

North America's Blue Carbon: Assessing Seagrass, Salt Marsh and Mangrove Distribution and Carbon Sinks



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Abstract

Blue carbon denotes the long-term storage of carbon within plant habitats growing in coastal lands and near-shore marine environments. North American blue carbon habitats—seagrass, salt marsh, and mangrove—were assessed to determine their spatial distribution and to document these sinks of blue carbon. Maps of habitat distributions on the coasts of Canada, Mexico and the United States were collected, verified and compiled to create the first continent-wide areal assessment of blue carbon. The maps show that seagrasses grow coastally throughout North America while mangroves are primarily tropical and salt marshes primarily temperate/arctic. A geodatabase was established, metadata were documented, and data and methodological gaps were assessed along with challenges in identifying the extent of these habitats. The maps compiled for North America document 24,200 square kilometers (km²) of seagrass, 13,500 km² of salt marsh and 10,100 km² of mangrove. Only half of the continent's seagrasses have been mapped; priority sites were identified for future mapping. Salt marsh mapping is incomplete for parts of Canada and Mexico. The area of blue carbon habitat within marine protected areas and terrestrial protected areas was determined.

Executive Summary

Seagrasses, salt marshes and mangroves were assessed to determine their spatial distribution and to document these sinks of blue carbon. A North America-wide database was created and maps of seagrasses, salt marshes and mangroves were compiled from international (UNEP), national (Conabio [Mexico], US Geological Survey, and the US National Wetlands Inventory), state and provincial sources, as well as from individual investigators for some seagrass distributions. Based on the areal information mapped to date, the total area of blue carbon habitat in North America is 47,776 square kilometers (km²): 1,191 km² in Canada, 17,588 km² in Mexico and 28,997 km² in the United States.

The compilation includes over one million features: 740,515 polygons (area denotations) for seagrasses, 209,771 polygons for salt marshes, and 45,872 polygons for mangroves, as well as 99,940 lines, which indicate the presence, but not area, of seagrasses. There are 24,190 km² of mapped seagrass habitat in North America: 14,352 km² in the United States (US), 9,193 km² in Mexico, and 645 km² in Canada. For salt marshes, there are 13,474 km² in North America, with 12,775 km² in the United States, 546 km² in Canada and 153 km² in Mexico. There are 10,112 km² of mangrove vegetation in North America, with 2,344 km² in the United States and 7,768 km² in Mexico.

The seagrass maps presented are the depiction of the most comprehensive distribution data compiled to date. Nonetheless, there remain many locations needing more-complete seagrass mapping, particularly the west coast of the United States, as well as parts of eastern Canada and tropical and temperate Mexico. Some of the seagrass data collected by shoreline surveys (line data) require a procedure for conversion to area (polygon) data. Sites where seagrass point data (individual observations) are documented represent high priorities for additional mapping. About half the seagrass in North America has not been surveyed for spatial distribution and area.

Tidal salt marshes have been mapped in all the US states and most of the Canadian provinces. An exception is Newfoundland, Canada, for which maps are presently being prepared. Mapped coverage of marshes in Quebec, Canada, is also incomplete. In Mexico, only marshes in Baja California have been mapped, but efforts to survey and map salt marshes in other areas of the country are underway. The salt marsh mapping presented is a compilation of the most up-to-date assessments of marsh blue carbon available.

The distribution of blue carbon habitat that has the most complete mapped coverage is that of mangrove. Mangrove distribution has been mapped using imagery available through remote sensing. The entire Mexican coastline has been mapped, as have the mangroves of the United States. Repeated mapping efforts have enabled a historical time series to assess change in mangrove area for Mexico's entire coastline.

For inclusion in this compilation, maps and data were assessed based upon currency, spatial resolution, field verification, and metadata, with every effort made to obtain recent and accurate information. However, information gaps exist. For example, in the map sources, salt marshes and mangroves might be identified as tidal but, without more-detailed salinity data, the mapping includes some freshwater tidal marshes.

Sustainability of mangroves and marshes is dependent upon the continued vertical accretion of soil to maintain their surface elevation with respect to sea level, which is expected to rise at increasing rates with global warming. Increased sediment supply enhances this process while increased nitrogen from watersheds is expected to cause a decline in production of roots, which are keys to soil accumulation and the blue carbon stock. Agriculture and urbanization of watersheds contribute to this threat. In addition, marshes and mangroves on urbanized coastlines are threatened by "coastal squeeze," a situation in which the wetland loses area at its seaward edge as vegetation succumbs to excessive flooding; accretion of wetland soil enables lateral migration of wetland over inland habitats, but this migration is prevented if barriers (walls, roads, and pavement) have been constructed at the upper

edge of the wetland. Preservation of these coastal wetlands and their carbon stocks will require intensive management, including subsidies of sediment.

Marine protected areas and terrestrial protected areas were evaluated for their proximity to or inclusion of blue carbon habitats, as well as their protective value. Few if any were found to provide adequate protection to conserve seagrasses and, to some extent, salt marshes or mangroves. Issues in mapping blue carbon were reviewed, including data gaps, methodological gaps, and challenges: completing mapped coverage is a priority.

All three habitats—seagrass, salt marsh, and mangrove—exist at the coastal margins and are vulnerable to a suite of anthropogenic stressors which threatens their health and survival, even as their blue carbon value is being recognized. In some cases, habitat area is lost before it can be mapped or evaluated. Continued efforts to complete the areal mapping of all three habitats and to protect them are critical to their preservation and to the blue carbon resources they represent.

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Status of Seagrass, Salt Marsh and Mangrove Habitats in North America

The term “blue carbon” was coined in recognition of the valuable role seagrasses, salt marshes, and mangroves¹ play in sequestering carbon dioxide. They are some of the world’s most efficient carbon sinks; on many coasts their soils have accumulated carbon for millennia. Situated at the margin of land and sea, seagrasses, salt marshes, and mangroves are sensitive to impacts from both. These habitats have suffered a past of neglect, exploitation and loss as they have been polluted and transformed to other land uses through drainage, dredging and filling.

1. Seagrasses

Seagrasses are underwater marine flowering plants that root in the sediments and produce flowers, pollen, and seeds below the surface of the coastal ocean. Being rooted in the bottom sediments, they rely on light that penetrates the water column for growth. Seagrasses are the basis of estuarine food webs that support many recreationally, commercially and ecologically important species in the estuary and beyond. Seagrass filters estuarine waters and improves water clarity, removing both nutrients and suspended sediments from the water column. Its roots and rhizomes bind and hold sediments in place, thereby reducing turbidity.

Seagrasses grow on all continents except Antarctica, in coastal and estuarine environments; they are found from the intertidal zone to 90 meters (m) deep. Seagrass meadows range in area from a few square meters to hundreds of square kilometers. Although there are relatively few species of seagrass (72 species in 14 genera and 6 families), the complex physical structure and high productivity of these ecosystems enable them to support a considerable biomass and a diversity of associated species. Seagrass meadows provide ecosystem services that rank among the highest on Earth. Their economic value is substantial, since highly valued commercial and artisanal/subsistence seafood catches such as prawns and fish are dependent on seagrass ecosystems. Seagrasses themselves are a critically important food source for dugong, manatee, sea turtles and waterfowl. Many other species of fish and invertebrates, including sea horses, shrimps and scallops, utilize seagrass for part of their life cycles, often for breeding or as juveniles. Seagrasses are considered to be one of the most important shallow marine ecosystems to humans—binding sediments, filtering coastal waters, and providing some protection from coastal erosion (Hemminga and Duarte 2000). There is a growing awareness of “seagrass blue carbon,” referring to the fact that seagrasses sequester and store carbon in their roots and sediments. Although seagrasses represent only a small area (0.2% of the surface of the ocean), it is estimated that they store 20% of oceanic blue carbon. Seagrasses are among the world’s most threatened ecosystems and yet are little known because they are usually submerged and not easily seen.

Seagrasses are an indicator of estuarine and coastal health worldwide (Orth et al. 2006a; Waycott et al. 2009). Rooted in place, seagrass integrates the influences of environmental conditions that it experiences within an estuarine system and therefore its health status acts as a barometer of impacts and changes to the estuary. Seagrass beds alter their distribution and biomass in response to changing water quality, nutrient inputs, and light levels. Seagrass change can be measured at the plant population level or by examining differences in plant physiology and chemistry. Knowing the distribution, monitoring the biomass or measuring plant tissue reveals seagrass as a multi-parameter indicator of estuarine health (Lee et al. 2004; Short et al. 2014). In the tropics, seagrass habitat is

¹ The term “mangroves” is applied herein to a variety of intertidal, arboreal vegetation species, including ferns, found in tropical climate.

often associated with mangroves, while in temperate zones, seagrasses occur in the intertidal areas of salt marshes as well as subtidally offshore of marsh habitat.

Seagrasses are in decline and, around the continent of North America, huge seagrass losses have occurred due to coastal development and anthropogenic pressures such as pollution, land-cover change, and direct physical impacts. Nevertheless, there are still vast areas of seagrass along North America's near-shore environment, but many have not been mapped, and without knowing the extent and area of these habitats, knowledge of the magnitude of blue carbon sequestration and storage is incomplete.

1.1 Seagrass Distribution

Seagrasses are widely distributed along the coasts and in the estuaries of North America, in both temperate and tropical environments. The seagrass bioregions defined by Short and Moore (2007) are used to separate North America into three units, based on seagrass species assemblages and geography: 1) Temperate North Atlantic, 2) Temperate North Pacific, and 3) Tropical America. The Temperate North Atlantic bioregion occurs on the east coast of Canada and the United States, from Labrador through to most of North Carolina. The Temperate North Pacific bioregion extends from north of the Seward Peninsula in Alaska south to below the Gulf of California. The reason this temperate zone extends so far south is that the cold upwelling waters of the Pacific Ocean mean that all the seagrass species within this bioregion are temperate. The Tropical America bioregion extends from the northern Florida coastline along the Gulf of Mexico to the Caribbean coast of Mexico. Also included in the Tropical America bioregion are seagrass locations on the Pacific coast of Mexico as far north as southern Baja, all of which contain three tropical seagrass species of Caribbean origin.

The seagrass distribution map (Figure 1), with the most up-to-date seagrass distribution data as of (January 2015) comprises three different measures of seagrass occurrence: first, actual measured areas of seagrass, referred to as *polygons*, typically mapped using remote sensing (photographs or satellites); second, lengths of surveyed shoreline, referred to as *lines*, using information that is obtained by detailed, low-elevation helicopter surveys along a length of coastline, using video to document the occurrence of seagrass; and third, individual observations of the occurrence of seagrass, called *points*. Polygons, lines and points represent rather different kinds of information. Polygon data are the only quantitative spatial data and are what is needed for accurate blue carbon assessments. Polygon data provide a direct measure of spatial extent, information that is needed to determine carbon storage capacity. Line data have been collected only along the west coast of North America in Oregon, Washington, British Columbia and Alaska. Because of the enormous distances, numerous shoreline details, and poor weather, which make it difficult to conduct remotely sensed surveys in these locations of deep seagrass, helicopter line surveys were developed and funded. Line surveys provide the linear extent of seagrass along a stretch of shoreline, but not the area. Point data indicate locations where seagrasses have been clearly documented as occurring, but no quantitative measures are included, merely presence of seagrass at a given point. In the seagrass distribution map, these three kinds of information are displayed using separate colors. Where polygon data are available, they replace line or point data and are displayed preferentially.

Seagrass area information is based solely on polygon data, since neither line nor point data give areal information (Table 1). For North America, the total seagrass area mapped and incorporated into the database is 24,190 square kilometers (km²), but much remains to be mapped. The largest area currently mapped is in the United States, and accounts for 59% of documented seagrass area in North America. The breakdown by bioregion and species is found in Table 2. In the United States, there is a lack of seagrass polygon information for the west coast and parts of the Gulf of Mexico, although most seagrass areas in the country have been identified with point and line data and only remain to be

mapped. Neither tropical nor temperate seagrasses have been found in the mid-Atlantic coast of the United States between the southern coast of North Carolina and northeast Florida. Alaska has extensive seagrass distribution which is fairly well documented, with line data for 23,233 km of shoreline with seagrass. For Mexico, there is 9,193 km² of mapped seagrass, with some point data on both coasts but also a large extent of shoreline unsurveyed by any means.

For Canada, there are polygon data for southern British Columbia. In northern British Columbia, there is nearly complete line data, providing shoreline length—but not area—of seagrass beds of almost 10,000 km. On Canada's east coast and in Hudson Bay, there is a fair amount of seagrass areal data available, but they are not included here due to a lack of time. Additionally, the Cree Nation is interested in supporting new mapping of seagrass in James Bay and southern Hudson Bay. At this time, for all of Canada, there is only areal information for 645 km² (Table 1). There are many point observations of seagrass occurrence in the Maritimes; for example, in the southern Gulf of St. Lawrence, as well as in locations that appear not to have been surveyed for seagrasses (Figure 1) but likely have extensive seagrass areas. Newfoundland has minimal seagrass observations in the database; polygon data exist which were not obtained in time for inclusion.

Figure 1. Seagrass distribution in North America, 2015



Table 1. Seagrass information summary for North America, 2015

Location	No. of Polygons, (% of Total)	Mapped Area (km²), (% of Total)	No. of Lines, (% of Total)	Length (km), (% of Total)	No. of Points, (% of Total)
North America	740,515	24,190	99,940	32,587	665
USA	713,627 (96%)	14,352 (59%)	82,473 (83%)	23,233 (71%)	112 (17%)
Mexico	540 (2%)	9,193 (38%)	0	0	54 (8%)
Canada	26,348 (4%)	645 (3%)	17,467 (17%)	9,354 (29%)	499 (75%)

Table 2. North American seagrass distribution by bioregion, country and species, 2007

Bioregion	Location	Quantity	Coverage Type	Seagrass Species ¹
		No. of Polygons	Area (km ²)	
Temperate North Atlantic	North America	19,811	1094	Zm, Hw, Rm
	USA	18,982	1092	Zm, Hw
	Canada	829	2.0	Zm
Temperate North Pacific	North America	603,582	2195	Zm, Ps, Pr
	USA	578,063	1553	Zm, Ps, Pr, Pt
	Canada	25,519	643	Zm, Ps, Pr
Tropical America	North America	117,122	20,900	Tt, Sf, Hd, Hw
	USA	116,582	11,707	Tt, Sf, Hd, Hw
	Mexico	540	9193	Tt, Sf, Hd, Hw
		No. of Lines	Length (km)	
Temperate North Pacific	North America	99,940	32,587	Zm, Ps, Pr
	USA	82,473	23,233	Zm, Ps, Pr
	Canada	17,467	9,354	Zm, Ps, Pr
		No. of Points	No. of Sites	
Temperate North America	Canada	487	487	Zm
Temperate North Pacific	North America	121	121	Zm, Ps, Pr, Pt, Hw
	USA	88	88	Zm, Ps, Pr
	Mexico	21	21	Zm, Hw, Pt, Ps
	Canada	12	12	Zm, Ps, Pr
Tropical America	North America	57	57	Tt, Sf, Hd, Hw
	USA	24	24	Tt, Sf, Hw
	Mexico	33	33	Tt, Sf, Hd, Hw

¹Zm = *Zostera marina*, Hw = *Halodule wrightii*, Rm = *Ruppia maritima*, Pr = *Phyllospadix serrulatus*, Ps = *Phyllospadix scouleri*, Pt = *Phyllospadix torreyi*, Tt = *Thalassia testudinum*, Sf = *Syringodium filiforme*, Hd = *Halophila decipiens*.

km² = square kilometer(s).

Photo 1. Seagrass (*Zostera marina*) habitat in Portsmouth Harbor, New Hampshire



Photo credit: D. Porter.

1.2 Threats to the Distribution of Seagrasses

The primary factor limiting seagrass distribution and biomass on the North American continent is the amount of light reaching the seafloor. As anthropogenic activities increase on land and in water, the clarity of coastal waters decreases, reducing the amount of light reaching seagrasses and posing a threat to these populations (Orth et al. 2006b; Grech et al. 2012). Another threat is temperature, which affects seagrass metabolic rates; the ideal growing temperature varies between species. Three predominant and widespread threats to seagrass are: 1) turbidity level in the water, 2) phytoplankton concentration in the water, and 3) seawater temperature. Both elevated turbidity and phytoplankton have a negative impact on the amount of light reaching seagrass growing on the seafloor, while temperature affects seagrass growth and species distribution.

Other threats to seagrasses resulting from human activities include dredge and fill, aquaculture, shoreline hardening, and nutrient loading. Considering the whole of North America, the two major threats that can be detected from satellite imagery in a broad-scale analysis are: turbidity (measured as total suspended solids) and primary production (measured as chlorophyll in the water column), driven in part by nutrient loading (Halpern et al. 2008). These threats are subject to anthropogenic enhancements which exacerbate the natural background conditions that affect seagrass. An additional factor that is important to consider is temperature: temperate and tropical seagrasses have different temperature ranges, ranges that limit their distribution in North America.

1.3 Seagrass Stressors

The main stressor of seagrasses in North America is nitrogen over-enrichment; this results in proliferation of phytoplankton blooms, which decrease water clarity and prevent light from reaching seagrass beds on the sea floor. Dense phytoplankton blooms in ocean waters derive primarily from high nitrogen concentrations, which fuel phytoplankton primary production and eutrophication (McGlathery et al. 2007). In near-shore areas, excess nutrients are derived from many human sources and promote the growth of competitive algae, both seaweeds and phytoplankton. Nitrogen-loading from wastewater treatment facilities, watershed runoff, agriculture, and loss of natural buffers results in excess nutrients, which in turn encourages phytoplankton and seaweed blooms, shading seagrasses and causing seagrass declines (Short and Wyllie-Echeverria 1996).

Climate change is also adversely affecting seagrass distribution and health, via bleaching, storms, temperature stress, and species migration (Short and Neckles 1999). Sea-level rise will have some effect on seagrasses, although the rate of seagrass vertical and horizontal expansion is greater than that of sea-level rise, so that, at most, sea-level rise will result in relocation of seagrass meadows. Seagrass response to increased UV-B is reddening of the leaves, which slows plant growth overall and will therefore reduce carbon production and storage (Novak and Short 2011). Other stressors are summarized in Table 3.

Table 3. Seagrass stressors resulting from human activities

Stressor	Cause
Reduced Water Clarity (Increased Turbidity)	Nutrient-loading
	Sediment-loading
	Contaminants
	Cumulative impacts
	Aquaculture structures and practices
Physical Damage	Dredge and fill
	Overwater structures
	Fishing practices
	Moorings and propellers
	Wakes from ships, ferries and boats
	Hardened shoreline
	Climate change (storms, tides, erosion, UV-B)

1.4 Data Gaps for Seagrass Maps

Seagrasses represent the greatest data gap in blue carbon habitat map data overall, and their spatial extent is largely underrepresented in the data from Canada, Mexico, the United States. The underrepresentation is largely due to the challenge involved in mapping seagrass habitat, which is generally submerged, while marsh and mangrove habitats are covered by emergent plants that are visible even at high tide. Both Mexico and Canada have large extents of seagrass unmapped as to area; additionally, in the United States, Alaska does not have area data for most of its coast. To complete the challenge of seagrass mapping of North America, data gaps must be filled, and some adaptive methods developed. Seagrass can be mapped by several methods in shallow or intertidal waters, but it is much more difficult to document how far seagrass extends into deeper water. Many sources indicate presence of seagrass, and not the areal extent.

About half the coastline of North America remains to be surveyed to complete our knowledge of seagrass areal distribution (Figure 1). Some polygon (areal) data from Canada as well as the northern Gulf of Mexico exist presently but were not available, and could be added to improve coverage. Additional mapping remains to be done in parts of the Canadian Maritimes, the Gulf of California, the west coast of the United States, and the Pacific coast of Mexico, where point data are available documenting presence of seagrass but no areal mapping has been done. The challenge also remains to develop a new methodology and make effective use of the line data from British Columbia, Oregon and Alaska for areal seagrass blue carbon estimates. In the case of Oregon, the data were only recently collected (2014) and are not yet available.

Besides additional mapping, there exists seagrass area data which are not yet in the database. Clearly, these should be obtained and entered in the database and maps. The data exist in several categories of availability and readiness:

- Data already available but not yet in this database exist from: Eastern Canada (New Brunswick, Newfoundland, Quebec) and United States (Texas, Louisiana, California).

- Data that could be made available with appropriate permissions exist from: Canada (eastern James Bay, parts of Nova Scotia) and United States (California).
- Seagrass distribution data for which the sources of the areal information have not been identified (maps have been seen, but not the data that are the basis of the maps) exist from: Canada (Gulf of St. Lawrence) and Mexico (some Baja Peninsula estuaries, western Gulf of Mexico).

1.5 Seagrass Mapping Priorities and Recommendations

Prioritized seagrass mapping is depicted in Figures 2 (A, B and C) and 3 (A, B and C). Surveying seagrasses to create new polygon maps is needed in many locations across the continent, as indicated by the seagrass point data, which show where seagrasses have been observed but not mapped in any way for areal extent. The most cost-effective way to expand knowledge of seagrass blue carbon is to first utilize the extensive aerial mapping of coastline distribution—the line data—from the Pacific Northwest (Figure 2, A). These line data represent a huge untapped knowledge bank of seagrass areal blue carbon information. Additionally, some areal maps of seagrass in all three countries were discovered toward the conclusion of this project and it is a priority to acquire these polygon data and add them to the North American seagrass map.

Beyond the conversion of line data to areal (polygon) data in the United States and Canada, new surveys and mapping are a priority for the Gulf of California, Mexico (Figure 2, C), where a few polygon maps exist but where many observations document much more seagrass that is not yet accounted for. In Canada, Prince Edward Island has extensive seagrass resources which are highly threatened by agricultural runoff. These are a priority to be captured in areal mapping since, presently, only point data of observations exist (Figure 2, B). The coast of California, USA, also has abundant point data but only some areal surveys. Areal maps of the California seagrass resource have focused on the major estuaries but many other seagrass areas exist both along the open coastline and offshore around islands (Figure 3, A).

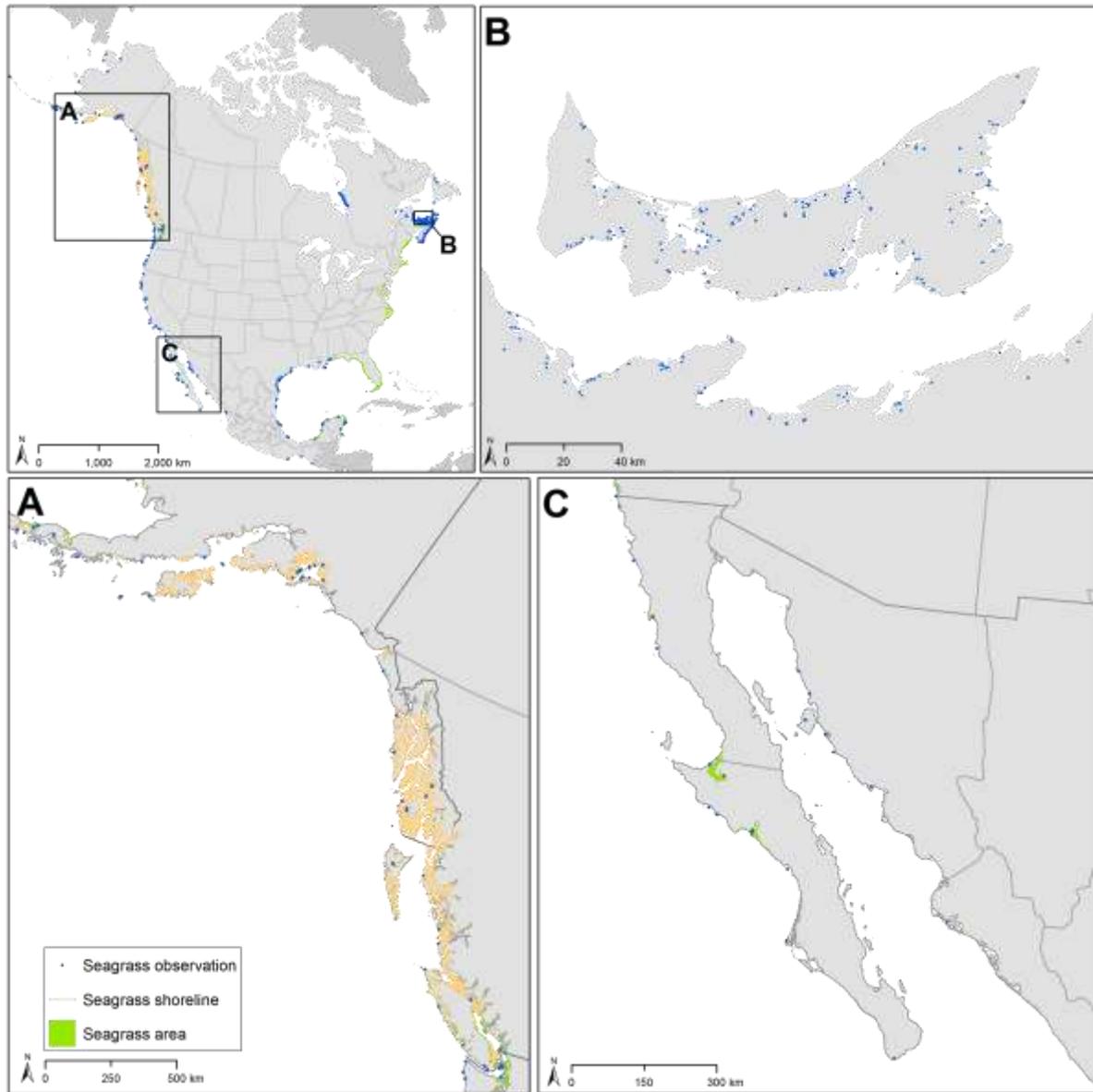
Additional priorities for seagrass surveys leading to areal seagrass blue carbon mapping occur along the east coast of the southeast Yucatan Peninsula (Mexico) and in Hudson and James bays in Canada. The Yucatan, from Sian Ka'an southward as well as around Cozumel Island, has only point data for seagrasses in the database (Figure 3, C). In Quebec and Ontario, the shores of Hudson Bay and James Bay (Figure 3, B) are reported to have extensive seagrass meadows for which only point data are available and these areas are a priority for the Cree nation, which is interested in jointly supporting new mapping of eelgrass in James Bay and southern Hudson Bay.

There are additional priorities to bear in mind in order to increase the quality and cost-effectiveness of the seagrass blue carbon database. First, a set of mapping guidelines is important, to identify, document and prioritize seagrass areas not yet mapped or only having anecdotal documentation of the seagrass blue carbon resource. A second priority is the formulation of a low-cost mapping plan, a “directed citizen mapping” effort that will use available imagery and trained volunteers for ground truth assessment.

In summary, mapping priorities for seagrasses are as follows:

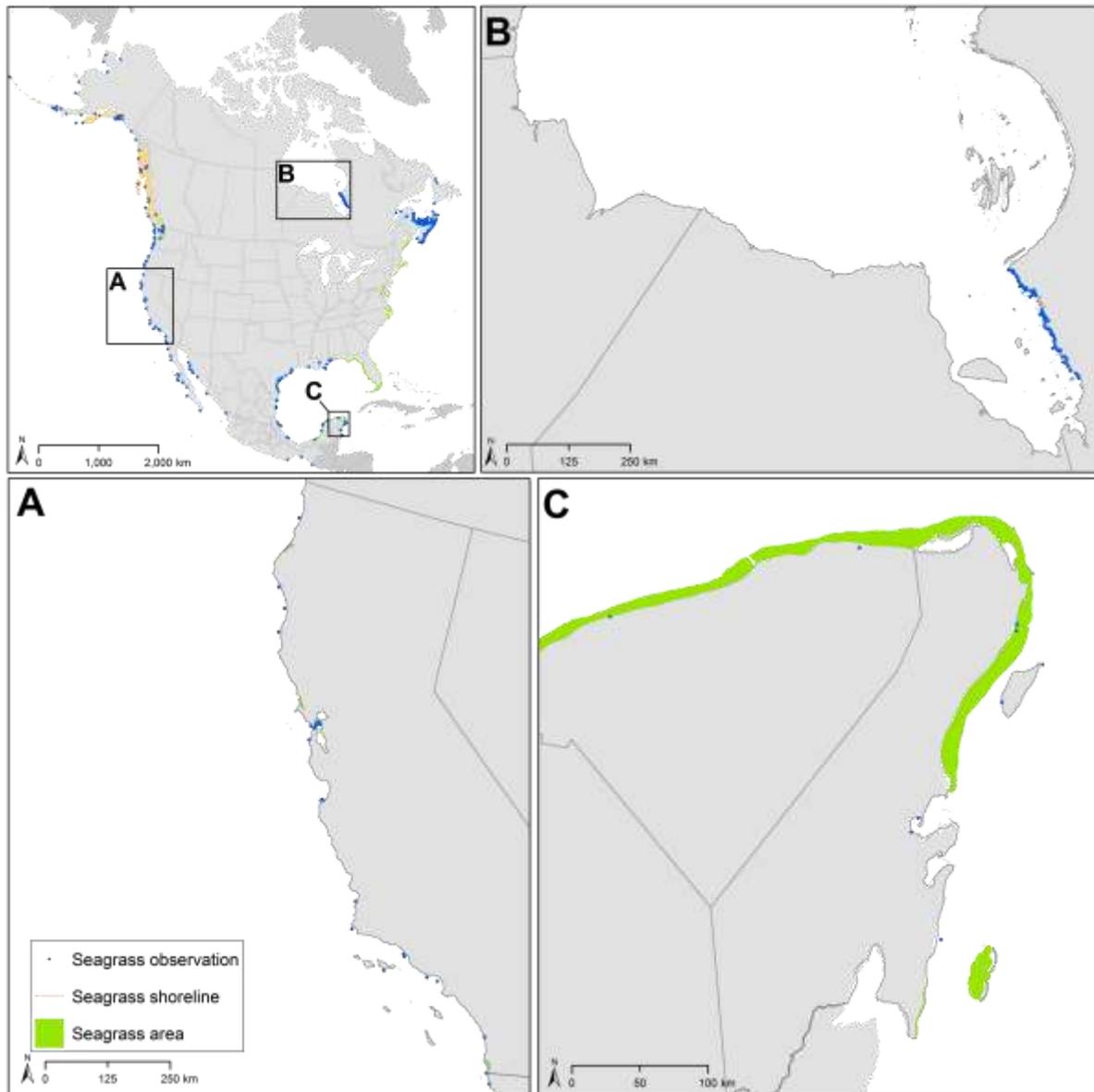
1. Create digital polygon maps for the blue carbon database from existing line data in (Figure 2, A) for:
 - i. British Columbia (Canada), and
 - ii. Alaska and Oregon (USA).
2. Obtain, verify and add already-existing digital polygon maps to the database from:
 - i. Newfoundland, New Brunswick, Quebec, parts of Nova Scotia (Canada);
 - ii. Texas, Louisiana, California (USA); and
 - iii. Sonora, Baja California Sur, Sinaloa (Mexico).
3. Survey sites where point data have already documented seagrass occurrence and then create digital polygon maps (Figures 2 and 3) for these sites in:
 - i. Prince Edward Island (Canada),
 - ii. Gulf of California (Sonora and Baja California Sur, Mexico),
 - iii. Hudson Bay and James Bay (Ontario and Quebec, Canada),
 - iv. California Coast between Eureka and Santa Barbara (California, USA), and
 - v. Yucatan Peninsula, from Sian Ka'an south (Mexico).
4. Develop mapping guidelines adapted for each country and for each bioregion.
5. Formulate a low-cost mapping plan that uses existing imagery and citizen verification.

Figure 2. High-priority areas in need of seagrass mapping for blue carbon resources



Note: A: Pacific Northwest. B: Prince Edward Island, Canada. C: Gulf of California, Mexico. Uncolored areas of shoreline are largely unsurveyed.

Figure 3. Second-tier priority areas in need of seagrass mapping for blue carbon resources



Note: A: California, USA. B: Hudson Bay and James Bay, Canada. C: Yucatan Peninsula, Mexico. Uncolored areas of shoreline are largely unsurveyed.

1.6 Seagrass Carbon Stocks and Rates

Seagrasses sequester carbon from seawater through plant uptake for photosynthesis, and incorporate carbon into leaf, root and rhizome tissues, as well as in flowering parts. Much of the leaf material remains in the seagrass meadow as the leaves age; roots and rhizomes remain buried in the sediment (Duarte et al. 2005). All seagrass tissues and a substantial amount of other organic matter filtered by seagrasses become incorporated and stored in place in seagrass meadows. All these organic inputs to seagrass meadows, minus whatever is decomposed, become the “blue” carbon stored in the sediment.

Seagrass carbon, both as leaves and roots, partially decomposes in sediments, enhancing the amount of organic carbon in the soils. In addition, seagrass beds accumulate organic carbon from surrounding environments as part of their water-filtering process (Short and Short 1984) and this filtered material amounts to about 40% of seagrass carbon storage. Large, dense seagrass beds can sequester and store large amounts of carbon but both sequestration and storage vary widely by seagrass species, as well as by density of the meadow. Seagrasses grow rapidly under good conditions, providing major carbon storage in relatively short time-frames.

Seagrasses form the basis of both a grazer food web and a detrital food web, contributing to the trophic cascade of coastal ecosystems. Consumption of seagrasses, their epiphytes, and nearby phytoplankton by grazers, and subsequent predation of grazers contribute to the complex carbon capture, storage and sequestration function in seagrasses (Lutz and Martin 2014). Seagrass biomass turns over in time-frames of from weeks to years for most species while, for some species, roots and rhizomes can persist for millennia, so that the role of seagrasses in the oceanic carbon budget is more significant than expected from their area.

Seagrasses are estimated to be responsible for 15% of total carbon storage in the ocean (Kennedy and Bjork 2009), although seagrasses occupy only 0.2% of the area of the oceans. They provide long-term carbon burial of 83 grams of carbon per square meter per year ($\text{g C m}^{-2} \text{yr}^{-1}$), which translates into global storage rates of between 27 and 40 teragrams of carbon per year (Tg C yr^{-1}), or 27 to 40 million metric tons of carbon (based on a global seagrass area of 300,000 km^2) (Nellemann et al. 2009; Fourqurean et al. 2012). Median soil carbon storage in seagrass meadows is 140 megagrams organic carbon per hectare ($\text{Mg C}_{\text{org}} \text{ha}^{-1}$) (Fourqurean et al. 2012).

In other terms, seagrass meadows annually store up to 83 metric tons of carbon per square kilometer—almost three times the amount stored by terrestrial forests—predominantly in sub-surface sediments. The below-ground roots and rhizomes are not grazed to any great extent and decompose more slowly than seagrass leaves. Root and rhizome tissue of seagrass can amount to 15–50% of primary production. The morphology and biomass of seagrass species vary widely, making the carbon storage in below-ground deposits highly variable.

The most comprehensive discussion of blue carbon to date, including seagrass blue carbon, can be found in Nellemann et al. (2009). Technical detail on rates of seagrass carbon sequestration, with somewhat more recent information, is found in Fourqurean et al. (2012). Pendleton et al. (2012) discuss the economics of blue carbon but warn that the science of blue carbon rates “...has high uncertainty. New research is needed to improve our estimates of how much carbon is trapped in these ecosystems, how much carbon is released into the atmosphere by their conversion, and where on the planet carbon loss is occurring most rapidly.”

North American seagrass blue carbon is clearly a critical resource experiencing many impacts and undergoing loss at a significant rate. Seagrass losses mean blue carbon is being depleted from sediment storage and the sequestration rate is reduced. Unfortunately, the data available for assessing blue carbon sequestration and storage are based on global estimates, not the suite of seagrass species in North America. Beyond this, the lack of an accurate areal estimate for seagrasses themselves across the continent further inhibits the possibility at present of a clear estimate of North American seagrass

blue carbon stocks and sinks. For better assessments of carbon stocks, mapping of seagrass by species is desirable.

1.7 Seagrass Carbon Stock Conservation

Temperate seagrass species recover and expand rapidly if source populations are available to provide seeds (Neckles et al 2005; Orth et al. 2006a; McGlathery et al. 2007). Tropical dominant climax species recover relatively slowly, but pioneer species develop rapidly and can provide initial revegetation of the habitat and begin restoration of blue carbon. In the past century, along the developed east coast of the United States, it is estimated that up to 50% of all seagrass habitat has been lost (Short and Short 2003) and the prospect for recovery in most of this area is low due to loss of water clarity, and severe coastal alterations and heavy use. The vast meadows of seagrass that once existed in the Canadian Maritimes supported an industry involving the collection of seagrass wrack from the beach, which was dried and used for a commercial home insulation product (the “Cabot’s Quilt”) (Cabot 1986). The occurrence during the 1930s of eelgrass wasting disease, which was responsible for a loss of 90% of eelgrass on both sides of the Atlantic, knocked back the habitat and it has never fully recovered (Milne and Milne 1951; Short and Short 2003), in part due to degraded near-shore conditions. There are places like Waquoit Bay, Massachusetts, where all the eelgrass has been lost due to eutrophication; despite decades of management, the conditions are still not adequate for seagrass recovery.

There is a changing baseline: As memory of the presence of seagrass is lost, and there is a general lack of awareness of its importance and distribution, the impetus to protect and restore the habitat is lost as well.

Seagrasses benefit tremendously from careful management and protection, and can be restored if water clarity and other conditions for their successful growth and sustainability are met. There is presently no seagrass that is entirely protected—that is, seagrass habitat that is legally protected from ecosystem degradation from water-borne stressors or other impacts. Even marine protected areas (MPAs), which are typically created for protecting the species within them, do not adequately prevent impacts to seagrasses and often are losing seagrass at rates similar to those in areas outside the MPAs.

Priority areas for seagrass habitat protection are designated based on their blue carbon stocks, on the potential for sustainability of the seagrasses if they receive protection, and on threats that could be reduced by protection. Some of these priority areas for protection are the following:

1. In Canada, the estuaries of Prince Edward Island have large seagrass beds that are very productive, are currently being degraded by agricultural run-off and aquaculture, and need protection. On the Pacific side of Canada, the coastal waters around Vancouver Island in southern British Columbia need protection.
2. In the United States, in the areas of New England where water clarity is being improved due to pollution reduction, the remaining seagrass needs protection so that it can rebound and thrive. On the west coast of Florida, areas where seagrasses are responding to management need protection for these trends to continue. The Big Bend area of Florida is highly threatened but supports extensive seagrass beds which also need protection.
3. In Mexico, the temperate Gulf of California has extensive seagrass meadows, many of which have not been mapped, but they are under threat from agricultural run-off and lack of management. They need protection. In the Yucatan and the coast of the Gulf of Mexico, extensive seagrass habitat, mostly climax species with large blue-carbon capacity, is being lost rapidly and is in dire need of protection.

Seagrass restoration is expensive and difficult. It should only be done in areas with sufficient water clarity that also meet other criteria of appropriate sediment, temperature, salinity and wave/current conditions. Candidate areas for seagrass restoration include the following:

1. In Canada, the Atlantic coast of Nova Scotia is an area where seagrass restoration is possible and would make a difference. In southern British Columbia, where land management is reducing threats to water quality, restoration would aid seagrass recovery.
2. In the United States, mid-coastal Maine and Morro Bay in central California are both areas with adequate water clarity for seagrass restoration if the localized threats are successfully managed.
3. In Mexico, increased management of local impacts would create better possibilities for seagrass restoration success.

In all cases, increased management at municipal, state, provincial, and federal levels is needed to conserve existing seagrass resources and to make any restoration activities more likely to achieve success for blue carbon storage. Retaining current stocks of seagrass blue carbon and encouraging the rebound of affected seagrass areas is much cheaper and has a higher likelihood of success than restoration of individual seagrass beds or meadows. Management first must focus on water clarity. All the impacts on water clarity from land-based activities ultimately affect seagrass habitat: deforestation, agriculture, impervious surfaces, waste disposal, vehicle emissions, power generation, non-point nutrient and sediment inputs, etc. Clearly, the challenge is huge, but concerted management efforts with citizen support can turn around a degrading marine coastal system, bring back the seagrass resource, and retain and expand the blue carbon in coastal waters.

2. Tidal Salt Marshes

Tidal salt marshes are found from the Arctic to the tropics, on marine coasts in areas protected from the direct actions of the sea—behind barrier islands, in sheltered bays and in river estuaries. On most of North America’s coastline they are vegetated by herbaceous plants, primarily grasses, sedges and rushes. Situated in the intertidal zone, emergent marsh vegetation at the lowest elevations is subject to tidal flooding once or twice a day, but is irregularly flooded at higher elevations—where the greatest salt stress can occur. Few species have evolved adaptations which allow them to tolerate both flooded and saline soils, so salt marsh plant diversity is low. Lack of competition allows for exceptional levels of productivity by the dominant species. Marshes provide a variety of ecosystem services: through provision of food and habitat for fish and bird populations, as sinks for pollutants, and as a first line of defense against the destructive force of storms. In the last decade, their soils have been recognized as exceptional carbon sinks, which help to mitigate climate change.

Photo 2. Salt marsh at high tide in Antigonish County, Nova Scotia, Canada



Photo credit: G. Chmura.

Tidal salt marshes are found on marine coasts in areas protected from the direct actions of the sea—behind barrier islands, in sheltered bays and in river estuaries. Bordering the coastal, lower edge of the marsh may be an intertidal mudflat devoid of vascular plants, or a seagrass bed grading into the continuously flooded sub-tidal zone, which also may be vegetated with seagrass. At the lowest elevations, emergent marsh vegetation is subject to tidal flooding once or twice a day, but in some regions, such as the Bay of Fundy, Canada, the upper elevations may be flooded by tidal waters only a few times in a year (Byers and Chmura 2007). Ponds within salt marshes may support small populations of submerged aquatic plants, such as the seagrass *Ruppia maritima*, which occurs on the northeast Atlantic coast.

Marsh plants are distinct from seagrasses in that their photosynthesis and respiration generally require exposure to the atmosphere. Submergence by seawater, and soils that are saturated with salt water stress the plants' physiology. Flooded soils place limitations on nutrient uptake and can result in accumulation of phytotoxins (Mitsch and Gosselink 2007). Species found in salt marshes have adaptations that help them to deal with the osmotic stress of saline soils by producing osmolytes, concentrating salts in organelles (a characteristic of some succulents), or excreting salt through special glands. Many have aerenchyma, a tissue that serves to channel gases between the atmosphere and soil, to oxygenate the root zone. Some species can even respire anaerobically, but the high energy cost associated with this limits the time plants can tolerate flooding. The number of plant species that have evolved adaptations which allow them to tolerate both flooded and saline soils is limited. Thus,

the vegetation of salt marshes is not diverse, and the number of species increases with decreasing salinity. Along much of the coast of North America, marshes are dominated by herbaceous vegetation (i.e., grasses, sedges, rushes and forbs), with a border of shrubs delineating the upper extent of the tides. At lower salinities, upstream in estuaries, salt marsh vegetation is replaced with other species which cannot tolerate the stress of saline soils but are better competitors. Transplant experiments have shown that when protected from competition, salt marsh vegetation grows better in freshwater, but species moved to higher salinities do not survive (e.g., Chapman 1960; Bertness 1991).

2.1 Tidal Salt Marsh Distribution

Many high-latitude marshes are situated on coastlines subject to isostatic rebound, where marshes may transition to freshwater or terrestrial habitats as they are lifted from the tidal frame (Pendea and Chmura 2012b). At the lowest latitude, marshes may be found interspersed with mangroves (McKee 2012).

Compilation of the distribution data available to date indicates that there are approximately 13,474 km² of tidal salt marsh in North America, a number which is an underestimate, for various reasons (Figure 4). For example, salt marshes are probably underestimated in Mexico because Conabio (the National Commission for Knowledge and Use of Biodiversity) has only recently begun a project to map them. The largest contribution is from the United States, which has about 12,775 km², much of which is in the Tropical Atlantic bioregion. Louisiana has twice as much area as any other state. The area of tidal marsh blue carbon habitat tallied for Canada is considerably lower, around 546 km², but this does not include parts of Hudson and James bays and parts of Quebec, such as the northern shore of the St. Lawrence.

In salt marshes, the pattern of species dominance is more complicated than among seagrasses or in zones in marine ecoregions. From the St. Lawrence River estuary south to Chesapeake Bay, *Spartina alterniflora* dominates at lower elevations and *Spartina patens* at higher elevations. The relative abundance of forbs, such as *Plantago maritima* and *Triglochin maritima*, and other graminoids increases with latitude (Chmura et al. 1997; Wigand et al. 2014; Desroches et al. 2013), and forbs can be dominant in high-latitude marshes such as those on the coast of James Bay (Pendea and Chmura 2012a). With decreasing latitude, from Chesapeake Bay south, *Juncus roemerianus* increases in importance (Baldwin et al. 2012), and along the South Atlantic coast of the US it dominates high marsh elevations (Pennings 2012). The high evapotranspiration rates at lower latitudes result in high soil porewater salinity in the middle marsh, a zone that is above the reach of regular tidal flooding and distant from freshwater sources inland. The higher soil salinity results in an unvegetated salt pan between low and high marsh.

Along the northeastern Gulf of Mexico coast the dominant marsh species is *J. roemerianus*, but *S. alterniflora* and *S. patens* are also found in restricted areas (Battaglia et al. 2012). The western side of the northern Gulf of Mexico is subject to increasing aridity, and when freshwater availability is restricted the succulents *Salicornia bigelovii* and *Batis maritima* will be the dominant species. Situated between the eastern and western marshes of northern Gulf of Mexico is the Mississippi Delta. The Mississippi River outflow reduces salinity of coastal waters here, but many of the same species found in more-saline marshes occur here. *Spartina alterniflora* is found in the most “seaward” marshes of the delta plain. The zone inland to this contains a mixture of *S. alterniflora* with *S. patens* and *Distichlis spicata* (Visser et al. 2013).

In the sub-tropics and tropics, tidal marshes may occur landward of mangroves, in patches where the tree canopy is open, or as a narrow fringe seaward of the mangrove stand (McKee 2012). Where freshwater input is low and evapotranspiration is high, soils are hypersaline and the vegetation is dominated by succulent forbs such as *Batis maritima*, *Sesuvium portulacastrum* and *Salicornia*

virginica, and grasses such as *D. spicata*, *Sporobolus virginicus*, *S. alterniflora* and *Spartina spartinae* (McKee 2012). As salinity decreases, the vegetation includes more sedges and rushes.

Differences in historical biogeography result in a largely different species pool available to populate salt marshes of North America's Pacific coast—in fact, *S. alterniflora* is considered an invasive species here. Marshes of the Pacific coast geographically have been classed into five groups by Macdonald (1977): Arctic (71–60°N), Subarctic (60–54°N), temperate (54–35°N), dry Mediterranean (35–32°N), and arid (35–32°N). On the Arctic coast of Alaska, sea ice disrupts vegetation, and marsh communities are mosaics of *Puccinellia* and *Carex* species. Diversity increases to the south in the subarctic marshes, where species of *Glaux*, *Sarcocornia*, *Spergularia* and *Suaeda* are found and important salt marsh components are *Carex lyngbyei*, *Deschampsia caepitosa*, *Plantago maritima*, *Potentilla pacifica* and *Triglochin maritima*. The temperate group loses some Arctic species but gains other species. Warmer and drier conditions in the summer means that vegetation must withstand high soil salinities during the growing season, thus high marsh plains in California are dominated by the very salt-tolerant succulent, *Sarcocornia pacifica*, while *Spartina foliosa* tends to be dominant in marshes at lower elevations. Other species may be present in marshes or on creekbanks where freshwater inflow reduces the salinity: *D. spicata*, *Schoenoplectus americanus*, *Schoenoplectus californicus*, *Bolboschoenus maritimus*, *Jaumea carnosa*, *Frankenia salina*, and *Limonium californicum*, among others (Callaway et al. 2012). Diversity is greatest in the dry Mediterranean salt marshes, but is sharply lowered in the arid group situated in Baja California.

In many areas, assessment of the quality of maps can be made by overlaying polygons on Google Earth imagery. This exercise has been performed for some of the coast in New Brunswick, Canada, and reveals that some marshes and parts of some marshes are missing. This may be the case within other jurisdictions but a complete review of individual polygons was beyond the scope of this project.

Figure 4. Salt marsh distribution in North America, 2015



2.2 Threats to the Distribution of Tidal Salt Marshes

Threats to the distribution of tidal salt marshes include:

- land development;
- lack of suspended sediment (low turbidity);
- excess nutrient; and
- coastal squeeze.

The cumulative threat to marshes on Canada's Atlantic and Pacific coasts is low, with sediment supply being the primary limitation on productivity. However, sediment supply is not the most significant problem in areas where large proportions of marsh have been lost to dyking. Consideration of increased costs of dike management in the face of rising sea level, and consideration of potential economic gains by selling credits on the carbon market may be effective incentives for future restoration of these blue carbon sinks. This will require research on the rates of return of carbon stocks.

For the United States, when looking at the total area threatened, nutrient levels in coastal waters stand out as a threat on the US portion of the Tropical Atlantic bioregion, south from North Carolina and west to the states bordering the Gulf of Mexico. Overall, marshes in this region are less threatened by surrounding land use, low suspended sediment supply (low turbidity), and coastal squeeze. Reduction of nutrient levels could make a substantial difference in the sustainability of the marshes there. Reduction of nutrient levels in coastal waters will require changes in agricultural management by actors who may be located quite a distance from the coastal wetlands and so not perceived as stakeholders. To reduce nutrient loading will require reduction in fertilizer use, or improved management of agricultural lands. Because these actions would help protect blue carbon sinks, there may be opportunities to give carbon credits to farmers who reduce the escape of nutrients from their fields.

Marshes of the urbanized coasts of New Hampshire, Massachusetts, Rhode Island and Connecticut are exposed to high threat levels associated with urbanization, as well as with lack of sediments. Two factors may contribute to the latter. Many of the region's rivers are extensively dammed, which results in limited delivery of sediments to the coast (Weston 2014). Dams have been removed from some sites and there are plans for more removals, but since the objective is often to restore fish populations, an alternative to removal is the construction or improvement of fish ladders—which will not prevent sediment from being trapped behind dams. Noting the possible impact of dam removal on sediment supply and sustainability of carbon sinks might help shift cost-benefit ratios to the side of dam removal rather than construction of fish ladders.

Threats in Mexico have not been well documented to date.

2.3 Data Gaps for Tidal Salt Marsh Maps

The area of salt marsh in Canada (with the exception of Newfoundland) and the United States is fairly well known, but there is little information on the area of salt marsh in Mexico. In Canada, local gaps have been identified that could be filled with additional mapping effort; across jurisdictions, mapping of salt marshes has not been consistent.

2.4 Tidal Salt Marsh Carbon Stocks and Rates

Although salt marshes may trap particulate organic matter transported in tidal floodwaters, it is generally recognized that the primary source of the carbon they store is through growth of the marsh vegetation (autochthonous carbon). The vascular plants of marshes uptake carbon dioxide from the atmosphere and, through photosynthesis, transform it into the organic matter. Although plants release some carbon dioxide through respiration, a growing plant will sequester increasing amounts of carbon in its tissues. Other photosynthesizers, such as benthic microflora (growing as a biofilm on the soil surface) and macroalgae (growing amongst grass stems), are common in salt marshes and contribute carbon to the soil, but their contribution to the salt marsh carbon sink has not been addressed.

Existence of a salt marsh is dependent upon soil carbon accumulation. Carbon accumulates in both the above-ground (leaves and stems) and below-ground (roots and rhizomes) tissues of salt marsh plants. A major reason why salt marshes are such effective carbon sinks is that they translocate large

amounts of carbon to below-ground tissue where it becomes part of the soil. In fact, rates of below-ground biomass production can surpass above-ground biomass production by a factor of more than four (Roman and Daiber 1984). While above-ground biomass may be consumed by herbivores or exported into estuarine waters, the biomass in the soil is generally protected from such losses. Above-ground production that is not consumed, exported, or decomposed will become buried in the marsh soil, but little research has addressed how much it contributes to the carbon sink. What is more widely recognized is the contribution of above-ground vegetation to the trapping of sediments from tidal flooding—a process that contributes to marsh elevation (e.g., Li and Yang 2009). Decomposition of soil carbon will result in carbon dioxide emissions from the soil or leaching of dissolved organic carbon into tidal water (e.g., Tzortziou et al. 2011). However, decomposition rates in the saturated soil are considerably slower than the aerobic decomposition that occurs above ground. Despite these two losses, there generally is net soil carbon storage, an increase in soil volume, and in turn, soil elevation. Over approximately the last 10,000 years, sea level has been rising on most coastlines. And for the last few thousand years, salt marsh soils have been accumulating and saturating; the surface of the marsh remains within the tidal frame, is subject to regular tidal flooding, and develops a growing carbon stock.

Since carbon storage is dependent upon plant growth, it is logical to assume that factors that stress plant growth will result in decreased carbon storage. Loss of vegetation will result in loss of carbon storage, loss of soil volume, and decreased soil elevation to a level below which marsh vegetation cannot survive the extended periods of tidal flooding that will occur. Minor stress, however, can shift the allocation of plant carbon stores preferentially to below-ground growth. For example, saline soil water presents a physiological stress, and in some species increased salinity causes a greater nitrogen demand. This greater demand can drive higher root production so as to obtain the limiting nutrient. Excessive nutrient supply can have the reverse effect, but field experiments have shown variable results. In a field experiment on the coast of Massachusetts fertilization caused a decrease in below-ground biomass (Deegan et al. 2012), but another fertilization experiment on the coast of Connecticut resulted in no change in below-ground biomass (Anisfeld and Hill 2012).

Rates of carbon storage in a salt marsh depend upon a balance between below-ground carbon inputs and decomposition. Increased decomposition rates of marsh soil organic matter, found with increasing temperatures, indicate that soil carbon stocks in low-latitude marshes could decrease with climate warming (Kirwan and Blum 2011). However, in higher latitude, higher temperatures may increase marsh plant productivity, countering the loss of carbon through increased decomposition (Kirwan et al. 2014).

Measurements of soil carbon density (% soil carbon times soil dry bulk density) and soil carbon accumulation rates vary widely, as revealed in the 2013 compilation by the International Panel on Climate Change (IPCC) (Kennedy et al. 2013). They can be as high in colder-climate, high-latitude marshes as in warmer-climate, low-latitude marshes (Chmura et al. 2003). Studies have yet to explain the variability in carbon density and accumulation rates, although Chmura et al. (2003) did find a relationship of soil carbon density to average annual air temperatures in eastern North American salt marshes. The variability may be due to lack of control with regard to marsh locations sampled amongst various independent studies. Salt marshes are spatially variable with respect to vegetation, soil elevation and soil flooding status, thus location of a site for sample selection could bias results—an issue yet to be addressed in an empirical study.

Since the IPCC publication in 2013, there have been few additional reports on salt marsh carbon storage in North America and they fall within the range shown in Table 4. Many estimates of marsh-wide carbon storage, such as the IPCC's (Kennedy et al. 2013), assume that marsh peat has a depth of one meter (m), yet few studies have examined carbon stocks beyond 50 centimeters (cm). Geomorphological studies of individual marshes reveal that marsh peat deposits vary considerably

within a marsh and can be as deep as 4 m, on the Chesapeake Bay in the United States (Tibert 2012) and 7 m, on Canada's Bay of Fundy (Shaw and Ceman 1999).

Table 4. Global average marsh soil carbon stocks and carbon storage rates

Soil Type	Stock (tonnes ha ⁻¹)	Range of Stock (tonnes ha ⁻¹)	n	Storage Rate (tonnes ha ⁻¹ yr ⁻¹)	Range of Storage Rates (tonnes ha ⁻¹ yr ⁻¹)	n
Organic	340	221–579	35			
Mineral	226	15.6–623	82			
All Types	255	15.6–623	117	0.91	0.05–4.65	66

Note: One-meter depth assumed for carbon stocks.

m = meter; n = samples taken; ha⁻¹ = per hectare; yr⁻¹ = per year.

Source: Adapted from Kennedy et al. 2013, Tables 4.11 and 4.12.

2.6 Tidal Salt Marsh Carbon Stock Conservation

Where sea level is rising (which is on most coastlines), maintenance of marsh peat volume, which controls soil elevation, is essential for marsh sustainability. On most coasts, mineral sediments also contribute to marsh soil volume. Yet, the existence of 2 m-thick deposits of organic marsh soil on the Rhode Island (USA) coast's Palmer River (Chmura 1982) indicates that marshes could survive past rates of sea level rise without receiving mineral deposition.

Morris et al. (1990) have shown that production of *S. alterniflora* varies with mean sea level. Sea level fluctuates yearly, as well as within an 18.6-year cycle, and marsh vegetation persists with this variability. If sea-level rise keeps below a threshold level, then marsh vegetation will continue to survive, accumulate carbon, and maintain marsh elevation. Loss of marsh in places such as the Mississippi Delta and Chesapeake Bay (e.g., Kirwan and Guntenspergen 2012) has been attributed to the inability in these places of marsh below-ground production to maintain marsh elevation with relative sea-level increases. This loss of elevation is due, in part, to high local rates of sea-level rise, but anthropogenic modifications have had significant impacts on plant production and survival. Hydrological modifications such as culverts or berms can impede drainage of tidal floodwaters, causing added stress to vegetation. Excessive nutrient-loading may result in reduced below-ground growth.

Over the past few centuries, marsh carbon stocks have shown to be surprisingly resilient in the face of direct human perturbations. Native marsh vegetation was harvested as hay along the east coast of North America, and relics of that activity are still evident in marshes—although there has yet to be a study directly assessing the impact of haying. A study of the impact of grazing, on soil carbon in a St. Lawrence River (Quebec, Canada) salt marsh, however, revealed that marsh carbon stocks in the surface soil were higher than in the ungrazed marsh (Yu and Chmura 2010).

It is likely that if sea-level rise reaches the apocalyptic rates predicted by many (e.g., Vermeer and Rahmstorf 2009), then marsh plant production, carbon accumulation, and marsh elevation, which depends on the carbon accumulation, will not keep pace, leaving unvegetated deposits subject to erosion. The presence of marsh peat submerged on the continental shelf off the east coast of North America (Emery et al. 1965) suggests that some of the submerged carbon stock will persist, but there has been no research directly addressing the fate of submerged marsh carbon stocks.

3. Mangroves

Photo 3. Mangrove in Marismas Nacionales, Nayarit, Mexico



Photo credit: P. Fajardo.

Mangrove swamps are found along marine and estuarine coasts in the tropics and subtropics, sometimes co-mixed with smaller areas of salt marsh. The term mangrove is applied to arboreal vegetation that grows in intertidal areas of this climate zone. Globally these are trees, shrubs, palms, and even ferns, such as the *Acrostichum aureum*, which can reach three meters in height and is found on the US coast (Spalding et al. 2010). The designation “mangrove” requires no phylogenetic relationship among the species, but is applied to any such species that tolerate salinity along the coasts of bays and estuaries.

Due to tidal flooding, mangroves face the same soil-based stresses as salt marsh vegetation. Like salt marsh vegetation, mangrove vegetation has aerenchymous tissue, tissue that helps to transport oxygen to flooded soils and reduce the stress. Mangroves, however, have additional adaptations that aid their survival in flooded soils, such as lenticels in their bark (allowing air exchange), and many have viviparous seedlings that germinate on the parent tree, thus avoiding high salinity and flooded

conditions that could hinder germination when seeds are buried in soil. Root growth provides a major input to the mangrove soil carbon stock. Mangrove swamps are critical habitat for fish and thus are indirect subsidies of offshore fisheries. The dense wood of the mangrove tree (*Rhizophora*) is highly valued for boat building and fishing gear and also for charcoal—activities which are strictly regulated in North America. Mangroves play an important role in wave attenuation and are even more valuable than salt marshes as a storm buffer, but this role can take a toll on mangrove stands. In Mexico, as elsewhere in the world, there are local communities that depend upon mangroves for their subsistence.

3.1 Mangrove Distribution

Mangroves can grow in soil where the salinity is higher than in seawater (albeit these are usually stunted, or dwarf forms) and extend inland to reaches of coastline that are freshwater tidal. They could thrive in soils saturated by freshwater, but in these locations mangroves cannot compete with other species.

The mapped distribution of mangrove in North America is shown in Figure 5. The regions of mangrove in Mexico and their area are listed in Table 5. For Mexico the polygons come from a single source, the National Commission for Knowledge and Use of Biodiversity (Conabio). Distribution data for US mangroves were available from the US Geological Survey through the United Nations Environment Program (UNEP). Both bodies used remote sensing to determine distribution.

The maps available document an overall mangrove area in North America of 10,094 km², and more than 76% of this is on the Mexican coast. Mangroves are present on the coasts of each of Mexico's coastal states. Within Mexico the three states of the Yucatán region hold the greatest area of mangrove, equivalent to about 55% of Mexico's total (Figure 6).

The northernmost extent of mangroves in North America is on the Atlantic coast, in the US states of Texas, Louisiana and Florida. At the northern extent of mangrove's range, mangrove survival is limited by the occurrence of frost (Mitsch and Gosselink 2007). Of the seven mangrove species in North America, three are dominant and widespread: *Avicennia germinans*, *Laguncularia racemosa* and *Rhizophora mangle*. *Avicennia germinans* has the northernmost range and is the least sensitive to frost. It is followed in distribution southward and in sensitivity to frost by *R. mangle* and *L. racemosa*. Expansion of mangrove populations has been reported in these states and attributed to a warmer climate (e.g., Bianchi et al. 2013; Henry and Twilley 2013). On the Pacific coast, the northernmost presence of mangroves is on the coast of the State of Baja California North, Mexico. Here, *R. mangle* has the northernmost extent, followed by *L. racemosa* to the south. In the State of Chiapas, on Mexico's Pacific coast, two other species are also found (Spalding et al. 2010; Rodríguez-Zúñiga et al. 2013): *Rhizophora Harrisonii*, a natural hybrid of *R. mangle* and *R. racemosa* (Cornejo 2013), and *Avicennia bicolor*. *Conocarpus erectus*, which occurs inland of these mangroves on both coasts, is considered by some a mangrove and by others a mangrove "associate."

Figure 5. Mangrove distribution in North America, 2015



Table 5. Regional distribution of mangrove area (in hectares) in Mexico in 1981 and 2005

Region	Year 1981		Year 2005		Net Change
	Area	% of Region	Area	% of Region	Area
Gulf of Mexico	89,581	10.5	87,470	11.3	-2,111
Yucatán Peninsula	452,962	52.9	421,399	54.5	-31,563
Northern Pacific	197,798	23.1	186,444	24.1	-11,354
Central Pacific	16,475.66	1.9	6,987	0.9	-9,489
Southern Pacific	98,750	11.5	71,555	9.3	-27,195
Total	855,566		773,854		-81,712

3.2 Threats to the Distribution of Mangroves

Most of North America’s mangroves occur on micro-tidal coasts and thus are highly sensitive to other threats. The most important threats are low levels of suspended sediment supply (low turbidity) and high levels of nutrient supply. All mangroves in Mexico’s Southern Pacific region and Central Pacific region are highly threatened by low levels of suspended sediment. Low levels of suspended sediment are also a high threat to portions of the Northern Pacific and Gulf of Mexico mangrove regions. The high-nutrient threat is the most prevalent in the Gulf of Mexico mangroves, but this threat does occur throughout the extent of North American mangroves.

In Mexico, the major drivers of mangrove loss have been land-use changes associated with shrimp- and fish-farming, agriculture, port infrastructure, tourism, and urban development, but also hurricanes. A threat analysis showed that most of Mexican mangrove areas are highly threatened due to high nutrient loads and low turbidity, but also many areas are highly threatened due to coastal squeeze, such as in the states of Jalisco, Colima, Quintana Roo and Yucatán. Despite Mexican laws to protect mangroves, there are several mangrove areas at the state level that are at high risk of irreversible loss due to coastal development, including areas in Jalisco, Colima, Nayarit, Michoacán, Guerrero, Baja California Sur, Quintana Roo and Yucatán. The existing infrastructure in these states leads to maintenance and expansion of existing urban, tourism and electricity infrastructure and, for some areas, new development (e.g., Baja California Sur). Mangroves in the Gulf of Mexico region are highly threatened by oil spills, and oil and natural gas extraction. Mangroves in the North Pacific are also threatened by the possible expansion of aquaculture activities, which have caused the loss and degradation of important mangrove areas.

Valderrama et al. (2014) show that loss of mangrove areas has been predominantly due to conversion to either wetland or terrestrial vegetation types. Transformation to agricultural and general anthropogenic land use types accounts for 80% of mangrove loss. Similarly, according to Mexico’s national assessment of mangrove change (Figures 7, 8 and 9), in recent years the most loss has been attributed to use of mangrove areas for agriculture. In at least some regions transformation of mangroves for agricultural use entails opening the canopy and planting grass for grazing animals. Tidal flooding of the sites persists. This kind of agricultural activity may result in minimal loss of blue carbon or even in gain of soil carbon (e.g., Yu and Chmura 2010), but research is required to assess the impact on the blue carbon sink. Under this type of agricultural use, other ecosystem services may be retained, at least in part. As long as opening lands to grazing does not lead to intensive modification, such as shrimp-farming, then carbon may be retained, as well as the potential for mangrove restoration, carbon sequestration and storage.

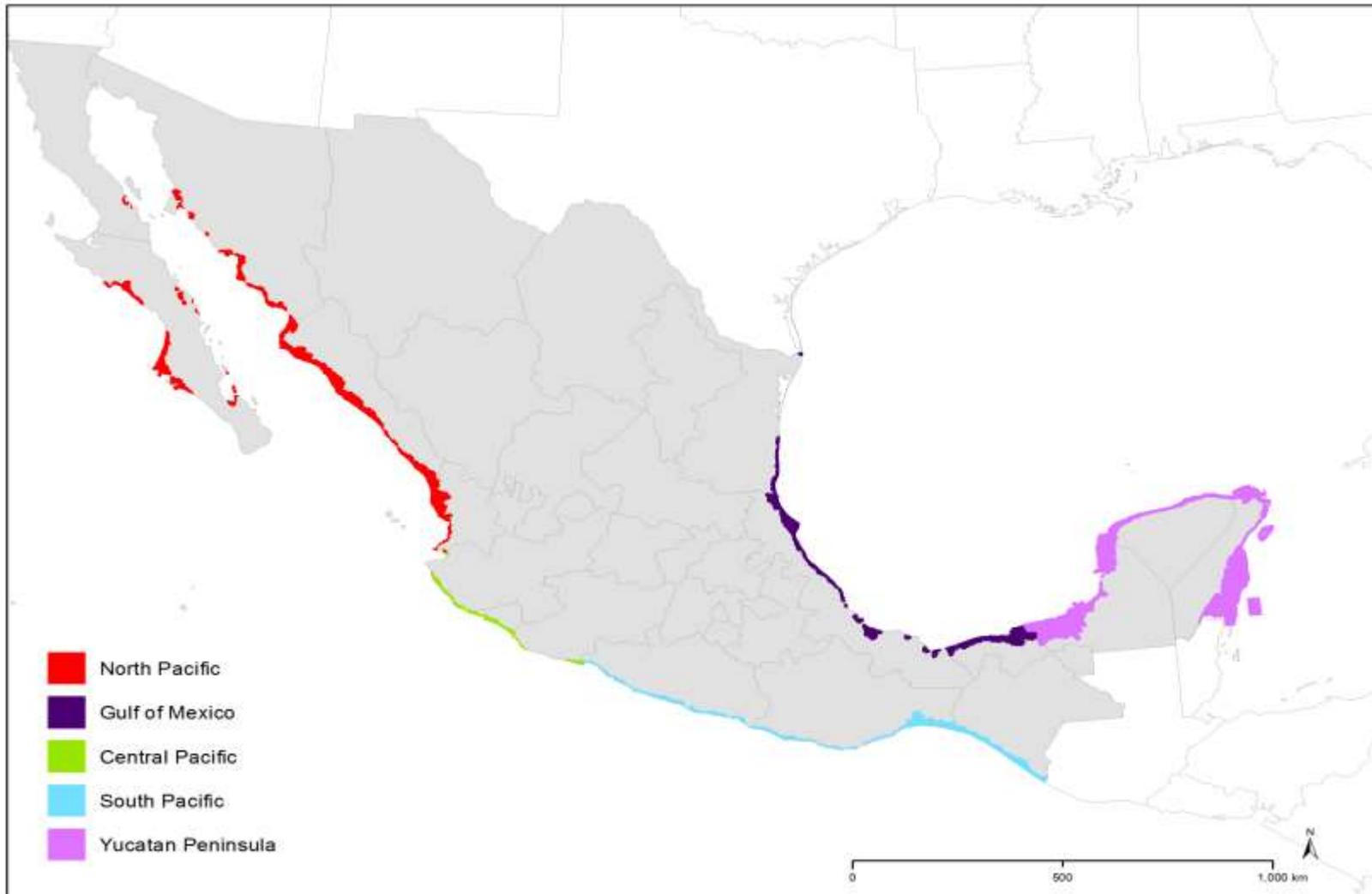
Compared to the salt marshes on the northeast coast of the United States, Mexican mangroves generally have a low level of threat attributable to land use and coastal squeeze. However, the states of Jalisco, Colima, Michoacán and Guerrero, which have small areas of mangroves (Figure 6), are proportionally facing a high level of threat by coastal squeeze.

With a warming climate, the range of mangroves is expected to expand poleward, with mangroves invading salt marshes. Researchers have begun to consider the impact of mangrove expansion on soil carbon. Bianchi et al. (2013) examined soils where mangroves had invaded *S. alterniflora* marsh on the Texas coast and found that where mangroves had invaded, soil carbon accumulation rates were twice that of nearby marshes without mangroves (2.7 vs. 1.0 tonnes per hectare per year), but were still within the normal range of carbon storage rates for marshes.

Restoration

Mexico's National Commission for Knowledge and Use of Biodiversity (Conabio 2013) lists 81 sites that are biologically relevant or where immediate restoration is necessary, ranging from 27 sites in the Gulf of Mexico to 10 sites in the North Pacific (Figure 10). Restoration is underway in many of these sites already.

Figure 6. Regions in Mexico with mangrove forest in 2013



Source: Adapted from Conabio 2013.

Figure 7. Distribution of mangrove forests in Mexico in 1970



Source: Adapted from Conabio 2013.

Figure 8. Distribution of mangrove forests in Mexico in 2005



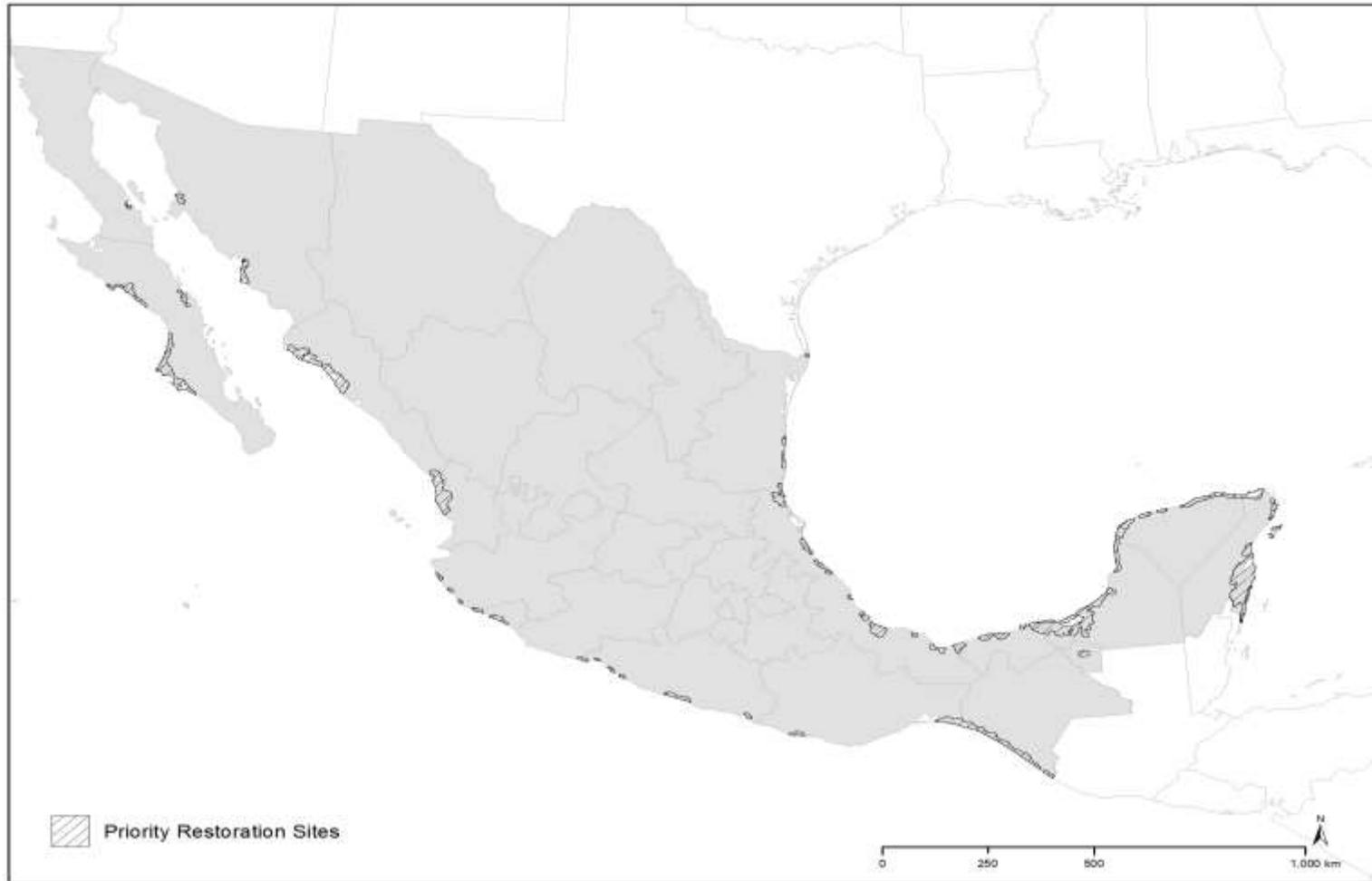
Source: Adapted from Conabio 2013.

Figure 9. Distribution of mangrove forests in Mexico in 2010



Source: Adapted from Conabio 2013.

Figure 10. Priority mangrove sites of biological relevance or in immediate need of ecological restoration, 2013



Source: Adapted from Conabio 2013.

3.3 Data Gaps for Mangrove Maps

In North America, mangrove mapping is the most complete of the mappings of the three blue carbon habitats. Repeated mapping has allowed development of time series demonstrating recent changes.

3.4 Mangrove Carbon Stocks and Rates

Like forest trees, mangroves capture, store and release carbon dioxide through photosynthesis and respiration and store considerable quantities of carbon in their wood (Table 6), while the above-ground biomass of other blue carbon habitats is considered negligible when accounting for carbon stocks (Kennedy et al. 2013).

Table 6. Global average above-ground carbon stocks in mangroves

Climate	Average Carbon Stock (tonnes ha ⁻¹)	Range of Carbon Stock (tonnes ha ⁻¹)	n
Tropical wet	86.6	3.9–173.6	49
Tropical dry	41.5	1.4–91.1	13
Subtropical	33.8	1.8–58.6	10

Note: One-meter depth is assumed for carbon stocks. ha⁻¹ = per hectare; n = samples taken.

Source: Adapted from Kennedy et al. 2013, Table 4.3.

However mangrove soils are generally in anoxic conditions (Twilley et al. 1992), thus soil organic matter is degraded at very low rates and stored in great quantities in comparison to that in terrestrial forests (McCleod et al. 2011). Organic carbon in mangrove soils originates from both local sources and external inputs (e.g., Bouillon et al. 2004). Local sources are mainly from roots and from litter from above ground, as well as from microphytobenthos (microflora) and macro algae (Bouillon et al. 2004; Kristensen et al. 2008). Root production is assumed to be a major contributor of organic carbon to the soil, but empirical studies assessing its importance relative to other local sources are needed. Inputs from tidal water include phytoplankton, seagrass-derived material (Bouillon et al. 2004), and organic sediments of riverine and marine origin.

The depth of mangrove soils is variable, and can be up several meters, therefore soils are the major carbon pool in these ecosystems, where organic carbon is stored for millenia. Mangrove soil carbon storage is influenced by different factors, such as the balance between production and organic matter decomposition rates driven by microbial activity, the net flux of carbon across mangrove boundaries (Twilley et al. 1992), and geomorphology. In mangrove swamps, the above-ground organic matter (particularly leaves) that is not decomposed may be permanently buried in soils but also exported by tidal forcing to surrounding ecosystems (Kristensen et al. 2008). Within the soil, decomposition of organic matter produces dissolved organic carbon that can be leached out of soils—a process controlled by water circulation within the ecosystem and external hydrological processes at watershed levels. Sedimentation processes in mangroves are extremely important as they will determine many ecosystem ecological processes such as carbon and nutrient burial and primary productivity, but more studies on this subject are warranted. Carbon accumulation plays an essential role in maintaining soil elevation with sea-level rise and thus sustainability of the mangrove ecosystem itself. In the process, mangroves effectively capture mud to create their own environment.

The relative importance of mangrove ecosystems as carbon sinks or sources can vary with the geographical location and with the ecological processes, which are highly sensitive to natural and anthropogenic influences. Excavation of mangrove soils can result in oxidation of organic matter that has been stored for as much as thousands of years. Knowledge of the carbon stocks in mangrove systems is needed in order to prioritize important areas for conservation of mangroves. However, studies have not reported how deep soils are within North America’s mangrove systems.

In the last five years, the recognition of mangroves as carbon sinks has drawn governmental, non-governmental, and scientific attention at local, national and international scales, in response to the need for data on carbon stocks and sequestration rates, for inclusion of mangroves in carbon incentive programs. The 2013 compilation of mangrove carbon stocks and accumulation rates (Kennedy et al. 2013) reveals that the range of carbon stocks in mangrove soil is wider and higher than that in salt marsh soils—assuming a one-meter depth (Table 7).

Table- 7. Global average soil carbon stocks in mangroves

Soil Type	Average Carbon Stock (tonnes ha ⁻¹)	Range of Carbon Stock (tonnes ha ⁻¹)	n
Organic	471	216–945	43
Mineral	286	16–623	82
All	386	55–1376	119

Note: One-meter depth is assumed for carbon stocks. ha⁻¹ = per hectare; n = samples taken.
Source: Kennedy et al. 2013, Table 4.11.

Since the IPCC publication, new data on carbon stocks in North American mangroves have been published. Locations studied include Sian Ka’an (Adame et al. 2013) and Laguna de Términos (Guerra-Santos et al. 2014) in Mexico, and the Florida Everglades in the United States (Breithaupt et al. 2014). The carbon stocks and rates reported all fall within the ranges shown in Table 7.

3.5 Mangrove Carbon Stock Conservation

As mangroves are intertidal ecosystems, their long-term permanence requires that they be able to maintain vertical accretion rates equal to or greater than the rate of sea-level rise. If mangroves are not able to adapt to these rapid changes through sediment inputs and below-ground root production, they will become vulnerable and may release important quantities of carbon into the atmosphere. Another consequence of climate change is the more frequent occurrence of tropical storms and hurricanes, which can affect the biogeochemical conditions of mangrove soils through excess deposition of sediments. Additionally, human activities (e.g., construction of dams and modification of hydrological conditions) may also affect the function of mangroves as carbon sinks, by interrupting the supply of sediment that enhances mangroves’ capacity to maintain equilibrium with sea-level rise.

4. Marine Protected Areas, Terrestrial Protected Areas and Blue Carbon Habitats

The areas of mapped seagrass, tidal marsh, and mangrove occurring in marine protected areas (MPAs) and terrestrial protected areas (TPAs) are listed in Table 8. In some cases blue carbon habitat

falls within the jurisdiction of a TPA but not an MPA, as in the case of Baja California (Figures 11 and 12). In other areas large proportions of the blue carbon habitat fall within both MPAs and TPAs, such as in the case of the mangroves of the State of Campeche, Mexico. In contrast, a very small proportion of the salt marshes of Nova Scotia and New Brunswick, Canada, fall within either MPAs or TPAs. Seagrass habitat sometimes falls within the boundaries of both MPAs and TPAs. As with salt marshes and mangroves, the mapping coverage varies, with close to 100% mapped coverage of the area in North Carolina, USA, to less than 12% elsewhere. In some locations MPA areas overlap TPA areas (Figure 11), but the degree of this overlap has not been calculated. MPAs also overlap with each other, as they are established to meet different objectives—for example, to regulate the catch of a single fish species, or for general protection, such as a national park.

For all of North America, the number of seagrass polygon data, line data and point data that occur within marine protected areas (MPAs) showed that 58% of polygons, 47% of lines, and 10% of points might be considered to have some protection via MPA status. These percentages represent only a small portion of the existing seagrass resource; and, as already stated, the protection granted to seagrasses within MPAs is minimal. The line data—representing length of shoreline containing seagrass—are limited to the Pacific Northwest and, of the 47% that falls within MPAs, almost all are from the Tongass National Forest in southeast Alaska. However, this status provides no protection for marine waters and, in fact, forest clear-cutting is permitted, contributing to upland erosion and loss of water clarity in the coastal waters.

A surprise finding with regard to seagrass was that virtually no MPAs provide real protection for seagrass habitat (Spalding et al. 2003). In fact, MPAs often occur in places where seagrass is declining, and the MPA designation as a protected area does not address the present-day threats to these submerged plants.

Neither MPAs nor TPAs are likely to afford the kind of protection required to sustain blue carbon habitats today. While protective legislation is in place in Canada, Mexico and the United States to prevent wetland loss, seagrasses are declining in many MPAs and TPAs, since runoff from watersheds and the resulting nutrient-loading are rarely under the purview of protected areas. Nevertheless, studies of marsh and mangrove wetlands show a recent decline in the rate of loss, compared with extensive historical losses to urbanization, construction of infrastructure for transportation, and agriculture. The projected increase in the rate of sea-level rise, however, will likely renew high loss rates, particularly if indirect threats to mangroves and salt marshes are not addressed. Such indirect threats are typically not part of MPA or TPA designations or management.

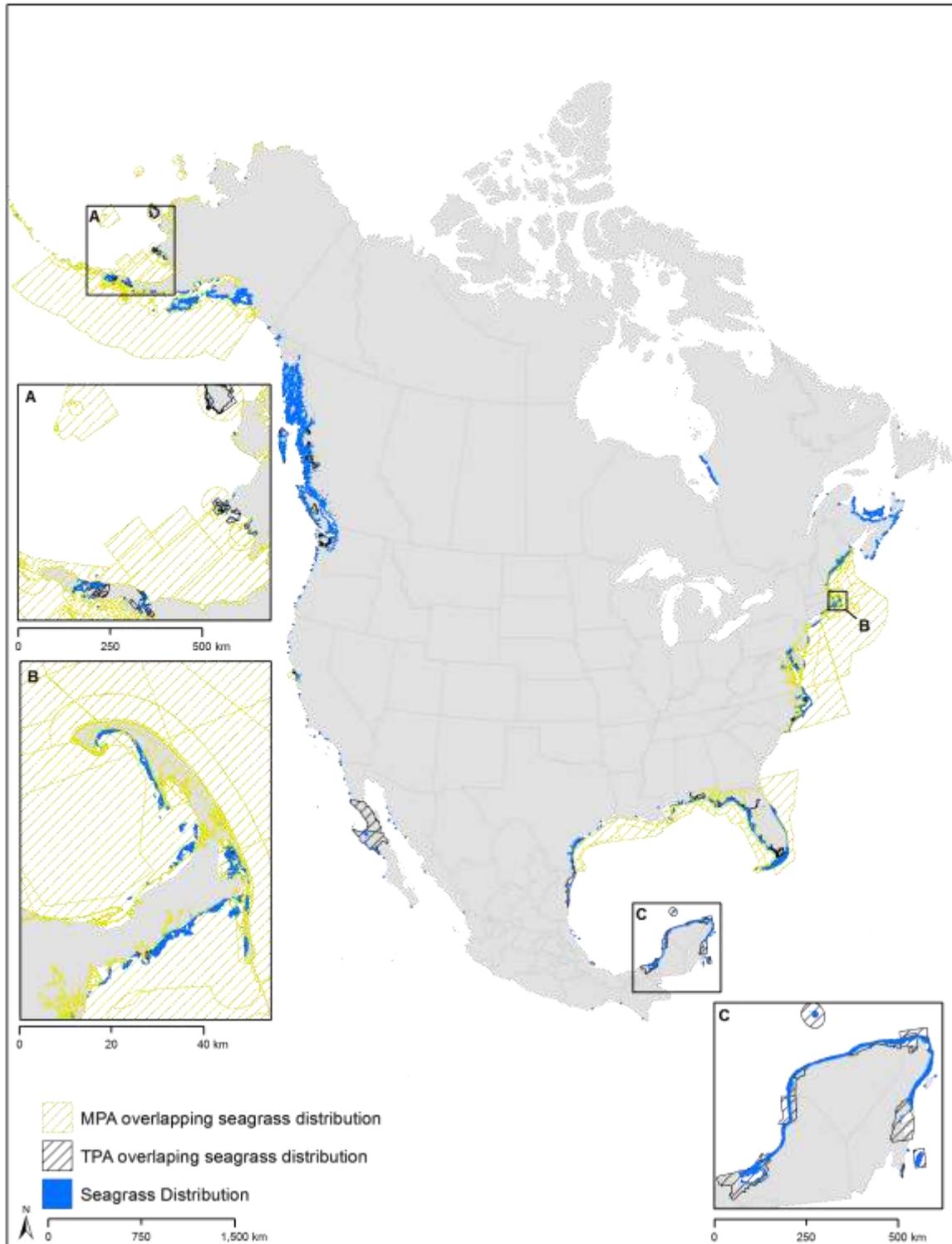
Table 8. Areas of mangrove, seagrass and salt marsh habitat within marine protected areas (MPAs) and terrestrial protected areas (TPAs), 2014

Country	State/Province	Mangrove			Seagrass			Salt marsh		
		All Habitats (km ²)	In MPA (km ²)	In TPA (km ²)	All Habitats (km ²)	In MPA (km ²)	In TPA (km ²)	All Habitats (km ²)	In MPA (km ²)	In TPA (km ²)
Canada	British Columbia	0	0	0	643	196	13	111	19	3
	New Brunswick	0	0	0	0	0	0	128	1	4
	Nova Scotia	0	0	0	0	0	0	158	6	3
	Prince Edward Island	0	0	0	0	0	0	70	0.3	4
	Quebec	0	0	0	2	2	0	79	55	11
Mexico	Baja California	0.3	0	0.3	24	0	3	30	0	21
	Baja California Sur	262	36	36	450	424	450	123	108	121
	Campeche	1,255	1,072	1,078	4,276	2,097	2,280	0	0	0
	Chiapas	395	276	294	0	0	0	0	0	0
	Colima	27	0	0	0	0	0	0	0	0
	Guerrero	76	0	0	0	0	0	0	0	0
	Jalisco	24	0	1	0	0	0	0	0	0
	Michoacán	15	0	0	0	0	0	0	0	0
	Nayarit	705	0	0	0	0	0	0	0	0
	Oaxaca	190	0	0	0	0	0	0	0	0
Quintana Roo	1,315	692	887	4,049	1,257	1,559	0	0	0	

	Sinaloa	832	0	91	0	0	0	0	0	0
	Sonora	113	0	12	0	0	0	0	0	0
	Tabasco	485	23	144	0	0	0	0	0	0
	Tamaulipas	18	0	3	152	0	145	0	0	0
	Veracruz	372	0	8	0	0	0	0	0	0
	Yucatán	1,683	918	1,434	716	265	268	0	0	0
USA	Alaska	0	0	0	405	361	316	948	284	330
	Alabama	0	0	0	4	0.02	0.10	129	33	26
	California	0	0	0	37	5	3	192	16	32
	Connecticut	0	0	0	8	1	0	37	11	6
	Delaware	0	0	0	0	0	0	293	141	67
	Florida	2,303	1,755	1,853	10,739	8,268	137	1,300	799	800
	Georgia	0	0	0	0	0	0	1,281	158	274
	Louisiana	22	4	5	65	19	8	2,954	75	200
	Massachusetts	0	0	0	125	125	0.1	170	105	28
	Maryland	0	0	0	153	129	8	399	55	104
	Maine	0	0	0	100	99	0.2	76	31	14
	Mississippi	0	0	0	1	1	0	226	166	145
	North Carolina	0	0	0	559	553	5	767	357	289
	New Hampshire	0	0	0	7	7	0	22	5	0.69
	New Jersey	0	0	0	53	52	0.09	608	366	232
New York	0	0	0	82	78	16	87	45	19	
Oregon	0	0	0	5	0	0.00	33	6	16	
Rhode Island	0	0	0	4	4	0.00	5	4	0.1	

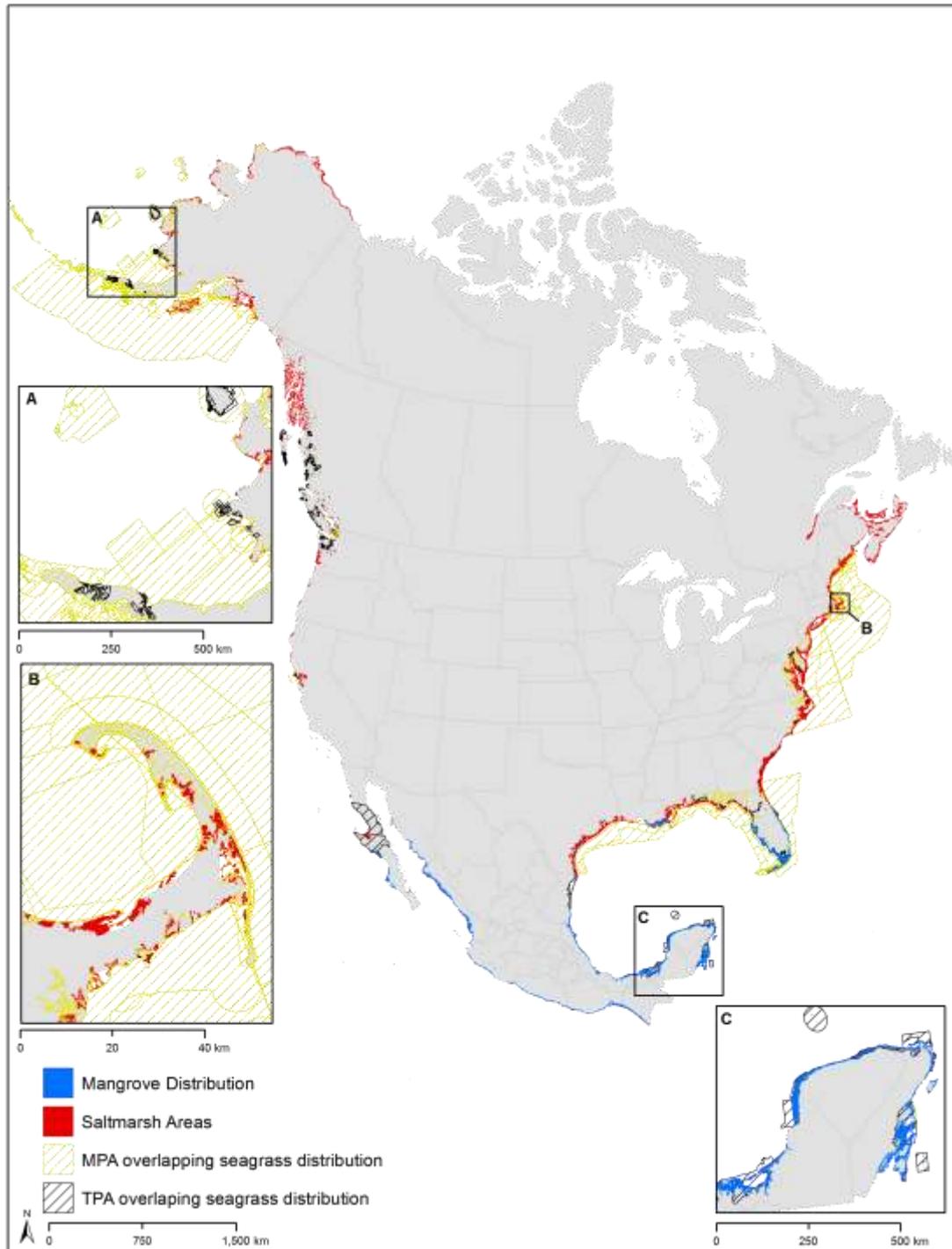
	South Carolina	0	0	0	0	0	0.00	1,304	299	371
	Texas	19	6	3	897	106	14	1,393	298	227
	Virginia	0	0	0	2	1	0.49	501	215	225
	Washington	0	0	0	632	82	34	49	2	6

Figure 11. Seagrass distribution within MPAs and TPAs in Mexico



Note: MPA = marine protected area; TPA = terrestrial protected area; A = Bristol Bay, Alaska, USA; B = waters around Cape Cod, Massachusetts, USA; C = Yucatan Peninsula, Mexico.

Figure 12. Salt marsh and mangrove distributions within MPAs and TPAs in Mexico



Note: MPA = marine protected area; TPA = terrestrial protected area; A = Bristol Bay, Alaska, USA; B = waters around Cape Cod, Massachusetts, USA; C = Yucatan Peninsula, Mexico.

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