policy brief

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Payments for Blue Carbon
Potential for Protecting Threatened Coastal Habitats

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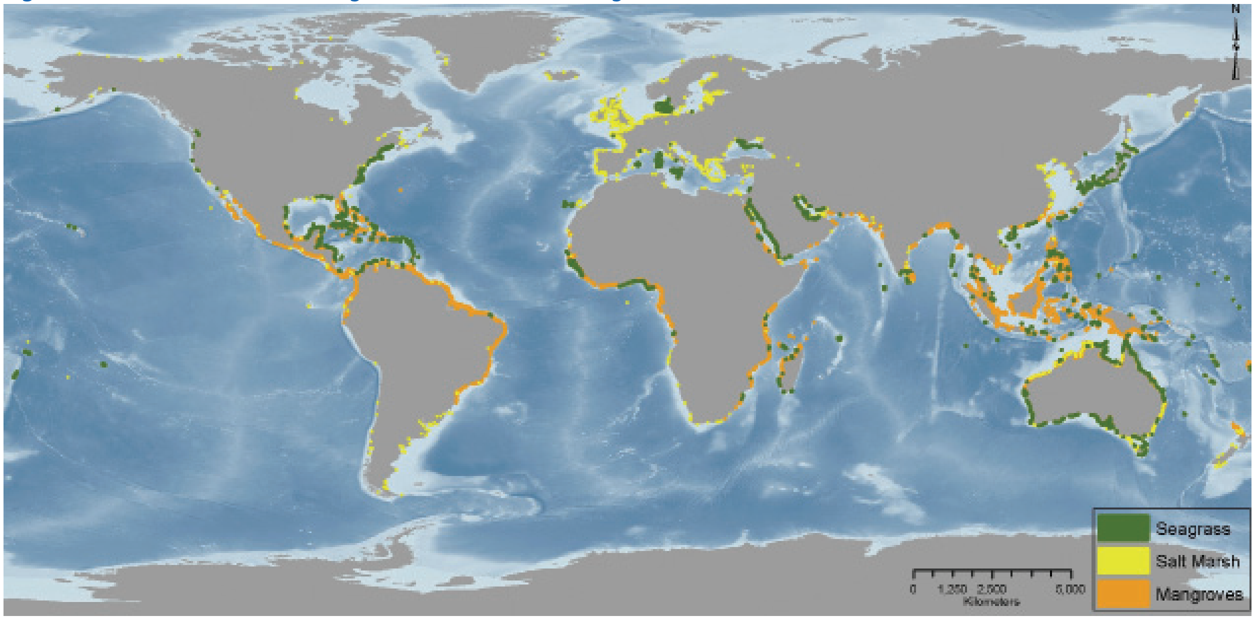
Coastal habitats worldwide are under increasing threat of destruction through human activities such as farming, aquaculture, timber extraction, or real estate development. This loss of habitat carries with it the loss of critical functions that coastal ecosystems provide: support of marine species, retention 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.[2] Because the value of these ecosystem services are not easily captured in markets, those who control these lands often do not consider these values when choosing whether to clear the habitat to produce goods that can be sold in the marketplace. This is a form of market failure that 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.

Figure 1. Global distribution of seagrass, salt marsh, and mangroves.



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

1 This brief is a modified version of a previous policy brief. It provides a preliminary assessment of research to date by the Nicholas Institute on the economics of blue carbon as a climate mitigation strategy. A final report of the analysis is forthcoming in early 2011. We are grateful to the Linden Trust for Conservation and Roger and Victoria Sant for their financial support of these efforts. We particularly wish to thank Roger Ullman and Vasco Bilbao-Bastida of the Linden Trust for their substantive insights on these issues, as well as participants in a workshop on the biophysical processes underlying the blue carbon issue held at Duke University on November 10–11, 2010, including Christopher Craft, Steve Crooks, Daniel Donato, James Fourqurean, Boone Kauffman, Núria Marba Bordalba, Patrick Megonigal, and Emily Pidgeon. We also acknowledge review comments and methodological contributions of Duke faculty members Randall Kramer and Jeffrey Vincent and the mapping of coastal habitats by Duke faculty member Patrick Halpin. David Cooley provided research assistance and Paul Brantley provided editorial support and document formatting.
2 E.B. Barbier, "Valuing Ecosystem Services as Productive Inputs," *Economic Policy* 49 (2007): 178–229.

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

A critical question is whether monetary payments for "blue carbon"—carbon captured and stored by coastal wetlands or avoided emissions from conversion—can rearrange economic incentives to favor protection of coastal habitats such as mangrove forests, seagrass meadows, and tidal 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 REDD, incentives to retain rather than emit blue carbon would preserve biodiversity as well as a variety of other ecosystem services at the local and regional scales. And like REDD, success will hinge on structuring the incentives to avoid negative impacts on the wellbeing of local populations who depend on these resources for their livelihoods.

Seagrass meadows are communities of underwater flowering plants found in coastal waters of all continents except the Antarctic. There are more than 60 known seagrass species and up to 10 to 13 of them may co-occur in tropical sites.

Tidal salt marshes are intertidal ecosystems occurring on sheltered coastlines ranging from the subarctic to the tropics, though most extensively in temperate zones. 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. They are estimated to occupy almost 40% of tropical coasts worldwide, down substantially from 75% in the recent past.

Coastal Habitats: Extent, Location, and Rate of Loss

Seagrass meadows can be found in shallow waters of all continents, while tidal 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. Tidal marshes and mangroves exist in the intertidal zone between land and sea, whereas seagrasses grow in shallow water (1–10 m) on the continental shelf and may be near or far from land. Current estimates of the global extent of these habitats are approximate at best, ranging from 137,000 to 170,000 km^2 for mangroves to up to 600,000 km^2 for seagrasses (see Table 1). Although together these habitats cover a relatively small area—less than 3% of the global coverage of forest (39.5 million km^2)[4]—they are some of the most threatened ecosystems in the world. Over the 1980–2000 period, mangroves have been experiencing annual loss rates of 0.8%–2.1% per year, driven mainly by aquaculture, forestry uses, and agriculture. Tidal marshes have historically been reclaimed for agricultural use and salt ponds, but continue to be converted at a high rate, approximately 1%–2% per year, often for real estate development. Globally, seagrass meadows are disappearing at a similarly rapid rate, at about 1.2%–2% per year since 1980, due predominantly 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 (km^2)	Conversion drivers	Annual Loss Rate (~1980–2000)	Total Historical Loss (%)
Seagrass	300,000–600,000[a]	Water quality degradation, mechanical damage	1.2%–2%[b]	29[c]
Tidal Marsh	400,000[d]	Historical reclamation for agriculture & salt ponds; real estate development	1%–2%[e]	Centuries of conversion[f]
Mangroves	137,000–170,000[g]	Aquaculture, forestry uses, Agriculture	0.8%–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;

3 The focus here is on CO_2 because methane, a more potent GHG with 25 times the global warming potential of CO_2, 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.

4 Food and Agriculture Organization (FAO), "Forest Resource Report" (Rome: Food and Agriculture Organization, 2005).

(d) Nellemann et al. 2009; (e) Adam 2002; Duarte et al. 2008; (f) Bromberg Gedan et al. 2009; (g) Valiela et al. 2001; FAO 2007; (h) Valiela 2001; FAO 2007; Giri et al. 2011; (i) Valiela 2001; Duke et al. 2007. See Appendix for complete references.

Carbon Storage in Coastal Ecosystems

Figure 2 and Table 2 provide estimates of carbon stocks and sequestration rates across the different regions of the world for each of the focal habitats. Coastal ecosystems remove CO_2 from the atmosphere via photosynthesis, return some to the atmosphere via respiration and oxidation and store the remaining carbon in two pools: living biomass (includes 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 stock, virtually all of the sequestration ends up buried in the soil carbon stock. This rate is assumed to be constant over time for the purposes of this paper.[5]

Figure 2. Global averages for carbon pools (soil organic carbon and living biomass) of focal coastal habitats.

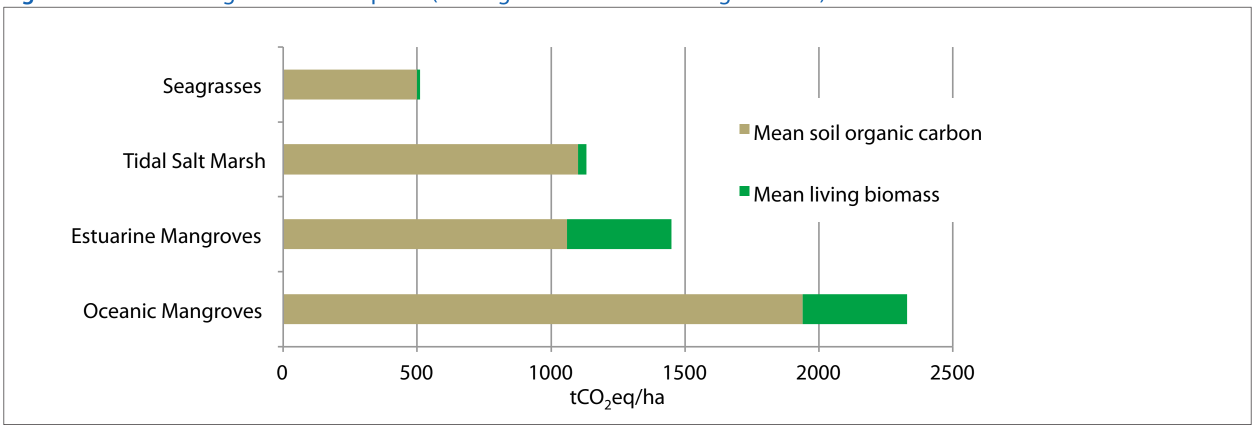


Table 2. Global averages and standard deviations of the carbon sequestration rates and global ranges for the carbon pools, by habitat type.

Habitat Type	Annual Carbon Sequestration Rate ($tCO_2eq/ha/yr$)	Living biomass (tCO_2eq/ha)	Soil organic carbon (tCO_2eq/ha)
Seagrass	4.4 ± 0.95[a]	0.4–18.3[b]	66–1,467[c]
Tidal Marsh	7.97 ± 8.52[d]	12–60[e]	330–4,436[f]
Estuarine Mangroves	6.32 ± 4.8[g]	237–563[h]	1,060[h]
Oceanic Mangroves	6.32 ± 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; 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. See Appendix for complete references.

Based on geomorphological differences, we divide mangroves into oceanic (or fringing), occurring along the ocean, and estuarine (or deltaic), growing where river deltas meet saltwater bodies. We present data on tidal *salt marshes only*, though *brackish* marshes may enter future research efforts. Although they are ecological communities with species compositions that vary geographically, seagrasses are reported here as one system type across their range.

Annual carbon sequestration rates do not vary much across the three coastal habitats, although there is great variation within each habitat type. Both tidal marshes and mangroves average between 6 and 8 tons of carbon dioxide equivalent (CO_2eq) per hectare per year, while seagrasses tend to accumulate carbon at a somewhat lower rate of approximately 4 $tCO_2eq/ha/yr$. These rates are about two to four times greater than global rates observed in mature tropical forests (1.8 to 2.7 $tCO_2eq/ha/yr$).[6] The amount of carbon held in living biomass is much more variable among the habitat types, with seagrasses containing 0.4 to 18.3 tCO_2eq per hectare and tidal marshes, on average, a few times higher than that at

5 In the case of mangroves, carbon burial rates seem to be keeping pace with sea level rise, though if sea level rise were to outstrip the elevation of the sediment surface, then the seaward and landward margins of the mangrove forest would retreat landward since the mangrove species need to maintain their preferred hydroperiod. D.M. Alongi, "Mangrove forests: Resilience, Protection from Tsunamis, and Responses to Global Climate Change," *Estuarine Coastal and Shelf Science* Vol. 76, No. 1 (2008): 1–13; E.L. Gilman, et al., "Threats to Mangroves from Climate Change and Adaptation Options: A Review, *Aquatic Botany* Vol. 89, No. 2 (2008): 237–250.
6 S.L. Lewis, et al., "Increasing Carbon Storage in Intact African Tropical Forests," *Nature* 457 (2009): 1003–1006.

12–60 tCO_2eq/ha. Mangrove forests, growing up to 8 meters tall, clearly lead in this area and maintain 237 to 563 tCO_2eq per hectare in living material. Soil organic carbon, however, is the biggest carbon pool for all the focal habitats, averaging from about 500 tCO_2eq/ha for seagrasses to almost 2000 tCO_2eq/ha for oceanic mangroves. In relative terms, over 95%–99% of total carbon stocks of tidal marshes and seagrasses are stored in the soils beneath them, while mangroves keep about 60%–80% in the soil carbon pool. It is important to note that these numbers only represent the carbon storage for the first meter of soil depth in order to facilitate consistent comparisons between habitat types. There can be great variability in the depth of the organic rich sediments underlying these habitats (some reaching depths of several meters), yet most have at least a meter. As a comparison, tropical forests with peat soils, as those found in Indonesia, may store around 2,900 tCO_2 per hectare, with about 75% of the total stock contained in the soil.[7] For non-peat soils, as common in the Amazon, wet tropical forests may harbor 1,250-1,400 tCO_2/ha, with soil carbon pool accounting for approximately 25% of the total.[8]

Carbon Loss after Disturbance

Two things happen after coastal ecosystems are disturbed or converted to an alternative use—the sequestration process removing CO_2 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. The magnitude and time pattern of the post-conversion CO_2 release depends on the type of coastal habitat disturbed as well as the type of disturbance. The disturbance class can determine to what depth the soil profile will be altered. This suggests how much soil carbon may potentially be exposed to oxygen and thereby emitted in the form of CO_2. Although meters of carbon-rich organic soils may underlie these coastal habitats, that carbon may persist if the habitat conversion only affects the top layers and the deeper layers remain inundated. Conversion of these habitats will often only disturb 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 (see case study below). In theory, following conversion, carbon in biomass is emitted to the atmosphere in the first few years. 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, high emission rates would be expected in the years immediately after disturbance, then dropping to lower rates later.[9] It should be emphasized that scientific understanding of post-conversion rates of CO_2 emissions is currently embryonic and, accordingly, we make conservative assumptions in the case study application.

Monetary Value of Blue Carbon Benefits

The potential monetary value of blue carbon activity is the product of the GHG emission reductions stream and the price received per unit of reductions. On the biophysical side, the GHG flux is the quantity of CO_2eq whose release will be averted through avoiding conversion of that habitat type. It is a function of carbon burial rates, carbon stocks in biomass and soil, and the rate and extent of GHG emissions from destruction.

In Equation 1, these GHG flows are monetized by multiplying the annual GHG fluxes by a stream of expected carbon prices ($/$tCO_2eq$) over a time horizon of length *n*. The resulting cash flow streams are then discounted using a financial discount rate to arrive at present value estimates for the blue carbon (BC) value of an avoided conversion project.

[1]
$$BC\ value = \sum_{t=0}^{n} \frac{GHG\ flux_t * Price(tCO2eq)_t}{(1+d)^t}$$

This provides a measure of the stream of monetary flows for keeping blue carbon intact and for being paid at the market rate for the emissions avoided over time.

The Cost 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

7 Only includes top meter of soil for consistent comparison. H. Keith, B.G. Mackey, et al., "Re-evaluation of Forest Biomass Carbon Stocks and Lessons from the World's Most Carbon-Dense Forests," *Proceedings of the National Academy of Sciences of the United States of America* Vol. 106, No. 28 (2009): 11635–11640; Wetlands International, Peatland Distribution in Sumatra and Kalimantan (2008).

8 Y. Malhi, L. Aragao, et al., "Comprehensive Assessment of Carbon Productivity, Allocation and Storage in Three Amazonian Forests," *Global Change Biology* Vol. 15, No. 5 (2009): 1255–1274.

9 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 tCO_2 is exposed to conversion and it is assumed to have a half-life of 5 years, then at year five 50 tons will remain, at year ten 25 tons will remain, at year fifteen 12.5 tons will remain (and so on).

of these costs has been limited to date.[10] Usually the largest cost factor, opportunity costs embody the forgone returns of the most profitable alternative use for the land that the coastal habitat occupies. This represents what the owner of the resource gives up by not converting the habitat.[11] In some cases, opportunity costs can be negligible, such as when government-owned coastal land lies far from human populations and has no immediate plans for alternative use, furnishing potentially low-cost protection possibilities.

Can Blue Carbon Make Protection Pay?

Put simply, a blue carbon investment can pay off if the monetary value of carbon payments exceeds the costs of protection:

[2] *BC benefit value > Costs of Protection*

Suppose the BC benefit value is less than the costs of protection. This does not necessarily mean that protection should not proceed on economic grounds. *Recall that blue carbon is proposed as a possible means to strengthen the economic incentive case for protection, thereby serving as a surrogate for the wide range of ecosystem services for which there are no economic incentive payments.* Yet these unpriced services do create economic value and should be considered where possible, especially if the resource is publicly owned and to be managed with broader environmental, social and economic objectives in mind. Then those additional values might be considered additive to the BC value to elucidate the size of an economic incentive required to conserve the habitat.

A further consideration is that participation in a market for blue carbon reductions will involve some cost 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, accounting adjustments put in place that might discount the value of credits, and enforcing contracts and monitoring transactions. These costs are unknown at this time, tend toward being "up front" in nature and should be carefully assessed before parties proceed with protection efforts.

Case Study: Conversion of Mangroves to Shrimp Farms

A shrimp farm in Malaysia.

Historically, the largest threats to mangroves have been over-exploitation for fuelwood or timber and conversion to rice production. In the mid 1970s, a new driver, intensive shrimp farms, began to displace mangrove forests. By the mid 1980s, large-scale mangrove conversions to intensive shrimp farming operations were common in Central and South America. This trend was pushed economically by rising demand in many developed markets, which drove up the price of shrimp and encouraged growth in supply. The high-intensity nature of these farms made them prone to disease epidemics and largely unsustainable in the long term.[12] World shrimp aquaculture production has rocketed, from about 500,000 metric tons in 1988 to over 2.8 million in 2008. Asia grew over 80% of that production in 2008, with China, Thailand, and

10 The exception being A. McCrea-Strub, D. Zeller, et al., "Understanding the Cost of Establishing Marine Protected Areas," *Marine Policy* Vol. 35, No. 1 (2011): 1–9.

11 For some conversion activities, profits may be high in the early years after converting a natural habitat and then taper off or disappear completely due to loss of productivity or unsustainable practices. Future profitless years need to be taken into account in the present value calculation of the opportunity costs.

12 As documented in Food and Agriculture Organization (FAO), *The World's Mangroves 1980–2005* (Rome, Food and Agriculture Organization of the United Nations: 2007), and C. Thornton, M. Shanahan, et al., "From Wetlands to Wastelands: Impacts of Shrimp Farming," *Wetland Science and Practice* Vol. 20, No. 1 (2003): 48–53.

Indonesia accounting for 26%, 18%, and 14%, 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.[13]

The nature of intensive shrimp aquaculture requires the complete removal of mangrove biomass, excavation and banking of the first meter of sediment, and complete alteration of the hydrology. This form of land use results in release of previously stored carbon as CO_2, from both biomass and soil, as well as an abrupt stop to the annual carbon sequestration. As shown in Figure 2, mangrove carbon stocks are variable across global region and mangrove type. In this case study, we focus on mangroves in the Southeast Asia-Indo-Pacific region, which contains over 40% of the world's mangrove forests.[14] Upon conversion to a shrimp farm, it is reasonable to assume that the carbon in the living biomass, both above- and belowground, returns rapidly to the atmosphere, while soil carbon takes longer. We assume that soil in the top 30 cm and 30–100 cm layers are excavated and piled together and thus should be treated equally.[15] The decay rate half-lives of biomass and of soil carbon are assessed to be 1 and 7.5 years, respectively (Table 3). This means that almost 94% of biomass carbon will be gone within four years, while it will take 30 years for this percentage of the disturbed soil carbon to return to the atmosphere. We conservatively assume that the considerable soil carbon stocks below 100 cm are not disturbed, remain inundated, and thus do not emit any CO_2. By modeling the carbon release over a 30-year period, overall carbon stocks are reduced from 3,098 and 3,743 tCO_2eq/ha to 951 and 2,186 tCO_2eq/ha for oceanic and estuarine mangroves, respectively. As a result, annual carbon losses average 72 $tCO_2eq/ha/yr$ (2.3% rate) and 52 $tCO_2eq/ha/yr$ (1.4% rate) for the two mangrove types. To arrive at the total creditable carbon, these avoided carbon emissions are added to the annual carbon sequestration rate for Southeast Asian mangroves, 5.02 $tCO_2eq/ha/yr$, for each year of the study period.

Table 3. Mean carbon pool values in oceanic and estuarine mangroves in the Southeast Asia-Indo-Pacific region.

	Oceanic mangroves (tCO_2eq/ha)	Estuarine mangroves (tCO_2eq/ha)	Decay rate half-life
Total carbon stored, prior to conversion	**3,098**	**3,743**	
Aboveground biomass	352	352	1
Belowground biomass	211	211	1
Total biomass	**563**	**563**	**1**
Soil C: top 30 cm	507	318	7.5
Soil C: 30–100 cm	1,183	742	7.5
Total soil C disturbed	**1,690**	**1,060**	**7.5**
Soil C: > 100 cm (undisturbed)	845	2,120	
C remaining after 30 yrs	951	2,186	

Source: D.C. Donato and J.B. Kauffman, pers. comm.

To analyze the economics, we take the streams of avoided carbon emissions and monetize them with a carbon price range of \$5–\$30 tCO_2eq and discount them back to the present with a 10% real discount rate (Figure 3). These BC values then are compared to the costs of protection to estimate the carbon price payments for blue carbon that could incentivize mangrove conservation. Among the costs of protection, we use \$232 per hectare as the upfront establishment costs to create a protected area.[16] The annual management costs involved in running a protected area are assumed to be \$100 per hectare per year, with the 30-year present value equaling \$943/ha.[17] Those are added to the opportunity costs of converting the mangrove forest into a shrimp farm, which are equivalent to the economic returns of that land use. We use \$11,735/ha for shrimp farming in Thailand.[18] This estimate reflects high profitability immediately after establishment of an intensive shrimp operation, which is ultimately unsustainable and has a productive life of only five years. As

13 I. Valiela, J.L. Bowen, et al., "Mangrove forests: One of the World's Threatened Major Tropical Environments," *BioScience* Vol. 51, No. 10 (2001): 807–815.

14 According to FAO (2007), 41% of global mangrove extent falls within seven countries in this region (Australia, Indonesia, Malaysia, Myanmar, Papua New Guinea, Philippines, and Thailand).

15 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.

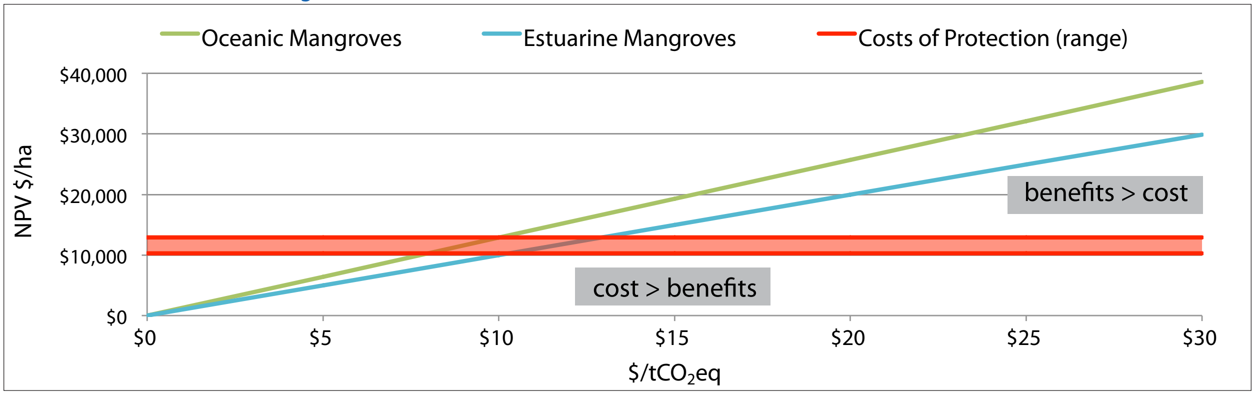
16 McCrea-Strub et al. (see note 10) find a range of establishment costs of \$20 to \$788 (2009 USD) per hectare for small marine protected areas in the tropics. The mean value was \$232/ha, which we use in our calculations.

17 Balmford et al. find management costs for a terrestrial protected area near populated areas in a developing country to be about \$13/ha (2009 USD). A. Balmford, K.J. Gaston, et al., "Global Variation in Terrestrial Conservation Costs, Conservation Benefits, and Unmet Conservation Needs," *Proceedings of the National Academy of Sciences of the United States of America* Vol. 100, No. 3 (2003): 1046–1050. Marsden and Sumaila find management costs for small marine protected areas (< 3,000 ha) in Central and South America to range between \$4 and \$35 per hectare. D. Marsden and U.R. Sumaila, "Investments in Marine Managed Areas: A Preliminary Analysis," Science2Action (2010). We use \$100/ha for protected area management costs to be conservative.

18 S. Sathirathai and E. Barbier, "Valuing Mangrove Conservation in Southern Thailand," *Contemporary Economic Policy* Vol. 19, No. 2 (2001): 109–122. Values adjusted to 2009 USD.

a point of comparison and a different opportunity cost estimate, a World Bank report[19] values a hectare of permanent cultivated land in Thailand at $9,068 when using a 10% discount rate (the same rate employed in the Thailand shrimp farm study). Using both opportunity cost estimates, we create a range of the costs of protection that runs from $10,243 to $12,910 per hectare.

Figure 3. Net present value analysis of blue carbon value and costs of protection of oceanic and estuarine mangroves in the Southeast Asia-Indo-Pacific region.



As can be seen in Figure 3, the break-even carbon price range is where the upper and lower bounds of the costs of protection (red bands) cross the BC value streams (green and blue lines). Thus, at the high end of the range, the BC values offset the costs of conservation at $10.04 for oceanic mangroves and $12.95 for estuarine mangroves, while, at the low end, conservation benefits equal the costs at $7.97 for oceanic mangroves and $10.27 for estuarine mangroves. Avoiding conversion should be financially attractive to a coastal landholder at carbon prices above the break-even points. In this example, oceanic mangroves would more likely be targeted for protection efforts that avoided habitat conversion than estuarine ones since the price point would be lower. However, if soil carbon stocks deeper than one meter were expected to be disturbed by land use conversion, then conserving either mangrove types would become more cost-effective as greater amounts of carbon would be avoided for the same costs of protection. Moreover, estuarine mangrove protection would begin to close the gap with oceanic ones because estuarine mangroves hold considerably greater amounts of carbon in soil layers below one meter than oceanic. Also, note that if the value of the forgone economic opportunity were higher than shrimp farming (e.g., real estate resort development), the required break-even price would be higher as well. The opposite would be true if the forgone opportunity cost were lower.

Policy Implications

Because they store large amounts of carbon and are severely threatened by the economic allure of conversion, coastal ecosystems could be an ideal target for carbon financing. Although data on carbon sequestration rates, on-site storage, emission profiles and the cost of protection are somewhat elusive at this time, preliminary analysis of the type presented here suggests that blue carbon protection may be economically viable in some important cases (mangrove protection in South-Southeast Asia) at the level of carbon prices we now see in world markets (approximately $20 per ton in the EU ETS at this time).

Any verdict on the economic viability of blue carbon is far from complete, however. To begin with, there are currently no markets for credits generated by blue carbon activity. A logical venue for considering blue carbon payments would be through the United Nations Framework Convention on Climate Change (UNFCCC) process. At this time, the only blue carbon activity that could be potentially be covered under the UNFCCC would be mangrove protection, possibly falling under the auspices of Reduced Emissions from Deforestation and Degradation (REDD+) advanced in the Copenhagen Accord, if that mechanism is adopted in full by the UNFCCC at its December 2010 meeting in Cancún, Mexico. However, even if mangrove 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 above ground carbon stocks. If soil carbon is left out, this could seriously hinder the attractiveness of mangrove protection.

19 World Bank, 2011, *The Changing Wealth of Nations: Measuring Sustainable Development in the New Millennium* (Washington: The World Bank).

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 seems in order. Meanwhile research in the natural sciences, economics, and policy should continue to explore the viability of blue carbon as an effective greenhouse gas mitigation tool and, possibly, a means to finance protection of these ecologically diverse and valuable ecosystems.

Appendix: Additional References

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