CHAPTER 4

Coastal wetlands: function and role in reducing impact of land-based management

G.L. Bruland
Natural Resources and Environmental Management Department, University of Hawaii Mānoa, USA.

Abstract

Coastal wetlands are among the most productive, valuable, and yet most threatened ecosystems in the world. They provide a variety of functions that reduce the impact of land-based management on the coastal zone such as slowing the flow of water from the mountains to the sea, trapping of sediments, and retaining or transforming nutrients. Numerous studies have reported that increased soil erosion and nutrient export from land-based management are threatening estuaries and coastal zones. Coastal wetlands are located at a critical interface between the terrestrial and marine environments and are ideally positioned to reduce impacts from land-based sources. There are various types of coastal wetlands including riparian wetlands, tidal freshwater marshes, tidal salt marshes, and mangroves. Some classification systems also consider seagrass beds and coral reefs to be wetlands. Coastal wetland ecosystems vary in their ability to reduce impacts from land-based management in both space and time. These wetlands can retain, and transform, or sometimes even act as sources of nutrients and sediments. Some wetland types are more effective at sediment retention and others at nutrient retention. Watershed size, climate, and position of wetlands in a watershed are other important factors that determine the effectiveness of coastal wetlands in reducing the effects of land-based activities. Wetlands do not appear to be infinite sinks for sediment or nutrients. Once critical sediment and nutrient loading thresholds have been crossed, coastal wetlands are subject to degradation and even loss. While many coastal nations have developed coastal-zone management policies and legislation, degradation and losses of coastal wetlands continue to occur due to altered hydrology, increased sediment and nutrient loading, urban development, agriculture, and aquaculture. While we have made significant progress in our ability to restore and create tidal marshes and mangroves, other coastal systems such as seagrass beds and coral reefs appear to

be much harder to restore. Thus, it is important that existing natural coastal wetlands be prioritized for conservation and that best management plans be developed to reduce sediment and nutrient losses from terrestrial watersheds.

1 Introduction and current status of coastal wetlands

Changes in terrestrial land-use patterns such as agricultural intensification and urban expansion have resulted in increased sediment and nutrient loadings that are transported into coastal areas [1–4]. For example, the change in nitrogen (N) loading to coastal areas since preindustrial times has increased fourfold in the Mississippi River, eightfold in rivers of the northeastern United States, and tenfold in rivers draining to the North Sea [2]. Human activities have increased sediment loads in rivers by 20% compared to preindustrial times [5]. However, reservoirs and diversions trap approximately 30% of the total sediment load in rivers from reaching the ocean. While the overall sediment load to the coastal zone has decreased by 10% [5], there are hotspots in places like the Philippines, Indonesia, and Madagascar where sediment loads to coastal zones have dramatically increased in recent years. Other pollutants such as herbicides and heavy metals, that are generated from land-based activities, have been shown to suppress photosynthesis in seagrasses and corals and suppress coral fertilization at concentrations of a few tens of parts per billions [5, 6, 7].

Coastal wetlands provide a critical interface between terrestrial and marine environments, and their importance to global sediment and nutrient budgets is much greater than their proportional surface area on earth would suggest [8]. It has been estimated that wetlands provide \$4.88 trillion (US) yr^{-1} in ecosystem services [9]. These ecosystem services include disturbance regulation, water supply, water quality maintenance, pollination, biological control, food production, and others. According to Costanza *et al.* [9], wetlands are 75% more valuable in terms of ecosystem services than lakes and rivers, 15 times more valuable than forests, and 64 times more valuable than grassland or rangelands. Despite the many ecosystem services that wetlands provide, they have been subject to conversion to other land uses for millennia and especially during the last 200 years.

Some of the best data on these conversion rates come from the United States. For example, it has been estimated that the state of California has lost over 90% of its original wetland area due to conversion to urban and agricultural land uses [10]. Iowa, Illinois, Indiana, and Ohio each have lost greater than 85% of their wetlands mainly due to conversion to agriculture [10]. In Florida, the Greater Everglades Ecosystem comprises less than half its original extent due to drainage and conversion to agricultural and urban land uses [11]. The state of Louisiana is expected to lose another 181,300 ha of wetlands in next 50 years or an area equal to the size of Washington DC./Baltimore metropolitan region [12]. Wetland losses in Louisiana continue to occur due to channelization of the Mississippi River, sea-level rise, herbivory from nutria (*Myocastor coypu*), storm surge from hurricanes, and impacts due to oil and gas production. Louisiana accounted for approximately 90% of the coastal marsh loss in the continental states during 1990s (not including Alaska).

On a more positive note, wetland loss rates in the United States have decreased over the last 30 years [13]. In the period from 1986 to 1997, 98% of wetland losses in the United States were from forested and freshwater wetlands, while only 2% of the losses were from estuarine wetlands [13]. While conversion rates of coastal wetlands have declined in the United States, globally conversion of coastal wetlands continues and in some places has even accelerated [5].

The coastal land at the continental margin accounts for less than 5% of the Earth's land area, yet 17% of the earth's human population lives within this zone [14]. Furthermore, approximately 4 billion people live within 60 km of the world's coastlines [15]. Table 1 lists the share of the total and coastal population that live within 50 km of different coastal wetland types.

Specifically, 27% of the earth's human population lives within 50 km of an estuary (Table 1). Coastal population densities have been estimated to be 100 people per square kilometer compared to only 38 people per square kilometer in inland areas [5]. This not only causes damage to coastal wetlands but also to adjacent seagrass beds and coral reefs (classified by some as wetlands and others as deepwater habitats; see the next section). Seagrass beds are currently threatened by physical disturbance, ship activities, dredging, landfill, erosion from terrestrial sources, growth of aquaculture, and eutrophication [16]. Large-scale declines in seagrass beds have been observed at over 40 locations, 70% of which were due to human-induced degradation [16]. Coral reefs are also in serious decline. Thirty per cent of all reefs are severely damaged and close to 60% may be lost by 2030; furthermore, it has been stated that there are no pristine reefs remaining [17].

The rest of this chapter will include information on wetland classification, compare and contrast different types of coastal wetlands, examine the coverage and position of wetlands in the watershed, explore the role of coastal wetlands in trapping sediment and retaining nutrients, compare soils of natural wetlands to created and restored wetlands in a case-study example, and identify future research needs and directions. Specifically, the major wetland classification systems will be identified and their classification of coastal wetlands will be discussed. This will be followed by comparing and contrasting the dominant types of coastal and deepwater

Table 1: Share of world and coastal populations living within 50 km of estuaries, mangroves, seagrasses, and coral reefs in 1995 [14]†.

Types	Human population [millions]	Share of world population [%]	Share of coastal population [%]
Estuaries	1,599	27	71
Mangroves	1,033	18	45
Seagrasses	1,146	19	49
Coral Reefs	711	12	31
Total ‡	5,596	–	–

† Based on spatially referenced population data.
‡ Due to overlap of some habitat types the figures do not add up to 100%.

wetlands including riparian wetlands, tidal freshwater marshes, salt marshes, mangroves, seagrass beds, coral reefs, and kelp forests. The coverage and position of wetlands in the watershed will then be examined in light of reducing impacts from land-based management. Next the methods for assessing sediment retention in coastal wetlands will be discussed followed by a summary of studies that investigated sediment retention. A similar section on methods for assessing nutrient retention and summary of nutrient retention studies will follow. This chapter will conclude with a case study that compares soils of natural wetlands to created and restored wetlands as well as a discussion of future research needs and directions.

2 Wetland classification

Before moving into a discussion of the function and role of wetlands in reducing the impacts of land-based management, it is important to understand that there are different definitions of wetlands and that various systems are used for their classification. Currently, the three most prominent hierarchical wetland classification systems include the U.S. Fish and Wildlife Service (USFWS) Classification System, the Canadian Classification System, and the Ramsar Convention System. The USFWS classification system, titled "Classification of Wetland and Deepwater Habitats of the United States" was published in 1979 [18]. This system includes both wetlands and deepwater habitats. The USFWS System defines wetlands as "Lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface." This definition does not include areas with permanent standing water greater than two m deep. Such areas would be defined as aquatic or marine habitats [18]. The levels of the USFWS System include *Systems*, *Subsystems*, *Classes*, and *Subclasses*. *Systems* are the broadest level of the classification scheme and include Marine, Estuarine, Riverine, Lacustrine, and Palustrine. Coastal wetlands are usually classified in the Estuarine *System*.

The Canadian Wetland Classification System defines a wetland as a "land that is saturated with water long enough to promote wetland or aquatic processes as indicated by poorly drained soils, hydrophytic vegetation and various kinds of biological activity which are adapted to a wet environment" [19]. Classification in the Canadian System is hierarchical and has three main levels: *Classes*, *Forms*, and *Types* [19]. The five *Classes* of wetlands in this system include Bogs, Fens, Swamps, Marshes, and Shallow Water Marshes. Coastal wetlands are found in all of the *Classes* except for Bog.

The Ramsar Convention defines wetlands as "Areas of marsh, fen, peatland, or water, whether natural or artificial, permanent or temporary, with water that is static, flowing, fresh, brackish, or salt, including areas of marine water where the depth at low tide does not exceed six m" [20]. This is a more inclusive definition of wetlands that incorporates coral reefs and seagrass beds that are not defined as wetlands in the USFWS or Canadian systems. The Ramsar System groups wetlands into *Classes* based on their location in the landscape and vegetation [20]. It has 32 *Classes* that are divided into marine/coastal and inland groups.

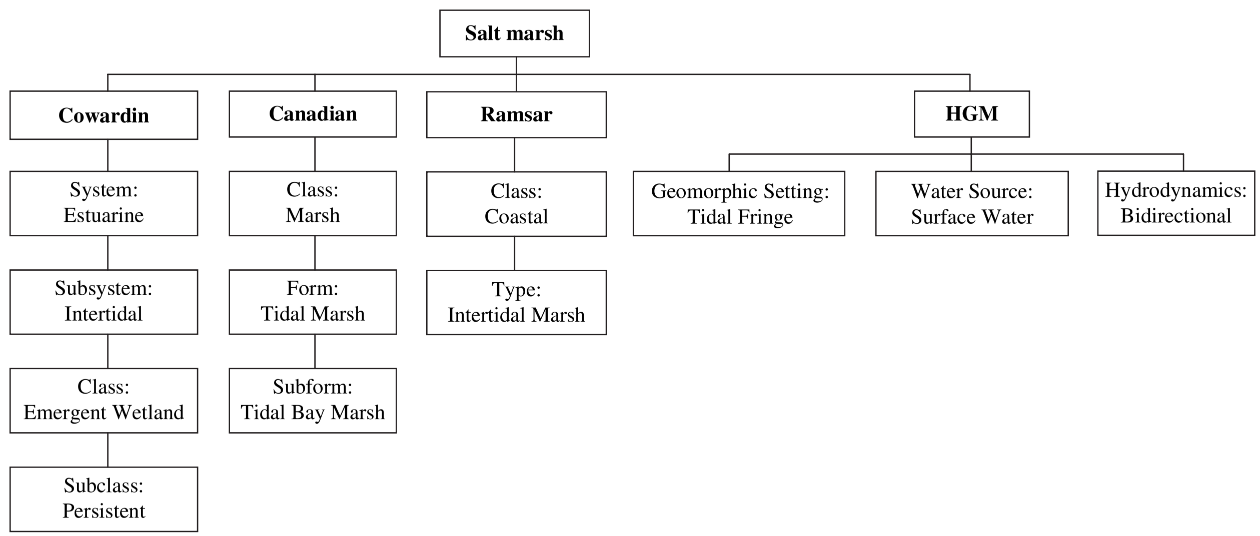


Figure 1: Classification of a common coastal wetland, the salt marsh, according to the Cowardin, Canadian, Ramsar, and Hydrogeomorphic systems.

The Hydrogeomorphic (HGM) Classification System [21] is functional classification system that is also worth mentioning. The HGM System is focused on evaluating physical, chemical, and biological functions of wetlands in the field simply, rapidly, and inexpensively [21]. The HGM system emphasizes two abiotic controls in maintaining wetland functions, hydrology and geomorphology [21]. Hydrology controls the amount, source, and season of water entering the wetland whereas geomorphology controls where the water comes from and whether or not it leaves. The HGM system includes 7 different *Geomorphic Settings*: depressional, riverine, lacustrine fringe, tidal fringe, slope, mineral soil flats, and organic soil flats [21]. It recognizes three *Water Sources*: precipitation, surface water, and groundwater as well as three types of *Hydrodynamics*: vertical, unidirectional, and bidirectional [21]. Coastal wetlands can have either riverine or tidal fringe geomorphic settings, all three types of water sources, and unidirectional (riverine) or bidirectional (tidal fringe) hydrodynamics. Figure 1 illustrates how the four systems mentioned above would classify a common coastal wetland, the tidal salt marsh.

3 Types of coastal wetlands

There are a variety of types of coastal wetlands that occur in different landscape positions, have different vegetative communities, and provide different functions in terms of reducing the effects of land-based pollution in coastal zones. Spanning a continuum from salt to fresh water, these include mangroves, tidal salt marshes, tidal freshwater marshes, and riparian wetlands (see Fig. 2a). Deepwater habitats include seagrass beds, coral reefs, and kelp forests. These ecosystems are often nested across the hierarchy of the larger estuarine-coastal system (Fig. 2b).

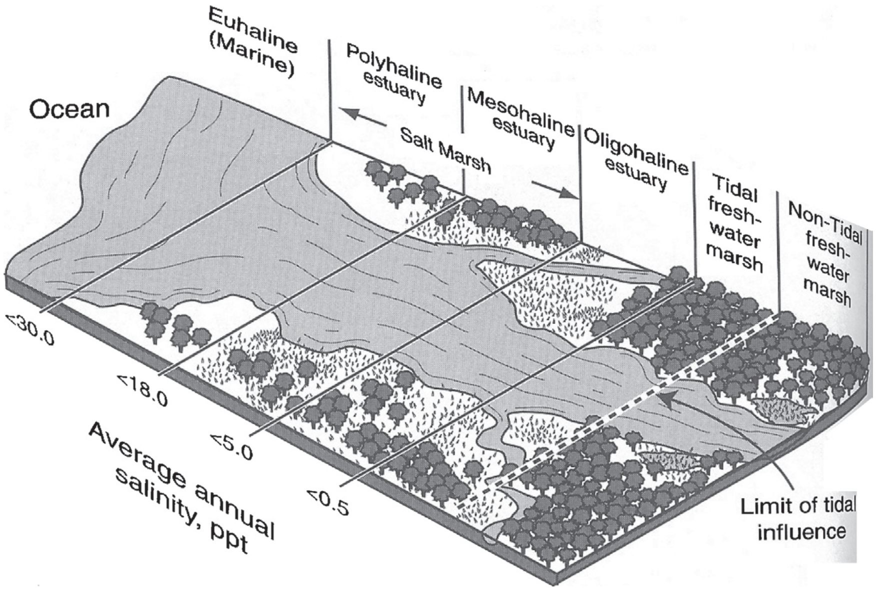
3.1 Riparian wetlands

Riparian wetlands occur as ecotones or interfaces between aquatic and upland ecosystems, have distinct vegetation and soil characteristics [22, 23], and perform important functions at the watershed scale [24]. Riparian wetlands are not easily classified or delineated but instead are comprised of mosaics of landforms and communities within the larger landscape [23]. Brinson *et al.* [24] stated that riparian wetlands are characterized by an abundance of water and fertile alluvial soils. They also list three major features that separate riparian wetlands from other types of wetlands and upland ecosystems that include:

1. Their linear form as a consequence of their proximity to rivers and streams.
2. Energy and material from the surrounding landscape [upstream watershed] converge and pass through riparian wetlands in much greater amounts than those of any other freshwater wetland systems.
3. Riparian wetlands are functionally connected to upstream and downstream wetlands and are laterally connected to upslope [upland] and downslope (aquatic) ecosystems [24].

Thus, riparian wetlands are dynamic, open systems that are subject to large inputs of surface water, sediments, and nutrients from forested, agricultural, and urban

(a)



(b)

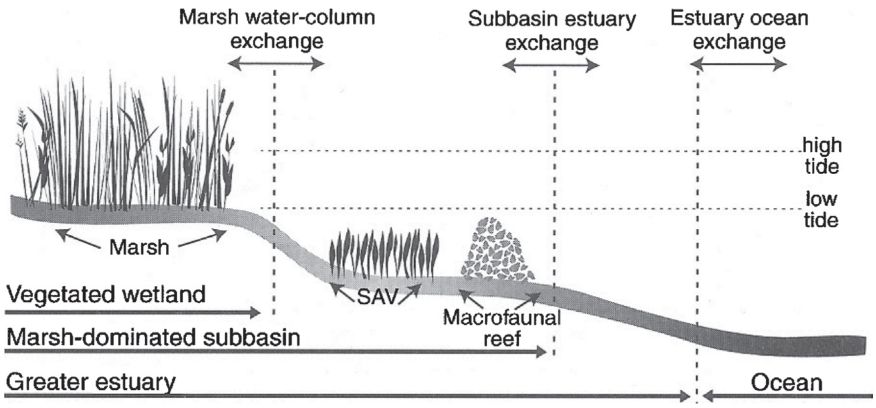


Figure 2: (a) The estuarine salinity gradient. (b) Hierarchy of the estuarine-coastal landscape. Copywrite (2000) from *Wetlands*, 3rd Edition by W.J. Mitsch and J.G. Gosselink [27]. Reproduced by permission of John Wiley and Sons, Inc.

areas upstream in the watershed. These riparian systems have the capacity to retain and transform large quantities of these inputs and keep them from being transported to coastal zones.

In the United States, the most extensive riparian wetland ecosystems are the bottomland hardwood forests of the Southeast. These bottomlands stretch across the Gulf and Atlantic coastal plains from Texas to Maryland and are associated with rivers such as the Mississippi, Appalachicola-Chattahoochee, Ogeeche, Altamaha-Ocmulgee, Pee Dee-Yadkin, Neuse, Tar-Pamlico, and Roanoke. Bottomland hardwood forests in the southeastern U.S. have been, and continue to be, converted to other land uses such as agriculture and urban developments. The Nature Conservancy [25] estimated that in 1991 about 2.0 million ha of bottomland hardwood forested remained in the Mississippi River Alluvial Plain; this area supported about 8.5 million ha of bottomland hardwood forest prior to European settlement. The Atlantic coastal plain that stretches from Florida to Maryland also has extensive areas of riparian hardwood forest lining many of the rivers that flow from the Piedmont to the ocean. Some of these riparian forests remain intact, others have been logged, others have regenerated, and many are in the process of being restored [26]. At the global scale there are various other forested wetland systems that are found in floodplains or riparian zones. The Amazon River floodplain in Brazil is an example of one of these types of systems as is the Zaire Swamps in Africa. These systems perform similar functions to the bottomland hardwood systems described above but are located in a tropical rather than a temperate setting.

Riparian wetlands located in temperate regions of the Western United States, or in tropical climates such as those in the Hawaiian and other Pacific Islands, are generally quite different from the bottomland hardwood forests described above. These riparian wetlands are located in steep, narrow, and dynamic riparian zones. While floodplain forests often have subtle changes in elevation and vegetation, the gradients in steep-sloped riparian wetlands are usually sharp and the visual distinctions between community types are clear [27]. These riparian wetlands have also been extensively modified by human activity [27]. Logging, grazing, conversion to agriculture and urban development have been widespread. These riparian areas are often the only flat lands available for cultivation and home building. They also tend to concentrate grazers due to the flat terrain and presence of water. Logging also continues to have a major impact on these wetlands. Often the streams in these riparian zones are used to move logs from the forest to estuarine holding pens. As many streams were too small to move logs efficiently, they were dammed and their banks were cleared to enhance logging and transport of timber [27]. This has lead to extensive erosion from upland areas and deposition in riparian ecosystems with subsequent transport of sediment into coastal zones. In some cases, destruction of riparian zones has resulted in floods and burial of natural estuarine habitats under tons of silt and enriched sediment [28]. The value of the ecosystem services provided by riparian wetlands and floodplains has been estimated to be \$19,580 ha^{-1} yr^{-1} [9], which is one of the highest values for any type of ecosystem.

3.2 Tidal freshwater marshes

Tidal freshwater marshes are close enough to the oceans to experience tides, but are above the reach of oceanic saltwater [27]. These coastal wetlands combine many of the features of salt marshes and freshwater marshes. They are similar in structure and function to salt marshes, with the major difference being a greater diversity in biota due to the reduction in salt stress. Plant diversity is high and more birds use these marshes than any other marsh type [27]. Often, the boundary between tidal freshwater marshes and salt marshes is difficult to determine.

Tidal freshwater marshes have been studied much less than salt marshes or inland freshwater marshes. Three major types of tidal freshwater marshes have been recognized: 1) mature marshes, 2) floating marshes, and 3) new marshes in prograding deltas [27].

In a cross-section, elevation in tidal freshwater marshes usually increases slowly from the stream edge to the adjacent upland areas. They typically have a slightly elevated levee along the stream bank where the overflowing water deposits much of its sediment load. The sediments in these marshes are fairly organic, especially in the floating marshes [27]. The European Union Habitats Directive has declared the conservation or coastal freshwater wetlands a priority [29].

3.3 Tidal salt marshes

Tidal salt marshes are found in coastal areas in the middle and high latitudes. They are common wherever accumulation of sediment is equal to or greater than the rate of land subsidence and where there is adequate protection from destructive waves and storms [27]. From afar, tidal salt marshes appear to be vast fields of a single species, often salt marsh cordgrass (*Spartina alterniflora*). While their physiognomy is much simpler than a bottomland hardwood forest, the vegetation of tidal salt marshes does vary across salinity and flooding gradients and provides habitat for a variety of plants, animals, microbes that are adapted to deal with the stresses of this environment. Numerous studies have shown salt marshes to be highly productive and to support the spawning and feeding of various marine organisms [27]. Thus, salt marshes represent a critical interface between terrestrial and marine ecosystems [27].

Chapman [30] divided the world's salt marshes into the following major geographical groups: artic, northern Europe, Mediterranean, Eastern North America, Western North America, Australasia, eastern Asia, Australia, South America, and the tropics. Although different plant associations are dominant in the different geographic groups, the ecological structure and function of salt marshes is similar around the world [27].

Salt marshes are predominantly intertidal and found in areas that are at least occasionally inundated at high tide but not flooded during low tide [27]. The upper and lower boundaries of these marshes are usually set by the tidal range. The lower boundary is determined by physical stresses such as depth and duration of flooding and the mechanical effects of waves, sediment availability and erosional forces [30, 31]. The upper boundary was thought to be set by the limit of flooding on extreme tides [32], but more recent research has indicated that the upper boundary is set by plant competition [31]. Based on elevation and flooding patterns, salt marshes can be divided into two zones: the high marsh and the low marsh [27]. The high marsh is flooded irregularly and can experience at least 10 days of continuous exposure to the atmosphere, while the low marsh is flooded almost daily and there are never more than 9 continuous days of exposure [27]. Competitively superior plants tend to dominate the high marsh habitats while stress-tolerant plants dominate the low marsh habitats [31].

Another prominent feature of salt marshes is the presence of tidal creeks. These creeks are often found in the low marsh. As the flow in tidal creeks is bidirectional, the channels tend to remain stable [27]. Generally, the banks of these tidal creeks are characterized by greater vegetative production than the interior areas of the marsh due to better flushing of salts and toxins from the tides, more oxygen in the soils due to higher elevation, and higher nutrient concentrations.

Tidal salt marshes are among the most productive ecosystems in the world, producing up to 80 metric tons ha^{-1} of plant material (8000 g m^{-2} yr^{-1}) in the southern Coastal Plain of North America [27]. They have also been estimated to be one of the most economically valuable ecosystems due to the many services that they provide. One study estimated the value of tidal salt marshes and mangroves to be \$9990 ha^{-1} yr^{-1} [9]. Unfortunately, due to increased construction of dams and reservoirs, sediment delivery to estuarine zones and salt marshes has decreased considerably, and in some cases large areas of salt marsh cannot keep pace with rising sea levels and are experiencing subsidence [5]. In the Southeastern United States, recent studies have shown that overharvesting of the blue crab (*Callenectes sapidus*) has led to explosive increases in the populations of the periwinkle snail (*Littoraria irrorata*) [33]. Freed of predatory control, the snails have engaged in destructive grazing of the salt marsh cordgrass and caused large-scale die-offs in salt marshes [33]. Thus, human alterations of salt-marsh trophic dynamics may decrease the ability of these systems to trap sediments and retain nutrients from land-based sources.

3.4 Mangroves

Mangroves swamps replace salt marshes along coastlines in subtropical and tropical regions [27]. In a few transitional situations (e.g. Florida) mangroves and salt marshes coexist. Global mangrove forest cover is estimated to be between 16 and 18 million hectares [34, 35]. In the region between 25° N and 25° S latitude, mangroves dominate approximately 75% of the world's coastline [36]. Mangroves are defined as areas of trees, shrubs, and other plants found in the intertidal zones and estuarine margins that have adapted to living in saline waters, continually or at high tides [37]. Mangroves are known for their seemingly impenetrable maze of woody vegetation and their unique adaptations to the double stresses of flooding and salinity [27]. Mangrove swamps provide many critical ecosystem services such as exporting organic matter to adjacent coastal zones, serving as nursery habitat for marine organisms, trapping sediments and nutrients, stabilizing shorelines, buffering land from storms, and providing safe havens for humans in the 118 coastal countries in which they occur [27, 34].

The frequency and severity of frosts are the main factors that limit the extension of mangroves beyond tropical and subtropical climates [38]. Mangroves are particularly dominant in the Indo-West Pacific region where they contain the greatest diversity of species [27]. Three main types of mangroves have been identified based on dominant physical processes and geomorphological characteristics: tide-dominated riverine mangroves, river-dominated riverine fringe mangroves, and interior basin mangroves [39]. Up to 95% of the detritus generated by the vegetation may be exported from riverine mangroves to adjacent estuaries and coastal zones, while only 21% of the detritus in basin mangroves may be exported [36].

Some of the most intact mangrove forests in the world are found in Malaysia and Micronesia. Recent studies have shown how important these ecosystems are to the local economies in these countries [39]. The importance of these ecosystems is related to the goods and services they provide, such as trapping of sediment, processing of nutrients and organic matter, providing food and habitat for animals, protecting shorelines, and providing plant products such as fuelwood, building materials, woodchips, tannins, honey, and medicinal products [39]. Mangroves are rapidly being converted to other land uses (i.e. urban development, rice fields, oil palm plantations, and aquaculture sites) in this region at an average rate of 1% per year [40]. In some countries, greater than 80% of the original mangrove cover has been lost due to deforestation [34]. Currently, the conversion to aquaculture accounts for 52% of mangroves losses, forest use accounts for 26% of losses, and freshwater diversions account for 11% of losses [35]. The value of ecosystem services provided by mangroves has been estimated to be \$9990 ha^{-1} yr^{-1} [9].

The Hawaiian Archipelago provides an interesting example of mangrove ecology and management. Hawai‘i has no native mangrove species despite having both suitable climate and geomorphic settings [41]. However, since their introduction in 1902, mangroves have flourished to such a degree that many people have been concerned about their impacts especially their dramatic effects on native plant community structure [42]. Expensive projects have been undertaken for their removal [41] with varying degrees of success.

3.5 Seagrass beds

Seagrass beds are defined as areas with aquatic flowering plants that live fully submerged in saline waters with a sediment substrate [36]. They occur over soft sediments worldwide, from the tropics to the boreal margins of every ocean [16]. In higher latitudes, eelgrass (*Zostera* spp.) forms dense meadows, while in the tropics, manatee grass (*Thalassia testudinum*) and turtle grass (*Syringodium filiforme*) are dominant [5]. Seagrasses are estimated to cover about 0.1–0.2% of the global ocean [43]. They are adapted to continuous and complete immersion in saline water, and as such, they grow in tidal zones although some can also grow in intertidal waters [16]. Seagrasses provide important ecosystem functions such as biological productivity, habitat for various marine organisms, trapping sediment, carbon sequestration, and buffering of wave action [36, 44]. The combined productivity of seagrasses and associated algae make seagrass beds among the most productive ecosystems on earth [45]. Seagrasses are also ecosystem engineers that provide physical structure that transforms featureless sediment bottoms into diverse and complex habitats for coastal biota [44].

As with salt marshes, the presence of a few plants has the effect of slowing water movement and permitting the settlement of even more sediment [36]. Since seagrasses grow submerged in seawater, they are more likely to be light-limited than salt marsh or mangrove vegetation [36]. Often, the distance to which seagrasses extend from the shoreline is related to the slope of the sea floor. When adequate sediment is present, seagrasses will extend seaward to the depth at

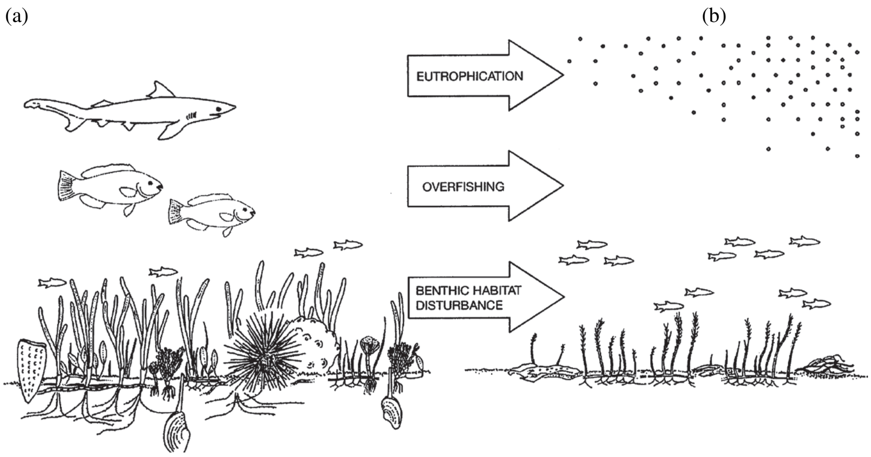


Figure 3: Comparison of a pristine (a) and a degraded (b) seagrass bed. Copywrite (2006) from J.E. Duffy [44]. Reproduced by permission of Inter Research.

which annual light flux is just sufficient to support a balance of photosynthesis over respiration [36]. However, if the water becomes more turbid due to increases in suspended sediment loads, light penetration is reduced, and the depth at which seagrasses can survive will also be reduced [46]. In undisturbed conditions, the maximum depth at which seagrass beds are found is about 30 m [47].

Increased sediment and nutrient loading from land-based sources are major threats to seagrass beds. Increased siltation is a particularly acute problem in coastal zones of Southeast Asia that receive the high amounts of sediment delivery as a result of soil erosion caused by extensive deforestation and changes in land-use in this region [48]. Under high nutrient loading from watersheds dominated by agricultural or urban land uses, growth of epiphytic algae on the seagrasses or blooms of phytoplankton in the water column can lead to decreased light availability to seagrasses and reduction in their extent [49]. Figure 3 shows the difference in structure of a pristine and a degraded seagrass bed.

The majority of remaining seagrass beds are found on the coastline of tropical countries, most of which are experiencing rapid rates of land-use conversion and environmental degradation [43]. The Pacific Islands are the only region where losses of seagrass beds are expected to be lower due to strict enforcement of zero-loss policies and small populations relative to the region's coastline [43]. An economic evaluation of ecosystem services provided by seagrass and algal bed ecosystems was quite high, estimated at \$19,004 ha^{-1} yr^{-1} [9].

3.6 Coral reefs and kelp forests

Coral reefs occur in tropical coastal waters with a minimum temperature of 18 ^{o}C, suitable light conditions, and high salinity [5, 36]. They can occur in association with tropical seagrass beds (as they do in coastal areas such as Florida Bay, USA or

Shark Bay and the Gulf of Carpenteria in Australia) and removed from them. Coral reefs are particularly abundant where sediment loading and freshwater inputs are minimal [5]. Thus, coral reefs are quite susceptible to changes in terrestrial land use and losses of riparian wetlands, tidal marshes, mangroves, and seagrass beds that can trap sediments from land-based sources. Major areas of coral reefs occur in the Pacific and Indian Oceans and the Caribbean Sea. Three main types of coral reefs have been defined [36]:

1. fringing reef: a reef found growing as a fringe attached to a land mass.
2. barrier reef: a reef that occurs at some distance out to sea and creates a shallow lagoon between the reef and the land.
3. atoll: an isolated structure surrounded by deep water that tends to form a ring of coral with a central lagoon (Fig. 4).

Coral reefs also provide a number of ecosystem functions such as serving habitat for a tremendous diversity of marine species, supporting coastal fisheries, and protecting coastal areas from storms and marine erosion [36]. Most coral reefs occur along the coasts of developing countries where the most intensive coastal degradation is occurring [50]. Coral reefs are at risk from global change processes such as bleaching and sea temperature increases, as well as various human activities such as coastal development, overfishing, sediment and sewage inputs that lead to eutrophication, dumping of debris and toxic wastes, and oil spills [5]. One study has suggested that all current coral reefs will disappear by 2040 due to warming sea temperatures [17]. Despite the fact that they provide habitat for a tremendous diversity of species, the valuation of ecosystem services provided by coral reefs was estimated to be \$6075 ha^{-1} yr^{-1} [9], which is less than one third of the amounts estimated for swamps/floodplains and seagrass/algae beds.

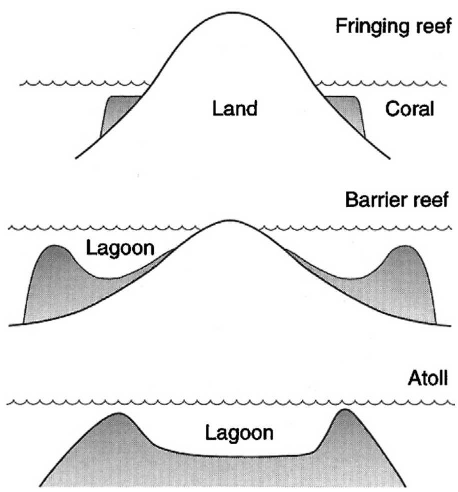


Figure 4: Types of coral reefs. Copywrite (2000) from *Ecology of Coastal Waters* by K.H. Mann. Reproduced by permission of John Wiley and Sons, Inc.

A temperate counterpart to the coral reef is the kelp forest. Kelp forests are temperate marine ecosystems dominated by large brown algae (*Macrocystis* spp.). They are characterized by high productivity and diversity [36]. For example, large brown algae can grow 45 cm d^{-1}, and extend to 60 m in length. Kelp forests are also remarkably resilient to disturbances from wave impacts, storm surges, and other extreme oceanographic events [51].

4 Wetlands in different types of watersheds

Wetlands in temperate versus tropical watersheds can differ in their ability to reduce impacts from land-based activities (i.e. logging, agriculture, urbanization) on sediment and nutrient loading to the coastal zone. Plant uptake and microbial activity in temperate and boreal wetlands has been shown to decrease dramatically, if not totally, during the cold winter months [52]. In contrast, plant uptake and microbial immobilization may occur throughout the year in tropical wetlands. However, the intense precipitation that is often experienced in tropical watersheds has been shown to be much more erosive and generate greater sediment export from terrestrial areas [53]. For example, on the island of Maui (Hawai‘i), erosivity can vary from less than 100 to greater than 1800 erosivity units over a span of 20 km [54]. Furthermore, in these small tropical watersheds, the terrestrial and marine environments are intimately connected. Here, human land-use activities are quickly translated to coastal areas because of high amounts of rainfall (sometimes over 5000 mm yr^{-1}) and steep stream gradients [55]. Rainfall on the ridgetop can result in increased surface water inputs to the estuary and coral reefs in a matter of hours. Thus, all lands may be considered coastal in the Pacific Islands [54]. This is quite a contrast to the Mississippi River Watershed that covers about 40% of the continental U.S. and for which it would take many days for precipitation in the headwaters of the upper Midwest to reach the rivermouth at the Gulf of Mexico.

Furthermore, many tropical watersheds are located in areas with steep topography and subject to intense land-use conversion. The combination of erosive rainfall, steep topography, and conversion of forests to agriculture and rangeland land has lead to massive increases in sediment transport through tropical watersheds. The coastal wetlands in these watersheds may be unable to handle these large fluxes of sediments and nutrients, or are themselves subject to conversion to more intensive land-use such as agriculture or aquaculture.

5 Coverage and position of wetlands in a watershed

In a study in the MidAtlantic United States, Novitski [56] found that when the per centage of the watershed in lakes and wetlands dropped below 10%, there were rapid increases in flooding. Thus watersheds appear to have critical thresholds of wetland area for flood control and most likely also for sediment and nutrient retention. The position of a wetland in the watershed also influences its ability to retain sediments and nutrients. Faber *et al.* [57] delineated three main watershed zones,

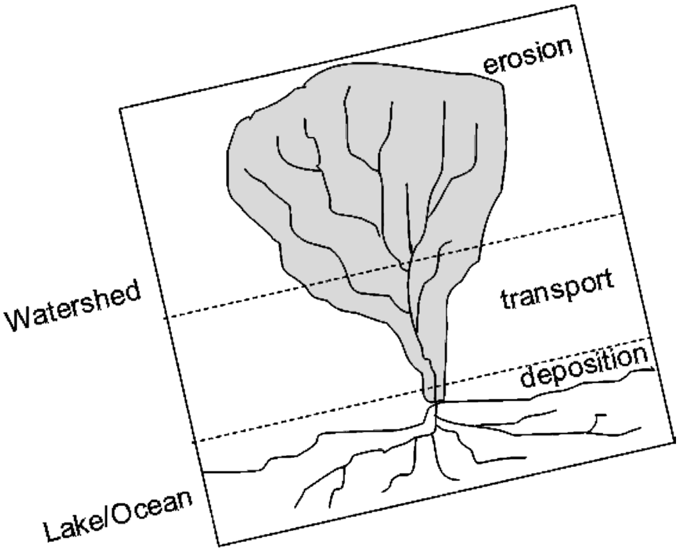


Figure 5: The erosional, storage and transport, and depositional zones of a watershed. After [57].

the zone of erosion, the zone of storage and transport, and the zone of deposition (Fig. 5). Most coastal wetlands are located in the zone of storage and transport (riparian wetlands, tidal freshwater marshes) or the zone of deposition (tidal salt marshes, mangroves, seagrass beds). As the human activity in a watershed increases, wetlands are often converted to urban, agricultural, or other land uses.

These changes often reduce the ability of the wetlands in the watershed to store water, sediment and nutrients.

Another interesting assessment of the position of wetlands in the watershed revealed that overbank flooding increases in importance with increasing stream order (Fig 6a), while overland flow is more important in headwater streams (Fig. 6b) [58].

Assuming that riparian transport is the most critical step in water-quality improvement of nonpoint-source runoff, Fig. 6c suggests that more emphasis should be placed on avoiding impacts to wetlands associated with lower-order streams than those associated with higher-order streams [58]. A given area of disturbance to these headwater wetlands will affect a greater proportion of the watershed than the same disturbance to a floodplain wetland. In other words, headwater riparian wetlands are converted to agriculture or other more intensive land uses, losses of sediments and nutrients are felt throughout the entire downstream watershed. Thus, it has been argued that it is not only the surface area of wetlands in a watershed, but also the length of wetlands along the streams that matters [58]. The large differences in average length with increasing stream order (Table 2) indicate that the riparian wetland length may be a better index of water-quality maintenance than wetland area.

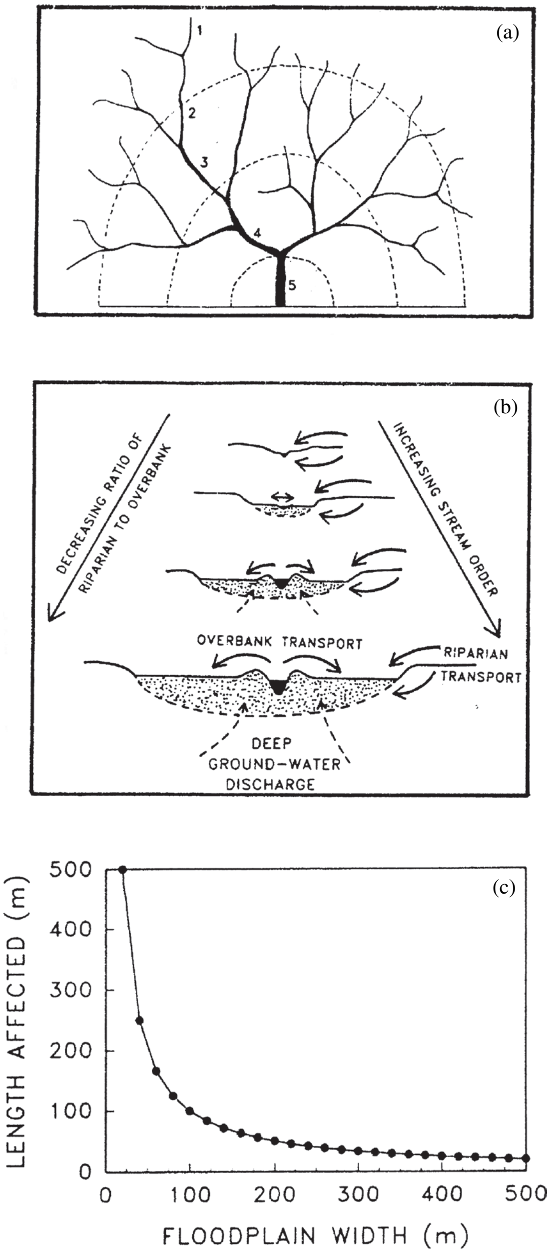


Figure 6: (a) A stream drainage network following the nomenclature of Strahler [174]. (b) Cross sections of riparian wetland floodplains showing how riparian transport and overbank flooding vary with Strahler stream order. (c) Change in length of floodplain affected by 1 hectare of disturbance as a function of floodplain width. Modified from [58] with permission of *Wetlands.*

Table 2: Relationships between stream order and other dimensions of stream configuration. First four columns are from Leapold *et al.* [173] and last two columns are from Brinson [58].

Stream order	Number	Average length [km]	Total length [km]	Estimated floodplain width [m]	Floodplain Surface Area [km^2]
1	1,570,000	1.6	1,526,130	3	7578
2	350,000	3.7	1,295,245	6	7771
3	80,000	8.5	682,216	12	8187
4	18,000	19.3	347,544	24	8341
5	4200	45.1	189,218	48	9082
6	950	103.0	97,827	96	9391
7	200	236.5	47,305	192	9082
8	41	543.8	22,298	384	8562
9	8	1250.2	10,002	768	7681
10	1	2896.2	2896	1536	4449

Ultimately, wetlands are needed in both the headwater and the depositional zones in terms of habitat and biodiversity but also in terms of flood control, water quality, sediment retention, and biogeochemical cycling. However, to meet the goal of improved water quality from a reduction in nonpoint-source pollution from land-based sources, restoration of wetlands along lower order streams may be the best strategy [58].

6 Methods for quantifying sediment accumulation in coastal wetlands

Short-term (monthly to annual rates) sediment accumulation in wetlands has been quantified by a combination of techniques that include horizon markers and sediment traps. Horizon markers have been used to quantify sediment accumulation in various wetland types [59–61]. This method involves laying down marker horizons at various points in the wetland. Feldspar, a white material composed of silt- and clay-sized particles [60], is commonly used as a marker horizon, although other studies have employed glitter or sand [62]. Sampling consists of taking a small core of sediments from the surface, down through the marker horizon, and measuring the thickness of sediments deposited above these highly visible marker horizons. A specialized coring device called a cryocorer can be used to collect the cores [63]. The amount of sediment that has been deposited on top of the marker corresponds to the sediment accumulation. Unfortunately, this method only estimates sediment depth and not the mass of sediment per unit area.

Mass of sediment per unit area can be measured with sediment traps. Sediment traps consist of tiles, petri dishes, or plastic containers that are deployed on the soil or sediment surface [64–67]. The material that collects on the trap can be quantitatively removed, weighed, and even analyzed for chemical composition.

Long-term sedimentation rates in wetlands have been quantified using radioisotopic dating techniques. Both ^{137}Cs and ^{210}Pb profiles have been shown to be effective for these purposes [68–70]. This process involves collecting deep soil cores (to bedrock when possible) and sectioning them into fine (i.e. 2 cm) intervals. The ^{137}Cs activity in each sample can be measured with a germanium detector. Peak concentrations of ^{137}Cs levels within each core correspond to peak fallout levels from atmospheric nuclear weapons testing in 1964 [69, 71]. The sediment located above the ^{137}Cs peak is equal to the amount of sediment that has been trapped in these wetlands since 1964. ^{210}Pb levels can be used to estimate sediment accretion rates over an even longer time intervals, up to 100 years before present [70–72]. Excess ^{210}Pb has been shown to accumulate in depositional environments from both atmospheric deposition and sedimentation [70, 73]. ^{210}Pb is considered more reliable than ^{137}Cs because it is polyvalent, and thus bound more tightly to mineral and organic soil particles [74].

7 Role of coastal wetlands in trapping sediment

In an analysis of eight temperate watersheds containing wetlands, Phillips [75] found that less than 65% of the sediment eroded from upland areas was transported out of the watersheds. Of the sediment that reached streams within the watersheds, 23–93% was retained by wetlands through which these streams flowed [75]. Thus, in watersheds that have sufficient wetland surface area and stream lengths associated with wetlands, sediment transport to coastal zones may be low. However, as watersheds develop, construction activities and the increase in the amount of impervious surface in the watershed have been shown to cause accelerated silt loading to neighboring estuaries [15, 76]. A number of studies in the tropics have reported that increased soil erosion in terrestrial watersheds is threatening estuaries and coastal coral reefs [77–81]. The steep topography, intense precipitation, and extensive land-use changes in these watersheds leads to high sediment loads that appear to have exceeded the sediment retention capacity of the coastal wetlands. When this occurs, sediment deposition within estuarine and reef zones can smother adult corals, kill juvenile corals, and prevent larval recruitment [81].

Mangroves have been shown to initially play a passive role in sediment accumulation [27]. Once mangrove vegetation has been established, it then acts to prevent erosion and trap sediments. The stems and leaves of mangrove and salt marsh vegetation slow water velocity and promote sediment deposition, roots and rhizomes increase the stability of the sediment, algae help trap fine sediments, oyster colonies modify the flow of water and sediments, and macroinvertebrates trap suspended detritus [27].

A study of a tropical watershed in Palau, Micronesia reported the mangrove fringe zone trapped about 44% of the riverine fine sediment flux, which was still not enough to prevent degradation of the associated coral reefs [79]. Another study on the island of Moloka'i (Hawai'i), reported that turbidity was lower on coral reefs adjacent to mangroves than on reefs with no adjacent mangroves [82]. A study of the Heeia Swamp in Kaneohe Bay, on the island of Oahu (Hawaii), reported that 10 cm of sediment was deposited in 16 months in the mangrove areas [83].

8 Methods for quantifying nutrient retention and transformation in coastal wetlands

A number of processes must be considered when quantifying nutrient transformation and retention in coastal wetlands. Plant uptake can be determined by harvesting plant tissue throughout the growing season and analyzing plant tissues for N and P concentration [36]. Microbial immobilization is difficult to quantify but can be measured in laboratory dosing studies and with radioisotopic techniques for P [84].

One of the dominant N transformation processes in coastal wetlands is denitrification. This process can be measured with *in-situ* core techniques but there are many difficulties associated with these methods due to the high background concentration of nitrogen gas in the atmosphere. The denitrification enzyme activity (DEA) [85, 86] is commonly used as an index of denitrification potential. The DEA is useful for site comparisons because it offers a method by which the denitrification potential can be compared across different soil types [87]. The DEA involves amending sieved, field-moist soils with solutions of glucose and potassium nitrate to ensure nonlimiting substrate conditions, and chloramphenicol to inhibit protein synthesis. The resulting slurries are made anaerobic by repeated flushing with N_2 gas. The anaerobic slurries are then injected with acetylene to inhibit N_2 production [85] and then shaken for a specific time (i.e. 90 min). At multiple time intervals (i.e. 30, 60, and 90 min), gas samples are collected from sample jars. Nitrous oxide (N_2O) concentrations of these gas samples can be determined with a gas chromatograph. Nitrous oxide fluxes are then calculated as the time-linear rate of concentration increase in the headspace of the sample jars. The DEA is then calculated as the short-term rate of N_2O production in the jars and is indicative of the size of the denitrifying enzyme pool present in the soil [85].

Another important nutrient retention function that occurs in wetlands is P sorption. The P sorption index (PSI) has been widely used to estimate the P retention capacity of wetland soils. The index was developed by Bache and Williams [88] and used by Richardson [89] in a seminal study of P sorption across various wetland types. Numerous studies have established that the PSI: 1) serves as a reliable gauge of a wetland soil's P sorption potential, 2) is less time consuming to measure than multiple-point P sorption isotherms, and 3) facilitates comparison with related soil properties [89–93]. The PSI can be determined by shaking a sterilized soil sample with a known P concentration for 24 h. The difference in concentration of inorganic P between the initial and final concentration represents the amount of P sorbed. The index is then calculated as $X \cdot (\log C)^{-1}$ where X = amount of P sorbed (mg P100 g soil^{-1}) and C = the final inorganic P concentration in solution (mg PL^{-1}).

Phosphorus fractionation is another common method that is used to quantify P storage in wetland soils and involves the quantification of the distribution of P across various soil pools. These pools are operationally defined, but have generally been equated with bioavailable-P, Ca and Mg bound P, Al and Fe bound P, and residual pools [93–96].

9 Role of coastal wetlands in retaining and transforming nutrients

As P is generally considered to be the limiting nutrient in freshwater ecosystems, N is considered to be the limiting nutrient in marine ecosystems, and coastal wetlands represent transitional zones of both N and P limitation [8, 97], the results described in this section involve processes dealing with transformation and retention of N and P. Retention of N and P in wetlands results from cumulative fluxes into storage compartments of wetland ecosystems such as microbes, vegetation, plant litter, and soils [98]. Uptake of N and P by emergent wetland plants may be high during the growing season, but much of this N is released upon senescence in the late fall and winter [99, 100]. Rooted wetland plants tend to preferentially take up reduced ammonium nitrogen rather than oxidized nitrate nitrogen [52, 101]. In terms of climate, plant uptake is lowest in cold northern-hemisphere zones and wetlands with stagnant hydrology, whereas plant uptake is greatest in tropical zones and wetlands with active hydrology [52]. Unlike emergent vegetation, trees in forested wetlands provide long-term nutrient storage [99, 102]. Microbial uptake of N and P has also been thought to be a short-term rather than a long-term nutrient sink [84, 89, 98]. A comprehensive study of the effect of wetlands on water quality in Minnesota found that wetlands were more effective in removing suspended solids, total phosphorus and ammonia during high-flow periods, but were more effective at removing nitrate in low-flow periods [103]. This suggests that the ability of wetlands to retain and store nutrients may not only vary on a seasonal scale but also during storm flow compared to the baseflow.

Organic matter accumulation and denitrification are two of the more dominant and long-term N transformation and retention mechanisms. Organic matter accumulation involves the storage of N and P in soil organic matter (SOM) pools. These pools are generated from litter inputs, and often escape decomposition due to the anaerobic soil conditions. Soils with high organic carbon generally also have high organic N and P [104]. This processes mentioned above are common to all coastal wetland types. In addition, different coastal wetlands vary in their ability to retain and transform nutrients. These differences will be explored in the next sections of the chapter.

9.1 Retention and transformation of N and P in riparian wetlands

As a result of their location in the landscape, riparian wetlands interact with both upstream and upslope sources of nonpoint-source runoff and have the ability to reduce inputs of N and P to coastal waters [58, 105, 106]. Thus, riparian wetlands may be sinks for N and P at the watershed scale and if so, play a central role in maintaining regional water quality [107]. Retention and transformation of N and P by wetlands involves a combination of biogeochemical processes such as denitrification, P sorption, sedimentation, and organic matter accumulation, plant uptake, microbial immobilization [27, 52, 108].

Riparian wetlands have been shown to support high denitrification rates and, in certain cases, to transform the majority of nitrate inputs to nitrogen gases [109, 110]. Denitrification in these riparian wetlands has also shown high spatial variability [111–115] due to the presence of patches of organic matter and anaerobic microsites in the soil profile [116–119]. The term "hot spots" was coined to describe these areas of high denitrification [116, 120]. Recent studies have shown significantly lower denitrification potential in wetlands with nutrient-poor substrates (sand, light till) than in wetlands with nutrient-rich substrates (alluvium, dark till) [121], significantly lower DEA levels in restored/created wetlands compared to natural wetlands [86, 115], and a positive relationship between plant species richness and denitrification potential [122].

Soils of riparian wetlands have been shown to have higher P sorption capacities than adjacent uplands or streambanks [90, 107]. Long-term P storage in wetlands is believed to be controlled by three main processes: (1) deposition of sediment-bound P; (2) sorption of dissolved phosphate; or (3) the storage of organic P by peat accretion [96, 108]. While significant amounts of P can be stored by sedimentation [98], these sediments may be resuspended in future hydrologic events. Consequently, sorption and peat accretion are believed to represent the most important long-term P-retention pathways [89, 108].

In alkaline wetland soils, P sorption has been shown to be significantly correlated with calcium and magnesium content, and to a lesser degree with aluminum, iron and SOM content [108]. In acid wetland soils, P sorption has been shown to be significantly correlated with amorphous Al and Fe content [89, 93, 106], and to a lesser degree SOM and particle size [90, 93]. While iron phosphates are solubilized when Fe(III) is reduced to Fe(II) under anaerobic conditions, aluminum phosphates are unaffected by changes in redox potential [123]. Thus, soluble iron and phosphate may be lost from wetland soils in reducing conditions, while aluminum phosphates persist in acid wetland soils [107].

Another layer of complexity is added when we consider that amorphous Al and Fe, SOM, and texture exhibit significant spatial and temporal variability in riparian wetlands [93, 107, 124, 125]. Beyond a locally random aspect, this spatial variability may be related to the combined action of physical, chemical, or biological processes that operate at different spatial scales [126]. In natural riparian wetlands, these processes might include overbank flooding, sediment deposition, surface runoff, erosion, groundwater inputs, fire, tree-throw, root activity, litter production, and activity of macro and micro soil fauna. Each of these processes may influence particular locations of the riparian zone with varying degrees of intensity. For example, Fig. 7 shows the spatial variability of per cent clay, oxalate

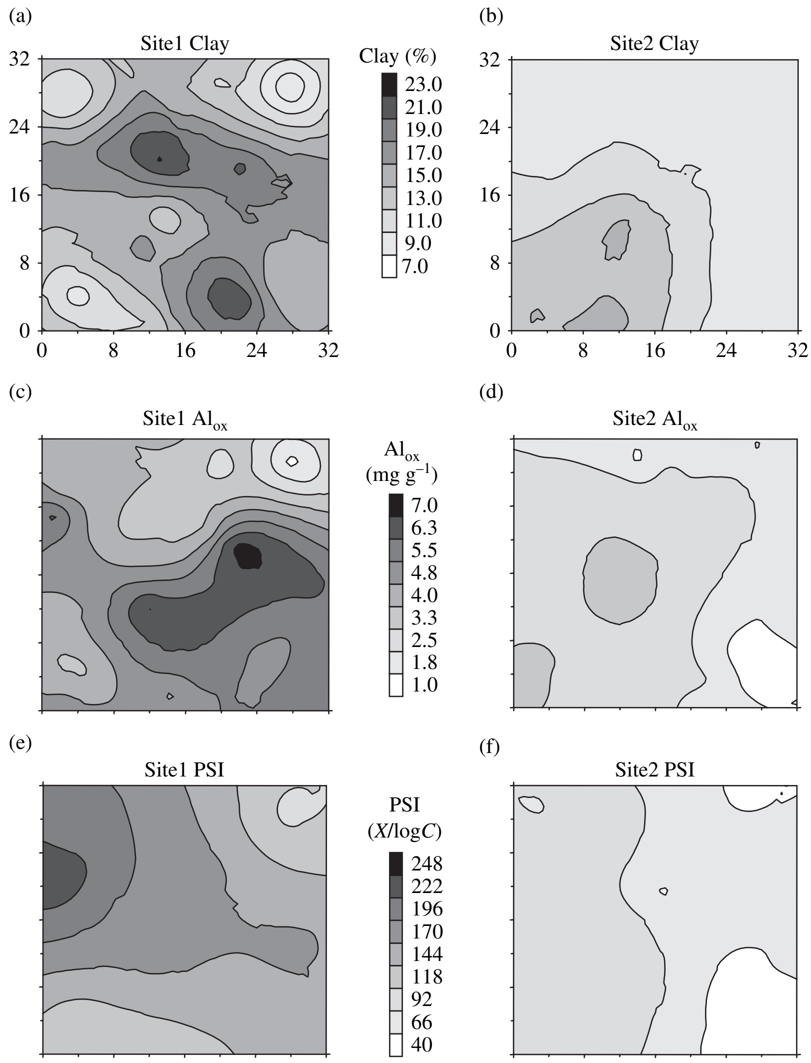


Figure 7: Spatial distribution of% clay (a), oxalate extractable aluminum (Al_{ox}) (c), and the phosphorus sorption index (PSI) (e) at Site 1, and% clay (b), Al_{ox} (d), and PSI (f) at Site 2. Plot size is 32 m by 32 m. Modified from [92].

extractable aluminum, and the PSI of two riparian wetlands in the North Carolina Coastal Plain [93].

9.2 Retention and transformation of N and P in tidal marshes

Salt marshes are thought to act as N transformers, importing dissolved oxidized inorganic forms of N and exporting dissolved and particulate reduced forms of N [127]. Salt marshes appear to be sinks for total P, but remobilization of phosphate in the sediments can lead to small net exports of phosphate from salt marshes [127].

Many studies have documented the fact that salt marshes are net exporters of organic material [27]. Salt marshes are generally thought to be N limited [3], but a recent study indicated that while the vegetation in the salt marshes was N limited, the microbial community was P limited [128].

In terms of P retention, a study along an estuarine salinity gradient along the Cooper River in South Carolina, demonstrated that there was a trend of decreasing P sorption capacity of intertidal marsh sediments with increasing salinity [129]. Specifically, the freshwater marsh site had the highest P sorption capacity followed by the two brackish marshes with intermediate P sorption. The salt marsh sedimentshad the lowest P sorption capacity. The results were attributed to the decrease in soil surface area across the salinity gradient as well as the changes in mineralogy, ionic strength, and redox chemistry of Al and Fe [129]. Interestingly, under freshwater conditions, Al and Fe hydroxides carry a net positive charge that facilitates P sorption, whereas under saltwater conditions, Al and Fe hydroxides carry a net negative charge that inhibits P sorption [130, 131].

9.3 Retention and transformation of N and P in mangroves

While the tidally dominated fringe mangroves provide protection of shorelines from marine wave action and storms, riverine and basin mangroves are more involved in trapping of sediment and retention of nutrients from terrestrial sources [39]. Mangroves are considered to be in a steady-state balance between N loss and N fixation, but they are capable of being stimulated to higher levels of production from local additions of fertilizers [36]. In addition to retaining sediments, Walsh [83] reported that the high nitrate and phosphate levels in Heeia Stream were reduced significantly in the upper reaches of the swamp, indicating that the mangroves may serve as sinks for these nutrients ask well. Mangroves have also been shown to have a high capacity to absorb and adsorb heavy metals and other toxic substances in effluents [132].

9.4 Retention and transformation of N and P in seagrass beds and coral reefs

While seagrass beds have been shown to trap sediments from terrestrial sources, their ability to retain and transform nutrients is largely unquantified [133]. Sediments underlying seagrass beds have been shown to have higher carbon content and lower redox potentials [134] indicating that they may be more effective at denitrification than adjacent areas without seagrasses or with rocky bottoms. Research on the deposition of seagrass castings and macroalgae remnants on beaches has shown these sources to be important for nutrient provisioning to coastal invertebrates and shorebirds [5]. Likewise, it was found that over 6 million kilograms dry weight of seagrass and algal detritus (20% of the annual production) is deposited each year on the 9.5 km beach of Mombasa Marine Park in Kenya [134]. These studies provide evidence that seagrasses may act as nutrient sources in coastal zones.

Nutrient cycling in coral reefs is quite complex. Primary producers have been shown to take up ammonium and nitrate from the waters surrounding the reef [36]. Consumers in the reef environment commonly excrete ammonia [36]. Reefs support a diverse community of bacteria that are involved in processes that produce, transform, or consume N such as N fixation, ammonification, nitrification, denitrification, and processing of organic N compounds [135]. Thus, it is difficult to generalize about the role the coral reefs plat in nutrient transformation and retention. However, it is clear that when reefs are subject to high sediment and nutrient loads, they do not respond favorably [17]. Thus, there is a great need to effectively manage the agriculture, pasture land, rangeland, and forestland in terrestrial watersheds, as well as to conserve and restore coastal wetlands that are so important for sediment and nutrient retention at the watershed scale.

10 Case study: comparison of soils from created, restored and natural wetlands

The United States Clean Water Act (CWA) of 1972 mandates mitigation whenever natural wetlands are impacted by development. The Army Corps of Engineers (ACoE) has jurisdiction over this process and requires created and restored wetlands to meet specific vegetative and hydrologic criteria during a five-year monitoring period to be considered successful [136]. Vegetative criteria require survival of a certain per centage of planted species per acre. Hydrologic criteria require that the water table be within 30 cm of the soil surface for a consecutive period of at least 12.5% of the growing season. The current process does not require any monitoring of soil properties or processes [136, 137]. It is interesting that soil has been omitted from the mitigation process, as soil plays an integral part in the definition of a wetland as stated in the USFWS Wetland Classification System [18]. Hydric soil, along with hydrophytic vegetation, and wetland hydrology are also the three criteria used to delineate jurisdictional wetlands as defined by the ACoE [138]. The lack of consideration of edaphic characteristics in the wetland mitigation process is a cause for concern for a number of reasons: 1) soil forms the foundation of these developing ecosystems; 2) inadequate soil properties can be detrimental to vegetative survival and the establishment of wetland hydrology; and 3) soil is the medium for biogeochemical processes that transform and retain nutrients [137]. Without suitable soil properties, created wetlands (CWs) and restored wetlands (RWs) may never replace the nutrient transformation and retention functions of the natural wetlands (NWs) that were destroyed.

As few CWs or RWs are assessed beyond what is needed to meet hydrologic and vegetative success criteria [139], the ability of these wetlands to replace natural wetland functions is a topic of considerable debate [86, 140–142]. It has been stated that the definitive test of success for CWs and RWs is how closely

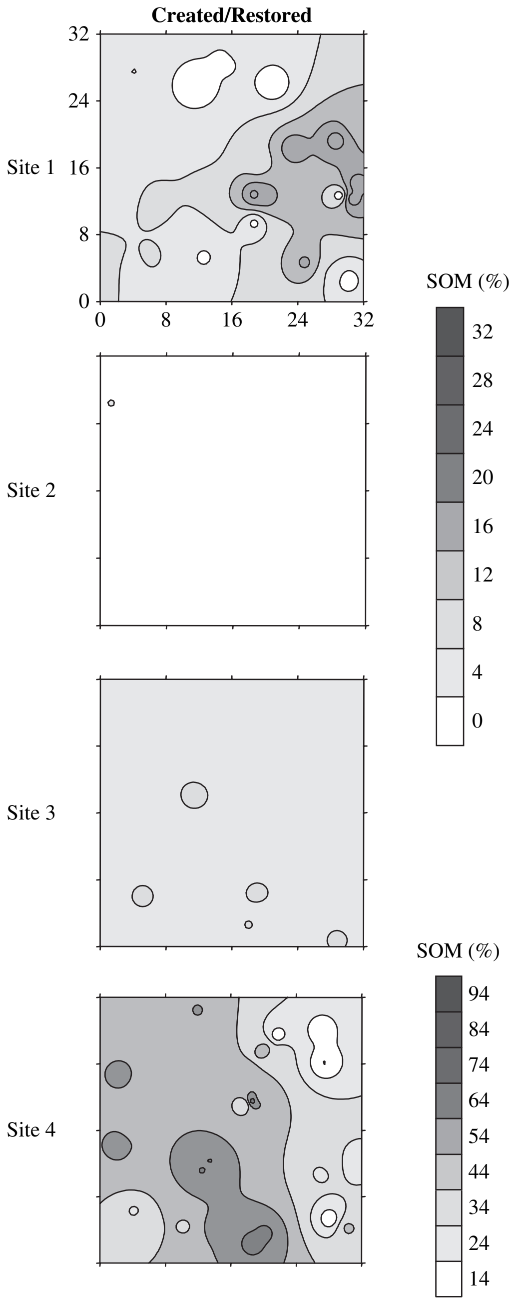


Figure 8: Spatial distribution of soil organic matter (SOM) at Site 1 restored (a) and natural wetland (b), at Site 2 created (c) and natural wetland (d), at Site 3 (e) and natural wetland (f), and at Site 4 restored (g) and natural wetland (h) plots. The upper scale applies to Sites 1–3, while the lower scale applies to Site 4. Plot size is 32 m by 32 m. Modified from [153].

they function like NWs [143]. Unfortunately, only a few studies have attempted to determine whether wetland functions in CWs and NWs are equivalent to those of NWs [86]. The assumption that wetland function follows wetland structure, which underlies the ACoE monitoring process, is also largely untested [144].

For example, soil properties of CWs and RWs have almost always been shown to differ from NWs [145]. Created wetlands typically have higher sand and lower clay content than NWs [141, 146, 147]. This has important implications for wetland function, as coarse-textured soils typically have lower water-holding and nutrient-retention capacities than fine-textured soils [148, 149]. Created wetlands and RWs also usually have lower levels of SOM and higher bulk densities than NWs [143, 146, 147, 150, 151]. Such soil conditions can lead to low growth and survival of planted and colonizing species. Litter layers in CW/RWs are often poorly developed or absent in comparison to that of NWs [86, 144]. As a result of low organic matter and sparse litter, it has been speculated that the microbial communities in the soil of CWs are much less viable than those of NWs [152]. Additionally, soil temperatures were reported to be significantly higher in CWs than in NWs as a result of a lack of shading by mature trees [141]. Furthermore, microtopography has been reported to be considerably lower in CW/RWs than in NWs [141, 151].

Bruland and Richardson [153] conducted a study in which they compared patterns of spatial variability of soil properties in CWs/RWs and paired NWs in the North Carolina coastal plain. They found that the spatial variability of SOM was lower at some but not all of the CWs/RWs (Fig. 8), and concluded that prior land use and mitigation activities could decrease, increase, or cause no change in the spatial variability of soil properties in CWs/RWs compared to NWs. In another study, Bruland *et al.* [115] found that the spatial variability of predicted DEA values in was much lower in the CWs/RWs than in the NWs for the riverine sites but not for the nonriverine sites

11 Future research needs and directions

Existing landscape patterns contain information about the processes that generated these patterns [154]. In various coastal wetlands around the world, the processes of hydrologic modification, sedimentation, and nutrient loading have not only affected the structure and function of these ecosystems but also their spatial extent and distribution [153, 155–158]. The use of remote sensing and geographic information systems (GIS) have enhanced scientists' capacity to describe patterns in nature over larger spatial scales and at finer levels of detail than ever before [159]. These methods can be used to quantify anthropogenic impacts to wetlands at multiple scales, providing valuable information to aid in the design of wetland and watershed restoration projects. These types of approaches may be especially useful for the monitoring of seagrasses and coral reefs. Traditional assessment of cover and density of seagrass and coral cover along transects and quadrats generally have an associated error $>30\%$ about the mean [43]. This makes it difficult to detect reliable changes in seagrass or coral cover. Typically, changes can only be detected when they are greater than 50–80%. Thus there is a great need to develop remote-sensing techniques to more accurately monitor changes in seagrass beds over large spatial scales [43].

Significant strides have been made in coastal-zone management in the last few decades and many of the world's 123 coastal countries now have some form of coastal management plans and legislation [5]. However, countries with well-developed coastal-zone management plans are still facing loss of coastal wetlands, overexploitation of coastal resources, user conflicts, and indirect degradation from activities occurring sometimes hundreds of km from the coastal zone itself [5]. Thus, management and policy have not been able to keep pace with increasing degradation of coastal ecosystems.

One bright spot in this picture is that coastal wetlands such as tidal marshes and mangroves can often be restored once they have been degraded. The science of salt-marsh and mangrove restoration is relatively advanced compared to restoration of other ecosystems. Research has shown that the establishment or re-establishment of proper elevation, topography, tidal flushing, and the planting of a few key species can restore salt marshes and mangroves on relatively short time scales [144, 160]. Mangroves appear to be the most amenable to restoration [15] of the coastal wetlands considered in this chapter. Restoration of mangroves has been accomplish by transplanting vegetative propagules, young trees, or mature trees with considerable success in India and Southeast Asia [161]. Coastal wetland restoration in the United States has been most successfully achieved in estuarine salt-marsh systems [162, 163]. However, studies have shown that, while the vegetative structure of created and restored marshes appears to approximate that of natural marshes, the soil properties and macroinverbrate communities in the created and restored marshes may take much longer time scales to develop [164, 165]. Restoration of seagrass meadows although increasingly common, has been fraught with difficulties; a worldwide success rate for restoration of these systems was estimated to be only 45% [161, 166]. Restoration of coral reefs may be the most difficult challenge of any ecosystem, and it has only been practiced at small scales and with limited success [133].

Future research, management, restoration, and policy needs to occur at the watershed or regional scale [167] and involve an interdisciplinary approach to assessing the role and functions of coastal wetlands in reducing the impacts of land-based management. According to Vivian-Smith [168], wetland restoration should consider ecological processes and structure at multiple spatial scales (Fig. 9). Hydrologic, vegetative, and edaphic heterogeneity increases the probability that an optimal habitat will exist at a restoration site for the intended species and better approximate the structure and function of natural wetlands [15, 169]. On a positive note, there are currently a number of landscape-scale coastal wetland restoration projects in various phases of planning and implementation that are designed to ameliorate the effects of hydrologic alteration and nutrient loading including the Chesapeake Bay [170], Mississippi River basin [171], Florida Everglades [172], and the wetlands of Iraq [157]. However, it has been estimated that a tripling of the

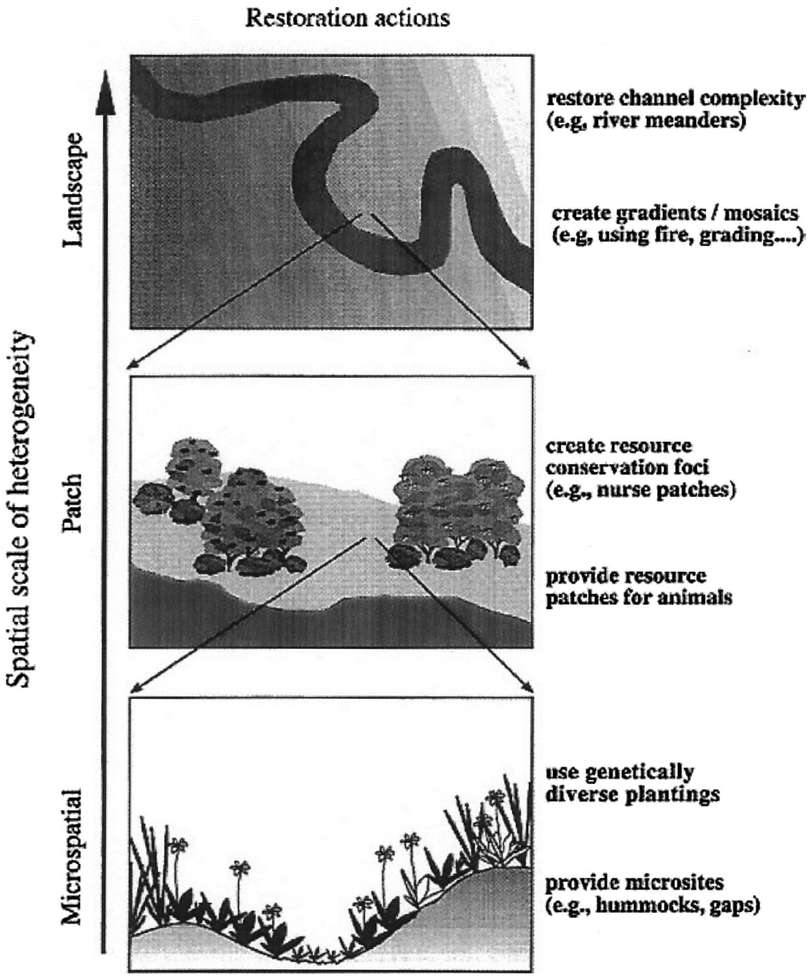


Figure 9: Restoration actions in the coastal wetland landscape focusing on habitat heterogeneity at different spatial scales (microscale, patch, and landscape) that influence key processes and accelerate ecosystem development. Copyright (2001) from, Developing a Framework for Restoration, by G. Vivian-Smith [168]. Reproduced by permission of Taylor & Francis, a division of Informa plc.

area of riparian forests and buffer systems would be needed in the Upper Mississippi River and Ohio River Basins in order to cause significant reduction in the N load of the Mississippi River to the Gulf of Mexico [171].

Another move in the right direction appears to be the coupling of coastal-zone management with watershed management as has occurred with the European Water Framework Directive and projects under the LOICZ (Land-Sea Interactions in the Coastal Zone) initiative [5]. Also, the U.S. Coral Reefs Task Force has established local action strategies to assess land-based threats to coral reefs specifically looking at solutions to prevent erosion and nutrient loading to coastal zones from terrestrial sources. With future population trends suggesting that the world's coastal population is expected to approach 6 billion people by 2025 [15], our ability to preserve, protect, manage, and restore watersheds, coastal wetlands, and near-shore habitats is becoming increasingly important.

References

[1] Nixon, S.W., Coastal marine eutrophication: a definition social causes, and future concerns. *Ophelia*, **41**, pp. 199–219, 1995.
[2] Howarth, R.W., Billen, G., Swaney, D., Townsend, A., Jaworski, N., Lajtha, K., Downing, J.A., Elmgren, R., Caraco, N., Jordan, T., Berendse, F., Freney, J., Kudeyarov, V., Murdoch, P. & Zhu, Z.L., Regional nitrogen budgets and riverine N and P fluxes for the drainages to the North Atlantic Ocean: Natural and human influences. *Biogeochemistry*, **35**, pp. 75–79, 1996.
[3] Vitousek, P.M., Aber, J.D., Howarth, R.W., Likens, G.E., Matson, P.A., Schindler, D.W., Schlesinger, W.H. & Tilman, D.G., Human alteration of the global nitrogen cycle: sources and consequences. *Ecological Applications*, **7**, pp. 737–50, 1997.
[4] Rablais, N.N., Turner, R.E. & Scavia, D., Beyond science into policy: Gulf of Mexico hypoxia and the Mississippi River. *BioScience*, **52**, pp. 129–142, 2002.
[5] Agardy, T. & Alder, J., Coastal Systems. *Millennium Ecosystem Assessment, Ecosystems and Human Well-Being: Current State and Trends* [Vol. 1], Island Press: Washington, DC, pp. 513–549, 2005.
[6] Reichelt-Brushett, A.J. & Harrison, P.L., The effect of copper, zinc, and cadmium on fertilization success of gametes from scleractinian reef corals. *Marine Pollution Bulletin*, **38**, pp. 182–187, 1999.
[7] Jones, R.J., Muller, J., Haynes, D. & Schreiber, U., Effects of herbicides diuron and atrazine on corals of the Great Barrier Reef, Australia. *Marine Ecology Progress Series*, **251**, pp. 153–167, 2003.
[8] Schlesinger, W.H., *Biogeochemistry*, 2nd edn, Academic Press: San Diego, 1997.
[9] Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P. & van den Belt, M., The value of the world's ecosystem services and natural capital. *Nature*, **387**, pp. 253–260, 1997.
[10] Dahl, T.E. & Johnson, C.E., Wetland status and trends in the conterminous United States mid-1970s to mid-1980s. U.S. Department of the Interior, Fish & Wildlife Service: Washington, DC, 1991.
[11] Noe, G.B., Childers, D.L. & Jones, R.D., Phosphorus biogeochemistry and the impact of phosphorus enrichment: Why is the Everglades so unique? *Ecosystems*, **4**, pp. 603–624, 2001.

[12] United States Geological Survey (USGS). 100 + Years of Land Change for Coastal Louisiana. National Wetlands Research Center, Lafayette. Map ID: USGS-NWRC 20030-03-085, 2003.

[13] Millennium Ecosystem Assessment (MEA), *Ecosystems and human well-being: wetlands and water synthesis*. World Resources Institute: Washington, DC, 2005.

[14] Center for International Earth Science Information Network (CIESIN). Gridded population of the world [GPW] version 3 beta, http://quin.unep-wcmc.org/MA/index.cfm.

[15] Kennish, M.J., Environmental threats and environmental future of estuaries. *Environmental Conservation*, **29**, pp. 78–107, 2002.

[16] Hemminga, M.A. & Duarte, C.M., *Seagrass ecology*, Cambridge University Press, Cambridge, 2000.

[17] Hughes, T.P., Baird, A.H., Bellwood, D.R., Card, M., Connolly, S.R., Folke, C., Grosberg, R., Hoegh-Guldberg, O., Jackson, J.B.C., Kleypas, J., Lough, J.M., Marshall, P., Nysrom, M., Palumbi, S.R., Pandolfi, J.M., Rosen, B. & Roughgarden, J., Climate change, human impacts, and the resilience of coral reefs. *Science*, **301**, pp. 929–933, 2003.

[18] Cowardin, L.M., Carter, V., Golet, F.C. & LaRoe, E.T., *Classification of Wetlands and Deepwater Habitats of the United States. U.S. Department of the Interior, Fish & Wildlife Service FWS/OBS-79/31*: Washington, DC, 1979.

[19] Warner, B.G. & Rubec, C.D.A., *The Canadian Wetland Classification System*, 2nd edn. Wetlands Research Center, University of Waterloo, Waterloo, 1997.

[20] Navid, D., The international law of migratory species: The Ramsar Convention. Natural *Resources Journal*, **29**, pp. 1001–1016, 1989.

[21] Brinson, M.M., A hydrogeomorphic classification for wetlands. Wetlands Research Program Technical Report WRP-DE-4. U.S. Army Corps of Engineers Waterways Experiment Station, Vicksburg, 1993.

[22] Johnson, R.R. & McCormick, J.F., Tech. Coords., Strategies for the Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. Proceedings of the Symposium, Callaway Gardens, Georgia, Dec. 11–13, 1978. General Technical Report WO-12, U.S. Forest Service: Washington, DC, 1979.

[23] Gregory, S.V., Swanson, F.J., McKee, W.A. & Cummins, K.W., An ecosystem perspective of riparian zones. *BioScience*, **41**, pp. 540–551, 1991.

[24] Brinson, M.M., Swift, B.L., Plantico, R.C. & Barclay, J.S., Primary productivity, decomposition, and consumer activity in freshwater wetlands. *Annual Review of Ecology and Systematics*, **12**, pp. 123–161, 1981.

[25] The Nature Conservancy, The forested wetlands of the Mississippi River: an ecosystem in crisis. *Nature Conservancy*: Baton Rogue, 1992.

[26] Bernhardt, E.S., Palmer, M.A., Allan, J.D., Alexander, G., Barnas, K., Brooks, S., Carr, J., Clayton, S., Dahm, C., Follstad-Shah, J., Galat, D., Gloss, S., Goodwin, P., Hart, D., Hassett, B., Jenkinson, R., Katz, S., Kondolf, G.M., Lake, P.S., Lave, R., Meyer, J.L., O'Donnell, T.K.O., Pagano, L., Powell, B. & Sudduth, E., Synthesizing U.S. river restoration efforts. *Science*, **305**, pp. 636–637, 2005.

[27] Mitsch, W.J. & Gosselink, J.G., *Wetlands*, 3rd edn, John Wiley & Sons, Inc.: New York, 2000.
[28] Teal, J.M. & Teal, M., *Life and Death of a Salt Marsh*, Audobon/Ballantine Books: New York, 1969.
[29] Ledoux, L., Vermaatb, J.E., Bouwerb, L., Salomons, W. & Turnera, R.K., ELOISE research and implementation of the EU policy in the coastal zones, http://130.37.129.100/english/o_o/instituten/IVM/research/eloise/pdf/btbch1.pdf.
[30] Chapman, V.J., *Salt Marshes and Salt Deserts of the World*. Interscience: New York, 1960.
[31] Bertness, M.D., Ewanchuk, P.J. & Silliman, B.R., Anthropogenic modification of New England salt marsh landscapes. *Proceedings of the National Academy of Sciences of the United States of America*, **99**, pp. 1395–1398, 2002.
[32] Beeftick, W.G., Salt marshes. *The Coastline,* (ed.) R.S.K. Barnes, John Wiley & Sons: New York, p. 93–121, 1977.
[33] Silliman, B.R. & Bertness, M.D., A trophic cascade regulates salt marsh primary production. *Proceedings of the National Academy of Sciences of the United States of America*, **99**, pp. 10500–10505, 2002.
[34] Spalding, M.D., Blasco, F. & Field, C.D., *World Mangrove Atlas*, The International Society for Mangrove Ecosystems: Okinawa, 1997.
[35] Valiela, I., Bowen, J.L. & York, J.K., Mangrove forests: One of the world's threatened major tropical environments. *Bioscience*, **51**, pp. 807–815, 2001.
[36] Mann, K.H., *Ecology of Coastal Waters*, 2nd edn. Blackwell Science: Malden, 2000.
[37] Duke, N.C., Mangrove floristics and biogeography. *Tropical Mangrove Ecosystems*, (ed.) A.I. Robertson & D.M. Alongi, American Geophysical Union: Washington, DC, pp. 63–100, 1992.
[38] Twilley, R.R., Mangrove Wetland. *Southern Forested Wetlands: Ecology and Management*, (ed.) M.G. Messina & W.H. Conner, Lewis Publishers: Boca Raton, pp. 445–473, 1998.
[39] Ewel, K.C., Twilley, R.R. & Ong, J.E., Different kinds of mangrove forests provide different goods and services. *Global Ecology & Biogeography*, **7**, pp. 83–94, 1998.
[40] Ong, J.E., The ecology of mangrove conservation and management. *Hydrobiologia*, **295**, pp. 342–351, 1995.
[41] Allen, J.A. Mangroves as alien species: the case of Hawaii. *Global Ecology and Biogeography Letters*, **7**, pp. 61–71, 1998.
[42] Ewel, J.J. & Bigelo, S.W., Plant life forms and tropical ecosystem functioning. *Biodiversity and Ecosystem Processes in Tropical Forests*, (ed.) G. Orians, R. Dirzo & J.H. Cushman, Springler Verlag: Berlin, 1996.
[43] Duarte, C.M., The future of seagrass meadows. *Environmental Conservation*, **29**, pp. 192–206, 2002.
[44] Duffy, J.E., Biodiversity and the functioning of seagrass ecosystems. *Marine Ecology Progress Series*, **311**, pp. 233–250, 2006.
[45] Duarte, C.M. & Cebrian, J., The fate of marine autotrohic production. *Limnology & Oceanography*, **41**, pp. 1758–1766, 1996.

[46] Anderson, E.E., Economic benefits of habitat restoration: Seagrass and the Virginia hardshell blue crab fishery. *North American Journal of Fisheries Management*, **9**, pp. 140–149, 1989.
[47] den Hartog, C., *The Seagrasses of the World*, North Holland: Amsterdam, 1970.
[48] Miliman, J.D. & Meade, R.H., World-wide delivery of river sediment to the oceans. *Journal of Geology*, **91**, pp. 1–21, 1993.
[49] Shepherd, S.A., McComb, A.J., Bulthuis, D.A., Neverauskas, V., Stefensen, D.A. & West, R., Decline of seagrasses. *Biology of Seagrasses*, (ed.) A.W.D. Larkum, A.J. McComb & S.A. Shepherd. Elsevier: Amsterdam, the Netherlands, pp. 346–393, 1989.
[50] Burke, L., Kura, Y., Kassem, K., Ravenga, C., Spalding, M. & McAllister, D., *Pilot Assessment of Global Ecosystems: Coastal Ecosystems*, World Resources Institute: Washington, DC, 2001.
[51] Dayton, P.K., The importance of the natural sciences to conservation. *American Naturalist*, **162**, pp. 1–10, 2003.
[52] Bowden, W.B., The biogeochemistry of nitrogen in freshwater wetlands. *Biogeochemistry*, **4**, pp. 313–348, 1984.
[53] Hudson, N., *Soil Conservation*, 2nd edn. Cornell University Press: Ithaca, 1981.
[54] El-Swaify, S.A., Operative Processes for sediment-based watershed degradation in small, tropical volcanic island ecosystems. *Integrated Watershed Management in the Global Ecosystem*, (ed.) R. Lal. CRC Press: Boca Raton, pp. 35–49, 2000.
[55] DeCarlo, E.H., Beltran, V.L. & Tomlinson, M.S., Composition of water and suspended sediment in stream of urbanized subtropical watersheds in Hawaii. *Applied Geochemistry*, **19**, pp. 1011–1037, 2004.
[56] Novitski, P.R. Wetland hydrology. *Wetlands Ecology and Conservation: Emphasis on Pennsylvania*, (ed.) S.K. Majumdar, R.P. Brooks, F.J. Benner & R.W. Tiner, Pennsylvania Academy of Science: Easton, pp. 47–64, 1989.
[57] Faber, P.A., Keller, E., Sands, A. & Masser, B.M., The Ecology of Riparian Habitats of the Southern California Coastal Region: A Community Profile. Biological Report 85 [7.27], U.S. Fish & Wildlife Service: Washington, DC, 1989.
[58] Brinson, M.M., Changes in the functioning of wetlands along environmental gradients. *Wetlands*, **13**, pp. 65–74, 1993.
[59] Hupp, C.R. & Bazemore, D.E., Spatial and temporal aspects of sediment deposition in West Tennessee forested wetlands. *Journal of Hydrology*, **14**, pp. 179–196, 1993.
[60] Kleiss, B.A., Sedimentation in a bottomland hardwood wetland in eastern Arkansas. *Wetlands*, **16**, pp. 321–333, 1996.
[61] Noe, G.B. & Hupp, C.R. Carbon, nitrogen, and phosphorus accumulation in floodplains of Atlantic Coastal Plain Rivers, USA. *Ecological Applications*, **15**, pp. 1178–1190, 2005.
[62] Harter, S.K. & Mitsch, W.J., Patterns of short-term sedimentation in a freshwater created marsh. *Journal of Environmental Quality*, **32**, pp. 325–334, 2003.

[63] Cahoon, D.R., Lynch, J.C. & Knaus, R.M., Improved cryogenic coring device for sampling wetland soils. *Journal of Sedimentary Research*, **66**, pp. 1025–1027, 1996.

[64] DeLaune, R.D., Reddy, C.N. & Patrick, W.H., Jr., Accumulation of plant nutrients and heavy metals through sedimentation processes and accretion in a Louisiana salt marsh. *Estuaries*, **4**, pp. 328–334, 1981.

[65] Boumans, R.M. & Day, J.W., Jr., The effects of two Louisiana marsh management plans on water and materials flux and short-term sedimentation. *Wetlands*, **14**, pp. 247–261, 1994.

[66] Fennessy, M.S., Brueske, C.C. & Mitsch, W.J., Sediment deposition patterns in restored freshwater wetlands using sediment traps. *Ecological Engineering*, **3**, pp. 409–428, 1994.

[67] Hensel, P.F. & Day, J.W., Jr., Short-term sedimentation dynamics in the Rhone River Delta France: The importance of riverine pulsing. *Estuaries*, **21**, pp. 52–65, 1998.

[68] Robbins, J. A. & Edgington, D.N., Determination of recent sedimentation-rates in Lake Michigan using Pb-210 and Cs-137. *Geochimicha et Cosmochimica Acta*, **39**, pp. 285–295, 1975.

[69] Craft, C.B. & Richardson, C.J., Peat accretion and N, P, and organic C accumulation in nutrient-enriched and unenriched Everglades peatlands. *Ecological Applications*, **3**, pp. 446–458, 1993.

[70] Craft, C.B. & Richardson, C.J., Recent [^{137}Cs] and long-term [^{210}Pb, ^{14}C] organic soil accretion and nutrient accumulation in the Everglades. *Soil Science Society of America Journal*, **62**, pp. 834–843, 1998.

[71] Richie, J.C. & McHenry, J.R., Determination of fallout Cs-137 and naturally occurring gamma ray emitters in sediments. *International Journal of Applied Radioactive Isotopes*, **24**, pp. 575–578, 1973.

[72] Armentano, T.V. & Woodwell, G.M., Sedimentation rates in a Long Island marsh determined by Pb-210 dating. *Limnology and Oceanography*, **20**, pp. 452–456, 1975.

[73] Brenner, M., Peplow, A.J. & Scheleske, C.L., Disequilibrium between ^{226}Ra supported ^{210}PB in a sediment core from a shallow Florida Lake. *Limnology & Oceanography*, **39**, pp. 1222–1227, 1994.

[74] Binford, M.W., Calculation and uncertainty analysis of ^{210}Pb dates for PIRLA project lake sediment cores. *Journal of Paleolimnology*, **3**, pp. 253–267, 1990.

[75] Phillips, J.D. 1989. Fluvial sediment storage in wetlands. *Water Resources Bulletin*, **25**, pp. 867–877.

[76] Edgar, G.J. & Barrett, N.S., Effects of catchment activities on macrofaunal assemblages in Tasmanian estuaries. *Estuarine, Coastal & Shelf Science*, **50**, pp. 639–654, 2000.

[77] Dubinsky, Z. & Stambler, N., Marine pollution and coral reefs. *Global Change Biology*, **2**, pp. 511–526, 1996.

[78] Wolanski, E. & Spagnol, S., Environmental degradation by mud in tropical estuaries. *Regional Environmental Change*, **1**, pp. 152–162, 2000.

[79] Golbuu, V., Victor, S., Wolanski, E. & Richmond, R.H., Trapping of fine sediment in an enclosed bay, Palau, Micronesia. *Estuarine, Coastal, & Shelf Science*, **57**, pp. 941–949, 2003.

[80] Wolanski, E., Richmond, R.H., Davis, G. & Bonito, V., Water and fine sediment dynamics in transient river plumes in a small, reef-fringed bay, Guam. *Estuarine, Coastal & Shelf Science*, **56**, pp. 1029–1040, 2003.

[81] Victor, S., Neth, L., Golbuu, Y., Wolanski, E. & Richmond, R.H., Sedimentation in magroves and coral reefs in a wet tropical islands, Pohnpei, Micronesia. *Estuarine, Coastal & Shelf Science*, **66**, pp. 409–416, 2006.

[82] Bigelow, K.A., Alspach, K., Lohle, R., McDonough, T., Ravetto, P., Rosenfeld, C., Stender, G. & Wong, C., Assessment of mangrove ecosystem of west Moloka'i, Hawaii, with additional surveys of Moauni Beach Park and Ualapu'e Fishpond. Marine Option Program, University of Hawaii: Honolulu, 1989.

[83] Walsh, G.E., An ecological study of a Hawaiian mangrove swamp. ed. G.H. Lauff, *Estuaries*, American Association for the Advancement of Science Publication No. 83: Washington, DC, pp. 420–431, 1967.

[84] Richardson, C.J. & Marshall, P.R., Processes controlling movement, storage, and export of phosphorus in a fen peatland. *Ecological Monographs*, **56**, pp. 279–302, 1986.

[85] Tiedje, J.M., Denitrification. *Methods of Soil Analysis, Part 2. Chemical and Microbiological Properties,* ASA-SSSA, Madison, 1982.

[86] Hunter, R.G. & Faulkner, S.P., Denitrification potential in restored and natural bottomland hardwood wetlands. *Soil Science Society of America Journal*, **65**, pp. 1865–1872, 2001.

[87] Groffman, P.M., Gold, A.J. & Simmons, R.C., Nitrate dynamics in riparian forests: microbial studies. *Journal of Environmental Quality*, **21**, pp. 666–671, 1992.

[88] Bache, B.W. & Williams, E.G., A phosphate sorption index for soils. *Journal of Soil Science*, **22**, pp. 289–301, 1971.

[89] Richardson, C.J., Mechanisms controlling phosphorus retention capacity in freshwater wetlands. *Science*, **228**, pp. 1424–1427, 1985.

[90] Axt, J.R. & Walbridge, M.R., Phosphate removal capacity of palustrine forested wetlands and adjacent uplands in Virginia. *Soil Science Society of America Journal*, **63**, pp. 1019–1031, 1999.

[91] Bridgham, S.D., Johnston, C.A. & Schubaurer-Berigan, J.P., P sorption dynamics in soils and coupling with surface and pore water in riverine wetlands. *Soil Science Society of America Journal*, **65**, pp. 577–588, 2001.

[92] Bruland, G.L. & Richardson, C.J., A spatially explicit investigation of P sorption and related soil properties in two riparian wetlands. *Journal of Environmental Quality*, **33**, pp. 785–794, 2004.

[93] Bruland, G.L. & Richardson, C.J., An Assessment of the phosphorus sorption capacity of wetlands in the Painter Creek Watershed, Minnesota, USA. *Water, Air & Soil Pollution*, **171**, pp. 169–184, 2006.

[94] Hedley, M.J., Stewart, J.W.B. & Chauhan, B.S., Changes in inorganic and organic soil phosphorus fractions by cultivation practices and by laboratory incubations. *Soil Science Society of America Journal*, **46**, pp. 970–976, 1982.

[95] Tiessen, H. & Moir, J.O., Characterization of available P by sequential extraction, *Soil Sampling and Methods of Analysis*, (ed.) M.R. Carter, Lew is Publishers: Boca Raton, pp. 75–86, 1993.

[96] Qualls, R.G. & Richardson, C.J., Forms of soil phosphorus along a nutrient enrichment gradient in the northern Everglades. *Soil Science Society of America Journal*, **160**, pp. 183–198, 1995.

[97] Correll, D.L., Phosphorus: a rate limiting nutrient in surface waters. *Poultry Science*, **78**, pp. 674–682, 1999.

[98] Johnston, C.A., Sediment and nutrient retention by freshwater wetlands: Effects on surface water quality. *Critical Reviews in Environmental Control*, **21**, pp. 495–565, 1991.

[99] Reddy, K.R. & DeBusk, W.F., Nutrient storage capacities of aquatic and wetland plants, (ed.) K.R. Reddy & W.H. Smith, Magnolia Publishers, pp. 337–357, 1987.

[100] Mitsch, W.J., Cronk, J.K., Wu, X. & Nairn, R.W., Phosphorus retention in constructed freshwater riparian marshes. *Ecological Applications*, **5**, pp. 830–845, 1995.

[101] Howard-Williams, C. & Downes, M.T., Nitrogen cycling in wetlands. *Nitrate: Processes Patterns and Management*, (ed.) T.P. Burt, A.L. Heathwaite, S.T. Trudgill, John Wiley & Sons, Ltd.: New York, pp. 141–167, 1993.

[102] Mitsch, W.J., Dorje, C.L. & Wiemhoff, J.R., Ecosystem dynamics and a phosphorus budget of an alluvial cypress swamp in Southern Illinois. *Ecology*, **60**, pp. 1116–1124, 1979.

[103] Johnston, CA., Deitenbeck, N.E. & Nieme, G.J., The cumulative effect of wetlands on stream water quality and quantity. A landscape approach. *Biogeochemistry*, **10**, pp. 105–141, 1990.

[104] Reddy, K.R., Kadlec, R.H., Flaig, E. & Gale, P.M., Phosphorus retention in Stream and Wetlands: a review. *Critical Reviews in Environmental Science & Technology*, **29**, pp. 83–146, 1999.

[105] Walbridge, M.R., Functions and values of forested wetlands in the Southern United States. *Journal of Forestry*, **91**, pp. 15–19, 1993.

[106] Lockaby, B.G. & Walbridge, M.R., Biogeochemistry. *Southern Forested Wetlands: Ecology and Management*, (ed.) M.G. Messina & W.H. Conner, Lewis Publishers: Boca Raton, pp. 149–172, 1998.

[107] Darke, A.K. & Walbridge, M.R., Al and Fe biogeochemistry in a floodplain forest: Implications for P retention. *Biogeochemistry*, **51**, pp. 1–32, 2000.

[108] Richardson, C.J., The role of wetlands in storage, release, and cycling of phosphorus on the landscape: A 25–year retrospective. *Phosphorus Biogeochemistry of Sub-Tropical Ecosystems*, (ed.) K.R. Reddy, CRC Press: Boca Raton, pp. 47–68, 1999.

[109] Jacobs, T.C. & Gilliam, J.W., Riparian losses of nitrate from agricultural drainage waters. *Journal of Environmental Quality*, **14**, pp. 472–478, 1985.

[110] Lowrance, R., Groundwater nitrate and denitrification in a Coastal Plain riparian forest. *Journal of Environmental Quality*, **41**, pp. 401–405, 1992.

[111] Schipper, L.A., Cooper, A.B., Harfoot, C. G. & Dyck, W. J., Regulators of denitrification in an organic riparian soil. *Soil Biology & Biochemistry*, **25**, pp. 925–933, 1993.

[112] Pinay, G., Black, V.J., Planty-Tabacchi, A.M., Gumiero, B. & Decamps, H., Geomorphic control of denitrification in large river floodplain soils. *Biogeochemistry*, **50**, pp. 63–182, 2000.

[113] Casey, R.E., Taylor, M.D. & Klaine, S.J., Mechanisms of nutrient attenuation in a subsurface flow riparian wetland. *Journal of Environmental Quality*, **30**, pp. 1732–1737, 2001.

[114] Dhont, K., Boeckx, P., Hofman, G. & Van Cleemput, O., Temporal and spatial patterns of denitirifcation and nitrous oxide fluxes in three adjacent vegetated riparian buffer zones. *Biology & Fertility of Soils*, **40**, pp. 243–251, 2004.

[115] Bruland, G.L., Richardson C.J. & Whalen, S., Spatial variability of denitrification potential and related soil properties in created, restored and paired natural wetlands. *Wetlands*, **26**, pp. 1042–1056, 2006.

[116] Parkin, T.B., Soil microsites as a source of denitrification variability. *Soil Science Society of America Journal*, **51**, pp. 1194–1199. 1987.

[117] Gold, A.J., Jacinthe, P.A., Groffman, P.M., Wright, W.R. & Puffer, R.H., Patchiness in groundwater nitrate removal in a riparian forest. *Journal of Environmental Quality*, **27**, pp. 146–155, 1998.

[118] Jacinthe, P.A., Groffman, P.M., Gold, A.J. & Mosier, A., Patchiness in groundwater nitrate removal in a riparian forest. *Journal of Environmental Quality*, **27**, pp. 156–164, 1998.

[119] Casey, R.E., Taylor, M.D. & Klaine, S.J., Localization of denitrification activity in macropores of a riparian wetland. *Soil Biology & Biochemistry*, **36**, pp. 563–569. 2004.

[120] Christensen, S., Simkins, S. & Tiedje, J.M., Spatial variation in denitrification: dependency of activity centers in the soil environment. *Soil Science Society of America Journal*, **54**, pp. 1608–1613, 1990.

[121] Groffman, P.M. & Hanson G.C., Wetland denitrification: Influence of site quality and relationships with wetland delineation protocols. *Soil Science Society of America Journal*, **61**, pp. 323–329, 1997.

[122] Chabrerie, O., Poudevigne, I., Burdeau, F., Vinceslas-Akpa, M., Nebbache, S., Aubert, M., Bourcier, A. & Alard, D., Biodiversity and ecosystem functions in wetlands: A Case study in the estuary of the Seine River, France. *Estuaries*, **24**, pp. 1088–1096, 2001.

[123] Ponnamperuma, F.N., The chemistry of submerged soils. *Advances in Agronomy*, **24**, pp. 29–97, 1972.

[124] Lyons, J.B., Gorres, J.H. & Amador, J.A., 1998. Spatial and temporal variability of phosphorus retention in a riparian forest soil. *Journal of Environmental Quality*, **27**, pp. 895–903.

[125] Johnston, C.A., Bridgham, S.D. & Schubauer-Berigan, J.P., Nutrient dynamics in relation to geomorphology of riverine wetlands. *Soil Science Society of America Journal*, **65**, pp. 557–577, 2001.
[126] Goovaerts, P., Geostatistical tools for characterizing the spatial variability of microbiological and physico-chemical soil properties. *Biology and Fertility of Soils*, **27**, pp. 315–334, 1998.
[127] Nixon, S.W., Between coastal marshes and coastal waters – A review of twenty years of speculation and research on the role of salt marshes in estuarine productivity and water chemistry. *Estuarine & Wetland Processes*, (ed.) P.K. Hamilton & K.B. MacDonald, Plenum Press: New York, pp. 437–525, 1980.
[128] Sundareshwar, P.V., Morris, J.T., Koepfler, E.K. & Fornwalt, B., Phosphorus limitation of coastal ecosystem processes. *Science*, **299**, pp. 563–565, 2003.
[129] Sundareshwar, P.V. & Morris, J.T., Phosphorus sorption characteristics of intertidal marsh sediments along an estuarine salinity gradient. *Limnology & Oceanography*, **44**, pp. 1693–1701, 1999.
[130] Stumm, W. & Morgan, J.J., *Aquatic Chemistry*, 2nd edn, Wiley Interscience: New York, pp. 625–640, 1981.
[131] Smith, K.S., Ranville, J.F., Plumlee, G.S. & Macalady, D.L., Predictive double-layer modeling of metal sorption in mine-drainage systems, *Adsorption of Metals by Geomedia,* (ed.) E.A. Jenne, Academic Press, pp. 521–547, 1998.
[132] Lacerda, L.D. & Abrao, J.J., Heavy metal accumulation by mangrove and salt marsh intertidal sediments. *Revista Basileira de Botanica*, **7**, pp. 49–52, 1984.
[133] Moberg, F. & Rönnbäck, P., Ecosystem services of the tropical seascape; interactions, substitutions and restoration. *Ocean & Coastal Management*, **46**, pp. 27–46, 2003.
[134] Ochieng, C.A. & Erftemeijer, P.L.A., The seagrasses of Kenya and Tanzania. *World Atlas of Seagrasses*, (ed.) E.P. Green & F.T. Short, University of California Press: Berkeley, 2003.
[135] D'Elia, C.F & Wiebe, W.J., Biogeochemical nutrient cycles in coral-reef ecosystems. *Coral Reefs*, (ed.) Z. Dubinsky, Elsevier: Amsterdam, pp. 49–74, 1990.
[136] Rheinhardt, R.D. & Brinson, M.M., An evaluation of the effectiveness of existing NCDOT wetland mitigation sites. CTE/NCDOT Joint Environmental Research Program, 2000.
[137] Bruland, G.L., An observational, geostatistical, and experimental assessment of edaphic properties and processes in created, restored, and natural wetlands of the Southeastern Coastal Plain. Ph.D. dissertation. Duke University; Durham, 2004.
[138] Environmental Laboratory. *Corps of Engineers Wetlands Delineation Manual*. U.S. Army Corps of Engineers, Wetland Research Program Technical Report Y-87-1: Washington, DC, 1987.
[139] Zedler, J.B., Progress in wetland restoration ecology. *Trends in Ecology & Evolution*, **15**, pp. 402–407, 2000.

[140] Zedler, J.B & Calloway, J.C., Tracking wetland restoration: do mitigation sites follow desired trajectories. *Restoration Ecology*, **7**, pp. 69–73, 1999

[141] Stolt, M.H., Genthner, M.H., Daniels, W.L., Groover, V.A., Nagle, S. & Haering, K.C., Comparison of soil and other environmental conditions in constructed and adjacent palustrine reference wetlands. *Wetlands*, **20**, pp. 671–683, 2000.

[142] Spencer, D.R., Perry, J.E. & Silberhorn, G.M., Early secondary succession in bottomland hardwood forests of southeastern Virginia. *Environmental Management*, **27**, pp. 559–570, 2001.

[143] Galatowitsch, S.M. & Van der Valk, A.G., Vegetation and environmental conditions in recently restored wetlands in the prairie pothole region of the USA. *Vegetatio*, **126**, pp. 89–99, 1996.

[144] Clewell, A.F. & Lea, R., Creation and restoration of forested wetland vegetation in the Southeastern United States. *Wetland Creation and Restoration: The Status of the Science*, (ed.) J.A. Kusler & M.A. Kentula, Island Press: Washington, DC, pp. 195–232, 1990.

[145] Verhoeven, J.T.A., Whigham, D.F., van Logtestijn, R. & O'Neill, J., A comparative study of nitrogen and phosphorus cycling in tidal and non-tidal riverine wetlands. *Wetlands*, **21**, pp. 210–222, 2001.

[146] Bishel-Machung, L., Brooks, R.P., Yates, S.S. & Hoover, K.L., Soil properties of reference wetlands and wetland creation projects in Pennsylvania. *Wetlands*, **16**, pp. 532–541, 1996.

[147] Campbell, D.A., Cole, C.A. & Brooks, R.P., A comparison of created and natural wetlands in Pennsylvania, USA. *Wetlands Ecology & Management*, **10**, pp. 41–49, 2002.

[148] Sopher, C.D. & Baird, J.D., *Soils and Soil Management,* Reston Publishing Company, Inc.: Reston, 1978.

[149] Poach, M.E. & Faulkner, S.P., Soil phosphorus characteristics of created and natural wetlands in the Atchafalaya Delta, LA. *Estuarine, Coastal & Shelf Science*, **46**, pp. 195–203, 1998.

[150] Shaffer, P.W. & Ernst, T.L., Distribution of SOM in freshwater wetlands in the Portland, Oregon area. *Wetlands*, **19**, pp. 505–516, 1999.

[151] Whittecar, G.R. & Daniels, W.L., Use of hydrogeomorphic concepts to design created wetlands in southeastern Virginia. *Geomorphology*, **31**, pp. 355–371, 1999.

[152] Duncan, C.P. & Groffman, P.M., Comparing microbial parameters in natural and constructed wetlands. *Journal of Environmental Quality*, **23**, pp. 298–305, 1994.

[153] Bruland, G.L. & Richardson, C.J., Spatial variability of soil properties in created, restored and paired natural wetlands. *Soil Science Society of America Journal*, **69**, pp. 273–284, 2005.

[154] Holling, C.S. & Gunderson, L.H., Resilience and adaptive cycles. *Understanding Transformations in Human and Natural Systems*, (ed.) L.H. Gunderson & C.S. Holling, Panarchy: Island Press: Washington, DC, pp. 25–62, 2002.

[155] DeBusk, W.F. Newman S & Reddy K.R. Spatial patterns of soil phosphorus in Everglades Water Conservation Area 2A. *Soil Science Society of America Journal*, **58**, pp. 543–552. 1994.
[156] Newman, S., Reddy, K.R., DeBusk, W.F., Wang, Y., Shih, G. & Fisher, M.M., Spatial Distribution of soil nutrients in a northern Everglades marsh: Water Conservation Area 1. Soil *Science Society of America Journal*, **61**, pp. 1275–1283, 1997.
[157] Richardson, C.J., Reiss, P., Hussain, N.A., Alwash, A.J. & Pool, D.J., The restoration potential of the Mesopotamian marshes of Iraq. *Science*, **307**, pp. 1307–1311, 2005.
[158] Bruland, G.L., Grunwald, S., Osborne, T.Z., Reddy, K.R. & Newman, S., Spatial distribution of soil properties in Water Conservation Area 3 of the Everglades. *Soil Science Society of America Journal*, **70**, pp. 1662–1676, 2006.
[159] Miller, J.R., Turner, M.G., Smithwick, E.A.H., Dent, C.L. & Stanley, E.H., Spatial extrapolation: the science of predicting ecological patterns and processes. *BioScience*, **54**, pp. 310–320, 2004.
[160] Kaly, U.L. & Jones, G.P., Mangrove restoration: A potential tool for coastal management in tropical developing countries. *Ambio*, **27**, pp. 656–661, 1998.
[161] Alongi, D.M., *Coastal Ecosystem Processes*, CRC Press: Boca Raton, 1998.
[162] Kusler, J.A. & Kentula, M.E., (ed.), *Wetland Creation and Restoration: The Status of the Science*. Island Press: Washington, DC, 1990.
[163] Kennish, M.J., (ed.), *Estuary Restoration and Maintenance: The National Estuary Programme*. CRC Press: Boca Raton, 2000.
[164] Sacco, J., Seneca, E.D. & Wentworth, T.R., Infaunal community development of artificially established salt marshes in North Carolina. *Estuaries,* **17**, pp. 489–500, 1994.
[165] Craft, C.B., Reader, J., Sacco, J.N. & Broome, S.W., Twenty-five years of ecosystem development of constructed *Spartina alterniflora* [Loisel] marshes. *Ecological Applications*, **9**, pp. 1405–1419, 1999.
[166] Berger, J.J., (ed.), *Environmental Restoration: Science and Strategies for Restoring the Earth*. Island Press: Washington, DC, 1990.
[167] Zedler, J.B., Coastal mitigation in southern California: the need for a regional restoration strategy. *Ecological Applications*, **6**, pp. 84–93, 1996.
[168] Vivian-Smith, G., Developing a framework for restoration. *Handbook for Restoring Tidal Wetlands*, (ed.) J.B. Zedler, CRC Press: Boca Raton, pp. 39–88, 2001.
[169] Bruland, G.L. & Richardson, C.J., Hydrologic, edaphic, and vegetative responses to microtopographic reestablishment in a restored wetland. *Restoration Ecology*, **13**, pp. 515–523, 2005.
[170] Teal, J.M. & Peterson, S., Restoration benefits in a watershed context. *Journal of Coastal Research*, **40**, pp. 132–140, 2005.
[171] Mitsch, W.J., Day J.W., Jr., Gilliam, J.W., Groffman, P.M., Hey, D.L., Randall, G.W. & Wang, N., Reducing nitrogen loading to the Gulf of Mexico from the Mississippi River basin: Strategies to counter a persistent ecological problem. *Bioscience*, **51**, pp. 373–388, 2001.

[172] Sklar, F.H., Chimney, M.J., Newman, S., McCormick, P.V., Gawlik, D., Miao, S., McVoy, C., Said, W., Newman, J., Coronado, C., Crozier, G., Korvela, M. & Rutchey, K., The ecological-societal underpinnings of Everglades restoration. *Frontiers in Ecology*, **3**, pp. 161–169, 2005.

[173] Leapold, L.B., Wolman, M.G. & Miller, J.P., *Fluvial Processes in Geomorphology*. W.H. Freeman & Co.: San Francisco, 1964.

[174] Strahler, A.N., Hypsometric [area-altitude] analysis of erosional topography. *Bulletin of the Geological Society of America*, **63**, pp. 1117–1142, 1952.