Chapter 15. Tidal Wetlands and Estuaries

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Key Findings

1. **Despite their small area, tidal wetlands and estuaries influence both terrestrial and oceanic carbon stocks and fluxes at global scales.**
2. **Carbon fluxes in tidal wetlands and estuaries are strongly influenced by land-use decisions upstream, as associated with nutrients, sediments, and hydrology.**
3. **Carbon fluxes in tidal wetlands and estuaries are strongly influenced by physical, chemical and biological coastal ocean processes**
4. **Estuaries vary in their relative distribution of tidal and subtidal components, thus influencing the direction and magnitude of C related exchanges between tidal wetlands and estuarine waters.**

15.1 Introduction

Estuaries and tidal wetlands are dynamic ecosystems that host high biological production and diversity (Bianchi 2007; ref)). They receive large amounts of dissolved and particulate carbon (C) and nutrients from rivers and uplands and exchange materials and energy with the ocean. Estuaries and tidal wetlands are often called biogeochemical “reactors” or “filters” where terrestrial materials are transformed through interactions with the land, the ocean and atmospheric forcings. Work conducted in the past decade has clearly showed that estuaries as a whole can be strong sources of carbon to the atmosphere – carbon dioxide (CO_2) and methane (CH_4) –despite the fact that we do not know how the strength of the degassing varies in space and time in many estuaries (Borges and Abril 2011; Cai 2011).In contrast, tidal wetlands represent a small area but per unit area are among the strongest long-term C sinks, due to continuous organic C accumulation in sediments with rising sea level. Estuaries were not included in the previous assessment of coastal C cycling in the SOCCR-1 report, but were reviewed in recent synthesis activities, particularly the Coastal CARbon Synthesis (CCARS) (refs). A consistent and important missing piece in previous fieldwork and synthesis is annual C exchanges (including CO_2 flux) across aquatic (hereafter estuarine) and intertidal (hereafter wetland) boundaries, which limit our understanding of their relative roles in the critical land:ocean margin. An updated synthesis of current knowledge and gaps in quantifying the magnitude and direction of C fluxes in dynamic estuarine environments is presented in this chapter.

Estuarine zones contain both intertidal (wetland and unconsolidated shore) and subtidal (aqueous and benthic) components. For the purpose of this chapter, the landward boundary of estuarine zones is defined herein as the head of tide (the maximal boundary of tidal expression in surface water elevation), and the seaward boundary as the continental shelf (the relatively shallow sea that extends to the edge of continental crust). This chapter is organized by seven sections. Section 15.2 provides a brief review of the historical and global context of C flux in estuaries and tidal wetlands. Section 15.3 assesses the current state of the C cycle and understanding of fluxes and stocks in tidal wetlands and estuaries. Section 15.4 synthesizes C fluxes of estuaries and tidal wetlands of the North America in the global context and from regional perspectives. We will also discuss new and relevant coastal C observations through indicators, trends and feedbacks in Section 15.6, and management and decisions associated with societal drivers and impacts in Section 15.6 within the C cycle context. Finally, we provide in Section 15.7 a synthesis that summarizes conclusions, gaps in knowledge and near future outlooks.

15.2 Historical Context

Tidal wetlands and estuaries are the subsystems that most influence the C dynamics of the coastal ocean. Not only are they the conduit and filter for all material passing between land and the sea, but they also exhibit the highest transfer rates of CO_2 with the atmosphere of any of the coastal ocean subsystems (Bauer et al. 2013, Benway et al. 2014). Tidal wetlands and estuaries of the US vary in relative area depending on coastal topography, historic rates of sea-level rise (SLR), and inputs of suspended solids from land. In drowned river valleys, (e.g. Chesapeake Bay, US) that are topographically steep, estuarine (e.g. aquatic) habitat is the dominant subsystem. In contrast, in topographically flat coasts (e.g. southeastern US coasts) with high rates of sediment inputs from land, tidal wetland habitat is the dominant subsystem (Day et al. 2013). C dynamics of both estuarine and intertidal subsystems are constantly changing, reflecting the geomorphic and ecological response to long and short-term perturbations and trends in external drivers, such as SLR, climate change and terrestrial land-use change.

The land-sea interface that defines the presence of tidal wetlands and estuaries (river-sea mixing zones) is itself extremely dynamic over broad spatial and temporal scales. The current configuration of wetlands and estuaries is the result of processes that have been occurring since the last glacial maximum, over 18,000 years ago. Over the past 4000 to 6000 years, when rates of SLR dropped to less than 1 mm/yr, tidal wetlands increased in size relative to open water estuaries, as bay bottoms filled with sediments from land and tidal wetlands prograded into shallowing open water regions and transgressed across uplands (**Figure 15.1**, Redfield 1967). Concomitant with increasing sea levels, tidal wetlands maintained their relative elevation as wetland plants trapped suspended solids from tidal floodwaters, and accumulated organic matter in sediments. Factors that affect tidal wetland area and relative elevation, through lateral and vertical erosion and accretion, include the rate of SLR, land subsidence or eustasy (glacial rebound), delivery and deposition of suspended sediment, the balance between wetland gross primary production (GPP) and respiration of all autotrophs and heterotrophs (R_{AH}), sediment compaction and the slope of land at the land-water interface (Cahoon 1997).

Tidal wetlands are among the most productive ecosystems on earth and continuously accumulate organic C in their sediments as a result of environmental conditions that inhibit organic matter decomposition. As a result, intact tidal wetlands are capable of storing vast amounts of both autochtonous C (produced on site) as well as intercepting and storing allochtonous carbon (produced off site). Documented C-related ecosystem services include significant uptake and storage of C in wetland soils, as well as ocean-export of organic matter that enhance productivity of coastal fisheries (Day et al. 2013). Globally, tidal wetlands are both poorly mapped and strongly variable in age and structure. Some of today’s tidal wetlands have persisted for more than 6,000 years, accumulating up to 10m of soil (e.g. McKee et al 2007), but some wetlands are young, due to recent human influences that enhanced sediment delivery (e.g. colonial era

East Coast; Kirwan et al 2011, gold rush in CA; Paliama 2007) to nearshore waters. Because human development is preferentially concentrated on coastlines, tidal wetlands have been subject to active loss, which has slowed in the US but is currently estimated at 2% annually worldwide. Loss of carbon stocks through wetland drainage and erosion remains poorly modeled due to limited mapping and quantification of initial C stock conditions.

Coastal waters represent 8% of the global ocean and about 25% of marine primary production (Walsh, 1989; Wollast, 1991). However, the role of coastal zones as sinks or sources of CO_2 is still not fully understood (Smith and Hollibaugh, 1997; Borges, 2005; Borges et al., 2005), resulting in an underestimation of global carbon budgets (Wollast, 1991). Estuarine waters have strong upland and ocean-based drivers, leading to strong seasonality in C behavior. Long term records suggest that estuarine C storage was enhanced in the past 6000 years by watershed activities, but the responses are varied. The effects of watershed alteration by direct and indirect human actions (hydrologic barriers, nutrient runoff, sediment redistribution) strongly affects water quality in estuaries. While human activities initially increased the delivery of organic materials to the estuaries and thus drove them to be more heterotrophic (respire more organic carbon), more recent human activities (dam construction) has greatly reduced organic matter delivery to many estuaries in the work (Bianchi and Allison 2009); thus it is expected that estuarine waters are and will be less heterotrophic. It has also been suggested that more carbon is buried in estuaries over past few decades (Regnier et al. 2013). The most important human induced changes to estuaries over the past century is however the increased nutrient loading, which has led to eutrophication and hypoxia in estuaries. It may have promoted C uptake and pH increase in estuarine surface environments (Borges and Gypens 2010) but also may enhance ocean acidification in subsurface or downstream waters (Feely et al. 2010; Cai et al. 2010).

15.3 Current Understanding of Carbon Fluxes and Stocks

Tidal wetlands have been long-term C sinks, but their net annual C exchanges, and the form of C fluxes, reflect changing environmental conditions (Kirwan and Megonigal 2013). Some of the key variables shown to influence today's magnitude and form of C fluxes in coastal wetlands include land subsidence (Allison et al 2016), extreme events (Crosswell et al 2014), nitrogen loading (Wigand et al. 2009), rising temperatures (Kirwan et al 2011), and sediment availability (Shields et al 2016). Geomorphic models, and empirical evidence from cores, illustrate the significant role of relative sea level rise in driving the rate of long-term C sequestration in soils (e.g. Drexler et al 2009; Morris et al 2012).

Vegetative and soil carbon stocks in coastal wetlands, vary widely with latitude, geologic and geomorphic setting, rates of productivity, and age of the wetland. Whereas mangrove ecosystems typically contain significantly more biomass carbon than salt marsh ecosystems (Adame et al. 2013; Doughty et al. 2016), mangroves themselves have marked variability in aboveground carbon stocks. In Florida alone, dwarf mangrove habitats may have biomass carbon stocks of only 8 Mg C ha^{-1}, whereas mature mangrove forests may have biomass carbon stocks of 110 Mg C ha^{-1} or more (Howard et al. 2014; Simard et al. 2006). Regardless of the aboveground biomass, C stocks in mangrove and salt marsh soils are longer lived and more dominant representing up to 98% of the total C pool (Donato et al. 2011).

Estuarine C dynamics reflect the balance of exchanges with terrestrial watersheds, tidal wetlands, and the continental shelf (Bauer et al. 2013). Processing of material inputs from land reflects the ecological structure of the receiving estuary and especially the quantity and relative ratio of organic C (primarily from tidal wetlands and land) and inorganic nutrient inputs (primarily from land) and estuarine water residence time, which reflects a balance in freshwater runoff and tidal mixing (Hopkinson and Vallino 1995, Kemp et al. 1997, Herrman et al. 2015). The relative abundance of pelagic (phytoplankton-dominated) vs benthic (seagrass or benthic algal-dominated) communities is also a major factor affecting estuarine C dynamics. The availability of light is perhaps the major constraint on the distribution of benthic autotrophic communities. Light availability to the benthos depends on estuarine depth and water clarity, which in turn is related to concentrations of suspended solids and phytoplankton in the estuarine water column. In N-enriched estuarine waters, high phytoplankton biomass, and epiphytic algae decrease light availability to benthic autotrophic communities, sometimes resulting in a complete loss of seagrass habitats (Howarth et al. 2000). In shallow systems, benthic macroalgae often dominates system dynamics. Seagrass is particularly important in estuarine C dynamics because it can accumulate excess production as blue carbon in its sediments (Duarte et al. 2005)). Seagrass, because of its ability to control wave and current strength, can play a major role in limiting sediment resuspension, thereby maintaining high water clarity (ref). Estuaries are typically heterotrophic, releasing excess CO_2 to the atmosphere, largely as a result of their processing of organic C inputs from watersheds (Raymond and Bauer 2001) and adjacent tidal wetlands (Bauer et al. 2013, Wang and Cai 2004). For example, US east coast estuaries as a whole are net heterotrophic (Hermann et al 2015); all but 3 of 42 sites in the US National Estuarine Research Reserve System were net heterotrophic over a year (Caffrey 2004), and a global survey concluded that 66 out of 79 estuaries were net heterotrophic (Borges and Abril 2011). At the same time, estuaries can serve as significant long-term OC sinks through sedimentation of terrestrial inputs and seagrass organic matter burial (Duarte et al. 2005, Nellemann et al. 2009, McLeod et al. 2011, Hopkinson et al. 2012).

Table 15.1. Average (range) values for ecosystem extent. Canada, Mexico, U.S. combined (CEC 2016)

	Tidal Freshwater Marsh	Tidal Freshwater Forest	Brackish/Saline Marsh	Mangrove	Seagrass	OpenWater* not sure this can be done
East-temperate						
Hectares						
Volume						
Soil C stock(1m)						
Gulf-temperate						
Hectares						
Volume						
Soil C stock(1m)						
West-temperate						
Hectares						
Volume						
Soil C stock(1m)						

	Tidal Freshwater Marsh	Tidal Freshwater Forest	Brackish/Saline Marsh	Mangrove	Seagrass	OpenWater* not sure this can be done
Boreal/Arctic						
Hectares						
Volume						
Soil C stock(1m)						

Table 15.2. Example C flux chart: Gulf of Mexico C budget for 37 U.S. estuarine systems)

	Input	NEP	Burial	Export
Area-integrated flux, Tg-C y^{-1}	9.8	-2.7	0.21	6.8
Area-normalized flux, mol-C y^{-1}	27	-7.5	0.57	19

15.4 Global, North American, and Regional Context

15.4.1 East Coast Estuaries and Tidal Wetlands

East coast estuaries are the most extensive and diverse in structure and function. Relatively shallow and driven primarily by landward forcings, they are strongly influenced by river flow and quality.

East Coast Estuaries

South Atlantic Bight. The South Atlantic Bight (SAB) is a typical passive, western boundary current margin with broader shelf areas and has extensive shoals and a series of barrier islands, behind which are lagoons with abundant salt marshes. Freshwater delivery in the SAB is through major rivers that are nearly evenly located along the coast. Many of the rivers are "black water" systems carrying high loads of dissolved organic carbon (DOC). SAB estuaries are mostly pristine saltmarsh lagoons and are pierced by the major rivers discharging along the coast. These marshes are the most extensive and productive of the US east coast (Schubauer and Hopkinson 1984, Gallagher et al. 1980) and evidence suggests that large amounts of marsh grass-derived organic matter (OM) and CO_2 are exported into the estuaries nearshore ocean where respiration and degassing occur (Wang and Cai 2004; Jiang et al. 2008). Many SAB rivers and estuaries have high DOC concentrations (several mM) and because of short transit times through many of the estuarine regions, much of the DOC is also discharged directly onto the shelf, supporting respiration and net heterotrophy (Hopkinson 1985, 1988) and CO_2 degassing on the inner shelf regions (Jiang et al. 2013). Much is known about the export of OM from SAB watersheds, including its magnitude, chemical composition and stoichiometry, and reactivity. We are relatively confident in concluding that the SAB salt marshes are tremendous sinks of CO_2 and organic carbon from uplands. We further conclude that estuarine waters are strong sources of CO_2 to the atmosphere, which are largely supported by OM and DIC export primarily from the very productive intertidal saltmarshes and secondarily from SAB watersheds (Hopkinson 1988, Cai 2011).

Mid-Atlantic Bight. The Mid-Atlantic Bight (MAB) coastline is dominated by numerous small to large size estuaries where C cycling is influenced by variable coastal morphology, climate and watershed characteristics. Inorganic carbon from carbonate weathering and OM remineralization accounts for the majority of the riverine C input to the MAB (Moosdorf et al. 2011; Hossler and Bauer 2013a). OM is generally higher in southern MAB rivers and can be more than half the riverine C load to estuaries that drain the piedmont and coastal plain, e.g. the lower Chesapeake Bay and Albemarle-Pamlico Sound (Stets and Striegl 2012; Tian et al. 2015). Lateral exchange with wetlands is an important C input to MAB waters and has been linked to net heterotrophy and air-water CO_2 efflux in narrow, marsh-dominated subestuaries (Raymond et al. 2000, Bauman et al. 2015). The latest evidence suggests that lateral export of DIC from tidal wetlands may be a primary source of C to the U.S. East coast, comparable to, or in some cases even larger than, the riverine C flux (Wang et al., 2016). However, larger MAB estuaries can be seasonal or annual sinks for atmospheric CO_2 due to stratification and high rates of internal production (Crosswell et al. 2014, Joesoef et al. 2015). Supporting this, recent modelling studies have estimated that that MAB estuaries are at near metabolic balance and that DOC export to the coastal ocean is approximately equal to riverine DOC input (Herrmann et al. 2015). However, data are lacking in most large estuaries in this region, for example, the air-water CO_2 flux is not known in the nation's largest estuary the Chesapeake Bay.

East Coast Tidal Wetlands

Despite some similarity in vegetation community composition (e.g. estuarine emergent *Spartina- spp.* dominant), East Coast tidal marshes are extensive and topographically varied in structure, from the more organic –rich MAB soils to the mineral rich plains of the South Atlantic Bight. Whereas organic content (% weight) shows regional increases with latitude, GPP decreases with latitude in tidal marshes, as does decomposition rates (Kirwan et al 2011). Perhaps more important than latitudinal patterns for C flux accounting are within-watershed patterns of marsh elevation (low marsh vs. high marsh), tidal range (e.g. microtidal eastern Florida vs. extreme macrotidal Bay of Fundy), and salinity regimes, whereby freshwater tidal wetlands (both marsh and forest) represent up to 30% of tidal wetlands. Data from chamber and eddy-covariance systems illustrate that vertical fluxes dominate CO_2 exchange (e.g. Forbrich et al 2015, Schafer et al 2012, Fuentes et al 2008). However, where measured hydrologic exchange is critical for closing C budgets, with lateral flows of DIC and DOC fluxes representing as much as80% of annual C inputs (Wang and Cai 2004, Wang et al 2016).. Further, the role of groundwater flows in driving C fluxes, as well as nutrient fluxes that alter estuarine processes, is varied and poorly understood (Moore et al 2011, Kroeger et al 2008).

15.4.2 Gulf Coast Estuaries and Tidal Wetlands

Gulf Coast Estuaries

Carbon fluxes into subhabitats of the Gulf of Mexico have been extensively studied at the land:ocean margin for years. Four major terms of estuarine organic C budgets for the Gulf of Mexico estuaries (excluding Mississippi and Atchafalaya Rivers) include:

1. riverine input,
2. burial in the sediment,
3. net ecosystem production (NEP), and
4. export to the ocean.

Gulf Coast estuarine environments are microtidal with winds and riverflow exerting strong control on water levels. On the extensive subtidal carbonate benthos, extensive seagrass meadows (e.g. Thalassia) persist and are known to recover rapidly from disturbance (McDonald et al 2016). We know poorly about air-water CO_2 flux in the Gulf of Mexico estuaries. Even though NEP can be estimated for the US portion of the Gulf, given the higher C/N ratio of riverine organic matter, it is expected that CO_2 flux is higher than NEP. Take the northwestern Gulf of Mexico (US side) for an example, there is a climatic gradient from northeast to southwest. Riverine freshwater export (much of it goes through lagoonal estuaries except the Brazos) decreases by a factor of two although net freshwater balance differs by two orders of magnitude from northeast to southwest (Montagna et al., 2009). Difference in riverine input coupled with climate condition in this compressed yet contrasting region clearly would result in different organic C delivery by rivers.

Gulf Coast Tidal Wetlands

Gulf Coast tidal wetlands are extensive, 52% of all US tidal wetland acreage (National Wetland Inventory) and embrace a wide range of salinity and elevation conditions, from tidal freshwater forests through floating marshes through extensive saline marshes. In addition to their extreme diversity and expansiveness, tidal wetland loss is greater in Gulf of Mexico than anywhere else in North America, especially due to consequences from land subsidence, coastal storms, sea level rise, and a lack of sediment delivery to compensate for ongoing compaction. Rising sea level and temperatures can also result in shifts in wetland cover. For example, mangroves have increased by 35%, encroaching on salt marshes in especially in Texas, Louisiana, and Florida (Krauss et al 2011, Bianchi et al. 2012; Saintilan et al. 2013).

Mangrove C sequestration rates can can range from 0-10 Mg ha-1 y-1, due to primarily to biomass variability, linked to disturbance status and hydrogeomorphic characteristics of the landscape; fringe, dwarf, riverine, and basin types (Adame et al, 2013; Breithaupt et al, 2014, Ezcurra et al, 2016; Marchio et al, 2016). Regular tidal flushing and allochthonus input from river and marine sediments generally provide more favorable conditions for above and below ground productivity. Similarly, the below-ground components of mangrove forests like coarse woody debris, soil, and pneumatophores can contribute between 45 and 65% of the total ecosystem respiration where inundation and soil salinity are the major drivers of variability (Troxler et al, 2015). Despite their short stature, dwarf mangroves, for example, may generate greater C pools belowground than taller statured mangroves (Hererra-Silveira et al 2009, Osland et al. 2012; Adame et al. 2013).

Both abrupt (hurricane) and chronic (SLR) changes are currently altering subtropical tidal wetlands of the northern Gulf of Mexico. Salt water intrusion into mangrove forests and coastal marshes associated with sea level rise, storm surge, and groundwater extraction can increase both inundation and salinity that impact both above and belowground carbon cycling (Chambers et al. 2013, 2016; Williams & Rosenheim 2015). The landward expansion of mangroves into salt marshes in two different regions of Florida has been attributed to salt water intrusion (Ross et al 2000; Krauss et al 2011). Organic carbon stocks face increased rates of mineralization and peat collapse with saline intrusion (Troxler et al 2015). In spite of these risks, carbon stocks may potentially increase as a result of statewide trends of mangrove expansion into salt marsh habitat (Doughty et al. 2015).Mangroves have already expanded their range both landward and northward as a result of altered hydrology and a lack of recent cold events (Krauss et al. 2011; Cavanaugh et al. 2014). This pattern is expected to continue with current trends in climate change and with increasing SLR rates. Severe damage from hurricane and freeze events can have distinct patterns based on the mangrove forest structure and coastal geomorphology (Barr et al, 2012; Lagomasino et al, 2014; Zhang et al 2016). Dwarf and basin mangroves, which generally have shorter canopies, are most effected by freezing temperatures, while hurricane damage has the strongest impact on fringing mangrove forests along the coasts (Zhang et al, 2016). Freeze and cold events drive the poleward advancement of mangroves along the east coast of Florida and the Gulf of Mexico (Giri et al, 2011; Cavanaugh et al, 2014; Saintilan et al, 2014). Though the actual extent of mangroves along these regions may not currently extend past their historical range limits (Giri and Long, 2014), the expansion and contraction of the mangrove forest is clearly documented.

15.4.3 West Coast Estuaries and Tidal Wetlands

West/Pacific Coast Estuaries

Pacific coast estuaries differ from other North American estuaries in that the C cycle dynamics tend to be dominated by ocean-sourced rather than river-borne drivers. North Pacific Ocean source waters are higher in CO_2 and nutrients and lower in oxygen than those in other open-ocean environments, predisposing Pacific coast estuaries and coasts toward hypoxia and acidified conditions, largely as a result of natural processes. Upwelling brings older, deeper water with these characteristics onto the shelf throughout the California Current System and into estuaries and bays, where the water is further modified by local estuarine processes.

From the Salish Sea to the north, glacially formed estuaries are characterized by sills restricting free circulation among estuaries and coastal waters that significantly alter estuarine physical and biogeochemical dynamics and can lead to hypoxic or anoxic conditions forming in deep water of these estuaries. South of the Salish Sea to the Southern California Bight, Pacific Coast estuaries tend to either be one of a few globally significant large river systems such as the Columbia or San Joaquin/Sacramento river estuaries or one of many “small mountainous rivers” (SMRs). These three types of estuaries have distinct C cycle characteristics and dominant processes. Major population centers line glacial and other major river estuaries, but human population centers along SMR-dominated coastlines are typically, but not always, smaller.

Phytoplankton productivity estimates across Pacific coast estuaries (San Francisco Bay to British Columbia, Canada) reflect an order of magnitude variation in median annual primary production rates, from 50 in the Columbia River estuary to 455–609 g C m-2 y-1 in the Indian Arm fjord near Vancouver, British Columbia (Cloern et al. 2014). A total water column primary production estimate for the Columbia River estuary summed to 30,000 metric tons C y-1 (Lara-Lara et al. 1990).

Terrestrial inputs to Pacific coast estuaries vary substantially along the steep rainfall gradient from very wet conditions in the north (Gulf of Alaska through northern California) to arid conditions in southern and Baja California, with increasing precipitation again from central Mexico through Panama. The Global NEWS 2 model estimated that total organic C inputs range from approximately 8.5 Tg C y-1 for the Gulf of Alaska through northern California to ~0.7 Tg C y-1 for southern and Baja California and the Gulf of California and 2.8 Tg C y-1 for Mexico south of Baja California and Central America (Mayorga et al. 2010). As with primary production and C burial in estuaries, monitoring efforts are inadequate and declining to track changes in terrestrial inputs of all forms of C or nutrients that would affect C cycling in estuaries and coastal oceans in the face of climate and other human-caused environmental changes (Boyer et al. 2006, Canuel et al. 2012).

The Southern California Bight (SCB) stretches from Point Conception in the north to the U.S.-Mexico border in the south. It has been estimated that prior to 1850 the coastline of SCB supported 20,000 ha of estuarine habitats composed of 40% vegetated wetland (e.g., salt-marshes), 25% salt- and mudflats, and 35% subtidal habitats (Stein et al., 2014). Since then, vegetated and unvegetated wetlands have radically decreased (-75% and -78%, respectively) while subtidal habitats have increased (5%; Stein et al., 2014). Estuaries and tidal wetlands in SCB are strongly influenced by the prevailing dry and arid Mediterranean climate of this region. Rain events are infrequent and mainly occur during winter (Dec. – Mar.) (Beller et al., 2014). Variations in dry/wet periods and subsequent material export to the coastal ocean are strongly influenced by annual and decadal cycles associated with the Pacific Decadal Oscillation (PDO) and the El Nino Southern Oscillation (ENSO) (Andrews et al., 2004; Beller et al., 2014). As a result, wetland habitats in this region are highly dynamic and vary substantially across space and time. Historically, many estuarine lagoons in this region naturally opened and closed to the adjacent coastal ocean depending on precipitation events, vegetation, sediment loads, tidal range and circulation, but has recently been actively managed to remain open (Elwany et al., 1998; Beller et al., 2014). Although significant efforts have been dedicated to the ecological history and status of SCB wetlands and estuaries, few studies have investigated their role in the coastal carbon cycle and little is known about the carbon fluxes to/from the atmosphere and the adjacent coastal ocean. Similar to estuaries and marshes on the US east coast, SCB estuaries are highly productive, but most likely act as sources of CO_2 to the atmosphere and net exporters of dissolved inorganic and organic carbon to the coastal ocean owing to input and decomposition of allochtonous carbon from surrounding land areas. Recent studies from lagoons and estuaries in the San Diego area all report consistent estuarine pCO_2 levels greater than atmospheric levels (Davidson, 2015; Paulsen, 2015; Paulsen et al., in prep.; SCCOOS: http://sccoos.org/data/oa/). Paulsen (2015; Paulsen et al. in prep.) assessed the seasonal export of DIC and TOC from a salt-marsh lagoon to the coastal ocean and approximated a total flux of 48×10^6 mol C yr^{-1} under baseline conditions, which increased to 62×10^6 mol C yr^{-1} owing to a winter storm event. More than 70% of the export of total carbon was comprised of DIC for most parts of the year. These authors also observed significant export of total alkalinity and estuarine waters entering the ocean having a pH similar or higher than the adjacent coastal ocean. It is currently unknown how these findings scale to other SCB wetlands and estuaries, and there is an urgent need for additional studies. In conclusion, the radical changes in wetland area that has occurred since the 19th century in this region imply that the role of SCB estuaries in the coastal carbon cycle has changed substantially over time. In addition, the dynamic nature of this region suggests that its role is also highly variable on seasonal and inter-annual timescales as well as between different wetlands and estuaries.

In Mexico along the 12,000 km shoreline has about 180 coastal lagoons and other estuarine areas, with about 10,000 km2 on the Pacific coast (Contreras, 1993; **Figure 15.2**).

In the coastal lagoons, located along the northwestern shores of Baja California, Mexico, the ocean is the most important external source of nutrients, organic and inorganic C (Hernández-Ayón et al., 2004; Camacho-Ibar et al., 2007; Hernández-Ayón et al., 2007a). Recurrent upwelling events provide nutrient-rich and high CO_2 levels to the coastal zone throughout the year. The semi diurnal tides, ranging between 1.0 and 2.4 m, play a key role in these systems, as they force new upwelled water into the embayment. While the importance of upwelled water, as a source of inorganic C, to the coast is being studied in coastal zones, few studies evaluate the fate of this inorganic C within the coastal lagoons (Camacho-Ibar et al., 2007; Hernández-Ayón et al., 2007a). The Pacific coast of the Baja California peninsula in Mexico, is one of the areas where the contribution of coastal C fluxes to the global synthesis of FCO_2, there is more information available (e.g. Hernández-Ayón et al. 2007; Ribas-Ribas et al. 2011; Reimer et al. 2013; Muñoz-Anderson et al. 2015). As a representation of Mediterranean lagoons from Baja, at San Quintin bay it is estimated that 85% at the oceanic end, acted as a source of CO_2 to the atmosphere due to the inflow of CO_2–rich upwelled waters from the neighboring ocean with high positive fluxes higher than 30 mmol C m-2 d-1. As a contrast, there was a net uptake of CO_2 and HCO_3- by the seagrass bed Zostera marina in the inner part of the bay, so the pCO_2 in this zone was below the equilibrium value and slightly negative CO_2 fluxes of -6 mmol C m-2 d-1. As of this bay, the NEP and ΔDIC positive values indicate that the Mediterranean bay used to be a net autotrophic system during the upwelling season. Based on mid to late summer observations, it was previously reported that San Quintín Bay is a net heterotrophic system, due to imports of labile phytoplanktonic C generated in the adjacent ocean during upwelling (Camacho-Ibar et al., 2003). Based on observations of nutrient dynamics during the upwelling seasons of 2004 and 2005 (Camacho-Ibar et al., in preparation). They suggest that this apparent contradiction can be explained based on a dependence of NEP in San Quintín Bay on upwelling conditions, including intensity and persistence.

South of Baja, in the oceanic region of the NETP off Mexico, the inorganic carbonate chemistry is a totally different scenario. The Oxygen Minimum Zones (OMZ) located south of Punta Eugenia of Baja California, and all over south of the Mexican Pacific area, are important sources of CO_2 when physical forcing brings subsurface water with high Dissolved Inorganic Carbon (DIC) to the surface. However, due to strong vertical stratification, the relationship between ΔpCO_2 at the air-sea interface and the oxycline/StSsW upper limit depth is weak (Franco et al., 2014). For example, during November 2009, the region was a weak source of CO_2 to the atmosphere (up to 2.5 mmol C m-2 d-1), while, during August 2010, a range of values were observed between -4.4 to 3.3 mmol C m-2 d-1. Strong stratification (> 1200 J m-3) prevented subsurface mixing of water from the OMZ to the upper layer also in the coastal. Results from the authors suggest that advection of surface water masses, reinforced by strong vertical stratification, controlled surface pCO_2 and air-sea CO_2 fluxes in these areas.

West/Pacific Coast Tidal Wetlands

The Pacific coast of North America south of Canada is a tectonically active coastal margin characterized by a narrow continental shelf and heavy surf. The coast is dominated by rocky headlands, broad sand dune complexes, sand beaches and spits. The extent of estuarine wetlands in 2010 along the Pacific Coast of the continental U.S. was estimated at 60,000 ha in 2010 (NOAA C-CAP 2015), while the extent of estuarine wetlands on the Pacific coast of Mexico between 1985 and 2000 was estimated at 890,000 ha (Contreras-Espinosa and Warner 2004). On the U.S. Pacific coast, areas of expansive estuarine wetlands are limited to the larger coastal estuaries, where major rivers enter the sea and where embayments are sheltered by sand bars or headlands (e.g., Coos Bay, Humboldt Bay, Elkhorn Slough, San Diego Bay). San Francisco Bay, which supports the largest extent of coastal wetlands along the Pacific Coast of North America, is a tectonic estuary: a down-dropped graben located between parallel north-south trending faults (Sloan 2006). In Mexico, high densities of coastal wetlands are found along the Pacific coasts of Baja California Sur, Sinaloa, Chiapas, and southern Oaxaca (Contreras-Espinosa and Warner 2004). These Mexican coastal wetlands are found in association with large barrier-island lagoon complexes where wave energies are reduced by headlands, offshore islands or the Baja California peninsula, and along the Gulf of Tehuantepec, where the continental shelf widens and intense off-shore winds originating in the Gulf of Campeche and moving south through the Isthmus of Tehuantepec limit fetch. Notably, hundreds of thousands of acres of brackish wetlands were formerly found in the Colorado River Delta (Sykes 1937; Leopold 1949). These wetlands were largely dewatered since the construction of dams on the Colorado River began in the 1930, and have been invaded by non-native tamarisk (Tamarix ramosissima; Glenn 1998). However, tens of thousands of acres of brackish wetlands have been incidentally created by water routing decisions in the United States and México (summing to 25,000+ ha), and these wetlands expand during years of higher than average precipitation, when runoff inputs exceed reservoir capacities (Glenn et al. 1992; Glenn et al. 1996). Additionally, relict intertidal wetlands in the Gulf of California remain (33,000 ha; Glenn et al. 1996). Environmental flows are now being returned to the Delta to help reverse habitat declines and recruitment of native wetland vegetation has resumed (Glenn et al. 2013). Assuming that published studies of soil C accumulation (0.79 Mg C ha-2 yr-1; et al. 2012; 3.0 Mg C ha-2 yr-1; Ezcurra et al. 2016) are broadly representative of conditions in the U.S. and Mexico, respectively, estimates of soil C sequestration by estuarine wetlands sum to 0.05 Tg C in on the Pacific coast of the U.S. and 2.67 Tg C yr-1 for the Pacific coast of México.

Although U.S. Atlantic and Gulf Coasts are known to support more organic-rich sediments, rates of C burial on the Pacific coast tend to be commensurately high to due to high rates of sediment accretion. Previous studies have reported accretion rates of 0.20 – 1.7 cm yr-1 in natural marshes along the Pacific Coast of North America (Jefferson 1975; Thom 1992; Patrick and DeLaune 1990; Cahoon et al. 1996; Watson 2004; Weis et al. 2001; Callaway et al. 2012), with many values at the higher end of this range. High rates of sediment accretion are a function of the active Pacific Coast margin: Pacific coastal watersheds tend be to high relief and support elevated erosion rates while providing limited opportunity for deposition of sediments along low-land floodplains (Walling and Webb 1983). This leads to high water column suspended sediment concentrations, often exacerbated by anthropogenic land use activities, such as agriculture, grazing, logging, development (Meybeck 2003), and thus, high coastal wetland accretion rates are common. High rates of sediment accretion are also known to promote high C burial rates: allochthonous organic C derived from upland sources is a sediment constituent (Ember et al. 1987); additionally organic C produced in situ is more quickly buried in the sediment anoxic zone in high accumulation environments (Watson 2004).

15.4.4 High-Latitude (Alaskan, Canadian, and Arctic) Estuaries and Tidal Wetlands

High-Latitude (Arctic) Estuaries

Salinity gradients are a defining feature of the Arctic Ocean well beyond the nearshore environment (McClelland et al., 2012), but climate change impacts are anticipated to have particularly profound effects near the land-sea interface. In many areas, nearshore ice conditions are changing (AMAP, 2011), and erosion of coastlines is increasing (Jones et al., 2009), and the duration and intensity of ocean acidification (OA) events are becoming more frequent (e.g. Fabry et al., 2009). In the Beaufort Sea, wave heights have reached unprecedented levels as fetch has increased with dramatic declines in ice extent. The urgency for improving our understanding of how these biophysical changes are interrelated, as well as their potential to impact society, biota, and the fate and transport of carbon, water and energy within the Arctic and beyond is well recognized.

In the Alaskan Beaufort Sea, lagoons, bounded by barrier islands to the north and Alaska's Arctic slope to the south, span over 50% of the coast. These lagoons link marine and terrestrial ecosystems and support productive biological communities that provide valuable habitat and feeding grounds for many ecologically and culturally important species. Beaufort Sea lagoons are ice-bound for approximately nine months out of the year; therefore, the brief summer open-water period is an especially important time for resident animals to build energy reserves (necessary for spawning and surviving for the winter months) and for migratory animals to feed in preparation for fall migrations.

Several relatively recent studies highlight these close linkages along coastal margins of the Arctic, especially how changes in sea ice extent can impact terrestrial processes (e.g. Bhatt et al., 2010), which can control coastal erosion (e.g. Aguirre, 2011) and the transport of carbon, water and nutrients to nearshore estuarine environments (e.g. Mathis et al., 2012; Pickart et al. 2013). In addition, recent work in the Beaufort Sea has demonstrated that nearshore waters are highly productive and resilient, sustaining year-round benthic invertebrate populations that support complex food webs (Dunton et al., 2012). Nearshore estuarine environments in the Arctic are critical to a vibrant coastal fishery (von Biela et al., 2012) and also serve as habitat for hundreds of thousands of birds representing over 157 species that breed and raise their young over the short summer period (Brown, 2006).

Arctic estuarine systems are primed to undergo dramatic state changes in functionality in the next few decades because of the enormous changes taking place on their landward and seaward margins. Shifts in carbon and nitrogen cycling, trophic linkages, and faunal diversity, driven by changes in freshwater inflow, coastal erosion, ice cover, and exchange with the coastal ocean promise to produce irreversible changes that effect the subsistence lifestyle of tens of thousands of native residents in the circumpolar Arctic.

High-Latitude (Arctic) Tidal Wetlands

High-latitude ecosystem C flux measurements tend to focus on for the abundant inland peatlands ,and thus much less is known about Arctic and subarctic tidal marshes. However, due to high sedimentation rates, continuous burial due to rising sea levels, and high C content, estuarine wetlands are estimated to sequester C at rates up to 10x higher per area than many other wetlands (Bridgham et al., 2006). Chmura (2003) estimated depths of 50 cm in calculations of depth, but some of these brackish marshes and uplifted/subsidence regions have deeper organic sediments. Very little research has been focused on the C dynamics in these vegetated coastal systems similar to what has been done at lower latitudes (Middleton and McKee 2001; Kristensen et al. 2008).

The Gulf of Alaska is characterized by very wide coastal shelves and high seasonality in very climatically sensitive region which has already experienced significant warming. The large impact of melting glaciers, including the Bering and the Malaspina piedmont glaciers (each the size of Rhode Island) will have an impact on sea level rise locally along with the global sea level rise. Many of these marshes are very fresh due to the large P-E regime of the Pacific northwest, as well as the high glacial meltwater that characterizes the region. River deltas such as the Yukon-Kuskokwim Delta are characterized by discontinuous permafrost and are difficult to access, but preliminary data in some of the permafrost area indicates sedge peat as deep as 3 meters with clear ice lenses.

One of the most important coastal Alaskan marsh system is the Copper River Delta, a critical habitat for migratory birds along the Pacific Flyway, which extends for more than 75 km and inland in some places as much as 20 km along the Gulf of Alaska. (Thilenius, 1990). While C storage estimates in these marsh locations is lacking, extensive research on the uplifted (and buried) peats by Plafker (l965; personal comm) indicate extreme events of subsidence and uplift (ie. yo-yo tectonics). Our examination of peat samples reveals marsh vegetation interspersed with intertidal muds, and freshwater coastal forest and moss peat, which extends to depths of greater than 7 meters. The 1964 earthquake raised the entire delta from 1.8 to 3. 4 meters (Reimnitz, l966). The role of Equisetum in these wetlands are critical to the nutrient cycling of Carex and shrubs such as Salix and Myrica, as they pump P and K from great depths (Marsh et al., 2000). It is clear that tectonic forces are the primary control on the C storage in these marshes, where sedges such as Carex lyngbyei (Lyngbyei sedge) and Eleocharis kamschatica (Kamchatka spikerush) are present, but the dynamic nature of the wetland with extensive sediment deposition from the Copper River ensures shrub growth of willows and shrubs, which contributes to the woody component of buried peats. Whether or not the areal extent of these wetlands will expand or decline with tectonic impact and sea level rise is dependent on many factors.

Hudson Bay Lowlands (HBL) tidal marshes are another understudied region of the north, as most of the HBL peatland storage is estimated (20% of the North American C pool) from the freshwater component (Packalen et al., 2014).

15.5 Indicators, Trends, and Feedbacks

All indications suggest that coastal environments are changing rapidly due to global and local-scale changes induced by human activities. The sustainability and quality of estuarine and intertidal wetland habitats, including their C sequestration services, is uncertain. Simulation models show the long-term sensitivity of coastal C fluxes to land-use and land-cover change, including land management practices while decadal and inter-annual variations of C export are attributable primarily to climate variability and extreme flooding events (Tian et al. 2015, 2016, Ren et al. 2015,2016).

Expected changes in climate for the remainder of the 21st century will likely have a major impact on C delivery to tidal wetlands and estuaries. Tidal wetland C dynamics are at extreme risk of change as a result of rapidly increasing rates of SLR, declining riverine inputs of sediments, a lowering of NEP as a result of a shifting balance between GPP and R that is temperature related, and coastal armoring, which limits wetland transgression (Kirwan et al. 2010, Morris 2016, Weston 2014, Hopkinson et al. 2012, Kirwan and Megonigal 2013). Salt water intrusion into oligohaline tidal wetlands will likely bring about substantial short-term increases in CO_2 emissions as additional terminal electron acceptors permeate freshwater sediments (Herbert et al. 2015, Neubauer et al. 2013, Weston et al. 2014). Sulfate inflows are also suggested to reduce CH_4 fluxes, despite potential for short term increases (Weston et al 2014). Seagrass C burial is also threatened as a result of eutrophication from ever-increasing N runoff from a variety of point and non-point sources (Bowen and Valiela 201). The loss of seagrass will be difficult to reverse even with extensive management intervention, as once the sediment stabilization by seagrass is lost, water clarity remains too turbid for seagrass re-introduction (ref). While terrestrial C loads will likely continue to drive system heterotrophy, extreme flooding events (Ren et al. 2016) might shunt material directly to the continental shelf, thus decreasing processing, transformation, and burial in the estuary. Overall it is likely that estuarine area will increase relative to that of wetlands (Fagherazzi et al. 2013, Mariotti et al. 2010, 2013) and estuaries will become more phytoplankton based relative to benthic algae and macrophytes (Hopkinson et al. 2012). By the end of the 21st century, it is quite likely that wetland and estuary net CO_2 uptake and storage as OC will be significantly reduced throughout the U.S.

The entire Gulf Coast is growing in population, an expected 144% from 1960-2010, and Texas has the second fastest growing coastal population out of the 5 Gulf States (Adams et al. 2004). There were five total states that had growth rates (averaged: 169%) that were greater than the total population growth (163%) of the United States (Adams et al. 2004). One of the two most affected areas was the coastal regions along the Gulf of Mexico, especially Louisiana which contains 40% of all the saltwater wetlands in the United States (Edwards and Proffitt 2003). Even in the absence of complete loss, fragmentation of a wetland can cause basic processes to fail and a decrease in biodiversity, leading to a delicate and weakened environment (Cline et al. 2007).

Examples of changes include:

- Areas of the Salish Sea and San Francisco Bay are strongly influenced by habitat change, human-caused nutrient and sediment budget changes, and shellfish cultivation, all of which have affected C stocks and fluxes of these estuaries significantly (Mauger et al. 2015).
- Anomalously warm ocean and atmospheric temperatures and river inputs during 2013–15 altered seasonal timing of biogeochemical and circulation dynamics in Puget Sound (Hartmann and/or Bond refs, PSEMP 2014, 2015, 2016 in press).
- Important calcifiers such as the non-native but economically important Pacific oyster, and pelagic pteropods (swimming snails important to juvenile salmon in the Gulf of Alaska) are known to have life stages sensitive to seawater pCO_2 levels and aragonite saturation states at present-day levels (Barton refs, Bednarsek refs).
- Information is rapidly and steadily emerging on new vulnerability to climate change and ocean acidification impacts. For example, Dungeness crab and other economically or ecologically important Pacific coast crustaceans and mollusks are known to be sensitive to levels of pH or aragonite saturation state that will increase in duration, extent, and intensity in estuarine waters in coming years (Miller et al. 2016, Hodgson et al. 2016, others).
- Seagrass production should be stimulated by higher CO_2 levels in seawater (ocean acidification) (Garrard & Beaumont 2014), which may overcome some of the consequences of higher projected temperatures (Zimmerman et al. 2015).
- Projected warmer sea surface temperatures and more acidified seawater in Pacific coast estuaries expected by 2050 are expected to increase growth rates, duration of favorable bloom conditions, and toxin production of various harmful algal bloom species (Fu et al. 2012?, Moore et al. 2015), which would likely have significant effects on population sizes of higher trophic level species (birds, mammals) [Cassin's auklet ref from 2015 HAB event or similar?].
- Projected increases in the frequency, duration, and severity of coastal and estuarine hypoxia and ocean acidification events (Bergamaschi et al. 2012 in Zhu et al USGS report, Hauri et al. 2013).

The neighboring ocean along the coastal lagoons, located along the northwestern shores of Baja California, who are under the influence of the California Current, is a typical wind-driven coastal upwelling system. The typical period of upwelling in the region is from April to August. In those areas, the seagrass Zostera marina is the dominant macrophyte, covering approximately 40% in lagoons such as San Quintín Bay surface (Ward et al., 2003). However, dense patches of the macroalga Ulva spp. are observed near the mouth during the upwelling season (Zertuche-González et al., 2009). During the peak of the upwelling season in spring and early summer, subarctic waters dominate off Baja California; by contrast, the influence of tropical and subtropical currents prevails during late summer and fall (Durazo

2009). The seasonality of upwelling conditions and its anomalies in the Northwest Pacific coast are influenced primarily by coastal winds, large-scale winds, and offshore and remote ocean conditions (García-Reyes and Largier 2012). Moreover, variations in upwelling conditions at the seasonal-to-inter-annual timescale in the CCS are linked to the variability in biogeochemical dynamics and biological components in coastal ocean ecosystems (Checkley and Barth 2009). During an anomalous warm oceanographic condition called 'the Blob', that occurred since late 2013 until at least late 2015 across the North Pacific basin (Hartmann 2014; Bond et al. 2015), the areas of the North Pacific have been as much as 5°C warmer than average, affecting regional weather and climate patterns (http://sccoos.org/projects/anomalies_workshop/). The impact of these regional climate variations and changing environmental conditions, in terms of trophic status and metabolic consequences in coastal ecosystems, is poorly understood (Testa et al. 2013).

According to Avila-Lopez et al. (In review), this oceanographic condition, for example, at San Quintin bay, led to a summer-like season (weak upwelling condition), which reached a maximum surface temperature anomaly of 2 ºC in September 2014. San Quintín Bay was close to equilibrium in 2014 (1.2 ± 1.9 mmol C·m-2·d-1) and served as a weak source of CO_2 to the atmosphere in summer months (6.0 ± 2.8 mmol C·m-2·d-1). The pattern of seasonal variations in C balance at San Quintín Bay appears to be linked to macroalgae metabolism, while the eelgrass community plays an important role in the annual whole-ecosystem metabolism. Net ecosystem production (NEP) switched seasonally between net heterotrophy and net autotrophy during the anomalous condition, with photosynthesis and respiration rates approaching a steady-state balance on an annual basis (0.1 ± 8.5 mmol C·m-2·d-1). Also, during anomalous ocean conditions, the NEP and FCO_2 estimates clearly pointed to a strong dependence of the San Quintín Bay on the upwelling conditions and benthic metabolism, where benthic C fluxes likely play a key role.

The capacity of estuarine wetlands to store C is a function of their continued integrity, which protects existing belowground C stocks (Hopkinson et al. 2012), and rates of sediment accumulation, as high rates promote C burial (Watson 2004; Kirwan and Mudd 2012). In contrast with coastal wetlands on the Atlantic and Gulf Coasts, sea level rise is less of a threat currently to the C sequestration capacity of Pacific Coast estuarine wetlands, due to lower rates of relative sea level rise and more robust sediment supplies. Long-term rates of sea level rise on the Pacific Coast tend to converge on ~1-2 mm yr-1 (NOAA 2016), with no evidence for recent SLR acceleration (Ryan and Noble 2007; Sallenger et al. 2012). Reported rates of sediment accretion measured for Pacific Coast estuarine wetlands generally range from 3+ mm yr-1, greatly exceeding contemporary rates of RSLR. In addition, Pacific coastal estuaries tend to have high water column suspended sediment concentrations, due high relief tectonically-active watersheds which support high rates of erosion and provide curtailed opportunities for floodplain sediment storage. In addition, estuarine wetlands in south San Francisco Bay survived subsidence caused by groundwater overdraft of 0.3-1.0 m during the early to mid-twentieth century, suggesting that wetlands – at least in South San Francisco Bay – were able to survive very rapid rates of RSLR exceeding 1 cm yr-1 (Poland and Ireland 1988; Patrick and DeLaune 1998; Watson 2004).

However, there are some indications that the outlook for Pacific coast wetlands with sea level rise might be not be so assured. Coastal wetlands at Elkhorn Slough in central California are drowning due at least in part due to increased flooding brought about largely by increased tidal exchange (Van Dyke and Wasson 2005). However, additional factors make this area of wetlands susceptible to loss, including low sediment inputs with loss of tributary rivers, low plant diversity, and high nutrient loads thought to be exacerbating edge loss (Van Dyke and Wasson 2005; Deegan et al. 2012). Also SLR rise rates might suddenly rebound from depressed values, meaning SLR rates might increase dramatically in coming decades (Bromirski et al. 2011).

One aspect of SLR that might challenge the continued integrity of Pacific Coast estuarine wetland C stocks include erosion and retreat of shoreward barrier spits. More open lagoon or river mouth structures will expose wetland edges to wave attack, and subsequent erosion and mineralization of C stocks. Alternatively, increases in tidal prism resulting from larger inlets are known to cause dramatic increases in tidal flooding, shifts in vegetation and marsh drowning (Van Dyke and Wasson 2005; FitzGerald et al. 2008; Smith 2009; Watson et al., in press), which also would lead to erosion and mineralization of C, in addition to a potentially smaller wetland footprint in coastal lagoons. Also, in the Pacific Northwest earthquakes are known to cause massive subsidence events (Atwater 1987; Bucknam et al. 1992): the combination of a subsidence or uplift event combined with changes in dune or spit integrity might cause similarly dramatic shifts in wetland extent and expose wetland deposits to erosion and mineralization.

Some previous work also suggest that sea level rise is expected to drive increases in coastal wetland extent as wetlands are able to persist in place and migrate inland (Kirwan et al. 2016). While empirical data supports this view in low slope coastal plain environments (Raabe and Stumpf 2016), but challenges this view in high slope coastal environments (Field et al., in press), transgression of estuarine wetlands is expected where not constrained by coastal development. To date, a remote sensing analysis has shown transgression of mangroves in Baja California Sur (México) (López-Medellín et al. 2011), and upland migration of wetland taxa in central California (Wasson et al. 2013), in association with flooding increases occurring in the highest part of the intertidal zone.

Altered coastal wetland C storage are expected to intensify with climate change due to changing precipitation patterns, warming temperatures, and physical forcings from extreme events. Perhaps the most pervasive environmental change occurring in Northern California north to Canada is changing salinity patterns. As temperatures increase and rain replaces snow in the headwaters of coastal estuaries (Knowles et al. 2006), earlier run-off peaks result in estuarine salinities that are depressed during winter months, but that are higher than long term averages through spring and summer (Knowles and Cayan 2002). As previous studies have suggested that salinity intrusion enhances organic matter mineralization (Weston et al. 2006) through increased availability of the terminal electron receptor (sulfate), estuarine wetland C storage may be negatively affected by increasing salinity. However, declines in methanogenesis with higher salinities (Poffenbarger et al. 2011) may offset declines in C storage with respect to the overall global warming potential of wetlands. The overall magnitude and impact of this change will be a function of current and future estuarine salinity patterns and the extent of wetlands subject to fresh, oligohaline, mesohaline, and polyhaline salinity regimes. Such impacts are unlikely to be significant in Pacific Mexico north to Northern California, where snowmelt is an non-significant contributor to estuarine salinity patterns.

Warming temperatures along the Pacific coast are likely to cause shifts in wetland C storage. As documented by litter mineralization measurements and soil C storage estimates along a gradient of soil temperatures, wetland C storage is are a strong function of soil temperature (Kirwan and Blum 2011; Crosby et al., in press) as microbial processing rates scale with temperature (Frey et al. 2013), and evidence suggests that climatic warming is amplified in shallow coastal waters (Roemmich and McGowan 1995; Oczkowski et al. 2015).

Lastly, increased intensity of frequency of storms and flooding events are likely to alter coastal wetland C storage patterns. Storms deliver a pulsed sediment deposition (Walters and Kirwan 2016), and previous work suggests that sediment accumulation in coastal wetlands in along rivers is highly dependent on flooding events (Watson et al. 2011; Gray et al. 2016). However, flood events are also linked with large scale channel avulsion events that reorganize the configuration of wetlands and distributary channels in such a way that C stored during the previous configuration maybe excavated and subject to mineralization. (Gray et al. 2016). Large events therefore may disrupt C burial patterns that have persisted for decades to centuries, leading to erosion and respiration of buried C.

Rising Temperatures, Altered Precipitation Patterns

Climatic changes affect the entire watershed, so the integration of small changes to terrestrial C cycling leads to a significant impact on the quantity, quality and seasonality of riverine inputs to MAB estuaries. Large MAB estuaries have long residence times and are strongly influenced by riverine input and internal biogeochemical cycling.

Sea-Level Rise

- More wetland flushing, Salinity intrusion and sulfate-enhanced respiration,
- Morphology shifts in shallow MAB estuaries: fewer barrier islands and increased tidal mixing,
- Enhanced sedimentation/C burial rate with SLR increase

Increased Storm Intensity

Mobilization of C stocks from terrestrial, wetland and estuary sediment that would otherwise remain stored during less-intense storms. Increase cycles of resuspension and deposition can enhance decay.

Ocean Acidification

- Changes in riverine input and circulation are probably more significant that OA in highly productive MAB estuaries with long residence times.
- Could increase net heterotrophy in estuary waters, but may be partially offset by autotrophic production elsewhere.

Linking cause and effect is difficult when coastal environments experience processes across variable timescales and spatial structure. Estuaries are complicated due to their presence at the intersection of so many contexts: past/future, land/ocean, managed/unmanaged.

Observational Approaches

A variety of observational approaches has been applied to study tidal wetland habitats and C fluxes and exchanges with the atmosphere and adjacent estuarine and ocean waters, however we currently lack a standardized, consistent methodology on wetland mapping, wetland C flux monitoring, and assessment. Wetland mapping, inventories and sampling efforts include the National Wetlands Inventory (USFWS), a national effort to map and classify the wetland resources in the United States (data updated at a rate of 2% per year), aerial photography, and high spatial resolution remote-sensing color infrared (CIR) imagery. LIDAR imagery has been applied to develop high-resolution digital elevation models for wetlands. Satellite optical (e.g., Landsat) and radar imagery has been used for mapping wetlands (**Figure 15.3**).

High resolution satellite ocean color observations can be used to examine wetland impacts on estuarine C dynamics and stocks, which combined with hydrodynamic models can provide information on lateral fluxes and wetland contributions to estuarine and coastal C budgets. Various ground-based approaches have been applied to validate mapped C stocks and inventories. Dated soil cores provide both quantification of carbon stocks and information on long-term rates of net carbon accretion or loss. Exchanges of CO_2 and CH_4 fluxes between the wetland-atmosphere and the estuary-atmosphere boundaries are typically measured using closed chamber systems. Deployment of automated water quality sondes and optical sensors at weirs draining tidal wetlands provides a method for continuous measurements of physicochemical and optical parameters that can be used as proxies for carbon.

Modeling Approaches

There are currently few regional hydrodynamic models that provide a realistic representation of the estuarine circulation that allows to capture some of the most important hydrological processes (e.g., wetting and drying) in tidal wetlands. With unstructured meshes that provide topological flexibility, the Finite Volume Community Ocean Model (FVCOM) (Chen et al. 2003) has been successfully applied to wetland-estuarine environments. Currently, there are no biogeochemical models that include accurate parameterizations for the sources and sinks that drive variability in C fluxes, amount, and quality at the wetland-estuary interface (e.g., allochthonous sources, photochemical transformation, viral lysis).

Key areas for further research and development are:

- to establish an unbiased, landscape-level sampling scheme to determine sediment carbon sequestration in tidal wetlands
- to develop networks for continuous measurements of wetland-atmosphere exchanges (CO_2 and CH_4 emissions) and wetland-ocean exchanges (dissolved and particulate C fluxes) and to better constrain these important fluxes.
- to develop new biogeochemical models that account for critical processes at the wetland-estuary interface

15.6 Societal Drivers and Impacts and Carbon Management

Societal drivers are at the heart of future projections. Estuaries are strongly influenced by land use, both directly and indirectly. Coastal wetlands are an actively managed landscape component, with increasing pressures from human and SLR drivers.

There is strong potential for coastal wetland ecosystems – tidal marshes, mangroves and sea grasses – to be integrated into climate change mitigation policy and financing mechanisms. To be viable for inclusion in policy or financing mechanisms (such as market or regulatory carbon financing), an ecosystem must be vulnerable to human-driven actions that will result in significant greenhouse gas emissions. Further, actions preventing or arresting significant greenhouse gas emissions (such as conservation, management or restoration) must be achievable.

These wetland ecosystems are characterized by high areal rates of carbon sequestration, low rates of methane and nitrous oxide emissions, and large soil carbon pools capable of oxidizing rapidly when disturbed and hence human activities resulting in the maintenance, degradation or restoration of these ecosystems have implications for greenhouse gas emissions. These large soil carbon reservoirs can become large CO_2 emission sources when disturbed through excavation or drainage (Pendelton et al. 2013). Although blue carbon ecosystems are often net sources of non-CO_2 trace gases, methane emissions are limited by sulfate-rich water, which favors sulfate-reducing bacteria over the methanogenic archaea they compete against for electron donors (Megonigal et al. 2006, Poffenbarger et al. 2012), and anaerobic conditions tend to reduce the ratio of N_2O:N_2 production compared to more aerobic conditions (Schlesinger 2009). Further, the longterm carbon sequestration and storage capacity of these ecosystems makes them particularly viable for climate change mitigation actions. Specifically, the capacity of these coastal wetlands to preserve carbon is unlimited provided that plant productivity remains intact as a carbon source, soils remain saturated to inhibit decay, and relative sea level rises gradually to create accommodation space for increasing soil stocks. The deep, carbon-rich soils that characterize these systems -- typically exceeding those of the most productive terrestrial habitats -- is evidence of the continuity of such conditions over thousands of years (Bridgham et al. 2006, Denato et al. 2010, McLeod et al. 2011, Fourqueran et al. 2012).

15.6.1 Carbon Policy

In 2013 the Intergovernmental Panel on Climate Change (IPCC) issued guidance on including seagrasses, tidal marshes and mangroves associated fluxes in national greenhouse gas inventories (IPCC 2013). Currently a number of countries, including the US, are in the process of implementing these guidelines. The inclusion of wetlands in national greenhouse gas inventories can support and accelerate implementation of policies favoring conservation and restoration of these ecosystems as countries work to meet climate change mitigation targets. Ongoing science is needed globally to support this inventory process including mapping and assessing current greenhouse gas fluxes in wetlands and determining the fluxes associated with specific human activities in these ecosystems. Largescale impacts in coastal wetland carbon inventories, such as that resulting from sea level rise or shifts in coastal hydrological patterns, must also be understood for their impacts on this accounting.

In the US, many federal policies require that impacts on ecosystem services be considered in the management, development or other use of coasts. While currently no federal statute, regulation, or policy specifically references the carbon held in coastal habitats, there are federal US statutes and policies which coastal carbon ecosystem services could feasibly be included in current environmental and ecosystem considerations (Pendelton et al 2013).

15.6.2 Carbon Accounting and Trading

Momentum has been growing to recognize the unique carbon cycling characteristics of coastal wetland ecosystems in order to secure financing for activities such as conservation, restoration and creation of wetlands. However, for such projects to be able to produce “carbon credits” that can attract financing through carbon trading markets, regulatory markets or other financing mechanisms, standardized broadly-applicable methodologies must exist for quantifying and monitoring the carbon and other greenhouse gas fluxes in these ecosystems. Such methodologies for granting carbon credits to activities in tidal wetland ecosystems have been approved under the American Carbon Registry and the Verified Carbon Standard (VCS). However, the success of these programs as a mechanism to mitigate climate change, and simultaneously enhance the many other ecosystem-services they provide, is hampered by significant knowledge gaps that can impact the certainty of the carbon flux value of specific actions and demand new research. For example, there is little understood about the fate of particulate organic carbon lost from eroding soils.

Carbon finance systems impose a range of constraints that must be satisfied in order to ensure that activities provide real climate mitigation. The requirement of *"additionality"* ensures that the greenhouse gas benefits of an activity would not occur without the actions to be supported through carbon financing. The VCS Methodology for Tidal Wetland and Seagrass Restoration (Emmer et al., 2015a; Emmer et al., 2015b) deems all wetland restoration activities in the United States as additional based on evidence that <5% of the total restoration potential that can presently be funded. *Leakage* requirements ensure that the activity does not lead to new activities outside of the project boundary that have negative ecological or greenhouse gas consequences. The VCS restoration methodology requires advocates to demonstrate that: (i) there are no relevant activities in the project area, (ii) relevant activities in the project area can continue at the same rate after the project), or (iii) the project is not given carbon credit for an activity that is displaced outside the project area. In the United States, there are relatively few activities in coastal wetlands that are likely to be displaced due to existing regulation that project tidal wetlands and the fact that regular flooding and salinity are s not compatible with common agricultural land use practices.

Conservation of existing stocks is potentially more important than restoring or creating new stocks, especially considering restoring the C content of the soil requires considerable time to gain equivalency to natural marshes (Edwards and Proffitt 2003), with some estimates of 20 years for surficial layers (Osland et al.) to thousands of years at depth. Even though salt marshes also require time to accumulate biomass and soil C (Craft et al. 1999), these systems can yield investors a continuing series of offset credits for many years into the future (Hopkinson et al. 2012; Chmura 2011).

All studies so far have been on small scales though. Since coastal wetlands can be so important concerning carbon, the increasing rate of wetland loss and degradation are concerning. The potential and current amounts of blue carbon in wetlands are needed for proper management and to understand the most important wetlands to protect.

15.7 Synthesis, Knowledge Gaps, and Outlook

15.7.1 Gaps in Knowledge of East Coast Estuaries and Tidal Wetlands

Observational and Modeling Scales: Diel, Seasonal, and Event-Based

The estimates of carbon lateral fluxes from wetlands were mostly based on discrete sampling events at monthly to seasonal intervals, with a sampling resolution from hourly to one half of a tidal cycle, leaving the majority of time unsampled, and thus requiring large interpolation between sampling events. Detailed water fluxes and budgets were often not considered. Such sampling strategies have been shown to produce substantial uncertainty in export fluxes (Downing et al., 2009; Ganju et al., 2012). The recent estimate of the DIC lateral flux from a pristine intertidal marsh on Cape Cod, MA with minute-scale resolution revealed that previous estimates of marsh DIC export may be several fold lower than what it actually is (Wang et al., 2016). It is thus reasonable to argue that previous studies may not fully resolve the export magnitude and temporal heterogeneity, which may be controlled by variability in both water flux and constituent concentration across time scales from minutes to tidal cycles to seasons. Future studies should be directed to capture appropriate variabilities of carbon exports from marshes in order to accurately define them. Observational and modeling efforts are both required to tackle this issue.

Spatial Variability in Burial Rates, Air-Water Flux

Other GHGs: CH_4, N_2O

15.7.2 Gaps in Knowledge of West Coast Estuaries

- Carbon cycle measurements have not been conducted in any Pacific coast estuary with sufficient resolution and duration to afford insight into short- or long-term changes associated with climate or other human-caused forcing.
- Observing and modeling gaps are largest in the Gulf of Alaska and Central American isthmus regions.
- Nearly no observational or modeling effort to date on methane and nitrous oxide in estuaries (subtidal).
- High-resolution models are just starting to incorporate carbon cycle biogeochemistry explicitly for estuarine ecosystems.
- Effects of current, short-term, and potential long-term human-caused changes (including climate and other human-caused drivers of change) in Pacific coast estuaries on carbon stocks and fluxes have not been quantified or projected with sufficient certainty to inform adaptation, mitigation, or policy planning.

In Mexican coastal lagoons, there is a lack of information on air-water CO_2 fluxes in coastal ecosystems, such as lagoons, where the few studies of carbon cycling have focused solely on the ecosystem metabolism (e.g., Carmouze et al., 1998; McGlathery et al., 2001; Hung and Hung, 2003). Moreover, very few studies have addressed CO_2 cycling in lagoons (Hernández-Ayón et al., 2007a; Koné et al., 2009). More information needs to be done in coastal lagoon in the tropical Pacific and Gulf of California of Mexico.

A better description of the CO_2 system in these regions is required. As a result, few information is available not only about CO_2 fluxes, but also about Ocean Acidification studies.

Air-sea exchange of climate reactive gases CO_2, CH_4, and N_2O in open waters of Pacific coast estuaries also suffer from inadequate attention for a systematic analysis. One study on the Columbia River estuary estimated that it was net neutral with respect to net annual atmospheric CO_2 exchange (Evans et al. 2012). A dissolved Columbia River estuary methane budget estimated losses of 42% to the atmosphere, 32% to the ocean, and 25% to methane oxidation (Pfeiffer-Herbert et al. 2015). Similarly, other relevant estuarine CO_2 or CH_4 studies stop short of generating quantitative budgets including gas exchange (for example PSEMP 2014, 2015, 2016).

15.7.3 Gaps in Knowledge of West Coast Tidal Wetlands

- Published studies on basic rates of wetland organic carbon storage are limited to estuarine marsh in the San Francisco Estuary (Callaway et al. 2012) and four lagoons in Pacific México (Ezcurra et al. 2016); further studies are needed.
- Trends in estuarine wetland extent along the Pacific Coast of the continental U.S. suggest that current rates of wetland loss due to coastal development in the U.S. are minor (CCAP 2012). However, mangroves have only been protected in Mexico since 2014; consistently gathered remote sensing analysis will be needed to assess the success of this new policy.
- Experimental research on the impacts of climate-forced changing salinity patterns on carbon and methane ecosystem exchanges are relevant for predicting changes in carbon stocks and global warming potentials of Pacific coast estuarine wetlands.

- Coupled research on wetland geomorphic change and carbon storage is needed where channel and wetland distributions are highly dynamic

15.7.4 Gaps in Knowledge Throughout the High Latitudes with Climate Change

- Both external vs. internal C stocks and fluxes…uncertainty will depend on hydrology, C exchange for the future
- Depths of brackish marsh peat along rivers – high uncertainty
- GHG temperature increase in coastal marshes will likely result in higher productivity and longer growing season, but whether decomposition will keep pace is an open question (Ellison, 2000)
- Uplift/subsidence for southern coastal region – unknown but has big impact
- Exchange of C, hydrology, location along tidal gradients\
- Rates of sea level rise are important – could increase, decrease, or maintain C sequestration along coastlines (McCloud et al. 2011) due to rates of sea level rise (erosion or burial), water temperature, and vegetation shifts, as well as migration inland.
- Quantification of present accumulation or loss of carbon in coastal wetlands
- Quantification of export of C to adjacent systems

Supporting Evidence

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Figures

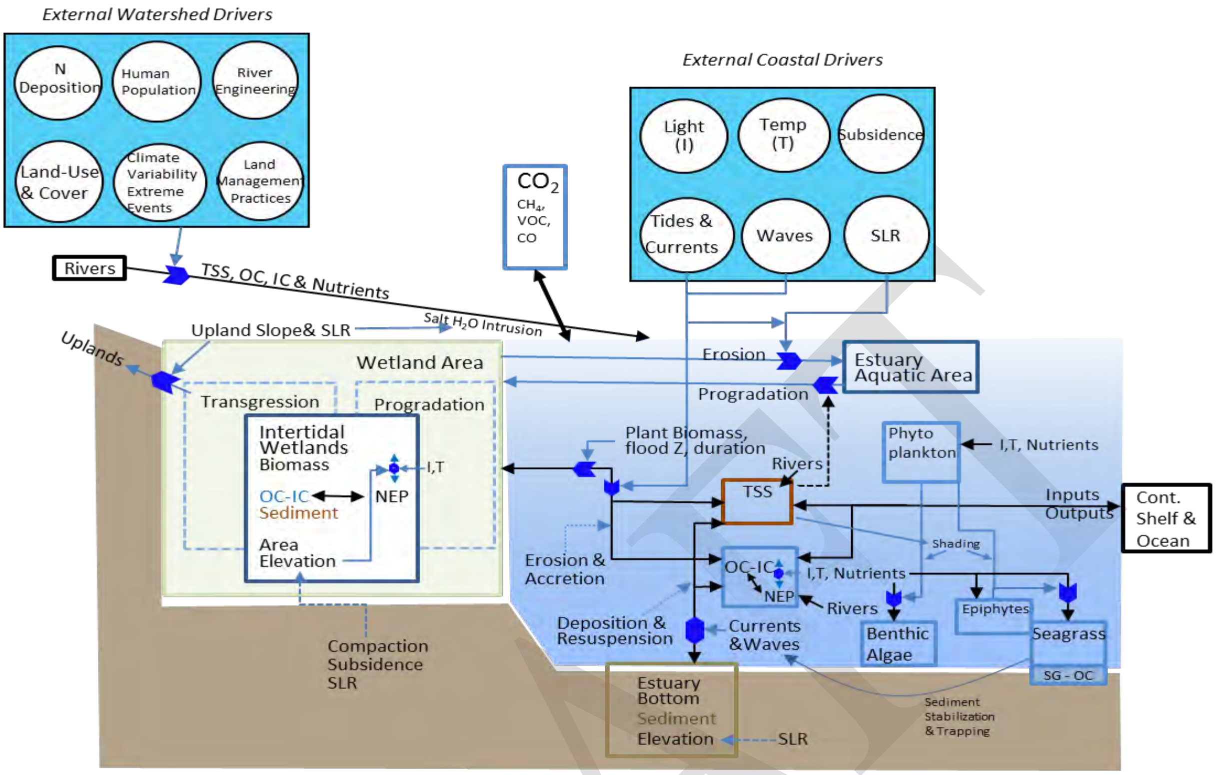

Figure x. Chapter 15. Conceptual model of coastal tidal wetlands and estuaries highlighting the linkage with adjacent terrestrial and oceanic systems and the drivers, processes and factors that largely control C dynamics. This model encompasses a range of temporal scales (e.g., millennial to daily) in an attempt to illustrate both long-term processes such as marsh progradation and accretion, medium-term processes such as trends in OC and sediment export from watersheds, salt water intrusion and eutrophication of estuarine waters that can alter the relative importance of planktonic vs benthic/seagrass habitats, and short-term processes, such as GPP, R, NEP, and storms that erode marsh shorelines or deposit large amounts of sediments onto marsh surfaces. The drivers that control watershed export and wetland/estuary dynamics are quite different, although events, such as storms, play can play a major role in both systems. OC – IC – NEP refer to the processes of GPP, R, and NEP and the link between inorganic and organic carbon dynamics. Elevation is an important variable because it changes over time and can thereby alter the flooding frequency and depth of wetland vegetation, which alters wetland production/biomass and the negative feedback that biomass play in marsh accretion, elevation gain and SLR. Compartments for intertidal wetlands, benthic algae, phytoplankton and seagrass have been simplified, so do not explicitly show GPP, R and NEP of the entire community present in these habitats. Modified from Hopkinson (CRC blue Carbon primer).

Figure 15.1. Conceptual model of coastal tidal wetlands and estuaries highlighting the linkage with adjacent terrestrial and oceanic systems and the drivers, processes, and factors that largely control carbon dynamics.

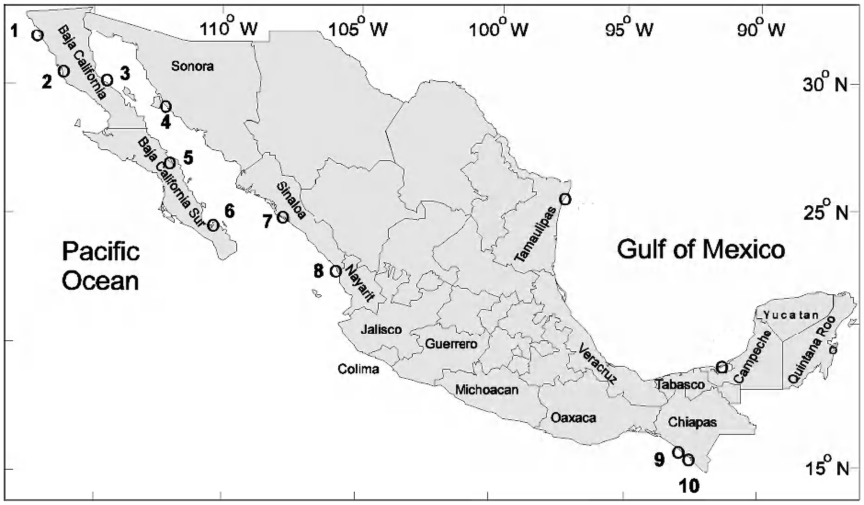

Figure 15.2 Key coastal lagoons along the Pacific Coast.

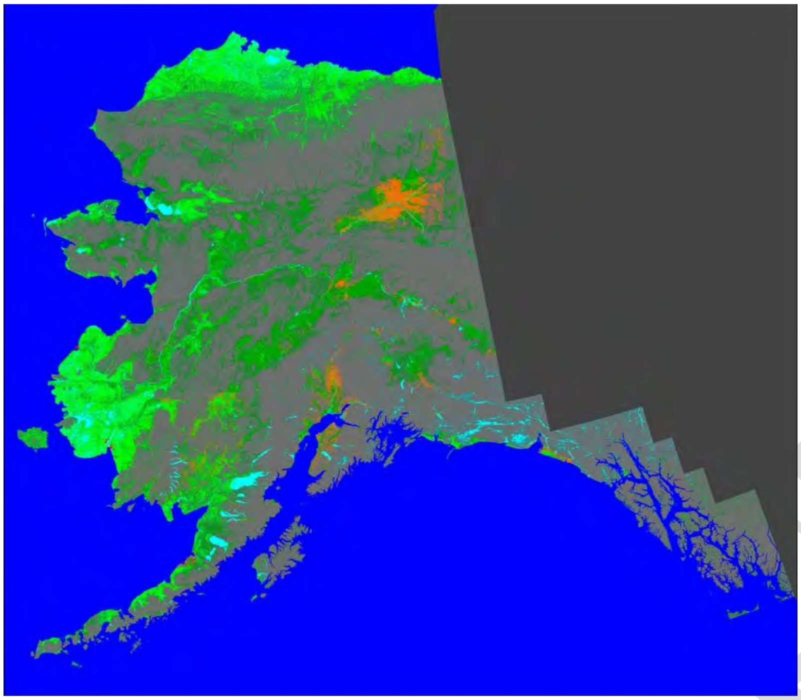

Figure 15.3 High-resolution map of Alaska wetlands using mosaics of Japanese Earth Resources Satellite (JERS-1) synthetic aperture radar (SAR) imagery (Whitcomb et al. 2009).