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Blue CARBON

Global Options for Reducing Emissions from the Degradation and Development of Coastal Ecosystems

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Executive SUMMARY

"Blue carbon" is shorthand for the carbon found in three major coastal and marine ecosystems: mangroves, seagrasses, and salt marshes. Mangroves occur in tropical intertidal areas and are generally considered forests. Seagrasses are ocean "meadows," consisting of different seagrass species. Salt marshes are found in intertidal areas; they are dominated by salt-tolerant shrubs, grasses, and other plants.

Although information on the amount of carbon stored by coastal ecosystems is limited, the available findings suggest that they may be a globally important surface reserve of carbon. But coastal ecosystems are also among the most threatened and rapidly disappearing natural environments worldwide (Valiela et al. 2001). As a result of continual conversion to other uses—-including those agricultural, aquacultural, residential, and industrial—much of the capacity of coastal ecosystems to store and further sequester carbon may soon be lost (FAO 2007a; Spalding et al. 2010).

Drawing from experiences with programs to reduce emissions from deforestation and degradation (REDD) in tropical regions, several international organizations and NGOs have proposed examining similar approaches to protect blue carbon ecosystems. Before developing such programs, and to take full advantage of the possible opportunities, decisionmakers need a better understanding of how blue carbon works and what role it can play in carbon markets and conservation programs. Although our overall knowledge of blue carbon systems is improving (Spalding et al. 2010; Giri et al. 2010; Murray et al. 2011; Donato et al. 2011; Pendleton et al. 2012), economic analyses that rigorously investigate the potential of blue carbon emissions offsets, similar to that for REDD programs, are lacking.

This report, together with a companion journal article (Siikamäki, Sanchirico, and Jardine 2012), begins to fill this information gap. In these two studies, we focus on evaluating whether blue carbon conservation actions would be warranted on an economic basis—that is, whether the benefits from investments in blue carbon conservation outweigh their costs. The economic feasibility of blue carbon critically depends on a broad range of other factors, some of which are economic—others, such as the biophysical availability of blue carbon, are noneconomic. Our assessment therefore draws extensively from both natural science and economics.

More specifically, this assessment involves

- identifying the locations of blue carbon ecosystems;
- estimating the volume of carbon currently stored and sequestered by them;
- projecting the risk of land conversions in blue carbon areas;
- examining potential carbon emissions due to land conversions; and
- estimating the opportunity cost of avoiding future emissions by reducing land conversions.

Because the geographic variations in biophysical and economic conditions are critical in this context, we conducted the first-ever fine spatial resolution assessment (9-by-9 km) of emissions from coastal ecosystems and the cost of avoiding them. Simultaneously, we maintained an overall global geographic scope to help comprehensively assess the potential of coastal conservation to mitigate carbon emissions.

Political considerations, such as ineffective governance, may cause concerns in the context of international systems for carbon offsets. We examined the potential role of host country governance in the supply of emissions offset credits from blue carbon ecosystems. We also examined the potential of blue carbon offset programs to generate co-benefits to biodiversity conservation.

Our main findings are summarized as follows:

Geographic Distribution of Blue Carbon Ecosystems

- The three primary blue carbon ecosystems—mangroves, seagrasses, and salt marshes—are spread across the globe, and at least one of the three can be found in almost every country that has a coastline.
- Mangroves are found in tropical areas and concentrated on both sides of the equator. Seagrass areas are more broadly distributed and include tropical, temperate, and high latitudes. Salt marshes are found everywhere outside the tropics; in the tropics, areas similar to salt marshes become occupied by mangroves.
- Global coverage of blue carbon ecosystems includes 139,170 km^2 of mangroves and 319,000 km^2 of seagrasses worldwide. Comprehensive data on the global distribution of salt marshes are not available, but their estimated total area is 51,000 km^2.
- Blue carbon ecosystems are heavily concentrated in a few countries and regions, with Southeast Asia as the unambiguous geographic center. Almost one half of all global mangroves and one quarter of all global seagrass areas are in this region.
- Indonesia is home to about one fifth of all global mangroves. The other top-five mangrove countries are Brazil, Australia, Mexico, and Nigeria.
- Countries with greatest seagrass areas include Australia, Saudi Arabia, the United States, Indonesia, and Guinea-Bissau.

Carbon Pools in Blue Carbon Ecosystems

- Globally, we estimate that mangroves, salt marshes, and seagrasses altogether store about 11.5 billion t C (about 42 billion t CO_2e).
- Seagrasses have the least amount of carbon per hectare, some 72 tons, but their large global coverage (319,000 km^2) results in a nevertheless substantial estimate of the total global carbon stock, 2.3 billion t C (about 8.4 billion t CO_2e).
- Relative to mangroves, salt marshes have slightly less carbon per hectare (about 393 t C ha^{-1}) and significantly smaller global coverage. As a result, we estimate a blue carbon pool in salt marshes at about 2 billion t C (about 7.3 billion t CO_2e). Current knowledge about salt marshes, including their areal coverage, is incomplete, however, so this estimate is subject to considerable uncertainty.
- Mangroves are the chief blue carbon resource. They contain on average about 470 tons of carbon (t C) per hectare. Given the global coverage of 139,170 km^2, we estimate a total global carbon stock in mangroves at about 6.5 billion tons (almost 24 billion tons of CO_2 equivalent, t CO_2e).
- Geographic variation in the amount of carbon stored by mangroves is substantial. This indicates the need to carefully target blue carbon conservation and also speaks to the advantages of a spatially detailed assessment.
- Most of the blue carbon pool is situated in the soils; they contain more than 80 percent of the overall blue carbon stock.

- Annually, carbon sequestration by blue carbon ecosystems is relatively small, estimated at about 53 million t C.
- Blue carbon ecosystems, especially mangroves, store larger amounts of carbon per unit area than do tropical and boreal forests, but their considerably smaller geographic extent makes the global carbon pool small relative to that of global forests.

Carbon Emissions from Blue Carbon Ecosystems

- Mangrove loss currently causes emissions of about 33 million t C annually (according to our central estimates). Using a CO_2 equivalent, this means that, yearly, about 120 million tons of carbon dioxide are released into the atmosphere.
- Estimating emissions from salt marshes and seagrasses is limited by the lack of information about their habitat loss rates. Using loss rates similar to mangroves, we estimate that emissions from salt marshes and seagrasses are each about one third of mangrove emissions (14.7 million and 10.6 million t C yr^{-1}, respectively).
- Overall, across all blue carbon ecosystems, roughly 215 million tons of carbon dioxide (58 million t C) is returned to the atmosphere from blue carbon habitat losses. Most of the estimated emissions, nearly 60 percent of them, come from the conversion of mangroves to other land uses.
- The three countries with the largest emissions from mangrove losses are Indonesia, with 12.0 million t C per year; Mexico, with 2.7 million t C per year; and Papua New Guinea, with more than 2.2 million t C per year. These three countries alone account for about 51 percent of all emissions.
- Southeast Asia, western Africa, and Mexico are the areas with the highest concentrations of carbon emissions.

Economic Potential of Avoided Blue Carbon Emissions

- We find that preserving mangroves has the potential to provide low-cost opportunities to mitigate carbon dioxide emissions. The majority of potential emissions could be avoided at roughly $4 to $10 per ton of CO_2.
- The economic potential of avoided carbon emissions varies by location, depending on the carbon released into the atmosphere after habitat loss, the risk of future habitat loss and its associated emissions, and the cost of avoiding habitat loss, including the opportunity cost of land and the cost of setting up and operating protected areas.
- The estimated cost per ton is below the recent emissions offset price—roughly between $10 and $20 per ton of CO_2 in the European Union's Emissions Trading Scheme (EU ETS)—suggesting that avoiding emissions from mangroves loss is an economically viable proposition.
- Asia and Oceania are the largest potential suppliers of blue carbon credits—more than 60 percent of total potential global offsets from mangroves come from these regions.
- The price per ton is relatively low for the Americas and the Caribbean, where relatively few blue carbon offsets would be available.
- Similar to the Americas and the Caribbean, the potential contribution from Africa and the Middle East to the global supply of blue carbon is fairly limited (less than 20 percent of global total).

Governance Considerations and Blue Carbon Offsets

- Potential host countries of blue carbon offset projects vary considerably in terms of government stability, reliability, and effectiveness.
- The attractiveness of blue carbon conservation investments across the globe based on the stability of government institutions and their effectiveness could greatly affect the prospective size of the blue carbon offset market.
- Limiting blue carbon supply to areas with relatively effective governments both reduces the supply of blue carbon offsets (less carbon available) and increases the price per ton. For example, excluding the countries ranked in the bottom half for government effectiveness (worst government effectiveness, according to the World Bank) removes about two thirds of the global supply of potential carbon offsets.

Biodiversity Co-benefits of Blue Carbon Conservation

- Conservation focused solely on generating carbon credits will not automatically target the areas most valuable for biodiversity conservation.
- Carbon-focused mangrove conservation will benefit biodiversity but to a lesser extent than if a more biodiversity-focused approach is taken.
- Mangrove conservation projects could be designed to focus more heavily on biodiversity, but the overall costs of such programs would increase. However, we find that the cost of following a co-benefit strategy over the most cost-effective carbon strategy is small compared with the overall program costs.

Blue Carbon in Climate Change Mitigation Policy Frameworks

- By and large, blue carbon has yet to establish a notable presence in international negotiations through the United Nations Framework Convention on Climate Change (UNFCCC). However, UNFCCC parties have acknowledged to some degree the potential benefits of maintaining stored carbon in blue carbon ecosystems.
- Overall, the similarities between blue carbon credits, especially mangroves, and REDD credits suggest that including blue carbon in REDD structures may be a viable path forward. Seagrasses and salt marshes, however, do not meet the current definition of REDD eligible ecosystems.
- Bilateral deforestation agreements are more flexible than the UNFCCC process and may be a more viable way to include blue carbon.
- EU ETS policymakers have serious reservations about land-use-based offsets, which will not be included in the ETS until after 2020.
- In the United States, the state-level program with the most potential for blue carbon is the Global Warming Solutions Act of California, also known as AB 32; it includes offsets, and mangroves might qualify. Currently, however, the credits must be located in the United States. AB 32 does not include salt marshes or seagrasses and does not consider soil carbon, where most blue carbon is sequestered.
- California has an agreement with the Pacific coast state of Chiapas, Mexico, which has mangroves, to develop offset programs, but those potential programs are still years away from providing credits.

- The Northeast's Regional Greenhouse Gas Initiative (RGGI), a regional cap-and-trade system in the United States, allows afforestation offset credits, but the project must be located within one of the member states, none of which have mangroves.

Discussion

- Although carbon is only one of the many benefits from mangroves, their preservation may often be warranted simply on the basis of reducing carbon emissions.
- Coastal conservation would also bring other benefits, such as protecting biodiversity and securing economic returns to fisheries and local communities. Such benefits can be considerable and they add further justification to protecting mangroves.
- Seagrasses and salt marshes are poorly understood in many aspects relevant to the assessment of their potential, including the extent of these ecosystems, their carbon pool, risks of development, and the opportunity cost of conservation.
- Future research needs also include better understanding of emissions profiles from blue carbon ecosystems after land conversion or other disturbances.
- In some locations, deviation between agricultural returns and land prices could be driven by urban and tourism development. These development pressures can result in higher prices for land than we considered in our study. Estimates of the opportunity costs of protected mangroves and other coastal areas, including economic returns from aquaculture (especially in the Asian Pacific region) generally need to be refined.

1.

INTRODUCTION

Coastal ecosystems are among the most threatened and rapidly disappearing natural environments worldwide (Valiela et al. 2001). These areas serve a wide range of ecological functions and provide people with economically valuable products and services (Spalding et al. 2010; Barbier 1994; Barbier et al. 2008, 2011). However, as a result of continual conversion to other uses, many coastal ecosystems have been degraded and their area has been substantially reduced (for example, FAO 2007a-f; Spalding et al. 2010). Even the recent increase in coastal and marine protected areas has not stemmed the current tide of gradual degradation and disappearance of coastal ecosystems worldwide (Lotze et al. 2006; Halpern et al. 2008; Waycott et al. 2009; Spalding et al. 2010). Additionally, new pressures are emerging, such as sea-level rise and climate change more generally.

Coastal ecosystems are well known to provide nursery habitats for fish, crustaceans, birds, and marine mammals (Twilley et al. 1996; Spalding et al. 2010; Mumby et al. 2004). They also provide considerable carbon storage and continuously sequester carbon dioxide through photosynthesis. "Blue carbon" is shorthand for the carbon found in three major coastal and marine ecosystems: mangroves, seagrasses, and salt marshes. Mangroves occur in tropical intertidal areas and are considered forests (FAO 2007a; Spalding et al. 2010). Seagrasses are ocean "meadows," consisting of different seagrass species. Salt marshes are found in intertidal areas, and are dominated by salt-tolerant shrubs, grasses, and other plants.

Information on the amount of carbon stored by coastal ecosystems is limited, but the available findings suggest that these ecosystems are a globally important surface reserve of carbon. For example, Donato et al. (2011) recently estimated that coastal mangroves could store up to 20 Pg (billion tons) of carbon, which is equivalent to roughly 2.5 times current annual greenhouse gas emissions globally. This is a striking observation, especially given that mangroves cover only some 0.7 percent of the tropical forest area worldwide.

The current loss rates of mangroves suggest that much of this carbon storage capacity may be lost, and soon (FAO 2007a; Spalding et al. 2010). Complete loss or less severe disturbance of coastal ecosystems leads to the release of all or some of the carbon they store (Donato et al. 2011). Moreover, coastal development also reduces or completely removes the capacity of the ecosystem to further sequester carbon dioxide from the atmosphere.

Similar concerns about emissions due to land conversion have elevated efforts to halt deforestation in the tropics. Deforestation is the second-largest anthropogenic source of carbon dioxide emissions—around 13 percent of all global carbon dioxide emissions is estimated to originate from deforestation (van der Werf et al. 2009)—and slowing it is integral to international climate policy. Most importantly, programs to reduce emissions from deforestation and degradation (REDD) have been proposed to encourage developing countries with high deforestation rates to reduce their emissions while receiving a stream of payments from developed countries. REDD programs are especially interesting because forest carbon emissions are massive and reducing them is seen as a low-cost way to lower global greenhouse gas (GHG) emissions (Angelsen 2008; Kindermann et al. 2008). Rather than adopting high-cost mitigation actions domestically, developed countries could meet their emissions reduction commitments by financing developing countries to achieve similar but less costly emissions reductions through REDD.

Drawing from experiences with REDD, several international organizations and NGOs (the UN Environment Programme, the International Union for Conservation of Nature, and Conservation International, for example) have proposed examining similar approaches to protect mangroves, seagrasses, and salt marshes—the most important blue carbon ecosystems. But before developing programs to reduce carbon emissions from coastal development, and to take full advantage of the possible opportunities, decisionmakers need a better understanding of how blue carbon works and what role it can play in carbon markets or conservation programs. Although our overall knowledge of blue carbon systems is improving (Spalding et al. 2010; Giri et al. 2010; Murray et al. 2011; Donato et al. 2011), basic economic analyses of the feasibility of a potential carbon credit system, similar to that for REDD programs, are lacking.

The main goal of this report and the companion journal article (Siikamäki et al. 2012) is to undertake primary research on the global-, regional-, and country-level economic potential of blue carbon. We synthesize current knowledge and data sources on blue carbon and then focus on determining the economic potential of blue carbon conservation in different areas around the world. Overall, our assessment requires a broad range of information: identifying the locations of blue carbon ecosystems, estimating the volume of carbon currently stored and sequestered by them, projecting the risk of land conversions, examining the carbon emissions due to land conversions, and estimating the opportunity cost of avoiding future emissions by reducing land conversions. Because the geographic variations in biophysical and economic conditions are critical for the robustness of our assessment, we develop the first-ever spatially explicit estimates (9-by-9-km) of the emissions from coastal ecosystems and the cost of avoiding them.

Although our assessment focuses on mangroves, we also summarize and highlight information on seagrasses and salt marshes to the extent that is feasible, given the available data and scientific understanding. While mangroves are scientifically the best understood blue carbon ecosystem, even the most elementary information, such as the total area and locations of mangroves, was limited until recently (Giri et al. 2010). However, researchers have by now developed a basic scientific understanding of mangroves' carbon storage, sequestration, and emissions. A considerable body of scientific evidence on the role of coastal ecosystems in the global carbon cycle has emerged and synthesized several recent assessments addressing specific ecosystems, ecosystem processes, geographic regions, and countries (for example, Chmura et al. 2003; Duarte et al. 2005; Bouillon et al. 2008; Kristensen et al. 2008; Donato et al. 2011). However, findings from this literature are not always directly applicable for policy assessment of the potential of blue carbon. Moreover, local and comprehensive estimates of carbon storage in coastal ecosystems are generally not available. We therefore focus on synthesizing and translating current scientific findings so that they provide localized estimates which are germane for the purposes of this assessment.

Parallel to this study, researchers at Duke University (Murray et al. 2011; also see Pendleton et al. 2012) have recently assessed the economic considerations of blue carbon. Their work is complementary to ours and also a helpful point of comparison. Our analysis, however, differs from the Duke study in four important ways.

First, unlike the Murray et al. (2011) study, which used global- and country-level data, we develop estimates at a high spatial resolution. Our assessment uses a 5-minute resolution, which means roughly a 9-by-9-km scale (about 5.6 miles squared), throughout. This fine scale allows us to capture within-country variations in the current carbon pool, the amount of avoided emissions, and the costs of conservation, which include the price of purchasing the hectares along with management costs. Our estimates are therefore based on micro-level data rather than country-level estimates. Simple observation of within-country land values reveals that the within-country variation could be equally significant to the economics of blue carbon as the variations among countries, or even more significant.

Second, we undertake a meta-analysis of the studies documenting the amount of soil carbon found in mangrove habitats from around the world to develop the first-ever set of country-level estimates of soil carbon. Because soil carbon is the leading source of stored carbon in mangroves, a more spatially refined estimate is critical for robust estimation of blue carbon.

Third, we filter the economics of blue carbon to account for the effectiveness of the government, as determined by the World Bank. The filter addresses the fundamental investment issue associated with the risks of engaging in long-term contracts with unstable and ineffective governments.

And finally, we combine our blue carbon data with species absence and presence information to understand the potential economic returns from bundling offset contracts with biodiversity conservation.

The rest of this report is organized as follows. In Section 2, we describe and map the geographic distribution of blue carbon ecosystems. Section 3 synthesizes current scientific knowledge about the amount of carbon stored and sequestered by blue carbon ecosystems, and details our spatially disaggregated measures of the soil carbon content of mangrove ecosystems. In Section 3, we also illustrate the global distribution of blue carbon, its projected emissions, and total emissions potential. Section 4 develops estimates of the opportunity cost of protecting mangrove ecosystems around the world. Combining our estimates of opportunity cost with the information on the carbon content, development risks, and opportunity cost of land, we estimate the marginal cost of avoiding blue carbon emissions. We present both regional and global estimates to illustrate the estimated supply of blue carbon emissions offsets. We also examine the potential effects of target countries' governance on the global and regional supplies of carbon offsets. In Section 5, we discuss and illustrate the potential for blue carbon conservation to provide co-benefits in biodiversity conservation. Blue carbon in the context of the international, U.S., and regional climate policy frameworks is discussed in Section 6, which includes a description of the most recent developments. We conclude with summary comments and areas for future research.

2.

GEOGRAPHIC DISTRIBUTION of Blue Carbon Ecosystems

Blue carbon, as noted, is shorthand for the carbon found in three major coastal and marine ecosystems: mangroves, seagrasses, and salt marshes. Mangrove forests are the most recognizable intertidal colonizer in the tropics: this habitat type is dominated by 73 species of trees and shrubs, including some ferns and at least one type of palm, that have evolved to thrive in anaerobic soils with varying levels of salinity (Spalding et al. 2010). Mangrove species have developed roots that can simultaneously exclude saltwater and transport oxygen into harsh intertidal soils. Stilt roots and other aerial root structures that grow from the main stem above the soil allow for the direct uptake of gases from the atmosphere and are present in most mangrove tree species (Spalding et al. 2010). Aerial roots can also act as nets to trap and suspend nutrients, peat, and sediments as they wash out from the land, as well as mute the energy of incoming tides that might otherwise cause inland erosion. Middelburg et al. (1997) discuss how the trapping of sediments can result in accretion and improve the ability of mangroves to naturally adapt to sea-level rise.

Photo © iStock.com

Examples of mangrove forests

The ability of mangroves to gather peat and sediments combines with the overall production and decomposition of the forest to enhance the carbon storage capacity of the habitat (Twilley et al. 1992) (see Section 3 for further discussion). Mangrove forests thrive best when they have access to water with diluted salinity and regular nutrient influx; thus they are found extensively in river deltas, estuaries, and coastal lagoons. Open coastlines with reasonable sedimentation and low wave energy can also present viable conditions for mangrove growth.

Salt marshes are the other intertidal habitat included in blue carbon ecosystems. They are often found in similar environments as mangroves, including estuaries, deltas, and low-lying coasts that experience low wave energy (Adam 2002). Salt marshes have a greater latitudinal extent and are dominated by herbaceous plants, like glassworts and cordgrasses, rather than trees. Along with having high salinity tolerances, these herbaceous plants must be able to withstand regular submersion, since marshes are inundated by high tides. A large proportion of biomass production in salt marshes is located in the subsurface—ratios of belowground to aboveground biomass can reach toward 50:50 (Chmura et al. 2003), which partially explains why salt marshes are responsible for significant carbon storage. They also work to trap nutrients and sediments deposited by tidal patterns.

Aerial view of salt marsh

Seagrasses differ from the other two blue carbon habitat types in that they have no terrestrial component; they are fully submerged in shallow coastal waters off all continents except Antarctica (Green and Short 2003). Seagrass meadows comprise almost 60 flowering species that provide shelter for aquatic animals and breeding grounds for various fishes (Kennedy and Bjork 2009). Additionally, seagrasses function as collection areas for sediments coming off the land and can provide important links between coral reefs and terrestrial systems like mangroves. Seagrass species are often separated into tropical or temperate, but there is a fair amount of overlap between the two categories. Light requirements for photosynthesis are higher for seagrasses than for other marine ecosystems, so they tend to occur in shallower waters (Hemminga and Duarte 2000).

Seagrass adjacent to mangroves

The three primary blue carbon ecosystems—mangroves, salt marshes, and seagrasses—are spread across the globe, and at least one of the three can be found in almost every country that has a coastline. Although the distributions of mangroves, seagrasses, and salt marshes overlap, the differences in their general geographic patterns have important implications for potential conservation efforts.

Figure 2.1 Global Distribution of Mangrove and Seagrass Ecosystems

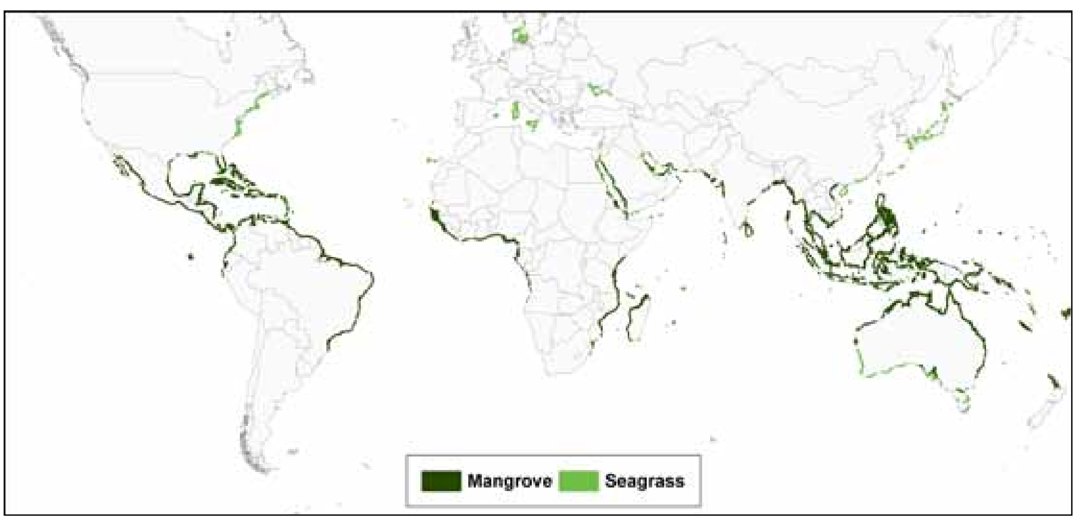


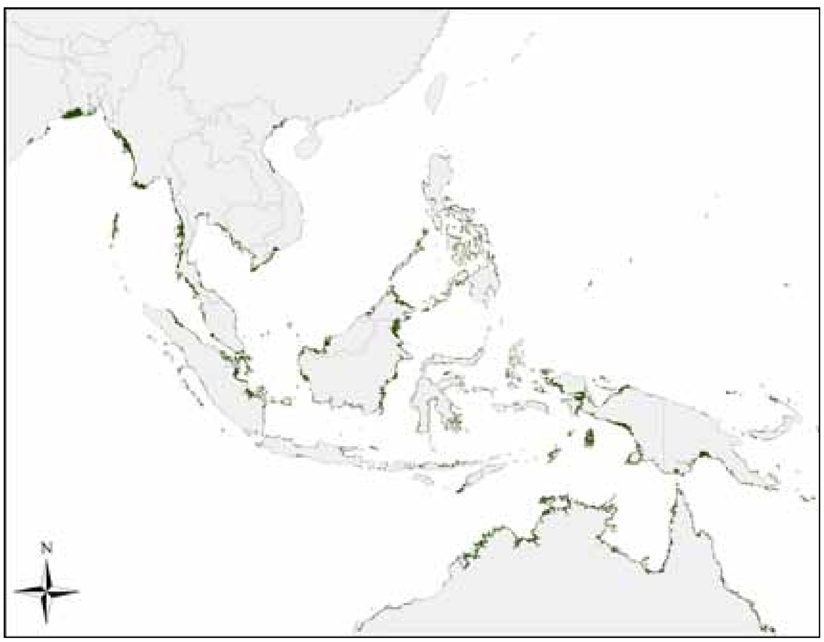
Figure 2.1 shows the global distribution of seagrasses and mangroves; salt marshes are not mapped because comprehensive and reliable distribution data are lacking. Seagrass areas are spread widely across the globe, including in some temperate zones, whereas mangroves are located exclusively on tropical coastlines. Geographic distribution of these ecosystems indicates that the chief opportunities for preserving blue carbon are in the tropics.

2.1. MANGROVES

Estimates of the global coverage of mangroves vary, but the most recent and rigorous spatial data on mangrove forests come from a study by Giri et al. (2010). We use and further process the land cover data from that study throughout this assessment. The data from Giri et al. (2010) study indicate a total area of 139,170 km^2 of mangroves worldwide,[1] which is about 12 percent less than the most recent estimate of the total mangrove area based on country-level information collected by the Food and Agriculture Organization (FAO).

The new spatial data from Giri et al. (2010) represent a notable update of mangrove area as well as an advance in the quality of available spatial information. Previously, the primary source for spatial data on mangrove coverage was the Global Distribution of Mangroves 1997 data set, a joint effort by the UN Environmental Programme–World Conservation and Monitoring Centre (UNEP–WCMC) and the International Society for Mangrove Ecosystems (ISME). For differences between the two data sets, see Box 1.

Figure 2.2 Distribution of Mangroves throughout Southeast Asia



Examining the global distribution of mangroves, we find that Southeast Asia stands out as the unambiguous leader in mangrove area, with almost one half of all global mangroves, about 66,687 km^2. Figure 2.5 shows the thick bands of mangroves spread along the shores of Kalimantan, Sulawesi, eastern Sumatra, and Irian Jaya in Indonesia. Additionally, there are extensive mangrove areas on the coasts of western Thailand, northern Australia, Burma, the Sunderbans in India and Bangladesh, and throughout the Philippines.

Western Africa is another important region, with an estimated 20,998 km^2 of mangroves,

1 This report estimates global mangrove area using our own calculations based on spatial data from Giri et al. (2010). Our estimate of the global mangrove area is about 1 percent greater than the estimate in Giri et al. (137,760 km^2). A small difference between the two estimates is not meaningful but can result from slight differences in the spatial data and data processing.

BOX 1. RECENT UPDATES IN MANGROVE AREA DATA SETS

Comparison of data coverage between UNEP–WCMC dataset and Giri et al. 2010. There are important differences between the mangrove data sets from UNEP–WCMC (UNEP–WCMC and ISME 1997; UNEP–WCMC 2003; UNEP–WCMC and Short 2005) and Giri et al. (2010), based on how they classify mangrove areas. UNEP–WCMC gathered information from a wide variety of maps that were hand digitized into a shapefile. After digitization, the data were manually checked to eliminate duplicate geometry, dangling nodes, and sliver errors. In 2007, the UNEP–Regional Seas Programme published a supplemental data set for the mangroves of Western and Central Africa (UNEP–WCMC 2007); it was based on U.S. Geological Survey Landsat TM 5 and Landsat 7 ETM+ images from 1999 to 2001.

In contrast, Giri et al. (2010) used data from the 2000 Global Land Survey supplemented by Landsat imagery. Giri et al. (2010) drew from approximately 1,000 Landsat images using hybrid supervised and unsupervised digital image classification techniques. These data have been evaluated against other global, regional, and local mangrove datasets. Local experts then provided a qualitative validation.

Hand digitization of paper maps is a time-consuming method to input geographic data. It involves tracing the contours of a map and geocoding the vertices of the resulting polygon and thus is subject to human error. The approach was common practice prior to the availability of remote sensing software and imagery. Satellite imagery is the basis for the modern and automated approach to land classification. Supervised classification is conducted by defining the spectral signature using a small validated area and then classifying all areas that share this unique spectral signature within a given image. Unsupervised classification clusters similar spectral signatures automatically and does not require a sample to classify the image.

It is important to note that none of the mangrove data sets have been fully validated to confirm the accuracy and precision of each classification, which is the final and perhaps most important step when performing any land classification. However, the Giri et al. (2010) data represent the most up-to-date estimates using state-of-the-art techniques, and the results have been cross-referenced with multiple global, regional, and local data sets.

or about 15 percent of the global total. The greatest concentration of mangroves in this region stretches from southern Senegal, continuing along the Gambia River and spreading farther south along the coast down through Guinea-Bissau, Guinea, and Sierra Leone (Figure 2.3). Small patches of mangroves crop up as the shoreline turns east before another major concentration of mangroves emerges around the Niger River delta in Nigeria and Cameroon. Notable mangrove expanses also appear in Equatorial Guinea, Gabon, and the Democratic Republic of Congo.

South America's 20,500 km^2 of mangroves account for 14.8 percent of the global total, putting it on par with Western Africa. Figure 2.4 shows the distribution of the mangroves along the northern coasts of South America and extending to southern Brazil. Although mangroves appear all along Brazil's long coastline, the greatest concentrations of mangroves are found on its northern shores, southeast of the Amazon delta. Venezuela, Colombia, Suriname, and Ecuador also possess notable mangrove resources.

Figure 2.3 Mangrove Distribution along the Western Coast of Africa

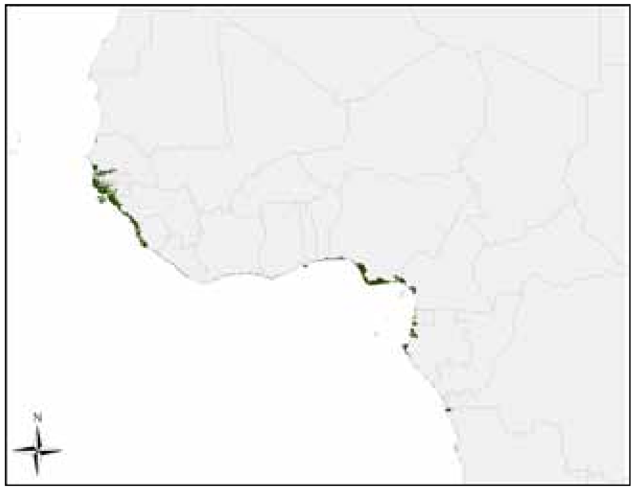


Figure 2.4 Mangrove Distribution along the Coast of South America

North America, including the Caribbean, cumulatively contains the next largest expanse of mangroves, accounting for roughly 17,796 km^2, or 12.8 percent of the global total. Figure 2.5 displays the distribution of mangroves on both the Pacific and Atlantic coasts of Mexico, Belize, and Honduras. Major areas include the Yucatan Peninsula, Cuba's northern barrier islands and western shores, the southwestern stretch of the Bahamas, and the Everglades in southern Florida.

Figure 2.5 Mangrove Distribution for North America, the Caribbean, and Parts of Central America

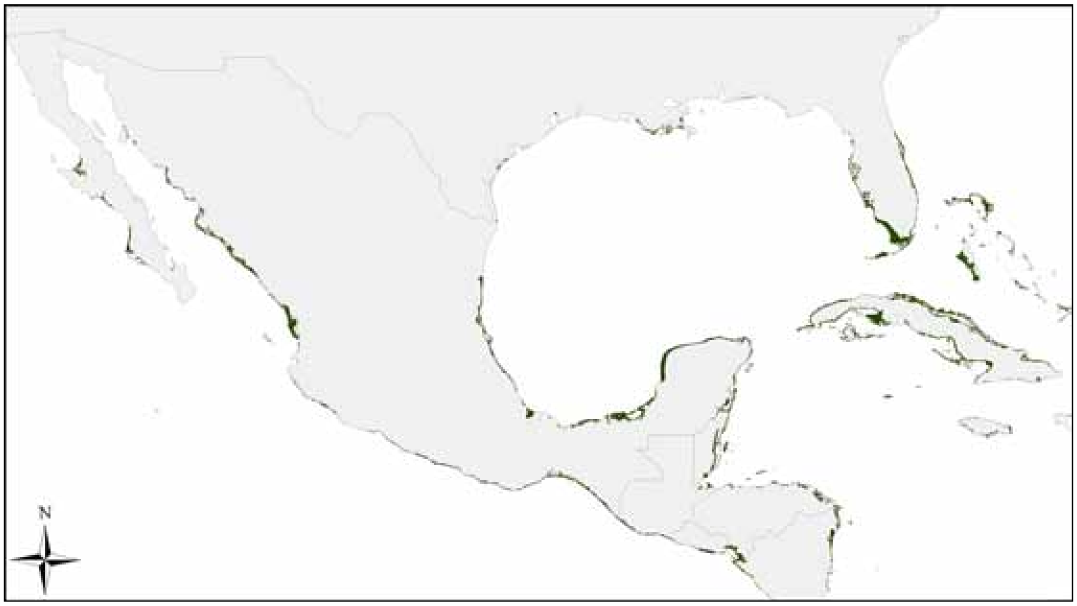
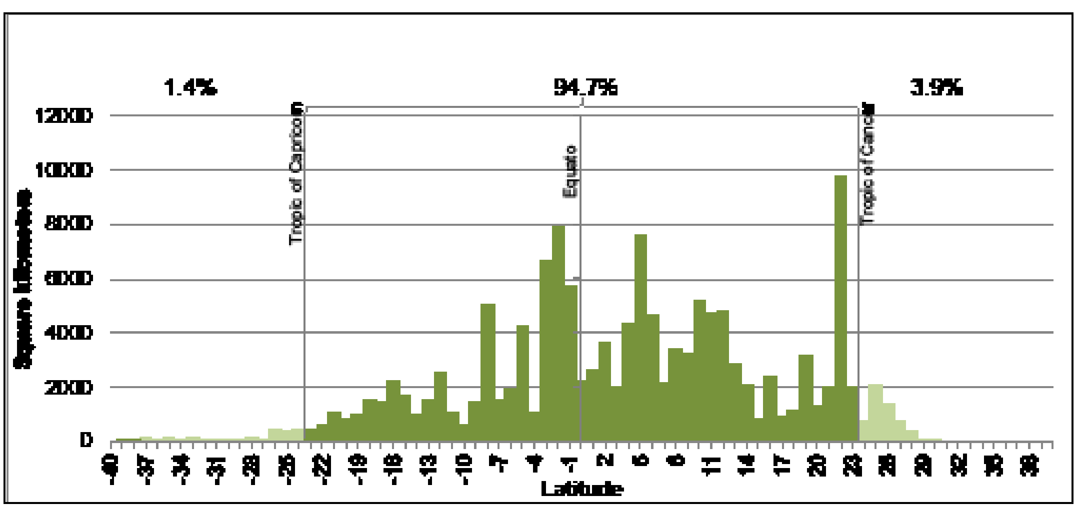


Table 2.1 breaks down the global distribution of mangroves by country, listing the top 20, by area, of the 111 countries with mangroves; these 20 countries account for over 80 percent of the total area. Southeast Asia is the region that has the largest area of mangroves and includes the top country, Indonesia. Indonesia alone accounts for nearly one fifth (27,072 km^2, or 19.5 percent) of the total. The next 4 countries—Brazil, Australia, Mexico, and Nigeria—are on other continents. The area of mangroves in these 4 countries varies between 10,630 km$_2$ (7.6 percent of the world's total) in Brazil and 7,047 km$_2$ (5.1 percent) in Nigeria. The top 6 countries have nearly half the world's mangrove area.

Figure 2.6 Global Area of Mangroves, by Latitude



Note: Each bar represents the area of mangroves (km^2) by 1 degree of latitude.

Table 2.1. Country Rankings for Mangrove Area

Rank	Country	Mangrove area (km²)	Percentage of global total	Cumulative percentage
1.	Indonesia	27,072	19.5	19.5
2.	Brazil	10,630	7.6	27.1
3.	Australia	9,525	6.8	33.9
4.	Mexico	7,302	5.2	39.2
5.	Nigeria	7,047	5.1	44.2
6.	Malaysia	5,616	4.0	48.3
7.	Myanmar	5,082	3.7	51.9
8.	Papua New Guinea	4,850	3.5	55.4
9.	Bangladesh	4,375	3.1	58.6
10.	Cuba	4,286	3.1	61.6
11.	India	3,870	2.8	64.4
12.	Guinea-Bissau	3,427	2.5	66.9
13.	Venezuela	3,360	2.4	69.3
14.	Mozambique	3,194	2.3	71.6
15.	Madagascar	2,731	2.0	73.6
16.	Philippines	2,596	1.9	75.4
17.	Guinea	2,519	1.8	77.2
18.	Thailand	2,496	1.8	79.0
19.	United States	2,360	1.7	80.7
20.	Colombia	2,147	1.5	82.3
	Word total	139,170 100		100

Mangroves are concentrated on both sides of the equator. Figure 2.6 illustrates the distribution of mangroves using a histogram to display the total area of mangroves (km^2) by degree of latitude. Using the underlying data, we calculate that 94.7 percent of all mangroves are found in tropical latitudes between the Tropic of Cancer and the Tropic of Capricorn. The remaining mangrove areas are in adjacent subtropical areas; only 1.4 percent and 3.9 percent of all mangroves are in the southern and nothern temperate regions, respectively.

2.2. SEAGRASSES AND SALT MARSHES

Globally, seagrass ecosystems are estimated to cover roughly 319,000 km^2. Seagrass ecosystems are broadly distributed: whereas mangroves occur mostly in developing countries around the equator, seagrasses have considerable worldwide coverage that spans both developing and developed countries (Table 2.2).

Southeast Asia is the leading region for seagrass area, with 81,348 km^2, or 25.4 percent of the world's total. Other important regions for seagrasses include North America, with 57,159 km^2 (17.9 percent of global seagrass areas), and Western Africa, with 47,993 km^2 (15 percent).

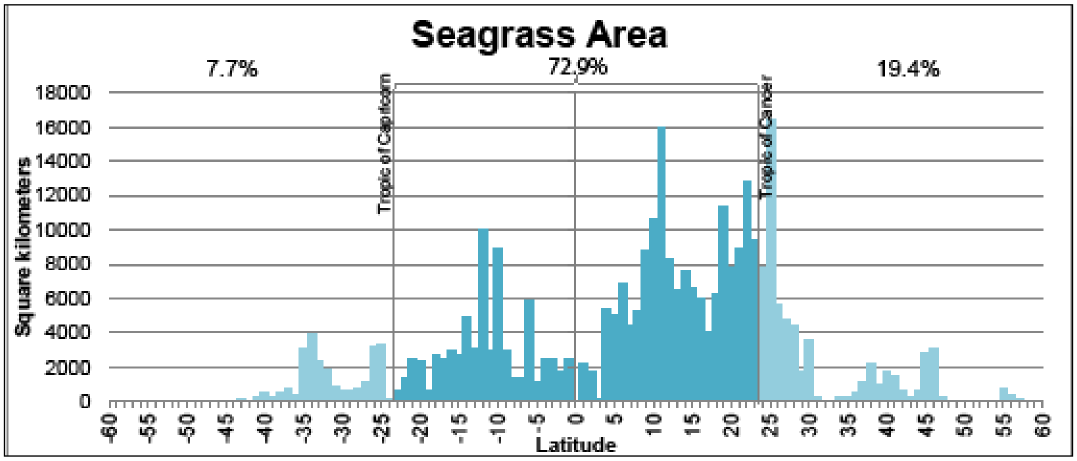
Table 2.2 Country Rankings for Seagrass Area (Top 20 Countries)

Rank	Country	Seagrass area (km^2)	Percentage of total	Cumulative percentage
1.	Australia	41,186	12.9	12.9
2.	Saudi Arabia	21,033	6.6	19.5
3.	United States	19,575	6.1	25.6
4.	Indonesia	17,714	5.5	31.1
5.	Guinea-Bissau	15,418	4.8	36.0
6.	Philippines	15,015	4.7	40.6
7.	Cuba	13,973	4.4	45.0
8.	Guinea	12,825	4.0	49.0
9.	Mexico	9,808	3.1	52.1
10.	Papua New Guinea	9,347	2.9	55.0
11.	Nigeria	8,889	2.8	57.8
12.	China	8,267	2.6	60.4
13.	Ukraine	5,963	1.9	62.3
14.	Madagascar	5,796	1.8	64.1
15.	Italy	5,712	1.8	65.9
16.	Nicaragua	5,566	1.7	67.6
17.	Yemen	5,464	1.7	69.3
18.	Belize	4,924	1.5	70.8
19.	United Arab Emirates	4,587	1.4	72.3
20.	Sierra Leone	4,500	1.4	73.7
	Word total	319,000		100

Using the available spatial data, we estimate that despite their relatively broad geographic distribution, most seagrasses are found between the Tropic of Cancer and the Tropic of Capricorn (Figure 2.7). According to these data, 72.9 percent of all seagrass areas are in the tropics; the southern temperate region accounts for 7.7 percent, and the northern temperate region for 19.4 percent. Whereas mangroves are concentrated around the equator, a noteworthy share of seagrasses (about 27 percent) is in temperate latitudes.

Comprehensive high-resolution spatial data on seagrasses do not yet exist, so their local or regional distributions cannot be mapped. For salt marshes, as noted, constructing even a comprehesive global map is altogether stymied by data limitations, although efforts are currently under way. Regardless, it is known that salt marshes are situated in temperate and high latitudes. In tropical areas, they give way to mangroves (Allen and Pye 1992). Overall, it can be expected that the geographic distribution of salt marshes is at least as broad as that of seagrasses (Allen and Pye 1992; Chmura et al. 2003). Salt marshes are estimated to cover roughly 51,000 km^2 worldwide (Chmura et al. 2003).

Figure 2.7 Global Area of Seagrasses, by Latitude



Note: Each bar represents the area of seagrasses (km²) by 1 degree latitude.

2.3. GEOGRAPHIC OVERLAP

Mangrove and seagrass areas overlap to some degree. For example, Southeast Asia is the world's region richest in both mangroves and seagrasses. Almost one half of all global mangroves (about 66,687 km^2) covers the coasts of this region, which also accounts for about one quarter of the known global seagrass area (81,348 km^2).

Country-level statistics on mangroves and seagrasses further highlight areas rich in blue carbon resources. In Figures 2.8 and 2.9, we illustrate for selected countries the total area of mangroves and seagrasses and the percentage of these areas of the world's totals. Indonesia and Australia possess considerable expanses of both mangroves and seagrasses. Other nations with significant mangrove-seagrass areas include Mexico, Nigeria, the United States, Brazil, Cuba, Guinea-Bissau, Saudi Arabia, and the Philippines. Many countries on this list rank high because of their extensive seagrass beds. In fact, only Indonesia and Brazil have more mangroves by area than seagrasses.

Figure 2.8 Combined Area of Mangroves and Seagrasses for Selected Countries

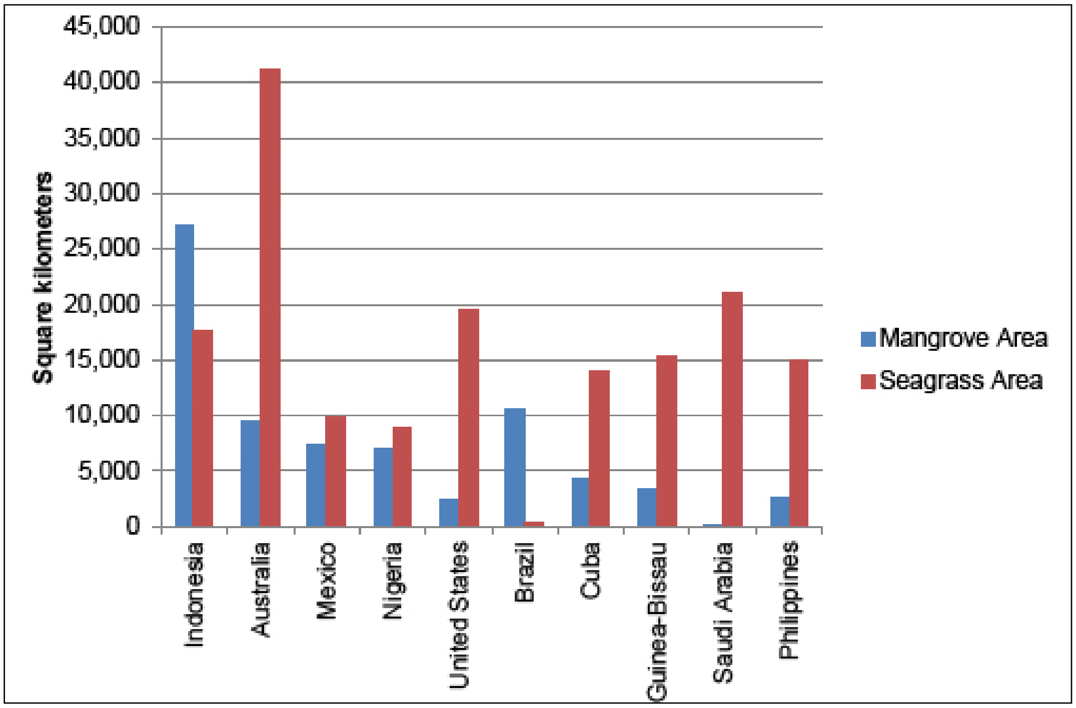
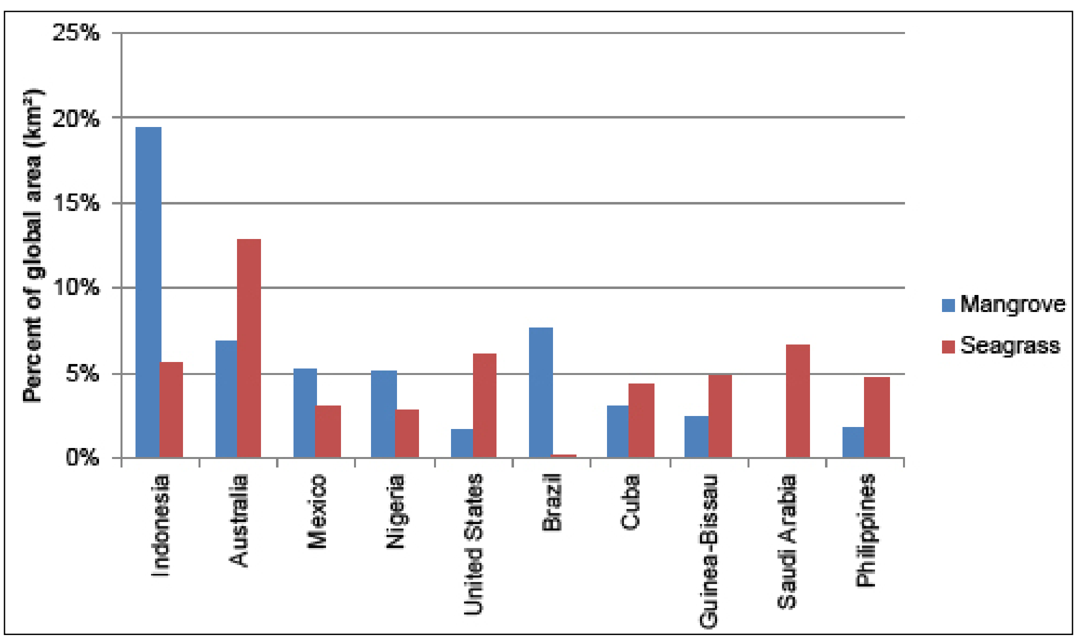


Figure 2.9 Percentage Area for Mangroves and Seagrasses in Selected Countries



3.

Carbon STOCKS and Current EMISSIONS

Evaluating options for blue carbon conservation requires developing a defensible and realistic methodology to estimate the volume of carbon reservoirs in coastal ecosystems. Blue carbon ecosystems differ from many other ecosystems, such as tropical forests, in that the majority of carbon in coastal ecosystems is trapped in their soils (Donato et al. 2011). Because of data limitations for seagrass and salt marsh ecosystems, we focus our carbon calculations on mangrove ecosystems.

We use available scientific information and the primary data underlying major studies to predict the volume of blue carbon in locations around the world, and find considerable evidence that the volume of carbon varies by location. To address this spatial variability, we develop spatially differentiated estimates of carbon stored in mangroves, including aboveground and belowground biomass and soil carbon, using data from a large number of studies conducted around the world.

3.1. ABOVEGROUND AND BELOWGROUND MANGROVE BIOMASS

Estimating the amount of carbon stored in biomass first requires predicting the volume of biomass and then projecting its carbon content. We predict mangrove biomass using a study by Twilley et al. (1992), which draws from a large number of studies to estimate the following regression equation of aboveground biomass (in Mg ha^{-1}) on latitude:[2]

$$(AG)=298.5-7.291\times Latitude$$

We combine predicted aboveground biomass with the mangrove coverage data to estimate the global distribution of aboveground biomass. Our primary scientific information regarding the volume of belowground biomass in mangroves comes from Twilley et al. (1992) and Donato et al. (2011). Using their results, we predict that the volume of belowground living biomass is, on average, 60.8 percent of aboveground biomass. Although some studies suggest that the volume of belowground biomass may be greater than aboveground biomass, these findings sometimes include both living and dead biomass in the belowground calculations (Komiyama et al. 2008). We separately account for the dead belowground biomass as a component of soil carbon and use the results from Twilley et al. (1992) and Donato et al. (2011) to project the volume of

2 The predictive model from Twilley et al. (1992) has been used by Bridgham et al. (2006), Bouillon et al. (2008), and Suratman (2008). Bouillon et al. (2008) assume a latitudinal distribution similar to that documented in Twilley et al. (1992) as the relationship was supported by new data.

living belowground biomass. This avoids confounding and double-counting different forms of belowground carbon.

Because the carbon content of biomass varies by plant and tree species, we also need to estimate the amount of carbon in mangrove biomass specifically. Following Bouillon et al. (2008), we estimate that 41.5 percent of the biomass is carbon.

Combining those findings with data on the locations of mangrove ecosystems and their extent, we estimate the global carbon pool in mangrove biomass at around 2.05 Pg C. Table 3.1 compares our estimate with previous estimates on the global carbon pool in mangrove biomass.

Our overall estimate of carbon in mangrove biomass is similar to that in Murray et al. (2011). The difference between our estimate and that of Twilley et al. (1992) is primarily associated with the difference in the total global mangrove area (139,163 km^2 in our study versus 240,000 km^2 in Twilley et al.); using similar figures for mangrove area would bring the Twilley et al. (1992) estimate very close to ours.[3] The differences between our estimates and those in Bridgham et al. (2006) and Laffoley et al. (2009) arise from a combination of factors, including data on the global mangrove area, biomass per hectare of mangroves, and the carbon content of carbon biomass. Our estimate is based on the most recent and detailed assessment of the global mangrove area, combined with parameters taken from the literature regarding the volume of biomass and its carbon content. Although our estimate of mangrove area is lower than that assumed by the first three studies in Table 3.1, our other parameters generally fall in the middle of the range of published estimates.

Table 3.1 Estimates of Global Carbon Pool in Mangrove Biomass

Mangrove biomass carbon	t C ha^{-1}	t CO_2e ha^{-1}	Global total (billion t C)
Twilley et al. (1992)[1]	167.9	615.7	4.03
Bridgham et al. (2006)[2]	220.1	810.3	4
Laffoley et al. (2009)[3]	79.9	293.0	1.22
Murray et al. (2011)[4]	153.6	563.0	2.12–2.61
This study[5]	147.49	540.8	2.05

[1]*Mangrove area 240,000 km^2.*

[2]*Mangrove area 181,000 km^2; root-to-shoot ratio 0.82; Twilley et al. (1992) biomass distribution; carbon-to-biomass ratio of 0.45.*

[3]*Mangrove area 157,000 km^2; additional assumptions not specified.*

[4]*Mangrove area 138,000-170,000 km^2; additional assumptions not specified.*

[5]*Mangrove area 139,163 km^2; aboveground-to-belowground ratio of 0.608; carbon-to-biomass ratio 0.415.*

3 For example, a simple scaling using the difference in the mangrove area reduces the Twilley et al. (1992) estimate to 2.34 billion tons of carbon in mangrove biomass, which is about 0.3 billion tons or 14 percent greater than our estimate.

3.2. SOIL CARBON IN MANGROVES

Soil carbon is the carbon in the organic matter of soil. We exclude carbon in the living biomass, such as roots, which is classified as belowground carbon. Soil carbon accounts for the majority of carbon stored in mangrove ecosystems (Donato et al. 2011; Bouillon et al. 2008; Kristensen et al. 2008). The volume of soil carbon per hectare in mangroves is also large relative to that in many other ecosystems, such as tropical forests (Donato et al. 2011). Therefore, developing a robust approach to estimating the volume of carbon in the mangrove soils is especially important.

Although research indicates that mangroves contain substantial amounts of soil carbon (Bouillon et al. 2008; Donato et al. 2011) and that the carbon density of mangrove ecosystems exhibits spatial variation (for example, Kristensen et al. 2008), the available assessments of the global carbon pool generally do not consider geographic differences in soil carbon. Instead, the global volume of mangrove soil carbon is obtained as the product of the estimated total global mangrove area and some assumed or otherwise derived representative estimate of the average carbon density of mangrove soils (Murray et al. 2011; Donato et al. 2011).

Accounting for the spatial variation of soil carbon may not be critical to estimating the total global volume of carbon stored by mangroves, but it significantly changes the estimated volume of carbon at different mangrove locations. Moreover, understanding the potential carbon supply at the subglobal level is of utmost importance in understanding the economic potential of different locations for blue carbon conservation. The geographic variation of soil carbon is therefore also critically important to our assessment.

In the absence of comprehensive spatial data on mangrove soil carbon, we systematically summarized available observations on the carbon density of mangrove soils, compiling data from more than 900 study sites in 30 countries, which together represent some 70 percent of the world's total area of mangroves. We first developed country-specific estimates of soil carbon in the 30 countries for which primary observations were available. Where primary data on soil carbon were missing, we developed estimates using observations from nearby areas. Finally, to determine the soil carbon per hectare of mangroves around the globe, we combined estimates of soil carbon density with calculations on the depth of carbon-rich soils. We discuss each of these measures in turn.

3.2.1. Country-Level Estimates of Soil Carbon Density

The steps used to develop geographically varied estimates of the soil carbon density in mangroves are as follows. First, we combined three data sets on soil carbon density: Chmura et al. (2003), Kristensen et al. (2008), and Donato et al. (2011). The Chmura et al. (2003) data set comprises soil carbon density measurements from 31 sites in Africa, North and South America, and the Pacific. The Kristensen et al. (2008) global data set was compiled from an extensive literature review of primary estimates from a large number of field studies. The resulting data set includes observations on the percentage of organic carbon in mangrove soils for 885 sites around the world.[4] Our third source, Donato et al. (2011), provides soil carbon measurements in 25 mangrove forests from the Indo-Pacific region.

Next, combining the above sources creates a data set with 941 observations on soil carbon

4 To combine data on mangrove soil carbon density with the Kristensen et al. (2008) measurements of the %OC in mangrove soils, we follow Donato et al. (2011), using an estimated relationship, $\%OC=3.0443*BD^{-1.313}$, where %OC denotes organic carbon (% weight) and BD is the bulk density (g cm^{-3}).

density from 30 countries. These countries are distributed around the world and together account for 70.4 percent of the world's mangroves. We calculated country-level mean and median soil carbon densities for each of the 30 countries. We use the median estimates for our main results throughout the assessment (see Siikamaki et al. [2012], Supporting Information, for country-level estimates).

After compiling country-level estimates of soil carbon, we summarized them according to the 10 biogeographic regions developed for mangroves by Spalding et al. (2010). Using this assessment, we found significant regional variation in soil carbon (Table 3.2).[5] For example, the mangrove soils of Southeast Asia (Indonesia, Malaysia, Thailand, the Philippines, and Vietnam) have an average density of 0.0418 g C cm^{-3}, whereas mangrove soils in South Asia (Bangladesh, India, and Sri Lanka) are estimated to have only half as much carbon (Table 3.2).

Table 3.2 Estimated mangrove soil carbon density, by biogeographic region

Region	Soil carbon density (mean)	Soil carbon density (median)	Observations	Coefficient of variation	Mangrove area (km^2)	Percentage of total area
East Africa	0.0230	0.0233	197	0.103	7,304	5.25
Middle East	0.0217	0.0217	0	N/A	2,391	1.72
South Asia	0.0205	0.0201	94	0.089	9,035	6.49
South East Asia	0.0418	0.0332	112	0.342	45,600	32.77
East Asia	0.0250	0.0248	100	0.080	0	0.0
Australasia	0.0328	0.0333	67	0.229	9,839	7.07
Pacific Ocean	0.0305	0.0294	146	0.168	6,536	4.70
North and Central America	0.0400	0.0373	43	0.183	20,203	14.52
South America	0.0322	0.0342	182	0.128	19,299	13.87
West and Central Africa	0.0361	0.0357	0	N/A	18,957	13.62

Source: Meta-analysis of data from Chmura et al. 2003, Kristensen et al. 2008, and Donato et al. 2011.

Importantly, there is considerably less variation in the soil carbon density within regions than among regions. The coefficient of variation for our entire data set is 0.355, but the within-region coefficients of variation range from 0.08 to 0.342, with a mean of 0.165. This suggests that the amount of carbon in mangrove soil may be relatively homogeneous within each region. In that case, accounting for the regional or country-level variation enables controlling for much of the underlying spatial variability in soil carbon. The finding also suggests that for countries

5 Our data set contains observations in 8 of the 10 mangrove regions. We imputed soil carbon density for the remaining two regions by taking an average between the two nearest geographic regions.

with missing primary data, soil carbon should be estimated by using regional—rather than global—averages. We therefore use regional estimates of soil carbon density to predict soil carbon pools for the remaining 30 percent of mangrove areas for which no primary data are available. For the Middle East and Western and Central Africa, where the entire region lacked data, we use the estimates from adjacent regions.

One way to compare our results with other global estimates is to develop a representative estimate of carbon density in mangrove soil. To do so, we first average our country-level estimates, which results in a soil carbon density of 0.038 g C cm^{-3}. A simple average, however, puts equal weight on observations of soil carbon from locations with large and small areas of mangroves in predicting the global average. When we combine data on soil carbon with the extent of mangrove area that each observation represents (Giri et al. 2010), we obtain mean and median mangrove area-weighted global estimates of soil carbon density equal to 0.036 g C cm^{-3} and 0.0319 g C cm^{-3}, respectively. These estimates are about 25 percent and 14 percent greater than the most recent and (to our knowledge) the most systematically developed global estimate 0.028 g C cm^{-3} (Donato et al. 2011).

The difference between our estimate and Donato et al.'s (2011) is linked to both data and methods. Donato et al. (2011) predict a globally representative soil carbon density using the median estimate in the primary data. We first added observations from Chmura et al. (2003) to the data used in Donato et al. (2011). We then summarized all individual observations by country, and finally, we used the country-level estimates to calculate a mangrove area–weighted global average. Our goal was to weight different observations according to how many hectares they represented. If primary data are not weighted, locations with large numbers of observations relative to their share of total global mangrove area can potentially bias the average.

3.2.2. Depth of Carbon-Rich Soils

The depth of carbon-rich soils is an important determinant of the total volume of carbon stored in mangrove soils. Unfortunately, both data and scientific consensus regarding the depth of mangrove soils are lacking. For example, when calculating the volume of carbon storage in mangrove soils, Chmura et al. (2003) assume a soil depth of 0.5 meters, although they note that average soil depths are closer to 1 meter. Donato et al.'s (2011) recent findings suggest that soil depths may be greater than 1 meter; measurements at 25 mangrove forests in the Indo-Pacific region show an average depth of carbon-rich soils of about 2 meters.

Table 3.3 Estimates of Global Soil Carbon Pool in Mangrove Ecosystems

Study	Representative soil carbon density	t C ha^{-1}	t CO2e ha^{-1}	Global total
Chmura (2003)[1]	0.055 g C cm^{-3}	275	1008	4.98
Murray et al. (2011)[2]	0.0354 g C cm^{-3}	280	1,027	3.86–4.26
Donato et al. (2011)[3]	0.028 g C cm^{-3}	280	1,027	3.86-4.26
This study[4]	0.032 g C cm^{-3}	320	1,173	4.45

[1] Coverage: 181,000 km^2; data from 26 studies and own data covering sites in 6 countries; soils are 0.5 m deep.

[2] Coverage: 138,000 km^2; assumes that carbon-rich soils are 1 m deep. The carbon density per cm^3 derived from Table 3 in the report is a weighted average of estuarine and oceanic mangroves.

[3] Coverage: 138,000–158,000 km^2; estimate comes from an adjustment of their own sample with a global mangrove soil concentration data set. Assumes soils are vulnerable down to 1 m depth.

[4] Coverage 139,163 km^2; uses a mangrove area–weighted average of (by country) soil carbon density estimates.

Although our assement is localized, we again provide a globally representative estimate for a point of comparison. First, we assume that land-use conversion will disrupt carbon in the top 1 meter of soil (Donato et al. 2011; Murray et al. 2011). Taking our area-weighted global estimate of soil carbon content, we find that mangroves contain, on average, 320 tons of soil carbon per hectare (1,173 t CO_2e ha-1). Combining this estimate with the global area of mangroves suggests that they contain altogether about 4.5 billion tons of soil carbon. This estimate is directly proportional to the assumption regarding the soil depth: increasing or reducing the assumed soil depth leads by a certain percentage to a similar increase or decrease in the amount of soil carbon per hectare.

Table 3.3 summarizes mangrove soil carbon pool estimates from global studies, including the area-weighted average calculated in this study. Although the underlying assumptions and data somewhat vary, the estimates of the global soil carbon pool in mangroves are remarkably consistent, varying between roughly 4 billion and 5 billion tons.

3.3. CARBON SEQUESTRATION IN MANGROVES

Mangroves are valuable not only for the carbon stored in soil and biomass, but also because they continually sequester carbon from the atmosphere and then store (bury) some in their soils. Several studies have estimated the amount of continual carbon sequestration and burial by mangroves, but most of these studies use data from earlier research (Table 3.4). Bouillon et al. (2008) conducts a literature review of the original studies estimating carbon sequestration and burial by mangroves, finding three primary studies that estimate mangrove carbon burial rates.[6] Bouillon et al. (2008) find that despite differences in methods, the three studies included in the assessment all yield similar results, or 1.15 t C ha^{-1} yr^{-1}.[7] We therefore use the Bouillon et al. (2008) burial estimate of 1.15 t C ha^{-1} yr^{-1}.

Table 3.4 compares our estimate of annual global carbon sequestration, a function of the sequestration rate and mangrove area, with the previous estimates. Our estimate falls in the lower range found in the literature, though the range of findings is fairly narrow. Moreover, the volume of carbon sequestered by mangroves annually is a small fraction of the carbon pool contained in mangrove biomass and soils. Therefore, estimates of annual carbon sequestration are not critical to the estimated magnitude of overall carbon in mangrove ecosystems.

6 One of the estimates considered in Bouillon et al. (2008) was from Duarte et al.'s (2005) analysis of the Chmura et al. (2003) data set. Duarte et al. (2005) use the geometric mean from this data set, noting that the data were skewed. Therefore, by way of the Duarte et al. (2005) study, the Bouillon et al. (2008) assessment also includes the Chmura et al. (2003) data.

7 None of the three studies provide standard errors for their estimates.

3.4. CARBON EMISSIONS FROM MANGROVE LOSS

3.4.1. Risk of Land Conversion

To model the risk of mangrove conversion, we use data on mangrove forest areas over time from the UN Food and Agriculture Organization (2007a). These data are available at the country level, and we use the change in the mangrove area by country between 1990 and 2005 to estimate an average annual mangrove deforestation rate by country. Although most countries experienced a loss of mangroves, the area in a few countries, such as Bangladesh, that actually increased during the study period.[8]

Countries subject to some mangrove loss account for about 92.4 percent of the global mangrove area. On average, the annual mangrove loss between 1990 and 2005 was about 0.7 percent. Where mangroves were lost, we use the FAO average annual deforestation rates along with country-level mangrove area to estimate the annual net loss of mangroves by country. Countries that did not experience a net loss of mangroves were not considered emissions sources in this assessment.

Table 3.4 Global Estimates of Annual Carbon Sequestration by Mangroves

Study	t C ha^{-1} yr^{-1}	t CO_2e ha^{-1} yr^{-1}	Global (billion t C yr^{-1})
Twilley et al. (1992)[1]	1	3.67	0.024
Jennerjahn and Ittekkot (2002)[2]	1.15	4.22	0.023
Chmura et al. (2003)[3]	2.1	7.7	0.038
Duarte et al. (2005)[4]	1.39	5.1	0.028
Bouillon et al. (2008)[5]	1.15	4.22	0.018
Nellemann et al. (2009)[6]	1.39	5.097	0.017-0.024
Murray et al. (2011)[7]	1.72±1.31	6.32±4.8	0.006-0.052
This study[8]	1.15	4.22	0.016

[1] Mangrove area 240,000 km^2; based on literature review.

[2] Mangrove area 200,000 km^2; litterfall rate 460 g C m^{-2}; calculates 25% of litterfall accumulates in mangrove sediments and applies this to get global estimate.

[3] Average from all sites in sample (difference between mangrove and salt marsh sequestration rates not found statistically significant).

[4] Mangrove area 200,000 km^2; takes geometric mean from Chmura et al. (2003) data.

[5] Coverage 160,000 km^2; based on estimates from 3 studies (Twilley et al. 1992; Jennerjahn and Ittekkot 2002; Duarte et al. 2005).

[6] Mangrove area 170,000 km^2; based on estimates from 3 studies (Duarte and Cebrian 1996; Duarte et al. 2005;

8 Additionally, the FAO (2007a) lacks data on mangrove area and deforestation rates for 24 countries, mostly small island nations representing 1.3 percent of total carbon in mangroves. Therefore, we are unable to include these countries in the carbon emissions calculations.

Bouillon 2008).

[7] Mangrove area 138,000 km²; based on literature review.

[8] Mangrove area 139,163 km²; based on Bouillon et al. (2008) literature review.

Some mangrove areas are already protected. Using spatial data from the World Database on Protected Areas (WDPA), which is a joint initiative of the International Union for Conservation of Nature (IUCN) and the World Conservation Monitoring Centre of the UN Environmental Programme (UNEP–WCMC 2010), we estimate that about 4 percent of the world's total mangrove area is under some type of protection (IUCN conservation categories I–VI). Using these data, we net out the mangrove hectares that are already protected.

3.4.2. Carbon Emissions after Land Conversion

There is a paucity of information regarding the effects of land conversion on carbon stored in mangrove biomass and mangrove soils. When estimating emissions from mangrove deforestation, Murray et al. (2011) assume that all carbon in the first meter of mangrove soils is exposed to oxygen and gradually released into atmosphere. Assuming a half-life of 7.5 years, about 90 percent of soil carbon in the top 1 meter is released into the atmosphere after 25 years. Donato et al. (2011) posit that 50 percent of soil carbon in the top 30 cm of mangrove soils is released, and that in the soils beneath that, 17.5 percent of soil carbon is emitted. The authors then project a range of potential emissions by assuming that the overall disturbance reaches either 1 or 2 meters in depth. Both Donato et al. (2011) and Murray et al. (2011) assume that 75 percent of the carbon in mangrove biomass is released upon conversion.

We drew from the above two studies to project a range of potential emissions. First, consistent with most of the literature on mangrove soil carbon, we considered that mangrove conversion affects soil carbon down to 1 meter. Second, and again consistent with previous studies, we predicted that 75 percent of carbon in the aboveground and belowground biomass is emitted. Third, we constructed a range of potential carbon emissions from the mangrove soils: with the Donato et al. (2011) approach, a total of 27.25 percent of the soil carbon in the top 1 meter is released; with the Murray et al. (2011) set of assumptions, 90 percent of soil carbon in the top 1 meter is released. We used those estimates as the low and the high, and the average of the two forms our middle estimate of the carbon that could be released from mangroves as a result of conversion.

3.4.3. Avoided Carbon Emissions Due to Additional Protection

We projected for each area of mangroves the total avoided emissions that could be credited as an emissions offset as a result of protection. We first estimated baseline carbon emissions from each area (grid cell) of mangroves without additional protection. Thereafter, the volume of potential carbon emissions offsets was estimated as the reduction of baseline carbon emissions due to additional protection. When estimating baseline emissions, we considered that each area of mangroves is potentially subject to deforestation and that the risk of deforestation is known ex ante only as a probability. But when aggregated up to the country level, the rate of deforestation equals what was observed by country between 1990 and 2005.

In projecting emissions from a certain parcel of mangroves, we considered a 25-year time horizon, and modeled the additionality condition by assuming that offset credits are granted only for the at-risk portion of the mangrove in each year. An example illustrates the approach. Suppose that a country has a deforestation rate of 1 percent, and we are considering protecting

100 hectares of mangroves. In year 1, the total avoided emissions (TAE) of blue carbon are equal to emissions from 1 hectare of deforestation. In year 2, TAE is equal to 1 percent of the 99 remaining hectares. If we continue this accounting from one year to the next over the horizon, we find that the TAE is characterized by a finite geometric series. The formula for TAE is as follows:

$$TAE_{ij} = \left\lceil 1-(1+\delta_i)^T \right\rceil * \left[M_{i0} * C_{i0})\right]$$

where δ_i is the FAO country mangrove deforestation rate, T is the horizon of the contract (25 years), M_o is the number of hectares of mangroves protected, and C_o is the carbon content of these hectares.

3.4.4. Methane and Nitrous Oxide Emissions from Mangroves

Mangroves sequester carbon dioxide from the atmosphere, but they also constitute a natural source of methane ($CH_{4)}$ and nitrous oxide (N_2O), both of which are potent and important GHGs (Purvaja and Ramesh 2001; Purvaja et al. 2004; IPCC 2007). The emissions of methane and nitrous oxide from mangroves are related to periodic anoxic conditions due to tidal flooding. Although estimates of methane emissions vary by study and location, current findings indicate that methane and nitrous dioxide emissions from mangroves are small relative to the volume of carbon stored in them (and potentially released to the atmosphere as a result of land development). For example, Krithika et al. (2008) estimate that mangroves in South India release, on average, about 0.09 tons of CH_4 ha^{-1} yr^{-1}. For Puerto Rico, Sotomayor et al. (1994) estimate emissions of about 0.16 tons CH_4 ha^{-1} yr^{-1}. These estimates suggest that methane emissions from mangroves are equivalent to emissions of roughly between 2.1 and 3.6 tons CO_2 ha^{-1} yr^{-1}. Nitrous oxide emissions similarly vary by study and location. For example, in India, Krithika et al. (2008) estimate N_2O emissions in mangroves are equivalent to between roughly 0.4 and 0.9 tons CO_2 ha^{-1} yr^{-1}.

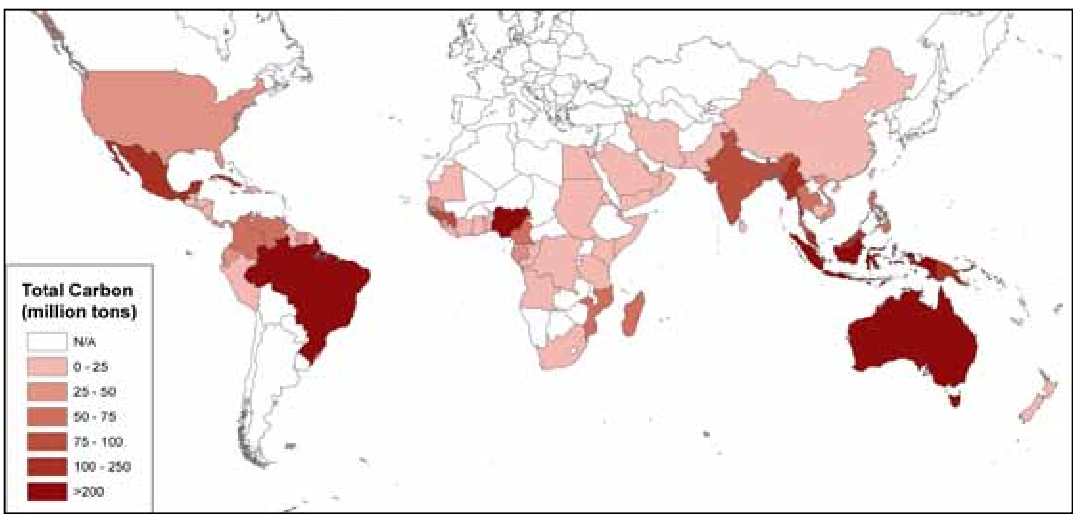
While emissions of methane and nitrous oxides from mangroves and other natural ecosystems are important to consider as part of overall assessments of greenhouse gas emissions, the critical aspect in the context of this assessment is to determine how potential land conversion would alter methane and nitrous oxide emissions. Potential offset credits from habitat protection would be determined as the difference between greenhouse gas emissions with and without habitat protection. Emissions without habitat protection provide a baseline against which emissions under habitat protection would be evaluated.

In our assessment, emissions under alternative land uses such as agriculture, therefore, provide an emissions baseline. Accordingly, if emissions of methane and nitrous oxides under habitat protection are greater than under alternative land uses, then the overall greenhouse gas benefits from habitat protection would be lower than when estimated by focusing only on carbon emissions. On the other hand, if alternative land uses cause greater methane and nitrous oxide emissions than natural mangroves, then the avoided emissions from mangrove protection would be greater than when estimated based on carbon only.

Comprehensive and globally representative estimates of methane and nitrous oxide emissions from mangroves converted into agricultural and other alternative land uses are not available, but the available estimates suggest that emissions under alternative land uses likely are greater than emissions from natural mangroves. In particular, rice farming on wetlands is considered one of the chief agricultural sources of methane (Sass 1999; Sass et al. 1999; IPCC 2000, 2007; Yan et al. 2003). According to the IPCC (2000), this activity annually generates between 20 and 100 Mt of methane emissions globally, equivalent to emissions of roughly 3 to 16 t of CO_2 ha^{-1} yr^{-1} (mean estimate 10 t CO_2 ha^{-1} yr^{-1}).[9] Moreover, agriculture and rice cultivation, especially, are chief alternative land uses driving mangrove deforestation (Giri et al. 2007), thus providing a useful point of reference here.[10]

Overall, the above evidence suggests that mangrove conservation likely would not increase but may decrease emissions of methane and nitrous oxides relative to their baseline emissions under alternative land uses. Therefore, mangrove conservation projects could potentially qualify for greater greenhouse gas offset credits than one would estimate solely based on avoided carbon emissions. However, any potential benefits from avoided methane and nitrous oxide emissions are extremely small relative to avoided carbon emissions. For example, methane emissions from natural mangroves are in the range of a few tons per hectare, while the avoided carbon emissions from mangrove protection are, on average, about 290 tons per hectare (our central estimate). Therefore, the inclusion or exclusion of methane and nitrous oxides from the emissions calculation does not critically affect the results of this assessment. Moreover, only limited information is available to support more precisely quantifying the potential greenhouse gas benefits associated with methane and nitrous oxides. Therefore, we assume similar emissions profiles of methane and nitrous oxides under habitat protection and alternative land uses. Under that assumption, the net effect from conservation of greenhouse gases is fully captured by avoided carbon emissions.

Figure 3.1 Total Carbon Emissions Potential from Mangrove Ecosystems, by Country



9 The estimate of global emissions corresponds to roughly 140,000 km[2] of cultivated rice. The current rice cultivation area is greater. For example, in 2009, it was about 158,000 km[2] globally (FAO 2011).

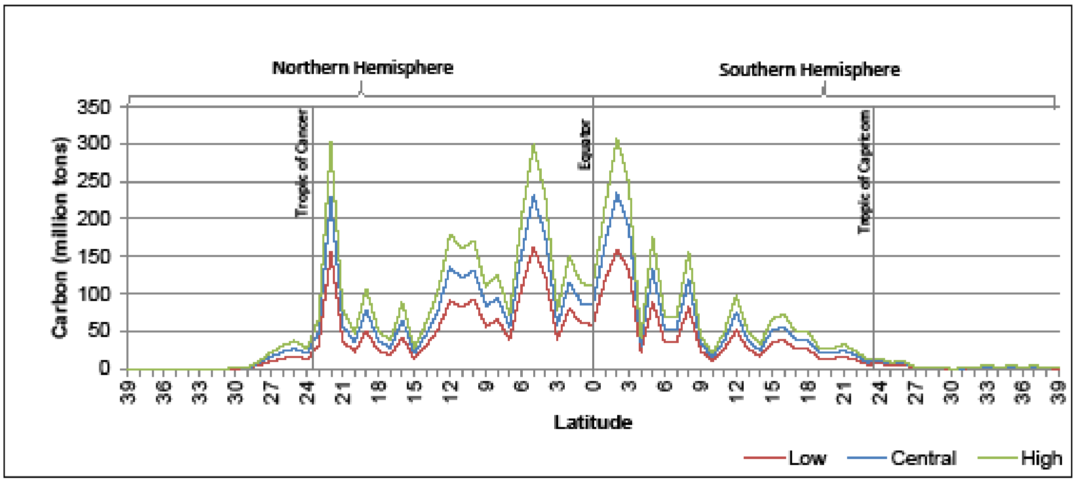
10 Rice cultivation also generates nitrous oxide emissions, but their overall volume and global warming potential is small relative to methane emissions.

3.5 GLOBAL DISTRIBUTION OF MANGROVE CARBON

3.5.1. Total Emissions Potential (Carbon Storage)

In this section, we first project the total carbon emissions potential of the world's mangroves. This is the amount of carbon that would be released to the atmosphere if all mangroves were converted to alternative land uses. While it is, of course, unlikely that all mangroves would suddenly be converted, this scenario demonstrates well the total emissions potential associated with mangroves. The emissions potential is somewhat lower than the total carbon pool in mangroves because only a fraction of the total carbon is anticipated to be released into the atmosphere as a result of land conversion.

Figure 3.2 Distribution of Carbon Storage in Mangrove Ecosystems, by Latitude



According to our middle estimate, mangroves contain about 4.4 billion tons of carbon vulnerable to being emitted, with our low and high estimates at 3.0 billion and 5.7 billion, respectively (Table 3.5). The top 15 countries account for about 74 percent of the world total, and the top 6 countries contain more than half of all potential carbon emissions from mangroves (Table 3.5). According to our middle estimate, Indonesia's mangroves contain some 963 million tons (about 22 percent of the world total) of carbon that would be released into the atmosphere if all mangroves were converted to alternatives land uses. Brazil's total emissions potential is 289 million tons (Mt C), followed by Australia, with 286 million; Nigeria, with 260 million; Mexico, with 1,217 million; and Malaysia, with 203 million.[11]

The country rankings for total emissions potential closely correspond to those for mangrove area, although countries near the equator and with high soil carbon move up in the ranking for total carbon. Almost 97 percent of total emissions potential from mangroves falls between the Tropic of Cancer and the Tropic of Capricorn (Figure 3.2). The only exception to the tight centering of this distribution around the equator is a spike at approximately 22°N latitude, where the Sundarbans in the Bay of Bengal is the site of the largest halophytic (saltwater) mangrove forest in the world.

11 The country rankings reflect our middle estimates of carbon.

Table 3.5 Country Rankings for Total Emissions Potential from Mangrove Ecosystems (Top 20 Countries)

	Estimated total carbon emissions potential from mangroves				
Country	Low (Mt C)	Middle (Mt C)	High (Mt C)	Percentage of world total	Cumulative percentage of world total
1.Indonesia	685	963	1,240	22%	22%
2. Brazil	207	289	370	7%	29%
3. Australia	191	286	381	7%	35%
4. Nigeria	181	260	339	6%	41%
5. Mexico	140	217	293	5%	46%
6. Malaysia	143	203	263	5%	51%
7. Myanmar	109	161	214	4%	54%
8. Papua New Guinea	110	154	198	4%	58%
9. Cuba	85	135	185	3%	61%
10. Guinea-Bissau	80	118	156	3%	64%
11. India	65	93	121	2%	66%
12. Guinea	60	88	117	2%	68%
13. Colombia	55	87	119	2%	70%
14. Bangladesh	61	83	105	2%	72%
15. Philippines	55	81	108	2%	74%
16. Venezuela	58	77	97	2%	75%
17. Mozambique	54	76	98	2%	77%
18. Vietnam	50	75	100	2%	79%
19. Thailand	52	71	90	2%	80%
20. Madagascar	49	68	87	2%	82%
World total	3,025	4,375	5,723	100	100

The country rankings for total emissions potential closely correspond to those for mangrove area, although countries near the equator and with high soil carbon move up in the ranking for total carbon. Almost 97 percent of total emissions potential from mangroves falls between the Tropic of Cancer and the Tropic of Capricorn (Figure 3.2). The only exception to the tight centering of this distribution around the equator is a spike at approximately 22°N latitude, where the Sundarbans in the Bay of Bengal is the site of the largest halophytic (saltwater) mangrove forest in the world.

3.5.2. Estimated Annual Emissions

Next, using recently observed mangrove deforestation rates, we project actual annual carbon emissions from mangroves. If the recent rates of mangrove deforestation continue, we estimate mangrove loss will cause the emission of 33.5 million tons of carbon (Mt C) annually. Of this total projected amount of emissions, the 15 countries with the highest estimated emissions represent 84 percent of the total, and the top 6 countries account for two thirds (Table 3.6).

Table 3.6. Country Ranking for Estimated Annual Emissions from Mangrove Ecosystems (Top 20 countries)

Country	Estimated actual carbon emissions from mangrove loss, annually				
	Low (Mt C)	Middle (Mt C)	High (Mt C)	Percentage of world total	Cumulative percentage of world total
1. Indonesia	8.6	12.0	15.5	36%	36%
2. Mexico	1.7	2.7	3.6	8%	44%
3. Papua New Guinea	1.6	2.2	2.9	7%	51%
4. Malaysia	1.2	1.7	2.2	5%	56%
5. Vietnam	1.0	1.5	2.1	5%	60%
6. Sierra Leone	1.0	1.4	1.9	4%	65%
7. Guinea-Bissau	0.9	1.3	1.8	4%	68%
8. Gabon	0.5	0.8	1.0	2%	71%
9. Senegal	0.5	0.8	1.0	2%	73%
10. Honduras	0.5	0.8	1.0	2%	75%
11. Philippines	0.5	0.7	0.9	2%	77%
12. Colombia	0.4	0.7	0.9	2%	79%
13. Myanmar	0.4	0.6	0.8	2%	81%
14. Venezuela	0.3	0.5	0.6	1%	83%
15. United States	0.3	0.6	0.8	2%	84%
16. Madagascar	0.3	0.4	0.6	1%	85%
17. Panama	0.3	0.4	0.5	1%	87%
18. Nicaragua	0.2	0.3	0.5	1%	88%
19. Brazi	0.2	0.3	0.4	1%	89%
20. Ecuador	0.2	0.3	0.4	1%	89%
World total	23.2	33.5	43.8	100	100

The top 3 countries for estimated annual carbon emissions are Indonesia, with 12.0 million tons; Mexico, with 2.7 million tons; and Papua New Guinea, with 2.2 million tons. These 3 countries alone account for about 51 percent of all actual emissions. The remaining countries of the top 20 account for approximately 38 percent of total emissions per year. Summarizing the findings by world region, we find that Southeast Asia, Western Africa, and Mexico are the regions with the highest concentrations of carbon emissions. Figure 3.3 maps carbon emissions from mangroves by country.

3.6. METRICS FOR SEAGRASSES AND SALT MARSHES

Information on seagrass and salt marsh ecosystems is much more limited than for mangroves. For example, the availability and maturity of spatial data on salt marsh and seagrass areas are not sufficient to allow a systematic, detailed assessment. Nevertheless, some researchers have developed estimates of biomass and carbon storage potential for both habitat types. Table 3.7 summarizes our findings in both carbon and carbon dioxide equivalent and is further discussed below.

Table 3.7 Summary of Carbon Stock and Burial Estimates for Seagrass and Salt Marsh Ecosystems

	Per ha (t C)	Per ha (t CO_2 e)	Globally (Pg C)
Salt marshes (51,000 km^2)			
Biomass[1]	3.315	12.2	0.017
Soil[2]	390	1,430.0	1.989
Total stock	393.3	1,442.2	2.0
Burial[3]	2.1	7.7	0.011
Seagrasses (319,000 km^2)			
Biomass[4]	1.54	5.6	0.049
Soil[5]	70	256.7	2.233
Total stock	71.5	262.3	2.3
Burial[6]	0.54	2.0	0.017

[1] *Mitsch and Gosselink (1993); Cebrian (1999).*

[2] *Chmura et al. (2003), assuming 1 m depth of carbon rich soils.*

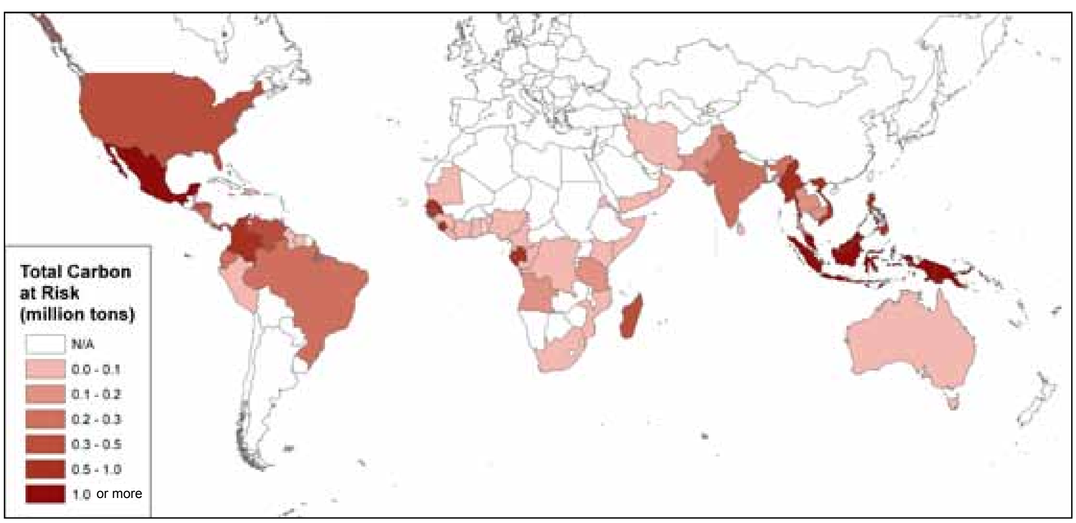
[3] *Chmura et al. (2003).*

[4] *Duarte and Chiscano (1999).*

[5] *Laffoley and Grimsditch(2009).*

[6] *Duarte et al. (2005).*

Figure 3.3 Estimated Annual Carbon Emissions from Mangrove Loss, by Country



3.6.1. Salt Marshes

To calculate biomass carbon in salt marshes, Cebrian (1999) compiled a data set with 25 estimates of aboveground and belowground marsh biomass measured in g C m^{-2}, finding that, on average, biomass in salt marshes contains about 3.45 t C ha^{-1}. Cebrian (1999) notes that the estimate is conservative, because in many cases the original data fail to account for belowground biomass. Using Mitsch and Gosselink's (1993) data on both aboveground and belowground biomass, Bridgham et al. (2006) estimated an average salt marsh biomass of about 3.18 t C ha^{-1}. The Bridgham et al. (2006) results are very similar to Cebrian (1999); we took the average of the two estimates.

Reviewing the literature on soil carbon in tidal wetlands, including salt marshes, Chmura et al. (2003) found 26 studies comprising 154 observations from the western and eastern Atlantic and Pacific coasts, Indian Ocean, Mediterranean Sea, and Gulf of Mexico, with about three quarters of the data addressing salt marshes. Averaging all observations on soil carbon density in salt marshes, Chumra et al. (2003) estimate 0.039 g cm^{-3}. Averaging observations on carbon burial rate gives an average salt marsh carbon burial rate of 210 g C m^{-2} yr^{-1}. On a per hectare basis, these estimates suggest a total volume of soil carbon of 390 t C ha^{-1} (assuming a 1-meter depth of carbon-rich soils) and a carbon burial rate of 2.1 t C ha^{-1} yr^{-1} (Table 3.7).

3.6.2. Seagrasses

For seagrass biomass carbon, we draw from a review by Duarte and Chiscano (1999), who compiled a data set containing 423 aboveground estimates and 250 belowground estimates for biomass, plus 128 aboveground estimates and 60 belowground estimates for production rates. Averages from these data address 30 of 60 known seagrass species.[12] The authors estimate an average biomass of 461 g m^{-2} and assume that 33.5 percent of dry weight is carbon. This implies that, on average, seagrass biomass contains 1.54 t C ha^{-1}.

Duarte et al. (2005) review the literature on the carbon burial rate of seagrasses and estimated an average carbon burial for seagrasses at 83 g C m^{-2} y^{-1}. Kennedy et al. (2010) find that 50 percent of this organic matter comes from seagrass plant tissue. These two findings imply that approximately 41.5 g C m^{-2} yr^{-1} of seagrass biomass is stored in seagrass sediments. On a per hectare basis, this estimate indicates a total volume of carbon burial rate of 0.54 t C ha^{-1} yr^{-1} (Table 3.7).[13]

For soil carbon in seagrass meadows, we use the findings from Laffoley and Grimsditch (2009), who estimate a soil carbon volume of about 70 t C ha^{-1}.

3.6.3. Global Estimates

The estimated total area of salt marshes is about 51,000 km^2, and that for seagrasses is 319,000 km^2. Using the available estimates of carbon in biomass, soil carbon, and carbon burial rate, we estimate that the total carbon stock in salt marshes and seagrasses is about 2.0 Pg C and 2.3 Pg C, respectively.

On a per hectare basis, salt marshes have about 5.5 times the amount of carbon in

12 Data are recorded in dry weight, which is assumed to be 33.5 percent carbon (Duarte 1990).

13 Duarte (2010) finds that seagrass ecosystems also bury carbon from neighboring ecosystems and estimates the net burial to be 120 g C m^{-2} yr^{-2}. We do not include the burial of external carbon in this assessment; however, the findings indicate that seagrass ecosystems may be more valuable carbon sinks than the sequestration rate in Table 3.7 implies.

seagrasses, but since the total area of seagrasses is more than 6 times greater, the estimated total stocks for these two ecosystems are nearly equal. Annually, global salt marshes and seagrasses are estimated to sequester some 11 and 26 million tons of carbon.

3.7. SUMMARY

We have compiled a large number of estimates to describe the volume of carbon stored and sequestered by the three blue carbon ecosystems. We focus on mangroves, in large part because of the data limitations for salt marshes and seagrasses, but also because of the prominence of mangroves in the context of this assessment. Table 3.8 summarizes our main findings.

Mangroves are estimated to be the primary source of blue carbon storage. They contain, on average, about 470 tons of carbon per hectare. Given the global coverage of 139,170 km^2, the estimated total carbon stock in mangroves is about 6.5 Pg C.

Salt marshes are estimated to have slightly less carbon per hectare, about 393 tons, and their global coverage (51,000 km^2) results in an estimated total carbon stock of about 2 Pg C. Because current knowledge of the areal coverage of salt marshes is incomplete, this estimate is subject to considerable uncertainty.

Seagrasses have the least amount of carbon per hectare, approximately 72 tons, but their large global coverage (319,000 km^2) results in a nevertheless substantial estimate of total carbon stock, 2.3 Pg C.

Table 3.8 Summary of Carbon Stock and Burial Estimates for Blue Carbon Ecosystems

	Per ha (t C ha^{-1})	Per ha (t CO_2 ha^{-1})	Global (Pg C)[10]
Mangroves			
Biomass[1]	147.5	540.8	2.05
Soil[2]	320.0	1,173.3	4.45
Total stock	467.5	1,714.2	6.51
Burial[3]	1.15	4.2	0.016
Salt Marsh			
Biomass[4]	3.315	12.2	0.017
Soil[5]	390	1430.0	1.989
Total stock	393.3	1442.2	2.01
Burial[6]	2.1	7.7	0.011
Sea grasses			
Biomass[7]	1.84	6.7	0.059
Soil[8]	70	256.7	2.233
Total stock	71.8	263.4	2.3
Burial[9]	0.83	3.0	0.026

[1] *Twilley (1992), Donato et al. (2011).*

[2] *Kristensen et al. (2008), Donato et al. (2011), Chmura (2003), assuming 1 m depth of carbon rich soils.*

[3] *Bouillon et al. (2008).*

[4] *Mitsch and Gosselink (1993); Cebrian (1999).*

[5] *Chmura et al. (2003), assuming 1 m depth of carbon rich soils.*

[6] *Chmura et al. (2003).*

[7] *Duarte and Chiscano (1999).*

[8] *Laffoley and Grimsditch (2009).*

[9] *Duarte et al. (2005), Kennedy et al. (2010), and Duarte et al. (2010).*

[10] *Estimated global coverage is 139,170 km^2 for mangroves; 319,000 km^2 for seagrasses; and 51,000 km^2 for salt marshes.*

Table 3.9 further summarizes our global estimates of carbon stock and annual burial and also lists our estimates of annual emissions from the three blue carbon ecosystems. We estimate that mangroves, salt marshes, and seagrasses together store about 10.80 Pg C (about 39.6 Pg CO_2e). Most of the blue carbon pool is in the soils, which contain more than 80 percent of the overall carbon stock.

Table 3.9 Estimates of Global Carbon Stocks, Storage, and Annual Emissions

	Mangroves	Salt marshes	Seagrasses	Total
Storage (Pg C)				
Biomass	2.05	0.02	0.06	2.13
Soil	4.45	1.99	2.23	8.68
Total stock	6.51	2.01	2.29	10.80
Burial	0.016	0.011	0.026	0.053
Emissions (millions tons C)				
Biomass	11.2	0.1	0.3	11.6
Soil	19.4	8.5	9.5	37.4
Burial (loss)	2.9	1.9	4.8	9.7
Grand total	33.5	10.5	14.7	58.7

Emissions estimates are mostly illustrative for salt marshes and seagrasses because data on habitat loss rates are lacking. The globally representative mangrove deforestation rate is 0.73 percent. Because information on the loss of seagrass and salt marsh areas is lacking, we use the estimated mangrove loss rate across the three ecosystems to illustrate potential emissions.

Estimating overall blue carbon emissions requires information on the rate of disturbance regarding all three blue carbon ecosystems. Unfortunately, no such information is available on salt marshes or seagrasses. For the sake of illustration, we apply the globally representative mangrove loss rate to salt marsh and seagrass areas. Using the mangrove loss rate across other blue carbon ecosystems is not completely arbitrary, however. Seagrass meadows often lie adjacent to mangroves, the loss of which likely will degrade the seagrass bed. Salt marshes are subject to similar land-use pressures as mangroves, though their much broader and different geographic range suggests specific caution when interpreting these estimates.

Our emissions estimates indicate that mangrove loss currently releases about 33.5 million tons of carbon annually. Using a CO_2 equivalent, this means that about 123 million tons of carbon dioxide is released into the atmosphere from mangrove loss. The estimated emissions from salt marshes and seagrasses (14.7 million and 10.6 million tons C yr^{-1}, respectively) are roughly 40 percent and 30 percent of the magnitude of mangrove emissions, respectively. Overall, and assuming that the mangrove deforestation rate also consistently describes the habitat loss rates of seagrasses and salt marshes, we estimate that, annually, roughly 2 million tons of carbon dioxide (58.7 million t C) are returned to the atmosphere from the loss of blue carbon habitat. Nearly 60 percent of the estimated emissions is from mangroves.

In the next section, we examine the economic potential for reducing blue carbon emissions. Because of information limitations, we focus on mangroves. For more information on the following analysis, please also see Siikamäki et al. (2012).

4.

ECONOMIC POTENTIAL for Reduced Emissions

4.1. CONCEPTUAL APPROACH

Whether blue carbon will help reduce carbon emissions depends in part on the potential economic returns to investment in blue carbon offsets. Besides the volume of offsets generated and the associated costs, the returns will depend on the potential revenue streams available for blue carbon credits. Future carbon emissions credit and allowance prices are difficult to predict, as they depend on such factors as domestic and international climate policy programs, general economic conditions, and availability of other carbon credits courses. Instead of attempting to project future carbon credits prices, we use the current prices in the European Union's Emissions Trading Scheme (EU ETS) as a benchmark. EU ETS is by far the world's largest trading mechanism for carbon emissions allowances. It determines the market price for allowances within the EU, and it also includes an offset market through which firms can purchase emissions reductions outside the EU to offset emissions within the EU. Although land management–based carbon offsets are not part of the EU ETS, the carbon offset price[14] is a useful indicator of the scale of potential revenues from blue carbon offsets.

The cost per ton of potential carbon credits generated from mangroves is a function of the opportunity cost of mangrove hectares and the current and future direct outlays from protecting the hectares. Unfortunately, a spatially explicit data set of land prices, which provide a measure of the opportunity costs of land, does not exist. In a global assessment, Murray et al. (2011) use country-level estimates of agricultural land values. Country-level data, however, do not reflect the sometimes substantial variation in land prices within a country.

Our goal in estimating the opportunity cost of land is to project the spatial variation of land values within each country while remaining consistent with country-level estimates of land values. We achieve this by combining a spatially explicit global data set on gross economic revenues per hectare from agriculture and rangeland management (Naidoo and Iwamura 2007) with country-level estimates of agricultural land values from the World Bank.[15]

Naidoo and Iwamura (2007) (hereafter NI) estimate the annual flow of gross revenues on a global \$/ha basis from 42 crop and 6 livestock types, based on crop productivity, livestock density, and global prices in 2000. The NI data encompass both coastal and inland areas; we use both in the calibration routine but draw from information specific to mangrove areas to

14 These offsets are called Certified Emission Reductions, or CERs, and they are available through the Clean Development Mechanism (CDM) and Joint Implementation (JI) frameworks set forth in the Kyoto Protocol.

15 The World Bank estimates of agricultural land values are obtained as an area-weighted average of cropland and pasture.

estimate their opportunity cost. The World Bank country-level data measure the present value of economic revenues minus costs and, as such, are a better measure of the price of land. The calibration exercise calculates the ratio between the country-level weighted average of the NI data and the World Bank data. The ratio of the two measures is then used to scale the NI data. The calibration accounts for the differences in the scale of the numbers due to the treatment of costs (NI data address gross revenues whereas we need information on net revenue streams).

The original NI data set covers approximately 38 percent of all 9-by-9-km grid cells in our mangrove data and 35 percent of total mangrove area. To increase coverage of the opportunity cost data, we also use NI estimates from areas near mangroves to fill in missing values.[16] The result is a smoother (less pronounced) surface of potential economic returns to land than in the original NI data. Depending on the maximum distance allowed when imputing missing opportunity cost data with data from nearby cells (we use 13, 26, and 39 km distances), the coverage of cells with comprehensive data on the global scale increases to 87 percent for 13 km, 94 percent for 26 km, and 95 percent for 39 km. Unless otherwise stated, the results that follow use NI estimates averaged using a 39-km distance (Siikamäki et al. 2012).

The one-time costs of setting up protection from mangroves and the annual management cost per hectare follow Murray et al. (2011). We convert the per-year management cost into the present value of a stream of annual costs over a 25-year period using a 10 percent discount rate.

Putting the pieces together, the full opportunity cost of a hectare of land is expressed as follows:

Opportunity cost per hectare$_{ij}$ (\$/ha) =

Net present value of estimated economic returns from most profitable land use (land price)$_{ij}$

+ One-time setup cost

+ Present value of annual costs of managing protected area,

where the *ij* subscript indicates that the measure varies within country (i) and cell (j). The price per ton of carbon is equal to the opportunity cost per hectare divided by the total avoided emissions of carbon associated with protecting the hectare. Specifically, we have

$$\textit{Per ton cost of carbon}_{ij}\ (\$/ton) = \frac{\textit{Opportunity cost per hectare}_{ij}\ (\$/ha)}{\textit{Total avoided emissions of carbon}_{ij}\ (tons/ha)},$$

where the total avoided emissions (TAE) of carbon varies across countries because of differences in soil carbon content and varies within and among countries because the aboveground biomass estimates vary by latitude.

Although our estimates are based on the opportunity cost of land for agriculture, some other land uses, such as certain types of aquaculture and especially residential development, could plausibly be much more valuable. In locations where hotel development is planned in mangrove areas, for example, the cost of avoiding the emissions could be an order of magnitude greater than the estimate based on agricultural revenues. Regardless, a hotel development

16 More specifically, we use a nearest-neighbor averaging routine on the calibrated estimates for three different distances (13 km, 26 km, and 39 km) to fill in missing values at the cell level. The averages use all the calibrated NI data within the specified distance, including cells that have no mangroves (for example, cells adjacent to coastal cells). In some cases, the nearest-neighbor routine will pick up cells from a different country. Cells with missing data are excluded.

usually affects a limited area, so opportunities to offset its emissions for a lower cost likely would arise elsewhere.

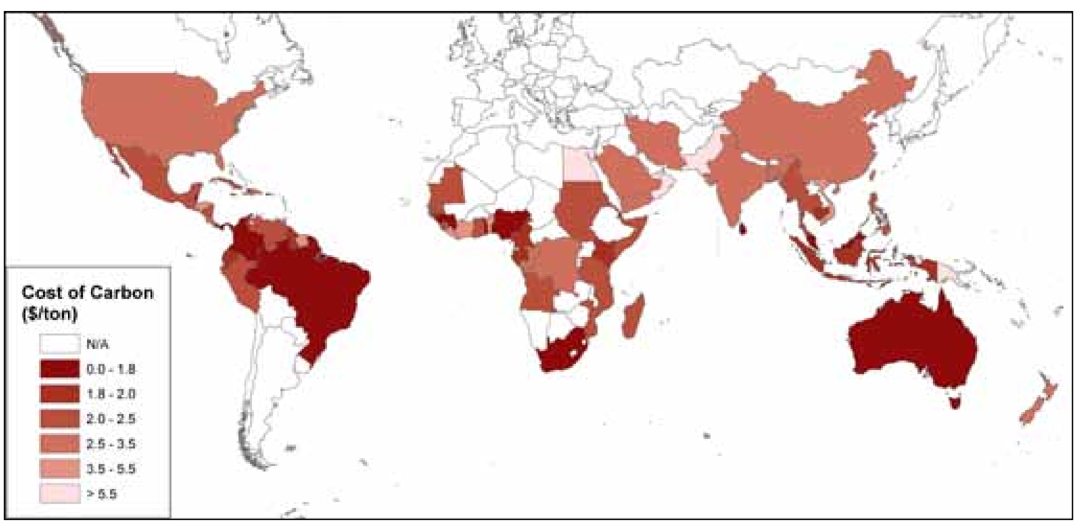
Moreover, while hotel development and shrimp farming have received considerable attention as drivers of coastal development, systematic assessments of actual land-use changes highlight agriculture as the main alternative. For example, Giri et al. (2007) find that between 1975 and 2005, more than 80 percent of the mangrove loss in several Southeast Asian countries was due to agriculture. We therefore consider it appropriate to use cost estimates based on potential agricultural revenues. In some areas, supplementing agricultural land value assessments with information on other possible land uses could provide additional information. Although incorporating such assessments into this report is not feasible, we examine the robustness of our findings using a broad range of alternative approaches to the estimation of the opportunity cost of avoided emissions.

4.2. GEOGRAPHIC DISTRIBUTION OF COST OF AVOIDED CARBON EMISSIONS

Figure 4.1 illustrates the country-level average of the estimated per-ton cost of avoided carbon emissions from mangrove loss. Because our data are at a subcountry level, in 9-by-9-km grid cells, we calculate a country-level average as a mangrove area–weighted average of the costs within each country. Each cell within a country is weighted by the hectares of mangroves in each cell relative to the country-level total. In Figure 4.1, countries whose cost per ton of avoided emissions is lowest are shaded the darkest.

Figure 4.1 Opportunity Cost of Avoided Carbon Emissions ($\$ \, t \, C^{-1}$), by Country

We find the global average cost is $682 per ha of mangroves protected. This cost includes



three components: the opportunity cost of land, a setup cost, and an annual operation and management cost.

Overall, on a per-ton of carbon basis, the lowest-cost opportunities to reduce emissions are distributed broadly across the world and include South America, Western and South Africa, and Southeast Asia. However, these estimates indicate the cost per ton of carbon emissions avoided without regard to the total potential volume of carbon emissions from a specific country. It is possible that a country with a low cost per ton of carbon also has a low total emissions reduction potential.

4.3. GLOBAL SUPPLY OF BLUE CARBON EMISSIONS OFFSETS

Using our spatial estimates of the per-ton costs and the potential volume of blue carbon offsets, we estimate a global supply curve for blue carbon. The supply curve represents the price per ton (minimum marginal cost) of avoiding blue carbon emissions under different total avoided emissions. The marginal cost of avoided carbon emissions depends on the amount of at-risk mangroves within each country, their loss rates, and the cost of preventing the loss of mangroves. To determine whether preserving blue carbon is economically feasible, we contrast the supply curve with the range of EU ETS prices in 2011.

A natural question involves identifying the mangroves that are most likely to be deforested in each country. Several authors have investigated land changes due to deforestation (for summaries of this literature, see, for example, Chomitz et al. 2006 or Kaimowitz and Angelsen 1998). Their conclusion is that deforestation is a complex process that depends on the variability in economic returns (for example, soil quality, distance to market), low-cost access to the frontier (for example, road networks), and government policies.

Examining the patterns and future risks of mangrove deforestation would require an in-depth study in each country. Because that is not feasible, we make simplifying assumptions to highlight the potential economic viability of blue carbon offsets. Because our data on deforestation risks come from country-level data, we assume ex ante that all mangroves within a country are subject to the risk of deforestation. As in earlier sections, we predict the risk of mangrove conversions using the FAO data on deforestation that is specific to mangrove habitats.[17] We also use the data on world protected areas (UNEP–WCMC 2010) to net out mangroves already protected.

We denote this supply estimate as a "uniform targeting" case, because the potential offset contracts would protect all at-risk mangroves within a specific data cell. The annually avoided emissions for each data cell are determined by the total area, deforestation risk, carbon pool, carbon sequestration rate of mangroves, and other factors within each cell (in other words, the TAE discussed in Section 3). The deforestation risk itself is constant across all mangroves within a country. We also consider other targeting cases (Section 4.6), but for illustrative purposes, we use the uniform targeting case as our base case.

To reflect scientific uncertainty about the amount of carbon in the soil along with uncertainty on how potential offset contracts would consider this carbon, we present the supply curve assuming high, low, and middle estimates of carbon dioxide (Figure 4.2).

Our middle estimate of supply indicates a maximum supply of slightly less than 34 million

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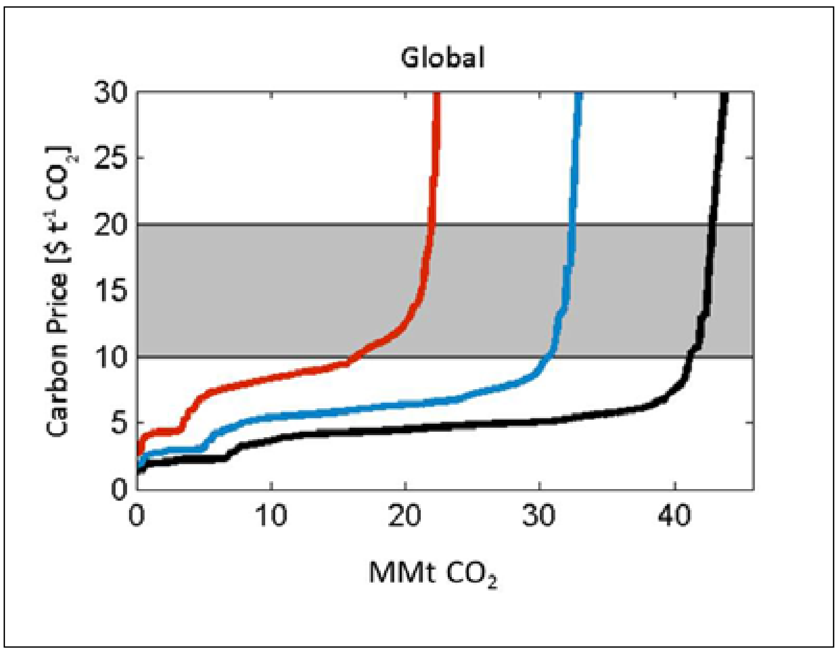
17 Because not every cell with mangrove hectares in a country is in our data set (because of missing data on opportunity cost), we adjust the FAO deforestation rate to account for the difference between the country-level at-risk hectares and our within-sample calculations of country-level at-risk hectares. The result is an FAO deforestation rate that is adjusted upward based on the ratio of total mangroves in a country to our sample estimate of total mangroves. The adjustment varies based on the averages for land prices that we use because of data coverage differences.

tons of carbon dioxide annually. Logically, the cases with low and high soil carbon lead to a lower and greater potential supply of carbon, respectively, in terms of both the total potential supply and the supply for a given price per ton. Whereas the total maximum supply in the low case is about 23 million tons of carbon dioxide, the high case projects a supply about twice that (about 44 million tons CO_2).

The approximate range of EU ETS emissions offset prices in 2011 (the shaded area in Figure 4.2) provides a reference point from an actual offset market. In all cases, we estimate that a majority of the blue carbon would be economically attractive at the minimum offset credit price of about $10, which was observed in the first half of 2011. These results are consistent with Murray et al. (2011).

Figure 4.2 Global Supply Curve for Blue Carbon with Low, Middle, and High Estimates of Soil Carbon



4.4. REGIONAL SUPPLY OF BLUE CARBON EMISSIONS OFFSETS

The global supply of potential blue carbon offsets is a product of their availability by region, by country, and within countries. We demonstrate this variation by deconstructing the global supply curve and illustrate the contribution of three regions to the global supply curve: the Americas and the Caribbean, Africa and the Middle East, and Asia and Oceania (Figure 4.3). These regions are broadly defined and based on longitude.

Using these three regions, Figure 4.4 graphs the global and regional supply curves. When mapping these curves, we plot only the high and low soil carbon cases. As expected by our previous findings, we see that the Asia and Oceania region is the potential largest supplier of blue carbon offsets. For example, in the case of high soil carbon, Asia and Oceania together could supply more than 25 million tons of offsets of the roughly 45-million-ton maximum annual global supply. We also find that although the price per ton is low for the Americas and the Caribbean, there is relatively little supply of potential blue carbon offsets available. Similarly, the potential contribution from Africa and the Middle East to the global supply of blue carbon is fairly limited.

Figure 4.3 Regions in Supply Assessment

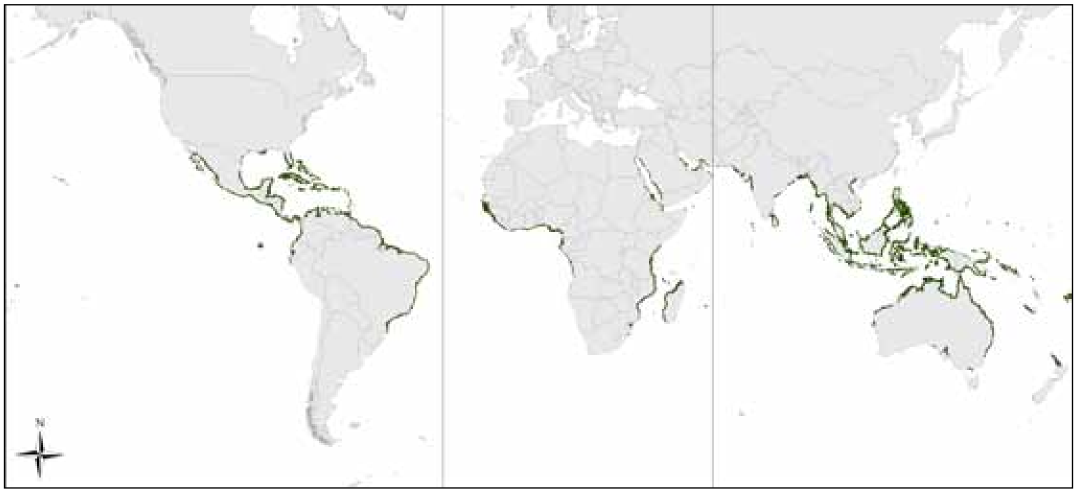
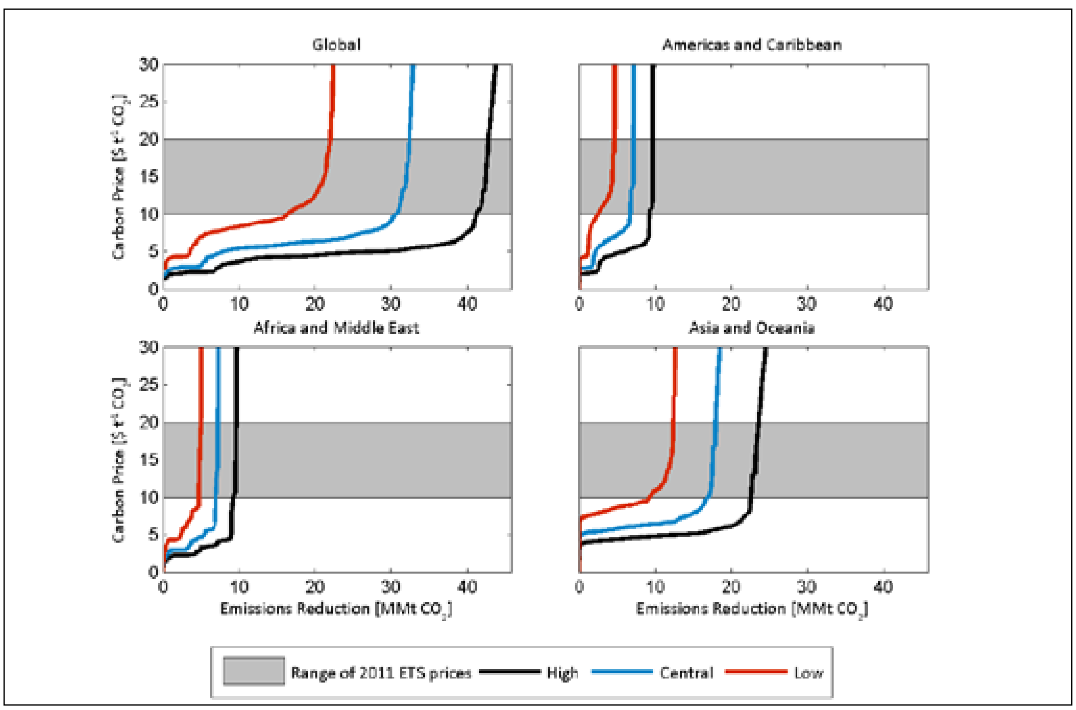


Figure 4.4 Global and Regional Supply Curves



Note: Compiled using results from Siikamäki et al. 2012.

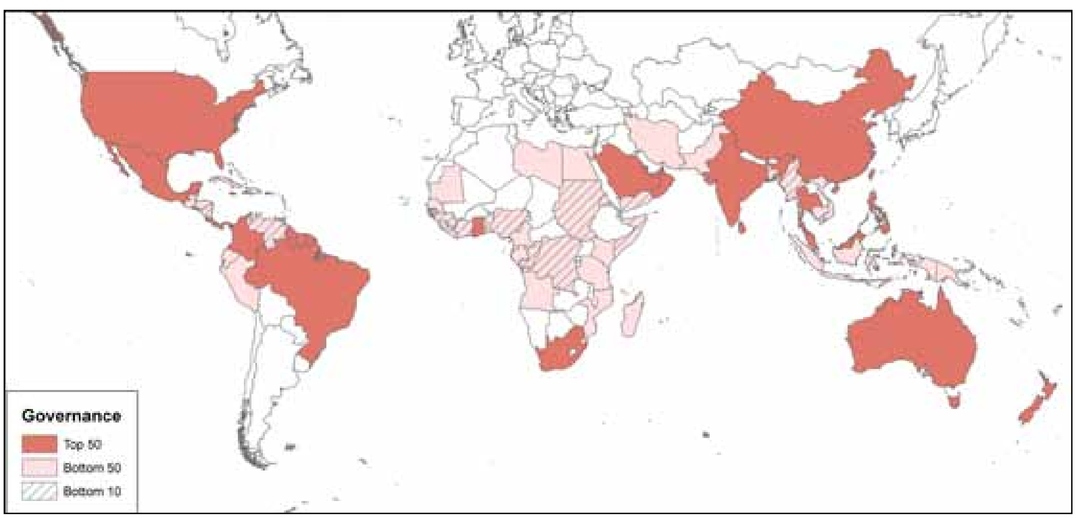
4.5. GOVERNANCE CONSIDERATIONS

Our supply estimation in the previous section assumes that the potential market for offsets includes the entire set of countries with emissions from mangrove losses. These countries vary greatly, however, in terms of stability of governing institutions and the corresponding risks associated with investing in long-term projects. Using the World Bank index on governmental effectiveness, we demonstrate two thought experiments illustrating the potential effect of host-country governance on the supply of blue carbon offsets. In these experiments, we restrict offset contracts to the countries in the top 50th or 90th percentile of the governance index (Figure 4.5). This both reduces the supply of blue carbon offsets (less carbon available) and increases the price per ton.[18]

Figure 4.6 illustrates the two cases using the central case. We find that excluding the bottom 10th percentile (countries with the worst governmental effectiveness, according to the World Bank) does not drastically change global or regional supplies. Not surprisingly, excluding the bottom half has a more dramatic effect: effectively all potential blue carbon in Africa and the Middle East is excluded from consideration, and the global supply of potential carbon offsets shrinks by two thirds.

Clearly, the stability of institutions and their effectiveness can have substantial effects on the potential size of the blue carbon offset market. This finding highlights the importance of governance considerations for REDD and other international programs to reduce land management–based carbon emissions (see, for example, Corbera et al. 2010; Johns and Schlamadinger 2009).

Figure 4.5 Countries with Mangroves, Categorized by World Bank Governance Index



4.6. ALTERNATIVE TARGETING OF CONSERVATION INVESTMENTS

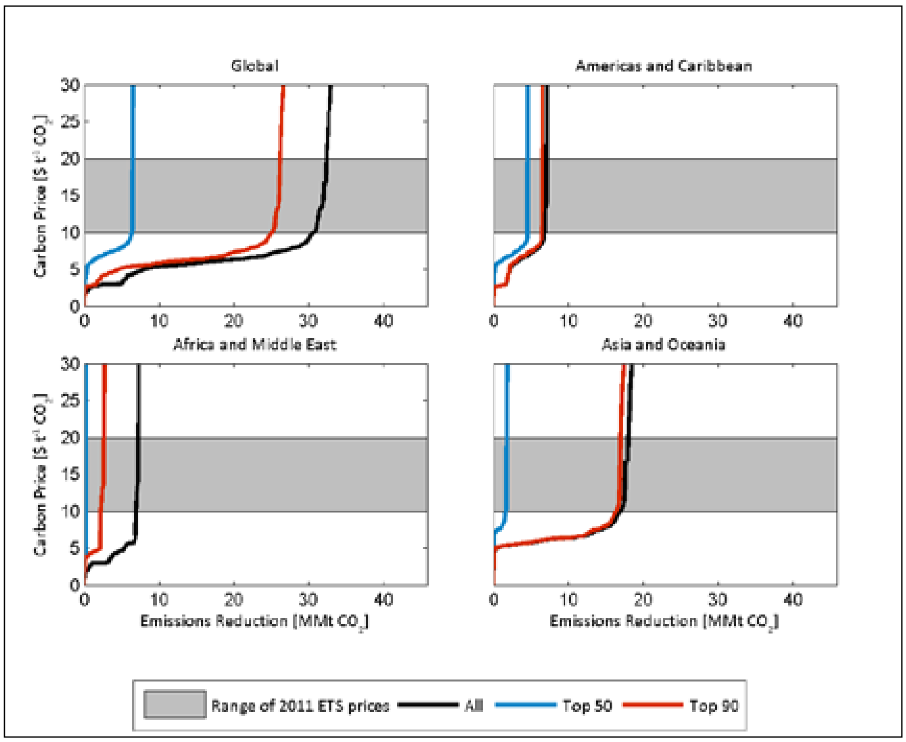
The analysis until now has focused on the uniform targeting case, where providers of offsets need to protect all at-risk mangroves within a cell even as the offsets are spread across all the cells within a country. The provider of the offset could also target the region within each cell that is subject to deforestation. Obviously, these assumptions are strong, but nevertheless the

18 The supply curve shifts up and to the left.

case is illustrative of the potential economics of blue carbon.

In this section, we introduce alternative targeting cases that further illuminate the potential range of outcomes. The first case, Cost per Ton Targeting, illustrates the lower envelope of costs, where targeting is based on the per-ton cost of avoided emissions. Within each country, the mangrove parcels are sorted by the carbon price, and only the lowest are included in the supply curve (until the country emissions baseline is met). This is a potential outcome if buyers of offsets have perfect information on the per-ton cost of avoided emissions. Of course, the information is subject to scientific, policy, and economic uncertainties, but the case highlights the possible returns from improved information.

Figure 4.6 Governance Considerations



Note: Compiled using results from Siikamäki et al. 2012.

The second case, Low Land Rent Targeting, focuses on the supply side of the offset market, where the risk of mangrove deforestation is perfectly and negatively correlated with land prices (returns from land). That is, mangroves in areas (in our analysis, cells) with the lowest opportunity cost are the locations where mangroves are at risk and are therefore the locations where offsets are available.

The third case reverses the Low Land Rent Targeting case by assuming that the mangrove deforestation risk is perfectly and positively correlated with land prices. That is, the mangroves in areas where the opportunity cost is highest (potential agricultural revenues are greatest or development is driving the price of land) are subject to deforestation and are therefore the locations where potential offsets are available through additional protections. We denote this case High Land Rent Targeting.

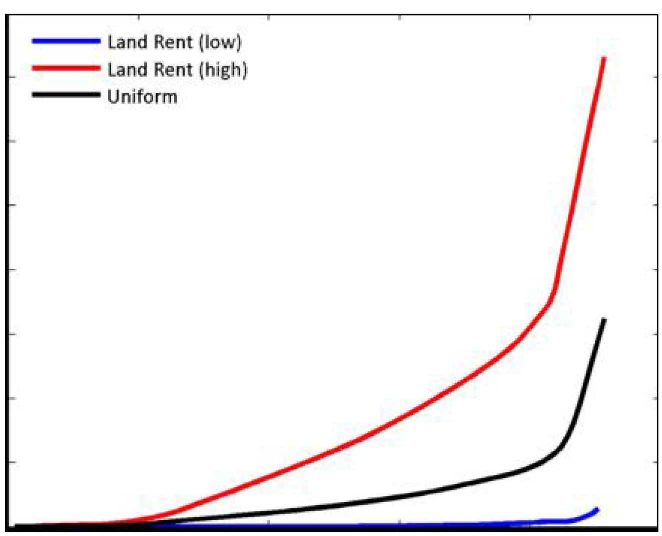
At the outset, we know that Cost per Ton Targeting, by definition, will provide the lowest cost per ton of avoided emissions, and that High Land Rent Targeting will represent the highest cost per ton of avoided emissions. Together these cases will highlight the potential range that could emerge in actual offset markets.

We next illustrate the relative cost of achieving different emissions offset goals using the alternative targeting methods. Across the targeting cases, we normalize the global supply contribution by each country to match the country-level total estimated area of mangroves that are deforested. For example, if we know that 1,000 hectares of mangroves are likely to be deforested in a country, the different targeting cases will protect mangroves in different locations (cells) until a total of 1,000 hectares is protected.[19]

Figure 4.7 summarizes the differences in the overall cost of blue carbon conservation across the cases for different levels of carbon offsets. We measure the differences from the lowest-cost case (cost per ton). As expected, different targeting cases result in different estimated costs of achieving the same targets. Using the lowest-cost potential case (cost per ton) for reference, we find that the increment cost associated with the low land rent case is small, on the order of $1 million to $2 million. The difference between the uniform targeting and cost per ton case is substantially greater, and the high land rent case further increases the cost differential relative to the cost per ton case.

Even though, empirically, the differences among the three cases could have been quite large, we find overall that the differences are relatively small given the potential size of a carbon market. Additional costs are largest at high carbon volumes and between the high land rent and cost per ton carbon cases, so we use them to illustrate the findings. When a program is designed to avoid nearly all carbon emissions from mangrove deforestation, seeking to avoid around 30 million to 35 million tons of carbon dioxide annually, the program under the high land rent scenario is about $30 million to $40 million more costly than in the lowest-cost case (cost per ton). Therefore, the cost increment is about $0.3 per ton carbon, or slightly more than $1 per ton CO_2.

Figure 4.7 Cost Differentials among Targeting Cases



19 Because we normalize by the amount of at-risk mangroves, the amount of total carbon offsets available in each of the cases can differ because of the within-country variation of aboveground biomass.

These additional assessments further strengthen our overall findings. Even though most emissions from mangrove deforestation could be avoided at costs well below $10 per ton of CO_2, we find that even the most disadvantageous assumption regarding the opportunity cost of land would add only around $1 to the estimated per-ton cost. Therefore, under a broad range of assumptions and within the relevant range of potential emissions reduction targets, the cost of avoided emissions is likely below the current carbon emissions offset prices in the EU ETS (between about $10 and $20 per ton of CO_2 in the first half of 2011).

5.

Potential Co-benefits of MANGROVE CONSERVATION

Our assessment suggests that the preservation of mangroves generally is warranted solely for the sake of preventing carbon emissions. Nevertheless, protecting mangroves will undoubtedly also contribute to the provision of other ecosystem services and goods. Mangrove ecosystems provide habitat for a wide variety of species, and the diverse landforms of mangroves themselves form an important component of biodiversity (Twilley et al. 1996). Our goal in this section is to highlight the major geographic trends in biodiversity associated with mangrove ecosystems and illustrate the potential implications for targeting the purchase of offsets to maximize one of the co-benefits from blue carbon.

We constructed country-level and within-country indicators of species richness by using species data on mangroves, amphibians, birds, reptiles, and marine mammals from IUCN (International Union for Conservation of Nature, 2010) and BirdLife International (2011). The original data sets indicate the habitat range of each species. Using GIS, we mapped the species' ranges and overlaid this information with data on mangrove locations. Using our global grid (approximately 9-by-9-km grid cells), we counted the species whose known ranges overlap with mangroves.

5.1. GLOBAL BIODIVERSITY PATTERNS

More than 70 different species of mangrove are known to exist, though the total number somewhat varies depending on the definition of mangrove. Mangrove species are often divided into eastern (Indo–West Pacific) and western (Atlantic–East Pacific) floral groups, with almost no overlap in species between them. The eastern group is considerably richer in mangrove species—more than 60, versus 12 in the western group. Nevertheless, all mangrove species share some basic characteristics. For example, they grow in or adjacent to intertidal areas and have adapted to cope with that environment, evolving different forms of aerating roots to transport oxygen to roots submerged in water or anaerobic soils.

Figure 5.1 maps mangrove species richness by country. The map shows that Southeast Asia and southern Asia are the regions with the greatest number of mangrove species. In the Western Hemisphere, the center of mangrove species diversity is in Central America and Colombia, especially on the Pacific coast. The map also shows the difference between the eastern and western groups in the number of species.

Southeast Asia is clearly the global center of mangrove species richness. In combination with other results from our study, this suggests that blue carbon conservation should concentrate in areas particularly rich in mangrove species. However, practically no species overlap exists between the western and eastern mangrove species groups (Spalding et al. 2010), so a global approach to preserving mangrove species richness might also suggest protecting western mangroves. Because conservation efforts focused on avoiding carbon emissions due to mangrove loss likely would concentrate in Southeast Asia, additional programs may be required to ensure the protection of western mangrove species.

Figure 5.1 Mangrove Species Richness, by Country

The importance of mangroves in biodiversity conservation obviously extends beyond mangrove species richness. Mangroves support a wide variety of other species, and considering the potential for their conservation is therefore also relevant. Among terrestrial vertebrates, birds are an important species group using mangroves for nesting and roosting sites as well as food. Using detailed spatial data on avian ranges, we calculate the number of bird species in mangrove areas by country (Figure 5.2.1). This assessment shows that Southeast Asia has the greatest number of bird species associated with mangroves. Looking only at the endangered bird species (Figure 5.2.2) somewhat evens the global distribution, but Southeast Asia nevertheless emerges as the global hotspot for birds associated with mangroves.

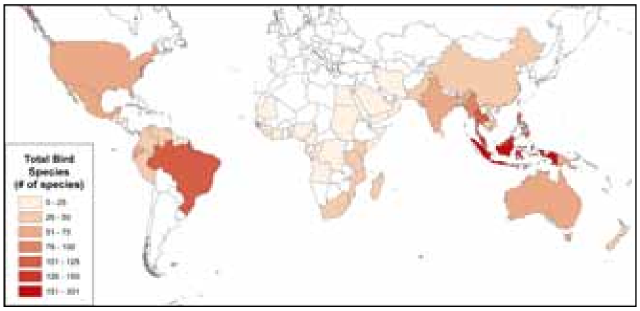
Figure 5.2.3 and 5.2.4 map the number of amphibian and reptile species associated with mangroves by country. Although amphibians are relatively broadly distributed and their number is particularly high in both South America and Southeast Asia, intertidal areas with relatively high salinity of water generally do not favor this class of vertebrates, which instead occur near or adjacent to mangroves. Efforts to protect amphibians will therefore not likely focus on mangroves.

5.2. POTENTIAL FOR BIODIVERSITY CO-BENEFITS

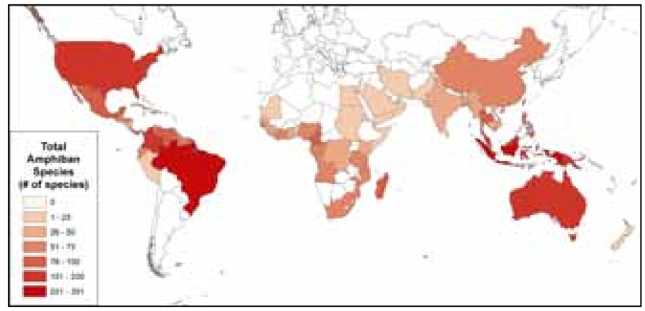
In this section, we investigate the potential for co-benefits to flow indirectly from blue carbon offsets and the potential costs of prioritizing offsets in areas with high biodiversity. We use a global analysis that pools all observations and highlights the local- or country-level variations in our data.

Figure 5.2 Bird, Endangered Bird, Amphibian, and Reptile Species Richness Associated with Mangroves, by Country

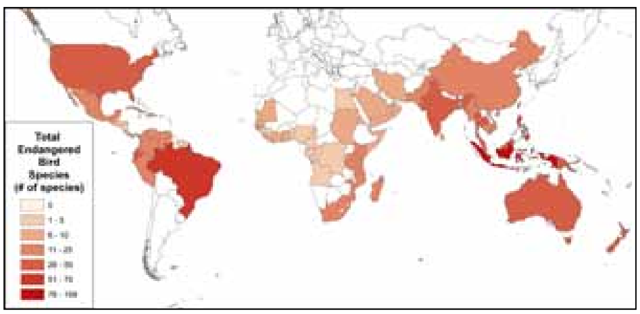
5.2.1 Birds



5.2.3 Amphibians



5.2.2 Endangered Birds



5.2.4 Reptiles

5.2.1. Global Analysis

At a global level, we examine whether three indicators of blue carbon—carbon per hectare, carbon emissions per hectare, and the cost of avoided carbon emissions—are correlated with indicators of biodiversity. Specifically, we examine blue carbon content (t C ha^{-1}), projected emissions in the area (that is, the product of carbon content and the rate of mangrove loss, or t C ha^{-1} yr^{-1}), and the opportunity cost per ton of avoided blue carbon emissions (\$ t C^{-1} yr^{-1}). Regarding biodiversity conservation, we measure the richness (total count) of all mangrove, bird, reptile, amphibian, and marine mammal species.

Table 5.1 shows global pairwise correlations that provide a lens into the potential attractiveness of mangrove areas for REDD-type programs and biodiversity conservation. The correlation coefficients in Table 5.1 measure the global correlation; that is, all the mangrove areas throughout the globe are pooled together. The global correlations indicate that the blue carbon content tends to be high in areas that are relatively rich in mangrove species (carbon/ha and mangroves are positively and relatively correlated; correlation coefficient 0.28). Bird species richness also is relatively high in areas with high blue carbon content (correlation coefficient 0.13). And, blue carbon emissions are relatively high in areas rich in mangrove and bird species.

Although the blue carbon content and projected emissions are mostly positively and statistically significantly correlated with the biodiversity indicators, there is practically no or very low correlation between the opportunity cost per ton of emissions avoided—the main measure that we would expect to influence blue carbon conservation investments—and the species richness measures. The estimated correlation coefficients between the cost per ton of emissions and all measures of species richness are negative and small, and two out of five correlation coefficients are not statistically significantly different from zero. A negative correlation between the opportunity cost of blue carbon conservation and biodiversity does, however, suggest that areas attractive to blue carbon conservation may be richer in biodiversity than mangrove areas on average.

Table 5.1 Correlations between Blue Carbon and Biodiversity Indicators

	Carbon per ha	Emissions per ha	Cost of avoided emissions ($/t C)	Mangroves	Birds	Reptiles	Amphibians	Marine mam-mals
Carbon per ha	1							
Emissions per ha	-0.026*	1						
Cost of avoided emissions ($/t C)	-0.170*	0.008	1					
Mangroves	0.283*	0.120*	-0.046	1				
Birds	0.134*	0.125*	-0.060	0.377*	1			
Reptiles	0.078*	0.057*	-0.046*	0.616*	0.465*	1		
Amphibians	0.063*	-0.014*	-0.074*	-0.321*	0.015*	-0.152*	1	
Marine mammals	0.031*	-0.001	-0.032*	0.151*	-0.00	0.265*	-0.170*	1

** Significant at 5% level*

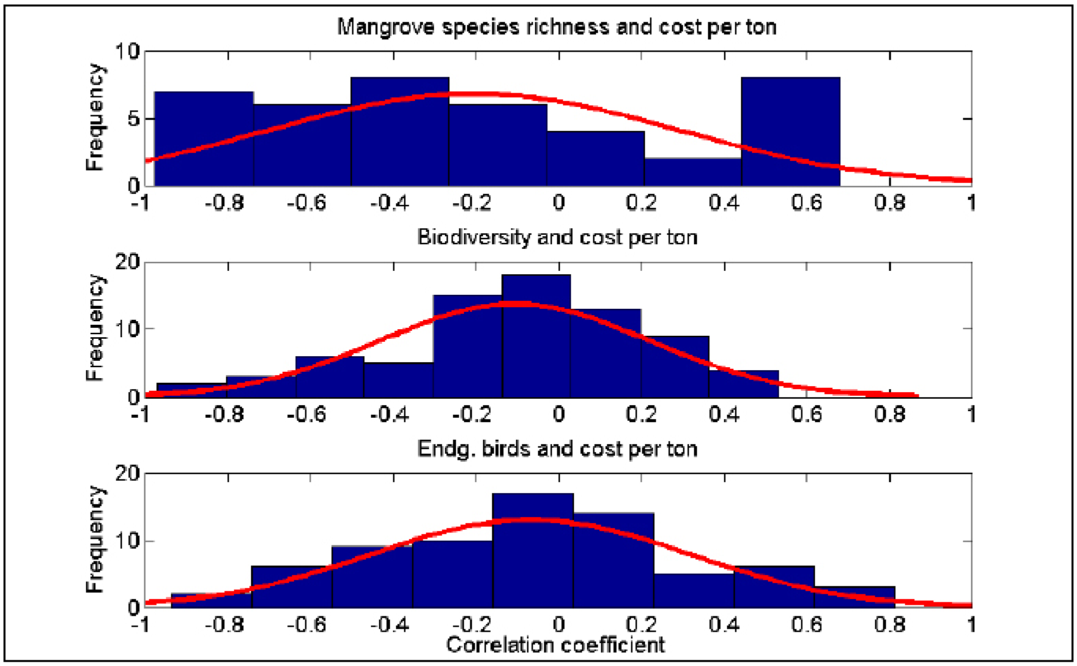
5.2.2. Local Analysis

The global-level correlations hint at the potential for co-benefits, so we used our data on within-country variation in biodiversity and blue carbon indicators to probe the question further. Figure 5.3 illustrates the distribution of correlation coefficients between per ton cost of carbon and mangrove species richness, biodiversity richness, and number of endangered bird species across the countries with mangroves at risk. Biodiversity richness measures the total number of mangrove, bird, reptile, marine and terrestrial mammal, and amphibian species in the each cell.

Overall, the local results are consistent with those in Table 5.1. The different distributions of correlation coefficients, however, clearly highlight the variability within each country. For example, whereas the global correlation of mangroves and per-ton cost is -0.046, the range goes from less than -0.8 to 0.6. The implication is that in some parts of the world, cost per ton is positively correlated with mangrove species richness, and the likelihood of co-benefits for mangrove species richness is small (that is, costs could be prohibitive). On the other hand, other countries have correlation coefficients greater than 0.5, implying that places with low costs per ton also have high species richness. The other indicators reveal similar patterns in terms of the correlation coefficients and potential for co-benefits.

Both the global and the local correlation coefficients suggest that mangrove conservation focused solely on generating carbon credits will not automatically target the areas most valuable for biodiversity conservation. That is, blue carbon conservation programs that focus exclusively on carbon benefits will be roughly as effective at producing biodiversity benefits as a program that randomly selects mangrove parcels for protection. Although carbon-focused mangrove conservation will benefit biodiversity, it will not achieve the same level of benefits that could be attained using an approach focused on biodiversity.

Figure 5.3 Distribution of Correlation Coefficients
in Countries with Current Mangrove Losses



5.2.3. Biodiversity-Focused Provision of Blue Carbon Offsets

In this section, we examine three alternative biodiversity goals for purchasing offsets. We contrast approaches that preserve mangroves based on mangrove species richness, combined species richness, and number of endangered bird species, by country.[20] As with the carbon price and land rent targeting cases, we assume that the at-risk hectares available for the offset market occur in the cells that are being targeted based on their co-benefits. Furthermore, cells within a country are included until the country total of at-risk mangrove hectares is met.

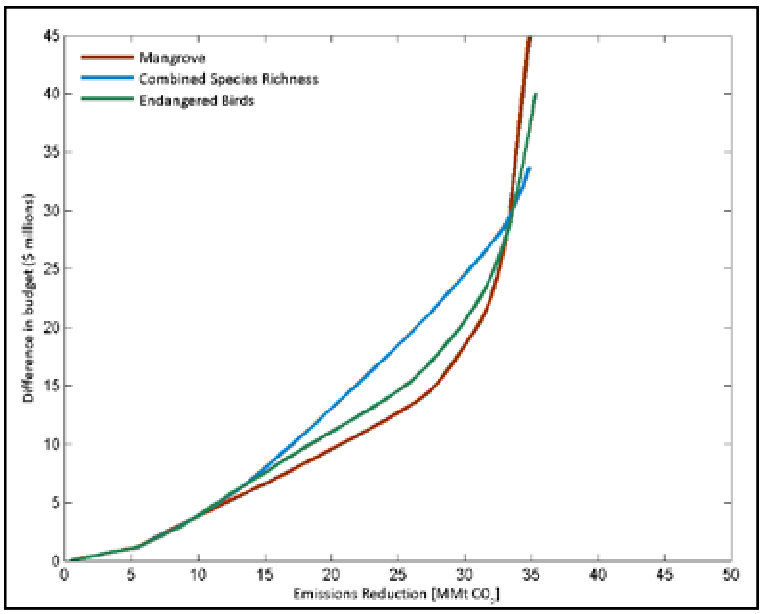
The cost of following this strategy is an increase in the price per ton of avoided emissions relative to the most cost-effective carbon price targeting scenario (avoided emissions at the least cost per ton of carbon). The additional cost represents a lower bound on the potential economic values associated with the biodiversity needed to make blue carbon offsets economically attractive (the benefits are greater than the costs).

The additional cost depends on the particular strategy and the amount of carbon per ton

20 In cases where two cells in a country have the same species richness, we break the tie by choosing the cell with the higher number of marine mammals.

(Figure 5.4). For example, for most of the potential range of carbon credits, following the combined species richness strategy results in the highest costs, except at the upper range of carbon. Why might there be a shift in the ordering? Recall that the supply curve of carbon in each case combines blue carbon from around the globe. Adding the additional carbon once we get close to the annual limit occurs in cells where the biodiversity indicator is negatively correlated with cost per ton. We also find that the cost differences between the uniform strategy and either the mangrove species richness or the number of endangered bird species are not significantly different.

Figure 5.4 Additional Cost of Targeted Approach Based on Co-Benefits



Overall, the magnitude of the additional total costs from following a co-benefit strategy over the most cost-effective carbon strategy is about $35 million annually, with significantly lower additional costs at lower levels of total carbon supply. In terms of the size of the potential carbon offset market, these additional costs appear to be nearly insignificant.

6.

Blue Carbon in CLIMATE POLICY Frameworks

Policymakers and advocates are becoming aware of the effect that blue carbon could have on climate change mitigation efforts, but the conversations about blue carbon are in the formative stages: viable conservation options have not yet been developed. Success in protecting these ecosystems will depend, in part, on the design and implementation of mechanisms to value their stored carbon. Existing structures for valuing carbon storage in natural systems may provide useful examples or even vehicles for blue carbon. Here we look at current proposals with room for the inclusion of blue carbon as well as some of the challenges that need to be overcome to make blue carbon credits available to the international community.

6.1. CURRENT INTERNATIONAL PROPOSALS

Blue carbon has yet to establish a notable presence in international negotiations, though parties to the United Nations Framework Convention on Climate Change (UNFCCC) have acknowledged to some degree the potential benefits of maintaining stored carbon in blue carbon ecosystems. The official proceedings of the convention have set a precedent for developing mechanisms through which ecosystems are valued and protected—specifically through the concept of reducing emissions from deforestation and forest degradation (REDD). Since its introduction to official proceedings of the UNFCCC in 2005, REDD has been a prominent aspect of international climate negotiations. The UNFCCC negotiations in Cancun in December 2010 formally established many important aspects of REDD+,[21] including basic guiding principles, a distinct scope for eligible activities, and initial frameworks for payment mechanisms (UNFCCC 2010). A larger agreement has not been ratified by the negotiating parties, so the REDD language is not yet binding.

The similarities between blue carbon credits, especially for mangroves, and REDD credits suggest that including blue carbon in REDD structures is a viable path forward. A coalition of marine-focused organizations and researchers called on UNFCCC to include blue carbon in its deliberations in the leadup to the Cancun meeting (BCC 2010). Similarly, a group of scientists and organizations called the International Working Group on Coastal "Blue" Carbon made recommendations in March 2011 to develop financial incentives to reduce emissions from coastal ecosystems and to include mangroves in national REDD+ strategies and actions (IWGCBC 2011).

21 REDD counts only deforestation and forest degradation; REDD+ is an expanded concept that includes conservation of forest stocks, sustainable forest management, and enhancement of forest carbon stocks.

Bottom-up efforts from these groups have increased the exposure of blue carbon issues enough to catch the attention of some non–Annex 1 countries. As a result, the issue of conducting more research on blue carbon and including it in systematic observations of important ecosystems was brought before the Subsidiary Body for Scientific and Technological Advice (SBSTA) during the UNFCCC intersessional meetings in Bonn in June 2011 (IISD 2011). Papua New Guinea introduced blue carbon into the agenda and, with the Coalition of Rainforest Nations, advocated for its inclusion. Although most parties approved more research on the topic, there was strong opposition from Bolivia and Venezuela, both of which feared that blue carbon would generate new market mechanisms that will not adequately protect natural systems (IISD 2011). The parties could not reach an agreement, and lacking consensus, blue carbon was not included as a subject for further research.

Despite that setback, opportunities to include blue carbon in UNFCCC processes will arise in the future. It is apparent, however, that understanding of blue carbon is not sufficiently mature to warrant a separate mechanism. Incorporating it into UNFCCC REDD+ structures may be a viable though currently limited option. Based on their height, density, and land cover, some mangroves can be classified as forests, depending on the definition established by specific countries. These qualified mangroves would be eligible to be included in national REDD+ plans, which all participating countries that receive funding are required to develop.

Mangrove forests share most of the same challenges facing terrestrial REDD+ programs: establishing the clear additionality of projects, ensuring the permanence of credits, identifying specific drivers of deforestation, and developing robust measurement and verification standards (Emmett-Maddox et al. 2010). Understanding the volume that mangrove losses contribute to overall deforestation in a REDD+ nation presents an extra obstacle to their inclusion in the program because that information will be a critical component in establishing baselines by which the performance of each country is measured. Further, the amount of carbon stored in the soil of mangroves proves a particularly important challenge because the Tier 1 standards, the most basic and least resource intensive of the Intergovernmental Panel on Climate Change (IPCC) accounting standards, address only the top 30 cm of soil or are based on emissions rates (O'Sullivan et al. 2011). Neither approach is comprehensive enough to count all the carbon stores in mangrove systems. Standards that would fully capture the carbon sequestration by extending the soil depth are more expensive (and for some developing countries, too difficult) to implement (O'Sullivan et al. 2011).

The inclusion of mangroves in REDD+ faces obstacles, but the other major blue carbon habitats, seagrasses and salt marshes, do not even fit in the current definition of eligible ecosystems in REDD+ programs. To include them, REDD+ programs would have to expand beyond forestry into other land-use types. Such expansions have been discussed in negotiations, but the parties have decided to focus on forestry for the time being. SBSTA's report on the role that land use plays in driving deforestation is due to be delivered in time for COP 18 in December 2012 (O'Sullivan et al. 2011). There may be room in those discussions to include blue carbon ecosystems besides mangroves, though the process for inclusion will be much like the lead-up to Bonn. Seagrasses and salt marshes may remain on the sidelines until REDD+ programs have more on-the-ground experience and monitoring techniques advance.

6.2. BILATERAL AGREEMENTS

Another and perhaps more promising vehicle for valuing and preserving blue carbon is bilateral deforestation agreements. In 2010, Norway agreed to support Indonesia's REDD efforts with up to US$1 billion, some of which will be used to develop a national REDD strategy (Kingdom of Norway and Republic of Indonesia 2010). The rest will be distributed based on Indonesia's performance in delivering actual, verifiable emissions reductions. This arrangement will continue regardless of the status of UNFCCC negotiations.[22]

The advantage of a bilateral arrangement is that it can more easily include many kinds of land-use practices. For example, the Norway–Indonesia partnership covers not only forests but also peatlands, which store substantial amounts of carbon in their soils. Indonesia has the most extensive blue carbon resources in the world and could take a major step toward protecting those resources by including blue carbon in its national strategy. Similarly, if other countries follow in the footsteps of Norway and Indonesia, they will be able to develop national strategies that are compatible with but more extensive than the UNFCCC guidelines. Blue carbon could easily be identified as a priority for funding if the participating countries define the rules for a project's eligibility accordingly.

The bilateral approach also may lay the groundwork for future agreements for developing countries to supply carbon offsets to help developed countries achieve domestic emissions reduction goals. In a hypothetical example, a developed country would look to offset some percentage of its emissions through crediting or another mechanism from a developing country with substantial carbon stock resources. As with the Norway–Indonesia partnership, the two countries would define the types of projects that are eligible to be counted as offsets as well as the rules for determining the quality of those offsets; blue carbon could be included. Such an approach has already found its way into proposed legislation in the United States. Specifically, language in the American Power Act of 2010, which would have established a cap-and-trade system, recognized the value of carbon stores in marine ecosystems and paved the way for their inclusion in future offset programs (BCS 2010). Although this bill did not pass, having such language within its provisions increases the likelihood that blue carbon will be part of U.S. offset programs with developing countries in the future.

6.3. REGIONAL AND STATE PROGRAMS

Although the international community has not yet established a comprehensive carbon market, some regional- and state-level programs in Europe and the United States are in operation or close to inception. Many incorporate the use of carbon offsets from land use and natural systems, including avoided deforestation. These programs may eventually provide another way to include blue carbon in climate mitigation efforts, but the current rules governing which offsets are allowable make it unlikely they will include blue carbon in the short term.

The largest and most comprehensive cap-and-trade system is the European Union Emissions Trading System, which has been in operation since 2005. With 30 participating countries, it covers close to 40 percent of the CO_2 emissions from the European Union (Ellerman and Joskow 2008). It accepts offsets in the form of Clean Development Mechanism (CDM) or Joint

22 Either Norway and Indonesia will continue their partnership as it is currently constructed, or the agreement will get folded into REDD under UNFCCC.

Implementation (JI) credits, excluding land use and nuclear power. The European Commission investigated including land-use offsets in the third phase of the EU ETS but had serious reservations about several issues, including the reliability of monitoring, the reporting and verification systems, and the permanence of credits (O'Sullivan et al. 2011). As a result, land-use credits will not be included in the EU ETS until after 2020; at that point, the ETS will accept only approved CDM or JI methodologies. A CDM methodology was accepted for mangrove restoration in June 2011, but no methodologies exist for avoided destruction of mangroves, nor have any been developed to cover seagrasses and salt marshes (UNFCCC 2011).

Despite the lack of progress toward a national cap-and-trade program similar to the EU ETS in the United States, smaller-scale programs exist at state and regional levels. The one with the most potential for blue carbon is California's Global Warming Solutions Act, also known as AB 32. The act aims to reduce California's emissions to 1990 levels by 2020 through a combination of regulations and cap-and-trade markets. Offsets are included in the market, and a number of offset design methodologies, including one for avoided deforestation, have been developed for the system. The provisions took effect at the beginning of 2012.

Mangroves could potentially qualify as a credit for Reduced Emissions from Deforestation (RED) under AB 32, where *forest* is defined broadly enough to include some mangroves. RED credits currently must be located in the United States, however, and this severely limits the areas of mangroves eligible for inclusion. California has signed agreements with Chiapas (on the Pacific coast of Mexico, where mangroves occur) and Acre (in inland Brazil) to develop offset programs, but those programs are still years away from providing RED credits.

Using our data, we more closely examined the situation in Chiapas. Overall, the region has some 497 km^2 (192 square miles) of mangroves. We estimate that the Chiapas region loses about 600 hectares of mangroves annually, and that each mangrove hectare contains about 936 tons of carbon dioxide (255 t C) that would be released into the atmosphere in case of land conversion. According to our opportunity cost estimates, the emissions from Chiapas can be avoided at just a few dollars per ton of carbon. Although the price ranges of emissions allowances and offsets under AB 32 are yet unknown, the available economic assessments (see, for example, California Environmental Protection Agency, Air Resources Board 2010) suggest that avoided emissions from mangrove deforestation could provide a cost-competitive emissions offset alternative. As such, mangrove offsets in Chiapas could be a natural area to develop demonstration projects for blue carbon offsets.

Another potential barrier to blue carbon's inclusion is California's current RED methodology, which does not consider soil carbon and thus makes it far more difficult for mangroves to compete with the other potential sources of credits. Moreover, the protocols do not include salt marshes or seagrasses at all.

The other regional cap-and-trade system in the United States, the Regional Greenhouse Gas Initiative (RGGI), allows afforestation offset credits, but the project must be located within one of the member states (RGGI 2008). All RGGI states are located in the Northeast and have no mangroves—the only blue carbon habitat eligible for afforestation under the program.

7.

Discussion and **FUTURE RESEARCH** Directions

Considerable progress has been made in recent years on international efforts to combat climate change, yet major challenges remain. Regulating land management related to greenhouse gas emissions only complicates the vast political difficulties in formulating a comprehensive and effective global climate policy framework. Recent advances in understanding how deforestation affects the global carbon cycle are significant, but much more remains to be done before emissions from deforestation can be effectively regulated. Nevertheless, emissions offset programs and voluntary country-level commitments to reducing emissions from deforestation are promising first steps for directly incorporating land management–based emissions into climate policy agreements.

Whether blue carbon can and should be considered in this context depends, in part, on whether carbon storage and sequestration of coastal habitats are sufficient from an emissions standpoint. Using a set of spatial estimates, our results suggest that on a per-hectare basis, carbon emissions from mangrove deforestation are large, especially in comparison to other terrestrial ecosystems. Although mangroves only occupy a tiny share of Earth's surface, the total volume of carbon they hold is substantial. Given the current rates of conversion, which in some areas of the world are extraordinarily high, much of this carbon pool in mangrove ecosystems is likely to be released into the atmosphere unless additional conservation efforts are undertaken.

Our assessment has focused on evaluating whether blue carbon conservation actions would be warranted on an economic basis: do the carbon benefits from investment in blue carbon conservation outweigh the costs? Inevitably, there will be cases where preventing mangrove loss could be excessively costly, but we find that, by and large, preserving mangroves may provide relatively low-cost opportunities to mitigating carbon dioxide emissions. Using a spatially differentiated, local-level assessment with a global scope, we find that reducing mangrove deforestation could generate economically competitive carbon emissions offsets. For example, almost all of the available emissions reductions from mangroves are initially in the range of $4 to $10 per ton of CO_2. In most areas of the world, the cost of preventing a ton of carbon emissions from mangrove deforestation is below the cost of reducing carbon emissions from industrial sources in developed countries.

Perhaps the most central finding of this assessment is that although reducing carbon emissions is only one of many benefits from mangroves, their preservation may often be warranted on that basis alone. But coastal conservation would also bring other benefits, such as biodiversity protection and benefits to fisheries and local communities (see, for example, Barbier et al.

2008 and Barbier et al. 2011). These additional benefits from conservation add further justification to protecting mangroves.

We would expect the cost of avoiding emissions to rise over time as opportunities to generate additional offsets become more constricted, but it is difficult to predict the rate of this cost increase. Additionally, if the emissions allowance markets are efficient and no major changes in the supply and demand of emissions allowances occur, a realistic prediction would be that the price of offsets would also rise at the rate of interest. Given a 5 percent interest rate, an offset with a price of $5 today will be equal to $10 (minimum price observed in the EU ETS between January and August 2011) in approximately 14 years. In the long run, the relative changes in the cost of generating offsets and their market prices will determine whether blue carbon offsets remain competitive in the market.

One potential concern regarding blue carbon conservation is host-country governance. We find that setting a minimum requirement for governmental effectiveness could drastically reduce the volume of offsets. Using the World Bank's governmental effectiveness indicator, we found that the reduction could be 17 to 75 percent less than is otherwise available; these determinations can take whole regions—particularly Africa and the Middle East—off the market. Whether investment decisions regarding blue carbon conservation could incorporate some combination of negotiating and enforcing long-term contracts with governments is an open question.

Finally, there is the critically important question of whether land management–based carbon offsets will expand to include blue carbon. In international climate negotiations, the current methodology for REDD+ offsets does not readily apply to blue carbon ecosystems. The prospects for including blue carbon in the EU ETS and RGGI are also not very promising. Under California's AB 32, however, mangrove offsets in Chiapas could be a natural area to develop demonstration projects for blue carbon. Furthermore, our results suggest that the pure economics of blue carbon could be an additional enabling factor in the adoption of blue carbon offsets.

Our analysis also shows that co-benefits in the form of biodiversity gains from blue carbon conservation for emissions offsets are possible but not necessarily particularly large. If the blue carbon offset market proceeds and offsets are provided at the lowest cost, some biodiversity gains will likely follow. However, a biodiversity-focused approach would achieve more. Not surprisingly, the more biodiversity-focused approach will be more costly, on average, per ton of carbon offsets generated. Therefore, the question becomes whether the additional costs of the more biodiversity-focused approach outweigh its additional costs, and whether biodiversity benefits from blue carbon conservation could somehow be appropriated by the offset provider. Carbon offsets that also guarantee specific co-benefits may be more valuable in the market, but experience in this context is limited.

Important areas for future work would include improving the estimates of the opportunity costs of protected mangroves. In some locations, researchers will need to consider the economic returns from aquaculture, especially within the Asian Pacific region (Murray et al. 2011). In other locations, the deviation between agricultural returns and land prices can be driven by urban and tourism development. These development pressures can result in higher prices for land than we considered in our study.

Another area of future research involves predicting the emissions profile from blue carbon ecosystems after land conversion or other disturbance. The current literature offers only very limited guidance, and much more needs to be done so that emissions from blue carbon ecosystems can be more accurately estimated. For example, available assessments of blue carbon emissions, including this one, posit that each type of land conversion in a given location has a uniform emissions profile. In reality, emissions will likely differ between, say, conversion to agriculture and development for urban uses. Emissions profiles of different forms of agriculture or aquaculture may also differ, and further information would help in estimating emissions and in configuring land-use changes, if otherwise unavoidable, to minimize emissions.

Yet another area for future research involves blue carbon ecosystems' economic value. Mangroves are known to deliver considerable benefits to fisheries by providing juvenile and adult fish populations with nursery habitat, food, and protection from predation. Mangroves and coral reefs are widely acknowledged to have an interactive relationship for fish migration and reproduction (Twilley et al. 1996). Several studies show that many fish species occur in both habitats (see, for example, Gilmore and Snedaker 1993; Sedberry and Carter 1993; Twilley et al. 1996; Mumby et al. 2004; Mumby 2006), and there is increasing evidence that coral reefs close to mangroves are considerably more productive fisheries than reefs in mangrove-poor areas. In some areas, seagrass meadows are situated near coral reefs and mangroves, thereby providing further connectivity of those areas and supporting fish species dependent on reefs and mangroves (Sanchirico and Springborn 2011). Our assessment did not consider these benefits, but future work should consider how blue carbon conservation programs can be configured to most effectively incorporate the beneficial effects of their ecosystems on fisheries.

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