have been scaled back by 10%. Because of the country-level FAO mangrove data, we are able to calculate annual loss rates for each country. We used the 1990–2005 period, which we consider the most pertinent to the potential loss rate in the future. To our knowledge, no dataset details mangrove area for all mangrove countries after the year 2005. As Figure 9 shows, Indonesia alone accounts for about one-third of the global mitigation potential of 160 million t CO_2e per year. Mexico accounts for more than 10% of that potential, and Papua New Guinea, a little less than 10%.

Figure 9. Biophysical mitigation potential for mangrove protection by key countries. These top 12 countries for mangrove mitigation potential are sorted according to current total stocks of stored carbon and current conversion rates. Total mitigation potential is scaled back to the potential for lands at risk for conversion to agriculture or aquaculture or to overexploitation of wood harvest.

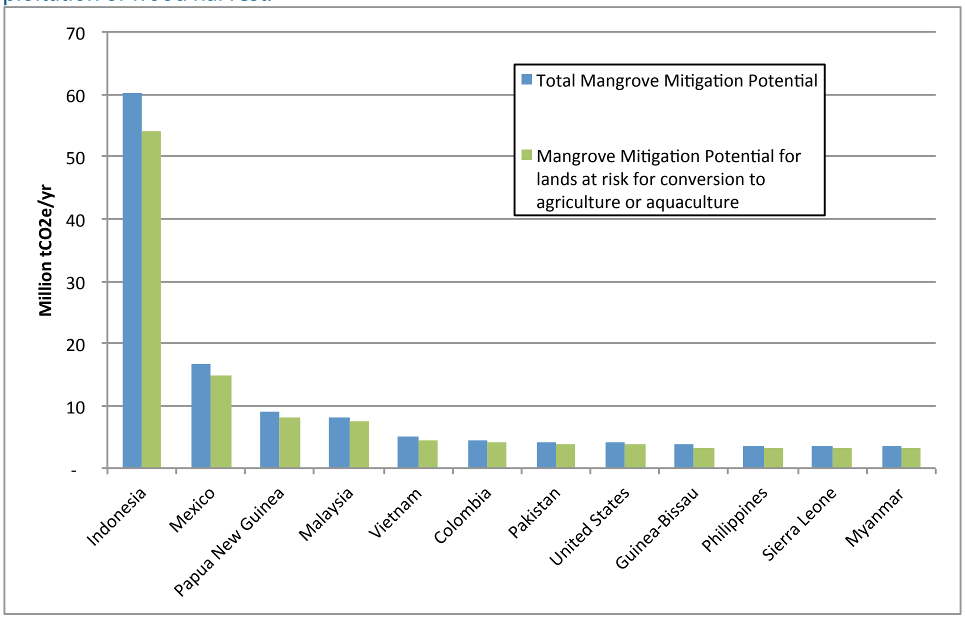
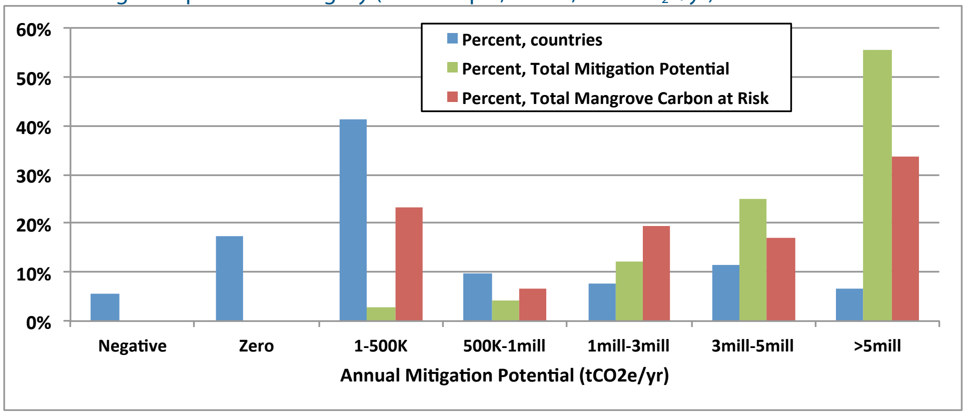


Figure 10 indicates how many of the 104 mangrove countries worldwide belong to each annual mitigation potential category. Because the costs of protecting habitat and the transaction costs of avoiding conversion projects will vary across countries, it may be useful for project investors to have an idea of the size of the mitigation market at different mitigation levels relative to the number of countries that make up that market. Just over half of mangrove countries have a relatively modest mitigation potential of less than 1 million t CO_2e/yr, while about 23% have no potential because their mangrove loss rate is near zero (for example, Nigeria) or negative (for example, Bangladesh). Approximately 80% of global potential resides in the few countries whose annual mitigation potential is over 3 million t CO_2e; over 55% of that potential resides in the seven countries with at least 5 million t CO_2e. Distribution of the relative proportions of total mangrove carbon at risk is somewhat more evenly distributed. Almost 30% of carbon at risk belongs to countries with low mitigation potential (<1 million t/CO_2e/yr), about 20% to countries with intermediate potential (1–3 million t/CO_2e/yr), and 50% to countries with high potential (>3 million t/CO_2e/yr). This final metric indicates that mitigation potential could rise substantially in countries that currently show low potential if their mangrove habitat loss rates were to increase, whether due to a new loss driver or to better data collection. Overall though, both Figure 9 and Figure 10 highlight the fact that much of the opportunity for blue carbon may lie in a small group of countries.

Figure 10. Biophysical mitigation potential for mangrove ecosystems worldwide. The biophysical mitigation potential shown here is based on current stored carbon stocks and current conversion rates. Bars represent the percentage of mangrove countries, the percentage of total mitigation potential, and the percentage of total mangrove carbon at risk in each annual mitigation potential category (for example, 1–500,000 t CO_2e/yr).



6. Magnitude and Timing of Carbon Loss after Habitat Disturbance

The magnitude and timing of post-conversion CO_2 release depends on the type of coastal habitat disturbed and the type of disturbance. The latter can determine the depth to which the soil profile will be altered. This depth suggests how much soil carbon may potentially be exposed to oxygen, be oxidized, and thereby be emitted in the form of carbon dioxide. Although meters of carbon-rich organic soils may underlie the focal coastal habitats, the carbon in those soils may remain if the habitat conversion only affects the top soil layers and the deeper layers remain inundated and their carbon intact, or if the unvegetated habitats that replace vegetated habitats continue to store organic carbon in their soils. Habitat conversion often disturbs only the top meter of soil, and so only the carbon stored there (plus the biomass) is likely to be emitted, as in the case of shrimp farming. In theory, following conversion, carbon in biomass is emitted to the atmosphere in the first few years, although organic carbon could be redistributed and redeposited to other environments. Release of soil organic carbon will take longer than biomass, and the deeper the soil carbon, the slower its rate of release. In each case, emission rates are expected to be high in the years immediately after disturbance and to drop later.[11] It should be emphasized that scientific understanding of post-conversion rates of CO_2 emissions is currently embryonic and, accordingly, we make conservative assumptions when we model carbon loss from the focal coastal habitats.

Each habitat is modeled in essentially the same way. We assume that the habitat is converted or damaged in year zero, and we model the release of carbon from the soil and biomass over a 25-year horizon. We use the values for biomass and soil organic carbon that were presented above. On the basis of guidance from collaborators and the scientific literature,[12] we assign exponential decay functions and associated half-lives for biomass and for soil organic to each habitat type.

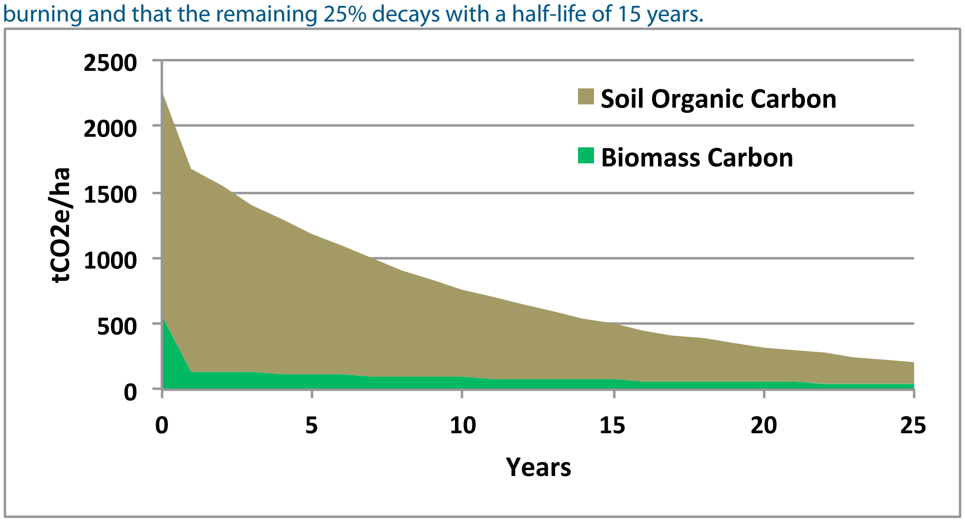
For the intertidal systems (salt marshes and mangroves), we assume that the top meter of soil is either excavated or drained during the habitat conversion and that all of that soil is equally exposed to oxygen. The excavation of mangrove soil would be standard procedure in construction of aquaculture ponds; salt marshes are likely to be diked and drained in their conversion to cropland. The assumption of one meter is a rough approximation of an average depth of soil disturbance across many types of habitat-destroying actions. We use a half-life of 7.5 years for both mangrove and marsh soil carbon and one-half year for the decay of salt marsh biomass. Regarding mangrove biomass, we assume that mangroves are burned when the habitat is converted, resulting in an immediate release of the majority (75%) of the biomass carbon to the atmosphere. Mangrove biomass includes both aboveground and belowground biomass, because roots could be uplifted, exposed, and burned away just like the trunk and branches of the trees during excavation. The remaining 25% of biomass is left to rot, which we approximate with a decay half-life of 15 years.

11. An exponential decay function may approximate this physical process, especially using the concept of half-life that denotes the time required for the carbon pool to fall to one-half of its initial value. For example, if 100 t CO_2 is exposed to conversion and it is assumed to have a half-life of 5 years, at year five, 50 tons will remain; at year ten, 25 tons will remain; at year fifteen, 12.5 tons will remain.
12. Marshes: Huang et al. 2010; C. Craft and P. Megonigal, pers. comm. Mangroves: D.C. Donato and J.B. Kauffman, pers. comm. Seagrasses: J. Fourqurean and N. Marba, pers. comm.

As a submerged ecosystem, seagrass meadows are subject to different stressors, but the carbon loss process is modeled similarly. The main mechanisms of habitat destruction are excavation or mechanical damage, as through dredging or trawling, and vegetative death resulting from degraded water and sediment quality. In the first case, we assume that the top 50 centimeters (cm) of soil would be immediately washed away and that the decomposition of the rest of the soil organic matter (another 50 cm) would occur in place. Limited evidence suggests that all seagrass carbon would be converted to carbon dioxide and be released to the atmosphere at the same rate, whether buried or on the surface. In the case of vegetative death, organic matter would decompose in place, and its carbon would make its way to the atmosphere. We apply decay half-lives of about 100 days (0.27 years) to the seagrass biomass carbon and one year to the soil organic carbon.

Additive to the stream of carbon emissions are the annual carbon sequestration or burial rates of intact habitats (Table 2). If the habitat were to be destroyed, carbon burial would cease because living matter would die. But habitat protection allows the process to continue apace. Whereas the quantity of avoided carbon emissions declines over the 25-year time horizon (and disappears in the case of faster-decaying biomass), carbon sequestration is assumed to continue at a constant annual rate, as the living matter continues to photosynthesize and fix carbon, a portion of which is stored in the soil each year. A process in these habitats that runs counter to sequestration from a climate protection standpoint is the release of methane (CH_4), a greenhouse gas that has 25 times the global warming potential of carbon dioxide. Methane release is not an issue for seagrass meadows, but small amounts of methane appear to be emitted from the intertidal systems each year. Thus, in our modeling, ongoing methane emissions equivalent to 0.4 t CO_2e/ha and 1.85 t CO_2e/ha are subtracted each year from the creditable avoided emissions for salt marshes and mangroves, respectively.[13]

Figure 11. Release of carbon to atmosphere from converted oceanic mangrove habitat in tropical Asia. The release is based on the assumption that only the top meter of soil is disturbed and reflects an exponential decay function whereby soil organic carbon has a half-live of 7.5 years. It is assumed that 75% of the biomass carbon is emitted immediately through burning and that the remaining 25% decays with a half-life of 15 years.



For illustration, we briefly describe and present values for the modeling of carbon loss in mangroves in tropical Asia, which contains about half of the world's mangrove forests (FAO 2007). We assume that on conversion to a shrimp operation, soil in the top 100 cm layer is excavated and banked and that the soil carbon and biomass decomposition processes begin immediately.[14] Figure 11 presents the decay curves for the soil organic carbon and biomass carbon. The precipitous drop in biomass carbon in the first year is due to the assumption that 75% of it is burned away on habitat conversion. Decay of the remaining biomass carbon is much slower, and 8% of it remains at the end of the 25-year period. Some of the soil carbon remains after 25 years as well. Overall, carbon stocks are reduced from 3,098 t CO_2e/ha and 3,743 t CO_2e/ha to 1,059 t CO_2e/ha and 2,172 t CO_2e/ha for oceanic and estuarine mangroves, respectively (see Table 4). As a result, annual carbon losses average 82 t CO_2e/ha/yr (2.6% rate) and 59 t CO_2e/ha/yr (1.6% rate) for the

13. Salt marshes: Poffenbarger et al. Forthcoming. Mangroves: Krithika et al. (2008). For mangroves, we use the mean of 1.85 t CO_2e/ha/yr calculated from a range of 0-10.75 t CO_2e/ha/yr adapted from Krithika et al. 2008.
14. All carbon stock values associated with mangroves as well as the best professional judgment regarding CO_2 decay rates are sourced to D.C. Donato and J.B. Kauffman, pers. comm.

two mangrove types. To arrive at the total creditable carbon, we add these avoided carbon emissions to the annual carbon sequestration rate for Southeast Asian mangroves (6.32 t CO_2e/ha/yr) for each year of the study period, and we subtract the annual effect of methane emissions (1.0 t CO_2e/ha/yr).

Table 4. Carbon pool values and losses for destroyed oceanic and estuarine mangroves in the Southeast Asia/Indo-Pacific region.

	Oceanic mangroves (t CO_2e/ha)	Estuarine mangroves (t CO_2e/ha)
Total carbon stored before conversion	3,098	3,743
Aboveground biomass	352	352
Belowground biomass	211	211
Total biomass	563	563
Soil C: <100 cm (disturbed)	1,690	1,060
Soil C: >100 cm (undisturbed)	845	2,120
Biomass C remaining after 25 yrs	46	46
Soil C (<100 cm) remaining after 25 yrs	168	105
Total C remaining after 25 yrs	1,059	2,172

7. Potential Payment Mechanisms for Blue Carbon

Mechanisms to pay for avoided emissions or enhancement of blue carbon stocks do not yet exist. A logical venue for considering blue carbon payments would be the United Nations Framework Convention on Climate Change (UNFCCC). This section briefly defines the UNFCCC, its role in creating a global carbon market, and the potential opportunity to include blue carbon as a covered activity under the UNFCCC. We also consider the prospect for blue carbon in other compliance markets and in the voluntary carbon market.

UNFCCC and the carbon market

The UNFCCC constitutes an agreement by over 190 countries to stabilize greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system. In 1997, the UNFCCC forged the Kyoto Protocol, a mechanism by which the world's most developed countries agreed to reduce GHG emissions approximately 5% below 1990 levels by 2012.[15] The Kyoto Protocol allowed for developed countries to essentially trade emission rights among themselves to meet those reduction commitments more cost-effectively. It also created the Clean Development Mechanism (CDM), which allowed developing countries to voluntarily undertake GHG reduction projects and to generate marketable credits that generate revenue for them and that help developed countries meet their commitments more cheaply. Together, the flow of emission rights within the developed world and between the developing and developed worlds has created a "carbon market" that is global in reach.

Blue carbon is not currently covered by the UNFCCC and therefore not included in a carbon market. Countries are not required to account for it, and they are neither responsible for any increased emissions from blue carbon, nor can they benefit financially from blue carbon emission reductions or restoration (for example, through the CDM). As a result, economic incentives are tilted in favor of converting blue carbon habitats to alternative uses—such as aquaculture, agriculture, and real estate development—from which parties can generate profits. But the UNFCCC is now negotiating a successor agreement to the Kyoto Protocol, which expires in 2012. The framework for the successor agreement was initially structured at the UNFCCC's Fifteenth Conference of Parties (COP 15) in Copenhagen in 2009, but it was not accepted by the UNFCCC until the following year's COP 16 in Cancún, which produced the so-called Cancún Agreement. The Cancún Agreement is a somewhat general document; details are to be ironed out before it takes effect.

The Cancún Agreement, blue carbon, and the possible applicability of REDD+ provisions

The Cancún Agreement could eventually include blue carbon in UNFCCC-covered activities. Leading up to COP 16 in December 2010, a group of 55 marine and environmental stakeholders representing 19 countries prepared an

15. The Kyoto Protocol was ultimately ratified by all of the developed countries, with the exception of the United States, that initially signed the agreement. Some of the world's largest emitters, such as China and India, were not required to assume binding emission reduction commitments owing to their then-status as developing countries. And some of the countries that did assume such commitments—Canada, for example—appear unlikely to meet those commitments by the end of 2012.

open statement (Blue Climate Coalition 2010) calling on the UNFCCC to take up the following considerations in its deliberations:

- Include the conservation and restoration of mangrove, saltwater marsh, seagrass, and kelp ecosystems in strategies for climate change mitigation and adaptation;
- Establish a global Blue Carbon Fund for the protection and management of these important coastal ecosystems;
- Include blue carbon sinks in national REDD+ strategies and greenhouse gas accounting; and
- Support coordinated scientific research to better quantify blue carbon's role in climate mitigation, including the development of protocols and methodologies for monitoring, reporting, and verification of coastal and marine carbon sinks.

Aside from the statement, several side events at the Cancún meetings were specifically dedicated to informing the negotiating community on blue carbon potential.[16]

The Cancún meeting did produce an agreement on the basic framework for moving the climate negotiating process beyond the Kyoto Protocol expiration date (2012) (UNFCCC 2010). Moreover, provisions on reduced emissions from deforestation and degradation (REDD+) were a substantial component of the Cancún Agreement. These provisions established a set of principles guiding REDD+; a defined scope of potentially covered activities; safeguards for environmental integrity, biodiversity, governance, and rights of local populations; and funding mechanisms for planning and implementation.

In the Cancún Agreement, the scope of REDD+ is defined along the following activities: reducing emissions from deforestation and forest degradation, conserving forest carbon stocks, sustainably managing forests, and enhancing forest carbon stocks. These activities reference only forests, so what do they mean for blue carbon? Answering this question is difficult because the agreement does not define forests—a task left to future deliberations, presumably with guidance from the Intergovernmental Panel on Climate Change (IPCC).[17] But it is conceivable—and many observers surmise—that forests could include mangroves, as they have above-ground woody vegetation. Inclusion of salt marshes and sea grasses, however, would presumably require a significant broadening of the terms agreed at Cancún. This inclusion could be part of efforts to seek a broader platform for emissions and sequestration activities from all land uses (forest, agriculture, grasslands, wetlands). This platform is referred to in some quarters as agriculture, forest, and land use (AFOLU).

Another critical issue is which carbon pools would count under REDD+, and most relevant for blue carbon, is whether the soil carbon pool (or just the aboveground biomass pool) would be included. Much of the emphasis on deforestation and forest degradation focuses on carbon above ground, rather than below ground. As the data throughout this report suggest, exclusion of belowground carbon from recognition would substantially undercut blue carbon mitigation potential. Again, inclusion/exclusion of soil carbon is a detail that remains undefined by the broad strokes of the Cancún Agreement—a detail to be resolved in post-Cancún deliberations.

Blue carbon under the CDM?

Another possibility would be for blue carbon to be included under the project-based Clean Development Mechanism. Indeed, a methodology for mangrove restoration has recently been proposed for inclusion as an afforestation/reforestation (AR) activity under the CDM.[18] Although inclusion under the CDM could be a start for those seeking market incentives for blue carbon, the opportunities are limited on a couple fronts. First, the proposed methodology applies to mangroves only; salt marshes and sea grasses would not appear to qualify. Second, the proposed qualifying AR activity is restoration; the much larger avoided emissions through protection of blue carbon stocks would remain outside the mechanism. Again, the parallel with forests is worth noting, as forest carbon coverage under CDM is limited to afforestation/reforestation. The inclusion of incentives for much larger-scale forest protection (emissions from deforestation and degradation) was the impetus for efforts to include REDD+ in the Cancún Agreement outside of the CDM. As with forests, limiting blue carbon to the restoration activities possibly covered under the AR provisions CDM would provide positive, but small, incentives to substantially change the blue carbon balance.

16. See IISD 2010.
17. The IPCC has a sub-body, the Subsidiary Body on Science and Technical Advice (SBSTA), that often roots out these more detailed technical issues left open by the broader negotiating text.
18. See Emmer and Silverstrum 2010.

Non-UNFCCC compliance markets

Although the UNFCCC has created the largest global GHG compliance mechanism and related carbon market, it is not the only one. In the United States, compliance markets exist through the Regional Greenhouse Gas Initiative (RGGI) in ten northeastern states. A compliance market will soon exist in California as a manifestation of that state's own GHG cap, which was signed into law in 2006.[19] Both these regional programs allow offsets from uncapped sources to be used for compliance, and California has already moved forward on inclusion of international offset sources, most notably through intergovernmental agreements on REDD+ with specific states and provinces in select forested countries such as Brazil, Indonesia, Nigeria, and Mexico.[20] Neither system references blue carbon, but each could in the future, either specifically or, in the case of mangroves, under the REDD+ provisions of the California system or in certain circumstances under RGGI.[21]

Voluntary markets

Outside the auspices of the UNFCCC and other compliance systems, the market for voluntary reductions offers opportunities to incentivize carbon activities. Although not required by law to do so, some parties will pay for emission reductions to offset emissions from their own activities or simply as an act of good corporate or individual stewardship. Another motive for some buyers in the voluntary carbon market is to create a hedge against the possibility that they may one day be subject to compliance obligations. Voluntary payments for emission reductions may one day count as an early compliance action.

At this stage, blue carbon does not trade on voluntary markets. But it might with efforts to propose and develop methodologies for inclusion in voluntary market systems such as the Voluntary Carbon Standard (VCS), Climate Action Reserve (CAR), or American Carbon Registry (ACR), all of which consider land use and forest practices as potentially creditable activities. These systems expand the suite of creditable activities beyond the narrow boundaries of CDM to include not only afforestation/reforestation, but also REDD+ and improved forest management. In doing so, they increase the possibility that blue carbon might be included in them.

Current range of carbon market prices

Table 5 provides a snapshot of current carbon prices for GHG emission reductions in compliance and voluntary markets across the world. Prices are expressed on a U.S. dollar-per-tonne CO_2-equivalent basis.

Table 5. Carbon prices in compliance and voluntary markets. Latest quotes prior to publication of report. Prices vary daily.

Market	Price ($/t CO_2e)	Comments	Source
Compliance markets			
EU ETS	$24.48	Full price of an EU allowance (EUA)	Point Carbon, 3 April 2011 www.pointcarbon.com
CDM	$18.64	Secondary market price of a CDM-certified emissions reduction (sCER)	Point Carbon, 3 April 2011 www.pointcarbon.com
RGGI	$1.89	Full price of a RGGI allowance. RGGI cap is currently loose in stringency and thus willingness to pay for additional allowances is relatively low.	3 April 2011 www.rggi.org
Voluntary market	$6–$7	Average price of over-the-counter voluntary carbon market transactions in 2009. The voluntary market does not include the Chicago Climate Exchange (once the largest single source of voluntary carbon market transactions), which ceased operating in 2010.	State of the Voluntary Carbon Markets, 2010. Ecosystem Marketplace and New Bloomberg Finance. http://moderncms.ecosystemmarketplace.com/repository/moderncms_documents/vcarbon_2010.2.pdf

Compliance market prices reflect current prices on the exchanges on which EU ETS, CDM, and RGGI credits are traded. The UNFCCC markets are the highest on the board, trading between about $18 for CDM credits to about $24

19. Smaller regional compliance markets exist in Australia (New South Wales) and Canada (Alberta), but they are in Kyoto Protocol signatory countries, so they are at least indirectly linked to the UNFCCC.
20. See Governors' Climate and Forests Task Force, http://www.gcftaskforce.org/.
21. Under RGGI, in the case of a stage-two trigger event (when the 12-month rolling average allowance price is equal to or greater than $10 in 2005 dollars), GHG reduction credits resulting from any mandatory carbon-constraining program outside the United States or certified through the UNFCCC will be eligible for up to 10% of compliance obligation. The general provision on international offsets is in the Model Rule, Section XX-10.3(a)(2)(i)(b). http://www.rggi.org/docs/Model%20Rule%20Revised%2012.31.08.pdf.

for regular EU allowances. RGGI allowances are about an order of magnitude smaller than the UNFCCC-related markets, reflecting differences in the scope and relative stringencies of the two regulations. RGGI applies only to electric power plants in the northeastern United States, and compliance obligations are not much different from those in a business-as-usual scenario, making marginal abatement costs low.

Credits on the voluntary market typically trade at prices lower than those found in compliance markets (the exception is RGGI) and at roughly a third or fourth of those in the UNFCCC markets, according to the most recent survey evidence. However, some credits on the voluntary market trade at a premium relative to the rest of the voluntary market, especially if the credits are compliant with what the market perceives to be higher standards of performance in terms of stringency of carbon reduction requirements or the inclusion of other desirable attributes such as biodiversity co-benefits or protection of human rights.

8. Economic Model of Avoided Conversion

Most of the remainder of this report focuses on the economics of emission reduction projects involving blue carbon. We begin by presenting our economic framework for evaluating these projects. We then consider the monetary valuation of benefits and costs that would be associated with the projects and offer case studies. Finally, we provide country-specific estimates of net economic returns (present value of benefits minus present value of costs) for mangrove mitigation as well as provide mangrove blue-carbon supply functions that aggregate this information across countries.

The economic model begins with the quantification of the GHG benefit—emissions that would be avoided and sequestration maintained through a blue carbon project that protects one of the focal coastal habitats. The GHG benefit fluxes of blue carbon avoided-conversion projects take this general form:

$$(1)\quad GHG\ Benefit\ Flux_{it} = CS_{it} + AvCO2_{it} - M_{it}$$

where i is the habitat type and t is time, expressed in years; CS is the annual carbon sequestration rate, which continues as the habitat is retained; $AvCO_2$ is the CO_2 emissions avoided from the habitat's conversion; and M represents the annual methane emissions that continue to be emitted as the habitat remains intact. Methane emissions reduce the GHG benefits of protecting habitat but tend to be small relative to the magnitude of the carbon sequestration rate and especially the avoided CO_2 emissions in conserved saltwater habitats. All quantities are expressed in CO_2 equivalent units.[22]

Thus, the annual GHG benefit fluxes are characterized as temporal flows, reflecting the dynamic nature of post-conversion GHG emission rates. Rates of carbon sequestration and methane emissions could also fluctuate temporally, though they are assumed here to be constant annual rates. All fluxes are computed on a per-hectare basis. As quantitative understanding of the relevant processes improves, complicating factors, such as sea level rise, could be integrated into the modeling of the GHG fluxes.

GHG benefits flows are monetized by multiplying the annual GHG fluxes by a stream of expected carbon prices ($/t CO_2e) over a time horizon of length n. The resulting cash flow streams are then discounted using a financial discount rate (d) to arrive at present value estimates for the blue carbon (BC) value of an avoided conversion project:

$$(2)\quad Blue\ Carbon\ value_i = \sum_{t=0}^{n} \frac{GHG\ Benefit\ Flux_{it} * Price(\$/t\ CO2e)_t}{(1+d)^t}$$

The results provide a measure of the present value of a stream of monetary flows to land managers for keeping blue carbon intact and for receiving the market rate for the emissions avoided over time.

We consider two types of comparison that can be conducted to determine whether blue carbon payments are sufficient

22. All carbon (C) fluxes are multiplied by 3.67 to be expressed in CO_2 equivalent units. All methane fluxes are multiplied by 25, which is the IPCC global warming potential value for methane to make one tonne of methane equivalent in radiative-forcing value to one tonne of CO_2.

to incentivize protection of the focal coastal habitats. In the first, the estimated blue carbon values will be compared to the costs to protect that habitat. This comparison will be the emphasis of the analysis in the following sections, which will demonstrate that an avoided conversion would be economically viable if:

$$(3) \quad Blue\ Carbon\ value_i > Protection\ costs_i$$

The results of this comparison indicate whether blue carbon payments would be sufficient to promote the protection of the coastal habitat in question, irrespective of the values of other ecosystem services (ES) provided. This finding gives us a useful first indication of the economics of blue carbon. If the carbon payments alone can cover the costs, they can be influential in protecting these ecosystems even without explicitly considering other ecosystem services. These results will allow us to identify specific habitats and locations where conservation and restoration efforts will be most economically beneficial. The main point with equation (3) is to see whether new markets or funds that would pay land managers to avoid GHG releases through habitat conversion could make a difference for some coastal habitats.

A second comparison would incorporate the estimated values, where available, of other ecosystem services provided by the intact ecosystem. These ES values could be varied across habitats and regions and could be greater than the blue carbon value. They show that the social returns of protecting coastal habitats extend beyond simply the blue carbon value. The overall ES values of the habitat are compared to the protection costs. Thus, avoided conversion projects would be economically justified if:

$$(4) \quad Blue\ Carbon\ value_i + ES\ values_i > Protection\ costs_i$$

In this comparison, we would ascertain whether the total ecosystem value (blue carbon payments plus other ES values) is great enough to exceed the various costs of protecting the habitat. This analysis would allow us to consider the effect of other incentives that could be combined with payments for blue carbon.

One difficulty in making this comparison is that ES values will often be location-specific and, as a result, values estimated in a study for one salt marsh may not be readily applicable to other salt marshes. In contrast, the blue carbon value (GHG reduction) is a global service, meaning that the avoided emission of a tonne of carbon dioxide has the same value no matter where the service occurs.[23] Because ES values for each habitat may not be generalizable to broader scales, as implied in equation (4), we do not add them to the blue carbon value in the following analysis. Relevant ecosystem services and their values are discussed in Section 11.

9. Monetary Value of Blue Carbon Benefits

Once the streams of creditable carbon have been modeled, blue carbon flows are monetized with carbon prices (expressed in \$/t CO_2e). As discussed in Section 7, future prices for carbon offset credits are uncertain, so we apply a range of possible carbon prices to capture that uncertainty. The most likely range for near-term carbon prices is \$0–\$30 per t CO_2e. Figure 12 shows the present value (PV) of gross financial returns to avoided conversion projects, that is, the benefits side of the project ledger (equation [2]). We use a real discount rate of 10% over a 25-year time horizon.[24] As implied by equation (3), net returns (net present values or NPV) also account for project costs, which are discussed in Section 10.

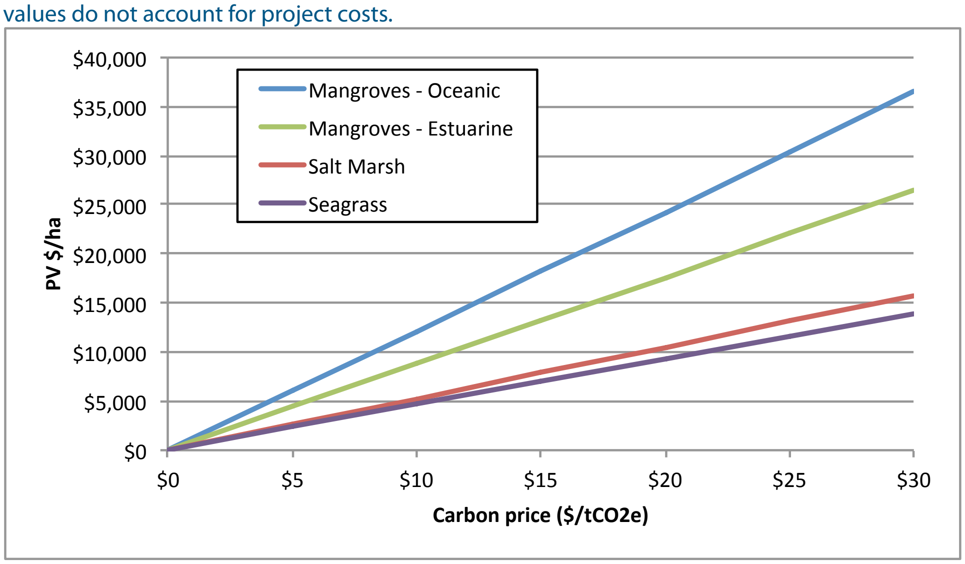
Gross blue carbon values would be zero at a price of \$0/t CO_2e, but they would begin to rise at varying slopes as the carbon price rises. In Figure 12, we present global blue carbon value streams for each focal habitat and divide mangroves into oceanic and estuarine types. The mangrove values are based on weighted averages of biomass and soil carbon across the four mangrove regions. Because regional data on carbon pool estimates for seagrasses and salt marshes are lacking, these values are assumed to represent those systems worldwide.

23. Another important difference between blue carbon value and other ES services is that the latter are often scale-dependent. That is, the per-hectare value of the ecosystem may change when the habitat area in question increases or decreases. The blue carbon value is independent of scale because each avoided CO_2 emission performs the same level of climate stabilization service.
24. Some readers may consider a real rate of discount of 10% to be on the high side, but we believe this rate better reflects the time preferences and risks associated with investments in developing countries, where many of the blue carbon mitigation opportunities exist.

Gross returns for mangrove avoided-conversion projects would be the most attractive returns for the three coastal habitats modeled. The present value per hectare exceeds $10,000 at carbon prices of about $8/t CO_2e for oceanic mangroves and about $11/t CO_2e for estuarine mangroves; salt marshes and seagrasses do not reach $10,000/ha until carbon prices approach $20/t CO_2e. Seagrasses have much less carbon in the first meter of soil than salt marshes, but that carbon returns to the atmosphere much more quickly when seagrasses are disturbed; thus, emission reductions accrue faster. As a result, seagrasses have approximately the same blue-carbon benefit curve as salt marshes do.

Oceanic mangroves tend to accrue greater blue carbon benefits than estuarine mangroves because their carbon stocks in the first meter of soil are much larger; biomass carbon is assumed to be the same in both types. However, estuarine mangroves often have deeper organic soils and would close the gap with oceanic mangroves if the disturbance went much deeper than one meter. Positing a deeper disturbance would also raise salt marsh returns because salt marshes can have several meters of organic soil beneath them. In contrast, seagrass meadows have about one meter of soil on average and thus would have the same gross returns even if the disturbance exceeded one meter in depth.

Figure 12. Gross financial returns to avoided conversion projects for the three habitat types over a range of carbon prices ($0–$30/t CO_2e). Values were calculated using equation [2], with a time horizon of 25 years and a discount rate of 10%. These values do not account for project costs.



Source: Authors.

Figure 13 reports blue carbon values for the two mangrove types in four regions. The figure legend lists the mangrove systems in decreasing order of their potential blue carbon values. Oceanic mangroves in tropical Africa and tropical Asia have the highest, and more or less equal, returns, though the subtropical and tropical Americas are not far behind (note that the lines for "Oceanic – Africa" and "Oceanic – Asia" in Figure 13 almost overlap). Within mangrove types, the largest gap in values is that between Asian and subtropical estuarine mangroves. This gap is driven wholly by differences in biomass carbon because the data show estuarine soil carbon stocks as constant across regions. Overall, all oceanic mangrove systems show greater blue carbon values than estuarine systems because the mangrove systems have higher carbon density in their first meter of soil. All mangrove values are notably higher than salt marsh values, as seen in Figure 12.

Figure 13. Gross financial returns to avoided conversion projects for mangroves in different regions. These values are calculated with same parameters as in Figure 12. The regions of Africa, Asia, and the Americas only include tropical areas and the Subtropical region accounts for all other mangrove areas, no matter on what continent.

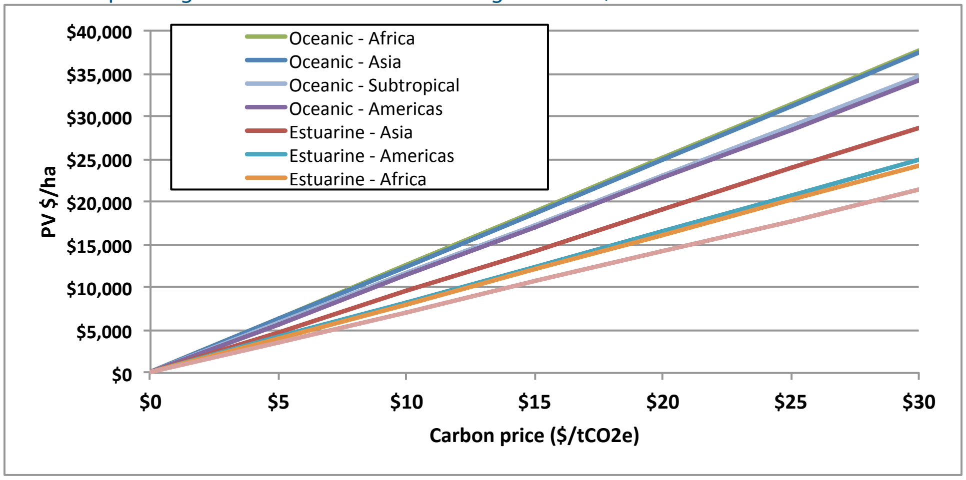
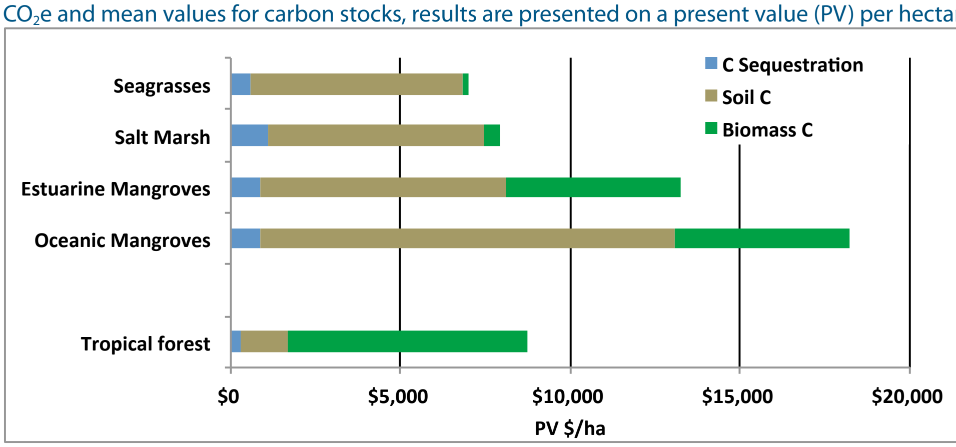


Figure 14 allocates the gross returns to different carbon fluxes within each habitat type. We use the midpoint of the carbon price range, $15/t CO_2e, as an illustrative price. Again, mangroves yield the highest values; estuarine mangroves exceed $13,000/ha and oceanic mangroves exceed $18,000/ha. At nearly $7,000/ha, seagrasses have just over half of the revenue potential of estuarine mangroves, but lag the revenue potential of salt marshes by only 12%. With the highest carbon burial rate, salt marshes have the greatest amount of carbon credit value attributed to carbon sequestration, almost double that of seagrasses. The vast majority of returns for seagrasses and salt marshes, which possess only minimal biomass, come from soil carbon. As a point of comparison, most of the value of avoiding the loss of tropical forests comes from retaining aboveground biomass. Soil carbon and biomass carbon contribute fairly equally to blue carbon value for estuarine mangroves, whereas soil carbon stocks outpace the contribution of biomass carbon for oceanic mangroves. Exerting a small negative effect are annual emissions of methane, a low-volume but potent greenhouse gas emitted by existing coastal wetlands. These emissions decrease the present values of blue carbon by less than 1% for salt marshes and less than 2% for mangroves.

Figure 14. Gross carbon credit revenue potential for coastal habitats and tropical forests. Assuming a carbon price of $15/t CO_2e and mean values for carbon stocks, results are presented on a present value (PV) per hectare basis.



Source: Authors:

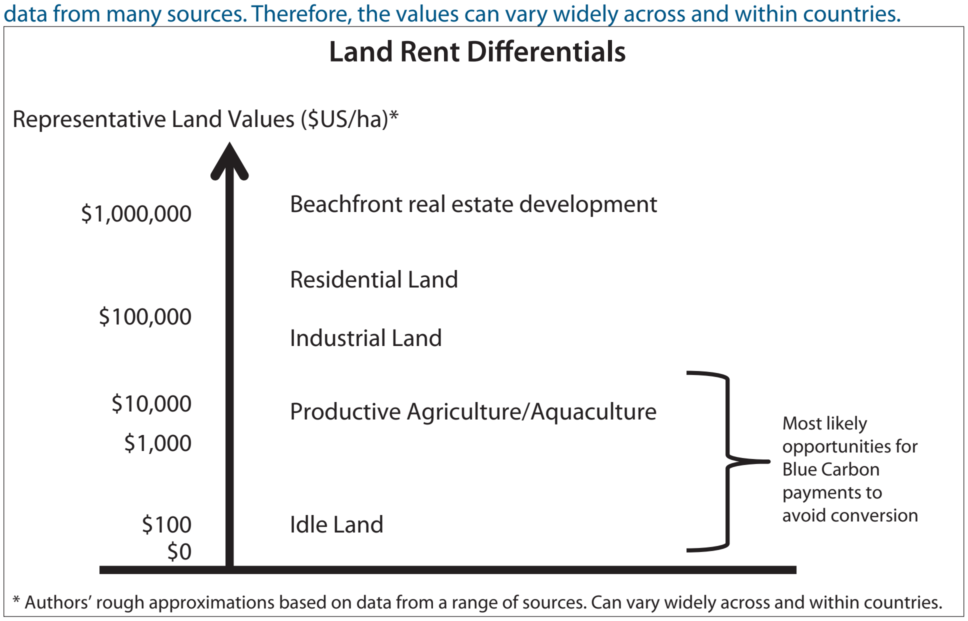
10. Costs of Blue Carbon Protection

Avoiding the conversion of a threatened coastal habitat to another use, such as aquaculture, will entail certain direct and indirect costs. Direct costs include the recurrent outlays for protection, which typically include administration, maintenance, and enforcement. Costs related to the establishment of protected areas may also apply, though examination of these costs has been limited.[25] Opportunity costs are usually the largest single cost factor. They embody the forgone returns of the most profitable alternative use for the land that the coastal habitat occupies, thus representing what the owner of the resource (see Box 1) gives up by not converting the habitat.[26] In some cases, opportunity costs can be negligible. When government-owned coastal land lies far from human populations and is not slated for an alternative use, the cost of protection possibilities may be low. Figure 15 illustrates the great variability in land values in coastal areas. Land rents for coastal areas, on which mangroves and salt marshes occur, may vary widely according to the location and potential alternative use of the land (typically influenced by proximity to population centers). The majority of opportunities for blue carbon projects will be on land currently or in the near future at risk for conversion to agriculture, aquaculture, or wood harvest. Costs tend to increase with the income of the country in question, though we use a fixed cost for establishment of protected areas because there is only one study (McCrea-Strub et al. 2011) from which to draw estimates.

Box 1. Concepts of ownership.

We use "owner" here to mean controller of the resource. The concept of ownership can vary greatly. Standard notions of private or government ownership could be supplanted by community rights (formal and informal), separate rights to land and to the resources on them, and weakly enforced or unenforced property rights. A range of institutional structures of ownership exist across and within countries. These ownership notions and institutional structures complicate matters and must be fully considered when assessing local implementation.

Figure 15. Land value differentials for coastal areas. These representative land values are rough approximations based on data from many sources. Therefore, the values can vary widely across and within countries.



Source: Authors.

Among the direct costs, we apply $232 per hectare as the upfront establishment costs to create a protected area, whether terrestrial or marine. McCrea-Strub et al. (2011) find a range of establishment costs of $20 to $788 (2009 USD) per hectare for small marine protected areas in the tropics, with a mean value of $232 per hectare. Regarding annual costs involved in managing a terrestrial protected area, Balmford et al. (2003) find such costs to be about $13/ha/yr (2009 USD) near populated areas in a developing country and under $1/ha/yr for more remote regions. For the developed world, Balmford et al. report that these running costs fell in the range of $60–$600/ha/yr. Using these numbers as a

25. The exception is McCrea-Strub et al. 2011.
26. For some conversion activities, profits may be high immediately following habitat conversion but may taper off or disappear several years later due to productivity losses or unsustainable practices. Future profitless years need to be taken into account in the present-value calculation of opportunity costs.

guide, we conservatively assign management costs according to relative national affluence by dividing countries into four groups. Least developed countries are assumed to have annual costs of $25/ha, emerging countries to have either $50/ha or $100/ha, and the wealthiest countries and some prosperous small island nations to have $250/ha. Because these costs are annual, present values are calculated using a 25-year time horizon and a 10% discount rate. For example, yearly costs of $25/ha lead to a present value of $227/ha. These costs are used for countries containing salt marshes or mangroves.

According to the scientific literature, marine protected-area management costs do not differ considerably from terrestrial protected-area management costs. Balmford et al. (2004) report a mean of $34/ha/yr from a wide range. Marsden and Sumaila (2010) find management costs for small marine protected areas (<3,000 ha) in Central and South America to range between $4 and $35 per hectare. To be conservative, we apply the four-part cost groups to seagrass countries that we used for terrestrial protected-area management costs.

Opportunity costs for the intertidal habitats—salt marshes and mangroves—are drawn from datasets from the World Bank (World Bank 2011) and the Global Trade Assessment Project (GTAP 2007). We use the land values from these studies as opportunity costs, which are well-approximated by land values because the stream of future economic returns is assumed to be embedded in the value of a hectare of land. Both datasets derive land values on the basis of potential agricultural returns for arable land. For the WB dataset, cropland wealth per hectare is calculated as the net present value of returns from crop cultivation, using a discount rate of 10% over a 25-year horizon. These returns are calculated as the product of rental rates and revenues from production; a constant profit margin of 30% across all considered crops is assumed. The GTAP dataset represents estimates of the value of the land in the next best alternative to forestry, which is typically agriculture, for countries of interest. Within each country, several forest types and the associated land rents are provided, allowing us to compute weighted averages for land value across the relevant forest types. Thus, unlike the WB data, GTAP does account for variation of land values within countries. We transform the GTAP values into present value using a 10% discount rate and 25-year time horizon so as to align them with the WB data and the rest of this analysis. Because estimating land values is an inexact science, the datasets are generally not in agreement and have very low statistical correlation. On one hand, the WB data are somewhat higher than one might expect, especially for the Latin American countries. On the other, the GTAP data generally seem to be on the low side.[27]

Because aquaculture, particularly shrimp farming, has increased in recent decades, its returns are arguably somewhat higher than those for agriculture. To account for aquaculture's potentially higher per-hectare returns, we increased the WB values, which are derived from agricultural rents, by 30%. This adjustment is based on the percentage difference in WB land values and net returns for shrimp farming for two countries, Thailand and Bangladesh, reported in Sathirai and Barbier (2001) and Islam et al. (2005). We use values averaged from the GTAP dataset and the WB dataset (adjusted for aquaculture returns) as mean estimates for opportunity costs for all countries.

Salt marshes often exist in developed countries, which have environmental laws that ensure their conservation or at least replacement elsewhere if affected (for example, U.S. Clean Water Act S. 404 requires no net loss of salt marshes). Some mangroves, such as those in the U.S. state of Florida, may enjoy this type of protection too. If so, landholders wishing to convert wetlands to another land use would incur costs associated with adhering to the law, such as going through the permitting process and paying for compensatory mitigation. These costs make the habitat conversion more expensive and thus less attractive to landholders and therefore would be *subtracted* from the net cost of protection. The additional legal costs of conversion could be quite high, as suggested by the admittedly scarce data on compensatory mitigation costs related to compliance with the statute.

Opportunity costs for seagrass meadows are more challenging to ascertain. Because degraded coastal water quality is the most significant factor in seagrass habitat loss, interventions to improve that quality would need to be characterized. These interventions would most likely occur in terrestrial areas within the watershed that outflows into the seagrass habitat. The principal options would be to reduce runoff from agricultural or urban areas by retiring cropland or implementing best management practices (for example, for tillage or fertilizer application) on it or by constructing wastewater treatment plants. Given differing effects of tides and differing distances of seagrass meadows from the coast, these actions would have varying levels of effectiveness that as yet are not well-understood quantitatively.

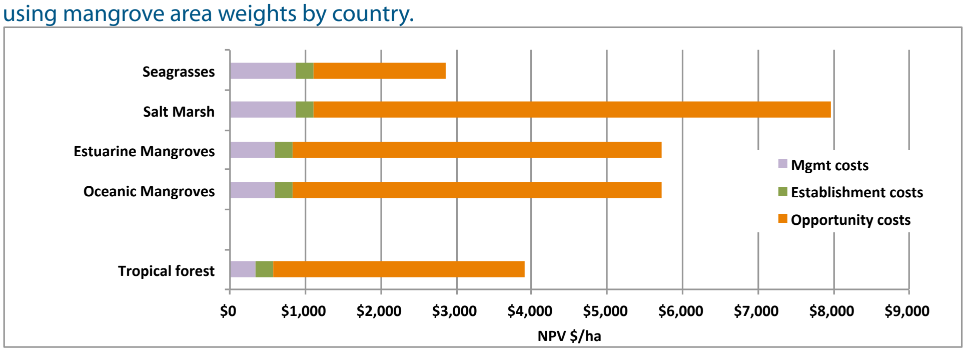
27. For example, the land value for a hectare in Colombia is valued at $19,651 in the WB dataset but at $701 in the GTAP dataset; a hectare in Papua New Guinea is valued at $11,922 in the WB dataset but at $290 in GTAP dataset.

An alternative approach to assessing seagrass opportunity costs would be to exclude extractive use (for example, commercial or artisanal fishing) of the seagrass areas, which would remove the main causes of mechanical damage from anchoring, trawling, and so on. Thus, the opportunity cost would be the forgone profits from fishing the protected seagrass areas. Ascertaining opportunity costs this way has a strong conceptual foundation (Smith et al. 2010), though somewhat limited application to date (Adams et al. 2011; Klein et al. 2010). One recent study examining the collection of invertebrates and fish in seagrass meadows in Indonesia found that collectors would forgo profits of about $45 to $140 per hectare per year if they were to be displaced by the establishment of a non-extractive marine protected area (Unsworth et al. 2010). The average across all site types was about $100/ha/yr, which yields a present value of over $900/ha over 25 years at a 10% discount rate. We could generalize to other seagrass countries by observing that Indonesia is in the third quartile in terms of GDP per capita in the world. Opportunity costs would likely be lower in lower-income countries and higher in higher-income countries. This approach appears to be an appropriate first-order option for finding opportunity costs for seagrass protection, though research that would provide empirical estimates to be placed into our framework is needed to conduct a proof of concept.

Figure 16 presents the average estimated costs for the three cost types across the focal habitats and, for comparison, tropical forests. Management costs fall within the same general range of about $350 to $900 per hectare and are somewhat higher for salt marshes and seagrasses because those habitats tend to occur more frequently in higher-income countries. Management costs for tropical forest reserves are the lowest of the group for the same reason—tropical forests are predominantly found in low-income countries. As mentioned above, the same protected-area establishment cost ($232/ha) is used for all habitat types. Due to higher land values in higher-income countries, opportunity costs are highest for salt marshes at over $6,800/ha. Opportunity costs for mangroves are about 30% lower than those for salt marshes.[28] They tend to be lower than those for other terrestrial habitats because mangroves are limited to tropical countries, most of which are lower income, and because the land values adjusted upward to account for aquaculture are not applicable to them (most forests would be upland from the coast). The opportunity cost for seagrass is the lowest among the habitats because per-hectare values for fishing are lower than those for agriculture. Of course, our estimate of opportunity cost may be a partial value that takes into account only the opportunity costs of fisherman, and it may even understate that value. Larger-scale approaches that seek to improve coastal water quality through changes in management of farmland or urban wastewater would likely be much more expensive. Additional research is needed to better understand potential effectiveness and costs.

One last consideration regarding costs is that participation in a market for blue carbon reductions will involve some costs itself. These costs include the costs of measuring, monitoring, and verifying coastal habitat loss and carbon stocks, establishing a baseline against which emission reductions are measured, making accounting adjustments that might discount the value of credits, and enforcing contracts and monitoring transactions. These transaction costs are as yet unknown, tend to be "up front" in nature, and should be carefully assessed before parties proceed with blue carbon projects.

Figure 16. Costs of protection (PV $/ha, 25-year horizon, 10% discount rate) for the focal coastal habitats and tropical forest. Average costs across each habitat type are presented for each cost category. For mangroves, averages are calculated using mangrove area weights by country.



Source: Authors.

11. Case Studies of Coastal Habitat Conversion

To demonstrate situations in which payments for blue carbon could make a difference in protecting coastal habitats, we present one case study for each focal habitat in a specific part of the world.

28. There are a few exceptions. Opportunity costs for U.S. tropical forests, for example, are $3,339/ha.

Seagrass meadows: Trawling in the Spanish Mediterranean

Seagrass meadows occupy shallow seas throughout the world, except those in the polar zones. But one of the only seagrass meadow areas to have been well studied is in the Mediterranean Sea. The Mediterranean has an estimated area of 1.35–5.0 million hectares (Mha) (N. Marba and J. Fourqurean, pers. comm.). Assuming a global area of 30 Mha, the area of the Mediterranean seagrasses equates to approximately 5% to 17% of world seagrass habitat. Although the conservation status of most seagrass meadows is unknown, annual rates of area loss may be 1.2% or more (adapted from Waycott et al. 2009), and they may be up to 5% within the Mediterranean (Marba 2009). Of the 46 Mediterranean sites monitored annually, 40 are found along the Spanish coast and are dominated by the species *Posidonia oceanica* (Marba 2009). Since 2000, nearly half of the *P. oceanica* meadows studied have suffered net losses of shoot density exceeding 20% (Marba 2009). Impacts on seagrass habitat directly related to human activities include eutrophication from nutrient pollution, alteration of coastal sediment balance, and mechanical perturbations (for example, dredging, trawling, and anchoring). Other impacts on seagrasses that are indirectly related to human activity include rising sea temperature (climatic change) and the introduction of invasive species (maritime traffic and aquaculture).

Trawling as a disturbance factor

Trawling, a method of fishing, has been identified as one of the most important direct causes of habitat destruction of *P. oceanica* meadows in the Mediterranean (Claudet and Fraschetti 2010). Trawling involves dragging large fishing nets, weighted down by an otter board, through shallow water. Repeated passes can destroy seagrass shoots and rhizomes (Gonzalez-Correa et al. 2005). Moreover, trawling resuspends sediment, thereby increasing water turbidity and reducing the amount of light reaching the seagrass, which is extremely sensitive to deterioration of water clarity (Duarte 1995).

The mechanical disturbance associated with trawling can rip out both the living biomass and sediments associated with seagrass habitats. The carbon in these habitats becomes more susceptible to suspension and can wash away, oxidize in the water column, and ultimately release carbon dioxide to the atmosphere. In the case of vegetative death, organic matter in the sediments decomposes in place; over 95% of its carbon returns to the water column and on to the atmosphere within less than five years. In addition, seagrass mortality results in the loss of annual carbon sequestration performed by the living plants. When Mediterranean seagrass habitats are destroyed, the oxidized carbon can result in the release to the atmosphere of more than 500 t CO_2e per hectare; the forgone annual sequestration from lost living seagrass would be approximately 4 t CO_2e per hectare per year.

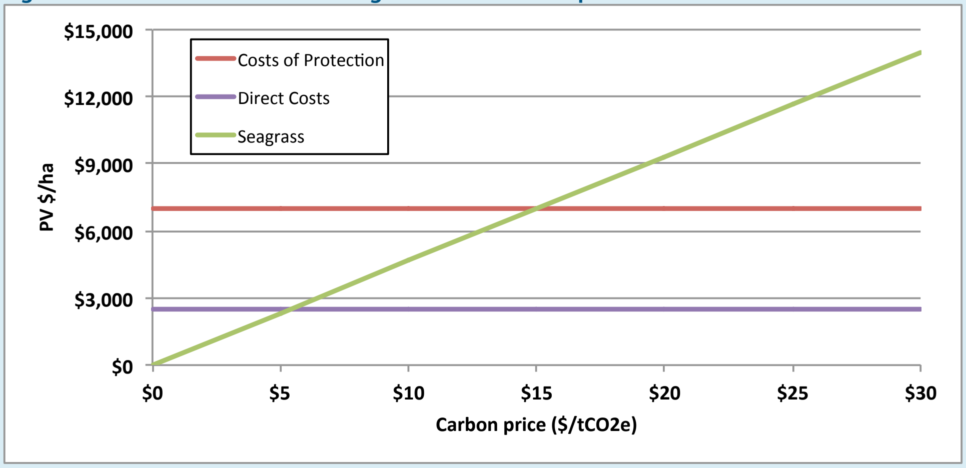
Actions to halt trawling over seagrass meadows would reduce seagrass mortality and the corresponding loss of soil organic carbon and sequestration capacity. The amount of CO_2 emissions avoided by such actions could potentially be credited in a carbon payment system.

Blue carbon valuation

The blue carbon value of a stop-trawling project would depend on the future expected amount of carbon that would be released under trawling and that would otherwise have been sequestered in an intact habitat. That value would also depend on the carbon price over time. Over a near-term carbon price range of \$5–\$30 t CO_2e, blue carbon benefits in a protected seagrass habitat could be valued between \$2,300 and \$14,000 per hectare (see Figure 17.)

To consider the financial viability of efforts designed to stop trawling, we compare the market value of carbon that could be emitted from seagrass loss to the costs of conserving seagrass, which may include the costs of establishing and managing a marine protected area as well as trawlers' opportunity costs. The estimated mean direct cost of establishing a marine protected area is approximately $250, and the present value of recurring costs is approximately $2,250/ha, for a total present value cost of $2,500/ha. If the direct costs were the only relevant ones for protection (that is, opportunity costs are zero), the break-even market price of carbon (the price at which blue carbon benefits equal costs of conservation) would be $5.36/ t CO_2e.

Figure 17. Blue carbon value for seagrass meadows compared to direct costs and overall costs of protection.



Nevertheless, it is likely that excluding trawling from seagrass meadows would result in lost profits for trawlers, which would be the opportunity costs (Smith et al. 2010). This exclusion would mean that trawlers would be forced to find other areas in which to fish, perhaps accruing greater travel costs to get to alternative areas or bringing in less catch per unit of effort if other trawling areas are less productive. Or the exclusion could force them to quit trawling altogether and search for alternative employment. These opportunity costs could be on the order of $4,500/ha for a high-income country such as Spain (see Section 10), though they have been little studied to date and could vary greatly among and within countries. In this case, overall costs of protection would be about $7,000/ha, and the carbon break-even price would be $15/t CO_2e. Therefore avoiding destruction of seagrass habitat could be financially attractive at carbon prices above $15/t CO_2e, which is about where CDM-certified emissions reductions were trading in early 2011.

Other ecosystem service values

Seagrass meadows store large quantities of carbon in their sediments, perhaps accounting for 15% of the total carbon buried in the ocean worldwide (Duarte, Middelburg, and Caraco 2005). In addition to that function, they perform many other critical ecosystem services. Seagrass meadows host many fish and invertebrate species at different life stages; they can generate both subsistence and commercial value in their intact state. In Indonesia, invertebrate and fish collectors harvest in the seagrass beds and this use has been valued at about $50 to $150 per hectare per year (Unsworth et al. 2010). In Cairns Harbor, Australia, juvenile prawns use seagrasses as a nursery and are commercially harvested as adults. On the basis of this offshore prawn fishery, one study valued a hectare of seagrass at around $1,500 per year (Watson, Coles, and LeeLong 1993). Other essential ecosystem services provided by seagrass meadows include nutrient cycling, coastal protection, and the export of sediment and nutrients to beach and associated dune systems that help sustain them.

Salt marshes: Reversing sediment starvation in the Mississippi Delta

Coastal marshes, including salt and brackish marshes, have been reclaimed for human uses from the times of early human settlements, resulting in an overall marsh habitat loss up to 67% of the original extent (Lotze et al. 2006). Wetland loss has continued in recent decades; a recent study found a total loss of 1.6 million hectares in 14 deltas worldwide from the mid-1980s through the early 2000s (Coleman et al. 2008.). Over two-thirds of the loss was attributed to conversion for agricultural or industrial use, while the other third was a net loss due to expansion of open water, which is often related to dams or other human modifications upstream of deltas. In North America, most historic coastal marsh loss was due to agriculture or urban uses; eastern Canada and the Pacific Coast have seen the greatest losses relative to the original extent (Gedan et al. 2009.) Nevertheless, the U.S. Gulf Coast has also experienced substantial wetlands loss—an estimated 450,000 hectares—much of it coastal marsh from the Mississippi Delta in Louisiana alone (Day et al. 2000). In recent years, wetland loss in the United States has slowed considerably due to national wetland legislation and public attention (Dahl 2006). Of the estimated 1.6 million hectares of salt and brackish marsh in the United States, about 13,450 hectares disappeared between 1998 and 2004, the majority of which was lost to open saltwater systems in coastal Louisiana (Dahl 2006).

The role of flood control

Originally, the Mississippi River was hydrologically linked to its delta through many distributaries, which provided river water to the deltaic plain and its vast wetlands during the high-river stage (Lane et al. 2006). Upon colonizing New Orleans in 1719, the French began constructing flood-control levees and closing distributaries, though it was not until after the great flood of 1927 that the levees were upgraded and the Mississippi River was completely separated from the delta. Dredging and construction for the oil and gas industry have further altered the natural hydrology of the delta. These human modifications prevent seasonal flooding and the introduction of nutrients and sediments into delta wetlands, contributing to high rates of marsh deterioration (DeLaune et al. 2003).

Long-term stability of marsh is sustained when the wetland surface elevation gain is equal to or greater than relative sea level rise. When a marsh receives little sediment, accretion processes will slow or stall.[29] Relative sea level rise is the sum of global sea level rise and the local lowering of the wetland surface, which may exceed 1 centimeter per year in the Mississippi Delta.[30] Thus, sediment starvation will make marsh more susceptible to "drowning" as relative sea level rise may outpace elevation gain. The annual deposition of sediment can be returned to marshes by breaching levees along the shipping canals and restoring tidal marsh accretion processes. This task can be accomplished with water diversion structures, some of which have been installed along the lower Mississippi River. These efforts typically involve state and federal agencies as well as private stakeholders. Monetizing carbon benefits from protecting coastal marshes could help finance those efforts.

Potential carbon losses

The coastal marsh can erode and become open water when relative sea level rise outpaces the vertical accretion of the marsh surface. This phenomenon leads to death of marsh vegetation and, by extension, the halt of annual carbon sequestration performed by the habitat. Carbon dioxide emissions are released as the vegetation decomposes and as sediments lose the stability provided by the vegetation and their soil carbon washes away or decays *in situ*. The

29. An extreme example of this phenomenon is the Yangtze River delta, where coastal wetlands has receded rapidly as riverine sediment delivery has dropped considerably since the 1960s, mostly due to dam construction. The sediment load was 35% of the 50-year average in 2004 before the full impact of the Three Gorges Dam (Yang et al. 2006).
30. This lowering is often termed "subsidence" and is due to natural factors such as sediment-compacting tectonic activity as well as human factors such as oil and gas withdrawal.

resulting carbon loss could be as high as 950 t CO_2e/ha; over 95% of this loss comes from the soil carbon stores. Annual sequestration of about 8 t CO_2e/ha would also be forgone. Efforts that avoided this marsh habitat loss would allow carbon sequestration to continue and keep marsh biomass and soil carbon intact.

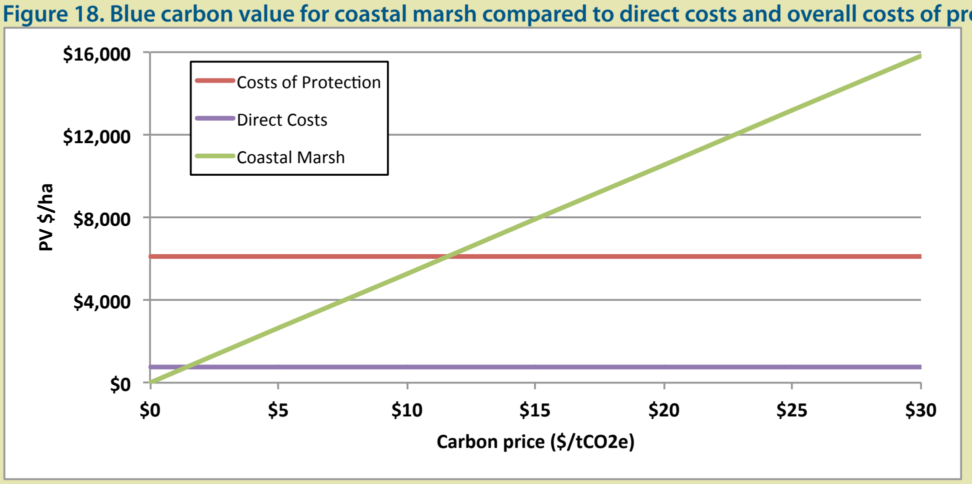
Blue carbon valuation

The blue carbon value of maintaining the coastal marsh habitat intact would depend on three factors: the future expected amount of carbon that would be emitted if the habitat were lost, the carbon that would be sequestered if the habitat were conserved, and the carbon price over time. Under scenarios in which carbon prices range from \$5 to \$30/t CO_2e, the market value of protecting coastal marsh would be between \$2,600 and \$16,000 per hectare.

To assess the financial viability of efforts to stop marsh loss, we compare the blue carbon value of habitat protection to the costs of protection. In this case, marsh habitat is sustained by a diversion structure that introduces freshwater and sediment from the main channel of the Mississippi River into the delta. The direct costs for this structure would be its construction costs as well as annual maintenance and operation costs. The Caernarvon Freshwater Diversion structure was completed in 1991 at a cost of \$41.1 million in 2009 USD (U.S. Army Corps of Engineers 1998). One study estimated that the Caernarvon diversion could maintain 54,100 to 92,300 hectares of fresh and brackish marsh (Lane 2006), making the per-hectare costs for construction between \$445 and \$760. Data on annual costs associated with the Caernarvon diversion were not available, so we consider the estimated costs of construction to be a lower bound on direct costs, as indicated by the purple horizontal line in Figure 18. If the \$760 estimate reflected the only relevant costs for implementing a water diversion structure, the break-even price would be under \$1.50 t CO_2e.

For a more complete assessment of costs, the opportunity cost of river diversions would be included. The wetlands that would receive Mississippi River water may be private or public land, but in either case, raising oysters appears to be the most profitable alternative use. Thus, the opportunity cost for protecting marsh habitat would be equal to the value of oyster leases for those wetlands, given that the freshwater diversions would kill the oyster beds when released. It has been estimated that the average Louisiana annual oyster lease is worth almost \$589/ha (Meitrodt and Kuriloff 2003). The present value of the estimated opportunity cost is \$5,350/ha; adding that value to the direct costs, we find the overall costs of protection to be about \$6,110/ha. Weighing the blue carbon value of intact coastal marshes against the total costs of protection yields a break-even price of \$11.58/t CO_2e.

Figure 18. Blue carbon value for coastal marsh compared to direct costs and overall costs of protection.



Other ecosystem service values

When salt marsh is protected, not only does carbon remain intact in sediments, but the provision of other ecosystem services continues. Salt marshes often serve as nurseries for commercial or recreational fisheries. For instance, in the blue crab fishery along the Gulf Coast of Florida, an additional hectare of coastal marsh produces about 6 pounds of blue crab annually and is valued at between \$1.50 and \$15.00 per year (Lynne et al. 1981; Ellis and Fisher 1987). Regarding the recreational catch of estuarine-dependent finfish in Florida, one study estimated annual values for the East and West Coast to be \$465 and \$3,066 per hectare, respectively (Bell 1997). Other coastal marsh ecosystem services include coastal protection, wastewater treatment, and wildlife habitat.

Pressure on mangrove forests: Shrimp farming in tropical Asia

Historically, the largest threats to mangroves have been over-exploitation for fuelwood or timber and conversion to rice production (FAO 2007; Thornton et al. 2003). In the mid-1970s, intensive aquaculture, especially shrimp farms, began to displace mangrove forests. By the mid-1980s, large-scale mangrove conversions to intensive shrimp-farming operations had become common in parts of Asia and central and South America. This trend was pushed by rising demand in many markets, which drove up the price of shrimp and encouraged growth in supply. The high-intensity nature of the shrimp farms made them prone to disease and other problems, often limiting periods of economic productivity to only 5 to 10 years per aquaculture site. Although some larger operations appear to have adopted more sustainable practices, and some shrimp farms have even located inland from the coast, much of the world's shrimp supply continues to come from poorer, smaller farms that have not undertaken these improvements (Stokstad 2010). World shrimp aquaculture production has increased substantially, from about 500,000 tonnes in 1988 to over 2.8 million tonnes in 2008, according to the NOAA Fishery Database. Over 80% of that production in 2008 was in Asia; China, Thailand, and Indonesia accounted for 26%, 18%, and 14% of production, respectively. A 2001 study estimated that about 38% of global mangrove loss is attributable to the clearing of mangroves for shrimp culture, while another 14% is due to other aquaculture (Valiela et al. 2001). In this case study, we focus on mangroves in tropical Asia, which contains nearly 50% of the world's mangrove forests (FAO 2007; Giri et al. 2010).

Carbon losses

Converting mangrove forests to shrimp aquaculture typically involves the burning of mangrove biomass, excavation and piling of approximately the first meter of sediment, and complete alteration of the local hydrology. Mangrove conversion of this sort results in the release of previously stored carbon into the atmosphere as gaseous CO_2, from both biomass and soil, as well as an abrupt stop to annual carbon sequestration. These emissions may amount to roughly 1,500 to over 2,000 tonnes of CO_2e. A project that avoided mangrove conversion would forgo habitat loss, thereby allowing carbon sequestration to continue while preventing the release of carbon dioxide from the mangrove biomass and soil. The avoided release of carbon and continued carbon sequestration associated with the project would then potentially be creditable under a carbon payment system.

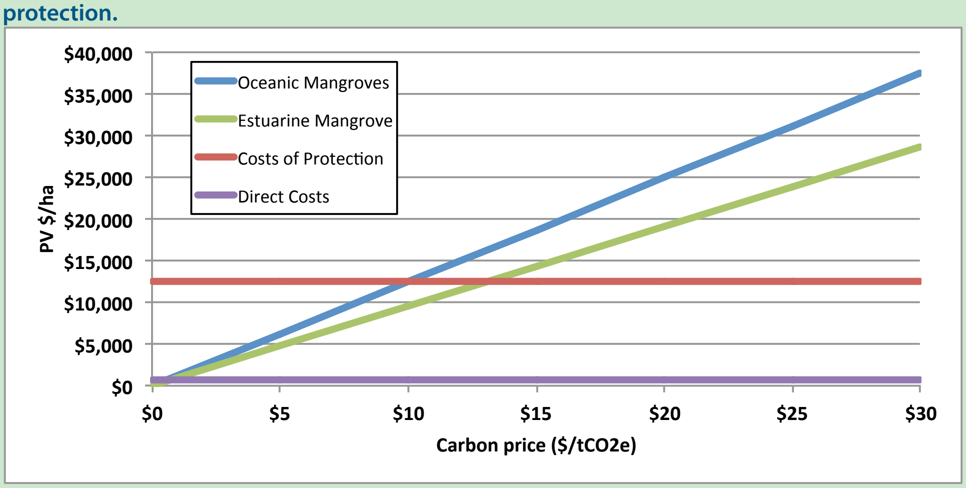
Blue carbon valuation

The carbon value of an avoided mangrove conversion project would depend on three factors: the future expected amount of carbon that would be released under conversion, the carbon that would be sequestered without conversion, and the carbon price over time. For oceanic and estuarine mangroves in tropical Asia, the market value of the carbon that would be kept from the atmosphere by forgoing conversion would be between $5,000 and $37,000 per hectare under scenarios in which the price of carbon ranges from $5 to $30/t CO_2e (see Figure 19).

Protection of mangrove habitats would make financial sense if the revenues that could be earned from carbon markets exceeded the costs of protection. In Figure 19, we compare the blue carbon value to the costs of protection. We estimate the direct costs of protection by considering estimates of the costs for the establishment and ongoing management of protected areas in coastal areas. Values from the literature indicate that these costs are about $232 and $454 per hectare, respectively. Total direct costs are $686 per hectare and are represented by the purple horizontal line in Figure 19. Break-even prices, at which the financial payment from avoiding carbon releases just offsets the direct costs of mangrove protection, are under \$1/t CO_2e for both oceanic and estuarine mangroves in this case.

A more complete comparison would include the opportunity cost of protecting habitat—the financial value that reflects the most valuable alternative use of the area. For mangroves in tropical Asia, shrimp farming is often that use. From the literature, the net present financial value (adjusted to 2009 U.S. dollars) of shrimp farming has been estimated at \$11,735/ha (Sathirai and Barbier 2001). Adding that value to the direct costs, the overall costs of protection become \$12,421/ha (red line), and break-even prices are \$9.95 for oceanic mangroves and \$13.00 for estuarine mangroves. Therefore, avoiding mangrove conversion could be financially attractive to a landholder at carbon prices above the break-even points, which are below where CDM-certified emissions reductions were trading in early 2011 (about \$15–18/t CO_2e).

Figure 19. Blue carbon value for oceanic and estuarine mangroves compared to direct costs and overall costs of protection.



Compared with estuarine mangroves, oceanic mangroves would more likely be targeted for protection efforts because positive net returns would be achieved at a lower price point. If the value of the forgone economic opportunity (for example, beach resort development) were higher than shrimp farming, the required break-even price would be higher as well. The opposite would be true if the forgone opportunity cost were lower.

Other ecosystem service values

When mangroves are conserved, not only have carbon emissions been avoided, but the provision of other ecosystem services continues. For instance, mangroves often serve as nurseries for fisheries of local or regional importance. In the Rekawa lagoon system in Sri Lanka, the mangrove-lagoon fishery engages up to 250 local people and is valued at \$294/ha (Gunarwardena and Rowan 2005). In addition, the catch of mangrove-dependent species in the coastal fishery yielded a value of about \$542/ha. Mangroves forests may also protect coasts and the lives and goods of people who reside there. Following a super cyclone that ravaged southern India, researchers found that villages with greater widths of mangroves buffering them from the coast suffered fewer deaths than those with lesser widths or no mangroves (Das and Vincent 2009). The life-saving effect of the remaining mangroves was estimated to be 0.0148 lives per hectare.[31] Other potential mangrove ecosystem services include sustained yield of wood for timber or fuel, biofiltration, habitat protection, and recreation.

31. Das and Vincent observe that agricultural land near the mangrove forests was worth about 173,000 rupees per hectare (in 1999), which means that "the average opportunity cost of saving a life by retaining mangroves was 11.7 million rupees per life saved. This opportunity cost is less than the value of reductions in mortality risks implied by wage differentials in India, which has been estimated as ranging from 13.7–14.2 million rupees to 55.5–60.6 million rupees per avoided death…"

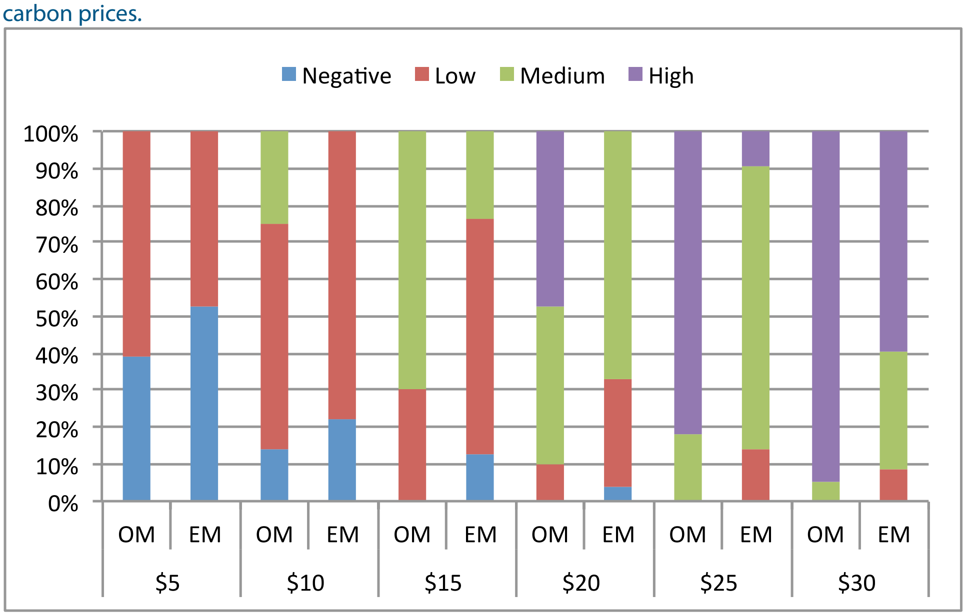
12. Global Market Supply Potential for Mangrove Ecosystems

Up to this point, we have compared the biophysical and economic dimensions of blue carbon mitigation broadly across the three habitat types. This section focuses on mangroves, for which blue carbon opportunities are greatest, for which the data are most detailed, and to which current policy appears most applicable. Lack of country-level data for the areal extent of seagrasses and salt marshes make meaningful, regional or country-specific analyses of net returns to blue carbon investments for those ecosystems difficult. However, as data become available, the approach used here for mangroves would serve as a template for assessing the potential mitigation supply of seagrasses or salt marshes at the country scale. The costs of protection used here are mean values for each country, and they consist of protected-area establishment and management costs plus opportunity costs that are the average of the World Bank and GTAP land values discussed above. These same costs serve as the foundational costs for all of the results discussed in Section 10.

Figure 20 provides some insights into the economic potential of mangrove protection for blue carbon across countries at different carbon prices. We can see that oceanic mangroves (OM in Figure 20) generally have net returns superior to those for estuarine mangroves (EM) because they harbor greater soil carbon stocks in the top meter of sediments (see Figure 7)—not because their protection costs (the mean values used here) are lower. We assume that these costs are the same for both types of mangroves.

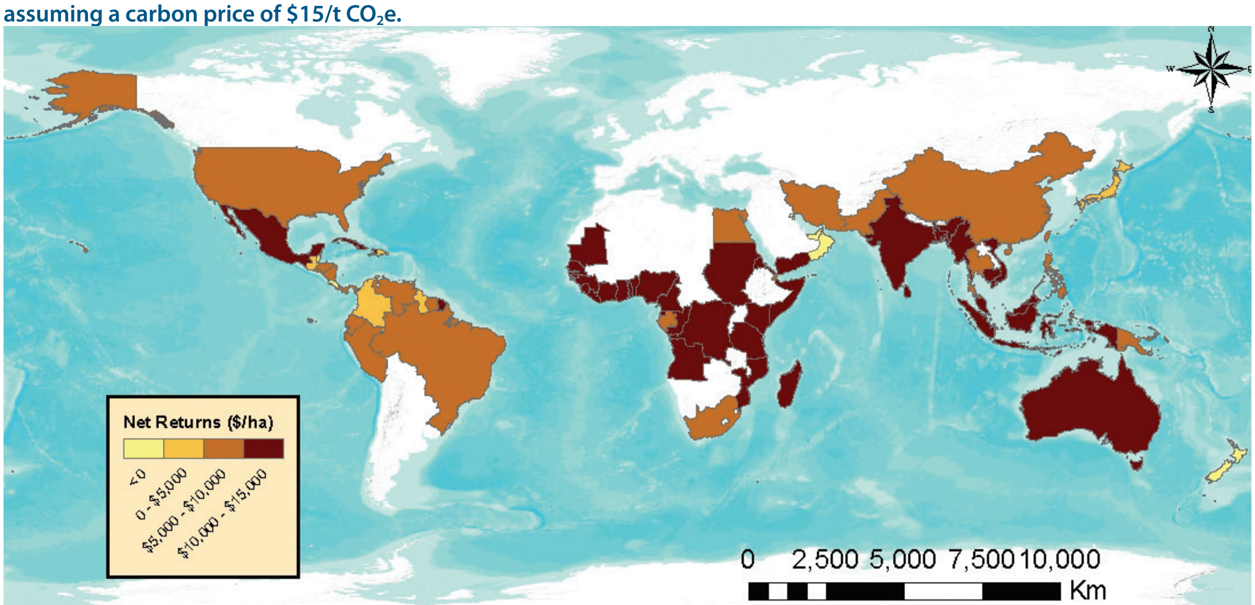
Net returns to blue carbon investments are negative in about 40% to 50% of mangrove countries at \$5/t CO_2e, though those percentages drop to about 15% or less when the carbon price reaches \$15/t CO_2e. Although low blue carbon value potential (\$1–\$10,000/ha) is common among countries at low carbon prices, the majority of countries have net returns of at least \$10,000 to \$20,000/ha starting at \$15/t CO_2e for oceanic mangroves and at \$20/t CO_2e for estuarine mangroves. High returns, those between \$20,000/ha and \$30,000/ha, are found for oceanic mangroves in 82% and 94% of countries at prices of \$25/t CO_2e and \$30/t CO_2e, respectively. Estuarine mangroves are not quite as strong a return; 10% and 60% of countries are in the high-return category at those same carbon prices.

Figure 20. Mangrove country-level blue carbon results. For oceanic mangroves (OM) and estuarine mangroves (EM), the percentages of countries with blue carbon net returns (benefits minus costs of protection) in various ranges for different carbon prices.



We continue our assessment of mangrove blue carbon on the global scale by examining the distribution of returns across the world. The map in Figure 21 displays the net returns per hectare (NPV\$/ha) for each mangrove country at a carbon price of \$15/t CO_2e; higher returns are depicted by the darker shades. We use a weighted average for mangrove blue carbon benefits—one based on estimates of the relative proportion of oceanic and estuarine mangroves in different regions. Overall, there appears to be economic potential for blue carbon projects worldwide; 31% and 54% of countries show net returns of \$5,000/ha to \$10,000/ha and \$10,000/ha to \$15,000/ha, respectively. The tropical Americas tend to have lower returns than other regions because estimates of their opportunity costs are generally higher.[32] A notable exception is Mexico. With net returns just over $10,000/ha, it has the second highest overall mitigation potential. Countries with subtropical mangroves, such as United States and China, have moderate returns to blue carbon investments. In the highest-return category are the majority of countries in Africa and tropical Asia, the regions with the most mangrove blue carbon opportunity on a country-level, benefit-cost basis. This finding is partially explained by the somewhat higher carbon pools in mangroves of those regions, though the main factor is that opportunity costs tend to be lower, on average, in countries of these regions.

Figure 21. Net returns ($/ha) to mangrove avoided-conversion efforts per hectare for all mangrove countries of the world, assuming a carbon price of $15/t CO_2e.

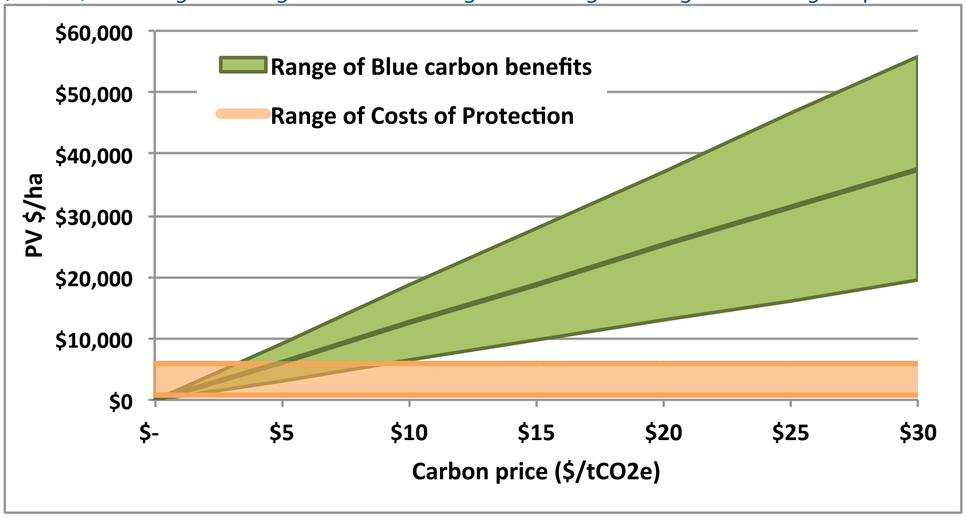


We highlight the most compelling regional results for mangrove blue carbon in Figure 22. This graph displays the range of blue carbon values for oceanic mangroves in tropical Africa over a carbon price range of $0/t CO_2e to $30/t CO_2e intersected by the range of costs of protection for the region. The top line of blue carbon value (the green triangle) represents a mangrove forest with relatively high carbon storage (50% above the mean); the bottom line represents one with low carbon storage (50% below the mean). The mean estimate for carbon storage, which is used in Figure 21 and Figure 22, bisects the green triangle. Regarding the costs of protection (orange rectangle), the top line of costs are derived from the average for tropical African mangrove countries from the World Bank land values as opportunity costs and the bottom line from the mean of the GTAP African land values; costs of protected-area management and establishment are the same in each case. The vast majority of the benefits triangle lies above the top line of the cost range, indicating that the returns would generally be positive for mangrove blue carbon investments over this price range.

Break-even prices, the point at which blue carbon value equals the costs of protection, can further elucidate key thresholds for blue carbon returns. When carbon stocks and costs of protection are both in the middle of the range, the break-even price is $2.65/t CO_2e. In the most optimistic scenario, carbon storage values would be high and protection costs would be low, equating to a break-even price of $0.39/t CO_2e here. On the other end of the spectrum, the least attractive scenario would involve low carbon stocks and high protection costs, and the break-even price would be $9.15/t CO_2e. Thus any carbon price above $9.15/t CO_2e would result in a positive return, whether the actual mangrove carbon storage ended up being low, high, or anywhere in between. Results for oceanic mangroves in tropical Asia are nearly as compelling, though higher expected costs of protection for Asian countries mean a higher cost range. The resulting break-even prices are $0.56, $4.91, and $17.37 for the most optimistic, mean, and least optimistic scenarios, respectively. These cost-per-tonne numbers are in line with earlier country-level average cost estimates for emission reductions from avoided deforestation in the tropics (Murray, Lubowski, and Sohngen 2009[33]).

32. World Bank land value estimates, which are averaged with the GTAP values to calculate opportunity costs, tend to be high for many Latin American countries.
33. Table 2.3 in that report shows carbon sequestration cost estimates from country-level research studies with average costs ranging

Figure 22. Blue carbon results for oceanic mangroves in tropical Africa. The green triangle represents blue carbon benefits (PV$/ha) for a range of mangrove carbon storage. The orange rectangle is the range of protection costs for the region.



We turn now to results for key mangrove countries. Table 6 lists total mitigation potential and break-even price estimates for the 25 countries with the largest current emissions from mangrove loss.[34] These 25 countries represent almost 92% of the global mangrove mitigation potential. For each country, the break-even price is the point at which blue carbon value equals the cost of protection. The cost of protection is equivalent to the costs of establishing and managing protected areas plus the average of the opportunity costs derived from World Bank land values and from GTAP land values. Annual mitigation potential for mangroves is the weighted average of oceanic and estuarine mangroves by mangrove region. The results are for the mean estimated values of carbon storage and protection costs for each country. A sensitivity analysis in Figure 24 provides a range of outcomes.

As indicated previously, Indonesia has the largest annual mitigation potential of any country, with nearly 30 million tonnes of emission reductions from avoided mangrove conversion, roughly equal to the yearly emissions from all sources (except land use) in New Zealand or Ecuador. The average break-even price for Indonesia is about $4 per tonne of emissions avoided, which is in the top third of all country-specific estimates in Table 6. Eight countries offer cheaper emissions reductions than Indonesia. Investors in carbon reductions will likely focus their efforts on countries that appear to have the least expensive potential, though countries with large-scale mitigation potential may have lower transaction costs. Overall, the cheapest reductions ($1.70 per tonne) are found in Senegal, while the most expensive are found in Colombia ($11.31). Five of the top seven countries with the lowest break-even prices are in tropical Africa. This result primarily reflects protection costs in Africa, which are the lowest of any region's, but also reflects the fact that African oceanic mangroves have the highest blue carbon values. The region with the most mitigation potential is tropical Asia. That potential—well over half of the world's—is largely explained by Indonesia's contribution, but three other tropical Asian countries are in the top five by mitigation potential: Papua New Guinea, Malaysia, and Vietnam.

from roughly $2 to $8 per tonne (2005 dollars). Subsequent supply function analysis uses more spatially refined data to show a wider range of costs, with prices of $10 to $30 per tonne needed for large-scale global supply.

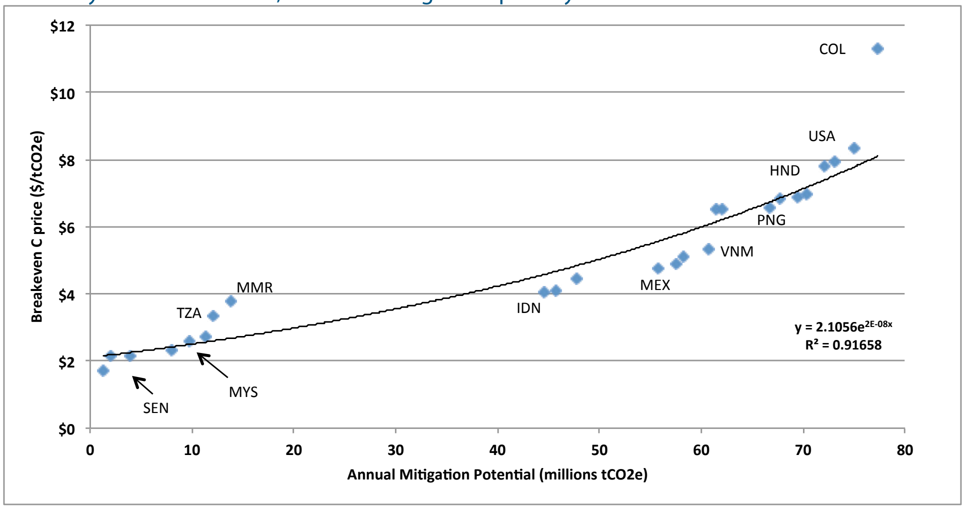
34. As with the results above, the mitigation potential estimates account for the time path of carbon release and thus have been discounted using a 10% rate over a 25-year period. In addition, the biophysical mitigation potential estimates in Table 6 reflect country-specific annual mangrove loss rates and have been adjusted to represent only lands that are at risk of conversion to agriculture, aquaculture, or wood harvests (see Figure 9).

Table 6. Top 25 mangrove countries by break-even carbon price ($/t CO_2e). Mitigation potential (t CO_2e/yr) is discounted to the present with a 10% rate as well as scaled down to represent only lands at risk from conversion to agriculture, aquaculture, or wood harvests.

Country	Break-even C price, avg cost	Discounted mangrove mitigation potential	Cumulative mangrove mitigation potential	Annual mitigation revenue potential, $/yr @$15/t CO_2	Total mangrove area, 2005 (ha)	Annual Revenue potential ($/ha)
Senegal	$1.70	1,342,843	1,342,843	$22,380,714	115,000	$194.61
Cambodia	$2.14	692,276	2,035,119	$11,537,939	69,200	$166.73
Guinea-Bissau	$2.16	1,832,201	3,867,320	$30,536,677	210,000	$145.41
Malaysia	$2.34	4,181,896	8,049,216	$69,698,271	565,000	$123.36
Sierra Leone	$2.60	1,716,291	9,765,507	$28,604,843	100,000	$286.05
Madagascar	$2.73	1,539,227	11,304,734	$25,653,783	300,000	$85.51
Tanzania	$3.35	755,870	12,060,604	$12,597,840	125,000	$100.78
Myanmar	$3.78	1,790,324	13,850,928	$29,838,734	507,000	$58.85
Indonesia	$4.04	30,679,644	44,530,572	$511,327,397	2,900,000	$176.32
India	$4.10	1,133,760	45,664,332	$18,896,005	448,000	$42.18
Pakistan	$4.46	2,026,638	47,690,970	$33,777,304	157,000	$215.14
Mexico	$4.74	8,137,233	55,828,204	$135,620,556	820,000	$165.39
Gabon	$4.90	1,698,338	57,526,542	$28,305,641	150,000	$188.70
Nicaragua	$5.13	681,651	58,208,193	$11,360,853	65,000	$174.78
Vietnam	$5.32	2,564,008	60,772,201	$42,733,462	157,000	$272.19
Ecuador	$6.53	684,104	61,456,305	$11,401,728	150,500	$75.76
Thailand	$6.53	603,800	62,060,105	$10,063,336	240,000	$41.93
Papua New Guinea	$6.58	4,570,866	66,630,971	$76,181,108	380,000	$200.48
Venezuela	$6.83	1,124,822	67,755,793	$18,747,035	223,500	$83.88
Philippines	$6.90	1,762,242	69,518,035	$29,370,699	240,000	$122.38
Brazil	$6.98	872,828	70,390,863	$14,547,128	1,000,000	$14.55
Honduras	$7.83	1,631,183	72,022,046	$27,186,382	67,200	$404.56
Panama	$7.95	1,056,887	73,078,933	$17,614,785	170,000	$103.62
United States	$8.34	1,953,947	75,032,880	$32,565,786	195,000	$167.00
Colombia	$11.31	2,261,764	77,294,644	$37,696,062	350,000	$107.70

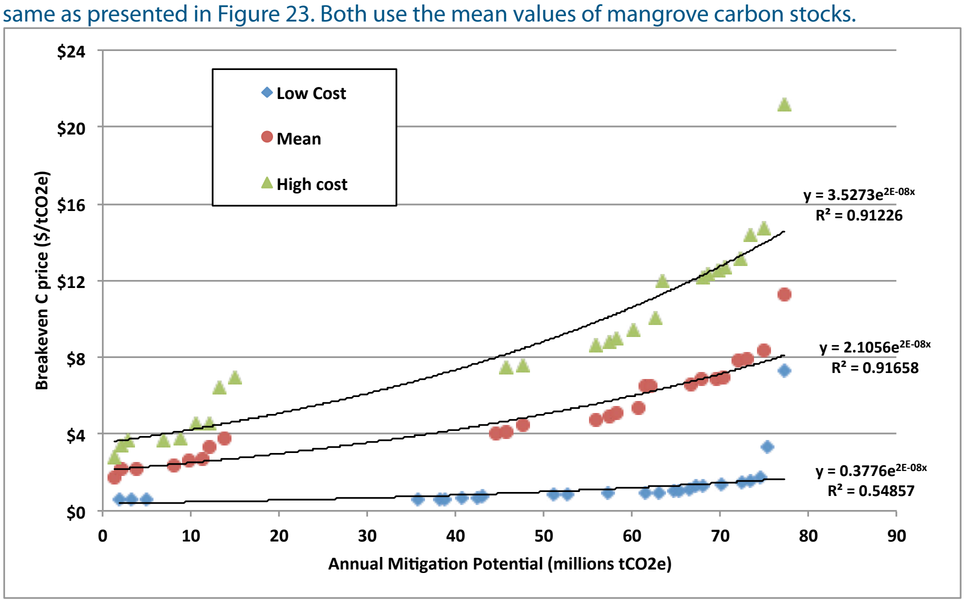
We combine the break-even prices and mitigation quantities to construct an emission reduction *supply curve* for avoided mangrove destruction in these 25 countries (see Figure 23). This supply curve roughly captures how much mitigation quantity could be provided by these countries if they were compensated at levels sufficient to cover their costs of protection. These estimates are rough, because each country is treated as if it has one break-even price and one mitigation quantity it can supply at that price. In reality, each country has its own internal supply function, reflecting its range of protection costs and mitigation quantities, but we do not have the data to explore each country individually. Therefore, the function in Figure 23 provides a relatively global sense of how rising levels of compensation could, on average, bring on more countries and more mitigation. For instance, at a carbon price of $4/t CO_2e, about 40 million tonnes of mitigation might be supplied. At $6, another 20 million tonnes would come on, pushing the total to 60 million t CO_2e. At $8, the total supplied would approach 80 million tonnes. Because this supply function incorporates almost 92% of global mangrove mitigation potential, we would not expect much change in the shape of the curve or mitigation supply if we expanded the number of included mangrove countries.

Figure 23. Supply function of mitigation potential. Relates the annual mitigation potential (millions t CO_2) to break-even carbon prices (\$/t CO_2), accumulated across each of the key mangrove countries. As the price rises, more countries can economically reduce emissions, and total mitigation quantity rises.



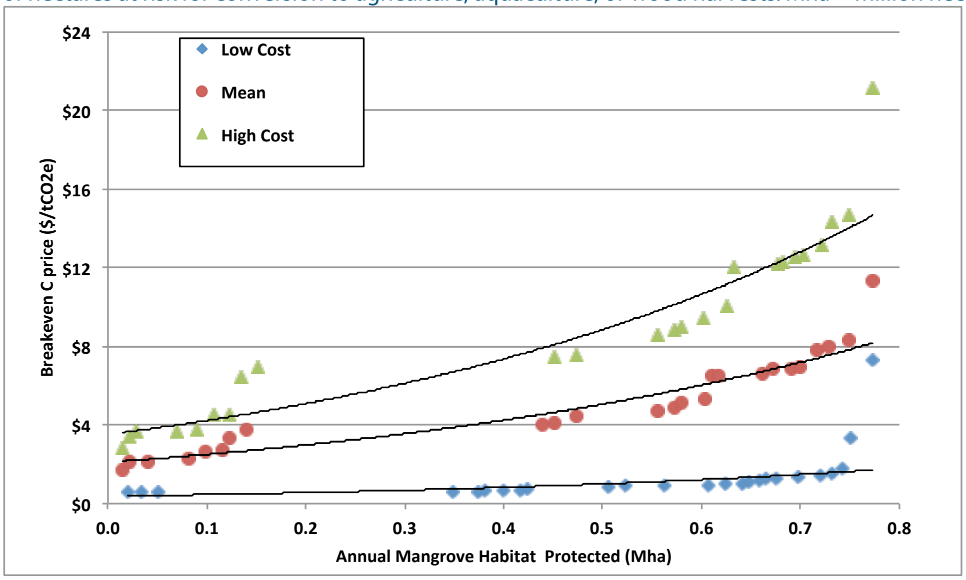
The supply function in Figure 23 reflects all economic and biophysical parameters at their mean values for each country. Figure 24 augments the story by showing the same "mean" supply function positioned between a version of the function that reflects the high-end protection cost estimate for each country as well as the low-end cost estimate. The result is quite a range of outcomes. For example, a price of \$4/t CO_2e is sufficient to supply around 40 million tonnes of mitigation in the mean supply case, but only about 10 million tonnes in the high-cost case. For the same comparison at \$8/t CO_2e, the mean case supplies nearly 80 million tonnes, while the high-cost case yields approximately 45 million tonnes. The low-cost supply function estimate "peaks" at about \$2 per tonne: prices above this level do not elicit more mitigation, as all economically viable mitigation opportunities have been exhausted.

Figure 24. Mitigation potential supply functions for low-cost, mean, and high-cost scenarios. The low-cost scenario assumes protection costs 50% lower than the mean costs. The high-cost scenario assumes protection costs 50% higher than mean costs. The mean supply function uses mean costs (the average of World Bank and GTAP opportunity costs) and is the same as presented in Figure 23. Both use the mean values of mangrove carbon stocks.



Shifting the focus to the total area of habitat that can be conserved, Figure 25 illustrates the quantity of habitat that might be protected each year through payments for blue carbon to the top 25 mangrove countries. This graph provides supply curves with the break-even prices shown on the Y-axis in Figure 24, but with millions of hectares of mangrove habitat at risk on the X-axis. Again, the range of supply from low- to high-cost scenarios is wide, especially at higher levels of habitat protection. At $4 for a tonne of CO_2e, the mean case would supply almost 0.4 Mha (or 400,000 hectares) of protected mangrove habitat, whereas the high-cost case would yield less than 100,000 ha. The $8/t CO_2e mark would yield nearly 800,000 ha for the mean case and around 450,000 ha for the high-cost case. In the low-cost scenario, almost all habitat at risk could be protected when the carbon price reaches $2. Overall, if a GHG compliance market allowing blue carbon offsets were to come about, a carbon price of $15/t CO_2e may not be unreasonable and could be sufficient to protect all mangrove habitat considered here, even assuming high costs.

Figure 25. Mangrove habitat protection (area) supply functions for low-cost, mean, and high-cost scenarios. This figure uses the same cost scenarios and mean mangrove carbon stock values as Figure 24. Mangrove habitat is the annual amount of hectares at risk for conversion to agriculture, aquaculture, or wood harvests. Mha = million hectares.



13. Research Needs

Both the natural science and economics of blue carbon are at a nascent stage. Additional work needs to be done in many areas, such as geography, biogeochemistry, and economics, to reduce uncertainties and move future analyses onto firmer ground. We identify these needs by category below.

Improved scientific understanding

Our research has uncovered a number of critical issues that need to be explored to improve the scientific basis for blue carbon assessment and solutions. They include

- Delineation of carbon pools:
  - Improve understanding of how soil organic carbon (SOC) density may change with soil depth.
  - Improve knowledge of variation in SOC, especially for seagrasses, and in biomass C across regions to arrive at estimates of representative carbon content by region (we lack carbon estimates for some of the most expansive focal habitats: seagrass in the Indo-Pacific and salt marsh C in Australia).
  - Determine the carbon stocks and rates of sequestration for the land uses that replace coastal ecosystems.
- Better understanding of the temporal profile of carbon losses, both by habitat and conversion type (many assumptions made in this study are based on limited evidence or best professional judgment).
- Better understanding of the effects of uncertainty on economic and ecological analyses of blue carbon.

More complete and comprehensive data

In some cases, scientific knowledge is strong in the places blue carbon systems are studied, but data to extend this knowledge to or interpret it in other places are lacking. And there are wide data gaps on disturbance factors and drivers and on other economic phenomena.

Areal extent
More spatially explicit datasets of areal extent, especially of seagrass and salt marsh habitats, are needed. Such datasets are essential even for mangroves, the most data-rich of these systems. Note the wide discrepancies between the FAO (2007) and Giri et al (2011) datasets on key countries such as Australia and Nigeria. Even if habitat areas are considered intact (depending on mapping criteria), they may be degraded to some degree, which would affect ecosystem functions, including carbon sequestration or storage. Estimating the proportion of extant habitat that is degraded would be a next step.

Drivers of conversion and disturbance
More information is needed on several of the factors underlying conversion drivers, including

- Amount of actually occurring disturbance for different drivers (we assume a disturbance depth of one meter as a rough estimate across many driver types)
- Land values/opportunity costs
  - Among the few existing datasets based on agricultural returns, land values vary widely.
  - There is no dataset for coastal areas (worldwide or otherwise), which are perhaps the most fragile and endangered habitats and a key to climate adaptation.
  - There has been no research in literature on opportunity costs for seagrass protection.
  - Study of local opportunities requires data for the area in question rather than the national numbers used in this study.
- Establishment costs
  - There are limited data on protected-area establishment costs and transaction costs for blue carbon projects and on how these costs might compare with those for other project types.

More targeted analysis
Better analyses of current drivers of habitat loss and quantification of the relative contribution of each are needed. Drivers, particularly those of seagrass and salt marsh habitat loss, are poorly understood. The information is a bit better for mangroves, though variation across countries or regions should be explored. Ideally, remote sensing studies would be complemented by ground truthing.

The average-cost break-even price analysis here provides a very rough first-order approximation of the economic potential for blue carbon mitigation, but it ignores much of the underlying heterogeneity of biophysical and economic processes. Every country will have a range of conditions that will make some blue carbon emissions relatively cheap and some blue carbon emissions relatively expensive to mitigate. Moreover, economic behavior is complicated: we cannot simply expect all agents to go "all in" if break-even price conditions are met. More sophisticated models of supply behavior at different prices are required to more realistically capture likely responses to economic incentives for blue carbon.

We need better understanding, both biophysical and economic, of habitat protection's co-benefits—that is, ecosystem services other than carbon storage and sequestration. These services (and especially their economic values) are often location-specific, so additional research should be conducted across various locations and regions.

Improved understanding of institutions
Legal or *de facto* ownership of mangroves, salt marshes, and seagrasses are not always well-established, yet the definition or pattern of ownership in individual countries will likely affect the design of payment systems. The issue of property rights raises several political/institutional questions, including:

- How do these rights relate to the global politics of international carbon transfers?
- How do they affect current habitat conversion behavior?
- How can country-level institutions be designed to efficiently mobilize carbon protection?

14. Conclusions

Because they store large amounts of carbon and are severely threatened in their natural state by the economic allure of conversion, coastal ecosystems could be an ideal target for carbon financing. Although data on on-site carbon storage, carbon emission profiles, and the costs of protection are somewhat elusive at this time, preliminary analysis of the type presented here suggests that blue carbon protection could be economically viable in some important cases (mangrove protection from agriculture and aquaculture uses in tropical Africa and tropical Asia) at the level of carbon prices we now see in world markets (approximately $15 to $25 per tonne in the EU ETS).

Any verdict on the economic viability of blue carbon is far from complete, however. To begin with, there are no markets for credits generated by blue carbon activity. A logical venue for considering blue carbon payments would be the United Nations Framework Convention on Climate Change (UNFCCC) process. At this time, the only blue carbon activities that might be covered under UNFCCC compliance markets are mangrove restoration through the CDM and mangrove protection under the auspices of the Reduced Emissions from Deforestation and Degradation (REDD+) initiative advanced in the Cancún Agreement. However, even if protection is allowed under REDD+, it is unclear whether avoided soil carbon emissions will be included, as the primary emphasis in REDD+ deliberations has been on aboveground carbon stocks. If soil carbon is left out, the attractiveness of mangrove protection would be diminished because a vast majority of the habitat's carbon is stored below ground. Regardless of whether REDD+ can incorporate mangrove protection, a broader consideration of all candidate blue carbon ecosystem types (including salt marshes and seagrasses) and certainly an emphasis on the belowground carbon components of these ecosystems would be in order to fully tap blue carbon mitigation potential. This broader consideration of blue carbon ecosystems could be coupled with stakeholder-driven efforts to include all wetlands, agriculture, and all other major land uses under the auspices of agriculture, forests, and other land uses (AFOLU) in global climate protection efforts.[35]

Aside from the compliance markets supported by UNFCCC action, there are other compliance markets in the United States, Canada, Australia, and elsewhere as well as voluntary markets in which blue carbon could be considered in principle. But as with the UNFCCC system, further work will be necessary to define the boundaries of blue carbon eligibility, to propose methods for quantification, and to seek approval for inclusion in the market. The science on blue carbon storage, drivers of conversion, and post-disturbance emission rates is nascent, and the available data are geographically incomplete. More effort will be necessary to expand the knowledge set, define gaps, and characterize the uncertainty attending serious consideration of blue carbon as a mitigation option in general. This effort will be particularly important if blue carbon protection and restoration efforts are to be tied to markets, multilateral funds, or other institutions in which money will change hands for specific outcomes such as tonnes of emissions avoided, carbon sequestered, or area protected or restored.

This report has focused on the biophysical and economic potential for emission reductions from the avoided conversion of blue carbon habitats. We focus on avoided conversion, which appears to have the highest potential for economic returns in the carbon market or in payment systems emerging internationally. Current rates of habitat loss are high, and the emissions effects are immediate. In principle, protection could be coupled with efforts to restore blue carbon by returning some areas converted to agriculture or other uses back to native habitats that once again become net carbon sinks and provide other ecosystem services. However, carbon removals associated with sequestration rates are much smaller than emission reductions from avoided conversion and, as such, restoration actions would likely have more modest economic returns, all else equal. Where habitat conversion has occurred recently and soil carbon releases are ongoing, restoration could gain carbon credits for halting emissions. Moreover, taking land out of agricultural or aquacultural development in places where it has already attracted migrating populations would be very difficult from a broader economic and social perspective.

Our focus on the value of carbon and climate regulation in coastal habitats is not meant to devalue the importance of the habitats' many other ecosystem services, such as providing wildlife and fish habitat and maintaining water quality. The value of these services, which are known to be substantial even if ignored by markets, may exceed both the habitats' developed value and value as a carbon store. But carbon is where the most readily available money to pay for protection may reside. If carbon payments create economic incentives sufficient to induce resource owners to forgo conversion or

35. See, in particular, the efforts of the Terrestrial Carbon Group to broaden the scope of land-based activities covered by international agreements, such as the UNFCCC.

undertake restoration efforts, these consequences are only improved by the fact that a whole range of other valuable ecosystem services would come along for the ride.

In many ways, the blue carbon issue is in the same place that REDD+ was about five to ten years ago. There is widespread recognition of the environmental and economic cost of coastal habitat destruction, frustration at the lack of success in curtailing these losses through traditional conservation measures, and hope that the value of these ecosystems as a store of carbon might align with international efforts to fight climate change and provide the economic incentives necessary to avoid continued loss. Economic incentives for REDD+ are moving toward reality as the UNFCCC has embraced it as part of the Cancún Agreement. For blue carbon to get to the same point as REDD+, the international community must grasp the extent of the problem and determine whether climate policy is an appropriate means to address it. This study, admittedly based on preliminary scientific and economic foundations, suggests that the scale of the problem is large enough to warrant attention, that there exists a logical and synergistic tie between coastal habitat protection goals and climate goals, and that there are economic opportunities to reduce blue carbon losses well in line with, or even less than, the costs of many other mitigation options in the land and energy sectors.

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