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Coastal Wetlands: Alteration and Remediation

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Preface

This volume of *Coastal Research Library* (*CRL*) deals with the general topic of coastal wetlands but specifically from within the purview of impacts that are deleterious to wetlands and kinds of restorative efforts that are deployed in attempts to correct wrongs resulting from human action. To this end, the volume is divided into three main parts: Part I, Impacts of Urbanization, Agricultural Occupation, Pollution, Climate Change, and Coastal Marine Influences; Part II, Impacts of Coastal Engineering and Environmental Degradation; and Part III, Restoration Techniques, Ecological Aesthetics, and Ecosystem Conservation (Sustainability and Biodiversity). These general subject area parts are in turn subdivided into chapters that are exemplars of degradation impacts or vignettes illustrating various approaches to restoration, either conceptually or in principle, and examples of new methodologies.

The geographical scope of this volume ranges from tropical to high-latitude coastal zones with various types of wetlands such as mangroves and salt marsh. A wide range of ecological considerations focuses on fisheries, intertidal benthic fauna, macrobenthic communities, and wildlife management. This selection of wide-ranging topics provides insight into the interconnectedness of various aspects of coastal wetlands. Provided here is a plethora of examples of successes and failures in attempts to correct the errors of human action when it comes to dealing with coastal wetlands. Sadly, many coastal wetlands around the world have been subjected to unwanted or unintended adverse impacts associated with urbanization, industrialization, and commercialization. With half of the world's coastal wetlands destroyed by such activities, it is imperative to absorb what is reported here in the following chapters that outline potential remediation efforts to save, conserve, or protect what is left of these valuable coastal ecosystems that are almost continually under threat from development.

Part I contains eight chapters that are examples of coastal wetlands adjacent to or overtaken by urbanization and agricultural occupation, which in turn result in degradation or destruction of coastal ecosystems. This dismal situation is further exacerbated by pollution, usually associated with urban development and/or agriculture, that compromises the integrity and in some cases the very survival of the remaining wetlands. Chapter 1 ("The Florida Everglades: An Overview of Alteration and Restoration"), by Charles W. Finkl and Christopher Makowski, discusses how urbanization, agriculture, and flood control destroyed about half of the Florida Everglades (a wetland of international importance [Ramsar Convention] and an international biosphere reserve [UNESCO]) and indicates failures of the world's most expensive reclamation effort that amounts to more than US$ 8 billion. Chapter 2 ("Recent Agricultural Occupation and Environmental Regeneration of Salt Marshes in Northern Spain"), by Ane García-Artola, Alejandro Cearreta, and María Jesús Irabien, deals with the reclamation of more than 50% of the original salt marshes that were degraded since the seventeenth century. This chapter illustrates how global temperate coastal wetlands with abundant sediment supply can be regarded as a soft adaptation measure that militates against consequences of climate change in the coastal zone. Chapter 3 ("Impact of Urbanization on the Evolution of Mangrove Ecosystem of the Wouri River Estuary [Douala, Cameroon]"), by Ndongo Din, Vanessa Maxemilie Ngo-Massou, Guillaume Léopold Essomè-Koum, Eugene Ndema-Nsombo, Ernest Kottè-Mapoko, and Laurant Nyamsi-Moussian, illustrates the deleterious effects of the urban environment on mangrove depletion around cities due to wood harvesting, sand extraction, and petroleum exploitation, in addition to coastal erosion and climate change. Unfortunately, the prognosis for a change in perception of mangrove degradation in this region is poor due to the absence of implementation of specific regulations to protect the mangrove forests. Chapter 4 ("Impacts of Coastal Land Use Changes on Mangrove Wetlands at Sungai Mangsalut Basin in Brunei Darussalam"), by Shafi Noor Islam and Umar Abdul Aziz Bin Yahya, continues in a similar vein by showing how increasing population pressure and economic development are detrimental to mangroves and salt marshes. Similar to the Everglades, there is the specter of conversion of water bodies and loss of open space where clearing of coastal mangroves and salt marshes result in a wide range of environmental issues and risks, not the least of which is severe pollution. This situation happens because the local authorities are unable to cope with the rapidly changing situations, internal resource constraints, and management limitations. Chapter 5 ("Land Use and Occupation of Coastal Tropical Wetlands: Whale Coast, Bahia, Brazil"), by Sirius O. Souza, Cláudia C. Vale, and Regina C. Oliveira, reports similar impacts in Brazil where coastal tropical wetlands are often compromised by the gradual expansion of population and economic cycles. Better planning proposals for land use and occupation are suggested for implementation. Chapter 6 ("Degraded Coastal Wetlands Ecosystems in the Ganges-Brahmaputra Rivers Delta Region of Bangladesh"), by Shafi Noor Islam, Sandra Reinstädtler, and Albrecht Gnauck, is perhaps the premier example in the world of population pressure on coastal wetland resources where 36.8 million people are living in the coastal region delta and who are dependent on coastal water resources. Unwanted impacts on the delta, tidal flats, mangrove forests, marches, lagoons, estuaries and other natural resources are elucidated in the light of ecosystem development and management strategies that are supposed to ensure communities with livelihood and sustainable development. Chapter 7 ("Handling High Soil Trace Elements Pollution: Case Study of the Odiel and Tinto Rivers and Accompanying Salt Marshes [Southwest

Iberian Peninsula]"), by Sara Muñoz Vallés, Jesús Cambrollé, Jesús M. Castillo, Guillermo Curado, Juan Manuel Mancilla-Leytón, and M. Enrique Figueroa-Clemente, verifies that salt marshes are one of the most prolifically heavy-metal polluted systems in the world. The interesting aspect of this chapter is its explanation of how key native halophytes are able to phytoextract or phytostabilize trace elements leading to the recovery of native prairies of low tidal marshes. Chapter 8 ("El Yali National Reserve: A System of Coastal Wetlands in the Southern Hemisphere Affected by Contemporary Climate Change and Tsunamis"), by Manuel Contreras-López, Julio Salcedo-Castro, Fernanda Cortés-Molina, Pablo Figueroa-Nagel, Hernán Vergara-Cortés, Rodrigo Figueroa-Sterquel, and Cyntia E. Mizobe, like the preceding chapters discusses adverse impact of human activities on coastal wetlands, in this case in central Chile, but additionally brings in the effects of natural disasters such as earthquakes, tsunamis, ocean swells, and ENSO. Field monitoring is also discussed with the objective of eventually implementing ecological restorations.

Part II contains five chapters and deals with direct impacts of coastal engineering and environmental degradation. The chapters here focus on clearly established and obvious links between construction works and degradation of coastal wetlands that are induced by ancillary effects. Chapter 9 ("Physical and Morphological Changes to Wetlands Induced by Coastal Structures"), by Germán Daniel Rivillas-Ospina, Gabriel Ruiz-Martinez, Rodolfo Silva, Edgar Mendoza, Carlos Pacheco, Guillermo Acuña, Juan Rueda, Angélica Felix, Jesús Pérez, and Carlos Pinilla, focuses on procedures that are used to better understand the relationship between modifications of coastal processes and the response of a coastal environment, in the case of civil works in the development of a new port in Barranquilla, Colombia. The interest here is to ascertain what changes in physical conditions will produce negative effects on the stability of natural systems in coastal wetlands. Chapter 10 ("Long Term Impacts of Jetties and Training Walls on Estuarine Hydraulics and Ecologies"), by Alexander F. Nielsen and Angus D. Gordon, probes inlet instabilities caused by the construction of jetties that in turn adversely impact the distribution of seagrass, salt marsh, and mangrove forests on the east coast of Australia. Chapter 11 ("Mangrove Degradation in the Sundarbans"), by Ashis Kr. Paul, Ratnadip Ray, Amrit Kamila, and Subrata Jana, investigates aspects of mangrove degradation in the Sundarbans and identification of contributing factors via extensive fieldwork, geospatial techniques, and factor analysis. This chapter shows hypersalinity, storm effects, fishery development, land erosion, and sediment deposition parameters are mainly responsible for mangrove degradations. Chapter 12 ("Assessment of Anthropogenic Threats to the Biological Resources of Kaliveli Lake, India: A Coastal Wetland"), by Krishnan Silambarasan and Arumugam Sundaramanickam, focuses on various threats to Kaliveli Lake, which is one of the largest wetlands in peninsular India and considered a wetland of international importance by the International Union for Conservation of Nature and Natural Resources (IUCN). Anthropogenic activities such as infringement from agricultural lands, wildlife poaching, loss of surrounding forests, increased salt pan and aquaculture farming, and recreation constitute important threats to the well-being of this wetland. This chapter also explores measures for conservation and protective management. Chapter 13 (“Egyptian Nile Delta Coastal Lagoons: Alteration and Subsequent Restoration”), by Ayman A. El-Gamal, identifies causes of wetland degradation in the Egyptian Mediterranean coastal region to be pollution, deterioration of water quality, eutrophication, habitat loss, overfishing, siltation, and climate change. Field studies are being conducted in efforts to determine management practices that will improve the resilience of these coastal lagoons.

Part III covers restoration techniques, ecological aesthetics, and ecosystem conservation with particular emphasis on sustainability and biodiversity. Although some of the previous chapters include discussion of remediation, this section highlights restoration efforts that promote sustainability and biodiversity in the broadest sense. This section is thus a logical follow-up to the previous two sections that primarily identified threats or risks to coastal wetlands. Determination or identification of the problem is obviously the first step in remediation; otherwise, it is impossible to remedy causes of unwanted conditions or situations. These chapters are examples of efforts in a diverse range of ecological setups where management strategies are proffered as means of conservation and protection within the realm of restoration and remediation. Chapter 14 (“Coastal Wetland Restoration: Concepts, Methodology, and Application Areas Along the Indian Coast”), by Ramasamy Manivanan, features a new concept that uses natural restoration techniques for coastal wetland restoration using the Chilika wetland ecosystem as a prototype. The idea here is to create conditions under which coastal ecosystem processes can withstand and diminish the impact of stressors. Chapter 15 (“Ecological Aesthetics Perspective for Coastal Wetland Conservation”), by LeeHsueh Lee, posits a new approach to the conservation of coastal wetlands where it is suggested that aesthetic preference provides a critical connection between humans and ecology. Promoted here is the prospect-refuge theory and the preference matrix of the bioevolutionary hypothesis, based on aesthetic experience, that could drive landscape change and pull with it ecological quality. Chapter 16 (“Estuarine Ecoclines and the Associated Fauna: Ecological Information as the Basis for Ecosystem Conservation”), by Mário Barletta, André R.A. Lima, Monica F. Costa, and David V. Dantas, is based on the definition of ecocline as a “gradation from one ecosystem to another where there is no sharp boundary between the two” where there are relatively heterogeneous communities influenced by gradual changes between river-dominated and marshlike waters. This chapter explains how to generate descriptors of reference conditions taking into account how human impacts affect coastal systems while providing steps to guarantee the sustainable use of estuarine resources. Chapter 17 (“Alteration and Remediation of Coastal Wetland Ecosystems in the Danube Delta: A Remote-Sensing Approach”), by Simona Niculescu, Cédric Lardeux, and Jenica Hanganu, demonstrates advantages of using remote sensing techniques to classify coastal wetland vegetation in the Danube Delta (a Biosphere Reservation), which was altered by human intervention in over one quarter of the entire delta surface. The random forest supervised classification algorithm was used to advantage for the Sentinel-1 and Sentinel-2 data collection. Chapter 18 (“Implementation of a Wildlife Management Unit as a Sustainable Support Measure Within the Palo Verde Estuary, Mexico: An Example of the American Crocodile [*Crocodylus acutus*]"), by Omar Cervantes, Aramis Olivos-Ortiz, Refugio Anguiano-Cuevas, Concepción Contreras, and Juan Carlos Chávez-Comparan, is a species-specific study in the Palo Verde Estuary (a Ramsar site) that recognizes that pollution, fragmentation of ecosystems, and habitat destruction due to human action incite the need for strategic management practices to encourage harvest sustainability. This chapter represents an opportunity to reconcile human activities with the environment based on an analysis made from the perspective of the conceptual Driving Forces-Pressure-State-Impact-Response model. Chapter 19 ("Mangrove Inventory, Monitoring, and Health Assessment"), by Ajai and H.B. Chauhan, identifies threats to mangroves from human activities (reclamation of mangrove areas for human habitation, aquaculture, agriculture, and port and industrial development) and shows how the use of remote sensing data can be used to develop a model for mangrove health assessment. The model developed here is demonstrated through a case study in India. Chapter 20 ("How Can Accurate Landing Stats Help in Designing Better Fisheries and Environmental Management for Western Atlantic Estuaries?"), by Mário Barletta, André R.A. Lima, David V. Dantas, Igor M. Oliveira, Jurandyr Reis Neto, Cezar A.F. Fernandes, Eduardo G.G. Farias, Jorge L.R. Filho, and Monica F. Costa, discusses fishery management in Brazilian estuaries while pointing out the need for better statistics to help avoid the impacts of overfishing. The main thrust of this chapter is the explanation of the need to improve fishery management by compliance of ecological data and biological research, obtaining robust data for landing stats, and establishing a social profile of the fishery community to build better rules of comanagement. Chapter 21 ("Returning the Tide to Dikelands in a Macrotidal and Ice-Influenced Environment: Challenges and Lessons Learned"), by Laura K. Boone, Jeff Ollerhead, Myriam A. Barbeau, Allen D. Beck, Brian G. Sanderson, and Nic R. McLellan, deals with the lessons learned from the design, implementation, and monitoring of salt marsh restoration in the upper Bay of Fundy, Canada. They found that the bioengineering species saltwater cordgrass (*Spartina alterniflora*) performed well and could be used again in similar situations. Chapter 22 ("Macrobenthic Assemblage in the Rupsha-Pasur River System of the Sundarbans Ecosystem (Bangladesh) for the Sustainable Management of Coastal Wetlands"), by Salma Begum, investigates a non-forestry product (benthic invertebrates) of the Sundarbans (the world's largest mangrove forest) and found that the combined effects of environmental and biological parameters influence relative species abundance. Chapter 23 ("Ecological Services of Intertidal Benthic Fauna and the Sustenance of Coastal Wetlands along the Midnapore [East] Coast, West Bengal, India"), the last chapter in the book, by Susanta Kumar Chakraborty, shows the value and functional contribution of benthic biodiversity (macrobenthos and meiobenthos) for the continuation of the Sundarbans mangrove estuarine complex. These bioindicators are indicative of the health of this disturbed coastal environment.

What is presented in this volume is but a snippet of the global situation confronting coastal wetlands today, which entails a universal threat from human action. The chapters illustrate the status of coastal wetlands from the geographical spread ranging from the tropics to high-latitudes via studies in Florida, Spain, Cameroon, Brunei Darussalam, Brazil, Bangladesh, Chile, Colombia, Australia, India, Egypt, Romania, Mexico, and Canada. These vignettes carry the common theme of coastal wetlands under stresses of variable types ranging dominantly from human action and less so from natural causes related to climate change. With about half of the world's coastal wetlands already destroyed by either urban expansion or the development of industrial and commercial infrastructure, the remainder are seriously threatened by a range of human activities (e.g., wood harvesting and loss of surrounding forests, sand extraction, petroleum exploitation, infringement from agricultural lands, wildlife poaching, increased salt pan and aquaculture farming, fishery development, land erosion, sediment deposition, and recreation) that usually fall under the radar of governing bodies that either turn blind eyes to what is happening or do not have the available resources to control the deterioration of the wetland ecosystems.

The other main theme of the various chapters is that the remaining coastal wetlands worldwide that have or continue to receive protection usually cannot remediate the damage that has already incurred. Although large areas have come under the "protection" of various types of statuses (e.g., Ramsar Convention, International Biosphere Reserve [UNESCO], International Union for Conservation of Nature and Natural Resources [IUCN]), this does not guarantee proper management by local authorities. Although the intent is laudable, the practicalities of the present world situation is that population growth is out of control in many regions that contain coastal wetlands. Human pressure on wetland resources is immense, and constructive efforts to protect, preserve, and conserve coastal wetland ecosystems are currently too weak to achieve goals that will maintain this valuable resource base for posterity. Several chapters point to new research that is being conducted into innovative ways of understanding and comprehending how these ecosystems function so they can be better managed. But the research and implementation of its findings are generally too slow compared to population growth with the result that coastal wetlands remain under threat from a wide range of human activities that eventually harken the death knell. What is required are more stringent protective measures that will secure a sustainable and unfettered future for the world's mangrove forests, fresh- and saltwater marshes, lakes, estuaries, and lagoons. All of the chapters in this book indicate in one way or another the present status and probable conditions of coastal wetlands as we look to the future.

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Part I
Impacts of Urbanization, Agricultural Occupation, Pollution, Climate Change, and Coastal Marine Influences

Chapter 1
The Florida Everglades: An Overview of Alteration and Restoration

Charles W. Finkl and Christopher Makowski

Abstract The Florida Everglades, currently designated as a Wetland of International Importance (Ramsar Convention), an International Biosphere Reserve (UNESCO), and a World Heritage Site in Danger (UNESCO), was administered around the turn of twentieth century by federal and state ditch and drain policies to 'reclaim' the coastal wetlands for urban sprawl, agriculture, and flood control. Today, the so-called 'river of grass' is only about half of its original extent; the remaining oligotrophic wetlands have been compromised by an ingress of nutrient-rich polluted and contaminated waters from agriculture and urban development. Furthermore, the spread of invasive flora and fauna have further compromised these wetland environments. In attempts to repair some of the damage wreaked upon this unique subtropical coastal ecosystem, numerous programs have been implemented to produce the world's most expensive reclamation effort that amounts to more than US$8 billion. Positionalities of special interest groups and hegemonial overthrusts by various governmental agencies have produced a bewildering array of projects that fail to address the real causes of degradation while treating only symptoms instead. Due to the lack of common sense approaches of the restoration that deal with causes rather than symptoms, such as further wetland alteration to naturalize surface flow patterns of water and the inability to hinder the introduction/spread of exotic alien species, the Florida Everglades has evolved into something quite different from pre-settlement conditions, with major doubts that the ecosystem can be put back together again.

Keywords Wetlands • Urban sprawl • Agricultural runoff • Oligotrophic • Eutrophification • Invasive species • Flooding • Everglades Agricultural Area • Pollution • Environmental remediation

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1.1 Introduction

For an estimated 5000 years, water in southern Florida Peninsula once flowed freely from the Kissimmee River southward to Lake Okeechobee and over low-lying wetlands leading to the estuaries of Biscayne Bay, Card Sound, the Ten Thousand Islands, and Florida Bay (Davis and Ogden 1994; Gunderson and Loftus 1993; Lodge 1994). This slow-moving, shallow expanse of water covered almost 28,500 km^2, and for thousands of years this complex hydrologic system created a finely balanced ecosystem of ponds, sloughs, sawgrass marshes, hardwood hammock woods, and forested uplands in the southern half of the state (Gleason 1984; McVoy et al. 2001; White 1970). However in 1847, only two short years after Florida was granted statehood from the Union, the first known proposal to drain the overflowed lands of the Lower Peninsula was put forth (Light and Dineen 1994). This proposal was based on the reconnaissance operations of Generals William S. Harney and Thomas S. Jessup, who were responsible for exploring areas that included the Everglades, Kissimmee River Valley, and Peace Creek (Fig. 1.1). Using these findings as leverage, the Secretary of the Treasury, H.J. Walker, under the administration of President James K. Polk, moved forward with drainage plans by appointing local St. Augustine resident, Thomas Buckingham Smith, the task of making a general inspection of the Everglades area and to report his findings to Congress (Light et al. 1995). Although Smith was neither a trained engineer nor an educated scientist, he accepted the challenge in June 1847. The results of five weeks of investigation include vivid descriptions of wildlife, as well as, detailed measurements and careful analyses of the terrain. Smith described the pristine Everglades with poetic prose:

> Imagine a vast lake of fresh water, extending in every direction, from shore to shore, beyond the reach of human vision; ordinarily unruffled by a ripple on its surface, studded with thousands of islands of various sized, from one-fourth of an acre to hundreds of acres in area…

Smith wrote of lilies and "other aquatic flowers of every variety and hue." He described the sensation of drawing near an island by saying "the beauty of the scene is increased by the rich foliage and blooming flowers of the wild myrtle and the honeysuckle" (Davis 1943). Obviously touched by the beauty of the Everglades, he also commented on the solitude and solemn silence pervading the marsh. He commented it all "awakened and excited curiosity feelings bordering on awe." Although Smith was captivated by the magnificence of the Everglades, he remained stubborn in his recommendation to drain the swamp and use it to grow citrus, sugar, and other produce usually imported from the West Indies at that time. He envisioned that the drainage of the Everglades would lead to a coast-to-coast canal, much as St. Augustine's founder Pedro Menéndez had once described. When Smith reported to the United States Senate in June 1848, he lobbied that the Everglades could be reclaimed with a sensible system of canalling and by deepening the various streams that flowed both east and west to the coasts (Light and Dineen 1994). He believed that drainage would insure the growth of a new agricultural empire in south Florida.

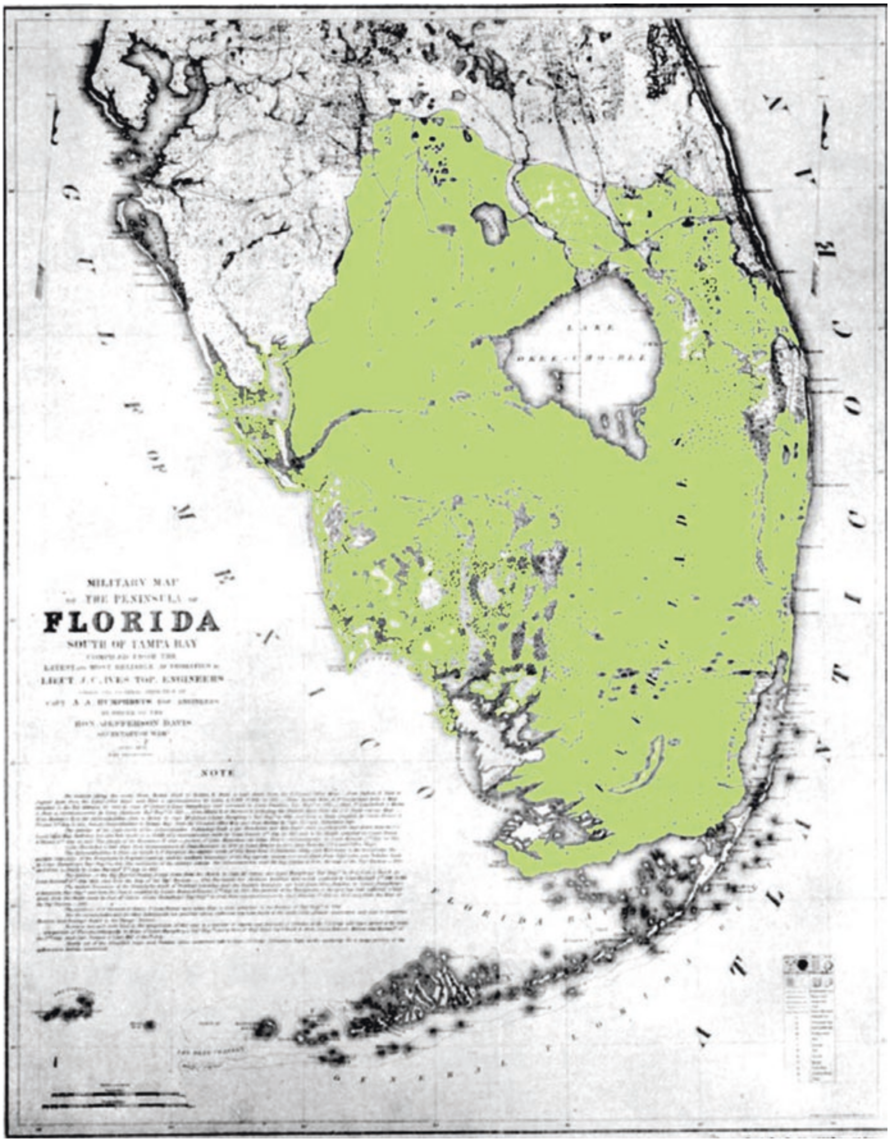

Fig. 1.1 United States military map of southern Florida drafted in the 1850s. Military campaign cartography, like this map created during the Seminole Wars, helped to understand the full geographic extents of the Everglades ecosystem. As shown above in a pristine state (*green* shading has been added to highlight this), this coastal wetland encompassed most of the southern tip within the Florida Peninsula and included not only all of the Everglades, but also the surrounding wetlands (e.g., Big Cypress Swamp and Kissimmee River Valley). Unfortunately, with this newly acquired knowledge of the Everglades, also came man's hubris to channelize and control this naturally dynamic environment (Credit: U.S. War Department)

His report to the 30th Congress was published as Senate Document No. 242 and proved to be the catalyst for the evitable drainage of the Everglades (Blake 1980; Snyder and Davidson 1994).

Even though the United States Congress immediately passed the "Swamp and Overflowed Lands Act of 1850," which conveyed the whole of Florida's swamp and overflowed lands to State ownership with the intent of draining them, it was not until the early twentieth century that large-scale alterations occurred (Finkl 1995; Light et al. 1995). The State Legislature wound up creating a Board of Drainage Commissioners and turned over to them the lands that were acquired in 1850 by the Swamp and Overflowed Lands Act (Light and Dineen 1994). This board was vested with government authority to:

> "...establish drainage districts and to fix the boundaries thereof in the State of Florida". Also, they were... "to establish a system of canals, levees, drains, dikes, and reservoirs…to drain and reclaim the swamp and overflowed lands within the State of Florida."

In 1906, the Trustees of the Internal Improvement Fund and the Drainage Commissioners purchased and operated dredges, and from 1906 to 1913, over 365 km of drainage canals were built, including the Miami, North New River, and South New River Canals by the Everglades Drainage District (EDD) (Light et al. 1995). Over the next 15 years, six large drainage canals and numerous smaller canals, totaling approximately 710 km, 75 km of levees, and 16 locks and dams were completed (Light et al. 1989; Sklar and van der Valk 2002). The newly constructed system of canals and locks was directly responsible for draining the northern and eastern parts of the Everglades region, with a majority of the canals originating at Lake Okeechobee and flowing easterly toward the Atlantic. This partial drainage of the Everglades subsequently opened the area to large farming settlements. By 1921, the population in the Okeechobee lake region was estimated to be around 2000 people, with newly cultivated lands in the glades yielding crops of sugar cane, tomatoes, beans, peas, peppers, and potatoes (Dovell 1947; Snyder and Davidson 1994).

Even with all the constructed canals, locks, and levees, which cost upwards of 20 million dollars, satisfactory drainage of Lake Okeechobee was not being met (Blake 1980). However, this was imperative since improper drainage did not afford sufficient protection to residents and farmlands from lake overflow during unusual weather. This impending hazard became all too apparent during the hurricane that struck Miami and the Lake Okeechobee region in 1926. Strong wind tides overflowed the banks of Lake Okeechobee causing over 200 deaths and millions in financial losses (Blake 1980; Chimney and Goforth 2001). Another hurricane in 1928 passed through the Palm Beach region on a path towards Lake Okeechobee with disastrous results. Wind-driven water off the lake, augmented by the torrential rains, overflowed the lakeshores and drowned approximately 2400 people, in addition to destroying a vast amount of property (Blake 1980). After these two catastrophes, the U.S. Federal government seized control under pressure from an outraged public. After a personal inspection of the area by President Herbert Hoover, the U.S. Army Corps of Engineers drafted a new plan that provided for the construction of floodway channels, control gates, and major levees along Lake Okeechobee's shores (Light et al. 1989). Construction of the Herbert Hoover Dike began in 1930 and in June of 1936, a national flood control policy was adopted by Congress (Light and Dineen 1994; Light et al. 1995). The Flood Control Act of 1936 established the policy that the Federal Government should:

> …improve or participate in the improvement of navigable waters or their tributaries for flood control purposes, if the benefits to whomsoever they may accrue, are in excess of the estimated cost, and if the lives and social security of the people are otherwise adversely affected.

Then, as if divine forces were at work to scorn the man-made decisions to excessively drain the Everglades, extreme dry spells of little to no precipitation occurred between 1931 and 1945. This successive lack of freshwater input resulted in a lowering of the groundwater, which then allowed seawater from the ocean to intrude on a multitude of wells that Miami and other coastal cities depended upon. In addition, the peaty, organic-rich soils of the Everglades that once depended upon the flooding ability of Lake Okeechobee to keep them moist were now subjected to desiccation (Snyder and Davidson 1994). Many areas caught fire in this dried-out state and were lost forever. The U.S. government was so focused on draining the Everglades, they never stopped to think that water conservation should be a necessary component of any drainage plan. In fact, structures designed to drain certain areas, thus protecting the people and farmland in time of floods, were also depriving the ecosystem of necessary moisture inputs during dryer periods.

Just as water conservation was being debated and proposed as part of the ongoing Everglades drainage plan in the aftermath of one of the worst droughts in Florida's history, the unthinkable happened. In 1947, over 254 cm of rain fell on south Florida, more than tripling the region's total recorded rainfall for 1945. The Great Flood of 1947 started on March 1 when a squall line dumped a welcome 15 cm of water on the parched agricultural lands of the upper Everglades. Rain was subsequently plentiful in April and May, and then in June became so heavy that chairman Dewey Hilsabeck called an emergency meeting of the Everglades Drainage District (EDD), which still had jurisdiction over a vast network of drainage canals, dams, levees, locks, water-control structures, and hurricane gates (Light and Dineen 1994). By opening the hurricane gates at Lake Okeechobee, for example, the EDD could drain excess water from the upper glades into the lake, which was purposely kept low for such an emergency (Light et al. 1989). But when the torrential rains came in 1947, the EDD was starved for funds to pursue its flood control program, and to compound the problem, a hurricane had formed offshore in the Atlantic. Slowly the hurricane worked its way across the Caribbean, battering Puerto Rico and the Bahamas, and then took direct aim at Florida. On September 17, it smashed into the mainland, with winds clocked at over 250 km h^{-1} at the Hillsboro Lighthouse. In Fort Lauderdale, the New River overflowed its banks and whitecaps broke over downtown, flooding luxury homes on the finger isles. Saltwater destroyed Dania's tomato crop and rainwater drowned the orange groves of Davie and the bean fields of Pompano Beach. Migrant workers near Lake Okeechobee were evacuated to higher ground. In West Palm Beach, the National Guard was called out and President Harry Truman declared Florida in a state of emergency. Especially destructive was the surge of water pouring into Lake Okeechobee from the Kissimmee River Valley, since the lake was already full and the water had just one place to go: southward toward Fort Lauderdale and Miami through already swollen canals. In the space of just 25 days, two hurricanes and a tropical disturbance dumped more water on a saturated area. When the rains finally ceased, 90% of southeastern Florida, from Orlando to the Keys, was literally underwater. Over 20,200 km^2 of water stretched from Lake Okeechobee across the Everglades and the Big Cypress Swamp southward through Broward and Dade counties, resembling an inland sea (Light et al. 1995). The total damage of this event was estimated by the Corps to be over $60,000,000 (Blake 1980).

Following the disastrous Great Flood of 1947, the problems of this area came to a climax and the government finally realized the current drainage protocols were not working. This flood, coupled with the experiences of the drought in 1945 and the intrusion of contaminating saltwater into potable and irrigational well water, made it imperative that corrective action had to take place immediately in order to prevent further loss of life, end excessive property damage, and to responsibly conserve water for beneficial uses during periods of drought. Therefore, in the wake of these catastrophes, a comprehensive plan for flood control and water conservation, which would encompass the entire Everglades and south Florida region, was put forth, thus starting the Everglades restoration movement in the interest of flood protection, drainage, and water control (Davis et al. 1994; Light and Dineen 1994; Light et al. 1995, 1989).

The following comments and discussion explores different components of the Everglades and the various projects under the guise of 'restoring' this ecosystem back to a resemblance of its former self. Will human action today be able to correct what took such a short amount of time to destroy? That is the question. The discussion section thus offers an injection of common sense when trying to answer that question and, possibly for the first time in the quest for atonement, presents a realistic solution to the possibility of "restoring" the Everglades.

1.2 Florida Everglades: Before and After Anthropogenic Alteration

The Everglades once covered almost 28,500 km^2 of southern Florida. Just a century and a half ago, water flowed down the Kissimmee River into Lake Okeechobee, then south through the Everglades marsh to the tidal flats of Florida Bay (Fig. 1.2), which was the final destination of the pure sheet flow (McVoy et al. 2011). Dubbed the 'River of Grass' (Douglas 1947) for the sawgrass that flourished throughout the marsh, the Everglades was a mosaic of freshwater ponds, prairies, and forested uplands that supported a rich plant and wildlife community (Davis et al. 1994; Douglas 1997). Known throughout the world for its abundant bird life, the

Fig. 1.2 View from Florida Bay looking northwards toward the southernmost margin of the Florida Everglades. The northern edge of Florida Bay or the southernmost edge of the Everglades in the interface or ecotone between the bay and the freshwater Everglades wetland ecosystems. In the northeastern part of Florida Bay, the ecotone is characterized by shallow marl soils that are dominated by scrub forests of red mangrove (*Rhizophora mangle*, background) and by open flats of broadhead spikerush (*Eleocharis cellulosa*). These estuaries (foreground) are nursery grounds for many species of juvenile fishes and invertebrates. The flow of freshwater from the Everglades area is the major determinant of the conditions in Florida Bay. However, the ecology of Florida Bay is currently showing signs of stress in the form of large die-offs of seagrass, with *Thallassium testudinum* (i.e. turtle grass) being affected the most. These die-offs are usually followed by a decline in conditions, including toxic blooms of phytoplankton and algae (Credit: C.W. Finkl)

Everglades was a safe haven for several species of large wading birds, such as the roseate spoonbill (*Platalea ajaja*), wood stork (*Mycteria americana*), great blue heron (*Ardea herodias*), and a variety of egrets (Ogden 1994). Also unique, is the brackish mix of salt and freshwater, making it the only place on Earth where alligators and crocodiles coexist side by side (Mazzotti et al. 2008).

Once a concerted effort to drain the Everglades was made in the early twentieth century, large tracts of swamp were transformed into productive farmlands and cities such as Miami and Fort Lauderdale began expanding along the coast and inland to the newly drained marshlands. As the population grew, so did the need to develop more land and provide flood control to the new residents of southern Florida. In 1948, the U.S. Congress created the most expansive water management system in the world, and today, this network of man-made canals, levees, and water control structures channel and discharge approximately 6400 m^3 of water daily from the Everglades into the ocean (Light and Dineen 1994; Perry 2008) (Figs. 1.3 and 1.4).

Fig. 1.3 Current map showing the segmentation of the Everglades in southern Florida. The Kissimmee River flows from the north into Lake Okeechobee, which is bordered immediately to the south and southeast by the Everglades Agricultural Area (EAA, *dark orange*). As water, and discharged effluent, flow southward from the EAA, water conservation areas, Loxahatchee National Wildlife Refuge (*NWR*), Big Cypress National Preserve (*NP*), Everglades National Park (*NP*), Biscayne National Park (*NP*), and Florida Bay are all negatively impacted. Overall, the Everglades ecosystems that remain today are only a fraction of the original extent (see Fig. 1.1) (Credit: NOAA)

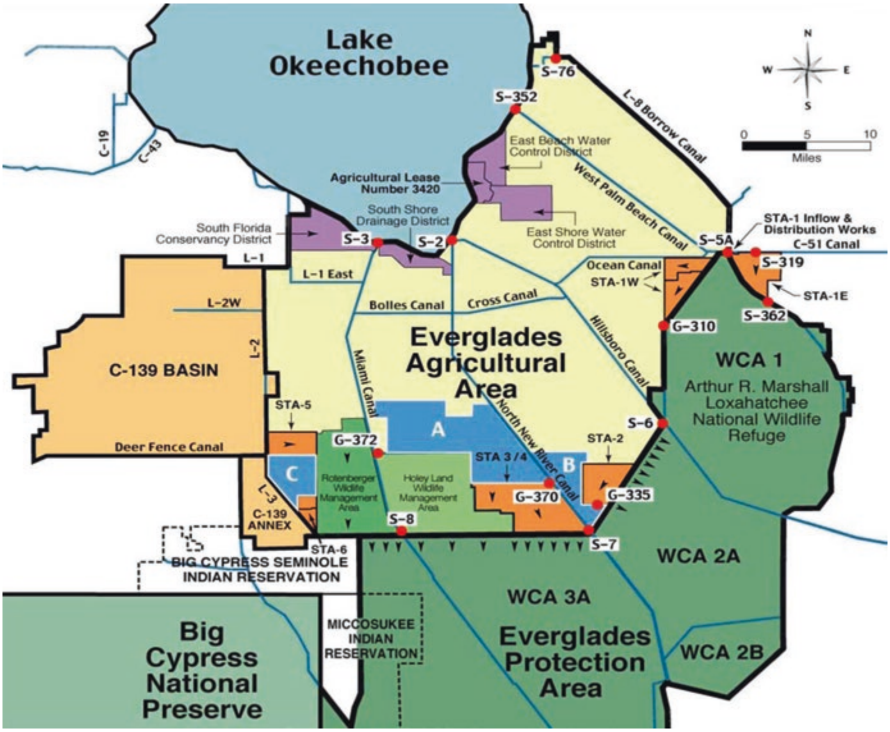

Fig. 1.4 Detailed managerial layout of the area surrounding the Everglades Agricultural Area (*EAA*), which includes a vast network of man-made canals, levees, and stormwater treatment structures to maintain water quality and control measures throughout the Everglades ecosystem. Notice how many different canals, basins, refuges, management areas, water conservation areas (*WCA*), stormwater treatment areas (*STA*), national preserves, and protection areas are needed to properly regulate one isolated agricultural area in the EAA (Credit: SFWMD)

However, this loss of water has forever changed the natural characteristics of the once pristine marsh. As the water receded, so did essential habitats of wading birds, fish, and dozens of animals. Saltwater intruded farther into the marsh from the coastal ocean and nonpoint-source pollution runoff flowed in from neighboring farms and cities. Changes in water quality also stifled the growth of native plants, allowing exotic species to take root and persist. As a result of these alterations continuing for the past 70 years, the Everglades is currently only half the size it was before the State of Florida was established (Browder and Ogden 1999; Ogden et al. 2005).

1.2.1 Geology

The Everglades sits on what geologists have named the "Floridian Plateau," an expanse of crust that includes the emerging portion of the tectonic platform (i.e. the Florida Peninsula) and the adjacent continental shelf region (Bryan et al. 2008;

White 1970). It is postulated that this limestone platform that has been developing since the late Triassic about 180 million years ago, mainly because of the plateau's composition, which includes a thick section of mostly under-formed carbonate rocks (Hine 2013; Hoffmeister 1974; Randazzo and Jones 1997). It is also believed that the development of the platform was controlled by regional subsidence of the passive margin and eustatic sea-level changes that allowed the deposition of a thick section of carbonate rock over many millions of years (Hine 2009; White 1970).

When evaluating the structure and stratigraphy of Florida, sedimentary formations resemble an anticline that plunges in a southeasterly direction from the Ocala limestone dome (Randazzo and Jones 1997). Just as the Everglades give way to the Gulf of Mexico, this Eocene limestone dips to a depth of approximately 365 m. Younger limestone formations of the Oligocene and Miocene ages, which become thicker as they approach the coastlines, can be found atop the Ocala formation (White 1970). Both the Pliocene and Pleistocene strata, being that they were the last to form, do not extend more than 50 m beneath the surface and are in fact are exposed in many places in and around the Everglades (Bryan et al. 2008; Hine 2013; Randazzo and Jones 1997).

The deposition of two Pliocene formations located in the Everglades, the Caloosahatchee marl and the Tamiami limestone, most likely began with an encroachment of the sea that extended beyond the latitude of Lake Okeechobee (Mansfield 1932; White 1970). It was this deposition of the Pliocene material that helped shape the present day topography of the Everglades. For example, the Caloosahatchee marl, exposed along the banks of the Caloosahatchee River, consists of mostly fine sand with a large proportion of unbroken shells (Mansfield 1932). The Munsell Color Index for the marl ranges from white to light gray, blue, or yellow. Deposited in a warm and shallow sea, this soil underlies a large portion of south Florida below the 27th parallel. Water from the marl has been recorded to have a high chloride content, mostly due to Pleistocene sea invasions and the influence of the Miocene rocks underneath it (Allison 1943; Cohen and Spackman 1984). The Tamiami limestone, on the other hand, comes to the surface in the lower reaches of the Big Cypress Swamp in a wedge-shaped formation, inclining toward the coast (White 1970). The calcareous sandstones and sandy limestones found in this formation are among the most permeable ever recorded by geologists (McVoy et al. 2011).

The first of the Pleistocene formations to be deposited, which are currently present in the southern and eastern parts of the Everglades, was the Miami Oolite (Hine 2013; Randazzo and Jones 1997). The oolite varies in thickness and is overlain by sand, muck, and marl. It can appear as a white or light yellow limestone with very high porosity as part of outcroppings along the east coast and in the banks of short rivers (Bryan et al. 2008). Additionally, the Anastasia and Pamlico formations, found mostly in the coastal ridge on the Atlantic and along the eastern borders of the Everglades, are composed of sand, sandy limestone, and calcareous sandstone (White 1970). In those areas where sands of these formations are mixed with organic soils of the Everglades, crop production is very optimal and sought after (Fig. 1.5; Finkl and Restrepo-Coupe 2007).

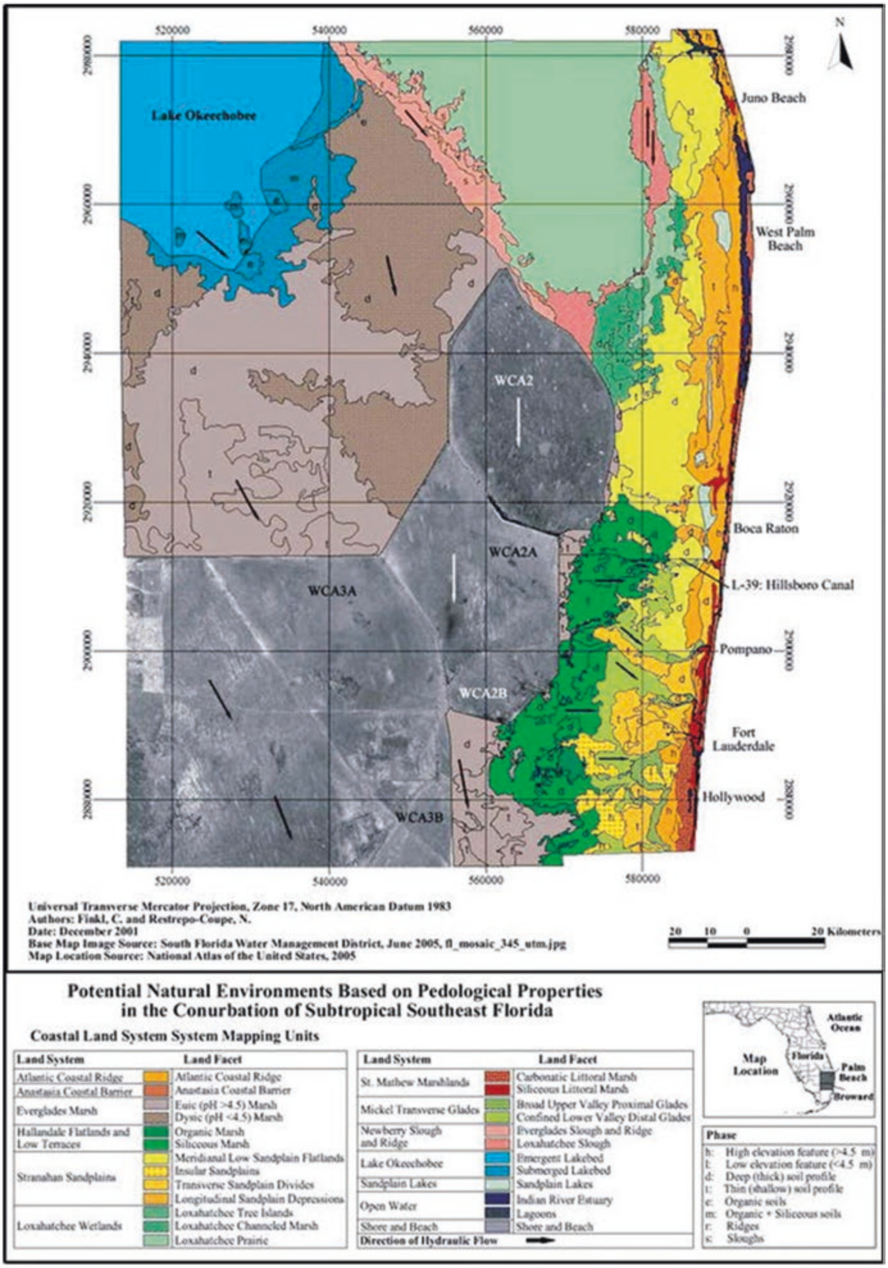

Fig. 1.5 Example of potential natural environments in and around the eastern and southern parts of the Florida Everglades. Because these environments have been modified by human action, they are referred to as potential natural environments based on the supposition that these are the ecosystems that existed, based on interpretations of soils, prior to human intervention and to which they could theoretically re-equilibrate. In some cases this is impossible due to urbanization, but in other cases the potential remains in protected areas such as the Loxahatchee Wetlands. This map includes 12 coastal land systems, 25 land facets, and 8 phases. The Water Conservation Areas were not mapped so that satellite image patterns could be visualized in the cognitive mapping process. This study area covered more than 6000 km^2 covered by water (Lake Okeechobee) and coastal wetlands (lakebed, freshwater marsh, littoral marsh, prairies, sloughs, lagoons, and estuaries) (Credit: Finkl and Restrepo-Coupe 2007)

The most recent geological formation in the Everglades is the Lake Flirt Marl, composed of soft gray marl or calcareous mud, which is almost universally present under the deeper muck of the upper Everglades (Hoffmeister 1974). The Lake Flirt Marl is unique because of its impermeability, which prevents the percolation of ground waters in the organic soils of the region. The top 15 m of rock strata in the northern half of the Everglades is relatively impermeable and restricts that northern section to natural water control through confining layers (McVoy et al. 2011). However, in the lower half of the Everglades, the strata becomes looser and highly water-bearing as the rim is approached, thereby, allowing canals to cut through the permeable strata and drain adjacent lands (McVoy et al. 2011).

The Everglades are presently bounded on the eastern side by a coastal fringe of sand dunes, on the western side by the Ocaloacoochee Slough and the Big Cypress Swamp, and extend to the southern and southwestern coast of Florida, where the salt-water marshes and mangrove swamps form the southern border (Gleason 1984; Myers and Ewel 1990). Currently, the glades constitute the third, or downstream, unit of the watershed that forms the interior of the Florida Peninsula below the twenty-eighth parallel (Light and Dineen 1994). The first, or tributary, unit of this drainage basin, the Kissimmee River, drains about two-thirds of the area. The second, or middle, unit of this watershed is Lake Okeechobee, a shallow body of fresh water whose surface elevation is now regulated between 4 and 6 m. The combined area of all three units once equaled 25,000 km^2 (Light and Dineen 1994). Under natural conditions, prior to the advent of artificial drainage, the outflow of the waters from the Kissimmee River and Lake Okeechobee naturally passed onto, and through, the Everglades (Davis 1943; McVoy et al. 2011).

Inside the Kissimmee-Everglades watershed basin is a wide and flat plain, flanked by natural drainage ways that have imperceptibly divided the area into a sawgrass plain bordered by a series of ridges and sloughs. The vegetative accumulation, or soil, varies from an average thickness of eight feet at Okeechobee's shores to the thinnest of deposits at the sides of the Everglades (Allison 1943; Cohen and Spackman 1984). The line of demarcation between the glades and adjoining areas is extremely irregular, which extends as a stretch of grassland that may be under 0.6 m of water at the end of the rainy season, but in most years is dry enough for the cultivation of a winter vegetable crop (Sklar et al. 2001). The actual boundary between the Everglades and the adjoining prairie is usually marked where the sedges of the glades are met by true grasses, cypress, salt marsh, or mangroves (Myers and Ewel 1990; Rutchey et al. 2008) (Fig. 1.6).

1.2.2 Nutrients

Even though water quality data prior to the 1940s is limited, the Everglades are believed to have been an oligotrophic environment, characteristic of extremely low levels of nutrients, such as phosphorus (P) (Davis and Ogden 1994). These low surface-water concentrations are naturally self-regulating and are a major reason

Fig. 1.6 Example of the Florida Everglades ecosystem under wet conditions during the rainy season when tree islands are surrounded by water. Wet conditions and higher temperatures bring about significant changes in the Everglades landscape. The wet season, which starts around the middle of May and continues through to November, is followed by a period of very little rain. The winter dry season or drought period extends from early December through late April or early May. Water levels in the Everglades can thus change drastically from month to month. Periphyton and other algae to proliferate during the wet season in contaminated nutrient-rich water. Note the discolored water in the foreground. Periphyton, a complex mixture of algae, cyanobacteria, heterotrophic microbes, and detritus, is attached to submerged surfaces in the Everglades aquatic ecosystem. This algal mixture, which is an important food source for invertebrates, tadpoles, and some fish, can also absorb contaminants by removing them from the water column and limiting their movement through the environment. Because periphyton is an indicator of water quality, responses of this community to pollutants can be measured at a variety of scales representing physiological to community-level changes (Credit: C.W. Finkl)

why the Everglades were able to maintain a delicate ecological balance that includes ecotonal successions between sawgrass, tree islands, and open water (Sklar and van der Valk 2002; Todd et al. 2010). For example, phosphorous is naturally removed from the system by accumulating in soils, algae, and plants, such as arrowhead, peat moss, and pickeral weed (Craft et al. 1995). The Everglades supports a variety of rare, threatened, and endangered species that have adapted to these low nutrient concentrations, as well as, other physio-chemical conditions (e.g., low dissolved oxygen [DO] levels) that are characteristic of the system (Davis and Ogden 1994; Lodge 1994), and whose survival depends on the original cycle of water and nutrients, particularly low levels of phosphorous (Gunderson and Loftus 1993). Historically, rainfall alone provided the primary source of nutrients into the Everglades (Newman et al. 1996).

However, agricultural development and urbanization since the 1800s have not only claimed almost two-thirds of the natural Everglades (only about 6000 km^2 now exist in their natural form) but have also dramatically increased phosphorus levels in surface water, at times exceeding the acceptable limit of 10 mcg/l (Childers et al. 2002; Gaiser et al. 2005; Payne et al. 2000). Improved drainage in the region during the turn of the twentieth century permitted a large tract of wetland (2830 km^2) immediately south of Lake Okeechobee, now referred to as the Everglades Agricultural Area (EAA), to be developed for agriculture. EAA runoff flows directly into the Everglades and carries elevated levels of nutrients and other constituents (e.g., total suspended solids, BOD, pesticides, bacteria) and causes decreased levels of dissolved oxygen, increased sedimentation, and food chain disruptions (Chimney and Goforth 2001; Payne et al. 2000; Snyder and Davidson 1994). Pollutant loads associated with stormwater runoff can be exceptionally high, with heavy phosphorous concentrations causing eutrophication in many areas (Gaiser 2009). This, in return, allows for the proliferation of species that disrupt the balance of the ecosystem, such as harmful algal blooms (Gaiser et al. 2005). Additionally, the chemicals in various agricultural fertilizers can lead to accumulations of toxic mercury in fish, birds, reptiles, and even mammals, including the endangered Florida panther (Doren et al. 2009a, b; Evans and Crumley 2005).

These changes in water quality and other environmental disturbances associated with agricultural development in the Everglades were first identified as early as 1938 (Snyder and Davidson 1994). Overall, alterations to the nutrient input of the Everglades have resulted in widespread changes to the ecology of the ecosystem, with dramatic declines in the size of wading bird populations (Frederick and Collopy 1989; Ogden 1994) and the invasion of species that include the cattail (*Typha* sp.) and duckweed into native sawgrass and slough habitats (Doren et al. 2009a, b; Miao and Sklar 1998; Newman et al. 1996; Rader and Richardson 1994). Any viable solution to improve conditions in the Everglades needs to reduce phosphorous concentrations from EAA runoff and improve the region's hydroperiod and hydropattern (Gwin et al. 1999; Payne et al. 2000). The South Florida Water Management District's long-term strategy for preserving and restoring the Everglades is intended to meet these needs by allocating approximately 167 km^2 of wetlands, referred to as Stormwater Treatment Areas (STAs), to treat runoff from the EAA and adjacent drainage basins in an effort to reduce devastating nutrient loading into the system (Guardo et al. 1995; Walker 1995).

1.2.3 Flora and Fauna

Even though the Everglades are situated in a semi-tropical climate, yearly variation between wet and dry seasons, as well as fluctuations in summer high temperatures (e.g., ~36 °C) and winter low temperatures (e.g., ~0 °C), has selected specific flora and fanua species that can tolerate such extremes (Gunderson and Loftus 1993). At first glance, one will observe that it is a region without many trees, rather the Everglades are dominated by grasses, sedges, reeds, rushes, and other herbs that take root in horizons of peat, marl, and even sandy soils that are nearly flooded or wet most of the year (Finkl and Restrepo-Coupe 2007; Gleason 1984; Lodge 1994).

Plant ecologists refer to these types of marshes as “low moors” which are aptly named for their resemblance to similar to bog-like environments with high peat accumulates (Douglas 1997; Rutchey et al. 2008).

In terms of floral-ecosystem classification, the Everglades have been compartmentalized into six general categories: (1) the sawgrass marsh plains of northern and central Everglades regions totaling approximately 4050 km^2; (2) the sawgrass and wax myrtle, or bay-berry thicket, areas along the flanks of the central Everglades plain equaling around 970 km^2; (3) slough and tree-island areas north of the Hillsboro Canal and west of the Miami Canal with approximately 3135 km^2; (4) mixed marshes and wet prairies east and west of Tamiami Trail (trans-Everglades highway at the latitude of Miami) totaling around 1215 km^2; (5) custard apple and willow-elderberry zone along the eastern and southern shores of Lake 0keechobee equaling approximately 570 km^2; and (6) bordering prairies with scattered hammocks and stands of trees along the borders of the Everglades with about 585 km^2.

A majority of the vegetation growing in the Everglades is derived from aquatic families, with sawgrass (*Cladium* spp.) being the predominant species to take root (Douglas 1997). Not actually a ‘true’ grass, as it is in fact a sedge, sawgrass can reach lengths over 3 m and have leaves lined with razor-sharp edges. Growing in tandem with the sawgrass along the canals of the Everglades, the water hyacinth (*Eichhornia crassipes*) has become one of the most abundant naturalized plants to grow in these hydrated soils (Gunderson and Loftus 1993). However, in the sloughs and deeper waters of the Everglades, where out-competition by sawgrass is diminished, other grasses and water plants thrive (Gunderson and Loftus 1993). For example, gama grass, water arums, giant foxtail, spider lilies, common reeds, boneset, elegant thalia, bull-rushes, and maiden cane help to form a vast area of native and naturalized vegetation growing on the lower mainland of Florida, many of which are located on the glades and island hammocks below Lake Okeechobee (Davis 1943; Lodge 1994; Sklar et al. 2001). The amazing growth of this flora, sometimes to a canopy height of nearly 7 m, often makes the Everglades look like an immense sea of green (Douglas 1997).

Along the eastern and western edges of the Everglades, marsh shrubs and trees grow predominantly in isolated clumps or on small islands (Sklar and van der Valk 2002) (Fig. 1.7). Most of these island banks are lined with a 3–5 km buffer of custard apple (*Annona reticulata*) and elderberry (*Sambucus* spp.), which can usually be found adjacent to one another (Wu et al. 2002). Growing on the small islands along the edge of the Everglades are the cocoplum (*Chrysobalanus icaco*), which forms a dense green foliage on the streams and once provided a dietary purple and white fruit to the native Seminole tribes (Davis and Ogden 1994). Other principal flora species found here include the amphibian willow, wax myrtle, and swamp bay. On the larger tree islands, or swamp keys, in the area, grow the live oak, cypress, maple, bay, and a few of the long leaf pine (Sklar and van der Valk 2002). Of special interest is one of the most utilitarian flora members of the Everglades community called the cabbage palm (*Sabal palmetto*). This palm can grow to great heights on the islands of the glades and adjoining prairies. The trunk has been timbered for building purposes, the leaves for thatch, and the tender bud at the heart of the

Fig. 1.7 Tree islands in the southeast Florida Everglades. The blanket term "tree island" refers to clumps of trees that are surrounded by smaller stature vegetation (i.e. sawgrass marshes). Tree islands tend in general to occur in plan view in the form of an elongated teardrop, with the long axis parallel to the direction of freshwater sheetflow. The bulbous end or "head" of the teardrop meets the waterflow. The head of the tree island is usually a tropical hardwood hammock, which rarely becomes inundated, that commonly contains tree species such as gumbo-limbo (*Bursera simarumba*), sugarberry (*Celtis laevigata*), mastic (*Mastichodendron foetidissimum*), pigeon plum (*Coccoloba diversifolia*), white stopper (*Eugenia axillaris*), paradise tree (*Simarouba glauca*), plus many other species. These hardwood hammock grade into bayhead forests in the direction of waterflow and as soil elevation decreases (Credit: C.W. Finkl)

uppermost end of the trunk serves as a succulent food when cooked properly (Gunderson and Loftus 1993).

Using a lot of these tree species as growing foundations, a large variety the climbing plants can be found in the Everglades. Among these species are a wide array of vines, orchids, and other epiphytes that include wild grapes, hunter's vine (whose sap filled stem is potable for drinking), cockspur, marsh orchids, Spanish moss, bromeliads, resurrection fern, and climbing brambles (Davis 1943; Douglas 1997).

The fauna of the Everglades is a unique collection of animals that includes multiple species of birds, mammals, fish, insects, and amphibians. However, perhaps the creatures that conjure up an immediate image inside the mind's eye at the mention of the Everglades are the reptiles, the largest of which are alligators and crocodiles. For example, American alligators (*Alligator mississipiensis*) found in the Everglades can grow to 4.4 m, weigh over 450 kg, and have been known to feed on practically any animal that passes within its reach (Mazzotti et al. 2008). Additionally, the Everglades are also home to a wide variety of snakes (Lodge 1994). The cottonmouth, or water moccasin (*Agkistrodon piscivorus*), which thrives in wet, swampy lands, is among the most predominant. This snake contains very poisonous venom and travelers through the glades have to be wary of its presence. The eastern diamondback (*Crotalus adamanteus*) and ground (*Sistrurus miliarius*) rattlesnakes are also dangerous and encountered occasionally, but both of these rattler species prefer a dry ground habitat. Garter, water, black racer, gopher, coachwhip, and green tree snakes are additional members of the snake family encountered in the Everglades. Lesser reptiles include lizards of the gecko, anole, iguana, and skink families (Gunderson and Loftus 1993). And finally, turtles, both freshwater and marine, and tortoises are also considered residents of Everglades National Park. These include the Florida snapping turtle, gopher tortoise, diamondback terrapin, green sea turtle, loggerhead sea turtle, and Atlantic hawksbill sea turtle (Lodge 1994; Myers and Ewel 1990).

Just as abundant as the reptile species, so are a wide variety of birds, fish, and insects. The diversity of bird species found in south Florida's Everglades is especially rich for a number of reasons, which include the presence of a mild climate, suitable water and food resources, and a protected resting area for migratory species. Ornithologists have noted that almost every bird species that frequents the states along the eastern seaboard are found in or near the Everglades at some time of the year (Ogden 1994). Only those distinctive bird members found in the Everglades year round are those who prefer a watery plain for a natural habitat. Examples of bird fauna found in or around the glades include the heron, crane, bittern, grebe, water turkey, duck, turkey vulture, limpkin, hawk, osprey, rail, gallinule, coot, dove, kite, thrush, and cardinal (Lodge 1994; Myers and Ewel 1990). No longer common among these species, the roseate spoonbill (*Platalea ajaja*) and the American flamingo (*Phoenicopterus ruber*), whose rose tint is derived from the carotenoid pigments they digest as dietary byproducts, have all but disappeared from south Florida due to human intrusion (Chimney and Goforth 2001). Likewise, the snowy egret (*Egretta thula*) and the white ibis (*Eudocimus albus*), once the prey of plume hunters and threatened with extinction, are now justly protected by law throughout the Everglades (Frederick et al. 2008).

Lake Okeechobee and the Everglades serve as ideal habitats for many fish species, especially in times of high water. During these flood-like hydroperiods, the fish usually venture into weedy sections bordering the lake and the glades in order to enjoy new feeding grounds. Among the fish species found in the various lakes, pools, and sloughs of the Everglades are catfish, shiners, kill fish, sunfish, bluegill bream, dogfish, alligator gar, and numerous minnows are found in the lakes, pools, and sloughs of the Everglades (Davis and Ogden 1994; Lodge 1994). One fish species that once thrived in the Everglades before the beginning of drainage operations was the black, or largemouth, bass (*Micropterus salmoides*). Specimens of over 10 kg were common before overfishing and water relocation in the glades occurred (Rader and Richardson 1994).

The Everglades are teeming with many species of invertebrates, such as insects, crustaceans, and gastropods. Among the insects are a large variety of spiders, termites, dragonflies, roaches, grasshoppers, beetles, moths, butterflies, ants, wasps, bees, hornets, centipedes, scorpions, flies, and mosquitoes (Lodge 1994). Whereas, gastropods, predominantly in the form of tree and marsh snails who display many different colors in their shells (Darby et al. 2002), and crustaceans, found mostly as crawfish species, both provide an important dietary niche in the complex food webs of the Everglades (Davis and Ogden 1994; Trexler and Goss 2008).

The mammals found within the Everglades are actually very few in number. Other than birds, fish, insects, and reptiles, the center of the Everglades is nearly devoid of other animal life (Davis and Ogden 1994). Deer will sometimes graze in open spots on the tender grass, Florida panthers are ever rarely seen, and an occasional wildcat will occupy one of the tree islands in order to prey upon rats and mice. Possums and raccoons can be found along the borders of the Everglades, but are most likely tempted to move towards urbanized areas where food sources are more abundant. The one mammal most likely adapted to the Everglades was possibly the Florida freshwater otter (*Lontra canadensis*), however, relentless hunting for their valuable pelts has all but eliminated this animal from its natural habitat (Gleason 1984).

The delicate balance of the Everglades ecosystem allows many species of flora and fauna to flourish. However, due to human alteration, that balance has been disrupted on a large scale. Nutrient pollution loading and overdrainage, while being major contributors to the problem, are not the only threats the Everglades faces. Alien species of plants and animals that have been introduced to the Everglades by humans can be just as destructive. Exotic plants, such as *Melaleuca*, Brazilian pepper (*Schinus terebinthifolia*), and Australian pine (*Casuarina* spp.) (D'Antonio and Meyerson 2002; Doren et al. 2009a, b), and invasive animals, such as Burmese pythons (*Python bivittatus*) and Nile monitors (*Varanus niloticus*), displace or eradicate native species and threaten to disrupt the entire ecosystem balance (LeSchiava et al. 2013) (Fig. 1.8).

Fig. 1.8 An example of a Nile monitor (*Varanus niloticus*) found within the Everglades. These non-native lizards can grow over 2 m in length and weigh close to 7 kg. Nile monitors are semi-aquatic and are often found basking or foraging for food along canal banks in South Florida. They feed on a variety of mammals, birds, reptiles, amphibians, fish, and eggs. Because the Nile monitors have very few predators, their population size and impact on the Everglades ecosystem cannot be naturally regulated. Their natural range is found in the Nile River delta and Sub-Saharan Africa (Credit: FWC)

1.3 Proposed Everglades Restoration Projects

1.3.1 Central and South Florida Project for Flood Control

The Central and Southern Florida (C&SF) Project, first authorized by Congress in 1948, was a multi-purpose project consisting of approximately 1600 km of levees, 1150 km of canals, and almost 200 water control structures (Finkl 1995; Light et al. 1995). This effort served as the genesis of water management in south Florida with the following goals in mind: adequate flood control; ample water resources for municipal, industrial, and agricultural uses; water influxes for Everglades National Park (ENP); the prevention of saltwater intrusion; and the protection of fish, plant, and wildlife resources (Perry 2008). Unfortunately, as was seen over time, the C&SF proved highly beneficial to many human interests involved with agriculture, water supply, and flood control, but not to wildlife residing in the Everglades ecosystem.

The C&SF Project was under the direct supervision of the Central and Southern Florida Flood Control District (FCD) and centered around three main components (Gunderson and Light 2006). First, there was the establishment of a perimeter levee through the eastern portion of the Everglades. The function of this levee was to sever approximately 16% of the eastern Everglades from the interior (Light et al. 1995). This allowed lands farther east to be protected from direct Everglades flooding through the blockage of sheet flow that would naturally occur. Second, the C&SF Project designed a large area of the northern Everglades, just south of Lake Okeechobee, to be drained and managed for the sole purpose of agriculture (Snyder and Davidson 1994). Aptly named the Everglades Agriculture Area (EAA), it encompassed about 27% of the historic Everglades and was a major driving economic factor of the C&SF Project (Davis and Ogden 1994). Third, the remaining Everglades between the EAA and the ENP were designated exclusively as water conservation areas (WCAs) (Kiker et al. 2001).

At that time, the newly formed FCD, which held their first official meeting in West Palm Beach in July 1949, was charged primarily with providing flood protection to the area, thus, they did not adhere to the natural hydrologic or geographic features of the Everglades (Rizzardi 2001; Sklar et al. 2005). In its first 5 years of existence, the FCD acquired, through either title or easement, approximately 3500 km^2 needed for construction and water storage, as dictated by the C&SF Project (Perry 2008). The acquired land, which cost \$1.5 million back then, included Everglades wetlands in the WCAs. The FCD was charged with keeping an optimistic schedule for the C&SF Project, which called for all the public works features to be completed within 10 years at a cost of \$200 million (Clark and Dalrymple 2003; Holl and Howarth 2000). However, as of 1960, \$68 million had been expended, with only about 20% of the work complete (Light and Dineen 1994). In the meantime, Florida's population was booming, increasing from three million to five million over the previous decade. Turner Wallis, who served as the FCD's first executive director from 1949 to 1956, called for a more rapid pace for the C&SF Project, undermining the importance of protection for Everglades fish, plant, and wildlife resources (Light et al. 1995). Between 1950 and 1960, the U.S. Army Corps of Engineers dug over 200 km of canals and improved or built over 450 km of levees (Light and Dineen 1994). During that construction, the East Coast Protective Levee was completed, which protected cities on the east coast from flooding while creating water storage in the newly formed WCAs. Work was also completed on a network of canals and levees in the EAA, creating close to 3000 km^2 acres of nutrient-rich peat and muck soil for farming uses (Snyder and Davidson 1994).

Realizing that south Florida's natural resources were not being properly managed, the Florida legislature, in 1972, enacted the Water Resources Act that changed the Flood Control District to the current day South Florida Water Management District (SFWMD) (Light et al. 1995; Water Resources Development Act 2000). The enactment of this legislation had drastic effects on the dynamics of water management in the Everglades region and allowed the SFWMD to function as the local co-operator for the federally authorized C&SF Project. Specifically, SFWMD's authority included flood protection, water quality protection, and environmental protection and enhancement of the Everglades (Light and Dineen 1994).

1.3.2 Kissimmee River Restoration Project

Located in the northern Everglades, the Kissimmee River Basin encompasses more than two dozen bodies of freshwater in the Kissimmee Chain of Lakes (KCOL), their tributary streams and associated marshes, and the Kissimmee River and floodplain (Lodge 1994). This basin serves as the driving force behind the hydrologic activity in south Florida and forms the headwaters of both Lake Okeechobee and the Everglades. In its pristine state, the Kissimmee River once meandered for over 160 km through central Florida, with a floodplain that reached up to 5 km wide when inundated by heavy seasonal rains (McVoy et al. 2011). Native wetland plants, wading birds, and fish thrived there under these conditions. However, prolonged flooding in 1947 prompted a public outcry for federal assistance to reduce flood damage to property and in 1948, the U.S. Congress authorized the U.S. Army Corps of Engineers to deepen, straighten, and widen the waterway, which included the construction of canals and water control structures to achieve flood control in the Upper and Lower Kissimmee basins (Light and Dineen 1994; Light et al. 1995).

From 1960 to 1971, the Kissimmee River was transformed from a beautiful, meandering river into a 90 km long ditch, 90 m wide and 10 m deep, known as the C-38 canal (Chimney and Goforth 2001). In addition, six water control structures were created to manage flooding within the central Florida basin (Gunderson 2001). As a result of this ditch-and-drain effort (see Finkl 1995), the wetland-dependent flora and fauna that once thrived in the Kissimmee River system declined drastically (Craft et al. 1995). While the project delivered on the promise of flood protection, it also destroyed much of a floodplain-dependent ecosystem that nurtured threatened and endangered species, as well as hundreds of other native fish and wetland-dependent animals (Childers et al. 2002; Gaiser et al. 2005). More than 90% of the waterfowl that once graced the wetlands disappeared and the number of bald eagle nesting territories decreased by 70% (Ogden 1994). After the waterway was transformed into a straight, deep canal, it became oxygen-depleted and the fish community it supported, such as largemouth bass fisheries, gave way to fish species more tolerant of low dissolved oxygen conditions (Chimney and Goforth 2001; Rader and Richardson 1994).

After realizing the devastating effects inflicting upon this area, the Kissimmee River Restoration Project was authorized by Congress in the 1992 Water Resources Development Act and authorized the restoration of the middle third of the channelized Kissimmee River (Gerlak and Heikkila 2011; Light et al. 1995). The Kissimmee River Restoration Project was projected to restore about 65 km of meandering river and more than 50 km^2 of wetlands, including Lake Kissimmee, Lake Cypress, and Lake Hatchineha, along with the Kissimmee River and associated floodplain between Lake Kissimmee and Lake Okeechobee (Light and Dineen 1994). This project was a 50/50 cost share agreement between the U.S. Army Corps of Engineers and the South Florida Water Management District (SFWMD). The SFWMD provided the state's funding and was responsible for the land acquisition, the biological monitoring, and public educational outreach as part of the plan.

Construction for the Kissimmee River Restoration Project began in 1999 and included backfilling of approximately one-third of the C-38 canal in order to reconnect and restore hydrologic flow to the river (Gunderson and Light 2006). Other construction associated with the project included levee removal, water control structure improvements, flood protection, and various infrastructure improvements. Upon completion, which was originally set for 2019, the Kissimmee River Restoration Project is projected to return natural flow to a portion of the meandering Kissimmee River and to once again allow seasonal rains and flooding to inundate the floodplain within restored areas (Fuller et al. 2008; Gunderson 2001).

1.3.3 Everglades Construction Project

The Everglades Construction Project (ECP) forms the backbone for one of the largest ecosystem restoration programs in the world. The ECP involves 12 inter-related construction projects, all located between Lake Okeechobee and the Everglades, to create man-made isolated wetlands in the form of stormwater treatment areas, or STAs (Guardo et al. 1995). Totaling over 190 km^2, these STAs use naturally occurring biological processes to become flow-through filtration marshes and reduce the excess levels of nutrients, mostly phosphorus, that enter the Everglades from the EAA (Perry 2008; Walker 1995).

Costing an estimated \$635 million for construction and operation of the STAs (Clark and Dalrymple 2003), plus an additional \$190 million for land acquisition (Holl and Howarth 2000), the primary goal of the ECP is to reduce phosphorus concentrations from EAA discharged water to 50 parts per billion (ppb), as well as, achieving the right balance of carbon, silica, calcium, natural bacteria and algae, and a myriad of other components that aquatic invertebrates require to create the necessary marsh water for the Everglades (Payne et al. 2000; Walker 1995). The STAs, while being constructed mostly on former farmlands and state wildlife management areas, are all located downstream of main agricultural discharge canals (Guardo et al. 1995). In addition to the STAs, landowners in the EAA have been required since 1996 to lower the phosphorus in their runoff by nearly 25% (Payne et al. 2000; Rizzardi 2001). These commercial farmers use a variety of techniques to accomplish this, which include applying fertilizer efficiently, preventing runoff with dikes, controlling erosion, and altering pumping operations (Snyder and Davidson 1994). Additionally, the ECP is responsible for the control of exotic plants and the improvement of hydropatterns (i.e. the flow of water over a surface, including the velocity, direction, location, and depth) within the Everglades (Guardo et al. 1995).

1.3.4 Modified Water Deliveries Project

The Modified Water Deliveries Project (Mod Waters), under the authorization of the Everglades National Park Protection and Expansion Act of 1989, is a controversial ecological restoration project in south Florida charged with improving water delivery mechanisms and restoring natural hydrological conditions within Everglades National Park (ENP) (Light et al. 1995; Walters et al. 1992). Mod Waters proposes structural modifications and additions to the construction of the Central and Southern Florida (C&SF) Project in order to improve the timing, distribution, and quantity of water flow to the Northeast Shark River Slough (Light and Dineen 1994; Walters and Gunderson 1994). Increasing water flow to this particular slough will initiate a cascade of enhanced water delivery into the ENP, thereby, improving the natural habitat and hydrology of the Everglades ecosystem (Perry 2008).

There are four main components to the Mod Waters project, which include: a 22 km^2 flood mitigation plan, Tamiami Trail modifications, conveyance and seepage control features, and combined structural and operational plan as part of project implementation support (Perry 2008; Walters et al. 1992). The flood mitigation plan includes the acquisition of approximately 20 km^2 of land located in south Miami-Dade County, about 10 km south of Tamiami Trail, and the construction of a levee, seepage canal, pump station, and detention area. Tamiami Trail alone creates an enormous physical barrier for water movement into ENP by preventing sheet flow from the L-29 canal to the Northeast Shark River Slough, located in the far northeastern corner of the park. Modifications involve the construction of a 1-mile eastern bridge, which would allow for water levels in the L-29 Canal to rise periodically and create a flow of water into the ENP and the Northeast Shark River Slough. The purpose of the conveyance and seepage control features was to reconnect freshwater flows and control seepage from west to east and out of the ENP. And finally, the combined structural and operational plan includes hydrological stream gauge monitoring, wildlife monitoring, and new project implementation procedures for restoring the naturally occurring ridge and slough land formation, vegetation, water flow, and depth patterns within the ENP.

Even though Mod Waters looks to help restore natural hydrologic conditions for the Everglades, the implementation schedule for the project has met with much scrutiny, especially since its completion is required before other Everglades restoration projects can proceed. Multiple concerns have been raised regarding the cost of implementing the project, project delays, and the U.S. Army Corps of Engineers' role in funding the project. Currently, the project is behind the originally scheduled timelines and will cost an estimated $400 million in total (Holl and Howarth 2000). A portion of the delays was caused by extended efforts to acquire land from private and state owners. Federal agencies have often used eminent domain to acquire some lands, a process that has been contentious at best. Furthermore, funding for the project is being conditioned on the fact that the State of Florida must prove water entering the Loxahatchee National Wildlife Refuge and Everglades National Park meet state water quality standards by reducing excessive phosphorus and other nutrients (Walters and Gunderson 1994).

1.3.5 Everglades Nutrient Removal Project

With a growing consensus at the South Florida Water Management District advocating for the treatment of runoff from the EAA, the Everglades Nutrient Removal Project (ENRP) was put into action during the 1980s. It involved an approximate 15 km^2 tract of state-owned land that was previously leased for agriculture to be converted into a biological treatment system for the purpose of reducing phosphorus levels in stormwater runoff (Perry 2004). The ENRP was charged by the District with three primary objectives: function as an operational treatment wetland and remove nutrients from EAA runoff before this water entered the Loxahatchee National Wildlife Refuge; serve as a prototype wetland providing the District with the operational experience and design data needed to maximize long-term nutrient removal performance in the larger stormwater treatment areas (STAs); and allow the District to develop and implement optimal nutrient removal technologies (Walters and Gunderson 1994; Walker 1995). Critical design issues for the ENRP included the optimal delivery of water to the project, sizing of the inflow and outlet pump stations, and construction materials and methods used for the levees. A separate design process addressed the interior features of the ENRP, due to many the scientific uncertainties regarding optimal sizing and configuration of the interior cells. Other key issues the District faced included whether to establish the wetland vegetation community either through natural recruitment or planting selected species, whether to actively manage the vegetation once established through harvesting, burning, or disking, and whether to maintain a prolonged hydroperiod versus alternate flooding and drying (Perry 2004; Walters et al. 1992).

The ENRP was built in three phases. Phases I and II included the construction of perimeter levees, pump stations, and other major structural elements associated with the project. Phase III, on the other hand, consisted of building interior levees, canals and water control structures, and establishing wetland vegetation (about 810,000 seedlings and shoots) in the growing cells. Some of the most challenging aspects of the construction included working with the thick muck topsoil, the hardness of the underlying caprock, managing surface runoff during construction, and the sheer magnitude of the planting effort (Perry 2008). The success of the ENRP would be based on the constructed wetlands producing the following performance requirements: achieve a 75% reduction in total phosphorus [TP] (i.e. measure of all the forms of phosphorus, dissolved or particulate, that are found in a sample) once waters are treated, producing effluent with a long-term average TP concentration no greater the 50 μg L^{-1} (Childers et al. 2002; Payne et al. 2000). This long-term outflow TP threshold was originally established as an interim goal for STA treatment performance, while the State of Florida has adopted a 10-μg L^{-1} limit to protect the ecological integrity of the Everglades (Payne et al. 2000; Perry 2008; Sklar et al. 2005).

1.3.6 Multi-species Recovery Plan

In 1999, the U.S. Fish & Wildlife Service, the U.S. Army Corps of Engineers, and the South Florida Water Management District all worked together to form the South Florida Multi-Species Recovery Plan (MSRP), which evaluated impacts to listed species in south Florida during the restoration of the Everglades. The MSRP is one of the first recovery strategies specifically designed to meet the needs of multiple species that do not occupy similar habitats and contains information on the biology, ecology, status, trends, management, and recovery actions for 68 federally listed species found in south Florida, as well as the ecology and restoration needs of 23 natural communities in the Everglades region (U.S. Fish and Wildlife Service 1999). It is also one of the first designed plans to restore the Everglades by specifically addressing the needs of entire watersheds, which included: the Kissimmee-Okeechobee-Everglades watershed, the Caloosahatchee River-Big Cypress watershed, and the Peace-Myakka River watershed. Using an ecosystem-wide approach, the MSRP identifies the recovery and restoration needs of threatened and endangered species, along with maintaining the about 67,500 km^2 of natural community biodiversity in the southernmost regions of Florida (Mazzotti et al. 2001; Perry 2004; U.S. Fish and Wildlife Service 1999).

The MSRP is outlined in a 2200-page document divided into two volumes. The first volume is entitled *The Species* and contains information on the biology, ecology, distribution, status, trends, management, and recovery actions needed for the 68 federally-listed species in the Everglades region. The second volume, entitled *The Ecosystem*, provides an overview of south Florida's Everglades ecosystem and discusses the biological composition, status, trends, management, and restoration needs of the area's 23 major ecological communities. It mostly describes a holistic approach to recovery by including recommendations on how to manage, reconstruct, or restore these communities in ways that will optimize benefits for the greatest number of imperiled species. By design, the MSRP is a living document, with the flexibility to accommodate changes identified through further research and to be compatible with other adaptive management strategies for the Everglades.

1.3.7 Comprehensive Everglades Restoration Plan (CERP)

The Comprehensive Everglades Restoration Plan (CERP) provides a framework and guide to restore, protect, and preserve the water resources over an 46,500 km^2 area of central and southern Florida, including the Everglades, while providing for other water-related needs of the region (Comprehensive Everglades Restoration Plan 2000; Perry 2004). The plan is actually a revision of the Central & Southern Florida (C&SF) Project, which was charged with providing water supplies, flood protection, and management to south Florida, but wound up causing adverse effects on the unique and diverse environments of the Everglades and Florida Bay. CERP is designed to provide the quantity, quality, timing, and distribution of water necessary to achieve and sustain those essential hydrological and biological characteristics that define an undisturbed Everglades ecosystem. It is projected that the restored ecosystems would be significantly healthier than the current status of those areas. Initial numerical and physical models showed most of the water generated by CERP would be allocated to the natural system for restoration goals, while the remainder of the water would be designated for use in the human environment.

At a cost of more than $10.5 billion and a 35+-year time-line, CERP has proven to be the largest hydrologic restoration project ever undertaken in the United States (Clark and Dalrymple 2003; Comprehensive Everglades Restoration Plan 2000; Holl and Howarth 2000). It was officially approved in the Water Resources Development Act of 2000, and entails more than 60 key elements, including: surface water storage reservoirs, water preserve areas, management of Lake Okeechobee resources, improved water delivery methods to the Everglades, underground water storage, treatment wetlands, removal of barriers to sheetflow, storage of water in existing quarries, reuse of wastewater, and improved water conservation (Perry 2004). Currently, the U.S. Corps of Engineers, the South Florida Water Management District, and other non-federal sponsors, in consultation with the Department of the Interior, the Environmental Protection Agency, the Department of Commerce, the Miccosukee Tribe of Indians of Florida, the Seminole Tribe of Florida, the Florida Department of Environmental Protection, and other Federal, State, and local agencies, have implemented CERP to ensure the restoration and protection of natural resources within the Everglades.

1.3.8 Florida Everglades Forever Act (FEFA)

The Florida Everglades Forever Act (FEFA), which was passed in 1994, outlines the state government's commitment to restoring the Everglades ecosystem in cooperation with the federal government's Comprehensive Everglades Restoration Program (CERP) (Comprehensive Everglades Restoration Plan 2000; Perry 2004). The primary goals of FEFA are to improve water quality by reducing the level of phosphorus that enters the Everglades ecosystem, to increase the quantity of water in the Everglades by restoring the hydrology of the ecosystem, and to restore and protect the native plants and animals of the Everglades by controlling the invasion of exotic species of plants and animals into the ecosystem (D'Antonio and Meyerson 2002; Gunderson and Light 2006). The secondary goals of FEFA include water resource development and supply, better public land management and maintenance, and acquisition of conservation easements. Under FEFA, both the Florida Department of Environmental Protection (FDEP) and the SFWMD are given enforcement power to oversee and evaluate various restoration efforts, including the Everglades Construction Project, water supply improvement and restoration, and the Everglades Nutrient Removal Project (ENRP) (Perry 2004; Walters et al. 1992). A major goal is to decrease the levels of phosphorus in the Everglades to acceptable levels that will improve the overall health of the Everglades Ecosystem and surrounding vicinities. Nonpoint source pollution, especially from Everglades Agricultural Area (EAA) sources, is a major contributor of the phosphorus contamination of the Everglades, and is addressed by FEFA through Best Management Practices (BMP) and stormwater treatment areas (STA) (Payne et al. 2000; Perry 2008; Sklar et al. 2005). In addition to water quality, the monitoring and controlling of exotic species is the exclusive duty of the SFWMD. FEFA requires that the SFWMD establish a biological monitoring network throughout the Everglades and perform a survey of exotic species at least every 2 years. SFWMD is also required to coordinate with federal, state, and/or other governmental entities the control of exotic species in the Everglades and EAA (D'Antonio and Meyerson 2002).

1.4 Discussion

The Florida Everglades is a compromised coastal wetland that has been abused since flood control efforts were initiated in the mid 1800s and when urbanization expanded after World War II (e.g., Douglas 1947; Finkl 1994, 1995). Initial insults to this unique environment took the form of ditch-and-drain engineering to make way for housing and commerce, after disastrous floods more than a century ago (Fig. 1.9). As development increased and moved inland from the coast per se to the higher Atlantic Coastal Ridge and thence into Everglades wetlands, its area was reduced until now only about one-third to one half of the original extent remains.

Fig. 1.9 Example of a drainage canal in the southeastern portion of the Florida Everglades that is also used for boat (barge) traffic. The C-111 Spreader Canal Western Project was designed to provide ecosystem restoration of freshwater wetlands, tidal wetlands, and near-shore habitat in addition to flood protection maintenance and recreation opportunities. Located in south Miami-Dade County, the project includes pump stations, detention areas, culverts, conveyance canals, and 10 plugs/water control structures. Note the presence of an elevated gravel service road (*left* side of photo) parallel the canal and airboat trails (*right* side of photo) on the other side. Exotic invasive species (e.g., *Casuarina*) have colonized the canal margins. *Casuarina* is a genus of 17 tree species in the family Casuarinaceae, native to Australia, the Indian Subcontinent, southeast Asia, and islands of the western Pacific Ocean. These evergreen shrubs and trees grow to 35 m tall. The foliage is slender, much-branched green to grey-green twigs bearing minute scale-leaves in whorls of 5–20. The flowers, simple spikes, are produced in small catkin-like inflorescences. Most species are dioecious, but a few are monoecious (Credit: C.W. Finkl)

Drainage of the wetland biomes set in motion a whole new set of conditions that resulted in the deterioration of a healthy ecosystem to the point that ecotonal successions resulted in the replacement of prior environments (e.g., Chimney and Goforth 2001; Finkl 1995; Finkl and Charlier 2003a, b; Finkl and Makowski 2013a, b). The new drier environments, which expanded at the sake of wetlands, created new biomes that little resembled the pre-geoengineered landscape. Some wetlands 'reclaimed' (by dredge and drain practices) from the Everglades, but now residing in the urban sprawl zone, are still subject to flooding in spite of land drainage (e.g., Finkl 2000; Finkl and Myers 1995, 1996, 1997). Subsequent encroachment by farming, as in the Kissimmee River Valley and by sugar cane plantations in Broward County (Everglades Agricultural Area, EAA), polluted waters through the use of fertilizers (e.g., nitrogen and phosphorus macroelements, microelements, and amendments) and pesticides. Eutrophication of Lake Okeechobee resulted and during wet years overspill sent nutrient-rich contaminated water through the Everglades as surface flow to Florida Bay, which in turn became polluted by nutrient-rich fresh water flowing into the estuary (Hansen 2016).

By the time the public began to take notice of the deteriorating conditions in the Everglades system, environmental problems were multifaceted and far-reaching. Public pressure and efforts by conservation groups moved federal and state governments to take action in the form of numerous projects that are outlined in the preceding paragraphs. The plans were to 'restore' the damaged ecosystem by throwing money at it, making the proposed effort one of the largest ecosystem restoration projects in the world (Clark and Dalrymple 2003). But there were problems stemming from the fact much groundwork and research was required to find out exactly how this unique oligotrophic wetland system functioned prior to settlement along the coast that eventually crept landward into the wetlands.

The restoration effort was initially plagued by lack of organization in the many multifaceted projects, and all the problems associated with that, and by researchers clamoring for funds. Congress allocated the money but there was no clear vision of what could or should be done in a logical, thoughtful, and temporal manner. The result was an initial helter skelter approach with research being conducted in unconnected or less than purposeful manners. The goal of institutions was to get contracts to do research and let someone else figure out if it was useful or not. That is to say, the project lacked a philosophy; there was little in the way of vision and coordination. As time went on, many programs were developed and put forward but they were disjointed and uncoupled. Great efforts were mounted to make the projects appear to be successful and legions of papers and reports were produced proclaiming the success of the project.

Part of the problem is philosophical and it is relevant to go back to the beginning to understand what the full nature and impact of the dying ecosystem would be (see Kiker et al. 2001) and what it would mean not only to the nation and State of Florida, but to the world as the Everglades was widely recognized as a unique environment, the care of which was entrusted to the United States and the State of Florida. Lack of oversight at all levels of government and the demand for more land by developers allowed wetlands to be gobbled up and destroyed. In a word, there was no overarching vision or appreciation of the Florida Everglades to the point where it was recognized that this unique environment needed to be protected and not destroyed. The history of how this coastal wetland was destroyed and remains impaired has been recounted innumerable times (e.g., Blake 1980; Chimney and Goforth 2001; Douglas 1947; Finkl 1994, 1995; Finkl and Charlier 2003a, b; Finkl and Kruempel 2005; Finkl and Krupa 2003; Finkl and Makowski 2013a, b; Gunderson and Light 2006; Gwin et al. 1999; Kiker et al. 2001; Light and Dineen 1994; Light et al. 1995; Perry 2008; Sklar et al. 2005) and now there are many explanations of how it should be put back together as a restoration process.

1.4.1 Lapis Philosophorium

Reviews of papers outlining the scope and objectives of the Everglades Restoration Project suggest that there was no well-defined philosopher's Rosetta stone that showed the way to a logical and well-thought-out plan or approach to remediation. There was recognition that something quite terrible had gone wrong in the multipronged management of this unique wetland, to the point that some observers might say that so-called environmental management here was and remains rather grotesque. Initially, in the later 1800s and early 1900s, the Everglades was viewed as something that impeded development and it therefore had to be eliminated by dredge and drain policies so that it could be used for urban development (see Finkl 1995; Finkl and Makowski 2013a, b). The philosophy of the time was based on what land developers viewed as potential building space that could make use of then-perceived useless swampland. There was money to be made by draining the wetlands and this philosophy became governmental policy that not only supported but also financed land drainage.

The policy of land drainage for urban and agricultural development continued more or less unabated until the advent of Marjorie Stoneman Douglas' book *The Everglades: River of Grass* (1947). Her book brought a new philosophy that redefined the popular conception of the Everglades as a worthless swamp to an appreciation of a treasured river of grass. The impact of her work cannot be overstated as she opened the eyes of many people, whose new philosophy demanded care, concern, and protection of this subtropical wetland. The impact of Douglas' book rests in a public perception of a coastal wetland environment that was degraded or destroyed by human action, the result of which has been compared to that of Rachel Carson's influential book *Silent Spring* 1962. Public awareness of the uniqueness of the Florida Everglades ecosystem eventually filtered up to the political realm causing the government to act.

The philosopher's Rosetta stone in this case arose from the foresight of a woman who changed public perception of useless swampland to realization of the value of environmental awareness. But, as this new philosophical outlook for the Everglades began to take hold, new vocabularies began to spring up as researchers and politicians grappled with problem of lost wetlands and what to do with those that remained in a damaged condition. For the half or so of the Everglades that was lost to development, there was nothing to do. It was too late. Those wetlands are lost forever. The state or condition of the remainder required some sort of sensible management and new environmental principles and practices were invoked in efforts to put things right. A new view or philosophy for the Everglades was needed and this is where projects were proposed and eventually implemented (e.g., Central and Southern Florida (C&SF) Project, Kissimmee River Restoration Project, Everglades Construction Project (ECP), Modified Water Deliveries Project (Mod Waters), Everglades Nutrient Removal Project (ENRP), South Florida Multi-Species Recovery Plan (MSRP), Comprehensive Everglades Restoration Plan (CERP), and Florida Everglades Forever Act (FEFA). The problem with projects is that they tend to stifle visions because they are goal oriented and become politicized over time by those who control the purse strings. And when projects become political they tend to lose their logical (scientific) foundation, philosophy, or vision as conditioned by the economics of perceived costs and benefits. Very much related to visions of what should be done to put the Everglades right again are environmental theories and ideas as to what should be done, how it should or could be accomplished, and by what means. The term most commonly used in this context is 'restoration' but there are many pitfalls in the application of this concept that affect visions of the Everglade's future.

Returning to the philosophy of restoration, it is perhaps worth the effort to consider what the word *restoration* means when it comes to the environment. The general meaning refers to the act or process of returning something to its original condition by repairing it, cleaning it, etc. That is, a restoring to an unimpaired or improved condition relative to the present (damaged or impaired) status. This broad scale definition mostly refers to physical objects, although the term has many other meanings with political, socioeconomic, or religious overtones (Baker and Eckerberg 2013). In reference to the Everglades, the focus of the context is environmental or more specifically, ecological. As defined by the National Research Council (NRC 1992), restoration of aquatic ecosystems is the "return of an ecosystem to a close approximation of its condition prior to disturbance." The concept of restoration may be further clarified by defining many types of restoration-related activities.

The Society for Ecological Restoration (SER) defines *ecological restoration* as "the process of assisting the recovery and management of ecological integrity where ecological integrity includes a critical range of variability in biodiversity, ecological processes and structures, regional and historical context, and sustainable cultural practices" (Lewis 1989). And, as pointed out by Sklar et al. (2005) there are societal underpinnings to ecological restoration. That is ecological restoration, at least in concept, is not only a scientific endeavor but one that must per force draw in aspects of the political spectrum which is part and parcel of socioeconomics. Balance between competing forces is thus an essential component of restoration efforts.

The holistic nature of restoration, including the reintroduction of animals, is important. The objective is to emulate a natural, self-regulating system that is integrated ecologically with the landscape in which it occurs. Often, restoration requires one or more of the following processes: reconstruction of antecedent physical conditions; chemical adjustment of the soil and water; and biological manipulation, including the reintroduction of absent native flora and fauna.

1.4.2 The Everglades Restoration Project: An Opus Magnum or Mors Spiralis (Death Spiral)?

Most restoration efforts, no matter where they are located, face many problems not the least of which is the politicization of original good intentions (see Finkl and Kruempel 2005, for example). But, after time true intentions are difficult to discern because visions or ideals of restoration become blurred by positionalities of those who are in the game to make money, achieve political gain, acquire gratuitous accolades from the general public, and who are not really concerned with putting things as right as they can be for the environment (e.g., Finkl and Kruempel 2005, 2006). When big government gets involved, which after all is a political instrument, project goals become the mantra without much thought or planning that integrates research efforts in widely disparate fields of research. The result, as seen in the Everglades Restoration Project, is the proliferation of myriad multifaceted projects that are disjointed. Part of this effort involves computer models that attempt to approximate conditions in the Everglades by which so-called adaptive management decisions are based (Gunderson and Light 2006; LeSchiava et al. 2013; Sklar et al. 2001). The problem here is that ecological processes and natural environments are generally poorly understood for pre-settlement conditions (see for example Finkl and Restrepo-Coupe 2007, for a description of potential natural environments based on soils). There are many examples of the unreliability of computer models (e.g., Pilkey and Pilkey-Jarvis 2007) and their application in the Everglades project may do little to provide guidance for management decisions that are useful to solving problems that plague the well being of the environment.

The other aspect of restoration efforts, with or without computer models and numerical simulations, is that without a clear vision of what is being attempted the project goals become a process rather than a destination. The process, from the viewpoint of big government, is to feed money into the machine that is largely composed of government agencies, universities, and consultancies. The institutionalization of the projects thus becomes the new vision, to feed public and private research groups that in turn help keep the economic wheels turning. Even though these are the facts of life when dealing with large projects administered by a centralized or totalitarian federal government (e.g., national agencies such as NOAA, EPA, NPS, COE), the public is offered glowing reports of success to justify the expenditure of large sums of money from the public purse. Yet, researchers struggle to show the viability and success of their efforts by referring to indicator species that have increased in numbers. The idea here is to represent the situation as successful because certain fauna seem to be making a comeback from near extinction or a severely degraded condition. Use of indicators (Doren et al. 2009a, b; Frederick et al. 2008; Gaiser 2009; Mazzotti et al. 2008; Trexler and Goss 2008) to make arm waving proclamations as to the success of the whole restoration effort are perhaps overzealous, to say the least. Other methods are also used to assess restoration progress, as reported by Rutchey et al. (2008) in the case of vegetation mapping and by Gwin et al. (1999) via wetland regulation through hydrogeomorphic classification.

As far as the process is concerned, the input of large sums of money to support numerous institutions, the purpose becomes one of maintaining the status quo at any cost. Government agencies and institutions become used to the inflow of cash for research, administration, and oversight, and as such they never quite come to conclusions or definitive results. They instead always suggest or indicate that just a little more research is required to get to the answer; that is, a solution of the problem. A good analogy perhaps might be the industry that has been built up around cancer research. It has long been rumored that solutions or cures for certain cancers have been found but are buried or not made generally available to the public because a cure curbs the industry (e.g., Trull 1993), which will do everything in its power to continue to exist. Another analogy might be the status of political systems and terms of individual politicians, both of which do whatever is necessary to ensure the survival and continuance of the *status quo*. That is to say, the end results justify the means. All of this, whether agreeable to most or not, is part and parcel of the socioeconomic-political system that exists today.

With these thoughts in mind, it may be fair to ask whether the Everglades Restoration Project represents an *opus magnum* of which all involved agencies and institutions can be justly proud as a showcase for the world or whether the process is in a death spiral. As it so often happens, when projects get so large and administered by many different and often competing heads, the overarching vision is lost in the administrative details and obvious solutions based on common sense are overlooked (for political hegemonic reasons). This effect or result is not to suggest some sinister plot or conspiracy but to note that common sense is not very common when it comes to implementation.

1.4.3 The Cause of Wetland Degradation

If one were to stand back and look at the problems facing the well being of the Everglades ecosystem, it would be quite obvious that the cause of ecological distress and decline is due to impacts associated with urban encroachment and agriculture (Fig. 1.10). Loss of wetlands due to urban expansion is obvious and now more or less limited by the levees up to which subdivisions have spread. Wetland loss to urbanization is now contained and the half of the Everglades built upon is gone forever, never to be restored.

Agriculture surrounding the headwaters of the Everglades overland flow and in the EAA is the other side of the problem than now threatens the remaining half of the Everglades. This is where the rubber hits the road, so to speak, and the problems of pollution and nutrient enrichment of the oligotrophic wetlands become intracta-

Fig. 1.10 Example of environmental degradation of wetlands by airboats in the southeastern-most part of the Everglades in Dade County, Florida. The airboat trails look like roads or tracks, but they are actually under water. Note the marshland in the foreground with tree islands in the background (Credit: C.W. Finkl)

ble. The obvious common sense solution would be to provide a buffer around headwaters and EAA seepage areas that feed into the Everglades by removing the cause of phosphorus enrichment, which is due to runoff, seepage, and ground water flow from agricultural lands. Modern agriculture as practiced today requires the application of soil amendments, fertilizers and pesticides (including algaecides, herbicides, fungicides, etc.) that are eventually leached into surface and ground waters that seep into the Everglades. These polluted, nutrient-rich waters are the cause of most problems and a sensible solution would be to focus on the cause rather than the symptoms (Perry 2008). Modern societies studiously avoid causes and focus instead on symptoms because it is easier to treat symptoms than causes. It is just that simple. Politicos avoid hard decisions because they are not concerned with providing rational solutions to problems, but to maintaining their status quo (continued term in office) while appearing to provide viable solutions. It is a gimmick or game, sometimes referred to as the 'narrative' which in a sense is really propaganda, that works every time as the public is generally scientifically illiterate and so it is easy to parlay solutions that are acceptable to the public. The solutions offered may be scientifically absurd, but they are nevertheless accepted by an ignorant public that does not want to think about hard decisions or which really does not care about what happens to the environment until it personally effects them in an adverse way.

1.4.4 Common Sense Approach

A common sense approach to the 'Everglades problem' focuses on the obvious fact that totalitarian agriculture and healthy coastal wetlands are incompatible. Byproducts from monocultures (e.g., sugar cane farming, citrus groves, cattle ranches, dairy farms) inadvertently pollute natural systems and if the ecosystem can't absorb the environmental insult of nutrient-laden waters, the system will evolve in response to the new conditions. Proposed and implemented counter measures to clean up polluted waters have little effect when agricultural nutrients continue to have unfettered ingress. Storm water treatment areas in constructed wetlands (e.g., Guardo et al. 1995) and deep well injection for underground storage of water may be viable methods in some situations but they will not solve the problem in the Everglades because the cause of the problem has not been removed. These are expensive stopgap measures that draw public attention to purported cures for what ails the Everglades ecosystem, polluted and contaminated wastewater from agriculture.

Sadly, the nature of the problem has more far reaching effects than generally recognized. It is now appreciated that water quality in Florida Bay is affected by runoff from the Everglades to the extent that the Bay experiences algal blooms and contains a so-called ecological dead zone as well as mercury contaminants in fish (e.g., Evans and Crumley 2005). These unwanted ecological responses occur due to nutrient-laden waters entering the bay from the Everglades. Equally disastrous is the upwelling of polluted EAA (Everglades Agricultural Area) water along the Florida Reef Tract. Degradation of coral-algal reef systems result from underground transport of nutrient-rich groundwater from the EAA to the coastal ocean (e.g., Finkl and Charlier 2003a, b; Finkl and Krupa 2003; Finkl and Makowski 2013a, b; Finkl et al. 2005).

This means that side effects of totalitarian agriculture adversely affect a wide range of protected areas ranging from Gulf and Atlantic cost estuaries, the Everglades National Park, to Florida Bay, and also to the Florida Reef Tract. Pulsed releases of contaminated fresh water from Lake Okeechobee adversely affect estuaries leading to both the Gulf of Mexico and the Atlantic Ocean via the Caloosahatchee Estuary on the west coast and the St. Lucie Canal to the Indian River Lagoon on the east coast. The degree of environmental degradation is legion and it is hard to believe that extant problems can be remediated until buffer zones can be established that remove agriculture from the perimeters of protected areas.

Humans live in alternative realities, which means that they perceive themselves as not part of Nature but apart from it and in control of it. What is required is a paradigm shift where humans go through a cognitive process so that they perceive themselves as part of Nature. This notion of separate identity, which is deliberately contrived via the education system (Gatto 2015) that stifles common sense and ratiocinative powers, leads to manifold environmental problems that seem unsolvable because it is inconceivable to consider attending to the cause of degradation of Everglades ecosystems. Preposterous as the suggestion may seem to those government officials and members of the general public who are recalcitrant and intractable, the suggestion of removal of surrounding agricultural lands needs to be considered as a possibility, or perhaps some kind of meaningful curtailment or amelioration of present activities to those that might be more environmentally (ecologically) benign. Consideration of possibilities based on sound scientific principles offers potential solutions in a real sense where there are ranges or degrees of limiting agricultural lands. In the real world there are always opportunities and possibilities for interventions, but in alternative realities there are always roadblocks or impossibilities. The latter are imposed by special interest groups and lobbyists who do their best to thwart perceived threats to the status quo, no matter what downstream environmental damage their special interest activities may induce. A paradigm shift is thus needed in the politico-socioeconomic system and government that allows all possibilities to be on the table and open for discussion and real-world action. Until that change occurs as a true intention and not a smoke screen, there will be no meaningful change in the course of human action that results in a cleaner Everglades environment.

1.4.5 Need for a Change in Perception

In order for a paradigm shift to take place to produce more acceptable ecoscenarios, that is a change in attitude towards what can be accomplished with a modicum of common sense, there must be a serious reality check. The realities of the Everglades ecosystem as it now stands is that the environment is severely degraded not only in the Everglades per se, but also in peripheral sanctuaries such as Florida Bay, the Florida Keys National Marine Sanctuary, and the Florida Reef Tract. The scope and degree of pollution is not a minor consideration as agriculture and urbanization have proven to be toxic to the natural environment in many different ways. Toxicity and environmental degradation take many forms but keynote observations show that runoff from agriculture is the major culprit followed by effluent disposal in ocean outfalls (during the tourist season and with high rainfall events) along the Florida Reef Tract (e.g., Finkl and Charlier 2003a, b; Finkl et al. 2005). The reef tract is additionally insulted by upwelling of nutrient-rich ground waters that are derived from the EAA (e.g., Finkl and Krupa 2003).

Still further insults to the natural environment result from the introduction of exotic plants and animals. Some exotic invasive species were deliberately introduced (e.g., Malaleuca, Casuarina, Brazilian Pepper tree, water hyacinth) while others were accidental as in the case during when some reptiles escaped from captivity. Pet owners who found snakes and iguanas too difficult to keep as they over time matured into larger animals deliberately released them into the Everglades. Whatever the mode of introduction to the Everglades ecosystem, these exotic invasive species have for the most part thrived in their new environment, finding it rather salubrious, to the detriment of native species. It is perhaps somewhat surprising that there is not more concern among Floridians that pythons, cobras, and mambas now inhabit these subtropical wetlands. Pythons are, for example, now challenging alligators as the top predator population in the Everglades. Common sense suggests that as their food supply dwindles, they will search for new sources and eventually work their way into adjacent urban areas. This problem is not only a threat to indigenous Everglades populations, but also to residents and their pets in urban areas.

This list of invasive exotic species is hardly exhaustive and could go on, but this vignette makes the point that the Everglades ecosystem is not only under threat from invasive flora and fauna that change the ecosystem but that the wetlands are changed forever. Attempts to control exotic populations have failed, even with removal programs for Malaleuca and Casuarina, there has been no reduction in the numbers of reptiles that are now proliferating unchecked in their new environment (e.g., Finkl and Makowski 2013a, b). The unfortunate fact is that eradication of invasive plants and animals is impossible and with these new species populating the Everglades, the ecosystem is now irreversibly changed from what it once was. It thus seems ludicrous to advertise 'restoration' of the Everglades. Environmental change induced by human action has gone too far. For the sake of simplicity, re-interpreting an old nursery rhyme can make the point as follows:

> Humpty Dumpty sat on a wall,
> Humpty Dumpty had a great fall.
> All the king's horses and all the king's men
> Couldn't put Humpty together again

This quatrain, with external rhymes and a trochaic meter, succinctly gets the point across that no matter how much time, effort, and money is put into 'restoration' of the Everglades, it can never be put back together again. This is reality, a sad fact of human interference with one of Nature's great triumphs in the form of a river of grass as so elegantly described by Marjorie Stoneman Douglas 1947. With this thought in mind, a serious reality check, it is perhaps worth considering the value of what is being attempted under the aegis of federal direction. That is, the expenditure of billions of dollars (Clark and Dalrymple 2003; Holl and Howarth 2000) to attempt something that cannot be done (restore the Everglades), for Humpty Dumpty cannot be put back together again. The situation in the Everglades is similar to so-called restoration of wetlands in coastal Louisiana where enormous sums of money are spent in efforts to save barrier islands and deltaic wetlands that have suffered environmental damage that can be traced back to human action (see Finkl and Khalil 2005).

1.4.6 Degradation of Wetlands Versus Restoration Potentials

This discussion is not meant to be a diatribe or attack on what has been done so far in the Everglades Restoration Project, but a call to common sense via a paradigm shift. Billions of dollars have been spent so far with little to show for it as far as the wellbeing of the wetlands are concerned. What needs to be prudently considered is the balance between what should be done from a scientific point of view to help clean up the Everglades; how to better organize, integrate, and organize so-called restoration in a logical sequence; how to satisfy governments' need for control and assurance from positive public opinion; how to control governments' desire to keep research (educational) institutions and consultancies going by throwing money at them in an effort to bolster local economies; and finally how to approach the real causes of the problems that now plague this wetland, designated as a Wetland of International Importance, an International Biosphere Reserve, and a World Heritage Site in Danger. The Florida Everglades was first inscribed on the danger list (UNESCO World Heritage Danger List) in 1993 because of the damage caused by Hurricane Andrew the previous year, as well as a marked deterioration in water flow and quality due to urban development and agriculture. It was removed from the list in 2007, but degradation of the site has continued with water inflows reduced up to 60% and pollution reaching the point where the site is showing significant indications of eutrophication. The degradation has led to the loss of marine habitat and a subsequent decline in marine species (http://www.un.org/apps/news/story.asp?NewsID=35490#.VyDLuCMrJAo), according to UNESCO. This endangered ecosystem was placed back on the list in 2010.

Without realistic consideration of all options to make the Everglades a better place, nothing will change. The status quo will win out, that is: researchers will be funded, institutions will thrive on overhead costs, local economies will maintain or grow, and totalitarian agriculture will continue to pollute and contaminate the river of grass, estuaries, Florida Bay, the Florida Keys, and coral reefs.

That is one part of the puzzle that deserves fair adjudication, but the other part that includes the proliferation of invasive species is in all likelihood an insurmountable or uncorrectable problem. That is say even with best management practices, adaptive management, and so on there is no way to correct, ameliorate, remediate, or rectify the damage caused by exotics that are mobile and impossible to track down or eliminate. The Everglades can never be put back together or restored to pre-settlement conditions and there is thus great uncertainty in the results of management scenarios (e.g., Fuller et al. 2008). The situation is out of control and has gone too far. Some conditions can be improved, such as providing clean water to the Everglades (Perry 2004, 2008), but conditions involving exotics are escalating and becoming direr with time. This aspect of the Everglades problem is intractable and humans will have to adapt to the new evolving expansion of invasive reptilians. For South Florida, this will become a brave new world. Many societies live with pythons, mambas, and cobras in other parts of the world. They have learned how to accommodate the risk and deaths associated with unexpected encounters with these deadly reptiles. South Florida will become another learning ground.

In sum, the idea of restoring the Florida Everglades is laudable and a worthy goal in concept. Attempts to restore wetlands are serious business (Conniff 2014), but they must be approached in the correct manner (e.g., NRC 1992). In practice, however, the problem is not solvable to the extent that is advertised to the public or reported in most of the scientific literature. The critical component or key to resolving environmental degradation lies with the willingness, or lack thereof, of those in power to provide a buffer zone around the Everglades that is devoid of agriculture. Last but not least, there is a responsibility of the federal government and State of Florida to protect what is recognized as a Wetland of International Importance, an International Biosphere Reserve, and a World Heritage Site.

The responsibility of clean up is strong, the task is enormous, and the goals of restoration momentous. Parts of the Everglades problem can be solved if there is willingness but other parts will remain problematic. What is happening in the Florida Everglades is not unique as coastal wetlands throughout the urbanized world are threatened by inaction or insufficient efforts, as governments are not prescient enough to recognize or deal with the environmental problems or lack the funding to do so. The Florida Everglades problem is huge by any standard, but other smaller coastal wetland projects elsewhere may show better success ratios by virtue of scalar factors. Additionally, time is not on the side of restoration efforts because polluted or contaminated water from agriculture continues to enter the Everglades ecosystem and because invasive and exotic populations will increase with time. And finally, there remains the case of the coral reefs that are degraded by upwelling of nutrient-rich ground waters that have taken several decades to reach tide from the agricultural areas (Finkl and Krupa 2003; Finkl et al. 2005). Even if the source areas of nutrient input (the EAAs) through organic soils into the aquifers are immediately eliminated, decadal transit times means that the coral reef system will be impacted for at least another half century. Time is thus of the essence to correct known deficiencies in plans to bring the Everglades ecosystem to a more robust and healthier state.

1.5 Conclusions

The unique Florida Everglades ecosystem came under assault from human action early in the nineteenth century. Initial insults, starting in the mid 1800s, to the subtropical ecosystem took the form of ditch and drain projects that converted the wetlands into dry land that could be used for agriculture and urban expansion. By the time people began to see the unwanted effects when more than half of the Everglades was lost to development, flora and fauna populations were in decline and the ecosystem was severely compromised to the point that nutrient-rich surface and ground water supplies to the coast caused algal blooms, dead zones as in Florida Bay, and degraded coral-algal reefs of the Florida Reef Tract in the Atlantic Ocean along the Florida east coast. Estuarine systems on the east, west, and southern coasts of the Florida Peninsula were also degraded by injection of nutrient-rich waters from the Everglades. The extent of ecosystem decline basically affects the entire southern part of the State of Florida. Numerous efforts put in place to remediate the environmental damage were only marginally successful at best and complete failures at worst. A recounting and evaluation of the largest restoration project on the planet stresses the point that government planners and managers eschew dealing with causes in preference to trying to fix symptoms. The latter approach appeases special interest groups such as the agricultural lobby to the point that it is not envisaged how the Everglades can be put back together again.

Restoration of the Florida Everglades is not an isolated case on the world scene as many coastal wetlands are being usurped by urban expansion and commercial/industrial development. Such approaches are not conspiratorial in nature but most likely result from ignorance and mismanagement of fragile ecosystems, as government agencies are influenced by special interest groups that promise income incentives to public coffers from activities on the drained and ‘reclaimed’ wetlands. Short sightedness on the part of governments create a lack of understanding of the value of these coastal wetlands to the environment and a willingness to ignore causes of environmental destruction on the part of the general public and its government creates situations that are untenable for future generations. What is evident today in the case of the Florida Everglades is that more than half of the ecosystem has already been destroyed and the remainder is in grave danger of despoliation and ecological succession into an environment that will be foreign to the zone that it was developed in. This estrangement from potential natural environments occurs because native species are lost or reduced to the brink of extinction because present management practices, adaptive or not, cannot put the system back together again. The almost kaleidoscopic piggybacking and partly overlapping restoration programs call for concision of effort to avoid duplicity and uncoordinated work. The present state of affairs is unsustainable and it remains unknown what the Everglades ecosystem will evolve into at this point in time. Such may be the case for other coastal wetlands elsewhere.

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Chapter 2
Recent Agricultural Occupation and Environmental Regeneration of Salt Marshes in Northern Spain

Ane García-Artola, Alejandro Cearreta, and María Jesús Irabien

Abstract Salt marshes reduce wave energy and offer natural protection from storms and floods. In the last centuries these coastal areas have been intensely impacted by human activities worldwide. In northern Spain, more than 50% of the original salt marshes have been reclaimed with agricultural purposes since the 17th century. However, many of these coastal wetlands have been recovered since the agricultural decline during the 1950s. Benthic foraminifera and sand content can be used as proxies to identify past episodes of salt-marsh reclamation and to analyze the environmental regeneration process of previously occupied lands. Foraminifera are absent in agricultural soils and increase in abundance during the regeneration of the area, until total recovery is reached. Similarly, sand content increases as tidal inundation takes place during the environmental regeneration period. The physical disturbance originated by reclamation presents a challenge for the ^{210}Pb dating method. Nevertheless, historical aerial photography provides a good record for age estimation. This can be supported by chronostratigraphic horizons of major pollution events and nuclear weapon testing (i.e. heavy metals and ^{137}Cs). In the current context of sea-level rise, sediment supply constrains the environmental regeneration of salt marshes. In northern

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Spain, abundant regional sediment input is available, allowing high sedimentation rates to happen during the regeneration process and facilitating adaptation to ongoing sea-level rise. Therefore, restoration of currently reclaimed tidal wetlands in global temperate coastal areas, with abundant sediment supply, can be considered as a soft adaptation measure against climate change consequences in the coastal zone.

Keywords Salt marsh • Sedimentary record • Human impact • Environmental regeneration • Sea-level rise

2.1 Introduction

Salt marshes consist of a gently sloping vegetated platform, intrinsically linked to the tides, as they develop between mean tide level (MTL) and highest astronomical tide (HAT). Between these lower and upper limits, salt marshes can be subdivided into elevational zones (i.e. low and high marsh) represented by vegetation changes that respond to the duration and frequency of tidal flooding. Salt-marsh surface elevation changes as a consequence of sediment accretion and compaction, being the latter more influential as the salt marsh gets older. The elevation of the salt marsh can also be affected by degradation of organic matter (Day et al. 2011; Kirwan and Blum 2011), which represents one of the main components of sediment accretion. In fact, salt-marsh vertical accretion is produced by sediment of allochthonous (i.e. fluvial and marine minerogenic and organic sediments) and autochthonous (i.e. plant organic matter) sources (Allen 2009). Vertical accretion is opposed by autocompaction or loss of porosity caused by the weight of overlying sediments (Bartholdy et al. 2010).

In northern Spain, these coastal ecosystems have been occupied initially with agricultural purposes and later to support the more recent urban and industrial settlement, covering around 50% of the original salt-marsh extension (Rivas and Cendrero 1991; Gobierno Vasco 1998). This human occupation has led to the destruction, size reduction, and degradation of the environmental quality of these coastal areas. Rivas and Cendrero (1991) conclude that human occupation of salt marshes and other intertidal areas represents the main geomorphological change occurred in northern Spain during the last two centuries.

Agricultural activities seem to be present in this coastal region since at least 5000 cal years BP (Zapata 2005/2006), although it was not until the beginning of the 18th century that these activities significantly affected coastal wetlands. During the last 300 years salt marshes have been drained and occupied, particularly intensely from the second half of the 19th century. Desiccation of salt marshes was enhanced with the implementation of the Cambó Law in 1918, founded on their suspected insalubrity (Gogeascoechea and Juaristi 1997). Since the agricultural decline and migration to the cities during the 1950s, some of these previously reclaimed lands have been naturally regenerated. The abandonment of agricultural activities, lack of dyke maintenance, and entrance of estuarine water has allowed recolonization by halophytic vegetation of these areas, once artificially isolated from tidal inundation. In the last three decades, local authorities have been taking political decisions for the preservation of salt marshes due to their ecological importance and high productivity. In this sense, the Spanish Coastal Law passed in 1988 included these coastal environments in the public domain, promoting their conservation. The Urdaibai estuary (Fig. 2.1) was declared a UNESCO Biosphere Reserve in 1984 (Monge-Ganuzas et al. 2008). Similarly, the Santoña salt marshes (Fig. 2.1) were designated as a Ramsar site in 1994 (Irabien et al. 2008b).

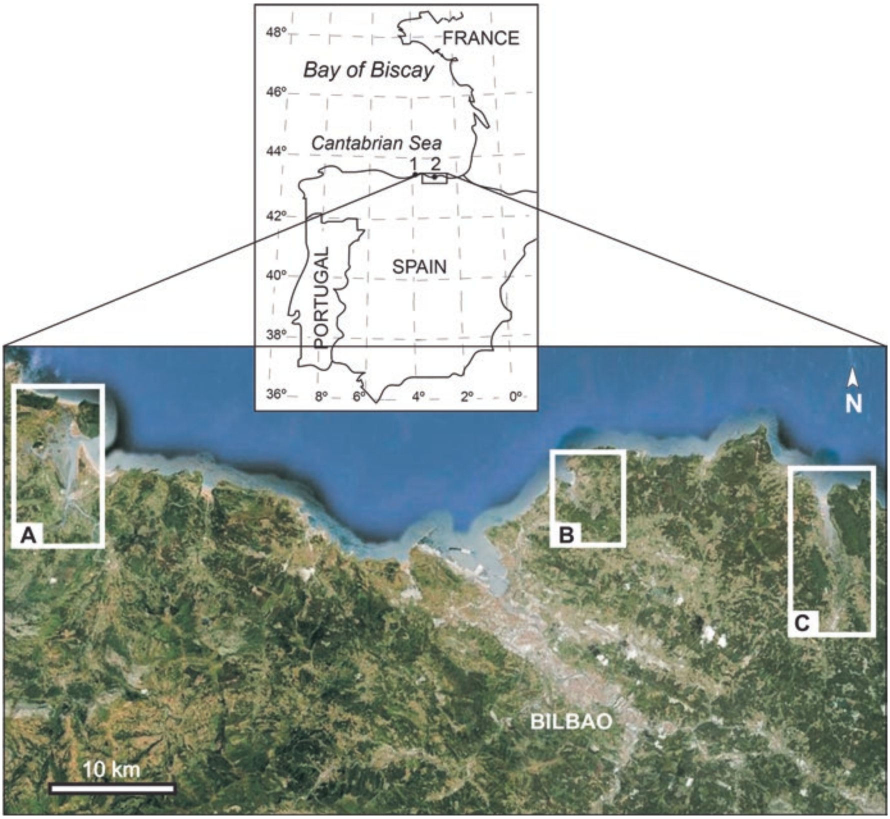


Fig. 2.1 Geographic location of the Santoña (*A*), Plentzia (*B*), and Urdaibai (*C*) estuaries in northern Spain. (*1*): Santander tide gauge. (*2*): Bilbao tide gauge (Source for snapshot is Google Earth)

The recovery of coastal wetlands is of great importance because these vegetated areas act as shoreline stabilizers, protecting coastal settings from storms and floods since these ecosystems are able to dissipate wave energy (Costanza et al. 2008; Weis and Butler 2009; Mudd et al. 2010; Gedan et al. 2011; Shepard et al. 2011). This is of particular interest in the current context of relative sea-level rise in northern Spain: the analysis of the nearest Bilbao tide-gauge (Fig. 2.1) record provides a relative sea-level rise rate of 2.98 ± 1.08 mm year^{-1} for the period between 1993 and 2005; and the analysis of the longer record from the nearby tide gauge of Santander (Fig. 2.1) provides a relative sea-level rise rate of 2.08 ± 0.33 mm year^{-1} for the period from 1943 to 2004 (Chust et al. 2009). Therefore, restoration of currently reclaimed salt marshes by reintroducing tidal flow is nowadays gaining importance as an adaptation strategy against ongoing sea-level rise (French 2006; Roman and Burdick 2012).

In order to predict future coastal evolution, it is essential to understand its environmental transformation in the past. In that sense, the recovery of coastal wetlands can be easily observed in recent regional aerial photography (Chust et al. 2007). Aside from graphical records, it is useful to study human fingerprints preserved in salt-marsh sediments. In fact, continuous deposition of sediment within estuaries occurs in sheltered areas such as salt marshes. Therefore, salt marshes are considered as precise geological archives of past environmental change (Fatela et al. 2009), providing a record of natural and anthropogenic processes through time. Hence, this chapter focuses on the identification of signals of human impact in salt-marsh sedimentary records from northern Spain. This information is of great utility in areas where anthropogenic activities took place before aerial photography was available.

2.2 Geomorphology of Northern Spain

The northern coast of Spain is dominated by high rocky cliffs that are constantly being eroded by high-energy waves. Therefore, salt marshes develop in the most protected areas within the small estuaries that make their way through the cliffs. Regional salt marshes first formed at 3000 cal years BP, when sea level stabilized after the Holocene marine transgression (Caballero et al. 2011; Leorri et al. 2012). Holocene sediments started to infill former river valleys at 8500 cal years BP (Leorri and Cearreta 2004) and lie above a bedrock mainly composed of sedimentary rocks of Mesozoic-Cenozoic age (Martínez Cedrún 1984; Irabien and Velasco 1999).

The Santoña, Plentzia and Urdaibai estuaries in northern Spain (Fig. 2.1) were infilled during the last thousands of years and are analyzed here to identify recent agricultural fingerprints and the later environmental regeneration. These estuaries present the main drainage channel located on the left side, while sedimentation of beaches and dunes takes place on the right side due to the prevailing northwest wind direction (Cearreta et al. 2004). The Santoña estuary is formed by the tidal part of the Asón River. The 11-km long and 0.5 to 3-km wide estuary covers an area of 2000 ha (Cearreta 1988). Around 63% of the total surface is exposed at low tide (Irabien et al. 2008b). The eastern part of the lower estuary is dominated by sand, where beaches and sand dunes develop, whereas the western part is mainly made of mud, where salt marshes evolve. The Plentzia estuary is formed by the tidal part of the Butroe River. The estuary covers a total area of 115 ha, with a 7 km length and an average width of 20 m (Gobierno Vasco 1998). Around 80% of the estuarine surface is exposed at low tide (Cearreta et al. 2002). Nowadays, salt marshes develop in a few isolated patches in the lower and upper estuary. Finally, the easternmost Urdaibai estuary has a total surface of 765 ha, where tidal waters are mixed with fresh waters from the Oka River. The estuary covers a 12-km long and 1-km wide alluvial valley (Monge-Ganuzas et al. 2011). The lower part of the estuary is dominated by sand, while in the middle and upper parts mud becomes dominant and salt marshes are more abundant. The three estuaries present a semidiurnal tidal regime and a mean tidal range of 2.5 m (Cearreta et al. 2008) that fluctuates between a minimum oscillation of 1 m during neap tides to a maximum variation of 4.5 m during spring tides (Villate et al. 1990; Monge-Ganuzas et al. 2008).

2.3 Collecting and Sampling Sedimentary Records

Sediment cores were collected from the central part of the salt marsh, far from wave action. Two 50-cm long and 12.5-cm diameter PVC tubes were inserted by hand into the sediment at each sampling point and the extracted core length depended on soil resistance (Fig. 2.2). Compaction of the sediment during sampling was negligible due to the minerogenic nature of the sediment in northern Spain. In fact, there is an important regional input of minerogenic material carried by estuarine waters that exceeds by far the organic matter derived from salt-marsh vegetation, which is rapidly consumed with depth (Cearreta et al. 2002, 2013; Fernández et al. 2010). Precise location and elevation of the cores was measured in the field using a Global Positioning System-Real Time Kinematic (GPS-RTK) or a total station, and referenced to the local ordnance datum (LOD: lowest tide at the Bilbao Harbour on September 27, 1878), which is located 1.727 m below the national leveling datum (Mean Sea Level in Alicante or NMMA).

In the laboratory, we divided each core replicate in halves and obtained four identical core replicates to carry out different analyses (i.e. foraminifera, sand, short-lived radionuclides, and heavy metals). Each replicate half was then subdivided in 1-cm-thick samples for analysis.

Fig. 2.2 Collection of two core replicates pushing down two PVC tubes in the Carasa salt marsh (Santoña estuary). Topographic elevation of the marsh surface is measured using a GPS-RTK

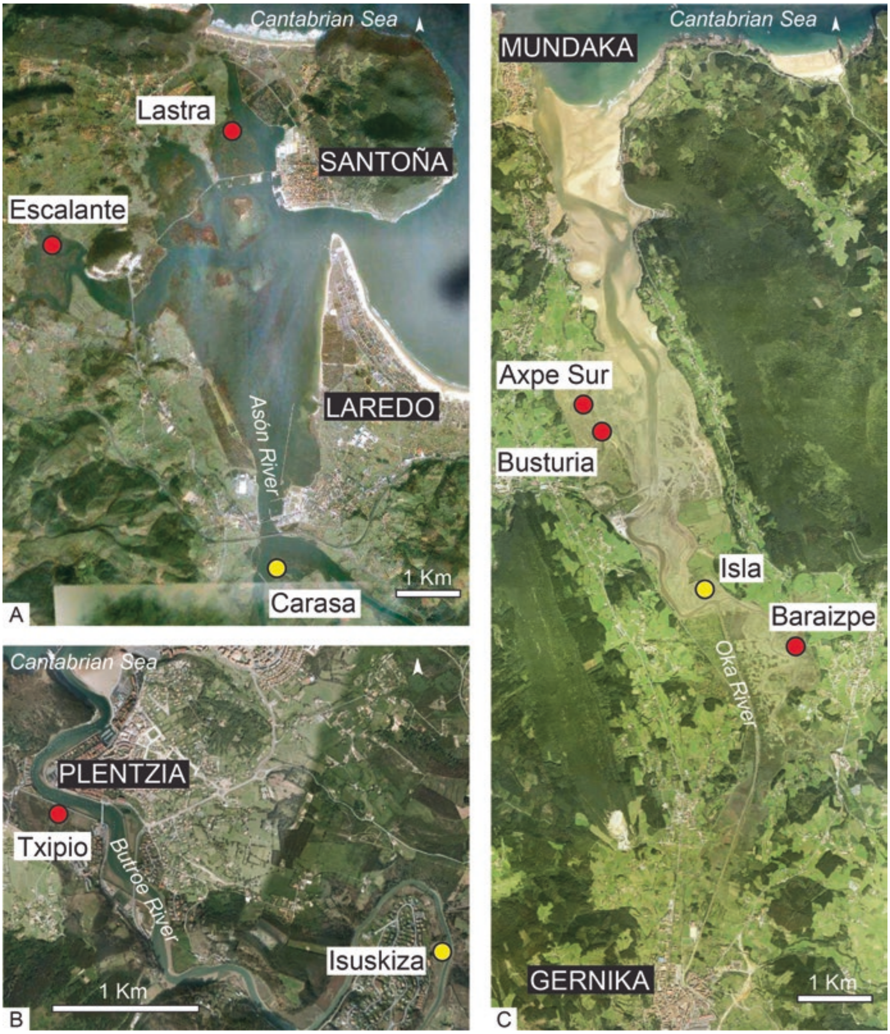


Fig. 2.3 Geographical location of the nine cores collected in the Santoña (***A***), Plentzia (***B***), and Urdaibai (***C***) estuaries. Cores in *yellow* are described in detail in this chapter (see Table 2.1 for information about cores in *red*) (Modified from García-Artola et al. 2016)

Following this strategy, we collected and studied nine cores from the Santoña, Plentzia and Urdaibai estuaries (Fig. 2.3). Three of them will be described in this chapter (yellow dots in Fig. 2.3) and details about the others (red dots in Fig. 2.3) can be found in previous publications referred to in Table 2.1.

Table 2.1 Precise geographical location and topographic elevation of the nine cores analyzed from northern Spain estuaries. Z is in m above local ordnance datum (LOD)

Estuary	Core	X	Y	Z	References
Santoña	Lastra	461,811.41	4,811,193.00	3.189	García-Artola et al. (2016)
	Escalante	458,908.46	4,809,383.99	3.553	Leorri et al. (2014a)
	Carasa	462,590.85	4,803,949.21	3.067	Irabien et al. (2015)
					García-Artola et al. (2016)
Plentzia	Txipio	503,996.11	4,805,512.61	2.507	Cearreta et al. (2002)
					Cearreta et al. (2011)
	Isuskiza	506,695.48	4,804,823.87	3.110	García-Artola et al. (2011)
					Cearreta et al. (2011)
					Leorri et al. (2013)
Urdaibai	Axpe Sur	525,218.33	4,802,812.65	4.054	Leorri et al. (2014a)
	Busturia	525,420.69	4,802,512.50	3.722	Cearreta et al. (2013)
	Isla	526,567.32	4,800,781.35	3.599	Cearreta et al. (2013)
	Baraizpe	527,558.44	4,800,144.72	3.659	Cearreta et al. (2013)

References from previous publications that describe the cores are shown

2.4 Proxies to Identify Human Intervention

2.4.1 Foraminifera

Foraminifera are protists, single-celled marine organisms that consist of a soft body (protoplasm) enclosed within a test (Lowe and Walker 1997). This test makes them geologically important, because it is preserved in the sedimentary record and can be used to study past environmental conditions. Foraminiferal tests are composed of a secreted organic material called tectin, secreted minerals (calcite, aragonite or quartz) or agglutinated particles (Armstrong and Brasier 2005). Agglutinated species are adapted to all marine environments from supratidal to deep oceans, although they are a minority in comparison with other wall types except in salt marshes and below the carbonate compensation depth (CCD) in the ocean. On the contrary, calcareous forms are generally dominant in modern neritic, bathyal and shallower abyssal assemblages, and their distribution is limited to the availability of calcium carbonate (Murray and Alve 2011).

Most foraminiferal species are benthic, except for a fewer number of planktonic taxa. They are mostly marine, covering a wide range of saline environments, from brackish to hypersaline locations. Their distribution in salt marshes is primarily determined by elevation, as a function of subaerial exposure, and a range of additional environmental variables such as salinity or pH (Edwards and Wright 2015). Due to their ecological sensitivity, foraminifera can be used in a variety of paleoenvironmental reconstructions, which require an understanding of the influence of infaunal populations and taphonomic loss on the foraminiferal assemblage found in the sedimentary record. Deep infaunal habitats have been reported in North American salt marshes (Goldstein and Watkins 1999; Patterson et al. 1999), even down to 60 cm (Hippensteel et al. 2000). On the contrary, analyses from Europe show a dominant shallow infaunal behavior of foraminifera (Horton 1999; Alve and Murray 2001), as in northern Spain where they live in the top 2 cm and do not significantly change the assemblage downcore (Cearreta et al. 2002; Leorri et al. 2008). Moreover, the taphonomic loss of calcareous species is not relevant in northern Spain, where foraminifera are well preserved due to the abundant supply of calcium carbonate to the environment by regional carbonate rocks (Cearreta and Murray 2000). Hence, the paleoenvironmental interpretation of buried assemblages can be easily performed by comparison with modern analogues.

Modern salt-marsh foraminiferal assemblages in northern Spain are dominated by agglutinated species (Cearreta 1988, 1989; Cearreta et al. 2002; García-Artola et al. 2015). The high marsh is dominated by the agglutinated taxa *Entzia macrescens* and *Trochammina inflata*, together with *Arenoparrella mexicana*, *Haplophragmoides wilberti*, *Miliammina fusca*, and *Scherochorella moniliformis* as secondary forms (Plate 2.1). In the low marsh, the agglutinated assemblage is combined with the calcareous hyaline species *Ammonia tepida*, *Haynesina germanica*, *Elphidium oceanense*, *Elphidium williamsoni*, and even the marine species *Lobatula lobatula*.

In northern Spain, foraminifera are very abundant in intertidal areas such as salt marshes. Thus, the unusual absence or near absence of foraminifera detected in some salt-marsh sedimentary records can be attributed to former agricultural activities following the graphical proof of historical aerial photographs (otherwise related to anthropogenic environmental pollution, as in the nearby Bilbao estuary; Cearreta et al. 2000). Foraminifera present a clear trend in response to this human intervention (Fig. 2.4): (1) extremely low numbers or absence of foraminiferal tests during the agricultural reclamation; (2) increasing foraminiferal densities during the environmental regeneration process; and (3) abundant foraminiferal tests in the regenerated salt marsh.

We studied foraminifera retained in the sand-size fraction (see Sect. 2.4.2) under a stereoscopic binocular microscope using reflected light. In order to accelerate the counting process, foraminifera were concentrated by flotation in trichloroethylene as described by Murray (1979). Foraminiferal tests were picked until a representative number of at least 300 individuals for each sample was obtained. Otherwise, all the available tests were counted. Foraminiferal results are expressed as a percentage (Table 2.2). In order to homogenize abundances, the number of foraminiferal tests in 50 g of dry sediment was calculated for standardization with regional values (Table 2.2).

2.4.2 Sand

The amount of sand in a sample is indicative of tidal influence, which declines during land reclamation (Cearreta et al. 2013). Immediately after reclamation ends, sand-rich tidal waters invade previously occupied areas. Therefore, we observe an increase in sand content during the regeneration process until the stabilization of the environment arrives once the salt marsh is regenerated (Fig. 2.5).

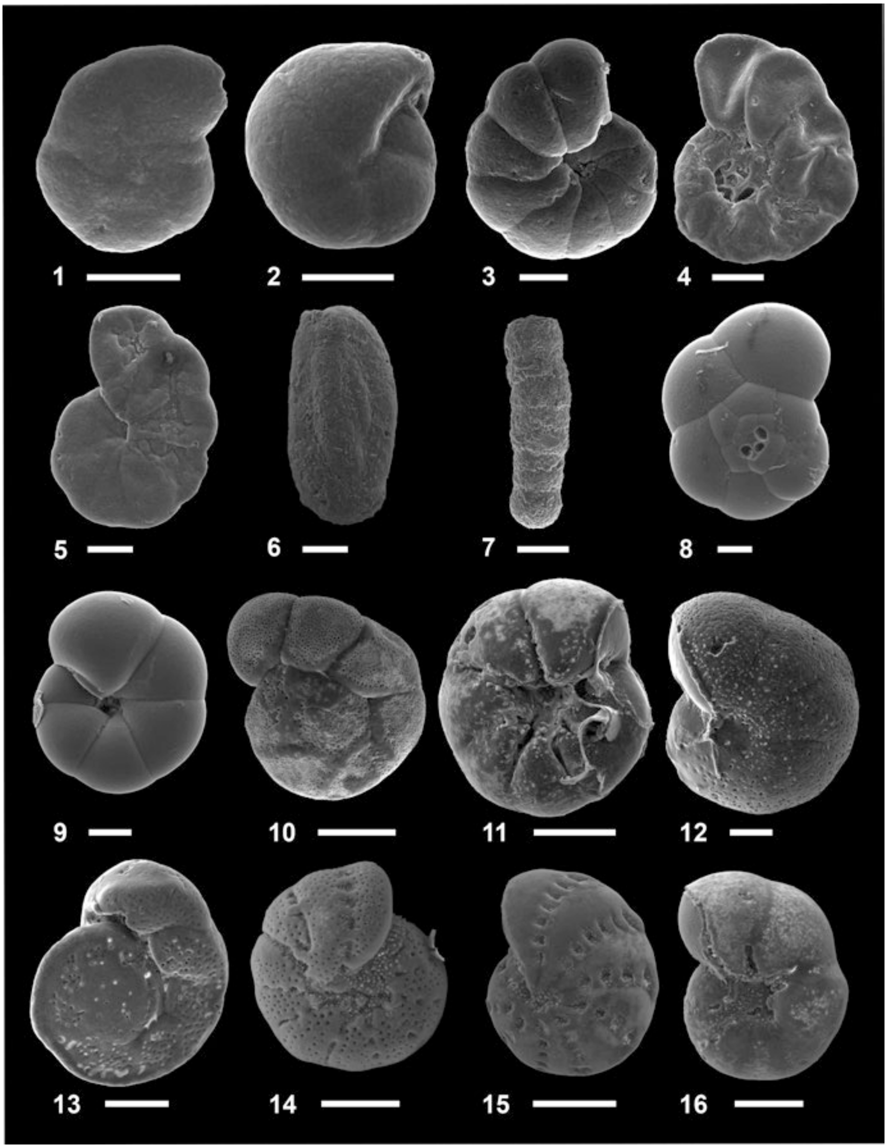


Plate 2.1 Scanning electron microscope (SEM) photographs of the main foraminiferal species found in cores from salt marshes of northern Spain. (*1*) *A. mexicana*, dorsal view; (*2*) *A. mexicana*, umbilical view; (*3*) *H. wilberti*; (*4*) *E. macrescens*, dorsal view; (*5*) *E. macrescens*, umbilical view; (*6*) *M. fusca*; (*7*) *S. moniliformis*; (*8*) *T. inflata*, dorsal view; (*9*) *T. inflata*, umbilical view; (*10*) *A. tepida*, dorsal view; (*11*) *A. tepida*, umbilical view; (*12*) *L. lobatula*, dorsal view; (*13*) *L. lobatula*, umbilical view; (*14*) *E. oceanense*; (*15*) *E. williamsoni*; (*16*) *H. germanica*. Scale bar, 100 μm

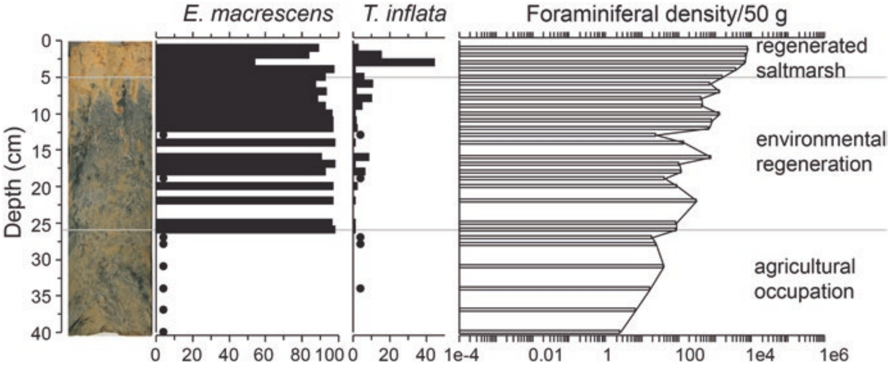


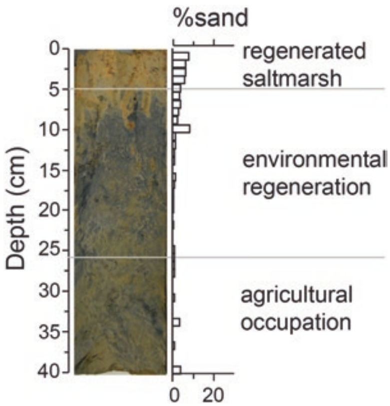
Fig. 2.4 Foraminiferal evolution showing upward-increasing abundance indicative of the environmental regeneration of the formerly reclaimed Axpe Sur salt marsh, Urdaibai estuary (Modified from Leorri et al. 2014a)

Table 2.2 Foraminiferal quantification of absolute and relative abundances of tests and species for estuaries in northern Spain

Abundance	Very low	Low	Moderate	High	Very high
Number of tests/50 g	<100	100–299	300–1999	2000–4999	≥5000
Number of species	<5	5–9	10–19	20–29	≥30
% allochthonous tests	<20	20–39	40–59	60–79	≥80
% dominant species	≥10				
% secondary species	2–9				
% minor species	<2				

Modified from García-Artola et al. (2016)

Fig. 2.5 Upward-increasing sand profile indicative of the environmental regeneration of the formerly reclaimed Axpe Sur salt marsh, Urdaibai estuary (Modified from Leorri et al. 2014a)



Sand-size fraction was obtained by wet sieving each 1-cm-thick sample through 2 mm and 63-micron sieves to remove large organic fragments and fine-grained sediments, respectively. Sand-size fraction (retained in the 63-micron sieve) was then oven dried at 50 °C and weighed to calculate the sand to mud ratio. Results are presented as a percentage of the total dry weight.

2.5 Proxies to Date Human Intervention

2.5.1 Lead-210

Recent sediments (last 100–120 years) can often be dated through the short-lived isotope ^{210}Pb (half-life 22.3 years), which is a natural radionuclide from the ^{238}U decay series (Appleby 2001). Its parent isotope ^{226}Ra in sediments decays into ^{210}Pb, through the emission of ^{222}Rn. Part of this ^{222}Rn escapes into the atmosphere and decays into ^{210}Pb, which is then deposited in the sediment. Therefore, ^{210}Pb appears in excess ($^{210}Pb_{xs}$ or unsupported, from atmospheric deposition) in the sediment respect to the supported ^{210}Pb. Supported ^{210}Pb derives from in situ decay of its parent radionuclide ^{226}Ra in secular equilibrium (Appleby 2001). The $^{210}Pb_{xs}$ activity in the sediment can then be used to estimate the age (Cochran and Masque 2005).

In the cores described here, the total ^{210}Pb activity was determined by alpha spectrometry of its granddaughter ^{210}Po in secular equilibrium (Sanchez-Cabeza et al. 1998). The ^{226}Ra activity was determined in a coaxial high-purity Ge detector (EG&G Ortec) by the gamma emission of ^{214}Pb (351 keV) in secular equilibrium. The $^{210}Pb_{xs}$ activity was calculated by subtracting the ^{226}Ra activity from the total ^{210}Pb activity. Finally, the Constant Flux model (CF model: Sanchez-Cabeza and Ruiz-Fernández 2012; also known as the CRS model: Appleby and Oldfield 1978) was applied to the Carasa core for age determination. Other cores (e.g. Lastra; Fig. 2.6) were analyzed following slightly different procedures (see García-Artola et al. 2016).

2.5.2 Cesium-137

In order to corroborate the accuracy of ^{210}Pb-derived ages, the artificial radionuclide ^{137}Cs (half-life 30 years) is commonly used (Andersen et al. 2011). ^{137}Cs is related to nuclear activity and provides three main chronohorizons: (1) the beginning of atmospheric-nuclear-weapon-testing record in 1954 (2) the peak of nuclear detonations in 1963; and (3) the Chernobyl accident in 1986 (Ritchie and McHenry 1990).

Spain is far from major nuclear facility discharges in Europe (e.g. Sellafield in Great Britain and La Hague in France), as well as from the Chernobyl plume. Therefore, the peak of ^{137}Cs, determined by gamma spectrometry (661 keV), is related to the maximum in nuclear activity in 1963 (Fig. 2.6). We discard the use of the oldest chronohorizon, indicative of 1954, because agricultural activities and the associated sediment disturbance occurred until the 1950s in most locations.

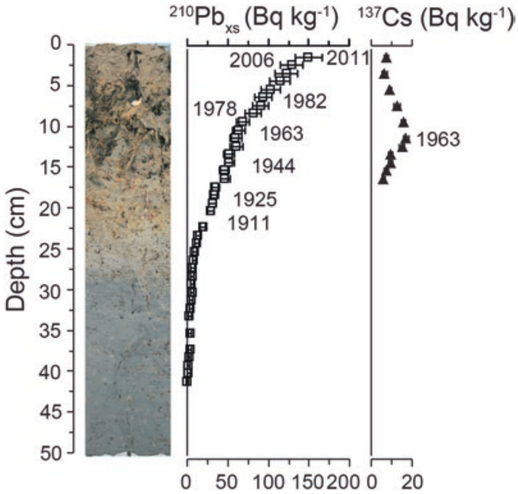


Fig. 2.6 Excess ^{210}Pb exponential decay profile and ^{137}Cs peak for age estimation of sediments from the Lastra salt marsh, Santoña estuary (Modified from García-Artola et al. 2016)

2.5.3 Pollution Fingerprints

Metals in estuarine sediments derive both from natural (e.g. rock weathering) and anthropogenic sources (e.g. mining operations, industrial effluents, sewage waste discharges, and fossil-fuel burning). Unlike other pollutants, they are not biodegradable and can accumulate in the sediment over time, preserving a valuable record of human impact. In addition, when the history of anthropogenic inputs is well known, metal pollution signals may be used to estimate deposition dates (Irabien et al. 2008a; Marshall 2015). Lead is the most widely scattered toxic metal in the world since ancient times (Cheng and Hu 2010). The geochemical fingerprint of Roman mining and smelting has been detected in ice cores, peat bogs, and lacustrine deposits (Elbaz-Poulichet et al. 2011), as well as in sediments from a nearby estuary in the northern coast of Spain (Irabien et al. 2012). However, the most dramatic increase in Pb production occurred from the Industrial Revolution onwards, with periods of enhanced atmospheric deposition at the turn of the 20th century and since the 1950s (Weiss et al. 1999).

In the Urdaibai and Plentzia estuaries historical Pb/Al profiles are broadly similar, with enhanced values throughout the 20th century that peak between 1965 and 1975 in relation to the main local industrialization period (Cearreta et al. 2002; Leorri et al. 2014b; Irabien et al. 2015). The local pollution history is in reasonable good agreement with maximum emissions of this metal to the atmosphere in Europe and Spain during the mid-1970s (Olendrzyński et al. 1996; Pacyna et al. 2007). This explains the observed enrichment of Pb/Al in the upper centimeters of the sedimentary records (Fig. 2.7).

Lead concentrations were determined using Inductively Coupled Plasma-Optic Emission Spectrometry (ICP-OES) after digestion with aqua regia. Although this technique does not achieve the total dissolution of the sample, it has been widely used for the analysis of recent sediments for environmental purposes (Landajo et al. 2004; Sarkar et al. 2004; Mil-Homens et al. 2006). Pollutants usually associate with fine-grained sediments due to their higher surface area to volume ratio and their compositional characteristics. In order to compensate for granulometric effects, results were normalized to aluminum concentrations (Ackermann 1980; Schropp et al. 1990; Cundy and Croudace 1996; Covelli and Fontolan 1997; Mil-Homens et al. 2006). This element shows a positive correlation with metals in pre-industrial materials from this area (Cearreta et al. 2002, 2013).

2.5.4 Aerial Photography

Geological determination of the recent environmental evolution of the studied areas can be supported by modern and historical aerial photography that procure approximate dates to the interpreted natural and anthropogenic processes.

Aerial photography presents a graphical proof of the agricultural occupation and later environmental regeneration process in northern Spain. In this sense, photographs from the 1950s commonly show tidal channels invading formerly reclaimed lands, providing evidence of the abandonment of agricultural activities around the mid-20th century in most locations of northern Spain (Fig. 2.8).

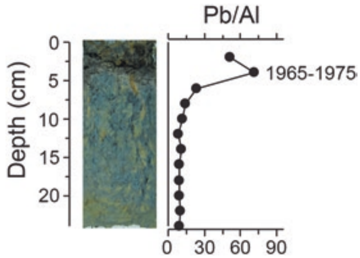


Fig. 2.7 Pb/Al profile for age determination of sediments from the Baraizpe salt marsh, Urdaibai estuary (Modified from Cearreta et al. 2013)

Fig. 2.8 Agricultural fingerprints in historical aerial photography (1957: *left side*) compared to a modern photograph (2009: *right side*) of the nowadays regenerated Busturia salt marsh, Urdaibai estuary (Modified from Cearreta et al. 2013)

2.6 Case Studies in Northern Spain

2.6.1 Carasa Salt Marsh in the Santoña Estuary

The Carasa salt marsh is located in the upper part of the Santoña estuary, where in 2010 a 50-cm-long core (two replicates) (coordinates X: 462,590.847, Y: 4,803,949.206, Z: 3.067 m above LOD) was extruded from an area dominated by *Halimione portulacoides* and *Juncus maritimus* vegetation (Fig. 2.9).

Historical aerial photography from 1946 exhibits agricultural lands covering the area, while it now appears as a regenerated salt-marsh environment (Fig. 2.10).

The Carasa core was composed of mud that switched from dark grey in the lower 27 cm to light brown in the upper 23 cm, and was accompanied by plant roots in the top 10 cm (Fig. 2.11).

2.6.1.1 Paleoenvironmental Interpretation

Benthic foraminifera were abundant in the upper 41 cm and considerably decreased in the lower 9 cm (Table 2.3; Fig. 2.11). Three depth intervals (DIs) were distinguished in this core in terms of presence, abundance and dominance of foraminiferal species. The lowermost 9 cm (DI 3) exhibited very low numbers of foraminiferal tests (mean 40 tests/50 g). *Entzia macrescens* and *T. inflata* were the main species

Fig. 2.9 General view of the Carasa salt marsh covered by *H. portulacoides* and *J. maritimus* vegetation. Location of core (*yellow dot*) in the Santoña estuary is shown (*top left*). Scale bar, 1 km

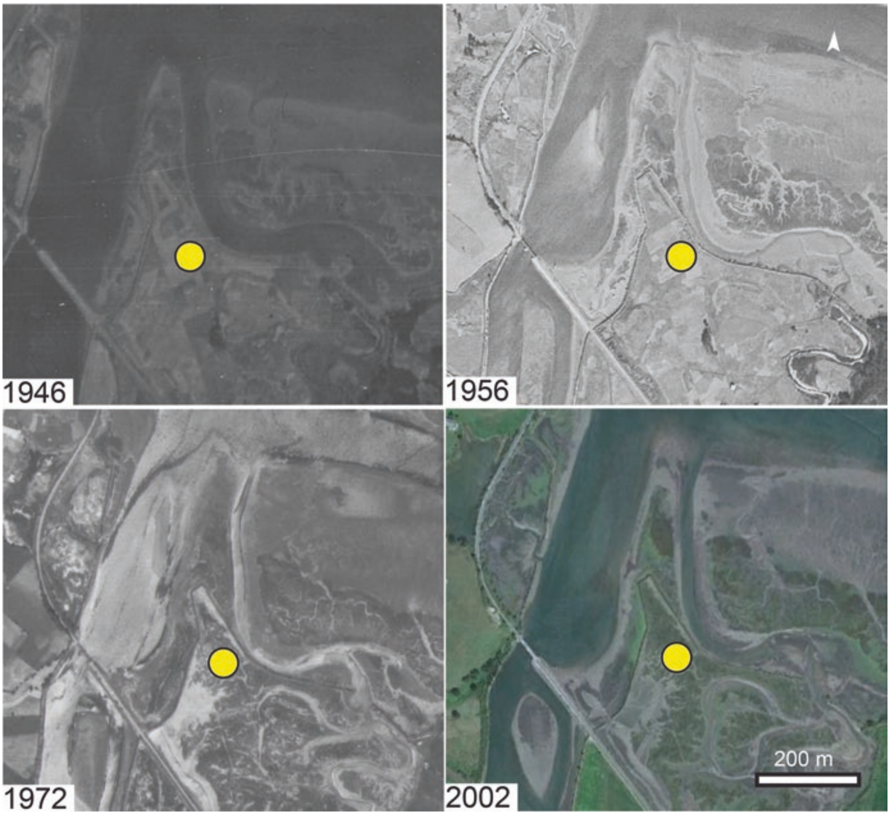


Fig. 2.10 Geographical location of the Carasa core (*yellow dot*) in historical (1946, 1956 and 1972) and modern (2002) aerial photography (Modified from Irabien et al. 2015)

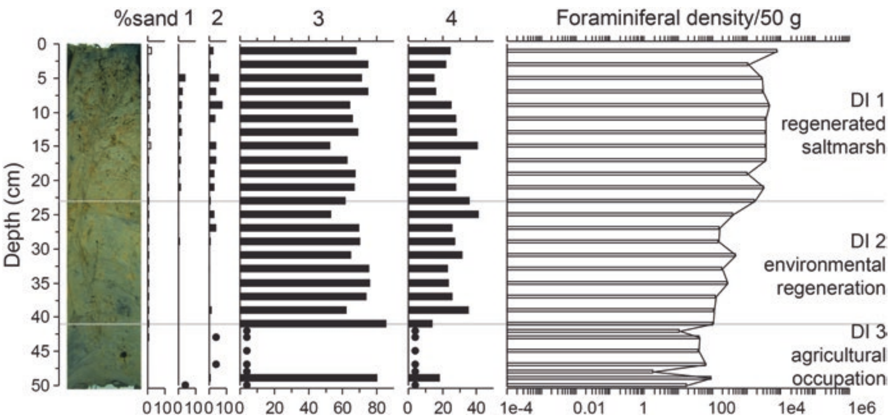


Fig. 2.11 Core photograph, sand content (%), main foraminiferal species (1: *A. mexicana*; 2: *H. wilberti*; 3: *E. macrescens*; 4: *T. inflata*) (%), and foraminiferal density/50 g of bulk dry sediment with depth (cm) in the Carasa salt-marsh core (Santoña estuary). *Black dots* in the foraminiferal species profiles represent samples with less than 100 tests. Depth intervals (DIs) are shown (Modified from Irabien et al. 2015)

Table 2.3 Summary of sedimentological and microfaunal data in the Carasa (Santoña estuary; García-Artola et al. 2016), Isuskiza (Plentzia estuary; Leorri et al. 2013), and Isla (Urdaibai estuary; Cearreta et al. 2013) salt-marsh cores

Carasa core	Isuskiza core	Isla core
DI 1	DI 1:	DI 1
Thickness: 23 cm	Thickness: 7 cm	Thickness: 13 cm
Elevational range: 3.067–2.837 m	Elevational range: 3.110–3.040 m	Elevational range: 3.599–3.469 m
Lithology: light brown mud with plant roots in the upper 10 cm	Lithology: burrowed brown silty clay	Lithology: brown burrowed silty clay
1 (1–3)% sand	35 (24–48)% sand	16 (14–18)% sand
3416 (1110–8189) tests/50 g	632 (261–1189) tests/50 g	5635 (3320–8466) tests/50 g
5 (3–7) species	11 (9–12) species	6 (4–8) species
E. macrescens 67 (53–75)%	*E. macrescens* 42 (35–53)%	*E. macrescens* 57 (37–73)%
T. inflata 27 (16–41)%	*T. inflata* 29 (22–46)%	*S. moniliformis* 31 (15–50)%
H. wilberti 4 (0–8)%	*M. fusca* 13 (1–33)%	*T. inflata* 8 (4–17)%
A. mexicana 2 (0.3–4)%	*A. tepida* 5 (1–9)%	*A. mexicana* 2 (0–5)%
	H. germanica 4 (1–9)%	*M. fusca* 2 (0–5)%
	S. moniliformis 3 (2–6)%	
	A. mexicana 2 (0.1–6)%	
DI 2	DI 2:	DI 2
Thickness: 18 cm	Thickness: 38 cm	Thickness: 19 cm
Elevational range: 2.837–2.657 m	Elevational range: 3.040–2.660 m	Elevational range: 3.469–3.279 m
Lithology: dark grey mud	Lithology: dark grey burrowed silty clay	Lithology: black and brown silty clay
1% sand	33 (17–55)% sand	14 (5–19)% sand
230 (107–492) tests/50 g	39 (0–102) tests/50 g	1317 (12–3373) tests/50 g
3 (2–6) species	4 (0–10) species	4 (2–5) species
E. macrescens 70 (53–86)%	*A. tepida* 38 (0–59)%	*E. macrescens* 94 (75–99)%
T. inflata 28 (14–42)%	*E. macrescens* 20 (1–48)%	*S. moniliformis* 3 (0–16)%
	E. oceanense 15 (0–26)%	*T. inflata* 2 (0–8)%
	T. inflata 14 (0–52)%	
	H. germanica 6 (0–10)%	
	E. williamsoni 3 (0–4)%	
DI 3	DI 3:	DI 3
Thickness: 9 cm	Thickness: 5 cm	Thickness: 12 cm
Elevational range: 2.657–2.567 m	Elevational range: 2.660–2.610 m	Elevational range: 3.279–3.159 m
Lithology: dark grey mud	Lithology: dark grey silty clay	Lithology: black silty clay
1 (0–1)% sand	8 (8–9)% sand	5 (2–7)% sand
40 (2–96) tests/50 g	No foraminifera	31 (2–8) tests/50 g
3 (2–4) species		2 (1–2) species
Few foraminifera		Few foraminifera

The single value represents the average and those in parentheses give the range but numbers were too low to produce percentage values. Species number (average 3 species) was very low. This interval can be interpreted as the anthropogenic deposit introduced during the agricultural reclamation of the area observed in historical aerial photography from 1946 (Fig. 2.10). The following 18 cm (DI 2) were characterized by low and upward-increasing numbers of foraminiferal tests (mean 230 tests/50 g) dominated by *E. macrescens* (average 70%) and *T. inflata* (average 28%). Species number (average 3 species) was very low. This interval represents the environmental regeneration process occurred as tidal water invaded the formerly occupied salt marsh, progressively improving the living conditions for benthic foraminifera. The last 23 cm (DI 1) showed high numbers of foraminiferal tests (mean 3416 tests/50 g), dominated by *E. macrescens* (average 67%) and *T. inflata* (average 27%). *Haplophragmoides wilberti* (average 4%) and *A. mexicana* (average 2%) appeared as secondary forms. Species number (average 5 species) was low and slightly higher than in the lower DIs. This interval with the highest foraminiferal abundance reflects the stabilization of the environment and characterizes the regenerated salt marsh.

Sand content was very low throughout the core (average 1%) that was mainly composed of mud (Table 2.3; Fig. 2.11). Therefore, the area represented a low energy environment since the human occupation period (DI 3) until the regenerated high-marsh environment (DI 1) was developed. A slight increase in sand content was observed from the bottom to the top of the core, which could be related to the entrance of sand-rich estuarine water as regeneration took place.

2.6.1.2 Dating the Sedimentary Record

The $^{210}Pb_{xs}$ profile showed a typical exponential decay (Fig. 2.12), apparently allowing the use of radiometric dating techniques. However, application of the CF model (Sanchez-Cabeza and Ruiz-Fernández 2012) indicated that by 1946 the salt marsh was already regenerated. This seriously contradicts aerial photography that shows agricultural lands in this area at that time (Fig. 2.10: *top left*). This contradiction can be explained by the occurrence of sediment mixing during land reclamation that causes the misinterpretation of the $^{210}Pb_{xs}$ inventory for age determination. In fact, historical photographs reflect how areas that appeared cultivated in the late 1940s were already undistinguishable from other surrounding salt-marsh areas by the early 1970s. On the contrary, the ^{137}Cs activity profile seems to offer a more reliable chronological benchmark determined by the peak indicative of 1963 located at 23 cm, in the limit between DI 2 (salt marsh in regeneration) and DI 1 (regenerated salt marsh). This age determination is also in agreement with Pb-derived chronology, as this metal exhibited maximum values (period 1965–1975) slightly above the bottom of DI 1. Pb values were normalized with Al to be comparable to previous regional studies, even though sand content barely changed throughout the core. Consequently, the regeneration process (DI 2) of the Carasa salt marsh took around 10 years (between the late 1940s and the early 1960s) based on aerial photography, the ^{137}Cs peak, and the Pb/Al profile. Sedimentation rates were calculated taking into account the thickness of the DI of interest and the age of its upper and lower limits.

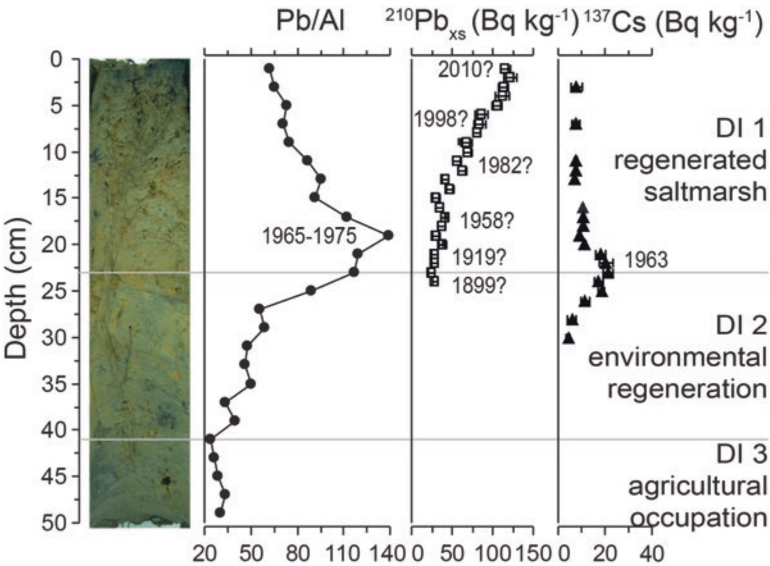


Fig. 2.12 Core photograph, Al-normalized Pb distribution, and excess ^{210}Pb and ^{137}Cs activities (Bq kg^{-1}) with depth (cm) in the Carasa salt-marsh core (Santoña estuary). Depth intervals (DIs) and ages based on the CF model are shown (Modified from García-Artola et al. 2016)

Thus, sedimentation rates during the environmental regeneration process (DI 2) reached about 18 mm year^{-1}, three times more than during the regenerated salt-marsh accretion (5 mm year^{-1}, DI 1).

2.6.2 *Isuskiza Salt Marsh in the Plentzia Estuary*

The Isuskiza salt marsh is located in the upper Plentzia estuary, where in 2006 a 50-cm-long core (two replicates) (coordinates X: 506,695.480, Y: 4,804,823.870, Z: 3.110 m above LOD) was extruded from an area dominated by *H. portulacoides* vegetation (Fig. 2.13).

Historical aerial photography shows evidence of human occupation in the recent past (1956, Fig. 2.14). According to Hormaza (1998), this salt marsh was reclaimed with agricultural purposes in the 1860s.

The Isuskiza core was made of dark grey silty clay, except for the top 7 cm that were brown with visible bioturbation (Fig. 2.15).

2.6.2.1 Paleoenvironmental Interpretation

The number of benthic foraminifera was high in the top 7 cm and gradually decreased until foraminifera completely disappeared in the lower centimeters. After the reanalysis of the samples and foraminiferal density determination, this

Fig. 2.13 General view of the Isuskiza salt marsh characterized by *H. portulacoides* vegetation. Location of core (*yellow dot*) in the Plentzia estuary is shown (*top left*). Scale bar, 0.5 km

core was reinterpreted and subdivided into three new DIs (Table 2.3; Fig. 2.15). In the basal 5 cm (DI 3) foraminifera were absent. This interval likely represents the anthropogenic deposit introduced during the reclamation period that occurred in this area until approximately the 1960s (Fig. 2.14). The upper 38 cm (DI 2) contained very low numbers of foraminiferal tests (mean 39 tests/50 g). The calcareous hyaline species *A. tepida* (average 38%) and *E. oceanense* (average 15%), together with the agglutinated species *E. macrescens* (average 20%) and *T. inflata* (average 14%) were dominant in those samples containing enough (≥100) foraminiferal tests. The calcareous hyaline taxa *H. germanica* (average 6%) and *E. williamsoni* (average 3%) were secondary forms. The number of species was very low (average 4 species). This middle interval was deposited during the regeneration process from the agricultural soil (DI 3) to the regenerated salt marsh (DI 1), as shown by the upward-increasing number of foraminiferal tests. The clear dominance of calcareous species (average 88%) and the moderate sand content suggest that these sediments were deposited in a low elevation environment such as a tidal flat (Cearreta 1988). In the top 7 cm (DI 1), foraminiferal tests were moderately abundant (mean 632 tests/50 g). Agglutinated forms (average 90%) were more important than calcareous hyaline forms (average 10%). *Entzia macrescens* (average 42%), *T. inflata* (average 29%), and *M. fusca* (average 13%) were dominant, while *A. tepida* (average 5%), *H. germanica* (average 4%), *S. moniliformis* (average 3%), and *A.*

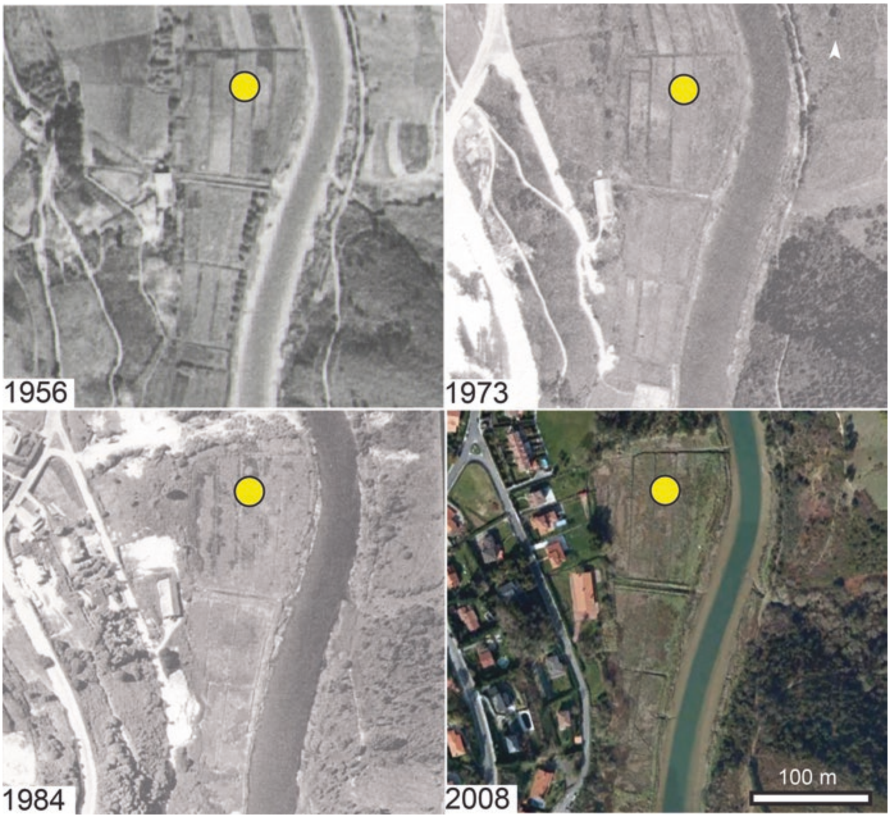


Fig. 2.14 Geographical location of the Isuskiza core (*yellow dot*) in historical (1956, 1973 and 1984) and modern (2008) aerial photography (Modified from García-Artola et al. 2011)

mexicana (average 2%) represented secondary forms. Species number (average 11 species) was moderate. This interval with stabilized foraminiferal content characterizes the modern regenerated salt-marsh environment.

The sand pattern along this core clearly illustrates that as regeneration progressed sandy (from 33% of sand in DI 2 to 35% in DI 1) tidal waters inundated the muddy (8% of sand) agricultural lands of DI 3 (Table 2.3; Fig. 2.15).

2.6.2.2 Dating the Sedimentary Record

The Isuskiza core did not show a clear $^{210}Pb_{xs}$ exponential decay profile discarding any age determination (Fig. 2.16). The ^{137}Cs activity maximum at 18 cm was located in the middle of DI 2. Historical aerial photographs agreed with the ^{137}Cs interpretation, as agricultural fields were observed in 1956 and a regenerated environment was already evident in 1973. All this suggests the regeneration process (DI 2) took

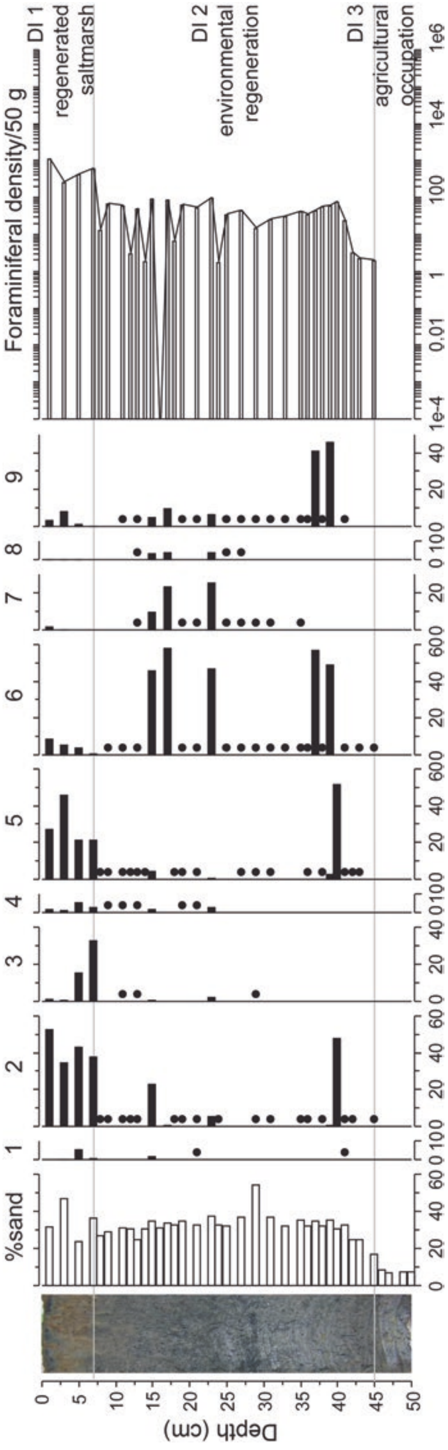


Fig. 2.15 Core photograph, sand content (%), main foraminiferal species (1: *A. mexicana*; 2: *E. macrescens*; 3: *M. fusca*; 4: *S. moniliformis*; 5: *T. inflata*; 6: *A. tepida*; 7: *E. oceanense*; 8: *E. williamsoni*; 9: *H. germanica*) (%), and foraminiferal density/50 g of bulk dry sediment with depth (cm) in the Isuskiza salt-marsh core (Plentzia estuary). *Black dots* in the foraminiferal species profiles represent samples with less than 100 tests. Depth intervals (DIs) are shown (Modified from Leorri et al. 2013)

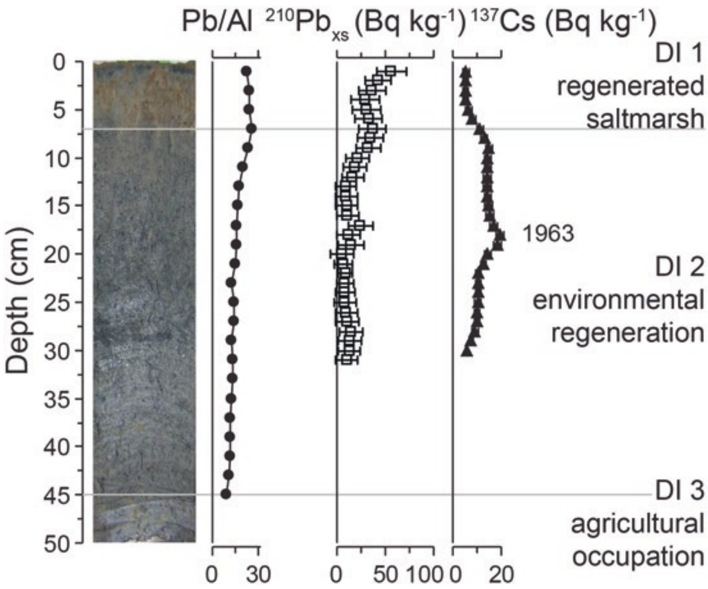


Fig. 2.16 Core photograph, Al-normalized Pb distribution, and excess ^{210}Pb and ^{137}Cs activities (Bq kg^{-1}) with depth (cm) in the Isuskiza salt-marsh core (Plentzia estuary). Depth intervals (DIs) and the ^{137}Cs maximum are shown (Modified from Leorri et al. 2013)

place approximately between the late 1950s and the early 1970s. This is in accordance with previous results from the nearby Txipio salt marsh, where the beginning of the environmental regeneration was established in the 1960s (Cearreta et al. 2002, 2011). The very high sedimentation rates (40 mm year^{-1}) observed in Isuskiza during DI 2 were responsible for the transformation of the tidal flat (lower part of DI 2) into the regenerated salt marsh (DI 1). The sedimentary "dilution" caused by these extremely elevated sedimentation rates could have been the reason why the ^{137}Cs and the Pb/Al peaks were not as obvious as in the previous Carasa core (Fig. 2.12). Besides, the latter presented elevated metal concentrations very likely unaffected by the high sedimentation rates occurred during the environmental regeneration process.

2.6.3 Isla Salt Marsh in the Urdaibai Estuary

The Isla salt marsh is located in the middle-upper part of the Urdaibai estuary, where in 2008 a 44-cm long core (two replicates) (coordinates X: 526,567.318, Y: 4,800,781.350, Z: 3.599 m above LOD) was obtained from an area covered by *H. portulacoides* and *Puccinellia maritima* vegetation (Fig. 2.17).

Fig. 2.17 General view of the Isla salt marsh dominated by *H. portulacoides* vegetation. Location of core (*yellow dot*) in the Urdaibai estuary is shown (*top left*). Scale bar, 1 km

Historical aerial photography shows that the area was already in regeneration by 1957, when tidal channels were crossing formerly reclaimed agricultural lands (Fig. 2.18).

The bottom 26 cm of the core were black silty clay and the upper 18 cm were brown silty clay with burrows of polychaetes.

2.6.3.1 Paleoenvironmental Interpretation

The benthic foraminiferal content was high except for the basal 12 cm. In general, although present, calcareous forms were very scarce in this core and foraminiferal assemblages were almost entirely made of agglutinated forms (only *E. macrescens* was dominant throughout). Three DIs were distinguished in this core (Table 2.3; Fig. 2.19). The lowermost 12 cm (DI 3) contained very scarce tests (mean 31 tests/50g) and species percentages were not produced. Species number was very low (average 2 species). This deepest interval represents the agricultural reclamation period. The following 19 cm (DI 2) were characterized by moderate numbers (mean 1317 tests/50 g) of upward-increasing foraminiferal tests. Assemblages were highly dominated by *E. macrescens* (average 94%), while *S. moniliformis* (average 3%), and *T. inflata* (average 2%) appeared as secondary species. Species number

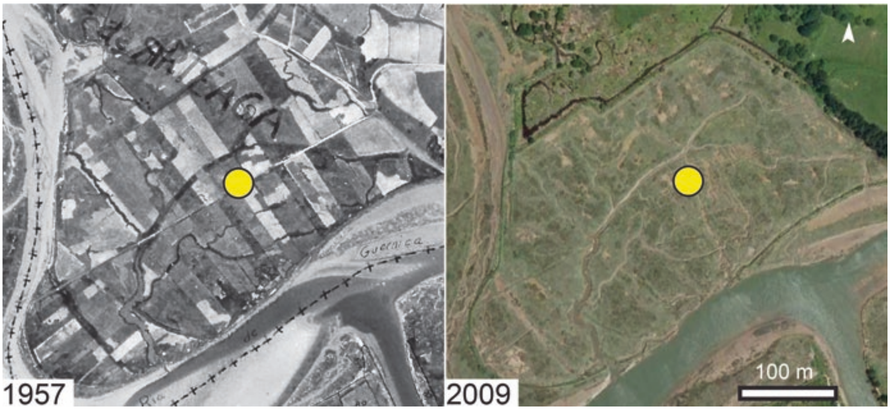


Fig. 2.18 Geographical location (*yellow dot*) of the Isla core in historical (1957: *left side*) and modern (2009: *right side*) aerial photography (Modified from Cearreta et al. 2013)

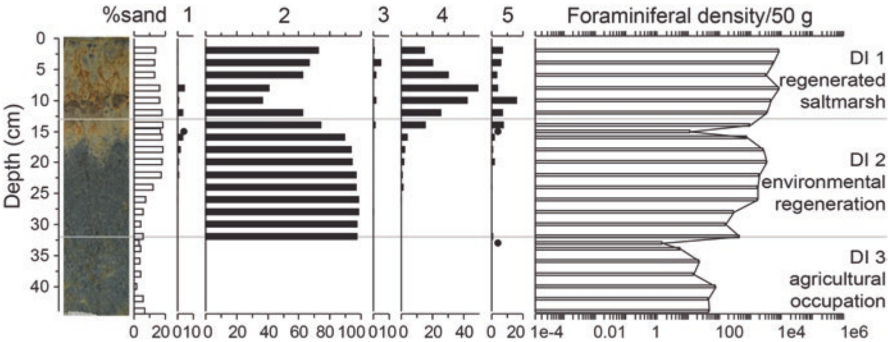


Fig. 2.19 Core photograph, sand content (%), main foraminiferal species (1: *A. mexicana*; 2: *E. macrescens*; 3: *M. fusca*; 4: *S. moniliformis*; 5: *T. inflata*) (%), and foraminiferal density/50 g of bulk dry sediment with depth (cm) in the Isla salt-marsh core (Urdaibai estuary). *Black dots* in the foraminiferal species profiles represent samples with less than 100 tests. Depth intervals (DIs) are shown (Modified from Cearreta et al. 2013)

(average 4 species) was very low. This middle DI can be interpreted as the sedimentary record deposited during the environmental regeneration of the former agricultural soil. The uppermost 13 cm (DI 1) were dominated by *E. macrescens* (average 57%) and *S. moniliformis* (average 31%), followed by *T. inflata* (average 8%), *A. mexicana* (average 2%), and *M. fusca* (average 2%) as secondary species. Species number was low (average 6 species) and similar to the previous interval. Foraminiferal density was very high (mean 5635 tests/50 g). This top interval represents the modern regenerated salt-marsh environment.

Sand content showed a general upward-increasing trend indicating the flooding of the formerly reclaimed land as regeneration continued (Table 2.3; Fig. 2.19): from a very low and variable content in DI 3 (average 5%), a low and upward-increasing content in the intermediate DI 2 (average 14%), to a slightly higher and relatively constant content in DI 1 (average 16%).

2.6.3.2 Dating the Sedimentary Record

$^{210}Pb_{xs}$ activities of the Isla core were not suitable for age determination, given that the profile was not strictly exponentially decreasing (Fig. 2.20). Aerial photographs clearly showed that this area was still in regeneration by 1957 with tidal channels flowing over former agricultural fields (Fig. 2.18). Thus, agricultural activities (DI 3) were active prior to the 1950s. The ^{137}Cs activity peak at 9 cm was located slightly above the transition between DI 2 and DI 1 (Fig. 2.20), pointing out that the salt marsh was already regenerated by 1963. Therefore, the environmental regeneration (DI 2) took place in ~10 years (between the 1950s and 1960s). Sedimentation rates during this process were high (18 mm year^{-1}), almost tenfold the calculated rates for the recently regenerated salt marsh (DI 1: 1.7 mm year^{-1}). Similar to the previous Isuskiza core, the high sedimentation rates apparently overprinted the Pb/Al profile. In fact, the Pb/Al maximum values exhibited in DI 1 could have been a result of a deceleration in the sedimentation rather than an increase in the input.

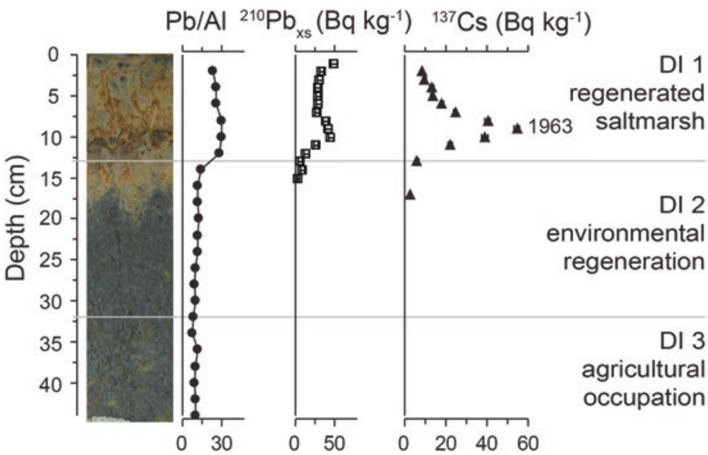


Fig. 2.20 Core photograph, Al-normalized Pb distribution, and excess ^{210}Pb and ^{137}Cs activities (Bq kg^{-1}) with depth (cm) in the Isla salt-marsh core (Urdaibai estuary). Depth intervals (DIs) and the ^{137}Cs peak are shown (Modified from Cearreta et al. 2013)

2.7 Regional Overview

Foraminifera and sand present a clear temporal pattern indicative of the agricultural reclamation and the subsequent environmental regeneration of salt marshes in norther Spain: agricultural soils can be recognized by the nearly absence of foraminifera; salt marshes under environmental regeneration show increasing amounts of foraminifera and sand, together with very high sedimentation rates linked to the inundation by tidal waters carrying sandy sediments; and, regenerated salt-marsh environments present abundant foraminifera.

These recently regenerated salt marshes challenge the use of ^{210}Pb for dating sediments. In fact, agricultural activities cause sediment mixing, which can create a hiatus, a misinterpretation of background levels, and an underestimation of the $^{210}Pb_{xs}$ inventory. This is common in recently regenerated environments where the agricultural occupation period coincides with the timeframe of the ^{210}Pb dating method (~100–120 years). In fact, radiometric studies are based upon undisturbed and continuous sediment sequences. Although these conditions are not always met in previously reclaimed salt marshes, it is common to observe that some $^{210}Pb_{xs}$ profiles show a typical exponential decay with depth (Fig. 2.12), pointing out that sedimentary disturbances may be often difficult to detect exclusively from $^{210}Pb_{xs}$ profiles. However, the ^{210}Pb dating method can be successfully applied in areas where human intervention was prior to the 1900s and the $^{210}Pb_{xs}$ inventory was not anthropogenically manipulated (García-Artola et al. 2016). In recently disturbed areas, we recommend a combination of multiple proxies such as ^{137}Cs and heavy metals for age estimation. Furthermore, historical aerial photography is an essential age-validation tool.

A correct age determination allows the estimation of sedimentation rates, which were very high (usually 14–18 mm year^{-1}; up to 40 mm year^{-1} at lower elevation environments) during the environmental regeneration process of formerly occupied salt marshes. This process occurred in less than 10 years, mainly between the 1950s and the 1960s. In the Plentzia estuary it took place slightly later, during the 1960s, and in some locations of the Santoña estuary even before the 1940s (see García-Artola et al. 2016 for additional regional examples shown in Fig. 2.1). The elevated sedimentation rates are responsible for the fast environmental regeneration and respond to the ability of salt marshes to reach equilibrium with the tidal frame. Salt marshes situated lower in the tidal frame tend to accrete faster than those located at higher elevations (van Wijnen and Bakker 2001; Temmerman et al. 2004; Goodman et al. 2007). As an example, the Isuskiza core presents sedimentation rates exceeding any previous records due to the low elevation of the reclaimed land. As the occupied salt marsh was invaded by tidal waters, a tidal flat was formed (clear dominance of calcareous foraminiferal species) that quickly gained enough elevation during the regeneration process (at around 40 mm year^{-1}) to become the modern regenerated salt marsh. This regenerated environment differs from a natural salt marsh in terms of organic matter content (Santín et al. 2009) and vegetation diversity that might take longer to fully recover (Garbutt and Wolters 2008; Chang et al. 2016).

Sediment availability has been crucial in this region for the environmental regeneration process of previously reclaimed estuarine areas and the adaptation to current sea-level rise. Sediment reaches the estuaries from inland, related to deforestation practices (mainly during the second half of the 20th century), but also from the marine environment, where high-energy coastal processes continually erode nearby cliffs. Therefore, the abundant sediment supply has allowed salt-marsh restoration in the past and will probably continue to do so in the future because deforestation is only a partial source of sediment, but coastal erosion represents a long-term process.

2.8 Conclusions

We demonstrated that temporal variations of benthic foraminifera and sand preserved in salt-marsh sedimentary records from northern Spain reflect former human occupation and environmental regeneration processes. The analysis of the environmental regeneration of salt marshes, occurred mainly in the mid-20th century, revealed that these ecosystems are able to accrete sediment very fast to reach equilibrium with the tidal frame; from 14–18 mm year^{-1} up to 40 mm year^{-1} at lower elevation environments such as tidal flats. This information is of great interest in the present scenario of sea-level rise because currently reclaimed salt marshes could be regenerated, increasing the estuarine surface available for tidal inundation and, consequently, acting as natural defenses. This cost-effective adaptation strategy can be utilized in worldwide temperate areas with enough sediment input to outpace ongoing sea-level rise.

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Ammonia tepida (Cushman) = *Rotalia beccarii* (Linné) var. *tepida* Cushman, 1926
Arenoparrella mexicana (Kornfeld) = *Trochammina inflata* (Montagu) var. *mexicana* Kornfeld, 1931
Elphidium oceanense (d'Orbigny) = *Polystomella oceanensis* d'Orbigny, 1826
Elphidium williamsoni Haynes, 1973
Entzia macrescens (Brady) = *Trochammina inflata* (Montagu) var. *macrescens* Brady, 1870
Haplophragmoides wilberti Andersen, 1953
Haynesina germanica (Ehrenberg) = *Nonium germanicum* Ehrenberg, 1840
Lobatula lobatula (Walker and Jacob) = *Nautilus lobatulus* Walker and Jacob, 1798
Miliammina fusca (Brady) = *Quinqueloculina fusca* Brady, 1870
Scherochorella moniliformis (Siddall) = *Reophax moniliforme* Siddall, 1886
Trochammina inflata (Montagu) = *Nautilus inflatus* Montagu, 1808

Chapter 3
Impact of Urbanization on the Evolution of Mangrove Ecosystems in the Wouri River Estuary (Douala Cameroon)

Ndongo Din, Vanessa Maxemilie Ngo-Massou, Guillaume Léopold Essomè-Koum, Eugene Ndema-Nsombo, Ernest Kottè-Mapoko, and Laurant Nyamsi-Moussian

Abstract Cameroon mangroves are protected over 20 years by both Management of Forest and Fauna and Environmental Management Legal Framework laws. However, these juridical tools are not efficient in the field regarding the rate of mangrove forest depletion around coastal cities in the country. This work aims to identify the main factors of mangrove degradation and to assess their effects on the dynamics and evolution of this ecosystem in relation with city development. Key abiotic parameters are favorable for mangrove progression. Natural disasters and anthropogenic activities have been identified as responsible of mangrove ecosystems depletion. Wood harvesting, urban settlement and infrastructures, sand extraction, petroleum exploitation, coastal erosion, and climate change appear to be the most important factors of mangrove degeneration. Secondary destructive factors such as dwellings, sustenance agriculture, collection of Non-Timber Forest Products, digging, landfill, dyke construction and large clear-felling also contributed widely to mangrove degradation. The realization of state projects had heavily impacted the

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evolution of mangroves in the Wouri river estuary. In the absence of law and specific regulation implementation strategies, populations have taken advantage of the authorities' tolerance to invade all mangroves areas around the Wouri river estuary. The management of Cameroon mangrove ecosystems faced the population conception of considering mangroves as an ordinary forest. Mangrove degradation along the Wouri river estuary does not seem raising advocacy in spite of the fact that this especial ecosystem could never change its coastal nature place like other artificial generated forests.

3.1 Introduction

Mangroves form a complex ecosystem, comprising several interconnected elements at the land-sea interface, which are in turn connected with adjacent coastal ecosystems such as coral reefs, seagrass beds and terrestrial vegetation. Mangrove forests prevent coastal erosion, contribute to the progression of the land towards the sea and react as a buffer in areas prone to cyclone or other ocean surges (Din 2001). In the absence of upwelling on the Cameroon coast, fresh water inputs from rivers and streams constitute the best source of nutrients to the coastal waters. Mangrove ecosystems are opened areas and the movement of materials (organic and mineral) affects not only the composition and structure of plant and animal groups, but also the soil characteristics.

Mangroves supply essential ecosystem services to tropical economies by contributing substantially to timber and charcoal supplies, productivity of near-shore fisheries, ecotourism, etc. (Nfotabong-Atheull et al. 2009). Degradation of natural resources is a major environmental issue that societies around the world are currently facing (Goodman 2010). Mangrove forests are experiencing long-term and severe decline. The rate of the deforestation is high in many developing countries, possibly higher than any other type of tropical forests (Spalding et al. 2010; Giri et al. 2011). The causes of such losses include not only natural disasters and sea-level rise, but also and especially land-use development. The intense human activities within the mangrove area (excessive harvest of mangrove trees for firewood, charcoal, clearing of mangrove areas for agricultural purposes, pollution), coupled with the rapid urbanization of the adjacent towns, have led to a gradual degeneration of these ecosystems (Din et al. 2008; Nfotabong-Atheull et al. 2009, 2013; Fusi et al. 2016). This phenomenon is expected to worsen in the years to come, with the expected vulnerability of mangrove to the rise of the sea level due to the climate change (Duarte et al. 2013; Alongi 2015; Din et al. 2016).

It is projected that anthropogenic climate change is likely to have adverse impacts on African ecosystems and their biodiversity, but projections of impacts based on a range of methodologies diverge widely (Midgley and Bond 2015). These differences relate to the extent to atmospheric CO_2 and disturbance on ecosystem structure and productivity, and relative strengths in accounting for temperature-versus water-related controls on biodiversity. As in certain areas of Africa, Cameroon is characterized by a strong climatic variation since 1960. The rise of pluviometry is palpable on annual, seasonal and monthly scales. Significant studies on variability and the climatic fluctuations in relation to the development and the environment showed rainfall deficits of about 20%. These values might be sometimes higher than 25% on the Atlantic coast and in the forest areas that confirms that "wet" Tropical Africa is regularly under the effect of climatic variability (WTO and UNEP 2008).

In Cameroon, human activities appear to be the main factor influencing the structure and dynamic of mangroves. Mangrove deforestation is occurring at a rate of 1–2% per year, which implies that most forests will disappear within this century (Alongi 2002; Feka and Manzano 2008). These disturbances in mangroves have been attributed to a combination of such factors due to the absence of adequate legislation regarding mangrove protection, and pollution in the peri-urban settings (Nfotabong-Atheull et al. 2013; Fusi et al. 2016). Rapid population growth has affected resources, including arable land, food supplies, water and energy, especially in developing countries where government policy-makers still pay little attention to protect these coastal ecosystems (Dahdouh-Guebas and Koedam 2008; Walters et al. 2008).

Cameroon mangroves are rather protected by both Law n°94/01 of 20th January 1994 on the Management of Forest and Fauna and Law n°96/12 of 05th August 1996 on the Environmental Management Legal Framework. Article 94 of the last law stipulates that "Mangrove ecosystems need particular protection in relation with their importance in the conservation of marine biodiversity and the maintenance of coastal ecologic balances". However, the application of these laws and other specific regulations is not effective in the field and consequently exposes Cameroon coastal natural resources and areas to overexploitation and degradation (Din 2001; Din et al. 2008; Nfotabong-Atheull et al. 2009).

Pollution which is neglected in coastal Africa constitute the principal form of environmental degradation in more industrialized countries. The scarcity of relevant data could justify the lack of interest in the effects of pollutants on mangrove ecosystems functioning. A recent survey shows that the degradation of Cameroon estuarine and marine environment is also due to pollutants (Fusi et al. 2016). The main source of contamination in the mangrove forest surrounding Douala is represented by uncontrolled discharge of urban wastewater and persistent, illegal and indiscriminate use of DDT. These contaminants, together with four specific heavy metals (As, Cr, Zn, Se) seem to affect the macrobenthonic assemblage, suggesting that Douala mangrove is subjected to a complex patchwork of contamination (Fusi et al. 2016).

Urbanization in this context has involved the development of infrastructures and voluntary resettlements. The lack of state and local relevant planning programs, the poverty and the permanent demographic pressure in towns accelerate the depletions easily appreciable on a short time. Mangroves suffer therefore natural and human pressures. Coastal erosion, invasion by weeds, sea-level rise and climate change damage these ecosystems. Massive degradation of mangroves in Cameroon is mostly observed near the coastal cities. High household demands for fuel-wood greatly affect plant populations. Uncontrolled wood cutting, unauthorized settlements, industrial discharges and various organic by-products, sand-pits, irresponsible factory building and warehouses, perpetual extension of sea-port, all have additional negative consequences on the performance of mangrove ecosystems. This work aims to describe the main factors of mangrove degradation and to assess their effects on the dynamics and evolution of this coastal ecosystem.

3.2 Mangrove Characteristics

3.2.1 Mangrove Distribution

Geographically, Cameroon belongs to both central and western Africa. The Atlantic front, with about 400 km long, is located at the southern part of the Gulf of Guinea, imbedded between Nigeria and Equatorial Guinea.

This country has an important biodiversity because of climate and relief variation. Mangrove areas represent about 1–1.5% of the tropical rain forest. Cameroon mangroves border the Atlantic Ocean, and occupy a surface area estimated from 2300 to 2700 km^2. The two main areas of mangroves are in the northern part of the coast. From the mouth of Sanaga river to the one of the Ntem river, mangrove stands are very reduced and could not appear clearly in aerial photographs or satellite imageries. The Rio del Rey estuary mangroves covered between 1400 and 1600 km^2 while the Cameroon Estuary mangroves occupied about 1000 km^2 (Fig. 3.1).

3.2.2 Climate

The climate of the coast of Cameroon is influenced in part by the proximity of both the Atlantic Ocean and Mount Cameroon, and secondly by the permanent presence of the meteorological equator where converge the Azores anticyclone and that of St. Helena. The main geographical factors in rainfall are similar to those generally determine the distribution of climates. It's admitted that in Cameroon, the distribution of climates depends on the precipitations and incidentally the thermal regime which essentially depends on altitude and latitude (Suchel 1972).

The climate is a specific equatorial, with a long rainy season (March–November) and a short dry season (December–February), often punctuated with showers. Rainfall is abundant and regular (more than 200 days of rain per year). Mean annual rainfall reaches 5 m and mean annual temperature is above 26 °C around Douala. Humidity remains high throughout the year (absolute maximum close 100%). In general, wind speed is low except during Saint Helene anticyclone setting phases (March–April), and dies back later in the season (September–October). Winds give almost a constant direction to coastal currents that carry and deposit large amounts of materials (mud, clay, sand). This situation occurs in "the mouths of Cameroon"

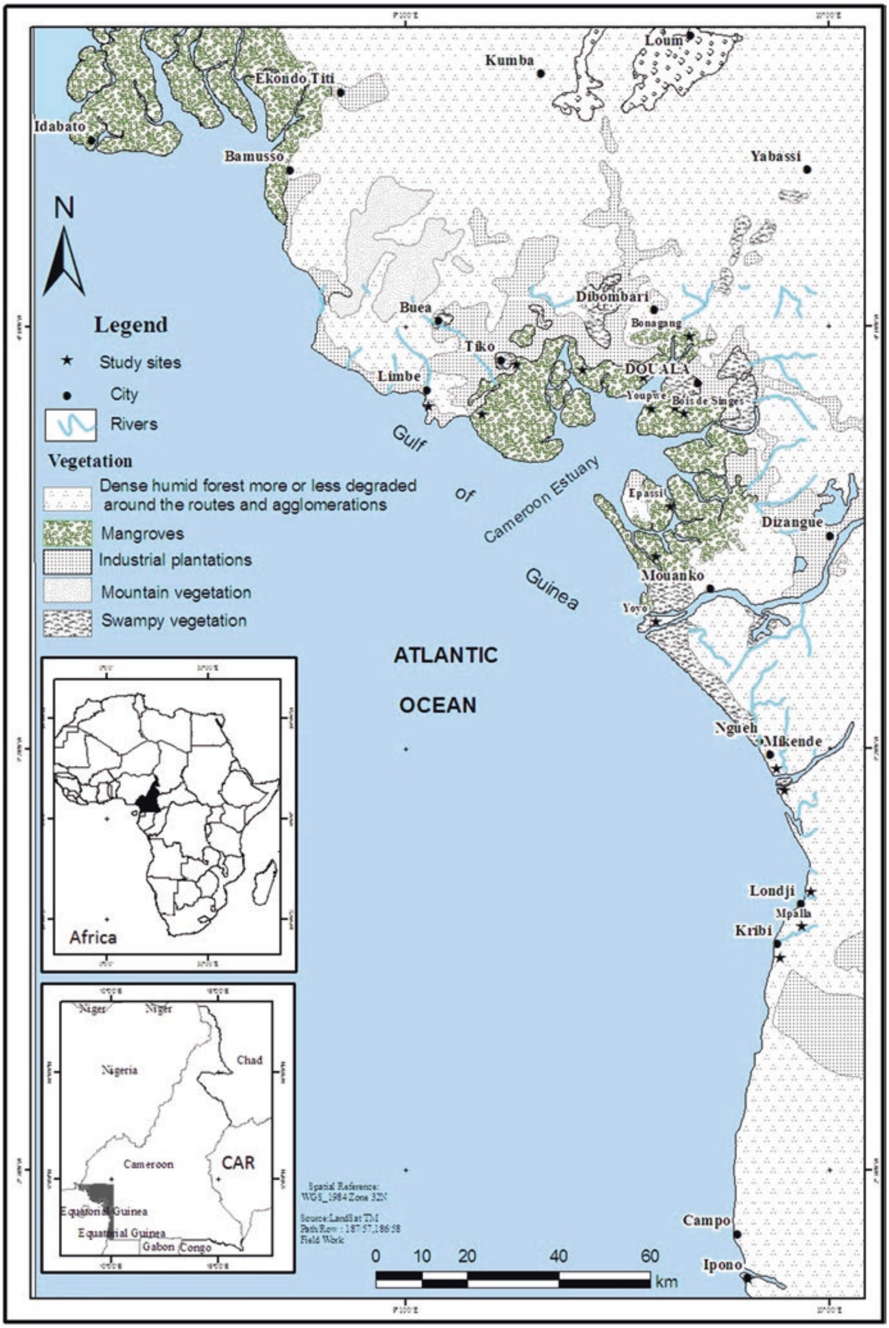


Fig. 3.1 Distribution of mangrove ecosystems in Cameroon coast (Modified after Din et al. 2016)

where silting up obliges the authorities to permanent dredging of Douala Port channel. All climatic parameters in the Wouri river estuary show favorable values for the progressive evolution of mangroves. No element of the climate does not seem likely to act as a limiting factor (Din 2001).

3.2.3 Substrate

Few data on soil analysis are available and this remains a major problem in the study of Cameroon mangroves. However, given the luxuriant vegetation, one can assume that edaphic conditions are favorable for the development of a mangrove forest ecosystem in this zone. The variation in edaphic parameters based on vegetation types was studied in the Wouri estuary mangrove (Abata 1994; Baltzer et al. 1995). The frontier area consists of a clay loam soil with blackish fluid consistency with total absence of litter. The area of young *Rhizophora* spp. growth is characterized by a significant felting roots and rootlets in the surface portion of the core, of fibrous and spongy structure (Fig. 3.2). The floors are dark gray fibrous peat in the first 50 cm and very plastic black undeveloped consistency in depth.

In the *Avicennia germinans* area, litter is thick on surface among many pneumatophores and seedlings (Fig. 3.2). The soil has marbled brown spots, red, gray dark or bright, black and rust along the pneumatophore cavities. In depth, the peat horizon in semi-fluid consistency has morphological relics soil with *Rhizophora*.

Soil on degraded areas of *Rhizophora* and *Acrostichum* are peat with a thin litter regularly moved by the tides. Between 0 and about 80 cm, the fibrous structure is spongy and very compact formed by felting of medium roots and rootlets. The color is reddish brown in surface parts and dark gray to black in depth.

In the *Guibourtia demeusei* area, soils have a medium close rooting soil well drained areas of inland areas. The texture is clayey, lumpy structure developed consistency. The color is dark brown on the surface and deep yellowish brown with gradual disappearance of radial maze of roots. The texture is sandy clay and lumpy structure more or less wet. From 60 cm, the relics of the spongy, fibrous structure of pioneer areas are observe. These horizons are plastic, adhere to semi-developed consistency, waterlogged and with numerous vertical roots of *Rhizophora* in advanced decomposition.

In shrubland of *Dalbergia ecastaphyllum*, *Drepanocarpus lunatus*, *Hibiscus tiliaceus*, *Ormocarpum verrucosum* and *Phoenix reclinata*, litter consists of decomposed leaves. In the superficial part, the floors are dark brown color, clay texture and crumb structure developed in consistency while in depth, the mottling of spots appear with color change (dark gray black) and structure (clayey silt) with morphological relics areas with *Rhizophora*.

The texture of mangrove sediments is rarely homogeneous, often characterized by a succession of clay beds on surface and sandy beds in depth or by alternating of these two layers. Because of the permanent flooding by tides, chemical ripening is more advanced than the physical. The pH is very fluctuating between two successive stages of engorgement (high tide) and drying (low tide). These mangrove soils

Fig. 3.2 Mangrove substrates under *Rhizophora racemosa* shrubs (**a**) and *Avicennia germinans* trees (**b**)

are generally characterized by a high C/N ratio due to the slowing of the biological activity, induced from anoxia.

A recent work on Cameroon mangrove soils has concerned their morphological characteristics through soil profiles (Fig. 3.3), the description of their physical and chemical characteristics in relation with the degradation level of forest vegetation and the ability to curb the phenomenon of climate change through carbon sequestration. The average stocks of carbon sequestered have been estimated at 2289.33 ± 407 $Mg.ha^{-1}$ in Bamusso (Rio del Rey group) and 2025.89 ± 165 $Mg.ha^{-1}$ in Campo (Ndema-Nsombo et al. 2016).

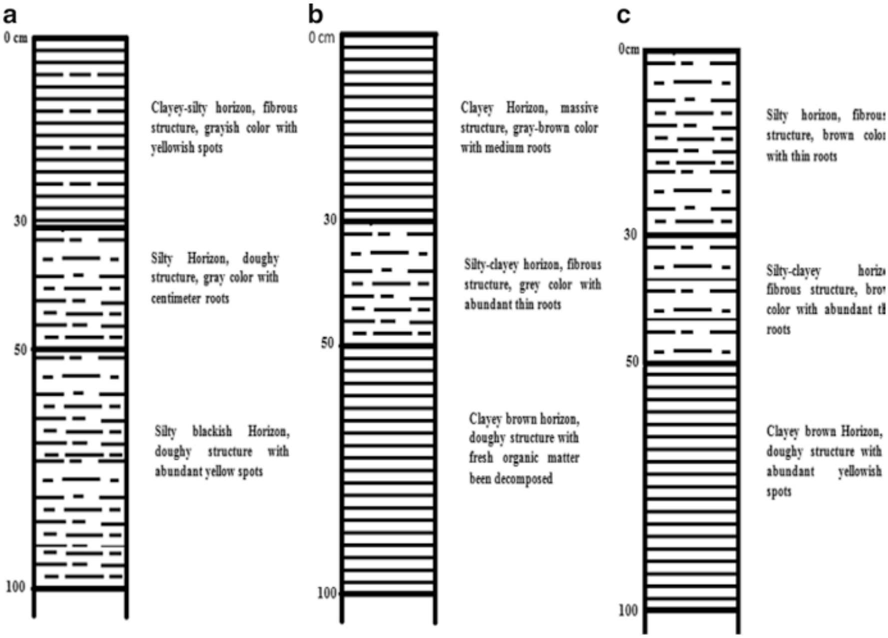


Fig. 3.3 Cameroon mangrove soil profiles: (**a**) Ntem Estuary; (**b**) Cameroon Estuary; (**c**) Rio Del Rey Estuary (Ndema-Nsombo et al. 2016)

3.2.4 Hydrology

The hydrology of Cameroon is influenced among other factors by the general scheme of the Biafra Bay, climate, topography and geomorphology (Olivry 1986). Hydrological conditions of the Biafra bay are relatively stable. The mass of surface water remains warm (25–28 °C) all the year with relatively low salinities (below 35‰). In estuaries, the salinity is still less than 20‰ and dilution is very fast. Less than 30 km from the ocean, salinity is null or almost during the most part of the year.

The estuary waters present a stratification with continental waters dominance, warm and rich in nutrients and sediment loads that float on marine water, saltier, colder and clearer (Baltzer et al. 1995). Coastal rivers constitute two subsets around the Sanaga river, the country main river which drains itself a watershed of about 135,000 km^2, with 920 km in length. During the long rainy season, seawater that waters daily twice these ecosystems is still below 10‰, while in the dry season, it varies between 4 and 20‰. Mangroves of the Cameroon Estuary are constantly watered by brackish water resulting from the dilution of seawater by fresh water from rivers and heavy rainfall.

Tides influence the structure, composition and distribution of mangrove vegetation. Similarly, they affect the salinity of the soil and the evaporation rate. Tide regime is semi-diurnal in Cameroon with a maximum amplitude that is approximately 3 m high in the Douala port (Wouri river). Baltzer et al. (1995) explained the local stability of salinity in the mangrove of Wouri river estuary by the existence of stabilizing factors. Din (2001) associated the distribution of salinity in this estuary with three areas of mangrove vegetation structures:

– Seawaters predominance with a salinity between 15‰ and 20‰ coincide with areas of pioneer vegetation largely dominated by *Rhizophora* spp. and sometimes *Nypa fruticans* in degraded recolonization zones;
– Intermediate waters with a salinity between 5‰ and 10‰ characterize the regressive series composed of small stunted *Rhizophora* generally associated with *Acrostichum aureum* and *Dalbergia ecastaphyllum* or *Hibiscus tiliaceus*;
– Low salinity waters with concentrations between 0‰ and 5‰ sheltered in the river front large *Rhizophora* spp. trees and in the transitional zone, a forest vegetation dominated by *Guibourtia demeusei* or *Cynometra mannii* or species of *Arecaceae* family in degraded areas.

The salinity of soil water is much higher. Important values were found in the *Avicennia germinans* stands. The distribution of forest plant species follows the salinity gradient. True mangrove plants tolerate medium level of salinity while mangrove associate plants are commonly found in low salinity areas (Din et al. 2002). Salinity also varies within the sediments (along a core), the surface portions being less salty.

3.3 Mangrove Composition

Mangrove landscapes in Cameroon are slightly different in structure from other mangrove ecosystems in the Gulf of Guinea, but the composition remain the same. The flora and fauna are not very diversified in comparison with other types of forest in the country.

3.3.1 Flora

The flora of the Cameroon mangroves and associates presents 63 species divided into 54 genera and 29 families. The herbaceous stratum represents less than 1% of all vegetation. There are seven indigenous species consisting of *Acrostichum aureum* L., *Avicennia germinans* (L.) Stearn, *Conocarpus erectus* L., *Laguncularia racemosa* (L.) Gaertn. F, *Rhizophora racemosa* GFW Meyer, *R. harrisonii* Leechman, and *R. mangle* L. *Nypa fruticans* Wurmb. is often considered as an introduced species from Indo-Malaysian mangroves (Din 2001).

These previous species, considered as true mangrove species live in association with others considered like associates (Tomlinson 1986). Most frequent plant species include *Alchornea cordifolia* Müll. Arg., *Annona glabra* L., *Anthocleista* *vogelii* Planch., *Bambusa vulgaris* Schrad. ex J.C. Wendn., *Cocos nucifera* L., *Cynometra mannii* Oliv., *Dalbergia ecastaphylum* (L.) Taub., *Drepanocarpus lunatus* GFW Meyer, *Elaeis guinensis* Jacq., *Eremospatha wendlandiana* Dammer ex Becc., *Hibiscus tiliaceus* L., *Guibourtia demeusei* (Harms) J. Léonard, *Ormocarpum verrucosum* P. Beauv., *Pandanus candelabrun* P. Beauv., *Paspalum vaginatum* Sw, *Phoenix reclinata* Jacq., *Raphia palma-pinus* (Gaertn.) Hutch., *Sesuvium portulacastrum* L., etc.

The diversity in mangrove ecosystems can be expressed in terms of types of vegetation encountered. The flora of the Wouri river estuary mangrove is made in the suburban area of monospecific stations dominated widely by *Rhizophora* spp. while *Avicennia germinans* and especially *Nypa fruticans* succeeded in large deforested areas. The next group, a blend of trees, shrubs, palms and grass, appears more diversified, consisted of *Acrostichum aureum*, *Cynometra mannii*, *Dalbergia ecastaphyllum*, *Drepanocarpus lunatus*, *Hibiscus tiliaceus*, *Ormocarpum verrucosum*, *Pandanus candelabrum*, *Phoenix reclinata*, *Raphia palma-pinus*, etc. which are associations of plants generally observed along the edges slightly muddy (Din 2001; Nfotabong-Atheull et al. 2013).

In the peri-urban areas, mangroves are converted into agricultural fields in which there are species such as *Zea mays* L., *Phaseolus vulgaris* L., *Elaeis guineensis* Jacq., *Musa* spp., *Saccharum officinarum* L. Other observations show that change of land use promotes presence of unusual species. This is among other: *Paspalum vaginatum*, *Anthocleista vogelii*, *Sesuvium portulacastrum*, and *Alchornea cordifolia*. The presence of these species in the intertidal zone is justified by physicochemical changing properties of the soil due to scarcity of water in these places once regularly flooded (Nfotabong-Atheull et al. 2013).

The structure of mangrove forest always characterizes this ecosystem worldwide by an organization of vegetation based on distinct bands or zones, parallel to the shore of the main water channel and each zone usually dominated by a single plant species or rarely a combination of two plant species (Tomlinson 1986; Din 2001; Marchand 2007; Din and Baltzer 2008; Nfotabong-Atheull 2011). The structure of the Wouri river estuary mangroves is especial in the dimensions of trees (Fig. 3.4). *Rhizophora* species have trees which can grow up to 50 m height in mature and undisturbed forest while *Avicennia germinans* trees have shown diameters upon 100 cm (Din and Baltzer 2008; Din et al. 2002).

In many places, mangrove ecosystems of the Wouri river estuary have presented a complete progressive series from pioneer stage to almost climax stands. The physiognomy of mangrove vegetation has been characterized by four landscapes (Fig. 3.5):

1. the pioneer vegetation is consisted of *Rhizophora* seedlings and shrubs which grow on an unstable substrate in the accretion areas installed in the center or on the edges of the river and channels. After the depletion of the mangrove forest, a recolonization is observed on abandoned and temporally flooded areas. *Nypa fruticans* generally dominated this stage and the evolution led to transform the original mangrove vegetation features;

Fig. 3.4 Measurement of mangrove tree structure parameters: (**a**) *Rhizophora* zone; (**b**) *Avicennia* zone

2. on the concave edges (accretion zones), after the pioneer stands, a monospecific wide band of *Rhizophora* spp. with tall individuals exceeding 40 m in height extends sometimes on hundreds of meters. Inside the opened forest, this species often formed mixed stands with giants *Avicennia germinans* and thickets *Nypa fruticans*;
3. the convex banks (erosion areas) are characterized by consolidated substrate consisted of low alluvial deposits. *Acrostichum aureum* is tightly mixed with

a

b

Fig. 3.5 Landscapes of Cameroon mangrove vegetation. (**a**) Pioneer progressive area; (**b**) recolonization by *Nypa fruticans*; (**c**) tall individuals of *Rhizophora* spp.; (**d**) setting up of *Avicennia germinans* band; (**e**) open forest with *Rhizophora* spp. (*background*) and *Acrostichum aureum* (*front*); (**f**) very low flooded swampy forest with *Pandanus* sp.

shrubs and short trees of *Rhizophora* spp. characterized by dwarfed and tortuous stems. The inner part of this zone is generally opened and often invaded by shrubs and young trees of *Conocarpus erectus*, *Dalbergia ecastaphyllum*, *Drepanocarpus lunatus*, *Hibiscus tiliaceus*, *Laguncularia racemosa*, *Ormocarpum verrucosum*;

4. the back mangrove areas or transitional zone looks like any tropical swamp forest marked however by the continuous presence of mangrove characteristic spe-

c
d

Fig. 3.5 (continued)

cies among swampy forest species. The last band is a low flooded area occupied mainly by species of *Annona glabra*, *Anthocleista vogelii*, *Cynometra mannii*, *Guibourtia demeusei*, *Pandanus candelabrum*, *Phoenix reclinata*, *Raphia palma-pinus*, and few true mangrove forest species.

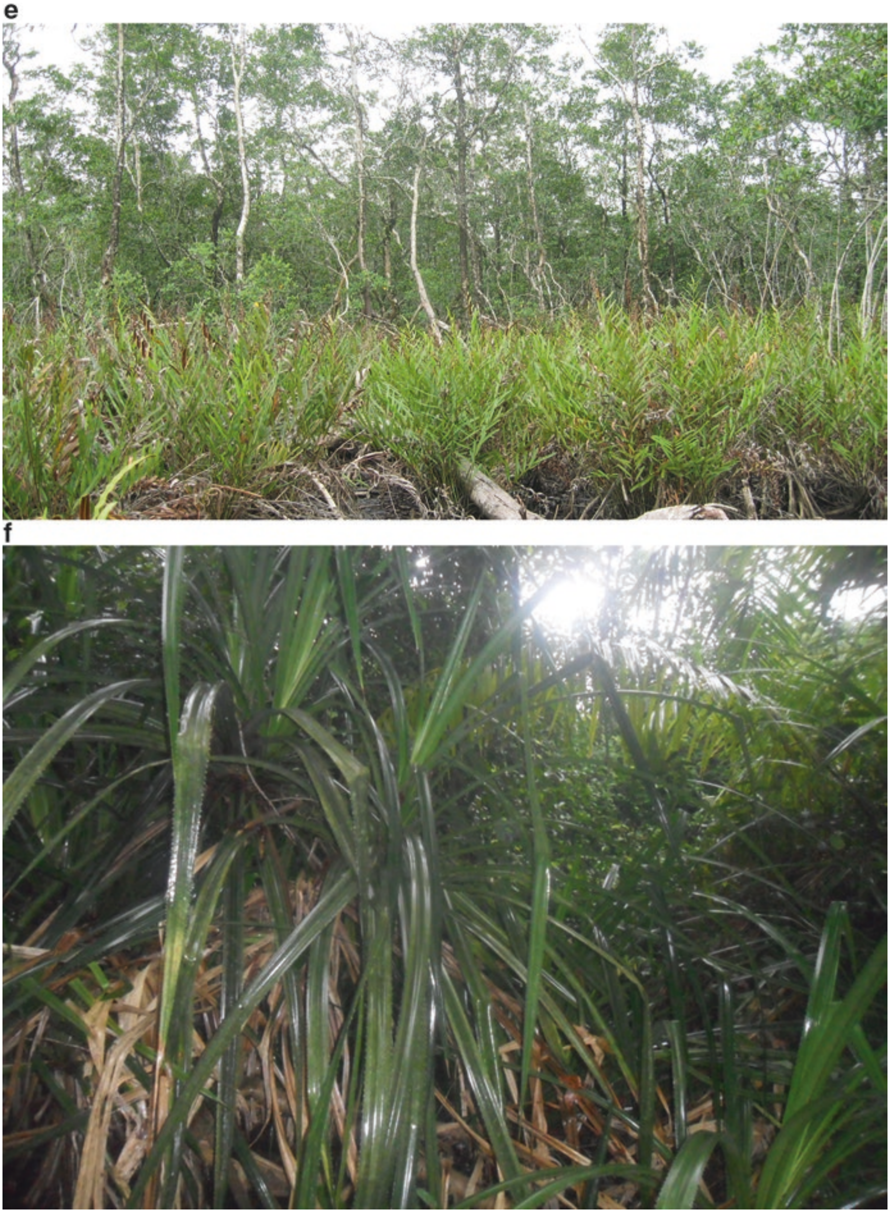


Fig. 3.5 (continued)

3.3.2 Fauna

Mangrove forests play a crucial role in providing suitable habitats for fauna, safe breeding and chick rearing grounds, nurseries for a diversity of fishes and shellfishes, as well as ideal foraging grounds for animals such as fishes, birds and aquatic invertebrates and refuge from predators (Macintosh et al. 2002). Faunal assemblages of mangroves are significantly less studied and documented than the forests they inhabit (Lee 2008). Animals found within mangrove environments include a variety of taxa, many of which are vulnerable or threatened as a result of human activities in the coastal zone (Nagelkerken et al. 2008). Some of the animals depend on mangrove areas their whole lives while others utilize them only during specific periods such foraging, shelter and breeding (Nyanti et al. 2012).

3.3.2.1 Invertebrates

Invertebrates break down leaf litter that act as fertilizer (Smith 1987), increase surface area of mud through burrowing (Kristensen 2008; Penha-Lopes et al. 2009) and increasing the diffusion rate of gases (Lee 1998) that ultimately affect the growth and productivity of the mangrove vegetation (Nielsen et al. 2003; Kristensen and Alongi 2006). In the Wouri river estuary mangroves as worldwide, the well-known invertebrates are crabs and molluscs. They are the predominant taxa in mangrove forests and are thought to play a significant ecological role in the structure and function of this ecosystem (Cannicci et al. 2008; Ngo-Massou et al. 2012). They form an important link between the primary detritus at the base of the food web and consumers of higher trophic levels (Sousa and Dagremond 2011).

Crabs

Many researchers were interested by mangrove crabs in Cameroon (Boyé et al. 1975; Guiral et al. 1999; Longonje 2008; Longonje and Raffaelli 2013; Ngo-Massou et al. 2012, 2014, 2016). Din et al. (2014) have presented a review of mangrove crabs evolution in Cameroon. The data have shown 33 species recorded grouped into 19 genera and 10 families (Table 3.1). Ongoing surveys (unpublished data) have improved these results with more species and new families of crabs in the coastal Atlantic mangroves of the country (Fig. 3.6).

Molluscs

Abundance and biomass of molluscs in mangrove habitats can be equally impressive than brachyuran crabs (Nagelkerken et al. 2008), although the number of comparative studies on mollusc diversity and structure in Cameroon mangroves is limited (Plaziat 1974; Boyé et al. 1975; Bandel and Kowalke 1999; Guiral et al.

Table 3.1 Mangrove crab species found in Cameroon

Family	Genera	Scientific name
Gercacinidae	*Cardisoma*	*Cardisoma armatum*
		Cardisoma guanhumi
Grapsidae	*Goniopsis*	*Goniopsis cruentata*
		Goniopsis pelii
	Grapsus	*Grapsus grapsus*
	Pachygrapsus	*Pachygrapsus transversus*
		Pachygrapsus gracilis
		Pachrygrapsus sp. 2
Macrophthalmidae	*Macrophthalmus*	*Macrophthalmus* sp.
Majidae	*Maja*	*Maja squinado*
Ocypodidae	*Ocypode*	*Ocypode africana*
		Ocypode cursor
		Ocypode ippeus
	Uca	*Uca tangeri*
Panopeidae	*Eurypanopeus*	*Eurypanopeus Blanchardi*
	Panopeus	*Panopeus africanus*
Pilumnidae	*Pilumnopeus*	*Pilumnopeus africanus*
Potunidae	*Callinectes*	*Callinectes amnicola*
		Callinectes pallidus
	Portunus	*Portunus validus*
Sesarmidae	*Armases*	*Armases elegans*
	Chiromantes	*Chiromantes buettikoferi*
		Chiromantes angolense
	Metagrapsus	*Metagrapsus curvatus*
	Perisesarma	*Perisesarma alberti*
		Perisesarma huzardi
		Perisesarma kamermani
	Sesarma	*Sesarma* sp. 1
		Sesarma sp. 2
		Sesarma sp. 3
		Sesarma sp. 4
Varunidae	*Helice*	*Helice* sp.

Modified after Din et al. (2014)

1999; Ngo-Massou et al. 2012). However, these works have provided relevant data concerning essentially the distribution of molluscs in the Cameroon Atlantic coast (Table 3.2). A relative recent census found 12 species among which 11 gastropods and one unknown species of bivalves (Ngo-Massou et al. 2012).

Throughout most mangrove habitats, molluscs live on and in the muds, firmly attached to the roots, or forage in the canopy. They occupy a number of niches and contribute to the ecology of the mangal in important ways (Kathiresan and Bingham 2001). The composition of molluscan community in Wouri estuary river mangrove forest consist of two main taxa as Gastropods and Bivalves (Fig. 3.7).

Fig. 3.6 Mangrove crab species encountered in Cameroon ecosystems

Table 3.2 Species of molluscs found on mangrove areas of Cameroon. Ongoing studies have found many undefined species (not present in this document) in order to improve the database

Species	Plaziat (1974)	Boyé et al. (1975)	Bandel and Kowalke (1999)	Guiral et al. (1999)	Ngo-Massou et al. (2012)	Ongoing research
Achatina achatina	–	–	–	–	+	+
Anadara senilis	–	–	–	+	–	–
Angiola lineata	–	–	+	–	–	–
Arca sp.	+	–	–	–	–	–
Assiminea hessei	–	–	+	–	–	–
Bivalve (Unknown)	–	–	–	–	+	–
Corbula trigona	+	–	–	+	–	–
Crassostrea gasar	+	–	–	+	–	+
Cyrenoida rosea	+	–	–	+	–	+
Cyrenoida rufa	+	–	–	–	–	–
Cyrenoida senegalensis	–	–	–	+	–	–
Egeria radiata	+	–	–	–	–	+
Fissurela sp.	+	+	–	–	–	–
Iphegenia deleserti	–	–	–	+	–	–
Iphegenia leavigata	–	–	–	+	–	–
Iphegenia rostrata	+	–	–	–	–	+
Littorina (Scabra) angulifera	+	+	+	–	–	+
Littorina sp.	–	–	–	–	–	+
Melampus liberianus	+	+	+	+	–	–
Melanoides pergracilis	–	–	–	–	+	+
Melanoides tuberculata	–	–	+	–	+	+
Melanopsis sp.	–	–	–	–	–	+
Murex sp.	–	–	–	–	–	+
Mytilopsis africana	–	–	–	+	–	–
Neritilia manoeli	–	–	+	–	–	–
Neritilia rubida	–	–	+	–	–	+
Neritina afra	–	–	+	–	–	–
Neritina glabrata	+	+	+	+	–	+
Neritina lineolata	–	–	–	–	–	+
Neritina oweniana	–	–	–	+	–	–
Neritina rubricata	–	–	+	–	–	–
Neritina senegalensis	+	+	–	+	–	+
Onchidium sp.	–	+	–	–	–	–
Ostrea tulipa	+		–	–	–	–
Pachymelania aurita	+	–	+	+	+	+

(continued)

Table 3.2 (continued)

Species	Plaziat (1974)	Boyé et al. (1975)	Bandel and Kowalke (1999)	Guiral et al. (1999)	Ngo-Massou et al. (2012)	Ongoing research
Pachymelania byronensis	–	+	+	+	–	+
Pachymelania fusca	+	–	+	+	+	+
Pachymelania Granifera	–	–	–	–	+	+
Pachymelania mutans	–	–	–	–	–	+
Pachymelania sp.	–	–	–	–	+	+
Potadoma lirincta	–	–	–	–	+	+
Potamopygus ciliatus	–	–	+	–	–	–
Pupura callifera	+	–	–		–	–
Pupura yetus	+	–	–	–	–	–
Scabra scabra	+	+	–	–	–	–
Semifusus moris	–	+	–	+	–	–
Sepia officinalis	+	–	–	–	–	–
Siphora mouret	+	–	–		–	–
Tangelus angulatus	–	–	–	+	–	–
Tectarius granosus	+	+	–	–	–	–
Terebralia palustris	–	–	–	–	–	+
Thais callifera	+	+	–	+	–	+
Theodoxus niloticus	–	–	–	–	+	+
Theodoxus sp.	–	–	–	–	–	+
Thiaridae (Unknown species)	–	–	–	–	–	+
Tympanotonus fuscatus	+	–	+	+	+	+
Tympanotonus radula	–	–	+	–	+	+

Other Invertebrates

Invertebrates as Insects, Annelids and other Crustaceans (prawns, shrimps) are inhabitants of mangrove forest, but there is not data about these invertebrates in mangroves of Wouri estuary river. Although the mangal may be a sink for settlement and early growth of shrimp and prawns, it may also be a source for larvae that are transported to other habitats (Kathiresan and Bingham 2001).

3.3.2.2 Mangrove Vertebrates

The distribution of the mammals in the mangroves is hardly better known. The West African Manatee (*Trichechus senegalensis*) occurred in the Wouri river estuary can be considered as one of the most sanctuaries of Manatee in Cameroon coast. The

Fig. 3.7 Some mangrove characteristic species of molluscs found in Cameroon

specie is common in rainy season. Since several years, the manatee is faced to many threats (Ayissi and Jiofack 2014). The carnivores are rather rare; all the species of monkeys are arboreal. The existence of Whales and Dolphins along Cameroon coast is known, but the species distribution is unknown (Ayissi et al. 2011). Whereas some species can be more or less frequently observed, others are much rare and make only incursions of short duration into the mangrove forest.

Mangrove waterways are rich fishing grounds and many commercial species can be found. The most frequently encountered species are: *Ilisha africana*, *Sardinella maderensis*, *Caranx* spp., *Tilapia* spp., *Dentex congolensis*, *Arius* spp., *Pomadasys* spp. and *Periophtalmus papilio*, the mud skippers (Fig. 3.8).

Fig. 3.8 *Periophtalmus papilio*, the most characteristic mangrove fish

Four species of sea turtles are common along the Cameroon coast: Leatherback turtle (*Dermochelys coriacea*), Green turtle (*Chelonia mydas*), Olive ridley (*Lepidochelys olivacea*) and Hawsksbill (*Eretmochelys imbricata*). But, just Green turtle and Hawskbill are common on mangroves (Ayissi and Jiofack 2014). These turtle species utilize mangrove areas for foraging and breeding purposes due to the richness and diversity of plankton and benthic food resources.

Few crocodile species exist in mangroves, estuarine, and adjacent rivers (Rajpar and Zakaria 2014), but the presence of species such as *Crocodilus niloticus* and *Ostealaemus tetrapis* all classified like species in danger by IUCN is surmised. A wide array of animals such as birds, fishes and mammals are prey of these crocodile species. Other reptiles founded are snakes and mangrove monitor lizard (*Varanus niloticus*).

The mangrove habitat plays a host role to a moderate number of bird species around the globe. Hundreds of bird species migrate to the mangrove forest for feeding, roosting, nesting and breeding, certain species are dependent on the mangrove ecosystem and they also play a vital role in maintaining the mangrove ecosystem through several activities mainly those of pollinator, seed disperser, and pollution regulation providing food for other animal predators and also contributing to nutrient recycling processes. However, few animals have been reported to feed on mangrove trees directly, whereas other parts of the mangrove, like dead leaves, stems and roots (Rohit et al. 2016).

Van der Waarde (2007) have found 300 birds along Cameroon coast among which less than a hundred of species are present in mangrove areas (Table 3.3). The water birds reported were categorised into four groups:

– Cormorants to Ibises which counted Little Egret, Grey Heron, Great Egret, African Openbill storks, White pelican, Pink-backed Pelican (*Pelecanus rufescens*),

Table 3.3 Water birds found in Cameroon mangrove areas

Family	Scientific name	Common name
Anatidae	*Dendrocygna viduata*	White-faced Whistling Duck
Anatidae	*Plectropterus gambensis*	Spur-winged Goose
Anatidae	*Pteronetta hartlaubii*	Hartlaub's Duck
Anatidae	*Nettapus auritus*	African Pygmy Goose
Anhingidae	*Anhinga rufa*	African Darter
Ardeidae	*Ardeola ralloides*	Squacco Heron
Ardeidae	*Bubulcus ibis*	Cattle Egret
Ardeidae	*Butorides striatus*	Green-backed Heron
Ardeidae	*Egretta gularis*	Western Reef Heron
Ardeidae	*Egretta garzetta*	Little Egret
Ardeidae	*Mesophoys intermedia*	Intermediate Egret
Ardeidae	*Casmerodius albus*	Great White Egret
Ardeidae	*Ardea purpurea*	Purple Heron
Ardeidae	*Ardea cinerea*	Grey Heron
Ardeidae	*Ardea melanocephala*	Black-headed Heron
Ardeida	*Ardea goliath*	Goliath Heron
Burhinidae	*Burhinus senegalensis*	Senegal Thick-knee
Charadriidae	*Charadrius dubius*	Little Ringed Plover
Charadriidae	*Charadrius hiaticula*	Ringed Plover
Charadriidae	*Charadrius marginatus*	White-fronted Plover
Charadriidae	*Pluvialis squatarola*	Grey Plover
Charadriidae	*Vanellus albiceps*	White-headed Lapwing
Ciconiidae	*Anastomus lamelligerus*	African Openbill Stork
Ciconiidae	*Ciconia episcopus*	Wolly-necked Stork
Ciconiidae	*Mycteria ibis*	Yellow-billed Stork
Glareolidae	*Glareola cinerea*	Grey Pratincole
Heliornithidae	*Podica senegalensis*	African Finfoot
Jacanidae	*Actophilornis africanus*	African Jacana
Laridae	*Larus fuscus*	Lesser Black-backed Gull
Laridae	*Larus cachinnans*	Yellow-legged Gull
Laridae	*Sterna nilotica*	Gull-billed Tern
Laridae	*Sterna caspia*	Caspian Tern
Laridae	*sterna maxima*	Royal Tern
Laridae	*Sterna sandvicensis*	Sandwich Tern
Laridae	*Sterna hirundo*	Common Tern
Laride	*Sterna albifrons*	Little Tern
Laridae	*Chlidonias niger*	Black Tern
Laridae	*Rynchops flavirostris*	African Skimmer
Pelecanidae	*Pelecanus onocrotalus*	Great White Pelican
Pelecanidae	*Pelecanus rufescens*	Pink-backed Pelican
Phalacrocoracidae	*Phalacrocorax africanus*	Long-Tailed Cormorant
Podicipedidae	*Tachybaptus ruficollis*	Little Grebe

(continued)

Table 3.3 (continued)

Family	Scientific name	Common name
Rallidae	*Amaurornis flavirostris*	Black Crake
Recurvirostridae	*Himantopus himantopus*	Black-winged Stilt
Recurvirostridae	*Recurvirostra avosetta*	Pied Avocet
Scolopacidae	*Calidris alba*	Sanderling
Scolopacidae	*Calidris minuta*	Little Stint
Scolopacidae	*Calidris ferruginea*	Curlew Sandpiper
Scolopacidae	*Philomachus pugnax*	Ruff
Scolopacidae	*Gallinago gallinago*	Common Snipe
Scolopacidae	*Limosa limosa*	Black-tailed Godwit
Scolopacidae	*Limosa lapponica*	Bar-tailed Godwit
Scolopacidae	*Numenius phaeopus*	Whimbrel
Scolopacidae	*Numenius arquata*	Eurasian Curlew
Scolopacidae	*Tringa tetanus*	Common Redshank
Scolopacidae	*Tringa stagnatilis*	Marsh Sandpiper
Scolopacidae	*Tringa nebularia*	Common Greenshank
Scolopacidae	*Tringa ochropus*	Green Sandpiper
Scolopacidae	*Tringa glareola*	Wood Sandpiper
Scolopacidae	*Tringa hypoleucos*	Common Sandpiper
Scolopacidae	*Arenaria interpres*	Ruddy Turnstone
Scopidae	*Scopus umbretta*	Hamerkop
Threskiornithidae	*Bostrychia hagedash*	Hadada Ibis
Threskiornithidae	*Threskiornis aethiopicus*	Sacred Ibis

Modified after Van der Waarde (2007)

Squacco Heron, Green-backed Heron and Sacred Ibis mainly concentrate in the Ndian Basin and Wouri estuary, the two main mangrove areas of the country;

- Ducks, Rails and Fin foots: in this group the Hartlaub's Ducks were found mainly around Sanaga river;
- Waders, the well represented group on the Cameroon coast which included Palearctic species such as Common Greenshank, Common Redshank, Curlew sandpiper and Common Ringed Plover. This group also counted the Grey Pratincole.
- Gulls, Terns and African Skimmer which are the most dominant group of water birds on Cameroon coast represented by Royal Tern and African Skimmer.

Mangrove fauna often show vertical and horizontal zonation. Some of them dominate in mud, some on the shrubs and leaves and the others around roots (Mauris 2005) and can be divided into three inhabitants such as (i) aquatic animals, (ii) semi-aquatic animals and (iii) terrestrial animals based on their living behaviour. These animal communities utilize mangrove areas for their daily activities such as foraging, breeding, and loafing. These animals play a significant role in the management of mangrove forests and in balancing nature in and around the mangrove areas (Spalding et al. 2010; Nyanti et al. 2012).

3.4 Major Factors of Mangrove Degradation

More than 90% of world's mangroves are located in developing countries where impoverished human populations depend on their resources for subsistence (Duke et al. 2007; Walters et al. 2008). Human impacts on mangroves, including climate change, have received much attention of late mainly because mangrove deforestation is occurring at a rate of 1.2% per year, which implies that most forests will disappear within this century (Alongi 2002, 2015). At a global level, natural and anthropogenic drivers of mangrove destruction and degradation include sea-level rise, the harvest of forest products for local (wood, charcoal, and tannins) and industrial (woodchips and lumber) consumption, conversion of mangrove forests into agricultural, aquacultural, industrial and urban areas (Di Nitto et al. 2008, 2014; Rakotomavo and Fromard 2010; Paul and Vogl 2011; Goessens et al. 2014; Santos et al. 2014) and other activities such as river damming and herbicide use (Abuodha and Kairo 2001; Koedam et al. 2007).

In Cameroon, natural disasters and several anthropogenic activities have been identified in relation with mangrove ecosystems depletion. Wood harvesting, urban infrastructures, fishing, sand and gravel extraction, conflicts, petroleum exploitation, coastal erosion, climate change, invaded aquatic plants and agriculture appear to be the most important factors of mangrove degeneration. In addition, most of these parameters are making place to secondary destructive factors, generally linked to poverty such as dwellings, livestock, sustenance agriculture, collecting Non-Timber Forest Products, digging, landfill, dyke construction and large clear-felling which also contributed widely to mangrove degradation (Din 2001; Feka and Manzano 2008; Nfotabong-Atheull et al. 2009; Din et al. 2016; Fusi et al. 2016; Ngo-Massou et al. 2016).

3.4.1 Logging

This activity began in the mangroves of Cameroon at the dawn of the twentieth century (Din et al. 2008). In 1919, a French company based in Manoka devastated the Cameroon mangrove estuary for the home country needs. In less than a decade, more than 3000 tons of wood were sold in the form of railway sleepers. The consequences of this activity are still visible in this borough.

The recent introduction of modern transportation and cutting equipments has accelerated the degradation of several primary formations. Mangroves of the Wouri estuary are strongly flattened. *Rhizophora* spp., flammable species fresh, are most at risk followed by *Avicennia germinans*. Several species of mangrove forests generally have a very hard wood which is flammable fresh and this feature causes the cutting of trees still standing and accelerates the degradation of the forest. The modern sector degradation and the urban demographic pressure induced poverty which increase the number of loggers and consumers who cannot buy kerosene or cooking gas.

In the 1990s, loggers organized clandestine microenterprises in order to destroy Wouri estuary mangrove ecosystem. In addition, they did not pay any taxes although they realized real incomes above the national average. Mangroves wood markets in Douala are found throughout the city. The most important are Bobongo, Youpwe, Deido, Bonaberi. At Tiko, mangrove wood is widely used in households. Firewood is also used by fishermen for drying fish. This is visible in all the fishing camps, even in the localities of the estuary of the Rio del Rey (Ekondi Titi, Bamusso, Meme, Andokat) where tree operating traces are well marked. In Kribi mangroves, collecting timber carried on a small scale is highly selective and often regard the straight trunks of small diameter which are suitable for the construction of precarious homes.

Loggers in the mangroves of Cameroon priority choose *Rhizophora* spp. for firewood and *Avicennia germinans* for lumber. By cons, they slaughter daily without discrimination all other species. Excessive cutting of forests in some mangrove background areas usually leads to the development of low vegetation not erect port. These dense thickets of *Hibuscus tiliaceus* associated with *Dalbergia ecastaphyllum*, *Acrostichum aureum*, *Drepanocarpus lunatus*, *Ormocarpum verrucosum*, etc. adverse shaded form clumps in the establishment of new propagules. In addition, the slaughter intensified of mature mangroves on mudflats in major or intermediate submersion often suggests a new landscape characterized by a canopy punctuated by gaps.

The wood exploitation constitutes one of the main reasons for the degradation of mangrove in the world. Almost 53.216 ha of the Cameroon's mangrove forests have been lost over the last 13 years (Spalding et al. 2010). The mangrove trees are used as an important or potential source of firewood and charcoal, in response to the increase in domestic needs of energy by urban populations in developing countries. Wood appropriation in the mangroves remains an activity which is fully controlled by the informal sector causing that ecosystem to lose its market value.

Mangrove wood is perceived and estimated as the major disruptive factor in the Cameroon estuary ecosystems. Wood extraction resulted mainly from domestic needs of neighboring city dwellers and fish smoking hearths installed inside mangrove camps (Fig. 3.9). Since the both last decades, significant changes in mangrove vegetation (diversity and structure) were observed in Wouri river estuary stands (Din et al. 2008; Nfotabong-Atheull et al. 2009, 2013). The large disturbed mangrove forests found in the early 1990s, which were easily accessible by foot, had been progressively clear-cut from the landward margin toward the main water channels. The degraded areas were subsequently developed for housing. Some areas like Youpwe, Mboussa Essengue, Wouri bridge, Bon'Ewonda and many areas in Bonaberi have been strongly affected and the rapid population increase has especially lead to large-scale deforestation due to a growing demand for housing land and logging cultivation (Din et al. 2008).

Rhizophora spp. is dominant and strongly marketed or directly used in the households for subsistence needs (Feka et al. 2009). Exploitation of mangroves for fuel wood, charcoal production, construction and other uses have been identified as an important pervasive and intrusive threat to this ecosystem (Feka and Manzano 2008). With about 350 loggers estimated in Douala mangrove areas, daily quantity of wood harvested from this ecosystem was about 500 m^3 and revenues was estimated around 11,500 USD (Din et al. 2008).

Fig. 3.9 Mangrove wood exploitation in Cameroon. (**a**) Irregular logging using motor chain-saws; (**b**) pile of poles for cooking and building; (**c**) collection of planks inside mangrove forest; (**d**) Mangrove wood service waiting for transportation to urban area

Fig. 3.9 (continued)

3.4.2 Sand Quarries

The exploitation of sand is one of the important activities in the mangrove, mainly around large cities. Sand quarries are visible throughout the estuary of Cameroon, particularly around the city of Douala (Youpwe, Bonaberi, Bonamouang, Bonagang, Bon'Ewonda and others) with an annual mangrove sand production estimated at 90,000 m^3. Unlike logging, it does not only affect the vegetation but can cause disturbances in the structure or even the nature of the substrate. This activity is practiced in low tide, which degrades the quality of the water or coves on previously bare soil, thus causing degradation of the environment. This activity is becoming increasingly important in Cameroon estuary (Fig. 3.10) due to the ever increasing demand of export sand towards Equatorial Guinea.

The surroundings and the main river beds are mined mainly in Douala and Tiko. The amount of sand taken is important and mangrove areas especially offer a wide variety of sand quality (fine sand to pebbles). In the 1990s, over 50% of the sand used in the city of Douala for various constructions derived from mangrove areas (Din 2001). The sand mining appeared to be an important economic activity which involved both the informal sector and dredging societies from public works. When

Fig. 3.10 Sand quarry in Douala degraded mangrove forest (**a**) partition of sands and gravels operation near a channel; (**b**) loading of a truck with mangrove sand near *Rhizophora* and *Avicennia* trees

the exploitable areas were detected, trees were felled in a radius of several hundreds meters. The waterways are being diverted or simply blocked. The immediate consequence is the drying of all the area to be harvested and also the whole rear portion which was fed by these pathways. Mangrove characteristic species are very sensitive to daily flooding by tides; stopping the water flow always triggers in short term, the degeneration of the forest composition and structure.

Sandpits change the structure and physiognomy of mangrove ecosystems. Attempts to regenerate mangroves observed in many quarries show that the destruction of mangrove forests due to this activity may be reversible if brackish water supply is restored degraded substrates. The new progressive series is always lead by *Nypa fruticans* but had never evolved to a climaxic mangrove forest.

3.4.3 Landed Distribution

Mangrove forests are experiencing long-term and severe decline (Valiela et al. 2001; Alongi 2002). The causes of such losses include the coastal land-use development leading to losses due to over-harvesting (Walters et al. 2008; Nfotabong-Atheull et al. 2009), expanded agriculture (Hossain et al. 2009), and conversion into shrimp farming ponds (Guimarães et al. 2010), to name but a few. From 1974 to 2009, mangrove forest area had decreased 53.16% around Douala concurrent with a substantial increase of settlements (60%), roads (233.33%), agriculture areas (16%), non-mangrove areas (193.33%), and open water (152.94%) (Nfotabong-Atheull et al. 2013). Deforestation appears to be one of the most ubiquitous forms of land degradation worldwide. Although remote sensing and aerial photographs can supply valuable information on land/use cover changes, they may not regularly be available for some tropical coasts (*e.g.*, Cameroon estuary) where cloud cover is frequent.

While the drivers causing this depletion/deforestation vary from one region to another, there is a general consensus that anthropogenic activities are the root drivers of this change. In addition, while the dilemma of mangrove ecosystem conversion for aquaculture is recognized as the greatest threat to mangrove forests, globally pollution, agriculture and urbanization seem to be making headway among developing countries. Furthermore, exploitation of mangroves for fuel wood, charcoal production, construction, and other uses have been identified as an important pervasive and intrusive threat to this ecosystem (Dodman et al. 2006), particularly within coastal developing countries, where local communities depend on the exploitation and use of these resources for their livelihoods (Focho et al. 2001).

Douala is the first most populous city in Cameroon, with a population that grew rapidly from 1,352,833 inhabitants in 1987 to 2,510,263 people in 2005 (Cameroon National Institute of Statistics 2009). The characteristic urban anarchic development, accompanied by demographic pressure in the peri-urban areas and emphasized by increase in poverty contributed hardly to forest depletion. This growth rates indicate an intense human pressure on the urban landscape. This expansion was therefore the driving factor for the "colonization" of new lands in spite of their physically hostile nature (Fig. 3.11).

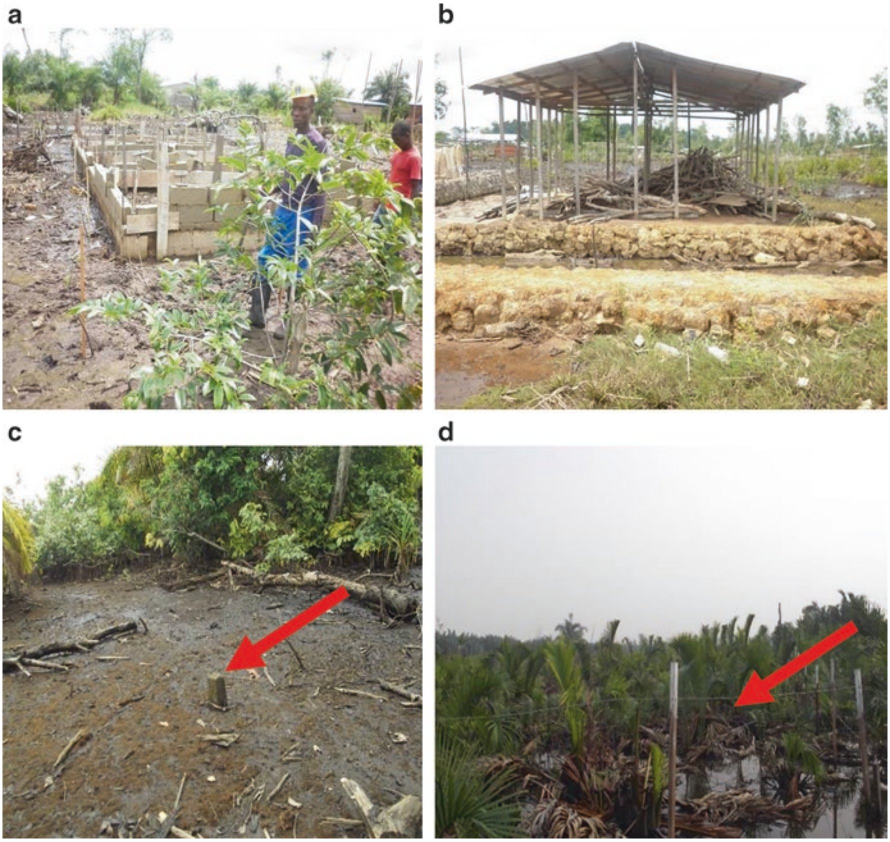


Fig. 3.11 Constructions in the mangrove areas (**a**) definitive material; (**b**) precarious material; (**c**) implanted boundary stone; (**d**) boundary line surrounded young *Nypa fruticans*

3.4.4 Anarchic Urbanization

The majority of coastal communities are dependent on the surrounding mangroves for both subsistence and commercial uses (Din et al. 2008; Nfotabong-Atheull et al. 2009, 2011). Here, mangroves are under severe pressure from housing development, agriculture, sand quarrying, road construction, lumber harvesting, and seaport maintenance and expansion. Actually, Douala mangroves undergo the second bridge construction during which important mangrove areas are being destroyed. However, this urbanization transformed hectares of forest characterised by mature stands of mangrove trees such as *Rhizophora* spp. and *Avicennia germinans* into highly degraded areas with a dramatic change in botanical assemblages (Nfotabong-Atheull et al. 2011).

Douala has a coastal nature, with the presence of water and mangroves, especially the river Wouri estuary. It is common to find dwellings surrounded by scanty or permanent water. Much of the city is less planned and is not built sustainably: less than 8% of the total area is occupied by condominium, multi-storey, villa and single storey, whereas mixed construction has a surface area of 23.5% and mud/wood construction, 20.4% (CLUVA 2010). This indicates a very poor population and hence a high pressure on natural resources and vegetation. Plantations, parks, and agriculture cover only a little part of the area. In the face of the rapid urban expansion for the past 20 years, it is certain that the demand for sand, soils and scoria would remain elastic. The profitability and others facilities offered by the Douala city favored this activity and induced the increment of quarries.

Urbanization is one of the most unfavorable factors to the conservation and evolution of terrestrial ecosystems in emerging or developing countries (Fig. 3.12). The Cameroon coast contains four major cities with mangrove stands border agglomerations from Kribi to Limbe. Mangroves of Kribi were heavily damaged by the installation of modern buildings at the seaside (Nfotabong-Atheull 2011). The construction of tourist complexes after the realization of the asphalt road that connects Kribi to Edea have spared no specimen of mangroves (Din 2001). The development of the city of Tiko destroyed mangrove mainly in the old port that was certainly attracted people in the area. The installation of new populations favored the destruction of a major band of mangrove.

The establishment of roads inside mangroves led to an ecological fragmentation and isolation of two sides of mangroves that can no more exchange materials between them. The general aspect of the vegetation (composition and structure) corresponds to a senescent stand where swamp forest species gradually invade the site. Mangrove ecosystems could survive after fragmentation if the derivative forests are regularly watered by daily tides. The zonation and the progressive series change and affect deeply the structure of the forest but the composition will remain unchanged even if the structure of trees is also affected. Species like *Nypa fruticans* take advantage of such phenomenon to occupy more space.

Although mangroves contribute considerably to the social and economic wellbeing of the Cameroon coastal inhabitants, their total surface area has decreased by 30% in 20 years (Spalding et al. 2010), mainly due to rapid and uncontrolled urbanization around Douala city (Din et al. 2002; Ellison and Zouh 2012; Nfotabong-Atheull et al. 2013).

3.4.5 Wastes

Wastes and wastewater discharged into nearby mangrove forest, marshes and rivers can finally get into the mangroves ecosystem due to rainfalls, currents and tide movements. Direct discharges into mangroves are frequently observed in the Wouri estuarine mangrove. The mangrove areas are daily transformed as solid and liquid waste disposals accepted by the Douala urban council without any restriction

Fig. 3.12 Anarchic urbanization destroying mangrove ecosystems. (**a**) Electrification project following population settlement in mangrove degraded areas; (**b**) building inside mangrove area; (**c**) state project converting mangrove ecosystem; (**d**) fragmentation of mangrove forest resulting from a road opened

Fig. 3.12 (continued)

(Fig. 3.13). Tankers poured out liquid wastes collected from private cesspools along the city directly in the mangrove areas between Youpwe and Bois Des Singes. The Douala urban council lay a tax on this hazardous activity.

In the SW part of the town (Mambanda), a sawmill had poured for several decades solid wastes into the mangrove forests and had taken advantage of the progressive degradation of the forest to get a considerable free space. Many other

Fig. 3.13 Deposit of solid (**a**) and liquid (**b**) wastes in mangrove areas

factories and population discharge their waste directly into the rivers of the Cameroon estuary. Mangroves of the Wouri estuary have progressively become potential dumps for solid materials of which the low decomposition will cause the forest depletion associated with the intoxication of the food chain.

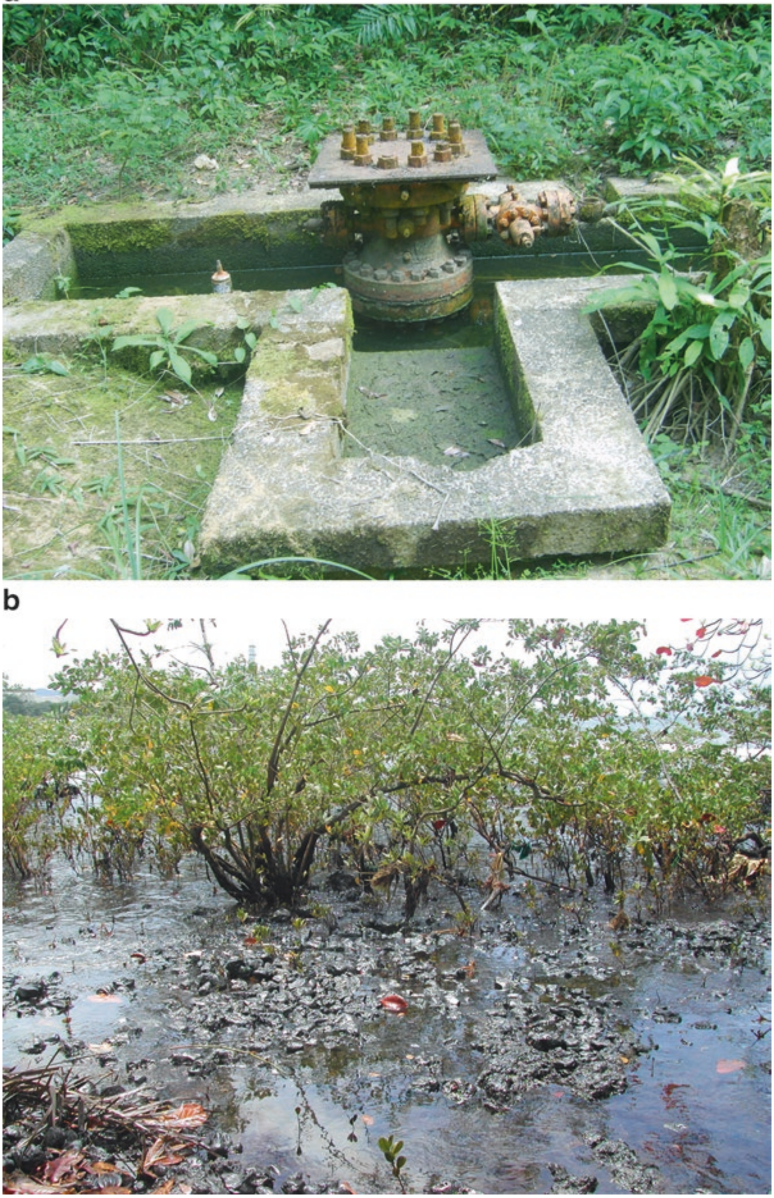
3.4.6 Petroleum Activities

Mangrove forests provide generally a variety of goods and services (Dahdouh-Guebas 2001; Walters et al. 2008). Oil spills seeping into coastal waters and rivers, covering exposed roots of mangroves and airy. It is difficult or impossible to respiratory plants lenticels to perform their essential functions when covered in oil, so they are slowly and progressively asphyxiated. The massive successive deaths of mangroves is a common phenomenon afflicting the mangrove areas where oil exploitation is practiced. All petroleum activities are source of risk. Due to the fact that oil spills often occur in remote areas, a large number of frequent accidents could go unnoticed for long periods of time and are not cleaned effectively and timely manner. According to Din (2001) such situations may occur in the mangrove of the Rio del Rey estuary as oil activities in this area are free from any control on their environmental impact.

Douala is one of the major shipping ports in the Guinea Gulf that serves the entire central Africa and refuels oil tankers to export locally extracted oil, another significant anthropogenic impact on the Wouri river estuary mangroves (Alemagi 2007; Duke 2016; Price et al. 2000; Van De Walle 1989). Many hazards from petroleum activities have been encountered in all mangrove areas of Cameroon (Fig. 3.14). Drilling activities destroyed many hectares of mangroves forest near Douala. A serious oil spill from the Limbe refinery had damaged all mangrove stands along the Atlantic Ocean. Seismic lines are responsible of mangrove depletion in the two mains mangrove stands of the country.

Non peri-urban mangroves located in the estuary of Cameroon, the mouth of the river Nyong and Kribi will face in the future to disturbance from exploration (seismic survey to identify potential petroleum reserves) and oil exploitation (Nfotabong-Atheull 2011). Devegetation of these intertidal forests resulting from seismic delineation leave bare soil may be eroded by wave movements (Osuji et al. 2010). The disruption of the structure of mangroves will be inevitable since the seismic exploration phase (cable routing) often require cutting trees up (Osuji et al. 2007). As has been shown in Nigeria, the recovery of disturbed areas will take about 3 years (Aston-Jones 1988).

a



b

Fig. 3.14 Effects of petroleum activities on mangrove ecosystem. (**a**) Abandoned well inside mangrove area; (**b**) mangrove degradation due to oil spill; (**c**) mangrove depletion after accumulation of wastes from drilling operations

c

Fig. 3.14 (continued)

3.5 Evolution of Douala City and Mangrove Degradation

3.5.1 Effects of State Projects

The implementation of projects for the urbanization of the city of Douala is one of the worst factors to mangrove conservation of the Wouri estuary river. The Douala city's urban plan had foreseen in the 1990s, the extension of the present "Boulevard Général Leclerc" to Bonamoussadi (northern part of the city). Its implementation should involve the disappearance of all coastal mangrove patches on the left bank of the Wouri river. In the same project, it provides a bridge that connects Bonamoussadi to Bonendale through Bonamatoumbè and Djebalè. That project combined with the "Sawa Beach project" will be an environmental disaster for the mangroves of the Wouri river estuary (Din 2001). The above projects are actually being executed. The predictions seem to be exact and the disappearance of mangrove stands are effective (Fig. 3.15). The construction of roads alone will cause a sharp reduction in mangrove areas. Building the bridges will result in more reduction surfaces, the deviation of the supply channels, blocking channels that feed the mangrove substrate and the installation of population. The construction of road infrastructure has an impact on the general aspect of the vegetation and structure of tree characteristics that match senescent stands where swamp forest species invade sites.

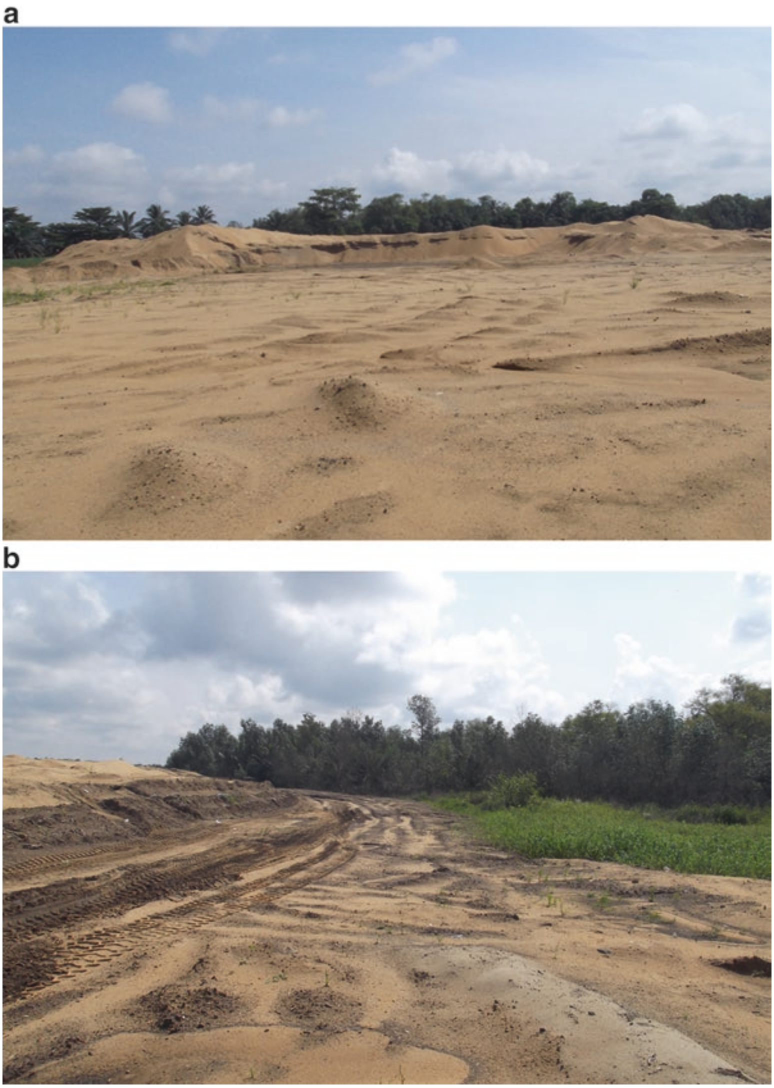
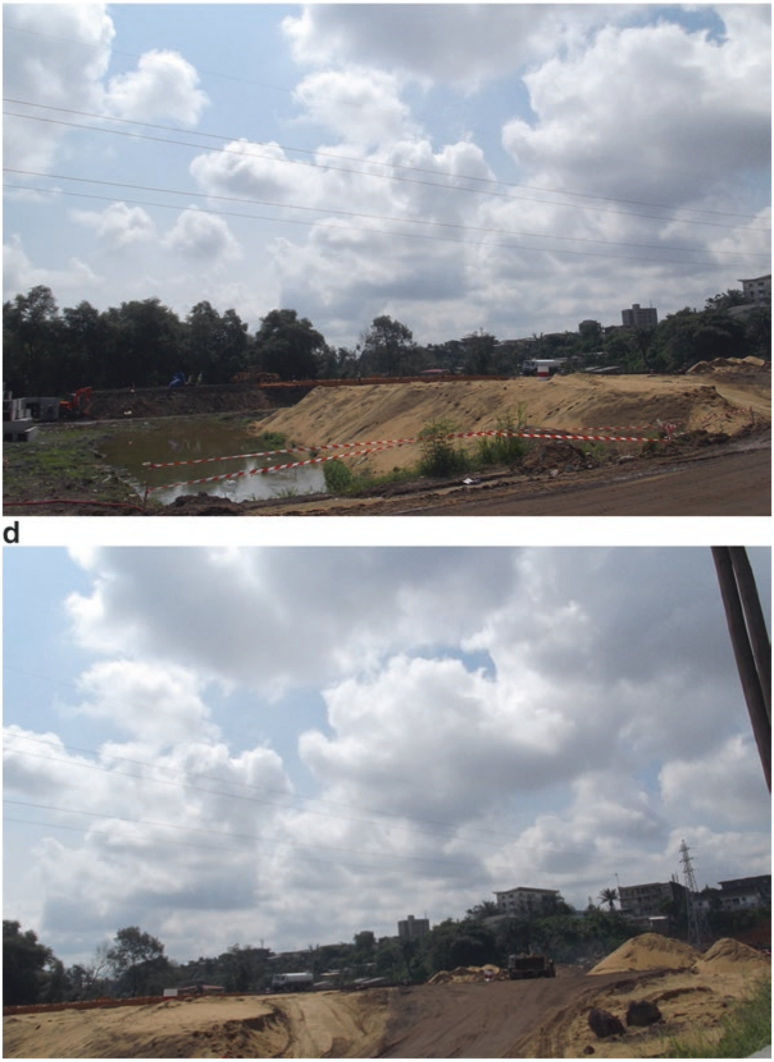


Fig. 3.15 Effects of the Wouri bridge rehabilitation on the neighboring mangrove ecosystems. (**a**) Bending sand on mangrove forests; (**b**) road opened inside mangrove area; (**c**) installation of equipment and live camps after mangrove area depletion; (**d**) extension of destruction and supply of solid materials; (**e**) progress of *Avicennia* band damaging; (**f**) total drainage of soil and trees which means mangrove disappearance

c



d

Fig. 3.15 (continued)

Fig. 3.5 (continued)

The construction of the road that connects Bonaberi to Bekoko caused the breakdown of many mangrove stands at the locality of Yapaki. Along this axis, shrubs of all mangroves characteristics species are frequently observed. The rehabilitation of the Wouri Bridge area in the 1980s, the creation and expansion of the port area and the creation of new residential areas including Mambanda (Bonaberi), Deido, Bonanloka, significantly reduced the surfaces of mangroves. This loss of vegetation cover of mangroves has ecological, economic and social effects.

The city's development projects have caused the loss of biodiversity of the mangroves of the Wouri river estuary with the loss of species in the Mboussa Essengue area include *Cassipourea barteri* (Hook. f. ex Oliv.) N.E. Br., and a proliferation of *Nypa fruticans*. The latter species is experiencing exponential colonization of the areas. The completion of several state projects had heavily impacted the evolution of mangroves in the Wouri estuary river. Ultimately, these achievements certainly cause the disappearance of important mangrove areas.

3.5.2 Effects of Demographic Pressure

In addition to a high rate of natural increase, estimated at 29‰, Douala receives each year, according to estimates from the Urban Community, a minimum of 120,000 new inhabitants. So it is every year more than 200,000 people in addition to the population, an increase of nearly 7%.

This significant demographic pressure leads to uncontrolled land use and therefore to the appearance of vast squatter settlements in mangrove areas (Fig. 3.16) and in swampy areas at risk. These spontaneous settlements not geometrized, the almost non-existent social and health infrastructure are precarious living conditions. Dense populations living there in promiscuity exposed to any health risks resulting from urbanization deficit and aggravating themselves the environmental degradation especially of mangrove ecosystems.

The development of these areas begins with the clearing of mangrove and continues with the blocking of channels that feed more or less saline substrate. The latter action is very harmful to these mangroves formation because it causes a dryness that causes the substrate in addition to the physiological drought, natural or climatic drought. This means of conquering new spaces has been used in many localities of the city of Douala including Mboussa Essengue, Youpwe, Bois-des-Singes, Bobongo and Bonaberi (Fig. 3.16a).

When the water supply channels of the substrate are created, herbaceous plants quickly disappear as soon as the water environmental conditions become unfavorable. The area is gradually replaced by species swamp forests (*Alchornea cordifolia*, *Anthocleista vogelii*) and later herbaceous plants of mainland settled, marking the disappearance of mangroves (Din 2001).

3.5.3 Effects of Poverty

Like many other countries, Cameroon has experienced in the 1990s, a severe economic crisis that has resulted in numerous job losses and income for the people. This has not been a favorable factor for the development of mangroves. Helping economic calculations, it was easy to understand that the free surfaces such as mangroves were predilection sites since the cost of the development was below the purchase or lease

Fig. 3.16 Population pressures on Bonaberi and Douala mangroves with the construction of houses (**a**) and roads (**b**)

of a yet more appropriate field. Many people who hoped to find the Eldorado in the economic capital have in the rural exodus invests Douala. These people have acquired housing inside mangrove areas and live in dire poverty (Fig. 3.17a).

One consequence of the great crisis in Cameroon in the 1990s, is an increase in the solicitation of the mangroves of the Wouri river estuary. Thus, people who had lost their jobs have returned to the mangrove ecosystem that showed no stress and no protection. These people are converted in the extraction of sand, cutting and sell-

Fig. 3.17 Population settlements in dangerous health conditions (**a**) house constructed in swamp area; (**b**) collection of poles (*Rhizophora* and *Nypa fruticans*) for house building

ing mangroves firewood and timber, making charcoal, cutting poles for house construction (Fig. 3.17b). They have mostly invested their savings to optimize their new business with the acquisition of cutting equipment and a highly efficient logistics including saws motor and large canoes with outboard motors.

Health services and sanitation of the Douala City Council are not yet involved in the maintenance of populated areas near the mangrove. In reality, these areas are not included in the development plans and their populations, despite the fact that they pay taxes, are still considered marginal. All domestic waste is directed to mangroves and organic pollution by domestic waste is highly noticeable. In addition to household waste, there is also considerable amounts of fecal waste.

3.5.4 Effects of Land Uses

3.5.4.1 Effects on Mangrove Forest

Several anthropogenic factors (logging, anarchic urbanization, agriculture, industrial pollution, sand extraction) are responsible for a decrease in the mangrove cover. Despite mangrove forests in Cameroon being legally protected since 1996 (Frame-law n°96/012 relative to management of environment in Cameroon), there was an effective loss of mangrove forests in the peri-urban settings of Douala since four decades and possibly persisting beyond. Nfotabong-Atheull et al. (2013) have documented the effect of anthropogenic land use in the reduction of mangrove cover around the Wouri estuary both qualitatively and quantitatively. The overall accuracy of the land use/cover map of 2009 was above 90% with an overall Kappa index of 0.87.

A trend of decrease in mangrove area and increase in human settlements was observed between 1974 and 2009. In 1974, mangrove forests covered 51.89% (3.01 km^2), where as in 2003 it was 31.20% (1.81 km^2), indicating a decrease of 39.86% over the 29-year period (Table 3.2). In the 2009 photograph, mangroves represented 24.29% (1.41 km^2), a decrease of 22.10% to that of 2003. Between 2003 and 2009, the coverage of less disturbed mangrove forests decreased from 0.80 to 0.29 km^2 representing a decrease of 63.75% over the 6-year period.

In contrast, the rate of change in large disturbed mangrove extend after 2003 ranged from 55.80 to 79.43%, amounting to an increase of 10.89% between 2003 and 2009. In 1974, settlements made up 18.11% (1.05 km^2), whereas in 2009 they extended to 28.94% (1.68 km^2) with an increase of 60% over the 35-year period. However, the measured rate of mangrove decrease is twice as high as that estimated for tidal forest degradation at Mngazana estuary, Eastern Cape, South Africa (Rajkaran and Adams 2010), which amounted to ~21% (~32 ha) of mangrove loss since 1982 or to a decline of about 1 ha/year. Although none of the mangrove species present in Cameroon have been listed as threatened (Polidoro et al. 2010), populations are at risk from habitat loss (Table 3.4).

Since 2009, the introduced mangrove palm *Nypa fruticans* Thurnb. Wurmb. was found growing in large open areas and along the muddy river front. This species was also found in the field under the canopy of less degraded mangrove forests, places where it had not been possible to accurately delineate its distribution on the airborne imagery. The increase in areas occupied by water, road, agricultural land, sand quarry, wood market, and other vegetation types (mainly composed of *Annona glabra* L., *Drepanocarpus lunatus* GWF Meyer, *Paspalum vaginatum* Sw., *Anthocleista vogelii* Planch., *Sesuvium portulacastrum* L., and *Alchornea cordifolia* Müll. Arg.).

3.5.4.2 Effects on Mangrove Fauna

Though such decrease in mangrove faunal composition was recorded in the Littoral region, many interviewees linked it to the anthropogenic disturbances prevailing. Local people unanimously shared the opinion that trawler activities along the shore

Table 3.4 Land use/cover changes derived from 1974, 2003, and 2009 aerial photographs of Douala (Cameroon) peri-urban setting

	Area at 1974		Area at 2003		Area at 2009		Change 1974–2003	Change 2003–2009	Total change 1974–2009
Land use/cover	(km^2)	(%)	(km^2)	(%)	(km^2)	(%)	(%)	(%)	(%)
Mangroves	3.01	51.89	1.81	31.20	1.41	24.29	−39.86	−22.1	−53.16
Settlements	1.05	18.11	1.57	27.06	1.68	28.94	49.52	7	60
Agriculture	–	–	0.25	4.31	0.29	4.99	–	16	–
Other vegetation types	0.15	2.59	0.34	5.86	0.43	7.41	126.66	29.41	193.33
Water	0.17	2.93	0.33	5.69	0.43	7.41	94.11	30.3	152.94
Road	0.06	1.03	0.15	2.58	0.2	3.44	150	33.33	233.33
Sand quarry	–	–	0.01	0.17	0.01	0.17	–	0	–
Youpwe wood market	–	–	0.002	0.03	0.006	0.10	–	66.67	–
Airport	1.36	23	1.34	23	1.35	23	−1.47	0	0

Nfotabong-Atheull et al. (2013)

resulted in a decline in local fish catch inside the mangroves. This negative change was reinforced by the increased number of local fishermen. A decrease in the population of silurid, tilapia, mullets, shrimp, oyster and crab species, was probably related to changes (siltation, reduced water circulation, etc.) that occurred in the waterways.

Crab assemblage has shown a considerable variation in terms of species richness and abundance according to deforestation (Ngo-Massou et al. 2016). Abundances of *Chiromantes buettikoferi* De Man, 1883 (from 4.5 to 2.6% of individuals), *Metagrapsus curvatus* Herklots, 1951 (from 12 to 7.68% of indiviuals), *Sesarma* sp. (from 4.3 to 2.7% of individuals), *C. angolense* Brito Capello, 1864 (from 10.73 to 3.30% of specimens) and *Perisesarma alberti* Herklots, 1951 (from 16.61 to 6.42% of specimens) decreased. The number of species in Sesarmid crabs (from 10 to 8 species) and Gecarcinid crabs (from 3 to 1 species) fallen down. The importance of vegetation and channel distance on crab distribution has been described (Ngo-Massou et al. 2014). Vegetation is highly dominated by *Rhizophora* plant and that zone is the most diversified (18 crab species) which, two crab *Helice* sp. and *Maja squinado* Herbst, 1788 were strictly pledged. Therefore, the overexploitation of *Rhizophora* spp. lead the loss of these two indigenous crabs. Furthermore, *Cardisoma* spp. and *Uca tangeri* Eydoux, 1835 species are the most terrestrial crabs, because they live essentially close to back mangroves. With regards to anthropogenic activities, these terrestrial crabs are mainly threatened more than *Callinectes pallidus* Rochebrune, 1883 and *Portunus validus* Herklots, 1951, crabs found only near channels (marine crabs).

3.6 Conclusion

In spite of the existing laws and regulations in order to protect mangrove ecosystems in Cameroon, the rate of deforestation is significantly high in Douala peri-urban mangroves. The depletion issue from natural disaster must be considered as negligible in compare with anthropogenic activities. The fact that state projects toward mangrove areas did not especially show example of the importance of biodiversity protection, neighboring population profited from the authorities' laxity to accelerate the depletion mainly by building up to the channels and tributaries.

Considering in addition the demographic pressure in the city, always coupled with human poverty and associated here with a high level of incivism, the mangrove ecosystem will disappear in less than two decades if the authorities continued to mismanage the population settlement in coastal peri-urban areas.

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Chapter 4
Impacts of Coastal Land Use Changes on Mangrove Wetlands at Sungai Mangsalut Basin in Brunei Darussalam

Shafi Noor Islam and Umar Abdul Aziz Bin Yahya

Abstract The chapter evaluates the rapid changes of coastal landscape and expansion and asymmetrical land use or cover changes in Sungai Mangsalut basin area in the coastal region in Brunei Darussalam. It is revealed that the coastal land use changing pattern of Sungai Mangsalut basin area is rapidly expanding due to natural environmental calamities, increasing population pressure and economic development and consequently led to the conversion of water bodies, cultivated lands, natural forest vegetation or open spaces in buildup areas. The exploitation of settlement and urbanization size through inhabiting the river mouth and low lying areas and clearing of coastal mangrove and coastal salt marches of vegetation without a corresponding expansion to infrastructures resulted in a wide range of environmental issues and risks such as severe pollutions, significant decrease of water bodies along with greenery and uncontrollable growth of settlement. In other words, these issues existed because the authority is unable to cope with the swift changing situations due to their internal resource constrains and management limitation. Hence, for that reason, this paper also highlights the consequences of the coastal wetland and basin landscapes changes in Sungai Mangsalut basin area in the coastal region in Brunei. The objective of this chapter is to analyze the coastal environmental impacts of the area surrounding Sungai Mangsalut (Mangsalut River). Also to identify the factors influence the degradation of the Sungai Mangsalut in the present time.

Keywords Sungai Mangsalut • Degradation • Coastal land use • Environmental impacts and ecosystem

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Fig. 4.1 The map in 1978 shows the river is surrounded with swamps forest man- grove forest (Modified from Google Earth, 2015)

4.1 Introduction

Brunei Darussalam is one of the country that have high proportion of primary forest and wetland among all countries around the world. Brunei Darussalam is well known to protect their forest and rivers and continue to maintain it high biodiversity and landscapes (Engbers 2010; Rhett 2013). A fluctuating sea level, and the geomorphologic evolution that it induces, generates changes of the ecological state of coastal wetlands (Sandal 1996). These changes modulate the transgression upslope from upland forest to sub tidal ecosystems in different part of the critical geomorphic regions as well as in the Sungai Mangsalut basin area in Brunei (Wolanski et al. 2009). Tropical and sub-tropical coastal wetlands represent particularly productive ecosystems that contribute to both terrestrial and marine biodiversity and landscape. In this chapter, the Sungai Mangsalut basin area laduse and the coastal wetlands that will be considered are associated with mangrove and salt marshes shorelines (Wooddroffe and Davies 2009). Mangrove characterize the upper intertidal zone on many low-energy tropical coasts, often with salt marsh and associated wetlands that can form landward of such halophytic vegetation which is displaying the similar scenario in Sungai Mangsalut basin area in the coastal region in Brunei (Fig. 4.1) (Rovertson and Alongi 1992; Wooddroffe and Davies 2009).

Sungai Mangsalut was best known in the past as "the river producing fortune" or in Malay *"Sungai yang mengalur keuntungan"* for the local people (Rehett 2013; Engbers 2010). In the past, the river was a famous area for fishing ground, hunting and leisure, this is because the river mouth sited on the coastal area (Rehett 2013; Engbers 2010). The river was seen as a primary source (economic) of living for the local people. The river is surrounded by swamp forest and mangrove forest in 1978 (Rehett 2013; Engbers 2010). However, this report will explain further on how the degradation of the wetland features surround the Sungai Mangsalut (Mangsalut River) in the present times and to show the comparison of the physical environment in 1978 with 2015 map (Rehett 2013; Engbers 2010).

4.2 Aims and Objectives

The aims and objectives of this study is to analyze the environmental impacts of the area surrounds Sungai Mangsalut. Also to identify the factors influence the degradation of the Sungai Mangsalut in the present time.

The specific objectives of this study is given bellow:

- To clarify the location of Sungai Mangsalut (*Mangsalut River*) and its present environment status.
- To recognize the factors that influences the changes of the coastal wetland landscape ecosystem and environment in 2015 and in 1978.
- To identify the direct and indirect ecological impacts of the Sungai Mangsalut in the coastal region in Brunei Darussalam.
- Based on the study findings prepare some practical recommendations for the better management and development of the case study area in the near future.

The aims and objectives above is seen as a guideline towards the study of the Sungai Mangsalut. It can be seen as the starting point for the study and fundamental in keeping the study clear and relevant.

4.3 Literature Review

Wetland that was stated by Mitsch and Gosse link (1986) they give a definition stated wetland as "areas of marsh, fen, peat land or water whether natural or artificial, permanent/ temporary with water that is stable or flowing, fresh, brackish or salt including areas of marine water the depth of which at low tide does not exceed 6 meters". They also added that wetlands have a great value because their functions have proved to be useful to humans. The value for some wetlands also increases with human development (agricultural and urban) because of increased use and/or increased scarcity (Mitsch and Gosselink, 1986; Islam et al. 2014). This can easily be overwhelmed in areas of heavy human development, thus lessoning those values.

Wetlands are multiple-value systems; they do not just do on a specific system. They perform many processes simultaneously and therefore they provide a suite of values to humans, which could also lead to culture and religious values towards the wetlands itself. The Sundarbans mangrove forest, one of the largest forests in the world that is formed at the delta of the Ganges, Brahmaputra and Meghna Rivers on the Bay of Bengal. The culture value of the area provides a livelihood for the people who live near the area, working variously as wood-cutters, fishermen, and gatherers of honey, golpatta leaves *(Nipa fruticans)* and grass (Christensen 1984).

In addition, the Bangladesh people also hold a religious value on the Ganges River. According to Muller and Blij (2010), they stated that the people view the Ganges River as Hinduism's sacred river, where they believe its ceaseless flow and spiritual healing power are Earthly manifestation of the Almighty. Therefore, wetland is part of human heritage and religious belief and also link to economic development as some society depends on the wetland area. Moreover wetland contains very rich components of biodiversity local to national.

Another good example is Mangrove areas in Peninsular Thailand that have been degraded through over-exploitation for the charcoal industry, fuel wood harvesting by local people (Plathong and Plathong 2010). In their study of Past and Present Threats on Mangrove Ecosystem in Peninsular Thailand by Sakanan Plathong and Jintana Plathong (2010), they said that the lacking knowledge on mangrove ecology and its importance, ignoring attention-grabbing of stakeholders according to the first threat, lacking of public participation in development projects, weakness of law enforcement and fishery pressure are the causes factors that responsible for the degrading of the Mangrove areas in Thailand. This is due the selfishness of mankind as they were blinded by the economic boost. It is reported that half of wetlands have disappeared since 1900. Development and conversion continue to pose major threat to wetlands. The conversion of wetlands for commercial developments, drainage schemes, extraction of minerals and peat, overfishing, tourism, pesticide discharge from intensive agriculture toxic pollutants from industrial waste and construction of dams and dikes. Globally more than 6% (6.0–8.6 million km^2) of land is made up of wetlands (Islam et al. 2014).

In addition, the main culprit in the destruction of mangroves is man. To achieve harmful supremacy over nature, human have destroyed this magnificent ecosystem almost irreparably. Land reclamations and industrial developments are the major causes of mangroves degradation. Systematic dumping of all kinds of waste and debris in the mangrove areas destroys them. Land reclamations and industrial developments are the major cause of mangroves degradation (Islam et al 2014; Islam and Gnauck 2008). This waste/debris creates a barrier preventing the seawater from entering the mangroves and eventually kills the mangroves. In many instances, this is done intentionally to reclaim land for construction activity.

4.4 Characteristics of Sungai Mangsalut (Mangsalut River)

Sungai Mangsalut is located in the Brunei Area of the Country of Brunei. The Stream is located at the latitude and longitude coordinates of 4.916667 and 114.933333. Fishing enthusiasts interested in fishing near or at Sungai Mangsalut. This could be one of the best fishing or outdoors adventure locations in the regions of Africa/Middle East. The fishing activities are also motivating and encouraging to start commercial and entrepreneurship activities to the near coastal region in Sungai Mangsalut Rivers. The young entrepreneur fishing and outdoor adventures are enhanced by our social fishing website, so we ask that you share your experiences with other fishing enthusiasts both young and old. Fishermen and Fisherwomen should also submit a comment or report on Sungai Mangsalut to help out their fellow anglers. The fishing site also provides users the ability to easily track fishing catches and experiences with exact GPS location on the body of water, create fishing groups to communication with your fishing buddies, find tackle shops fishing schools and marines and port located near your fishing sites and last be not least enter fishing contests to win fishing equipment and supplies.

The Mangsalut River is an estuary. Estuary is an area where freshwater river or stream meets the ocean. The type of estuary is a coastal plain estuary, in the study of classification of estuaries done by Oberrecht (ed.), it was formed at the end of the last ice age. As ice melted and water warmed up, sea level rose. The rising sea level swamped low-lying coastal river valleys. These valleys usually have a shallow depth with gentle sloping bottoms. Their depth increases towards the river's mouth (Fig. 4.2).

The Sungai Mangsalut is interconnected to seven rivers in Brunei:

1. Sungai Antong (Antong River)
2. Sungai in Kampong Lambak (Kampong Lambak River)
3. Sungai Salambigar (Salambigar River)
4. Sungai Orok (Orok River)
5. Sungai Hanching (Hanching River) (Fig. 4.2).
6. Sungai Tilong (Tilong River) and
7. Sungai in Kampong Tanah Jambu (Kampong Tanah Jambu River)

From Fig. 4.2, it shows the seven river's is interconnected with the river mouth of the Sungai Mangsalut. In 1978, it shows that there's a huge catchment area of swamp and forest swamp. Mangrove trees surround the Mangsalut River; prove by referring to the map in Figs. 4.3a and 4.3b.

According to Stewart (1986a, b), Mangrove forest or best known as "Bakau" in Malay dialect, best developed on extensive mud-flats or tidal mud-flats. The area itself is surrounded by hilly area, which could explain the process of water run-off and infiltration resulting the swampy area at the area (Stewart, 1986b).

The soil type around the area is peat swamp soil. This peat forms in wetland or water logged conditions, where flooding obstructs flow of oxygen from the atmosphere, slowing rates of decomposition. The lower peat part of the soil is brackish

Fig. 4.2 The above portion shows the location of the river that is interconnected and the lower portion shows the extension of river catchment (Modified from Google Earth, 2015)

while the upper layer is strongly acidic. The features of the peat is soft and easily compressed, leading to the under pressure water in the peat is forced out. Peatlands are content of toxic elements and low availability of plant nutrients adapted to the extreme conditions of high water and low oxygen.

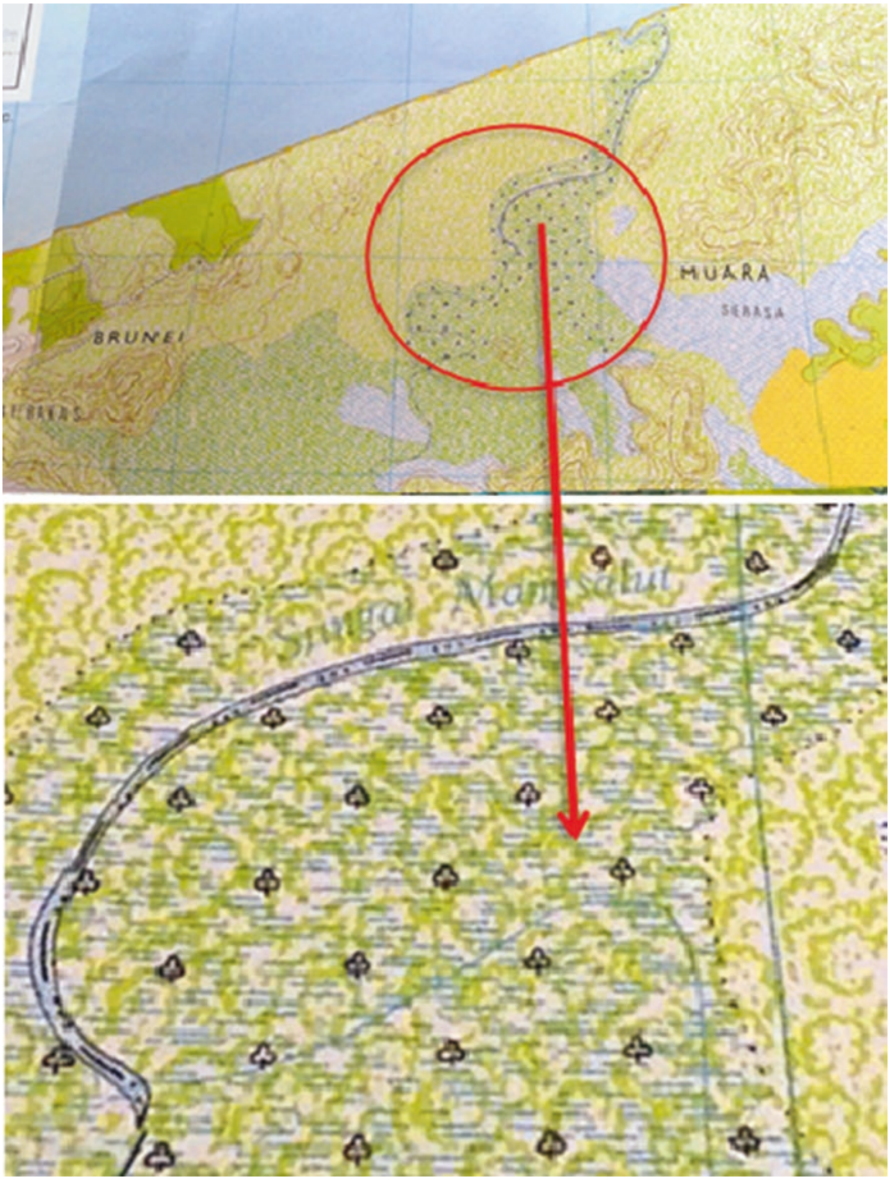

Fig. 4.3a Shows the legend and map of Sungai Mangsalut in 1978 and 2015 (Modified from Google Earth 2015) (Close up of the Sungai Mangsalut Mangrove Swamp in 1978)

By referring to the map in Figs. 4.3a and 4.3b, the river was meandering towards the river mouth. The river mouth of Mangsalut River is seated at the coastal area of Brunei Muara district. The coastal area is also the beaches and estuaries linking with each other on the landform. By using Google Earth, the identification of deposition

Fig. 4.3b Shows the legend and map of Sungai Mangsalut in 1978 and 2015 (Modified from Google Earth 2015) (Close up of the existing Sungai Mangsalut Mangrove Swamp in 2015)

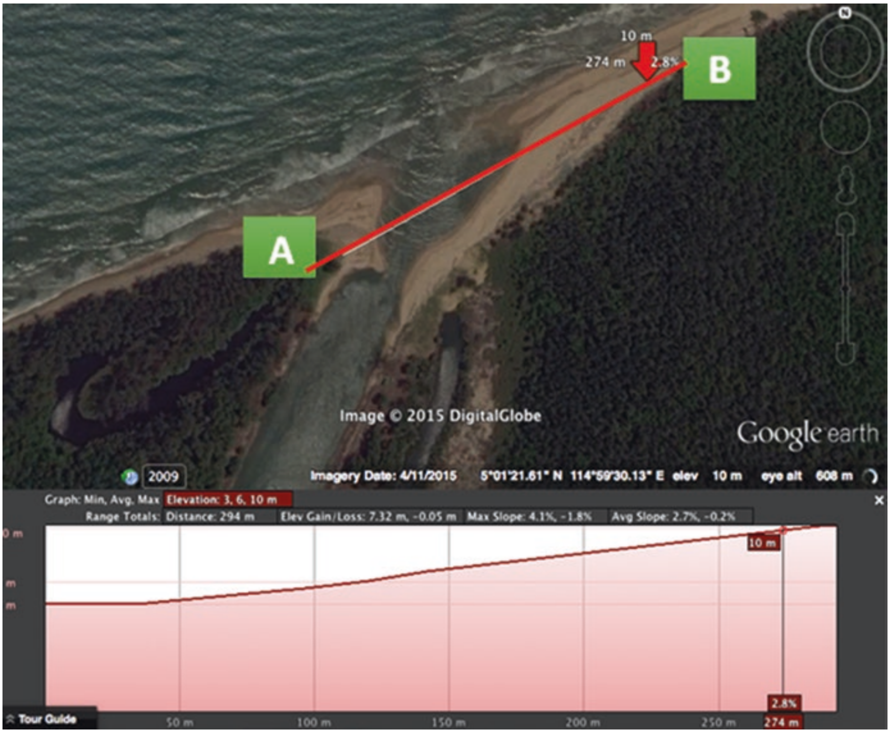

Fig. 4.4 Shows the elevation profile of the Mangsalut River (Modified from Google Earth, 2015)

process occurs on the right side of the river estuary forming beaches that is steeper than the left side. This is proven with the elevation profile using the Google Earth by referring to Fig. 4.4. By drawing the line across the river mouth with the help of the Google Earth elevation features, the result of the elevation is shown in Fig. 4.4. The deposition process is due to the constructive wave regime that has stronger swash than backwash. B.

The deposition is a process in which rocks, soil and sediments are transported and added to a certain location to form a landmass. The process takes place when the river becomes flooded after a heavy rain. Debris is deposited along the riverbank as water spreads. From Fig. 4.4, the profile shows the deposition area at the beach is stated at 10 m of height at A and 3-m height at location B.

4.5 Coastal Wetland Condition

Coastal wetland includes saltwater and freshwater wetland is located within coastal watersheds. Wetland types found in coastal watersheds include salt marshes, fresh marshes and mangrove swamps. According to the webpage of U.S. Environmental Protection Agency (2008), the Non-Tidal freshwater coastal wetlands (coming from the in land) and the Tidal saltwater coastal wetland coming from the sea connected at the "Head of Tide" where it is the limit for the tidal influence. These exchange of water Tidal and Non-Tidal water only occur within the watershed boundary (Fig. 4.5).

For Sungai Mangsalut, the watershed boundary is shown in Fig. 4.6. It is within the river catchment area that is interconnected with the Mangsalut River. Water comes

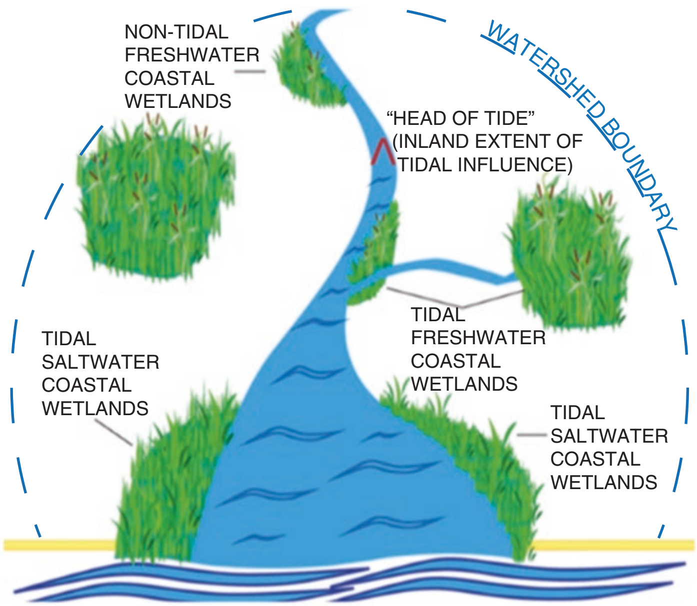

Fig. 4.5 Above shows the watershed boundary and "Head of Tide" (Photo taken from United States Environmental Protection Agency)

Fig. 4.6 The *yellow* boundary represent the water catchment and the *red* symbol represent the "Head of Tide" (Photo taken from Google Earth)

from inland and flow towards the Mangsalut River, however the mouth of the river mangsalut also receiving the tidal saltwater coming from the sea. These water will meet up at the "Head of Tide". Figure 4.6, will show the location of the "Head of Tide". According to Bird (1984a, b) an estuary may be defined in terms of tidal conditions, as the lower reaches of a river subject to tidal fluctuations or in terms of salinity where fresh water meets and mixes with salt water from the sea. Study done by Lines and Bolwell (1983), currents from inlands are also important carrying agents particularly in estuaries where tidal currents may clear away deposits, depositing them further along the coast or out to the sea. The "head of tide" where the freshwater and saltwater meets is known as the brackish water (Fig. 4.6). The Fig. 4.6 shows the Sungai Mangsalut river mouth where the fresh water and saline water is mixing. The Sungai Mangsalut River's catchment area is potential for mangrove wetland ecosystem, as the seven rivers are connected and carrying fresh water to the mouth and estuaries.

This brackish water is suitable for the growing of mangrove trees and the ecosystem will be maintain as long as there is no interference from other factors such as human factors.

At the mouth of Mangsalut river is the coastal area that not stable because it kept on changing the through time. Studied done by Davis and FitzGerald (2004) on Coastal Landforms, he states that the coast is driven by the force of waves, currents, tides and climate along with the process of deposition and erosion that plays huge part of the coastal landforms. This process can also relate with the Mangsalut River. By referring to the Fig. 4.7, shows the evidence of the coastal landform being a dynamic process as it changes through the years. In 2009, the sand stretches towards the sea, however, in 2011 the send is receding backwards due to the process of erosion by the waves (Fig. 4.7).

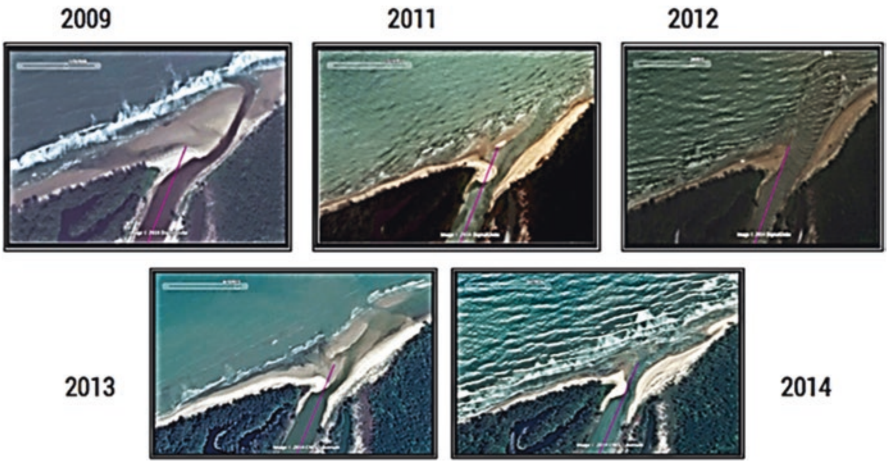

Fig. 4.7 Coastal dynamic that changes through the years (Photo taken from Google Earth)

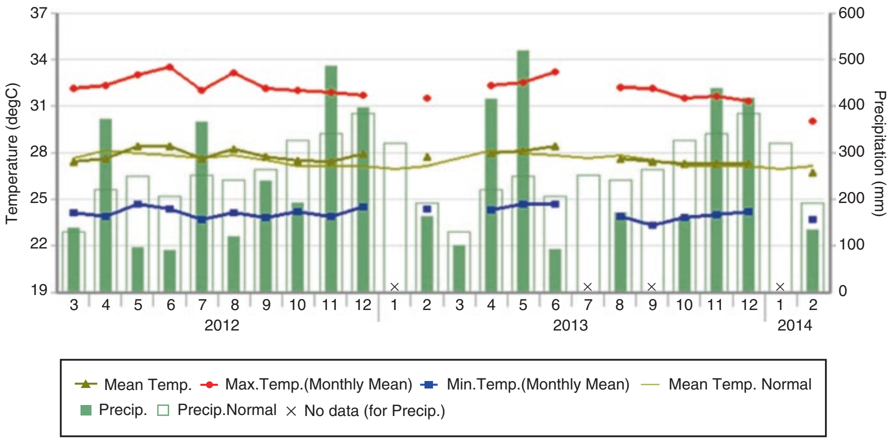

Fig. 4.8 Brunei weather graph, which was taken from the Japan Meteorological Agency

In 2012 the river from the inland over powered the tidal waves from the sea as the river cuts through and enlarge the width size of the river. This is probably due to the heavy rainfall inland as Brunei is the country experiencing tropical climate. These cause the high density/velocity of river flow capable to carry deposition of smaller material. Furthermore, in 2013, the sand stretches back towards the sea stabilizing the coastal area similar to the coastal formation in 2009.

To solidify the statement regarding the high intensity of changes of the coastal landform, the graph in Fig. 4.8 will explained more in depth of the months where Brunei experienced high precipitation in the Brunei Weather Graph retrieved from the Japan Meteorological Agency record.

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By referring to the graph in Fig. 4.8, in 2012 shows an abnormal received of precipitation compare to the normal precipitation. The highest record data was in November 2012, where Brunei received the highest precipitation in that year. However, in 2013 the highest precipitation recorded was in May, but in that year Brunei experience a dry month in January, July and September. Furthermore, the coastal landform at the Sungai Mangsalut River is proved to be a coastal dynamic process. The change in coastal landform will constantly change throughout the year due to the rainfall received. Beaches are sedimentary bodies situated at the land–sea interface and usually made up of non-cohesive particles of mainly sand size, although coarser sediments (such as shingles) are dominant in some specific settings (Bird 1996).

Bird's statement can explained the reason to the constant changing of coastal landform. This statement can link to studies done by Davis and FitzGerald that the coastal landform is driven by the process of waves, currents, tides and climate along with the process of deposition and erosion.

4.5.1 Features of the Coastal Area at Sungai Mangsalut (Mangsalut River)

This section will show the snapshot taken during the fieldtrip to Sungai Mangsalut in 2015. All of the pictures on this section will only related to the coastal environmental scenarios and geomophological features on the changing scenarios of the Mangsalut River basin area (Figs. 4.9 and 4.10 and Fig 4.18) The below geomorphic features in the Mangsalut River catchment areas are displaying the changing pattern in the basin area as well as in the whole catchment area.

4.6 The Past and Present of "Sungai Mangsalut Catchment Area"

In this section will show the comparison of the Past and Present features. It is aim to show the changes of the physical environment for Mangsalut river. The methods used to identify the physical environment is by using map (from 1978 maps of Brunei), Google Earth, interview with the local people and also by self-trip to the Mangsalut river. According to Stewart (1986a, b), the post-war (the end of Second World War in 1945) there is an expansion of the economy in Brunei Darussalam that has led to high standard of living, better education and improved standard hygiene and nutrition. Numerous primary and secondary schools have been built (Fig. 4.11).

Fig. 4.9 This is the buried forest found at the coastal area around Sungai Mangsalut (Photo taken by authors in 2016)

Fig. 4.10 The present label above is the migrating bars at the Mangsalut River (Photo by the authors in 2016)

After the of Second World War in 1945, Brunei growth of population boost triple times more rather than the usual (Fig. 4.12). This is proved by the table shown above, in 1960 about 100,000 population in Brunei. This is during the era where Brunei Darussalam economic sources is the Oil and Gas Industry and during this era where the building of modern hospitals and clinics has resulted in better medical care. Many of the diseases responsible for a high infant death rate have been brought under control for example cholera, typhoid and tuberculosis (Stewart 1986a).

Lily Kong et al. (2000) regarding the study of unity and diversity in South East Asia stated that the population of Brunei is mostly along the coast. A high rate of population growth caused by low infant mortality and death rates, high nutrition

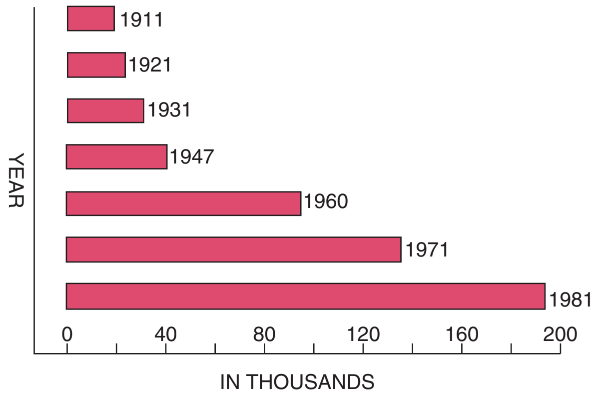

Fig. 4.11 The table above shows the number of population in Brunei (Modified from Stewart 1986)

Fig. 4.12 Shows the nearby settlement located 2.61 km away from The Mangsalut River (Modified from Google Earth, 2015)

levels and high life expectancy means that population pressure on the fragile coastal ecosystem is increasing (Japan Meteorological Agency 2002). For example, the urban construction industry has been responsible for the destruction of vegetation, soil erosion and siltation of streams (Davis and FitzGerald 2004). The increase in wealth has in turn prompted the development of construction industry. Brunei's environmental problems are more result of affluence than of poverty (Coles et al. 2011).

4.6.1 Results of Field Investigation

For the Mangsalut River, there is no data that resemble the past activities in the past or recorded in books. The only knowledgeable and reliable data on Mangsalut River is the experience of the local people in the past. By doing interview, this could get a more valid data on the study area and with this qualitative method; we could get the aesthetic values on the Mangsalut River. The interview with one of the local people Dato Paduka Haji Ahmad bin Kadi, he stated that the Mangsalut River was known for the fishing activities and also for hunting. He added, during his youth age that was the famous place for the youth his age to go for fishing as they sees it as a source of entertainment, economic sources and the male was seen holding a responsibility of a breadwinner in the family by providing everyday needs such as foods and clothing. The lifestyle for the Bruneian people back then in the 1970s, they still practicing their traditional lifestyle despite Brunei in the era where oil and gas was their main source.

In 1970s the development was visible in Brunei, however the society in 1970 is still practicing their traditional ways of living where their main source for consumption was agriculture, fishing and hunting. The interviewee also added and emphasize that the local people still holds their culture and traditional belief very dearly until now. He quoted*, "Despite the modern lifestyle, Bruneian still treasure the culture and even **The Sultan** (The King for Brunei Darussalam) treasure the Royal Customs that they carry from the past until now"*. This shows how *The Sultan* emphasize in treasuring the unique Malay culture of Brunei to the world and by being a role model to its country and the local people.

4.6.2 Past Scenario Since 1978

By referring to Fig. 4.13, it is proved that during the year 1978 the Mangsalut River was surrounded by the huge catchment of Mangrove swamps. According to Bird (1984a) where mud is being deposited mangroves spread on to it and create a sheltered environment enabling other salt marsh plants to colonise when the mud surface has built up to suitable level. He also added that the mangrove encroachment is impeded by strong wave action or tidal scour, so mangrove swamps is most extensive in sheltered areas. The Mangsalut River itself was surrounded by hilly area, which could lead to the process of water run-off and infiltration, resulting in the swampy area. Moreover the area itself was untouched by the development or construction activities and the river was naturally flowing in meandering formation to the river mouth. However, about 2.61 km there is a nearby settlement, which is located at Kg Salambigar away from the main Mangsalut River. Only few houses back then and little human development and infrastructure development such as schools and industries (Bird 1984b; Bird 1996).

Fig. 4.13 Above is the compilation of the snapshot taken during the fieldtrip to Mangsalut River (Photo taken by the authors in 2016)

During 1978, The Mangsalut River was not heavily touched by development despite the growing of economic in Brunei. Local people back then still practiced their traditional culture, as they believe it is their source of living and support for their family. In addition, there is no picture or data recorded that tells about the activity of the people in that area. However, the only information that is reliable is the stories of the local people that have experienced in the area (Bird 1984a, b; Bird 1996).

The Mangsalut River Features in 1978:

1. The river surrounded by the mangrove swamps
2. The river of Mangsalut is meandering characteristics
3. There is little development or construction activity during the era.

4.6.3 The Present Scenario in 2015–2017

In the Present time, due to population expanding around 300,000 the demand for the people is increasing. Housing scheme were given to the local people, highways built for easy accessible and construction activities surrounded the Mangsalut River. During the self-trip to the Mangsalut River, there is no sighting of mangrove swamp around

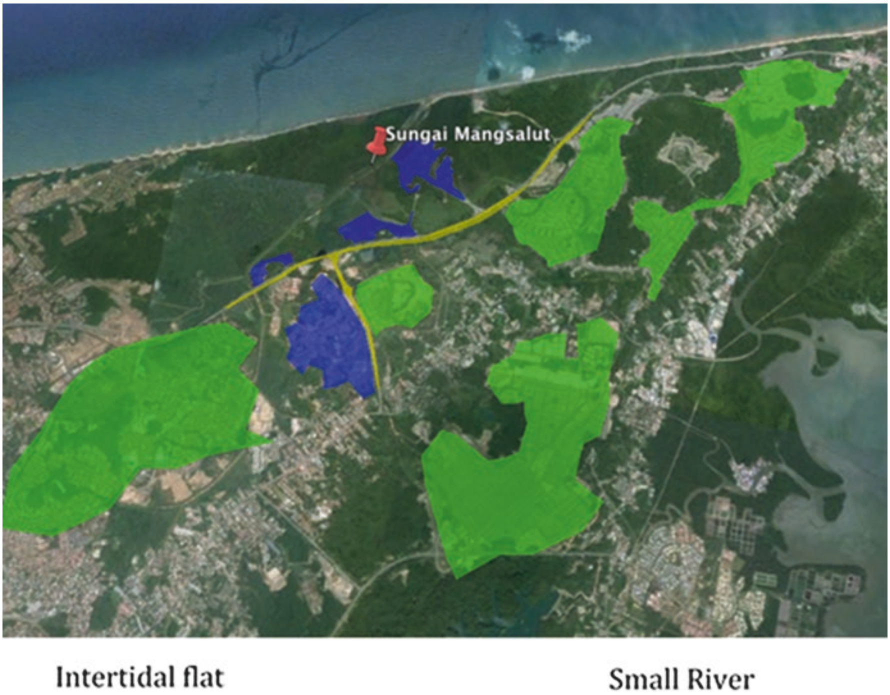

Fig. 4.14 This is the plotting of urban development surrounds the Mangsalut River (Modified from Google Earth, 2015)

the area itself. The only feature present was the Nipa Trees (*Nipa frutican*), Swamp, Peat Swamp Forest, Intertidal flat and Small River (Christensen 1984) (Fig. 4.13).

By referring to Fig. 4.13, there is no sighting of the Mangrove Forest surrounds the Mangsalut Area. Only Nipa Trees that conquer the area of Mangsalut River and some patches of swampy area. This is because by the development that is occurring surrounds the area of the Mangsalut River (Silvestre et al. 1992). The development is overbearing in the present time compare with the era in 1978. The land clearance for major highway, housing scheme and buildings followed by heavy construction activities that is sited east side beside the Mangsalut River (Roger M and John EE 2009).

Figure 4.14 explain the development occurrence around the Mangsalut River. The blue plotting indicates the heavy construction activity and land clearance near the Mangsalut River. The yellow plotting indicates the main/major highway that link Brunei-Muara to Tutong and Belait. Finally the green plotting indicates the area of housing scheme for the local people. By referring to Fig. 4.14, shows a clear image of urban developments that conquer the Mangsalut River and with these developments the changes in the Mangsalut River is visible (Fig. 4.14).

In developing and development process, there will be extreme changes in the environment and changes in the ecosystem. For example, due to urban development the amount of water that the mangrove receive will be more as the flowing of water will

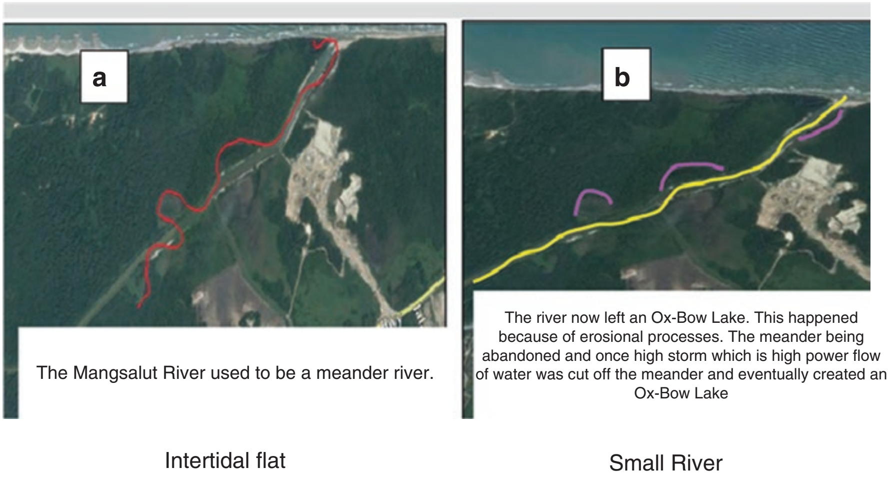

Fig. 4.15 Shows the shape of the Mangsalut River in the past (**a**) and the present shape of the Mangsalut River (**b**) (Modified from Google Earth, 2015)

flow faster because of the concrete river bed (Robertson and Alongi 1992; Islam et al. 2014). Mangrove swamps survive with saline water (Stewart 1986a, b). If the amount of water is domineering at the mangrove area, the saline water will receive more freshwater resulting the destruction of the mangrove area (Robertson and Alongi 1992).

As this process goes on, there will be lesser mangrove forest. Part of the Mangrove forest characteristics is to infiltrate water and also help to slow down the impact of incoming water. However, with the degrading of mangrove forest through time, the flow of water will be much faster and the meandering river will be cut through (Islam et al. 2014). The meandering cut through could also cause by natural causes such as heavy rainfall that result in high storm. High storm can trigger fast flowing water and rough river channel that could affect the river environment (US EPA 2008; Wolanski et al. 2009; Woodroffe and Davies 2009).

In Fig. 4.15 shows the comparison of the shape of the river. Figure 4.15a, shows the original shape of the river in 1978. The red line in Fig. 4.15a represent the meandering river. Moreover, by comparing with Fig. 4.15b, which is taken in the present time, shows the meandering river is cutting through the river through time. This is shown with the yellow line that represent in Fig. 4.15b.

The purple line in Fig. 4.15b represent the Ox-Bow lake features in the present time. According to Leong (1998), the Ox-Bow lake feature only occur on the lower course where broad plain heading to the sea. It's the most predominant function of deposition process occur on the lower course of the river according to Leong. In Fig. 4.2 on page 6, shows the listed rivers that are connected to the Mangsalut River. These rivers are important source for the Mangsalut River. However, due to the urban development occur at near the river and the shaping of the river channel, causing drastic changes in the Mangsalut River (Thia-Eng 1987; Silvestre et al. 1992; Sandal 1996).

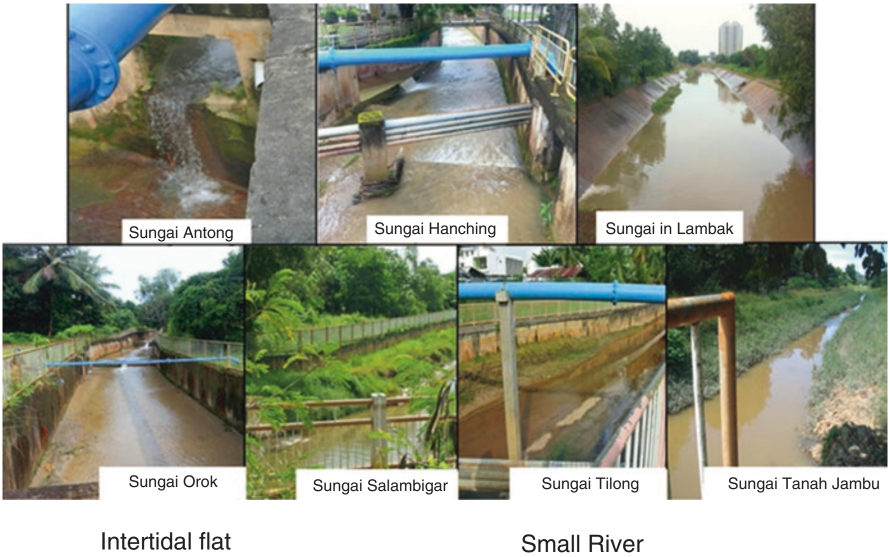

Fig. 4.16 Shows the rivers that are urbanized and are connected with the Mangsalut River (Photo taken by authors, 2015)

By referring to Fig. 4.16, all the seven rivers that were connected to Mangsalut River was being developed by men. Almost all of the rivers change from river to huge sewer system as shown at Fig. 4.16. The riverbed was also replaced to concrete bed that would not allow water to infiltrate through the riverbed. During heavy rainfall, this urbanized river will experience water flowing much faster comparing the natural river. Along the way, the water will not be able to carry sediments or nutrients that could satisfy the survival of Mangsalut River. This can be compared with the study done by Coles et al. (2011) regarding the Metropolitan areas in the United States where the development hinder the stream ecosystem. This is to provide the local people with employment, housing, medical services, education and recreation. Their study emphasize on the effect of urban development are particularly evident in the physical, chemical, and biological characteristics of local streams. The natural vegetation and native topsoil being replaced with impervious cover (rooftops and pavement, such as roads, parking lots and drive-ways) that block the infiltration of rain into the ground.

There is also problem with the urban development on the branch river that is connected to the Mangsalut River. In Sungai Lambak, the river was now being altered to a concrete riverbed and river channel. Beside the Lambak River, there is land reclamation occurred to make way for buildings and houses (Stewart 1986a, b; Robertson and Alongi 1992).

However, one of the building during the self-trip to the sight, we bumped to a house that was abandoned (Refer to Figs. 4.17 and 4.18). As we study the area, the house was clearly tilted and unsafe for people to make it as resident area. The house in Fig. 4.17 shows the evidence of a failed urban development. The

Fig. 4.17 Abandoned house due to the instability of land (Photo taken by authors)

Fig. 4.18 Sediments is accumulated on the concrete river and colonizing the manmade river (Photo taken by authors 2016)

land reclamation to make way for houses is not suitable to be built in this area. Brunei Darussalam consist of two weather which is dry and wet climate. As rainy season occur, this area will face an unstable land that could lead to the destruction of the house.

The Mangsalut River basin and surrounding features in 2015–2016:- the following features of the Sungai Mangsalut are displaying in time to time which is depend of the condition of the surface features and natural hazards and anthropogenic influences;

(a) No existence of mangrove swamps, altered river system at the moment;
(b) The tributaries and distributaries (Branches Rivers) that connect with Mangsalut River change into small sewer system in the Brunei Muara coastal region (Woodroffe 2009; Wolanski et al. 2009).
(c) Urban development such as houses, buildings and roads surround the river.
(d) The area is a sub urban which developed into one of the suburban in Muara district in Brunei Darussalam.
(e) The present features of the Sungai Mangsalut estuary and surrounding well known due to their major attractions land use which people may resides to the areas with the land uses or facilities available for them to have a better standard of living.
(f) Despite the continuous construction and development of the area, there were no complains or impact towards the people in the area in which the development are going smoothly.
(g) Construction activity of land clearance activities are the common features in the estuaries region as well as within the catchment area of Sungai Mangsalut.

4.7 Assessment of the Past and Present Scenario

The present scenario of the estuary of the Sungai Mangsalut (Mangsalut River) mouth and the basin area displaying the real scenario of the mouth area of the river (Wolanski et al. 2009). Table 4.1 displaying the comparative scenario on the estuary and basin areas.

The comparison between the past and the present Scenario of the Mangsalut river basin area in the coastal wetland region in Brunei coast. The table shows the scenario in 1978 and in 2015 (Al-Dousari 2012; Bird 1984a, b). As a whole the scenarios are very different within 38 years. Including the physical features, water flows and erosion pattern is also very changeable in the estuary area as well as in the catchment areas (Leong 1998; Lines and Bolwell 1983).

Table 4.1 Displays a comparative scenario on Sungai Mangsalut basin

1978 (past)	2015 (present)
The river was surrounded by huge catchment of mangrove swamps	The huge catchment of mangrove swamp disappears in 2015. Nipah palm and peat swamps
The river was surrounded by hilly area that leads to the process of water run-off and Infiltration. This resulted the occurrence of the swampy area	The hilly area is being level off/cut down for development purposes. This disturb the river ecosystem
The area itself was untouched by development/construction activities	The development and construction activities are overbearing to suit the demand of increasing in population size
The river was meandering near to the river mouth	The river is not meandering near the river mouth due to the development surrounds the river and the modification on the river itself
The mangrove trees survive with the right amount of saline water	The amount of water is domineering at the mangrove area as they receiver more freshwater

4.8 How Development Could Change the Environment

The area is highly impacted by anthropogenic factors, showing several factors such as rubbish, construction materials, wood, paint cans and other aspects of soil degradation such as erosion. Studies done by Longo et al. (2012) regarding the Indicators of Soil Degradation in Urban Factors: Physical and Chemical Parameters, they stated that the degradation of certain environment can promote significant changes in physical and chemical parameters of forest soil could hinder the process of restoring its ecosystem. Development plays a huge role in the changes of the environment. The process of Urbanization also link with a lot of process such as deforestation, land reclamation and land clearance (Longo 2012). Urban development also includes heavy machinery that could affect heavily on the environment (Meyer 1988). Rapid urbanization has altered hydrology. Storm (Heavy rain) will increase the surface runoff over the urbanized area due to less infiltration and percolation into the ground and less interception by vegetation. Hence, resulting in more flash flood (Meyer 1988; Mead 1988).

Rivers, over time become smaller and smaller due to vegetation colonization with sedimentation and erosion. Urban river systems are constantly being filled in. Heavy rain has caused higher buildup of sediments and vegetation colonizing mudflat (Muller and Blij 2010).

Moreover, there is a study done by Al-Dousari et al. (2012) regarding the Effects of Atmospheric Lead on Soil Microbiota in Kuwait, they said that the roadside soil is a major sink of heavy metals (e.g. lead, zinc, cadmium, chromium, nickel and copper) and hydrocarbons pollutants (e.g. gasoline, gas, oil and kerosene) in the environment such as roadside soils (Sandal 1996; Robertson and Alongi 1992; Silvestre 1992; Thia-Eng 1987). These factors/discharged that had a direct impact to the environment and cause harmful effects to human and all living organisms. This study can relate with the degradation of Mangsalut River, due to the construction of the major highway sited near to the Mangsalut River (Muller and Blij 2010; Oberrecht 2010; Plathong and Plathong 2010).

4.9 Fifty Years into the Future

Urbanization is one of the strongest influences that could affect the environment. Infrastructures, educational institutions areas to support human activities, this could have an indirect impact to the environment that could cause a flash flood because of the high built up of sediments that colonizing along the urban rivers (Stewart 1986a, b). Human development might exacerbate river pollution and also coastal pollution as well. The local people treat the urban river and label them as a sewer system. If this continues, the unwanted sediments such as rubbish could be carried to the Mangsalut River and could affect the area itself and the size of Mangsalut River will become smaller and eventually will not be a river catchment in the near future (Stewart 1986a, b; US EPA 2008). A strict environmental regulation is proved to have gained great significance. According to study done by Meyer (1988) regarding the Hazardous Waste Management, that the avoidance of waste is the "first commandment" in a modern management system. With this strict environmental regulation could limit the impact of the Mangsalut River that was known to produce fortune. However the relationship between development and human will always align as we move towards the future (Wolanski et al. 2009; Woodroffe and Davies 2009). Mead points out:

> As a people we have developed a life-style that is draining the earth of its priceless and irreplaceable resources without regard for the future of our children and people all around the world. (Mead 1988, p 149)

Mead saying that the human extracted the resources from the earth only for the "now" without prioritizing the resource for the future needs.

4.10 Discussion and Conclusion

In concluding, discussion it can be stated that if human are capable to adapt to changes in lifestyle through time, they also need to realize that over extracting of earth resources can have a huge effect on the environment and to themselves.

Therefore, it is the time to consider and keep the neutrality of the natural river system, water flows and coastal and estuaries landuse system in the case area in Brunei Darussalam.

On this basis of the lanuse analysis conservation and management perspective the following recommendations can be included;

(i) Sustainable land use is that which meets the needs of the present while, at the same time, conserving resources for future generations. This requires a combination of production and conservation.

(ii) The production of the goods needed by people now, combined with the conservation of the natural resources on which that production depends so as to ensure continued production in the future.
(iii) A community that destroyed its land forfeits it future. Landuse has to be planned for the community as a whole because the conservation of soil, water and other resources is often beyond the means of individual land users.
(iv) Equity and acceptability, landuse must also be socially acceptable. Goals includes food security, employment and security of income in rural areas.
(v) Land improvements and redistribution of land may be undertaken to reduce inequality or, alternatively, to attack absolute poverty.
(vi) As a recommendation, there is a need of a proper and detail information of the area. Especially the Sungai Mangsalut estuaries and catchment area which is already occupied by the people and government agencies. Some new private and commercial intervention have been developed as unplanned basis.

In conclusion, the land cover and land use in the coastal region of Sungai Mangsalut is changing over the past few years. The land that mostly still cover with forested area provided with the facilities that needed by communities at the area such as school, mosque and business area. Traffic congestion and flooding are the main issues at this area and have been overcome by the authorities by making the improvement at the coastal region of the Sugai Mangsalut estuary region in Muara district in Bandar Seri Begawan of Brunei Darussalam.

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Chapter 5
Land Use and Occupation of Coastal Tropical Wetlands: Whale Coast, Bahia, Brazil

Sirius O. Souza, Cláudia C. Vale, and Regina C. Oliveira

Abstract This chapter aims to discuss the process of land use and occupation in coastal tropical wetlands, particularly the ones that occur at the Whale Coast, in the extreme south of the state of Bahia – Brazil, between 1984 and 2014. For this purpose, orbital images of the Landsat-TM satellite were used, which were subsequently integrated and processed in the Geographic Information System using the Envi™ software, through object-targeted classification. Throughout the text, the current standard of occupation in the Brazilian territory is addressed, which is co-validated by the gradual expansion of population and economic cycles, such as the expansion of the eucalyptus forestry, the urban area and pasture in the Whale Coast. At the same time, there is a reduction of forest occupied areas. This study has the objective of contributing with landscape evolution studies and subsiding better planning proposals for land use and occupation in coastal tropical wetlands.

Keywords Coastline • Coastal zone • Occupation • Wetlands

5.1 Introduction

For being in direct contact with constructive and destructive forces of ocean and inland waters, coastal wetlands are characterized as transition zones with numerous interfering factors and a myriad of complexities. This intense natural dynamics renders them frail when faced with the prevailing natural processes, thus making them naturally unstable areas.

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Ever since human beings left nomadism behind to form settled communities, man-nature relationship grew more complex, with man no longer being part of the landscape, thus assuming the role of modifier agent. Not merely transforming nature for survival, men caused imbalance in the natural dynamics that reigns over environmental systems.

Globally, coastal tropical wetlands has high population densities, well above those found inland, which results in environmental degradation, in some cases with irreversible destructuring (Small and Nicholls 2003).

The history of territory appropriation proves that coastal areas were prioritized to the detriment of inland areas, since they have particular characteristics, such as expressively flat morphology, which facilitate agricultural activities and community settlement, and privileged location near the ocean, which provides them with better circulation of goods between continents.

Taking into consideration that, in all peripheral colonial spaces, European colonizers arrived at the new territories by sea, it is understandable why coastal areas were the first to establish population centers in developing countries. Thus, every conquest flow started out at coastal centers and progressed towards inland regions.

This recurrent pattern of territorial conformation is described by Moraes (2007) as occupation by drainage basin. This concept illustrates and confirms the existing territorial organization, based on the drainage of natural resources that are then taken to specific spaces in the Coastal Zone.

As urbanization processes advanced, these areas had their landscape reconfigured according to the growing needs of human beings, which increased the instability of natural systems and collaborated with the intensification of environmental impacts and risk situations, which can be better described by Muehe (1995, p. 254):

> Under the geomorphological point of view, the coastline is characterized by instability due to changes caused by natural and anthropogenic effects, which result in changes in the availability of sediments, in the wave climate and of relative sea level. The coastline and, especially, the beaches respond with changes in form and deposition that may have undesirable economic consequences, when they result in the destruction of property or high costs, in the attempt of stopping or delaying the geomorphological adjustment process […]

The increase in urban agglomerations is a current trend in global urban development. This has been particularly significant throughout coastal wetlands, due to the advantages in location for industrial and commercial activities. Currently, over 50% of the world's population lives up to 100 km from the coastline and eight of the ten largest urban agglomerations in the world are within the coastal region (Chhabra and Geist 2006), with high growth rates when compared to other countries, and with a tendency to rise up to 75% by the year 2020 (Charlier and Bologa 2003).

The interaction between natural processes and the various human activities is growing in coastal wetlands, which are considered transition areas between land and sea. The growth in the number and size of coastal cities in countries of peripheral economy has caused intense changes in the dynamics of environmental systems. These agitations have been accelerating the results of environmental changes, which, among many other consequences, promote alterations in the behavior and frequency of extreme events.

This coercion by use and occupation has been ever increasing, ignoring the importance of maintaining balance in the natural systems that control the morphogenetic processes, which in turn assume the role of risk factors, implying the deterioration of landscapes and ecosystems, as well as the unfeasibility of economic activities, culminating in irreversible damage.

Thus, Muehe (1998) stresses the need to prepare specific diagnoses for each area, not only for the identification of causes, but also for the proposing of mitigation and management measures concerning environmental impacts. In accordance with Muehe, geographer Ab'Saber (2003) defends the urgency of developing technogenic insertions that are compatible with the strengths and weaknesses of coastal landscapes. In this sense, a brief discussion on the current conditions of use and occupation of the Brazilian coastal wetlands is presented, based on the context of socioeconomic formations that established themselves throughout their evolution, forging a path characterized by environmental and cultural diversity, and by social and economic imbalances. Then, the analysis is deepened and the operating variables in the use and occupation of the *Whale Coast* – state of Bahia are presented, in order to represent and synthesize the current forces operating in the use and occupation of the Brazilian coastal tropical wetlands.

5.2 Distribution of the Wetlands Coastal Tropical in Brazil

The Brazilian Coastal Tropical Wetland shown in Fig. 5.1 is a continuous challenge for management due to the diversity of associated issues.

It extends from the mouth of the Oiapoque River (04°52′45″N) all the way to the mouth of the Chuí River (33°45′10″S) and the limits of the cities on the coastal strip, to the west. The land strip, of variable width, its natural recesses considered, extends for about 10,800 km throughout the coast, and has an approximate area of 514,000 km^2, of which 324,000 km^2 correspond to the territory of 395 cities (MMA 2008). Generally, it has very few indentations, with the largest being the Amazonian Gulf (PA) and the *baía de todos os santos* (BA).

It is a variable relief area, where a quarter of the Brazilian population lives, according to the Interministerial Commission for Sea Resources (CIRM), which results in a density of about 87 inhabitants per square kilometer, a rate five times higher than the national average in 2010 (Ibge 2010). This narrow continental strip includes 17 states and concentrates 13 of the 27 Brazilian capitals, some of which are metropolitan areas where millions of people live, an indicator of the high level of anthropic pressure to which natural resources are subjected (Nicolodi and Peterman 2010).

It is within this dynamic scenario of high mobility, both physical and socioeconomic, that approximately 30% of the population lives, with 16 of the 28 metropolitan areas concentrating on the coast (Fig. 5.2). These areas of high population density coexist with large expanses of scattered and sparse populating, such as the habitats of artisanal fishing communities, the remnants of the afrodescendant

Fig. 5.1 Location of the Brazilian coastal wetland

communities, the indigenous tribes and other groups immersed in genres of traditional life (Ibge 2010).

According to Mitsch and Gosselink (1986), definitions and terms related to wetlands are many and may vary from author to author, especially due to the scale and the area of study. However, wetlands are within a continuum between aquatic and terrestrial environments, so it is possible to identify some common characteristics, such as the presence of shallow water or water-saturated soil, accumulation of organic material due to vegetation and presence of plants and animals adapted to aquatic life.

Fig. 5.2 Map of the Brazilian coastal States

For this purpose, in this chapter, we adopted the concept of "wetlands" according to the Convention on the International Importance of Wetlands (RAMSAR), published by the Decree no. 1.905, May 16, 1996:

> […] wetlands are areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres. (Brasil 1996, p. 2)

This concept is in accordance with the inventory of wetlands for the Neotropical region published by Scott and Carbonell (1986) who listed 19 types of tropical wetland environments:

1. Shallow and narrow bays;
2. Estuaries and deltas;
3. Small islands near the coast, islets;
4. Rocky coastlines, headlands;
5. Marine beaches (sand, pebbles);
6. Intertidal marsh zones, sand;
7. Lagunas and coastal marshes of brackish or salt water, saline;
8. Mangroves, mangrove forests;
9. Slow running rivers, streams (permanent lowlands);
10. Fast-flowing rivers, streams (permanent upland);

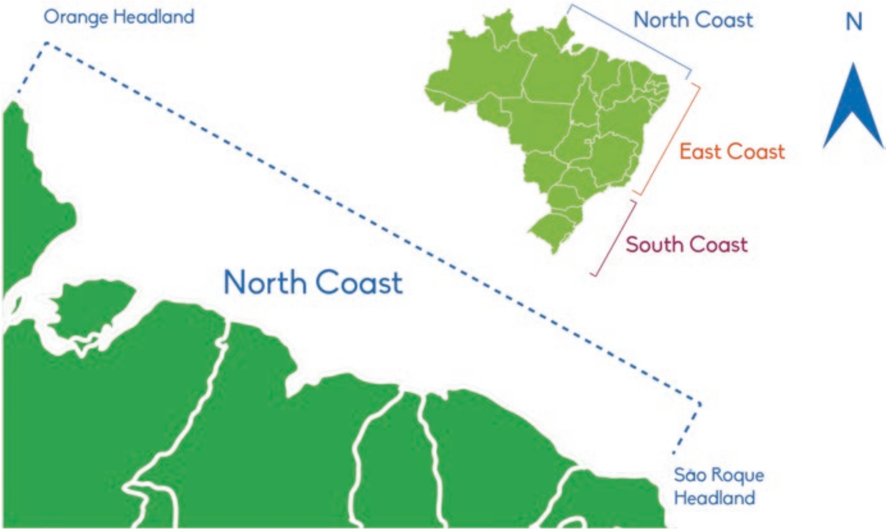

Fig. 5.3 Compartmentalization of the Brazilian coast – highlight on the northern section

11. Fluvial lakes (including backwaters of the river), floodplains;
12. Freshwater lakes and associated wetlands (lake area);
13. Freshwater ponds (less than 8 ha), wetlands, mudflats;
14. Saltwater ponds, saline (continental interior systems);
15. Dams, barrages;
16. Fields seasonally flooded, savannas, palm groves;
17. Flooded crop fields, irrigated terrain;
18. Swamp forest (swamp forest) temporary flood forest;
19. Mires, wet Andean meadows, flooded areas by melting snow.

In an inventory specific for Brazil, Diegues (1990) classified 22 types of wetlands that meet the categories proposed by Scott and Carbonell (1986), and some others: river islands, coral reefs, sandy plains and dunes (interior) and – a separate category for lagoons – salt marshes. It is evident that coastal wetlands in Brazil conglomerate various ecosystems such as freshwater ponds, brackish water ponds without marine influence; mangroves, Atlantic forests, coastal vegetation, savannah, fields, temporarily or permanently flood forests, among others.

According to Fig. 5.3, the physical and biotic characteristics of the Brazilian tropical coastal wetland can be clearly divided into three sections. Bellow the figure, we have a brief discussion on these sections and their respective features.

5.2.1 The North Coastal Tropical in Brazil

The north coast (Fig. 5.3) begins at the Cabo Orange, in the state of Amapá, and extends to the Cabo de São Roque, in Rio Grande do Norte. It is marked by large tropical wetlands with ecosystems of halophytes and pneumatophores species (Fig. 5.4), interspersed with dunes, sandbanks and coastal lagoons – for instance, the Lençóis Maranhenses National Park (Fig. 5.5), in the state of Maranhão, fortunately still well-preserved.

Fig. 5.4 Detail of the coastline of maranhão – Brazil (Source: Google Earth 2016)

Fig. 5.5 Detail of the Lençóis Maranhenses National Park – Brazil (Source: Rapideye 2014)

5.2.2 The East Coast Coastal Tropical in Brazil

The east coast (Fig. 5.6) extends from the Cabo de São Roque, in Rio Grande do Norte, all the way to the Cabo de São Tomé, in Rio de Janeiro. A striking feature of this section is the presence of sandy beaches, barrier island systems and sandy-clay sediments of the Neogene age, locally known as the *Formação Barreiras* (Barrier Formation).

This formation stands out due to the existence of deep valleys and steep edges, and a general surface inclined towards the coast, thus creating *Tabuleiros Costeiros* (Coastal Tablelands), characteristic of this section as illustrated in Fig. 5.7 (Andrade and Dominguez 2002).

It is worth noting that the east coast has sandstone reefs and corals, such as the Abrolhos Reef Complex, illustrated in Fig. 5.8, which is considered the largest and the richest coral reef system in the South Atlantic (Andrade and Dominguez 2002).

Fig. 5.6 East coast of the Brazilian coastline

Fig. 5.7 Coastal tablelands in Alagoas – Brazil

Fig. 5.8 Abrolhos Archipelago (Source: IBAMA 2012)

Fig. 5.9 Image of the cliffs in the east coast of Brazil (Source: Souza 2015)

Another point of attention is the meeting of coastal tablelands with the sea, originating cliffs (Fig. 5.9). They have sedimentary origin and include erosion scarps of the Coastal Boards caused by marine abrasion. It is a beautiful landscape with tourist appeal and still suffering the erosive action of waves.

5.2.3 The South Coast Coastal Tropical in Brazil

And lastly, the south coast begins at the Cabo de São Tomé, in Rio de Janeiro, and extends to the Arroio do Chuí, Rio Grande do Sul (Fig. 5.10). In this section, we find tropical wetlands with rugged coastline with rocky shores and crystal cliffs, where the ancient folds are very close to the sea (Fig. 5.11). The coastal plain tends to be small and discontinuous, with the common creation of Baixada Fluminense and Santista, as well as the coastal lagoons.

In this last section, wetlands are associated with estuarine-lagoon systems, a wide variety of macrophyte plant communities that vary according to the water regime, morphometry and other physical characteristics of each system (Schwarzbold and Schäfer 1984).

Fig. 5.10 South coast of Brazil

Fig. 5.11 Image of the South coast – Serra do Mar, SP, Brazil (Source: Instituto Florestal 2014)

5.3 Land Use and Occupation of the Brazilian Tropical Coastal Wetlands: Impacts and Current Scenarios

The approximately 10,800 km of Atlantic coastline places Brazil among the countries with the largest coastal areas of the world. This latitudinal coverage, with extensive climatic and geomorphological variety, is one of the main factors that explain the diversity of species and ecosystems along the Brazilian coastline. Despite its size, most of the Brazilian coastal wetlands are characterized by low nutrient concentrations and reduced productivity, contradicting the common sense that the region is an abundant and inexhaustible source of resources (Mma 2010).

Although the fishing activity in Brazil has unquestionable socioeconomic importance as a provider of animal protein, while generating around 800,000 jobs, for four million people that are, directly or indirectly, linked to the activity. In recent years, studies have revealed this mistaken perception regarding the abundance or inexorability of these resources (Mma 2010).

The Coastal Zone is, strictly speaking, an ecological transition region and has an important role for the development and reproduction of many species and genetic exchanges that occur between terrestrial and marine ecosystems. Moreover, the Brazilian Coastal Zone has significant territorial overlap with the Amazon area and the Atlantic Forest, and to a lesser extent, with the Caatinga, Cerrado and the Pampas, being not an ecological unit but a mosaic of ecotone ecosystems with highly complex ecological environments, extremely important to sustain life.

Besides the tropical and subtropical characteristics prevalent along the entire coast of the country, the oceanographic and climatological conditions of the region provide distinctive characteristics to biodiversity. The sea area by the coast has warm waters on the north and east coasts, and cold water on the south coast, supporting a wide variety of ecosystems, such as dunes, beaches, marshes and wetlands, estuaries, sandbanks, mangroves, rocky shores, lagoons and salt marshes, housing numerous species of flora and fauna, many of which are endemic and in danger (Mma 2002a, b).

Nutrient concentrations and other environmental conditions, such as thermal gradients and salinity alterations – under extraordinary conditions of shelter and support to the early stages of reproduction and nutrition of most species that live in the oceans –, make the coastal tropical wetlands environments one of the main targets of environmental preservation to preserve biodiversity.

However, following the global trend, the Brazilian coastal wetlands are also marked by population conglomerates and productive activities. Data from the last census of the Brazilian Institute of Geography and Statistics (Ibge 2010) show the two important processes that influence how Brazilian geographical spaces have been configured in the past decade (2000–2010): the inland-advancing process, driven by agriculture and livestock expansion, and the process of "coast revaluing" related to the expansion of economic activities linked to tourism, oil exploration and port and air logistics, that go "beyond population density and urban centers located near the sea, reinforce the historical process of coastline urbanization" (Ibge 2010).

Baeninger (2003) adds that, despite the current trend of population decentralization and new non-metropolitan urban territories in the country, the coastline urbanization does not seem to have affected the population density in the Coastal Zone. In over 70% of the coastal cities, the urban population prevails over the countryside. Data from 2010 indicate that 45.6% of the 395 coastal cities have urbanization rates between 80 and 100%. The urbanization percentage in the rest of the cities is 27.2% (Ibge 2010).

Thus, it is clear that Brazilian coastal area occupation has intensified in recent decades, due to three priority vectors of development: urbanization, industrialization and tourism activities (Moraes 1999). In addition to these vectors, we also have intraregional migration, productive restructuring and municipal emancipations with the inclusion of new population groups, uses and activities of coastal cities. The contextualization of these processes is essential to understanding the present dynamics of the Brazilian coastal tropical wetlands.

Coastal urbanization as illustrated in Fig. 5.12 is a national and international phenomenon, linked to coastal valuing for historical, economic and, more recently, cultural and environmental reasons.

The historical and economic background reinforces the concentrating character of urbanization in Coastal Zone s. However, within a decade, the number of metropolitan areas increased in the country from 9 to 28, and in the coastal area from 5 to 16. This growth, although controversial in some cases, indicates a trend of metropolitan structures expansion that makes coastal areas even more complex, due to the anthropic pressure on natural systems, and sanitation, housing and transportation needs, in addition to the basic public services human conglomerates require (Ibge 2010).

Fig. 5.12 Urbanization of the Barra da Tijuca – Rio de Janeiro – Brazil (Source: Google Earth 2016)

The environmental and cultural features emphasize the unique character of the Coastal Zone, urbanization, identifying it as a public space for recreation and, in some cases, preservation. The urbanization process changes the space with allotments, vertical and horizontal condominiums for second homes near large urban centers, hotel complexes and resorts for domestic and international tourism in different beautiful areas (Strohaeckeri 2008).

According to Strohaeckeri (2008), in the last two decades, the productive restructuring and modernization of port industrial complexes in the east and south coast highlight the concern in adjusting the country to the demands of the global economic market. This suggests an overlap and interconnection of active processes in the Brazilian Coastal Zone: industrialization, urbanization, tourist development and real estate, productive restructuration, migration and, to a lesser extent, emancipations (Strohaeckeri 2008).

In the current Brazilian coastline, we can distinguish two main economical macro sets (Strohaeckeri 2008):

(a) the first includes the South and East Coasts and part of the North Coast – from Rio Grande do Sul to the metropolitan area of Fortaleza – and is mainly characterized by urban occupation and the revaluing of land use, including urban interstitial spaces that have resulted from land reserves for commercial purposes or for economic activities such as forestry, rice cultivation, agriculture and extensive livestock;
(b) the second covers the rest of the North Coast, from the metropolitan area of Fortaleza, in Ceará, to Amapá, on the border with the French Guiana. It is sparsely occupied, with low population density, large unoccupied areas and traditional extracting and collecting communities. Highlight to some regional centrality poles, like Parnaíba (PI) and Macapá (AP), and metropolitan concentrations in Belém (PA) and São Luís (MA).

In the last intercensal period (1991–2010), shown in Table 5.1, the resident population of the Brazilian coastal tropical wetlands hada total population increase of 5,465,581 inhabitants, corresponding to approximately 47.87% since 1991 – a growth higher than the national growth recorded for the same period, approximately 29.92%.

The development vectors that guide land use and occupancy rate are still industrialization, urbanization and tourism, being a milestone of the Brazilian Coastal

Table 5.1 Percentage distribution of the Brazilian population living in the Coastal Zone

Year	Resident population		Proportion of the resident population in the Coastal Zone (%)
	Total	Coastal zone	
1991	146,825,475	34,315,455	23.37
1996	157,070,163	36,204,278	23.05
2000	169,799,170	39,781,036	23.43
2010	190,755,799	50,745,044	26.58

Source: Ibge (2010)

Zone development (Moraes 1999). These vectors are related to activities that, on one hand, have different types of implementation – occupation standard and space along the coast – while on the other hand follow the same logic of resources exploitation and the natural and locational potentials of the territory.

Strohaeckeri (2008) discussed vector development in the coastal zone and declared industrialization as an eminently concentrating process, achieved through financial, energy, human and infrastructure resources that allow space fluidity and optimization of networks interaction. The author points out that the use of industrial plants in the coastal zone is rare and subject to State policies that demand accessibility and both physical and virtual connections.

Oil exploration is an example of industrial activity of the utmost importance for the Brazilian economy. Estimated as one of the economic acceleration growth factors, this activity showed significant growth in recent years, contributing to increase the GDP per capita of coastal cities responsible for the production. Oil exploration and offshore natural gas account for the major part of the national production, mainly Campos (RJ) and Santos (SP).

The Brazilian coastal tropical wetlands stand out regarding population, industrial and economic concentration, being also responsible for a wide range of ecological functions: flood prevention, saline intrusion and coastal erosion; protection against storms; nutrients and pollutants recycling, and the direct or indirect provision of habitats and resources for a variety of exploited species.

Oliveira and Souza (2014) deem the Brazilian Coastal Zone an area of multiple uses by various activities, with specific valuing of spaces, without following a pattern of homogeneous use and occupation. Regarded as “a space endowed with specificities and locational advantages” (Moraes 1999), the Coastal Zone has unique natural characteristics that emphasize the great environmental and landscape value of the area, as well as being the focus of considerable economic interest, with a dense infrastructure where strong population densities prevail.

In Brazil, environmental and social impacts are part of the current Coastal Zone scenario. The pace and intensity of transformation invariably cause imbalances and sometimes uncontrollable social tensions, leading to, for example, socio-cultural processes of destructuration of the traditional local communities and accentuate vulnerability levels of many natural systems.

The high environmental vulnerability levels seen particularly in coastal systems with important environmental services and the negative changes in the life of traditional communities are considered, by rule, as a consequence of unplanned land uses or ineffective political actions (Souza and Vale 2016).

Disorderly occupation and unbridled tourism growth increasingly emphasize the mosaic of inequalities, supporting an exploitation model that degrades ecosystems. These actions, as a rule, do not follow decision processes based on knowledge of natural and social dynamics, do not use future projections and, thus, incorporate unbalanced practices of land management (Oliveira and Melo Souza 2013). Bellow we present the survey on use and occupation of Whale Coast, used as a tool for analysis and discussion on the management and planning of coastal tropical wetlands.

5.4 Case Study of the Land Use and Occupation of Whale Coast: Bahia: Brazil

As in other countries, population growth is considered the main cause of the environmental changes observed in the Brazilian coastline (Ibama 2002). In the state of Bahia's Coastal Zone, the average population density is 96 inhabitants per km^2, which is low when compared to Pernambuco, with 913 inhabitants per km^2, and Rio de Janeiro, with 806 inhabitants per km^2 (Ibge 2010). The coast of Bahia, therefore, can better control occupation, to minimize coastal environmental impacts.

Among the coastal states, Bahia stands out as having the longest coastline in the country, with approximately 1,183 km and high socioeconomic potential emphasized by dense population occupation, large agricultural use, high tourist demand, presence of sedimentary basins, alkaline bodies that are economically exploitable and so on (Nicolodi and Petermann 2010).

According to Anjos (2005), Bahia coastline is currently "sold" as the New Caribbean, the New Mediterranean, the New Florida. Government initiatives result in significant flow increases – both domestic and international –, while foreign capital injected into the sector assumes an important role (hotels and resorts).

The coast is an interesting reality as a study topic and the basis for public policies, since the coastal regions of Bahia receive international capital and are marked by considerable social and environmental contrasts. This study opted to focus particular attention on the coastal region known as Whale Coast,[1] a name given by the Bahia Tourism Development Program (Prodetur 2003) (Fig. 5.13).

Whale Coast is on the extreme South of Bahia, between the geographical coordinates 17°15′ and 18°20′ S, and 39°08′ and 39°40″ W. The region has about 130 km of coastline, being part of the Alcobaça, Caravelas, Nova Viçosa and Mucuri. Originally part of the Colonial Captaincy of Porto Seguro, this area has great diversity of landscapes, with sandy beaches, beach sections coated with sandbank vegetation, and tidal plains covered by mangroves, among others. The area has been the target of disorderly real estate speculation, for being considered the "Entryway" to the Abrolhos Marine National Park and due to a multitude of environmental impacts derived from human intervention.

Geotechnology allowed us to diagnose the main forms of land use and occupation in Whale Coast between 1980 and 2014. This study aimed to contribute to the planning of the territory's use and occupation. Based on this, the following questions guided this work: what have been changing in the use and occupation of the area in the last 34 years? How lands are used today and how they relate to each other? Is there a link between tropical wetlands and areas of anthropic use? The answers to these and other questions are the issues discussed below.

The methodological procedures were divided into three stages: the first was a literature review on land use and occupation of coastal tropical wetlands; the second

[1] The name refers to ecological sanctuaries such as the Abrolhos Marine National Park, nursery of the Humpback Whale.

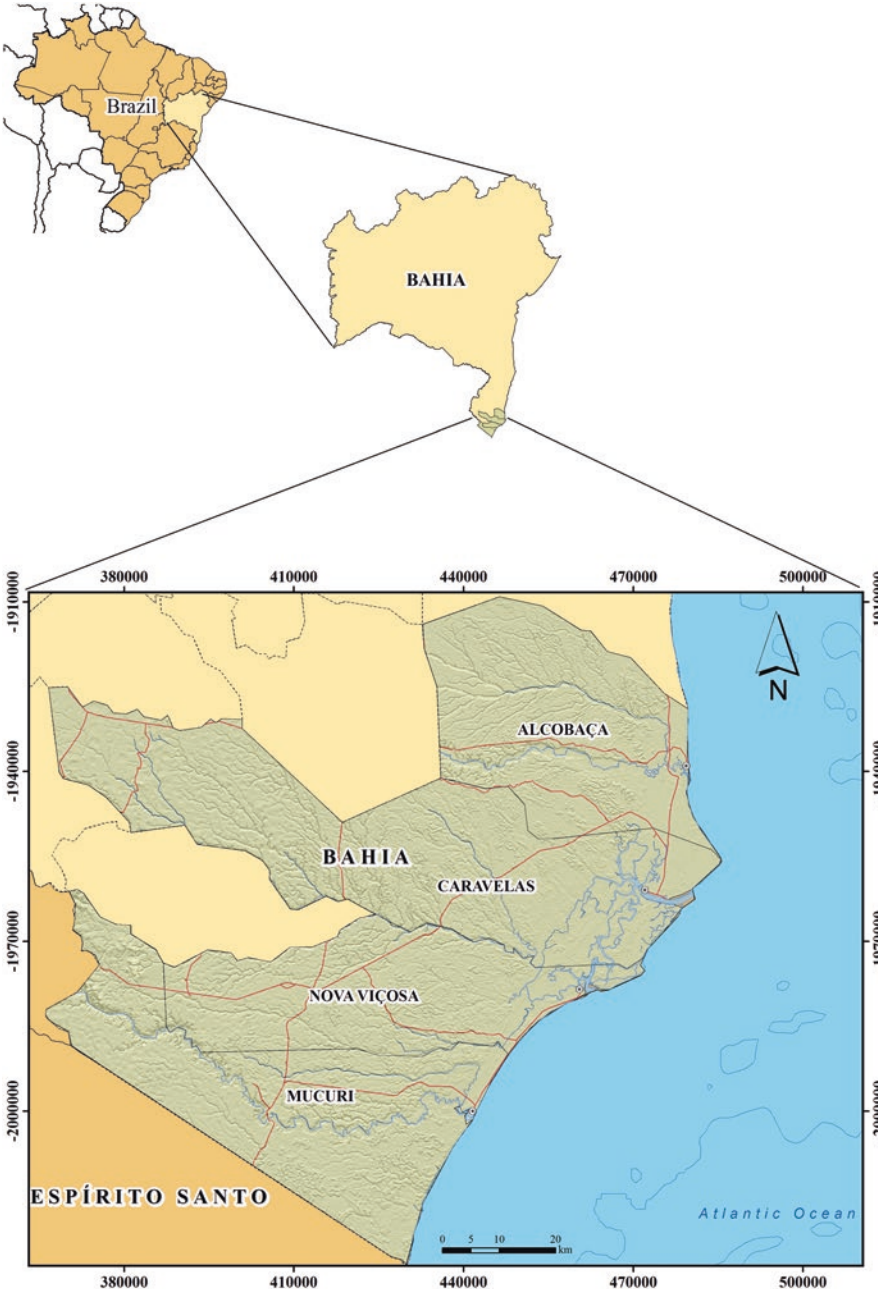

Fig. 5.13 Location of the Whale Coast

was the acquisition of Landsat TM images and work in the field with observations and class records on the present land use and occupation; the third stage involved the integration of data collected in the field with the data from the Landsat TM sensor in GIS environment, using the ArcGIS™ 10.1 and the ENVI™ software, followed by the final draft of the research. In the next lines we detail the second and third stage procedure.

The Land Use and Occupation Maps (1984 and 2014) were made with a 1:100,000 scale, using object-targeted classification of orbital images. Basically, this type of segmentation and classification consider various characteristics such as spatial (medium) and spectral (color) heterogeneity and the difference of the surrounding objects; in other words, it uses the region's growth method to add neighboring pixels.

Moreira (2003) states that the classification uses algorithms and the spectral patterns in the image are recognized in a sample from the training area provided by the analyst, thus validating the need to know the area of study and improving the quality of the map generated.

For this research, four images were used from the Thematic Mapper sensor TM, orbit 215 point 072 and 073 and orbit 216, point 072 and 073, with satellite passing dates of April 1980 and 2014. The TM sensor aboard LANDSAT 5 and LANDSAT 8 satellites makes images of the Earth's surface, producing images of 185 km (width) of the ground, with spatial resolution of 30 m and 7 spectral bands. The satellites revisit time to capture the same portion of land is 16 days.

Orbital images were chosen considering the smallest possible amount of clouds, lower excess of brightness and higher spectral normality. During this search, it became clear that the region had a predominance of orbital images covered by clouds, which somewhat limited the choice of images, especially regarding available dates.

After selection, these images were georeferenced through Envi® 5.0 software with the IBGE's topographic sheet as basis, which covers the area of study, plus a 1:100,000 scale in digital format. Then images were grouped into a false-color composition (R5G4B6). The representative interpretation keys of each class of interest were chosen for the algorithm used in the object-targeted classification. Nine classes were defined based on the Technical Manual on Land Use of the Brazilian Institute of Geography and Statistics (Ibge 2013): Urbanized Areas, Inland Water, Ocean Water, Temporary Cultures, Pastures, Eucalyptus Forestry, Forest Area, Mangrove Areas and Uncovered Areas. The use of symbols and standard forms also followed the same manual.

The next step was generating the maps using the Envi feature extraction module that extracts information and classifies images based on spectral, spatial and texture characteristics. Initially, image segmentation was applied on homogeneous regions. For the item "edge" threshold 40 was applied, and 50 for the "merge setting", both chosen by trial and error, until a satisfactory result was obtained through visual observation. Among the types of object-targeted classification algorithms, there are Support Vector Machines (SVM) and K-Nearest Neighbor (K-NN). In this study, based on Garofalo et al. (2015), SVM were used, since it provided the best accuracy classifications for the area of study and scale used.

The SVM algorithm aims to determine decision thresholds that produce optimum separation between classes by minimizing errors (Vapnik 1995). SVM is a computational technique for pattern recognition problems, based on the optimal separation of classes (Huang et al. 2002). Further details of the SVM method, as well as applications in remote sensing area, may be found in Waske et al. (2010), Liesenberg and Gloaguen (2013), and Garofalo et al. (2015).

5.4.1 Discussion

As a case study of a tropical coastal wetland it presents the following discussion of the results on the use land and occupancy in Whale Coast region, later followed by a proposition of possible diagnoses and scenarios.

5.4.2 Land Use and Occupation in Whale Coast, Bahia

Land use and occupation between 1984 and 2014 were observed by analyzing the spectral behavior of the targets and the fieldwork performed in Whale Coast. Figures 5.14 and 5.15 show the spatialized results, based on the categories previously described.

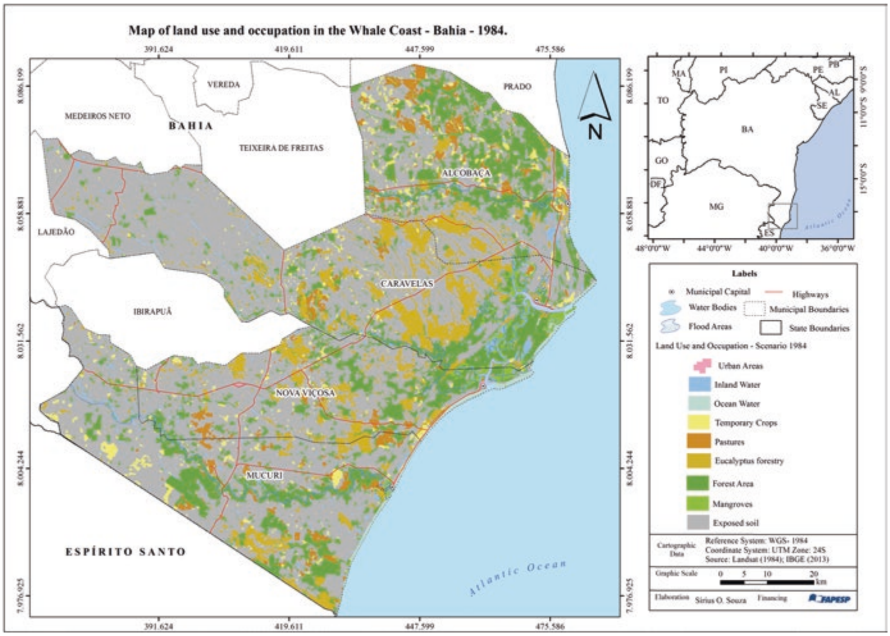

Fig. 5.14 Map on the use and occupation in the Coastal Plain of Caravelas in 1984

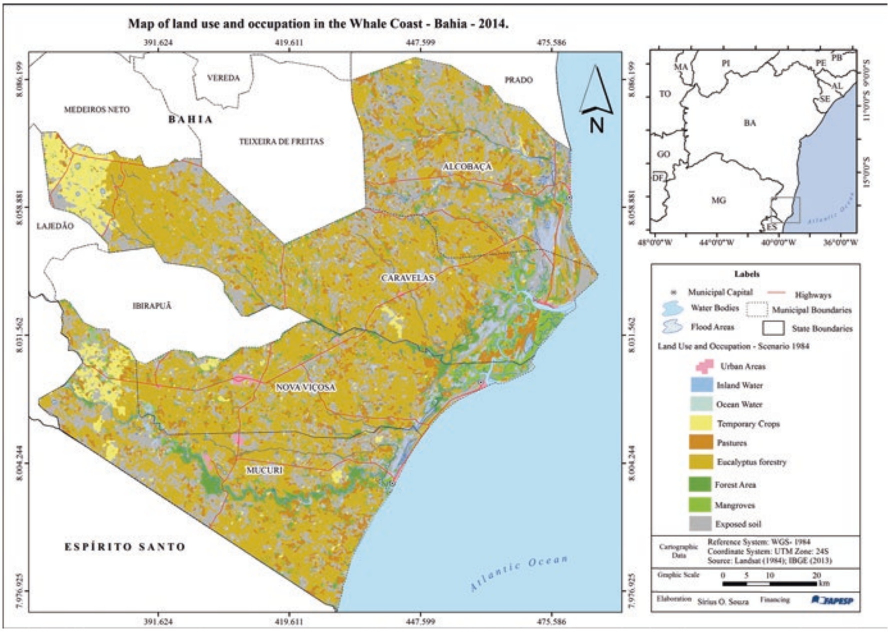

Fig. 5.15 Map on the use and occupation of the Whale Coast in 2014

Eucalyptus forestry is currently a matrix of region, occupying about 70.48% of the total area (4,858.61 km^2), and represented by the massive presence of subsystems responsible for the production of seedlings, planting, cultivation, silvicultural treatment, exploration, budding management, among other subsystems of the cellulose chain production. Authors such as Souza (2013, 2015), Amorim (2011), Pedreira (2008), Limonad (2008), and Fontes and Silva (2005) among others confirm the predominance of this class in the region studied.

The first map on use and occupation, referring to the 1984 scenario, reports a new cycle of investments in the region, characterized by implementation and expansion of forest regions, linked to the agro-industrial sector of paper and cellulose, which occupied about 9.73% of the total area (about 670.74 km^2). According to Ima (2008), in the region, the eucalyptus cultivation had productivity five times higher than in other regions of the country. This fact caught the attention of large companies, since the area is one of the most productive in the world.

Given these information, it is important to consider that in Whale Coast the areas occupied by eucalyptus forestry (Fig. 5.16) grew from 670.74 km^2 in 1984 to 3140.63 km^2 in 2014.

This means a growth from 9.73% to approximately 45.56% of the occupied area. In only 34 years, this class had a total growth of 368.23%. Knowing this type of use is important, since it is one of the main roles of the economic and spatial vector in the area studied.

Many authors defend the positive aspects of eucalyptus forestry, like Vital (2007): improved on-site water regime of degraded areas; improvements in nutrient concentration in degraded soils, particularly litter mineralization; increases in local

Fig. 5.16 Details of eucalyptus forestry in Alcobaça – Bahia – Brazil (Source: Google Earth 2016)

biodiversity rates and the possibility of building ecological corridors and carbon dioxide sequestration.

As described in Figs. 5.14 and 5.15, eucalyptus cultivation occupied mostly previously uncovered areas, or areas already deforested – mainly degraded pastures –, meaning environmental improvements over the previous use, visible in a short period of time. An approximate reduction of 42% was observed, equivalent to 2,271 km^2. Floriano (2004) defends that forest plantations are less impactful than pastures, with more benefits during periods when the tree layer is closed.

However, there are many negative aspects – whether ecological or social. It is noteworthy that the planting of eucalyptus (*Eucalyptus* sp.) occurs through the insertion of exotic and transgenic species. This fact alone is a great demerit for the local environmental development. It is known that invasive alien species have a significant impact on environmental biodiversity and are considered the second biggest threat to biodiversity (Espínola and Júlio Júnior 2007), after the destruction of habitats, thus directly affecting coastal wetlands, local economy and human health, also leading to environmental homogenization, destruction of unique characteristics and alterations in essential ecological properties.

During the study period, we noticed a continuous increase of areas occupied by pastures, represented by the existence of introduced herbaceous/granitoids fields or by deforestations with the purpose of becoming pastures. This class had an absolute increase of 394 km^2 (about 154%), which is explained by the local economic reality, supported mainly by activities related to agriculture and breeding. This use occupied, in 2014, approximately 648.49 km^2, or 9.41% of the area of study.

Agricultural uses, represented by the class Crops, as seen in Fig. 5.16, concentrate approximately 593.85 km^2 of the area. Generally, there is a predominance of temporary crops over permanent crops. Except for the city of Alcobaça that had more permanent crops, such as coco-da-baía (coconut), passion fruit, papaya, annatto and coffee.

Caravelas and Nova Viçosa are mainly involved with production of sugarcane, watermelon, cassava, corn and black pepper. Mucuri, in 2013, had approximately 13,600 ha of permanent and temporary agricultural areas. Its production focus on sugarcane, cocoa, papaya, cassava and *coco-da-baía*. Mucuri showed, in 2013, a harvested area larger than the total area harvested in Alcobaça and Nova Viçosa combined. This fact proves the existence of investments, agricultural policies and, especially, forestry partnership programs between farmers and forestry units.

Urban areas in Whale Coast cover approximately 0.49% of the total area, and are part of urban centers in Alcobaça, Caravelas, Nova Viçosa (Fig. 5.17) and Mucuri. This class also shows urbanized areas in Posto da Mata district, which is part of Nova Viçosa, and Itabatâ, which is part of Mucuri. According to the mapping carried out, during the last 30 years this class showed an increase of approximately 632% (about 29,26 km^2).

This growth occurred thanks to new residents attracted to the area by the eucalyptus plantation or by the third sector of the regional economy. The growth was also due to the construction of second homes and tourism facilities. Urbanization was mainly observed in the city areas, justified by the land routes available, when compared to other parts of the region.

Forest areas, largely represented by the presence of Dense Ombrophylous Forest – in advanced stage of anthropization –, Mata Seca de Restinga and other vegetation associated with coastal wetlands, showed a drastic reduction in the

Fig. 5.17 Details of Urban areas in Nova Viçosa- Bahia – Brazil (Source: Google Earth 2016)

period of study, with estimated 351.22 km^2 devastated. This reduction may be associated with the sinuous growth of areas occupied by eucalyptus plantations, agricultural areas and urban areas.

Considering mangroves and other coastal wetlands, there is a slight growth of 95.62 km^2 in the area, as well as a slight increase in the areas occupied by inland water (about 61.63 km^2). A fact that can be associated with the possible recovery of mangrove stretches in the Permanent Preservation Areas and Legal Reserves, and also the gradual recovery of areas from Cassurubá Extractive Reserve, illustrated in Fig. 5.18, created by Presidential Decree on June 5, 2009, comprising an area of 100,687 ha, divided into 31,996 ha of estuary and 68,665 ha of marine area.

5.4.3 Land Use and Occupation of Whale Coast: Diagnoses and Possible Scenarios

In this section, the classes of land use and occupation found during fieldwork are described and in Table 5.2 and in Fig. 5.19 some possible scenarios are listed, according to the spectral behavior in an expansion or reduction environment.

Field diagnoses allow us to infer that currently the eucalyptus forestry is the matrix of use and occupation in Whale Coast and tends towards expansion (Table 5.2). Simultaneously, the areas occupied by pastures also tend to expand, and are in various stages of degradation, with large areas having been abandoned and subjected to an extensive process of laminar erosion and erosion in grooves, which contributes to the silting of the surrounding rivers.

In agricultural areas, crops tend to expand and reveal, at different points, degradation of ecosystems due to plantations and crops near the rivers, what disrespects the Brazilian Contemporary Forest Code (Law 12.651, of May 25, 2012) that considers as Permanent Preservation Area the marginal tracks of any natural watercourse, perennial and intermittent, from the edge of the regular bed rail, with a minimum width of 30 m.

Table 5.2 Evolution of land use and occupation in the Whale Coast

	1984		2014		
Types of use	Area (km)	Area (%)	Area (km)	Area (%)	Diagnostics
Pastures	254.44	3.69	648.49	9.41	Expansion
Forest area	1,396.79	20.26	490.98	7.12	Reduction
Mangroves	124.12	1.80	219.74	3.19	Expansion
Temporary crops	234.20	3.40	359.65	5.22	Expansion
Inland water	239.87	3.48	301.50	4.37	Expansion
Urban area	4.62	0.07	33.88	0.49	Expansion
Eucalyptus forestry	670.74	9.73	3,140.63	45.56	Expansion
Exposed soil	3,969.00	57.57	1,698.91	24.64	Reduction
Total	**6,893.78**	**100.00**	**6,893.78**	**100.00**	

Fig. 5.18 Details Cassurubá extractive reserve in Caravelas – Bahia – Brazil (Source: Google Earth 2016)

Interfering in agricultural areas, urban areas grow disorderly and reveal the recent land concentration process, the decrease of the number of rural workers and socioeconomic reorganization, a direct consequence of the implementation of eucalyptus forestry in the region. Both tend towards expansion.

Forest areas are moderately conserved and tend to decline as shown in Fig. 5.19, an issue already discussed by Dominguez (2008). During the fieldwork, bushfires (extensive areas of Mata Seca de Restinga*)* was observed, and this justifies the studies on susceptibility of forests to bushfire.

Mangroves and water bodies that result from the creation of present conservation units, like the already mentioned RESEX Cassurubá, have a tendency to expand. They cover large areas of Whale Coast and group ecosystems marked by commercial use and by artisanal fishing of crustaceans and mollusks. During fieldwork, we observed the indiscriminate use of these areas, with bushfires, illegal abstraction of water and grounds. This fact validates the constant need of studies and monitoring, since human interference is frequent.

The uncovered areas – also known as areas of bare soil – tend to expand and have large pastures and abandoned agricultural lands due to wrong soil management. Sandy beaches are also areas of bare soil that represent an important economic region of the Whale Coast.

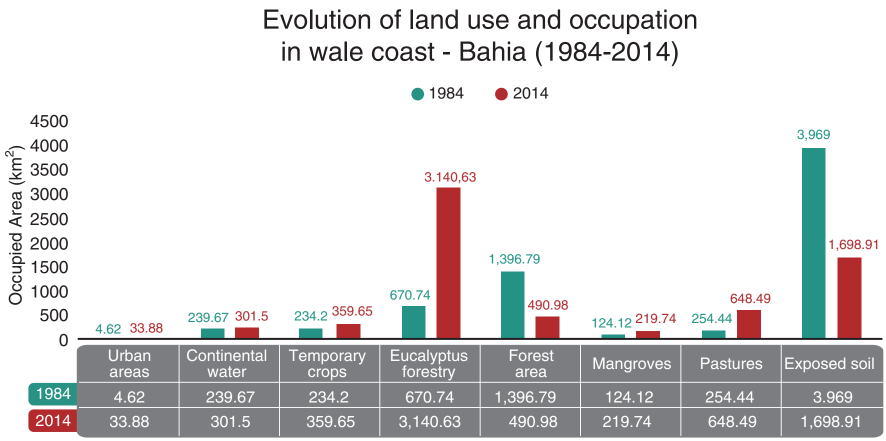

Fig. 5.19 Chart with the evolution of land use and occupation in Whale Coast

5.5 Conclusions

It is clear that occupation of Brazilian coastal tropical wetlands are linked and subordinated to external interests related to strategies between private and State interests that affect the region and the Brazilian coastline occupation. The evolution and problems related to the regional inequalities and dynamics indicate that some cities have been gradually losing their social and economic position.

In Bahia, as well as in Brazil, the equalization of infrastructure and welfare conditions remain therefore on the list of priorities. This equalization can create a new regional development model that rooted in local, solid and secure foundations, against the tendency of productive reconcentration and interstate income inequality.

With the analysis of the use and occupation of Whale Coast lands in the last 34 years, we observed that the changes reflect an economic model current in Brazil – occupation of coastal tropical wetlands occurs in a disorderly manner, disregarding the importance of ecosystems. This study proves the gradual expansion of the agricultural, urban and pasture areas, with different trends. At the same time, there is a reduction of the areas occupied by forests and wetlands and an overgrowth of areas occupied by eucalyptus forestry.

It is not only a technical problem or lack of planning in tropical wetlands, or even the lack of regulatory mechanisms, but a choice related to the management of these areas. The decisions must meet regional and local interests with balanced and fair urbanization and territorial occupation.

Further research in this area is necessary, aiming to increase valuing and preservation of the natural environment and the historic and cultural heritage, based on instruments that minimize the negative effects of human interference.

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Chapter 6
Degraded Coastal Wetland Ecosystems in the Ganges-Brahmaputra Rivers Delta Region of Bangladesh

Shafi Noor Islam, Sandra Reinstädtler, and Albrecht Gnauck

Abstract The Ganges-Brahmaputra-Meghna Rivers carry 6 million m^3/s water and 2.4 billion tons of sediments annually into the Bay of Bengal, but it shows no tendency toward rapid seaward progradation. Bangladesh coastal region is gifted with vast natural resources, delta, tidal flat, mangrove forests, march, lagoon, bars, spilt, estuary and coastal ecological environment which is very much potential for communities survival. There are 36.8 million people are living within the coastal region and dependent on coastal water resources. The coastal water resources are drastically reducing due to unplanned use by the community and the stakeholders. The coastal zone of Bangladesh is enormously important for the development and management of natural resources. The coastal water resources are playing an important role to protect the coastal ecosystems and socio-economy. The present situation stated that an integrated natural resource management plan is necessary for the protection of coastal ecosystem and coastal community livelihoods. The paper prepared based on primary and secondary data sources. The objectives of this study are to analyze the present coastal natural resources management status. The study seeks the deltaic wetlands ecosystem development and management strategies for ensure communities livelihood and sustainable development of coastal resources in Ganges-Brahmaputra Rivers deltaic coastal floodplain region in Bangladesh.

Keywords Coastal wetland • Ecosystem • Ganges-Brahmaputra delta • Degradation and management

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6.1 Introduction

The coastal wetlands provide numerous ecological services to humanity. They protect the coast against erosion and guard against loss of capital infrastructure and human lives (Wolanski et al. 2009). Coastal ecosystem also provide ecosystem services that have socioeconomic benefits to the human population, including fuel, forage, building material, timber, fisheries, and protection of commercial, recreational, and naval vessels (Williams 1919; Wolanski et al. 2009). The tropical and sub-tropical coastal mangrove wetlands provide another important ecosystem service to the population living in the hinterland, by sheltering it from storm winds and capturing salt spray and improving crop production in arid coastal areas (Wolanski 2007). The coastal region of the Ganges-Brahmaputra Rivers Delta is the largest floodplain wetland region is located in the South Asian Region (Morgan and Mcentire 1959; Khandoker 1987; Adeel 2001; Sarker et al. 2003; Goodbred and Nicholls 2004; Islam 2016).

Bangladesh is a land of water but water is one the most critical and major problems in country, especially the scarcity of upstream fresh water has created serious environmental problems in the coastal region of Bangladesh (Islam et al. 2016). The location of the landward boundary of the coastal zone is a function of three basic geophysical processes: tidal fluctuations; salinity; and risk for cyclone and storm surges. The coastal zone Bangladesh, affected by these processes, cover an area of 47,210 km^2, or 32% of the country, being the landmass of 19 districts (Fig. 6.1) abound 36.8 million people, representing 29% of the population, live in the coastal zone (MoWR 2005; Goodbred and Nicholls 2004; Islam 2016). Coastal zone is the area on both sides of the actual land-sea interface, where the influences of land and water on each side are still in determining factors-climatically, physiographically

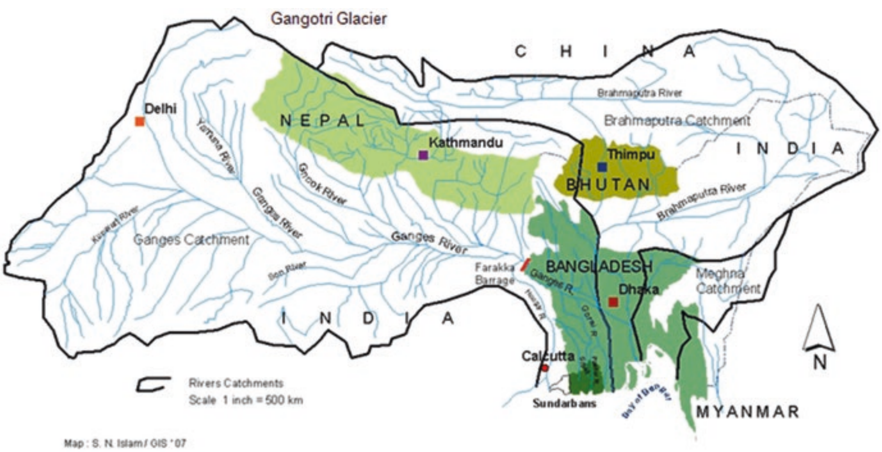


Fig. 6.1 The geographical location of Bangladesh and its coastal region (Map Source: Islam and Gnauck 2008)

and ecologically. The deltaic coastal areas of Bangladesh is about 710 km in length, extend along the Bay of Bengal from the mouth of Teknaf river in the south East to the Mouth of Raimangal River in the south west and includes the greater districts of Chittagong, Noakhali, Barisal, Patuakhali and Khulna (Nishat 1988). The question for sustainable development is a key concept in strongest driving force of the water sector (Grigg 1996; Gleick 1998; Costanza et al. 1992; Islam and Gnauck 2007a, b).

Coastal wetland environment and ecosystem services play an important role in socio-economic development of the Bengal Deltaic coastal region of Bangladesh (Islam 2016). About 25% (35 million people) of population live in the deltaic coastal areas and most families dependent on coastal natural resources for their livelihood (Hidayati 2000). Most part of the Bengal deltaic coastal areas of Bangladesh are an active or nascent delta with vigorous dynamism and the Islands or chars are formed, eroded and reformed (Islam 2016). The Swatch of No Ground (SNG) is also a major feature only 24 km far from the coast water and from the Sundarbans mangrove forest (Rahman 1988). The coastal region is composed of the land and the sea including estuaries and islands adjacent to the land water interface of south east Bangladesh and the coast can divide into three distinct regions (Pramanik 1983). It is at the coast where the heavily sediment-laden river water with very little salinity meets the saline sea water. This mixing creates a unique ecosystems which have given rise to the species of flora and fauna that exhibit their presence here. Amongst these the most important are the Sundarbans with its unique tropical mangrove forest and the wild life, the fisheries dominated by economically important species such as Hilsa and Prawns (Rahman 1988). Upstream fresh water supply to the coastal region is playing a potential role to make a balance of ecosystems. Therefore sustainable deltaic coastal wetland ecosystem management is essential for the coastal community's livelihoods as well as other stakeholder's benefits in Bangladesh (Pramanik 1983; Rahman 1988; Costanza et al. 1992; Islam and Gnauck 2007a, b, 2016).

6.2 The Aims and Objectives of the Study

The aims of this study is to understand about unique characteristics of the coastal deltaic wetland region of Bangladesh and its potentiality of coastal socio-economy and community development and the role for the protection of coastal wetland ecosystem.

There are some specific objectives of this study have been recognized as follows;

(i) To understand the characteristics of the Ganges-Brahmaputra Rivers deltaic coastal wetland ecosystems and its role in the coastal region.
(ii) To investigate the water salinity intrusion trends in the lower tidally active mangrove wetland regions and analysis of the degraded wetland ecosystem in the deltaic floodplain regions in Bangladesh.

(iii) Study seeks the adequate coastal wetlands ecosystem management strategies and
(iv) Makes recommendations for future development policies for the coastal natural resources, protection and management of coastal deltaic degraded ecosystem in the Bangladesh coastal region.

6.3 Data and Methodology

The present study was carried out based on primary and secondary data sources. The primary data collected from field investigation in 2003 and 2008. PRA practices were arranged with the local people at Munchiganj near Sundarbans, Koira Upazila, and Mongla port area. The information was collected from Galachipa and Kalapara upazilas of Patuakhali district. Besides these a various reports and published articles in journals and conference proceedings have been used for this study. Published materials, reports and journals were collected from different government and non-governmental organizations in Bangladesh. For secondary data collection, CARDMA reports on coastal resource development and management was used very openly. The especial research reports of CARDMA, FAP 24 report, BWDB, CEGIS reports and BCAS reports were also used for this study. Beside these some interviews were arranged with water engineers, environmentalist, geographers, geologists and sociologists and expert peoples on river systems and its ecology. Besides, the secondary data were collected from the relevant published research works in the country and outside of the country. The published papers in journals and books in Bangladesh and outside of country were received. The collected data were processed, analyzed, and visualized through MS Excel interpolation, VISIO 32, ArcGIS 9.3 and 10.1 software. They were used for water salinity approximation to investigate the degraded deltaic coastal floodplain wetland ecosystems in the Ganges-Brahmaputra Rivers Deltaic region in Bangladesh.

6.4 Geography, Geological Setting of the Bengal Deltaic Coastal Region

Bangladesh is situated in the Ganges-Brahmaputra-Meghna Rivers catchments. The deltaic coastal region of Bangladesh is located in the south of the country. The coastal length is 710 km and the coastal area comprises 47,210 km^2 with a population of 36.8 million (BBS 1979, 1981, 1985, Islam 2016). There are 19 costal districts are included within the coastal region such as Jessore, Narail, Satkhira, Khulna, Bagerhat, Gopalganj, Shariatpur, Chandpur, Pirojpur, Jhalokati, Barguna, Patuakhali, Barisal, Bhola, Lakshmipur, Noakhali, Feni, Chittagong and Cox's Bazar (Figs. 6.1 and 6.2). The coastline of the Ganges delta extends from the month

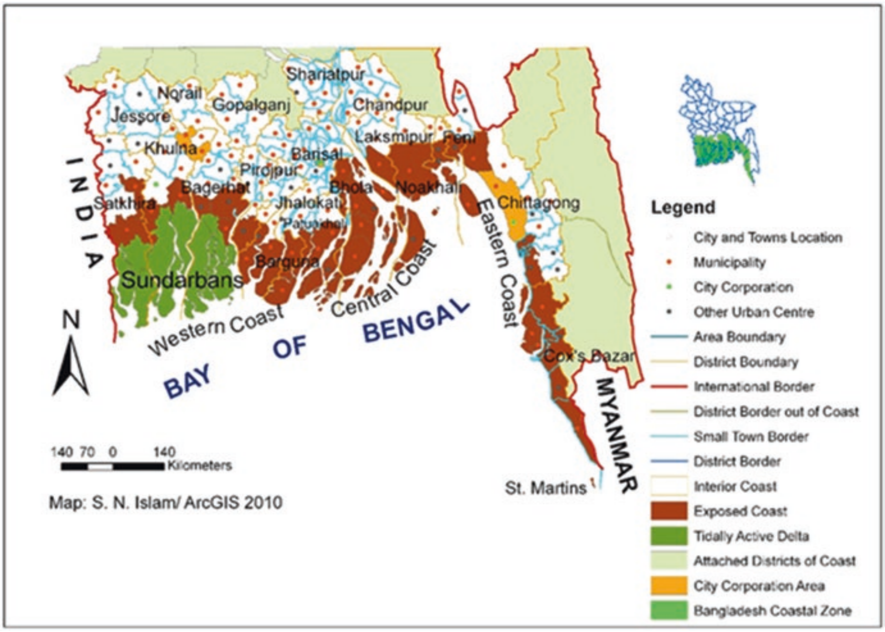


Fig. 6.2 Coastal morphology and ecosystems in Bangladesh (Source: Islam et al. 2016)

of mouth of Hoogly- Bhagirathi river in the west to the mouth of the Padma-Meghna river in the east (Figs. 6.1 and 6.2) (Islam 2016). The coastal area of the delta is confined roughly in an area between longitudes 80° 0′ E to 91° 0′E and 21° 30′N to 22° 30′N latitude. The coastal area of Bangladesh is deltaic active and hydrologically dynamics. The coastal areas of Bangladesh have been defined and categorized by Coastal Area Resource Development and Management Association (CARDMA) in three different types of coastal regions on the basis of physiographic characteristics. These three divisions are: The Eastern Region, The Central Region and The Western Region (Figs. 6.1 and 6.2). The eastern coastline extending from the Big Feni River to Badar Makam along Chittagong is regular and unbroken and protected along the sea by mud flats and submerged sands (ESCAP 1988; FAP-4 FPCO 1999; Halcrow 1993; Goodbred and Kuehl 2000). The Cox's Bazar sand beach about 145 km long is part of this coastline (Fig. 6.2).

The central region runs east from the Tetulia River to the big Feni River estuary and including the mouth of the combined flows of the Ganges-Brahmaputra-Meghna Rivers (Akter et al. 2010). This is why this region is characterized by heavy sediment input, formation of new *chars* and river bank erosion and accretion. The western region covers the coastline westward from the Tetulia River to the international boundary located at the Hariabhanga River (Figs. 6.1 and 6.2). This region is mostly covered with dense mangrove forests with reduce the River bank erosion. The rivers of the region are mostly stable; land accretion does not occur massively (Jalal 1988).

These three categories are defined three types of divisions based on the physiographic and ecological characteristics such as;

Atlantic Type: The western region is part of the Ganges deltaic floodplain. The Sundarbans is largely covered by mangrove forests. The western region is known as Atlantic type coastal region in Bangladesh. This western part of coast is more stable than other part of the coastal region; its character is like the Atlantic characteristics (Figs. 6.1 and 6.2).

Central Region: The Ganges-Brahmaputra-Meghna (GBM) River systems fall into the Bay of Bengal through the Meghna estuary. The Meghna estuary is located in the central region of the coastal Bangladesh. The central region is more unstable and most vulnerable. Coastal char formation, river erosion and accretions are the regular activities and character of region (Figs. 6.1 and 6.2).

Pacific Type: The eastern region of Chittagong specially a narrow strip with a long sandy beach in Cox's Bazar, known as Eastern Region. It is a long strip until Teknaf and St. Martins Coral Reef Island. The eastern part of Bengal coastal region is hotspot of biodiversity and displaying coastal narrow floodplain and mountainous characters. This eastern region has Pacific characteristics (Figs. 6.1 and 6.2).

The GBM system is discharged through the low-lying area of the central coastline, heavy sediment at inputs from the rivers result in a morphologically dynamics coastal zone. The central region has deltaic characteristics (Pramanik 1983; Miah 1989). The coastal zone is highly affected by high spring tides when water is saline during the dry season. The coastal areas are highly prone to cyclone induced storm surges that bring about the most catastrophic damage here (Chowdhury 2009).

The geology of Bengal basin began 130 million years ago. The initial breakup of Gondowana land possibly occurred in early Cretaceous (130 million years ago) with separation of India from combined Australian and Antarctica, the sea floor was created in what is now the Bay of Bengal (Anwar 1988). The coastal morphology is a characterized by different features like vast networks of rivers and channels, heavy water discharge carrying sediments, many islands in the coastal channels, strong tidal waves, cyclone and surges, physiographically funnel shape and Swatch of No Ground is located in the continental shelf about 24 km south of the Bangladesh coast (Figs. 6.1 and 6.2). There is a coastal mangrove landscapes and wetland ecosystem in the south western region of Bangladesh coast. Beside there are some other delineate of the coastal region are high salinity, tidal waves, influence of shrimp cultivation, and administrative management of the coastal region.

The GBM Rivers discharge is 1.5 million m^3/s during the peak period (Hasan and Mulamoottil 1994). The Ganges-Brahmaputra-Meghna (GBM) rivers system is the largest river systems in the world which is passing through Bangladesh on its way to the Bay of Bengal and carries sediments 1.8–2.4 billion tons (Nishat 1988), These are subject to coastal dynamic process generated by the river flow, tide and wide actions leading to accretion and erosion in the coastal area of Bangladesh (Nishat 1988).

6.5 Degradation of Coastal Wetland Ecosystems in the Ganges-Brahmaputra Rivers Deltaic Region

The coastal zone is a large and diverse area comprising a rich array of social, economic and environmental resources (Iftekhar 2006). The continental shelf of Bangladesh coast is about 66,000 km^2 and the coastal water is very shallow less than 10 m depth covering about 24,000 km^2 and (up to 200 m depth) covers about 70,000 km^2. The area of EEZ is about 1, 64,000 km^2 which is larger than the land area is 1,47,570 km^2 of the country (Islam 2016; Islam et al. 2016). The sector contributes 22% of the total animal protein intake of the country. The Bangladesh coastal zone also provides a natural and cultural heritage, the Sundarbans natural world heritage site, indigenous life style, and tangible and intangible cultural heritage and historic events. It contains natural resources that are a natural inheritance of immense economic, cultural and intrinsic value. Coastal communities enjoy the coastal resources such as; mangrove forest, indigenous cultural events, sand shore beaches, sun and the eco-tourism in the Sundarbans natural heritage site. There are lots of cultural activities in the coastal region which are dependent on coastal environment and its elements (Islam and Gnauck 2007a, b).

Damage of coastal ecosystems poses a direct threat to human survival in the coastal region in Bangladesh. This is primarily because coastal resources are an important source of both food and income and critical for business through ports, industrial development and coastal-marine biodiversity resources (Fig. 6.2). In recent years coastal area is becoming attractive for tourism and recreation purposes. Fish and sea food are primary sources of animal protection in the coastal region (Tan et al. 1997; Islam and Gnauck 2007a, b). In the coastal region of Bangladesh where shrimp farms are densely concentrated, such natural disasters have coast thousands of lives. The destruction of mangroves also alters the regional water system and coastal habitats, affecting fisheries coastal ecosystem balances and climate stability.

All of these effects reduce the fertility of land located in nearby areas (Biksham and Andrea 1996). It is estimated that 90% of all marine organisms spend some portion of their life cycle within mangrove system (Adeel and Pomeray 2002). The Sundarbans mangrove forests furnish ecosystem services made all the more valuable by climate change. They protect coastal communities from cyclone and storm damage, and this function may become even more important as climate change intensifies. Globally, mangroves are being cleaned or damaged at the alarming rate of 1–2% annually and in Bangladesh case it is the same. The Chakoria Sundarbans mangrove forest of Cox's Bazar in Bangladesh already cleaned up one decade ago which is occupied by shrimp culture and looks like a saline desert. The Challenge of coastal environment and water resources management in Bangladesh is to identify the trends impacting on coastal resources, balance the multiple and competing demands and responds with strategies to achieve sustainable management of the coast.

6.5.1 Changes of Coastal Ecosystem and Biodiversity

The salinity investigation results shows that the south west Bengal coastal regions are carrying the highest rate of water salinity which is unbalancing the coastal ecosystem and ecologies. According to salinity approximation this high rate is harmful to for urban biodiversity as well as for urban drinking water (Fig. 6.3). The Fig. 6.4 demonstrates the water salinity intrusion trends in the south west region of the Ganges deltaic region, where four major cities and 136 small towns are located in the coastal region in Bangladesh. All most all the towns are effected through salinity intrusion and sea level rise impacts in the region. Therefore, the investigation result of salinity modelling in the South west coastal deltaic regions are under threat for ecosystems and coastal urban ecosystem goods and services. The coastal mangrove and agro biodiversity loss is a common scenario in the Ganges-Brahmaputra-Meghna Rivers deltaic region between Bangladesh and India (Chowdhury and Haque 1990). The Fig. 6.3 demonstrates the scenarios of the coastal mangrove wetlands region in the Sundarbans in Bangladesh. The quality of wetland water and soil are rapidly degrading due to high saline water intrusion and anthropogenic influences (Fig. 6.3).

Total number of employed to be about 600,000 people for 6 months. Beside this 120,000 tourists visit the Sundarbans heritage site yearly, 50,000 people work daily.

The mangrove reduction rate is 45% in both countries. Deforestation and land-cover is changing due to shrimp farming, salt farming, agricultural land extension, urbanization extension and settlement development are adversely affects coastal fish production and leads to a loss of agrobiodiversity and coastal floodplain biodiversity and of livelihood to cover 3.5 million people who dependent on natural resources in the coastal region in Bangladesh (Anon 1995). The mangrove wetland

Fig. 6.3 The anthropogenic influences on coastal wetlands in the Sundarbans mangrove in Bangladesh (Source: Islam and Gnauck 2007a)

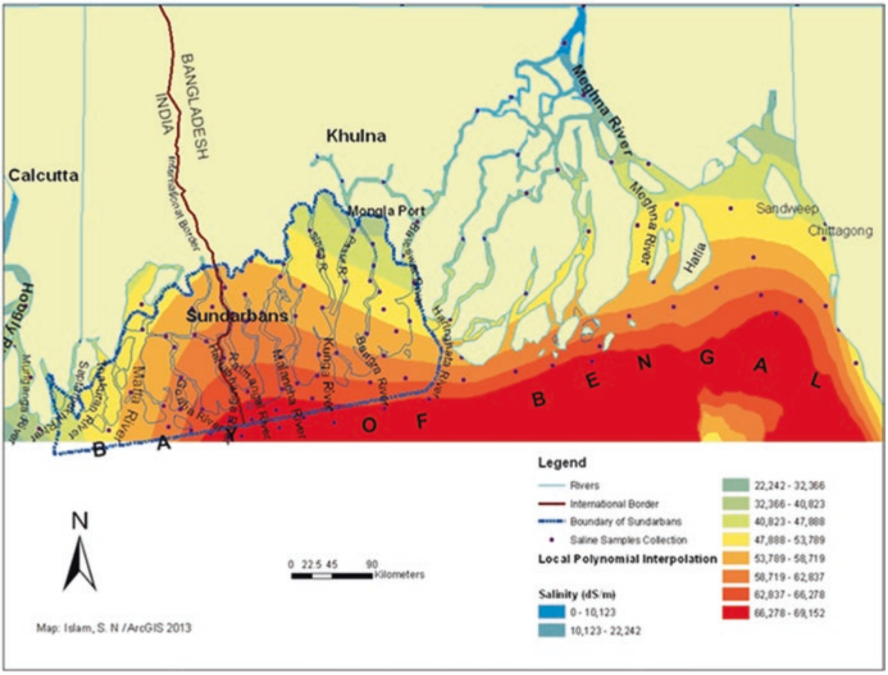


Fig. 6.4 The salinity intrusion pattern in the Bengal coastal region (Source: After Islam 2014)

ecosystems are dependent on the water and soil salinity. Almost all of the mangrove forest need freshwater supply from the upstream. In the Sundarbans Ganges deltaic coastal region the two potential rivers such as the Passur-Mongla and Chunar-Munchigannj are carrying the high rate of salinity intrusion. The Fig. 6.4 shows the high salinity intrusion trends in the coastal wetland region. The salinity model also demonstrates that the salinity trends is much higher in the south western region of the Sundarbans mangrove wetlands regions. The salinity rate was 42,000 dS/m in 2003, whereas in 2010 the salinity rate is 53,000 dS/m in the Passur-Mongla river point. On the other hand, the south western corner is showing the highest rate of water salinity in 2010, which is over 53,000 dS/m (Ferdra and Feoli 1999).

The coastal wetland ecosystems have been recognized as a driving force for biodiversity conservation and coastal urban socio-economic improvement (Nishat 1988; Ahmed et al. 2008). In the Ganges deltaic region of Bangladesh more than 60% people are dependent on agricultural crops production (Fig. 6.4). At present 30% of agricultural productions are reduced in the coastal region due high to medium level of salinization. The salinity penetration in the upstream areas of the coastal zone is one of the main obstacles to maintenance of water quality for drinking, irrigation and fisheries purposes (Islam and Gnauck 2009a).

Sixty six different species of mangroves growing in Sundarbans where 70 species has recorded in the world, 12 species of plats and animals already vanished and

Table 6.1 The coastal land area affected by high salinity intrusion (in 1973 and 2000)

Districts	1973 (ha)	2000 (ha)
Patuakhali	115,000	139,000
Barguna	103,000	104,000
Bhola	40,000	93,000
Pirozpur	20,000	28,000
Barisal	8,000	10,000
Jhalokhati	1,500	3,000

Javan rhinoceros, Single horn rhinoceros, Water buffalo, Swamp deer, Mugger crocodile, Gaur and Hog deer are extinct animals (Table 6.1).

In the Ganges-Brahmaputra rivers deltaic floodplain alone approximately 2.1 million ha of wetlands have been lost due to flood control, drainage, and irrigation development (Khan et al. 1994), therefore, coastal urban wetlands biodiversity is facing serious challenges from salinity intrusion, environmental changes and anthropogenic impacts (Sarkar 1993; Sarker et al. 2003; Nair 2004; Ahmed et al. 2008). The wetlands in the coastal region includes rivers, estuaries, mangrove swamps, marsh (haor), oxbow lake (baor) and beels, water storage reservoirs, fish ponds, and some other lands are also facing the similar environmental problems (Milliman et al. 1989; Khan 1993; Hughes et al. 1994; Gopal and Wetzel 1995; Islam and Gnauck 2008). In such problematic complicated situation the Smart-use of natural resources and salinity desalination processes can solve the coastal floodplains biodiversity and ecosystem problems in the Ganges-Brahmaputra coastal surface areas in Bangladesh.

6.5.2 *Threatened Coastal Deltaic Agricultural Crops Production*

The coastal region of Bangladesh is very important for the country because of natural resources, newly formed land area and agricultural and fisheries resources. Total 47,000 km^2 Land area is are declared as coastal region where agriculture and aquaculture are the most popular and potential agro crops and food production of the country. It is covering 34% of the country's food demand and a major portion of this 34% is covering from the coastal urban region. The major role of wetland are nutrient retention/removal, support for food chains, fisheries production, habitat for wildlife, recreation, natural heritage values, biomass production, water transport, biodiversity presentation and micro-climate stabilization. An in-depth analysis on this particular issue leading to coastal urban natural resources and biodiversity degradation in the coastal urban regions of Bangladesh has been carried out.

The Figs. 6.5 and 6.6 shows the water and soil salinity intrusion pattern in the south western Ganges deltaic region of the Bengal coast. There the salinity trend is imposing in the upstream areas from south direction to north direction. In the north

Fig. 6.5 The mangrove vegetation and biodiversity in the coastal wetland areas in Sundarbans region in Bangladesh

direction most part of the agricultural lands are converting from agricultural crops to shrimp cultivation due to high salinity intrusion. As a result the soil fertility of agricultural lands are losing its fertility.

The major cities like Calcutta is still located in the upper north and Mongla, Bagerhap, Sathkhira, Khulna cowns are located under this salinity threshold line (Fig. 6.6). On the other hand most of the coastal towns in the deltaic region are also affected due to high salinity intrusion. The traditional urban vegetation and crops are also changing due to salinity intrusion in the urban areas of the Bengal coastal region. In Fig. 6.6 the red line illustrates the salinity thresholds line for agricultural crops and vegetation productivity.

The threshold line which is indicating that the contour of exceeded threshold value (43,220 dS/m), which is harmful and threat for normal agricultural crops production and coastal urban vegetable production as well as high threat for safe drinking water availability in the coastal towns in Bangladesh. The coastal region of the Bengal delta of Bangladesh which is declared as Bengal coastal region is 47,000 km^2 is now under threat for agricultural crop production and urban biodiversity conservation. As well as the present condition is penetrating the coastal urban food security when agricultural productivity is gradually reducing in the coastal region in Bangladesh.

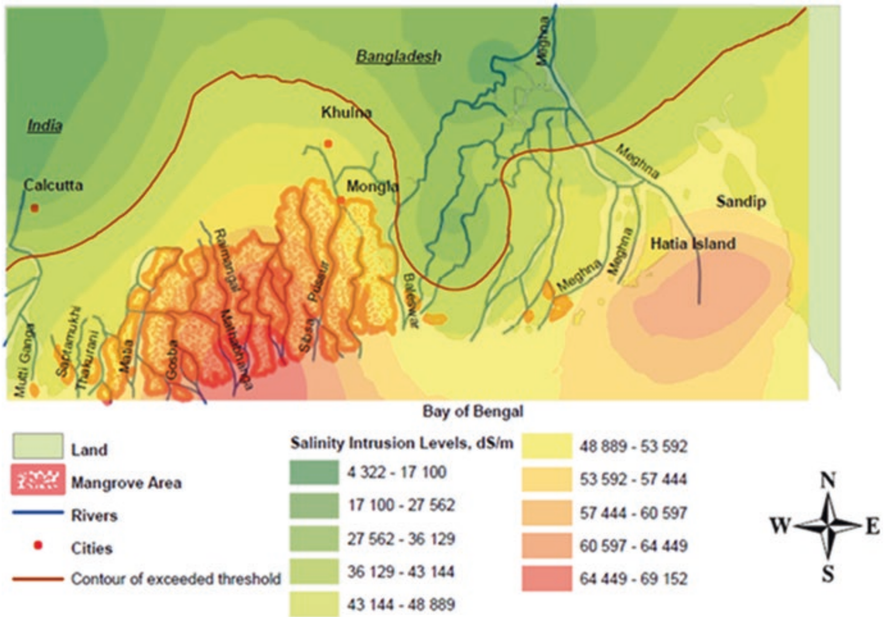


Fig. 6.6 The lower Ganges-Brahmaputra-Meghna deltaic coastal wetlands region is affected by salinity intrusion (Source: Islam 2014)

6.6 Coastal Wetland Resources Management Strategies

National water policy in Bangladesh (1999) also gives due importance on research and development and in article 3 the objectives of national water policy is to develop a state of knowledge and capability that will enable the country to design coastal water resources and wetland management plans by itself with economic efficiency, gender equity, social justice and environmental awareness. Article 4.15 Research and information management where strengthen and promote the involvement of public and private research organizations and universities where some specific objectives should be ensure such as; Appropriate technologies innovation and implementation, Develop and promote coastal wetland and water resources management techniques and Produce skilled professionals for water resource management.

The coastal wetland resources are totally dependent on the upstream fresh water supply and availability in the coast, therefore some potential issues should be incorporated in national water policy in Bangladesh. If the present coastal wetland resource management strategies analyses and compare with the national policies, it will show the real coastal wetland resource scenarios. There are 257 rivers in Bangladesh and they have important role to supply fresh water to the coastal areas (Islam and Gnauck 2007a). The Ganges-Brahmaputra-Meghna Rivers carry almost 6 million m^3/s of water and 13 million tons of suspended sediment per day during flood season to the Bay of Bengal; it is three times the amount of borne by the Mississippi River (Coleman 1969; Anwar 1988).

It is estimated that some 1.5–1.8 billion tons of sediment is denounced in the Bay of Bengal per year (Nishat 1988; Rahman 1988) but Anwar (1988) and Jabbar (1979) mentioned that the mighty rivers Ganges, Brahmaputra and Meghna transports about 2.4 billion tons of sediments annually to the Bay of Bengal. Curray and Moore (1971) show an enormous sub aqueous delta beneath the Bay of Bengal formed by the sediment derived from the Ganges-Brahmaputra-Meghna Rivers, the Bengal Deep Sea Fan, and the largest deep sea fan in the world. Sediments deposited on top of the upper most peat are fluvial controlled. These are the sediments deposited by the Ganges-Brahmaputra-Meghna rivers system (Islam 2001). Erosion and accretion are common phenomena in the coastal zone, where nearly a billion tons of sediment are brought by the rivers. About two thirds of these sediments are discharged into the Bay of Bengal, while the rest contributes to the formation of new land and islands (Iftekhar 2006). However river bank erosion is a severe problem. The annual rate of erosion in the Meghna river estuary alone is around 3199 ha/year (Islam 2016). Aspects of the sedimentary history of the Bengal Basin since Ca. 9000 years BP can be interpreted from data provided in this study.

It is undoubtedly true that the Ganges-Brahmaputra-Meghna system has been carrying sediments from upstream and depositing them in the basin, but their depositional environment needs to be considered (Islam 2001). The Fig. 6.7 shows a model of integration of coastal wetland water resources and ecosystem services in the coastal region. Water related sectors such as agriculture, land use, water navigation sector, industrial sector, mangrove wetland ecosystems, forest sector, fisheries and livestock and coastal community livelihood are directly dependent on coastal wetland resources (Fig. 6.7).

Bangladesh produce with 1 million tons fisheries products, which is more than twice Indian aquaculture production per capita, Nepal, Pakistan and Sri Lanka, lag far behind. In the coastal Bangladesh aquaculture is emerging as a prime rural industry, contributing to employment, food security, poverty reduction and export earnings. Export of aquaculture products earn Bangladesh half a billion dollars per year, making them the country's second-biggest earner of foreign exchange after textile. All these water resources sectors and sub sectors are inter-connected for their own sectoral development and management. Therefore coastal water resources sector is a potential sector in the coastal region and it is playing as a driving force of sustainable development of the coastal region of Bangladesh.

In most development, water resources management and sustainable livelihood discourses need for integration at different levels are recognized and emphasized (Rahman 2003). The soil of the entire coastline is generally sediment deposit of the GBM and other rivers. Such soils are fertile but recently it has been affected by high salinity intrusion. It has been estimated almost half of the coastal arable land can be classified as saline zone, within this saline affected areas such as the greater Khulna and Patuakhali districts are being the most affected region in the coastal region of Bangladesh (Jalal 1988).

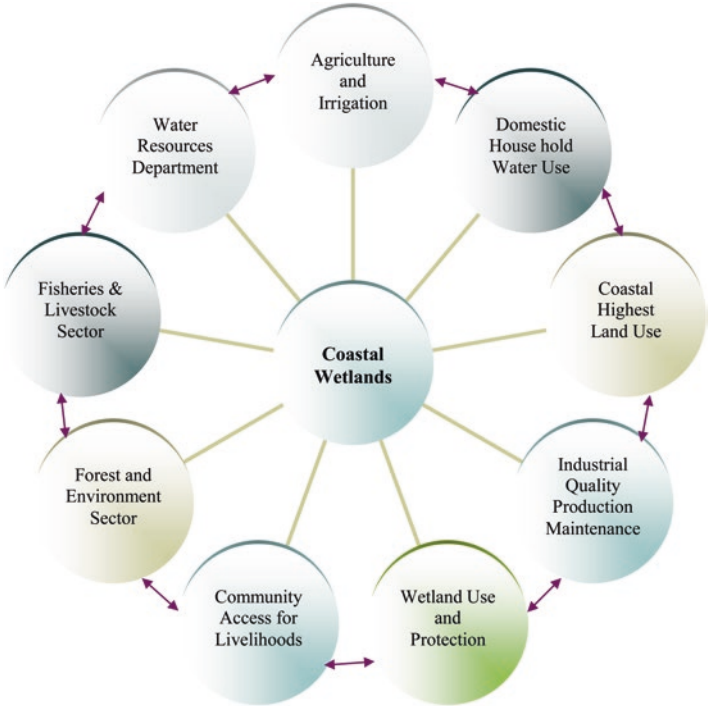


Fig. 6.7 Interlinkage of coastal wetlands, water resources, ecosystem services and livelihoods (Source: Islam et al. 2016)

6.6.1 Present Threats of Coastal Water Resources

The Bangladesh coast constitutes the low-lying landmass of the Himalayan river basin ecosystems (Iftekhar 2006). The formation of the delta plain commenced some 12,000 years ago when the sea level started to rise rapidly due to the melting of ice sheets to the north. About 6000 years ago, sea level rise slowed down and coastal plains started to form (Islam 2001; Iftekhar 2006). The large rivers especially the Brahmaputra, Meghna and the Ganges are playing an important role in the coastal region. Biogeographically, the coast is part of the Bay of Bengal large marine ecosystem (Nishat et al. 2002). The Ganges fresh water plays a vital role in coastal economy, floodplain fertility and keeps the balance of mangrove wetland ecosystem services in south western coastal part of Bangladesh (Islam 2009a).

The Ganges River receives over 60% of its discharge from its tributaries. The Gomti, Damodar, Yamuna Rivers, Mohananda are the major tributaries of Ganges Rivers in the Indian portion (Helmer and Hespanhol 1997). The main distributaries of the Ganges River in Bangladesh are the Baral, the Gorai, the Arial Khan, the Bhairab, the Mathabhanga, the Kumar, Chitra and the Ichamati River (Miah 2001). Their average annual combined discharge into the Bay of Bengal is 100,000–

140,000 m^3/s carried by Ganges-Brahmaputra-Meghna (GBM) Rivers (EGIS 2000). As a whole the Brahmaputra (Jamuna) River carries water flow 60,000–100,000 m^3/s. The minimum water flow of the Meghna River is 3750 m^3/s (Fig. 6.6). The Ganges River is the major source of silt deposition and delta formation in the Bay of Bengal (Joseph 2006; Islam and Gnauck 2008). The Brahmaputra carries every year, about 600 million tons of sediments, with alluvial sand constituting large proportion (FAP 24 1996). The sediment is deposited as dunes in the river bed, which is about 350, in some cases up to 600 m long and they can move very fast as 17 m/s (FAP 24 1996; EGIS 1997).

The Meghna Delta is the main output of the Ganges, the Brahmaputra and the Meghna to the Bay of Bengal (Chowdhury 1990). The sediment load is extremely high, with suspended sediment load during flood stage reaching as high as 13 million tons per day (Coleman 1969). The strong south Asian monsoon and high Himalayan source area supports one of the world's largest riverine sediment loads (1.8–2.4 billion tons annually) for the Ganges-Brahmaputra-Meghna dispersal system (Anwar 1988; Umitsu 1993; Goodbred and Nicholls 2004).

The study finding has asserted that India's diversion of fresh water had resulted in a loss of biodiversity and rice output of 236,000 metric tons in 1976 just after construction of Farakka Barrage in India, this loss figure in 2007 became three times more than the loss of 1976 (Islam and Gnauck 2009a). The previous study result of CEGIS then EGIS (Environmental Geographical Information System) showed that the water quality of Sundarbans region has degraded in dry season (February–June), when 60% water of Sundarbans rivers are poor quality where EC dS/m is 5532 and 40% is good quality and the EC dS/m is 2766 (Islam and Gnauck 2009a). Coastal water pollution such as oil spills are new threat and could cause immense damage, especially to aquatic fauna, seabirds, mangrove biodiversity and coastal agricultural cropping systems. The yearly natural calamities, global warming and its impacts are new challenge and threats for coastal food security and agro-biodiversity (Islam and Gnauck 2009b).

As per an earlier soil investigation conducted by SRDI in 1970, the soil salinity was mainly found in the Ganges tidal floodplain of the coastal region. The Ganges River floodplain and the peat basins areas were classified as a non-saline. Soil salinity occurred south of Khulna and Bagerhat districts. The surface and subsurface salinization is very common, with saline water intruding into freshwater aquifers. Each year, a new area of 146 km^2 is affected by salinity, where a significant reduction in biodiversity is observed (Dutta and Iftekhar 2004). Salinity range was between 8644 and 17,288 dS/m. A rise of soil and water salinity has been noticed in 1975 when Farakka Barrage constructed on the Ganges River and withdrawal fresh water from the basin. At present soil salinity level has recognized at south of Khulna and Bagerhat town ranges between 17,288 and 32,415 dS/m during the dry season (November–May). Soil and water salinity is rapidly increasing in the coastal region, currently, river water salinity moves up as far as Kamarkhali River port in Magura district; (SRDI 1997, 2000; EGIS 2000; Islam and Gnauck 2008). The salinity trends is higher in the Sathkhira, Khulna, Bagerhat, Borguna, Jhalokhati, Potuakhali, Bhola and the southern part of Noakhali districts. The trends are higher in the south western region of Bangladesh and gradually eastern region is comparatively less saline intrusion in the coastal region.

At present, tidal waves, surges and coastal flooding are common annual natural phenomena in Bangladesh (Islam 2001). Most coastal regions show uneven sediment compaction which is controlled by differential sedimentation rates, sediment composition, water content, depth of overburden sediment layers and tectonic activities (Greensmith and Tucker 1986; Islam 2001). A national strategy for coastal resource management in Bangladesh is necessary to identify and analysis the problems and opportunities and future development. A study of ESCAP (Economic and Social Council for Asia and the Pacific) in 1988 and GOB (Government of Bangladesh) has referred five sets of constraints to the development of a strategy for coastal water and wetland resource management in Bangladesh such as;

- Policy making and strategies implementation
- Planning for coastal water and wetland resources maintenance
- Integrated coastal wetland resources management
- Coastal wetlands and marine resources uses and sustainability
- Local environmental and coastal ecological perspective and
- Lack knowledge of coastal wetland environment and better understanding (Jalal 1988).

The water and soil salinity penetration in the upstream area of the whole coastal zone is one of the main obstacles to maintenance of wetland water quality for drinking, irrigation and fisheries purpose. The shortage of upstream fresh water supply to the coast and tidal waves are facilitating to penetrate salinity intrusion (SRDI 1997). In the coastal belt there are 81,000 ponds are scattered locations which are the main source of fresh water supply for the domestic consumption in the rural areas. The quality of wetland's water is the major constraint in water supply (Jalal 1988). The coastal wetland lands are fertile for shrimp cultivation because the water is mostly saline. A total of nearly 150,000 ha were under shrimp cultivation of the region, representing 11.8% of the land in the region respectively. Almost 60% of the biomass comes from agricultural residues in the coastal region. Around 200 industries in industrial zones of Chittagong discharges their untreated wastes to the Karnafuli River and 190 industries are located in three industrial zones of Khulna, discharging untreated toxic waste into the Bhairab River which is carrying into the coastal region. There are 50 coastal towns are located in the coastal region and are affected by high salinity intrusion in drinking water and arsenic contamination in ground water aquifer.

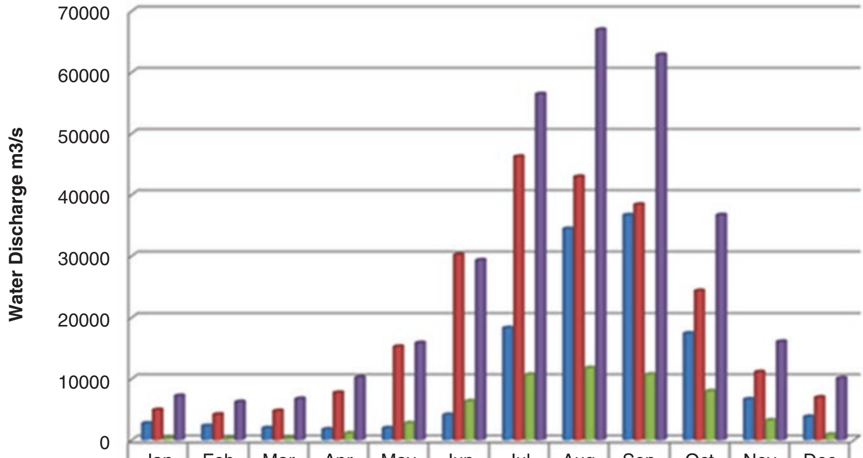
Beside these two international ports are situated in the coastal region. The ships at the ports are spillage of crude oil and discharge of ballast water, sewage from ships and other activities are causing severe coastal water pollution of the coastal marine ecosystems. The tourism activities in the coastal region of Bangladesh are also distracting water quality in the coastal region. As a whole the Bangladesh coastal region is the most vulnerable coast in the world (Jalal 1988). There is no integrated management plan for the coastal natural resources or for the coastal communities. The coastal land cultivation, paddy and shrimp production policy is still dependent on the local initiatives. Fresh water flows, drinking water supply, water for irrigation and fishing are severely affected. Considering the present water resource management condition a newly policy framework guideline is essential to protect and maintenance the coastal water resources in Bangladesh.

6.6.2 Sea Level Rise and Its Impacts on Coastal Region

Climate change and sea level rise, induced by global warming, also compromise the ecological stability of the coastal zone; in sum, due to various natural and anthropogenic factors, the natural resource base of the zone is declining. Failing ecosystem productivity further degrades the quality of life of the local population (Dasgupta 2001). Relative Sea-Level Rise (SLR) movement has an immediate and direct effect on the coastal inter tidal ecosystems, particularly on vegetation. Arise of relative SLR decreases the influences of terrestrial processes and increases the influence of coastal marine processes (Islam 2001). The world great deltas are among the most densely populated and most vulnerable of coastal areas are threatened by sea level rise (Broadus 1993). Global warming and sea level rise and vulnerability of coastal wetland ecosystems are factors that have to be considered the long term management strategy for dealing with the coastal mangrove wetland issue. The impacts of climate change in any given region depend on the specific climatic changes that occur in that region. Local changes can differ substantially from the globally averaged climate change (Harvey 2000). The global warming and climate change a predicted sea level rise, accelerated by global warming will cause a further 'Squeezing' of the natural tidal land. In Bangladesh case it has been projected by IPCC (2007) and MoEF that 3 mm/year sea level rise which will occurs before 2030 and 2500 km^2 land (2%) will be inundated. About 20% of the net cultivable area of Bangladesh is located in the coastal and offshore island (Fig. 6.9).

A very recent study on coastal area in Bangladesh by Wolanski et al. (2009) shows that the mean tidal level at Hiron Point is showing an increase of 4.0 mm/year which is higher than the global rate. Soils in this area are affected by different degrees of salinity (Rahman 1988). About 203,000 ha very slightly, 492,000 ha slightly, 461,000 ha moderately and 490,200 ha strongly salt affected soils are assessed in south western part of the coastal area (Fig. 6.9). The climate change impact issue is a new threat for the coastal area of Bangladesh (Fig. 6.9). In the Sundarbans case, sea level rise would result in saline water moving further into the delta which would be the major threat for mangrove and coastal wetland ecosystems (IECO 1980). The fate of the Sundarbans with different sea level rise the potential impacts on environment if 10 cm SLR will inundate 15% of the Sundarbans and with 1.5 m SLR about 17 million (15%) of the population will be affected and will have to be displaced or homeless, whereas 22,000 km^2 (16%) land will permanently be inundated (Fig. 6.9).

The Figs. 6.8 and 6.9 shows that with 3 m SLR would be worse scenario for Bangladesh when almost one third of land could be inundated by saline water. The



	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Ganges	2744	2336	1984	1781	1971	4136	18324	34466	36699	17439	6682	3819
Brahmaputra	4944	4209	4786	7752	15280	30225	46242	42962	38418	24335	11137	6980
Meghna	370	264	362	1106	2778	6348	10691	11785	10722	8008	3240	919
Padma	7243	6242	6756	10220	15867	29337	56466	66980	62889	36716	16078	10104

Fig. 6.8 Major rivers annual water discharge flow in the Ganges Deltaic region (Data Source: EGIS 2000; Miah 2001; Goodbred and Nicholls 2004; CEGIS 2010)

reduction rate of mangrove areas will from 50 to 75% which would be more harmful for coastal ecosystems in the estuaries (IPCC 2007). Besides, other environmental problems will arise in the coastal belt such as; water pollution and scarcity, soil degradation, deforestation, solid and hazardous wastes, loss of bio-diversity estuary landscape damage and river bank erosion which will create a lot of new challenging problems for human livelihood in the coastal region. In such situation it will further create an unstable agricultural crop production, damaging fisheries and livestock and food security in the coastal riverine islands in Bangladesh (MoWR 2005). Therefore the climate change impacts on coastal region such as on the estuaries in the Sundarbans will create new threats for estuaries ecosystems and landscapes.

6.6.3 Changes of Biodiversity in the Bengal Coastal Region

The salinity investigation results shows that the south west Bengal coastal regions are carrying the highest rate of water salinity which is misbalancing the urban ecosystem and ecologies. According to salinity approximation this high rate is harmful to for urban biodiversity as well as for urban drinking water (Fig. 6.9). The Fig. 6.9 demonstrates the water salinity intrusion pattern in the south west region of the Ganges deltaic region, where 4 major cities and 136 small towns are located in the coastal region in Bangladesh. All most all the towns are effected through salinity

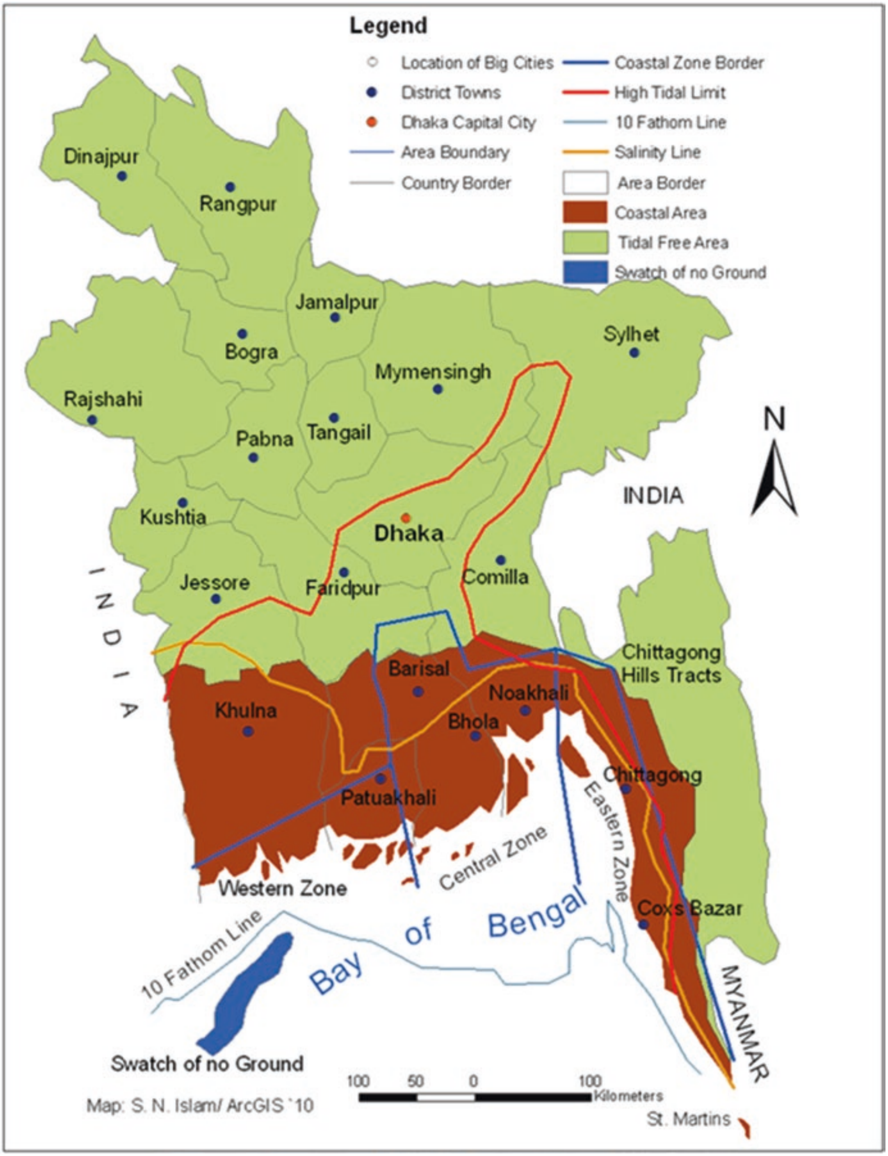


Fig. 6.9 Climate change impacts in the coastal region of Bangladesh (Source: Akter et al. 2010)

intrusion and sea level rise impacts in the region. Therefore, the investigation result of salinity modelling in the South west coastal deltaic regions are under threat for ecosystems and coastal urban ecosystem goods and services.

The coastal mangrove and agro biodiversity loss is a common scenario in Bangladesh and India. The mangrove reduction rate is 45% in both countries. Deforestation and land cover is changing due to shrimp farming, salt farming, agricultural land extension, urbanization extension and settlement development are adversely affects coastal fish production and leads to a loss of agrobiodiversity and coastal urban biodiversity and of livelihood to cover 3.5 million people who dependent on natural resources in the coastal region in Bangladesh (Anon 1995).

Moreover these are support rich biodiversity of flora and flora, contributing substantially to the socio-economic life of millions of people of rural areas of different remote areas of the world by providing opportunities of employment, food and nutrition, fuel, fodder, transportation and irrigation (Nishat 1988). The coastal forest and wetlands resources in Bangladesh has suffered drastically from the impacts of growing human population and anthropogenic activities on natural resources. The water resources and mining resources are similarly using for economic development. Furthermore natural resources are recognized as a driving force for biodiversity conservation and coastal urban socio-economic improvement (Nishat 1988; Ahmed et al. 2008). In the Ganges deltaic region of Bangladesh more than 60% people are dependent on agricultural crops production. At present 30% of agricultural productions are reduced in the coastal region due high to medium level of salinization. The salinity penetration in the upstream areas of the coastal zone is one of the main obstacles to maintenance of water quality for drinking, irrigation and fisheries purposes.

In the Ganges-Brahmaputra-Meghna floodplain alone approximately 2.1 million ha of wetlands have been lost due to flood control, drainage, and irrigation development (Khan et al. 1994), therefore, coastal urban wetlands biodiversity is facing serious challenges from salinity intrusion, environmental changes and anthropogenic impacts (Sarkar 1993; Sarker et al. 2003; Nair 2004; Ahmed et al. 2008). The wetlands in the coastal region includes rivers, estuaries, mangrove swamps, marsh (*haor*), oxbow lake (*baor*) and *beels*, water storage reservoirs, fish ponds, and some other lands are also facing the similar environmental problems (Khan 1993; Hughes et al. 1994; Gopal and Wetzel 1995; Islam and Gnauck 2008). In such problematic complicated situation the Smart-use of natural resources and salinity desalination processes can solve the coastal urban biodiversity and ecosystem problems in the coastal surface areas in Bangladesh.

6.7 Approach for Coastal Wetland Ecosystems Management and Sustainability

The coastal water resource usages and maintenance is the fundamental management aspect for local community. The fringing of this study identified a number of environmental obstructions for proper use, reuse and maintenance of their coastal natural resources. Most of the developing countries have not yet developed the coastal water and natural resources management policies and strategies. Therefore, shortage of upstream fresh water supply, saline water intrusion and climate change impacts on the coastal regions are the big challenge for agricultural crop production

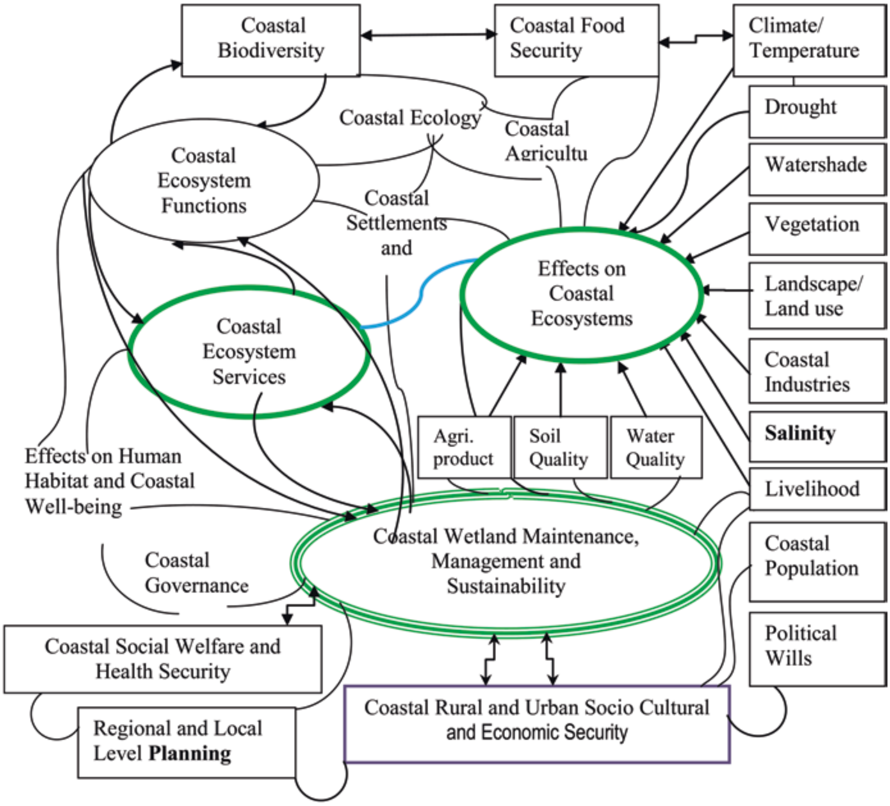


Fig. 6.10 Model of coastal wetland resources protection, management and sustainability

and maintain food security in the coastal region of Bangladesh. These preoccupations need to be recognized and each given due weight in order to integrate them into policy and decision making in coastal water resource management to fight against climate change impacts, food insecurity and poverty for ensure sustainable development for the coastal communities; which means development that meets the needs of the present without compromising the ability of future generation to meet their own needs. What is important is to carry out a programme for making the people aware of the impending dangers and to develop, along with them, methods of coastal adaptation and migration to promote water resources management and properly use for food security in the coastal region of Bangladesh using the vulnerability adaptation framework.

Figure 6.10 is demonstrating a local level planning approach for sustainable local level natural resources management. The model (Fig. 6.10) has prepared based on the present situation of coastal wetland resources scenarios in Bangladesh case. Considering the findings this long-term management plan for the sustainable use of coastal water resources is necessary and it urgently needed. This model could be implemented in the coastal region in Bangladesh as well as other coastal region which are similar characters like Bangladesh could be considered as well. Besides, some practical recommendations are as follows for better management of coastal wetland resources in Bangladesh. Therefore, applied research, awareness education, monitoring and evaluation are the key potential issues of a successful coastal wetland resources management and conservation.

- Increase salinity due to the lack of fresh water flows in the transboundary river basin has caused damage the vast area of coastal region and its ecosystems.
- Upstream rivers water supply should be ensured to the coastal region in Bangladesh coast. Whenever Ganges water supply is disputable problem between India and Bangladesh, therefore other rivers water and monsoon water harvesting and make a reservoir in the upstream area and supply fresh water to the south western coastal region could solve high salinity problem (Begum 1987).
- A sharp rise in water levels generally begins in May and continuously until July. Therefore, some potential steps should be taken into consideration for sustainable water resources management in the coastal region such as;
  - Monitoring of ground and surface water quality and
  - Modelling for its sustainable management of coastal wetland resources.

Coastal community management plays a major role and becomes the key to success in integrated coastal resources management. It has been realized that coastal natural resources relies on the active participation of the coastal communities in protecting, maintaining and conserving their surrounding coastal resources. Coastal community base management could be the best strategy in integrated coastal management that is the involvement of the coastal society, and stakeholder in implementation stage. The indigenous coastal community is ecologically friendly and is an important source of intellectual asset in management the coastal environment (Hidayati 2000). The community, as the prime users of the coastal wetland resources, through community involvement, local technology should be considered in protecting practices and maintaining wetland resources in the coastal region in Bangladesh.

6.8 Conclusions and Recommendations

Coastal wetlands can be considered as driving force for community social and economic enhancement. They have the ability to focus tremendous energy and to generate significant creative and economic betterment. In general, the country's coastal wetland natural resources as well as the ecosystems are degrading due to anthropogenic influences and natural calamities. Considering the scenarios the Ganges-Brahmaputra deltaic coastal zones of Bangladesh have great importance for the country's economy, industrial, ecological, socio-economic and cultural context. Moreover it supports rich biodiversity is contributing substantially to the livelihoods of millions of people through creating opportunities of employments. The climate change impact has added as additional negative impact on coastal wetlands as well as the local communities and stakeholders in the Ganges-Brahmaputra rivers deltaic region in Bangladesh.

The coastal region of Bangladesh encompasses about one fifth of the country's landmass and supports livelihood of 36.8 million people. The region is becoming more vulnerable due to natural and anthropogenic causes. Growing anthropogenic and climatic impacts have been found to put multifarious adverse impacts on coastal water resources, environmental and livelihood of the inhabitants of the coastal region. It is estimated that the requirement of the Ganges fresh water supply to the Sundarbans is 49 MAF and other rivers also supplying the similar quantities of surface water to the Sundarbans. Any endeavour through to be suitable for the wellbeing of the people as well as for the coastal environment, has to be viewed with options considering the socio-economic and cultural traditions of the society. Any particular adjustment to be included in the final plan ought to be within the technological capability of the people.

The study findings show that water and soil salinity intrusion is a severe threat for agricultural crops production, mangrove ecosystems and its services as well as coastal settlements and population. National and International political commitments and wills should be ensured properly. Therefore an integrated coastal natural resource management policies and guideline framework for livelihood is necessary and it is urgent for Ganges –Brahmaputra rivers deltaic region in coastal Bangladesh.

Considering the influence, problems, prospect and their impacts on socio-economic condition of the coastal community and on the ecosystems of coastal natural resources, the strategies and intervention in planning is necessary. There is a potential realization that effective coastal water resources management at local level is essential and appropriate management structure should be developed.

The study would suggest that most mangroves along the coast of Bangladesh would disappear under sea-level rise scenarios when rate of rise up to 1.5 cm year^{-1} have been predicted. Therefore an integrated coastal water resources management policies and guideline framework is necessary and it is emergency. The finding of this study could help to the policy and decision makers to prepare a guideline framework for coastal wetland resources protection, conservation and management in Bangladesh.

Besides the above elements and factors the following important issues should be included in the national coastal wetland resources management and conservation plan, which should be developed based on the changing tendency of ecosystems in the deltaic coastal region of Bangladesh; the following recommendations could be considered in making the policy framework for better management such as; For achieving the goal for sustainable coastal wetland water resources management plan the following steps should be undertaken and included in policy making;

- Increase coastal community socio-economic welfare and alleviate poverty for sustainable livelihood in the vulnerable coastal region. Provide alternative income generation opportunities to the coastal communities that should ecologically friendly and profitable.

- Increase public awareness concerning environmental training and education, particularly related to the importance of the sustainable use of coastal wetland and other natural resources. Provide information to the coastal communities concerning natural resources collection, properly uses and long-term protection measures.
- Coastal wetlands biodiversity maintenance in the region; mangrove ecosystem functions and services should maintenance in the coastal region for the better interest of coastal communities in the Ganges-Brahmaputra Rivers deltaic wetland region.
- To develop and trained up capacity building of the local community groups, local government, stakeholders, NGOs and national policy makers and planners those are involved in coastal resource management activities in the Ganges-Brahmaputra deltaic coastal wetland region in Bangladesh.

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Chapter 7
Handling High Soil Trace Elements Pollution: Case Study of the Odiel and Tinto Rivers Estuary and the Accompanying Salt Marshes (Southwest Iberian Peninsula)

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Abstract Salt marshes are being increasingly polluted by trace elements, and the design and implementation of management actions adapted to each particular situation are necessary. Salt marshes developed at one of the most heavy metal-polluted systems in the world, the Odiel and Tinto joint estuary, are threatened by high pollution levels, erosion and the invasion of the alien plant species *Spartina densiflora*, despite the high ecological values recognized by regional to international protection figures. Soft management on these marshes tries to preserve the equilibrium between conservation and decontamination. The ability of key native halophytes in the area to phytoextract or phytostabilizate trace elements has been taking into account. A local restoration project has resulted in a rapid recovery of the native prairies of low tidal marshes, dominated by *S. maritima*, becoming a promising tool to phytostabilize eroding areas in European marshes. These prairies also seem to stop the advance of the alien *S. densiflora* invasion and prevent erosion. On the other hand, areas invaded by *S. densiflora* are difficult to manage due to the acidity and pollution level of sediments preventing the establishment of any other plant species. Despite its invasive character, *S. densiflora* avoid at present the removal of highly toxic sediments and the trace element release to the food chain in this area.

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7.1 Introduction

7.1.1 A Brief View to Coastal Marsh Ecosystems and Its Significance for Conservation

Coastal salt marshes are complex ecosystem developing in the intertidal zone of middle and high latitudes worldwide, regularly flooded by tides and subjected to sedimentary dynamic of accretion. Developed within the last 8000 years (Milliman and Emery 1968), they are characterized by harsh environmental conditions, including flooding, sedimentation, high salinity and anoxia. This abiotic matrix imposes certain limits on organisms living in such habitats. A low vegetation type, mainly composed by salt-tolerant species particularly adapted to this environment (halophytes), grows in salt marshes, where the majority of species are perennials, with only few annuals, mainly pertaining to the Chenopodiaceae family (Ranwell 1972). Vegetation is generally distributed in relatively well defined belts or bands parallel to the tidal line, along the intertidal gradient defined by elevation, showing a typical vertical zonation leaded by the particular abilities of the species to establish and survive to the environmental constrains, mainly related with the degree of tidal influence (flooding frequency and permanence timing, nutrient availability, degree of anoxia and salinity, sedimentation rates and mechanical damage), and the associated level of accretion and soil maturation (Álvarez Rogel et al. 2001; Davy and Costa 1992; Olff et al. 1997; Ranwell 1972; Vince and Snow 1984). Nonetheless, biotic interactions, both positive and negative, also play a relevant role in determining plant species distribution in these systems (Bertness 1991; Castillo et al. 2008a; Costa et al. 2003; Pennings and Callaway 1992).

Coastal salt marshes provide notable ecosystem services to society. They play a relevant role as natural barriers to front coastal hazards such as storms, flooding and coastal erosion, by attenuating wave energy and storing floodwaters, and so decreasing the impact of extreme events such as hurricanes and tsunamis (Costanza et al. 2008; Gedan et al. 2011; Möller et al. 2014). They act as a sponge absorbing and sequestering nutrients, microbes and pollutants from runoff and riverine discharge, thus improving estuarine and coastal waters quality (Gedan et al. 2009; Mitsch and Gosselink 2007), and play a major role in the global carbon cycle by acting as carbon sinks (Chmura et al. 2003). Salt marshes are recognized for their importance in the aquatic food web and the cycling of nutrients in coastal waters, and they are crucial to global biodiversity maintenance, providing habitats for feeding, resting and breeding for many bird species, both migrator y and resident (Ferns 1992; Howe 1987; Hughes 2004; Laegdsgaard 2006), as well as for juvenile fishes, crabs, and shrimps (Shenker and Dean 1979; Zimmerman et al. 2000). The harsh environment in salt marshes to living beings imply a high level of specialization of the marsh species of fauna and flora, which contributes to the ecological value and vulnerability of these ecosystems. In addition, salt marshes have a recognized geomorphological, historical, scenic, cultural and socio-economic interest, and they are important locations for nature tourism and recreation, education and research.

At present, it is estimated that less than 50% of the world's original wetlands, among which are the saltmarshes, remain, showing a current loss of 1–2% per year (Bridgham et al. 2006; Mitsch and Gosselink 2007). Salt marshes are seriously threatened by both natural and human-induced drives, and changes in land use and land take, alteration of coastal hydrology, effects associated to climate change (increases in atmospheric CO_2, global warming, increases in storms intensity and sea level rise), biological invasions, eutrophication and pollution by trace elements from urban, industrial and mining areas have been identified as the main hazards for salt marsh conservation (Curado et al. 2010; Deegan et al. 2012; Gedan et al. 2009).

7.1.2 Soil Contamination in Coastal Salt Marshes

Coastal marshes are highly vulnerable to contamination by trace elements due to their location at the river mouths (Beeftink 1977; Williams et al. 1994), particularly in those cases where mining and industrial areas exist close to estuaries or upstream their associated rivers, since both nutrients and contaminants are transported through river systems. In addition, rivers and salt marshes pollution by trace elements is increasing in extent and number of affected areas (Handa and Jefferies 2000).

The capacity of trace elements to be transferred between soil phases under particular soil conditions is one of the main factors driving their behavior and bioavailability in each specific case. In particular, marsh sediments are characterized by fine size of grain, mainly composed by silt and clay, with a high capacity for adherence and retention of trace elements. In addition, both pH and redox potential are relevant drivers for trace elements solubility and thus bioavailability in this environment. It can be generalized that, in well-aerated (oxidizing) acidic soils, several trace metals (particularly cadmium and zinc) are easily mobile and available to plants, while metals are substantially less available in poorly aerated (reducing) neutral or alkaline soils (Kabata-Pendias 2004). In this regard, marsh sediments are considered a sink for trace elements. Very high metals concentrations in a reduced state may be contained in the anoxic zone, although they show a reduced bioavailability in comparison with oxidized soils of terrestrial systems (Kabata-Pendias 2004; Weis and Weis 2004). Nevertheless, acidification of sediments promoted by pollution from mining activities as well as by terestrialisation of marshes and accumulation of organic matter in the development of ecological succession favors the bioavailability of trace elements. In addition, plant species able to withstand high metal concentrations in soil are also able to influence the trace element mobility through their accumulation in their tissues or their immobilization in their rhizosphere, as it is further described in this chapter. On top of all, salt marshes are usually under other types of pressures, including human use. As a result, all these considerations make difficult the management of specific areas by standardized actions, and imply that each case may be particularly considered, beyond general common guidelines for ecological restoration.

7.2 Salt Marshes Developed in One of the Most Polluted Estuaries in the World

The rivers Odiel and Tinto jointly flow into the Atlantic Ocean in the central area of the Gulf of Cádiz, near the city of Huelva (37°15′–37°37′N, 6°57′–6°58′W, SW Iberian Peninsula, Spain; Fig. 7.1). The climate in this coastal area is Mediterranean warm-temperate with an influence from the Atlantic Ocean. Mean annual temperature is around 18 °C and mean annual rainfall is about 580 mm, with a pronounced period of drought from May to September (29 year record, 1974–2003; Gibraleón Meteorological Station, Huelva). The prevailing SW winds in the coastal frame bring 74% of the swell from the SW, with a medium-low energy regime (Rodríguez Ramírez et al. 2003), and the tidal regime is mesotidal and semidiurnal, with a mean range of 2.10 m and a mean spring tidal range of 2.97 m above the Spanish Hydrographic Zero (SHZ) (Castillo et al. 2000).

Fig. 7.1 Location of the Odiel and Tinto rivers joint estuary in the Gulf of Cádiz (SW Iberian Peninsula, Spain), considered one of the most polluted estuaries in the world. Odiel marshes are delimited by a discontinuous *red line*

Fig. 7.2 Tinto river (Huelva, Southwest Iberian Peninsula); the characteristic *red color* of waters is related with the high Fe content. In the picture, a dike crossing the river creates a water reservoir and cuts the tidal influence upland

This joint estuary is considered as one of the most heavy metal-polluted systems in the world (Nelson and Lamothe 1993; Sainz et al. 2004) (Fig. 7.2). The heavy metals content is the result of both the industrial activity occurring on the banks of the estuary and the mining activities taking place upstream, in the Iberian Pyrite Belt (IPB), an important metal-rich sulphide deposit. This latter source is responsible for more than 99% of the total metal content in the estuary (Pérez-López et al. 2010).

The Odiel Marshes develop in this estuary (Fig. 7.1), occupying some 1758 ha, being one of the largest areas of salt marshes in the Iberian Peninsula (Castillo et al. 2000) and the second most relevant wetland in Spain (Aranda and Otero 2014). They comprise a group of islands (Saltés, Bacuta, Enmedio, Marismas del Burro), channels where fishing and shell-fishing occurs, beaches and dunes, supporting a wide variety of habitats. A raised dike built in 1977 divides the south areas of the Odiel Marshes into two main zones, where the eastern part continues to perform rapid drainage into the joint estuarine channel, receiving a low sediment load, whereas sedimentation dominates in the western part, where the development of a sand spit prevent the drainage so standing water persist long after high tides (Castellanos et al. 1994). Marsh channels are used by boats, what has leaded to serious erosion problems due to the swell that this kind of traffic promotes. In addition, the American species *Spartina densiflora* Brongn. is invading a wide range of the marsh habitats, efficiently occupying the space and competitively displacing native species (Nieva et al. 2001).

Table 7.1 Habitat types present on the Odiel Marshes included in the habitats list of the EU Habitat Directive (1992)

Habitat Code	Habitat name
1140	Mudflats and sandflats not covered by seawater at low tide
1150*	Coastal lagoons
1310	*Salicornia* and other annuals colonising mud and sand
1320	*Spartina* swards (*Spartinion maritimae*)
1420	Mediterranean and thermo-Atlantic halophilous scrubs (*Sarcocornetea fruticosi*)
1510*	Mediterranean salt steppes (*Limonietalia*)
2110	Embryonic shifting dunes
2120	Shifting dunes along the shoreline with *Ammophila arenaria* ("white dunes")
2130*	Fixed dunes with herbaceous vegetation ("grey dunes")
2250*	Coastal dunes with*Juniperus* spp.
2270*	Wooded dunes with *Pinus pinea* and/or *Pinus pinaster*

The sign '*' indicates priority habitat types

Despite this situation, these marshes have merit their protection under different figures and levels, particularly due to their interest for birds conservation. They are protected as Natural Park (Paraje Natural; 1984), with two areas where spoonbills (*Platalea* leucorodia) nest (Isla de Enmedio and Marisma del Burro) declared Singular Natural Reserves (Reservas Naturales). The Odiel Marshes have been declared Natural Reserve of the Biosphere by UNESCO (1983), and they are included in the European Natura 2000 Network as SCI (Site of Community Interest; 2016) and SPA (Special Protection Area for Birds; 1987), and proposed for EU Special Area of Conservation (SAC). Odiel Marshes are considered a site of international importance for migratory waders through the East Atlantic flyway (Sánchez et al. 2006), and they are catalogued as Wetland of International Importance (Ramsar Convention 1989). In this area, a totality of 11 EU habitats of interest (Habitat Directive; Council Directive 92/43/EEC) have been identified (Table 7.1), which are relevant for the conservation of a high number of bird, mammal, amphibian, reptile and plant species of interest (see the Natura 2000 Standard Data Form for Site ES0000025). In addition, the Tinto and Odiel rivers joint estuary maintains the remains of an old Islamic city with origin in the twelfth to thirteenth centuries and Roman salting factories (second to fifth centuries), together with other materials dated from proto-historic times (seventh to third centuries B.C.).

7.2.1 Pollution in the Estuary and Effects on Vegetation

Pollution by trace elements in the Odiel and Tinto marshes is mainly originated from two clearly differed sources, as it has been previously stated. On one hand, the industrial activities located at the Odiel and Tinto joint estuary (Elbaz-Poulichet et al. 2000), particularly phosphate fertilizer industry. Radioisotopes associated to this industry have been found in the estuary tidal salt marshes as well as in areas outside the tidal influence (Bolívar et al. 2000; El Mrabet et al. 2001; Elbaz-Poulichet et al. 2000). In addition, a phosphogypsum stack created since the 1960s is located in the right bank of the Tinto river mouth, covering about 1200 ha of formerly salt marshes. This dump, with an average height of 5 m and containing about 100 Mt of phosphogypsum (Bolívar et al. 2000), acts as an emission source of contaminants and releases highly toxic pollutant such as arsenic, uranium, lead or cadmium, among others (Pérez Lopez et al. 2010). On the other hand, long-term mining activities have occurred in the Iberian Pyrite Belt for a long time, and wastes have been drained by both Odiel and Tinto rivers (Elbaz-Poulichet et al. 1999; Leblanc et al. 2000; Leistel et al. 1997; Morillo et al. 2002; Van Geen et al. 1997). In this regard, the Tinto river ore, considered as one of the largest metal-rich sulfide deposits in the world, started to be mined before Roman times (Van Geen et al. 1997), and metal concentrations its basin reach higher values in comparison with the basin of the Odiel river, mainly due to a more intensive mining activity (Elbaz-Poulichet et al. 1999).

Pollution in the Odiel and Tinto joint estuary is accompanied by very low pH values along both rivers as consequence of the sulfuric acid production by drainage of pyrite wastes and slags oxidation, as well as due to the metabolism of specific bacteria. Therefore, pH of both rivers remains low (approx. 2.5 in the Tinto river and 3.0 in the Odiel river) outside the tidal influence (López-Archilla et al. 1993; Van Geen et al. 1997). This creates a gradient of pH from very acidic sediments at the mining area to neutral sediments (pH ca. 7) in the salt marshes down to the estuary due to the buffering capacity of dissolved salts from seawater and dilution of the polluted waters (Elbaz-Poulichet et al. 2001; Galán et al. 1999; Grande et al. 2003) (Table 7.2). This sediment acidification in the estuary promotes the increase on bioavailability of many metals, becoming toxic for plants and animals and affecting their distribution (Heckman 1990; Vázquez et al. 2000).

The referred acidity gradient, together with changes in salinity due to tides, promotes a marked vegetation zonation along the banks of the main channel of the Tinto river. Thus, the vegetation in the Odiel Marshes shows a typical zonation pattern of species distribution related with the elevation gradient close to the mouth of the estuary. Species such as *Zostera noltii* Hornemann, *Spartina maritima* (Curtis) Fernald (small cordgrass), *Sarcocornia perennis* (Mill.) Scott ssp. *perennis* (*Arthrocnemum perenne* (Miller) Moss). and *Salicornia ramosissima* J. Woods establish in the low marshes (Figs. 7.3 and 7.4), *Sarcocornia fruticosa* (L.) Scott, *Sarcocornia perennis x fruticosa* (Figueroa et al., 2003), *Halimione portulacoides* (L.) Aellen, *Suaeda maritima* L. (Dumort) and *Spartina densiflora* grow in middle marshes, and *Arthrocnemum macrostachyum* (Moric.) Moris and *Suaeda vera* Gmelin dominate high marshes (Fig. 7.5).

In contrast, only a few marsh plant species, such as *S. densiflora*, *Typha dominguensis* (Pers.) Steudel, *Phragmites australis* (Cav.) Trin. and *Scirpus maritimus* L., are able to colonize more acidic salt marshes (pH ca. 4), where many areas are devoid of vegetation. Acidic banks (pH ca. 2.5) free from the tidal influence are colonized by other plant species, tolerant to certain salinity and flooding degree, such as *Juncus* sp., *Scirpus* sp. and *Tamarix* sp., *Cynodom dactylon* L. (Pers.) or *Chaetopogon fasciculatus* (Link) Hayek (Curado et al. 2010).

Table 7.2 Trace elements concentrations in five sampling sites along the Tinto river channel (Southwest Iberian Peninsula) and pH values in the different sampling sites

Site	Distance from the river mouth (km)	pH	Trace element concentration (ppm D.W.)								
			Zn	Cu	Cd	Pb	Cr	Ni	Al	Fe	As
1	0	7.5 ± 0.0	770 ± 121	109 ± 4	≤ 0.1	32 ± 3	13 ± 1	≤ 0.1	577 ± 23	333 ± 11	25 ± 2
2	1.7	4.0 ± 0.1	1062 ± 114	413 ± 36	≤ 0.1	1085 ± 179	35 ± 5	6 ± 0	1254 ± 77	4659 ± 565	967 ± 129
3	10.0	2.9 ± 0.1	778 ± 41	289 ± 55	≤ 0.1	811 ± 87	35 ± 7	6 ± 0	1714 ± 391	3587 ± 166	906 ± 96
4	15.1	2.5 ± 0.1	1233 ± 114	276 ± 38	≤ 0.1	1195 ± 92	40 ± 3	11 ± 1	1856 ± 132	3820 ± 193	766 ± 36
5	28.9	2.8 ± 0.1	3118 ± 131	504 ± 24	1.0 ± 0.0	7575 ± 232	61 ± 1	15 ± 1	2476 ± 49	8698 ± 89	3872 ± 107

Modified from Curado et al. (2010)

Fig. 7.3 The native European cordgrass *Spartina maritima* colonizes the rivers bank of Tinto salt marshes close to the mouth of the estuary, where pH is close to neutrality but it is absent from acidic marshes upland

7.3 The Odiel Marshes Key Species as Phytoremediation Biotools

Most of the dominant plant species in salt marshes are halophytic, that is, they are resistant to high concentrations of salts. Their ability to survive in saline environments renders these species with the additional ability to survive in locations with high metal contamination (Lutts and Lefèvre 2015; Mendez and Maier 2008), due to the particular adaptations developed to avoid or withstand salts. This lend them a critical role to the dynamics of the estuarine ecosystem, since they are able to alter the chemistry of the sediment in contact with their roots (rhizosediment) (Almeida et al. 2006), so influencing metal mobility in an extent that depends on the existing physicochemical properties in the salt marsh sediment (Jacob and Otte 2003), as well as strongly influence the processes of heavy metal accumulation (Weis et al. 2002; Windham et al. 2003). Depending on the properties of the particular plant species, some halophytes may be of special interest for phytoextraction or phytostabilization purposes In this regard, phytoextraction is understood as the ability of plants to remove pollutants from soil, sediment or water and assimilate them into their biomass, while phytostabilization is referred to those mechanism that cause the sequestration of pollutant in zones of the soil, preventing their mobilization and

Fig. 7.4 Colonization of a highly polluted area with metals by the annual *Salicornia ramosissima* in theTinto salt marshes. Red soils are the result of high Fe content

circulation into the food web. In addition, metal uptake by plants depends on various factors such as the plant species, the age and growth stage of the plant, seasonal variations, the existence of an iron plaque on the roots, metal speciation and bioavailability in the environment, and metal characteristics (Caçador et al. 2000).

The tolerance and accumulation capacity of heavy metals and metalloids have been explored in different halophytes present in the Odiel and Tinto joint estuary. For example, Cambrollé et al. (2008) analyzed As, Cu, Fe, Mn, Pb and Zn contents in sediments and rhizosediments from Odiel and Tinto marshes in different tissues of the European native *Spartina maritima* and the invasive alien *S. densiflora*, concluding that both plants influenced the distribution of metals in the marsh sediments, showing to have potential use for metal phytostabilization. In this study, it was also detected that *S. densiflora* shows higher capability to retain metals around its roots and to control the uptake or transports of metals than *S. maritima*, which could be related with a higher formation of plaques of Fe/Mn (hydro) oxides on its roots. A ferric oxide/hydroxide precipitate, commonly known as iron plaque, envelops the roots of a number of wetland plants (Smillie 2015), including various species of *Spartina* and *Salicornia* genus. These plaques consists mainly of Fe/Mn (hydro) oxides with a large capacity to adsorb metals (Kabata-Pendias and Pendias 2001; Ye et al. 2003), with the ensuing accumulation of metals in the rhizosphere (Doyle and Otte 1997). Both *Spartina* species seem to have the ability to oxygenate their rhizo-

Fig. 7.5 Salt pan during summer time colonized by *Arthrocnemum macrostachyum* in the Tinto salt marshes. Salt is accumulated in the soil surface, what results in *white areas* of bare soil

sphere due to a well-developed aerenchyma in roots and rhizomes, thus generating an oxidizing soil environment around their belowground tissues (Castillo et al. 2000). The high concentrations of Fe in the Odiel and Tinto joint estuary (Egal et al. 2008), coupled with the oxidation of the root zone by both *Spartina* species, seem to be promoting the formation of high amounts of Fe-oxides in the rhizosphere and thus of iron plaques on the surface of their root, leading to the immobilization of metals through sorption processes (Cambrollé et al. 2008). Regarding *S. maritima*, it has been recently investigated the benefits of the inoculation of this native cordgrass with indigenous metal-resistant endophytes. This could accelerate both adaption and growth of *S. maritima* in polluted estuaries in restoration operations but may not be suitable for rhizoaccumulation purposes (Mesa et al. 2015). However, Paredes-Páliz et al. (2016) showed how the gram-negative bacteria *Pantoea agglomerans* RSO6 and RSO7 exhibited good results for resistance to and bioaccumulation of heavy metals. These abilities make them very interesting as inoculants for phytoremediation processes but further work is still necessary. Together with *Spartina maritima*, *Sarcocornia perennis* has been also described as a valuable biotool for phytoremediation projects for a wide range of heavy metals, (Curado et al. 2013a, 2014a). The soil–plant transfer coefficient for a set of metals has been explored for *S. perennis* in the Odiel Marshes, showing values higher than 1.0 for most of them and indicating hyperaccumulation, except for Pb and As (Curado et al. 2014a).

On the other hand, *Halimione portulacoides* (L.) Aellen is a halophytic shrub frequently found on sandy and muddy sea-shores and middle salt marshes around the coasts of Europe, North Africa and South-West Asia. This species is frequently the physiognomic dominant on well-drained middle marshes, often fringing channels and pools that are flooded at high tide (Chapman 1950). In several estuaries of the Iberian Peninsula, *H. portulacoides* grows in sediments featuring extremely high concentrations of metals. Several studies have explored the phytoremediation potential of *H. portulacoides* under different factors, such as the presence of organic pollutants (e.g., Duarte et al. 2007) and heavy metals (e.g., Andrades et al. 2013). Moreover, *H. portulacoides* have been highlighted as having the potential for phytoextraction (Milić et al. 2012). Cambrollé et al. (2012a, b) evaluated the effects of Zn and Cu on this species in greenhouse experiments. They concluded that this species can tolerate extremely high tissue concentrations of Cu and Zn without suffering adverse physiological effects, producing significant amounts of biomass while sequestrating high concentrations of these metals. In this regard, Sousa et al. (2008) stated that compartmentation and detoxification mechanisms are crucial to allow *H. portulacoides* to tolerate high levels of heavy metals, and found that this halophyte is able to retain a considerable quantity of metals in its root cell wall. Latest approaches investigating *H. portulacoides* potential for remediation are focused on the use of stem cuttings directly planted in the marsh. In this sense, Cambrollé et al. (2016) found that this species is able to survive and grow at external Zn concentrations of 130 mmol l^{-1} (approximately 9000 mg kg^{-1}) during a greenhouse experiment, and determined that the use of stem cuttings with a minimum size of approximately 10 cm in length and a minimum biomass of 100 mg dry weight would be advisable to use to remediate soils highly polluted with Zn showing concentrations from 50 mmol l^{-1}. Nevertheless, despite these promising results, this approach still needs to be comprehensively tested on the field.

Although known to be highly salt tolerant, *Salicornia* spp. tolerance to heavy metals has been poorly studied (Rosso et al. 2005). Compared to results from uncontaminated sites, it has been observed that *Salicornia* spp. *accumulate* high levels of certain metals when growing in metal-polluted sites (Bryan and Gibbs 1983). In contrast, other studies show that metal concentration in the *Salicornia* spp. tissues is generally lower in comparison with values recorded in the associated sediments (Williams et al. 1994), with the exception of Zn, which has been observed to be hyperaccumulated. In this sense, the annual *S. ramosissima* has shown high capacity for Cd accumulation (Pérez-Romero et al. 2016). A recent study stated that *Salicornia* spp. appears to be a suitable tool for biomonitoring Zn and Cu (Smillie 2015).

The potential of the extreme halophyte *Arthrocnemum macrostachyum* has been analyzed by Redondo-Gómez et al. (2010) to determine its tolerance and ability to accumulate Cd for phytoremediation purposes. This species, growing at high salt marshes and salt pans, has demonstrated hypertolerance to stress by Cd, showing no phytotoxicity at shoot concentration as high as 70 mg kg^{-1}. Therefore, it has been suggested to be a valuable species for restoring Cd-contaminated sites.

On the other hand, the high salt marsh plant species *Limoniastrum monopetalum* could be used in the revegetation of Cu-contaminated soils since it is able to tolerate high Cu concentrations due to its capacity to accumulate the metal in its roots and effectively prevent its translocation to photosynthetic tissues (Cambrollé et al. 2013).

Finally, it has been suggested that *Atriplex halimus* L. may be useful in the restoration of Cu-polluted areas (Mateos-Naranjo et al. 2013), and it has been found that its stem cuttings are able to survive, root and grow at concentrations up to 9 mmol l^{-1} under greenhouse conditions (Mancilla-Leytón et al. 2016). In this sense, *A. halimus* could be also considered to be implemented in restoration projects in metal-polluted soils.

7.4 Dealing with the Case of *Spartina densiflora* in the Odiel Marshes: Between Invader and Phytoremediator

The dense-flowered alien cordgrass *S. densiflora* was unintentionally introduced to the Gulf of Cádiz (SW Spain) probably in the sixteenth century from South America, where it is a salt-marsh dominant of wide latitudinal range (Nieva and Luque 1996). In Europe, this species has extended its range into the Mediterranean and into Portuguese marshes (Castillo et al. 2000). Odiel Marshes are extensively invaded by this species, where it has colonized a wide range of habitats including dunes, high marsh and levees, salt pans, and intertidal flats, where it has competitively displaced most of the native halophytes, getting into contact with the indigenous low-marsh dominant, *Spartina maritima* (Castillo et al. 2000; Nieva et al. 2001) (Fig. 7.6).

Spartina densiflora tolerates a wide pH range from 4 to 9 (Carnevale et al. 1987; Curado et al. 2010), as well as the high metal concentrations present in this estuary (Cambrollé et al. 2008; Curado et al. 2010) This situation favors the cordgrass expansion in tidal mudflats, where no native perennial halophytes can grow as fast and vigorously as *S. densiflora* (Fig. 7.7). In fact, the common reed (*Phragmites*

Fig. 7.6 Progression of the colonization of the invasive cordgrass *Spartina densiflora* on an intertidal mudflat in the Odiel Marshes (Southwest Iberian Peninsula) between 2008 and 2016. During these 8 years, *S.densiflora* covering has notably increased to almost cover the whole surface in this narrow frame

Fig. 7.7 The invasive *Spartina densiflora* colonizes middle and high salt marshes in along the Tinto estuary, among other habitats including dunes and intertidal flats, being able to stand acidic and polluted sediments under tidal influence

australis) (Fig. 7.8) is the only native plant species growing healthily together with *S. densiflora* in areas with pH ca. 4 (Curado et al. 2010), although some other species from the genera *Juncus*, *Typha* and *Scirpus* can grow in these areas although occupying much less surface than the two mentioned dominant species. At present, *S. densiflora* is able to form monospecific communities in some areas under tidal influence of the joint estuary of Odiel and Tinto rivers (Fig. 7.9). In contrast, this species is not able to colonize by seeds areas outside the tidal influence where pH and heavy metal pollution limit its expansion along the Tinto river, and it has been found that some metals such as Cr or Al may affect *S. densiflora* seed germination dynamic (Curado et al. 2010). On the other hand, it has been observed that this species hardly colonizes by rhizomes the Tinto river upstream areas, outside the tidal influence (Curado, unpublished data). In the lower marshes limits, in addition, well developed *S. maritima* prairies have showed to be able to efficiently prevent *S. densiflora* invasion at low elevation in the tidal gradient (Castillo et al. 2008a). Considering this, it seems that the invasion area of *S. densiflora* is almost limited in the Odiel Marshes.

As it was previously described, *S. densiflora* shows a higher capability to retain metals around its roots and to control the uptake and transport of metals, mediated by higher formation of plaques of Fe/Mn (hydro) oxides on its roots. This fact would be quite helpful for phytostabilization purposes and it should be taken into account

Fig. 7.8 Common reed (*Phragmites australis*) colonizes very acidic and highly polluted sediments with metals at the Tinto salt marshes. This is the only native species surviving and dominating acidic and polluted areas together with the invasive alien *Spartina densiflora*

for implementing new restoration projects in its native distribution range (Cambrollé et al. 2008; Curado et al. 2013a). However, this species has demonstrated negative ecological impacts in the Odiel and Tinto joint estuary. It directly and negatively affects native vegetation, having also negative impacts on bird and macroinvertebrates communities, as well as on sediment dynamics, among others (Curado et al. 2012, 2013b, 2014c; Nieva et al. 2001). In addition, the long-term implications for marsh development of the invasion of *S. densiflora* are uncertain, but continued spread of this alien threatens a significant loss of biodiversity in southern European salt marshes.

At this point, *Spartina densiflora* plays a double key role in the Odiel and Tinto marshes. On one hand, its invasive ability makes it an important threat for the ecosystem conservation, so it should be eradicated from the area, while natural native vegetation should be recovered. On the other hand, it acts as an ecosystem engineering stabilizing both sediments and soil contamination. Removing the present biomass of *S. densiflora* from the occupied marshes would have, therefore, two major impacts on the coastal marsh ecosystem. Firstly, soil contamination would avoid native species from establish in those areas presently invaded by *S. densiflora*, so natural recovery may be really difficult or even not possible by itself. This would mean a relevant loss for the maintenance of the associated fauna. Secondly, sedi-

Fig. 7.9 The invasive cordgrass *Spartina densiflora* is able to form monospecific communities in some areas under tidal influence of the joint estuary of Odiel and Tinto rivers. Nevertheless, it spread is limited by both well-developed *S. maritima* prairies in the lower limits of the tidal gradient, and acidic pH and heavy metal pollution outside the tidal influence

ment erosion favored by the absence of stabilizing vegetation would result in the mobilization of highly polluted sediments. In view of these alternatives, the environmental administration has a difficult dilemma: decontamination or conservation (Sainz and Loredo 2005).

7.5 Restoring Salt Marshes with Plantations of *Spartina maritima* to Phytostabilize and to Phytoextract Metals

Soft engineering projects are needed to restore, rehabilitate, or recreate degraded salt marshes, with cordgrasses (genus *Spartina*) being one of the most popular biotools. Phytoremediation of wetlands polluted with metals may be included among the aims of these restoring projects. Our studies in the Odiel Marshes, located in one of the most polluted estuaries with heavy metals all around the world, show that plantations of the European cordgrass *Spartina maritima*, and accompanying halophytes, stabilize polluted sediments and phytoextract metals. Our results are particularly relevant since *S. maritima* is the only native European cordgrass. Until now, most of the studies had work with *Spartina alterniflora* and *Spartina* *foliosa*, native from the Atlantic Coast of North America, but little was known about the possibilities of ecological restoration and phytoremediation using *S. maritima*.

A novel ecological restoration project was carried out in the Odiel Marshes from November 2006 to January 2007. Previous to the restoration works, the sediments of the low marshes where *S. maritima* was planted were polluted with metals (Van Geen et al. 1997) and an area was also polluted by an historic oil deposit due to oil spills from neighboring industries. In addition, the alien *Spartina densiflora* was actively invading these degraded marshes and had already occupied more than 2.01 ha. On top of all, one location exposed to high energy waves and currents was suffering high erosion rates, evidenced by detachment of substrate blocks from an erosive bank (Castillo et al. 2002).

The approach was supported by previous results obtained in different studies that enabled the development of a suitable methodology to restore these salt marshes using *S. maritima*:

Firstly, a transplant experiment was used to investigate the means by which physical and chemical factors determined lower distribution limits of native *S. maritima* and invasive *S. densiflora* along the intertidal gradient. Neither species survived for a year at elevation lower than 1.04 m relative to SHZ. The lower distribution limit was +1.41 m SHZ for *S. maritima* and +1.46 m SHZ for *S. densiflora*. Moreover, tiller growth rates of both species increased with elevation, but that of *S. densiflora* was more sensitive to low elevations (Castillo et al. 2000). This study was very useful to understand the elevations where *S. maritima* had to be transplanted.

At the same time, the role of *S. maritima* and *Sarcocornia perennis. ssp. perennis* on the ecological succession was analyzed. *S. maritima* colonization of low-lying mud flats promotes sediment accretion sufficiently to ameliorate stressful conditions related with sediment anoxia, facilitating the colonization of *S. perennis*. This halophyte becomes increasingly dominant as accretion progress, until *Spartina* is virtually eliminated from all but the expanding edges of its own tussocks (Castellanos et al. 1994, 1998). Later on the succession, *S. perennis* facilitates the in situ formation and the colonization of its hybrid *Sarcocornia perennis x fruticosa* as a result of pollen flow from high-marsh *Sarcocornia fruticosa* to the stigmas of the established dominant *S. perennis*. Succession might therefore be genetically facilitated (Figueroa et al. 2003). These studies enabled us to know how restored salt marshes using *S. maritima* and *S. perennis* would evolve over time.

On the other hand, the shoot height of *S. maritima* was also study since this species tend to exhibit a wide range of phenotypes, often with short and tall growth forms. Our results showed that height variation in *S. maritima* appears mainly to be a result of phenotypic plasticity, with hypoxic sediments stimulating stem growth (Castillo et al. 2005). The discovery of this highly plastic growth form of *S. maritima* showed that there was no need to select specific clumps from natural populations for transplant since every populations showed similar heights in a common environment.

The role of *S. maritima* and *S. densiflora* on the establishment of salt marsh ecological zonation were also studied in areas where both cordgrass species are dominant. *S. densiflora* invaded the upper areas of the marsh at the centre of the circular tussocks of *S. maritima*, where above-ground biomass of *S. maritima* dropped drastically. The competitive potential of *S. densiflora* at higher elevations was reflected in high above- and below-ground biomass and higher shoot densities, accompanied by elevated wrack accumulation and the absence of other marsh plants. However, the alien invasion may be limited by the presence of the autochthonous cordgrass at lower elevations (Castillo et al. 2008a). These results were useful for restoration projects since they showed that healthy *S. maritima* prairies would be an adequate way to limit the invasion of *S. densiflora*.

Finally, during all these above mentioned studies, we carried out pilot transplant experiments with *S. maritima* were carried out at different locations in the Odiel Marshes, and variations on above- and below-ground biomass of *S. maritima* and on abiotic environment were subsequently analyzed along a chronosequence of six marshes created from 1997 to 2003, showing different sediment dynamics, and adjacent natural marshes and unvegetated tidal flats. Results showed that *S. maritima* behaves as an autogenic engineer, as its colonization of bare sediments in created salt marshes yiels marsh level rise accompanied by higher oxygenation and salinity. These modifications of the abiotic environment were site-specific, depending mainly on sedimentary dynamics. At the same time, abiotic environmental changes determined biomass production rates of *S. maritima* that were higher in more-accreting marshes; however, constant above-ground biomass was kept from early in its development (2 years) (Castillo et al. 2008b). These results pointed to the importance of the sedimentary dynamic when setting realistic expectations for success criteria of created and restored wetlands. In addition, we already knew that transplants of *S. maritima* will develop similar above-ground biomass levels than natural populations within 2 years and that the accumulation of below-ground biomass happens more slowly.

Once the methodology was tested at small-scale, a large restoration project was carried out with four specific goals: (1) to recover native vegetation, restoring the degraded landscape; (2) to phytostabilize oil-polluted sediments; (3) to prevent erosion and stabilize banks; and (4) to promote the conservation of *S. maritima*, an endangered species included on some European red lists.

Plantation zones were delimited with small wood stakes between +1.50 and +2.30 m SHZ (8.37 ha) based on the lower general distribution limit of *S. maritima* in the tidal range (1.41 m SHZ) (Castillo et al. 2000). Planting was carried out at a density of 1 clump per m^2 using a triangular-shaped herringbone planting method to maximize sediment occupation. *Halimione portulacoides* clumps were planted at the edges of interior marshes to accelerate its colonization and many *S. maritima* clumps were accompanied by *Sarcocornia perennis ssp. perennis* since the extraction areas where natural populations where *S. maritima* was being outcompeted by *S. perennis* (Figueroa et al. 2003).

The monitoring of this restoration project offered very interesting results to remediate metal pollution in estuaries. Our results showed the success from the point of view of vegetation of restoring European low salt marshes using *S. maritima* and *S. perennis* plantations since they are able to reproduce, 2.5 year after restoration, the typical plant zonation pattern (Curado et al. 2014b) (Fig. 7.10). At the same time than vegetation development, erosion was lower in

Fig. 7.10 Development of plantations of the European native cordgrass *Spartina maritima* in 2005 (*left*), 2007 (*center*) and 2016 (*right*) in two restored salt marshes in the Odiel Marshes (southwest Iberian Peninsula). The *S. maritime* prairies were recovered after ca. 2 year with a relative cover of 62%, being a useful tool to phytostabilize eroding areas

the *S. maritima* zone (mean accretion rate between +10 and +27 mm year^{-1}) than in bare intertidal mudflats (−14 to +12 mm year^{-1}). Thus, extensive salt marsh plantations using *S. maritima* behaved in a similar way to natural preserved marshes after ca. 2 year with a relative cover of 62%, this being a useful tool to phytostabilize eroding areas in European marshes, since they reduce erosion and increase accretion (Curado et al. 2012).

Iron, aluminum, copper and zinc were the most concentrated metals in the restored area. Every metal, except nickel, showed higher concentration in the root zone than in the sediment surface, with values as high as ca.70 g Fe kg^{-1} (Fig. 7.11). The highest metal concentrations in *S. maritima* tissues were recorded in its roots (maximum for iron in *Spartina* roots: 4160.2 ± 945.3 mg kg^{-1}). Moreover, concentrations of aluminum and iron in leaves and roots were higher than in superficial sediments. Rhizosediments showed higher concentrations of every metal than plant tissues, except for nickel. Our results showed *S. maritima* to be a useful biotool for phytoremediation projects in European salt marshes (Curado et al. 2013a). In addition, *S. perennis*, that was accompanying *S. maritima* transplants, also appeared to be a useful phytoremediation tool able to hyperaccumulate Al, Cd, Cr, Cu, Fe, Ni, and Zn (Fig. 7.12). The highest metal concentrations were recorded in *S. perennis* roots (translocation coefficient lower than 1.0 for every metal) (Curado et al. 2014a).

In addition to metals phytostabilization and phytoextraction, the plantations of *S. maritima* and *S. perennis* were sequestering atmospheric carbon, therefore providing some mitigation for global warming, and were capturing nitrogen, what reduces estuarine waters eutrophication (Curado et al. 2013c, 2014a).

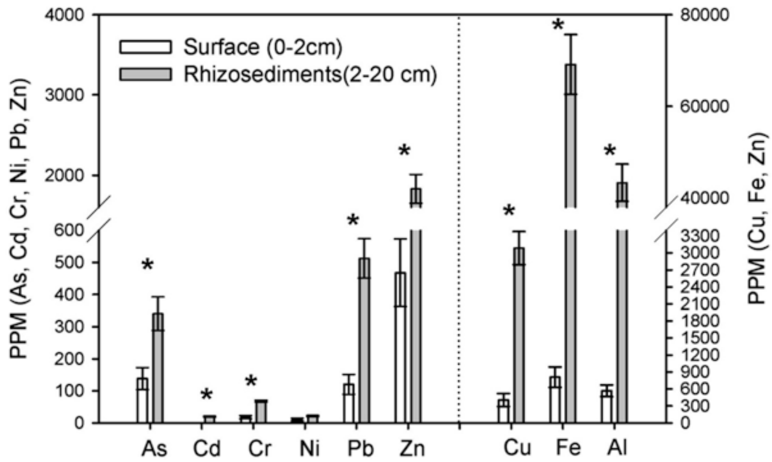

Fig. 7.11 Metal concentration (n = 10; parts per million DW-ppm-) in superficial sediments (0–2 cm deep) and sediment colonized by *Spartina maritima* roots (rhizosediments) (2–20 cm deep) in restored salt marshes in Odiel Marshes (Southwest Iberian Peninsula) described by Castillo and Figueroa (2009) 2.5 years after restoration. "*" indicates significant differences between depths (*t*-test or *U*-test, $P < 0.01$)

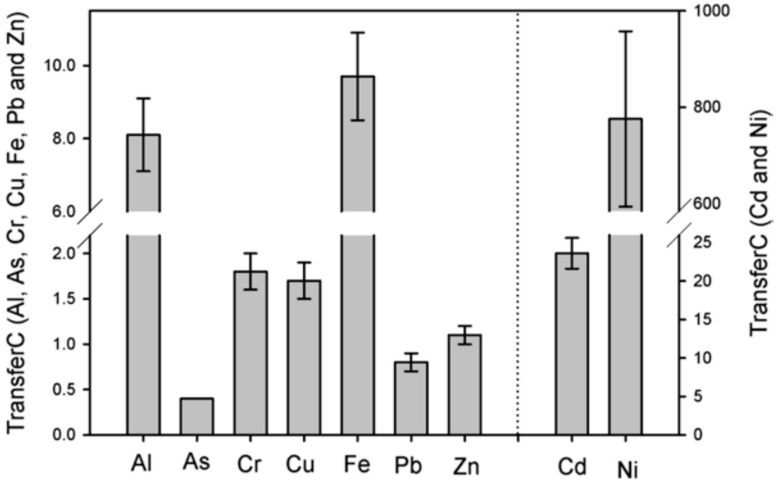

Fig. 7.12 Soil–plant transfer coefficient (TransferC) for *Sarcocornia perennis* in restored salt marshes in the Odiel Marshes (Southwest Iberian Peninsula) described by Castillo and Figueroa (2009) 2.5 years after restoration. TransferC higher than 1.0 indicates hyperaccumulation

7.6 Discussion

General soft management actions in coastal ecosystems are commonly focused on the eradication of exotic species and recovery of native species, the recovery of both the structure and function of plant communities and of natural sedimentary dynamics, and the minimization or elimination the existing impacts and perturbations. Regarding soil contamination by trace elements, phytoremediation has become a main green technology in recent years, due to its notable efficiency in terms of costs and efficacious (Montpetit and Lachapelle 2016). General soft management actions regarding the rehabilitation of contaminated soils by trace elements have preferentially tried to stabilize pollutant in the soil rather than extract them. Nonetheless, phytoextraction has received increasing attention since the discovery of hyperaccumulator plants, able to phytoextract and accumulate high levels of particular metals in their above-ground biomass (Vamerali et al. 2010). In such a dynamic ecosystem as salt marshes, phytoextraction would be limited to the use of herbaceous native species, particularly selected to each specific pollutant and objective. Additional desirable characteristics that these species should present are fast grow and high biomass production, extended root system for exploring large soil volumes, good tolerance to high concentrations of metals, high translocation factor and easy management (Vamerali et al. 2010). Such species should ideally avoid biomass (leaves) release along their life cycle, since polluted biomass circulating through marshes channels and even littoral sea currents would easily enter the food chain. On top of all, further biomass management should be planned, regarding harvesting and safe storage of polluted biomass. All this brings to the idea that management of polluted salt marshes needs for the design of particular actions adapted to each case.

In the case of the salt marshes in the Odiel and Tinto joint estuary, other two main threats are identified in addition to soil contamination: the invasion by the alien *Spartina densiflora* and erosion of marsh channels. A local restoration project has resulted in a rapid recovery of the native prairies of low tidal marshes, dominated by *S. maritima*, becoming a promising tool to phytostabilize eroding areas in European marshes (Curado et al. 2012). These prairies seem to stop the advance of the *S. densiflora* invasion and prevent erosion. In this regard, extensive *S. maritima* plantations in the lower marshes along the protected area would be desirable. Plantations in other upper areas with other native species need to be further studied, although a relation of suitable native species for soil contamination phytoremediation in European salt marshes is available at present. Although further research concerning plant establishment and biomass production and limitations is still necessary, also the use of stem cuttings of certain species, such as *Halimione portulacoides* (Cambrollé et al. 2016), provides a cost-efficient way to accelerate the phytoremediaton process.

On the other hand, some protected areas of the Odiel and Tinto marshes are invaded by the alien *S. densiflora*. In this regard, and in contrast to generalized criteria, it seems that maintaining the *S. densiflora* prairies in areas where it has become dominant is a better option than removing them, since the latter action would lead to an important release of presently stabilized pollutants. A solution could be related with getting ready neutral sediment salt marshes using *Spartina maritima* and *Sarcocornia perennis* plantations, which have already been tested as a successful methodology for ecological restoration in polluted sediments in the same estuary (Castillo and Figueroa 2009). The most complicated step would be to restore those polluted areas where *Spartina densiflora* tussocks would be eliminated. They should be planted with native *Phragmites australis* due to its capacity to grow in the same acidic and metal polluted sediments than *S. densiflora*. In fact, planting *P. australis* in estuaries and river banks in areas already invaded by or susceptible to invasion by *S. densiflora* has been suggested as viable option for creating a barrier to dense-flowered cordgrass expansion (Mahmoud-Abbas 2012). On the other hand, these plantations should be accompanied with *Typha dominguensis, Juncus* sp. and *Scirpus* sp. to increase biodiversity and resilience of the restored marshes. Further actions should be focused on planting phytoextracter species and their progressive substitution by native species, once the contamination has decreased to safe levels. These actions need to be carefully designed, minimizing any impact on such ecosystem that has begun to be considered as a highly complex ecosystem with high scientific interest. In any case, this alien is being controlled in other areas along the SW Iberian Peninsula coastal frame.

7.7 Conclusions

Soil contamination in salt marshes and estuaries is an increasing problem, and proper management actions are needed. Such actions may take into account the particular contamination traits and other threats in each case, as well as the sedimentary and successional dynamics, so concrete management designs can be developed and implemented. Soft management identifies phytoremediation as one of the main green tools, showing notable efficiency in costs and efforts. Recovery of the lower marsh limits through plantations with the European cordgrass *S. maritima* seems to be an efficient approach to phytoremediate trace elements pollution, avoid erosion and recover the natural plant community in European salt marshes. In addition, it imposes certain limits to alien invasion. A wide variety of salt marsh species have been identified as suitable tools for phytoremediation of the main polluting trace elements along the elevation gradient in salt marshes, either as phytostabilizer or phytoextractor, including the alien *S. densiflora* that can be indeed used inside its natural range.

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Chapter 8
El Yali National Reserve: A System of Coastal Wetlands in the Southern Hemisphere Affected by Contemporary Climate Change and Tsunamis

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Abstract El Yali is a complex wetlands system composed by more than 14 waterbodies, located in central Chile and delimited by two basins of the most important rivers of the region. Among the waterbodies is a coastal lagoon, some estuaries, artificial wetlands, salt mine and inner lagoons that were coastal lagoons in the past but due to tectonic processes have been moved and raised to their current location.

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The damming of the river that delimits the system to the south in 1968 cutoff the natural the sedimentary supply to the extensive beach and dunes, leaving the wetlands in a situation of vulnerability before climate change and variability, anthropic pressure, ocean swells and tsunamis. In the present chapter is illustrated the degradation that the wetlands system is suffering, by means of the estimation of tendencies of the long term records available in the zone, antecedents about its natural history, anthropic pressure and natural disasters like earthquakes, tsunamis, ocean swells and ENSO, along with field monitoring that has been carried out with the objective of implementing an ecological restoration. These antecedents show a decrease of precipitations and river discharges, an increase of ambient temperature and sea surface temperature, a rising of sea level and a change of the incident waves.

El Yali was severely affected by the earthquake and tsunami in 2010, which destroyed 800 ha of beach and coastal dunes that provided natural protection. In many opportunities, El Yali was affected by earthquake and tsunamis, intense ocean swells and ENSO, all phenomena that dissipate energy destroying the natural barrier the dunes represent. However, before the construction of Rapel reservoir, the sedimentary supply allowed a rapid recovery of the beach, sandbar and dunes. When this sediment supply was cutoff, the dunes has enough material to support the equilibrium of the system for some decades, but when tsunami waves destroyed the dunes, there was no sand supply to restore the beach nor the dunes. This generated a substantial change in the system, turning it more vulnerable to tsunamis, even smaller, ocean swells, and sea-level raising associated to ENSO Kelvin waves and climate change. For this reason, the first proposed restoration action is to restore the dunes and plant a vegetal cover with native species that reinforce and maintain the dunes.

Keywords Ramsar site • ENSO • Dune • Beach • Sandbar

8.1 Introduction

Coastal wetlands in central Chile are extremely dynamic and fragile environments, whose existence is conditioned by a number of natural and anthropic factors, including hydrologic and climatic variability, high littoral energy content, variability of sediment deposition, seismicity, and tectonic processes on the Chilean coast, which generate large morphological changes in coastal areas. This so particular combination only has similar referents in some areas of South Africa, Australia (Cienfuegos et al. 2012) or New Zealand (Nichol et al. 2007).

This chapter aims to show the degradation that El Yali coastal wetland system (33°45′S; 71°43′W) is currently suffering, from a multidisciplinary perspective, as result of a combination of different factors:

(a) The effects of the contemporary climate change by a diminution of precipitations and the increase of temperature seem to be changing the hydrologic equilibrium of these systems and this represents a pressure forcing changes in agriculture and use of soils that increase the demand for hydric resources.

(b) Climate variability effects, such as ENSO phenomenon that modifies the seasonal evolving and other intra-annual cycles, generating a significant inter-annual variation in sea-level, sea surface temperature, air temperature, rainfall and river discharges;
(c) Anthropic effects, some of the contemporary, such as real estate demand, change of land uses, competition for water, the increase of recreation activities associated with the transit of vehicles on coastal dunes, as well as past activities, like the construction of a damn in Rapel river (basin that represents the southern limit of El Yali), which represented a cutoff in the contribution of sand to beaches and coastal dunes in this area; and
(d) The recurrent earthquakes, tsunamis and ocean swells that have affected this zone, producing morphological changes due to co-seismic vertical movements, eroding beaches and sand dunes and changing the characteristics of the water body by seawater intrusion.

These changes appear to have accelerated as consequence of the earthquake and tsunami of February 2010. Even when in 2009 some problems of environmental quality were already detected in the wetland system, when Figueroa et al. (2009) utilized the index of conservation of lentic ecosystems (ECELS, proposed by the Catalan Water Agency 2004) and found that El Yali characteristics qualified as regular to bad, some bird census showed a high biodiversity (Vilina et al. 2014) and there were no signs of severe degradation. After that event, a permanent monitoring is being carried out in this area. This monitoring program allowed registering a dramatic decrease of waterbodies extension, especially in inner lagoons, the loss of regeneration capacity of beaches and coastal dunes, the appearance of new vegetal species and presence of invasive flora and fauna, and the massive death of the amphibian *Calyptocephalella gayi* (Acuña-O et al. 2014; Mizobe et al. 2014), known as the big Chilean frog, a specie in danger of extinction and endemic of Chile. This chapter is based on the information obtained from this monitoring program. Additionally, an initiative funded by Parks Canada allowed carrying out a study to plan the ecological restoration of the area. That study provided information to define the base line and objectives for restoration. Currently, the restoration plan is in its initial stage of implementation.

In the littoral zone of central Chile there is a confluence of at least two aspects that have influenced the formation of numerous coastal wetlands:

(a) The oceanic Nazca tectonic plate is slipping beneath the continental South America plate at a relative rate of 8 cm/year and with a convergence angle of 78° at NE (Pardo et al. 2001) provokes an imperceptible but continuous lifting of the South American Plate. On the other hand, this same process is responsible of episodic big earthquakes subduction (Mw >8.0) in this zone. These earthquakes produce important horizontal and vertical co-seismic movements (Farías et al. 2010; Quezada et al. 2012a; Wesson et al. 2015). This process alone is able to form a coastal lagoon, as in the case of El Yali. But this process can also destroy coastal wetlands, as in the case of Tubul-Raqui wetland (37°13′S; 73°26′W), which, with an extension of 2000 ha, suffered a co-seismic lifting of 1.6 m during the earthquake occurred on February 27th, 2010, causing a partial desiccation of the wetland that has been well documented (Valdovinos et al. 2010, 2012).

(b) On the other hand, river in this zone, with snow-rain regimes and important slopes and discharges, are able to transport large quantities of material from the Andean cordillera to the coast (Cienfuegos et al. 2012). In the recent past, there have been cases where the alteration of hydrodynamic equilibria has an immediate morphological response, as that documented by Pomar (1962), which describes the accretion of a sand beach and consolidation of lagoon Llolleo (33°36.4′S; 71°37.4′W) nearby Maipo river mouth, whose basin represents the northern limit of the wetlands system, after the construction of a sheltering structure in the port of San Antonio. In other cases, a rapid recovery has been observed, as in the case of the 8 km length sand bar in Mataquito river mouth (34°52′S, 72°09′W), which almost completely disappeared under the combined action of tsunami waves and the relatively large land subsidence (Lario et al. 2016; Vargas et al. 2011). This zone river mouth is located right off the rupture zone where most of the February 2010 earthquake energy was liberated (Lay et al. 2010). After the earth quake and tsunami, the coastal evolution of this zone has been monitored by using field work techniques and satellite imagery (Cienfuegos et al. 2014), evidencing that most of the original sand bar recovered in less than 18 months, even showing a rapid recovery in less than 6 months (González et al. 2012). These examples show that changes produced by anthropic alterations as well as tectonic and seismicity can provide conditions for beaches accretion and dune and sandbar formation (Martínez et al. 2015; Veas et al. 2016) that also favor consolidation of coastal wetlands. This explains the fact that there are more than 400 coastal wetlands along the coast between 30°S and 44°S (Marquet et al. 2012), conforming a, ecological corridor for migratory birds and other species. However, these wetlands, that are fragile environments due to those conditions that influence their construction and destruction, are under a growing pressure due to a persistent anthropic activity around them: using wetlands as sink of liquid residues, receiving contamination from agriculture products, draining wetlands to extract water or expansion of real estate projects. Moreover, contemporary climate change represents an additional pressure, influencing the delicate equilibrium of these systems.

Besides describing a so far poorly known system, the novelty of this chapter is based in two main aspects:

(a) To incorporate the effect of tsunamis and resilience to the discussion about coastal wetlands. During more than 50 years after the construction of the damn in Rapel River, the system of beaches to the north maintained an unstable equilibrium, where the coastal dunes that protected the coastal lagoon and beach stopped receiving a continuous supply of sand. In spite of that, the material accumulated over years was able to stay in fragile equilibrium, not showing significant changes. However, the tsunami of 2010 swept all the coastal dunes and removed a significant part of the sand form the beach, triggering a loss of functionality that will be detailed next.

(b) To describe the changes that the El Yali wetland is experiencing as response to Climate Change and the influence of multiple anthropogenic pressures. El Yali coastal wetland is located in the southern limit of the semi -arid wetland (Figueroa et al. 2009; Vidal-Abarca et al. 2011), but on its RAMSAR file it has been described as a Mediterranean wetland (RAMSAR 1996). This confusion is because of its location in the limit where the semi-arid climate is extending southwards, over the Mediterranean Central Chile. This situation offers an opportunity to study the effects of these changes on waterbodies and associated biotopes.

8.2 Geological Context and Natural History of the Site

El Yali is a complex system of coastal wetlands located in Central Chile (Fig. 8.1), between Maipo River (to the north) and Rapel River (to the south) basins. The latter was dammed in 1968, what caused an important decrease in the sand supply that used to maintain the important dunes system nearby the wetland and the coastal lagoon (Vergara 2014). This system is composed of more than 14 waterbodies, from which three of them are part of the protected RAMSAR site N°878. These three protected water bodies are the Albufera, Matanzas lagoon and Colejuda lagoon (Fig. 8.2). The wetland is characterized for the differences in the chemical composition and seasonality of its waterbodies (Figueroa et al. 2009), what explains the richness and biodiversity in this area. Thus, this condition highlights El Yali as one of the most important coastal wetlands in Central Chile (Vilina 1994; Dussaillant et al. 2009; Dussaillant 2012; Fariña et al. 2012).

El Yali wetland sustains 28% of the bird species in Chile (about 139 species) and this Mediterranean eco-region poses 3% of the endemism in aquatic bird population (Victoriano et al. 2006). Approximately 29 migratory and visiting bird species, 18 coming from the northern hemisphere and 11 form the southern hemisphere,

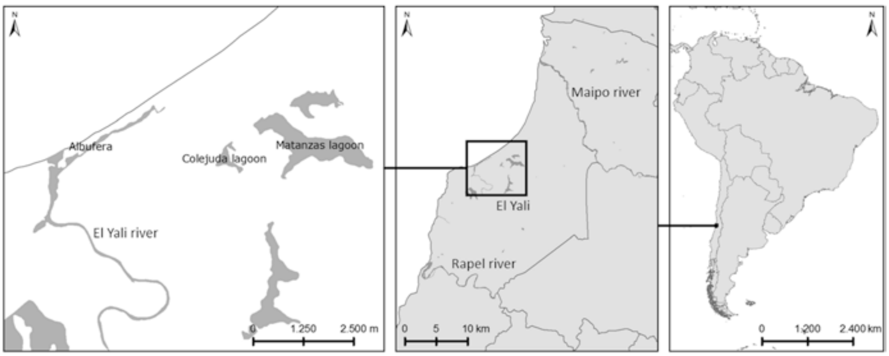

Fig. 8.1 Map of El Yali and its location on the coast of the central Chile and South American context. Three protected waterbodies are shown: Matanzas lagoon, Colejuda lagoon and Albufera (coastal lagoon). El Yali River mouth discharges at the SW extreme of the Albufera

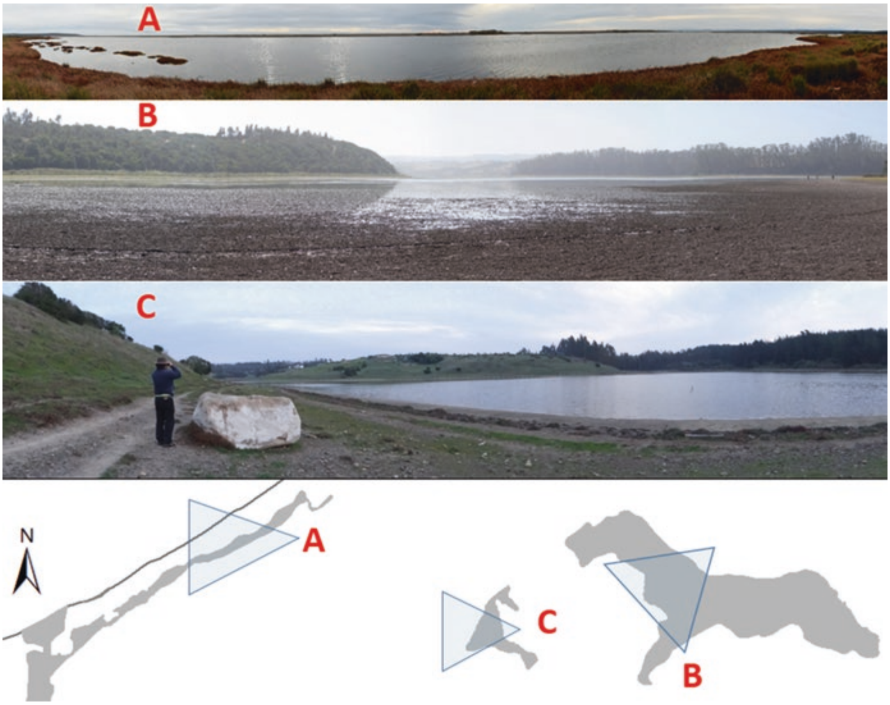

Fig. 8.2 (**a**) Albufera in August 2015. It is possible to observe a remnant of the coastal dunes on the horizon. (**b**) Matanzas lagoon in December 2014. Desiccation being suffered by this water body is easily observed. (**c**) Colejuda lagoon in July 2014. At the forefront is the gneiss block located in this place. *Below* are shown the positions where these photos were taken and the estimated direction of the sights

respectively, arrive in this wetland (Vilina 1994; Brito 1999; Victoriano et al. 2006; Fariña et al. 2012; Vilina et al. 2014).

The inner lagoons were coastal lagoons in the past, but due to seismic displacements and elevations, they are in their current position (10 or more meters above sea level and some km inland). However, they still conserve saline characteristics that favor a high abundance and biodiversity of flora and fauna.

Contreras-López et al. (2014) described the current configuration of El Yali basin, taking into account the big tectonic movements that occurred by the middle of the Tertiary Period, and that provoked strong dislocations by faults on the coast of central Chile, along with a lifting of the Andean cordillera to its current level, the sinking of the longitudinal valley and the formation of the coastal cordillera (Noguera 1956). The small streams that originated in the coastal cordillera and flowed toward the sea with tilted slopes deposited diverse coarse sediment whereas finer material was swept along to the sea. New violent tectonic movements lifted the adjacent seabed and formed the successive systems of marine terraces that can be observed nowadays along these coasts. In particular, five main movements can be identified in

El Yali area: two raisings, one sinking followed by a new raising (Noguera 1956), and a rising associated to the 1985 earth quake (Barrientos and Kausel 1990).

Thus, the development of El Yali and the system it belongs to, obeys to the interaction of several dynamic factors that have determined the geomorphological evolution that the central Chilean coast has experienced during the last 18,000 years (Fariña et al. 2012).

Marine transgressions occurred during the last glaciation (14,000 years ago) and during the middle Holocene (6000 years ago) allowed the accumulation of marine and/or fluviomarine sediment throughout the basin (Fariña et al. 2012). The tectonic activity expressed in the co-seismic vertical movements that characterized the last 3000 years (Lower Holocene), produced the regional raising of the coast that caused the emersion of a littoral terrace where is located the wetland. A sign of this process is gneiss block present to the southwest of the lagoon Colejuda (Contreras-López et al. 2014).

As result of the coastal raising, a marine regression occurs and initiates the progradation of the coastal line by means of eolic and fluvial sedimentary processes (Paskoff et al. 2000). The existence of fluvial refills and transgressive and parabolic dunes in the area indicates that progradation occurred through sedimentary loads from Rapel and Maipo rivers, by means of littoral drift and eolical transport (Paskoff and Manríquez 2004).

Paleoclimatic studies based on the analysis lacustrine sediment cores from lagoons Matanzas (Jenny et al. 2000; Villa-Martínez 2002) and Colejuda (Valero-Garcés et al. 2010, cited by Fariña et al. 2012) suggest that the evolution of the wetlands complex has been mostly modeled by changes in eolic sedimentations and significant variations in precipitations associated to ENSO (El Niño – Southern Oscillation), which increases in frequency and intensity during the Holocene.

Pollen, sediment, geochemical and chironomid fossils analyses show that 5000–3300 years ago lagoon Matanzas was a littoral lagoon with moderate to low salinity conditions (Villa-Martínez 2002). During this period, precipitations had important fluctuations, showing two rainy intervals, 4900–4800 years ago and 4500–3300 years ago, separated by an dryer interval 4800–4500 years ago (Villa-Martínez 2002).

From 3300 to 2600 years ago there would have occurred an important increment in salinity, probably representing a change in the system from a littoral lagoon a hyper-saline lake (Villa-Martínez 2002). Likely this change would be related with the closure of lagoon Matanzas basin, some 2600 years ago, due to predominance of eolic sedimentary processes that formed a dunes chord that presently dam this system (Fariña et al. 2012; Paskoff and Manríquez 2004; Martínez 2009), and that is locate immediately to the north of Navidad Formation (Encinas et al. 2006), a geological formation that attracted great interest from the naturalist Charles Darwin when he passed by this area in 1834 (Darwin 1846). This consolidated dune (El Convento), with an extension of over 1000 ha, is one of the most important in the region (Castro-Avaria 2015). Nowadays, it is difficult to appreciate its extension, due to the efforts realized during the twentieth century to consolidate its advance by planting pines and eucalyptus (Albert 1900). These plantations contribute to hide the presence of the dune.

During the gradual closure process of lagoon Matanzas, fluvial sedimentation associated to floods of Las Rosas brook would have also been an important factor (Fariña et al. 2012). Appearance of species intolerant to salinity along with low chlorine concentrations found in sediments suggest that 2400 years ago existed moderate to low salinity conditions in lagoon Matanzas (Jenny et al. 2000). During that lapse, sediments show at least 20 different flood periods, what would indicate marked fluctuations in winter rains (Villa-Martínez 2002).

During the last 150 years, Villa-Martínez (2002) recognizes a deterioration of vegetation associated to forest surrounding lagoon Matanzas, what is attributed to anthropic activity. Notwithstanding, Fariña et al. (2012), point out that an alternative but not excluding explanation is the decrease in precipitations detected in the sediment core from lagoon Colejuda by Valero-Garcés et al. (2010).

8.3 Some Aspects About Climate Change in Central Chile

El Yali is located in a region threatened by the effects of the contemporary climate change. An analysis about the registers of temperature and precipitation in this region show that temperature has risen 0.5 °C in the last 50 years, whereas annual precipitations have decreased in a 12% during the same period, which is consistent with global warming (Fig. 8.3). On the other hand, a recent analysis about the response of bio-climates of the region to different climate change scenarios showed a decrease in precipitation and an increase of temperature in Valparaiso region by 2080 (Luebert and Pliscoff 2012).

For the case of annual precipitations, it is expected to observe a latitudinal pattern of decrease in the coastal and Andean areas (a decrease that can be ~280 mm), whereas the inner areas would present a less decrease (50–80 mm per year). Similarly, the mean annual temperature shows a longitudinal increase pattern, with lower values in the coastal zone (1 °C), reaching an increase of 2.4–3.5 °C in the Andean area. Under these climatic conditions it is expected in the long term an ascendant altitudinal displacement of vegetation floors, especially in the Andean area, as well as a latitudinal displacement, with the intrusion of species typical of semi-arid and arid zones in central-north Chile, especially in the coastal zone. The recent finding of *Suaeda foliosa* (typically found in northern Chile) in El Yali wetland (Flores-Toro and Contreras-López 2015), extending its distribution in 300 km, seems to be an early expression of this phenomenon. As an additional element, centuries of agriculture in central Chile have produced a progressive degradation of soils and a severe fragmentation of the sclerophyll forest, which has consequences on the habitat (Grez and Bustamante-Sánchez 2006), causing a reduction of biodiversity and making the system more vulnerable to climate fluctuations and a decrease of the available hydric resources.

A series of historical records collected in the vicinity of El Yali wetland is shown in Table 8.1. Except the annual series of precipitations in Valparaiso (1900–2015), which corresponds to the total precipitation during 1 year, all series were averaged or accumulated to transform into monthly averages. Combined by means correlations, the

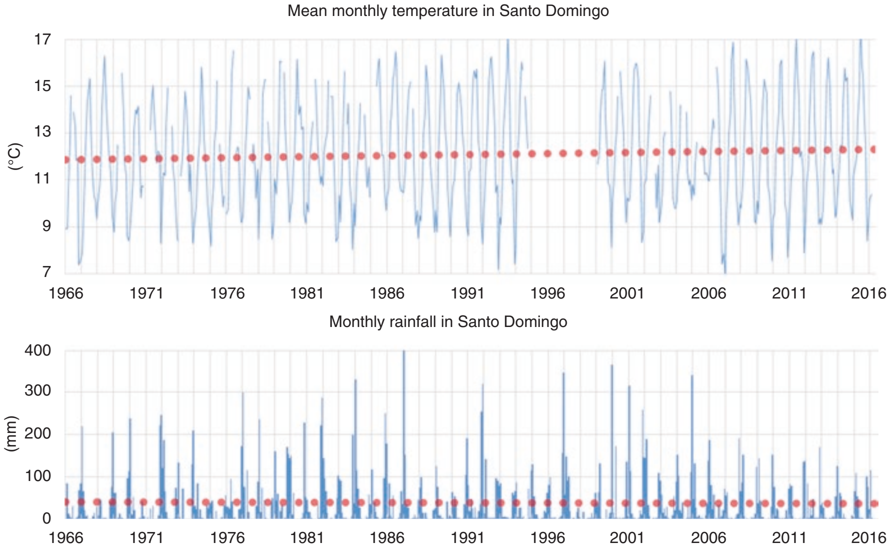

Fig. 8.3 Time series of temperature monthly averages (*top*) and monthly accumulated rainfall (*down*) registered during the last 50 years at aerodrome of Santo Domingo, far 15 km from the wetland. A 0.5 °C increment can be observed in ambient temperature and a decrease in rainfall, even when the latter present large intra-annual fluctuations associated to ENSO

hourly sea level record for almost 30 years, air temperature for 50 years, precipitations for 76 years, and three series of coastal water surface temperature allow to construct a series of 71 years, 45 years of water discharge nearby Maipo river mouth (delimiting the wetland to the north) and a 41 years record of Rapel dam (delimiting the wetland to the south) discharges. During the last years it has been possible to monitor the wetland by combining different instruments, measuring atmospheric variables with two weather stations and four temperature loggers. One of the weather stations was located in lagoon Matanzas. The available data is summarized in Table 8.1.

To estimate linear trends, all series were transformed to monthly series and a linear fit was applied by the minimum square method, determining the uncertainty at 95% confidence.

From 2010 on, some parameter of the water bodies have been measured in situ (temperature, pH, conductivity, dissolved oxygen). Additionally, with monthly frequency, the surface of coastal lagoons and lagoon Matanzas was measured by walking along their perimeter with a GPS whose positioning error is less than 3 m. This was complemented with satellite imagery and maps from the Military Geographical Institute, that allow having an almost inter-decadal sequence to observe the morphological changes experienced by the coastal lagoon and El Yali brook mouth.

El Yali National Reserve is located in a coastal region with template-warm climate, with winter rain and long dry season (7–8 months). This climate is characterized by a high cloudiness almost all year long, with major intensity in winter, associated to fog and drizzle and low thermic amplitude. Atmospheric humidity during this period is high. Precipitation is relatively more abundant than in northern regions, with over 350

Table 8.1 Available time series to characterize El Yali costal wetland

N	Parameter	Location	Start (D/M/Y)	End (D/M/Y)	Length (years)	Frequency of original record	Source
1	Sea level	Puerto San Antonio	02/08/1985	16/11/2014	29	Hourly	SHOA
2	Air temperature	Santo Domingo	01/08/1966	21/09/2016	50	Daily average	DMC
3	Air temperature	Albufera	09/03/2015	11/08/2016	1	Hourly	UPLA
4	Air temperature	Mirador Matanzas	09/03/2015	11/08/2016	1	Hourly	UPLA
5	Air temperature	Administración	09/03/2015	11/08/2016	1	Hourly	UPLA
6	Air temperature	El Yali-01	2013	2016	3	Hourly	UPLA
7	Air temperature	El Yali-02	2014	2016	2	1 Minute	UPLA/SERVIMET
8	Water temperature	Laguna Matanzas	09/03/2015	16/06/2016	1	Hourly	UPLA
9	Sea surface temperature	Valparaíso	01/01/2002	31/08/2016	14	Monthly	SHOA
10	Sea surface temperature	Valparaíso	01/01/1945	31/05/2011	66	Daily	SHOA
11	Sea surface temperature	Montemar	01/01/1961	31/12/1998	37	Daily	UV
12	Precipitation	Fundo Las Dos Puertas	01/06/1990	30/06/2016	26	Daily	DGA
13	Precipitation	Rapel	01/07/1940	31/07/2016	76	Daily	DGA
14	Precipitation	Cabimbao	01/09/2010	31/05/2016	6	Daily	DGA
15	Precipitation	Faro Punta Panul – San Antonio	01/01/1988	31/03/2016	28	Daily	DGA
16	Precipitation	Santo Domingo	01/08/1966	30/08/2016	50	Daily	DMC
17	Precipitation	El Yali-01	2013	2016	3	Hourly	UPLA
18	Precipitation	El Yali-02	2014	2016	2	1 Minute	UPLA/SERVIMET
19	Precipitation	Valparaíso	01/01/1900	31/12/2015	115	Annual	SERVIMET
20	River discharge	Río Maipo en Cabimbao	1971	2016		Daily	DGA
21	N° of days with discharge	Embalse Rapel	01/01/1975	01/08/2016	41	Daily	ENDESA
22	Wave parameters	Valparaíso	1963	2013	50	Every 3 hours	ULEAM

Long time series of sea surface temperature, ambient temperature, river discharge, rainfall, sea level I nearby locations and meteorological parameters in the wetlands system (although of shorter extension) are available. The series of ocean wave parameters correspond to summary statistics (significant wave height, wave direction, and wave period) reconstructed for this region

Sources: Universidad de Playa Ancha (UPLA), Servicio Meteorológico de la Armada de Chile (SERVIMET), Servicio Hidrográfico y Oceanográfico de la Armada de Chile (SHOA), Universidad de Valparaíso (UV), Dirección General de Aguas del Ministerio de Obras Públicas (DGA), Dirección Meteorológica de Chile (DMC), Empresa Nacional de Electricidad Sociedad Anónima (ENDESA) y Universidad Laica Eloy Alfaro de Manabí (ULEAM) mm/year, although there are 8 months with less than 40 mm precipitation. Precipitation concentrates from May to August, with more of 80% of the annual precipitation. Intensity of precipitations and wind in winter can even reach intensities proper of storms.

The main source of climatic variability in this zone is ENSO, a natural climatic event that is cyclic and recurrent (Espino 1999), but not periodic (Capel 1999), that occurs in the central Equatorial Pacific from 7000 years ago (Arntz and Farhbach 1996). This phenomenon is characterized by unusually warm and humid conditions in the eastern equatorial Pacific Ocean that generate sea surface temperature and wind anomalies in the tropical Pacific. These anomalies affect the whole planet (Ramesh and Murtugudde 2013) and represent the most important source of interannual climatic variability (Guevara 2008), perturbing the meteorological conditions through droughts, floods, and heat and cold waves (Johnson 2014); what turns into severe environmental impacts and effects on the economic activities around the world.

There are several indexes to characterize the ENSO phenomenon. The southern oscillation index (SOI) measures the difference of the monthly mean atmospheric pressure between Tahiti and Darwin. El Niño 3.4 measures the anomaly of the sea surface temperature (SST) in the 5°N–5°S, 120°–170°W region. Finally, the Oceanic Niño Index (ONI) allows to identify El Niño (warm) and La Niña (cool) events in the tropical Pacific. It is the running 3-month mean SST anomaly for the Niño 3.4 region. Events are defined as five consecutive overlapping 3 -month periods at or above the +0.5 °C anomaly for warm (El Niño) events and at or below the −0.5 anomaly for cold (La Niña) events. The threshold is further broken down into Weak (with a 0.5–0.9 SST anomaly), Moderate (1.0–1.4), Strong (1.5–1.9) and Very Strong (≥2.0) events. There is uncertainty about how ENSO is going to respond to contemporary global warming, even though and increase in its frequency is being observed.

A comparison between the monthly anomalies of the SST off Valparaíso and the El Niño 3.4 from 1945 to 2015 is shown in Fig. 8.4. Very strong El Niño 1982/1983 and 1997/1998 events are associated to important positive anomalies of SST. However, the last ENSO 2015/2016 did not show a significant SST anomaly.

The warm phase of ENSO, known as El Niño, presents an increase of the SST and a decrease of the easterlies in the Pacific Ocean. These anomalous conditions generate strong precipitations (Fernández and Fernández 2002; Aceituno 1992), and notable changes in climate (Pizarro and Montecinos 2004) and fisheries (Parada et al. 2013: Arcos et al. 2004; Cañón 2004; Valdivia and Arntz 1985; Alvial et al. 1994). These anomalies are observed in riverine countries of the Southeastern Pacific (Colombia, Ecuador, Peru and Chile) and other parts of the continent, like Panama (Corredor-Acosta et al. 2011), Mexico (Aguirre-Gómez et al. 2012), Bolivia (Miranda 1998), Brazil (De Meló 1998; Coutinho et al. 1998), Argentina (Capüto et al. 1998) and so distance places as the Indian Peninsula (Yadav 2012).

The cold phase of ENSO, known as La Niña, is characterized for presenting lower SST than normal, intensification of easterlies in the eastern Pacific Ocean and periods of droughts.

Precipitations registered in Rapel location, along with the ONI and the adjusted linear trend are shown in Fig. 8.5. A decrease rate can be observed. In general, large intra-annual variations in precipitation can be explained by variations of ONI, what is

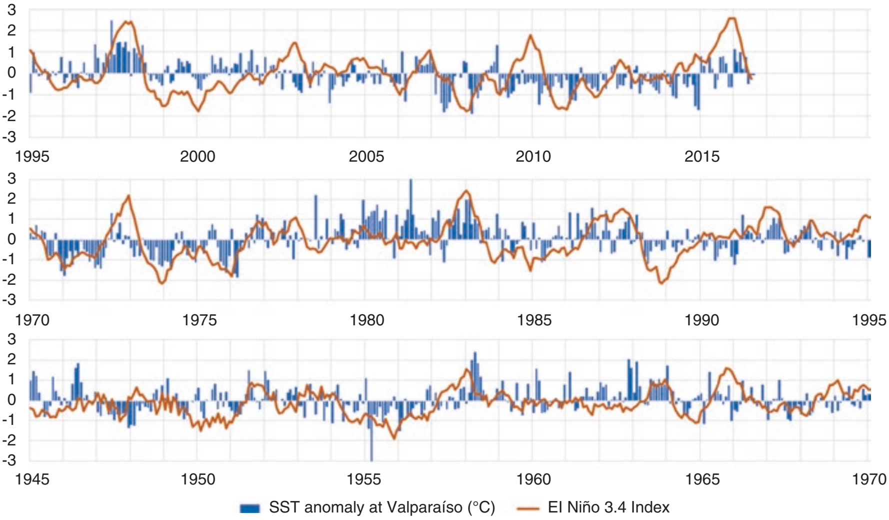

Fig. 8.4 Monthly SST anomaly (°C) off Valparaiso between 1945 and 2016. This series was constructed by means of correlations between Valparaíso and Montemar locations, in central Chile. It is also shown the monthly series of El Niño 3.4 index. In general, a good coherence exists between the monthly anomaly of SST off Valparaiso and El Niño 3.4 index

coherent with ENSO. Moreover, sharp oscillations in precipitations are observed but they are not always related to the ONI, as what was observed between 2001 and 2002.

Time series of Maipo river discharge at its mouth (20 km north of El Yali) is shown in Fig. 8.6. It is observed that the historic average of 200 $m^3 s^{-1}$ is exceeded many times, mostly associated to El Niño. The higher swellings correspond to years 1987/1988 and 2002/2003, which do not correspond to intense ENSO events. On the other hand, ENSO 1982/1983 and 1997/1998 have associated important swellings but not as severe as those observed during 1987/1988 y 2002/2003.

The mean se level (MSL) has increase during the recent decades due to the effects of thermic expansion and glacier, ice caps and polar ice melting. According to the IPCC (Magrin et al. 2014), the global mean sea level rose 0.19 m (0.17–0.21 m) during 1901–2010 and it is expected an increase of 0.26–0.82 during 2081–2100. In Chile, however, this variation is conditioned by the seismic activity between Nazca and South America plates. An analysis of 60 years records along Chile (Contreras et al. 2012) indicates that mean sea level variations significantly differ along Chile. In northern Chile, the MSL is decreasing at −1.4 mm/year (Arica, northernmost extreme of Chile), whereas in the center-south region the MSL is increasing at a rate of 2.2 mm/year (Puerto Williams, southernmost extreme of Chile). Predictions of conservative climate change scenarios for 2100 indicate a MSL rise between 0.2 and 0.3 m for different latitudes along the country; values that coincide with rates estimated by CEPAL (2015) and IPCC report (Magrin et al. 2014). By using numerical models, Albrecht and Shaffer (2016) Project MSL increase of 34–52 cm for RCP4.5 and 46–74 cm for RCP8.5 by the end of the twenty-first century.

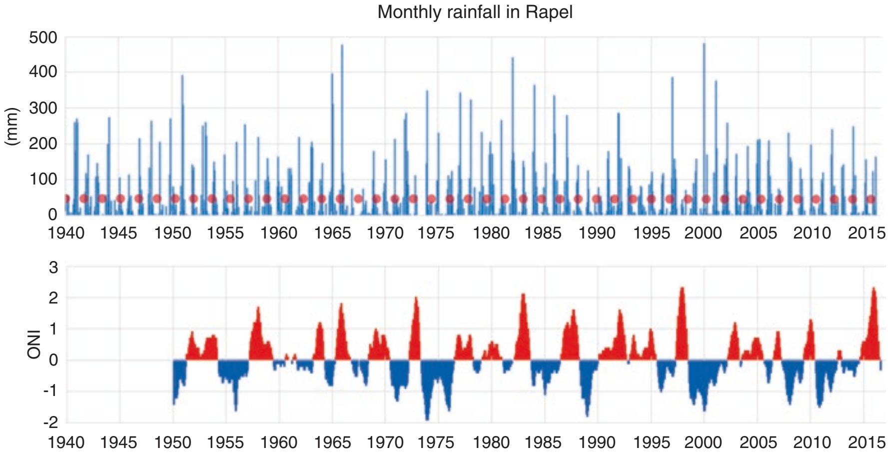

Fig. 8.5 Monthly precipitations in Rapel (1940–2016) and Oceanic Niño Index (ONI) (1950–2016). A negative lineal trend for precipitations fitted by means of minimum squares is also shown. Large intra-annual variations in precipitation can be explained by variations of ONI

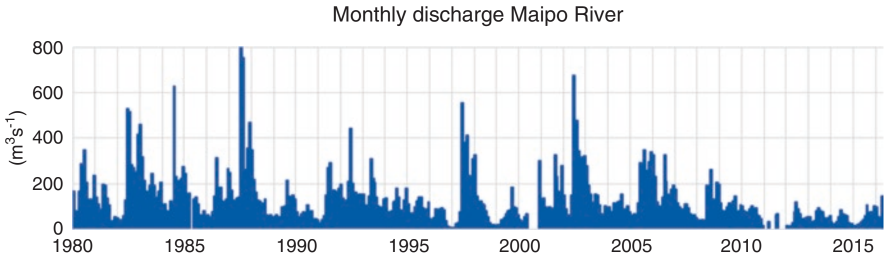

Fig. 8.6 Monthly river discharge (m^3 s^{-1}) at Maipo river mouth between 1939 and 2012. The higher swellings correspond to years 1987/1988 and 2002/2003, which do not correspond to intense ENSO events. On the other hand, ENSO 1982/1983 and 1997/1998 have associated important swellings but not as severe as those observed during 1987/1988 y 2002/2003

It must be considered that MSL records are modified by geologic processes, due to the convergence of tectonic plates (Wyss 1976; Albrecht and Shaffer 2016). Within seismic cycle, co-seismic (during earth quake) and inter-seismic (between earth quakes) movements result in coastal raising or subsidence. Cases where the instrumental record shows descend of NSL can indicate the tide gauge is being raised faster than the MSL raising caused by climate change. Co-seismic vertical movements can produce descend of meters in the MSL (e.g. Farías et al. 2010), which equals centuries of raising associated to climate change.

MSL is also affected by irregular cyclic oscillations like ENSO. Contreras et al. (2012) observed that ENSO can increase MSL monthly averages up to 30 cm during El Niño and reduce these values in the same order during La Niña. The increase of the MSL during El Niño is therefore comparable to the raising that would be observed by the end of the century.

In Fig. 8.7 are shown the variations of the MSL in San Antonio, 20 km to the north of El Yali wetland. An increase rate of 3 mm/year is observed (95% confidence). However, El Yali coastal lagoon is far more threatened by an earth quake that could raise it or sink it, and more frequently by ENSO. In this sense, the same figure shows the raising caused by the strong El Niño 1997/1998.

Waves are the main driver of littoral processes on the open coasts of central Chile. This stressor is described by means statistic parameters like significant height (SH), direction and period, which have experienced historic variations due to contemporary climate change. Molina et al. (2011) indicate a 10 cm increase has been observed in HS in central Chile and an alteration of 12° in wave direction. This agrees with Church et al. (2013, AR5), who foresees a 5% increase in the average SH for most part of Chilean territory, and with Izaguirre et al. (2013), who detected a positive trend in extreme wave heights for all South America coasts. Variations and trends of SWH and direction off Valparaíso are shown in Fig. 8.8. This wave corresponds to reconstructed deep water climate.

Different antecedents show that ocean swell is a frequent phenomenon in Chilean coasts. Campos-Caba et al. (2015); Campos-Caba (2016) identified 201 events with effects on the Chilean coasts between Valdivia and Arica, during 1823–2015. Sixty four of these events occurred before 1979. The other 137 events occurred after 1979, with an average of 4 swell/year and marked seasonality. Brito (2009), described several events of swells and storms between Valparaíso and San Antonio, based on historic antecedents. Available studies, however, have no enough statistic robustness – data prior to 1979 is essentially qualitative – to assess the vulnerability of coastal settlements to contemporary climate change. On the other hand, the lack of a permanent network of wave records along the cost makes it difficult to capture

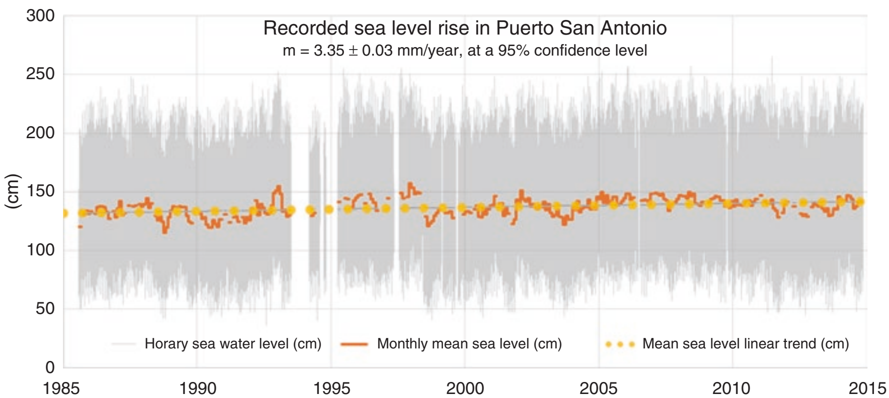

Fig. 8.7 Hourly sea level record in San Antonio port (1985–2014), 20 km to the north of El Yali wetland, along with monthly estimations of MSL and linear fit. An increase rate of 3 mm/year is observed (95% confidence). The figure shows the raising caused by the strong El Niño 1997/1998

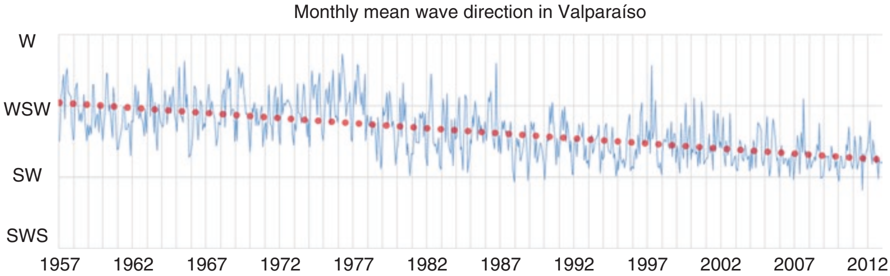

Fig. 8.8 Variation of incident wave direction off Valparaíso (1957–2013). This series corresponds to summary parameters reconstructed from satellite altimetry and records of wind and atmospheric pressure (Source: ULEAM)

Fig. 8.9 Comparison of Albufera (coastal lagoon) shore and costal dunes before and after the ocean swell in August 2015. Northward sight

storm events that are not represented by models used nowadays (Beyá and Winckler 2013; Vicuña et al.2014).

In August 2015, a strong ocean swell hit central Chile and severely affected the beach and coastal dune of El Yali, eroding the scarce material accumulated after the tsunami that occurred in 2010 (Fig. 8.9).

8.4 Earthquakes and Tsunamis

El Yali was severely affected by the earthquake and tsunami in 2010, when about 800 ha of beach and dunes were swept (Contreras 2014; Contreras-López 2014). The absence of these dunes has turned the wetland system more vulnerable to ocean swells, as occurred in August 2015 (Winckler et al. 2015). The recent earthquake and tsunami in September 2015 flooded about 280 ha too (Contreras-López et al. 2016).

The littoral zone of El Yali Natural Reserve is located 367 km far from the point where plates converge and that is part of one of the segments of the South American subduction area (von Huene et al. 1997; Pardo et al. 2001). In zone occurs earth quakes with important magnitudes and whose epicenter is close to the coast or above the deep seabed. These two conditions are sufficient to consider the area as of high tsunamigenic danger, due to the generation of near field tsunamis. Tsunamis are not always destructive when reaching the coasts, being able to reach from a few centimeters to several meters above sea level. A determinant factor is whether the arrival is under low-tide or high-tide conditions (mitigating or amplifying its effects). Likewise, it is necessary to consider a far field tsunami (generated in distant zones), whose contemporary example is the earth quake originated in Japan on 11 March 2011.

When consulting (a) Historic database about tsunamis in the Pacific (Gusiakov 2001), HTDB/Pac (2016); (b) World historic tsunami database (HTDB/WLD 2016), and (c) NOAA/WDC tsunami database (NOAA 2016), it is possible to infer that the coastal zone of El Yali Natural Reserve has been affected by eleven tsunamis since 1900 to present time: (1) Valparaíso 1906 Mw 8.2; (2) Constitución 1928 Mw 7.6; (3) Alaska 1946 Mw 8.1; (4) Rusia 1952 Mw 9.0; (5) Alaska 1957 Mw 8.6; (6) Valdivia 1960 Mw 9.5; (7) Alaska 1964 Mw 9.2; (8) Valparaíso 1971 Mw 7.5; (9) San Antonio 1985 Mw 8.0; (10) Cobquecura 2010 Mw 8.8; (11) Japón 2011 Mw 9.1 and (12) Illapel 2015 Mw 8.3 (Table 8.2).

As mentioned, El Yali National Reserve was very affected by the Mw = 8.8 27 February 2010 earthquake (Rubio and Basic 2011; Fariña et al. 2010). The tsunami wave entered, on average, more than 1 km inland, reaching 2 km in some cases and flooding about 800 ha, 200 ha corresponded to protected areas. This event affected mostly the lagoon, due to the break of the sandbar, producing a sever loss of the equilibrium in the ecosystem.

This event provoked changes in the wetland ecosystem, destroying the vegetation and leaving many dead birds, besides altering nesting and resting sites. Moreover, the landscape was altered because the wave transported algae and clastic debris of a wide granulometric spectrum, domestic garbage, fishery material and wreckage amounting 10 tons.

Due to the destruction of the dune by the tsunami, the first notorious change that was observed was the permanent connection of the Albufera with the sea in at least three sites. These connections would gradually close create, 6 months later, a drastic

Table 8.2 Summary of earthquakes generating tsunamis and that possibly have affected El Yali wetland

N°	Year	Month	Day	Mw	Country	Field	Runup Valparaíso	Runup San Antonio	Runup El Yali	Uplift
1	1906	8	17	8.2	Chile	Near	>0	>0		>0
2	1928	12	1	7.6	Chile	Near	>0			
3	1946	4	1	8.1	EEUU	Far	1.60			
4	1952	11	4	9.0	Russia	Far	1.80			
5	1957	3	9	8.6	EEUU	Far	2.10			
6	1960	5	21	9.5	Chile	Near	1.70			
7	1964	3	28	9.2	EEUU	Far	2.20			
8	1971	7	9	7.5	Chile	Near	1.20			+0.10
9	1985	3	3	8.0	Chile	Near	1.15	3.50		+0.50
10	2010	2	27	8.8	Chile	Near	2.61	3.90	5.20	−0.35
11	2011	3	21	9.0	Japan	Far	1.54	0.89	>1.00	0
12	2015	9	16	8.3	Chile	Near	2.00	1.00	2.80	

Source: NOAA (2016), HTDB/Pac (2016), HTDB/WLD 2016, IOC (2016), Fritz et al. (2011), Contreras-López et al. (2016), and Quezada et al. (2012b). 2011 run-up record corresponds to the autor observation that has not been published yet
Only the last three events have been registered with instruments so as to prove the occurrence of tsunamis hitting the wetland. Runup and uplift values are in meters (m)

segmentation of the coastal lagoon into two water bodies, one of which would completely disappear 2 years later.

In 2011, the far-field tsunami generated in Japan also hit El Yali river mouth and the coastal lagoon. When the sandbar and coastal dunes did not exist anymore, the tsunami waves trespassed the south part of the beach and flooded some meadows that were not reached like this since middle of the twentieth century.

During these years, it was possible to appreciate a moderate recovery of coastal dunes, at lower rates than in other places that were also affected by the tsunami 2010. However, any disturbance provoked the disappearance of the small coastal dunes being formed.

In 2015, a month after the occurrence of a severe ocean swell that erodes the beach and coastal dune, a local tsunami that had destructive effects 200 km to the north overpassed the Albufera (Contreras-López et al. 2016). In Fig. 8.10 is shown the beach and remnants of the coastal dune before and after the ocean swell, and before and immediately after the last tsunami that affected the wetland.

It has not been possible to find testimonies of what occurred in this site in 1985, when a Mw 8.0 earthquake had its epicenter nearby the wetland. It is known that this earthquake generated a tsunami in San Antonio port and, therefore, we can infer that it was also destructive in El Yali. Likewise, it is possible that the transoceanic tsunami that followed the Valdivia earthquake Mw 9.5 in 1960 must have had similar effects to what occurred in 2010.

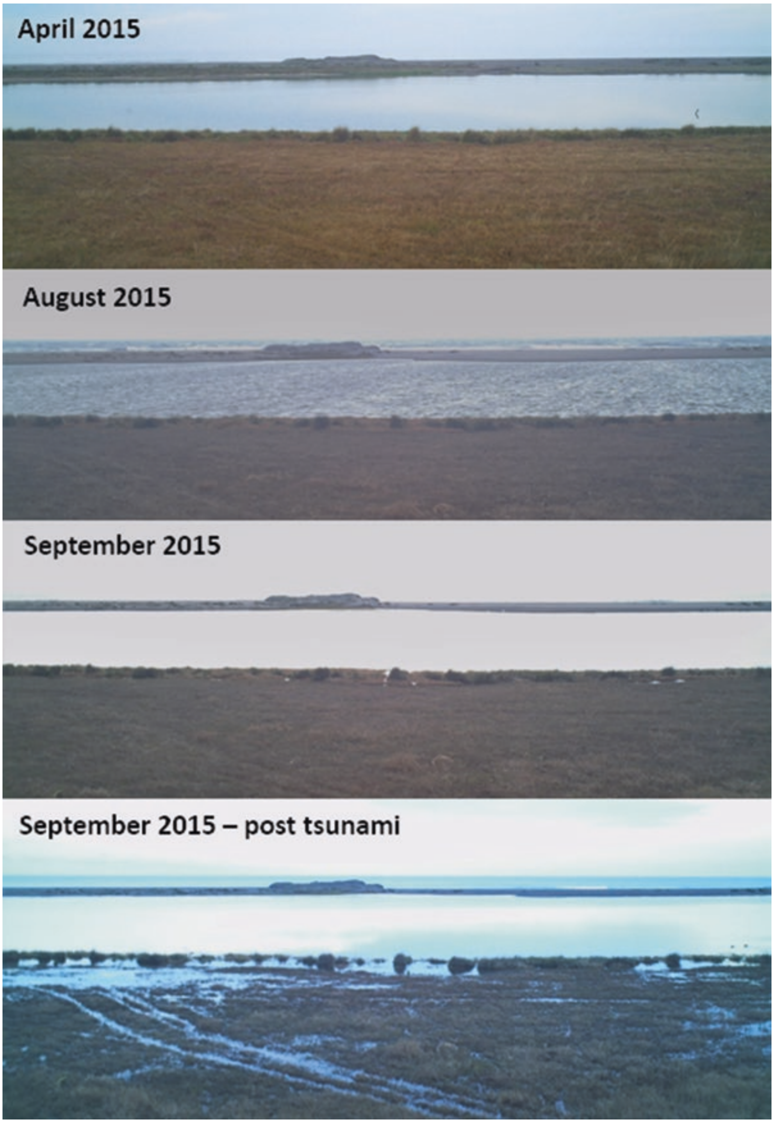

Fig. 8.10 Sights from a camera located on the south shore. It is possible to observe the effect of the ocean swell between April and August. The image captured in September was taken the day before the tsunami, whereas the next one was taken right after the waves

8.5 Evolution of Waterbodies

As it has been pointed out, El Yali brook mouth and the lagoon have experienced drastic changes in their recent morphology due to the effect of the 2010 tsunami. The destruction of coastal dunes implied an increase in the vulnerability of the lagoon ecosystem before ocean swells and tsunamis, as was demonstrated with the far field tsunami of Japan 2011, the ocean swell in August 2015 and the tsunami occurred in September 2015. In the case of El Yali, coastal dunes were doubly affected: first, by the impact and second, by the creeping mechanism; water eroded a volume of sediments from the area which was no quantified but estimated as significant when comparing the current dune field with pre-tsunami images. However, when analyzing historic geographic charts from the Military Geographic Institute and the recent evolution of the dune field, it is possible to appreciate the instability and fragility of the system since 1923 until present days (Fig. 8.11). The coastal lagoon is located where El Yali brook mouth used to be. This mouth moved to the south. Presumably, in many opportunities during the past these dunes were destroyed by intense tsunamis and/or ocean swells. However, they were reconstructed with the important sediment load from Rapel River. After Rapel dam was constructed (which allowed damming Rapel river in 1968), imposing a definitive cutoff of this continuous sand supply, the current situation suggests that this system lacks its natural regenerative capacity or requires a longer time to recover, since the volume of sediments that Rapel river used to supply experienced a sharp decrease after the dam.

Rapel hydroelectric central (34°02′S; 71°35′W) consists of a concrete vault which upper part has a curvature radius of 174 m and 350 m long. The wall height is 112 m and the dam can content 832 million m^3, with a maximum depth of 90 m (Vila et al. 1986). The importance of this reservoir for El Yali wetland arises because the material forming the dunes and the long beach surrounding it corresponds to material originally coming from Maipo River (Paskoff and Manríquez 2004). Because the river was dammed, this process does not longer occur in a natural fashion since 1968 (Paskoff et al. 2000).

The morphological evolution of the Albufera seems to react to the occurrence of tsunamis that affect the wetland. For instance, when comparing Table 8.2 with Fig. 8.11, it is observed that the most drastic changes in the shape of this wetland coincidently occurred right after a tsunamigenic earthquake. The segmentation that the Albufera suffered in 2010 was well documented (Contreras 2014), but also drastic changes in the shape of the river and the Albufera after tsunamis of 1960 and 1985 can still be observed. However, even when the tsunami of 1960 must have had similar characteristics to the tsunami of 2010, Rapel River was not dammed yet and therefore the beach and dunes recovered as rapidly as other similar systems, as Mataquito river, did after the earthquake of 2010. In 1985, a new tsunami occurred in this area and produced a 50 cm vertical co-seismic raising (Quezada et al. 2012b). Somehow, this co-seismic raising, unlike the sinking observed in 2010, mitigated the lack of sedimentary material discharged by Rapel River. On the other hand, due to the high precipitations occurred before and after 1985, the dam gates were opened in several occasions (Fig. 8.12.)

It is presumed that the poor recovery of the coastal dune and beach that used to protect the wetland is related to the closure of the dam gates since 2009 (Fig. 8.12). On the contrary, the recent recovery of this waterbody seems to respond to the opening of the dam gates, allowing supply of sedimentary material. The monthly evolution of Matanza lagoon and the Albufera surfaces is shown in Fig. 8.13.

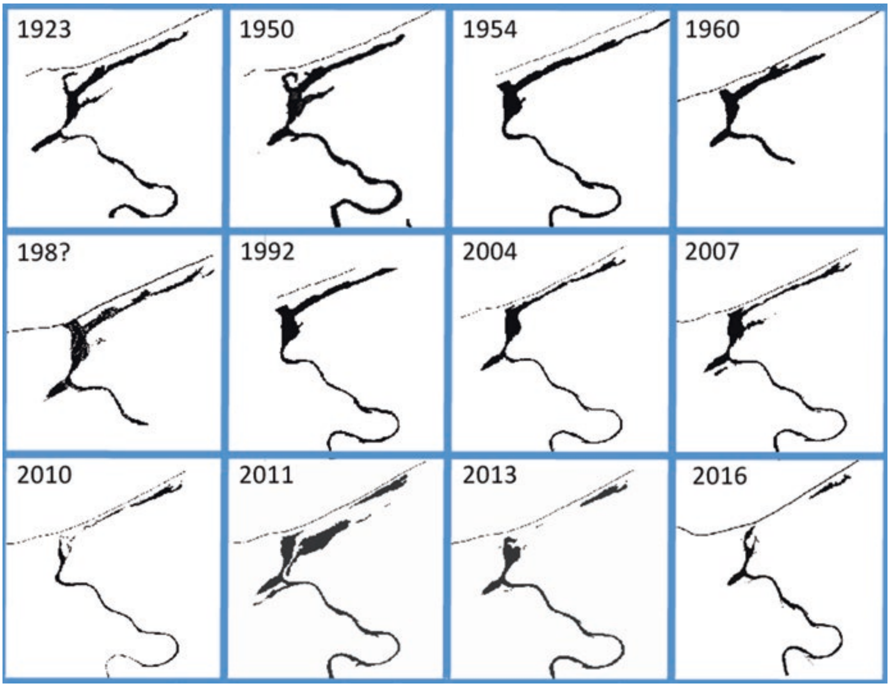

Fig. 8.11 Morphological evolution of El Yali brook mouth and the Albufera (1923–2016). Important changes are observed during the 1960s, after the 80 decade and during 2010

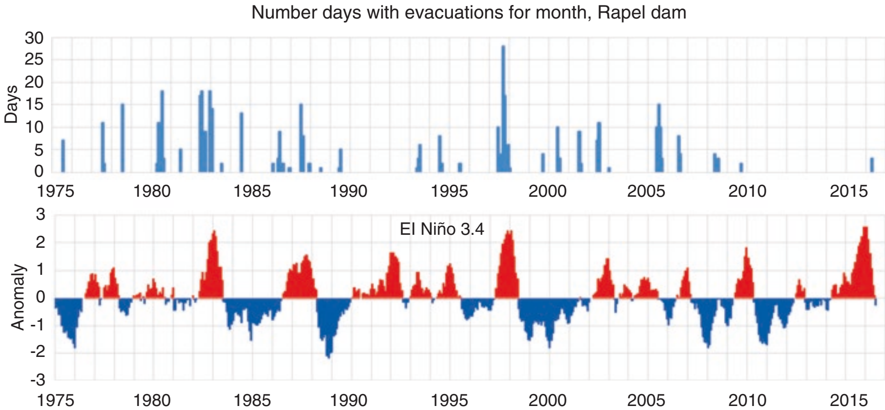

Fig. 8.12 Number of days by month when Rapel dam gates were opened, allowing the supply of sediment to the beach of El Yali wetland. ENOS 3.4 index is also shown and it is observed that, especially in the past century, there used to be a correlation between the opening of the dam gates and the index that indicates the anomalous increase of temperature in the 5°N–5°S, 120°–170° W region

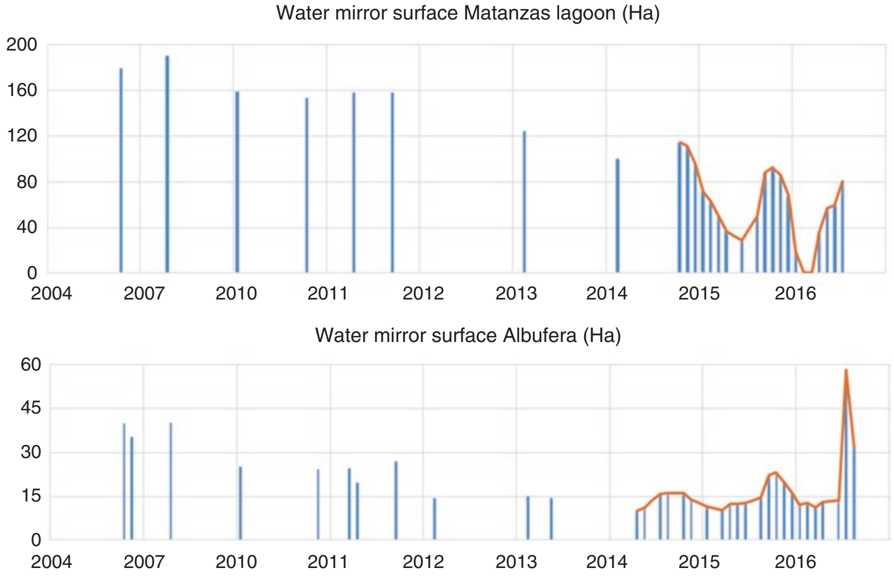

Fig. 8.13 Monthly evolution of Matanzas lagoon and the Albufera surfaces. The Surface extension is estimated from tracks along the border of the waterbodies with a GPS. It is observed that Matanzas lagoon has reached historical minimal levels, whereas the Albufera has experienced a decrease after 2010 and has shown a slight recovery only during the last months

In Fig. 8.13 is possible to observe how the Albufera experienced a reduction in its size after the tsunami 2010, which is explained by the segmentation and desiccation of the lagoon. A peak is observed during the last months, which corresponded to a flooding of the lagoon after the ocean swell. However, unlike past events, the recent opening of the dam gates (Fig. 8.12) seems to have delivered enough material to retain more water in the Albufera. A recovery of the old desiccated segment has been observed during these months.

8.6 Proposal of Ecological Restoration

As it is known, a proposal of ecological restoration must begin by defining the object on which the restoration will be focused. This is particularly difficult in coastal wetlands like El Yali, located in one of the zones most intervened by the man in the littoral of central Chile. There are antecedents about how different pre-Hispanic cultures, including Inca, Aconcagua and Llolleo cultures (Falabella and Planella 1980, 1985), have altered these waterbodies. Afterwards, with arrival of Spaniards, the first farms were established in this area. Matanzas (“slaughters”) lagoon owes its name because it used to be the place where many heads of cattle were sacrificed to produce salad meat to be sent to Lima, Peru (Brito 2009).

However, the wetland system continued playing a fundamental role for migratory birds and endemic biodiversity of central Chile until the beginning of the twenty-first century. With the tsunami in 2010, it was clear that the construction of Rapel dam was a serious disturb because it cut off the sediment supply that sustained beaches and the dunes that protected this wetland. In many opportunities, this wetland was affected by quakes and tsunamis, intense ocean swell and ENSO, all phenomena able to dissipate their energy by destroying that natural barriers represented by the dunes. Before the dam construction, the continuous sediment supply was able to rapidly reconstruct the beach, sandbar and dunes, as was observed, for instance, in Mataquito river mouth.

When the sedimentary supply was cutoff, the dunes still had enough material to sustain the system equilibrium for some decades but when tsunami waves destroyed these dunes, there was no material to recover neither the beach nor dunes. This generated a substantial change in the system, turning it more vulnerable to other (even smaller) tsunamis, ocean swell, sea level rise associated to ENSO Kelvin waves, and climate change. For this reason, the first proposed restoring action is to reconstruct the dunes and reinforce them with a vegetal cover composed by native species, which used to cover the dunes destroyed by the tsunami (Caldichoury 2002). The other two steps are: (1) management of exotic forest species located around Matanzas lagoon and that appear to increase the sensibility to water scarcity and (2) planting native vegetal species around the waterbodies, as green fences that contribute to generate buffer areas in the wetland complex.

8.7 Discussion

As observed in records from around El Yali wetland, coastal wetlands in central Chile are facing pressure from the contemporary climate change: Besides the rising of the ambient temperature, the decrease of precipitations and the sea level rise, it possible to observe an increase of the sea surface temperature, a decrease of nearby rivers discharge, and a modification of the incident wave direction and significant wave height. These effects are proved by fitting a linear trend with positive or negative slopes, depending on the particular variable (Figs. 8.3, 8.4, 8.5, 8.6, 8.7, and 8.8). However, due to the occurrence of ENSO, it is possible to distinguish an important intra-annual climatic variability in those series. Years of intense precipitations are followed by years of hydric scarcity, due to the warm and cold ENSO phases, respectively. However, this typical behavior has deviated during the present century, where a strong El Niño occurred but has not been accompanied by intense rainfall around El Yali. This explains the continuous decrease of the water surface observed in the inner Matanzas lagoon (Fig. 8.13).

The delicate equilibrium at the Albufera was severely affected in two situations: (a) the cutoff of the sedimentary supply after the damming of Rapel River in 1968 and (b) the removal of sediments from beach, sandbar and coastal dunes of El Yali after the tsunami 2010. Thus a combination of anthropic alteration along with the occurrence of a natural phenomenon was able to neutralize the natural mechanism of protection and dissipation of energy that El Yali wetlands system used to have.

As result of climate change, it is possible to infer that another important alteration is occurring: From El Yali River northward, the beach is oriented from SSW to NNE and the littoral border runs parallel to the incident direction of waves from SW, increasing the attenuation of wave energy. As the marine terrace is very wide, it provides space to create an extended system of dunes cordons, transversal free dunes, dunes restrained by vegetation, along with coastal dunes on depressions and wetlands around the Albufera. However, if there is a change in the incident waves (Fig. 8.8), the beach orientation will lose this characteristic, implying a change in the costal morphology. This way, there is a natural experimental design: the system was completely reset after the earthquake and tsunami 2010, there are no continuous sediment discharges since 1968 but the only occur when the dam gates are open and consequently it is possible to know the exact moment when new sedimentary material is delivered to the system. Finally, the direction of the incident waves is gradually changing.

It is important to observe that sea-level raising in central Chile coasts seems to be no relevant as threat to coastal wetlands, considering the alterations represented by co-seismic vertical movements associated to earthquakes occurring in this region. As observed in Table 8.2, the 1985 earthquake meant a rising of 50 cm in San Antonio port. When comparing this elevation with the actual increment rate of the sea-level, about 3 mm/year, it means that the earthquake made the sea-level rise to retrogress the equivalent to 167 years. On the other hand, the descent of 35 cm during the earthquake in 2010 implied an advance equivalent to about 100 years.

Finally, besides climate change, ENSO, tsunamis and earthquakes, it must be considered the intense anthropic pressure that seeks the accelerated conversion of these wetland systems to real estate, agriculture and recreational systems. In central Chile, wetlands are the subject of anthropic perturbations due to the land use (e.g. exotic forest plantations, specific and diffuse contamination, etc.), resulting in alterations of their structure and functionality (Muñoz-Pedreros and Merino 2014).

According to Fariña et al. (2012), currently, in spite of management plans (CONAF 2009), El Yali wetland is under a series of menaces that put its biodiversity in risk: extraction of groundwater, deviation of water streams for agriculture or industry, urban expansion, contamination and eutrophication of streams, lagoons and wetlands, reduction of native vegetation by fires, agriculture, livestock and forestry activities, furtive hunters, 4WD vehicles, powered parachute disturbing birds, and electric cables causing birds dead.

8.8 Conclusions

The natural history of the wetlands complex in El Yali basin suggests that the equilibrium of this system is sensitive to the occurrence of natural phenomena like the tsunami in 2010, ENSO and climatic variations in general. It is expectable, therefore, that contemporary climate change produces alterations due to the change in the hydric regime forced by precipitations, modifications of ambient temperature and alterations in the coastal zone by the elevations of the MSL, what, added to anthropic pressure, put the wetland sustainability in risk in the long term.

Previous lagoons were also coastal lagoons but due to displacements and co - seismic raising, they are in their current location (10 or more meters above sea level and some km inland). However, they still conserve the saline characteristics that support the abundance and biodiversity of animals and plants.

El Yali was severely affected by the earthquake and tsunami in 2010, destroying 800 ha of beaches that used to be protected by a dune now disappeared (Contreras 2014). The absence of thus dune has turned the system more vulnerable before any ocean swell, as those occurred in August 2015 (Winckler et al. 2015). The recent earthquake and tsunami of 16 September 2015 also flooded about 280 ha in this area (Contreras-López et al. 2016), showing once more the vulnerability of the system.

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Part II
Impacts of Coastal Engineering and Environmental Degradation

Chapter 9
Physical and Morphological Changes to Wetlands Induced by Coastal Structures

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Abstract This document is focused on the establishment of a methodology to assess erosive processes in a coastal wetland. Particularly, it analyses the spit that separates the lagoon from the sea, elaborating a diagnostic process that helps to characterize the effect of the coastal infrastructure in morphological changes in a short and medium-term. Elements such as the morphology, the wave climate, the hydrodynamic and the evolution monitoring of coastline are key elements to understand whether a coastal wetland is on equilibrium or in the contrary, its state of vulnerability is such that in the slightest change in physics conditions will produce negative effects by the system instability.

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Generally, it describes the procedure performed to properly understand the relation between modifications of coastal processes and the response of a coastal environment. It uses numerical and theoretical models to assess the behavior of the waterfront, considering the historical changes that have occurred to ultimately predict variations of the spit as consequence of the establishment of new civil works. Finally, it concludes with this method of analysis that the evaluated study case will be affected by the works of action to be developed for facilitating the navigability conditions of a new port currently under construction in the city of Barranquilla, Colombia.

Keywords Ciénaga de Mallorquín • Coastal wetland • Coastal erosion • Morphological changes • Port • SANDY® • Wave climate

9.1 Introduction

A coastal lagoon is a body of water that is separated from the ocean by a coastal bar, barrier island or any coastal formation, as a coastal spit. It is nurtured by fresh water from the continental part; and seawater from the marine border (Kjerfve 1986). The coastal lagoons can be connected to other bodies of water through natural channels, or in some cases through artificially built channels.

Wetlands are areas located underwater during different seasons; have hydrophilic vegetation; and are regulated by hydrologic regime and by influence of marine zone, in particular by the tide rise. Within the classification of coastal wetlands it can be found: (a) swamp; (b) estuary; (c) coastal lagoons; and (d) mire. They have an important role in the hydrologic equilibrium and the regulation of the planet climate. They are diverse systems that harbour a large number of ecosystems (Buenfil 2009).

According to the Ramsar Convention (1999), a coastal wetland contains the extensions of marshlands, swamp and mire; surfaces covered by water in natural or artificial regimen, with permanent or transitory flow, stagnant or with continuous flow; freshwater, saltwater. Besides, it includes maritime extensions where the minimum tide level (low tide) that do not exceed 6 m of depth.

From the hydrological perspective, coastal wetlands can be classified into:

Marines. Coastal wetlands and lagoons, including coral reefs.
Estuarines. deltas, tidal marsh, and mangroves.
Lacustrine. Wetlands associated to large continentals.
Riverine: Wetlands adjacent to rivers and streams.
Palustrine: Swamp and Lagoon.

According to Mitsch and Gosselink (1993), the identification of coastal wetlands is regulated by three criteria: (a) it must be a covered water surface; (b) it has hydrophilic, marshy, saturated or flooded lands; and (c) it has hydrophilic vegetation that are plants that live in flooded zones.

The most important physical processes that involve in coastal wetlands are the currents tidal and the sediment transport (Brunn 1994). Furthermore, there the variations of phreatic level associated with the contributions of river and maritime waters, the residues discharges and even, the wind action.

In relation to landforms, the bars of coastal wetlands can be continuous or discontinuous given the oscillation of the astronomical tide. In some situations by the action of the meteorological tide and processes of short scale that modify the hydraulic behaviour of the body of water such as the salinity intrusion.

Analysing these water bodies, in terms of the mass balance equation, it is possible to identify contributions that influence in the hydraulic of wetlands, such the flow deviations by hydraulic works and the precipitation. The marine border not only provides a transitory flow of seawater due to the tide; so it has a saline layer that notably affects in the wetland hydrodynamics. The outputs are system losses as product of the evaporation; the tide ebb current; and in lesser extent, the opening of artificial mouths in bars and edges that surround the body of water.

Coastal processes have a major role in the dynamics of a coastal wetland, influencing not only in the variation of the water level or in the hydrodynamics behaviour as mentioned in the previous paragraph, it also can change the physic-chemical features of a wetland. Changes in salinity levels are giving by the tidal rise flow through a lagoon mouth that generates discontinuity in the coastline. Dimensions of this natural channel regulate the amount of flow entering, determining the flow velocity that dominates the mouth behaviour and the stability. According to studies made by Brunn and Gerritsen (1961), flows can reach in this tide channels up to 1 m/s.

Despite its importance from the physical and environmental perspective, these systems are under strong anthropogenic impacts. The most common degradation activities are: (a) deforestation; (b) urbanization; (c) water contamination; and (d) restriction of water flows.

In virtue of the aforementioned, this work is focused on the definition of a methodological approach that allows assessing the affections that negatively impact in the coastal bars stability of wetlands. Particularly, the works of coastal infrastructure can generate an imbalance that entails to the wetland disappearance in long-term. The effects of the defence and shelter works such as breakwaters or jetties perpendicularly arranged to the coastline are characterized by the methodology of proposed analysis.

Synthetically, the document seeks to address basic concepts but relevant for the coastal geomorphology which allows defining the study zone through the establishment of a littoral cell. Subsequently, it assesses sediments features in order to understand how the material is transported by currents from the source to the end of calculus domain. No diagnostic study that seeks to understand the integral behaviour, can omit the wave climate weather of a littoral zone so the waves, the tidal and the currents, not only model the shape of the bar; but also, they have a fundamental role in the wetland equilibrium. Alongside, the study of maritime weather is carried out a hydro-morphological model that allows observing how currents interact with sediments. Then, a temporal evolution of the bar is made, analyzing the role that have the structures to modify the sediments sources and the bar configuration that protect the wetland. Finally, it is developed a plant modelling of the bar to represent morphological changes that can generate a maritime work in long-term.

9.2 Geomorphological Aspects of Coastal Wetlands

9.2.1 Mallorquín Lagoon

Geomorphology is the science that focuses its study on the terrestrial and marine relief present on the planet (Ruiz 2009). This science centres its interest in the detailed analysis of beaches configuration, phenomena and processes that regulates its evolution and physical conditions that, from present to the future, condition the behaviour of a coastal environment (Table 9.1).

The coastal zone is a dynamic environment from many perspectives. From the geomorphology, it is possible to observe different variations made in a wide range of temporal and spatial scales (Del Río and Gracia 2009). Changes that occurred in a wetland spit are described by advances and relapses of coastline that seasonally vary when the beach profile is on dynamic equilibrium. However, the erosive process appears like a modification of natural state of this evolutionary process which has an anthropic and natural origin (Van Rijn 1993).

Conceptually, the word coastline is used to define the boundary that exists between the land and the surface of a body of water, independently of a temporal defined scale; and it assumes that the coastal zone is the land and water area that borders the coastline, extending to a point where it covers all coastal processes that

Table 9.1 Phenomena and process interaction in geomorphology process

Transport	River
	Erosion
	Wetlands
	Storms
Wave climate	Temperature
	Precipitation
	Evapotranspiration
	Heat flows
	Wind
Level variations	Climate change
	Subsidence phenomena
	Eustatic
Anthropogenic impacts	Coastal infrastructure
Coastal process	Currents
	Water waves
	Tides

are involved in the marine dynamic. Beaches of these environments, like those associated to wetlands, can be characterized as emerged or submerged thanks to geological processes, as the subsidence or raising, that cause that the medium level migrates above or below to the average of the historic sea level.

The shoreline of coastal lagoon is under the classification of barrier island formed by a coastal bar whose geometric configuration in plant and profile is function of wave features that arrives to its coastal zone. Generally, these barrier systems are integrated to a body of water with salty features or estuarines when they have communication with any river source. These systems, that are known as wetlands or coastal lagoons save a large biodiversity and have a fundamental role as weather regulators. From the marine environment perspective, these natural elements have a beach profile of thin width, an alignment that allows the dissipation of wave energy and also, permits the sediment transport along the littoral.

Currently, one of the most common problems in littoral zones of coastal wetlands is the coastal erosion caused by anthropic or natural impacts. For the analysis, it is used morphologic temporal domains that vary in function of the shore affectations. Therefore, it defines three stages known as short, medium and long-term. In the long-term (years-thousand of years), the erosion process is manifested in a special domain of regional context and it is associated with events derived from the sea level rise and geological changes (Cooper and Pilkey 2004; Javrejeva et al. 2012). In intermediate scales (years, decades), usually the erosive phenomena is influenced by an unbalance in sediment transport rates as product of the inappropriate establishment of coastal structures or interventions in the upper part of the basin, as the reservoirs to the energy generation or flood abatement (Komar 1998; Cooper and Pilkey 2004). In the short-term, it is related with waves with a high content of energy, wave trains that directly influence in the beach where all their energy is dissipated, and any coastal process resulting in the loss of material from the beach.

The resilience of coastal bar that protects a wetland is assessed from its regulatory capacity with external forces, or from a mechanical perspective, when it is capable to maintain a mass balance in sedimentary terms. On this subject, it can mention the positive or negative regulation to characterize processes through which the beach assures a proper functioning (Ruiz 2009). The positive behaviour occurs in a waterfront that remains invariable in spatial and temporal terms; regardless of the solicitations exposed. The negative scenario is reflected when exists a temporal variation of sediments so the system adapts to the forcings of maritime weather without losing control.

The interpretation of that resilience is what defines the concept of equilibrium in a bar or in a coastline given that it can be affirmed with certainty that a beach has equilibrium when the quantity of gained or loosed material remains unchanged in time, and mechanically represents a conservative phenomenon. It is a concept that allows to assess the variability of beach shape in a temporal threshold and it is used as support to predict changes that will happen in the bar for variations in conditions that govern the coastal dynamic. Therefore, the static equilibrium is the state in which the bar is allocated when not appreciably variance in time. The slightest change in the configuration during a period of short and medium-term implies a transference to the domain of dynamic equilibrium.

There is another behaviour related to seasonal variations during a time interval in the spit. The coastline of the bar changes its morphological aspect until reaching the equilibrium (static or dynamic), but this is a regular state which is infinite, given the incidence and effect of meteo-marine conditions. According to Woodroffe (2003), a spit that has these features can be classified as metastable.

According to Ruiz (2006), the geometric configuration of coastline is a physical representation which shows the sediments response to process that act on it. In consequence, the spatial domain of a coastline can be analyzed from two approaches; for one hand, assessing the variation in plant and for the other one, with the transversal section of a beach.

The development of plant analysis must be performed by dividing the spit bar or the beach that separates the wetland in differential spatial increments within a coastal cell (Fig. 9.1) that by physical factors presents some morphological features of that particular environment. The idea of carrying out this analysis is to find any erosion sign or sediments accumulation throughout time.

Analyzing the wave climate, it can be observed that the main modeller of marine environment in terms of erosion is the waves generated by the wind. There are other secondary variables with a bold influence in erosion phenomena as: (a) weathering of rocky environments; (b) the tidal range; and (c) the configuration of beach.

By virtue of described processes, the most significant morphological features or frequently more observed in coasts and that are associated with the topic of coast erosion are cliffs, coastal terraces, natural arches, caves, bays or coves and the abrasion platform (Thornbury 1954).

In relation to the material deposition on the coast, this process owes its existence to the sustained action and medium intensity of waves that carry the material towards the beach. Most of coasts in the world are composed by sand and in some cases, by gravel; but sediment sources are originated from the material transported by rivers that can generate deltas that sustain extensive beaches. Another scenario consists in beaches defined by cliffs that are affected by erosion and also support the coastal zone. Based on these sediments, there are different coastal shapes such as beaches, bar, spit, tombolo and submerged bars.

Currently, the coastal bar of Mallorquín Lagoon has 3.5 km of length and an average width of 96.7 m of variable section. The recent lagoon mouth that connects the sea has a length of 88.41 m and is located at the east of the groin, at a distance of 1.4 km. The littoral coast is confined between Madalena River discharge (close to the coastal groin) and León stream (to the west), which intermittently flows into the sea and commonly finalizes its flow in the Mallorquín wetland (Fig. 9.1).

In the littoral bar of the coastal lagoon can be found three types of sedimentary environments: (a) beach environment; (b) narrow and open lagoon environmental ambient; and (c) restricted and protected lagoon environment. With regard to the first feature, it can observe that the beach environment is affected by a swell perpendicularly arriving at the coast and waves diffracted by the coastal structure. Besides, it can observe a coastal spit 2 km away from the coastline which is adhered to the

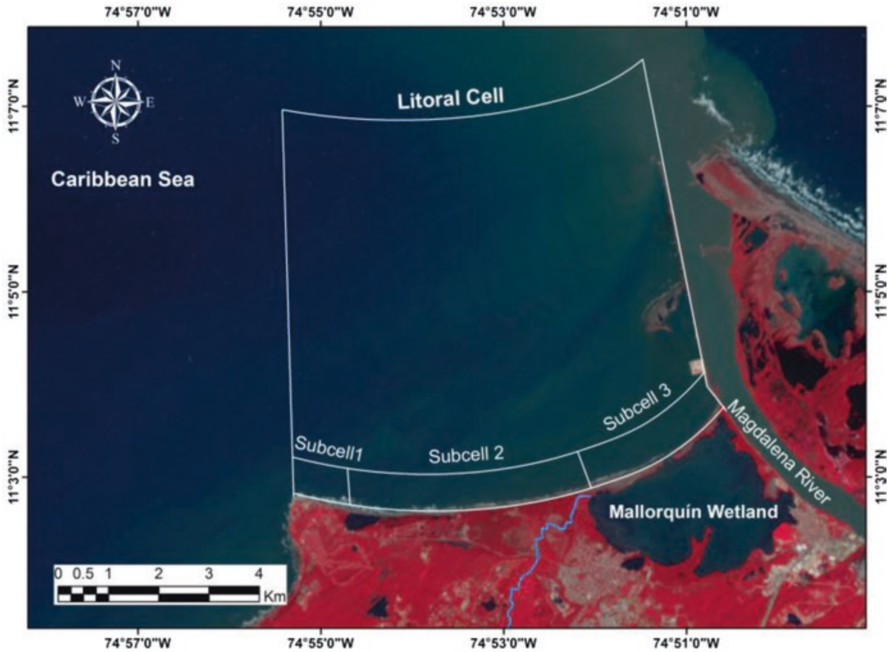

Fig. 9.1 Littoral cell of the Mallorquín Lagoon based on Landsat imaging 2015

coastal protection work (Fig. 9.2). The coastal spit behaviour is very dynamic, increasing its size during the dry season and inversely, it loses materials during the winter season when the waves has a high energetic content.

Analyzing the shoreline lagoon, it can be observed that it is part of a strip of sediments that flows parallel to the coast from river mouth ("Bocas de Ceniza"). The waves overcomes the coastal protection and sediment is transported by the diffraction to the coastal spit and subsequently, to the beach. Therefore, the morphological configuration of the bar is function of diffracted and refracted waves that travel in a submarine platform relatively low.

The characterization of the wetland morphological configuration is based on a classification that considers the processes and phenomena of marine environment, through the establishment of differential increases or littoral cells. This methodology allows identifying the relations among physical variables, forcings and natural restoration mechanisms of the system. The level of detail of this multilevel approach depends on the focus, the project or the processes scale that require to be assessed.

There are three approaches or methods of analysis to understand the relation between the natural variables of marine environment: White-box testing, Grey-Box testing and Black-box testing.

White-box Testing. It is located at an intermediate level between the three types of approaches, since through this, it is possible to develop an analysis on how exter-

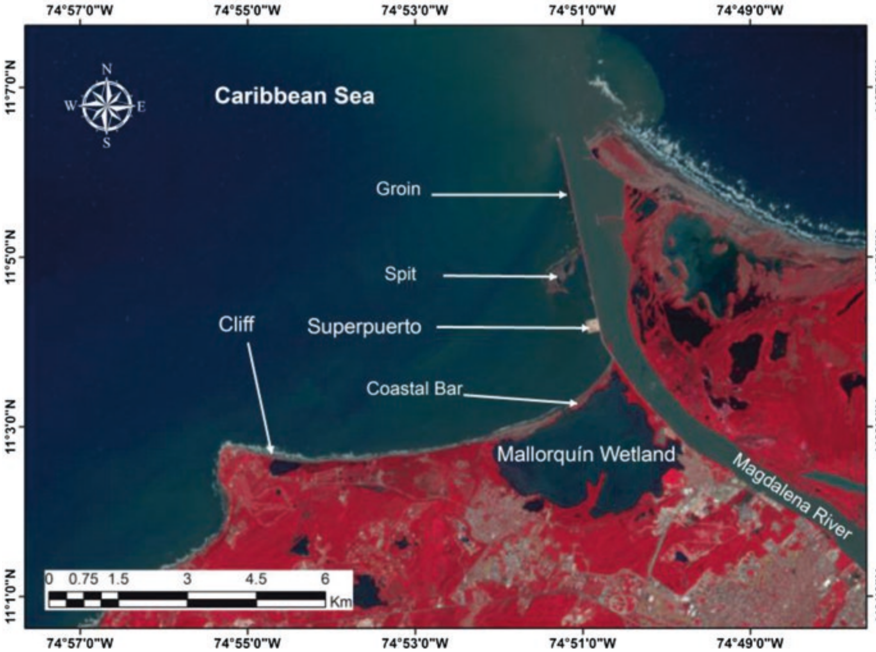

Fig. 9.2 Morphological features of the Mallorquín Lagoon based on Landsat imaging 2015

nal agents (natural element), phenomena (forcings), and processes (effect) correlate qualitatively within the natural environment.

Black-box Testing. It is the most generalist approach since it only allows making a description of more relevant elements of the system from a physical perspective.

Grey-box Testing. This analysis is the most detailed of all because within the qualitative description, it allows to identify the elements contained not only in general but in every littoral cell or subsystem and so establishing physical relations between the natural variables.

In virtue of the aforementioned, we have selected the gray-box testing for the analysis of the Mallorquín lagoon, where it allows to identify not only of in a general manner but also the elements contained within subsystems (subcells) or littoral cells that compose it, with the purpose to understand the relations between natural variables and physical processes that are part of the wetland dynamic. The process identified elements such as cliffs, shallow, rivers, bars, coastal arrows, tidal mouths, infrastructure works, alignment changes, presence of sand dunes, among others.

The model developed for the lagoon description considered among other aspects, those with higher relevance from the physics perspective in the identification of change tensors in the wetland: (a) geographical coordinates of boundary conditions that delimitate the littoral cell; (b) physics dimensions of the beach (plant-profile); (c) the characteristic material of the coast; (d) the configuration of the beach on plant.

The most important morphological features identified in the stage of characterization, correspond to coastal spits; the Magdalena River Delta; the submarine canyon; the sources of sediments (Magdalena River and small continental streams); coastal dunes; and coastal infrastructure. The selection of these morphological attributes is related to the fact that they correspond to those physical entities that more influence in the shape of beach and their evolution in time. Therefore, the borders of littoral cell were defined to considering in the left extreme of Bocas de Ceniza Groin; and the other side, the cells ends in the cliffs located at “Punta Roca”. It has a subsystem (subcells) that ends in the intersection zone with the León Stream when it opens the lagoon mouth.

Another aspect to consider, in detail, in the morphological analysis is to determine the marine structures built by men, identifying the affection grade that this intervention has had on the current shape of the beach. To give more physical sense to the study, results of morphological analysis are used to give a diagnosis that allows understating the causes and effects that led to the configuration of the coastline and, in general, of the littoral cell.

The bar of Mallorquín Lagoon is contained in a littoral cell confined between two physical borders: (a) Bocas de Ceniza Groin (Breakwater); and (b) Punta Roca (Cliff located at the west of the wetland). It is not observed along the coastline any type of physical, natural or artificial element that can influence in the morphological configuration of the whole cell or in the shape of beach. The most relevant feature from the geomorphologic perspective is related to the formation of a coastal spit in proximities of the groin to maintain the navigability activities (Bocas de Ceniza Groin). The beach presents a parabolic profile where the sand is the predominant material. Just in the western region of the littoral cell, it is identified rocky elements (cliffs or abrasion platforms) in the beach as shown in Fig. 9.2.

The beach has a variable section in plant throughout its length, with a mean width of 121 m. Additionally, it describes a defined discontinuity by a lagoon mouth that connects the wetland with the sea. This mouth has a dynamism associated with the wave climate variability and the oceanographic processes (currents, waves and tides). This is closed during the most part of the year, holds a width of 24.78 m approximately, but commonly it generates a channel at the end of the year (November, December) by with the influence of North Winds, Cold Fronts and extreme hydro-meteorological phenomena. Its position has varied in time to the present, and it is not expected to change physical conditions that have endured during last decades.

9.2.1.1 Impacts and Diagnosis

Perhaps, the event that triggered main changes in morphological level was the construction of the coastal structure as shown in Fig. 9.3. In ancient times, the system was composed of a cove and a set of bars generated by sediments from the Magdalena River. Subsequently, in the 1930s began the construction of a navigability channel in the Magdalena River, the Port Terminal and the works of shelter of the external

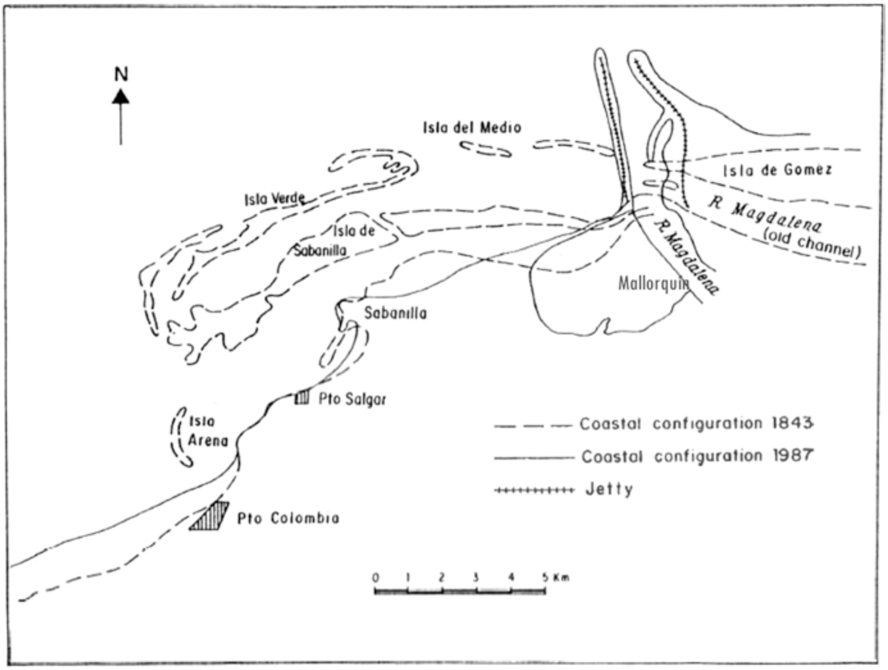

Fig. 9.3 Hystoric coastline modifications (Source: Martínez et al. 1990)

channel. This generated a change in the marine dynamic since before of the construction, the system could be considered, from the morpho-dynamics perspective, as in stable equilibrium.

After the mentioned event, the coastal system presents the natural tendency to adjust to new physical conditions trying to restore its state of equilibrium, but when it is altered, the feeding mechanism does not allow returning to the initial condition. This entails to the new configuration with a coastal bar and body of water in its upper part, with islands and internal lagoons generated during the events of Magdalena River rise. However, the fact of losing the main source of sediment brings associated unexpected effects since the marine environment lost its capacity of self-generation and it is erode continuously; so it can define that until now, the beach of Mallorquín Lagoon presents a metastable equilibrium. This is an indicator of the close relation between the river and the coastal zone near to the estuary, including the lagoon.

The physical parameters, which intervene in the formation of the configuration in plant for the wetland bar, are very different; but it could demonstrate that the longitudinal transport and the washing material in transversal direction are more relevant elements for the configuration of a costal system.

At present, the coastline of all nations that have border with the ocean is subject to erosion and sedimentation. As it was mentioned before, the coastal zone is a dynamic environment where many processes from the environmental, physical and biological perspectives occur. In consequence, any intervention made will produce a change in the balance that the system has leading to a response from the natural environment in search of new balance between processes and physical conditions. However, it is important to note that the activities developed by the human being in the coast, do not consider relations between meteo-marines processes and the physical variables that involve the beach, so the regulatory and restore process is interrupted.

When large infrastructure developments are carried out the morphological response of the coast and the effects derivate from the physical alteration are usually don't taken in account, and many others cannot be predicted a priori by the complexity of involved phenomena. In the studied zone, it was found as principal anthropogenic impact with negatives effects in the equilibrium of system since its establishment, the protection works of the external navigability channel for the Barranquilla Port. This resulted in a disruption of flowing sediment due to its perpendicular disposition to the coastline, with the consequent erosion associated not only in the unprotected area; besides, the sediment that flows as bedload through the principal channel of the river ending in a submarine canyon (Fig. 9.4). Only a small number of sediment reaches to overcome these two obstacles during certain times of the year, in function of the wave direction and the effects of the diffraction of western groin.

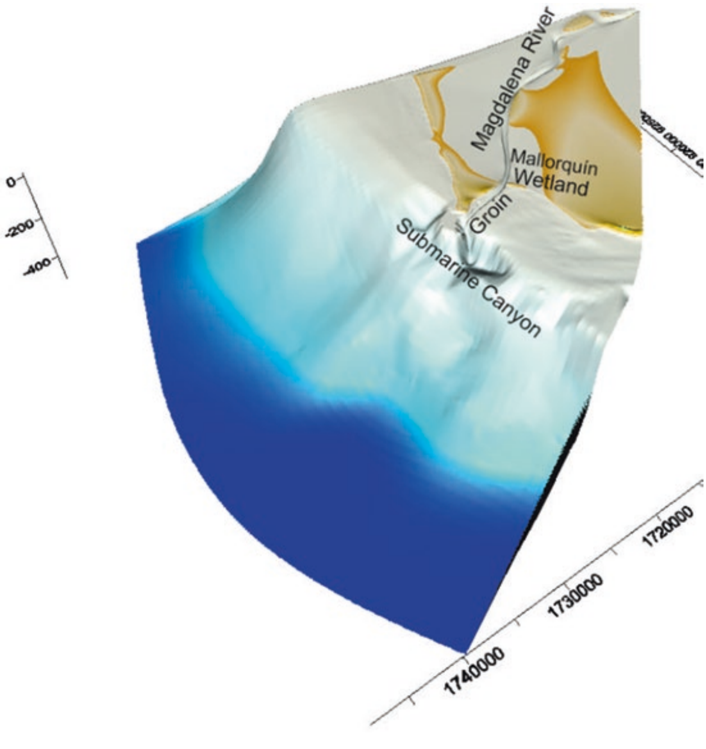

Fig. 9.4 Digital Elevation Model (DEM) of Submarine Canyon at Bocas de Ceniza Mouth

Furthermore, it is important to note as a positive element that the actual coastal dune associated with the Mallorquín coastal wetland remains unchanged. During the field-work, it was evident just one event of anthropic nature. It was observed that fishermen from the adjacent communities to prepare the pass of their vessels from the lagoon to the sea irregularly build craft channels (Fig. 9.5), altering the natural dynamic of the coastline. These accesses are built in two locations: (a) the first one is made by residents of the neighbourhood "La Flores" precisely in the point where the tidal channel opens naturally during storm events; (b) a second channel was built by residents of the neighbourhood "La Cangrejera" in proximities of the León stream inside the wetland, where the coastal bar has a minimum average width of 47 m. These channels affected directly the hydraulic conditions of wetland since allow the access of sediments from the marine border, interrupt the path of sediments through the bar, and the sediment enters in the water body reducing its hydraulic depth.

The analysis performed to date, can be used as monitoring indicators and permit to affirm that the coastal zone related to the Mallorquín Lagoon is a highly vulnerable system because it has lost its capacity of natural self-regeneration, and any change made in this system will have a direct and negative consequence, also hard to measure in short-term.

The nearest anthropogenic impact is due to the construction of the "Superpuerto" (Fig. 9.6). This mega work will produce important changes in dunes and in the sediment transport, that constitutes fundamental variables to maintain the quasi-balance

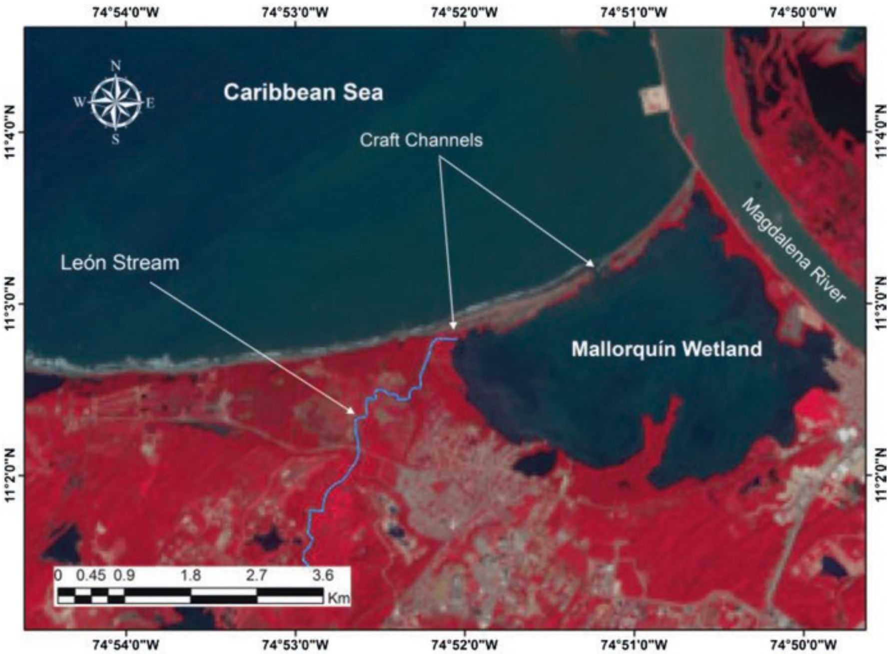

Fig. 9.5 Craft channels to communicate the coastal wetland to the sea

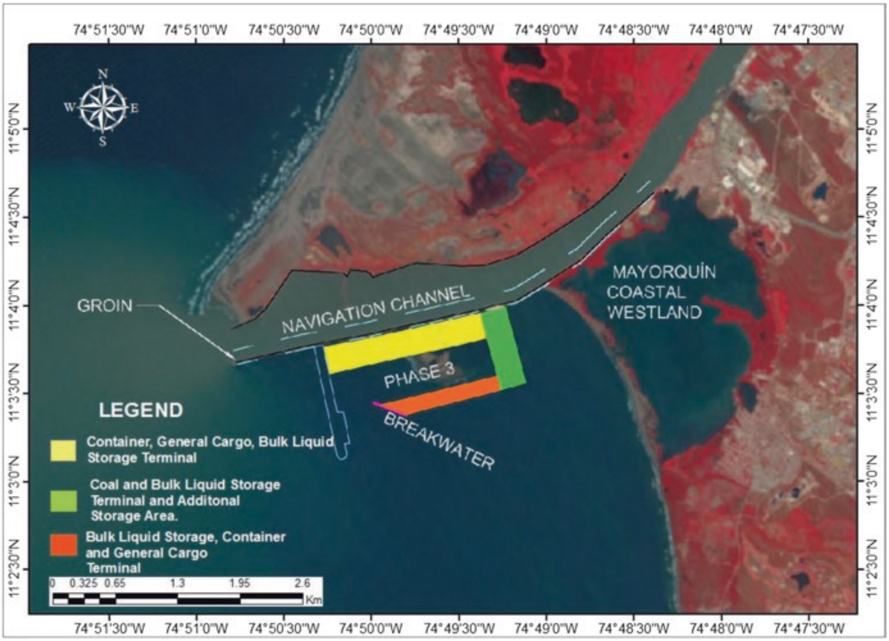

Fig. 9.6 Superpuerto Port

of sediments that has currently the beach but if coastal processes are modified, huge and relevant changes will be observed in the wetland.

Finally, it is recommended as in any other coastal engineering project, the understanding of those variables that allow the comprehension of the problems that coastal wetlands experiment. The recognition of the morphological behaviour will serve as a comparison element during the development of an intervention work, determining with this, the way to understand the capacity of self-regeneration of the bar that protects the wetland, and it will be an indicator of how it can evolve in time since it will be possible to maintain the ecosystemic services of these bodies of water.

9.3 Physical Properties of Sediments

From the engineering perspective, one of the elements that must be considered in the analysis of anthropogenic impacts of the bar of Mallorquín Lagoon with meteo-marine conditions and anthropogenic impacts, are the coastal processes that model the zone.

It is properly the interaction between the processes and physical contour of the coastline what origins the active dynamic of the beach, determining its configuration on-site and in profile. In this regard, we must consider the effect of currents, astronomical tide (if applicable), and the proximity to the continents, since they relate directly to the sediments transport and in particular, with its main manifestation, the erosion and accretion of the beach phenomena.

9.3.1 Sediment Sampling

Along the littoral of Mallorquín Lagoon we recollected 65 sediment samples in 23 locations, following a transverse and longitudinal path throughout the bar.

The methodological procedure made to each sediment sample considered the following elements (Fig. 9.7):

The sediment transport in a beach is a complex process to asses and comprehend. Firstly, because it is facing a two-phase flow conditions (water-sediment); and on the other hand, during the transport to the beach, the flow can be classified under the turbulent regime, where it is very important the friction between layers of fluids and a fully developed boundary layer is present. For this reason, currently there are a larger number of theoretical and empirical models in order to try to quantify the transport of material and the form how it correlates with the morphology of the seabed. There are mathematical models that approach effectively the real problem, but they require intensive field data to be validates and calibrated.

The above-mentioned aspects make the selection of a sampling site complex but that well evaluated will lead to important and correct conclusions regarding the state of equilibrium of the beach. This physical bases enables to say that several samples must be displayed and characterize different locations of a beach, and therefore, the samples taken have to be consistent with the physics of transport phenomena.

Consequently, samples were extracted from three points in each profile. In the Berm Zone (B); in the Washing Zone (L); and finally, in the submerged zone (S) that correspond to a depth of one meter (1.0) roughly. With the selection of these points, it covers a broad spectrum of phenomena related to the sediment transport in a beach, as the waves and the currents in the submerged beach, and the wind in the dry beach. By sampling site, they were recollected 3 kilos of sand in a straight line from the berm to the submerged area, with spaces between line and line of 50 m.

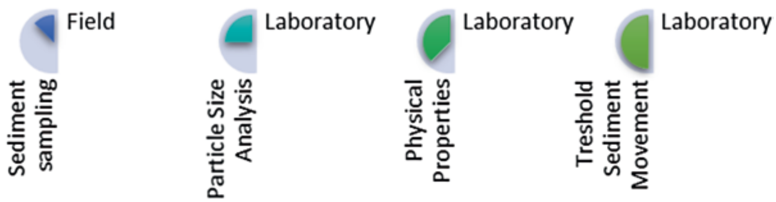

Fig. 9.7 Sediment samples analysis process

9.3.2 Sample Statistics Distribution

Analytical methods to quantify the form, the size and the composition of sediments are based on statistical assumptions that have been developed during the years. However, the significant fact is that under the principle of characterization of the grains of material a detailed study of their origin and form can be reached.

A statistical procedure is applied to problems where it is necessary to make the classification of materials (composition and size), to establish mechanism that originated the beach and know the way how the sediments were deposited. The classification is made from the main statistical parameters of the sample and based on it, it can determine whether the material dispersion or depositation was created by the waves and/or the currents, as well as, to define which are the physical factor that intervene in the configuration of the beach. This information is very relevant at the moment to carry out the diagnosis and the analysis of agents that can generate erosion in a littoral system and to value the sediment interaction with physical processes in the coastal zone.

For the representation of the mechanical analysis of sediment, we have adopted logarithmic functions. This gives a lot of practicality and relevance to the study, allowing that the sample properties are expressed in terms of statistical deviations from the normal distribution function. It is possible to express the frequency analysis from statistical and mathematical principles covering the entire distribution of sample sizes, without practical or theoretical limitations, when are applied to the full spectrum of sizes of sediment (Inman 1952). Parameters that depend on the order statistics such as mean, median, standard deviation and asymmetry are applied using different methodologies (moments, quartiles, logarithmic methods graphs, etc.).

An adopted method for the analysis of size distribution of the sediment sample consists in express the statistics though a logarithmic expression that describes the diameter conversion (d) in millimetres to units phi (ϕ), using the equation $\phi = -\log_2 d$ (Krumbein 1936). In reaching this classification, and in the particular case of sediments, measures of the central tendency of the mean and the median of the samples diameter is taken into account, where one of these two variables can acquire a greater importance than the other. In a normal distribution, the mean is the diameter value that represents the centre of gravity of the frequency distribution while the median, from a geometric perspective, is the amount that divides systematically the frequency curve (Inman 1952).

The influence of the median is relevant when it is about avoiding in the analysis affections in the extremes at the moment to adjust the distribution; instabilities that can potentially be originated by dispersions with respect to the mean for samples that have a large number of sizes (Inman 1952). For its algebraic character, the central value of a large quantity of sediments samples can be obtained by means of the analysed group, which reflects the average size that the sediment has in the source where it was founded. Although, it must be remembered that it is more recommended for quantification porpoises the use of the arithmetic mean.

In physical terms, this procedures allows to verify whether the consideration of percentiles 16 and 84 in the analysis of sediment properties, the volume distribution of the sand sample is symmetrical, whereas if we want to know how is the symmetry behavior and the continuity in the distribution extremes, the analysis must be performed for percentiles 5 and 95 (Inman 1952). It is vital to consider in the analysis other significant points in the cumulative curve of the sediment distribution in terms of ϕ, simplifying with this the original mechanical analysis of sediment, these are:

$$\varnothing_5\ ,\ \varnothing_{16}\ ,\ \varnothing_{50}\ ,\ \varnothing_{84}\ ,\ \varnothing_{95}$$

The theoretical formulations to determine the statistical behavior of the data in Phi units are:

$$\phi = -\log_2 d \tag{9.1}$$

$$d = 2^{-\phi} \tag{9.2}$$

d: diameter of the sediment particle in millimetres.
ϕ: Phi units.

To determine the principal moments or order of the data series associated to the sediment samples, it was used the following formulations:

$$\bar{d} = \frac{\phi_{16} + \phi_{50} + \phi_{86}}{3} \tag{9.3}$$

$\bar{d}$: Average diameter.

$$s = \frac{\phi_{84} - \phi_{16}}{4} + \frac{\phi_{95} - \phi_5}{6.6} \tag{9.4}$$

s: standard deviation.

$$Sesg = \frac{\phi_{16} + \phi_{84} - 2\phi_{16}}{2(\phi_{84} - 2\phi_{16})} + \frac{\phi_5 + \phi_{95} - 2\phi_{50}}{2(\phi_{95} - \phi_5)} \tag{9.5}$$

Sesg: Statistical Bias.

The sediment classification based on the statistical parameters suggests a scale ranging from very platykurtic to extremely leptokurtic (Fig. 9.8); from highly biased for the coarser material to very fine bias; and finally, it is given a description of the material sample from very well sorted to very poorly sorted, according to the standard deviation (Fig. 9.9).

By employing the SANDY script (Ruiz et al. 2016) we determined the statistical data, the size distribution and the material classification (Fig. 9.10). It is important to underline that this programme consider for the classification the following rules: (a) American Section of International Association for the Testing Materials (ASTM);

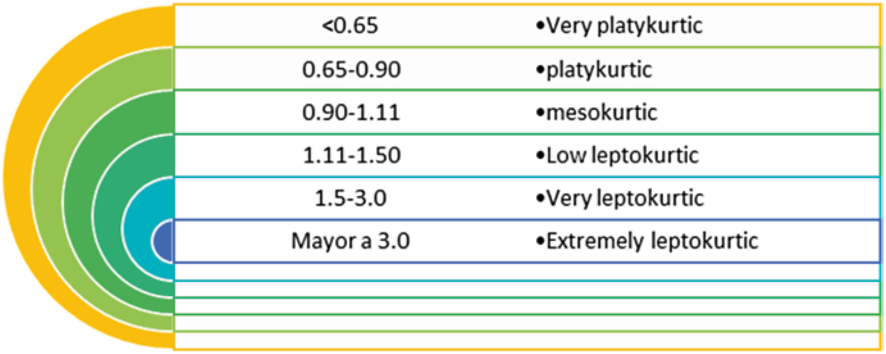

Fig. 9.8 Material classification respect to the kurtosis

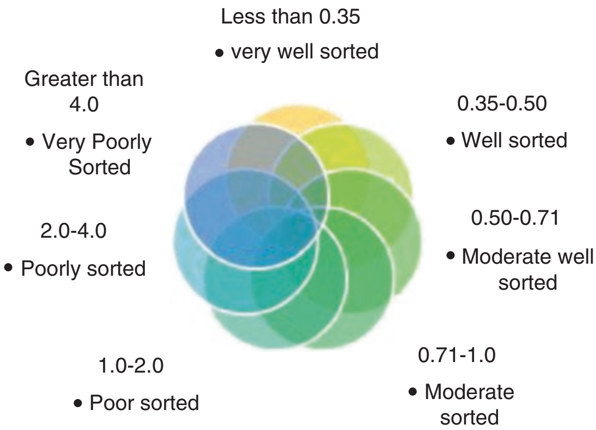

Fig. 9.9 Material classification respect to the standard deviation

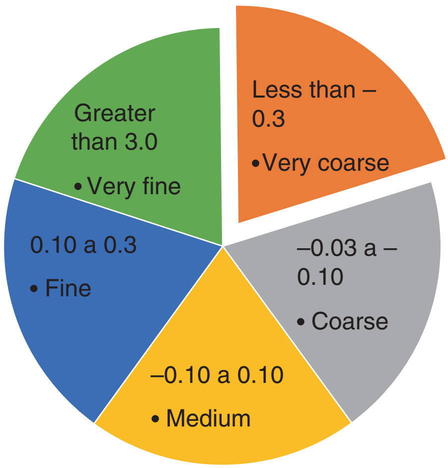

Fig. 9.10 Sediment Classification with respect to statistical bias

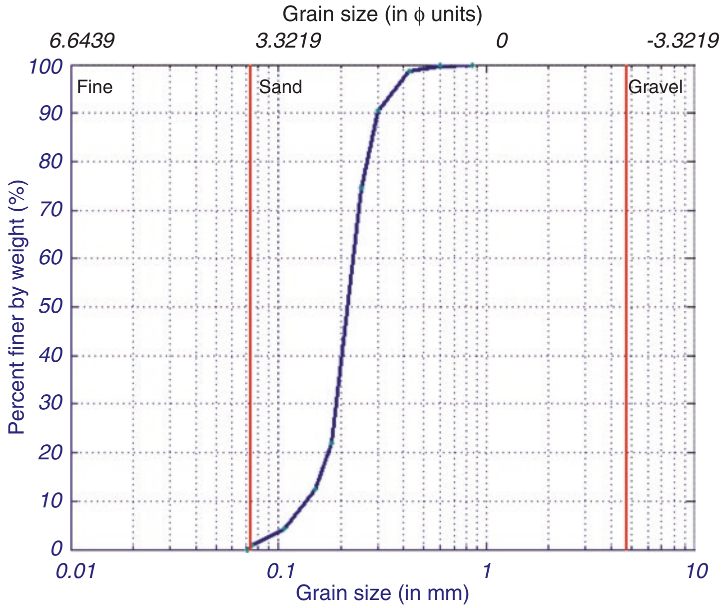

Fig. 9.11 Cumulative mass retained berms zone. Coastal bar Mallorquín Lagoon

(b) Wentworth classification; and (c) The Unified Soil Classification System (SUCS) (Fig. 9.11).

9.4 Wave Climate Analysis

The objective of the wave climate analysis is to provide detailed knowledge, both in deep water and in specific points of the area of interest, the current situation of the wave characteristics, sea level and currents that allow helping to understand the hydrodynamic operation of the study area. Without this knowledge, it is not possible to determine the causes and possible solutions to the problems of coastal erosion of a beach.

In search of characterizing the atmospheric conditions in the area of the Mallorquín coastal lagoon, it was extracted the climate information from the database NCEP North American Regional Reanalysis (NARR). The NARR database starts from January 1st, 1979 until December 31st, 2013 with information every 3 h daily, hourly and monthly of the Central America and the Caribbean hemisphere. The NARR information contains the main physical parameters of the meteo-marines conditions such as the wind, heat flows, solar radiation, and humidity, among others.

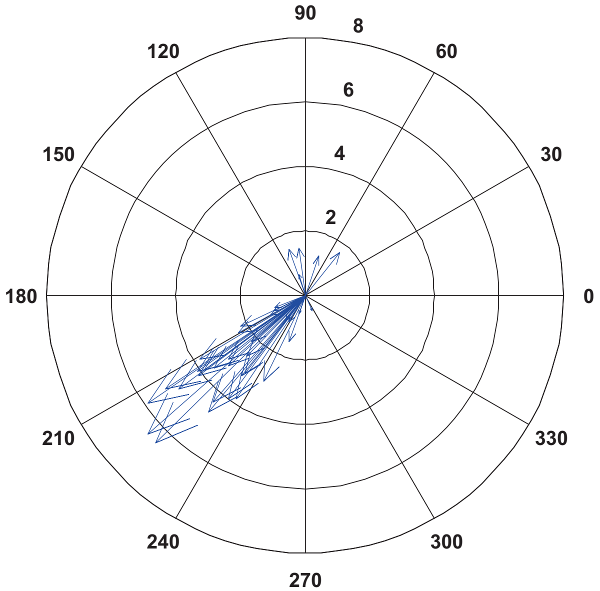

Fig. 9.12 Annual average wind rose (1979–2013) for October month (Source: NCEP North American Regional Wave Reanalysis: NARR)

9.4.1 Wind

Most of the year, the study area presented prevailing north-easterly winds (Fig. 9.12), october were a months with the great magnitude of the wind, with records between 8 m/s and greater variability, registering northwest, north and northeast winds.

9.4.2 Wave Conditions

From the point of view of the coastal engineering, one of the main objectives to advance a restoration project or to make a diagnosis of the condition of a coastal area, must be determined sea states that represent marine conditions, mainly the waves and wind, for both mean regime and during extreme events.

Through the use of mathematical statistics and probability adjustment functions are estimated and classify the sea states, their direction, magnitude and recurrence.

For the study was used the synthetic wave information of virtual buoys near to the zone developed by the company *Buoyweather*. The series of synthetic waves were generated by the reanalysis technique with the WAVEWACTH III model, which have an hourly and temporary resolution and a 30 years period.

9.4.2.1 Virtual Buoy 11.5 °N −74.5 °W

It can develop a mean regimen analysis of waves to analyse the behaviour of the maritime climate in a coastal wetland. For this, it analyses the probability of any storm is overcome in medium time. Besides, it is possible to classify physical variables by a joint probability analysis in order to characterize the phenomena magnitude with its persistence in different season of the year. This allows construction waves and wind roses and frequency histograms.

The potential of bars erosion that protect wetlands can be determined by an analysis of unfavourable weather conditions, known as extreme analysis. This is focused on the events characterization that have the larger energetic amount and that also, generate drastic changes in littoral zones, appearing during each winter season.

These analysis were developed with the data sample of a virtual buoy 11.5 °N, 74.5 °W, it identified an average waves associated to the 50% of exceedance probability, with a significant height of 1.78 m, and an extreme waves (95%) with a significant height of 2.87 m (Fig. 9.13a).

Based on the results of joint probability curve of significant height- peak period (Fig. 9.13b), obtained from the virtual buoy, the higher probability is defined by significant height of 2.1 m and a peak period of 6.9 s. The joint probability between the significant height and the direction (Fig. 9.13c), it has been found a value of 1.8 m and 49° respectively.

From the curve of exceedance probability (Fig. 9.13a), it was defined the threshold to the identification of maximum values in the time series, represented by the straight line on the Fig. 9.14.

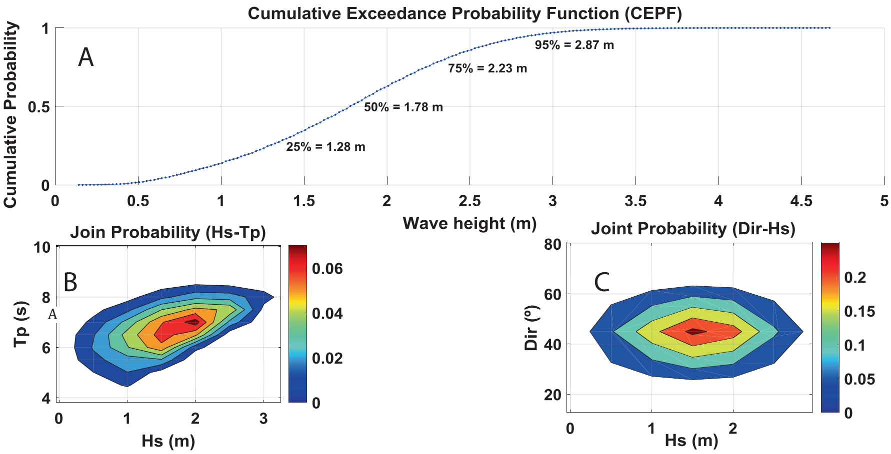

Fig. 9.13 (**A**) Exceedance probability of wave height; (**B**) joint probability of significant wave height against peak period; (**C**) joint probability of significant wave height against wave direction. At virtual buoy 11.5 °N y −74.5°W. Time period: 01-jan-1979 to 31-jan-2014

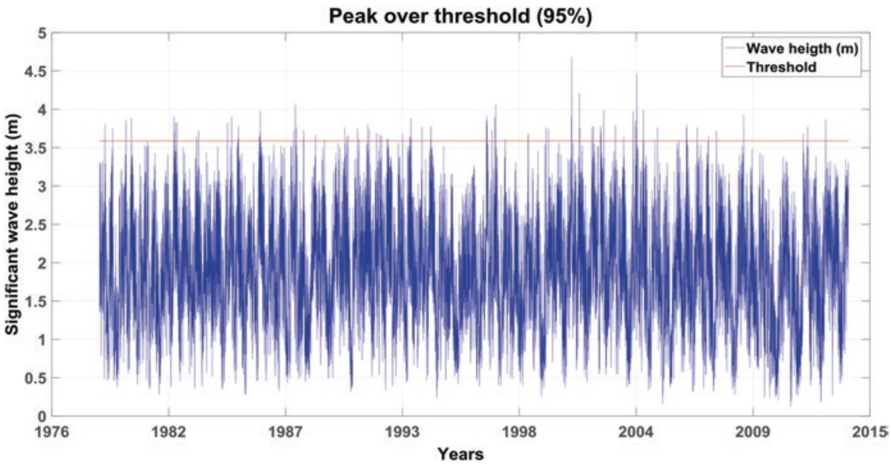

Fig. 9.14 Maximum wave height based on threshold

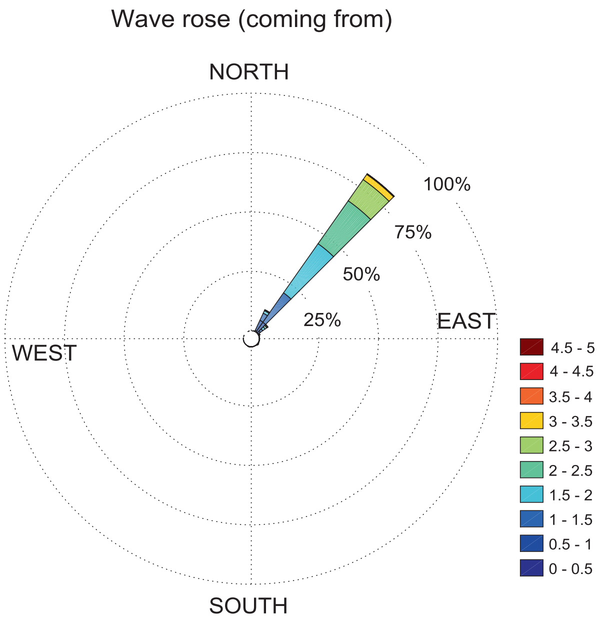

Fig. 9.15 Wave rose in the study zone

The wave rose of synthetic series (Fig. 9.15) indicated that the predominant wave direction was to the southwest, between the 35 and 49°, an average of the significant height value between 1.5 and 2 m, and maximum values of wave between 4.5 and 5 m.

Based on the probability curve of extreme exceedance, which has been built from the data above the threshold (Fig. 9.16) of 95% (2.87 m), it was found the

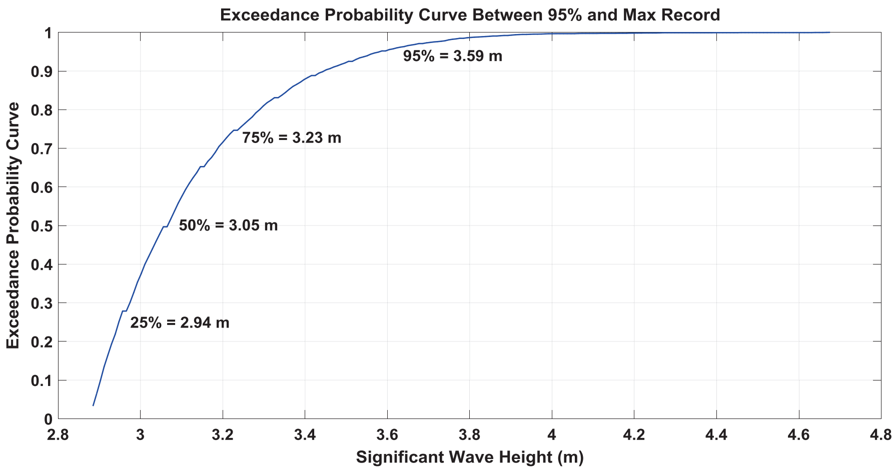

Fig. 9.16 Exceedance probability of wave height at virtual data series 11.5°N and −75°W. Time period 01-jan-1979 a 31-jan-2014

extreme wave more recurrent has presented a height of 3.05 m, and a critical extreme wave, which exceeds the 95%, comprises from the 3.59 m to the maximum height register of 4.68 m.

9.4.3 The Astronomic Tide

The astronomic tide is defines as a group of regular movements of ascendants or descendants sea level with periods of 12 or 24 h that are caused by the gravitational attraction made by the moon and the sun on the Earth, as well as, the effect from other planets. When there are investigations on the beach, or coastal studies are carried out, it is necessary to know the behaviour of the tide wave, especially for predicting its amplitude given that it is a determinist phenomenon. To approach the study of its behaviour consists in considering the astronomic tide as the sum of a finite number of waves, whose amplitude and phase are known; because of they have been associated with planetary movement.

To the study, it is developed a decomposition through the astronomic analysis considering The Earth, The Moon and the Sun movements, in order to determine the relative frequency and importance of each one of components (Cartwright 2001). Taking into account the tide amplitudes, it is possible to identify two types of tide: spring and neap tide.

9.4.3.1 The Harmonic Analysis of Astronomic Tide

The tide description and prediction is realized by the tide harmonic analysis which consists in decomposing the registers of sea level in a finite number of waves whose period and phase has been perfectly established, coinciding with periods of some astronomic movements related to the Earth, the Moon and the Sun, as it was described previously.

The method of harmonic analysis consists in obtaining from an hourly register all data of sea level, the amplitudes and phases of component waves. These parameters are known as harmonic components, due to the assumption where is considered that the responses of seas and oceans to the tidal forces do not change in time (Dronkers and Schonfeld 1959).

The astronomic tide is the sum of constituent waves from the Fourier coefficients

$$y(t_n) = a_o / 2 + \sum_{k=1}^{M} a_k \cos(\omega_k t_n) + b_k sen(\omega_k t_n) \tag{9.6}$$

The determination of the tidal regime from the observation point, was made by the equation used for the calculus of the factor "F", equation which takes the amplitudes of the harmonic constituents.

$$F = \frac{K_1 + O_1}{M_2 + S_2} \tag{9.7}$$

F: phases of harmonics
F = 0.00 a 0.25, represents a semi-diurnal tide.
F = 0.25 a 1.50, represents a mixed tide predominantly semi-diurnal.
F = 1.50 a 3.00, represents a mixed tide predominantly diurnal.
F = Greater than 3.00, corresponds a diurnal tide.

It was conducted the modelling of astronomic tide for a period of 52 years (2008–2032), so for this time series were determined their respective harmonics.

According to determined harmonics, it was obtained a factor F of 1.86, and a value z of 0.27 m, ranking the tidal regimen of the study zone as mixed micromareal tide predominantly diurnal.

9.5 Hydrodynamic in Coastal Wetlands

9.5.1 Model of SWAN

The SWAN model is based on the Action Balance Equation. The model can be used without the effect of the current, reducing the equation in terms of the energy balance equation for calculus of local scale (9.8).

$$\frac{\partial N_{(\sigma,\theta;x,y,t)}}{\partial t}=\frac{\partial c_{g,x}N(\sigma,\theta;x,y,t)}{\partial x}+\frac{\partial c_{g,y}N(\sigma,\theta;x,y,t)}{\partial y}+\frac{\partial c_{g,\theta}N(\sigma,\theta;x,y,t)}{\partial \theta}+$$
$$\frac{\partial c_{g,\sigma}N(\sigma,\theta;x,y,t)}{\partial \sigma}=\frac{S_{(\sigma,\theta;x,y,t)t}}{\sigma} \tag{9.8}$$

Where,

N = E/ω, is the density spectrum of action; E = density spectrum of energy; ω = absolute frequency of wave; θ = wave direction; σ = relative frequency; x, y = spatial coordinates in the horizontal; $N(\sigma,\theta)$ = density spectrum of action; $c_{g,x}$, $c_{g,\mathbf{y}}$ = prorogation velocity in the coordinates x, y.

The term $S_{(\sigma,\theta;)}$ is the density source of energy and the division $\frac{S_{(\sigma,\theta;)}}{\sigma}$ is the density action; these terms represent the effects of the generation during the interaction and dissipation of irregular waves (nonlinear).

9.5.2 The Hydrodynamic Delft3D Model

The numerical model with the computational module "Flow" solves the Navier-Stokes equations, considering the Boussinessq assumption, in which the effect of the density variable is taking into consideration in terms of pressure. The turbulence is considered by the Reynolds efforts and is represented through the k-l y k-ε approximations (Uittenbogaard et al. 1992). The model uses the temporary scheme Alternating Direction Implicit (ADI).

The continuity and momentum equations in the direction of x and y are expressed respectively as: (9.9, 9.10, and 9.11).

$$\frac{\partial u}{\partial x}+\frac{\partial v}{\partial y}+\frac{\partial w}{\partial z}=0 \tag{9.9}$$

$$\frac{\partial u}{\partial t}+\frac{\partial u^2}{\partial x}+\frac{\partial uv}{\partial y}+\frac{\partial uw}{\partial z}-fv+\frac{1}{\rho_0}P_x-F_x\frac{\partial}{\partial z}\left(v_v\frac{\partial u}{\partial z}\right)=0 \tag{9.10}$$

$$\frac{\partial v}{\partial t}+\frac{\partial vu}{\partial x}+\frac{\partial v^2}{\partial y}+\frac{\partial vw}{\partial z}-fu+\frac{1}{\rho_0}P_y-F_y\frac{\partial}{\partial z}\left(v_v\frac{\partial v}{\partial z}\right)=0 \tag{9.11}$$

The turbulence closure model k- ε is engaged to transport equation, without the interaction of the waves, and it is expresses as (9.12, 9.13)

$$\frac{\partial k}{\partial t}+u\frac{\partial k}{\partial x}+v\frac{\partial k}{\partial y}+w\frac{\partial k}{\partial z}=\frac{\partial}{\partial z}\left[\left(v_{mol}+\frac{v_{3D}}{\sigma_k}\right)\frac{\partial k}{\partial z}\right]+P_k+B_k-\varepsilon \tag{9.12}$$

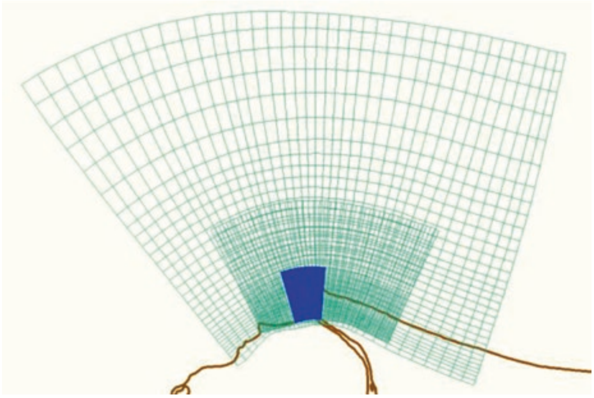

Fig. 9.17 Computed wave grid with Wave module. Nested grid at Mallorquín Lagoon

$$\frac{\partial \varepsilon}{\partial t} + u\frac{\partial \varepsilon}{\partial x} + v\frac{\partial \varepsilon}{\partial y} + w\frac{\partial \varepsilon}{\partial z} = \frac{\partial}{\partial z}\left[\left(v_{mol} + \frac{v_{3D}}{\sigma_{\varepsilon}}\right)\frac{\partial \varepsilon}{\partial z}\right] + P_{\varepsilon} + B_{\varepsilon} - \varepsilon_{\varepsilon} \qquad (9.13)$$

Where Prandtl-Schmidt numbers are $\sigma_k = 1$ and $\sigma_\varepsilon = 1.3$, P_k represents the production of turbulent kinetic energy in cutting flows, ε is the dissipation by turbulent energy (m^2/s^3) and it is the kinetic energy (m^2/s^2). The flow of thrust reversal B_k represent the transformation from kinetic to potential energy, and it is associated with the dissipation of energy and the edge flow respectively.

9.5.2.1 Meshes Deneration; Initial and Boundary Conditions

The numerical model Delf3D permits to use structured, flexible and unstructured meshes. For this case, it was used flexible structured meshes for the Flow module, and curvilinear grid for the Wave module considering a general area for the Mallorquin wetland as shown in Fig. 9.17.

Meshes of wave module are rectangular and curvilinear (Fig. 9.17), composed of square and rectangular cells. The largest mesh has cells with a size roughly of 1.8*1.8 km. The intermediate mesh has cells with sizes of 600*600 m, and the smallest mesh, which covers the sector of Bocas de Ceniza has cells of about 90*90 m.

The mesh of Flow module is flexible and is composed of square and rectangular cells with an average size of 30*30 m for the sector of Bocas de Ceniza mouth and for the Mallorquín coastal wetland, and 550*210 m for other sectors.

Bathymetry was constructed from the ETOPO1 database information and the bathymetries developed in the river mouth of the Magdalena River and navigable channel (Fig. 9.18) and cover all the coastal area associated to the Mallorquín lagoon.

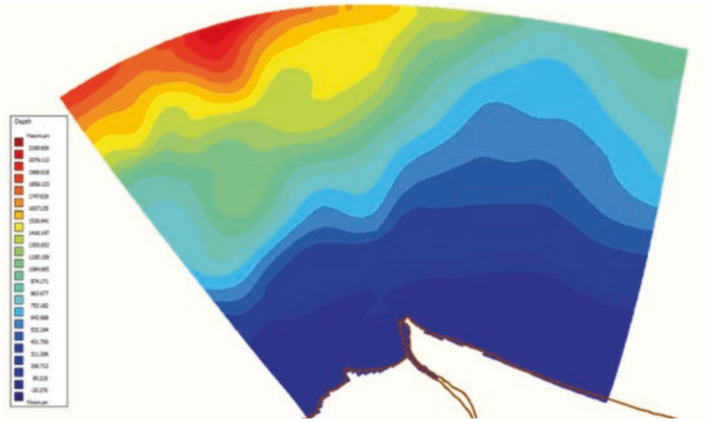

Fig. 9.18 Bathymetry of the study zone

9.5.3 Calibration of the Numerical Model

Calibration of the numerical models ensures that modelling results approaching to ocean-atmospheric conditions present in the study zone. For this project, it was used reference information which allows calibrating the hydrodynamic and wave model. Besides, it was used information of levels variations of the limnimetric station of Casa Pilotos (Fig. 9.19) and wave information available of the directional buoy closer to the study area used by the Maritime General Direction (DIMAR), *in spanish*), . For the calibration of the morphodynamic model, a longitudinal transect of these seabed within the access channel of Magdalena River was used.

Hydrodynamic calibration results according to the variations of sea level (Fig. 9.19) demonstrate a large adjustment between measured and modelled data. Despite available in situ information is subject to diverse processes and variations such as the flow, waves and the wind, it was possible to hourly simulate the variations of the sea level during a 4 months period. Based on the characterization of the astronomic tide, previously made, it was observed a pattern of mixed semidiurnal tide.

Results of wave calibration (Fig. 9.20), made possible to approximately reproduce the variation of the significant wave height of the study area, evidenced by the comparison between the variation of the significant wave height measured by the buoy, and the results simulated by the model. Based on the above mentioned and the general behaviour of time series, it was possible to adapt the numerical model to represent the conditions of average wave of the study zone.

After calibrating the hydro-morphodynamic model is necessary to identify the representatives modelling scenarios for the analysis of oceanographic conditions of the study zone, for purposes of establish the diagnosis of erosion problems of the coastal zone.

Consequently, the analysis of the hydro-morphodynamic analysis in 2010 was focused on characterized climate seasons of the study zone: Dry (February); Humid (October) (CIOH 2010), and according to the highest and lowest tides of each season, predefined from the Grenoble Model (Le Provost et al. 1994).

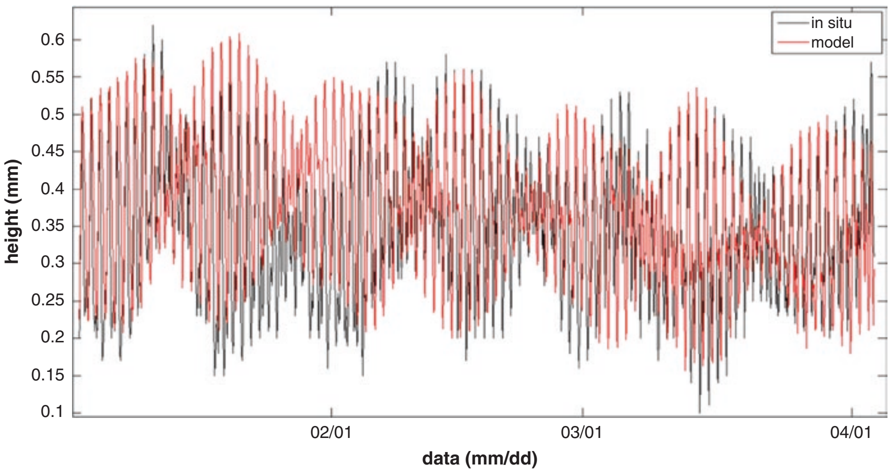

Fig. 9.19 Water levels calibration from Casa Pilotos station

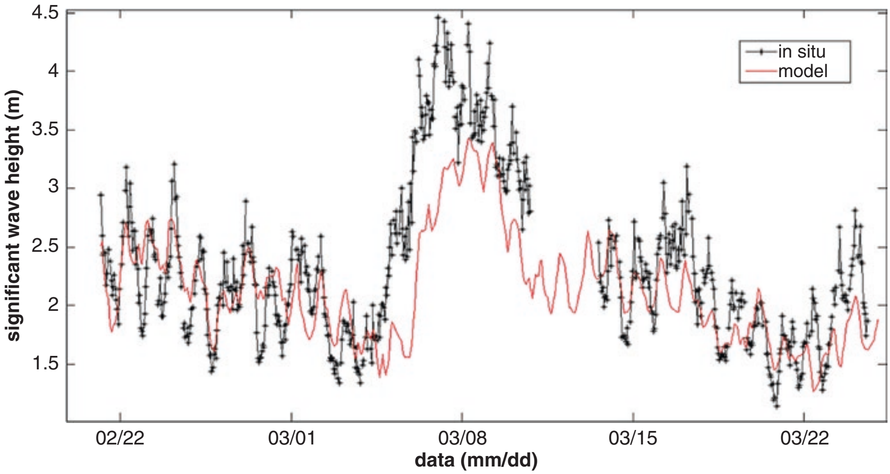

Fig. 9.20 Water waves calibration from DIMAR data buoy

9.5.4 Dry Season: February

9.5.4.1 Coastal Currents

The velocity rate in surface during the phase of high tide describes paths dominated by the river in the estuary zone (Fig. 9.21a). There is a vectors change due to the wave action and the diffraction phenomena that is generated in the edge of the groin, reaching speeds of the order of 1.6 m/s. Subsequently, they flow in the current direction, reducing the speed up to 0.3 m/s and align with bathymetric contours to initiate the movement in a south-westerly direction.

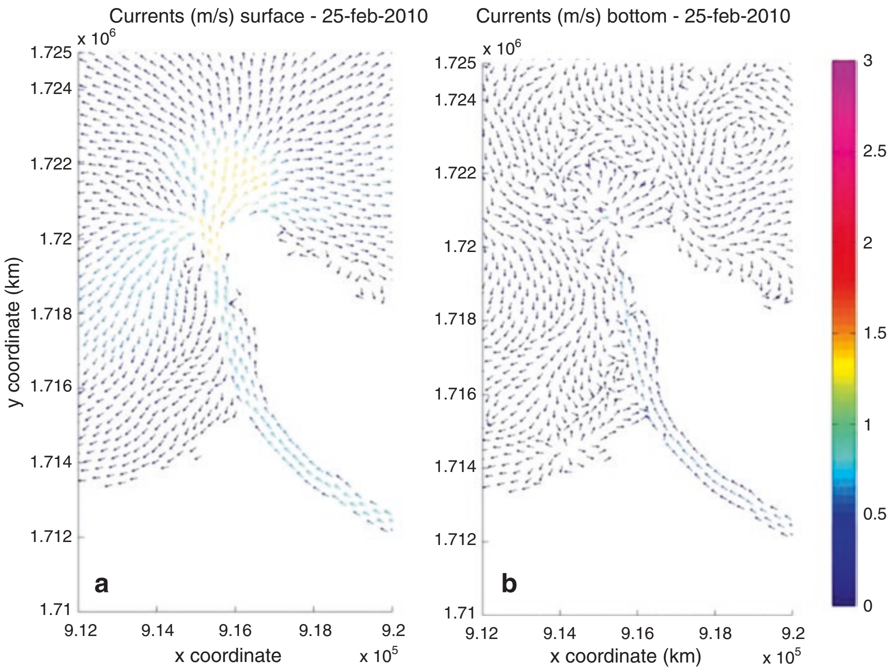

Fig. 9.21 Velocity field during highest astronomical tide during dry season (February, 2010) (**a**) surface (**b**) bottom

In the bottom layer as it was expected (Fig. 9.21b), the cascade of turbulence is fully developed; and besides, from the interaction between particles of different paths, arise rotational movements due to the shear stresses effect in the turbulent boundary layer. There, the viscous stresses have a great relevance to change the direction of the velocity fields, enabling the formation of vortices and chaotic movements that create a two-phase flow without a defined trajectories. This can be one of the reasons why the spit is originated next to the Western Groin, where the influence of the depth is so important that supported by the diffraction effect permit the deposition of coarser sediments.

9.5.4.2 Wave Height

In the Mallorquín lagoon arrived wave fronts of 0.8 m maximum and minimum of 0.2 m (Fig. 9.22a). This is because the groin generates a warm area that changes the behavior of wave trains, creating conditions of less intensity in terms of wave energy. The behavior of wave propagation during the dry season of 2010 showed wave heights in front of the access channel with maximum values up to 2 m (Fig. 9.22b).

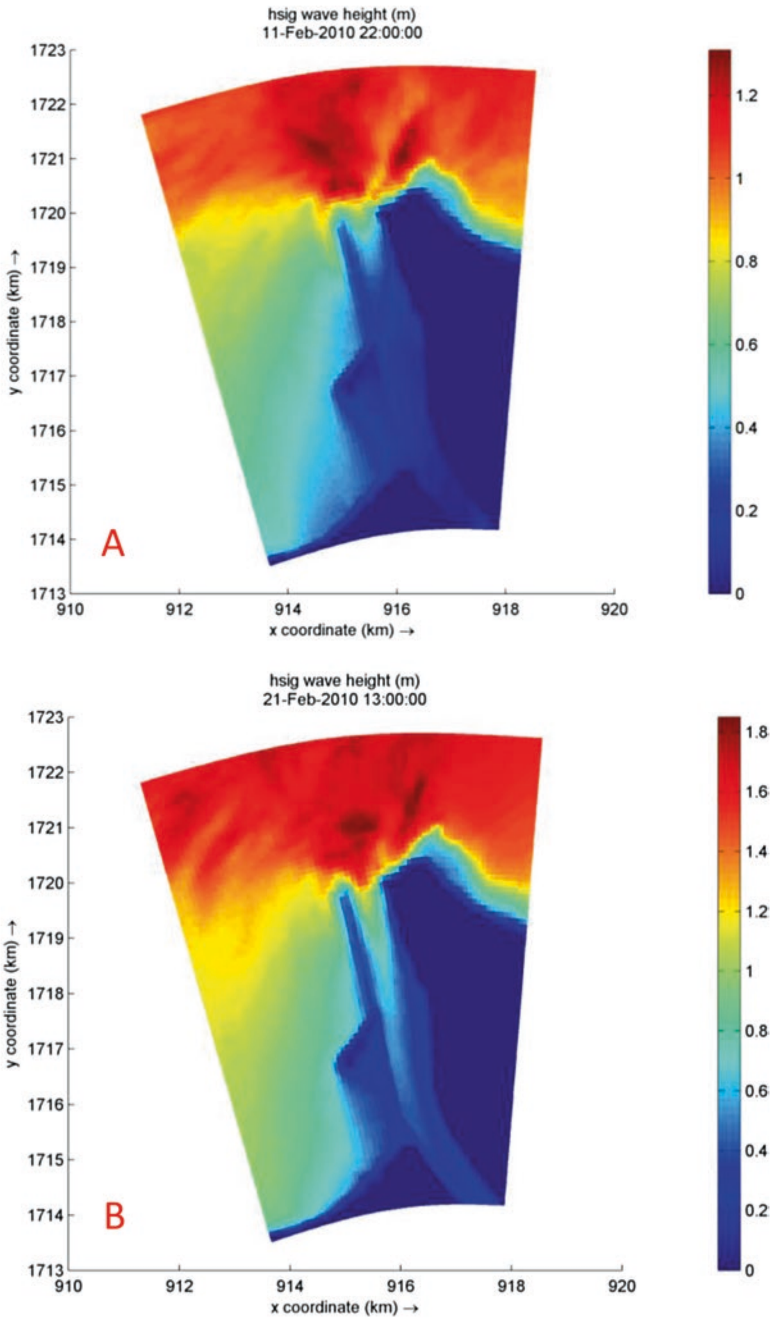

Fig. 9.22 Significant wave height **(A)** lowest and (**B**) highest. (February, 2010)

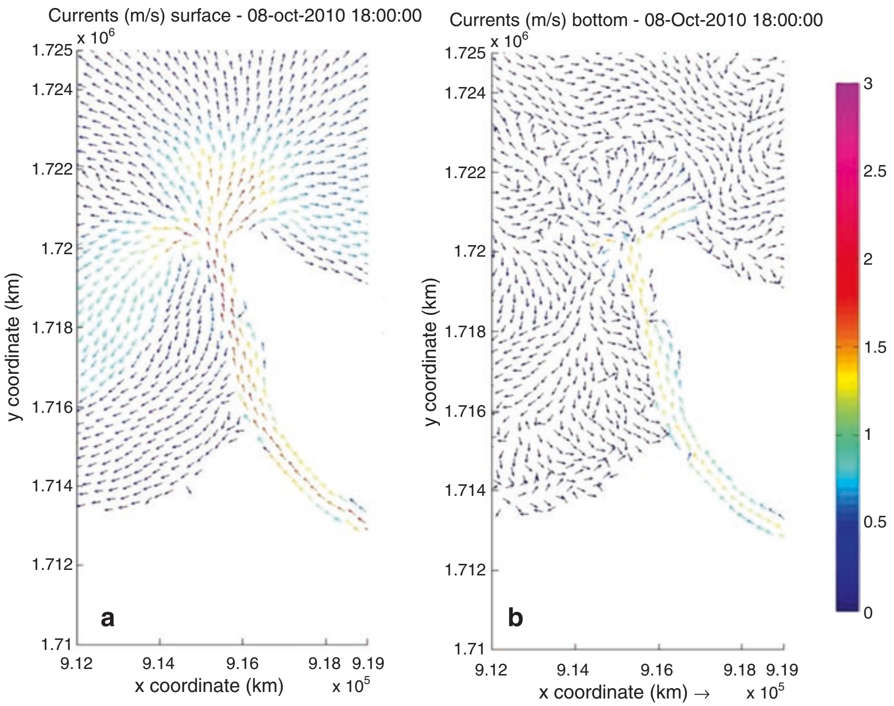

Fig. 9.23 Velocity field during highest astronomical tide during dry season (October 2010) (**a**) surface (**b**) bottom

9.5.5 Humid Season: October

9.5.5.1 Currents

The surface current field during the maximum level at the time described intense wet conditions compared to the dry season (Fig. 9.23a). There are flow velocities between 0.13 and 2.33 m/s, and a mean of 1.59 m/s; the maximum velocity values in the estuary surface. Similarly, velocities in the lagoon coastline increase, reaching values of 0.6 m/s.

The bottom layer has speeds between 0.02 and 2.21 m/s, with a mean of 1.57 m/s (Fig. 9.23b). The most turbulent flow is not only in the river mouth, but also, along the coastal protection works, and in front of the lagoon, paths follow the same direction with great velocity.

9.5.5.2 Wave Propagation

The behavior of the significant wave height during the humid season (Octubre) 2010, presented records up to 1.69 m in front of the access channel (Fig. 9.24a). In front of the wetland, there is not high values since the protection of the breakwater

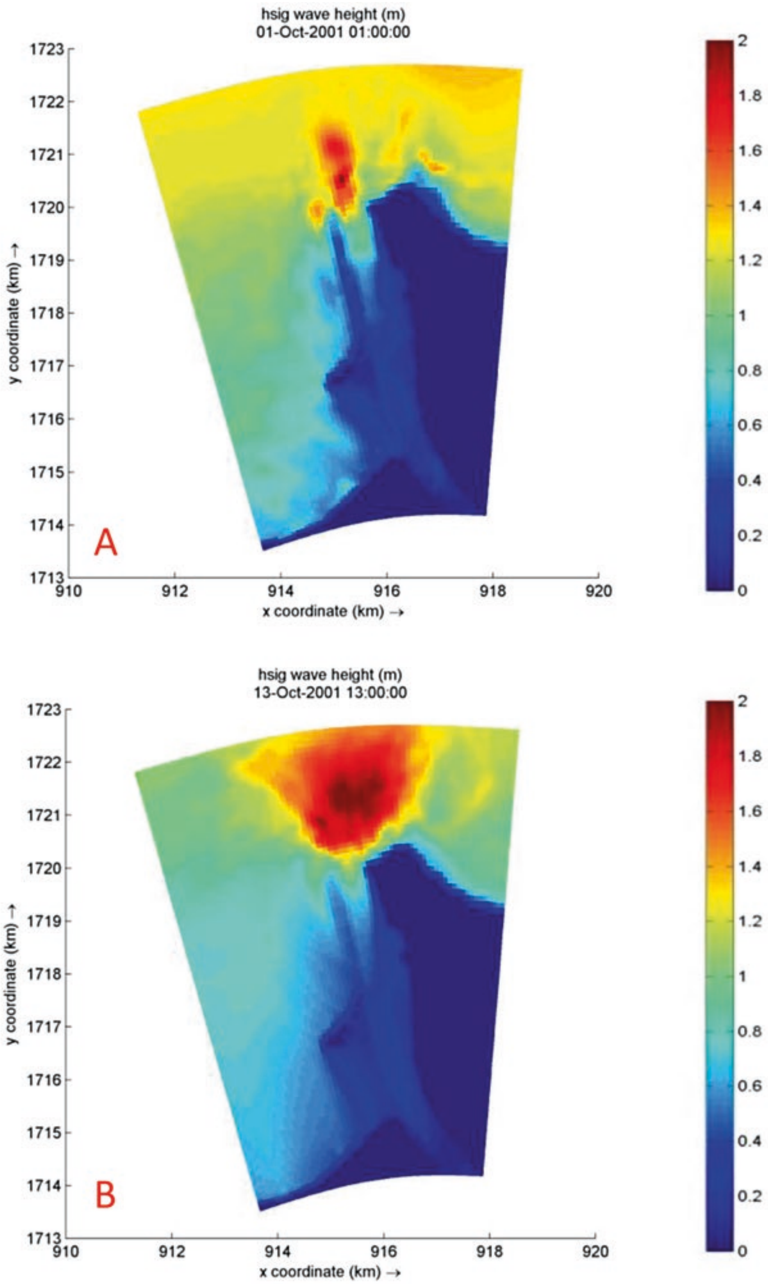

Fig. 9.24 Significant wave height (**A**) lowest y (**B**) highest (October 2010)

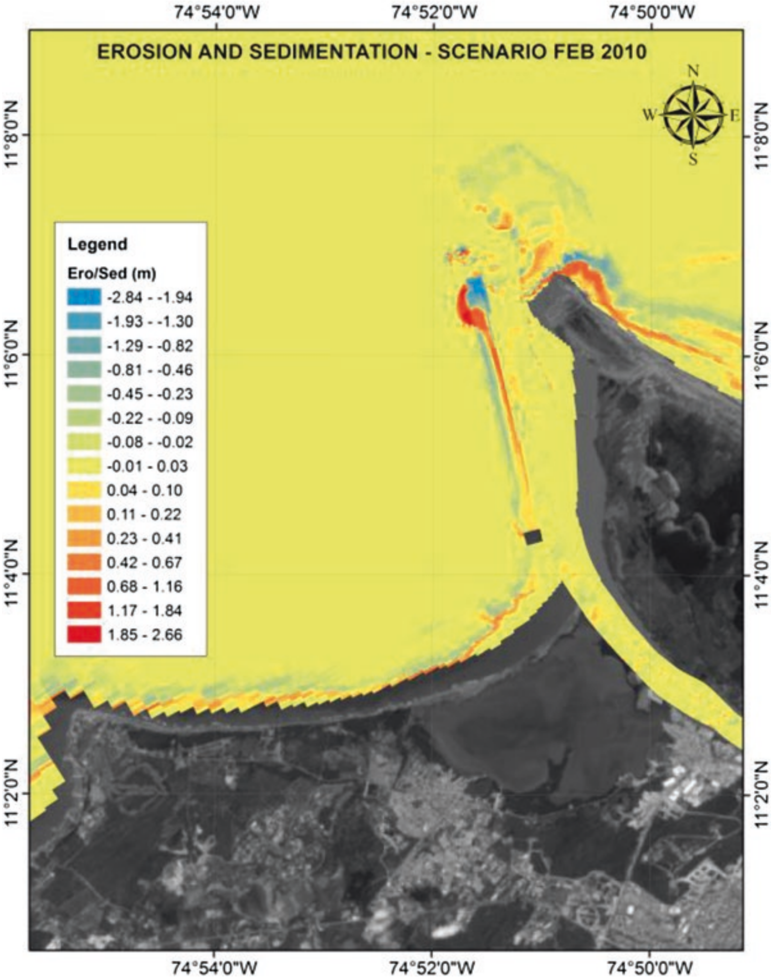

Fig. 9.25 Erosion and accretion of Bocas de Cenizas zone after 30 days of hourly modelling, since October 1st, 2010 to November 01st, 2010

values, the waves during this time changes its direction so that only reach maximum wave heights of 0.6 m and a minimum of up to 0.1 m (Fig. 9.24b).

9.5.6 Hydromorphological Modelling

Results of the erosion (blue color)/accretion (red color) modelling in the bottom (Fig. 9.25) for February on 2010, allow observing that predominates the material accumulation of up to 0.10 m in external borders of both the Eastern and Western Groin. However, at the extreme of the western groin is generates an erosion point

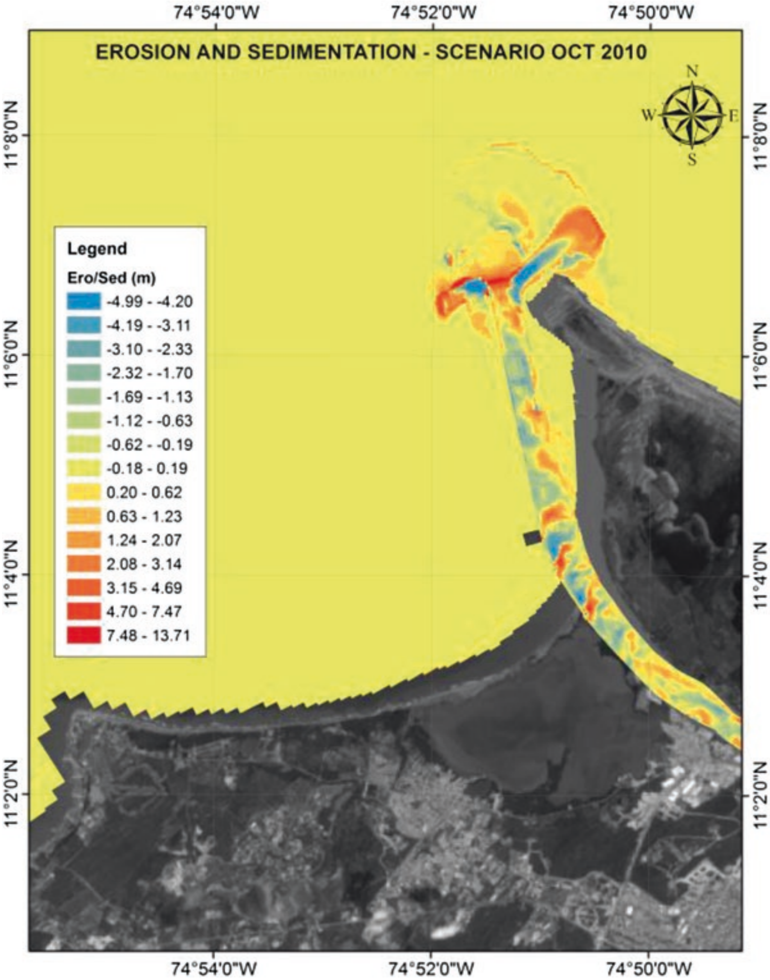

Fig. 9.26 Erosion and accretion of Bocas de Cenizas zone after 30 days of hourly modelling, since October 1st, 2010 to November 01st, 2010

with loss of material of up to −2.84 m. That material is deposited at the same height in the protected border where the generated currents by the diffraction, allow the sediment transport to the Mallorquín Lagoon. Even, it can observe how the first part of the "Superpuerto" works interrupt the sediment transport to the coast. Nevertheless, part of the sediment reaches bordering the construction work and travels in parallel direction to the coastline, supporting in lesser extent to the coastal bar of the Mallorquín Lagoon. It can observe in its proximities, accumulations of sediments from 0.31 to 0.46 m.

Based on results of erosion and accretion during October of 2010 (Fig. 9.26), it is generated deposit zones in the river mouth product of the highest waves that arrive to the zone by the effect of cold fronts and even extreme hydrometeorology phenomena.

There are points with material accumulation up to 14 m, and losses up to 4.9 m towards the sector of the border of the western groin. The strong meteo-marine activity creates a change in the local hydrodynamic helped with the change in the wave direction, which affects the sediment transport to the wetland. For this reason, during this season, the bar diminishes in some sectors its dimension in plant.

9.5.7 Analysis of Evolution in Plant of Mallorquín Lagoon

One of the most important activities for the professionals who study coastal processes are geomorphologic changes that are produced as result of the establishment of coastal infrastructure, or in general, natural changes that suffers a littoral environment of a rigid physical configuration of geological origin.

For the changes analysis that commonly presents a beach, it is necessary to know in detail which is the equilibrium state of the coastal bar Ruiz (2004). This includes not only a typical beach but also a barrier island that protects a wetland of the wave action. For that, since some decades ago, it is implementing theoretical model that allows from the establishment of geometric functions known to determine the final shape of the beach configuration.

The concept of static equilibrium, proposed by Tanner (1958), is defined by a confined coast in a condition of natural or artificial border that allows the generation of a soft beach in harmony with the conditions of maritime climate.

The shape in plant is adjusted to the local wave conditions in order to promote the sediment transport in the beach and for defining a configuration that achieves the proper energy dissipation that guarantees the equilibrium state in the time. Apart from the aforementioned, a water front that presents an unbalance condition, will vary its shape until finds the proper condition to guarantee the sediment transport and deposition process that travels to the beach. In this sense, the diffraction and refraction processes; and in some cases, the reflection take a fundamental role in the shape that will have the beach.

In relation to the beach shape, there are a lot of definitions in the state of the art. Krumbein (1947) denominate them as salient beaches, Silvester et al. (1980) describes them as pocket beaches and Yasso (1965) know them as salient beaches, to name just a few of them.

According to the described above, the static equilibrium represents that condition where the beach neither gains nor losses sediment, only transports and distributes along the coastline as a result of a symmetry between the isobaths, the coastline and wave fronts. This, in physical terms, allows concluding that the breaking wave takes place around the beach profile, thus avoiding excessive erosion. There is not littoral transport in the beach, and in accordance with Leblond (1979) there is a condition of proportionality between the sediment, the profile and the wave that travel to the coast.

In accordance with Hsu and Evans (1989), a beach has two types of equilibrium condition: static and dynamic. The dynamic equilibrium exists naturally when the beach has a continuous supply of sediment that induces a change in the width of recurring beach. Physically, this represents a coastline that follows a logarithmic profile, with a fixed point on a border where begin a curved section to the end section that is parallel to the coastline. It can underline with precision that the shape of a beach in plant, is a measure of how the currents transport the material.

The types of beaches, such as the found in the Mallorquín Lagoon, have been formed by the morphological evolution resulting from the action of waves and erosion processes over decades, obtaining progressively the shape presented today. This document attempts to reach a level of understanding of how evolved the beach front that protects this coastal wetland and attempt to infer the processes of change and loss of stability that can be achieved by establishing new physical elements in the littoral system.

9.5.8 Model of Parabolic Curve

For analysis on plant, a parabolic model developed by Hsu and Evans (1989) was applied, which fits very well to the planimetric shape of a beach. In their model, it is defined a point of diffraction from which wave fronts change direction. This point must be located on land, in a salient, a natural barrier or a coastal structure. Whether the beach is in equilibrium, it should be considered the time that the wave takes to propagate from the point of diffraction to the coastline. Whether, there is also a wide area of breaking waves since the submarine platform is very shallow, it should take

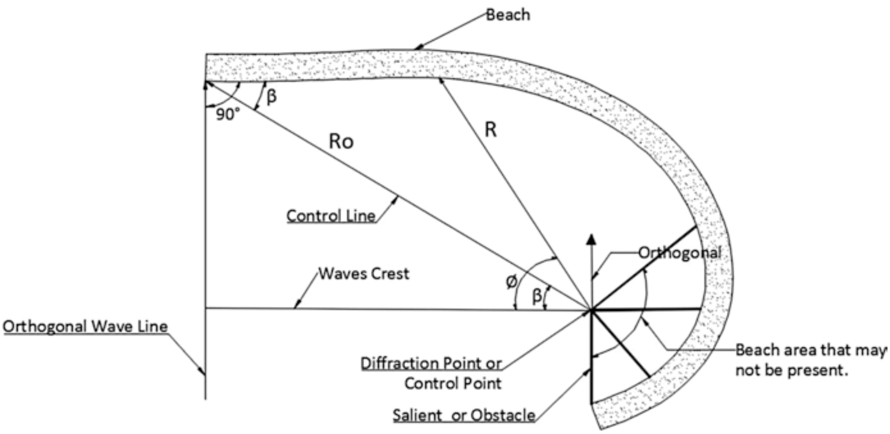

Fig. 9.27 Model of parabolic curve

into account the propagated wave refraction (Fig. 9.27). The general expression describing the parabolic model is:

$$\frac{R}{R_o} = C_o + C_1\left(\frac{\beta}{\theta}\right) + C_2\left(\frac{\beta}{\theta}\right)^2 \tag{9.14}$$

$$C_o = 0.0707 - 0.0047\beta + 0.000349\beta^2 - 0.00000875\beta^3 + 0.00000004765\beta^4$$

$$C_1 = 0.9536 - 0.0078\beta + 0.0004879\beta^2 - 0.0000182\beta^3 + 0.0000001281\beta^4$$

$$C_3 = 0.0214 - 0.0078\beta + 0.0003004\beta^2 - 0.00001183\beta^3 + 0.00000009343\beta^4$$

Where C_0 , C_1 , C_2 C_o, C_1 *y* C_2 are dimensionless coefficients that vary with β. β is the angle between the control line and the tangent point where the coastline becomes straight. This angle, generally, is between the ranges from the 10° to the 80°, because it covers most of obliquities with the waves arrives recurrently to salient beaches R is the radius from the diffraction point to the coastline which has an angle θ with respect to the wave front. R_o that starts in the salient to the point where the coastline becomes straight.

It is important to note that if the modelling line is adjusted to the actual coastline, the physical meaning of this result indicates that the beach is in static equilibrium.

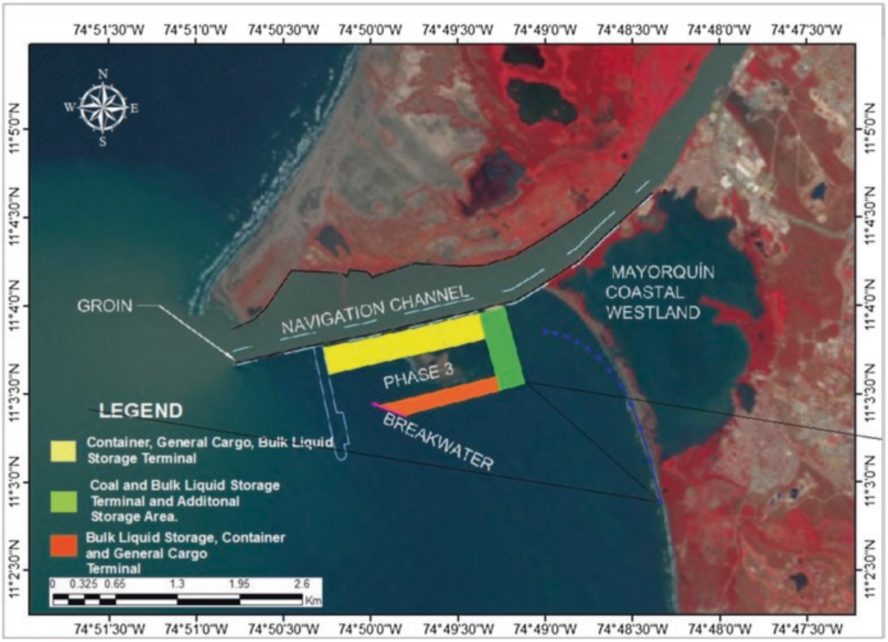

Fig. 9.28 Results of MepBay model. Evolution of the coastal bar due to the Superpuerto works

In case of variations in the model adjustment, there is an indicator that reflects a dynamism on the beach, which can be interpreted as dynamic equilibrium. Therefore, the beach is unstable and can erode quickly at the slightest change in the behaviour of the coastal system.

The modelling of the plant configuration and application of theoretical parabolic model was carried out with the MepBay model as shown in Fig. 9.28, developed by the University Of Santa Catarina Brazil. The importance of using the MepBay lies in the fact that efficiently determines morphological changes in plant and perform a physical interpretation of the equilibrium state of the beach (Raabe et al. 2010).

The modelling results (Fig. 9.28) indicate that with the establishment of the Maritime Terminal "El Superpuerto", the balance of the beach will be seriously affected. The coastline undergoes significant changes, especially in the shadow area generated by the port. However, this condition of the coastline gain should be analysed carefully, because only it will be presented if the current sediments source remains unchanged of the coastal zone (Fig. 9.28). However, the reality is very different, because for the entry of large ships will be built a navigable channel of 20 m of deep. This indicates that the little sediment currently crosses by the effect of diffraction of the salient in the groin (Tajamar) will end at the bottom of that navigable channel.

9.5.9 Identification of Potential Erosion

The erosion rate of the coastline from many countries has dramatically increased in recent decades. These forces to, the professionals dedicated to the study of coastal processes, carry out a regular monitoring of changes that are seasonally generated on a beach. This is not unrelated to the bars that protect coastal wetlands, even if they have been affected over the years by the establishment of infrastructure for the development of human activities in coastal environments.

A simple and cost-effective way to elaborate a changes diagnosis and the quantification of potential erosion in a coastal bar, it is based on the analysis of satellite images. With images of LANDSAT project and the use of the application of Geographic Information Systems (GIS), it was effected a variability analysis of the spatial and temporal coastline, from1973 until today. The analysis of the coastline and its changes over time are a powerful tool to determine processes of erosion and accretion in a littoral cell, evaluating differences between space increments of each of the lines drawn.

The methodology consists in drawing for each year and with a seasonal frequently, the polyline representing the coastline for the period measured. In one image, the different polylines overlap to get a map like the one shown in Fig. 9.29.

Then, to determine the potential for erosion suffered by the wetland littoral bar, the Digital Shoreline Analysis System (DSAS) developed model by the US Geological Survey was used (Thieler et al. 2009). Based on selected segments in the model it described the most representative changes on the coastline. For its applica-

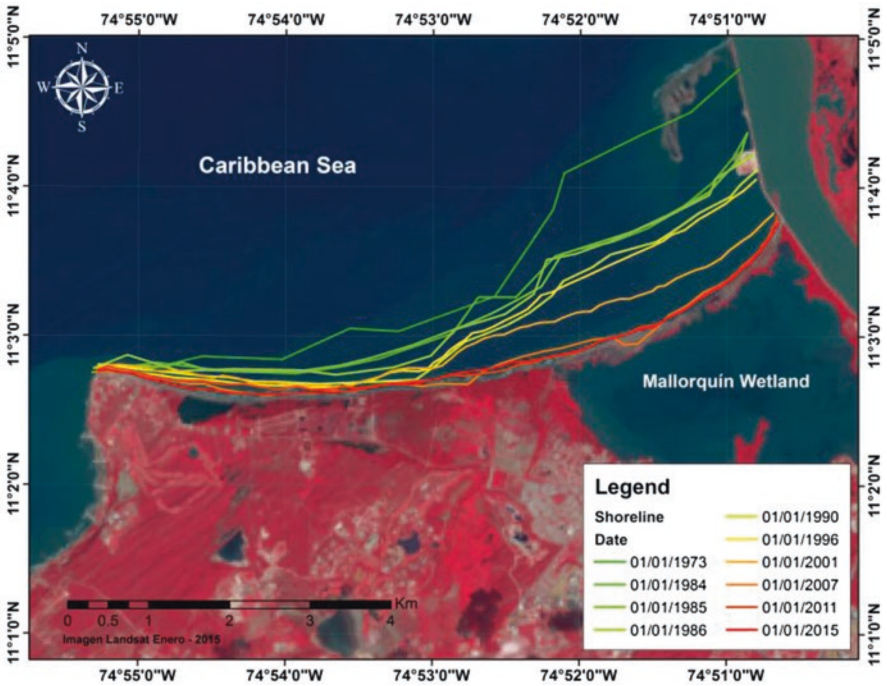

Fig. 9.29 Historic shoreline evolution of Mallorquín Coastal Wetland, Atlántico. From 1973 to 2016

tion, it is requires the establishment of transversal sections spaced uniformly in the sector where will be evaluated the coastal erosion.

After an attributes revision between the generated lines, satellite imagery and polylines, it is develop the model to statistically determine the change rate of the coastline and consequently the erosion potential.

This model calculates the potential erosion using different methods: (a) a linear regression; (b) weighted least squares regression; (c) the change rate on an end-point; and (d) jack-knife iterative regression techniques

For this study, it was used the No. 1 method because there were a complete data series, representative statistical of the historical variability of the coastline.

As it shown in Fig. 9.30, the most intense changes that have taken this coastal lagoon are in the vicinity of Bocas de Ceniza groin. It can appreciate that historically this location has presented an erosion rate per year of 0.14 m which extends along the entire bar that protects the Mallorquín Lagoon. In the place where finishes the subsystem or sub cell in the discharge of León stream, the rate decreases to 0.11 m/year (Fig. 9.30). The erosion continues to decline until it reaches the border of the cell in the area known as Punta Roca, where have cliffs and an abrasion platform of more than 8 m of height.

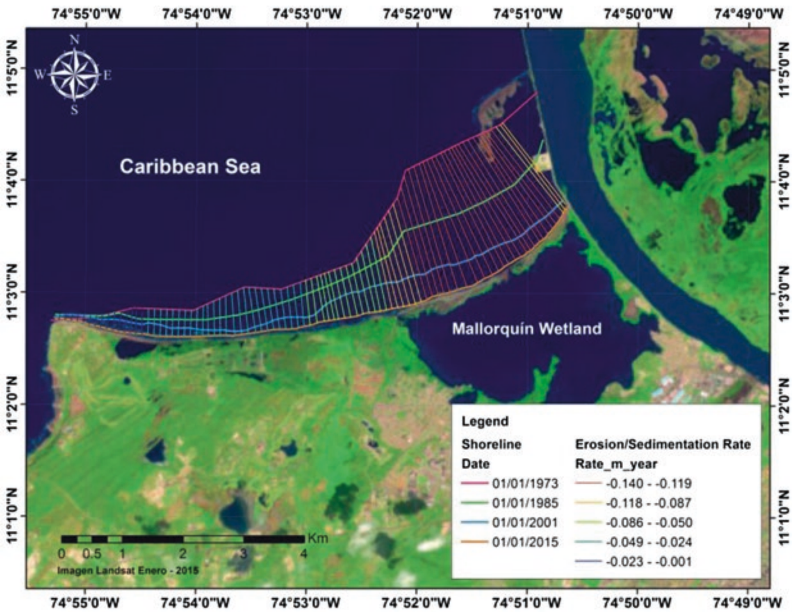

Fig. 9.30 Rates of shoreline change computed using linear regression

9.6 Conclusions

Based on the results obtained, it can be concluded that the associated littoral bar to the coastal wetland “Mallorquín Lagoon” is in dynamic equilibrium. It should be noted that any change in the marine dynamics will produce drastic changes in the behaviour of the wetland, reducing the width of the bar and perhaps its total loss.

From the hydrodynamics perspective, it is clear that the lack of sediment in the system, whose contribution rate varies depending on season weather conditions. There is a gain during the dry season and a reduction of the sedimentary contribution during the humid season by the change in wave direction.

The temporal evolution of the coastline confirms that the wetland has lost its water mirror and has undergone an intense erosion process. The coastline has moved about 2.5 km, with rates recession 0.14 m/year in the most critical area.

For all these reasons, it is recommended to take preventive measures that take into account the way how will be reduced environmental impacts for the loss of sediment that arrives to the bar, since it is expected that the navigable channel of 20 m traps this sediment that comes from the estuary of the Magdalena River. There are two potentially negative effects. A back siltation process of navigable channel and in the port; and a loss of the coastal wetland.

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Chapter 10
Long Term Impacts of Jetties and Training Walls on Estuarine Hydraulics and Ecologies

Alexander F. Nielsen and Angus D. Gordon

Abstract Data and theory show that the inlets of several large estuaries on Australia's eastern seaboard that appeared to be stable within a range of entrance conditions are demonstrating unstable scouring modes and have been doing so for decades, if not centuries, since entrance jetties had been constructed. Jetties have increased the hydraulic conveyance of the entrance channels by removing sand bars and extraneous littoral currents that impeded ebb tide discharges. Field data comprising comprehensive water level monitoring in the bays, enabling the definition of tidal planes to a high resolution, have shown that the spring tidal ranges of these bays has been increasing steadily for decades with high tide planes rising and low tide planes falling. The field data have indicated that these changes show no signs of stabilizing and Escoffier analyses have indicated that it could take centuries for these inlets to reach new stable hydraulic regimes. Implications have included extensive scour in the entrance channels requiring channel erosion protection works, subsidence of road bridges, collapse of foreshores including buildings, sedimentation in the bays and on adjacent beaches and permanent changes to fringing marine ecologies and fisheries. Changes to the distribution of seagrass, saltmarsh and mangrove forests have been observed to coincide with and confirm the expectations of impacts on marine ecology that could derive from jetty construction. While jetties have improved flood conveyance significantly the increases in ebb tide velocities have resulted in navigational hazards for recreational boating.

Keywords Tidal constituents • Amplitude • Phase • Prism • Inlet • Equilibrium area • Escoffier stability analysis • Jetties • Training walls • Marine ecology

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10.1 Introduction

Continuously, natural inlets on littoral drift shores comprising entrance bars, shoals and channels are in a state of flux, changing in response to variations in the controlling hydrodynamic forces such as floods, variations in the spring-neap tidal range, varying wave climates and rates of littoral drift transport. Typically, they scour during flood events but, subsequently, trend towards closure as littoral drift reforms the entrance bars and shoals, potentially closing the inlet. For much of the time the wetlands associated with natural inlets are subjected to small tidal ranges, reflecting shoaled entrances, which control the extent and diversity of the wetland ecology.

The sensitivity of estuarine responses to such variations depends primarily on the size of the estuary. The most noticeably sensitive estuaries are the small bays and lagoons, the ocean entrances of which, generally, are closed but can be opened abruptly to wave and tidal forcing following floods. Such small estuaries are the least stable when open (Brown 1928; Bruun 1978; Gordon 1990). On the other hand, large estuaries, while open to the ocean for much, if not all, of the time and, hence, exposed to a greater range of variable hydrodynamic forcing, often have a tidal discharge and a channel cross-sectional area that appear to fluctuate about stable average values.

For the larger estuaries, such as those commonly used for recreational boating or commercial fishing, ever-changing bars, shoals and channels present uncertainty and risks to navigation. Further, shoaled entrances can result in the backup of floodwaters causing inundation of waterfront properties. Often the response to navigational and flooding issues has been to construct entrance channel improvements such as training walls and jetties. In most cases these works have achieved their intended results, often spectacularly, but many have been implemented without an understanding of the potential long term impacts. Training walls and jetties can alter estuarine hydraulics significantly, increasing hydraulic conveyance, inducing scouring of the channels, changing tidal planes and, hence, changing the environmental conditions of the associated wetlands.

Large estuaries respond slowly to perturbations at their entrances and the signature of any change to their stability may go undetected for many years to decades. However, once set in motion, a change to the dynamic stability of a large estuary, which may have been occasioned by jetty construction, for example, or from a rising mean sea level, will be difficult to predict both in the degree of change and time to reach a new state of dynamic equilibrium.

As a result of technological advances in water level data loggers there is now a large and growing body of empirical data that allows for a closer examination and definition of estuarine hydraulics and inlet stability. Hourly water level recordings allow the determination of high-resolution, objective, statistical estimates of the tidal constituents on an annual basis that can be used to define accurately the relevant parameters of tidal range, phasing, prism and levels that are used in estuary stability theories. Examining the time histories of amplitude and phase of tidal constituents within estuaries where jetties have triggered unstable scouring modes, along with classic estuary stability theory, informs the hydraulic and ecological impacts of jetty and training wall construction and their future prognosis.

This chapter outlines theories related to the hydraulics of tidal channels and bays in communication with the ocean, the impacts of training walls and jetties on estuarine hydraulics, sediment transport, channel stability and the subsequent impacts on the marine ecology of the associated estuaries. The impacts are illustrated with three well-documented examples from two large and one smaller estuary on the Australian eastern seaboard.

10.2 Estuary Characteristics

10.2.1 Conceptual Model

The particular situations described herein apply only to specific estuary types and do not include large drowned river valleys or long narrow rivers that, generally, are not as susceptible to these processes. The natural elements that characterize the behavior of the estuaries of interest include an infinitely large water source (the ocean) with a periodic tide of reasonable amplitude connected to a relatively large coastal bay or lagoon that has the potential to generate a large tidal prism, by a relatively narrow and shallow erodible sandy channel; the ocean entrance of the channel being exposed to waves and the ingress of littoral drift.

In their natural state, on shores experiencing relatively high rates of littoral drift transport and in the absence of any significant precipitation, such estuaries trend towards closure (Brown 1928; Bruun 1978; Gordon 1990). However, flood events and, sometimes, large spring tides may scour the surf zone bars, the entrance channel and shoals, thereby removing the littoral drift that had been deposited over time and had choked the flow. The resulting reduction in hydraulic impedance at the entrance produces greater tidal discharges and, hence, the tidal ranges in the bay increase for a time. However, during ensuing dry periods, waves and currents re-establish the entrance bars and move littoral drift back into the entrance, resulting in the reformation of the marginal shoals, thereby increasing the impedance to the penetration of the tidal wave with the resulting loss of hydraulic efficiency to drive flows to and from the bay. The tidal range in the bay reduces progressively until either the entrance closes or the tidal flushing and occasional rainfall events are sufficient to keep a channel open, albeit in a shoaled state.

The action that alters fundamentally the natural cycle described above is the construction of entrance jetties that both increase depths over the offshore bars and limit or eliminate the ingress of littoral drift to the entrance channel. This produces a more hydraulically-efficient entrance allowing greater penetration of the tidal wave into the estuary. This increase in hydraulic conveyance depends upon the degree to which the works modify the behavior of the entrance bar. Experience on the Australian eastern seaboard has shown that a single jetty tends to have only a

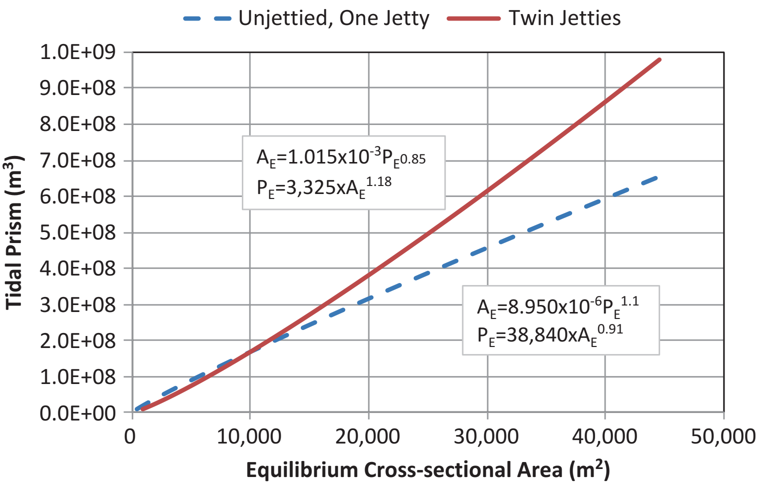

Fig. 10.1 Tidal prism vs. channel cross-sectional area for Pacific coast inlets (After Jarrett 1976). The boxes present the original equations (in metric units) with an inverted format presenting the equilibrium tidal prism as the dependent variable plotted on the ordinate. For the larger cross-sectional areas the tidal prisms for twin-jettied inlets is always larger than those with one or no jetty

modest impact on improving the overall hydraulic efficiency of an entrance. This is because a component of the gross littoral drift can enter the entrance channel from the unprotected side. However, twin jetties that intersect the surf-zone increase the hydraulic conveyance significantly. Such differences are found also on both the USA Atlantic and Pacific coasts where, for the same cross-sectional areas, twin jetties generate larger tidal prisms than do single jetty entrances or inlets without jetties (Jarrett 1976; see Figs. 10.1 and 10.2).

With the reduction of hydraulic impedance at the entrance due to twin jetty construction, the channel connecting the ocean to the bay begins to scour under steepened hydraulic gradients that increase channel velocities (Nielsen and Gordon 1980, 2008, 2011, 2015). This establishes a positive feedback loop; as channel depths increase, channel friction reduces and hydraulic gradients become steeper resulting in ever greater tidal discharges and velocities. Finally, but after a long time, the process begins to slow down as progressively increasing bay tidal ranges and reducing bay tidal phase lags begin to reduce hydraulic gradients and, hence, scour potential. As channel velocities approach the equilibrium velocity (O'Brien 1931, 1969; Jarrett 1976), channel scour ceases and a new stable hydrodynamic regime is reached. The larger the bay the longer it will take to reach a new equilibrium. Constraints to this runaway situation created by the construction of twin jetties can include bedrock controls, bank protection works or the imposition of relatively large structures, such as bridge abutments or a marina, in the entrance channel.

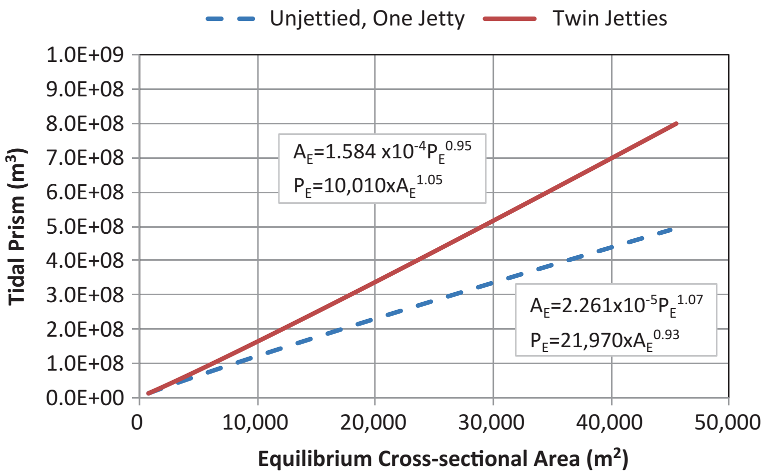

Fig. 10.2 Tidal prism vs. channel cross-sectional area for Atlantic coast inlets (After Jarrett 1976). The boxes present the original equations (in metric units) with an inverted format presenting the equilibrium tidal prism as the dependent variable plotted on the ordinate. For the larger cross-sectional areas the tidal prisms for twin-jettied inlets is always larger than those with one or no jetty

10.2.2 Tidal Constituents

The tidal signature in a waterway is a characteristic, sinusoidal oscillation comprising either two main cycles per day (semi-diurnal tides), one cycle per day (diurnal tides), or a combination of the two (mixed tides).

The underlying principle of tidal analysis is that a time series of tidal oscillations can be deconstructed into a series of regular sinusoids, usually represented by the cosine function, each having the period of oscillation of the celestial forcing that gives rise to it. The tidal harmonic constants comprise the amplitude and phase of the individual cosine waves, each of which represents a tidal constituent identified by its period.

While there may be scores of tidal constituents computed for a tidal harmonic analysis, the following are the major contributors to the astronomical tidal stage variation:

M2 – principal lunar semi-diurnal constituent
S2 – principal solar semi-diurnal constituent
K1 – luni-solar declinational diurnal constituent
O1 – lunar declinational diurnal constituent.

Empirical and analytical formulations for estuary hydraulic analyses rely on good estimates of the estuary's tidal planes, either to determine accurately the spring tidal prism or to determine accurately the bay-to-ocean tidal range ratio. The spring tidal range, mean high water springs (*MHWS*) minus mean low water springs (*MLWS*), is defined as:

$$MHWS - MLWS = 2 \times \left(M2_{amplitude} + S2_{amplitude} \right) \tag{10.1}$$

Accurate and consistent estimates of these tidal constituent parameters can be obtained only from long-term continuous tide recordings. Usually, such data are available only from government-operated sites, often managed by hydraulic laboratories.[1]

10.2.3 Channel Flow and Training Walls

Training walls may be constructed along the banks of an entrance channel, often to manage bank erosion. Such works can change the hydraulic characteristics of a channel and, invariably, increase its hydraulic conveyance by reducing the impedance to flow.

The basic equation for open channel flow is termed the Manning (or sometimes Strickler's) equation thus, in metric units (for example, see Henderson 1966):

$$v = \frac{R^{\frac{2}{3}} \times S^{\frac{1}{2}}}{n} \tag{10.2}$$

where:

v = channel velocity (m/s)
R = hydraulic radius (m)
  = A_C/P
A_C = channel cross-sectional area (m^2)
P = wetted perimeter (m)
S = energy or water surface slope (-)
n = Manning's bed roughness coefficient (-).

The instantaneous channel discharge, q, is the product of the average channel velocity, v, and the cross-sectional area, A_C, thus:

$$q = v \times A_C \tag{10.3}$$

Figure 10.3 presents schematic diagrams of three channel types; a natural channel in sand with typical side bed-slopes of 1:10 (vertical:horizontal), a channel in

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[1] The field data upon which the research herein was based was provided generously by the NSW Government Public Works Department Manly Hydraulics Laboratory. The authors take responsibility for its analysis and interpretation.

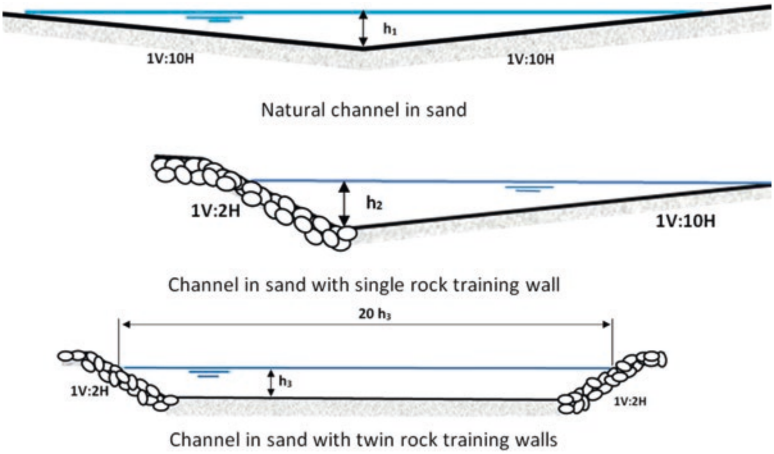

Fig. 10.3 Schematic diagrams portraying (1) a typical channel in sand (*top*); (2) a typical channel in sand with a single rock training wall (*middle*); (3) a typical channel in sand with twin rock training walls (*bottom*). Fundamental equations for channel hydraulics applied to these sections with equal areas demonstrate that training walls enhance hydraulic conveyance

sand with a single rock training wall of side slope 1:2, and a channel in sand with twin rock training walls. It can be demonstrated using Eqs. 10.2 and 10.3 that, for channels with equal cross-sectional areas, bottom roughness and water surface slopes, the channel with a single training wall would discharge 16% more water than the natural channel and the channel with twin training walls would have a 20% higher discharge than the natural channel. Thus, training walls alone improve significantly the hydraulic conveyance of tidal channels, without taking into account the impact that jetties may have on altering littoral drift processes.

10.2.4 Inlet Stability Theory

The understanding of estuary inlet hydraulics and stability is based on the synthesis of empirical and analytical formulations.

Empirical formulations (O'Brien 1931, 1969; Jarrett 1976; Bruun 1978) comprise the identification of relevant parameters, such as tidal prism, entrance cross-sectional area and rate of littoral drift transport to the inlet, relating cause and effect, and the development of relationships between these parameters using coefficients derived empirically from many field observations.

Analytical approaches comprise the development of generalized formulae from mechanism understanding relating bay tidal range (tidal prism) and lag, entrance channel area and velocity, channel head losses, friction and the forcing ocean tidal range (Brown 1928; Escoffier 1940; Keulegan 1951, 1967; O'Brien and Dean 1972; Czerniak 1978; van de Kreeke 1992; Seabergh 2003).

10.2.4.1 Empirical Formulations

From empirical data, O'Brien (1931) proposed that the stable inlet cross-sectional area could be determined from the tidal prism using the relationship (in metric units):

$$A_E = 9 \times 10^{-4} P_E^{0.85} \tag{10.4}$$

where:

A_E = equilibrium cross-sectional area below mean sea level (m^3)
P_E = equilibrium spring tidal prism (m^3).

Many other similar relationships with different constant and exponent values have been developed for various sites and differing entrance jetty configurations (Jarrett 1976; van de Kreeke 1992; Seabergh and Kraus 1997).

Of particular interest are the data from Jarrett (1976) on inlet configurations comprising twin jetties, a single jetty and/or no jetty, presented in Fig. 10.1 for the US Pacific coast and Fig. 10.2 for the US Atlantic coast, which show that estuaries with twin jetties invariably have larger tidal prisms than do those with a single jetty or no jetty, implying that twin jettied entrances have greater hydraulic conveyance.

These relationships between the spring tidal prism and the channel cross-sectional area of stable inlets imply that equilibrium channel velocities can be determined for inlets on various coasts with differing tidal regimes and with differing entrance configurations. If the form of the discharge curve can be assumed to be sinusoidal, the tidal prism can be related simply to the peak ebb tide discharge and, hence, the peak (maximum) channel velocity. Therefore, the equilibrium velocity for any stable inlet can be determined from these relationships thus:

$$P_E = \frac{v_{E_{\max}} A_E T}{\pi} \tag{10.5}$$

where:

P_E = equilibrium tidal prism (m^3)
v_{Emax} = maximum equilibrium channel velocity (m/s)
A_E = equilibrium channel cross-sectional area (m^2)
T = tidal period (s)

Combining Eqs. (10.4) and (10.5), for semi-diurnal tides – tidal periods of 12.42 h – the following relationship between the equilibrium flow area and equilibrium maximum channel velocity is derived from O'Brien's (1931) equation (in metric units):

$$v_{E_{\max}} = 0.269 A_E^{0.176} \tag{10.6}$$

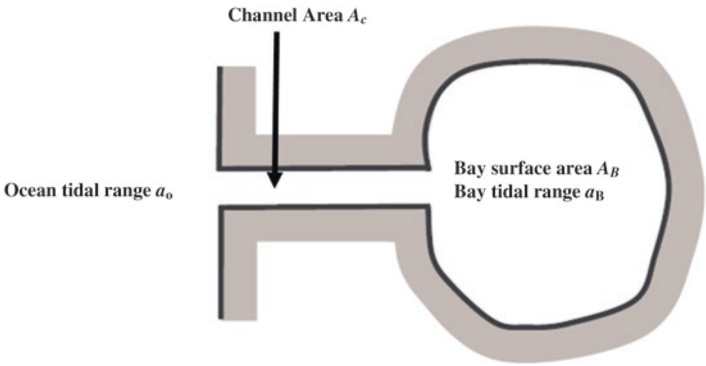

Fig. 10.4 Idealized estuary for Escoffier inlet analysis. The analysis assumes short regular entrance channels without surf zone bars and shoals

Similar equations can be derived from the myriad of prism/area relationships that have been developed since that of O'Brien (1931). For example, from Jarrett (1976) for twin jetties on the Pacific coast, the relationship is (in metric units):

$$v_{E_{\max}} = 0.509 A_E^{0.081} \tag{10.7}$$

Bruun (1978) presented a simple empirical relationship, based on the ratio of the tidal prism to the rate of littoral drift transport to the entrance, which determined whether an estuary may be prone to closure or may remain open. Low values of the ratio denoted an unstable shoaling mode (in the terminology of O'Brien and Dean 1972). However, this method does not predict an unstable scouring mode or a stable channel cross-sectional area. Further, the method does not apply to entrances where the littoral drift transport to the inlet has been interrupted by jetties.

10.2.4.2 Analytical Formulations

Brown (1928) and Escoffier (1940) presented a generalized analytical approach for a simple idealized estuary system, as shown in Fig. 10.4, relating the bay tide phasing and amplitude to that of the ocean tide through the hydraulic characteristics of the entrance channel. The method is applicable to small and large tidal inlets on shorelines where the rates of littoral drift transport are small. It favors estuaries with relatively short and regular entrance channels connecting the bay to the ocean.

Through the construction of an Escoffier Diagram, the status of an inlet at any point in time can be examined and, theoretically, the ultimate stable cross-sectional area of the entrance channel can be predicted. The method is outlined as follows (after O'Brien and Dean 1972).

Keulegan (1951, 1967) developed the relationships between the tidal phase lag, the ratio of the bay-to-ocean tidal amplitudes (a_B/a_O) and what is termed the

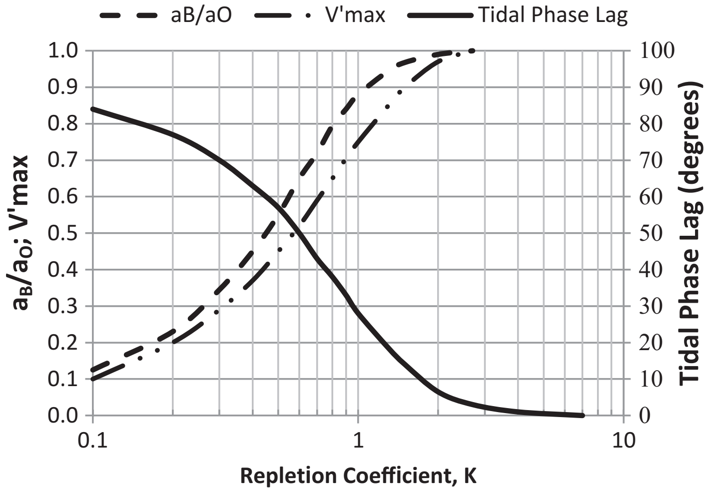

Fig. 10.5 The relationship between the Repletion Coefficient, K, and the ratio of the bay tide to ocean tide, a_B/a_O, the tidal lag and the dimensionless maximum velocity, v', in the entrance channel (From Keulegan 1951, 1967). Increasing K values denote more efficient repletion or filling of the bay storage volume as channel velocities increase, bay tidal ranges increase and tidal phase lags decrease, the high tide level in the bay being reached sooner

Repletion Coefficient (K; increasing K implies a more efficient repletion or filling of the bay storage volume), as shown in Fig. 10.5. Keulegan (1951, 1967) presented also the relationship between the repletion coefficient and the dimensionless maximum velocity in the inlet (v'_{max}), shown in Fig. 10.5, where the maximum velocity through a specific inlet is given by:

$$v_{\max} = v'_{\max} \frac{2\pi}{T} a_O \frac{A_B}{A_C} \tag{10.8}$$

The Repletion Coefficient, K, may be expressed as a function of the hydraulic and geometric properties of the estuary as follows:

$$K = \frac{T}{2\pi a_O} \frac{A_C}{A_B} \sqrt{\frac{2ga_O}{k_{en} + k_{ex} + \frac{fL_c}{4R}}} \tag{10.9}$$

where:

T = tidal period (s)
a_o = amplitude of the ocean tide (m)
A_C = inlet cross-sectional flow area (m^2)
A_B = surface area of the bay (m^2)
g = gravitational constant (m/s^2)

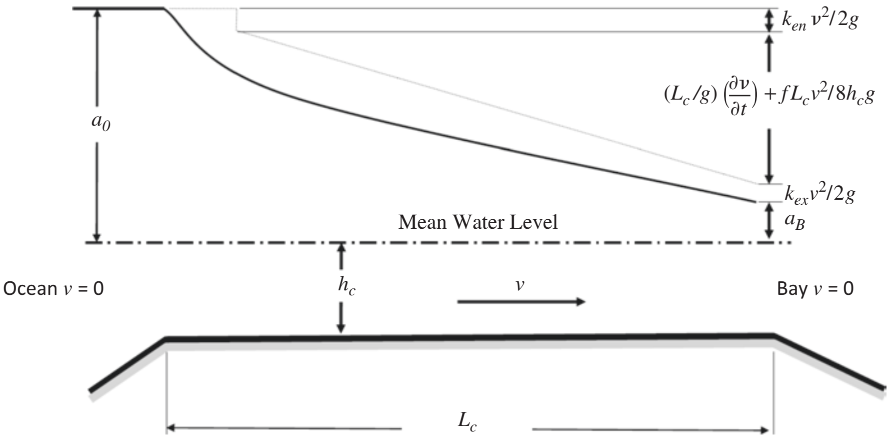

Fig. 10.6 Idealized entrance channel head losses assumed in the Escoffier inlet analysis depicting channel entrance (k_{en}) and exit (k_{ex}) losses and friction losses along the channel length for a channel velocity v. No entrance bar losses are assumed

k_{en}, k_{ex} = channel entrance/exit head loss coefficients (Fig. 10.6)
f = friction factor (-)
 $= 0.113(k_s/R)^{1/3}$ (Henderson 1966)
k_s = height of surface roughness (m)
R = hydraulic radius of entrance channel (m)
L_c = friction length (m; Fig. 10.6)

Where Eqs. 10.8 and 10.9 and Fig. 10.5 are used together, a range of inlet hydraulic conditions can be calculated in terms of the maximum velocity versus cross-sectional flow area. The inlet mechanics are portrayed by the inlet stability curve or Escoffier Diagram (Fig. 10.7).

It can be seen from this diagram that an induced change in the cross-sectional flow area of an hitherto hydraulically stable inlet, which has a flow area (A_E) in equilibrium with the tidal prism (P_E), will result in either a change in inlet current velocity that will work to return the inlet towards its equilibrium flow area by appropriate deposition or scour or, if the induced area change is so large as to reduce the cross-sectional area below the critical flow area, A'_c, making the inlet hydraulically unstable. An hydraulically unstable inlet is characterized by increasing friction with decreasing cross-sectional area or vice versa. The result is that if any natural or man-induced change in flow area occurs, this is accompanied by a change in the flow velocity that will, by inducing scour or deposition, perpetuate the induced area change. Since area changes are perpetuated, an hydraulically unstable inlet will either scour continuously until a stable flow area is achieved (unstable scour mode) or it will shoal continuously until inlet closure (unstable shoaling mode).

Other interpretations of the Escoffier Diagram place significance on the first (or lower) intersection of the “closure” curve with the equilibrium P_E/A_E relationship, classifying unstable inlets as only those having cross-sectional areas smaller than

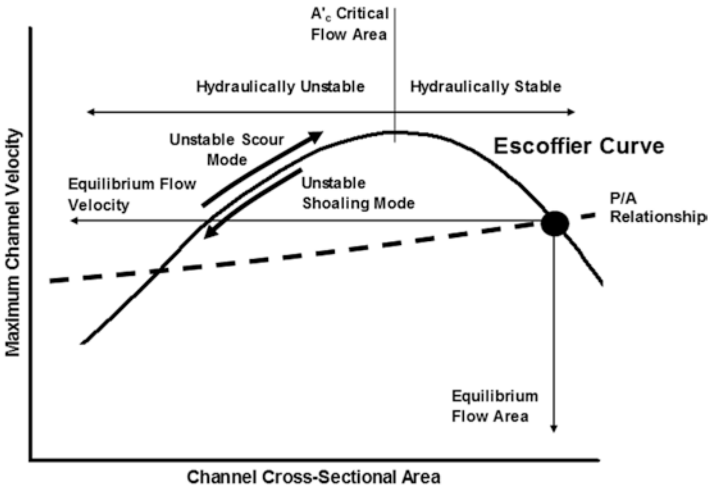

Fig. 10.7 Inlet stability curve or Escoffier diagram. An hydraulically stable inlet will strive to have a tidal prism/channel cross-sectional flow area as determined by the equilibrium *P*/*A* relationship. Natural tidal inlets with shoaled entrances may be induced to either scour to reach the equilibrium flow area or shoal to closure

that value (van de Kreeke 1992). Escoffier (1940) dismissed any relevance of this lower root.

Seabergh (2003) discussed approaches other than those using an equilibrium P_E/A_E relationship for determining the stable equilibrium condition on the Escoffier Diagram. Mota Oliviera (1970) proposed that the stable point may be reached when the Repletion Coefficient reached a value between 0.6 and 0.8. This was based on calculations that included the consideration of tidal stage with maximum ebb tide scour velocities, which occurred on low waters rather than on the Keulegan (1967) assumption of average depth. Another approach of Skou (1990) proposed that the most optimum situation for an inlet to remain stable was where the cross-sectional area coincided with the maximum gradient of the Escoffier curve.

When constructing the Escoffier Diagram for a particular estuary the following approach can be used (after Czerniak 1978):

- a_B/a_O is calculated from existing data and K is determined from Fig. 10.5
- Equation 10.7 is solved for L_c using K and known values of A_c, R, T, a_O, A_B, f, k_{en} and k_{ex}
- The hydraulic stability curve is computed using Eqs. 10.6 and 10.7 and Fig. 10.5 (for v'_{max}) with only A_c (and, consequently, R and, hence, f) varying over the entire range, maintaining the ratio of channel width to depth used in the calibration.

In Eq. 10.7, four head loss parameters (k_{en}, k_{ex}, f, L_c) are used to describe the total impedance of the entrance channel to the flow. A typical value for $k_{en} + k_{ex}$ is 1.3 (O'Brien and Dean 1972) with $f = 0.02$ being adopted for the calibration conditions adopted herein, thereafter f varying with R as indicated in Eq. 10.9.

There are two major sources of head loss between the forcing function, a_O (the amplitude of the ocean tide) and the bay response, a_B. These are the head loss over the entrance bar and the losses associated with the hydraulic characteristics of the entrance channel. As the forcing function is applied immediately outside the inlet, a_o is not a true ocean tidal amplitude unless entrance bar losses are negligible. Otherwise, bar losses need to be accommodated in the analysis (Nielsen and Gordon 1980).

Many limitations of the generalized analytical modelling approach are documented in Bruun (1978). A significant limitation of the approach is that it is an inaccurate predictive tool in situations where significant perturbations are to be made to the inlet impedance, such as those associated with the construction of entrance jetties (Nielsen and Gordon 1980). This is because the estuary stability relationship (entrance channel velocity versus cross-sectional area) cannot be constructed accurately if the inlet and channel head losses vary significantly from the "natural" calibrating condition. Difficulties can arise also where the dimensions of an entrance channel vary significantly along its length, where there are multiple channels or where significant abrupt head-losses are encountered at severe bends, constrictions and bridge crossings; that is, where the basic assumption of a short regular entrance channel in unconsolidated sediment is violated. Further difficulties can arise also where changes are introduced to channel conveyance through rock armoring and groin construction. Finally, the method can assess only the potential for change; it cannot, of itself, indicate whether or not an estuary is in a process of change.

Nevertheless, the development of these empirical and generalized analytical formulations for estuary stability presents a sound basis for an understanding of inlet tidal hydraulics. Of particular note is that the spring tidal prism is a common and most important parameter to all of these stability criteria. This can be defined accurately, objectively and consistently from the tidal constituents.

10.2.5 Marine Ecology

Estuarine macrophytes (saltmarsh, mangrove, seagrass) grow within the sub-tidal and inter-tidal zones where their presence is affected by physical, chemical and hydrodynamic conditions (Kailola 1993). Estuarine macrophytes are fundamental building blocks of estuarine ecology as they create new tissue from sunlight and, hence, initiate estuarine food chains, they provide habitat for fish, crustaceans and molluscs in which to shelter from predators as well as forage for food. Most of the commercially and recreationally important fish species on Australia's eastern seaboard are dependent at some stage of their life cycle on estuarine habitats.

A generalized schematic diagram of the distribution of macrophytes around the fringes within tidal estuaries is presented in Fig. 10.8. Rising tidal planes within estuaries are likely to impact these fringing ecologies. Saltmarsh habitat is very sensitive to tidal levels and increasing levels are likely to result in excessive flooding and loss of saltmarsh habitat, which would then be colonized by mangrove species.

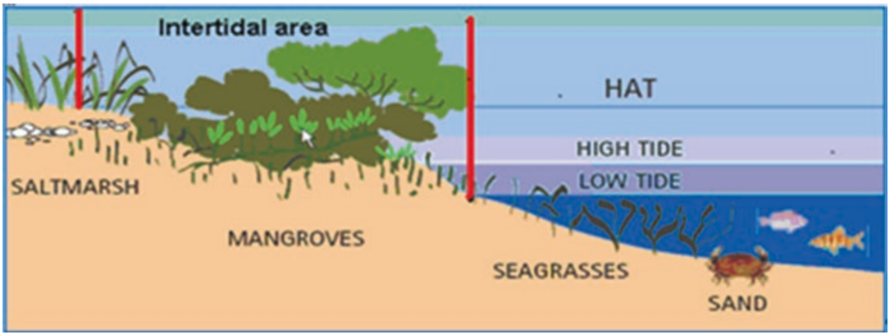

Fig. 10.8 Common fringing habitat zones in an estuary (Kailola 1993). HAT is highest astronomical tide. Saltmarsh is sensitive particularly to tidal levels. Various seagrass species are sensitive to water depth

Alternatively, if the topography is favorable, saltmarsh communities may migrate upstream into tributaries as may mangroves.

10.3 Impacts on Estuarine Hydraulics

10.3.1 Introduction

An estuary experiencing an unstable scouring or shoaling mode will have a tidal prism, channel cross-sectional area and hydraulic radius that are all varying with time, even if the banks are fixed by training walls, in which case the bed of the channel is the variable parameter for cross section and hydraulic radius. While it may be possible using the Escoffier inlet stability theory to identify that an estuary may be prone to such instability, it is not possible always to predict if an estuary is unstable, what the ultimate stable configuration may be and when that may be reached.

Should the amplitudes of the major tidal constituents measured within an estuary be increasing, this would be an indicator of increasing tidal prism. Similarly, should the phase lags of the major tidal constituents be decreasing, this would be an indicator also that the tidal wave is penetrating the estuary more efficiently, implying increasing tidal prism or improved efficiency of the tidal conveyance in the entrance channel. Conversely, should the amplitudes of the major tidal constituents be decreasing and/or their phase lags increasing, this would be an indicator of decreasing tidal prism and, possibly, an unstable shoaling mode. The rates of change of these parameters may be used to estimate the time it may take to reach a stable state.

In the following, the time histories of the variations in the spring tidal constituents of three estuaries with jettied entrances in New South Wales, Australia are examined; two are relatively large and the other an order of magnitude smaller. The

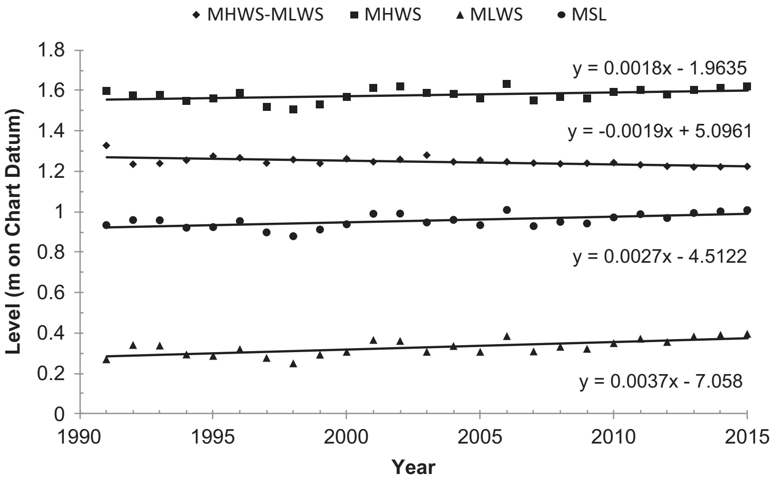

Fig. 10.9 History of ocean tidal planes at Sydney 1990–2015 (Data from NSW Public Works Manly Hydraulics Laboratory). These data are relevant to compare with changes to the tidal planes examined in the estuaries of interest exemplified herein. For the 25 years since 1990 the mean sea level (MSL) has increased at an average rate of 2.7 mm/a whereas the spring tidal range (MHWS-MLWS) has decreased at an average rate of 1.9 mm/a as measured in Sydney Harbor

data have been used to develop Escoffier analyses for each and to investigate if the time to reach stable configurations can be estimated.

10.3.2 Ocean Tidal Constituents Control Data

If the tidal constituents in the ocean vary over time then it could be expected that those within estuaries would follow suit. Therefore it is important to normalize the basis for tidal analysis. For the NSW estuaries exemplified herein, the selected control data set for the histories of amplitude and phase of the major ocean spring tidal constituents was that represented by the Sydney Harbor tidal data, a large, relatively deep, drowned river valley estuary with a reliable and stable tidal gauging station. The baseline information from the Sydney gauge is presented in Figs. 10.9 and 10.10.

Figure 10.9 shows that for the period 1990–2015, mean sea level (*MSL*) rose over the period at an average rate of 2.7 mm/a and, while mean low water springs (*MLWS*) rose at a higher rate of 3.7 mm/a, mean high water springs (*MHWS*) rose at a slower rate of 1.8 mm/a, resulting in an average decrease in the ocean's spring tidal range of 1.9 mm/a over that period. This reduction in ocean spring tidal range is an important consideration when comparing it with the increases in spring tidal range of the estuaries exemplified herein.

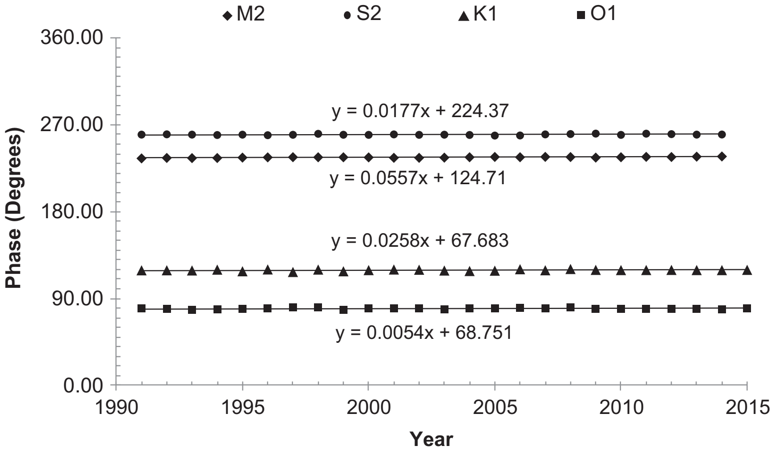

Fig. 10.10 History of major ocean tidal constituent phases at Sydney 1990–2015 (Data from NSW Public Works Manly Hydraulics Laboratory). For the 25 years since 1990 the phases of the major ocean tidal constituents as measured in Sydney Harbour have all increased

Figure 10.10 shows that over that period the phase of all of the major tidal constituents increased at average rates varying from 0.005 to 0.05°/a, with the major tidal constituent, *M2*, increasing at an average rate of 0.017°/a.

As the ocean tidal signature is the main forcing agent of the estuarine hydraulics, these data form an important control data set with which the tidal signatures within the case study bays are compared.

10.3.3 Wallis Lake

The Wallis Lake estuary on the mid-north coast of NSW is a complex system of bays and rivers with inter-connecting channels that separate the towns of Tuncurry and Forster, located adjacent to the ocean inlet (Fig. 10.11). The bay has a plan area of some 100 km^2 (Fig. 10.12) and has a mean spring tidal range of around 0.15 m. On very large spring ebb tides near solstices, peak velocities through the entrance channel between the jetties, which are 100 m wide, exceed 3 m/s, presenting challenging conditions for recreational boaters. The tidal prism on these higher spring tides has a discharge in the order of the annual average flood (Nielsen and Gordon 1980).

Prior to any training wall and jetty construction, the ocean entrance to the Wallis Lake estuary was choked with littoral drift (Fig. 10.13). The ruling depth on the ocean bar was 0.6 m (2 ft) on low waters. The estuary was reported to have been closed for many years in around 1831 (Pennington 1877). Over the time of recorded history only fresh water floods kept the inlet open until jetty construction modified the entrance.

Fig. 10.11 The Wallis Lake Estuary entrance at Forster/Tuncurry, looking west (Photo courtesy NSW Government). Tidal communication between the ocean and the bay (Wallis Lake) is effected through a myriad of channels. The piling foundations for the road bridge in the foreground have been compromised severely by channel scour

The southern (Forster) jetty was constructed in 1898. While it improved the tidal conveyance somewhat, often entrance navigability was compromised still so, in 1966, the southern jetty was extended some 90 m and a 460 m long jetty was constructed on the northern (Tuncurry) side.

Dramatic changes to the Wallis Lake estuary ensued. These were attributed to the increased hydraulic conveyance of the inlet occasioned by the construction of the northern jetty (Nielsen and Gordon 1980). Since 1990, when consistent and reliable data became available, the bay's spring tidal range, as measured at Tiona (Fig. 10.12), has continued to increase at a rate of 1.8 mm/a ($R^2 = 0.89$) and, by the year 2015, the ratio of the bay range to that of the ocean had risen from 0.09 to 0.14, a 55% increase, at a rate of 0.0016/a ($R^2 = 0.92$) and showing little signs of abating (Fig. 10.14).

The history of the major spring tidal constituent ($M2$) phase lag (bay phase minus ocean phase) from 1990, presented in Fig. 10.15, shows a weak decreasing trend indicating increasing efficiency in tidal wave penetration of the estuary.

An Escoffier Diagram (Fig. 10.16) was constructed for Wallis Lake following the method of Czerniak (1978), assuming the most constricted width of the inlet channel (Seabergh and Kraus 1997) of 100 m at the entrance with a channel depth of 5 m. This gave an effective friction length for the channel of 3,400 m. It was assumed also that the tidal discharge curve was sinusoidal with period 12.4 h to enable the

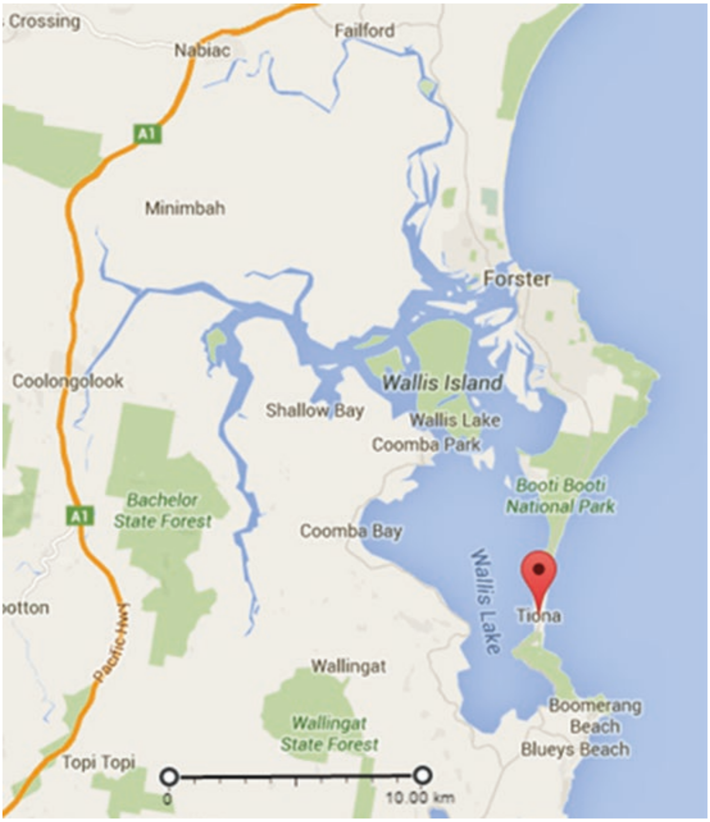

Fig. 10.12 A plan of the Wallis Lake estuary showing the ocean entrance at Forster and the location of the bay (Wallis Lake) tide gauge at Tiona (Courtesy Google maps). To note are the rivers that kept the natural entrance open by debauching flood waters. Historically, the inlet had closed during periods of prolonged drought

stable equilibrium flow velocities to be determined from the O'Brien (1969) and Jarrett (1976) prism-area relationships.

The Escoffier Diagram (Fig. 10.16) indicated that the estuary channel was in an unstable scouring mode. In accordance with O'Brien (1931), an equilibrium condition would be reached when the flow area reached some 6,000 m^2, a tenfold increase. By that stage the bay-to-ocean spring tidal range ratio (a_B/a_O) would have reached 1.0, increasing from 0.14. At an average rate of increase for a_B/a_O of around 0.0016/a (Fig. 10.14), it would take some 540 years for the equilibrium cross-sectional area to be reached. However, taking the approach of Mota Oliviera (1970), where stability could be reached when K reached a value between 0.6 and 0.8, a_B/a_O would have a value of between 0.65 and 0.78 and it would take some 300 to 400 years for the equilibrium cross-sectional area to be reached. It is noted that as scour progresses, the Escoffier diagram indicates that the rate of change may increase, which would reduce the time estimated to reach equilibrium.

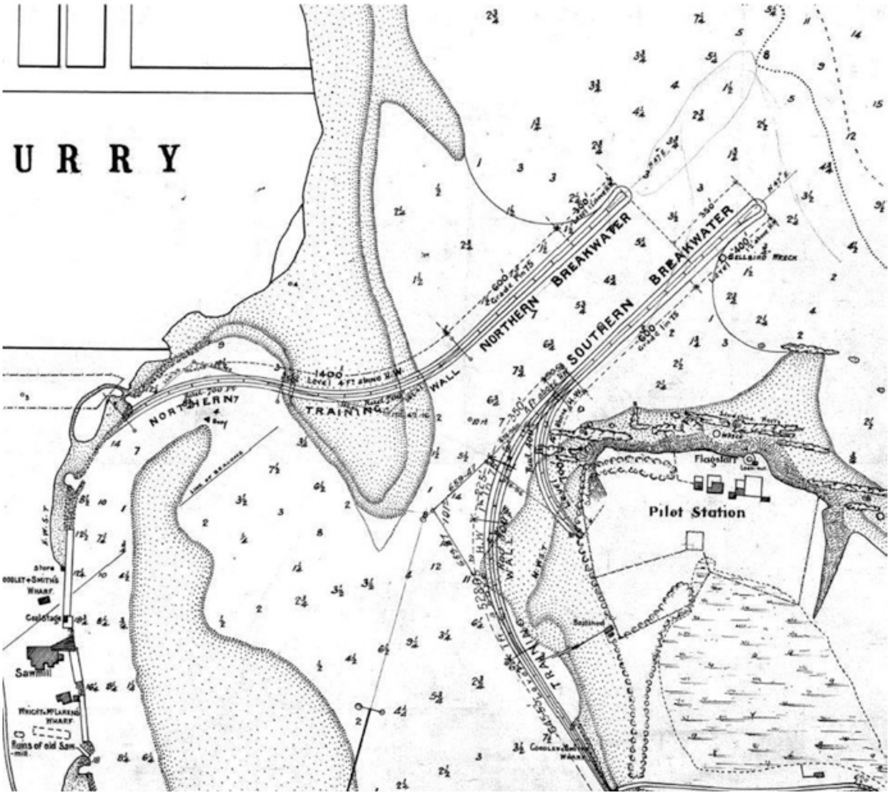

Fig. 10.13 Wallis Lake inlet prior to training wall and jetty (breakwater) construction and showing proposed jetties (water depths given in feet to Indian Springs Low Water – ISLW; Survey Plan courtesy of NSW Government)

These changes are difficult to contemplate as they assume that the jetty lined entrance can scour sufficiently to accommodate the additional flow. At present the field data give no hint of stabilization and the changes in tidal range and channel scour continue unabated.

10.3.4 Lake Macquarie

Lake Macquarie is on the NSW central coast some 25 km south of Newcastle (Fig. 10.17). The bay has a plan area of around 110 km^2 and is connected to the ocean by an irregular 4,700 m long Swansea Channel (Fig. 10.18) of average depth around 2.0 m, average width around 400 m and with various forms of rock bank protection along much of its length. It has dual highway bridges crossing the entrance channel that constrain the tidal flow. The bay spring tidal range is around

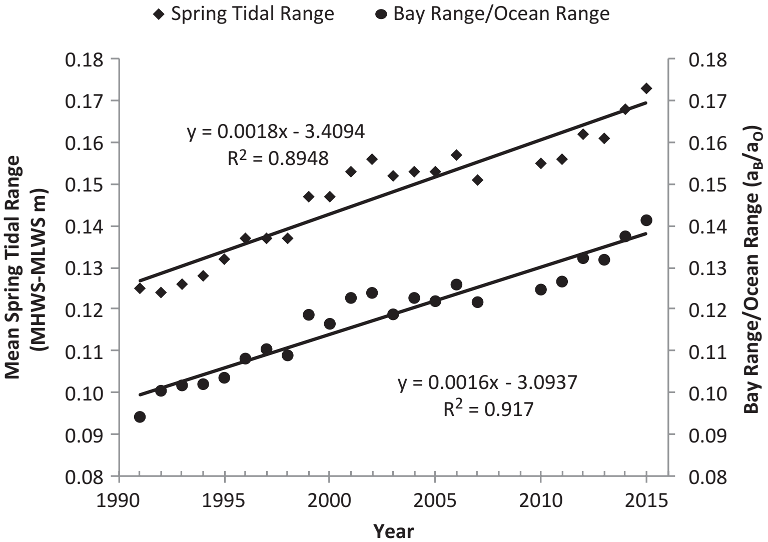

Fig. 10.14. Change history of the mean spring tidal range in Wallis Lake and its ratio to the ocean range. The trends are strong (R^2 = 0.9) showing no signs of abating. With the bay area some 100 km^2, the increase in the spring tidal range of 1.8 mm/a equilibrates to an annual increase in the spring tidal prism of 180,000 m^3

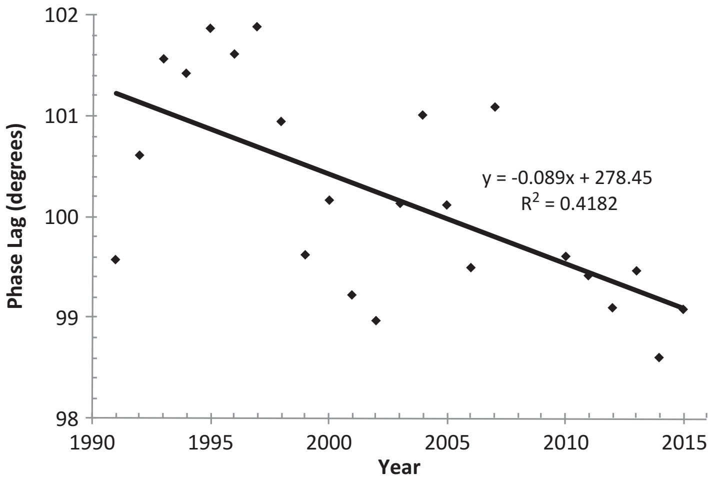

Fig. 10.15 Change history of the phase lag (bay phase minus the ocean phase) of the major spring tidal constituent (M2) in Wallis Lake. There is a weak reducing trend (R^2 = 0.4) indicating high tide arriving sooner in the bay

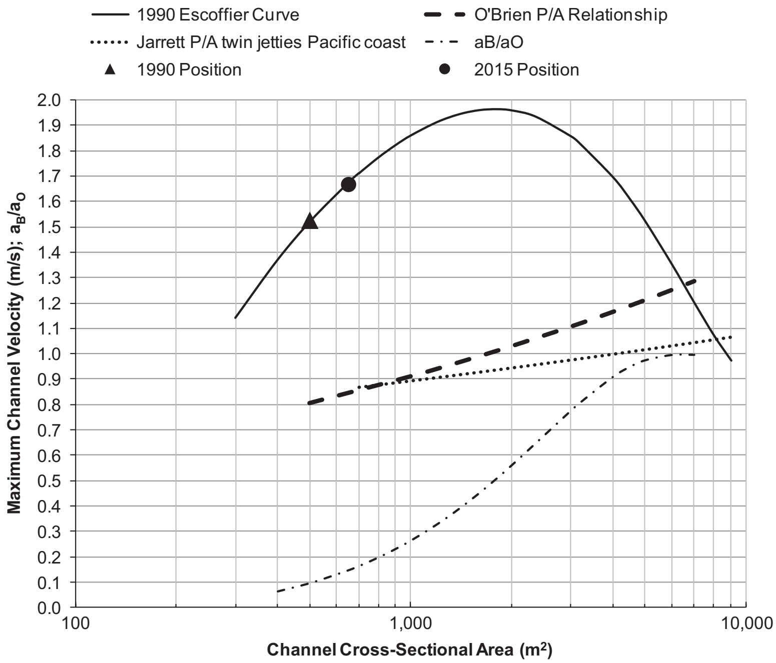

Fig. 10.16 Escoffier diagram for Wallis Lake indicating an unstable scouring mode. The O'Brien and Jarrett *P*/*A* relationships indicate a tenfold increase in channel area for the estuary to reach equilibrium

0.13 m and the maximum spring tide channel velocities are around 1 m/s (Watterson 2010).

Jetties were constructed from 1878 to 1887 (MHL 1994). Utilizing the available tidal records since 1990, the spring tidal range in the bay has increased steadily at an average rate of 1.7 mm/a (R^2 = 0.96; Fig. 10.19) with the bay-to-ocean spring tidal range ratio increasing steadily at a rate of 0.0015/a (R^2 = 0.94).

Figure 10.20 shows a decreasing trend in the phase lag of the major spring tide constituent of 0.5°/a. These data are strong indicators of an unstable scouring mode. If it is assumed that the tidal range grew steadily between 1887 and 1990 to reach 0.09 m from an initial value of zero, assuming the inlet being closed in 1887, the average rate over that period could not have been greater than 0.9 mm/a, suggesting that the current rate represents an increasing trend.

An Escoffier Diagram for Lake Macquarie is presented in Fig. 10.21, which indicated an unstable scouring mode. Without geomorphologic and/or anthropogenic constraints, based on the O'Brien (1931) equilibrium cross section, the channel area could continue to scour, increasing some fivefold, allowing for the tidal range in the bay to reach around 80% of the full ocean tidal range. At current rates of change, this could take a further some 450 years. However, taking the approach of Mota Oliviera (1970), it would take 350 to 400 years for the equilibrium cross-sectional

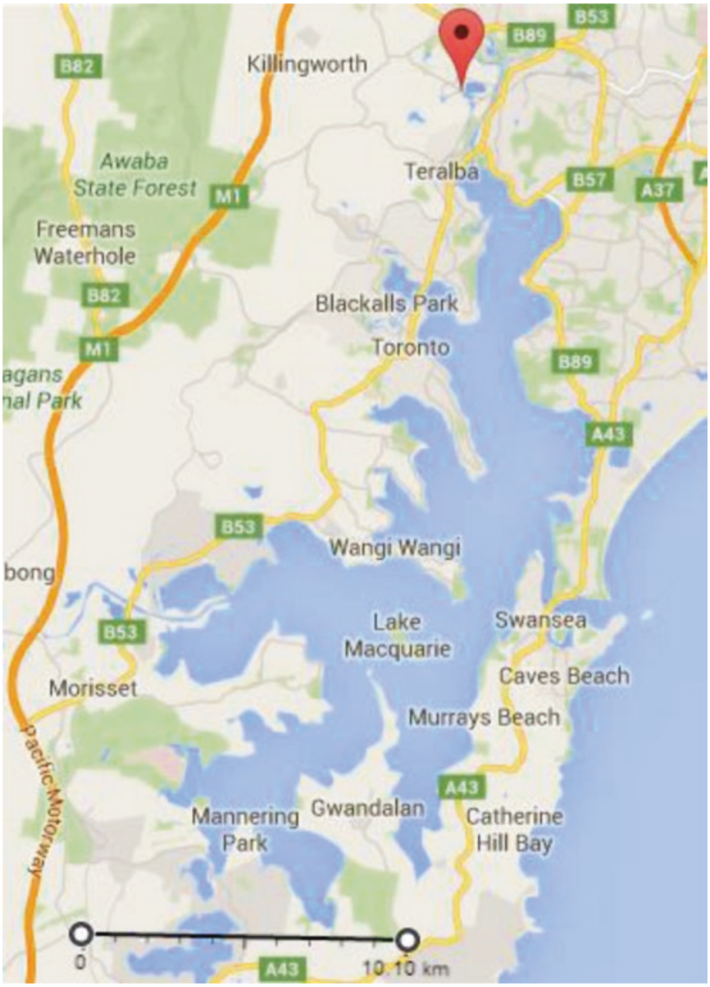

Fig. 10.17 Plan of Lake Macquarie (Courtesy Google maps) showing the location of the bay tide gauge. The ocean inlet is at Swansea. The plan area of the bay is 110 km^2

area to be reached. However, other factors such as a coal seam in the entrance channel and the constraints posed by the two bridges will produce an increasing relative roughness as velocities increase and, hence, the final equilibrium of the system may occur within a shorter timeframe.

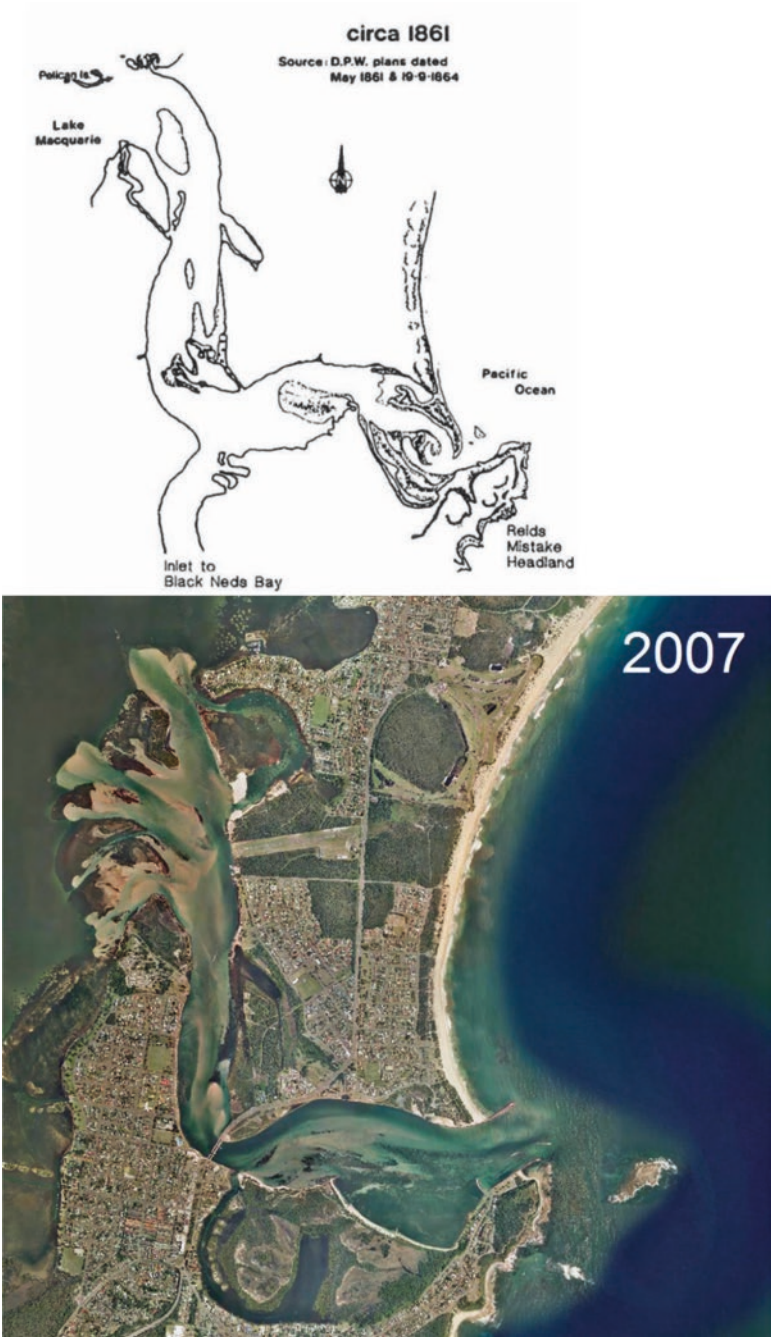

Fig. 10.18 Lake Macquarie entrance, Swansea Channel; *top* 1861; *bottom* 2015 (Watterson 2010). In 1861 the inlet virtually was closed with littoral drift choking the entrance. The beach on the southern side of the channel near the entrance is shown to have eroded significantly. There has been considerable extension of the flood tide delta northward into the bay

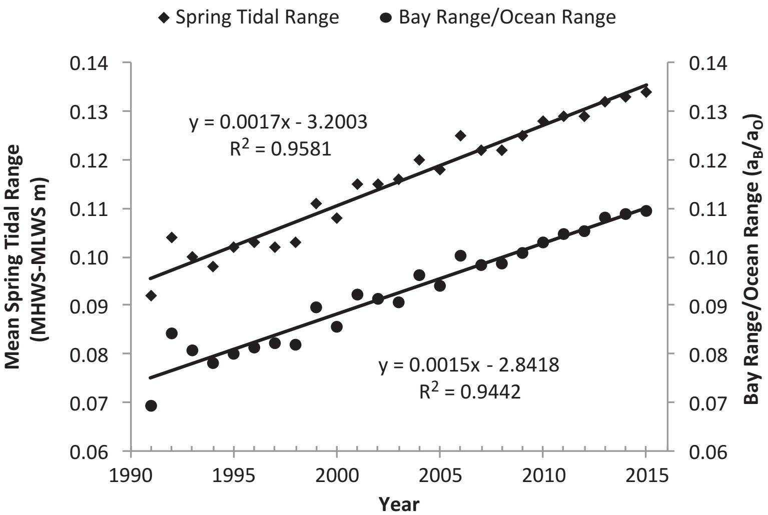

Fig. 10.19 Change history of the mean spring tidal range in Lake Macquarie and its ratio to the ocean range. The trends are strong ($R^2 > 0.9$) showing no signs of abating. With the bay area some 110 km^2, the increase in the spring tidal prism of 1.7 mm/a equilibrates to an annual increase in the spring tidal prism of 190,000 m^3

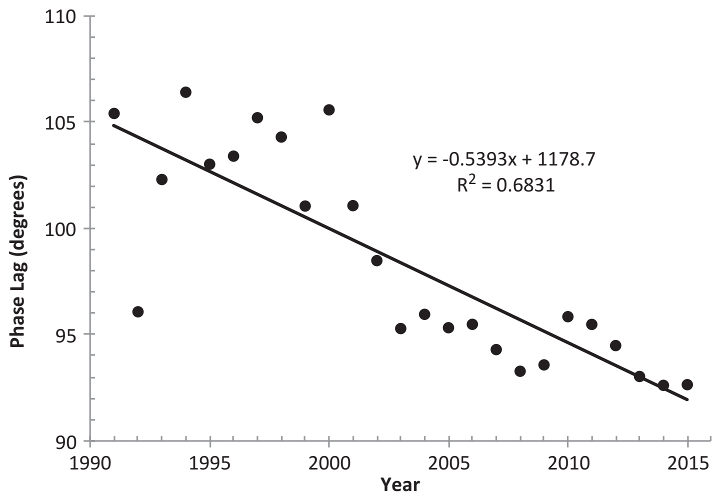

Fig. 10.20 Change history of the phase lag (bay phase minus the ocean phase) of the major spring tidal constituent (*M2*) in Lake Macquarie. There is a clear reducing trend ($R^2 = 0.7$) indicating high tide progressively arriving sooner in the bay

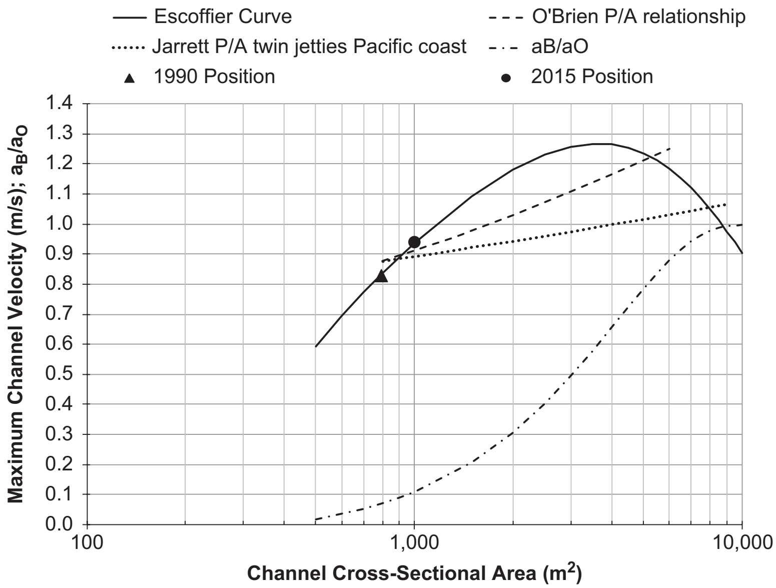

Fig. 10.21 Escoffier Diagram for Lake Macquarie indicating an unstable scouring mode. The O'Brien and Jarrett *P*/*A* relationships indicate a sixfold increase in channel area for the estuary to reach equilibrium

Fig. 10.22 Lake Wagonga, Narooma NSW (Courtesy Google earth). The plan area of the bay is 7 km^2. The jetties intersect the surf zone, the channel appears deeply scoured and there are significant flood tide sand deltas entering the bay

10.3.5 Lake Wagonga

The Lake Wagonga estuary is situated at Narooma on the NSW south coast (Fig. 10.22). Twin entrance jetties were constructed in 1976–1978, primarily to improve entrance navigability for the commercial fishing fleet (MHL 1994). The estuary comprises a steep-sided bay of area around 7 km^2 (MHL 2001); an order of magnitude smaller than Wallis Lake and Lake Macquarie. A regular 3,250 m long entrance channel has a depth around 2.0 m, an average width of around 100 m and has intertidal training walls constructed of rock rubble. The spring tidal range in the bay is around 0.7 m and on the higher spring ebb tides the peak channel velocities approach 2 m/s (MHL 2001).

Regular tidal stage measurements are available from 1997. As shown in Fig. 10.23, the spring tidal range has increased steadily over the period of record at an average rate of 3.0 mm/a ($R^2 = 0.84$) and the bay-to-ocean spring range ratio has been increasing annually at an average rate of around 0.0033/a ($R^2 = 0.91$). The change history of the major spring tidal constituent phase lag is in Fig. 10.24, indicating a steady reduction of around 0.2°/a ($R^2 = 0.80$).

The regular features of this estuary allow for a considered derivation of an Escoffier Diagram, which is presented in Fig. 10.25. The Escoffier Diagram confirms the trend in the field data, indicating that the estuary channel is in an unstable scouring mode.

Without limitations, such as the influence and behavior of the channel training walls, the indications are that the channel could scour to more than double its present cross-sectional area, leading to the bay achieving almost full ocean tidal range. At current rates and without constraint, full ocean tidal range in the bay is predicted to be achieved within 120 years. However, taking the approach of Mota Oliviera (1970), at an average rate of increase for a_B/a_O of around 0.0033/a (Fig. 10.23), it would take some 20 to 50 years, rather than 120 years, for the equilibrium cross-sectional area to be reached.

The trends, based on the relatively short 18 years' record, currently are linear with no indications of any decreasing rates of change to the amplitudes or phase lags of the major spring tidal constituents. However, as indicated by the Escoffier Diagram, once the critical flow area has been exceeded, the channel velocities are predicted to decline and the rate of change of the tidal range in the bay also may decline, extending the time required to reach stability.

10.4 Impacts on Coastal Processes

10.4.1 Wallis Lake

Prior to training wall and jetty construction, the entrances exemplified herein were plagued with shifting sand shoals and, occasionally, were closed to navigation. Jetty construction intersected the surf zone bars and eliminated the marginal flood tide

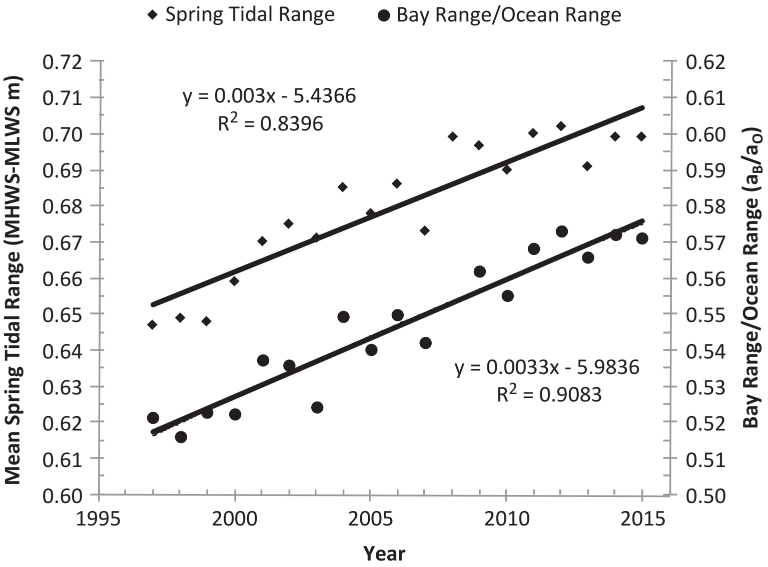

Fig. 10.23 Change history of the mean spring tidal range in Lake Wagonga and its ratio to the ocean range. The trends are strong ($R^2 = 0.8–0.9$) showing no signs of abating. With a plan area of 7 km^2, the rate of increase in the spring tidal prism of 3 mm/a equilibrates to an annual increase in the spring tidal prism of 20,000 m^3

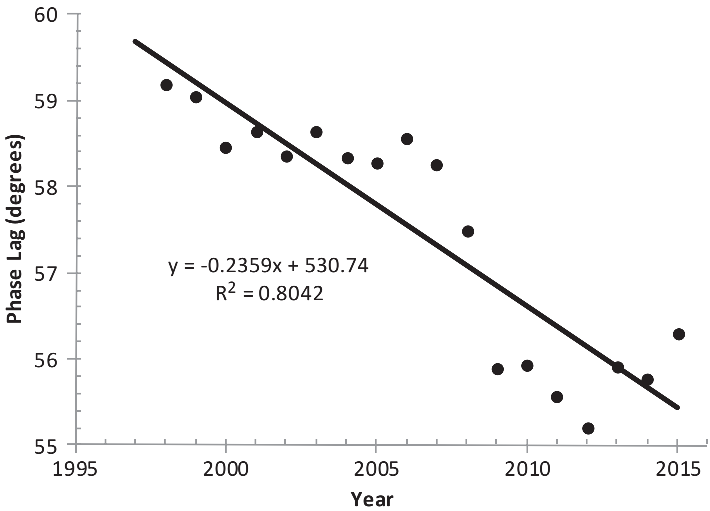

Fig. 10.24 Change history of the phase lag (bay phase minus the ocean phase) of the major spring tidal constituent (M2) in Lake Wagonga. There is a strong reducing trend ($R^2 = 0.8$) indicating high tide arriving sooner in the bay

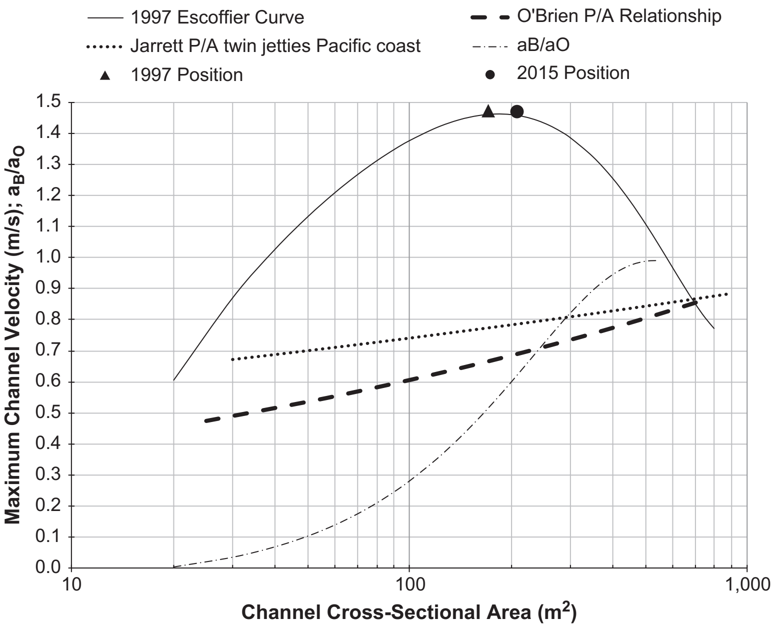

Fig. 10.25 Escoffier Diagram for Lake Wagonga indicating an unstable scouring mode. The O'Brien and Jarrett *P/A* relationships indicate a threefold increase in channel area for the estuary to reach equilibrium. The rate of change of the bay spring tide amplitude is likely to decrease as the equilibrium area is approached

channels and shoals. In each case, jetty construction has occasioned scouring of channels and changes to tidal planes, the reasons for which are exemplified in the case study of the Wallis Lake entrance improvements (after Nielsen and Gordon 1980).

Prior to the construction of the northern (Tuncurry) jetty at Wallis Lake entrance, the asymmetrical nature of the ebb jet expansion resulted in the development of shallow ocean shoals. A large ebb tide separation eddy had developed on the northern side of the inlet resulting in an inlet-directed current through the northern marginal flood tide channel during ebb tide (Fig. 10.26). This opposing current carried sediment into the entrance channel on ebb tides and induced a significant head loss to the ebb tide flow. The effect was enhanced during the flood tides as sand, which was entrained into the flow by wave action on the asymmetric entrance bar and beach on the northern (Tuncurry) side of the entrance, was transported into the entrance channel. This encouraged the southward growth of the spit on the non-jetty side, thereby tending to close the inlet.

The construction of the northern jetty intersected the marginal flood tide channel extending southwards along the beach. As indicated in Fig. 10.27, this eliminated the ebb tide separation eddy and reduced significantly the rate of littoral drift transport into the entrance channel, which would have had a major impact on keeping the inlet open (Bruun 1978). Further, this focused all of the tidal flows onto deepening

Wallis Lake Inlet 1952 (Photograph courtesy NSW Government)

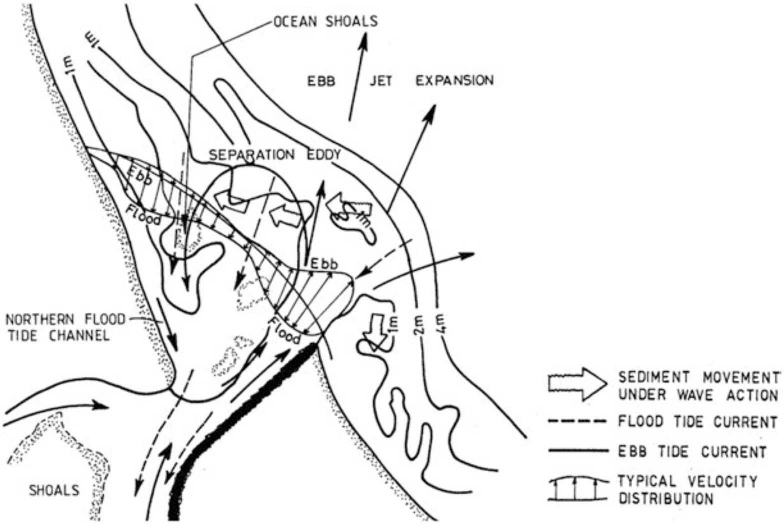

Fig. 10.26 Schematic representation of the hydrodynamics at a typical asymmetrical entrance (Nielsen and Gordon 1980). The entrance asymmetry created by a single jetty induces a separation eddy on the ebb tide discharge creating an inlet directed current along the beach, which constricts the ebb tide flow and contributes littoral drift to the entrance channel. Flood tide flow across the surf zone bars contributes a considerable amount of suspended sediment to the entrance channel. Wallis Lake Inlet 1952 (Photograph courtesy NSW Government)

Wallis Lake Inlet 1974 (Photograph courtesy NSW Government)

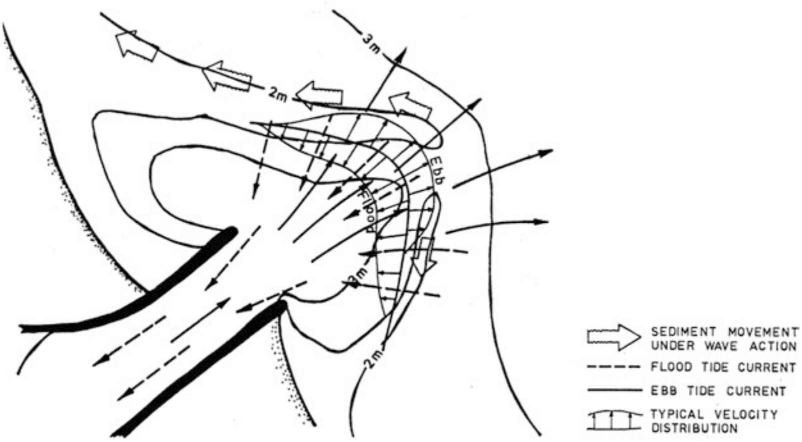

Fig. 10.27 Schematic representation of the hydrodynamics at a typical symmetrical entrance (Nielsen and Gordon 1980). The symmetrical jettied entrance configuration has improved hydraulic conveyance and reduces the sediment feed into the entrance channel. Wallis Lake Inlet 1974 (Photograph courtesy NSW Government)

the entrance bar and it eliminated the processes maintaining the shallow marginal ocean shoals. The resulting symmetrical and deeper entrance bar and channel reduced significantly the head loss for both the flood and ebb tide flows. The reduction in head loss across the entrance bar has improved the hydraulic efficiency of the entrance and has enhanced tidal propagation into the estuary. This has increased the

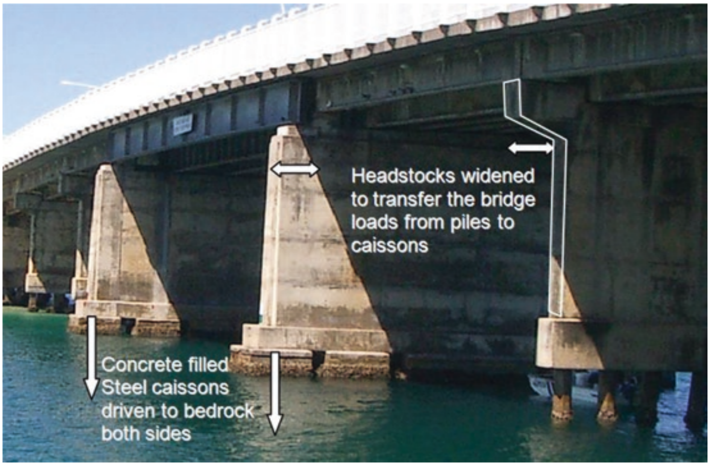

Fig. 10.28 Underpinning of the Forster-Tuncurry Bridge piers following channel scour. The channel piers had subsided almost 0.5 m. The headstocks were underpinned with steel caissons driven to bedrock and the headstocks were widened to transfer the bridge loads to them

velocities and sand transporting capabilities of the tidal streams in the estuary channels, leading to their scour and to the deposition of sand on the flood tide deltas leading into the bay as well as exporting sand onto the adjacent ocean beaches, thereby nourishing them and adding sand to their sediment budget. The scouring and progressive deepening of the entrance bar has increased further the effective tidal forcing, creating a positive feedback loop which, in turn, deepens the entrance bar, leading to more channel scour.

Interestingly, the study linking jetty construction to channel scour was initiated by an investigation that, by happenstance, used a main road bridge across the entrance channel as a measuring station for flow velocities and cross-sectional areas. When the channel cross-section was plotted on the works-as-executed drawing of the bridge it was found that the channel had scoured almost to the toes of the bridge piles, which had been designed as friction piles in sand, rendering the bridge potentially to become unstable. Subsequent surveys showed that the bridge pile supports in the main Forster Channel had settled some 475 mm and, at significant expense and disruption, led to underpinning the headstocks with steel caissons driven deep to bedrock (Fig. 10.28). Fortunately, the roadway deck had been simply supported, which allowed for considerable tolerance in settlement and for the subsequent jacking up of the roadway bridge deck to its original levels following underpinning.

10.4.2 Lake Macquarie

Since the jetties were constructed at Lake Macquarie, the Swansea Channel has been scouring and its foreshores have been eroding, necessitating groin and revetment construction, some of which is collapsing. Detailed bathymetric surveys undertaken in 1996 and 2008 (Fig. 10.29) indicated that the channel was scouring over that period at an average rate of around 25,000 m^3/a, with the bay's flood tide

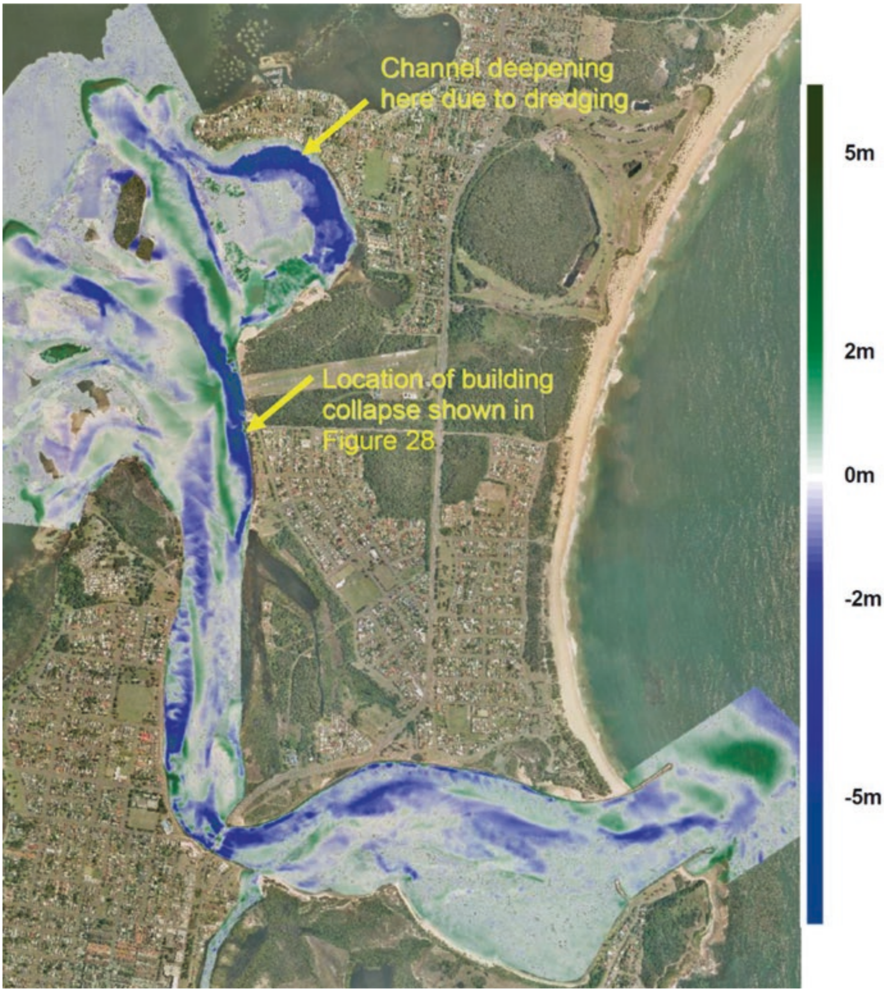

Fig. 10.29 Bed level changes in Swansea Channel 1996–2008 (Watterson 2010; courtesy Lake Macquarie Council). Up to 5 m of sand accretion has been measured on the leading edges of the flood tide deltas entering the bay and the ebb tide delta at the ocean entrance. Sand has come from channel deepening and erosion of the beach on the south side of the channel near the ocean entrance

Fig. 10.30 Collapse of Swansea Channel foreshore building comprising restaurant, offices and residences on 8 February 2016 (Photo: Fire and Rescue NSW). The location is shown in Fig. 10.29. Particularly severe erosion has occurred along a revetment constructed to protect the western end of the airstrip. The revetment has attracted the channel thalweg, which has led to further scour both upstream and downstream, undermining the pile foundations of the building

deltas growing at an average rate of around 12,000 m^3/a, the remainder being deposited on the ocean bar and, subsequently, as nourishment, onto the ocean beach (Watterson 2010). The piling foundations of the main road bridge spanning the entrance channel at Swansea village have been compromised by the channel scour, necessitating the placement of large volumes of rock scour protection, which requires frequent maintenance. In accordance with Sect. 10.2.3, it appears that a revetment constructed to protect the nearby airstrip has attracted the thalweg of the channel, increasing velocities there. This, along with the three-dimensional flow induced by the "cross-over" downstream, has induced additional scour and, hence, a deepening of the channel at that location.

The changes have been progressive over many decades, if not centuries. At Lake Macquarie, the data indicated that the rate of change may be accelerating. The acceleration of an unstable scouring mode may be caused by a sea level rise. However, that the rate may be increasing is predicted by the Escoffier Curve. The implication is for continued scouring of Swansea Channel. Progressively, this is threatening other assets as shown by the collapse of the foreshore marina building due to undermining of the pile foundations (Fig. 10.30), perpetual scour to the Swansea Bridge foundations, progressive failure of the marginal rock rubble revetments and groins. That this is occurring still some 125 years after the major perturbation of jetty construction indicates that the time for large estuaries to reach stabilization can be considerable and in the order of centuries.

10.5 Impacts on Marine Ecology

10.5.1 Wallis Lake

In Wallis Lake, trends in seagrass colonization between 1988 and 2002 have been mapped using satellite imagery (Fig. 10.31). The seagrass trend in Wallis Lake seems to be an overall decline in the shallow water seagrass *Zostera* with deeper water *Posidonia*, *Ruppia* and *Halophila* seemingly stable with no gross changes in the 14 year period (Dekker et al. 2003).

Of particular note is the entire loss of *Zostera* in the channels between the entrance to Wallis Lake and the ocean. We have attributed this to channel scour and deepening. Here there was some gain of *Posidonia*, which colonizes deeper waters. There was a slight increase in *Posidonia* also in the northern channel areas where we would predict channel deepening. Large areas of *Zostera* were lost also within Wallis Lake but no reasons for this were given in Dekker et al. (2003).

10.5.2 Lake Wagonga

Long term changes to the distribution of aquatic flora have been identified and mapped in the Wagonga estuary (Burrell 2012; Duchatel et al. 2014). These studies have shown that Lake Wagonga has experienced a significant decrease in the distribution of macrophytes over the 25 year period between mapping campaigns (Duchatel et al. 2014). Figure 10.32 shows seagrass mapping in the entrance

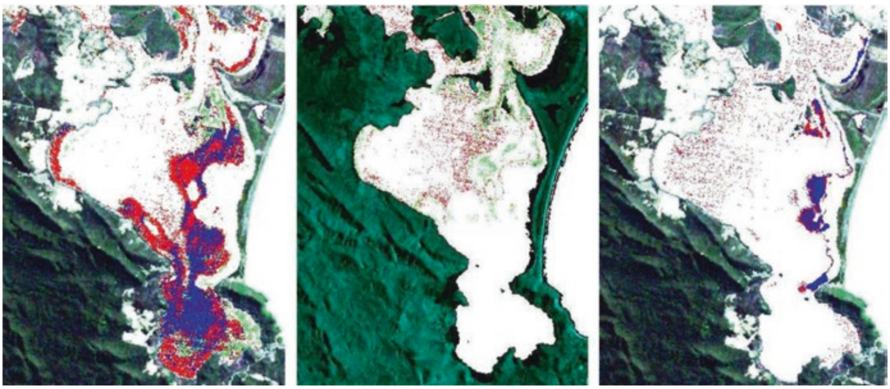

Fig. 10.31 Changes in Wallis Lake to aquatic flora (from *left* to *right*) *Zostera*, *Posidonia* and *Ruppia*/*Halophila* from 1988 to 2002 with *red* = loss, *green* = gain and *blue* = no change; *white* pixels within the lake indicate a class not identified as seagrass (Dekker et al. 2003). Channel scour between the lake and ocean has resulted in loss of *Zostera*. Some of the deepened areas have been colonized by *Posidonia*

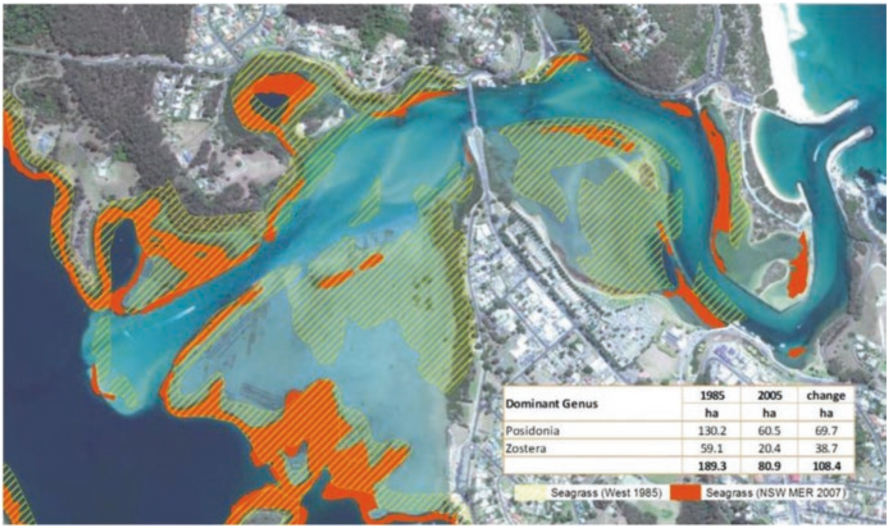

Fig. 10.32 Change in the extent of seagrass at the entrance to Lake Wagonga 1979–2005 (Duchatel et al. 2014). Loss of seagrass in the channel has been attributed to channel scour whereas on the flood tide delta it has been attributed to sand inundation

channel, where changes to seagrass distribution and abundance has occurred as a result of increased tidal velocities causing channel bed scour and accreting flood tide deltas causing seagrass smothering. The data indicated that the total abundance of all seagrass throughout the estuary decreased by 57% (from 189.3 to 80.9 ha). Approximately 64% of this loss was *Posidonia* and the remaining 36% predominantly *Zostera*. In this case some of the Posidonia beds were covered as the delta spread into the bay. Over 33% of the total seagrass loss occurred within the channel with approximately 72% of that being the loss of *Zostera.*

Since 1957, throughout most of the Wagonga estuary the mangrove communities were found to be either stable or expanding with expansion occurring through incursion up-slope into the saltmarsh communities, laterally along the foreshore and down-slope onto prograding deltas and sandbars (Burrell 2012). The data from Burrell (2012) showed that the rate of change had increased threefold since the jetties were constructed in 1978 (Fig. 10.33) with the greatest increase occurring in the upper part of the estuary.

Since 1957, saltmarsh has decreased throughout the estuary. Foreshore reclamation for a caravan park, urbanization and draining of wetlands has accounted for the most significant loss of saltmarsh within the Inlet (Burrell 2012). Taking into account those major impacts, the data from Burrell (2012) indicated that the natural rate of saltmarsh loss increased threefold since the jetties were constructed (Fig. 10.34).

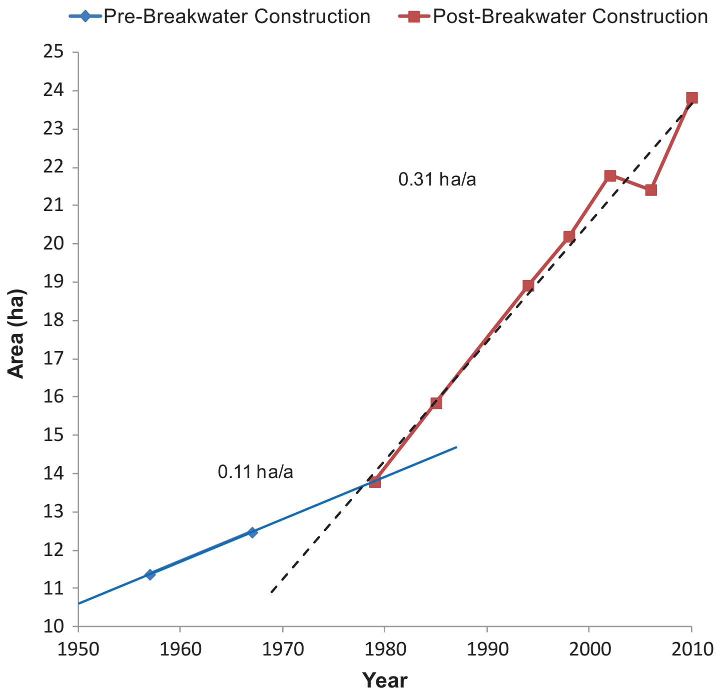

Fig. 10.33 Change in area of mangrove and natural rates of change at Lake Wagonga inlet prior to and after entrance jetty construction (Data from Burrell 2012). Marked differences in the rates of change coincide with jetty construction in 1978

10.6 Implications

The construction of training walls and jetties at the ocean entrances to coastal bays and lagoons has the potential to alter fundamentally estuarine hydraulics, inducing changes to channel flows, morphology and tidal planes. While in the case studies exemplified herein these works have improved flood conveyance and inlet navigability considerably, as intended, they have induced channel scour compromising roadway bridge foundations and bank stability. The subsequent requirements to protect foreshore assets with further training walls and revetments have exacerbated the scouring processes with additional unintended adverse impacts. Implications have included the continual maintenance of scour protection to prevent further loss of foreshore assets, including buildings, and scour protection for and underpinning of bridge pylons. The prognosis is that it may take decades to centuries for these estuaries to attain ultimate inlet stability, implying long term maintenance costs.

Such major long-term changes to estuarine hydrodynamics have significant potential to alter marine ecologies. The changes to the distribution of aquatic flora within the bays of Wallis Lake and Lake Wagonga are likely to have resulted from a combination of many factors. Nevertheless, they are consistent with and as would be expected from the changes to the geomorphology, tidal planes and flows that

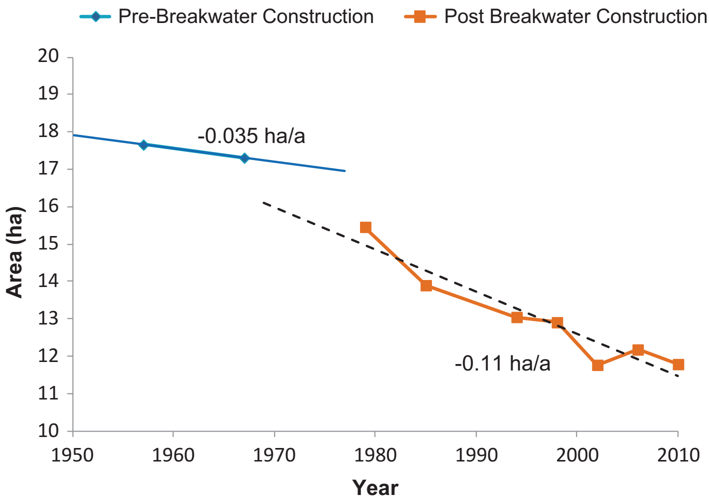

Fig. 10.34 Change in area of saltmarsh and natural rates of change at Lake Wagonga inlet prior to and after entrance jetty construction adjusted for shore-based land-use changes (Data from Burrell 2012). Marked differences in the rates of change coincide with jetty construction in 1978

have been induced by the construction of the jetties at the ocean entrances. Increasing tidal ranges induce wetland communities to migrate into tributaries and upland shorelines (Duchatel et al. 2014). However, often there will be limits to this occurring in the form of natural topographic features and anthropogenic constraints, such as shore protection works, foreshore roads, weirs, levee banks and flood gates constructed to limit salt water incursion into farm lands. The loss of these wetland areas has adverse impacts on fisheries and the overall health and water quality of estuaries.

It is vital to understand how estuary tidal planes may be modified by the construction of training walls, revetments and entrance jetties; such an understanding being the basis for designing and implementing necessary compensatory actions to secure assets and to retain wetlands in jetty protected estuaries.

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Chapter 11
Mangrove Degradation in the Sundarbans

Ashis Kr. Paul, Ratnadip Ray, Amrit Kamila, and Subrata Jana

Abstract Mangroves are most opportunistic plant to find out the favorable environment for adjustment with changing natural conditions of the coastal processes over time and space. The salt loving halophytic plants of intertidal environment are also known as a perfect biological indicator of coastal environmental changes. They act as natural buffer or bio-shield against the wind breaks, tidal waves and coastal erosion. Surface stability of younger deltaic sediments is achieved in the Sundarban coast due to the location and luxuriant growth of mangroves. Presently, Sundarban mangroves are affected by multiple ways of degradations that will produce the significant risks or vulnerabilities to the deltaic coasts occupied by land hungry people of South-Asia.

The present study reveals with an attempt to prepare a checklist for the assessment of mangrove degradations with special reference to south west Sundarban coast. The significance of mangrove conservation will be strongly supported by such degradation check lists for the coastal managers. The present work is conducted by extensive field works over a prolonged period, use of professional experiences of the authors, and application of Geospatial Techniques for database generation and management to achieve the purposes. However, factor analysis (PCA) method is also utilized to justify the ideal sequential factors those are responsible for mangrove degradations for each study area of the islands as per their regional location characters.

So far as, seven major factors and their total 56 sub-factors of mangrove degradations have been identified in southwestern parts of the Sundarban from the temporal field observations, remote sensing studies and explored historical documents in the study areas. From the present study it is revealed that hypersalinity, storm effects and sediment deposition parameters are mainly responsible for mangrove degradations in Patibania Island (Susnir Char); and for the Fredrick Island fishery development,

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land erosion and hypersalinity parameters are liable to mangrove degradations; and finally, fishery development, sediment deposition and land erosion parameters are sequentially responsible for mangrove degradations in Henry's Island.

Keywords Mangrove degradation checklist (MDC) • Sundarban • Biological indicator • Geospatial techniques • Factor analysis (PCA) • Vulnerability • Mangrove restoration

11.1 Introduction

The upper intertidal and supra-tidal zones of the Sundarban contain mangroves with traditional impenetrable jungles ruled by the natural forest guard as the Royal Bengal Tiger. Sediment accretion rates, surface stability of younger alluviums, unique ecosystem, natural bio-shield, and many other physico-chemical functions are controlled directly by the growth and distribution of mangroves in such tropical hot and humid environment. Presently it is proved that the nutrient recycling and carbon exchange between terrestrial and marine environments are strongly supported by dense growth of mangrove. They are important habitats for fish and crustaceans on which humans are dependent (Woodroffe et al. 2013). Mangroves are distributed in a sea word zone, land word zone and in a transitional zone dominated by several salt tolerant species. In Sundarban coast, brackish water community of *Nypa fruticans* and *Heritiera littoralis* often dominate the transition to inland vegetation with peaty swamp forest; but unfortunately, both the vegetation species are degraded due to salinity in the southwestern parts of the region (Fig. 11.1).

Mangroves can adapt themselves against sea-level rise by increased rate of siltation or sediment accretion and by inland migration through estuaries and other associated wetlands (Thom 1982). However, the present study shows that mangroves of the Sundarban are affected by degradations as estimated through the experiences of repeated observations over a time and space using photographic documentations and remote sensing techniques.

Several factors and sub factors are estimated as diversity of mangrove degradation (over uses by humans, development of fisheries, hypersalinity, sediment movement, storminess or storm effects, shoreline erosion, and regeneration problems) in the Sundarban coast predominantly influenced by hydrodynamic and morphodynamic changes at present. The cyclone 'Sider' in November, 2007 and cyclone 'Aila' in May, 2009 resulted in drastic changes in mangrove communities of the Sundarban, and increased frequency of storminess of the Bay of Bengal may result more devastation of mangroves on the coast in the near future.

The Sundarban without having mangrove buffers will be periodically affected by salt water floodings, rapid rate of shoreline erosion, non-availability of nutrients, costlier and unsustainable protection structures and increased human vulnerabilities

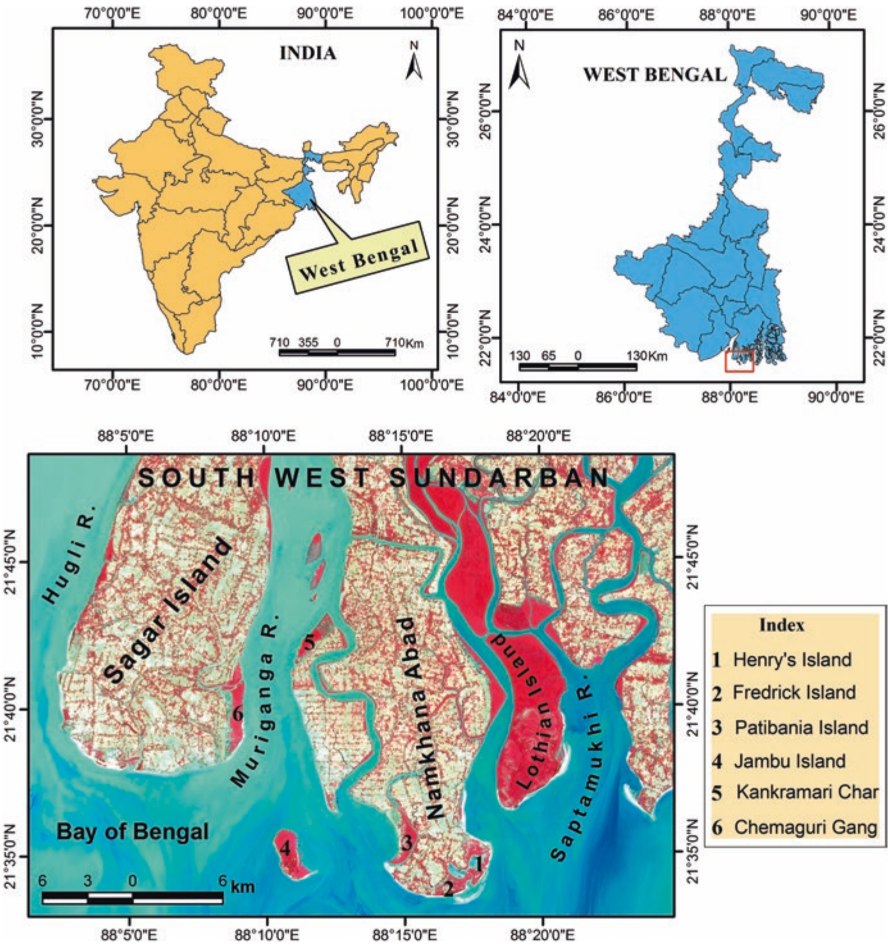

Fig. 11.1 The location of study area in coastal zones of Southwestern Sundarban

with poverty stricken people in the future. Conservation of mangroves is possible if the causes of their degradations are thoroughly estimated in the Sundarban.

Some study shows that development of plant habitat as a result of geomorphic processes in the deltaic depositional landforms evolve over time (Thom 1967), and modification of habitat conditions by the nature of geomorphic change ultimately control the ecology of mangroves in tropical humid coasts of macro tidal environment (Thom et al. 1975), and micro tidal environment of western Australia. Heathcote and Thom (1979) also views that plant habitat as landform may be affected by complexities which are subject to continual change due to the result of dynamic processes of the deltaic coasts and other marine forcing factors. The brackish water environment of Chilika lagoon, Odisha, India support a wide variety of coastal wetland habitats along the seashores of the Bay of Bengal. The coastal habitats dominated by island mangrove colonies are seriously damaged due to the magnitude of geomorphic changes resulted by the landfall of Phylin cyclone in 2013 (Paul et al. 2014).

Mangroves prevail in the fragile environment of the Andaman Islands (Samanta and Panda 2012) dominated by carbonate sediment depositional platforms maintain their species richness. According to their studies, the true mangrove species are mostly found to grow in the intertidal zones along the tidal creeks of Andaman Islands. Currently, the mangroves of carbonate platforms are damaged in many places due to the magnitude of tsunami waves in Andaman in 2004 (Paul 2005). The Sundarban mangroves however, have been suffering from rapid destruction and degradation (Ghosh and Naskar 2012) by direct and indirect interference of human systems with the mangrove ecosystem of the largest delta. The views of some workers (Acharya and Mohapatra 2012) represent the observable causes of degradation of Bhitarkanika mangroves in Odisha state of India. The tropical deltas mangroves of the region are significantly affected by anthropogenic threats (overuse and conversion). In one of the study made by Majumdar et al. (2012) it is noted that the holophytic grasses of the younger eastern island platform of the Hugli downstream section are luxuriant in growth in comparison to the swampy mangrove forests of the island (Nayachara Island). However, the higher platform with mangrove vegetation covers has reached a self maintaining stage in equilibrium with the present geomorphic processes, loss of tidal drainage, present tidal range with mature channel creeks and inundation frequency.

The coastal wetland loss can be assessed in the regions of Sundarban if the mangrove degradation checklist is prepared by exhaustive survey work on spatiotemporal basis. Environmental protection and conservation of mangrove wetlands depend on consideration of degradation checklist with estimated sequential factors.

11.2 Materials and Methods

The study was conducted in the southwestern part of the Sundarban coast (Henry's Island, Fredrick Island, Patibania Island, Freserganj and Bakkhali) with special emphasis on the forested and reclaimed areas of Namkhana Abad in between river Muriganga and river Saptamukhi (Fig. 11.1).

During the field survey, hypersaline tracts and over wash sand fan lobes, as well as the exposed consolidated mud banks (with presence of mangrove tree stumps) were identified for various measurements and monitoring changes of shoreline characters. Soil pits were constructed in the hypersaline tracts and on the surfaces of over wash fan lobes with the help of forest department for sediment sampling, for finding the productive swamp mud layers and underlying peat swamp, and also for minor measurements and photographic documentations. Total station survey was conducted in some selected areas of the shores in which mangrove dieback was also evident. Contour plan with erosion-deposition sites were prepared to find out the coastal morphometric behavior of the low lying shores formerly dominated by

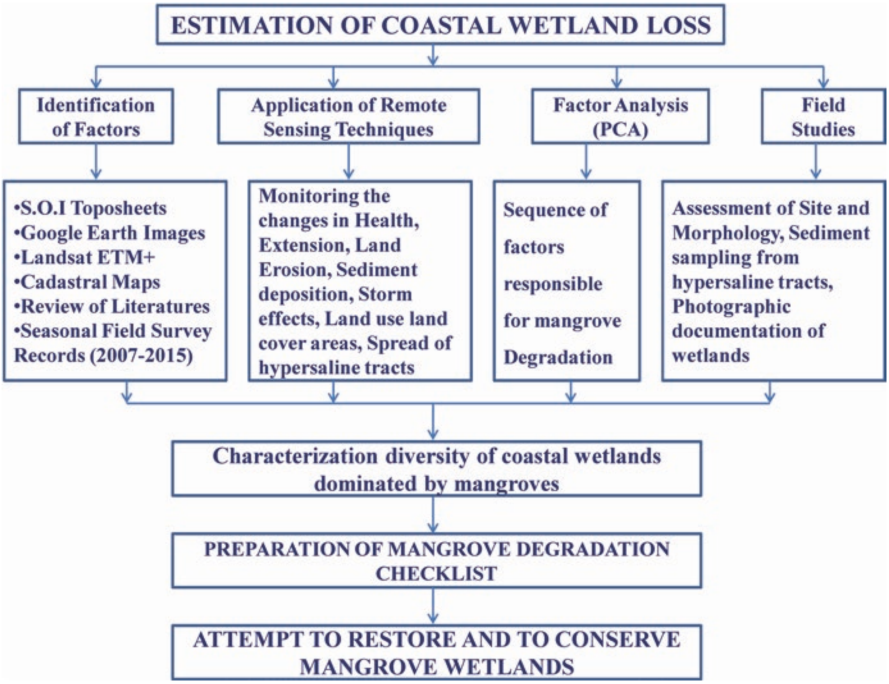

Fig. 11.2 Methodological flow chart to highlight the sequences of the work for assessment of coastal wetlands in Southwest Sundarban

dense mangrove forests. Various maps and images, available secondary data from different sources, database generation and data analysis supported the present work in identifying the multiple causes of mangrove degradations of the coast. Finally, the 40 years of professional experience of the present workers in the field of coastal research, particularly on the critical issues of Sundarban coastal tracts helped to achieve the outcomes of the work (Fig. 11.2).

11.2.1 Mapping of Vegetation Density and Health Status Through the Methods of Remote Sensing Technology

In this present study, to assess the mangrove forest covers dynamics, an object oriented enhancement algorithm has been designed using mathematical operators, which is supervised in nature and expressing the characteristics of plant chlorophyll ‘a’. Two considerations have been taken in this respect like, pixel value having chlorophyll influence will be greater in near-infrared band than red band and pixel value having no chlorophyll influence will be greater in red band than near-infrared band. On the basis of these considerations following supervised enhancement algorithm has been developed and is named as the Modified Advance Vegetation Index (MAVI), (Ray et al. 2013). That can be calculated as:

$$\left[\left\{\left(\rho\mathrm{NIR}+\mathrm{L}\right)\times\left(256-\rho\mathrm{Red}\right)\right\}\times\left(\rho\mathrm{NIR}-\rho\mathrm{Red}\right)\right]^{(1/3)} \tag{11.1}$$

(Where, ρRED = Reflectance value of Red band of TM5 Sensor, ρNIR = Reflectance value of Near-infrared band of TM5 Sensor, L = Coefficient, varies with the vegetation cover, Here L = correctional slope between NIR and Red bands).

It differs from NDVI in the perspective of physiognomic vegetation classes, though there is almost positive relationship between these two. It is more sensitive than NDVI as it is using the power degree of Infrared response. This algorithm has been applied both of the images of the year 1989 and 2015.

In the second phase the vegetation density (VD) has been calculated for both of images of the year 1989 and 2015 by synthesizing the Eq. 11.1 and bareness index (BI) on Principle Component (PCA) basis and it is level sliced into five density zones having distinct areal extent. The bareness index (BI) is calculated using supervised enhance algorithm as:

$$\left[\left(\mathrm{Band5}+\mathrm{Band3}\right)-\left(\mathrm{Band\,4}+\mathrm{Band1}\right)/\left(\mathrm{Band\,5}+\mathrm{Band\,3}\right)+\left(\mathrm{Band\,4}+\mathrm{Band1}\right)\right] \tag{11.2}$$

11.2.2 Mapping of Spatio-Temporal Changes of Hypersaline Tract Within Forest Floor (Southwest Sundarban)

Mature mangrove swamps usually submerged in the high stage of tidal cycles. The seepage channels or embryo creeks carry silts mainly from the interior of the floodplains. By scouring caused by the ebb-tides. This continual scouring without compensatory silting during the flood tends to lower the interior of the floodplains and gradually from the pan areas and enlarge them. In the tidal swamps of the abandoned delta, where the influence of fresh water is almost nil in the dry period, the rapid evaporation of water between two high tides has led to the formation of the salt encrusted flat in the landward zone. Generally wide areas of the high marsh or swamp surface are liable to flood only in the equinoctial tides and in the storms, exposing the top soils to hypersaline environment through prolonged evaporation and salt concentration, through which even halophytes disappear. Such areas remain as blanks without any vegetation. Few areas of tidal flood plain in their initial swamp surface were probably left un-vegetated and the high spring tide water which lay in these embryo pools evaporated to produce highly saline conditions in which no plant life could survive. The salt pans area also developed in the linear depressions of abundant creeks, small pools of supra-tidal flats and even at the elongate chutes of the channel banks (Paul 2002).

Following aforesaid discussion, in this present study, the methodology has been developed to extract the hypersaline tract within the forest floor. Here a PCA (Principal Component Analysis) approach has been carried out between salinity status (Eq. 11.3) and bareness status (Eq. 11.2) of the area and has been used for thresholding to extract the signature of hypersaline tract.

Here like Bareness Index, a supervised enhancement technique has been carried out over the image bands to extract the soil salinity information following a mathematical function which can be said as Salinity Index (SI).

$$SI = \sqrt{\left(\mathrm{R}_{\mathrm{Blue}} \times \mathrm{R}_{Red}\right)} \tag{11.3}$$

11.2.3 Mapping of Shoreline Changes Through Remote Sensing Techniques

Various methods for coastline extraction of optical imagery have been developed. Coastline can even be extracted from a single band image, since the reflectance of water is nearly equal to zero in reflective infrared bands, and reflectance of the absolute majority of land-covers are greater than water. This can be achieved by histogram thresholding on one of the infrared bands of TM or ETM+ imagery. Experience has shown that of the six reflective TM bands, mid-infrared band 5 is the best for extracting the land-water interface. TM/ ETM+ band 5 exhibits a strong contrast between land and water features due to the high degree of absorption of mid-infrared energy by water (even turbid water) and strong reflectance of mid-infrared by vegetation and natural features in this range. Of the three TM infrared bands, band 5 consistently comprises the best spectral balance of land to water. The histogram of TM band 5 ordinarily displays a sharp double peaked curve, due to tiny reflectance of water and high reflectance of vegetation (Cahoon and Hensel 2006). The transition zone between land and water resides between the peaks. The transition zone is the effect of mixed pixels and moisture regimes between land and water. If the reflectance values are sliced to two discrete zones, they can differentiate water (low values) and land (higher values). But the difficulty of this method is to find the exact value, as any threshold value will be exact on some area, not all. Another method is to use the band ratio between NIR (TM/ ETM+ band 4 and OLI band 5) and Green (TM/ ETM+ band 2 and OLI band 3) bands and also, between SWIR (TM/ ETM+ band 5 and OLI bands 6) and Green. With this method water and land can be separated directly. Finally, if the aforesaid two combinations are multiplied, the coastline can be delineated through thresholding approach (Ray and Mondal 2014).

11.3 Results

Major factors and their sub-factors of mangrove degradations are categorized after the field investigations and monitoring shoreline changes over a time and space in the region of Sundarban coasts. Southern shores of Namkhana-Henry's Island, Jambu Island, Lothian Island, Mousumi Island and Sagar Island are eroded after the impacts of previous cyclones. Younger bars are emerging in the shallow sea parallel to the shoreline that playing an important role for concentrating the tidal energy creating or directing the long shore currents into the shorelines of concavities and convexities at present. The low-lying shores with a high tidal range (>3.9 m) are liable to erosion under such conditions in which mangroves are lost and sediment stability is affected.

The current extent of mangroves has been dramatically reduced from the original extent nearly every country (Burke et al. 2001). The leading human activities that contribute to mangrove loss are 52% of aquaculture (38% shrimp plus 14% fish production), 26% by forest use, and 11% by fresh water diversion (Valiela et al. 2001). Restoration has been successfully attempted in some places, but this has not kept pace with wholesale destruction in most areas. A large number of poor people are directly or indirectly dependent on the forest uses in the Sundarban. One estimate shows that about 1 lakh people in West Bengal and 2 lakhs people in Bangladesh enter into the forest areas of the Sundarban in the month of April for the collection of non-timber forest products and other types of forest materials. Original extents of mangrove forests are converted into fish ponds in the northern fringes and south-western fringes of Sundarban in West Bengal (Fig. 11.3).

11.3.1 Studies of Mangrove Degradation in the Coastal Wetlands of Southwest Sundarban

Mangrove degradation checklists are prepared for a number of islands of southwestern Sundarban (Henry's Island, Fredrick Island, Bakkhali, Patibania Island, Jambu Island and Mousuni Island) for the present study. However, only three islands are selected for application of such method for assessing the nature of mangrove degradation at present.

11.3.2 Islands of Namkhana Abad with Extent of Mangroves

The extent of mangroves is still evident in the buffer areas of reserve forest of the Sundarban. Forest interface society is located in the fringe areas of such extent of mangrove forest of the southwestern Sundarban.

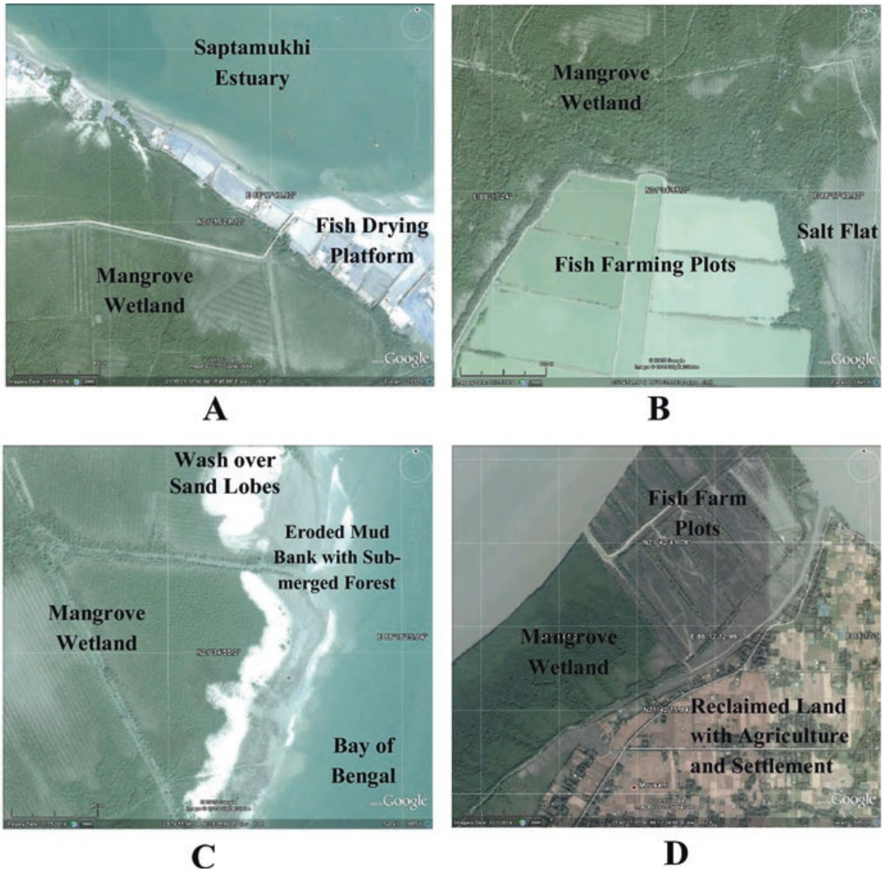

Fig. 11.3 Causes of mangrove degradations identified from the southwestern part of the Sundarban (India). (**a**) Location of fish drying platforms on the spit back surface adjacent of mangrove wetlands; (**b**) encroachment of fish farming plots into the inner part of mangrove wetlands; (**c**) landward movement of wash over sand fan lobes submerging the mangrove wetlands and (**d**) land reclamation processes, removing the natural mangrove wetlands of the Sundarban

Jambu Island of this region is reduced in size by land erosion, but currently forested by mangrove. Fisherman community was settled in the south central part of the island, and fish drying platforms were also extended over a large area of the island surface in the expanse of mangrove in the decades of 1980s and 1990s and even up to 2007. One of the permanent settlement units (J.L. No. 420) was also located at the northern limit of the island within the mangrove forest since 1966. However, the forest department has removed all the settlers from the island at present. Mangrove forests are now well preserved in the distal island, though a large part is affected severely by erosion.

Henry's island on the shores of the Bay of Bengal and Saptamukhi river estuary is another extent of mangrove forest in which fishery project, mangrove ecotourism

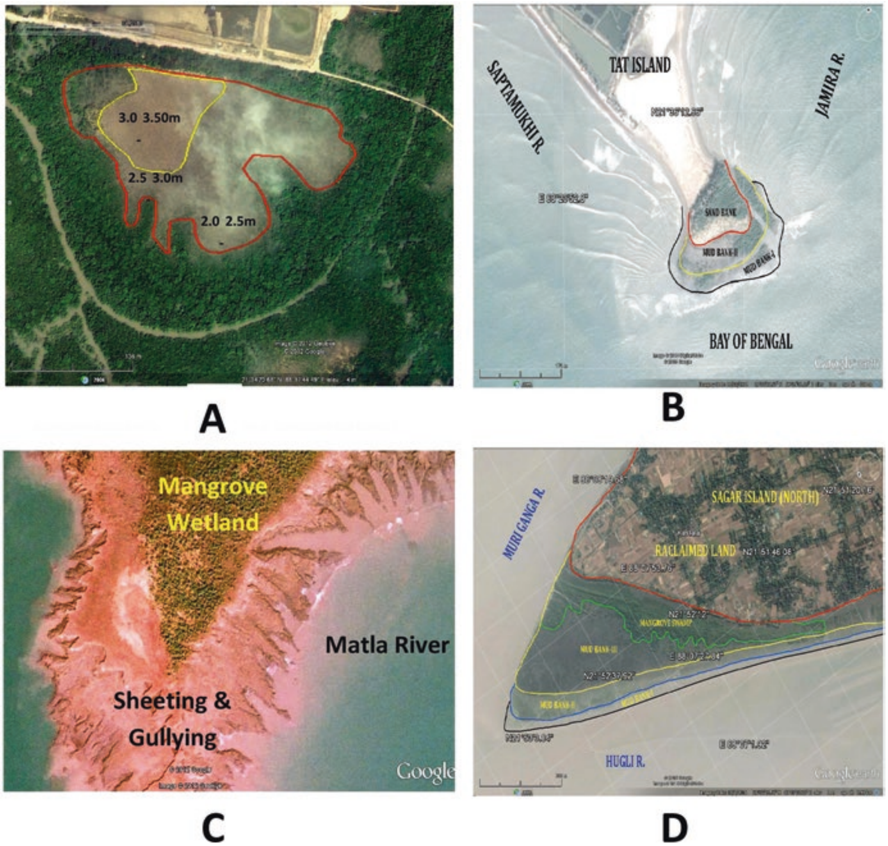

Fig. 11.4 Coastal wetland loss resulted from hyper salinity and erosion and mud banks in the Sundarban. (**a**) Spread of hyper saline tracts within the inner part of mangrove wetland affected by loss of tidal drainage under evaporative environment; (**b**) fragmentation of mangrove wetlands resulted due to erosion and sand deposited in the sea word face of the Sundarban; (**c**) active sheet and gully erosion on the bare mud banks (once colonized by mangrove wetlands) of Hally Day Island of Malta estuary and (**d**) mangrove swamps degradations due to erosion of the mud bank with active sheeting process in Sagar Island

complex and fish drying platforms are located. Mangrove forest with their zonation patterns and areas of fisheries projects encircled by degraded forests is seen from the watch tower of ecotourism complex. Land erosion along the shoreline and hypersaline tracts within the forest areas of drainage loss are evident on the island (Fig. 11.4).

Large areas of mangrove forests are cleared from the northern fringes of Fredric Island and Bakkhali Island during 1960s and 1970s for the extent of human settlements. Bakkhali emerged as a tourist destination in the middle part of the 1970s when Fresarganj and Lakshmipur village were badly damaged by cyclone landfall.

Mangroves were lost due to shorefront erosion and wash over sand deposits in the peat swamps of inner parts in these islands.

Patibania island and Susnir Char are among the part of reserve forest that extends from southwest of Fresarganj to northwest of Fresarganj across Patibania creek. Mangroves are dwarfed in the inner parts of the forest due to the extensive growth of hypersaline tracts resulted from drainage loss of tides in the region. Ridge forest is extended all along the tidal creeks with extent of active tidal flat in the region. Species richness of mangrove is positively related to the soil types and age of tidal flat in the island. Richness is reduced in the inner part or older tidal flat of the island that occupies high salt tolerant, salt marsh plants in the expanse of younger and matured mangroves.

Mahisani or Mousuni Island is located along Muriganga river estuary on the west of Patibania Island and separated by Pitt's creek. Mangrove forests are largely degraded in the island due to extent of human settlements, agricultural lands. Fish farm plots and direct human uses of various land resources. Remaining forests are extended towards northwest, south-southwest, and along a narrow strip parallel to the active tidal bank of Pitt's creek.

The above mentioned islands dominated by partial coverage of mangrove acts as a bio – shield or natural buffers against the effects of cyclone and shoreline erosion. Sediment stability is achieved in the younger deltaic alluviums by extensive growth of mangroves. The dynamic shaping and reshaping of the shorelines of Fresarganj and Bakkhali are the result of mangrove degradations of surrounding areas, once dominated by a luxuriant growth of mangrove that arrested sediments derived from marginal erosion and long shore current transport during cyclone breaks in the coast (Fig. 11.5).

Currently, the mangrove afforestation program is implemented for restoring the drainage in the old tidal flat by the forest department in Fredrick Island and Henry's island, but the extent of hypersalinity and regeneration problems of mangroves in the dynamic configurations of the shorelines disturbed the above attempt in the region.

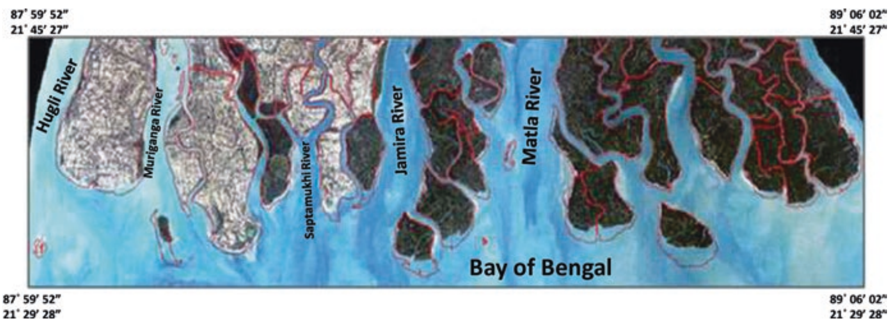

Fig. 11.5 Mangrove wetland loss resulted from shoreline change activities on the seaward side of the Sundarban, India (2000–2010)

11.4 Discussions

The above results are discussed in the following ways;

11.4.1 Causes of Mangrove Degradation

Several causes of mangrove degradations have been identified by the present study for the region of Sundarban under tropical hot and humid climate.

1. **Human uses of mangroves:** Historical land reclamations and permanent settlement started in the Sundarban since 1810 when the areas of mangroves (about 20,000 km^2) were distributed in three administrative districts (Buckerganj, Khulna and 24-Parganas) of undivided Bengal at the coastal zone of Ganga delta. The entire forest belt was separated into several islands by intricate networks of tidal creeks, and active distributary channels of the Ganga delta supported by seasonal discharges of fresh waters with frequent river flood, and seaward transport of sediments. At present, mangroves are thriving only in areas of 4962 km^2 land surface occupied by the West Bengal portion of Sundarban (North and South 24-Parganas districts) in India, and the remaining areas are distributed in the deltaic islands of Bangladesh Sundarban (Buckerganj and Khulna districts).

   The poverty stricken people use non-timber forest resources in the buffer areas of mangrove forest to support their livelihood. Various tourism infrastructures have been emerging within the forest belt in the expanse of mangroves. Mangroves are also lost from many places due to the dispersal of wastes nearby the harbors and market places adjacent of tidal creeks. As population pressure is increasing day by day at the northern and southwestern administrative blocks fringed by existing mangrove forests, the human uses of mangroves also may be rapidly increased in the buffer areas.
2. **Fishery development:** Construction of various aquaculture ponds within the forest belt, clay mining from the tidal flat for building the mud built walls to protect the fish ponds, emerging number of dry fish processing platforms, as well as the development of temporary colonies of fishermen along the interface of back shore mangroves and sandy shores have severely reduced the areas of mangrove forests in the Sundarban. Hence the release of untreated waters from the fish farm ponds into the tidal flats and construction of harbors, jetties, sluice gates and bridges like engineering structures also have disturbed the growth and survival of mangroves in the pristine environment of Sundarban. The rate of encroachment of fishery development in the forest and the forest fringe areas has been rapidly increased in the previous decades (1990s and 2000 onward).
3. **Hypersalinity:** Supra-tidal flats of islands and shore fringed areas of the Sundarban are affected by infrequent tidal inundation due to configuration changes on the island surface with uneven distribution of sediments in the sedimentary depositional environment. Such immersion of certain portions of the

forest floor from the active tidal flat may provide longer exposure to the evaporative environment that increases the salinity of soil to hinder the growth and survival of mangroves.

Such hypersaline patches are developed in different forms in entire Sundarban in between the swamp forests. They may be categorized as **salt ponds** (salt water ponds of linear and semi circular sizes), **salt pans** (salt encrusted pans with a cover of thinner vegetations), **salt flats** (highly salt affected flat surface with the remains of dead stumps and extremely dwarfed mangroves), and **saline blanks** with algae encrusted surface. Seasonal modifications with rain water storage, evaporative environment, supply of salt water and wash over sediments during storm events may take place in the low lying deltaic platforms dominated by mangrove forests and segmented by tidal creeks. The spread of such hypersaline patches is observed in the seaward face of islands probably due to the immediate effects of storminess of sea and prolonged evaporation rate. Mangroves are severely affected by dieback process in and around the supra tidal salt affected surface of the Sundarban at present (Fig. 11.6).

4. **Sediment deposition:** Currently, the sea sands from low lying sand dunes, spits, sea beaches and sand bars are transported or drifted into the forest areas by over wash fan sediment deposition particularly by storms and tidal waves. The mangroves on the sea ward side are submerged by advancing blankets of sand fan terrace in many places of the Sundarban.

   The delta front sand bodies with different shapes and sizes are now unable to hold the surge waters of advancing storm waves that can transport the bulk of

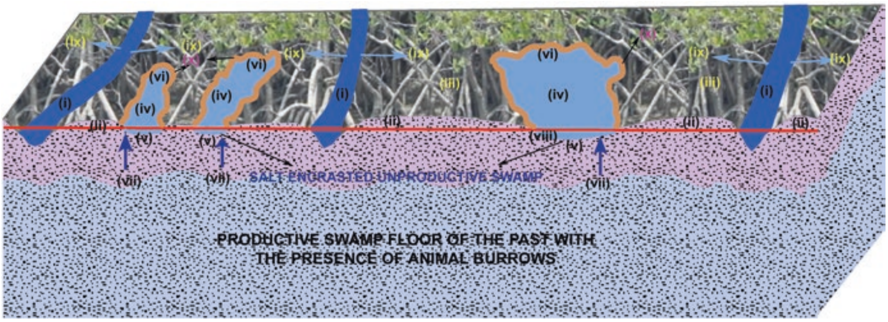

Fig. 11.6 Formation of hypersaline patches under evaporative environment into the inner part of mangrove swampy tract. (*i*) Tidal channels with High Tide and Low Tide levels; (*ii*) Siltation and natural levee formation with repeated inundations; (*iii*) Ridge forest development along the channel fringes with submergence and emergence of tide water levels; (*iv*) Areas of tidal emersion away from the active drainage condition; (*v*) Evaporation and loss of soil moisture with capillary rise and fall of water table; (*vi*) Encrustation of salts over the evaporated patch with seasonal supply of storm waters and sediments to support the temporal moisture level of the soil; (*vii*) Capillary rise of water table; (*viii*) Areas remain as surface depression due to loss of drainage and regular supply of sediments; (*ix*) Salt water encroachment with submergence in high energy events and (*x*) Vegetation dieback condition under the effects of salinity stress and transformation of salt tolerant vegetations into dwarfed condition along the fringes of hypersaline patch

sediments from seaward face to inner parts of the delta plains. The landward advancement of sediment lobes submerges the forests, wetland surface and tidal creeks at a steady rate with increased frequency of storm events in the deltaic coast at present.

5. **Storm effects:** The tropical cyclone breaks in November 2007, April 2009, November 1988, October 1989 and November 1991 devastated mangroves on the coast of the Bay of Bengal in Bangladesh and West Bengal (India). The stumps of mangrove trees on the shore front mud banks are the remains of the devastation. Sediments have been stripped from the forest floor in which the root system of mangrove trees is exposed in unearthing condition in the foreground. The top breakage of mangroves in the inner forest records winds thrown activities of cyclone landfall on the deltaic coast fringed by Bay of Bengal.

   Such storm effects have been monitored along the Sundarban shores after the events in many places to estimate the wetland losses. Prolonged inundation by salt water flooding and fire ball activities during storm events also devastated mangroves in the Sundarban coast in the past.

6. **Land erosion:** Low-lying deltas flat is the place of geographical habitat to support the vegetated tidal flat along the seaward parts of Sundarban. The rate of land erosion between 1989 and 2015 shows a great loss of mangrove forests as well as coastal wetlands in the Sundarban (Figs. 11.5 and 11.11). Various mechanisms of land erosion resulted from hydrodynamic stress have produced significant damages to the mangrove wetlands along the coastal belt. All the sea front islands with mangrove forests have been reduced in size and shape due to erosion in the previous decades. Erosion and flooding related activities will be increased in steady state if the current sea level rise takes place as per predictable rates in the Sundarban Delta (Table 11.4).

   It is established by the present study made by Raha et al. (2014) that other than sea level rise, many factors like sediment deposition, lack of fresh water flow in the deltaic part and natural subsidence of the deltaic platform are also playing a major role in the loss of mangrove wetlands in the Sundarban.

7. **Mangrove regeneration problems:** Large number of mangrove seeds is regularly drifted into the unfavorable sandy substrate after the events of storms, tidal waves; southwest monsoon brace and HAT (Highest Astronomical Tides) phase currents along the shores of the Bay of Bengal. Litters and huge amount of viviparous mangrove seeds are usually deposited along the high tidal limit of the sandy shores of south west Sundarban during monsoon months. Large number cattle population with other livestock generally feed on mangrove seeds along the sandy shores (Fig. 11.7).

   The wave beaten compact mud banks on the foregrounds of high tide shoreline do not provide the anchorage facilities for mangrove seeds in the dynamic current condition. Tidal dispersal of mangrove seeds cannot reach the supra tidal flat affected by drainage loss, particularly in the inner parts of islands away from channels and shorelines. Other activities like crab potting, clay mining, and unscientific netting across channels and intertidal flats also disturb the process of seed anchorage. Mangrove trees in the sapling stage are also affected by insect

Fig. 11.7 Mangrove seeds are drifted by currents from the forests along the line of the tidal litter zone. The cattle population and other livestock animals feed on such mangrove seeds

attacks and eutrophicated water bodies with surface encrusted algae mats. Regeneration problems of mangroves are also achieved due to surface instability of swamp tract under frequent salt water flooding incidences.

11.4.2 Mangrove Degradation Checklists for Southwest Sundarban

The vegetation vigourness status analysis along with the identification of vegetation density differences through Geo-spatial technology reveals the nature of coastal wetland degradations in buffer areas of Sundarban reserve forest in West Bengal (India). Other human effects as well as the effects of marine, forcing factors related to sea level rise (Raha et al. 2014) are visible in the region for the rapid rate of mangrove wetlands degradations at present (Figs. 11.8, 11.9, 11.10, 11.11, and 11.12). Monitoring coastal geomorphology in the West Bengal portion of Sundarban over the decades of 1980s, 1990s and 2000 millennium years, considering with the review of existing literature documents for Sundarban land reclamation process and cyclone effects, it is established that coastal wetlands of Sundatban have been

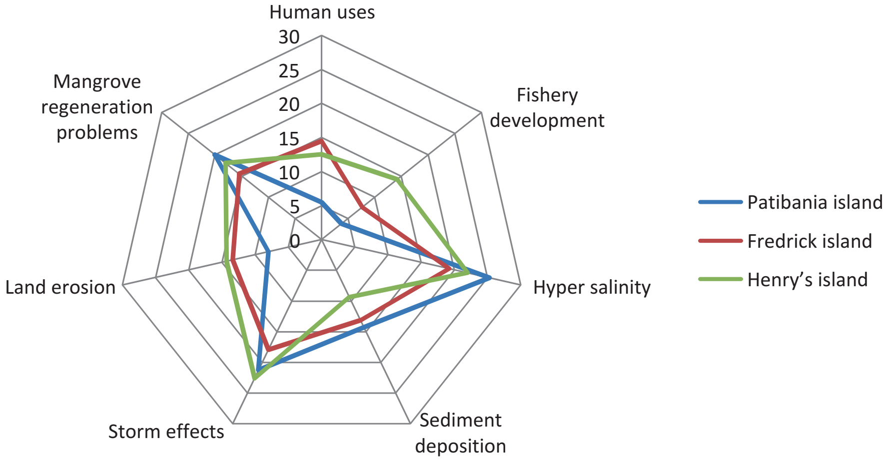

Fig. 11.8 Diagrammatic representation of the factors of mangrove degradation after the result of total weighted values for three sample islands

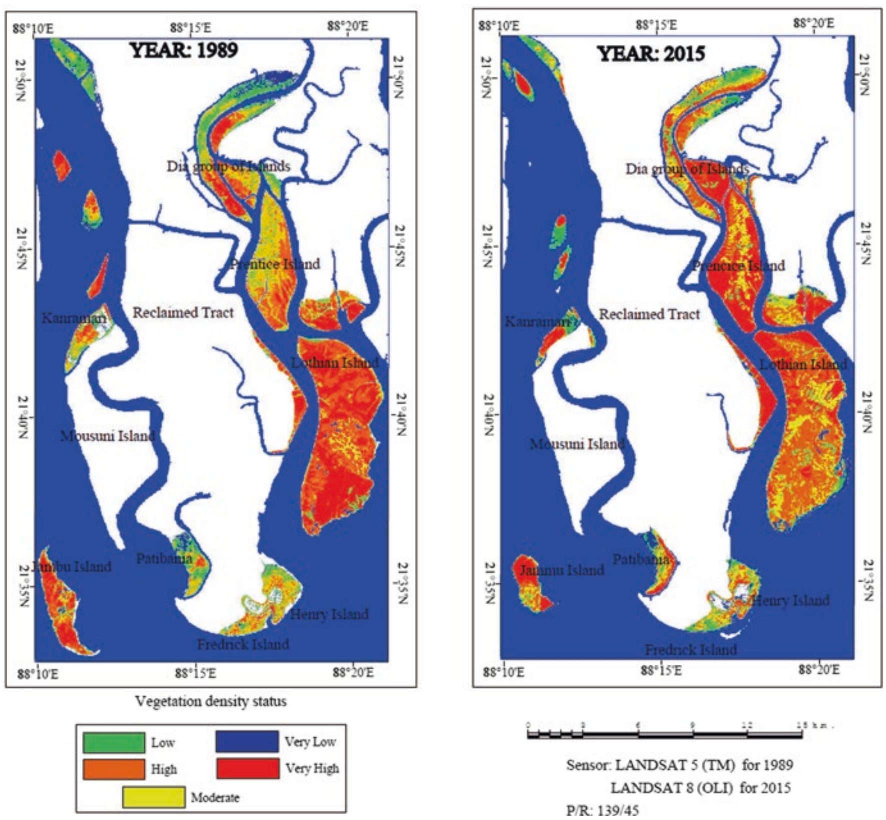

Fig. 11.9 Vegetation density differences in between 1989 and 2015 (the mangroves of Southwest Sundarban) represent a sharp reduction of mangrove density in the year 2015 in comparison to the year 1989. Low canopy coverage, dwarfed growth of mangroves and isolation of vegetation patches by the spread of hypersaline tracts are major changes that took place in the mangrove dominated wetlands as drivers of low density in forest cover

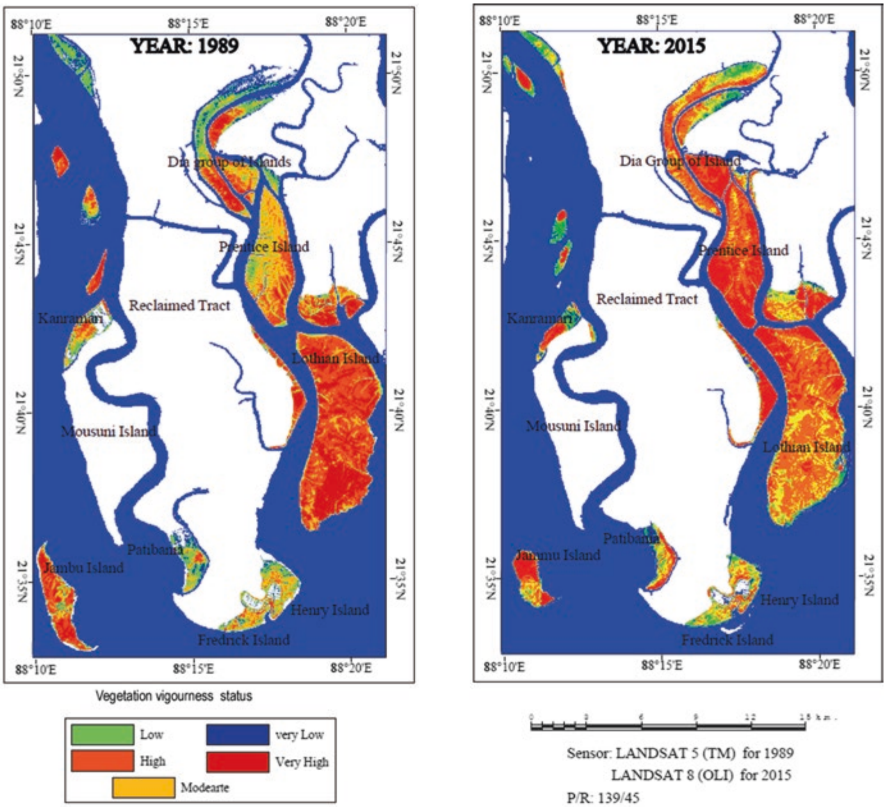

Fig. 11.10 Vegetation vigourness status (Health) of mangroves in the coastal wetlands of south-western Sundarban (1989–2015) is positively correlated with density differences of mangroves in spatio-temporal condition. The attractive health of mangroves is recorded only in those areas of the islands where fresh silt accumulation and tidal drainage are present

affected by multiple factors of degradations. Mangrove degradation checklists are prepared on the basis of the above investigations on recent environmental changes of Sundarban for identification of potential threats and formulation of conservation strategies of mangroves (Tables 11.1, 11.2, and 11.3) (Table 11.4).

11.4.3 Factor Analysis (PCA) for Mangrove Degradation Related Factors and their Sub-factors

Several factors of mangrove degradation have been identified from the above analysis in the case of Southwest Sundaraban (India). These factors and sub-factors are analyzed on the basis of the PCA method to categorize them in a sequential order for the case studies of individual islands according to the following statistical values.

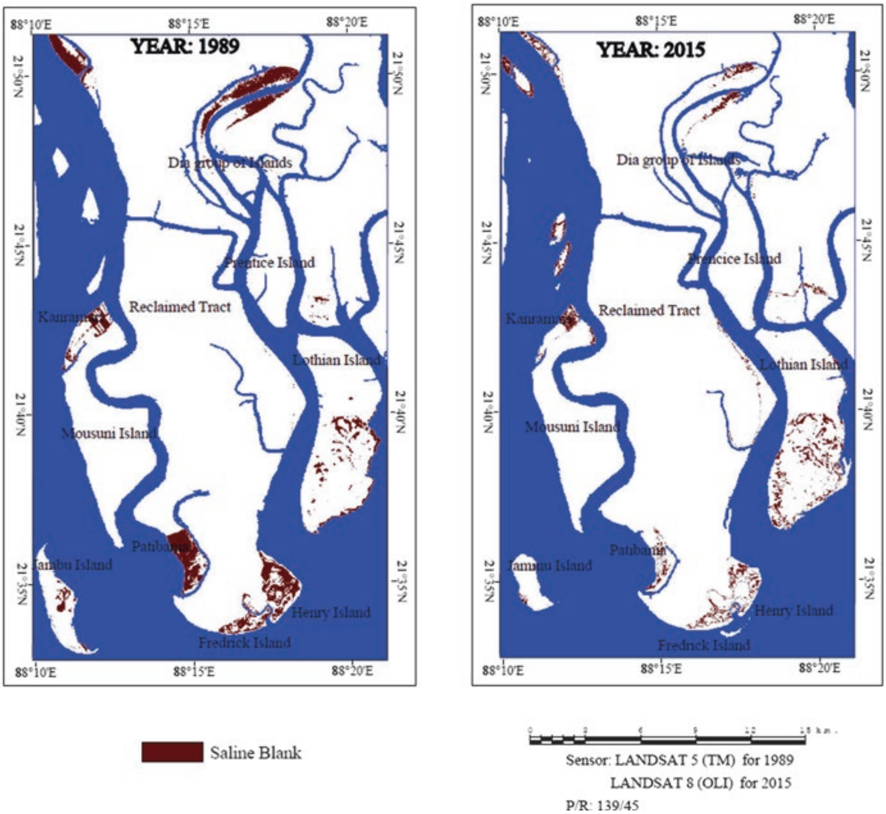

Fig. 11.11 Temporal changes of hypersaline tracts in between 1989 (17.1819 km^2) and 2015 (9.3078 km^2) at and around southwestern Sundarban. Saline blanks were concentrated in few islands, but they are now expanding over the seaward zones of mangrove wetlands

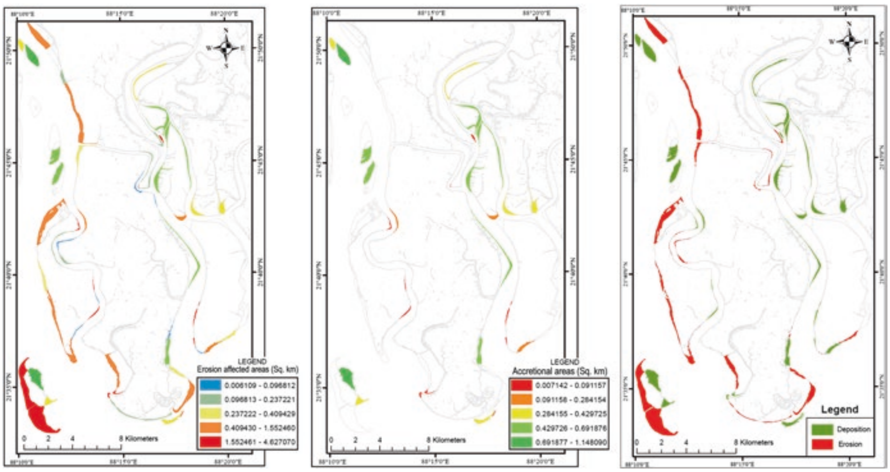

Fig. 11.12 Erosional and accretional areas of the shorelines adjacent of southwestern Sundarban estimated through spatio-temporal image analysis

Table 11.1 Identification of factors and sub-factors of the nature of mangrove degradations in the Sundarban coastal tract

Sl. No.	Island area surveyed in Southwest Sundarban	Identification of factors for each island	Identification of sub-factors (in number)
1	Patibania Island	Sediment deposition	8
2	Fredrick Island	Fishery development	8
3	Henry's Island	Human uses	8
4	Bakkhali	Hyper salinity	8
5	Jambu Island	Storm effects	8
6	Mousuni Island	Land erosion	8
7	Susnir char	Mangrove regeneration problems	8

Table 11.2 Preparation of mangrove degradation checklist by weighted values of factors and sub-factors

Sl. No.	**Human uses:(parameters)**	**Patibania Island**	**Fredrick Island**	**Henry's Island**
1	Fuel use and fodder use	1.0	2.0	2.0
2	Wood cut	0.5	1.0	0.5
3	Reclamation for agriculture and housing	0.5	4.0	3.0
4	Fixing nets by mangrove wood	0.5	1.0	1.0
5	Thatching roofs of mangrove leaves	0.5	0.5	0.5
6	Recreational uses for mangrove ecotourism	1.0	1.0	3.0
7	Coastal tourism expansion and industrialization	0.5	4.0	2.0
8	Waste disposal and pollution	1.0	1.0	0.5
Total scores		*5.5*	*14.5*	*12.5*
Sl. No.	**Fishery development:(parameters)**			
1	Construction of fish pond	0.5	2.0	4.0
2	Wetland for fish culture	1.0	2.0	3.0
3	Mari culture ponds	1.0	1.0	2.0
4	Dry fish processing platform	0.5	1.0	2.0
5	Construction of mud built bank wall and clay mining	0.3	0.5	1.0
6	Fisher man's colony development (temporary structures)	0.2	0.5	0.5
7	Release of untreated water from the fish farms	0.1	0.5	1.5
8	Construction of fishing harbors and jetties; sluice gates, bridges etc.	0.2	0.10	0.20
Total scores		*3.8*	*7.6*	*14.2*
Sl. No.	**Hypersalinity:(parameters)**			
1	Formation of saline blank	4.0	2.0	2.0
2	Sub-aerial exposure of mangrove swamps and rapid evaporation	3.0	2.0	2.0
3	Salt spray on the sea front mangroves	2.0	1.0	2.0

(continued)

Table 11.2 (continued)

4	Tidal drainage loss from the matured surface	4.5	3.0	4.0
5	Areas devoid of fresh silt accumulation	3.8	3.2	3.5
6	Episodic tidal waves and encroachment of tide water in inner island depressions	3.5	3.5	4.0
7	Extensive dry spells in a year	2.0	2.0	2.0
8	Low infiltration rate on the clay pans	2.5	2.5	2.5
Total scores		*25.3*	*19.2*	*22.0*
Sl. No.	**Sediment deposition:(parameters)**			
1	Wash over fan lobes on to the mangrove wetlands	3.0	3.0	2.0
2	Roll over process of barrier bars in the inland areas in the rising sea level	3.0	1.0	1.0
3	Formation of wind-tidal flat by blowing sand	3.0	2.0	1.0
4	Over bank splays at the distributaries flood basin	3.0	2.0	1.0
5	Channel decaying by in-channel siltation	1.0	1.8	1.5
6	Mobility of dune sediments in the inland areas	0.5	2.5	2.0
7	Mass transport of sands with placer minerals in the storm	0.6	0.8	0.8
8	Rapid siltation on the mud flats with slumping effects	0.2	0.1	0.1
Total scores		*14.3*	*13.2*	*9.4*
Sl. No.	**Strom effects: (parameters)**			
1	Salt water flooding over a time and space	4.0	2.0	4.0
2	Wind damage activities at the magnitude of storms	2.0	2.0	2.0
3	Fireball activities at the high magnitude storms	0.5	0.5	0.5
4	Storm dispersal of mangrove seeds	4.0	4.0	4.0
5	Storm surge elevation changes	4.5	3.5	3.8
6	Bank instabilities due to undermining the process by storm waves and currents	1.8	1.5	2.5
7	Sheet erosion and removal of fresh silts from the swampy tract	1.5	1.5	2.8
8	Increasing storminess with energy stress	3.0	3.0	3.0
Total scores		*21.3*	*18.0*	*22.6*
Sl. No.	**Land erosion: (parameters)**			
1	Bank erosion by shifting channels	2.0	3.0	3.0
2	Shoreline erosion by waves and currents (horizontal)	2.0	4.0	3.0
3	Significant cliffing along the shores (vertical erosion)	0.5	2.0	1.0
4	Spurs and furrows along and across the mud banks	0.5	1.0	2.0
5	Erosion protection measures by engineering structures	0.5	2.0	0.5

(continued)

Table 11.2 (continued)

6	Reduction of land margin swamps by erosion	0.8	0.8	2.0
7	Current concentration and increased wave heights during landfall of cyclones	1.5	0.5	2.5
8	Slumping and collapsing banks	0.2	0.1	0.3
Total scores		*8.0*	*13.4*	*14.3*
Sl. No.	**Regeneration problems: (parameters)**			
1	Unfavorable substrate	4.0	3.0	1.0
2	Change in island elevation and drainage loss	4.0	3.0	3.0
3	Trapping the seeds by fishing nets	1.0	1.0	2.0
4	Crab potting in the mud banks	2.0	1.0	2.0
5	Insect attacks in the mangrove trees	1.0	1.0	1.0
6	Clay mining on the mud banks	1.0	2.0	2.0
7	Cattle grazing, trampling effect and seed eating habit by live stocks	2.5	1.5	3.0
8	Storm dispersal of seeds to the inland areas	4.5	3.0	4.0
Total scores		*20.0*	*15.5*	*18.0*

Table 11.3 Total scores resulted from the estimation of weighted values against the identified factors of selected islands in the buffer areas of Southwest Sundarban

	Total scores		
Factors	Patibania Island	Fredrick Island	Henry's Island
Human uses	5.5	14.5	12.5
Fishery development	3.8	7.6	14.2
Hyper salinity	25.3	19.2	22
Sediment deposition	14.3	13.2	9.4
Storm effects	21.3	18	22.6
Land erosion	8	13.4	14.3
Mangrove regeneration problems	20	15.5	18

Table 11.4 The data base on vegetation density and vegetation health generated from the analysis of Remote Sensing images (1989 and 2015)

	Vegetation density		Vegetation health	
Category	Area (m^2) in 1989	Area (m^2) in 2015	Area (m^2) in 1989	Area (m^2) in 2015
Very low	50,230	31,208	3944	7702
Low	85,561	86,190	14,309	22,232
Moderate	102,721	105,166	34,950	46,793
High	108,554	106,901	77,392	79,385
Very high	110,336	107,964	106,895	101,229

Table 11.5 Descriptive statistics for seven major factors of three sample islands

Sl. No.	Factors	Analysis N	Patibania Island		Fredrick Island		Henry's Island	
			Mean	Std. deviation	Mean	Std. deviation	Mean	Std. deviation
1	Human uses	8	0.688	0.2588	1.813	1.4126	1.563	1.0836
2	Fishery development	8	0.475	0.3536	0.950	0.7111	1.775	1.2725
3	Hyper-salinity	8	3.163	0.9395	2.400	0.8159	2.750	0.9258
4	Sediment deposition	8	1.788	1.3141	1.650	.9532	1.175	0.6386
5	Strom effect	8	2.663	1.4282	2.250	1.1650	2.825	1.1913
6	Land erosion	8	1.000	0.7251	1.675	1.3339	1.788	1.0710
7	Regeneration problems	8	2.500	1.4880	1.938	.9425	2.25	1.035

11.4.3.1 Descriptive Statistics

The first output of this analysis carried out a table of descriptive statistics, which is attentive to all the variables under investigation. Classically, the mean, standard deviation and number of respondents (N) which contributed in the assessment are given below. The number of cases used in the study will be less than the total number of cases in the data file if there are missing values in any of the variables used in the factor analysis, because, by default, this investigation does a list wise deletion of deficient cases. Looking at the mean, one can conclude that the hyper salinity is the most important variable that influences the degradation of land, as it has the highest mean of 3.163. At the same time the highest mean value of Fredric Island and Henry's island is 2.400 and 2.825 respectively (Table 11.5).

11.4.3.2 The Correlation Matrix

The next output from the analysis is the correlation coefficient. A correlation matrix is simply a rectangular array of numbers which gives the correlation coefficients between a single variable and every other variables in the investigation. The correlation coefficient between a variable and itself is always 1; hence the principal diagonal of the correlation matrix contains 1. The correlation coefficients above and below the principal diagonal are the same. The determinant of the correlation matrix is shown at the foot of the tables below. Therefore, it is seen from the table the determinant value is not 0. If the determinant value is 0, then there will be computational problems with the factor analysis, and SPSS may issue a warning message or be unable to complete the factor analysis. However, the determinant values calculated for Patibania Island, Fredric Island and Henry's Island are 0.009, 0.002 and 0.004 respectively (Tables 11.6, 11.7, and 11.8).

Table 11.6 Correlation matrix for seven major factors of Patibania Island

Factors	Human uses	Fishery development	Hyper-salinity	Sediment deposition	Storm effect	Land erosion	Regeneration problems
Human uses	1.000	−0.410	0.151	−0.349	0.157	0.000	0.371
Fishery development	−0.410	1.000	−0.098	0.832	−0.327	0.251	0.041
Hyper-salinity	0.151	−0.098	1.000	0.292	0.803	0.031	−0.072
Sediment deposition	−0.349	0.832	0.292	1.000	1.025	0.381	0.084
Storm effect	0.157	−0.327	0.803	0.025	1.000	−0.070	0.188
Land erosion	0.000	0.251	0.031	0.381	−0.070	1.000	0.457
Regeneration Problems	0.371	0.041	−0.072	0.084	0.188	0.457	1.000

Determinant = 0.009

Table 11.7 Correlation matrix for seven major factors of Fredrick Island

Factors	Human uses	Fishery development	Hyper-salinity	Sediment deposition	Storm effect	Land erosion	Regeneration problems
Human uses		0.49	0.013	0.478	0.021	0.402	0.252
Fishery development	0.49		0.161	0.146	0.334	0.001	0.17
Hyper-salinity	0.013	0.161		0.384	0.044	0.194	0.413
Sediment deposition	0.478	0.146	0.384		0.363	0.215	0.315
Storm effect	0.021	0.334	0.044	0.363		0.34	0.424
Land erosion	0.402	0.001	0.194	0.215	0.34		0.231
Regeneration problems	0.252	0.17	0.413	0.315	0.424	0.231	

Determinant = 0.002

Table 11.8 Correlation matrix for seven major factors of Henry's island

Factors	Human uses	Fishery development	Hyper-salinity	Sediment deposition	Storm effect	Land erosion	Regeneration problems
Human uses	1.000	0.043	−0.053	0.457	−0.477	0.192	−0.271
Fishery development	0.043	1.000	−0.491	0.358	0.069	0.726	−0.396
Hyper-salinity	−0.053	−0.491	1.000	0.302	0.421	−0.263	−0.298
Sediment deposition	0.457	0.358	0.302	1.000	0.223	0.413	−0.832
Storm effect	−0.477	0.069	0.421	0.223	1.000	0.101	−0.307
Land erosion	0.192	0.726	−0.263	0.413	0.101	1.000	−0.126
Regeneration problems	−0.271	−0.396	−0.298	−0.832	−0.307	−0.126	1.000

Determinant = 0.004

Table 11.9 KMO and Bartlett's test for seven major factors of three sample islands

Tested substance		Patibania Island	Fredrick Island	Henry's Island
Kaiser-Meyer-Olkin measure of sampling adequacy		0.359	0.331	0.361
Bartlett's test of sphericity	Approx. Chi-square	17.903	23.782	21.037
	df	21	21	21
	Sig.	0.655	0.304	0.457

11.4.3.3 Kaiser-Meyer-Olkin (KMO) and Bartlett's Test

The KMO measures the sampling capability which should be greater than 0.5 for acceptable factor analysis to proceed. If any pair of variables has a value less than this, it may be considered the declining one of them from the investigation. The all off-diagonal elements should be very Small (close to zero) in a good model. Looking at the table below, the KMO measure shows values 0.359 for Patibania Island, 0.331 for Fredric Island and 0.361 for Henry's Island after the calculation.

The Bartlett's test is an additional indication of the power of the relationship along with variables. These trials represent the null hypothesis that the correlation matrix is a distinctive matrix. An identity matrix is the matrix in which all of the diagonal elements are 1 and all off-diagonal elements are 0. For which one can desire to reject this null hypothesis. From the same table (Table 11.9), one can see that the Bartlett's test of sphericity is significant, and at the same time its connected probability is less than 0.05. In fact, it is actually measured for Patibania Island is 0.655, is 0.304 for Fredric Island and 0.457 for Henry's Island; by which the significance level is small enough to reject the null hypothesis in this experiment (Table 11.9).

11.4.3.4 Communalities

The next item from the output is a table of communalities which shows how much of the ariance in the variables has been accounted for by the extracted factors. For instance, over 93.5% of the variance in Hypersalinity is accounted for while 65% of the variance in Human uses is accounted for Patibania Island. Similarly, over 95% of the variance in Human Uses is accounted for while 74.5% of the variance in Strom Effect is accounted for Fredric Island, and over 93% of the variance in Sediment Deposition is accounted for while 73% of the variance in Land Erosion is accounted for Henry's Island (Table 11.10).

Table 11.10 Communalities for seven major factors of three sample islands

Factors	Initial	Extraction		
		Patibania Island	Fredrick Island	Henry's Island
Human uses	1.000	0.650	0.951	0.890
Fishery development	1.000	0.862	0.938	0.918
Hypersalinity	1.000	0.935	0.888	0.830
Sediment deposition	1.000	0.926	0.870	0.934
Strom effect	1.000	0.898	0.745	0.877
Land erosion	1.000	0.673	0.871	0.739
Regeneration problems	1.000	0.810	0.800	0.820

Extraction method: Principal Component Analysis

11.4.3.5 Total Variance Explained

The variations of all the factors are extracted from the analysis along with their Eigenvalues. The percent of variance attributable to each factor, and the cumulative variance of the factor with the previous factors are added on the basis of Eigenvalues. It is noticed that the first factor accounts for 32.872% of the variance, the second one is 27.635% and the third one is 21.686% of Patibania Island. Whereas, in the case of Fredric Island it is noticed that the first factor accounts for 39.195% of the variance, the second one is 29.534% and the third one is 17.886%; and for Henry's Island the first factor accounts for 37.396% of the variance, the second one is 27.747% and the third one is 20.680%. All the remaining factors are not significant. The numbers of rows in this panel of the table correspond to the number of factors retained. In this study, it is requested to retain the three factors, so there are three rows, in which one for each is retained as a factor. The values in this panel of the table are calculated in the same way as the values in the left panel, except that here the values are based on the common variance. The values in this panel of the table will always be lower than the values in the left panel of the table, because they are based on the common variance, which is always smaller than the total variance (Tables 11.11, 11.12, and 11.13).

11.4.3.6 Scree Plot

The scree plot is a graph of the Eigenvalues against all the factors consider for such experiment. The graph is useful for determination of factors that will retain with the experiment. The point of interest is where the curve starts to flatten in the Scree plot for each island. It can be seen that the curve begins to flatten between factors 3, 4, 5 and 6 for Patibania Island, 4 and 5 for Fredric Island, and 3 and 4 for Henry's Island. It is also noticed that, factor 3 has an Eigenvalue of less than 1 so only one factor has been retained for Patibania Island; factor 4 has an Eigenvalue of less than 1 so only

Table 11.11 Total variance explained for seven major factors of Patibania Island

	Initial eigenvalues			Extraction sums of squared loadings			Rotation sums of squared loadings		
Factors	Total	% of variance	Total	Total	% of variance	Cumulative %	Total	% of variance	Cumulative %
Human uses	2.301	32.872	32.872	2.301	32.872	32.872	2.230	31.863	31.863
Fishery development	1.934	27.635	60.508	1.934	27.635	60.508	1.901	27.162	59.025
Hypersalinity	1.518	21.686	82.194	1.518	21.686	82.194	1.622	23.169	82.194
Sediment deposition	0.609	8.694	90.888						
Strom effect	0.502	7.175	98.063						
Land erosion	0.075	1.078	99.141						
Regeneration problems	0.060	.859	100.000						

Extraction Method: Principal Component Analysis

Table 11.12 Total variance explained for seven major factors of Fredrick Island

	Initial eigenvalues			Extraction sums of squared loadings			Rotation sums of squared loadings		
Factors	Total	% of variance	Cumulative %	Total	% of variance	Cumulative %	Total	% of variance	Cumulative %
Human uses	2.744	39.195	2.444	2.744	39.195	39.195	2.444	34.915	34.915
Fishery development	2.067	29.534	2.363	2.067	29.534	68.729	2.363	33.764	68.679
Hypersalinity	1.251	17.876	1.255	1.251	17.876	86.605	1.255	17.926	86.605
Sediment deposition	0.595	8.496	95.101						
Strom effect	0.239	3.407	98.509						
Land erosion	0.079	1.128	99.637						
Regeneration problems	0.025	0.363	100						

Extraction Method: Principal Component Analysis

Table 11.13 Total variance explained for seven major factors of Henry's Island

Factors	Initial eigenvalues			Extraction sums of squared loadings			Rotation sums of squared loadings		
	Total	% of variance	Cumulative %	Total	% of variance	Cumulative %	Total	% of variance	Cumulative %
Human uses	2.618	37.396	37.396	2.618	37.396	37.396	2.357	33.667	33.667
Fishery development	1.942	27.747	65.143	1.942	27.747	65.143	2.066	29.510	63.178
Hypersalinity	1.448	20.680	85.823	1.448	20.680	85.823	1.585	22.645	85.823
Sediment deposition	0.577	8.246	94.070						
Strom effect	0.252	3.598	97.667						
Land erosion	0.135	1.923	99.590						
Regeneration problems	0.029	0.410	100.000						

Extraction Method: Principal Component Analysis

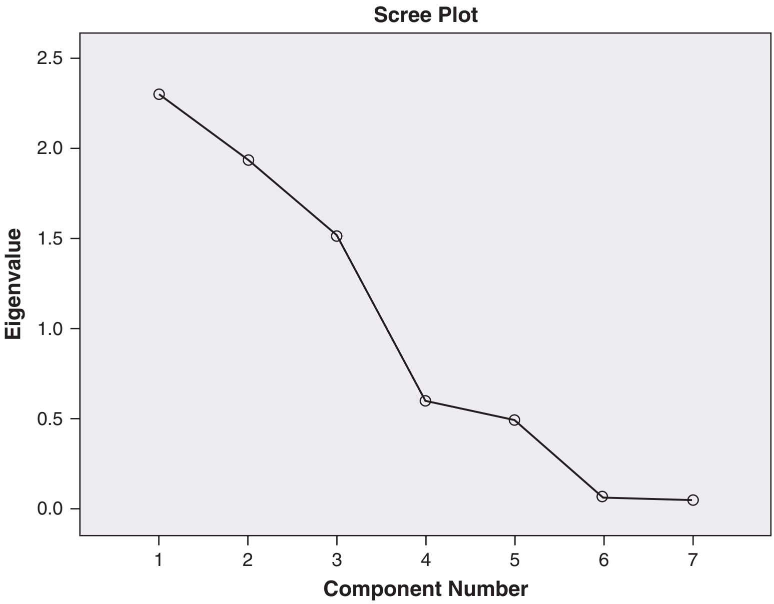

Fig. 11.13 Scree plot for Patibania Island based on total variance explained against seven major factors

two factors have been retained for Fredric Island; and at the same time factor 3 has an Eigenvalue of less than 1 so only three factors have been retained for Henry's Island (Figs. 11.13, 11.14, and 11.15).

11.4.3.7 Component Matrix

The table shows that the three factors are extracted from the loadings of the eight variables. The higher the absolute value of the loading, the more the factor contributes to the variable. The gap on the table represent loadings that are less than 0.5, and this makes the table easier to read for component matrix. In the same case all loading values less than 0.5 are suppressed in this table (Table 11.14).

11.4.3.8 Rotated Component Matrix

The idea of rotation is to reduce the number factors on which the variables under investigation have high loadings. Rotation does not actually change anything but makes the interpretation of the analysis is easier. The rotated component matrix, sometimes referred to as the loadings, is the key output of principal components analysis. It contains estimates of the correlations between each of the variables and

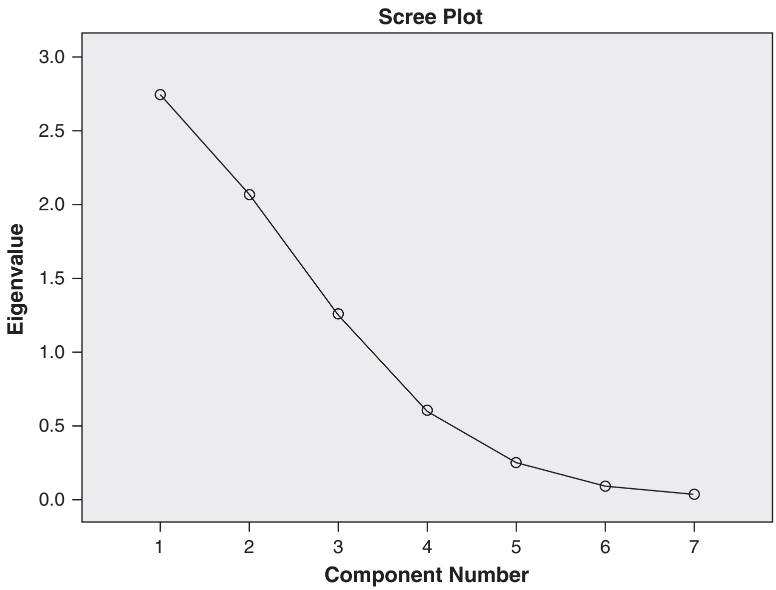

Fig. 11.14 Scree plot for Fredrick island based on total variance explained against seven major factors

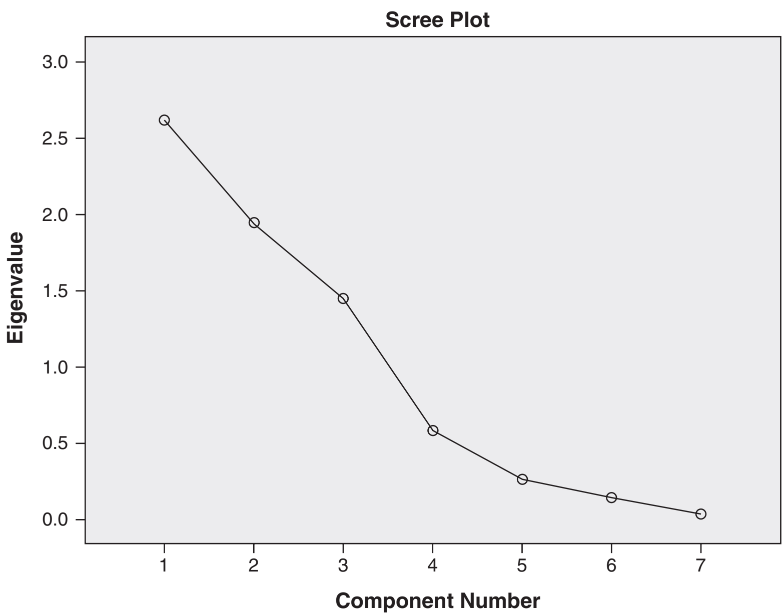

Fig. 11.15 Scree plot for Henry's island based on total variance explained against seven major factors

Table 11.14 Component matrix for seven major factors of three sample islands

	Component								
	Patibania Island			Fredrick Island			Henry's Island		
Factors	1st	2nd	3rd	1st	2nd	3rd	1st	2nd	3rd
Human uses	−0.583		0.512	−0.949					−0.814
Fishery development	0.922				0.958		0.702	−0.508	
Hypersalinity		0.835		0.884				0.896	
Sediment deposition	0.821					0.797	0.897		
Strom effect		0.791		0.848				0.627	0.653
Land erosion			0.568		0.929		0.663		
Regeneration problems			0.796			−0.748	−0.807		

Extraction Method: Principal Component Analysis, three components extracted

the estimated components. Looking at the table below, one can see that land erosion and regeneration process is substantially loaded on Factor (Component) 3 while hypersalinity, storm effect and sediment deposition are substantially loaded on Factor 1 and 2 for Patibania Island. Sequentially those three factors are much more effective for mangrove degradation of this island, and other factors can be used as variables for further analysis. However, for Fredric Island, one can see that sediment deposition is substantially loaded on Factor (Component) 3 while Fishery development, land erosion and hypersalinity are substantially loaded on Factor 1 and 2. Sequentially those three factors are much more effective for mangrove degradation of this island, and other factors can be used as variables for further analysis. In the case of Henry's Island, it is observed that human uses and storm effects are substantially loaded on Factor (Component) 3 while Fishery development, sediment deposition and land erosion are substantially loaded on Factor 1 and 2. Sequentially those three factors are much more effective for mangrove degradation of this island, and other factors can be used as variables for further analysis (Table 11.15).

11.5 Field Evidences in Support of Mangrove Degradations in Sundarban

Areas of mangrove degradations are observed in the coastal wetlands of Southwest Sundarban over a long period through field survey and photographic documentations. Mangroves are degraded along the shorelines, channel banks and around the salt affected surface by the physical processes of erosion, shifting channel banks, over wash sand deposits and increased soil salinity. Damages of mangroves are also recorded in many islands after the storm events (1988, 1989, 2007 and 2009) in the

Table 11.15 Rotated component matrix for seven major factors of three sample islands

Factors	Component								
	Patibania Island			Fredrick Island			Henry's Island		
	1st	2nd	3rd	1st	2nd	3rd	1st	2nd	3rd
Human uses	−0.634			−0.949					−0.833
Fishery development	0.902				0.958			0.935	
Hypersalinity		0.962		0.884			0.534	−0.659	
Sediment deposition	0.917					0.797	0.936		
Strom effect		0.930		0.848					0.875
Land erosion			0.739		0.929			0.828	
Regeneration problems			0.894			−0.748	−0.892		

Extraction Method: Principal Component Analysis; rotation method: varimax with kaiser normalization; rotation converged in five iterations

Sundarban (Fig. 11.16 and 11.17). Historical land reclamation and encroachment of fish farming plots into the mangrove dominated areas are anthropogenic causes of mangrove degradations. In few places of low lying areas of shore fringed islands, the remains of salt water to support the spread of eutrophication by the growth of algal bloom continuing with the mixing of rain water process hinder the growth of mangroves and mangrove regeneration systems. Many salt flats and saline blanks are monitored in the monsoon months (August–September) to identify the algal blooms in the areas of wetland depressions (Fig. 11.16).

11.6 Attempt to Restore and Conserve the Mangrove Forest in Coastal Wetlands (Southwest Sundarban)

Restoration of mangrove wetlands is a challenge at present by generating artificial drainage into the supra tidal flat are a supply of moistures in the evaporative environment. Mangrove dieback process in the areas of tidal drainage loss with a spread of hyper saline tracts and dwarfed growth along the fringe of saline banks reduces the health and density of swampy forest in the deltaic coast. There are other ways in the coastal wetland to restore mangroves for conservation of ecosystems. Many emerged bars and inter tidal flats are utilized for seedling mangroves at present by the foresters in the Sundarban. An attempt is also made to protect mangroves along the margins of embankments, roads, ponds and fish farm plots for achieving surface stability of tidal sediments through social forestry. People's awareness about the significance of mangroves against storm effects and land erosion may protect some areas of wetlands in favorable conditions. The mangrove nurseries should play a

Fig. 11.16 Algal bloom in the low lying depressions of coastal wetlands encircled by mangroves in parts of Henry's Island

vital role in planting saplings in the tidal wetlands as a purpose of agro forestry development in the low lying deltaic coast.

Existing mangrove forests in the buffer areas of the Sundarban Reserve Forest (National Park) need immediate cares and also require the strict implementation of CRZ rules in the coastal zone to protect and avoid conversion from further human uses.

11.7 Conclusions

Mangroves are not only degraded in the buffer areas of reserve forest, they also have been reduced from the inner parts of forests by increased salinity stress around hypersaline patches.

Through the remote sensing studies it is established that both the density and health of mangroves are degraded in between 1989 and 2015. Hypersaline patches are only reduced in the landward sides of mangrove dominated islands after storm effects and river floods. However, the seaward face of islands is affected by the growth of such saline blanks at smaller scales. Loss of tidal drainage, reduction in

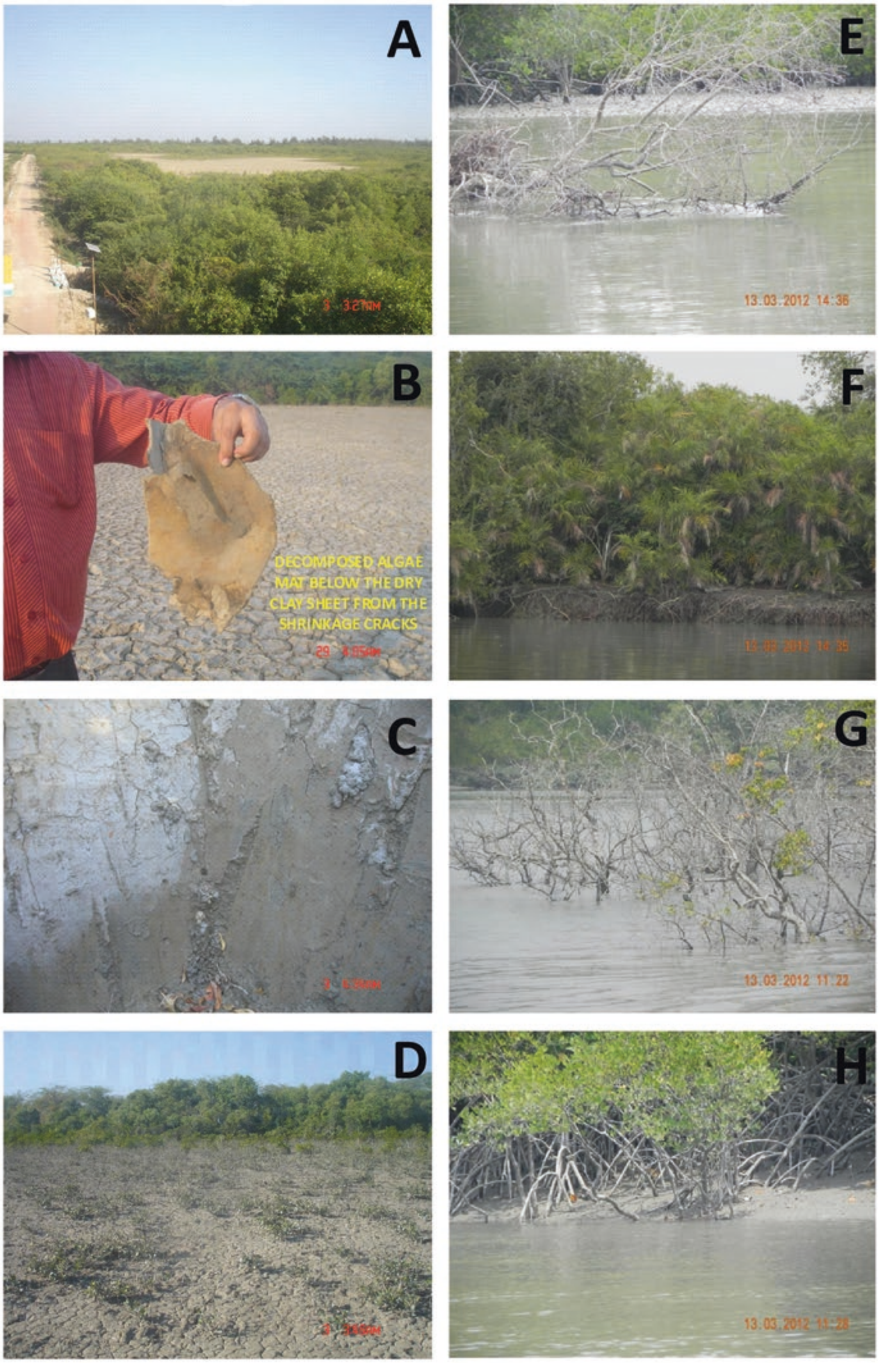

Fig. 11.17 Types of mangrove degradations in the parts of south west Sundarban. (**a**) Formation of the hypersaline patch into the inner part of the forest dominated by mangroves on the Henry's Island; (**b**) the un-vegetated bare surface of the hypersaline patch in the period of dry winter phase. The flake of desiccation mud crack is encrusted with salt layer on the top and a thin filmy layer of

flushing rate of deltaic streams with high evaporation rate, and increased storminess of the sea control the fate of mangrove wetlands by spread of hyper-salinity. Seaward zones of Island, estuarine Islands of younger origin and inner parts of mangrove dominated Islands are mostly affected by degradations.

All the factors and sub-factors are identified with the observations of spatio-temporal characters of coastal wetlands in and around the South western Sundarban. Identification of potential threats of the coastal wetland is possible by the preparation of mangrove degradation checklists. The MDC can provide thus a perspective for management opportunities and challenges to restore the coastal wetlands dominated by mangroves in the Sundarban. The Principal Component Analysis (PCA) for identified multiple factors reveals that hypersalinity, storm effects, land erosion, sediment deposition and fishery development are mainly responsible for mangrove degradations in the coastal wetlands of Southwest Sundarban.

The coastal wetlands of Sundarban dominated by mangroves are potentially threatened by degradations in a multiple way of risk factors. This is a great challenge to the forest department and coastal zone management authorities of India and Bangladesh to achieve the desired conservation objectives for the sea face of Sundarban wetlands in an even more sustainable manner against the upcoming threats with predicted sea level rise in the region. The assessment of geomorphological perspectives of mangrove ecology in regional settings and preparation of mangrove degradation checklists (MDC) can provide ideal tools for identification of potential threats to the coastal wetlands of Sundarban.

The physical effects of sea level rise in the form of storminess of the sea, land erosion at the shore fronts and along the estuarine banks, over wash sand depositional lobes behind the beaches and bars, high evaporation rate and hypersalinity, conversion of wetlands for fish farming plots by local people to adjust with salt water flooding, and spread of eutrophication in the surface depressions of swampy tract are drivers of mangrove wetland degradations in the Sundarban. The zoning of wetland management should not be restricted only in the emerged hyper saline patches or in areas of tidal drainage loss by creating artificial drainage lines for supply of moistures and fresh silts through salt water inflows but, other areas of emerged bars, river flats, buffer areas of reserve forests and emerging areas of salt water flooding incidences also should be considered for restoration of wetlands by mangrove afforestation at present.

Fig. 11.17 (continued) decomposed algae mat into the lower part; (**c**) the pit cutting of a hyper saline patch showing the location of animal burrows into the lower part of vertical profile underlain by salt encrusted top layer. The mangrove swamp was produced in the past when the animal burrows were developed into that layer; (**d**) the die back of mangrove vegetations and conversion of some vegetation into dwarfed condition due to increased salinity of top soil; (**e**) the mangrove trees uprooted from the wetland areas affected by slumping mud banks of the Island fringed tidal channel; (**f**) the loss of mangrove vegetation due to the erosion and retreating mud banks of compact clayey platform dominated by Hental trees *(Phoneix peludosa)*; (**g**) the chunk of forest patch is submerged due to undermining and collapsing process of Island platform fringed with the active tidal channel; and (**h**) sediments removed from the swampy surface dominated by mangrove trees *(Rhizophora sp.)* during storm events may produce damage to the vegetations in the near future

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Chapter 12
Assessment of Anthropogenic Threats to the Biological Resources of Kaliveli Lake, India: A Coastal Wetland

Krishnan Silambarasan and Arumugam Sundaramanickam

Abstract Kaliveli Lake is a coastal lagoon wetlands in the Viluppuram District, near to Bay of Bengal, Tamil Nadu state, Southeast coast of India. The lake is one of the largest wetlands in peninsular India, and is considered a wetland of international importance proposed by International Union for the Conservation of Nature and Natural Resources (IUCN). Kaliveli Backwaters is 12.5 km long and 370 m broad. The average depth is 1.75 m in the high tide. At some place it shows 3.5–4.0 m deep. It covers an area of 3940 acres with a gradient from freshwater to brackish water. It is a semi-permanent, fresh to brackish water lagoon, which empties into the sea through a narrow channel connecting the wetland with the Yedayanthittu estuary to the northeast. This wetland is one of the most significant habitats suitable feeding and breeding ground for migratory birds. More than ten thousand migratory birds are visiting this wetland every year. At present, this wetland is threatened by many anthropogenic activities such as infringement from agricultural lands, wildlife poaching, loss of the surrounding forests, increased saltpan and aquaculture farming and recreational activities. The present study focuses on identification and assessment of the various threats faced by the Kaliveli wetland and we affirm following suggestions regarding the adequate measures for its conservation and management.

Keywords Kaliveli wetland • Anthropogenic threads • Biological resources • Salt pan

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12.1 Introduction

Wetlands, meeting many crucial needs for life, are an essential fraction of human civilization. Wetland plays some significant roles in regional ecosystem, such as regulation of the climate, cleansing of the environment and balancing of the regional water. The wetland provides a critical habitat to a large number of flora and fauna. Based on several estimates, the extent of the world's wetlands is generally thought to be from 7 to 9 million km^2 or about 4–6% of the land surface of the Earth (Mitsch and Gosselink 2000). Ramsar Convention has described wetlands as the "regions of marsh, fen, peat land or water, whether natural or artificial, permanent or temporary with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six meters".

The wetlands around the world are receiving increased attention since these contribute to a healthy environment in many ways. They hold water during summer seasons, thus help to maintaining the water table high and relatively stable. However, during monsoon seasons, they mitigate stream and trap the suspended solids and associated nutrients. They buffer stormy seas, reduce shoreline erosion and take up intemperance nutrients, which help to reduce eutrophication and contribute to maintain the condition of mangrove habitat. Mangroves are highly productive components and an excellent natural renewable resource; it also protects coastal areas from sea erosion and forms the violent effects of cyclones and tropical storms (Kathiresan 2003). It serves as considerable repositories of aquatic biodiversity (Prasad et al. 2002). Most of the coastal finfish and shellfish species spend their early life stages in mangrove habitats. They provide shelters, foods and nesting sites for migratory birds and reptiles. Most of the migratory birds have adapted to life in or near mangrove wetlands water in many ways.

12.2 Distribution of Wetlands in India

Almost 86% of the estimated total natural wetland area is found in tropical, subtropical regions and boreal regions of the world. Whereas, temperate zone wetlands contribute only about 14% of the world's total natural wetlands. The wetlands in India are distributed in various ecological regions, totally about 27,000 wetlands are there of which 4000 are coastal wetlands. The coastal wetlands occupied about 6750 km^2, and are mainly dominated by mangroves (4461 km^2). Indian mangroves make up 3.1% of the total global; it spread alongside the east (59.6%), west (27%) coasts of India and the Andaman and Nicobar islands (13.3%) respectively.

India, by virtue of its geographical extent, varied geography and climatic conditions, supports a rich diversity of inland and coastal wetland ecosystems. Although, the significance of wetlands has been known for a long time, their role in maintaining ecological balance is still lesser understood. The Ramsar Convention of IUCN held during 1971 in Iran raised global awareness of the conservation and management

Fig. 12.1 Showing the Kalveli Lake, the Uppukali creek and the Yedayanthittu Estuary

of wetlands (Chopra et al. 2001). Ministry of Environment and Forests, Government of India, has identified 94 important wetlands in India, of which three are in Tamil Nadu state viz., Point Calimere, Pallikaranai and Kaliveli (Kazhuveli) wetlands.

The Kaliveli watershed is the second major brackish water lake in South India it is a semi-permanent, fresh to brackish wetland that drains 776 km^2 along the Coromandel Coast. It is located 18 km towards the North of Pondicherry in the state of Tamil Nadu (South India) with coordinates ranging between: 11°55′N, 79°35′E and 12°10′N, 79°55′E (Fig. 12.1).

It is considered a wetland of international importance proposed by International Union for the Conservation of Nature and Natural Resources (IUCN). Kaliveli Backwaters is 12.5 km long and 370 m broad. The average depth is 1.75 m in the high tide. At some place, it shows 3.5–4.0 m. deep. It covers an area of 3940 acres with a gradient from freshwater to brackish water.

The Kaliveli wetland ecosystems have often been seen as unproductive and stinking, and thus removed to make land areas for human settlements, infrastructure development, aquaculture, agriculture saltpan and other industrial activities and tourism development. Owing to their numerous uses, the people have oppressed the mangrove forests indiscriminately and caused extensive damage to the mangrove ecosystem, resulting in ecological decline in the Kaliveli wetland ecosystems. Hence, this article highlights the threats, conservation and management statuses of Kaliveli wetland ecosystems. Mis-management, changing socio-economic and policies have led to degradation of these rainwater-harvesting structures, thus altering the properties of the wetland itself.

12.3 Ecological Importance of Kaliveli Wetland

This watershed is an ancient representation of what was created 1500 years ago. Indeed, between the sixth and the tenth century of our era, a system of tanks was prepared in Tamil Nadu to supply the water to local population (D'Souza et al. 2007). The watershed landscape is largely agricultural and composed of a complex network of 238 tanks and their channels (Woistencroft et al. 1987; Gopinath and Srinivas 2004; Ramanujam and Anbarasan 2007; Bhalla and Prasadm 2009). These tanks not only to supply of water for irrigation and domestic purposes as they renew the ground water table during the monsoon period, although serve many other purposes to local populations such as cattle grazing ground, timber wood, roof thatch with the growing reeds and fishing. In addition, as the tanks are inter-connected with each other by a means of channels, all the superfluous water can flow from one tank to another, preventing the floods and the water loss during the rainy season.

In the Kaliveli catchment area, the water level fluctuates due to the precipitation, with the highest level during the end of the northeast monsoon. However, in case of little rainfall, it dried out fully during the summer season (Woistencroft et al. 1987). These regions have a wide variety of sedges and grasses, interspersed with barren sandy areas and muddy margins. Filling of the lake with the fresh water in November results in germination of numerous aquatic plants. *Enteromorpha intestinalis* is particularly common, among other algal species in the brackish areas. There are extensive reed beds and sedges in the less saline areas. A few straggly mangrove bushes are all that remain of what must once have been a large mangrove forest (Pieter 1987). The wetlands are situated amidst an agricultural land and an arid thorn scrub and it is broadly divided in three distinct zones.

12.3.1 Kaliveli Flood Plains

The water catchment area for this Kaliveli wetland (12°5′–12°3′–12°9′N and 79°47′–79°51′–79°53′E) is about 740.89 km^2 the south end is begin with the Auroville plateau, Marakkanam to the north and Tindivanam and beyond to the north west. Lowe-Mc Connell (1987) and Keddy (2000) delineated that it is floodplain. The aerial view of this water body is looks like petal shaped, there are several reservoirs present in the catchment area. All the tanks gets fill up with abundant precipitation of southwest monsoon and attains the highest level at the end of northeast monsoon. But, it totally dried out during the summer seasons.

12.3.2 Uppukali Creek

It is a narrow channel (12°9′N – 70°53′ E and 12°12′N – 79°56′ E) which connecting the floodplain into the Yedayanthittu estuary. Due to its estuarine links, the nature of this region is divergent from that of Kaliveli flood plain. There is stable inflow of water from the estuary throughout the year, but the water level and its quality is influenced by the waves and tidal actions (Ramanujam 2005).

12.3.3 Yedayanthittu Estuary

Yedayanthittu estuary (12°12′N – 12°15′ E and 79°56′N – 80°0′ E) situated about 3 km to the northeast of this wetland. This estuary has large areas of inter-tidal mudflats, but only small relicts of the once extensive mangrove forests now remain. There are some 500 ha of saltpans alongside the estuary immediately to the north of the Marakkanam road bridge across the channel from Kaliveli wetland. Aforetime 25 years ago, the entire area was heavily forested, but at present most of the forest area has been removed, the reservoir and the estuary are currently bounded with cultivation and scrubby thorn woodland. There are some stumpy sand dunes adjacent to the channel linking the tank to the estuary.

12.4 Diversity of Flora and Fauna

Kalivelli wetland is a haven for hundreds of species of flora and fauna, Kalivelli wetland nurtures several species of birds, fish, reptiles and mammals. The common aquatic vegetation plants in the study area are *Aponogeton natans, Eichhornia crassipes, Hydrilla verticillata, Limnophyton obtusifolium, Monochoria vaginalis, Vallisneria spiralis, Aristida adscensionis, Chloris barbata, Chloris Montana, Polygala arvensis, Lindernia crustacean, Scoparia dulcis, Waltheria indica, Acacia nilotica, Alternanthera sessilis, Bacopa monnieri, Coldenia procumbens, Cyperus distans, Eclipta prostrate, Heliotropium indicum, Hygrophila angustifolia, Ludwigia perennis, Phyla nodiflora, Polygonum barbatum, Typhya angustata, Prosopis juliflora, Barringtonia, Acacia nilotica*, and *Avicennia marina* (Ramanujam 2005).

Information regarding the Ichthyofaunal diversity of this wetland is still very meager; Ramanujam (2005) reported 42 species of fishes representing 25 families and 09 orders. Out of these 42 species, 6 were confined to the flood plain, 19 were estuarine and 17 occurred in both floodplain and creek. Both anadromous and catadromus species were present. Balachandran (1994) and Perennou and Santharam (1990) thoroughly studied the avifauna richness of this wetland. About 179 species of birds have so far been recorded in this wetland. Of the 179 species, 30 are shore birds and waders, while 13 species are ducks. During March–April, the wetland attracts pelicans, herons, egrets, storks and ibises. Besides these the wetland serves as an important corridor for the migratory birds they visit the Point Calimere Bird Sanctuary during winter. Large congregation (in thousands), of wetland birds can be seen from October to March, since the ecological conditions of the wetland during this period are highly suitable for the migratory birds. Nevertheless, this wealthy ecosystem has presently come under threat due to land encroachments, increasing shrimp farms and other anthropogenic activities. Because of this, there is an urgent need for the government to act and protect this wetland to ensure that the wetland ecosystem remains a safe haven for its diverse flora and fauna.

12.5 Potential Threats

The Kaliveli wetland has been converted to intensive agriculture, with few enclaves of forest remaining. As a result, of inhabitant's pressure, half of this eco-region's mangrove forests have been cut down to supply fuel wood and other natural resources. Human activities, including settlements in newer areas, indiscriminate use of natural resources and diversion of water have caused immense damage to the delta. Significant losses have been resulted from its conversion, threats from agricultural, industrial and various urban developments. Incidents of industrial wastes disposal in and around the lake have also been reported. The villagers living around the areas they engaged themselves in fishing and related activities for their income and utilize the grass and mangrove for firewood, fodder and building purposes. Salt panes and shrimp farming are on the increase in the brackish part of the lake.

12.5.1 Human Interface

Nearly 20 small and medium sized hamlets are located in this area with a meager population of around 2,500. Many of these people in these areas depend upon the Kalveli reservoir and Yedayanthittu lagoon for their sustenance in one way or other. The economic dependence is small scale, as they collect reeds and sedges to make some household items, fishing for their own consumption as well as to sell in the local market. Continues rising intrusion of land for various commercial activities is one of the major problems threatening the existence of Kalivelli. Urbanization in the region for its part also exerts a lot of pressure on the wetland.

12.5.2 Salt Pan

The salt marshes and mangrove regions around these areas are converted into salt-pans. Even some of the fresh water regions of the lake is being converted into salt-pans (Figs. 12.2 and 12.3a, 12.3b, 12.3c, 12.3d).

This alter the salinity gradient of the soil, which also influence the flora and fauna of this wetland ecosystem. Most of the salt pans are located in this regions are belonging to the Government. Owing to extended use the ground water has become saline. Most of them, they discharges partially or untreated wastewaters are into wetland ecosystem which leads to rise of biological oxygen demand and turbidity of the water also increases due to the organic particulate materials which conse-

Fig. 12.2 Showing the dense mangrove vegetation area

quences in fall of dissolved oxygen levels. This leads to the mortality of aquatic life's.

People were involved in salt pan construction as basic activities in salt using solar energy. The salts pans located nearer to mangrove forests areas hinder mangrove trees regeneration due to reduce the fresh water flow and rise of soil salinity. The reduction in fresh water and tidal water inflow increases the salinity of these areas, resulting in poor germination, growth and regeneration of mangroves (Fig. 12.4a–c). Which results, formation of unpleasant environmental conditions that damage the growth, regeneration of mangrove ecosystem. In addition, people reside in the salt pan areas utilizing the mangrove trees to build huts for store up salt and also they are using as firewood for cooking lunch at salt pan areas. These activities led to the decline of both mangrove vegetations and its ecosystem.

12.5.3 Agriculture and Aquaculture

Agricultural runoff water, contaminated with many pesticides and fertilizers used by the farmers to enhance the yield, is being mixed in Kaliveli watershed, thus distressing its fishery leads to affect the livelihood of fishermen. Most of the agricultural and other marsh lands are being transformed to aqua farms (Fig. 12.5).

People living in the surrounding villages started the shrimp farms and their area have increasing rapidly. The tidal waters from the Bay of Bengal flow up to them. Hence, the subsurface water in these region has become saline.

Fig. 12.3a Newly constructed saltpans

Fig. 12.3b Destruction of shell beds for salt pan construction

Fig. 12.3c Salt pan under practice

Fig. 12.3d Salt pan during harvest

Fig. 12.4 (**a–c**) Hindering the mangrove trees regeneration

Fig. 12.4 (continued)

Fig. 12.5 Aqua farm near to Uppukali Creek

A shrimp farm effluent poses another threat to the water quality, soil and ecology of the wetland. Aquaculture and its effluents in wetland areas is an significant problem on regeneration and survival of mangrove seedlings. Similarly, waste water discharged from aqua farms contains excess nutrients which rise the plankton population which serve as the potential food source for many species of coastal fish and invertebrates. However, increase of nutrient wastes leads to toxic algal blooming that may cause eutrophication.

Stalinization of soil attributable to shrimp farming has also been reported in Kaliveli watershed and adjacent areas. Formation of sand bar proceed as a barrier for immigration and emigration of fish and other fauna for completion of their life cycle, thereby contributing to the list of threats affecting the avifaunal diversity of the lake.

12.5.4 Birds Poaching

Kalivelli wetland is a distinctive birding hotspot; this reservoir forms a important wintering site for most of the migratory birds. The availability marshy grounds, mangrove vegetations, large verity of fishes and other food items attracts a diversity of migratory birds to visit Kalivelli during winter (Fig. 12.6a, 12.6b).

Several birds coming to Kalivelli wetland from Siberia for winter and some of them halt here on their way to Point Calimere and Sri Lanka. The migratory period extends between October to March depending upon the availability of water. According to 2004 assessment of IBCN and Bird Life International, Kalivelli supports more than 20,000 birds every year. The lake has been a feeding ground for

Fig. 12.6a A view of agriculture field in Kaliveli wetland with many migratory birds

Fig. 12.6b Migratory birds resting in salt pan

the longest distance migrants from the cold subarctic regions of the Central Asia and Siberia. During the migratory season, as the number of birds increase, poaching and hunting also increases. In migratory season several amateur hunters comes to this wetlands from the adjoining areas. Most of them are tribal people are causing wounds and kill number of birds with their regular operation to catch them for food. During night times, they trap the birds either with plastic wires or with thin wires. After falling in traps, the birds strive to escape and undergo injuries. It is most common to see tribal selling birds in the local markets (Fig. 12.7a, 12.7b). There are a number of eateries in the nearer towns and villages they have birds items in their menus to attract customers.

Many of these species have been listed in the IUCN Red List under various categories, viz., *Anguilla bengalensis* as endangered species, *Hoplobatrachus tigerinus* and *Pelecanus philippensis* as vulnerable, etc. (Ramanujam and Anbarasan 2007). The riparian regions linking the water tanks, although provide potential corridors for the movement and dispersal of these species, but are themselves endangered owing to agricultural encroachments, urban development, destruction of drainage channels and absence of regular maintenance (filling up of tanks and channels by sedimentation, collapsing of bunds), etc. (D'Souza et al. 2007).

There has been a controversial suggestion to develop Kaliveli as a tourist spot focusing on conservation and a bird sanctuary. Although, the Government of Tamil Nadu notified the Kaliveli wetland as reserve land to create a bird sanctuary under Tamil Nadu Forest Act 1882 of Section 26 dated 16.4.2001, but not much progress has been made afterwards, except the declaration.

Fig. 12.7a Showing poaching of some rare species (Darter, Pelican, Grey heron, Open bill stork, Painted stork)

Fig. 12.7b Black headed Ibis trapped by poachers are brought for sale at roadside market

12.5.5 Cattle Grazing

Grass covers the lakebed and grazing starts once the water level begins to retreat. The paucity of fodder drives more than 30,000 livestock into the lakebed. In many villages Dalits are entrusted with the grazers' job. They send their cattle or sheep from each house at morning and ensure that they return in the evening. Cattle graze in the peripheral region of mangrove wetland in the monsoon season at that time of new seedlings are budding up, and growth of young mangroves. Cattle grazing at during that time leads to poor regeneration and poor growth of mangrove vegetation in the grazing areas. Hence, the intensive study is required to reduce grazing pressures on the wetland.

12.6 Conclusion and Recommendations

Impact assessment is the necessary requirement for various management and planning activities on a regional or global level. It has assumed greater importance in view of the shrinkage and degradation of wetlands. Significant impact was observed in Kalivelli wetland ecosystem due to various anthropogenic activities. The United Nations and Ramsar Convention very clearly linked the importance of wetland for the agriculture production and livelihood. It is believed that Kaliveli watershed provides an ideal site for a model study of integrated watershed development. Indeed Kaliveli ecosystems get adjusted to moderate disturbances' that humans have been causing over centuries; an equilibrium or stabilization gets achieved. The management plans for Kaliveli should take into account the diverse uses made by villagers, of this wetland. Traditional activities such as fishing, reed cutting, grazing, desalting etc., if periodically monitored to prevent overuse, can cause little disturbance to the lake.

Overall observation indicated that aquaculture and salt pan has contributed to the major degradation of this wetland ecosystem. The financial benefits of aquaculture form, specifically foreign exchange earnings and provision of employment, are highly important to the Indian economy, but there is a need to minimize further its social and environmental impacts. A number of policy measures are recommended to increase enforcement and to decrease the environmental and social costs of shrimp aquaculture in India. These measures have a focus on mangrove protection; additional measures would be required to address other environmental and social issues.

12.6.1 Legal Status

There are number of laws within the constitution of India that can be directly applied in protecting Kalivelli wetland ecosystem. Even now, the Kalivelli is not declared as a protected area. It is unfortunate that a water tank with such importance does not fall under the control of department of environment and forests. India is a signatory of the Ramsar Convention for protection of wetland ecosystems, which it has approved. Protecting Kalivelli could be an important starting point. To protect the wetland from habitat loss, degeneration and alteration, the following approaches are suggested.

(1) regular observation of the wetland by using different modern systems, (2) alternative livelihood options to those poor people which are mainly depend upon this ecosystem, (3) proper enforcement of the policies, (4) increasing afforestation around the lake area, (5) minimizing the encroachment around the lake area, (6) eradication of invasive species, weeds and algal blooms at regular interval, and (7) banning illegal fishing and poaching in the catchment area. (8) Several organizations including French Institute at Pondicherry, Pondicherry University, and NGOs such as COPDANET (Coastal Planning and Development Action Network) have recommended that Kaliveli should be converted into a bird sanctuary. (9) Re afforestation of mangrove vegetation has also been recommended, (10) the wetland and the entire watershed have also been recommended to be designated as a Biosphere Reserve under the UNESCO Man and Biosphere (MAB) Programme. (11) Development of an educational programme to demonstrate to the local people the long term benefits of the management of Kaliveli. (12) A detailed study on the vegetation structure and dynamics of the lake is necessary. (13) Planting of trees around Kaliveli would provide nesting habitat to water birds. (14) The research and academic bodies of this area should also make efforts for continuous monitoring of this wetland.

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Chapter 13
Egyptian Nile Delta Coastal Lagoons: Alteration and Subsequent Restoration

Ayman A. El-Gamal

Abstract Wetlands and coastal lagoons are valuable and sensitive environments as recognized by Ramsar Convention. Egypt has many forms of wetlands. Mariout, Edku, Burullus and Manzala are considered as the most important lagoons and wetlands in the Egyptian Mediterranean coastal area. The Egyptian Ministry of Environmental Affairs (MENA) updated the National Biodiversity, Strategy and Action plan (NBSAP) for the years (2015–2030). One of the goals of this strategy is to minimize the rate of wetlands loss by 50%. The challenges of the Egyptian coastal lagoons were summarized to include pollution, water deterioration, lake of management, reduction of area, aquatic plants, habitat loss, climate change, siltation of the outlets, eutrophication, awareness, illegal fish practice, over fishing and decline of fish yield. Egyptian Environmental Affairs Agency (EEAA) initiate monitoring program to check the water quality of the coastal lakes and its adjacent marine area. The northern lakes have discussed to describe their morphology, environmental status & stress and their water quality according to recent measurements of the EEAA. Many efforts have been done for environmental conservation and socioeconomic development to the coastal lagoon such as El-Burullus Lake is protected by the Egyptian Prime Ministerial Decree 1444/1998 and is a Ramsar Site. Edku and Mariout Lagoons still need more efforts to environment conservation for sustainable development. In order to improvement of coastal lagoons resilience, periodical monitoring of water quality and pollution sources, quantities and type of discharges from these sources became perquisite to determine its impacts of these lakes.

Keywords Coastal wetlands • Lagoons • Nile Delta • Water quality • Environment resilience

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13.1 Introduction

Wetlands can be defined as the areas of marsh, fen, peat-land or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed 6 m. It includes also areas, which may incorporate riparian and coastal zones adjacent to the wetlands, and islands at low tide lying within the wetlands (Fahmy et al. 2007). The key is the presence of water for some significant period of time, which changes the soils, the microorganisms and the plant and animal communities, such that the land functions in a different way from either aquatic or dry habitats (Barbier et al. 1997).

Wetlands are amongst the Earth's most productive ecosystems (Barbier et al. 1997). Wetlands and coastal lagoons are valuable and sensitive environments and their important role has been widely recognized at the international level within the framework of the Convention on Wetlands, which known as Ramsar Convention. The Ramsar Convention is an international treaty for the conservation and sustainable use of wetlands (Ramsar 2014). Historically, many wetlands have been treated as wastelands and drained or otherwise degraded. The Ramsar Convention on Wetlands of International Importance was created to promote the conservation of wetlands and their wise use and management. Ramsar is promoting new methods of economic valuation to demonstrate that wetlands are valuable and should be conserved and wisely used (Barbier et al. 1997).

Wetlands have an imperious bio-ecological function through maintaining everlasting fit place for distinctive groups of biodiversity, especially migratory water birds (EEAA 2016a). Wetlands have been described as "the kidneys of the landscape", because of the functions they perform in the hydrological and chemical cycles, and as "biological supermarkets" because of the extensive food webs and rich biodiversity they support (Mitsch and Gosselink 1993).

Scott (1989) defined 30 groups of natural wetlands and nine manmade ones. However, It can be categorize into five broad wetland systems:

- Estuaries – where rivers meet the sea and salinity is intermediate between salt and freshwater (e.g., deltas, mudflats, salt marshes)
- Marine – coastal water not influenced by river flows (e.g., shorelines and coral reefs)
- Riverine – land periodically inundated by river overtopping (e.g., water meadows, flooded forests, oxbow lakes)
- Palustrine – where there is more or less permanent water (e.g., papyrus swamp, marshes, fen)
- Lacustrine – areas of permanent water with little flow (e.g., ponds, kettle lakes, volcanic crater lakes)

Wetlands continue to decline globally, both in area and in quality. As a result, the ecosystem services that wetlands provide to society are diminished. Contracting Parties and their policymakers are urged to take immediate action to meet the

Ramsar Convention's objective to stop and reverse the loss and degradation of wetlands and services to people (Ramsar 2015).

13.2 Wetlands in Egypt

Wetlands are some of Egypt's most important habitats in terms of biodiversity. Wetlands in Egypt can be classified according to its location as demonstrated by Baha El-Din (2002) as coastal wetlands either in the Mediterranean or Red sea and as inland wetlands. Concerning the Mediterranean coastal wetlands, the most important are the six major coastal lagoons on the Mediterranean: Bardawil, Malaha, Manzala, Burullus, Edku and Mariout. The remainder of the Egyptian Mediterranean coast is of rather limited importance. The Red Sea coastal habitats and wetlands include mudflats, reefs, mangroves and marine islands. There are six major inland wetland areas in Egypt: the Bitter Lakes, Wadi El Natrun, Lake Qarun, Wadi El Rayan Lakes, Nile River and Lake Nasser. In addition, there are many smaller wetlands dispersed in the Nile delta and valley, and in oases in the Western Desert (Baha El-Din 2002).

Wetlands in Egypt can be recognized into 14 generic types (MedwetCoast 2004):

1. The Bardawil-Manzala-Burullus-Idku, Mareotis-Mallah of Port Fouad. These are lakes of North Egypt of different origins and ecology. They all are bird sites and have access to the Mediterranean.
2. The Matrouh lagoons, close to the Mediterranean, and receive their water through the narrow limestone barriers.
3. The Moghra-wadi Natrun Lakes. These are shallow depression in the northern sector of the West Desert. They receive water from underground seepage. It is the eastern lobe of the Qattara Depression and is lied on the western outskirts of the Nile Delta.
4. The Qarun-Wadi Rayan lakes. These are two depressions of the West Desert. Lake Qarun receives drainage water of the Fayoum area. Evaporation makes it hypersaline. Wadi Rayan depression was connected to the agriculture drainage system of the Fayoum Governorate.
5. A number of small lakes scatter in the Delta and its outskirts. Abasa in the east and Dahshoor in the west. Abasa accommodates a fish-farming research and training center. Dahshoor become reed swamps.
6. The Moses Springs site in south-western Sinai form patches of saline moist soil with small ephemeral ponds, reed and rush swamps.
7. The main channel of the Nile between Aswan and Cairo embraces numerous islands.
8. Lake Nasser is the Egyptian part of the Aswan High Dam reservoir-lake (496 km long – total area 5000 km^2).
9. The Mediterranean coast outside the Delta provides little room for developed littoral salt marches.

10. The Red Sea and the Gulf of Suez has extensive littoral salt march formations all along the coast.
11. The Red Sea and the Gulf of Aqaba coastal lands have extensive patches of mangroves (c. 400 ha). It is an elaborate ecosystem with rich biota.
12. The Red Sea coral reefs form long stretches parallel to the shoreline, and comprise diversities of coral species with associated biota.
13. The Red Sea islands with the Egyptian exclusive economic zone comprise coral formation and volcanic islands.
14. The Suez Canal system includes a small Lake Temsah and a larger, further south, Bitter Lake. The whole system connects the Red Sea and the Mediterranean, and provides a causeway for migration of biota

Cataudella and his group (2015) divided the Mediterranean coastal area of Egypt into three sectors, according to the coastal lagoons presence and typology. The western sector from west of Salloum to Alexandria, the central area from Alexandria to Port Said and the third sector from Port Said to Rafah. The second and the central area contains the most important lagoons and wetlands: from west to east Mariout, Edku, Burullus and Manzala as shown in Figs. 13.1 and 13.2 (Cataudella et al. 2015).

The Egyptian Mediterranean coastal area has different categories of water, which mainly affected by Nile water and through coastal lake outlets and various drainage effluents. These effluents continuously discharge water with a complex of varied waste materials into the sea (Hamza 2006; El-Gamal 2016).

The Nile Delta has different kinds of water such as:

1. Fresh water as in the Nile River branches (Rosetta and Damietta).
2. Estuarine water, which is mixed between fresh and marine water.
3. Brackish water as in the northern lakes in the Nile Delta regions such as Edku, Elburullus, Marioute and Manzala Lakes.
4. Marine Water as the water of the Nile Delta coastal area.

The Ministry of Environment of Egypt updated the National Biodiversity, Strategy and Action plan (NBSAP) for the years (2015–2030) (MENA 2016). This report discuss the wetlands habitats in Egypt. Strategic Goal number 2 concerning the sustainable use of natural resources stated that by 2021 the rate of wetland loss is reduced by 50%, water efficiency in farming is improved by 50%, and BMP in development of inland water ecosystems are available to policy makers. The main topics of the National Plan and programs of action are as following (MedwetCoast 2004; UNEP-MAP RAC/SPA 2009):

1. Establishment of the national council for wetlands.
2. Survey of wetlands in Egypt.
3. Selection of sites for wetland nature.
4. Research program in representative wetland sites.
5. Program of studies for formulating management plans for each of the selected sites.
6. Program for materials for education

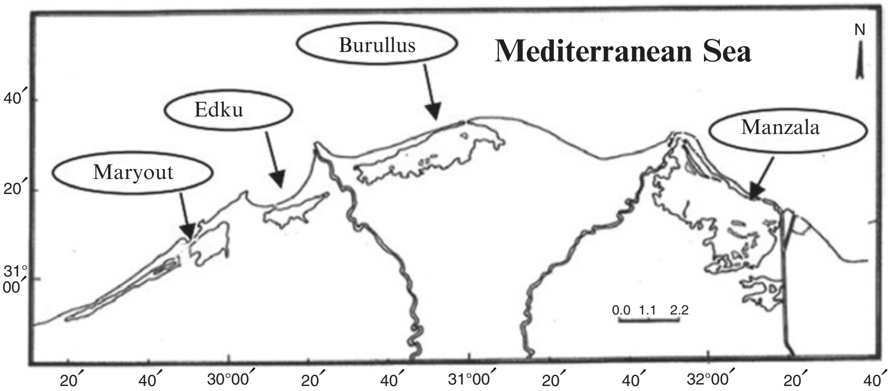

Fig. 13.1 The Mediterranean coastal lagoons in Egypt. The brackish water lagoons under investigation are located in the Nile Delta coastal area Maryout, Edku, El-Burullus and Manzala Lakes

7. Establishment of national wetland databank.
8. Program for training man power capacity building.
9. Program for inventories of cultural heritage and indigenous knowledge of wetlands of Egypt.
10. Consolidated national law for wetlands.

Egypt's wetlands are subject to a variety of human induced threats that are leading to the degradation of this valuable national resource. The main threat facing Egyptian Northern coastal lakes and their vulnerability to climate change is habitat loss and degradation driven by significant reduction in area. This reduction is a result of agriculture and settlements, abstraction of water for irrigation, coastal erosion, water pollution, over fishing and illegal fishing activities, introduction of alien species, spreading of aquatic plants and the blockage of their connections with the sea. Different kinds of pollution among various lakes from untreated or partially treated industrial and domestic wastewater in addition to the agricultural wastes, which loaded with eutrophication parameters (N and P), fertilizer, pesticide and herbicide residues.

The Egyptian Ministry of Environmental Affairs stated that wetlands in Egypt are facing many threats that lead to their degradation (MENA 2016), such as:

- Excessive expansion in scooping coastal lakes for implementing development projects.
- Intrude of different kinds of pollutants discharged to the coastal lakes from domestic, agricultural and industrial wastes generated from cities and villages located along coastal lakes such as El-Burullus, Edku and El-Manzala that affect biota and decrease services and resources of these lakes.
- Accretion and sand creep are natural threats exposed the wetlands, which may play an important part for siltation of the outlets of the lakes.

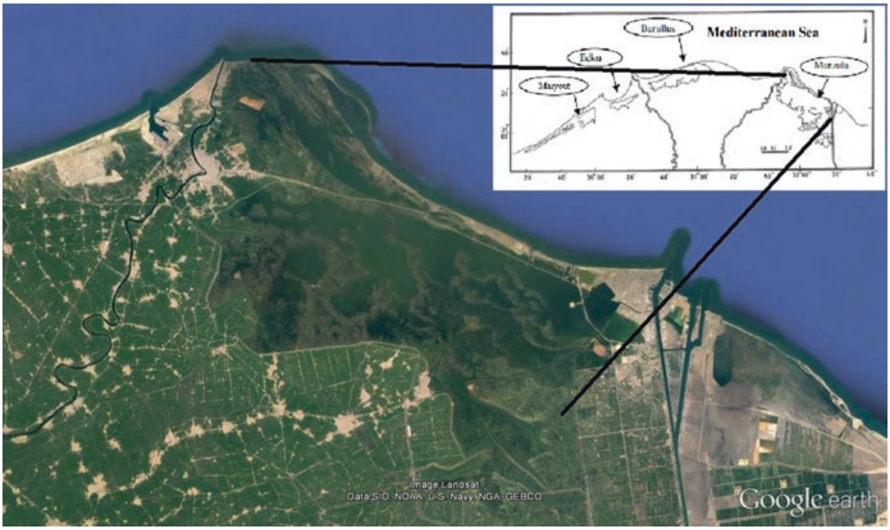

Fig. 13.2 Google earth image of El-Manzala Lagoon during 2016

- Climate change, mainly sea level rise.
- Overgrazing and erosion of vegetation coverage, in addition to drought episodes associated with low rainfall and poor management of rangelands.
- Mines spread along large areas of North Coast and Western Desert, left from the II World War in El-Alamein area, this area (more than quarter million feddans) is suitable for agriculture.

In this chapter, we will discuss the Nile Delta lagoons, which are the most important Mediterranean coastal lagoons Manzala, Burullus, Edku and Mariout. Morphology, environmental parameters and water quality of the four lakes under investigation with its adjacent Mediterranean water are discuss. Vulnerability and environmental stress of these wetlands will be presented with its conservation and sustainable development.

13.3 Morphology and Environmental Parameters

Lagoons are an integral part of the coastal landscape and have peculiar ecological conditions that characterize its ecosystems (Cataudella et al. 2015). It recognized by significant environmental heterogeneity, highly resilient, productive ecosystems and efficient trophic transfer (Cataudella et al. 2015). The delta contains a network of irrigation canals and drainage systems. Drainage waters finally flow into several

Table 13.1 State and water characteristics of Egyptian coastal lagoons during 2008

Lagoon	Area (ha)	Water sources	Water characteristics
Edku	8000	Drainage + sea	Freshwater- brackish
Burullus	41,000	Drainage + sea	Freshwater- brackish
Manzala	78,000	Drainage + sea	Freshwater- brackish

Abdel Rahman (2008)

wetlands and lagoons that face the Mediterranean Sea. These lagoons contribute significantly to the economics, environmental aspects and fishery production of the country (Cataudella et al. 2015).

The three delta lagoons (Edku, Burullus and Manzala) have almost the same pattern of water salinity. Each of them receives large volumes of drainage water and has one or more narrow connections to the sea (Table 13.1) (Cataudella et al. 2015). During the year except at the winter enclosure period, freshwater enters the lagoons in order to keep the water level in the lagoons higher than the sea level and preventing the intrusion of seawater. All lagoons are shallow water bodies with average depth ranging between 0.8 and 1.0 m (Shaltout and Khalil 2005).

13.4 The Northern Lakes: Manzala, Brullus, Edku and Mariout

The Northern Lakes under investigation are Manzala, Brullus, Edku and Mariout. These lakes are in contact with the Mediterranean Sea and have a direct or indirect relation with the Nile River. These lakes are natural environment for fauna and flora and a home for emigrant birds.

13.4.1 Manzala Lagoon

13.4.1.1 Site Description

Lake Manzala, the largest of Egypt's Mediterranean wetlands. Located within the boundaries of five governorates: Dakahliya, Damietta, Port Said, Ismailia and Sharkia, and bordered from the east by Suez Canal and from the west by Damietta Branch and the Mediterranean from the north (EEAA 2010). The importance of this lake is not only dependent on its size but also on its high productivity (Zakaria et al. 2007). It is shallow lake with average depth 1.3 m and located in the northeastern corner of the Nile delta. They reported that the area had drastically reduced from 1100 km^2 in 1973 to 1052 km^2 in 1984 and finally became 720 km^2 in 2003. They attributed this shrinking to the reclamation activities and to the construction of the coastal highway (Hereher 2014). It is predicted that existing reclamation plans will reduce its area further. Manzala is generally rectangular, about 60 km long and 40 km wide (Baha El-Din 2002). Its coastal line is about 293 km, with maximum width of 30 km (EEAA 2010). It is separated from the Mediterranean Sea by a narrow sandy fringe (Shakweer 2005), through which it is connected to the sea by four channels (bughaz) (El-Gamil, Ashtom El-Gamil, El-Diba and El-Baghdadi). Bughaz El Gamil is the main connection between the lake and the Mediterranean. The lake is connected to Suez Canal through Boughaz Alkaboty and Damietta branch through El-Ratama and El-Safra Canals (EEAA 2010).

Large areas in the north-west of the lake have been turned into fish farms. 3.7 km^3 of fresh water flow annually into Lake Manzala from nine major drains and canals. The most important of these are Faraskur, Al Sarw, Baghous, Abu Garida and Bahr El Baqar. Of all the drains discharging into Lake Manzala, the Bahr El Baqar drain is the most polluted (Baha El-Din 2002). Artificial wetland has been established to reduce the pollution of this drain (El-Quosy 2005).

13.4.2 Burullus Lagoon

13.4.2.1 Site Description

Lake Burullus, the second largest Delta lakes, with total area of about 70,000 feddans. It is elongate in shape extending for c.54 km from east to west with a width of 6–21 km and an estimated average depth of 75–100 cm. The mean annual fish production from the lake is 48,000 (Baha El-Din 2002). Lake Burullus is a shallow brackish water basin. The lake lies in the north of the Nile Delta, along the Mediterranean Coast of Egypt between Long. 30°30′ & 31°10′ E and Lat. 31°35′ N (Fig. 13.3). The lake is separated from the sea by to 5.5 km. The lake is connected directly to the Mediterranean Sea through El-Burullus outlet, which is about 250 m wide and 5 m deep. There are some 50 islands scattered throughout the lake with a total area of 0.7 km^2. The lake receives drainage waters with anthropogenic materials from agricultural areas through seven drains in addition to the fresh water from Brembal Canal (El-Sammak and El-Sabrouti 1995). The amount of the drainage water discharged annually into the lake fluctuates from 1 year to the other (Samaam et al. 1989).

The north shores of the lake are dominated by saltmarshes and mudflats, while the southern shore is bordered by an extensive fringe of reed-swamps (mainly Phragmites and Typha), which currently covers more than 25% of the lake area. Lake Burullus has abundant submerged vegetation, dominated by Potamogeton, which is densest in the southern portion of the lake (Baha El-Din 2002).

Fig. 13.3 Google earth image of El-Burullus Lagoon during 2016

13.4.3 Edku Lagoon

13.4.3.1 Site Description

Edku Lake is coastal wetland located in the north of the Nile Delta west of the Rosetta Nile branch and approximately 35 km east of Alexandria. It extends for about 17 km in the east-west direction. It lies between latitudes of 31°10′ and 31°18′ N, and longitudes of 30° 8′ and 30° 22′ E (Fig. 13.4) (Youssef and Masoud 2004). It is a shallow inland eutrophic lake with an average depth of 1 m. The mean annual fish production from the lake is 9000 ton. The width of the lake (N-S direction) is about 11 km at its widest part, where the narrowest part is only about 5 km (Masoud et al. 2005). The lake is connected to the sea by El-Maadia outlet. The total area of the lake is decreased from 30.000 to about 12.000 ha as a result of agricultural reclamation (Masoud et al. 2005). Two main drains namely Magrour Edku and Barsiek discharge huge volumes of drainage water to the Lake. Magrour Edku Drain is joined to different sources of drains. These Drains carry mainly agricultural, domestic and to less extent industrial effluents. The drainage water of more than 300 fish farms is also disposed to El-Khaiy Drain directly before being connected with the Lake. Barsiek Drain transports mainly agricultural drainage water to the Lake as well as the waste of Barsiek fishing ponds (Youssef and Masoud 2004). The water in the lake is mainly fresh (brackish), but increases in salinity towards the Bughaz and during the summer (Baha El-Din 2002).

Fig. 13.4 Google earth image of Edku Lagoon during 2016

13.4.4 Mariout Lake

13.4.4.1 Site Description

Lake Mariout is situated along the Mediterranean coast of Egypt south of Alexandria city (long. 29°51′ 00″–29° 56′ 15″E, lat. 31°04′ 15″–31° 10′ 45″ N) as shown in Fig. 13.5. It has been divided artificially by international roads and railway lines into four basins; the fish farm, the north-western, the south-western, and the main basins (EEAA 2010). The main basin is the heavily polluted part of the lake. The major water sources of the lake are El-Omoum Drainage, El-Kalaa Drainage and Nubaria Canal. The most important of these drains is the Kalaa Drain, as well as large quantities of municipal and industrial effluent from the city of Alexandria. El-Kalaa Drain disposes of an average of 920,000 m^3 wastewater per day in the main basin of the Lake Mariout. Since 1993 El-Kalaa Drain has received partially treated wastes from the Eastern Treatment Plant as well as mixed sewage from El-Kalaa tributary drains (Youssef 1999; Youssef and Masoud 2004).

In the nineteenth century, the western half was cut off by a railway embankment and transformed into an extensive salina, now known as Malahet (Brine) Mariout. What remains of the lake proper is brackish, receiving agricultural drainage-water through several drains. The lake has no direct connection with the Mediterranean, and is maintained at a level of c.2.8 m below sea level and discharge its water to the Mediterranean by a pumping station at El Max. Much of the lakeshore is fringed by extensive Typha/Phragmites marshes. The lake still supports a fishery, with Tilapia sp. making up most of the production (Baha El-Din 2002).

Fig. 13.5 Google earth image of Mariout Lagoon during 2016

13.4.5 Water Quality of the Coastal Lakes

The aquatic ecosystem of the coastal lakes is a major concern since good ecosystem health is essential not only for sustaining lake services to local populations but also for maintaining biodiversity (Zakaria et al. 2007). Egyptian Environmental Affairs Agency (EEAA) initiate monitoring program to check the water quality of the coastal lakes. The results of the monitoring of water quality of the lakes under investigation during August 2009 carried out by Central Department for Water Quality-the national program for monitoring the Egyptian Lakes-EEAA are listed in Table 13.2 as reported in EEAA report of Egypt state of Environment of year 2012 (EEAA 2015). Table 13.3 listed the maximum and minimum values of annual average of the chemical parameters of water and sediments of Edku Lake according to Masoud et al. (2005).

13.5 Environmental Indicators

13.5.1 Monitoring Dissolved Oxygen

Dissolved oxygen (DO) levels in lake water are influenced by many factors, including water temperature, the concentration of algae and other plants in the water, and the amount of nutrients and organic matter that flow into the water body from the watershed. Oxygen is produced through plant metabolism (photosynthesis), and it

Table 13.2 Ranges of water quality parameters of the coastal lakes under investigation during 2012

Parameter	Manzala	Burullus	Edku	Mariout
pH	7.85–8.6	7.9–8.89	8.08–8.68	7.82–8.37
Salinity (ppt)	1.75–22.6	0.9–13.93	1.01–3.81	1.63–5.89
DO (mg/L)	1.9–13.03	2.6–13.96	3.4–11.5	3.75–8
Ammonia (mg/L)	0.14–3.77	0.08–2.65	0.06–1.93	0.27–11.92
Total nitrogen (mg/L)	3.36–8.26	2.77–7.51	3.62–5.82	2.8–18.5
Total phosphorus (mg/L)	63.61–881.96	247.14–1059.03	485.8–1055.86	57.9–2184.7
BOD (mg/L)	13.27–45.13	6.68–22.77	10.91–19.94	24.21–63.29
COD (mg/L)	72.28–329.1	102.67–243.88	125.28–294.13	93.5–414.12
Total coliform bacteria (cells/100 ml)	48–65,825	120–4700	157–2875	498–1,053,500
Polychlorinated biphenyls (ng/L)	2.29–15.13	4.02–25.8	6.47–18.94	2.1–13.82
Pesticides (ng/L)	1.27–4.93	2.46–13.4	4.56–9.9	1.33–14.95
Hydrocarbons/petroleum origin (μg/L)	0.71–1.46	0.86–2.66	0.68–1.98	0.83–3

Table 13.3 Maximum and minimum values of annual average of the chemical parameters of water and sediments of Edku Lake

Type		Fe	Mn	Zn	Cu	Cd	Cr	Co	Pb	Ni
Water		mg/L	μg/L	μg/L	μg/L	μg/L	μg/L	μg/L	μg/L	μg/L
	Max	0.82	75.25	57.55	31.5	11.75	16.50	14.00	95.50	51.50
	Min	0.30	28.83	8.85	20.00	11.25	2.5	5.00	18.50	26.25
Sediments		g/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
	Max	36.29	273.17	71.25	2.57	1.56	60.31	2.65	2.41	18.35
	Min	27.40	226.35	52.00	1.83	0.62	45.83	9.73	1.15	13.79

Masoud et al. (2005)

is consumed during respiration and decomposition. Oxygen in lake water is also influenced by wind and wave action through weather events and the exposure of surface water to atmospheric sources.

An adequate supply of dissolved oxygen in lake water is essential to fish and other aquatic life forms. DO is also a sensitive indicator of change in water quality, and of the ability of a water body to support aquatic life. The loss, over time, of DO in the deep areas of a lake, especially during summer months, may indicate that the ecosystem is stressed and changing (VLMP 2016).

13.5.2 Temperature Profile

Water temperature plays an important role in determining the amount of oxygen found in the lake. Oxygen is more soluble in cold than warm water. Most lakes over 20 ft deep stratify during the summer into a warm, lighted upper layer (epilimnion) and a cold, dark lower layer (hypolimnion). Thus, the cold lower layer can potentially hold more oxygen than the warmer upper layer. it is important to define the thermal layers in a lake when characterizing dissolved oxygen conditions (USEPA 2002).

13.5.3 Salinity

Coastal seawater salinity affected by the discharge of the fresh or low saline water from inland source such as the coastal lakes.

13.5.4 Monitoring Sedimentation

Sedimentation problems occur when erosion is taking place in the watershed. Surface runoff washes sand and silt into the lake where it settles to the bottom and creates shallow areas that interfere with lake use and enjoyment. In addition, sediments often carry significant amounts of nutrients that can fertilize rooted aquatic plants and algae.

Concerning the suspended solids, some of the silt and organic matter that enters a lake does not settle to the lake bottom. Instead, it remains suspended in the water. These suspended solids decrease water transparency and can affect the suitability of the lake habitat for some species. One can monitor the suspended sediment condition by measuring two parameters: water transparency using a Secchi disk; and total suspended solids (USEPA 2002).

13.5.5 Monitoring Acidification

The measurement of pH is the detection of lake acidity and alkalinity status. The pH is measured on a scale of 0–14. The lower the pH, the higher the concentration of hydrogen ions and the more acidic the solution. Acid rain typically has a pH of 4.0–4.5. In contrast, most lakes have a natural pH of about 6.0–9.0. The pH of a lake sample can be easily determined by using a portable, battery-powered pH meter.

13.5.6 Monitoring Bacteria

The indicator organisms most often used to indicate sanitary conditions at bathing beaches are fecal coliform bacteria and enterococcus bacteria. Coliforms belong to the enteric bacteria group, Enterobacteriaceae, which consists of various species found in the environment and in the intestinal tract of warm-blooded animals. Fecal coliforms are the part of the coliform group that are derived from the feces of warm-blooded animals. The fecal test differentiates between coliforms of fecal origin and those from other sources (USEPA 2002).

13.6 Adjacent Marine Area and Their Outlets

Water quality of the adjacent marine area of the coastal lakes under investigation was monitored under the national program “The Environmental Information and Monitoring Program” (EIMP) which operate through the Egyptian Environmental Affairs Authority (EEAA). The purpose of the Coastal Water Monitoring Program is to obtain baseline knowledge of the quality of the Egyptian coastal waters and to establish a continuous survey of these waters. The outputs is also used to establish quantitative and causal relations between pollution sources and pollution impacts. The proposed water sampling Program focuses on measurements of marine water samples and the outlets from the river Nile and the major lakes are sites of interest in this program (EEAA 2016b). Continuous bimonthly and annual reports were presented the main results of this program such as EEAA (2016c, 2015). Figure 13.6 shows the dissolved oxygen values at the adjacent marine water of the lakes under investigation during May 2016. It shows that all the values are accepted as higher than accepted limit of 4 mg/L. Effect of water salinity of the coastal lakes under investigation to the adjacent marine area during May 2016 is presented in Fig. 13.7. The salinity in front of the outlets are decreased than the normal ambient seawater in this area. The water of El-Gamil outlet is the relatively higher effects of the adjacent seawater. The domestic wastewater discharged from the coastal lake outlets to the adjacent seawater was detected by coliform bacterial indicator. Figure 13.8 shows the counts of the Coliform bacteria in front of the coastal lakes outlet. Mariout Lake was recognized as the most polluted lake from the domestic pollution during May 2016. Moreover, scientific articles in specific journals and conference were submitted from the program output such as Haslund and his group (Haslund et al. 1999).

The water quality parameters of the drains discharged to Edku Lake and the lake water with its outlet have been summarized by Gharib and Soliman (1998) and listed in Tables 13.3 and 13.4. The quality of the adjacent marine water to the lake outlet is also presented during 1995–1996.

Coastal Research Institute (National Water Research Center, Egypt) established monitoring program for the water quality of the coastal lakes outlets since 16 years.

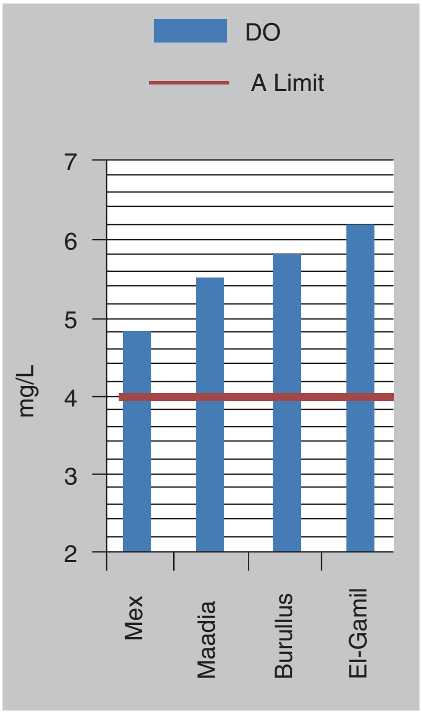

Fig. 13.6 Dissolved oxygen (DO) during May 2016 of the Mediterranean Sea Water adjacent to the coastal lakes under investigation against the accepted limit (Mex = Mariout Lake, Maadia = Edku Lake, Burullus = Burullus Lake and El-Gamil = Manzala Lake) (Data from EEAA 2016c)

The monitoring parameters includes the critical and the important water quality parameters such as water temperature, salinity, conductivity, pH, Dissolved oxygen, turbidity, transparency, TSS, NH_3, NO_2, NO_3, TN, PO_4, TP, SiO_2 and Coliform bacteria. Water quality index was calculated for the Mediterranean water adjacent to the coastal lakes outlets and it revealed temporal enhancement of their water quality (El-Gamal 2009). This is due to the governmental efforts to reduce the pollution through the establishment of the environmental law 4/1994 and his upgrading version 9/2009.

13.7 Vulnerability and Environmental Stresses

The rapid population growth in Egypt, especially in the Nile Delta covered by several wetlands, poses serious problems. Recently, coastal lagoons have become a matter of concern due to the detrimental impact of several human activities. There is close relationship between lagoons with terrestrial ecosystem from different environmental conditions. These conditions are hydrological modifications (freshwater diversions or drainage discharges), water pollution and habitat loss (Gamito et al. 2005; Pérez-Ruzafa et al. 2005). Finally, it make a deep change of the structure dynamic of the lagoons ecology (Cataudella et al. 2015). Accordingly, conservation

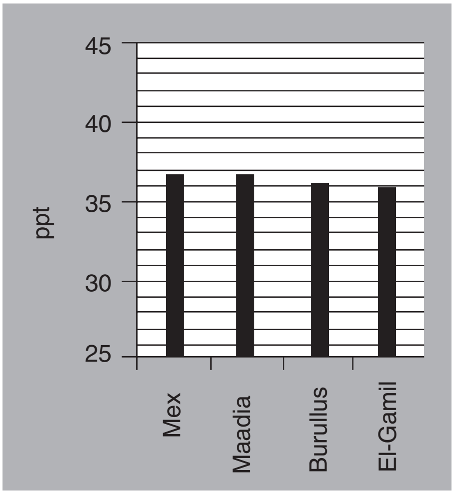

Fig. 13.7 Salinity during May 2016 of the Mediterranean Sea Water adjacent to the coastal lakes under investigation (Mex = Mariout Lake, Maadia = Edku Lake, Burullus = Burullus Lake and El-Gamil = Manzala Lake) (Data from EEAA 2016c)

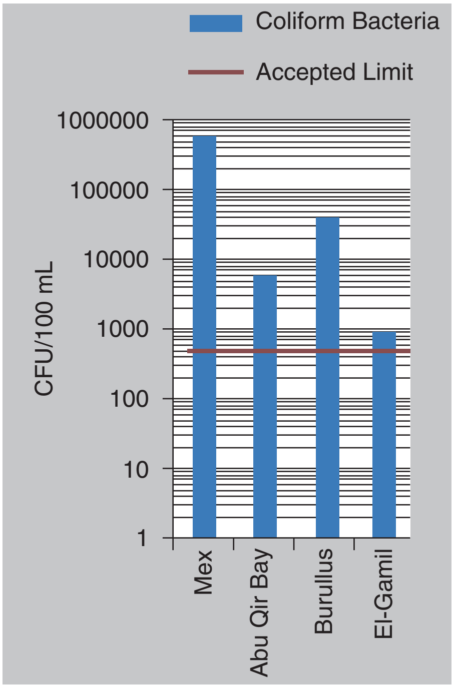

Fig. 13.8 Coliform Bacteria during May 2016 of the Mediterranean Sea Water adjacent to the coastal lakes under investigation against the accepted limit (Mex = Mariout Lake, Abu Qir Bay = Edku Lake, Burullus = Burullus Lake and El-Gamil = Manzala Lake) (Data from EEAA 2016c)

of these water bodies is important and became necessary for better management, mainly for the benefit of the Egyptian economy and public health. The Nile Delta lakes, as transitional zone between land and sea, are considered as the most productive natural systems in Egypt (Saad 2003).

Table 13.4 Range and (mean) values of the physic-chemical in the Edku Lake water and its drains with the adjacent seawater and the outlet water during 1995–1996 (Gharib and Soliman 1998)

Parameter	Sea	Outlet	Lake	Drain
Secchi depth (cm)	12.5–35 (29.6)	15–50 (30.2)	10–32.35 (25)	5–17.20 (15)
pH value	7–8.6 (7.85)	7.91–8.42 (8.05)	7.98–8.67 (8.32)	7.4–8.2 (7.68)
Dissolved oxygen (mg/L)	0.0–9.6 (3.94)	1.5–9.5 (5.25)	4.7–15.7 (10.83)	0.8–10.5 (6.99)
Ammonium (μmol/L)	0.0–22.45 (6.07)	0.0–18.96 (6.97)	0.0–22.99 (7.15)	0.0–34.79 (9.16)
Nitrate (μmol/L)	6–28.17 (11.16)	0.0–47.82 (16.44)	0.0–41.19 (50.2)	0.0–58.71 (26.16)
Nitrite (μmol/L)	0.88–51.44 (6.91)	2.48–28.80 (8.45)	0.64–8.12 (2.13)	2.1–23.60 (11.94)
Phosphate (μmol/L)	0.07–2.23 (1.29)	0.90–8.30 (4.78)	1–6.65 (2.57)	0.7–12.24 (7.99)
Silicate (μmol/L)	14.3–107.56 (47.92)	54.16–159.84 (126.15)	0.16–146.68 (88.1)	54.12–159.20 (140)

All Northern Delta lakes are more or less facing the same challenges according to many researchers as:

1. Pollution (Mehanna 2008; Cataudella et al. 2015), lagoons have even been considered as dumping areas for urban and industrial wastes (De Wit et al. 2011). Nile delta lagoons are the most polluted areas in Egypt. They receive great amounts of industrial, municipal and agricultural wastewaters without treatment.
2. Deterioration and decrease in available healthy water supplies (Saad 2003).
3. Lack of management (Cataudella et al. 2015) have strongly modified both the structure and the functioning of these sensitive coastal ecosystems.
4. Significant reduction in area (Mehanna 2008), most Egyptian wetlands have been degraded drastically during the past 50 years (Baha El-Din 2002). Filling up and drought; which lead to a decrease in size of all delta lagoons by over 70% of their original areas.
5. Spreading of aquatic plants (Mehanna 2008), several kinds of plants are found in the coastal lake such as *Myriophyllum spicatum*, *Potamogeton pectinatus*, *Phragmites australis*, *Typha domingensis* (Younis and Nafea 2012).
6. Habitat loss (Mehanna 2008), through land reclamation.
7. Climate change. Lagoon environmental features such as depth, connections with the sea, sediment dynamics, size, as well as water temperatures and productivity, shall all be affected by global climate change and the rise of sea level (Cataudella et al. 2015; Bianchi and Morri 2004; De Wit 2011; Nicholls et al. 2007).
8. The blockage of Boughazes (Mehanna 2008), siltation of the outlets is one of the important feature resulting from coastal processing and it will affect the ship sailing.

9. Eutrophication. Eutrophication can be accelerated with human consequences in the watershed. If proper controls are not in place, pollutants from agricultural, urban, and residential developments can easily be carried out to lakes and their tributaries (Cataudella et al. 2015; USEPA 2002).
10. Low awareness of fishermen about environmental issues (Mehanna 2008),
11. Declining of fish yield and fish quality, (Mehanna 2008)
12. Over-fishing (Mehanna 2008),
13. Illegal fishing practices and illegal harvesting of fish fry (Mehanna 2008),

13.8 Environmental Conservation and Socioeconomic Development in the Nile Delta Region

13.8.1 Manzala Lagoon

The lake is unprotected, apart from Ashtum El Gamil Protected Area (declared by Prime Ministerial Decree 459/1988), which encompasses a small area (c.35 km^2) located along the sandbar at Bughaz El Gamil, the largest connection between the lake and the sea, near Port Said. The main purpose for creating this protected area was the protection of gravid fish and fry during their passage in and out of Manzala, through Bughaz El Gamil (Baha El-Din 2002).

El-Salam canal is the main land reclamation project in the Eastern Nile Delta and Sinai Peninsula in Egypt (currently mostly destined for Manzala). The project aim is to reuse agricultural drainage water from Bahr Hadous and lower Serw drainage to irrigate about 620,000 feddans through El-Salam canal (EL-Sayed and Omar 2013). This is expected to lead to a significant increase in the salinity of the lake from the current 3–8 ppt, consequently changing its whole ecology (Baha El-Din 2002). Efforts have been done to decrease the water salinity in Bahr Hadous drain in order to increase the amount of the reused drainage water discharged into El-Salam canal (EL-Sayed and Omar 2013).

Lake Manzala serves as a final repository for much of the municipal and agricultural wastewater of the eastern Delta, including the wastewater of most of Cairo. The main contributors to the lake are the Bahr El Bakr Drain, Hadous Drain and the drainage water delivered by some adjacent pumping stations. The Bahr El Bakr Drain carries sewage effluent from Cairo and the polluted drainage water of more than 200,000 ha of agricultural land (Wahab and Badawy 2004). Engineered Wetland was constructed to improve the water quality of Bahr El-Baqar drain, before entering the Lake Manzala (El-Quosy 2005).

13.8.2 El-Burullus Lagoon

Burullus is protected by Prime Ministerial Decree 1444/1998 and is a Ramsar Site. The area of the lake is decreased as a result of ongoing drainage and reclamation of the lake's margins, and also due to the proliferation of emergent and submerged vegetation. It is anticipated that Burullus, along with other coastal delta wetlands, will be further reduced in area because of landward migration of coastal sandbars. Despite being the least polluted of the northern delta lakes, increasing quantities of agricultural drainage-water with heavy fertilizer and pesticide loads are being released into Burullus, contributing significantly to the eutrophication and pollution of the lake (Baha El-Din 2002).

13.8.3 Edku Lagoon

Edku Lake suffers from the same ailments that affect other delta wetlands: drainage and land-claim, pollution, disturbance, water bird catching, etc. Habitat loss through land-claim is certainly the most serious of these threats. Edku Lake has been reduced to less than half its original size (Baha El-Din 2002).

13.8.4 Mariout Lagoon

Lake Mariout has been reduced by more than 75% from its original area, and is still decrease in size. The main causes for the diminishing area today are urban encroachment and solid-waste dumping from the rapidly growing city of Alexandria. The lake is eutrophic and is the most polluted wetland in Egypt. The level of disturbance is particularly high because of the very close proximity of Alexandria's urban and industrial sprawl. The outlook for the future of this wetland is rather grim (Baha El-Din 2002).

13.9 Improvement of Coastal Lagoons Resilience

The Egyptian Ministry of State for Environmental Affairs has set a priority to protect the northern lakes from pollution and to maintain their sustainable development. This carried out within its priorities and strategy for water resources protection. So that periodical monitoring of pollution sources, quantities and type of discharges from these sources; also monitoring water quality and sediments in these lakes became a perquisite to determine impacts of various pollutants, and to set priorities for rehabilitation and development to ensure their sustainability to maximize their benefits (EEAA 2010).

Legal framework and constraints have been set to control the human activities in the coastal lagoons. Law 124/1983 regulates capture fishery and aquaculture activities in Egypt. The Law 4/1994 and its enhanced version 9/2009 on the protection of the environment constitutes the main legislative act in the field of environmental protection and promotion.

Two coastal lagoons in Egypt were designated as Ramsar Sites. The first lagoon is the Burullus wetland, which was declared as a natural protectorate in 1998. The protectorate includes the lake, its islets, as well as the sand bar between the Mediterranean and the lake. This habitat is very important for migrant birds for foraging, refuge and breeding. The second lagoon is the Bardawil lagoon, also designated as a Ramsar Site in 1988. Very limited eco-friendly human activities are allowed in the surroundings of the lagoon. All development activities are forbidden inside the protectorate except salt production. Other coastal lagoons are unprotected, apart from the Ashtum El Gamil Protected Area (declared by Prime Ministerial Decree 459/1988), which encompasses a small area (c.35 km^2) located along the sandbar at Bughaz El Gamil, in the Manzala lagoon. Constrains include land reclamation, pollution and illegal fishing practices (Cataudella et al. 2015).

In order to establish a successful volunteer monitoring program, the necessary steps to plan and manage were mentioned in the international guidance of EPA's "Volunteer Water Monitoring: A Guide for State Managers". Topics in this guide include how to establish goals, identify data uses and users, assign staff responsibilities, establish a pilot program, prepare a quality assurance plan, and fund a program (USEPA 2002).

The Egyptian Ministry of Water Resources and Irrigation (MWRI) has a wetlands policy that copes with the Ramsar Convention to strength the objective of protection and enhance water resources management and the overall aquatic environment on a sustainable basis. The policy includes promoting and encouraging fresh water conservation and the efficient water utilization in different water use categories.

Natural wetlands rehabilitation is in the Egyptian policy is worked with the wetlands protection in the process of water resources management (USEPA 2002; Fahmy et al. 2007).

Establishing quality assurance and quality control in many environmental issues was distributed and now well known. There are five major areas of uncertainty that should be evaluated when formulating data quality objectives as listed by USEPA (2002). These areas are accuracy, precision, representativeness, completeness and comparability. Accreditation for environmental laboratory that holding ISO 17025 certificate is recommended.

13.10 Summary and Conclusion

Wetlands and coastal lagoons are valuable and sensitive environments as recognized by Ramsar Convention. They described wetlands as the kidneys of the landscape. Egypt has many forms of wetlands. Different efforts have been done to classify the

Egyptian wetlands. One classified wetlands in Egypt according to its location to be coastal either in the Mediterranean Sea or Red Sea or inland wetlands. Another classification was adopted the wetlands in Egypt into 14 generic types. The northern lakes are grouped in one of these types. Moreover, classification of the northern lakes as coastal wetlands has been performed into three sectors from Salloum to Alexandria to Port Said to Rafah. The second and the central area contains the most important lagoons and wetlands: from west to east Mariout, Edku, Burullus and Manzala.

The Egyptian Ministry of Environmental Affairs updated the National Biodiversity, Strategy and Action plan (NBSAP) for the years (2015–2030). One of the goals of this strategy is to minimize the rate of wetlands loss by 50% and improved the water efficiency for farming by 50%. In general, the wetlands in Egypt are facing many threats to their degradation such as pollution, reduction of its area, accretion and erosion, climate change and overgrazing. Nile Delta coastal lagoons Manzala, Burullus, Edku and Mariout have discussed in some kinds of details to describe their morphology, environmental status & stress and their water quality according to recent measurements of EEAA team work. These coastal wetlands are shallow lakes with direct or indirect contact with the sea.

El-Manzala Lagoon was characterized by enlargement of its size, which reduced with the time, and its high productivity. The lake received its water from different polluted drains. Artificial wetland has been established to reduce the pollution of Bahr Elbakar Drain, which is the most polluted drain discharge its water to El-Manzala Lake. El-Burullus Lake is the second largest coastal lake. El-Burullus Lake receives fluctuated amounts of drainage waters with anthropogenic materials from agricultural areas through seven drains. Edku Lake is coastal wetland located in the north of the Nile Delta west of the Rosetta Nile branch. Two main drains namely Magrour Edku and Barsiek discharge huge volumes of drainage water to the Lake. Lake Mariout is situated south of Alexandria city. It divided artificially into four basins; the fish farm, the north-western, the south-western, and the main basins. The major water sources of Edku lake are El-Omoum Drainage, El-Kalaa Drainage and Nubaria Canal.

EEAA initiated monitoring program to check the coastal lakes water quality. The results of this program are presented in different EEAA publications such as Egypt state of environment and summarized in this chapter. The critical environmental indicators have been identified and monitored in the coastal lagoons under investigation. Dissolved oxygen, water temperature, salinity, sedimentation, acidification, coliform bacteria. This program indicated that hypoxia (dissolved oxygen <4 mg/L) condition has been detected in different sites in the four brackish coastal lagoons (Manzala, El-Burullus, Edku and Mariout). Coliform bacteria was found in parts of the four coastal lagoons under investigation higher than the accepted counts level, which indicated of the domestic pollution.

National program organized by EEAA has been established to monitor water quality of the adjacent marine area of the coastal lakes under investigation to obtain baseline knowledge of the quality of the Egyptian coastal waters. The program focus on the estuaries and outlets of major drains and the coastal lakes. Selected water quality parameters values were presented. During 2016, dissolved oxygen showed that all the values are accepted as higher than the accepted limit of 4 mg/L. Enhancement of water quality in the adjacent area of the outlets of the coastal lakes under investigation was recognized. This is due to the enrolling of the Egyptian environmental law 4/1994 and its updated version 9/2009. The four coastal lagoon are vulnerable to environmental stresses such as pollution. The challenges of the coastal lakes were summarized to include pollution, water deterioration, lake of management, reduction of area, aquatic plants, habitat loss, climate change, siltation of the outlets, eutrophication, awareness, illegal fish practice, over fishing and decline of fish yield.

Many efforts have been done for environmental conservation and socioeconomic development to the coastal lagoon. El-Manzala Lake is environmentally unprotected, but apart from it at the site Ashtum El Gamil is considered as Protected Area (declared by Prime Ministerial Decree 459/1988). Engineered Wetland was constructed to improve the water quality of the most polluted drain (Bahr El-Bakar drain) which discharge its water into Lake Manzala. El-Burullus Lake is protected by the Egyptian Prime Ministerial Decree 1444/1998 and is a Ramsar Site. Edku and Mariout Lagoons still need more efforts to environment conservation for sustainable development.

In order to improvement of coastal lagoons resilience, periodical monitoring of water quality and pollution sources, quantities and type of discharges from these sources became perquisite to determine its impacts of these lakes. In addition, the setting of the priorities for rehabilitation and development to ensure their sustainability is highly required.

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Part III
Restoration Techniques, Ecological Aesthetics, and Ecosystem Conservation (Sustainability and Biodiversity)

Chapter 14
Coastal Wetland Restoration: Concepts, Methodology, and Application Areas Along the Indian Coast

Ramasamy Manivanan

Abstract This paper emphasizes and endorses the use of natural restoration techniques for Coastal Wetland Restoration in Indian scenario. Natural techniques that restore an ecosystem's ability to approach a pre-disturbance condition are distinct from treatment technologies or structures that are inserted into the system to be acquired sustainable equilibrium. Natural restoration techniques use materials indigenous to the ecosystem and are incorporated into the dynamics of an coastal ecosystem in an attempt to create conditions in which coastal ecosystem processes can withstand and diminish the impact of stressors on the coastal environment. In Chilika wetland ecosystem, the salinity is restored due to open cut of the sand bar and entered much volume of sea water and increased the salinity concentration in the wetland. Subsequently, the fish population increased in the Chilika wetland ecosystem. Similarly, Vembanad wetland ecosystem, also restored the fisheries population by operating the gates of the weirs in the region.

Keywords Coastal wetland restoration • Chilka wetland • Vembanad wetland • Coastall • Dynamics • Salinity • Fisheries • Restoration technology

14.1 Introduction

Water has many wonderful, novel and unique properties. Due to this characteristic of water gets polluted by various sources like point and non point from industrial and domestic waste water. Present day scenario, the water ecosystem is damaged, contaminated, polluted even not for designated best use in the world. India also witnessing the development of industries, over population, releases industrial effluents and domestic sewage in to the coastal water body. The water is over exploited

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and the day come pure water is not available but recycled water is available for best designated uses. This is a right time to restore the ecological status of the coastal water body before contamination and erosion. Coastal wetland Restoration is solely applicable to severely degraded ecosystems such as rivers, lakes, reservoirs, streams, oceans and coastal environment. Although it can be used as an effective tool to return the degraded ecosystem to a pre-disturbance stage, coastal wetland restoration is also an important tool for preventing environmental degradation. Strengthening structural and functional elements through restoration can be helped to improve a coastal wetlands ecosystem's health.

Coastal wetland restoration is the process of returning a damaged ecosystem to its original condition prior to disturbance (Cairns 1991; Berger 1991). The long-term goal of restoration is to imitate an earlier natural, self-sustaining ecosystem that is in equilibrium with the surrounding landscape (Berger 1991). A National Research Council (NRC) report (1992) defines restoration technology as a holistic process as follows:

> *Coastal restoration is the return of an ecosystem to a close approximation of its condition prior to disturbance. In restoration, ecological damage to the resource is repaired. Both the structure and the functions of the ecosystem are recreated. The goal is to emulate a natural, functioning, self-regulating system that is integrated with the ecological landscape in which it occurs.*

Restoration is not yet a perfected approach with accurate and precise predictive capabilities and, in fact, is still "... an exercise in approximation" (Cairns 1991). The practicality and attainability of restoration depend on many factors, including adequate tools, site-specific ecological conditions, social consent, legal authority, and availability of resources i.e., personnel and funding. As with other water resource management alternatives, restoration must address questions concerning practicality, predictability of outcomes, and overall effectiveness of specific technology. Additionally, coastal ecological systems are complex and may take years to reach equilibrium or fully demonstrate the effects of restoration and other management activities, measuring results of restoration efforts may take a longer time than any other method of treatment technologies.

14.2 Scope of Restoration

Coastal Wetland Restoration must consider all sources of factors on an ecosystem and is therefore not restricted to any mitigation of impacts. The health and protection of a water body cannot be separated from the coastal ecosystem, and restoration must address all coastal processes that degrade an ecological system, e.g., sediment loading from littoral drift or development or increased polluted runoff from impervious areas. The intimate connection of rivers and estuaries is succinctly expressed by Doppelt et al. (1993): Most people think of rivers simply as water flowing through a channel. In the past 5–10 years many scientific studies and reports have

Fig. 14.1 A typical river ecosystem

documented that riverine systems are intimately coupled with and created by the characteristics of their catchments basins, or watersheds. The concept of the coastal wetland restoration includes four-dimensional processes that connect the longitudinal (upwelling-downwelling), lateral (longshore), and vertical (cross shore) dimensions, each differing temporally.

Coastal wetland restoration is an integral part of a broad, region-based approach for achieving federal, state and local goals (Lewis 1982a, b). Specifically, coastal wetland restoration is the re-establishment of chemical, physical, biological and microbiological components of an coastal ecosystem that have been compromised by stressors such as point or nonpoint sources of pollution, habitat degradation, hydro modification, and others.

14.3 Types of Ecosystems

The hydro ecosystems are classified into various types based on their flow, salty nature, volume etc. The following sub-chapters are discussed about the nature, characteristics, current speed, direction, status like polluted, non polluted etc.

14.3.1 River Ecosystem

A river is a large natural waterway. The source of a river may be a lake, a spring, or a collection of small streams, known as headwaters. From their source, all rivers flow downhill, typically terminating in the ocean. The mouth, or lower end, of a river is known as its base level (Fig. 14.1).

A river's water is confined to a channel, made up of a stream bed between banks. Most rainfall on land passes through a river on its way to the ocean. Smaller side streams that join a river are tributaries.

Fig. 14.2 A typical lake ecosystem

14.3.2 Lake Ecosystem

A lake is a body of water or other liquid of considerable size surrounded entirely by land. A vast majority of lakes on Earth are fresh water, and most lie in the Northern Hemisphere at higher latitudes. In ecology the environment of a lake is referred to as lacustrine. Large lakes are occasionally referred to as "inland seas" and small seas are occasionally referred to as lakes. Smaller lakes tend to put the word "lake" after the name, as in Green Lake, while larger lakes often invert the word order. Many lakes are artificial and are constructed for hydro-electric power supply, recreational purposes, industrial use, agricultural use, or domestic water supply (Fig. 14.2).

14.3.3 Stream Ecosystem

Stream Ecosystem Restoration (SER) means to restore degraded ecosystems to a level that can be permanently sustained through protection and conservation. More realistically, SER mitigates effects or remediates degraded ecosystems to ones that have a higher order of ecological sustainability. Restoration can occur naturally, but usually involves reductions of stresses such as nutrient or contaminant loads. Waste heat or different management. Assessments of degraded stream ecosystems allow decisions to be made as to what to control in order to remediate effects, or how much can be relied on nature to clean itself. To achieve ecosystem stability or sustainability requires decisions on what to do, including in situ options such as bioremediation or biomanipulation, as well as the development of ecosystem indicators of progress towards restoration. Restoration towards a less degraded, but not necessarily pristine ecosystem, requires decisions as to how far to go or how clean is clean. To arrive at such conclusions and to monitor progress towards them requires

Fig. 14.3 Typical stream ecosystems

the development of indicators of ecosystem health, stability and sustainability. If these criteria are met, the ecosystem can be declared as remediated to acceptable conditions, perhaps even restored.

The level of protection or control or regulation required to reach this state is dependent on the recovery or restoration of the ecosystem and the state that needs to be maintained. Restoration means that degraded aquatic ecosystems are remediated to some level of stability or sustainability involving minimization of stresses, in situ treatments, and probably conservation of components of the total aquatic ecosystem that have not yet been degraded. The typical stream ecosystems are shown in Fig. 14.3.

14.3.4 Ocean Ecosystem

Oceans are saline waters that cover almost 71% of the surface of the Earth.

The area of the oceans is 361 million sq. km., and nearly half of the world's marine waters are over 3,000 m (9,800 ft) deep. Though somewhat arbitrarily divided into several "separate" oceans, these oceans are in fact one global, interconnected body of salt water, often called the World Ocean. Figure 14.4 show the typical ocean area and associated rock coastal ecosystem.

14.3.5 Coastal Ecosystem

The coast is one of the most important boundaries in the world. It separates the marine part of the world from the continents. It separates the salt-water communities from fresh water communities; The following figures (Fig. 14.5) show the typical coastal area of rocky coast and sandy coast).

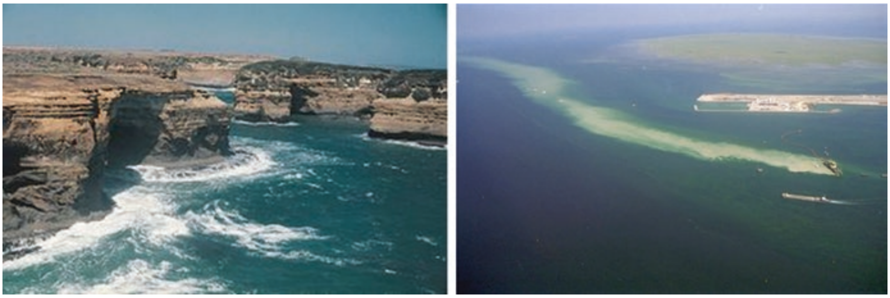

Fig. 14.4 Typical oceans and seas

Fig. 14.5 Typical coastal area of rocky and sandy coasts

It separates land life from water borne life but amphibians living both environments. Some of the marine organisms move towards estuary for reproduction with less water salinity. The coastline as such is not a stable. It fluctuates in place over considerable distances due to erosion and sedimentation, but also due to the relative change of sea level with respect to land. The coastal zone is thus a very mobile part of the earth from a geographical point of view. Due to its specific geophysical properties, the coast is also a very important area for the development of flora and fauna. There is hardly any other part of the world that houses such a variety of species, both in flora and fauna. The abundance of opportunities along the coast has also attracted a lot of human activities. The coastal zone has become an indispensable resource for mankind.

It provides excellent opportunities for fisheries, housing, recreation, transport, defense purpose, industrial activities and many others. The influx of man with their social and economic activities, their needs for safe housing, food, drinking water, and with the tremendous quantities of waste they are producing poses at the same time a serious threat to the unique features of the coastal zone. It is therefore necessary to study this valuable region so as to prevent that competing activities are gradually causing depletion of the resources in the coastline.

14.4 Coastal Wetlands of India

Coastal wetland Lakes of brackish water (Salt and Fresh Water Mix) (1) Ashtamudi Lake, Kerala – 61,400 ha. (R) (2) Chilika Lake, Orissa – 116,500 ha – Largest brackish water lagoon in Asia. (R) (3) Kuttanad lagoon, Kerala, Five major rivers drain – Most area consists of freshwater – 'kayal' or backwaters – 'One of the few places below sea level with farming' (4) Pulicat Lake, Andhra Pradesh & Tamil Nadu, – 77,000 ha – Second largest brackish water lagoon in India – Unique for its multi-ecosystem. (5) Vembanad-Kol Lake system, Kerala – 151,250 ha – Fed by ten rivers -Two distinct segments of fresh water and salt water (R). In this paper it is reported two coastal wetlands namely Chilika lake, Orissa and Vembanad Lake Kerala.

14.4.1 Restoration of Chilika Lake, Orissa

The basic approach adopted for restoration is to be facilitated a community based eco-management system for an integrated terrestrial and aquatic resource management programme, with a major emphasis on the capacity building at the community level. This is to be done through a series of training and exposure visits, to pave the way for preparation of the micro plan of watershed blended with indigenous knowledge at the community level for optimum utilization of the natural resources to increase in the productivity. The increase in the productivity level is also helping in the poverty alleviation in the catchment area. The watershed community also shares the part of the cost of the treatment. This is creating an enabling situation for the local community to take decision on the natural resource management within the catchment. The self initiated good practices are indication of the confidence regained by the stake holders on the lagoon ecosystem. The restoration of the Chilka Lake started with the following objectives:

- To restore the ecological balance of the lake
- Provide the livelihood to the villages surrounding the lake/lagoon.

The Chilka Lagoon (Fig. 14.6) was placed under the Montreaux Record of Ramsar due to adverse changes in its ecological character caused by pollution and other anthropogenic pressure. The entire issue of better lagoon management and the restoration programme was initiated.

The major interventions for the restoration were Desiltation of the outer-channel of the lagoon by dredging and Opening of an artificial mouth along the sand spit at a distance of 11 km from the lake. The Following restoration measures would be taken for restoration of Chilika lake.

- Control of Silt load by plantation in catchment area
- Improvement of water exchange and maintenance of salinity gradient. (Dredging;
- Bathymetry and Hydrologic monitoring)

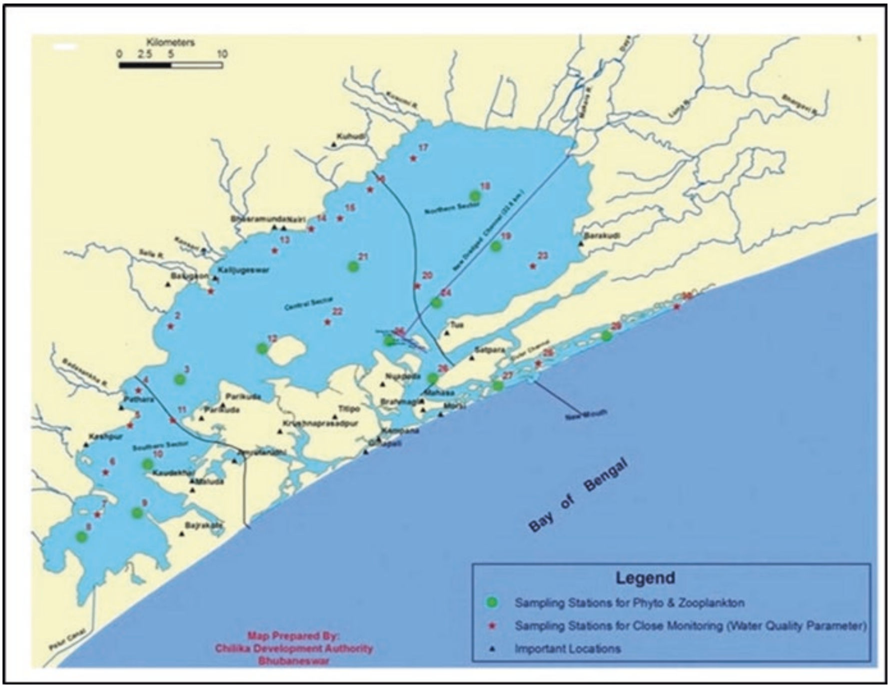

Fig. 14.6 Chilika wetland ecosystem

- Nalaban Island Ecosystem Conservation Programme for Migratory Bird
- Weed management with community participation and use in bio gas generation
- Hydrobiological monitoring
- Mass awareness programme
- Fishery Resource Development programme (net mesh-size regulation, prohibiting catching brood fish and prawn juveniles, identification)
- Upliftment of Socio Economic Condition of Rural Community around the lake
- Setting up Wetland Management Research Centre at Balugaon.
- Improvement of road network around Chilka.

Due to the restoration measure the following improvements were observed in the chilika lake wetland ecosystem. There have been remarkable changes in the ecosystem of the Chilka Lake after the restoration process. The major achievements were:

1. Salinity Change: Salinity level in the northern sector changed from 0.5 to 2.5 ppt recorded in the last decade to 0.1–36.00 ppt.
2. Fishery productivity: The fish land, which had declined to 1,600 mt before intervention changed to 11,877 mt in 2001–2002 and crab landing increased from 3 mt in 1994–1995 to 150 mts in 2002–2003. There is also a marked change in ranking order (Fig. 14.7).

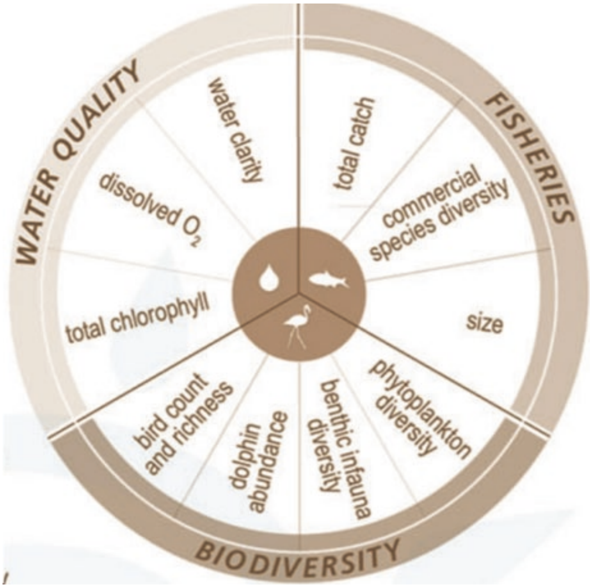

Fig. 14.7 Relationship of water quality with fisheries population

3. Auto recruitment of fish, prawn and crab: The fish and crab landing data further indicate the significant level of increase in auto recruitment from the sea into the lake after intervention. The shrimp species Paenaus indicus alone showed a record yield of 438 mt, which is higher than any other figure in last one-decade.
4. Decline in weed infestation: The trend of invasive fresh water species reaching an area of 523.01 sq. km., in 2000 (October) leaving an weed free area of 333.82 sq. km., was greatly changed following a reduction of 172 sq. km., of weed spread area.
5. Depth of Channel: The opening of mouth and consequent changes in tidal flux led to significant flushing of sediment from the channel thereby increasing the depth to 30–45 cm level.

Apart from ecology the other impacts from this restoration exercise were:

- Changes in local community income: lake restoration resulted in enhanced fish and crab yield thereby increasing the per capita average annual income by Rs. 50,000/–
- Improved stakeholder awareness: the stakeholders have self adopted good practices like regulation of mesh size, discouraging juvenile catches.
- A better linkage has been established between CDA and fisher community following implementation of action plan.
- Improved Chemical, Physical and Biological parameters

14.4.2 Restoration of Vembanad Lake, Kerala

The Vembanad wetland system covers an area of over 2,033.02 km^2 thereby making it the largest wetland system in India. Of this, an area of 398.12 km^2 is located below the MSL and a total of 763.23 km^2 area is located below 1 m MSL. The lake is bordered by Alappuzha, Kottayam, and Ernakulam districts. It is situated at the sea level, and is separated from the Arabian Sea by a narrow barrier island. Canals link the lake to other coastal lakes in the north and south. The lake surrounds the islands of Pathiramanal, Perumbalam and Pallippuram. The Vembanad Lake is approximately 14 km wide at its widest point. The lake is a part of Vembanad-Kol wetland system which extends from Alappuzha in the south to Azheekkode in the north, making it by far, India's longest lake at just over 96.5 km in length. The lake is fed by ten rivers flowing into it including the six major rivers of central Kerala namely the Achenkovil, Manimala, Meenachil, Muvattupuzha, Pamba and Periyar. The total area drained by the lake is 15,770 km^2, which accounts for 40% of the area of Kerala. Its annual surface runoff of 21,900 Mm accounts for almost 30% of the total surface water resource of the state.

The most popular location on the shores of the lake is the Kumarakom Tourist Village situated on the east coast of the lake. The Kumarakom Bird Sanctuary is located on the northern fringes of Kumarakom village. The Vembanad Wetland system was included in the list of wetlands of international importance, as defined by the Ramsar Convention for the conservation and sustainable utilization of wetlands in 2002. It is the largest of the three Ramsar Sites in the state of Kerala. Vembanad lake (Fig. 14.8) has been heavily reclaimed over the course of the past century with the water spread area reducing from 290.85 km^2 in 1917 to 227 km^2 in 1971 and 213.28 km^2 in 1990. In the same period almost 63.62 km of erstwhile water spread were reclaimed primarily for formation of polders and to enlarge the extent of the Wellington island of Cochin port. The lake faces a major ecological crisis and has reduced to 37% of its original area, as a result of land reclamation.

A unique characteristic of the lake is the 1,252 m (4,108 ft)-long Thanneermukkom salt water barrier constructed as a part of the Kuttanad Development Scheme to prevent tidal action and intrusion of salt water into the Kuttanad low-lands. It is the largest mud regulator in India and essentially divides the lake into two parts – one with perennial brackish water and the other with fresh water from rivers draining into the lake. This barrier has helped farmers in Kuttanad by freeing the area of salinity and allowing them an additional crop in the dry season. The Thanneermukkom barrier is located at one of the narrower parts of the Vembanad Lake. Only two-thirds of the original number of gates are opened in July to release flood flow. These gates remain closed until mid-November. The main drawback of the structure has been the loss of opportunity for fish and prawns to migrate upstream, and also an increase in weed growth in the upstream, severely restricting the natural flushing of pollutants. The Thanneermukkom bund has also created ecological problems, primarily, the rampant propagation of the Water Hyacinth in fresh water. Vembanad Kol Wetland was included in the list of wetlands of international importance, as

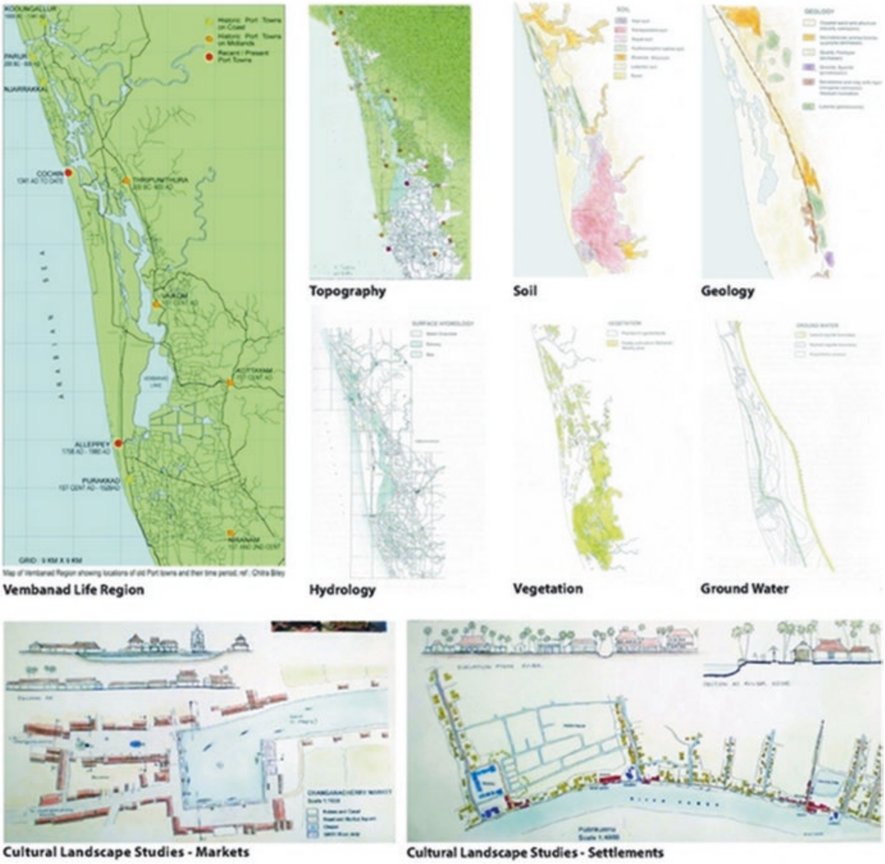

Fig. 14.8 Location plan of Vembanad wetland ecosystem

defined by the Ramsar Convention for the conservation and sustainable utilization of wetlands. It is home to more than 20,000 waterfowls – the third largest such population in India. It is also an ideal habitat for shrimps. Major livelihood activities of the people living on the shores of the lake include agriculture, fishing, tourism, inland navigation, coir retting, lime shell collection. The uncontrolled mining of shells from the lake bed is also posing a threat to the eco-system. The sewage effluents and the heavy load of organic material released from the neighboring areas including a medical college at Alappuzha is let into the water and are responsible for the decrease in dissolved oxygen content in the water in the water body.

The major threat to fishes of Vembanad is habitat alteration due to changes in waterscape. Thousands of hectares of the waterbody had been converted to land over past 150 years. According to one study, 23,000 ha of the lake had reclaimed between 1834 and 1984, mainly for agriculture and aquaculture. The depth of the lagoon has reduced by 40–50% in all zones and as a result the drainage capacity of the lake has been reduced to 0.6 km^3 from 2.4 km^3, a decline of 75%. Illegal Conversion and encroachment of the lake is visible around the lake even today. The problems faced by organic pollution of the lake is discussed earlier. The inorganic materials like plastics that are dumped to the lake gets settled to the bottom of the lake has adversely affected the bottom feeders like gobids, and has also affected the fishes that attach their eggs to the bottom soil as the habitat changed. Fishes get attracted to the plastic bags that are thrown to the lake with food remnants get entangled and dies. Many such bags with dead fishes entangles in it were found during the VFCs and according to fisherman fish mortality due to plastic bags are increasing day by day High concentration of plastics in the lake bed affects the fish by destroying its natural environment. Pollution from pesticides, weedicides and fungicides too affect the fishes as the rigorous use of these agrochemicals in to which the fishes are sensitive. Earlier many fishes were using paddy fields as breeding and nursery grounds Than cradles of fishes, the over use of pesticides has eventually converted the paddy fields to a graveyard of fishes Use of these non target chemicals destroys the aquatic invertebrates which prove vital for the survival of many fishes. Use of pesticides and weedicides also resulted in the decline in larvivorous fishes which are known to harbor small canals and channels inside and adjoining the fields.

14.4.2.1 Restoration Measures Taken in the Vembanad Wetland Ecosystem

1. Thanneermukkom Barrage is identified as a major threat to the ecosystem health and fish fauna of Vembanad. Hence the barrage should be kept open at least for 9 months to allow saline influx to the lake. To attain this, definite crop calendar should be followed in Kuttanad and paddy harvesting should be completed before March every year.
2. A comprehensive study on the ecological Impact of Thannermukkom barrage and its further development should be done by keeping the barrage experimentally open for at least 2 years.
3. Any non-wetland use of the lake should be strictly regulated.
4. Increase awareness among fishermen in and around Vembanad on ethical fishing practices.
5. Fishery enhancement program adopted by the authorities should be done in accordance with the fishermen and the suggestions by them should be incorporated in any such management plans.
6. Organic pollution from untreated sewage from the nearby towns and house boats and resorts are a major reason of concern. The government should take utmost care not to open any untreated sewage into the lake.
7. Strict measures should be adopted to prevent the deposition of plastic and other solid waste into the lake. 8. Ranching of exotic or transplanted fishes in Vembanad should be stopped as these can affect the diversity of native fishes.

8. Any developmental or modification works on critical fish habitats in vembanad lake should be done with proper impact assessment studies.
9. A democratic institution should be set up with ample representation for the traditional stakeholders and local panchayaths with due legal powers for the governance of the Vembanad lake and associated socio-ecological system.

14.5 Restoration of Coastal Water Bodies

The climate and easy transportation in coastline of India has resulted in an increasingly large human population. The natural landscapes of the area have been greatly modified to provide a setting for development. Coastal marshes have flat topography and occur near waterways. These attributes have made them prime sites for commercial development. Seventy five percent of the southern India coastal wetlands have been destroyed by development (Lonzarich et al. 1992). Changes in the watershed and levels of pollution have altered these remaining areas as well (Lonzarich et al. 1992). These losses have a catastrophic effect on the migratory waterfowl utilizing the area. These habitat losses are especially critical because stopping places for waterfowl are rare in the arid landscape of east and west coast of India. In addition, losses and alteration of wetlands have resulted in dissected marshes with barriers to animal movements in the estuary region. Barriers include increased noise levels, buildings, and changes in water circulation and quality. These barriers have hampered the ability of birds to reach alternate resting and feeding sites near tidal wetlands like Chilka wetlands ecosystem. In addition, barriers are preventing the dispersal of plants (Zedler 1982). Human caused disturbance has occurred for so long and to such a great extent that it is difficult to distinguish natural and unnatural features in coastal marshes. There are no pristine examples of coastal wetlands left in India to use as a template to reconstruct marshes (Zedler 1982). Nevertheless, restoration projects have been undertaken. This paper will discuss the use of beach nourishment as a restoration technique in southern east coast of India.

Coastal salt marshes occur in the intertidal zone of moderate to low energy shorelines along estuaries, bays, and tidal rivers. The coastal marshes of southern and central east coast are very distinct from the marshes found on the other coastal areas (Lewis 1982a).

The principal value of southern east coast wetlands is providing habitat for the several plant, animal, vertebrate, and invertebrate species that depend entirely on their estuaries. Providing habitat for these species helps to maintain biodiversity in the area. Nitrogen is an important aspect of wetland function. Its availability and supply affects plant biomass, reproduction, productivity and quantity of plant species. Vertebrate and invertebrate animal species are in turn affected by how nitrogen inadequacy alters primary productivity, decomposition, and the food chain hierarchy. Approximately 6% of marsh nitrogen requirement is met by nutrient poor tidal import. Though floodwaters are high in nitrogen content, they are infrequent and move too quickly through the coastal wetland for significant plant uptake of nitrogen. The remaining nitrogen requirements must then be met by recycling or fixation from atmosphere.

14.5.1 Soil Nutrient Pools

Reef ecosystems have suffered from a variety of anthropogenic stresses during the past several decades. Offshore recreational activities, siltation, sewage discharge, excessive nutrient input, thermal pollution, (Rinkevich 1995a, b) and overfishing (Luoma 1996) have contributed to the declining health of many reefs. Divers, snorkelers, and boat and cruise ship anchors can physically damage corals by breaking them, abrading large coral forms, and burying coral fragments in sediment (Rinkevich 1995). Overfishing of algae-eating fish can lead to accelerated algae growth that in turn smothers corals (Luoma 1996). Anthropogenic stresses have contributed to the loss of 10% of the world's coral reefs and an additional two-thirds are at risk of serious decline (Luoma 1996). Zoologist Marjorie Reakea-Kudla predicts the loss of 70% of the world's coral reefs by the year 2036.

Various techniques have been employed to create habitats for reef-dwelling organisms. In Indian coastal waters, most efforts have focused on increasing fish populations to provide enhanced experiences for recreational fishing and sport-diving (Hutchings 1996). Activities such as the deliberate sinking of ships or oil drilling platforms successfully attract fish. However, it is unclear whether these structures provide suitable habitat for reproduction or just enhance the local population by attracting fish from other areas (Weisburd 1986).

New techniques have begun to focus on the restoration of coral reef ecosystems. Of all marine ecosystems, coral reefs are the most diverse and complex. The building blocks of the reef are symbiotic animals that rely on specialized algae called zooxanthelae for nutrients. Through photosynthesis, the algae produce sugars and starches. In exchange for these products, the corals share a portion of the nitrogen and phosphorus they obtain by capturing tiny organisms (Luoma 1996a, b). Some coral species produce calcium carbonate skeletons specially in the region of Gulf of Mannar, Tamil Nadu. The accumulation of these skeletal structures forms the coral reef. Each coral species, of which there are hundreds, has its own growth pattern (Sorokin 1993). Staghorn, elkhorn, and finger coral colonies produce branching shapes while flower, star, and brain corals produce rounded structures of various sizes (Greenberg 1986). Coral formations provide the framework for a diverse ecosystem that includes a multitude of fish, crustaceans, invertebrates, and other marine organisms.

The world's first, although unintentional, large artificial reefs were sunken ships. Ships, as well as military tanks (Heins 1995), a retired Boeing 727 (Fritz 1994), junked automobiles, old toilets (Weisburd 1986), tires and other scrap materials have been used to construct artificial reefs along the eastern and southeastern coasts of the United States. The procedures for the construction of these reefs vary depending on the types of materials used. Ships, tanks, planes, and cars are drained of gasoline, oil, and other polluting fluids. Generally, the engine and any buoyant materials are also removed (Meier and Martin 1985). The removal of brass fittings, commonly found in ships, is important because brass is toxic for some marine organisms (Reef Ball Development, Ltd. 1997). Deployment of ships is usually done with explosives. Ships are transported with tugboats to the target location and the explosives are detonated. Barges are used to transport cars and other materials to the artificial reef site.

Scrap tires have been used in artificial reefs off the coasts of Virginia, North and South Carolina, (Meier and Martin 1985) and Florida. One of the largest scrap tire reefs, the Osborne Reef off the Florida coast, contains nearly two million tires. An essential part of the preparation process includes punching holes in the tires to eliminate flotation problems. Goodyear has developed a device that punches three large holes in the circumference of the tire. Other devices, such as electric drills, are also effective. After the tires are punched, they are compressed into bundles of ten to twelve tires and bound together using nylon banding materials. If the location of the artificial reef is prone to stormy weather or strong currents, the tires must be filled with cement ballast. Boats or barges can be used to deploy the tires. More fish are attracted to the reefs if the bundles are stacked on top of one another (Candle 1985).

A relatively recent development (1993–1994) in artificial reef construction is the use of prefabricated concrete structures called Reef Balls. These hollow dome-shaped structures resemble natural coral heads produced by some coral species. They have holes of different sizes which allow fish and other marine organisms to enter the interior. The holes are designed to create a whirlpool effect inside the ball to feed the invertebrates and corals. Reef Ball design recognizes the fact that the profile of a structure alone will not support fish. Surface areas need current, light or both to be productive.

The early initiatives by the MSSRF even before the tsunami. The MSSRF launched a major programme in 1996 for restoration of MSSRF launched a major programme in 1996 for restoration of mangrove wetlands of the east coast of India, with financial mangrove wetlands of the east coast of India, with financial support under the India Canada Environment Facility (ICEF) and support under the India Canada Environment Facility (ICEF) and in collaboration with the Ministry of Environment and Forests in collaboration with the Ministry of Environment and Forests and State Forest Departments of Tamil Nadu, Andhra Pradesh, and State Forest Departments of Tamil Nadu, Andhra Pradesh, Orissa and West Bengal. In Gujarat, mangrove restoration project (REMAG) was In Gujarat, mangrove restoration project (REMAG) was implemented by the State Forest Department along with Gujarat implemented by the State Forest Department along with Gujarat Ecology Commission (GEC) during 2002 Ecology Commission (GEC) during 2002–2007 with financial 2007 with financial support from the ICEF. Palk Bay region in south-east coast of India is known for its mangroves, sea grass beds and fishery productivity. This bay separates India and Srilanka. Our project is aimed to establish community based mangrove protection sites. This mangrove sites are called as "seed banks" which is divided into several mangrove thickets. The sites are located on the mouth of rivers that enhances the natural seed dispersal through yearly monsoonal river flows into Palk Bay. Capacity building in local village representatives was the first phase, which includes site selection, species identification, adjusting land elevations for natural hydrology in the de-graded areas. The sites were selected and after obtaining per-mission from village management, the trained villagers has been involved in fencing, seed collection, nursery management. Several environmental awareness programmes like street plays, field trips, field research training programmes have been conducted to create awareness among youths in the coastal areas.

These successful, pilot mangrove restoration sites of Man-green Project are now emerging as the platform for local NGOs and University Research students who learn and implement the similar activities in other coastal areas. The villagers team are continuously are visiting other coastal areas to provide consultancy on mangrove restoration techniques. Indian coast harbors richly diverse and critical coastal habitats like coral reefs and mangroves. Mangroves form one of the most important ecosystems of coastal and marine areas. It safeguards the ecology of the coastal areas and provides livelihood opportunities to the fishermen and pastoral families living in these areas. In real sense, mangrove is the Kalpvriksh (divine tree which fulfills all the desires) for the coastal communities. The restoration and plantation of mangroves have received a lot of attentions worldwide.

To assess the impact of mangrove plantation activities and to monitor the mangrove regeneration and restoration in various villages, a joint study under the Integrated Coastal Zone Management Project (ICZMP) was taken up by Gujarat Ecology Commission (GEC) and Bhaskaracharya Institute for Space Applications and Geo-Informatics (BISAG) in the Gulf of Kachchh, Gujarat State. The major objective of this study was to monitor the increase in mangrove cover in coastal areas of Gulf of Kachchh using the Indian Remote Sensing Satellite data of 2005, 2011 and 2014. The mangrove regeneration was monitored using multi-temporal Indian Remote Sensing Satellite (IRS) LISS-III and LISS-IV digital data covering Gulf of Kachchh region. The multi-temporal IRS LISS-III data covering Gulf of Kachchh of October-2005, November-2011 and LISS-IV data of April-2014 was analyzed. The mangrove density and mangrove area in different talukas was estimated based on the analysis of IRS LISS-III digital data. The mangroves have been delineated based on the pink colour observed on satellite images and the area was estimated in the Geographic Information System (GIS) environment. The taluka or block-level mangrove areas were estimated and changes in the areas were monitored during the period of 6 years from 2005 to 2011. It was observed that the areas where mangrove regeneration activities were carried out with active participation of Community Based Organizations (CBOs), mangrove density as well as mangrove area have substantially increased in the Gulf of Kachchh region.

Under the large and medium size projects, mangrove restoration in approximately 200 km2 areas carried mainly with the support of Government of India. The restoration strategy is adopted based on the tidal amplitude. Consequently, the entire coastal area in the five States is divided in to (i) high tidal amplitude area; and (ii) low tidal amplitude area. In the low tidal amplitude area, ‘canal bank planting’ technique with ‘fish bone’ design is preferable for restoration; and, in the high tidal amplitude area, restoration will be made by direct seed sowing and seedling planting in the mud flats. Wherever communities are willing and can undertake restoration operations, the communities may be mobilized and funds may be transferred to them. In other areas, where communities are not involved or not dependant on the mangroves, the Government departments may have to step in to undertake restoration operations.

Coastal sand dunes (CSD) floras were under constant anthropogenic and natural pressure due to rapid elimination of sand dunes and its associated vegetation; as a result, its associated indigenous knowledge with them is also gradually disappearing. Such biodiversity rich and useful ecosystems need immediate restoration and conservation actions. Cuddalore coastal area is prone to both anthropogenic and natural disaster. Cyclone Thane hit Cuddalore coast on 29th and 30th of December 2011 with wind speeds of up to 135 kmph (83 mph) and tidal surges reaching 1.5 m (5 ft), is worth mentioning apart from the tsunami hit during December 2004. Industrialization has occupied nearly 500 acres of Coastal land which causes pollution and destruction of sand dune vegetation. Restoration of degraded area by propagation of plants (*Ipomoea pescaprae* and *Spinifex littoralis* (which are natural sand binders) by plucking a portion of the creeper from the denser area and planting it in pits dug at a depth of 30–40 cm. The planting of creepers was made at an interval of 2 m distance each in six pits. The restoration work was started from October 2012, 90% survival was found during the restoration study. The best season for this program in this area was between October and January. After 3 months, 30 cm of growth was observed in the plants. As that of mangroves restoration programmes this sand dune vegetation flora should also be encouraged by all the countries in the world.

14.6 Conclusion

Restoration is solely applicable to degraded ecosystems. Although it can be used as an effective tool to return the degraded coastal ecosystem to a pre-disturbance stage, coastal wetland restoration is also an important tool for preventing environmental degradation in India. In India, two coastal wetland ecosystem were restored with the ambient natural conditions. One is salinity is restored and another one is fish population is restored. Strengthening structural and functional elements through restoration can be helped to improve a coastal ecosystem’s health. India also witnessing the coastal restoration programms are under way in the coastal states of India.

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Chapter 15
Ecological Aesthetics Perspective for Coastal Wetland Conservation

LeeHsueh Lee

Abstract Ever increasing amounts of coastal wetlands are being destroyed; therefore, public participation in wetland conservation is important. Aesthetic preference provides a critical connection between humans and ecology that could greatly promote public awareness regarding conservation actions. The prospect-refuge theory and the preference matrix of the bioevolutionary hypothesis illustrate that aesthetic experience could drive landscape change, and pull with it ecological quality. Based on these concepts, a healthy coastal wetland with beautiful scenery might have high aesthetic value and generate positive emotional reactions that are preferred by people, thus encouraging them to venture further into such environments to explore. The comprehensive attributes of physical landscape and ecological function affect the aesthetics of coastal wetlands. In particular, 15 factors spread among four attributes influenced peoples' perception of the aesthetic and ecological quality of coastal wetlands. Attribute 1, the quality of the waterbody, is a predominant influence on the health of coastal wetlands, which appearance of waterbody affects aesthetic value. Attribute 2, the natural water edge of coastal wetlands could maintain the health of the wetland well and ensure the quality of the ecotone, which is perceived naturalness and high aesthetic value. Attribute 3 is the quality of terrestrial plants, which affects the diversity of wetlands and benefit to protect the terrestrial and aquatic habitats. Visual penetration of terrestrial plants is the main influential attribute of aesthetic preference. Attribute 4, the overall landscape of wetlands, shows peoples' attitudes toward wetlands, related to the quality and size of the wetland and the challenge of aesthetics.

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15.1 Landscape Aesthetics and Coastal Wetland Conservation

Natural scenes of waterbodies and mountains have been extolled and admired for thousands of years. However, wetlands are the most misunderstood landscape, often viewed in the past as wasted land because they are swampy, with bug-infested water. In Italian, the word for wetland means "bad air" because originally it was believed that swamp air caused terrible disease. For these reasons, many wetlands worldwide were drained and filled. Today, anthropogenic activities like as agriculture, commercialization, residential development, road construction, inpoundment, resource extraction, mining etc. results in continual wetland loss and degradation that can substantially reduce human well-being (Assessment 2005). The world has lost 64% of its wetlands since 1900 (Ramsar 2015). Wetlands were the most threatened of all landscape types, therefore it is important to study ecological aesthetics of wetlands to develop strategies to conserve them.

The aesthetics of wetlands have long been suspect; people say there is nothing picturesque in a dismal swamp. Usually, the only thing that might be of interest in wetlands is the bird population: waterfowl are the redeeming feature of wetlands (Rolston 2000). Some philosophers have expressed aesthetic appreciation for wetlands. Henry Thoreau (Giblett 1996) and John Muir (Giblett 1996, 2014) noticed and praised the beauty of wetlands from floral and wildness perspectives. Leopold studied the evolutionary natural history and ecology of wetlands and found them aesthetic. Both Callicott (2003) and Rolston (2000) set out to show the diversity and prolificness of wetland and greatly admired wetlands. For their diversity, dynamic stability, spontaneity, and life-support capacity, wetlands are a miracle of creation with marvelous ecosystem aesthetics.

> *Any fool can appreciate mountain scenery, it takes a man of discernment to appreciate the fens.* (Coles and Coles, 1989 p. 8)

High quality and well-maintained landscapes more accorded with the requirement in increasingly urbanized and industrialized environments. Such landscape planning, design, management, etc. might establish a desirable relationship between aesthetics and ecology, because ecology plays an important role in landscape management and sustainability development. Soaring populations is a main impetus of land requirement and induces global warming and climate change; that makes wetlands protection more and more challenging. Nevertheless, an ecologically aesthetic landscape is important for ecological conservation and for humankind's well-being (Gobster 2010; Gobster et al. 2007; Kovacs et al. 2006; Carlson 2008). The integration of ecology and aesthetic of wetland is needed to improve the awareness of wetland protection.

Ecologists and environmentalists believe that environmental education could raise environmental awareness of the public and impel them to take environmental protection actions, further nudging the ecological environment toward a sustainable future. Knight's (2008) research finding showed that when educational and interpretive learning programs were performed for protected species, the ecological protection awareness increased 35% for that species even if the people did not consider it beautiful. When people agreed the species was beautiful, protection awareness approached 90%. A healthy ecological environment with beautiful scenery might benefit conservation actions and sustainable management, and so on. Gobster et al. (2007) also indicated that a beautiful and healthy ecological landscape aroused positive emotions, which fueled a desire to protect such a landscape. Therefore, ecological aesthetics not only are of fundamental importance for conservation, but also a key factor in the success of conservation actions. Once a relationship between ecology and the aesthetics of coastal wetlands has been established, the conservation efforts are more effective thereafter.

15.2 Landscape Aesthetic Preference and Ecological Aesthetics

The aesthetic preference of a landscape is thought to be influenced by the bioevolutionary preference theory, a concept that explains aesthetic behavior from a survival perspective. First, the prospect-refuge theory states that individuals and groups are attracted to those environments that appear most favorable to live in. To see real potential dangers or search for their needs, people should visually control their surroundings, and preferred locations are found at interfaces between prospect-dominant and refuge-dominant areas (Appleton 1984, 1996). Landscapes that appear most satisfying to humans are those that provide an ability to see (prospect) without being seen (refuge) (Appleton 1984). The capacity of an environment for such ensures immediate achievement of aesthetic satisfaction. Based on this hypothesis, a choice of landscapes is necessary for the survival and promotion of the well-being of humans. People inhabited these kinds of landscape and change them implicit aesthetic preference of prospect-refuge theory also considers the influence on landscape management. Therefore, aesthetic experience not only drove landscape change, but also influenced ecological quality (Fig. 15.1). This kind of landscape was described regarding certain characteristics, such as naturalness, coherence, stewardship, and beauty.

In addition to evolutionary theory being an important concept of aesthetic preference environments, information-processing theory states that the arrangement of content affects people's aesthetic preference for environments (Kaplan et al. 1998; Kaplan and Kaplan 1989). The content arrangement provides an understanding and allows for exploration of an environment. Understanding provides a sense of security, as people feel distressed if they cannot comprehend a situation. People also have a desire to explore the environment to obtain more information. Together, understanding and exploration form the information preference matrix (Table 15.1) (Kaplan and Kaplan 1989; Kaplan et al. 1998).

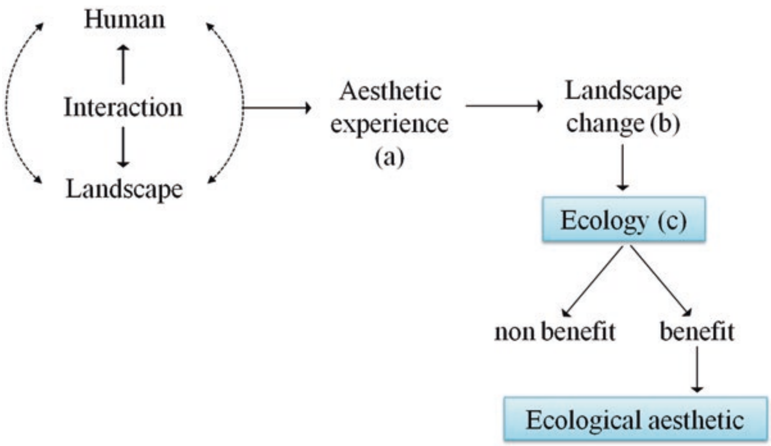


Fig. 15.1 Ecological aesthetic processed by human-landscape interaction
(a) Landscape aesthetics provides the linkage between human and ecological process, i.e., aesthetic experience drives landscape change
(b) Landscape change responded to the spatial-temporal milieu of the landscape; this response might appropriately be thought of as an aesthetic experience
(c) The aesthetic experience led people to change the landscape, either benefitting its ecological function or not

Table 15.1 Relationship between informational factors predicting environmental preference

	Understanding (provide information that can help people make sense of the environment)	Exploration (The promise of additional information imply there may be more to be seen)
Immediately	Coherence	Complexity
Inferred, predicted	Legibility	Mystery

Kaplan and Kaplan (1989, p. 53)

The four informational factors of the preference matrix are coherence, legibility, complexity, and mystery. Different environmental elements arrayed together, provide a sense of order; people would sense that environmental information with coherent perception. When people infer the environmental information, the elements are distinctive and easily identified, forming a sense of legibility. Richness in information offers many different kinds of distinctive elements in the environment, which relay complexity. A natural environment with high richness and diversity of natural landscapes would encourage people to explore it. A mysterious setting could compel people to venture in further to gain more information.

Specific landscape types evoke different aesthetic responses. Coastal wetlands have many forms and vary remarkably in size and character. According to the preference matrix, when environmental information is too complex, it leads to a lack of legibility that would make people feel perplexed and anxious, such as mangrove forest swamps. However, when information is too tidy and lacks variety, people may

Photo 15.1 (**a**) Gravel shore. (**b**) Marsh. (**c**) Coastal freshwater forest swamp. (**d**) Mangrove forest swamp

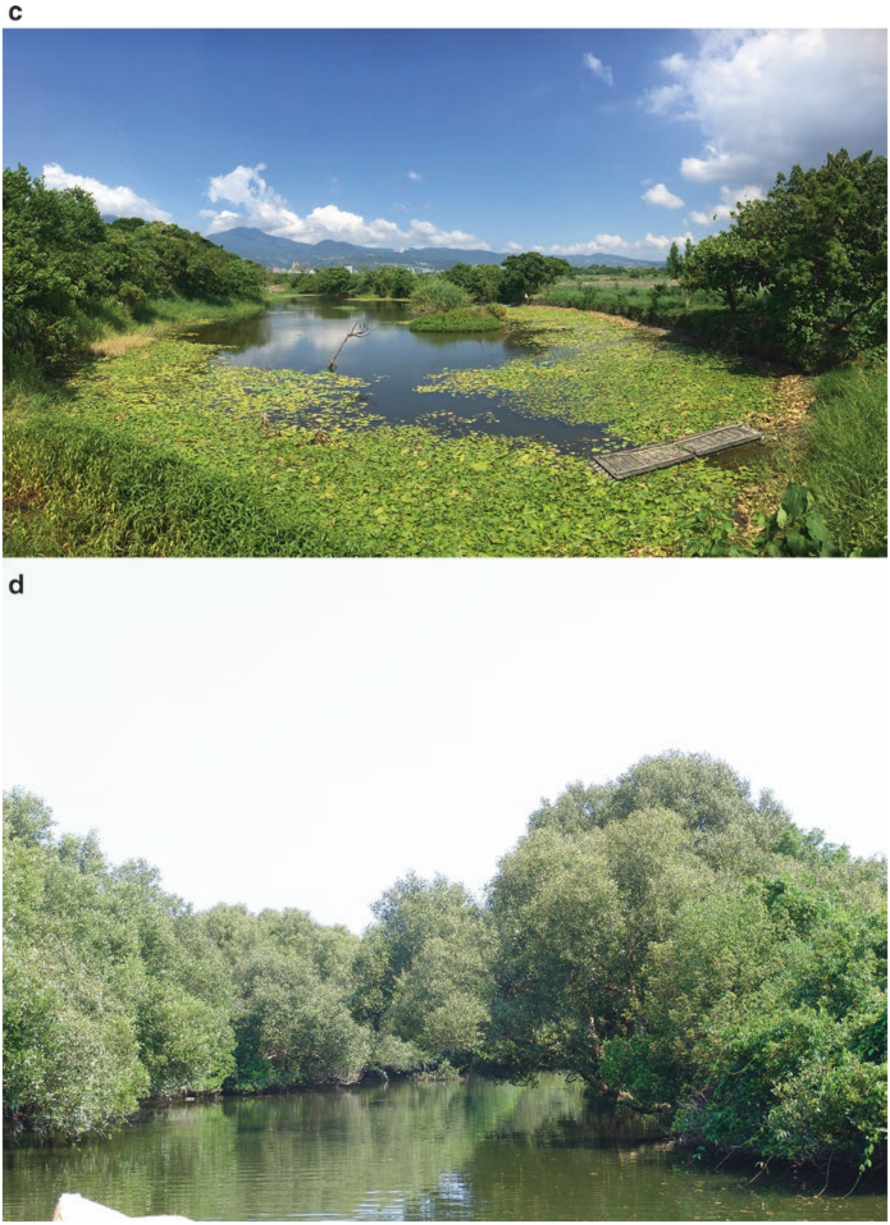


Photo 15.1 (continued)

want to quit because of the boring setting, such as mudflats. When information appears coherently and mysteriously, as in landscapes such as coastal freshwater marshes, it is likely to be favored because people may feel secure and interested in these types of settings (Photo 15.1a–d). Thus, the environmental information regarding various wetlands might allow the determination of their ecological aesthetic values. Aesthetic preference would make people change landscape, and thus ecosystem (Gobster et al. 2007). People desire to view, live, and visit beautiful places, and avoid or take measures to improve places that are perceived as unattractive (Kovacs et al. 2006). Sense of safety is also an influential factor on landscape preference (Hagerhall 2000; Herzog and Kropscott 2004; Herzog and Kutzli 2002). When a certain wetland made people feel in danger, people stayed away from it, or even worked to reclaim it.

Although wetland conservation actions are launching globally, wetlands are still at risk. The conversion of wetlands for various development and utilization projects is a major threat to wetlands everywhere. Healthy coastal wetlands of beauty could distinctly promote awareness of wetland conservation. It is imperative to give the ecological aesthetics of coastal wetlands a place of prominence in conservation efforts.

Photo 15.1a–d Environmental information from less to abundant, from coherence to complexity, respectively is gravel shore, marsh, coastal freshwater forest swamp and mangrove forest swamp, according to that, the openness of vision and accessibility is variable. An unobstructed foreground, water body may sense spaciousness, which setting is perceived coherence and legibility; and with complex middle to background, would induce people go deeper to explore (Photo 15.1c). The plants diversity is lower at mangrove forest swamp, which setting is perceived high coherence with low visual penetration. The enclosed of vision often blocked people's travel (Photo 15.1d).

15.3 Relationship Between Ecological Aesthetics and Coastal Wetlands

Most people think that landscape and ecology differ, with different contents of environmental experience as well. In general, we think that the public enjoys the landscape, whereas ecological experts are interested in understanding the ecology. Traditionally, we get a sense of stewardship, coherence, order, etc. from landscapes and diversity, complexity, and disorder from the ecological environment. Scenic appreciation focuses on the arrangement of visual elements that appear in a picturesque landscape and cause a pleasurable emotional reaction. The process of perceiving is immediate and the benefits of aesthetic response are taken more at face value, and in the short term. Ecological experience is engaged to the interaction between humans and the ecology, and its view is biocentric. The knowledge base of ecological experience is a process to understand and enjoy the intact ecosystem and arouses a pleasant emotional reaction. The benefit is long lasting and may encourage people to participate actively in environmental conservation.

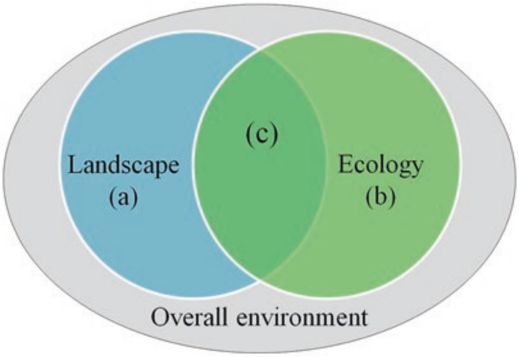


Fig. 15.2 The hypothetical relationship between landscape and ecology
(a) Landscape: It is easy to overrate the arrangement of visual elements. In general, the public appreciates the visual aesthetic of the landscape, but ecologists dislike it
(b) Ecology: The major focus is on the ecological functions and life-support services of a wetland, which is favored by ecologists, but not preferred by the public
(c) Ecological aesthetics setting: Both ecologists and the public alike prefer a healthy wetland with beautiful scenery

Aesthetic experiences, the outcomes of human-landscape interaction, have caused human-influenced environments and some different aspects of these environments. Because people so powerfully affect environmental phenomena in a way, landscape and ecology need to be considered together, from a holistic view, with landscape and ecology overlaid. It has a setting in both a healthy ecology and a scenic landscape (Fig. 15.2), which both ecological experts and the public might prefer. Seeking and promoting sustainable coastal wetlands agrees with both ecological experts and the public, who tend to be the least satisfied with management and conservation practices, whose success depends on healthy and attractive wetlands. The quality of ecological function is important not only for landscape setting, but also on people's cognition of the setting. The emotionally arousing experience is brought with landscape aesthetic preference (Callicott 2003; Cottet et al. 2013; Dobbie 2013; Rolston 2000; Sheppard 2001; Ulrich 1983). People become emotionally attached to a place, and such emotional preference might lead to choices of environmental behavior (Gobster et al. 2007; Kovacs et al. 2006; Sheppard 2001). This has been identified as one of the most important drivers of landscape change that strongly influences ecology and environmental management aesthetic preference, determining whether landscapes are viewed as beautiful or ugly, and sustainable or threatened.

A healthy wetland ecosystem can be defined by it wetland, including all of its biological and physicochemical parameters and their interactions that provide ecological and economic functions (Mitsch and Gosselink 2000). The appearance of landscape setting might change the perseverance of ecological statues of wetlands, which lead the stimulation of different emotional response in human. Ecological experts favor a healthy wetland with high diversity and good water quality. Such a wetland appears natural, alive, and beautiful, leading to the arousal of positive emotions. The public also prefers this kind of wetland. Conversely, for a threatened wetland, the public would perceive its water as unclear, suggesting that the water was polluted. This might not only arouse negative emotions, but also would not be admired aesthetically. In general, the public may not so care about the future of the wetland. Overall, ecological aesthetics were clearly a main influential factor on wetland conservation.

15.4 Landscape Perception and Aesthetics of Coastal Wetlands

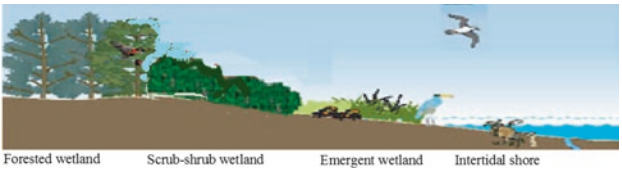
Many different kinds of classifications have been applied to coastal wetlands. A landscape is a complex phenomenon, which evolves continuously through time and space. A coastal wetland landscape is formed by a combination of physical features of the environment, such as tide, current, and waves etc., which determine the geomorphology, vegetation, habitats, and related to biological characters. Coastal wetlands greatly vary in landscape. Landscape perception is the interaction of humans and the landscape. One conceptualization of landscape perception is a full range of landscape experiences for people that can lead to the attribution of meaning and the valuing of specific landscapes.

In general, coastal wetlands are low-lying areas, periodically flooded by tidal waters for varying lengths of time (Tiner 1993), covered by vegetation of different heights, and configured to make landscape room (human's perceptual unit), which has a deep stake in the overview and visibility of the coastal wetland landscape and relates to human landscape preference. Hydrology, landform, vegetation cover, etc. are all concerns for the visible area of a waterbody. Water has long been recognized as an important element in landscape preference, particularly regarding water quality and visible area. More importantly, how water relates to the sustainability of a wetland has affected how people approach recreational activities at the shoreline. Substrate material is affected by seafloor slope; generally consists of mud, sand, gravel, or cobbles; and not only determines the characteristics and functions of a coastal wetland, but also affects recreational activities undertaken by people.

When people go into a landscape, the experience of overall landscape context and composition draw out scenic beauty value and emotional reactions. A coastal wetland landscape contains a variety of environmental features; vegetation cover, substrate, and the visibility of water, etc. all of which contribute to the overall scene and create different aesthetic values and emotional arousals.

The landscape of a coastal wetland is categorized as follows from water to land: intertidal shore, emergent wetland, scrub-shrub wetland, and forest wetland. These four not only differ in ecology and habitat, but also in landscape and aesthetic perception and recreational activities. People perceive the openness, complexity, visual penetration, etc. as considerably varied for these four types of coastal wetland landscapes. According to Kaplan's (Kaplan et al. 1998) preference matrix concept, the environmental information of these four types of wetlands that serve the people by letting them understand or explore the coastal wetland are different, too. Details of these wetlands are explained further below and are shown in Table 15.2.

Table 15.2 Landscape perception and environmental information for coastal wetlands

			
Landscape elements	Varied/numerous	←·······→	Uniform/few
Arrangement of landscape elements	Complicated	←·······→	Uncomplicated
View shed	Limited	←·······→	Expanded
Depth of view	Near	←·······→	Distant
Landscape openness	Enclosed	←·······→	Open
Visual penetration	Low	←·······→	High
Environmental information	Complex	←·······→	Coherent
Landscape room	Diverse	←·······→	Nondiverse

The intertidal shore gives a sense of large horizons and a feeling of openness, which spread environmental coherence information to people. However, various substrates (e.g., mud, sand, shingles/pebbles, cobbles and rocks) of the intertidal shore could create distinct landscape experiences. Compared to mud and sand flats, a rocky shore is more attractive because of the jagged rocks and beautiful waves that spray. Rich benthos and migratory birds abound, with shorebirds at the mud flats and recreational activities popular at this area.

Emergent wetlands are in the upper coastal intertidal zone between land and open salt water or brackish water that are often dominated by dense stands of salt-tolerant plants, such as grasses, herbs, or low shrubs (Adam 1993). Marshes have scenic beauty, abundant of natural resources and accessibility could provide diverse recreational activities, hence marshes are highly attractive natural feature to people. As the environmental information is coherent and people perceives complexity in the spacious marsh. The visible area of the waterbody also influences the aesthetic assessment. The large waterbody, with birds, mammals, and fishes in the marsh, would make the people's preference and aesthetic value high.

Scrub-shrub wetland dominated by woody vegetation less than 6 m tall. Wetland visual penetration may be obstructed that could not provide a sense of security and is reasonable to interrupted people further explore in deep. As the visible area of waterbody is broader and lies at foreground, and some conceivable and memorable landscape feature to help people perceive legibility. Scrub-shrub wetland landscape could encourage people to move deeper to gain more environmental information. This kind of scrub-shrub wetland ought to get high aesthetic value and preference. The wetland is good habitat for fish, avifauna, wildlife etc., and support a wide range of recreational activities.

Coastal forest wetlands are generally known as swamps, which dominated by woody vegetation six meters or taller. For visual character, diversity of landscape elements and land cover are two important dimensions that also related to habitat heterogeneity. Complexity and legibility, both are seen to have effects on human landscape perception and preference at swamp and attract people explore deeper into the setting. Meanwhile, the scale of landscape room is a concept in ecology and aesthetic. Landscape room concerns to depth of view and degree of visual penetration. Both increases may associate with higher aesthetic and landscape preference like as coastal fresh-water swamp. Landscape room has a great deal with the edge and core habitat, and species, age and vertical structure of vegetation biodiversity are important attributes in ecological functions in landscape ecology at coastal forest wetland (Fry et al. 2009). These attributes have discernible effect on human landscape perception of coherence and complexity, i.e. relates to understanding and exploration of coastal forest wetland.

15.5 Influential Factors on the Ecological Aesthetics of Coastal Wetlands

The influence factors of ecological aesthetic were conducted in some common grounds between landscape-visual and ecological character, and which factors could be used to connect both together. The compositional factors demonstrate landscape form pattern and ecological function; integration of factors is a critical linkage of landscape and ecology, and shows the spatial-temporal milieu of landscape, i.e., ecological aesthetic of environment. So, bases on the relationship between physical landscape-ecology attribute to scenic beauty, which has an effect on ecological status and aesthetic estimate.

The quality of ecological functions is important for not only landscape setting but also human cognition. Its aspects could arouse emotional reactions with landscape aesthetic preference (Callicott 2003; Cottet et al. 2013; Dobbie 2013; Rolston 2000; Sheppard 2001; Ulrich 1983). Certainly, a healthy wetland has rich natural diversity of plants and animals; meanwhile healthy wetlands make high aesthetic value, no doubt about it.

Physical landscape of coastal wetland relates to ecological aesthetic is divided into four attributes, that is waterbody, water edge, terrestrial plants and overall landscape, total include 15 factors. For more detailed description of each factors as shows below (Table 15.3).

Table 15.3 Physical landscape attributes and factors of coastal wetlands and how they relate to landscape aesthetics

Low◄……………	Aesthetics	……………► High
	#. Attributes/(#) Factors	
	1. Waterbody	
Small	(1) Visible area	Large
Low	(2) Water transparency	High
Yellow-brown	(3) Water color	Blue
Thick	(4) Sediment	Thin
Floating	(5) Aquatic plants	Submerged
Turbulent/larger	(6) Wave action/size	Peaceful/small
	2. Water edge	
Artificial Sparse to none	(7) Natural shoreline	Natural
	(8) Natural vegetation	Plentiful
	3. Terrestrial plants	
Low	(9) Diversity	High
Low	(10) Richness	High
Low	(11) Evenness	High
Simple	(12) Stratification of vegetation	Multilayered
	4. Overall landscape	
Enclosed	(13) Openness	Open
Foreground/near	(14) View distance	Middle ground to background/distant
High-density	(15) Surrounding land-use	Low-density

15.5.1 Waterbody

Water quality and wave is a predominant influence in the health of coastal wetland, and which opened out in landscape appearance of waterbody would affect aesthetic value. Waterbody usually locates at foreground of coastal wetland landscape, the factors of waterbody composition provides environmental information relates to coherence and legibility of cognitive aspects.

The attribute of waterbody include six factors that is water area, water clarity, water color, sediment, aquatic plants, and wave size.

The water quality and wave action have effects on dissolved oxygen of seawater. In water, dissolved oxygen is an essential factor for aquatic life that also disturbs sediment and causes an increase in suspended solids. Scattering from suspended particles plays an important role in the water color of coastal wetland.

Water clarity is affected by water quality, high water clarity with high landscape aesthetic value. Alga, aquatic plants, and suspended particles also are the appearance of water quality, and has a lot of pull with water clarity, water color and the visible water area, all are main factors that determine human's aesthetic and preference perception on coastal wetland. Blue water color attracts people closes to water

a

b

Photo 15.2 (**a**) The vertical stratification location on middle-back ground make wetland has greater visible water area. (**b**) The vertical stratification location on foreground could block human's visual penetration

Photo 15.3 (**a**) The *flat blue-green* calm of the sea could get high aesthetic value. (**b**) The choppy or boiling sea might arouse people's negative emotion

and could get high aesthetic value. Landscape visual quality increases with the area of water visible (Photo 15.2a–b).

Furthermore, the wave size not only affects people access to water, but also affects the landscape aesthetic of water of coastal wetland. The smooth, sparking, or small wave spring on the coastal wetlands that is more preferred and could make high aesthetic value. On the contrary, the big waves might arouse people's dangerous emotion, and people generally do not want to approach to coastal wetlands (Photo 15.3a, b).

Photo 15.2a,b The visible water area has effect on ecological aesthetic. When water area is covered by numerous aquatic plants that degrade water quality, and water clarity, water color. Vertical stratification of vegetation affects animal communities; meanwhile that has a dominant influence on landscape room. The vertical stratification location on middle-back ground (Photo 15.2a) could get high aesthetic value than located on foreground (Photo 15.2b).

Photo 15.3a, b Wave size, it affects human's perception of coastal wetland landscape. Compared to boiling wave, the calm or sparkling water might arouse people's safety and peace emotion. Wave disturbed suspended particles. The processing affects the scattering of light, and water color, that also influence preference of coastal wetland.

15.5.2 Water Edge

Wetland, itself has been considered as ecotones or ecotonal habitats; meanwhile a nature water edge of wetland is one of main influence factor on healthy coastal wetland. In general, the water edge of waterbody is related to the naturalness perception of landscape, and coherence and complexity in human cognition also could be traced into this attribute.

The water edge means the status of shoreline that includes natural shoreline, and the water edge is well defined by vegetation. The good quality water edge not only has effect on landscape aesthetic, but also relates to the quality of waterbody (Photo 15.4a, b).

The water shoreline may be natural or artificial; bare, construction or planting. The natural shoreline is more aesthetically pleasing than artificial, sharp straight edges. Meanwhile the functions of natural shoreline and natural vegetation along the shoreline of coastal wetland include water purification, erosion protection and improved wildlife habitat etc.

Bare and arid shoreline enables people considers the wetland was threatened, and have negative impacts on the quality of ecology and aesthetic. Aquatic and terrestrial planting seemed to blend into each other, or the shoreline planting is disordered. The obscure shoreline and waterbody would make people perception unsafe in this setting, and dislike it. The shoreline is well defined by natural vegetation would make high aesthetic value of coastal wetland.

a

b

Photo 15.4 (a) The shoreline stood out in relief, and which was well defined by vegetation. (b) Shrubs and emergent plants stretched so far forwards as almost to blend the land with the watershore

Photo 15.4a, b Natural shoreline, with well defined by vegetation could get high aesthetic value (Photo 15.4a); conversely, aesthetic value is lower (Photo 15.4b). That is related to accessibility of coastal wetland, due to the former made people perceive safety, people may feel instability in the latter.

15.5.3 Terrestrial Vegetation

Some species need both terrestrial and aquatic habitats, and terrestrial areas perform highly many ecosystem functions important to the provision of ecosystem service. Terrestrial plants quality has an impact on the diversity of wetland, and landscape aesthetic preference. Terrestrial plants area surrounds coastal wetland. As people go into this setting, terrestrial plants area usually locates at middle-background, its composition could make people perceives complexity and mysterious that induce people go deeper to explore or not.

The terrestrial vegetation closely related to ecological aesthetic of coastal wetland that concerns with diversity, richness, evenness, and stratification of vegetation.

Diversity, richness, and evenness of terrestrial vegetation provide availability of information for people, and arise concerning people's cognition aspects at coastal wetland landscape. Their composition related to habitat quality and ecosystem of wetland.

The diversity of species, age and shape of trees (Minckler 1980), and the vegetation diversity of grassland (Lindemann-Matthies et al. 2010), all have attractiveness and high aesthetic valve at wetlands. The evenness make appreciation of coastal wetland landscape is the coherent perception to people; that environmental information provides an understanding in the setting. The diversity and richness are considered about the complex and mysterious perception; that impels people goes deeper into and explores this landscape.

Perceived species richness had a strong influence on people's aesthetic appreciation, particularly in the evenness experiment (Lindemann-Matthies et al. 2010; Tilman et al. 2001), and increase visual complexity with increasing species richness might explain the high aesthetic appreciate. Humans prefer landscape scenes with levels of moderate to high complexity (Kaplan and Kaplan 1989; Kaplan et al. 1998; Leder et al. 2004; Lindemann-Matthies et al. 2010; Ulrich 1986). People felt discomforted as the view is blocked in dense vegetation of swamp that wetland has rich contents it lacks of clear focus. The plant diversity of swamp must be kept moderately mystery as much as possible. On the whole, diversity, richness and evenness can help people make sense, and has effects on ecological aesthetic of coastal wetland (Photo 15.5a–d).

Psychological theories of aesthetic response are used to deduce a variable, i.e. visual penetration. Stratification of vegetation means the arrangement of vegetation in layers refers to vertical layering of a habitat. Terrestrial vegetation of coastal wetland formed by upper story and under story; the visual penetration is better in

a

b

Photo 15.5 (**a**) The species evenness of plants in marsh. (**b**) The coastal wetland with arid and yellowing grassland. (**c**) The emergent plants are too higher to visual penetration for people. (**d**) A highly complex landscap of coastal wetland that setting has too environmental informations made people could not clear grasp and understand

c

d

Photo 15.5 (continued)

this setting. According to Kaplan's information-seeking preference behavior, these kinds of coastal wetlands are the most preferred landscapes have elements of mystery and involvement, and could make higher aesthetic value. If a coastal wetland only has under story of terrestrial vegetation that is low aesthetic value, especially yellowing vegetation of under story, which does not coincide with the pattern of humankind's preference landscape (Photo 15.2a, b).

Photo 15.5a–d The vegetation has species evenness that landscape would be appreciated, such as marsh (Photo 15.5a); however, arid and yellowing vegetation has low aesthetic value (Photo 15.5b). Vegetation composition and location also have influence on ecological aesthetic. The landscape aesthetic preference is low at an lack visual penetration coastal wetland, due to people perceive complex, but no legibility (Photo 15.5c, d); and too complexity landscape not only gave people a sense of insecurity, but also stopped people access to the setting, such as mangrove forest swamp (Photo 15.5d).

15.5.4 Overall Landscape

Evolution-base theories, research has shown that most preferred natural landscape typically possess open areas with low groundcover a water source directly or indirectly apparent and scattered clumps of trees and shrubs, savanna-like setting (Hill and Daniel 2007). That overall landscape of coastal wetland could put on this archetype landscape. The overall landscape includes openness, view distance, and surrounding land use.

Openness is an aspect of the visual landscape that is strongly related to perceived visual quality and landscape preference (Herzog 1987; Tveit et al. 2006; Nijhuis et al. 2011). According to prospect-refuge theory, people prefer a landscape that offers various options for cover with an overview of large open space. Thus, a balance of open and enclosed landscape is more preferred, moderately openness made people immediately understand and inferring the environment, and a moderately enclosed view at a distance could induce people further explore.

On the other hand, an open area of landscape is bordered by such as strips of taller vegetation, topography, forest etc. that related to the size of patches and affects the ecological of individual patch through its effect on edge and core habitat (Tveit et al. 2006; Baguette and Van Dyck 2007). The environmental information of wetland is high-coherence, people may perceived monotonous at this environmental like as mudflat, estuary marsh. These types of wetlands are unstable and unpredicted environment due to tides variation and climate effect is explicit that r-strategist species are more flourishing. In contrast, environmental information is too complicated and is not in the hands for people, like as mangrove swamp, coastal forest wetland; that environment is more stable for provide resistance to climate such as high winds. The K-strategist species would be vigorous. Like as coastal fresh-water swamp and a marsh with background of forest have a balance of open and enclosed landscapes. According to information-seeking theory and environmental information preference matrix, people perceives this wetland's foreground is coherence and middle-background is complexity and mysterious. The environmental information is easier to understand to people; furthermore, complex environmental information in the distance made people goes deep into exploration. This wetland setting is preferred by people. Meanwhile, this kind of wetland in both changing and stable environment, K and R strategist species coexist in this wetland, and species diversity and abundance is higher than others. The above imply the openness of landscape of coastal wetland has discernible effect on ecological aesthetic.

View distance is a critical factor in landscape preference (Jorgensen et al. 2002), and viewshed is deep and wide could determine visual quality (Appleton 1996). In terms of aesthetic preference, the long view distance of landscape is more interesting for people because the landscape itself contain more environmental information. Visual penetration is also dependent on the spatial arrangement of vegetation and view distance. The interaction of view distance and vegetation density has effect on landscape preference. Whereas sparser vegetation was more preferred to denser vegetation in close views, the visual penetration of former is better than latter.

Wetland's ecological aesthetic is not only wetland itself but includes the surrounding landscape of wetland. The surrounding land use has impacts on wetland landscape and ecology, especial the direct people's field of view. Human engage in environment that encompasses on-site and off-site area. The interactions give to raise aesthetic experience.

Land use activities adjacent to wetlands can affect wetland habitat by altering inputs of sunlight, sediment, nutrients, hydrology etc. (Batzer and Sharitz 2014) that changing the quantity and quality wetland (Houlahan and Findlay 2003; Karstens et al. 2016). Different surrounding land use is reflection of people-social situation and concern for wetlands that exposed environmental information would affect people value ecological aesthetic of wetland.

Land use intensity affects the patch size of wetlands; meanwhile, has effect on the scale of the people's naturalness perception. Vision field of high-intensity land use adjacent to wetlands will make people arouse negative emotional reaction such as dislike, bored because of perceived unnaturalness, species-poor. Conversely, lower-intensity land use is positive contribution to scenic beauty and usually benefits to wetland ecology. Effective buffer zone is important to ensure the quality of ecological aesthetic of coastal wetland (Castelle 1992; Castelle et al. 1994; Office of Long Island Sound Programs 1994).

15.6 Conclusion

Coastal wetlands are not only the important habitats but also the beautiful places; coastal wetlands are too valuable to lose (National Oceanic and Atmospheric Administration [NOAA] 2016).

Coastal wetlands play a number of roles in the environment and are considered the most biologically diverse of all ecosystems. Coastal wetlands are various in forms and scale that also made the landscape is interesting, and this diversity is a conservation challenge. Nevertheless, many coastal wetlands are being threatened in everywhere. General public does not dislike land; general public does not dislike water. But we often view the land-water with great skepticism. Due to legislate is often used in practice protection of coastal wetlands. However, public participation is also very important.

The healthy ecology and aesthetic landscape are combined together has been thinking deeply. Landscape aesthetic and ecology of collaborative ecological aesthetic to help understand how the importance of aesthetics affects landscape change, thus ecological function.

The perspective of ecological aesthetic will make people change the mistakes of coastal wetlands, which would be more ideal approach to appreciate coastal wetlands; in this way, how best to protect coastal wetlands.

The possibility of ecological aesthetic affects coastal wetland conservation, planning, design, construction, and management; while these issues are closely linked with ecological aesthetics that is conducive to coastal wetlands conservation, and correspond with Ramsar's voice "the wise use of wetlands".

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Chapter 16
Estuarine Ecoclines and the Associated Fauna: Ecological Information as the Basis for Ecosystem Conservation

Mário Barletta, André R.A. Lima, Monica F. Costa, and David V. Dantas

Abstract Ecocline is defined as a "gradation from one ecosystem to another when there is no sharp boundary between the two" containing relatively heterogeneous communities influenced by gradual changes between river-dominated to marine-like waters. It creates heterogeneous habitats, differing in abiotic characteristics, mainly water salinity. Estuarine fish fauna is highly influenced by the major annual water quality shifts resulted from seaward river flow during the rainy season and upstream coastal water inflow during drier seasons. Thus, faunal communities change seasonally in terms of quali-quantitative variables or living strategies. Estuarine ecocline can also set the seasonal retention, bioavailability or sinking of dissolved oxygen, pollutants and microbiological contaminants whose effects are crucial to determine the pattern of use, fish entering, tissue contamination and survival of early stages. Abrupt changes in climatic patterns or in the river flow induce changes in the ecocline and fishes will respond by modifying assemblage structures. The use of robust and consistent scientific information regarding fish fauna and their ecocline can provide reliable ecological information. This generates descriptors of reference conditions taking into account how human impacts affect coastal systems, providing steps to guarantee the sustainable use of estuarine resources.

Keywords Environmental gradients • Marine pollution • Coastal management • Anthropogenic impacts • Planned sampling design • Estuarine resources conservation

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16.1 Introduction

An ecocline can be defined as a "gradation from one ecosystem to another when there is no sharp boundary between the two" containing relatively heterogeneous faunal communities influenced by progressive changes between the two systems (Attrill and Rundle 2002; Barletta and Dantas 2016). To these boundaries, it is usually attributed environmental grades as upper, middle and lower reaches. Each of these habitats can actually shift horizontally, moving seaward and back according to rainfall patterns (Barletta et al. 2005, 2008) or upstream human interventions (Morita and Yamamoto 2002; Dallas and Barnard 2009; Barletta et al. 2016).

Estuarine ecocline have been always defined as the strongest forcing structuring faunal populations and communities within estuarine ecosystems (Bolton 1979; Calder and Maÿal 1998; Barletta and Blaber 2007). A study in the estuary of the Silver Burn, Castletown, Isle of Man, U.K., demonstrated that *Pilayella littoralis* (brown algae, Phaeophyta) presented a gradual transition from moderately to markedly euryhaline populations according to the salinity regimes of the sampling sites (Bolton 1979). Another study regarding hydroids (Cnidaria – *Clytia gracilis*, *Obelia bidentata*, *Garveia franciscana*) distribution patterns along the ecocline of a small estuary of northeast Brazil (Formoso River), demonstrated that changes in hydroid species composition were attributed primarily to differences in salinity characteristics from one site to another (Calder and Maÿal 1998).

Therefore, estuarine ecocline produces boundaries that result in a suite of faunal communities along the gradient, changing seasonally in terms of quali-quantitative variables or living strategies. Estuarine ecocline can also set physico-chemical conditions that determine the bioavailability or sinking of dissolved oxygen, and the concentration of pollutants (plastic debris, metals, emerging pollutants) (Lima et al. 2014; Costa and Barletta 2015; Costa et al. 2009; Reis et al. 2016) and microbiological contaminants (from sewage, urban runoff and animal farming) throughout the salinity ecocline (Costa and Barletta 2016).

Considerable information on estuaries and their fish fauna are available for the South American Atlantic border (Barletta et al. 2010; Blaber and Barletta 2016; Costa and Barletta 2016). However, to compile and interpret this body of literature turning it into useful managerial information is a daunting task for anyone, but a most needed step to prompt well-informed decision-making. This is especially complex if transitions among environments need to be considered, as is the case of the Western Atlantic, which extends from Equatorial to Sub-tropical regimes (Beck et al. 2001; Costa and Barletta 2016). Estuaries along this latitudinal gradient are transitional themselves, and provide a variety of habitats for fishes, at all life stages, that take advantages of available resources, especially the high productivity and shelter, to complete their life cycles (Able 2005). These systems can be characterized by high densities of few species (catfishes, croakers, drums); larval fishes and; smaller fishes (anchovies, herrings, sardines) that are trophic links to fishes of commercial and subsistence interest (Barletta-Bergan et al. 2002a, b; Dantas et al. 2012; Ferreira et al. 2016; Lima et al. 2015, 2016; Ramos et al. 2016).

On a regional scale, the Western Atlantic coast bordering South America encompasses a variety of estuaries that result from drainage basins crossing from tropical to temperate climates, and debouch into coastal and marine regions subject to different oceanographic forcing (Blaber and Barletta 2016). In most of these systems, the major annual water quality shifts result from seaward river flow during the rainy season and upstream coastal water inflow during the dry season (Barletta et al. 2005, 2008, 2010; Lima et al. 2015). River flow, and the extent of its influence over the estuarine system, is proportional to rainfall patterns and human intervention aiming at the use and control of water resources, such as damming and hydropower stations (Almodóvar and Nicola 1999; Morita and Yamamoto 2002). Although rainfall patterns changes among estuaries, environmental variability responds to a combination of factors that include riverine and estuarine basin geomorphology, microclimates and habitat modification. Thus, the main channel of estuaries are a composition of heterogeneous, temporary, habitats which differ among themselves in abiotic characteristics (mainly water salinity) varying from freshwater to marine water, known as estuarine ecocline (Barletta and Blaber 2007; Barletta and Dantas 2016).

When rainfall and river flow increase, pollutant loads move together with shifting abiotic habitats to other estuarine areas downstream. Although somewhat diluted, pollutants become bioavailable for a greater diversity of organisms over the whole ecocline. Abrupt changes in climatic patterns or in the river flow regime (dredging operations) will induce changes in the estuarine ecocline and fish communities will respond to these changes by modifying assemblage structures (Barletta et al. 2016; Prestrelo and Monteiro-Neto 2016). However, despite all impacts suffered for these systems, climate and hydrodynamics contribute to reduce pollution, especially during the rainy season, through biodilution and transport of contaminants out of the system to adjacent coastal waters, in a process known as environmental homeostasis (Elliott and Quintino 2007; Barletta et al. 2016).

Studies regarding estuarine ecocline have increased in term of quality and quantity aiming to provide consistent recommendations for conservational issues (Barletta et al. 2005, 2008; Barletta and Blaber 2007; Storm et al. 2005; Prestrelo and Monteiro-Neto 2016; Slater 2016). However, efforts of governmental agencies for conservation, recovery and sustainable use of estuaries are almost always implemented regardless of basic concepts of estuarine variability and the influences of human interventions, leading to poor management practices (Elliot and Whitefield 2011; Dauvin and Ruellet 2009; Barletta et al. 2010; Machado-Allison 2016).

Three important and well-studied examples exist of estuaries that represent the different stages of the latitudinal gradient along the western Atlantic, from the Equator to subtropical climates. The Caeté Estuary (0°45′–1°07′S; 46°25′–46°50′W), the Goiana Estuary (7°32′–7°35′S; 34°50′–34°58′W) and the Paranaguá Estuarine Complex (25°15′–25°35′S; 48°20′–48°45′W) are systems where the fish fauna were year-round characterized along their ecocline, and from where information to support managerial decision is ready to be used. These three systems represent the humid tropics, the dry tropics and the warm temperate, respectively. Different policies for the conservation of coastal ecosystems and traditional populations in the Amazon, northeast and southern Brazilian wetlands still exist (Barletta

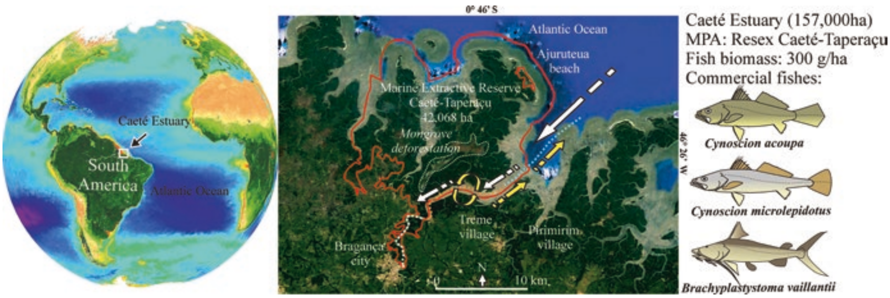

Fig. 16.1 Caeté Estuary. *Doted lines* indicate the portions of the main channel [(■) *upper*; (■) *middle*; (■) *lower*]. *White* and *yellow arrows* shows the main fish movement during the dry and rainy seasons, respectively. *Rounded yellow arrows* indicate the prefered habitat in the late rainy season. *Red line* shows the total area of the conservation unit Resex Caeté-Taperaçu. In the *right side* is presented the total biomass of the most important commercial fishes produced during a seasonal cycle

and Costa 2009; ICMbio 1995, 2012). However, the inefficient communication between academy and institutions responsible for structuring the fisheries conservation scenario, and the sustainable use of estuarine and marine resources are a main concern (Barletta et al. 2010; Costa and Barletta 2016).

Here in this chapter there is suitable information on how to work in favour of estuarine ecocline and how robust sampling designs can provide consistent ecological and statistically testable data, taking into account the overlapping of abiotic variables and life cycles of key species for different estuarine reaches and seasons. This generates descriptors of reference conditions and ecological information that must be used as the basis for Western Atlantic regional coastal systems conservation.

16.2 Study Areas and Fluctuation of the Salinity Ecocline According to Local Rainfall Patterns

16.2.1 Caeté Estuary

The Caeté Estuary (~157,000 ha) is a representative of the northern Brazilian basins (Eastern Amazon) (Fig. 16.1). Its climate is hot and humid and annual precipitation exceeds 2545 mm. The rainiest period is concentrated in the first semester of the year. Due to its near-Equatorial latitude, temperature remains above 25 °C. Salinity and main channel width increase from the upper to the lower estuary (Barletta et al. 2005). The middle estuary is characterized by mesohaline conditions, and the lower estuary is strongly influenced by coastal waters (Fig. 16.2). Vertical stratification of the water column occurs mainly during the rainy season, in the outer part of the

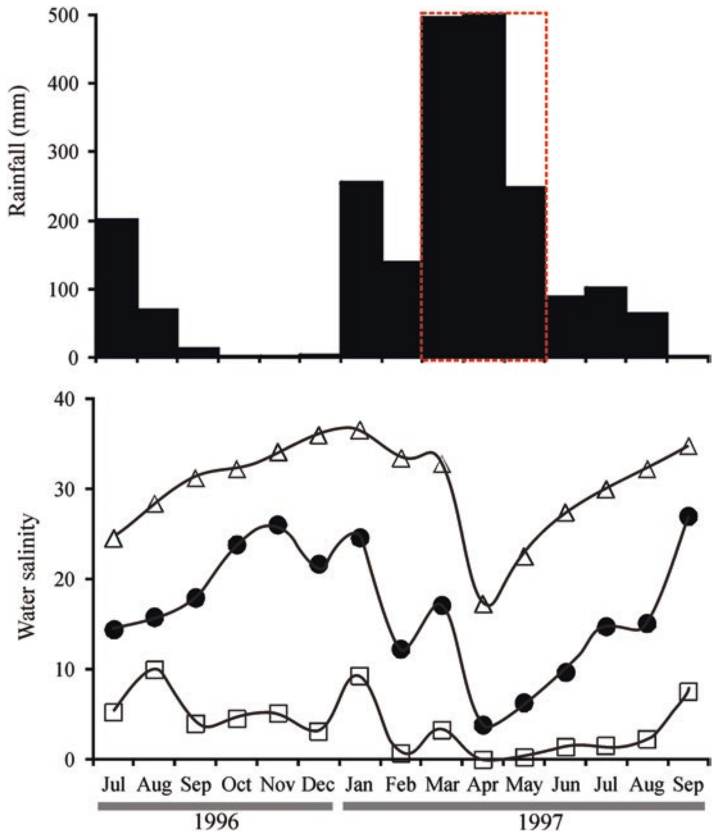

Fig. 16.2 Patterns of average monthly rainfall and average water salinity in the Caeté Estuary. *Red dashed boxes* (----) represents the rainy season

Caeté Estuary, and the seasonal salinity fluctuation is the main factor structuring the fish assemblages in the system (Barletta et al. 2005).

Most species captured by the artisanal and subsistence fisheries in this estuary require estuarine conditions to complete their life cycle (Barletta et al. 2005). During the dry season, hydrological conditions create a defined salinity gradient, and therefore estuarine and marine fish species fall along a horizontal gradient according to their capacity to tolerate salinity variations (Figs. 16.1 and 16.2). During the rainy season, freshwater runoff dramatically increases seaward, and salinity decreases in the upper portion of the estuary, which is then favourable for euryhaline freshwater fish species (Figs. 16.1 and 16.2) (Barletta et al. 2005).

Fig. 16.3 Goiana Estuary. *Doted lines* indicate the portions of the main channel [(■) *upper*; (■) *middle*; (■) *lower*]. *White* and *yellow arrows* shows the main fish movement during the dry and rainy seasons, respectively. *Rounded yellow arrows* indicate the prefered habitat in the late rainy season. *Red line* shows the total area of the conservation unit Resex Acaú-Goiana. In the *right side* is presented the total biomass of the most important commercial fishes produced during a seasonal cycle. *ATR* Atlantic Rain Forest, *Aqc* aquaculture, *SgC* sugarcane platation

16.2.2 Goiana Estuary

The Goiana Estuary (~4700 ha) represents a wide group of estuaries that border the eastern-most portion of South America, where only a few hundred kilometres of Atlantic Rain forest used to separate the coast from a wide expense of semi-arid continent (Fig. 16.3). Therefore, its climate is tropical, with well defined dry and rainy seasons (Barletta et al. 2010), however, temperature changes relatively little along the year. River basins of this region are thus highly sensitive, since freshwater flow to the systems during the dry season is very small and governed by the year's own rainfall amounts. In Goiana Estuary, the environmental quality of the system reflects the inland conditions of the river basin, with strong dependence on short and uncertain periods of rainfall (Barletta et al. 2010). Consequently, the freshwater-related fish assemblages is not as species rich diverse as marine-based assemblages in this system (Lima et al. 2015).

The vertical stratification of salinity, water temperature and dissolved oxygen are stable most of the year (Dantas et al. 2010; Lima et al. 2015). Fish are distributed along spatial gradients by temporal factors (Dantas et al. 2010; Ferreira et al. 2016; Lima et al. 2015; Ramos et al. 2016). Estuarine morphology allows coastal waters to influence even the upper-most portion of the estuary, and marine demersal fishes can reach areas next to the river (Lima et al. 2015). During the late rainy season, when continental runoff finally increases seaward, marine fish migrate from the upper to the middle and lower portion of the system to take advantage of greater salinity stability (Figs. 16.3 and 16.4) (Dantas et al. 2010; Ferreira et al. 2016; Ramos et al. 2016).

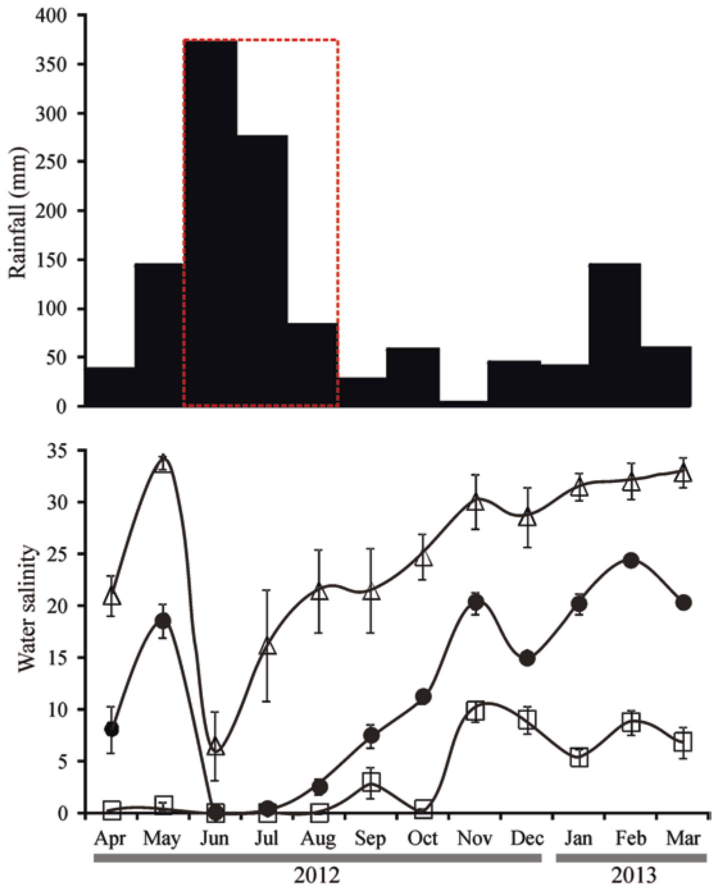

Fig. 16.4 Patterns of average monthly rainfall and average water salinity in the Goiana Estuary. *Red dashed boxes* (----) represents the rainy season

16.2.3 Paranaguá Estuary

The Paranaguá Estuarine Complex (~61,200 ha) is in the tropical to subtropical transition of the Western Atlantic (Barletta et al. 2008) (Fig. 16.5). The warm temperate climate shows two distinctive rainfall seasons (rainy and dry), but is also more susceptible to wide temperature changes. Mean annual rainfall is 2500 mm (Possatto et al. 2016). The Paranaguá Estuary shows a vertical salinity gradient in addition to its horizontal salinity-dominated ecocline. The upper estuary has meso to oligohaline characteristics (Fig. 16.6). The middle estuary has intermediate salinities, and during the rainy season is more strongly influenced by meso and oligohaline waters (Fig. 16.6). On the other hand, during the dry season, coastal waters influence this area (Barletta et al. 2008). Marine waters dominates the lower estuary throughout the year (Fig. 16.6).

Fig. 16.5 Paranagua Estuary. *Doted lines* indicate the portions of the main channel [(■) *upper*; (■) *middle*; (■) *lower*]. *White* and *yellow arrows* shows the main fish movement during the dry and rainy seasons, respectively. *Rounded yellow arrows* indicate the prefered habitat in the late rainy season. *Red line* shows the total area of the conservation unit "APA-Guaraqueçaba" and the Superagui National Park. In the *right side* is presented the total biomass of the most important commercial fishes produced during a seasonal cycle. *ATR* Atlantic Rain Forest, *NP* National Park, *PFP* Pontal do Félix port, *PRP* Paranaguá port

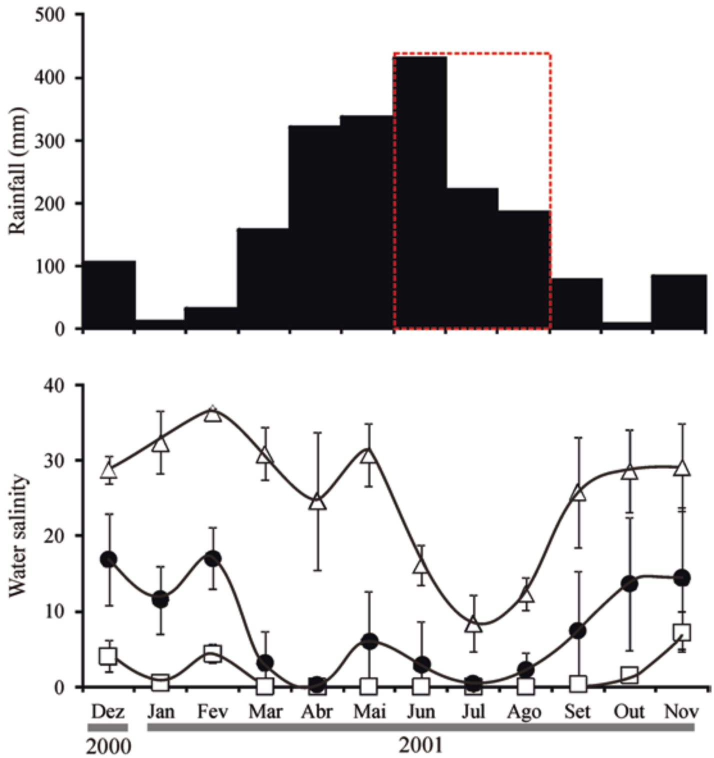

Fig. 16.6 Patterns of average monthly rainfall and average water salinity in the Paranaguá Estuary. *Red dashed boxes* (----) represents the rainy season

During the late rainy season, dissolved oxygen is also important in structuring fish assemblages. But, for the late dry season, both salinity and dissolved oxygen were responsible for it. In the Paranaguá Estuary, seasons are also responsible for defining salinity gradients and habitat movements. Environmental instability during the late rainy season induces estuarine fish to move outwards in search of greater salinity stability in the middle and lower portions, rather than favouring the freshwater species to move upstream (Figs. 16.5 and 16.6) (Barletta et al. 2008). The main channel of the Paranaguá Estuary supports inflow of marine waters even during high river discharges and marine fishes remain within its bounds until the late rainy season (Barletta et al. 2008). This corroborates the proposed model for tropical and subtropical estuarine fish movement patterns worldwide, where the salinity ecocline induces the fish movement in different estuarine portions (Barletta et al. 2005; Barletta and Blaber 2007). However, if natural or anthropogenic effects, such as dredging processes, modify the estuarine dynamics of current it will cause consequences on fish assemblage composition and therefore for fisheries practices (Barletta et al. 2016; Blaber 2000, 2002, 2013).

16.3 Estuarine Ecocline

16.3.1 Influence of Estuarine Ecocline on Fish Fauna of the Western Atlantic Estuaries

Being estuarine ecocline zones of gradual changes between river-dominated to marine-like waters, these systems result from the intrusion of coastal waters (function of tidal amplitude, channel depth, etc.) and freshwater inputs (Barletta and Dantas 2016) (Figs. 16.7 and 16.8). Turbulence zones mix these two different water bodies, producing boundaries that result in a suite of faunal communities along the salinity ecocline (Barletta and Dantas 2016). The fluctuation of the estuarine ecocline is strongly influenced by seasonal changes in freshwater flow and diel tidal cycles, forcing faunal communities to adapt, in terms of quali-quantitative variables or living strategies, to the different time scales of these modifications (Barletta et al. 2008; Dantas et al. 2010; Watanabe et al. 2014; Ferreira et al. 2016) (Figs. 16.2, 16.4, 16.6, 16.7, and 16.8). At a local scale, each estuary exhibits its own morphology and climatic distinctions, resulting in different abiotic demands through space and time for a variety of organisms that depends on estuaries for their ecological and biological requirements.

Estuarine-dependent fish species (Able 2005; Potter et al. 2013) are distributed according to their physiological abilities to withstand variations in the abiotic environment during different lengths of time during their life cycles (Elliott et al. 2007; Lima et al. 2015) (Figs. 16.7 and 16.8). It defines functional groups of species as estuarine, freshwater and marine guilds; as well as migratory taxa, as anadromous, catadromous, amphidromous and freshwater/marine migrant groups (Elliott et al.

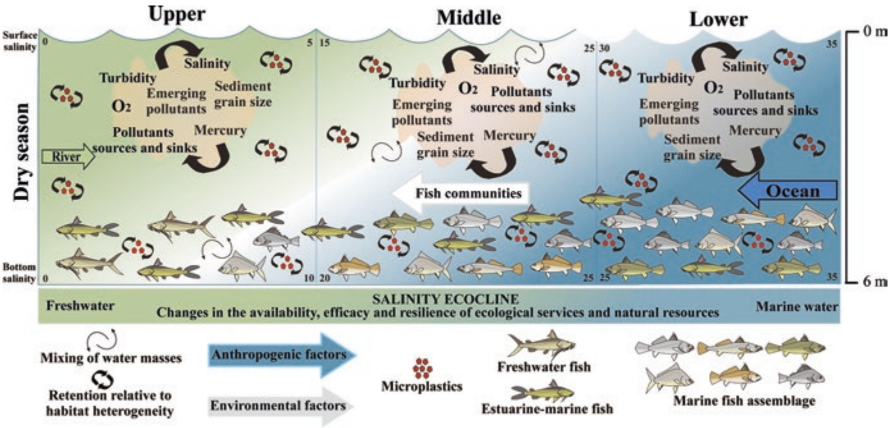

Fig. 16.7 Conceptual model. The influence of salinity ecocline on abiotic variable, fish communities and microplastics. During the dry season, the *vertical* stratification of water masses creates temporary habitats (*upper*, *middle* and *lower*) with distinct environmental characteristics according to the salinity gradient. It causes the retention of pollutants and physico-chemical factor, as well as habitat variability for fish assemblages

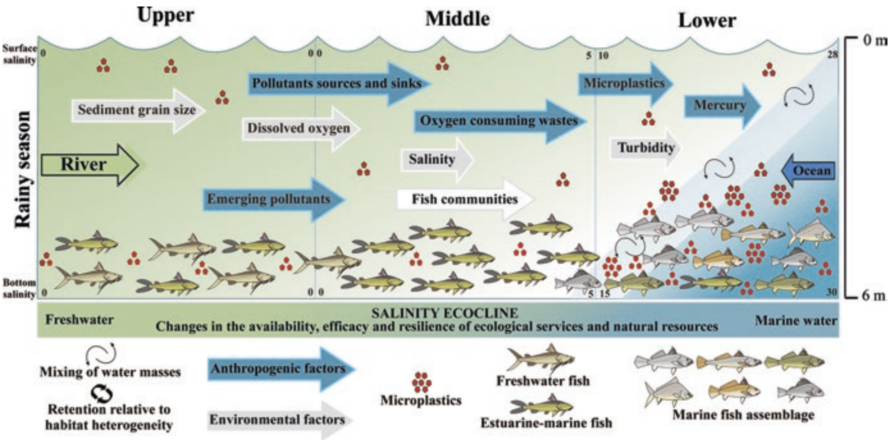

Fig. 16.8 Conceptual model. The influence of salinity ecocline on abiotic variable, fish communities and microplastics. During the rainy season, the incluence of river flow increases seward, habitats shift, and marine fish assemblages migrate to the estuarine reach with more stable salinity condition (middle or lower estuaries). During this period, pollutants, physico-chemical demands, microplatics and fish biomass are exported to the marine coastal evironment and high seas

2007). The former are euryhaline and exhibit wide ranges of effective estuarine use during the entire life cycle, being less affected by ecocline heterogeneity in tropical ecosystems (Elliott et al. 2007). Most marine and freshwater taxa are stenohaline and remain limited to areas of coastal or river influences, respectively (Elliott et al. 2007). Migratory fishes are euryhaline species that cruise the estuarine ecocline to maintain fulfil trophic and biological functions. Many marine/freshwater migratory taxa enter estuaries sporadically, especially as juveniles, only to take advantage of shelter from predators and food abundance. Some are highly euryhaline and move through the full length of the system (Elliott et al. 2007). In tropical and subtropical estuaries of the western Atlantic, salinity-dominated ecocline is a major driver influencing spatial and temporal distribution of fish and other fauna. Changes in fish sizes, diversity and abundance are strong indicatives of environmental variability into a particular habitat and are associated to seasonal movements of these habitats due to river flow switches from the dry to the rainy season and back (Barletta et al. 2005, 2008; Dantas et al. 2012; Lima et al. 2015; Ramos et al. 2016).

The management and conservation of estuaries (and consequently of its resources) should take into account the overlapping of environmental variables and life cycles of key species for different estuarine reaches and seasons (Figs. 16.7 and 16.8). Abrupt changes in climatic patterns or in the river flow regime will induce abrupt changes in the estuarine ecocline trends and fish communities will respond to these changes by modifying assemblage structures. In the Paranaguá Estuary, for example, during dredging operations, the upper estuary is deepened and alterations in the geomorphology of the system is responsible to the intrusion of a greater amount of coastal waters upstream, resulting in changes in fish community (Barletta et al. 2016). These concepts need to be taken as the basis of descriptors of reference conditions for regional coastal systems, considering the resilience of the system, including when under anthropogenic interference (Barletta et al. 2010).

Also important to note, is that tropical and subtropical estuaries usually have a greater diversity and complexity of habitats, including main channel, flood plain, intertidal creeks and surrounding mangrove forest (Fig. 16.9a–c) (Reis et al. 2016). The connectivity of these habitats guarantee that estuarine-dependent fishes complete their life cycle by accessing all the resources and using all the protection they need. Patterns of estuarine use can differ during the ontogenetic development of fish species. Larval, juvenile, sub-adult and adult stages of the same species respond differently to seasonal fluctuations of estuarine ecocline, as well as in the use of available effective habitats (Barletta-Bergan et al. 2002a, b; Barletta and Barletta-Bergan 2009; Ferreira et al. 2016; Lima et al. 2015). This trait is suggested to be a strategy to avoid competition and predation during the earlier, critical stages. The estuarine ecocline defines spatial and temporal patterns in community diversity and structure, population abundance, periods of reproduction and temporary nursery habitats for juveniles of many socio-economically important fish species (Dantas et al. 2010; Ferreira et al. 2016; Ramos et al. 2014, 2016).

Fig. 16.9 Main tropical estuarine habitats. (**a**) estuarine main channel showing the bordering mangrove forest; (**b**) flood plain; (**c**) intertidal mangrove creeks

16.3.2 Importance of Estuarine Ecocline to Predict Water Quality Changes

Seasonal and spatial changes in environmental gradients as estuarine ecocline influence both biotic and abiotic features of an estuary (Barletta et al. 2012; Costa et al. 2009; Lima et al. 2014). The estuarine ecocline can set physico-chemical conditions that determine the bioavailability or sinking of the amount of dissolved oxygen in the waters along the estuary, as well as the concentration of pollutants (e.g. plastic debris, metals, emerging pollutants) and microbiological contaminants (e.g. from sewage, urban runoff and animal farming) throughout the transition between freshwater and marine habitats (Figs. 16.7 and 16.8). South American countries have historically neglected basic sanitation problems in both urban and rural areas; the same was still generally observed for industrial and agro-industrial effluents until recently (Huang et al. 2014; Costa and Barletta 2016). Sources of industrial effluents, sewage and solid wastes are observed all along river basins, estuarine courses and adjacent coastal and marine waters (Pereira et al. 2010). Usually, pollutants reach critical levels near urban centers and industrial estates most of the year. When hydrological condition changes prompted by increased rainfall and river flow, pollutant loads move together with shifting abiotic habitats to other estuarine areas downstream, depending on the efficiency of the flushing (Figs. 16.7 and 16.8). In this way, although somewhat diluted, pollutants become bioavailable for a greater diversity of flora and fauna over the whole extent of the ecocline, and are eventually transported to adjacent coastal waters, from where they gain access to the continental platform and open ocean (Lima et al. 2014) (Figs. 16.7 and 16.8).

Although dissolved oxygen varies with the intensity of mixing and water renewal, levels of dissolved oxygen are higher in areas near the estuarine mouth, where photosynthesis is not affected by high turbidity next to the river. Nutrient and organic matter loads increased by anthropogenic effects can cause eutrophication when aerobic respiration and the oxidation of the organic matter overpass the recovery capacity of the estuarine system, even in short seasonal periods of little dynamic (Costa and Barletta 2016). Thus, oxygen depletion creates temporary or permanent zones of hypoxia and anoxia, negatively influencing the estuarine community due to physiological stress and reduced faunal diversity leading fishes to seek for other habitats or even to fish death (Yakushev 2016). An additional problem to oxygen depletion in the western South Atlantic is a permanent water temperature around 25 °C, reducing the water capacity to dissolve gases. Whereas estuarine ecocline is a heterogeneous gradient, zones of hypoxia and anoxia can be retained in particular habitats of the main channel for a period, and then, levels of dissolved oxygen can increase again due to the complexity of estuarine dynamics at seasonal scales (Figs. 16.7 and 16.8).

Salinity suspended particulate matter loads and dissolved oxygen levels are all strong controllers of metal biogeochemistry and bioavailability in estuaries. In the Goiana Estuary, for example, fish and bivalve when used as bioindicators of the influence of the estuarine ecocline in total mercury loads, showed that the metals availability changes with water quality along the estuary and seasons (Costa et al. 2009; Silva-Cavalcanti et al. 2016). For this estuary, mercury contents decreased in fish tissues during the late rainy season, when increased rainfall might be responsible for biodilution of this contaminant (Barletta et al. 2012) (Figs. 16.7 and 16.8).

Another marine pollution issue that affects estuarine gradients by following its spatio-temporal changes is plastic pollution, especially microplastics (>5 mm) (Thompson et al. 2009; Browne et al. 2010; Costa and Barletta 2015) (Fig. 16.10a–d). The most obvious source of plastic pollution is the inadequate disposal practices observed along to river basins and coastal areas (Fig. 16.10a–d). Interactions between plastics and marine biota include entanglement and ingestion, which might cause from minor physical damage to the death of animals (Costa and Barletta 2016), both already observed in estuaries and coastal environments of the Western Atlantic (Dantas et al. 2010; Guebert et al. 2013; Wright et al. 2013; Ferreira et al. 2016; Ramos et al. 2016). The fragmentation of large plastics into smaller fragments lead to the reporting of comparable amounts of microplastics and plankton in habitats of the Goiana Estuary main channel. The salinity gradient, or ecocline, was found responsible for the distribution pattern of the microplastic fragments by promoting retention of fragments from the river basin in the upper portion, and fragments of marine or local sources (e.g. fishery) in the lower portion when hydrological conditions were stable (Lima et al. 2014) (Figs. 16.7 and 16.8). During the dry season, the middle estuary is dominated by turbulence and stratification processes that

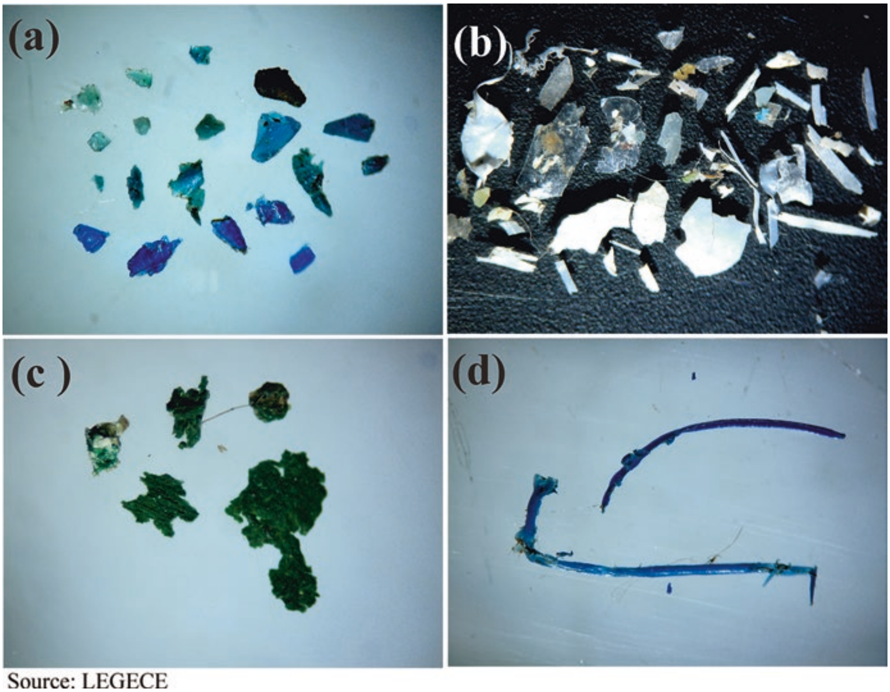

Fig. 16.10 Examples of microplastics (<5 mm) found within the main channel of the Goiana Estuary. (**a**) *blue* hard plastics; (**b**) *white* soft plastics; (**c**) *green* paint chips; (**d**) *blue* threads

do not allow the passing of microplastics from the upper to the lower system or in the opposite direction (Lima et al. 2014) (Figs. 16.7 and 16.8). However, in the late rainy season, river flush is strong enough to push microplastics seaward, and higher densities of this pollutant are observed in the lower estuary (Lima et al. 2014) (Figs. 16.7 and 16.8). The specific sources of microplastics are difficult to determine due to their ubiquity and because they continue to further fragment and mischaracterize almost indefinitely (Barnes et al. 2009; Costa and Barletta 2016), but findings at the Goiana Estuary confirms that river basins are important exporting sources of microplastic to the coastal seas (Cheung et al. 2016; Lima et al. 2014).

In addition, microplastics have large surface: volume ratios, and are important media for the transport of persistent organic pollutants (Frias et al. 2010). The smaller the fragments, the better for pollutants adsorption and the easier the ingestion by fauna, which will then be harmed through physical and chemical processes. Their abundance in the water column will determine bioavailability to estuarine communities by direct ingestion and/or trophic transfer (Farrel and Nelson 2013). In face of the generalized and unprecedented poor disposal practices of plastics (Barnes et al. 2009; Cole et al. 2011; Eriksen et al. 2014), it is possible to conjecture that every estuarine system in the Western Atlantic is experiencing this same problem of microplastic contamination and exportation (Chen 2015; Costa and Barletta 2015, 2016). But efforts to prevent are still in slow course.

Uncontrolled and synergetic human-driven changes in estuarine water quality might leave these systems increasingly vulnerable to global changes. Increasing water temperature impairs dissolved oxygen levels in the water column. Intense El Niño southern oscillations (ENSO) for example, may cause more severe, frequent and lasting climatic disturbances, resulting in longer dry seasons and/or excessive rainfall and river flushes (Ríos-Pulgarín et al. 2016) from which estuarine systems will find more and more difficult to recover if they have to face it under already precarious water quality conditions.

Despite all impacts, climate and hydrodynamics of western Atlantic estuaries contribute to reduce pollution, especially during the rainy season, through biodilution and transport of contaminants out of the system, in a process known as environmental homeostasis (Dyke and Weaver 2013; Elliott and Quintino 2007) (Figs. 16.7 and 16.8). In estuaries with larger river basins and which are more environmentally variable, such as Caeté and Paranaguá, unless severe, anthropogenic interventions are more difficult to detected (Barletta et al. 2016; Monteiro et al. 2011). However, the capacity of Goiana Estuary to withstand modifications is still a hesitation, since it is located in a short, low discharge, river basin only hundreds of km long and, therefore, the freshwater influence is small trough most part of the year (Lima et al. 2016).

16.3.3 Implications for the Conservation of Western Atlantic Estuaries

The connectivity among river basin, coastal zones and coastal waters promoted by estuarine gradients (ecocline) is currently widely discussed in the scientific literature (Able 2005; Barletta and Blaber 2007; Barletta et al. 2010; Watanabe et al. 2014), and estuaries are well known as important areas for providing biological and geochemical demands to other environments (Costa and Barletta 2016). These well established environmental processes need to face a mighty contender, that is intense societal and economic use and occupation. Estuaries are in coastal zones where population density is highest, pressures of agricultural and industrial expansion, improper disposal of wastes and sewages significantly alter natural environments, water quantities and quality, upsetting local and regional biogeochemical processes, flora and fauna (Barletta et al. 2010). Therefore, managerial plans should aim at improve resilience before the modifications experienced by these ecosystems overpass their services capacity (Williams and Crutzen 2013). To support estuarine and coastal management in an adaptive way, there are robust sampling designs that can provide consistent ecological and statistically testable data (Kurup et al. 1998; Worm et al. 2009; Blaber and Barletta 2016; Reis et al. 2016).

Properly logged, preserved and used fisheries landing data are a type of long term information that can reflect environmental health, since it shows (although in an indirect manner) what resources are available, how much, where and when. So, it is possible to know if estuaries are still able to provide their specific ecological services of food and shelter to fish populations that depend on their integrity to complete life cycles. The effect of managerial actions taken based on these consistent data can result in further food safety and environmental conservation, in order to guarantee that estuaries continue to function as a transition environment that links and separates land and sea.

Due to limited marine fisheries data in Brazil, the country with most territorial responsibility at the Western Atlantic, it is not possible to predict how many fishes are produced by the artisanal and industrial fleet for each of its regions. *Micropogonias furnieri* (whitemouth croaker), *Cynoscion acoupa* (acoupa weakfish) and *Mugil* spp. (mulletts) were responsible for ~83 tonnes of the marine extractive fishery in 2011 in Brazil (MPA 2011). Although artisanal fishery has more importance in the northeast (704,000 fishermen), and industrial fishery in the south/southeast Brazilian coast (133,000 fishermen), both regions produced ~280,000 tonnes of fishes from marine fishery (MPA 2011). Estuaries along the Western Atlantic are acknowledged to be responsible for the current fishing status of Brazilian coasts (Blaber and Barletta 2016). the three model estuaries approached in this chapter, Caeté, Goiana and Paranaguá, are important exporters of juveniles in both number and biomass for the recruitment of fish stocks in coastal waters and high seas, and therefore, efforts are arising to protect these ecosystems (Barletta-Bergan et al. 2002a, b; Barletta et al. 2005, 2008; Ferreira et al. 2016).

Caeté Estuary supports at least 50 species of marine, estuarine and freshwater fishes (e.g. Ariidae, Achiridae, Engraulidae, and Clupeidae). Few are of commercial importance, and most are used mainly for subsistence (Barletta et al. 2005). However, its main channel supports a biomass of more than 42.4 tons (300 g ha^{-1}) of commercial fishes in the earlier life stages during a seasonal cycle in 1996/1997 (270 samples) (Barletta et al. 2005) (Fig. 16.1). This included 14 marine species, such as the Sciaenidae *C. acoupa* and *C. microlepidotus* (smallscale weakfish) and a freshwater species, the Pimelodidae *Brachyplastystoma vaillantii* (piramutaba catfish) (Barletta et al. 2005) (Figs. 16.1 and 16.11a–c). Most of these species were also in tidal creeks of the flooded mangrove forest at the mouth of the estuary (Barletta et al. 2003). If the many other creeks that drains the mangrove forest are considered, these habitats are indeed very important in terms of biomass, especially for *C. acoupa* and species of the genus *Mugil* (mullets) (Barletta et al. 2003). These habitats were in fact important ecological grounds for larval stages of most species (Barletta-Bergan et al. 2002a, b). Amongst commercial taxa, larval *C. acoupa* was the most abundant species in terms of density in both main channel (3372 ind. 100 m^{-3}) and mangrove creeks (10,946 ind. 100 m^{-3}) (Barletta-Bergan et al. 2002a, b).

The Caeté Estuary supports intense commercial and harbour activities and, unplanned urban settlements are a problem (Fig. 16.12a–d). The main income source of 83% of households around the estuary derive from mangrove living resources. Families (42%) collect and sell the mangrove crab (*Ucides cordatus*)

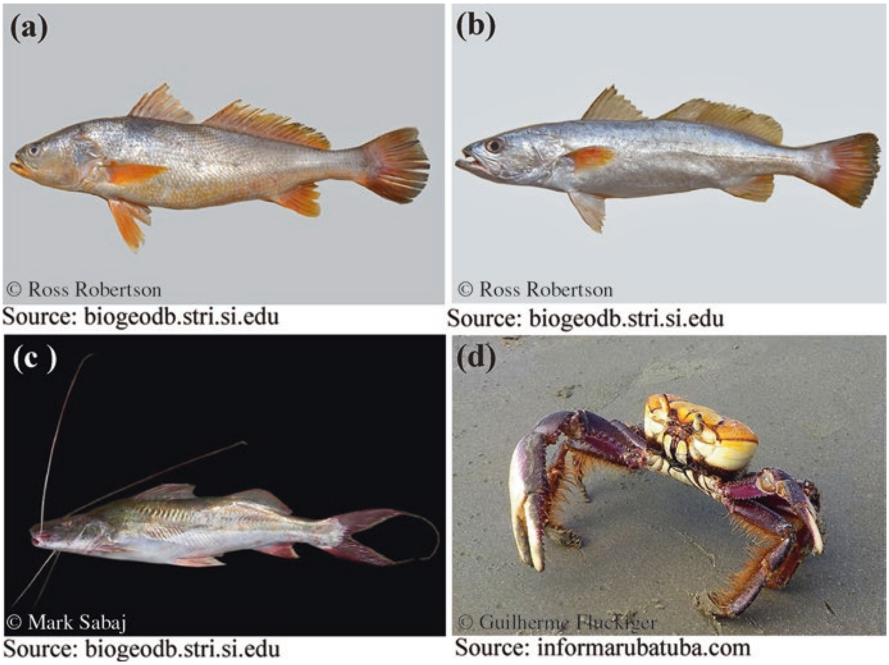

Fig. 16.11 Examploes of important commercial fish resources used for livelihood produced insede the Caeté Estuary. Sciaenidae: (**a**) acoupa weakfish (*Cynoscion acoupa*; (**b**) small scale weakfish (*Cynoscion microlepidotus*); Pimelodidae: (**c**) laulau catfish (*Brachyplastystoma vaillantii*); Ocypodidae: (**d**) uça-crab (*Ucides cordatus*)

(Fig. 16.11d) and engage in commercial fishing into the estuary or coastal waters (30%) (Wolff et al. 2000; Glaser 2003). This ecosystem also represents an important geographic region to the genetic patrimony of Western Atlantic sciaenid assemblage (Vinson et al. 2004) and *Epinephelus itajara* (Atlantic Goliath Grouper) (Silva-Oliveira et al. 2013). Since 1990, the goliath grouper fishery has been closed throughout the southeast region of the United States (SEDAR 2004). It is currently listed as critically endangered on the International Union for the Conservation of Nature (IUCN) Red List (SEDAR 2004). In Brazil, this species is protected since 2002 by the Chico Mendes Institute for Biodiversity Conservation (ICMBio). Still, wastewater discharges directly into the estuary, drop the levels of dissolved oxygen and local water quality during the dry period (Fig. 16.12a) (Pereira et al. 2010).

The main channel of Goiana Estuary supported a biomass of 91.6 tons (400 g ha^{-1}) of fishes important for subsistence during a seasonal cycle in 2005/2006 (216 samples). This included three Ariids [*Cathorops spixii* (madamango sea catfish), *C. agassizii* (catfish) and *Sciades herzbergii* (pemecou catfish) (Fig. 16.13a–c)] and two Sciaenids [*Stellifer brasiliensis* (stardrum) and *S. stellifer* (little croaker) (Fig. 16.13d–e)] (Dantas et al. 2010, 2015) (Fig. 16.13a–e). During the same period, its main channel was responsible for a total biomass of

Fig. 16.12 Anthropogenic impacts affecting the Caeté Estuary. (**a**) Pollution by solid wastes (plastic debris) and direct sewage discharge in the upper reach of the estuary related to riverside communities of Bragança city; (**b**) artisanal fishing harbour; (**c**) illegal collection of mangrove tree wood; (**d**) urban settlements (Bragança city)

more than 19 tonnes of two commercially important fishes in earlier life stages, *C. acoupa* (Fig. 16.11a) and the Gerreidae *Eugerres brasilianus* (Brazilian mojarra) (Fig. 16.13f) (Ferreira et al. 2016; Ramos et al. 2016) (Fig. 16.1). All these species were also found in mangrove tidal creeks near the mouth of the estuary (April/May 2008), together with another seven commercially important marine species (Centropomidae, Lutjanidae and Mugilidae) (Ramos et al. 2011). Larval stages of Sciaenidae, Gerreidae and Centropomidae were also found in the main channel and mangrove tidal creeks of the Goiana Estuary (Lima et al. 2015, 2016).

Possibly 450–1000 families explore living natural resources from the Goiana Estuary as a mean of traditional livelihoods (Barletta and Costa 2009). The main income sources are subsistence fishing of ariid catfishes (*C. spixii*, *C. agassizii* and *S. herzbergii* – Fig. 16.13a–c); commercial fishing of sciaenids, gerreids, centropomids, lutjanids and mugilids; and exploitation of crabs, prawns, oyster and shellfish (bivalves *Anomalocardia brasiliana* and *Tagelus plebeius* – Fig. 16.13g–h) (Barletta and Costa 2009). Overexploitation of fish stocks is a problem, especially for the most profitable catch as lobster (Guebert-Bartholo et al. 2011). Fishing with gillnets and interaction with plastic debris were considered the most important threat to the sea turtle populations that feed, mate and nest in the area (Guebert et al. 2013).

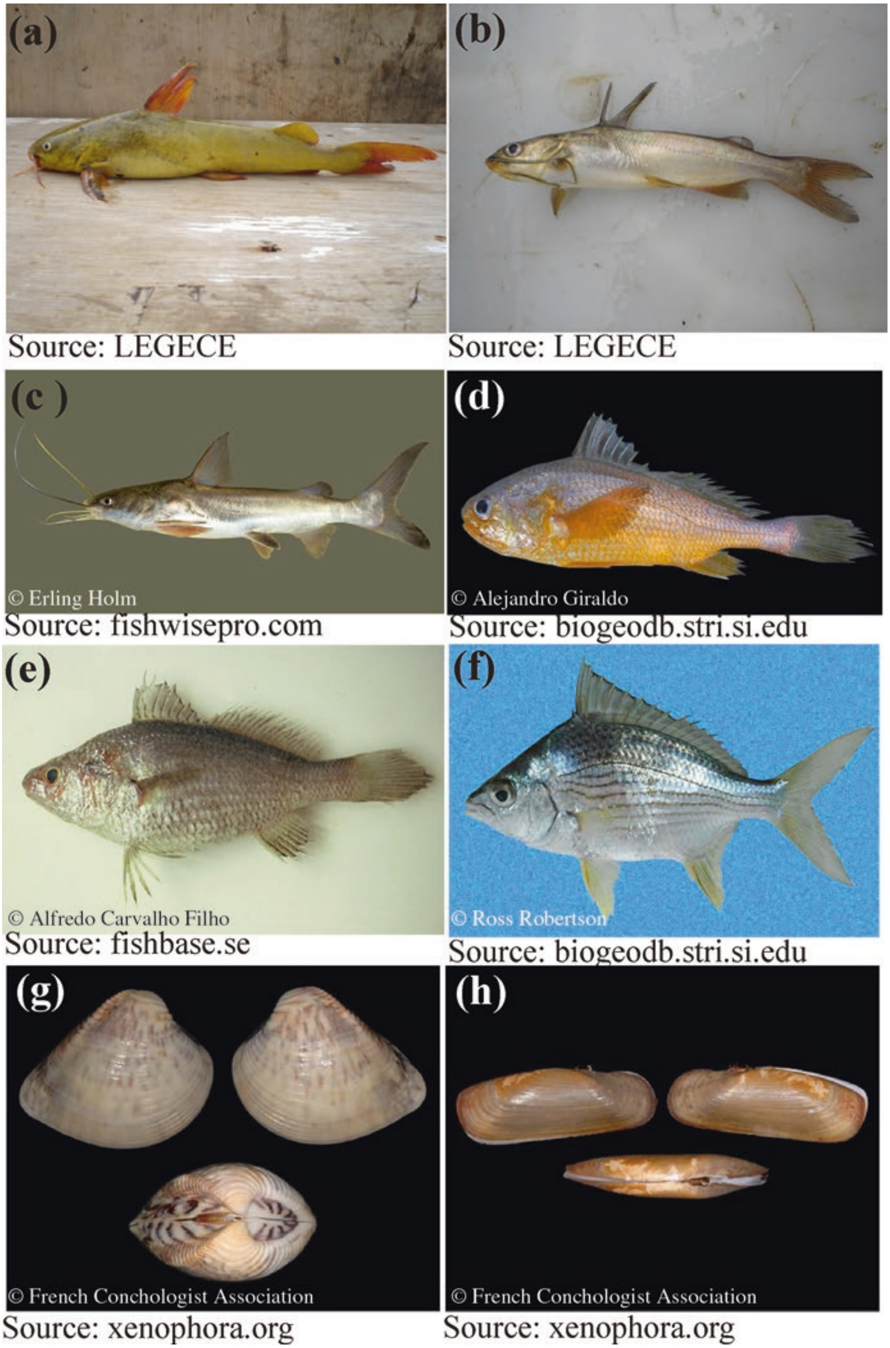

Fig. 16.13 Examploes of important commercial fish resources used for livelihood produced insede the Goiana Estuary. Ariidae: (**a**) madamango sea catfish (*Cathorops spixii*); (**b**) catfish (*Cathorops agassizii*); (**c**) pemecou catfish (*Sciades herzbergii*); Sciaenidae: (**d**) little croaker (*Stellifer stellifer*); (**e**) stardrum (*Stellifer brasiliensis*); Gerreidae: (**f**) brazilian mojarra (*Eugerres brasilianus*); Veneridae: (**g**) caribbean venerid clam (*Anomalocardia brasiliana*); Psammobiidae (**h**) stout mussel (*Tagelus plebeius*)

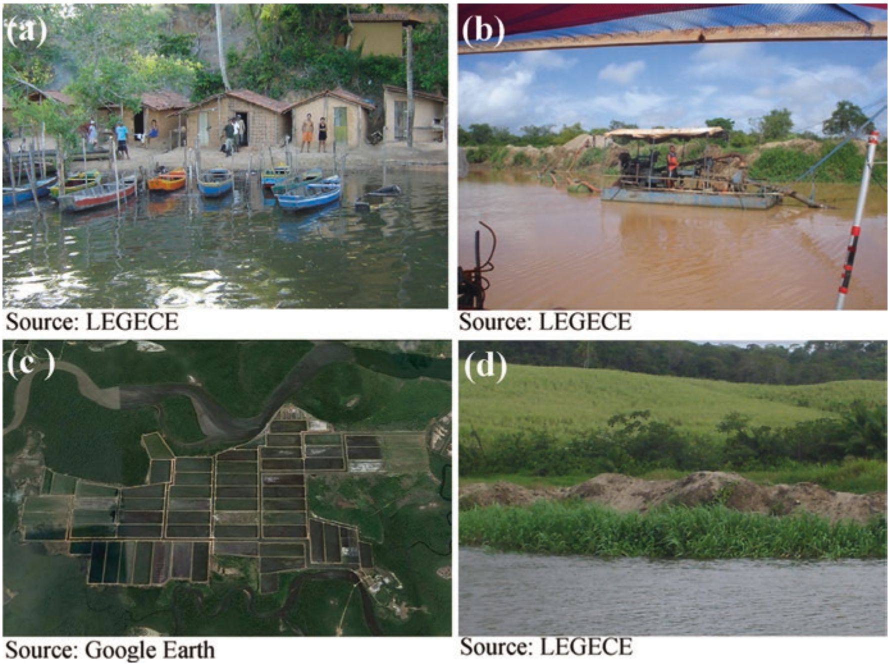

Fig. 16.14 Anthropogenic impacts affecting the Goiana Estuary. (**a**) Artisanal fishing villages in the river margin; (**b**) dredging activity for sand; (**c**) aquiculture farming next to the lower reach of the estuary; (**d**) Mangrove deforestation for sugarcane plantation

Economic activities such as sugarcane plantations and milling, aquaculture, mining, dredging for sand and land development are among the modifications that influence the estuarine ecosystem and water quality (Barletta and Costa 2009) (Fig. 16.14a–d). The Goiana Estuary provides a wide variety of ecological services to the local population and surrounding areas. It also includes important habitats for endangered species such as *E. itajara* (Atlantic goliath grouper), *Trichechus manatus* (manatee), *Chelonia mydas*, *Lepidochelys olivacea* (sea turtles) (Barletta and Costa 2009).

The Goiana Estuary receives wastewater effluents from the sugarcane planting and milling industry and sewage from urban settlements (Fig. 16.14a, d) (Costa et al. 2009). These are potential sources of mercury to subsistence and commercial fish (*Trichiurus lepturus* – cutlass fish) (Barletta et al. 2012; Costa et al. 2009), in which tissues mercury levels increased with size and weight, and decreased with increased rainfall (biodilution) (Barletta et al. 2012; Costa et al. 2009). Although this contamination are, in general, below WHO (World Health Organization) critical limits for human consumption, further contamination might increase mercury levels, making this fish species unsafe for pregnant women and children (Costa et al. 2009). Mercury bioaccumulation was also studies in the shellfish *A. brasiliana* from the Goiana Estuary; however, seasonal clear trends were not observed

(Silva-Cavalcanti et al. 2016). Contamination by plastic debris and microplastics was assessed in the main channel and mangrove creeks of Goiana Estuary (Fig. 16.10a–d) (Ivar do Sul et al. 2014; Lima et al. 2014, 2015, 2016).

Microplastics were comparable with half of the total fish larvae and with total fish eggs in density along the main channel (Lima et al. 2014). The ingestion of plastic threads by the demersal fish fauna was observed in 20% of ariid catfishes (*C. spixii*, *C. agassizii, S. herzbergii*) (Possatto et al. 2011) and 13% of Gerreids (*Diapterus rhombeus*, *E. brasilianus*, *Eucinostomus melanopterus*) (Ramos et al. 2012). In sciaenids, it was registered a contamination of 8% of *Stellifer* (*S. brasiliensis*, *S. stellifer*) (Dantas et al. 2012) and 64% in *C. acoupa* (Ferreira et al. 2016).

The estuarine complex of Paranaguá supports, at least, 60 species of fish used for subsistence among the local population (Barletta et al. 2008). The main channel yielded >49 tons (800 g ha^{-1}) of commercial marine fishes in earlier life stages (Barletta et al. 2008) (Fig. 16.1). This included ten marine species, as the Sciaenidae *C. leiarchus* (smooth weakfish) and *M. furnieri*; and the Haemulidae *Pomadasys corvinaeformis* (roughneck grunt) (Barletta et al. 2008) (Figs. 16.1 and 16.15a–c). The Paranaguá Estuary is a key ecosystem of the western Atlantic (Guebert et al. 2013; Possatto et al. 2016). Its natural resources attract fishing, tourism and conservation sectors; and the strategic position resulted in the development of the largest Latin America maritime terminal for grain and other agro-industrial products (Paranaguá port) (Fig. 16.16a). Therefore, channel maintenance is constant, and fish diversity decreases during dredging. Ariid catfishes benefit from damage on microbenthic fauna and, thus, increase their densities, while other species disappear (Barletta et al. 2016). Recently another port facilities was built in the estuary (Pontal do Felix port) and intense dredging activities are observed in the upper estuary (Barletta et al. 2016) (Figs. 16.1 and 16.16b). Alterations in water quality due to dredging and other anthropogenic impacts are also observed (Barletta et al. 2016) (Fig. 16.16c–d). Faecal steroid concentrations presented higher level closer to Paranaguá port-city, reflecting significant water and sediments sewage contamination (Martins et al. 2010). Plastic debris contamination was also more significant near the port (Possatto et al. 2015), and ingestion by juvenile green turtles *Chelonia mydas* feeding in the area was found to be frequent (Guebert-Bartholo et al. 2011). Characterizing yet another form of pollution, sediments near the Paranaguá port showed high levels of Arsenic (As) and other heavy metals (Zn and Ni) than WHO critic limits, suggesting inputs from human activities in addition to a geochemical anomaly (Sá et al. 2006). Fortunately, the Paranaguá Estuarine Complex is large, and so far has shown the capacity to withstand environmental modifications and is in homeostasis throughout the year (Barletta et al. 2016; Dyke and Weaver 2013; Elliott and Quintino 2007).

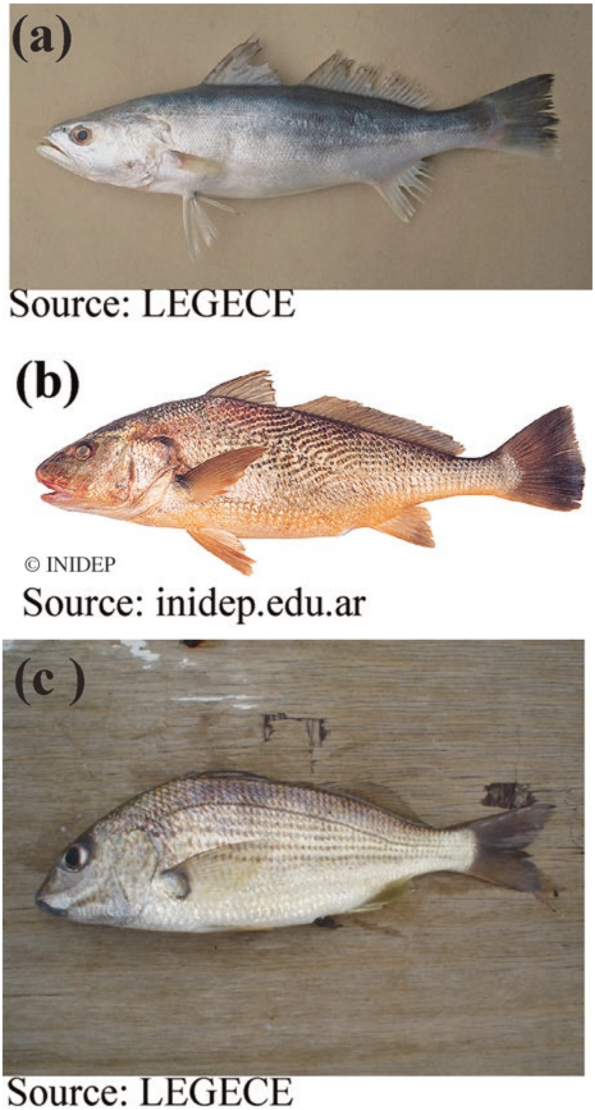

Fig. 16.15 Examploes of important commercial fish resources used for livelihood produced insede the Paranaguá Estuary. Sciaenidae: (**a**) smooth weakfish (*Cynoscion leiarchus*); (**b**) Whitemouth croacker (*Micropogonias furnieri*); Haemulidae: (**c**) roughneck grunt (*Pomadasys corvinaeformis*)

16.4 Discussion

16.4.1 Ecological Information as a Basis of Conservation Plans for Western Atlantic Estuaries

Although scientific literature on ecology, uses of estuaries and their ecocline is always increasing in volume and quality (Blaber 2000; Lotze et al. 2006; Elliott and Quintino 2007; Costa and Barletta 2016; Blaber and Barletta 2016; Reis et al. 2016),

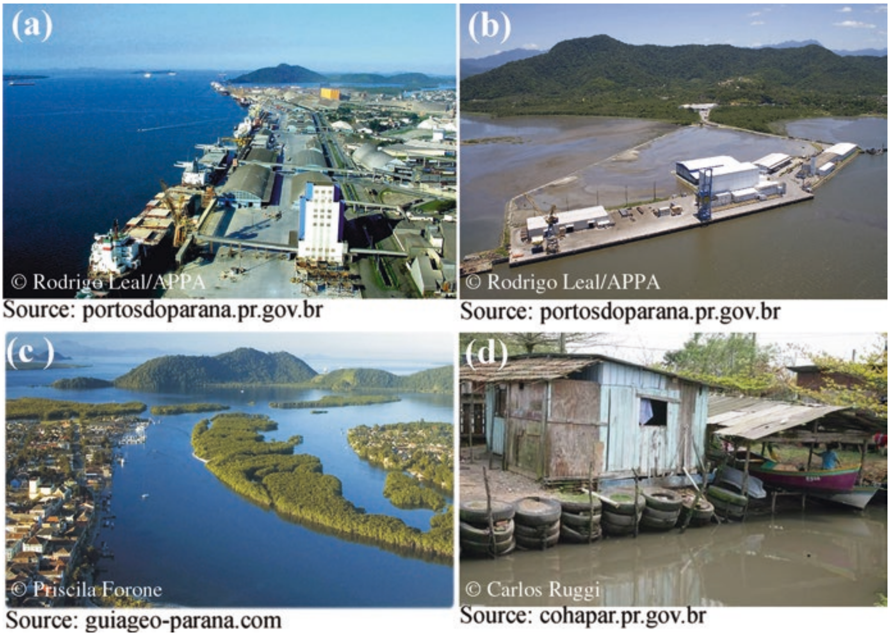

Fig. 16.16 Anthropogenic impacts affecting the Paranaguá Estuary. (**a**) Paranaguá port, a maritime terminal for grain and other agro-industrial products; (**b**) Pontal do Felix port, a recently port facility with intense dredging activities; (**c**) urban settlements (Paranaguá city); (**d**) direct sewage discharge in the upper reach of the estuary related to riverside communities

to compile the most relevant information remains a challenge in aquatic ecosystems conservation. The time elapsed from generation of information by the scientific community and the establishment of management plans is the main concerning. In Brazil, the longest coastal zone facing the Western Atlantic, only recently estuaries have received attention with the creation of the first Marine Protected Area (MPA) in the early 1980s (Diegues 2008), and despite the number and variety of coastal ecosystems including estuaries, the number of officially and effectively protected areas is still insufficient. MPAs are important, among other things, to conserve biodiversity, promote eco-tourism in areas where this talent can be developed and to ensure the replenishment of fisheries stocks (Diegues 2008; Jennings 2009). However, MPAs are only one of the many coastal conservation options, and could be regarded as exceptions in terms of territory covered. Ideally, conservation measures should also, and principally, be in place outside these specially protected areas. Such measures, although less severe, could guarantee the conservation of much wider coastal areas.

The main concern for estuarine conservation issues is that estuaries (and other transitional waters) were ignored for many years by freshwater and marine scientists due to the confusing natural processes that rule the ecosystem (Elliot and Whitefield 2011). Only recently, over the past four decades, estuaries are regarded as being ecosystems in their own (Elliot and Whitefield 2011). Therefore, the negligence of governmental agencies in considering the basic concepts previously established for estuaries worldwide has been debated over the last decades (Ducrotoy and Dauvin 2008; Dauvin and Ruellet 2009). An example is the definition of water quality status of water bodies, including transitional environments, that is estuaries, due to the implementation of the European Water Framework Directive (Dauvin and Ruellet 2009). This directive uses multiple bio-indicators and indices to define estuarine system regardless of the natural variability and the influence of anthropogenic effects in water quality (Dauvin and Ruellet 2009). Whereas estuarine fauna have adapted to withstand seasonal physico-chemical changes, not taking into account scientific reference conditions hampers any possibilities of estuarine conservation (Dauvin and Ruellet 2009; Barletta et al. 2010).

A short review on the ecological importance of the Orinco River basin (Colombia and Venezuela), although being a freshwater system, revealed that the high specie richness (>1000 species), fish diversity and production are closely related to the heterogeneity of habitats, providing multiple nurseries grounds (Machado-Allison 2016). However, the unregulated construction of dams, dredging, contamination from herbicides, mercury, oxygen-consuming effluents, deforestation for agricultural, industrial and urban purposes, overfishing and the introduction of exotic species are rapidly damaging the local fauna to a high extent (Machado-Allison 2016). The author asserts that although efforts through research, publishing of results and recommendations are intensive, conservation, recovery and sustainable use of resources are being affected by social, economic and political clashes (Machado-Allison 2016).

Different policies for conservation of coastal ecosystems and traditional populations exist in the Amazon, northeast and southern Brazilian wetlands to preserve natural resources and values thorough co-management of extractive and sustainable development reserves (Barletta and Costa 2009; ICMbio 1995, 2012). In the eastern Amazon a management unit to promote integral protection and sustainable use of the Caeté Estuary resources, the Marine Extractive Reserve Caeté-Taperaçu (42,068 ha and 3000 families), was created in 2005 (Fig. 16.1). Its management plans initiated became operational late in 2013. In northeast Brazil, the Marine Extractive Reserve Acaú-Goiana (6700 ha) was created in 2007 to guarantee the protection of livelihoods and resources as *A. brasiliana*, a typically female/family fisheries (Barletta and Costa 2009; Guebert et al. 2013) (Fig. 16.3). Management plans for extractive reserve Acaú-Goiana still do not exist, and stakeholder decisions are ineffective (Guebert et al. 2013), and until recently, fishers did not know what an extractive reserve is (Guebert et al. 2013). The Estuarine Complex of Paranaguá is one of the largest estuaries of the Western Atlantic. Guaraqueçaba bay and surrounding remnants of the Atlantic rainforest form its main settings (Fig. 16.5). It is therefore recognized as a Natural World Heritage Site (UNESCO 1999). The Atlantic rainforest encompasses the Saint-Hilaire/Lange National Park, of restricted use. The bay encompasses a conservation unit of restricted use (Superagui National Park) and another of sustainable use (Environmental Protected Area of Guaraqueçaba) (Guebert-Bartholo et al. 2011; Possatto et al. 2016) (Fig. 16.5). The consolidation of action plans for this mosaic of protected areas initiated in 1991 and extended until 1999.

Plans and actions aiming at estuarine conservation must take into account the seasonal fluctuations of ecocline along cycles of retention and flush of dissolved and particulate components, including environmental contaminants and, consequently, how hydrodynamics affect water quality. In Western Atlantic estuaries, annual rainfall patterns, vertical salinity structure during dry periods and river flow to the systems are very important factors to develop explanatory and predictive models aiming at conservation (Figs. 16.7 and 16.8). However, the large majority of studies consider only patterns of spatial changes of ecological variables, and sampling is based on distance to sources of interference. Not accounting for temporal variability of water quality leads to faults in predicting whether estuaries are able to withstand human modifications, or not. This is also important if considered that the main cause of habitat loss in Western Atlantic estuaries is related to control and uptake of freshwater flowing into the systems.

16.4.2 Recommendations for Future Estuarine Conservation

Understanding and working in favour of estuarine ecocline is also important to guarantee the conservation of socio-economically relevant estuarine-dependent fish species. Each ontogenetic stage exhibits different strategies to bear the spatial and temporal variability of abiotic compartments across the changeable estuarine habitats in order to complete their life cycles. Reliable ecological data are fundamental in managerial actions because planed ecological data can explain how environmental gradients influence trophic organization, distributional patterns, reproductive timing, peaks of abundance and temporary nursery habitats within estuarine fish assemblages, and estuarine communities (Table 16.1).

The concepts and importance of working in favour of ecocline are gaining attention, and recommendations for conservation in protected areas are being restructured (Storm et al. 2005; Slater 2016). This is the case of the Natural Park S'Albufera in Majorca (Balearic Islands, Espain), a Wetland of International Importance under the Ramsar Convention, which recently was defined as a marsh aquatic system with a saline-freshwater gradient (ecocline) containing a series of ecotones (communities), resulting in dynamic niches for plants and animals (Slater 2016). In the Rhine-Meuse estuary, one of the most important wetland in the Netherlands, a program of remedial action has been emerged aiming a healthier ecosystem by recovery the estuarine processes (Storm et al. 2005). In this study, it is suggested that the restoration of tidal and river dynamics in polders are the best option for ecological rehabilitation (Slater 2016).

Even when managerial plans are consistent with ecological processes, they may enter in conflict with the lack of basic sanitation, the worst concern in estuarine systems pollution and degradation. Ecological information concerning interactions among biota and contaminants along estuarine ecocline provides information on critical periods for bioaccumulation and its further consequences, risk of higher plastic debris ingestion (macro or micro) and unsuitable water quality due to oxygen

Table 16.1 Steps to the use of ecological information as the basis for the conservation of estuarine ecosystem of Wester Atlantic estuaries

Steps to conservational issues	Knowledge flow	Objectives	Effective outcomes	Marine protected areas	Managerial actions
Planned and consistent models to assess seasonal and spatial patterns in biotic and abiotic factors into estuarine systems	Scientists	Provide background information on the importance of the study	Importance	–	–
Project approval for scientific project development	Scientists Agencies Government	Gain financial support by scientific agencies	Relevance	–	–
Data information on the importance of the estuarine system for fish resources, fisheries and access to human impacts	Scientists Technical experts	Provide robust reference conditions for regional wetlands	Legitimacy	–	–
Peer reviews and consolidation of reliable data and testable models	Scientists	Make the scientific information available for worldwide decision makers and scientific communities	Credibility	–	–

Propose models to predict influence of estuarine ecocline on the interactions among estuarine fauna, the abiotic environment and human modifications	Scientists	Provide recommendations for ecosystem and biodiversity conservation	Boundary process	No-take zones	Limited mesh size to reduce juvenile captures;
	Technical experts	Practices of sustainable use of estuarine resources		Genetic legacy of species	Reduced bycatch and ghost fishing;
	Decision makers			National parks	Closed periods for fishing different groups;
	Managers			Extractive reserve	Fishing seasons;
	Stakeholders				Control fishing effort and capture;
	Government				Quotas for capture of common resources;
					Reduction of untreated wastewater disposal;
					Monitor, control and surveillance;
					Re-examine environmental operation licenses;
					Care with risk assessments

depletion, hypoxia/anoxia. In addition, by fully characterizing ecocline according to biotic and abiotic parameters, it is possible to predict if, and when, estuaries will recover from exceptional hydrodynamic and climatic events, still maintaining its natural capacity to support cycles of seasonal changes.

Advanced technologies are being in course to offer the potential of nanostructured catalytic membranes, nanosorbents, nanocatalysts, and bioactive nanoparticles for the treatment of surface water, groundwater, and wastewater contaminated by toxic metal ions, organic and inorganic solutes, and microorganisms (Theron et al. 2008; Qu et al. 2013). Other models use of aquatic plants (Gersberg et al. 1986), green algae (Wang et al. 2010) and activated sludge (Gernaey et al. 2004) for remove nutrient loads (especially nitrogen) from primary municipal wastewaters, as well as electrochemical technology to oxidation of organic pollutants (Martínez-Huitle and Ferro 2016). In this sense, efforts for a national plan of sewage treatment networks are urgent. This is special for uncontrolled riverside settlements, whose sewages are directly discharged along all courses of the estuaries (Costa and Barletta 2016). In addition, governmental agencies need to propose specific reference values for wastewater disposal for each type of oxygen-consuming effluents (i.e. agriculture, aquiculture and industrial). It is necessary to control the maximum loads of pollutants that can be discharged to guarantee the minimum contamination of estuarine ecosystems.

Changes in estuarine morphology induced by human intervention can alter the natural inflow of coastal water and lead to ecocline disruption along the whole system (Barletta et al. 2016; Prestrelo and Monteiro-Neto 2016). In the Piratininga and Itaipu coastal lagoons (southeastern Brazil), the opening of an artificial channel increased the salinity intrusion and now marine species dominate the system, while former dominant freshwater species decreased in terms of diversity and abundance (Prestrelo and Monteiro-Neto 2016). In addition, the higher salinity reduced the environmental heterogeneity and decreased the species richness and biomass, leading to reduced fish production (Prestrelo and Monteiro-Neto 2016).

Some important information to guarantee the sustainable use of estuarine resources available in the literature can already be highlighted as priorities for Western Atlantic estuaries (Table 16.1). Limited mesh sizes to decrease capture of juvenile stages; closed periods for fishing different groups/fishing seasons; no-take-zones; quotas for capture of common resources; efforts to reduce/eliminate untreated wastewater disposal (Table 16.1).

However, human impacts derived from urbanization, fishing and industries also generate financial income and some social profits, at least on the short term (Figs. 16.12, 16.14, and 16.16). To re-examine environmental operation licenses can be another important step to prevent habitat loss and/or degradation, especially in areas around estuaries, where shipping ports, industries and aquaculture cause damages that impair fisheries production and therefore the exportation of biomass from regional seas (Lotze et al. 2006; Dallas and Bernard 2009; Huang et al. 2014; Barletta et al. 2016). Fault in risk assessment of companies next to river basin is another important point to consider. Very recently, in 2015, in southeast Brazil, the disruption of a containment dam of mining tailings contaminated the Doce River basin (Minas Gerais and Espírito Santos states) by mood flood that reached the coastal marine areas (Jacobi and Cibim 2015). It caused a total imbalance in important ecosystems associated to the river, the estuarine system and the coast of many Brazilian states, killed tonnes of fishes and livelihood losses (Jacobi and Cibim 2015). The losses of the tragedy cannot be measured, because they still lack scientific basis to propose recovery plans, if this is still possible. The unique information is present in environmental technical reports (ICMbio 2016).

A study regarding global fisheries affirmed that, in half of all studied ecosystems, the average exploitation of marine resources has declined and is now at or below the rate to maximum sustainable yield (Worm et al. 2009). Nevertheless, the authors assert that worldwide fish stocks need attention, especially in order to protect vulnerable species. Surprisingly, information on the Western South Atlantic are not available in this study. It denounces the historical negligence of authorities in building reliable datasets for the fishery statistics in the region (Barletta et al. 2010; Reis et al. 2016). In addition, it shows the inefficient communication between academy and institutions responsible for structuring the fisheries conservation scenario, including the sustainable use of estuarine and marine resources along these coasts.

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Chapter 17
Alteration and Remediation of Coastal Wetland Ecosystems in the Danube Delta: A Remote-Sensing Approach

Simona Niculescu, Cédric Lardeux, and Jenica Hanganu

Abstract Wetlands are important and valuable ecosystems; yet, since 1900, more than 50% of wetlands have been lost worldwide. An example of altered and partially restored coastal wetlands is the Danube Delta in Romania. Over time, human intervention has manifested itself in more than one-quarter of the entire Danube surface. This intervention was brutal and has rendered ecosystem restoration very difficult. Studies for rehabilitation/re-vegetation were begun immediately after the Danube Delta was declared a Biosphere Reservation in 1990. Remote sensing offers accurate methods for detecting changes in restored wetlands. Vegetation change detection is a powerful indicator of restoration success. The restoration projects use vegetative cover as an important indicator of restoration success. To follow the evolution of the vegetation cover of the restored areas, images obtained by radar and optical satellites, such as Sentinel-1 and Sentinel-2, have been used. The sensitivity of such sensors to the landscape depends on the wavelength of the radar or optical detection system and, for radar data, on polarization. Combining these types of data, which are associated with the density and size of the vegetation, is particularly relevant for the classification of wetland vegetation. In addition, the high temporal acquisition frequencies used by Sentinel-1, which are not sensitive to cloud cover, allow the use of temporal signatures of different land covers. Thus, to better understand the signatures of the different study classes, we analyze the polarimetric and temporal signatures of Sentinel-1 data. In a second phase, we perform classifications based on the Random Forest supervised classification algorithm involving the

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entire Sentinel-1 time series, proceeding through a Sentinel-2 collection and finally involving combinations of Sentinel-1 and -2 data. The supervised classifier used is the Random Forest algorithm that is available in the OrfeoToolbox (version 5.6) free software. Random Forest is an ensemble learning technique that builds upon multiple decision trees and is particularly relevant when combining different types of indicators. The results of this study relate to the use of combinations of data from different satellite sensors (multi-date Sentinel-1, Sentinel-2) to improve the accuracy of recognition and mapping of major vegetation classes in the restoring areas of the Danube Delta. First, the data from each sensor are classified and analyzed. The results obtained in the first step show quite good classification performance for only one Sentinel-2 data (87.5% mean accuracy), in contrast to the very good results obtained using the Sentinel-1 time series (95.7% mean accuracy). The combination of Sentinel-1 time series and optical data from Sentinel-2 improved the performance of the classification (97.1%).

Keywords Coastal wetlands • Danube delta • Alteration and remediation of ecosystems • Remote sensing • Synergy of radar time series Sentinel-1 and optical image Sentinel-2

17.1 Introduction

According to United Nations Environment Programme research, 40% of the global economy depends on the adequate functioning of ecosystems. Many ecosystems have been sacrificed in the name of economic development in various fields such as agriculture and other production systems and as a result of urbanization, industrialization, resource extraction, transportation and other infrastructures. Given the increasing number of anthropogenic perturbations of natural and semi-natural ecosystems worldwide, the mere preservation of these ecosystems is no longer sufficient. Ecological restoration may thus prove to be an essential complement to their preservation. The dramatic depletion of biological diversity and the degradation of ecosystem services allowed legitimizing a marked evolution of the international, European and Romanian political context. As a result, the preservation and restoration of natural ecosystems has been recognized to be essential and constitutes part of Romania's future political commitments, more precisely of the National Biodiversity Strategy 2013–2020.

Over the past few decades, efforts to raise awareness of the environmental, functional and patrimonial interest of natural areas, as well as the evolution of protection policies, with a view in particular to the implementation of the "Habitats-Fauna-Flora" European Directive of 1992, have led to the carrying out of numerous ecological restoration operations. Ecosystem restoration is becoming a global priority at various levels of decision-making, with the goal of achieving specific political and technical objectives (Aronson and Alexander 2013).

17.2 Scientific Background

The concept and principles of ecological restoration emerged in Europe and in the United States of America at the beginning of the twentieth century, the goal being to look after the "wounded landscape". Thus, ecological restoration is a relatively new concept that has undergone considerable development over the last 20 years both at the theoretical level and at the level of concrete field applications. Additionally, in the last 10 years, the science and practice of the discipline have progressed significantly, generating knowledge, creating and applying tools, designing procedures and promoting networks worldwide. The first actions consisted of measures designed to control mountain land erosion and forest environment degradation. These actions have been extended and diversified to allow proper management of multiple environmental degradation situations in various natural environments. Certain systems may be restored directly according to the initial botanical composition theory; marshes or wetlands may thus be restored over a period of several years under optimal conditions. For other ecosystems, the process that takes place between the moment when they recover their self-regulating powers thanks to their restoration and the moment at which they achieve the environmental maturity stage of their target system is lengthy (decades or centuries).

In most cases, the ecosystem that needs restoring has been degraded, damaged, transformed or completely destroyed as a direct or indirect result of human actions. The human being who used to destroy without considering the consequences of his actions now wishes to repair the damage through the ecological restoration concept. Ecological restoration is an action that initiates or enhances the self-repair mechanism of an ecosystem and at the same time observes its health, integrity and sustainable development (Bouzille 2007). Restoration is generally thought to allow an ecosystem that has been degraded or destroyed by natural and/or human causes to resume its prior condition. The faithful recreation of original habitats still generates lively debate; many definitions of ecological restoration have been suggested, and other related terms such as rehabilitation and reallocation are also often used.

Ecological restoration is one of the means used to maintain ecosystem services and stop biodiversity loss. As a matter of fact, one of the objectives of the Convention on Biological Diversity (CBD) established during the Conference of the Parties held in Nagoya in 2010 was the restoration of at least 15% of the degraded areas worldwide by 2020. The Society for Ecological Restoration (SER) (2004) defines ecological restoration as "the process that could assist the regeneration of an ecosystem which has been degraded, damaged or destroyed". Ecological restoration is an intentional activity designed to accelerate or restore a historic ecosystem in connection with the original resident species, the structure of communities, the functioning of the environment, and the ability to shelter living organisms and connect them with the surrounding environment (Aronson 2010). This supposes and requires thorough knowledge of the functional and progressive ecology of the target ecosystems, of the history of the anthropogenic degradation of the ecosystem and, finally, of the choice of a reference ecosystem meant to guide the planning, implementation, follow-up and assessment of the restoration project (White and Walker 1997; Egan and Howell 2001). Restoration project guidelines (Clewell and Aronson 2013) and practical guidelines (Perrow and Davy 2002, have recently been developed, and more fundamental works have been published (Walker and del Moral 2003; Temperton et al. 2013; Suding and Hobbs 2009).

Ecological restoration should become a priority so as to limit the process of degradation of the environment, to contribute to the preservation of fragile habitats and of critically endangered species and to ensure the valorization of natural resources. When the economic, social and cultural dimensions are taken into account, ecosystem restoration becomes a strong area of intervention for development actors. Thus, the major restoration and rehabilitation objectives are, as concerns ecosystems, the preservation or increase of their primary or secondary productivity and the improvement of their biological diversity and stability, and, as concerns landscapes, the support of their reintegration when they are severely fragmented. With a view to sustainable development and, in particular, to building a sustainable environment, the natural environment sciences have recently begun to develop approaches that combine the restoration or preservation of ecological processes and functions and the biodiversity and productivity of ecosystems with economic and social uses.

These restorations are currently the object of numerous projects that are more or less ambitious and more or less expensive. As a result, in addition to the willingness to maintain and preserve ecosystems, there is a certain pressure from society concerning the assessment of the success of ecological restorations. Many of these projects concern wetlands worldwide. The wetlands of the world provide more ecosystem services per area than any other habitat type (Costantza et al. 1997; Dodds et al. 2008). Wetlands are important and valuable ecosystems, yet, since 1900, more than 50% of wetlands have been lost worldwide. The loss of ecosystem services when wetlands are degraded or converted to other land use types is well documented, as are global rates of wetland loss that range from 30% to 90% by region (Dahl et al. 1990; Junk et al. 2012; Zedler and Kercher 2005). Coastal wetland regions are under serious threat and have been suffering from severe degradation.

17.3 Restoration of Coastal Wetlands and the Danube Delta

The projects in wetlands are costly and achieve variable success. Although many coastal wetland restoration projects are conducted every year, wetland degradation has not been retarded worldwide because of the limited success in wetland restoration (Zedler 2000). Wetland restoration refers to the return of wetland from a disturbed or altered status caused by anthropogenic activities to a pristine status (Mitsch and Gosselink 2007; Jarzemsky et al. 2013). The degradation of coastal wetlands is often accompanied by direct or indirect changes in hydrology. Hydrology modification is more widely adopted as the appropriate hydro period, a key factor determining success in wetland restoration (Turner and Lewis 1996; Wortley et al. 2013; Jarzemsky et al. 2013). Chemical restoration refers to the removal of pollutants in inflow or the control of sources of pollutants in such a way as to restore the quality of coastal water and sediment (Wilcox and Whillans 1999), whereas biological restoration targets restoring the microorganisms, vegetation and fauna of degraded wetlands. Vegetation studies provide insight into the effectiveness of restoring wetland ecosystem functions. Although governments have put much effort into coastal wetland protection practices, most wetland restoration projects focus on the restoration and regulation of vegetation, and there is a lack of systematic studies on the mechanisms of coastal wetland degradation and ecohydrological processes, especially hydrological and biological connectivity on a large scale (Harttera and Ryan 2010). Therefore, ecohydrological environmental indicators need to be further integrated for successful evaluation of coastal wetland restoration based on the holistic restoration of wetlands (Ellison 2000; Allen 2003).

An example of altered and partially restored coastal wetlands is the Danube Delta in Romania (Fig. 17.1).

The total area of the Danube Delta Biosphere Reserve is approximately 5800 km^2 in Romania and more than 50 km^2 in Ukraine (Hanganu et al. 2002). In the Romanian portion, the reserve includes the upstream Danube floodplain of Tulcea-Isaccea, the Razim-Sinoe lagoon complex and the marine coastal waters (20 isobaths). The Danube Delta itself refers to the area between the three main branches of the Danube River (from north to south, these are the Chilia, the Sulina, and the Sfântu Gheorghe (St. George) branches), which are located in Romania with a total area of 3510 km^2. The Danube Delta is the third largest delta in Europe, after the Volga Delta (13,000 km^2) and the Kuban Delta (4300 km^2). The Danube Delta, Romania's youngest landmass, is a fluvial-maritime floodplain covering two floristic provinces, the lower Danube (ponto-sarmatic) and the Black Sea (euxinic) (Ciocârlan 1994). The diversified geomorphology, soils and hydrological conditions favor the proliferation of a large number of aquatic, semi-desert and saline habitats.

Each habitat is part of a unique nature conservation network. The flora in the Danube Delta Biosphere Reserve (both the Romanian and Ukrainian sectors) are characteristic of a steppe bioregion with a temperate climate featuring almost 1400 species of vascular plants (Hanganu et al. 2002), of which five species (one subspecies) are endemic (0.51% of the total number).

A vegetation map of the delta produced by [66] shows 44 types of vegetation grouped into eight categories in the Romanian delta and a significant part of the Ukraine delta. These units consist of the following: flood plain forests, beach/sea vegetation and dune vegetation, salt-tolerant vegetation, sandy steppe meadows, river elevation meadows, dune forests, marsh vegetation and aquatic vegetation. Natural marsh vegetation and aquatic vegetation are the most widespread in the Danube Delta. Vegetation cover of this type occupies 398,676 ha within the delta, of which 362,965 ha is in the Romanian section and 35,711 ha is in the Ukrainian section (Hanganu et al. 2002). Reed marshes, covering more than 220,000 ha, are by far the Danube Delta's dominant vegetation type. The dominant species is *Phragmites australis*, which is usually accompanied by hydrophilous species such as *Typha angustifolia*, *Schoenoplectus lacustris*, *Sparganium erectum*, and *Thelypteris palustris*. Most of the plant communities and species are terrestrial, and

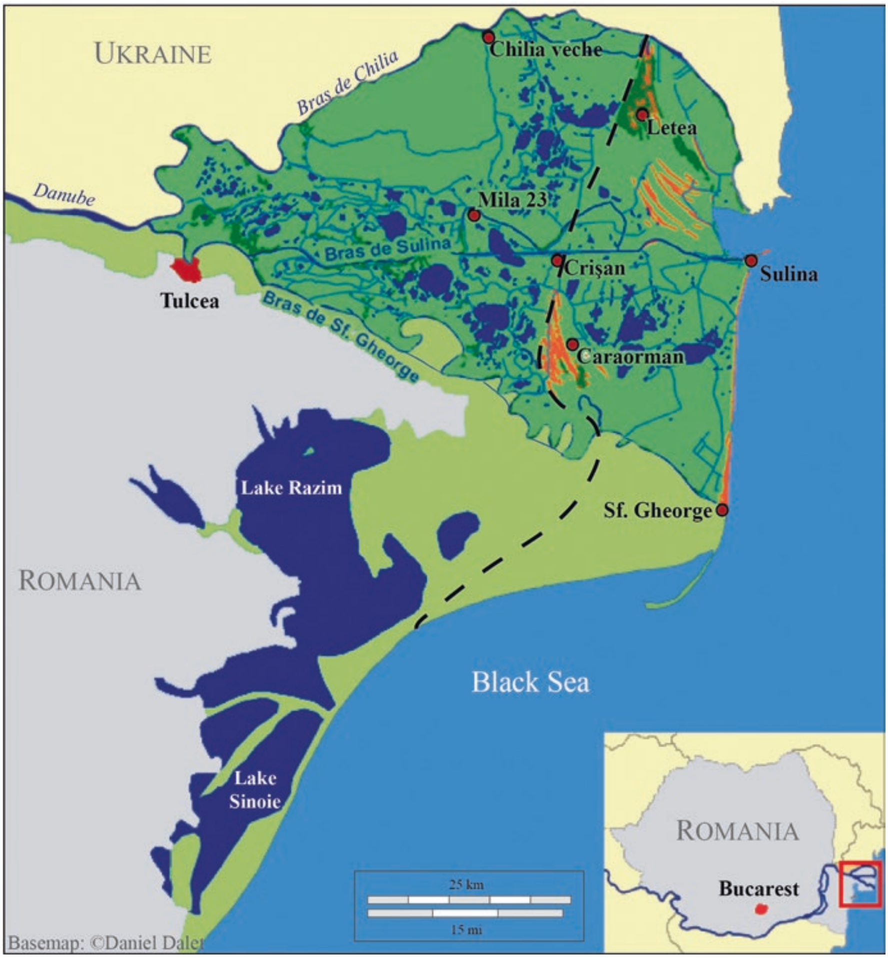

Fig. 17.1 Danube delta. Geographic localization

they can be found on the elevations and barrier beaches. Within the aquatic plant communities, the characteristic flora include Eurasian and circumpolar vegetation. The terrestrial plant community belongs to the Eurasian, continental, Pontic and Mediterranean classes.

Over time, human intervention has manifested itself in more than a quarter of the entire Danube surface. This intervention was brutal and has rendered ecosystem restoration very difficult. The ecological conditions of the Delta have also been influenced by the human activities carried out in the entire Danube basin: the building of flood plain dykes and dams, hydrotechnical accumulations, erosion control works and catchment works (especially for irrigation purposes) and development of economic activities in the Danube basin (industry, agriculture, energy, transportation, and other activities).

17.4 Improvements, Transformations and Alterations of the Danube Delta Ecosystems

Over time, the development of fluvial-maritime navigation and of resource use policies applying to the Danube Delta (fish, agricultural, forestry, etc. resources) has determined the main water system and landscape transformations in the delta. The first interventions in the water system of the Danube branches occurred in 1857–1858 after the creation in 1856 of the European Commission of the Danube (ECD) following the signing of the Treaty of Paris at the end of the Crimean War. The branch altered between 1857 and 1902 was the Sulina Branch in the central part of the delta. After the shortcuts in the meanders of this branch, in particular the "great M" (Dunarea Veche), the branch was shortened by 20.8 km, from 83.4 to 62.6 km, and deepened from 2.4 to 7.2 m, to allow the navigation of heavy-tonnage ships of up to 55,000 tdw. The alteration of the Sulina Branch also required the consolidation of the banks of the Danube and the building of a jetty at the entrance to the Black Sea. This jetty offers protection against clogging with sediments brought by the Chilia Branch. In 1895, also under the auspices of the European Commission of the Danube, a small marshy area next to the Mahmudia Locality and close to the Sfantu-Gheorghe Branch was altered for agricultural purposes. This area is also known as the "Gradinile franceze si olandeze[1]".

Other interventions in the water system of the delta date back to the 1900–1935 period, when, following the suggestions of the hydrobiologist Grigore Antipa (1914a), several canals were built to facilitate the flow of water between the Danube branches and the lake complexes within the delta or between these inner complexes and the Razim-Sinoe complex. The goal of these improvements was the enhancement of fishing productivity in a natural environment (Antipa 1914b). Some of the most important canals were built during this period, including the Dunavat (initially called Regele Carol I) in 1907, the Dranov (Ferdinand Canal, which connects the Sfantu Gheorghe Canal to the Razin-Sinoe complex) between 1912 and 1914, the Enisala (Elisabeta Canal) in 1913, the Litcov (Carol II Canal) between 1929 and 1932, the Crasnicol (Voeivodul Mihai Canal) between 1930 and 1934, and the Sireasa, which is parallel to the Sulina Branch connecting the Sontea-Furtuna Complex to the Danube branch (Fig. 17.2).

Another alteration that may be traced back to before the Second World War is the Tataru Polder (2500 ha), a small, dammed, drained and irrigated island on the Chilia Branch; on this island, cereal production is considerable (up to 10,000 kg/ha of corn). The Magearu and Eracle-Batacu Canals between the Chilia and the Sulina branches were also built prior to 1950, as were the Rosulet-Garla Imputita and Buhaz Ciotic Canals between the Sulina Branch and the Sfantu-Gheorghe Branch and the Gotca and Iacob-Batacu Canals in the Pardina Depression.

All these hydrological interventions carried out in the delta were accompanied by changes in the deltaic landscape. The issue of the use of the reed (*Phragmites*

[1] The French and Dutch Gardens.

Fig. 17.2 Chronology of the hydro-technical works in the Danube Delta during the 1900–1994 period. The most important canals built during this period: Dunavat (initially called Regele Carol I) in 1907; Dranov (Ferdinand Canal) between 1912 and 1914, which connects the Sfantu Gheorghe Canal to the Razin-Sinoe Complex; Enisala (Elisabeta Canal) in 1913; Litcov (Carol II Canal) between 1929 and 1932; Crasnicol (Voeivodul Mihai Canal) between 1930 and 1934; Sireasa, parallel to the Sulina Branch connecting the Sontea-Furtuna Complex to the Danube branch

australis) here was raised after the Second World War. An experimental research station designed to study the possible uses of the deltaic reed was set up in Maliuc in 1956. The main objective of this initiative was to alter certain depressions in such a way as to allow semi-guided reed growth and development (dammed depressions endowed with pumping stations designed to regulate the water levels depending on the height of the reed) and its subsequent industrial use. Other reed- and reed-use-related topics, such as reed physiology, the advantages and disadvantages of reed fires, the influence of heavy equipment (crawler tractors) on reed regeneration, the influence of dyke building on reed growth and development, etc., were studied at this experimental research station.

The Danube Delta reed had been used long before 1956 but in a traditional manner (on the ice, during winter, when the migratory birds had left the area and the fish took refuge in deep waters) without any intervention in the deltaic environment. Thus, the reed was used for construction purposes (roofs and fences), as a fodder supplement and even in the cellulose production process (a cellulose factory, which was destroyed during the war, operated in Braila between 1908 and 1916). After 1956, more precisely during the communist regime, reed was essentially used in the cellulose production process; therefore, the cellulose factory was rebuilt (1958) close to the town of Braila (Chiscani), approximately 100 km from the delta.

The 1960–1970 period, which is called the “reed age”, is also the first period of deeper alterations of the deltaic ecosystems. First, many canals were built, and the resulting alluvium was used to build 50 to 100 m-long and 2 to 3 m-high reed storage platforms above the Danube (Gastescu and Stiuca 2008). In the 1960s, most of the polders were designed for reed use, in particular in the fluvial delta (eastwards); these areas included the Pardina depression, the loop of the great M (DunareaVeche) and the depression located west of the Caraorman levee, located between the Sulina and the Sfantu Gheorghe Branches. These polders were irrigated by pumping between March and October and were then drained for the end of autumn-winter harvesting. This decade saw the peak of reed production in the Danube Delta. Thus, reed production reached 226,000 tons in 1965, decreased to almost 100,000 tons in the 1970s and did not exceed 20,000 tons in the 2000s.

What could be the explanation of this drop in reed production in the Danube Delta? One of the most plausible explanations, which is adhered to by many authors, is that the use of heavy machinery to harvest the reed destroyed its rhizomes. Other possible causes have also been suggested: reduction in the reed’s regeneration and natural drainage period and diminution of the sediments and nutrients in the water pumped into the altered depressions. Several opportunistic hydrophilic species such as the cattail (*Typha sp.)*, the rush (*Juncaceae* family) and the sedges (*Carex sp.*) appeared at this time and took advantage of the situation, proliferating to the detriment of the reed.

Due to the failure of reed processing in the delta, several reed-processing facilities (Rusca, Balteni, Maliuc, Obretin) were converted to fish-farming developments. In fact, the 1970–1980 decade is considered to be the period of fish farming development in the delta. The 1980s were marked by the passing of a decree by the State Council in 1983 (Programme for the Full Development and Exploitation of the

Natural Resources of the Danube Delta) according to which a considerable part of the delta was to be altered for agricultural purposes (crops and farms for animal husbandry), and the development of fish and forest exploitation was to be extended. Thus, the delta and the Razim-Sinoe lake complex were divided and shared among six business ventures exploiting delta resources. These ventures were subordinated to the Danube Delta Office seated in Tulcea (DDO).

A whole set of fish farming developments were subsequently created during that period, including the Popina, ChiliaVeche, Stipoc, Dunavat, Holbina I, Holbina II, Periteasca, and Ceamurlia developments, which together occupy approximately 40,000 ha. The fish farming system suggested for the delta was a closed one, which meant that all the feed had to be supplied by the livestock farmers. Things did not progress as expected because, due to economic recession after 1973, maintenance and servicing of the water supply and drainage facilities became difficult; consequently, fish production did not measure up to the initial investment. Some authors (Gastescu and Stiuca 2008) note that the location of these fish farming developments was not very wise to begin with. Either they were located on fertile land suited for agriculture (Stipoc), the soil salinization brought about by these alterations led to the extension of the French tamarisk species *(Tamarix gallica)* (Popina II, located eastward of the Letea levee), or the peat in the soil was harmful to the fish fauna (Holbina I and Holbina II, located between the Razim Lake and the Dranov Lake) (Fig. 17.3).

There are five agricultural development areas in the Danube Delta Biosphere Reserve, with an overall surface area of 39,974 ha; these are located in various places in the reserve on the territory of five communes (Table 17.1).

This intensive delta development period saw the setup of the many agricultural developments that still exist on the right bank of the Sfantu Gheorghe Branch from Tulcea down to Mahmudia. These agricultural developments do not significantly influence the ecological balance of the delta because there are few lakes and marshes in this area. These developments are considered profitable from an agricultural point of view. The Pardina and Sireasa developments, which are located in the fluvial part of the delta (Fig. 17.4), are considered to be the most radical.

Twenty-seven thousand hectares of the Pardina polder, which make up 10% of the deltaic plain and 67.62% of the agricultural development area of the delta, were fully altered (in the 1960s this depression was developed for fish farming) for agricultural purposes in 1983. This time, the natural landscape and the natural water flow disappeared completely after the major development works conducted here (drainage canals, irrigation canals) (Figs. 17.5 and 17.6).

Thus, five intensive agriculture farms were created in the Pardina polder. They had the following land use structure: 84.32% tillable land, 15.46% grassland and 0.2% vineyards and orchards. Legally speaking, 95.3% of the polder's surface area is currently included in the public domain managed by Tulcea County Council, and only 384 ha are owned by Pardina Commune inhabitants. The former Public Agricultural Company (IAS in Romanian), which owned the exploitation rights during the communist regime, was privatized after 1989. Nevertheless, the resulting private companies soon went bankrupt, and the land became the property of the

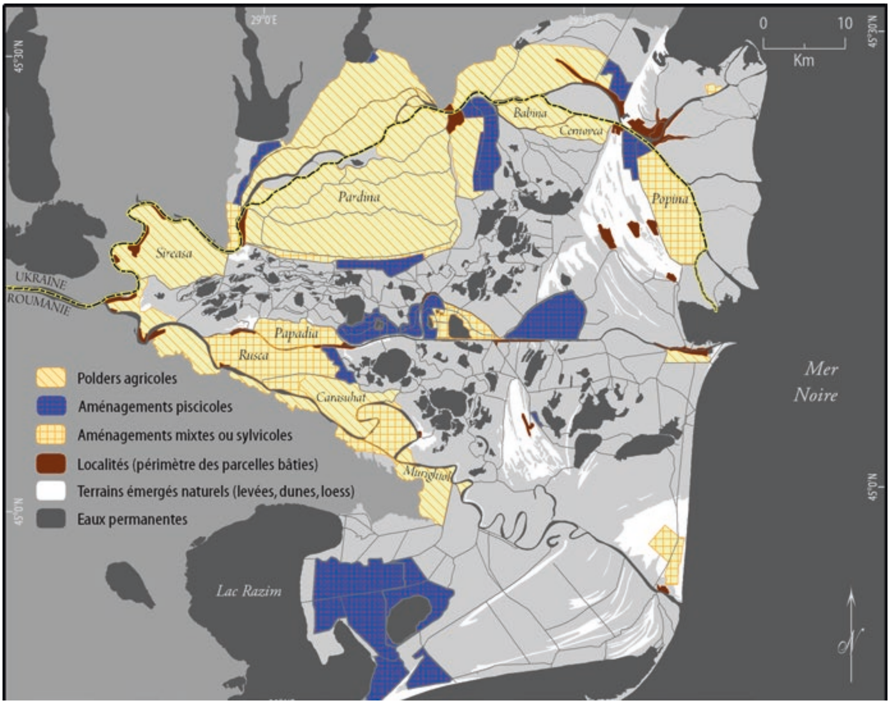

Fig. 17.3 Various Danube Delta improvements prior to 1994. Over time, human intervention has manifested itself in more than a quarter of the entire Danube surface

Table 17.1 Agricultural developments and their surface areas

	Locality	Agricultural development	Surfaces/ha
1	Ceatalchioi	Sireasa	5480
2	ChiliaVeche	Pardina	13,766
		Tataru	2061
		Total	15,827
3	Mahmudia	Carasuhat	2863
4	Murighiol	Dunavat-Murighiol	2538
5	Pardina		13,266
	Total area of agricultural developments		39,974

County Council. The policy consisted of granting concessions to private investors for large pieces of land by public tenders (Niculescu et al. 2015b). The County Council is accused of not taking into consideration the local specificity of Pardina and of giving away the land too easily, the farmers' only obligation being to practice only agriculture on that land. These changes attracted many foreign (French, English, and Italian) investors, who took over the farming of more than two thirds of the polder. Additionally, several shepherds who came to Pardina from Transylvania during the communist regime progressively took up the southern part of the polder.

Fig. 17.4 Example of agricultural development in the Sireasa polder (*top*) and example of fish farming development in Caraorman (*bottom*) (Photos S. Niculescu, September 2010)

The development of the Pardina polder has raised problems for the local and county authorities because they do not share the same vision. The ecological or economic benefits have taken the spotlight, and the local population seems to have been forgotten in these development projects, although the local population should be the

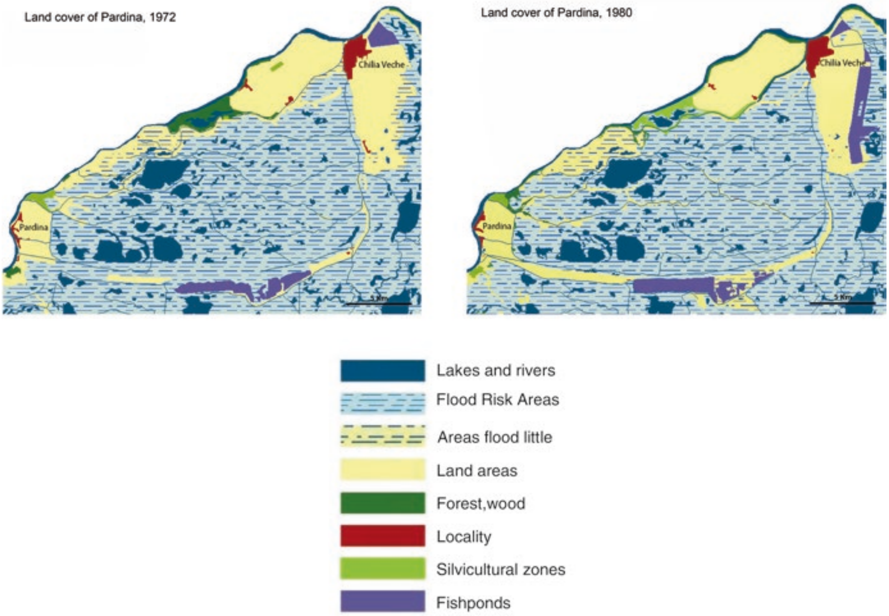

Fig. 17.5 Land cover of Pardina (Data Source: topographic map 1 /100,000, 1972 (*left*) and 1/25,000, 1980 (*right*))

most important topic in discussions of the sustainable development of the land where they live. The scientists working for the National Danube Delta Research Institute support the idea of flooding the polder and restoring it to its initial state, i.e., its state prior to the 1960s. The researchers working in this institute claim that a considerable portion (50–60%) of this polder is not cultivated and that crop rotation is not performed properly, with the result that the crops harvested here are quite small in comparison to the investments that have been made. However, the most clear-cut position on this issue is that of the Tulcea General Council. According to this council, polder flooding is merely utopian. They claim that the flooding would ruin Pardina's economic environment and lead to further degradation of the standard of living of the local population. Each year, the General Council collects several hundreds of thousands of Euros in concession fees. One of the problems that make the application of this project difficult is the fact that compensations would have to be paid to the polder farmers if they are expropriated. The Romanian Government has tried to find solutions to this problem, but given the current crisis, this project is no longer considered a priority (Niculescu et al. 2015b).

Seven thousand five hundred fifty hectares of the Sireasa polder have undergone similar radical agricultural development (Fig. 17.7). Such development has also occurred in Murighiol-Dunavat (on the Sfantu-Gheorghe Branch) (Fig. 17.8) as well as in Babina and Cernovca (located on the Chilia Branch) (Fig. 17.9), where

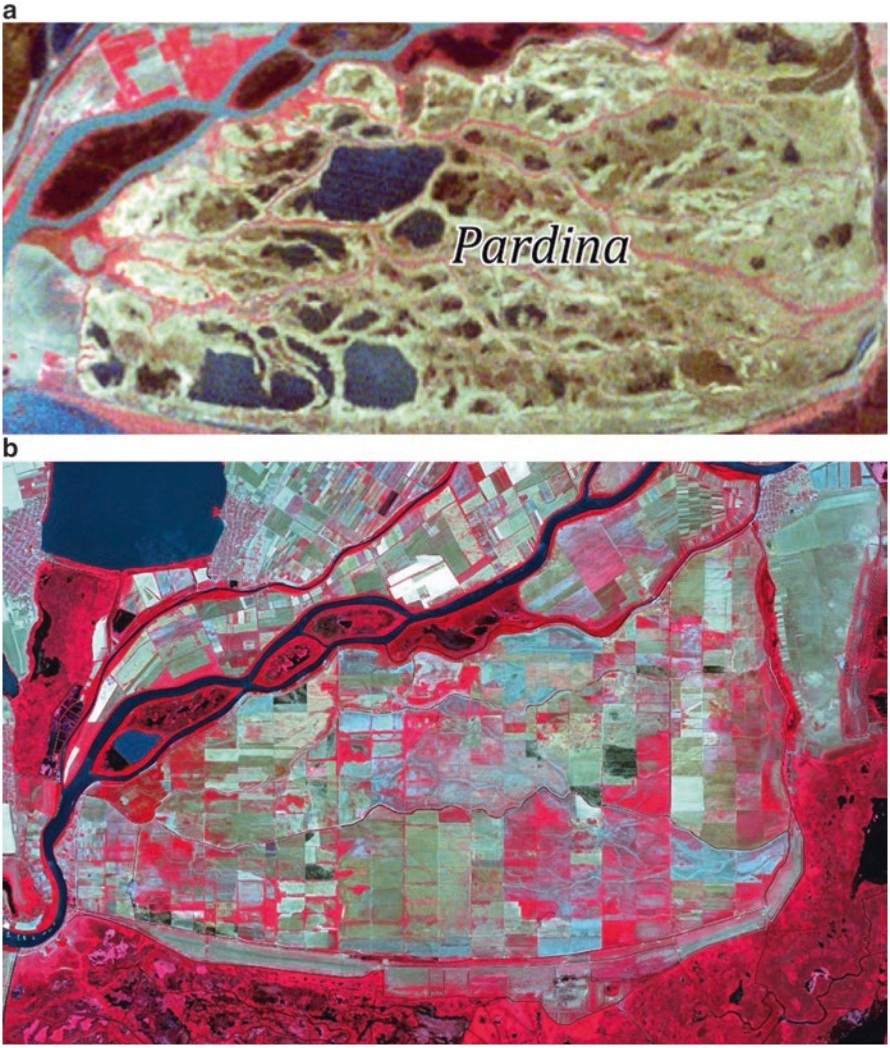

Fig. 17.6 Land cover of Pardina (**a**) Data source: Landsat-2 MSS, 23/04/1979). Pardina before the transformation in polder agricol (**b**) Data source: ALOS AVNIR-2, 10/06/2010. Polder agricol af Pardina after the fall of communism

the development involves smaller areas intended for rice production. This type of development also includes the polder Popina (Fig. 17.10).

According to various studies conducted by the researchers of the National Danube Delta Research and Development Institute, only 40% of the whole Murighiol-Dunavat polder was originally cultivated, whereas most of the surface of this polder is covered by organic deposits. The various studies carried out at that time (Munteanu 1979) clearly noted the major difficulties encountered during the

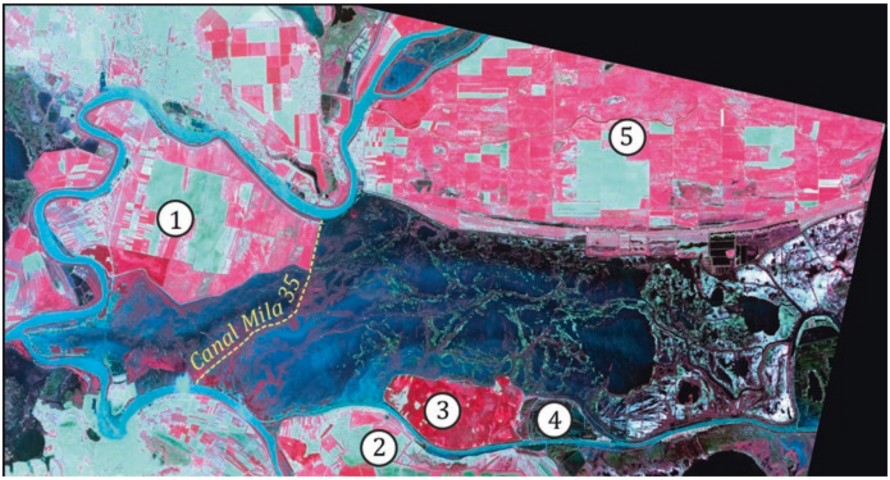

Fig. 17.7 Agricultural development in the northeastern part of the delta: *1* Sireasa development; *2* Rusca development; *3* Papadia development; *4* Maliuc development; *5* Pardina development (Data source: SPOT-5, 23/05/2006)

Fig. 17.8 Holbina-Dranov fish farming development (Data source: Landsat TM5, 20/08/1989)

Fig. 17.9 Babina (*1*) and Cernovca (*2*) agricultural developments before the restoration sites (Data source: Landsat TM5, 21/05/1992)

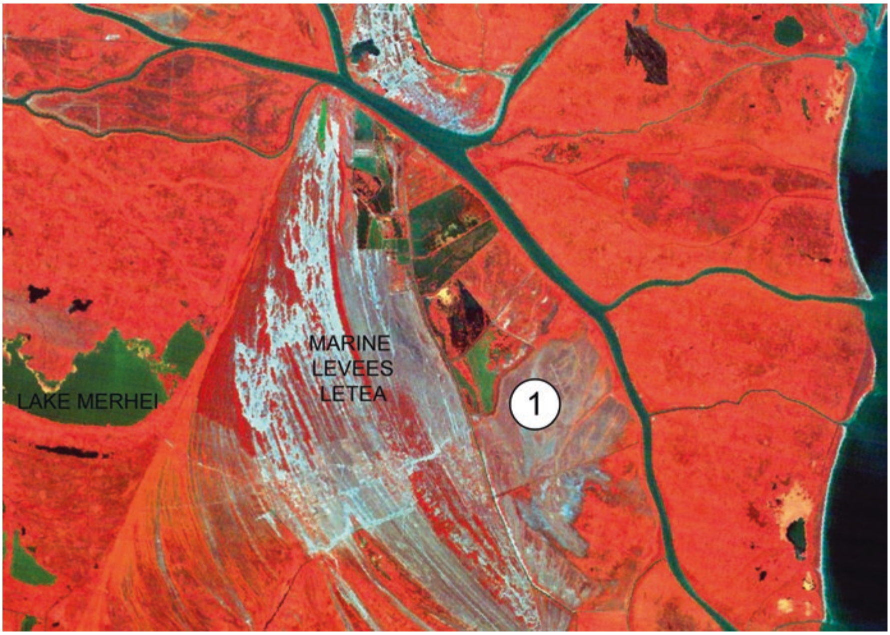

Fig. 17.10 Popina fish farming and agricultural development before the restoration site (Data source: Landsat TM5, 21/05/1992)

improvement of the quality of these soils; these include desiccation/drainage, soil salinization prevention and control, land leveling, soil texture homogenization, soil acidity problems, and reed removal problems.

The hydrological factor is the main aspect that disturbed the ecosystems of the spaces that were altered for agricultural purposes during the communist regime. The changes in the natural hydrological regime brought about by dykes have resulted in a series of physical and chemical changes in the soils and in the vegetation structure. In turn, these changes triggered changes in local fish and bird fauna habitats. The main alterations that were carried out here were damming and construction of a network of desiccation canals (hence, a substantial modification of the water circuit in these areas) and the replacement of wetland plants with agricultural crops, leading to profound changes in the vegetation structure and composition, marsh soil degradation (organic matter loss), increased soil salinity, reduction in the number of bird habitats, and loss of the filtration role of sediments and nutriments.

In the areas of concern, forestry developments are quite small in comparison to agricultural developments; the former only occupy 4650 ha. Canadian poplar plantations (*Populus canadensis*) are present in the Papadia, Rusca, Carasuhat, Pardina and Murighiol developments or along the main Danube branches. These plantations have reduced flower diversity, reflecting reduced delta biodiversity, and have led to the degradation of the forest ecosystems, especially those of the Letea and Caraorman forests.

All these developments, together with the full delta resources exploitation policy, have also stimulated the construction of canals around these facilities of economic interest. Examples of such canals include the Crisan-Caraorman canal connecting the Sulina branch and the Caraorman locality, where a sand exploitation facility was designed (nevertheless, this industrial facility was never built), the 10-km Mila 35 canal connecting the Tulcea and Chilia branches, which allows communication with and transportation to localities along the Chilia branch, other branch-straightening works designed to facilitate reed transportation, water supply in the fish farming developments and water supply for irrigation purposes. One of the last large changes during the communist regime occurred on the Sfantu-Gheorghe branch, which was straightened between 1985 and 1990; during this process, its length was reduced by 38 km from its original length of 108 km. The straightening of the branch, which was accomplished by shortcutting six meanders, has increased the water flow; consequently, the sediment flow has helped reduce the erosion of the southern coastline at the mouth of the branch.

The communist regime has left deep “scars” on the delta. The desire to exploit at all costs the deltaic resources has led to gigantic and ambitious projects that were disproportionate in comparison to the natural and fragile space represented by the delta. These projects displayed no regard whatsoever for the ecological component and for the natural balance, the only goal being economic development. Nevertheless, the natural balance preservation problems raised by delta resources exploitation were clearly noted as early as 1927 by Grigore Antipa in his *Report on the cultivation of the swampy areas of Romania*,[2] mentioned by Bethemont 1975.

Despite the financial investments made in deltaic resources exploitation during the communist regime, many of these developments failed, and others were progressively abandoned in the last years of the communist regime and especially after the fall of the regime. Given these facts, Bethmont's question (1975) is still very much current: "How could we exploit the wealth of the delta without making its fragility show?"

17.5 Ecosystem Alteration and Restoration in the Danube Delta Biosphere Reserve

In the early 1990s, after the fall of the communist regime and after the construction of various fish farming and agricultural development works as well as reed exploitation and branch shortcutting to facilitate navigation, 97,400 ha, i.e., 26.7% of the surface area of the delta, were excluded from the natural delta circuit. The ecosystems were completely upset, and the deltaic system was disrupted. The excessive exploitation of the deltaic natural resources in this region has resulted in the disappearance of reproductive areas used by the native fish and other animal species (20 bird species have disappeared) and in the clogging of the natural channels. The building of oversized canals (e.g., the Mila 35 Canal and the Crisan-Caraorman Canal) has trivialized and transformed the deltaic landscape. This ecosystem disruption and the need to protect an exceptional biodiversity – the heterogeneous mixture of habitats developed in the delta shelters a large community of plants and animals, the number of which is estimated to be approximately 5380 – and restore these ecosystems were arguments in favor of declaring the Danube Delta a Biosphere Reserve. Thus, in 1990, the Danube Delta, including the Razim-Sinoe lagoons area, was declared a Biosphere Reserve (by the Government Decision no. 983, article 5) with an independent management and scientific council.

That same year, i.e., in 1990, the Danube Delta (more precisely, its waterfowl habitat) was recognized as a Wetland of International Importance as defined by the Ramsar Convention of 1971; consequently, the Danube Delta has been included on UNESCO's list of World Heritage Sites. Also under the aegis of UNESCO, the Danube Delta was included in the MAB (*Man and Biosphere*) Programme in 1991. Finally, the law of 1993 (Law n°82 of December 7) and the Government Decision n°248 of 1994 stipulate the reserve law, its operation, scientific council, management board, and security and control team. This law was amended and adapted by Law no. 136/2011. According to these legislative documents, the Danube Delta

[2] According to him, "the potential of each section of the floodable area should be enhanced by devoting it to the production for which nature itself created it, thus achieving its productivity and profitability peak; the development system should also take into consideration that the works performed for this purpose should not alter the natural balance and trigger disastrous consequences."

Reserve is an ecological area of national and international importance. This area includes the deltaic space, the Murighiol Plopu salt marshes, the Razim Sinoe lagoon system, the Isaccea-Tulcea section and the Black Sea coastline from the Chilia Branch to Cap Midia, also including the territorial waters.

Within this perimeter, the Danube Delta shelters a large number of species belonging to a large number of systematic units. Moreover, the Danube Delta stands out due to its very high density of rare species or species that do not exist elsewhere on the continent. The protection of fauna, more precisely of birds, and also of forests became a must approximately 80 years ago, after the First World War. After 1930, there were two natural reserves: Letea Natural Park (1930) and Rosca-Buhaiova-Hrescica Area (1940). In 1950, thanks to the creation of the Commission for Natural Monuments by the Romanian Academy, the number of natural reserves increased to six (three bird reserves, one forest reserve and two bird and forest reserves). At that time, the overall surface area of the reserves was 41,046 ha.

Whereas other protected areas (national parks, natural monuments, strict natural reserves, etc.) enjoy strict protection, the Danube Delta Biosphere Reserve has several objectives. These are the preservation of the ecosystems (flora and fauna), the stimulation of traditional economic activities that are not harmful or are minimally harmful to the ecosystems, and the information and education of the population about the scientific importance of the existing ecosystems and the importance of their preservation. This concept of a reserve should be integrated as a deltaic space management tool meant to harmonize the traditional economic activities of the local population with the requirements of nature preservation. According to this concept, three types of zones were defined in the Danube Delta Biosphere Reserve (Fig. 17.11).

The strictly protected zones have an overall surface area of 506 km^2 (i.e., 8.7% of the total surface area). These zones include 20 reserves in which any economic activity is forbidden. Human access is permitted only for scientific research or environmental monitoring purposes. Buffer zones (2233 km^2, i.e., 38.5% of the total surface area) are generally defined around the strictly protected zones. Certain traditional natural resources exploitation activities are allowed in the buffer zones. These zones are intended to reduce anthropogenic pressure and to ensure a smooth transition to the economic zones. The economic zones (3061 km^2 or 306,100 ha, i.e., 52.8% of the area of the reserve) include floodable land, land protected by dykes for agricultural, fishery or forestry use, and localities. All economic activities are allowed in these zones; however, certain restrictions imposed by the ARBDD[3] may apply. Whereas all the zones which were previously altered are included in the reserve, these "altered" zones are not on UNESCO's list of World Heritage Sites; therefore, its surface area is only 312,440 ha.

[3] Management of the Danube Delta Biosphere Reserve.

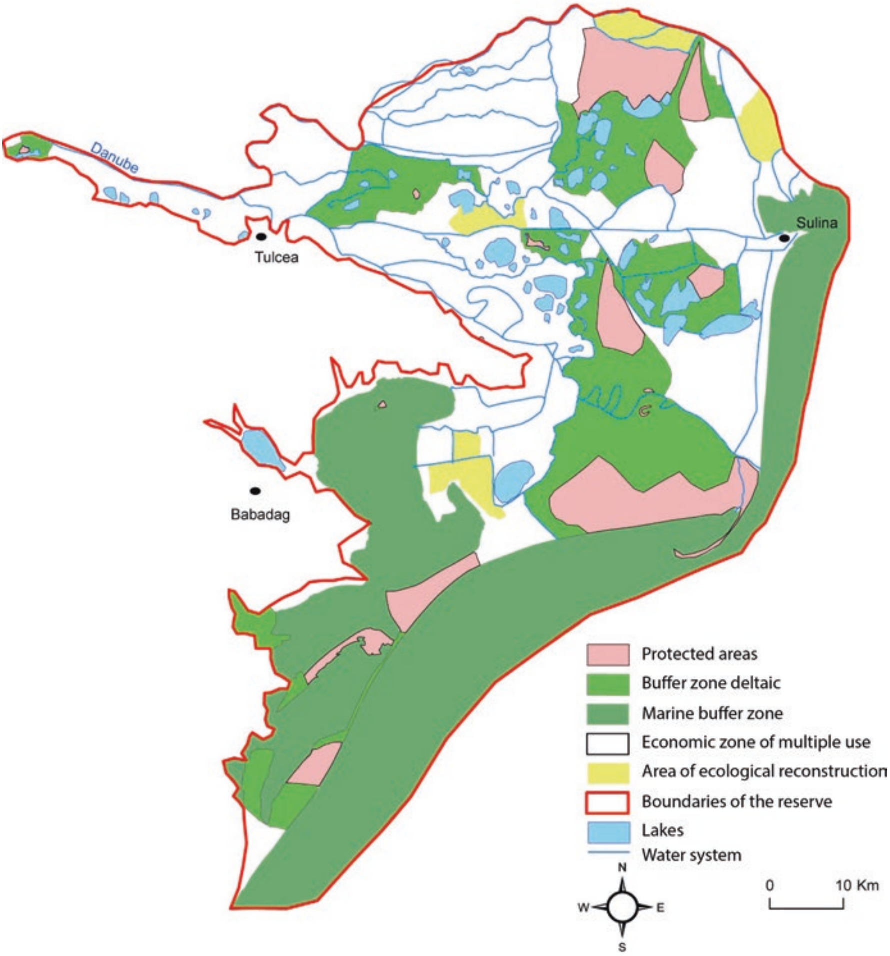

Fig. 17.11 Danube delta biosphere reserve zoning: protected zones; buffer zones: deltaic and marine; the economic zones

17.6 Site Restoration in the Danube Delta

In the last decades of the twentieth century, the Danube Delta has suffered due to anthropogenic interventions that led to dramatic changes in some areas. These interventions were the impoundment of large areas for intensive agricultural, fishery and forestry use, which led to dramatic alterations and changes in water balance. These interventions also affected natural processes, the ecological balance and the specific ecological functions of wetlands and led to alteration or even additional specific loss of wetland habitats. When work was halted in 1990, the impounded areas occupied an area of 97,408 ha (22% of the total area of 482,592 ha). Studies for the rehabilitation/re-vegetation of this area were initiated immediately after the Danube Delta was declared a Biosphere Reserve in1990.

Within the reserve, there are ecological reconstruction zones, that is, areas in which the ARBDD has initiated ecological balance restoration projects using adequate technical means. This ecological reconstruction policy concerns all the previously dammed areas (97,408 ha, i.e., 27.6% of the current area of the delta) that are devoted to agricultural and fish farming developments or to reed exploitation. Considering that the Danube Delta includes 30 types of ecosystems that are highly dependent on the oscillation of river levels, the main objective of this ecological recovery is to restore the natural hydrological circuit of the economically developed areas. A solution to these efforts of reconnection to the hydrological regime of the delta was suggested in 1994; it consisted of digging holes in the dykes, thereby allowing the water to flow freely in these dammed areas. Other types of ecological restoration development include the calibration and closing of canals if the water flows directly towards the lakes without being filtered by the reeds.

So far, 328 km of canals have undergone ecological reconstruction development as well as drainage and unclogging, whereas 15,712 ha have been reconstructed to be environmentally friendly. The ecological reconstruction developments were conducted in Popina (southern part), Babina, Cernovca, Fortuna and Holbina-Dunavat (Fig. 17.12). Other types of developments (on the canals) may be found in Matiţa-Merhei, Magearu-Cardon, Gorgova-Uzlina, Şontea-Fortuna, Dunavăţ-Dranov, Roşu-Puiu and Somova-Parcheş. The ecological progress made after these redevelopments includes the establishment of new bird and animal habitats, widening of the fish and waterfowl reproduction areas, increased hydrological flow and storage capacity of water, and increased sediment retention.

After the political changes in Romania in the early 1990s, the first proposed project in the Danube Delta Biosphere Reserve was Babina. The goal of this project was to switch from an intensively used, unspecified area to a state close to that of nature. Thus, in spring 1994, abandoned agricultural land in Babina in the northeastern Danube Delta has been reconnected to the natural regime of flooding of the Danube (Fig. 17.12). A monitoring program has also been developed and implemented to answer major questions raised by the recovery process and to check the ecological success of the reconstruction work.

After the commissioning of the Babina and Cernovca dykes (1994) and after the recovery of the floodability index, these two polders saw the emergence of different plant associations that depended on the Danube level fluctuations and on land morphometrics. The following habitats were restored in the Babina Polder: aquatic habitats, low and midland levee habitats and land habitats (high levees). Aquatic habitats are represented by hydrophilic species that are also found in other natural areas of the delta. These include *Hydrocharismorsus ranae*, *Lemna minor*, *Lemnatriscula*, *Nymphea alba*, *Salvinia natans*, etc. The low and midland levee habitats are represented by floodable land during major floods. Various hydrophilic species such as *Carex riparia*, *Carex acutiformis*, *Menthe aquatica*, and *Lycopus europaeus* are left behind by the floods on the midland levees located in the middle and in the eastern part of the polder. These species are subsequently replaced by the

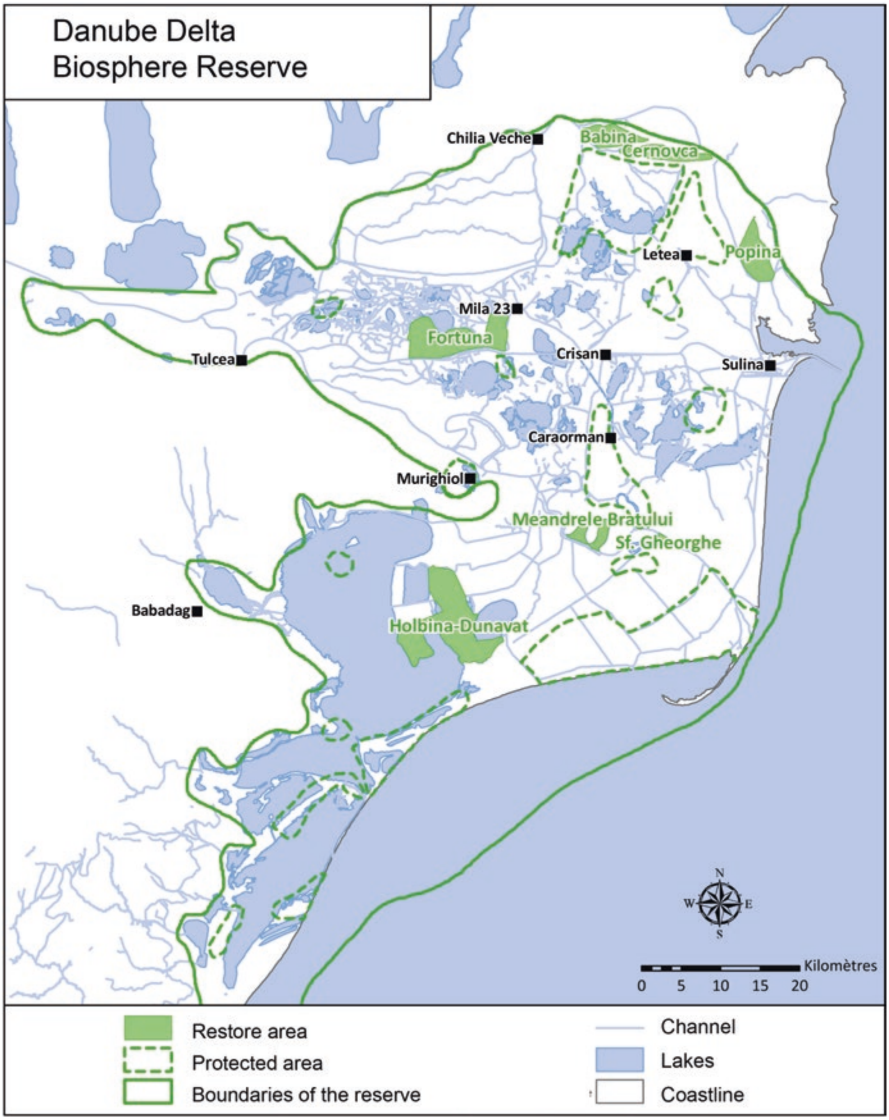

Fig. 17.12 Ecological restoration areas: the ecological reconstruction developments were conducted in Popina (southern part), Babina, Cernovca, Fortuna and Holbina-Dunavat

meso-hydrophilic species *Tanacetum vulgare*, *Atriplex tatarica*, *Puccinelia limosa*, and *Tamarix ramosissima*. Halophile species such as *Astertripolium* also grow on these types of levees. Low-levee vegetation is represented by hydrophilic species where *Phragmites australis* predominates; however, since 1996 this has been replaced by *Thypa angustifolia*. High levees have similar vegetation, and this has not changed after site restoration. Mixtures of *Hordeum hystrix*, *Cynodon dactylon*, *Atriplex tatarica*, *Torilis arvensis*, *Lotus tenuis*, *Verbascum blattaria*, *Artemisia annua*, and *Lactuca tatarica* grow on this type of levee. These species provide pastures for animals.

An analysis of the ecological characteristics of the plant species identified on the polders after their restoration was conducted. Specific biodiversity and remarkable hydrophilic species diversity were noted.

17.7 Plant Cover Estimation in the Restored Sites in the Danube Delta: A Remote-Sensing Approach

17.7.1 Remote Sensing and Restoring Wetland Habitats

Wetland habitat is being restored throughout the world (Zedler and Kercher 2005); however, achieving conservation goals and objectives requires knowledge of vegetation composition, structure, and change over time with respect to attributes such as percent cover, biomass, and plant diversity (Phinn et al. 1999). Therefore, there is a need to further develop, refine, and disseminate site- and landscape-level monitoring methods (Simenstad et al. 2006). Having developed criteria for selecting wetland sites to be restored or enhanced, wetland managers must prioritize the sites based on ecological and economic considerations (Klemas 2013). Remote sensing techniques can provide a cost-effective means for selecting restoration sites and observing their progress over time.

Remote sensing involves the acquisition of information about the Earth's surface at a remote distance, usually by airplane or satellite (Jensen 2000). It offers tools to map, measure, model, and evaluate wetland restoration efforts in a cost-effective manner. The use of this technology in the ecological sciences is rapidly increasing because ecosystems such as wetlands can be monitored at various spatial and temporal scales (Jensen et al. 1995; Guo and Psuty 1997; Michener and Houhoulis 1997; Apan et al. 2002; Heinl et al. 2006; Papa et al. 2006; Niculescu et al. 2016).

Despite the increasing use of remote sensing for wetland inventory and monitoring, there has been limited use of this technology in the restoration of wetlands (Phinn et al. 1999; Hinkle and Mitsch 2005). Remote sensing is ideal for monitoring restored wetlands because it provides a high spatial and temporal intensity of measurements in relatively inaccessible and sensitive sites without the potential invasiveness that traditional field methods present to delicate habitat conditions, bird-nesting territories, and endangered species habitat (Shuman and Ambrose 2003). In an ideal situation, remotely sensed images are acquired when decisions can be made about imagery specifications and field data collection that will make change detection accurate and applicable to the monitoring of a restoring wetland.

Recent advancements in imaging science have provided finer spatial, spectral, and temporal resolution as well as reduced price. In addition, non-optical data sources such as radar data (e.g., SAR, RADAR) and laser altimetry (e.g., LiDAR), have been shown to add value when combined with optical remote sensing data (Ramsey et al. 1998; Rosso et al. 2005; Niculescu et al. 2016). Change detection is an important tool for wetland restoration monitoring because it provides measurements of incremental changes that can be used for inventory and benchmark purposes; knowledge of these changes can then be integrated with adaptive management plans and used to target specific restoration goals (Tuxen et al. 2008).

The success of restoration, however, is difficult to assess. The degree of success for many of these restoration sites is still being debated, especially since there is no full agreement on criteria used to measure success. The creation, enhancement, or restoration of coastal habitats requires much time and constant attention (Klemas 2013). Remote sensing offers accurate automated methods for detecting change in restored wetlands. Vegetation change detection is a powerful indicator of restoration success. The restoration projects use vegetative cover as an important indicator of restoration success.

Synthetic aperture radar (SAR) technology provides the increased spatial resolution that is necessary in regional wetland mapping, and SAR data have been used extensively for this objective (Bourgeau-Chavez et al. 2005; Lang and McCarty 2008; Novo et al. 2002). Radar has the capability of penetrating the plant cover canopy and detecting submerged sectors and soil surface moisture. Although the spatial resolution of radar images does not allow thorough and detailed habitat mapping, these images are useful for mapping wetland vegetation. The radar polarimetry and polarimetric parameters contribute significantly to the improvement of vegetation identification based on polarization channels. Multipolarization and multi-frequency radar devices are also used for the classification of wetland vegetation depending on their wavelengths, polarizations and backscattering mechanisms and can be used to estimate the density and size of the vegetation. Microwave radiation polarization, like radar beam incidence angle and wave frequency, has long been acknowledged as an important parameter for object recognition and understanding object features. Access to the scattering matrix permits several analytical approaches and hence various ways of assessing the potential of multi-polarized radar images. One approach consists of synthesizing pixel-based signal strength, which should have been measured at the same frequency for any polarization configuration (linear and/or circular) (Niculescu et al. 2015a). The sensitivity of microwave energy to water and its ability to penetrate vegetative canopies makes SAR ideal for the detection of the hydrologic features under vegetation.

SAR image time series such as those provided by the Sentinel-1 satellite allow significant improvements in vegetation classification. The key advantage of satellite-borne SAR imaging is its independence of cloud cover, and because it is an active sensing system, its independence of sun-induced reflection. Consequently, SAR imagery has become an important tool for distinguishing different vegetation classes. Recently, polarimetric SAR images have been analyzed using decomposition theorems such as alpha/entropy decomposition, which increases the accuracy of vegetation analysis from microwave data. However, there is a wide choice of remote-sensing satellites, radar, and optical. Whereas optical satellites usually operate in one imaging mode, radar satellites can be programmed to work indifferent configurations. The user must choose the polarization configuration, the incidence angle, and the spatial resolution associated with the chosen imaging mode. Combined approaches of using optical and microwave images can improve the vegetation analysis.

Airborne laser instruments such as LiDAR represent innovative tools for management applications, including flood zone delineation, monitoring beach nourishment projects, and mapping vegetation (Niculescu et al. 2016) and changes along sandy coasts and shallow benthic environments due to storms or long-term sedimentary processes (Klemas 2013). Identifying potential restoration sites and prioritizing them using ecological and economic criteria is by no means a simple task (Russell et al. 1997; Thayer 1992; White and Fennessy 2005). The combined use of LIDAR, radar, and multispectral imagery can improve the accuracy of monitoring vegetation species discrimination and provide a better understanding of the topography/bathymetry and hydrologic conditions.

17.7.2 Dataset

We used the following satellite images in this study: 20 Sentinel-1 images acquired between 9.10.2014 and 01.04.2016 (Table 17.2) and one Sentinel-2 image acquired on 28.04.2016 in the restored areas in the northern part of the delta (Babina and Cernovca). The Sentinel-1 data were acquired in a time series that covered the entire growth season of 2015 and part of 2016. This enabled us to determine the influence of the time dimension and of the polarimetric dimension (VV and VH polarization are available) on the characterization and classification of the vegetation in the restored delta areas.

Since it was first launched in April 2014, the Sentinel-1 satellite has allowed specialists to monitor the earth's surface day and night regardless of weather conditions and has transmitted high-resolution space images free of charge. The Sentinel 1 SAR mission is part of the Copernicus Programme – European Earth Observation Programme, which was previously called GMES (Global Monitoring for Environment and Security), of the European Space Agency. Placed on an orbit at an altitude of 693 km, Sentinel-1 operates in four data collection modes: the StripMap (SM) mode, the Interferometric Wide swath (IW) mode, the Extra-Wide swath (EW) mode and the Wave (WV) mode. Each mode provides different products with respect to spatial resolution and imaging swath. Sentinel-1 images are captured in C band (5.5 cm), and they may exhibit simple HH or VV polarization or double HH+HV or VH +VV polarization. The data used in our research were collected in the IW mode. This mode includes three sub-swaths, namely IW1, IW2 and IW3, which correspond to cyclical antenna deviations. This mode provides GRD (Ground Range Multilook Detected) and SLC (Single Look Complex) images made up of three IW. The GRD images are Multilook images (five looks for the IW mode) with less speckle noise and coarser space resolution. Although the SLC products have finer resolution, it is difficult to use them directly due to the phase information,

Table 17.2 Sentinel-1 imagery used in this study

Date	Incidence angle	Orbit
09-10-2014	38.055	Ascending
02-11-2014	38.786	Descending
26-11-2014	38.653	Descending
13-01-2015	39.215	Ascending
26-03-2015	39.856	Ascending
07-04-2015	38.569	Ascending
01-05-2015	38.421	Descending
13-05-2015	39.654	Descending
30-06-2015	39.478	Ascending
05-08-2015	38.665	Descending
17-08-2015	37.789	Descending
29-08-2015	38.669	Ascending
10-09-2015	39.285	Descending
22-09-2015	39.456	Ascending
09-11-2015	38.721	Descending
03-12-2015	38.451	Ascending
27-12-2015	39.885	Ascending
20-01-2016	38.411	Descending
13-02-2016	39.662	Ascending
01-04-2016	39.453	Ascending

which seems useless as it prevents extraction of additional information in certain cases.

GRD image calibration is vital for viewing the maximum amount of information on an image. In our research, the $\sigma 0$ value is extracted using Calibration Tools of the OrfeoToolbox software, which provides us with the backscattering coefficient of the area. These values depend on the targets illuminated by the beam, on ground roughness and moisture and, in the end, on the vegetation density.

Sentinel-2A is the second satellite of Europe's Copernicus Programme, following the Sentinel-1A radar satellite launched last year. In partnership with the European Commission and within the frame of the Global Monitoring for Environment and Security (GMES) program, the European Space Agency (ESA) is developing the Sentinel-2 optical imaging mission, which is devoted to the operational monitoring of land and coastal areas. Sentinel-2 is the operational mission devoted to the observation of continental surfaces in decametric resolution. The Sentinel-2 mission ensures a systematic full land cover with 10-day repetitiveness by a single satellite and 5-day repetitiveness by two satellites. Sentinel-2 has 13 spectral bands, 3 of which are in the near infrared (SWIR). These images have a 290-km-wide field of view and 10-m, 20-m or 60-m resolution depending on the spectral bands. The Sentinel-2 mission is a land and coastal areas monitoring constellation of two satellites (Sentinel-2A, which was launched on 23 June 2015, and Sentinel-2B, which will follow in the second half of 2016) that provide high-resolution optical imagery and continuity for the current SPOT and Landsat missions. The mission will provide global coverage of the Earth's land surface every 10 days with one satellite and every 5 days with two satellites, making the data of great use in ongoing studies. Sentinel-2 delivers high-resolution optical images for land monitoring, emergency response and security services. The satellite carries a multispectral imager with a swath of 290 km. The imager provides a versatile set of 13 spectral bands spanning from the visible and near infrared to the shortwave infrared, featuring four spectral bands at 10-m, six bands at 20-m and three bands at 60-m spatial resolution. The imager's 13 spectral bands, from the visible and the near infrared to the shortwave infrared at different spatial resolutions, take land monitoring to an unprecedented level. In fact, Sentinel-2 is the first optical Earth observation mission of its type to include three bands in the 'red edge', which provides key information on the state of vegetation. The 13 spectral bands span from the visible and the near infrared to the short-wave infrared. The four bands at 10 m are the classical blue (490 nm), green (560 nm), red (665 nm) and near infrared (842 nm) bands dedicated to land applications. The six bands at 20 m include four narrow bands in the vegetation red edge spectral domain (705 nm, 740 nm, 775 nm and 865 nm) and two large SWIR bands (1610 nm and 2190 nm) dedicated to snow/ice/cloud detection and to vegetation moisture stress assessment. The three bands at 60 m are dedicated to atmospheric correction (443 nm for aerosols and 940 nm for water vapor) and to cirrus detection (1380 nm) (Baillarin et al. 2012).

17.7.3 Remote-Sensing Methodology

The chosen methodology is associated with multi-data radar and optical image classification methodology. We began with the preliminary processing of the radar and optical images (Figs. 17.13 and 17.14 show the radar data).

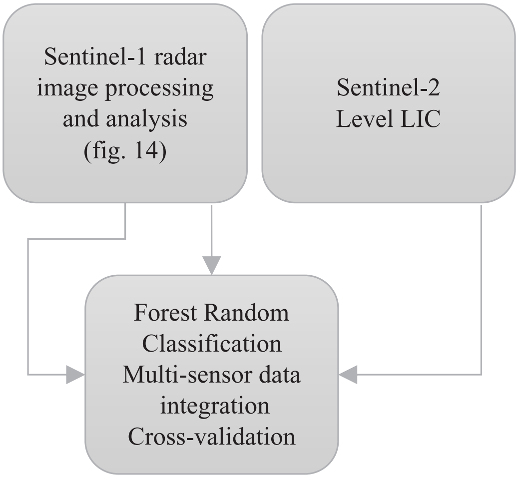

Fig. 17.13 Data processing procedure

A typical processing sequence applied to SAR data entails radiometric calibration, speckle filtering, and orthorectification. Radar signals require pre-processing to account for geometric distortions and for differences in illumination conditions due to topography and the surface being illuminated to one side of the satellite. An additional step is needed to remove noise caused by reflections from features that are not of interest. This is called *speckle noise* and is removed by a process called *speckle filtering*. The filtering applied is filtering of the Lee type (Fig. 17.14). Adaptive filters use local statistics to filter the data and so reduce image speckle and, in some cases, preserve or enhance edges and other features. At the same time, the backscattering coefficient was analyzed for the two different polarizations depending on a set of parameters related, on the one hand, to the RSO characteristics (acquisition frequency, polarization and geometry) and, on the other hand, to the attributes of the target (geometric structure, dielectric constant, biomass, etc.). The backscattering coefficient is usually expressed in decibels (dB), yielding a normalized value comparing the observed power to the rated power for an equivalent 1-m^2 surface and corresponding to the distance to the ground. The backscattering coefficient is also very much influenced by factors related to the sensor configuration and collection geometry.

The optical image (Sentinel-2) was already orthorectified in the UTM 35N cartographic system by ESA (level 1C). The geometric correction of image data is an important prerequisite that must be performed prior to using images in geographic information systems (GIS) and other image-processing programs. To process the data with other data (radar) or maps in a GIS, all the data must be based on the same reference system. Using a combination of different sensors, we resampled the data to the smallest pixel size between optical and radar. All the datasets were orthorectified, resampled to a 10-m pixel size and separately classified.

We then performed synthetic Random Forest classifications, first for all the Sentinel-1 radar data and then using combinations of the Sentinel-2 image. The

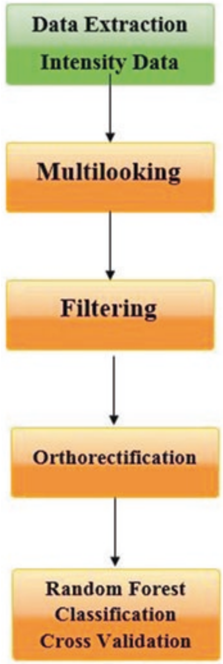

Fig. 17.14 Radar data processing procedure

supervised classifier used is the Random Forest algorithm, which is available in OrfeoToolbox (version 5.0) free software. Random Forests offers high-quality mapping of different vegetation types with much faster computation compared to other state-of-the-art classifiers such as, for instance, Support Vector Machines with Gaussian kernels (Inglada et al. 2016). Random Forest is an ensemble learning technique and builds upon multiple decision trees. Each decision tree is built using a subset of the original training data and is evaluated based on the remaining training features. New objects are classified as the class that is predicted by the most trees. According to Rodriguez-Galiano et al. 2012, the classifier has three main advantages for land cover classifications from remote-sensing images: (i) it reaches higher accuracies than other machine-learning classifiers; (ii) it has the ability to measure the importance level of the input images; (iii) it makes no assumptions about the distributions of the input images (cited by Hütt et al. 2016). We use the following parameters for the Random Forest algorithm: 100 trees, maximum depth of the tree 25 and minimum number of samples in each node 25.

The final stage of image processing relates to the integration of several images from two satellites (Sentinel-1 and Sentinel-2), which have different spatial resolutions. Image integration is a method for combining information from various sources. The combined analysis of optical and microwave imagery uses the advantages of both systems for vegetation classification.

17.7.4 Field Observation and Validation of Results

Another method is field observation and validation of results. Field observations are vital in remote sensing. In our research, the data collection stage prior to validation of the results of supervised classifications includes two categories of surveying methods, random (probabilistic) methods and empirical (non-random) methods. The survey was based on the satellite imaging document. Point sampling was used during this data processing stage. For each class, 1000 training points and 1000 control points (not the same points) were randomly chosen. This survey determines whether an observation unit belongs to a sample by random draw. In this case, the probability law is known. The random draw is stratified starting from all the homogeneous thematic areas. The stratification was initially performed prior to the field investigation phase. This first stage stratification is a morphological stratification that relies on textural homogeneity, backscattering and thematic homogenization criteria. As concerns field observations, the ground surveys (20 floristic surveys per thematic class) carried out in the restored delta areas allowed us to determine the vegetation typology in the surveyed area. Vegetation description is physiognomic and includes land cover rate estimation. Depending on the size of the homogeneous area, the size of the observation unit is more or less significant. The vegetation structure and type were measured at each point within a 100-m radius of the observer. Some floristic information was also gathered, including a list of species classified by physiognomic layers (trees, shrubs, and grasses).

The results of the evaluation are summarized in a confusion matrix. Based on the confusion matrix, statistical accuracy parameters are calculated. One is the overall accuracy, which counts pixels that are correctly classified in the reference divided by all pixels that are taken for reference. This procedure is used for both optical and microwave image classification.

17.7.5 Remote Sensing and Restoration Areas in the Danube Delta

The results of this study relate to combinations of data from different satellite sensors (Sentinel-1 time series, Sentinel-2) that are used to improve the accuracy of recognition and mapping of major vegetation classes in the restoring areas in the Danube Delta. First, the data from each sensor are classified and analyzed. The results show quite good classification performance (87.5% mean accuracy for Sentinel-2; 95.7% for the Sentinel-1 time series) in this first step. The combination of the Sentinel-2 time series and optical data from Sentinel-2 improved the performance of the classification (97.1%) (Fig. 17.16).

The vegetation types were labeled according to ten classes (figures classifications). These classifications allowed us to distinguish several classes of reeds in the 'large marsh vegetation' class (reed vegetation on salinized soils, pure reed

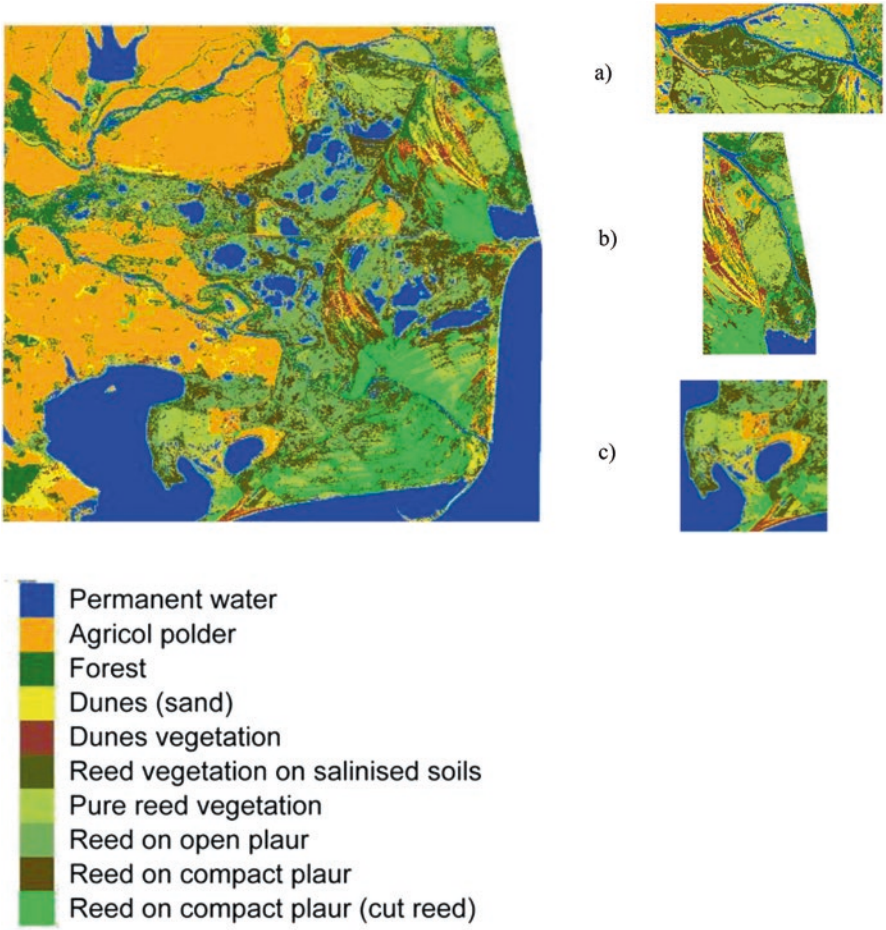

Fig. 17.15 Random Forest classification in ten classes vegetation: Optical image Sentinel-2.5 classes of reed vegetation with some confusion between these classes. The restored areas: (**a**) Babina-Cernovca; (**b**) Popina; (**c**) Dranov-Holbina

vegetation, and reed vegetation on open plaur (floating vegetation called *plaur* (floating reed bed) is an association of reeds and other wetland plants that grow on a one-meter thick cover made up of roots, soil and various organic materials) and two classes of reed vegetation on compact plaur (one class with cut reeds).

The classification accuracy of the Sentinel-2 image (Fig. 17.15) was estimated to be 87.5%, which was inferior to that of the time series from the radar data provided by Sentinel-1. The Sentinel-1 images time series classifications (95.7% mean accuracy) display very good accuracy.

The classification precision analysis per class proves that the Sentinel-2 images allow the identification of all ten classes of vegetation considered in this study. The following classes exhibit satisfactory accuracy for some of the restoring areas: reed vegetation on salinized soils (81.4%), pure reed vegetation (76.9%), reed vegetation on open plaur (87.3%). On the other hand, the class 'reed on compact plaur' exhibited lower performance in the mapping results, yielding an accuracy of 59.7% (Table 17.3).

By integrating the Sentinel-1 time series with optical images such as Sentinel-2, the quality of the habitat maps of the restoring areas in the Danube Delta can be considerably improved (Fig. 17.16).

Data integration between the Sentinel-1 and Sentinel-2 images provides classification with an overall accuracy of 97.1% and very good class accuracies ranging

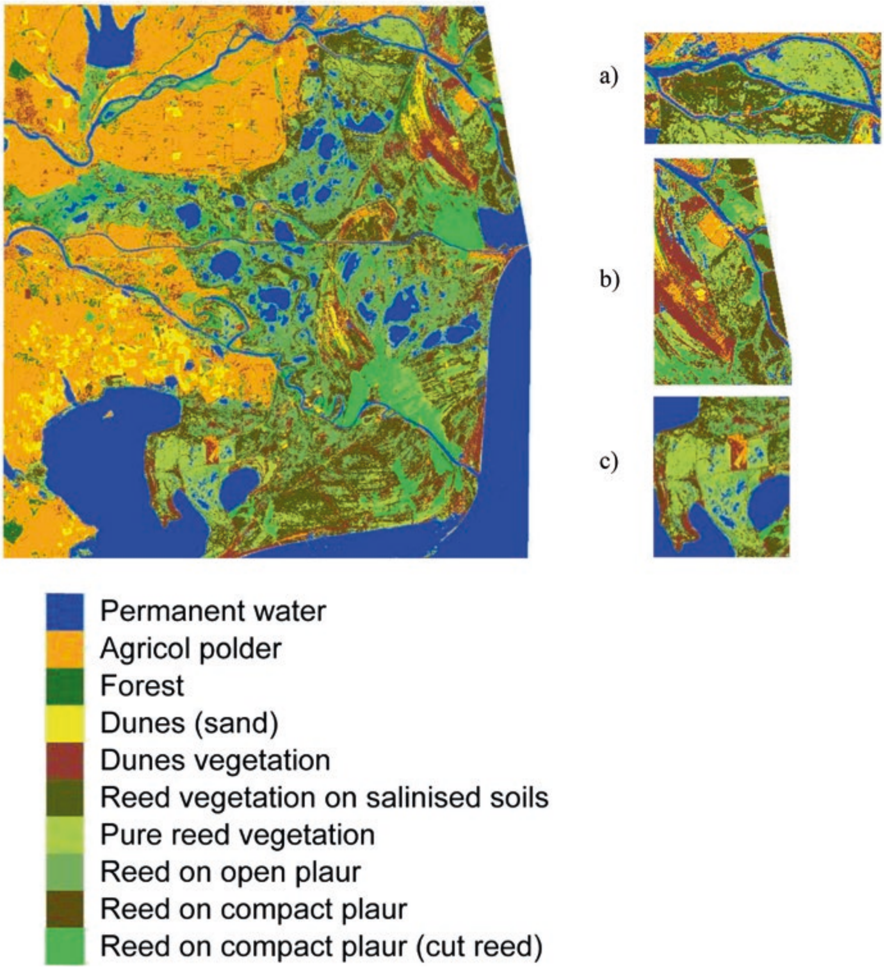

Fig. 17.16 Multi-sensor data integration Sentinel-1 radar time series and Sentinel-2 optical sensor. Five classes of reed vegetation and less confusion between these classes. The restored areas: (**a**) Babina-Cernovca; (**b**) Popina; (**c**) Dranov-Holbina

Table 17.3 Performance of the classification by class and all classes of the Sentinel-2

Class	Sentinel-2 per cent accuracy
1	100.0
2	91.5
3	90.0
4	98.4
5	94.4
6	81.4
7	76.9
8	87.3
9	59.7
10	96.4
Mean	**87.576**

Table 17.4 Performance of the classification by class and all classes of multi-sensors and time series of Sentinel-1

Class	Sentinel-1 per cent accuracy	Radar + optical per cent accuracy
1	99.5	99.8
2	95.1	98.3
3	96.4	98.1
4	94.5	99.6
5	96.8	99.5
6	96.8	97.1
7	92.1	91.1
8	96.7	97.9
9	91.5	90.3
10	99.4	99.9
Mean	**95.87**	**97.151**

from 90.3% to 95.8%. The classes 'reed vegetation on salinized soils' (97.1%), 'pure reed vegetation' (91.1%), 'reed on open plaur' (97.9%), and 'reed on compact plaur' (cut reed) (99.9%) were well mapped and show good accuracy (Table 17.4 and Fig. 17.17).

The mapping accuracies were summarized using confusion matrices and statistics including user, producer and overall accuracy and Cohen's K (Fig. 17.18). Classification accuracy was assessed using global and Kappa indices. Very good Kappa indices were obtained; for the optical data, the Kappa index was 0.86, and for the multi-sensor data integration, the Kappa index was 0.96. The classification accuracy was estimated using cross-validation and by calculating the percentage of correctly classified pixels on the resulting maps. These present the reference class labels in rows and the labels predicted by the classifier in columns. The results are expressed in percentages with respect to the reference labels, and therefore, values in the diagonal represent Producers Accuracy.

Figure 17.18 shows the confusion matrix for the optical data. The matrix reveals many confusions of reed classes involving different forms (reed on salinized soils,

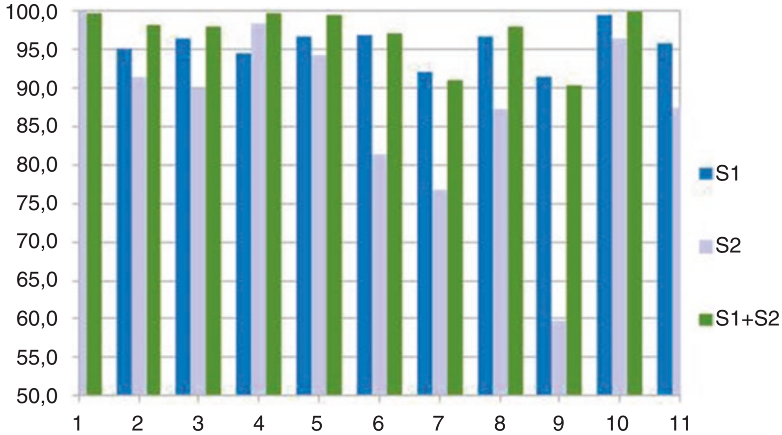

Fig. 17.17 Performance for the multi-sensor and time series classification by class (per cent accuracy) (S1 = radar sensor time series Sentinel-1, S2 = optical sensor Sentinel-2 and S1 + S2 = multi-sensor data integration)

pure reed, reed on compact plaur). The most important confusions concern the various reed classes that characterize the habitats in the restored areas. 'Reed on compact plaur' has a Producers Accuracy of 55.1%, with confusions with the 'pure reed vegetation' class (16.26%) and the 'reed vegetation on salinized soils' class (10.42%). Other confusions concern the 'pure reed vegetation' class, which displays a Producers Accuracy of 78.65%. The most important confusion in this class, 21.50%, is represented by the 'reed on compact plaur' class. Thu, even when optical data are used, the distinction between the plant formations of these wetlands is not always easy. Prior research has revealed that when optical imaging is used there is spectral confusion between wet and dry environments and also between various types of wetlands. Marsh and swamp identification in the spring usually causes fewer problems than identification of wetlands with drier water regimes, such as peat bogs or swamps with considerable foliar biomass (Ozesmi and Bauer 2002).

The confusion matrix of the classification resulting from the Sentinel-1 time series processing reveals very good Producers Accuracy values; most classes show values ranging from 90.01% to 99.72%. The most substantial confusions concern the 'pure reed vegetation class', with a Producers Accuracy of 90.01%. This class is mixed with the 'reed on compact plaur' class (4.21%) and with the 'reed vegetation on salinized soils' class (1.26%). Radar data provide information especially on plant physiognomies. This analysis supplies information on polarimetric data in relation to the geometric characteristics of the physiognomies of the plants growing in the restored areas of the delta and enables us to draw conclusions about ways to distinguish among the various plant physiognomies.

Finally, the confusion matrix of the multi-sensor data integration revealed excellent classification results when the Producers Accuracy rates were higher than 90%, i.e., between 90.4% and 99.91%. The low confusion values shown by this matrix

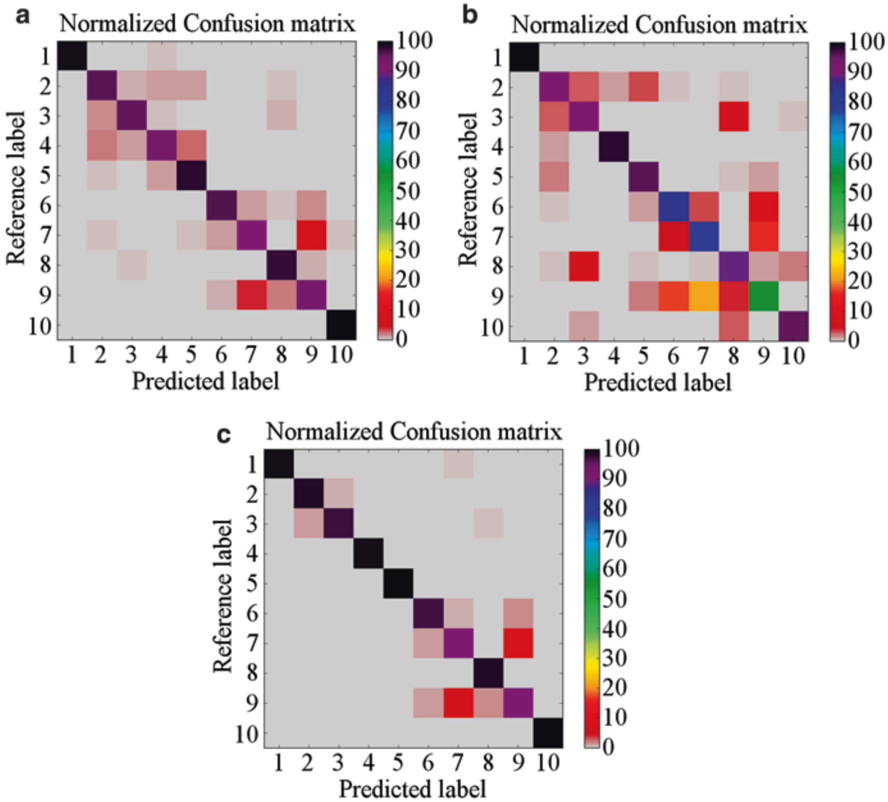

Fig. 17.18 Confusion matrix of random forest classifications: (**a**) Sentinel-1 time series; (**b**) optical data Sentinel-2; (**c**) Multi-sensor data integration Very good Producers Accuracy values for the confusion matrix of Sentinel-1 radar: most classes show values ranging from 90.01% to 99.72%. The confusion matrix of optical data with many confusions of reed class. The confusion matrix of the multi-sensor data integration: the Producers Accuracy rates were higher than 90%, i.e. ranging from 90.4% to 99.91%

concern the two classes for which we also read confusions in the previous matrices: reed on compact plaur and pure reed vegetation.

17.7.6 Temporal Intensity Radar Data Signature

Our analysis will primarily address the different reed classes (Figs. 17.19 and 17.20).

On average, the temporal variation is similar whatever the polarization, VV or VH; from 2014 November to 2015 January, the radiometry is not really changing because at this period the landscape is not changing very much. In March in early

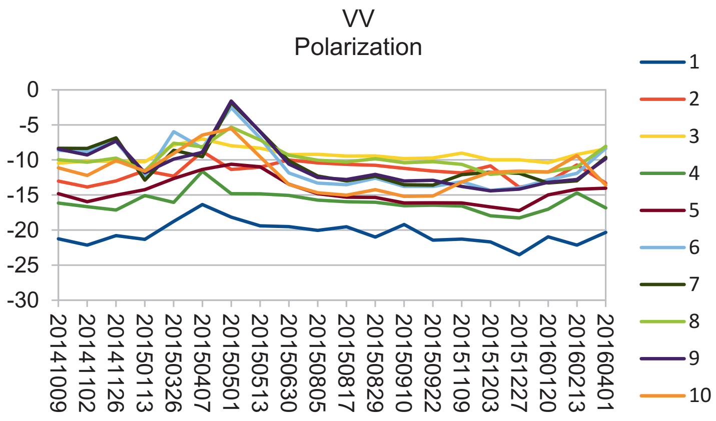

Fig. 17.19 Temporal intensity radar data signature. Polarization VV

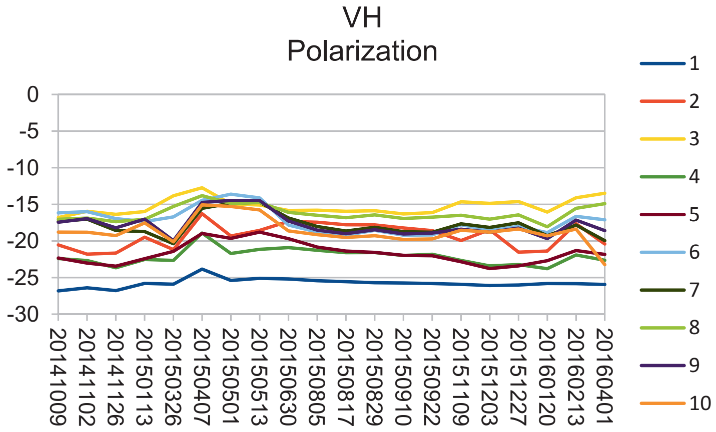

Fig. 17.20 Temporal intensity radar data signature. Polarization VH

spring, the different reed sites are characterized by surface backscattering with poor symmetric backscattering values. This surface backscattering is supported by the low intensity values of the VV polarization channel. The polarization channel values increase between late April and early June, indicating a transition from surface backscattering to dipolar backscattering. Between early June and late September, this dominant dipolar backscattering becomes almost representative of the total backscattering. In April and May, the backscattering values of different reed sites increase significantly due to the combined action of mature biomass and denser and taller vegetation. The decrease in the water level from ≈2 to ≈1 m between July and August–September also accounts for this backscattering decrease. Signal saturation in band C and the difficult substrate access, due to water drainage at most of the sites, led to a decrease in all the intensity parameters. The main backscattering source thus shifts towards the upper part of the canopy, where the large, raised reed leaves enhance rather than reduce signal backscattering. An important observation concerns the temporal evolution of backscattering. We noted that the backscattering peak is reached when consistent backscattering mechanisms are in place (in May/June), correlating with the increased aerial biomass.

The foregoing observations show that there is a transition from surface backscattering in early spring, during which the first plant growth phase occurs (May–June), to dipolar or double-bounce non-dominant backscattering. This mechanism continues to be dominant during the second phase (July–August) up to plant maturity; it then turns into volume backscattering during the senescence phase.

On VH polarization, the species of the various reed classes make up a very homogeneous group, and there are few differences between the various seasonal signatures. For these classes, the backscattered power peak is reached in May and June, when consistent backscattering mechanisms are in place and the aerial biomass reaches its peak height.

17.8 Conclusion

According to research conducted by the United Nations Environment Programme, 40% of the global economy depends on the proper functioning of ecosystems. In most cases, the ecosystem that needs restoring has been degraded, damaged, transformed or completely destroyed as a direct or indirect result of human actions. Ecological restoration should become a priority so as to limit the process of degradation of the environment, to contribute to the preservation of fragile habitats and of critically endangered species and to ensure the valorization of natural resources. Over time, human intervention has manifested itself in more than a quarter of the entire surface of the Danube. This intervention was brutal and has rendered ecosystem restoration very difficult. Over time, the development of fluvial-maritime navigation and of resource use policies applying to the Danube Delta (fish, agricultural, forestry, and other resources) has determined the main water system and landscape transformations in the Danube Delta.

After the fall of the communist regime, ecologic restoration actions were conducted in the delta. This ecologic reconstruction policy concerns all the dammed areas (27.6% of the current surface area of the delta) that had been previously developed for agriculture, fish farming and reed processing. Considering that the Danube Delta includes 30 types of ecosystems that are highly dependent on river level oscillation, the main objective of this ecological recovery is to restore the natural hydrological circuit of the economically developed areas. A solution to these efforts of reconnection to the hydrological regime of the delta was suggested in 1994; it consisted of digging holes in the dykes to allow the water to enter and flow freely in these dammed areas. For the observation and analysis of the restored ecosystems in these areas, we relied on state-of-the-art Sentinel-1 and Sentinel-2 radar and optical satellite imaging and remote sensing methodology. Remote sensing offers accurate automated methods for detecting change in restored wetlands. Vegetation change detection is a powerful indicator of restoration success. The restoration projects use vegetative cover as an important indicator of restoration success. Our research, which relies on several series of radar images captured especially during the growth period, enables us to improve plant formation recognition by exploiting the temporal dynamics of the various plant classes of the restored areas of the delta. Temporal analyses revealed that no single date allows satisfactory characterization of all the vegetation classes. Thus, the temporal dimension, which is represented by seasonal evolution, is an essential component if we intend to draw up a detailed inventory of the restored vegetation classes in the delta.

The synergy of a time series of radar satellite observations with the optical data and radar data can be exploited to improve monitoring and analyze the vegetation in the restoration areas of the Danube Delta. Information from different sensors may assist in the variable retrieval by limiting potential ambiguities. The temporal resolution of the optical sensor Sentinel-2 does not provide temporally frequent products of vegetation characteristics due to the cloud coverage. Application of a multi-temporal radar, multi-sensor approach to a temporal sequence of data acquired by different sensors can improve mapping and monitoring of vegetation state variables over time. By integrating the Sentinel-1 time series with optical images such as those obtained by Sentinel-2, the quality of the habitat maps of the restoring areas in the Danube Delta can be improved considerably (97.1%). Very good Kappa indices were obtained; for the time series radar, the Kappa index was 0.96, and for multi-sensor data integration the Kappa index was 0.97. The reliable Producers Accuracy and K coefficient results prove the complementarity of the two satellites for the observation, analysis and spatial representation of the deltaic plant ecosystems. The Producers Accuracy analysis by class shows that the Sentinel-2 sensor has its limits concerning the detection of similar plant classes, such as, for example, the different classes of reed. Although this sensor detects these classes, the mapping precision is not always high (on some occasions, it is approximately 55% for the 'reed on compact plaur' class). In contrast, the use of a Sentinel-1 time series reveals an interesting C band radar time signature in the Danube Delta ecosystem. Moreover, the combination with Sentinel-2 data resulted in considerable reduction of the observed confusions for both Sentinel-1 and Sentinel-2 with, for instance, a Producers Accuracy value of the 'reed on compact plaur' class of 90.46%, as well as increased accuracy for other reed classes.

As revealed by the data collected by the satellites used in our research, the plant cover of the restored areas appears to be normal and to consist of plant formations similar to those found in the natural areas of the delta. Therefore, we could conclude that plant ecosystem restoration in the Danube Delta has been successful.

Acknowledgment This work was supported by the CNES.

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Chapter 18
Implementation of a Wildlife Management Unit as a Sustainable Support Measure Within the Palo Verde Estuary, Mexico: Example of the American Crocodile (*Crocodylus acutus*)

Omar Cervantes, Aramis Olivos-Ortiz, Refugio Anguiano-Cuevas, Concepción Contreras, and Juan Carlos Chávez-Comparan

Abstract It is recognized that pollution, fragmentation of ecosystems and habitat destruction due to human activities, make necessary the sustainable use of natural resources ensuring a balanced development with legal certainty by alternatives that promote the protection, preservation and proper use of natural resources. Mexico is not the exception, the Palo Verde Estuary is a RAMSAR site that supports various socio-economic activities, and has a tendency towards deterioration. It has been the focus of many environmental studies. This paper describes the implementation of a Wildlife Management Unit for sustainable commercial harvesting of crocodile *Crocodylus acutus* as an alternative project in the site known as *Ecological Center Tortugario of Cuyutlan*. Thus, providing a standard for integrating an economic aspect to biodiversity and ecosystem protection in the decision making process. Finally, an analysis is made from the perspective of conceptual framework Driving Forces-Pressure-State-Impact-Response where the institutional context and

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participative management can be the main support of sustainable development on this site. The expansion of the a Wildlife Management Unit is an alternative project for the sustainable use of *C. acutus* present in the Palo Verde Estuary that can be replicated in other Mexican coastal systems. Additionally, in itself it represents an opportunity to reconcile human activities with the environment.

Keywords Environmental policy • Regulatory instruments • Sustainable management • Estuary

18.1 Introduction

The coast zone (CZ) is a set of complex ecosystems highly productive, biodiverse and vulnerable, conformed by a mosaic of micro ecosystems in continuous interaction that form the great system along time and space (Azuz-Adeath 2008). The coastal water bodies constitute a part of that mosaic of ecosystems, which are recognize by the ecosystem services that they provide (support, regulation, supply and cultural services) (MEA 2005), and for their fragility and sensitivity to the high anthropogenic pressures (Coelho et al. 2007; Alves et al. 2013; Jennerjahn and Mitchell 2013) that threat their integral functionality with ecological, economics and social costs (Bricker et al. 2003; Cotovicz-Junior et al. 2013).

The decision making related to natural resources require functional information to learn the actual situation of a particular ecosystem of interest, which not always is neither transmitted nor correctly understood. This is a problem that the potential users of the resources have to deal with (Lundberg 2013), and it is sharpened due to disengage between public administration periods (Azuz-Adeath and Rivera-Arriaga 2004; Santos-Martín et al. 2015).

In this sense, environmental evaluation and planning to achieve conservation constitute objects of interest and increases the necessity of the integral coastal management supported by legal and planning instruments with the purpose of protecting, mitigating and controlling the accelerated expansion in the CZ (Milanés-Batista 2012; Costa et al. 2013). Nevertheless, to link all these factors in a consensual manner (institutional, social, economic and ecological) is not an easy task (Espinoza-Tenorio et al. 2014).

Regarding Mexican case, among environmental policy instruments that are used for managing and planning of the coastal-marine zone are the ecological zoning, the protected areas (PA) and the Wildlife Management Units (WMU or UMA Spanish acronym) among other, which emanate from constitutional and legal provisions in the General Law of Ecological Equilibrium and Environmental Protection (LGEEPA, for its acronym in Spanish) (Zárate-Lomelí 2004; CONABIO 2009). Its rationale stems from the concern of different social sectors to the environmental crisis being experienced by mankind, and it has led to the search for new alternatives that promote the protection, preservation and proper use of natural resources, especially in regions under strong environmental, social and economic pressures, as pointed by Bocco et al. (2000).

Moreover, it is recognized that economic development (e.g. productive projects as alternative strategies) can provide protection for the environment and replace human activities with the potential detriment of other more sustainable conferring socio-ecological and cultural benefits (OECD 2001; Yáñez-Arancibia et al. 2013, 2014). In this sense, the WMU is recognized as one of the privileged instruments to diversify rural productivity and generating income which allows avoiding compromising the resilience of natural ecosystems (CONABIO 2009).

The experiences concerning cases of crocodile WMUs operating in Mexico, suggest that they are achieving the production target (Serna-Lagunes et al. 2013). This almost certainly with adherence to the guidelines established in the "Management Plan Type" identified by the SEMARNAT (2014), with which the viability of the harvesting of crocodiles such as *C. acutus* is demonstrated. Near the PVE there is a WMU operated by a non-governmental organization (NGO) through "The Tortugario" Cuyutlán Ecological Center (ECTC), which among its functions include the PVE protection. From this fact arises the interest of proposing the installation of a full cycle extractive WMU, attached to the same area as an alternative project by extending the terms of operation of the existing WMU. This paper proposes by this legal concept that the commercial harvesting of crocodile species *C. acutus* can be achieved as a key factor for the declaration as Protected Area (PA) of the PVE, which is a necessary and complementary tool for the rational use and conservation of territorial biodiversity (Pereira-Corona et al. 2015) as a feasible project as a response to the sustainable use of the PVE, consistent with the applicable legal framework and from the perspective of the Driving Force-Pressure-State-Impact response model (DPSIR) (OECD 2001), which is analyzed in this paper.

18.1.1 PVE Features

The PVE (Figs. 18.1 and 18.2) is located in the coastal area of Colima state, Mexico, between the coordinates 18°915′ and 18°89′N; 104°045′ and 104°015′W. It has an area of 734,968.75 m^2. It is part of the Cuyutlan Lagoon (LC) named Basin IV, which has been fragmented and partially isolated by infrastructure (roads, railways and hydraulic pathways) that negatively affect its natural hydrological flow in the northeastern part of the PVE drastically reducing water circulation and increasing the water residence time in the system. In the southwestern part it has sporadic communication with the Pacific Ocean through a natural opening of a sandbar as a result of the increment of pressure due to the large volume of water coming from the basin in presence of extreme weather events that burst in the region (Anguiano-Cuevas et al. 2015).

The PVE functions as part of the biological corridor of resident and migratory birds of the Mexican Pacific region (Mellink and Riojas-López 2009). It protects flora and fauna under some protection status in the Mexican Environmental

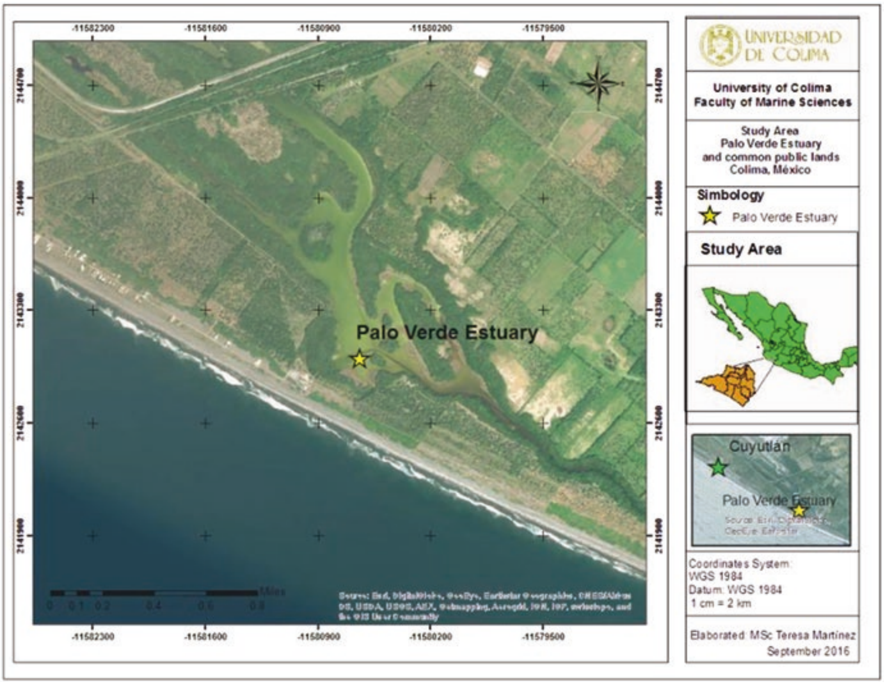

Fig. 18.1 *Palo Verde* Estuary (PVE) and common public lands (named *ejidos*) that sets the adjacent agricultural zone

Standards (NOM-059-SEMARNAT-2010) and it has been declared a RAMSAR site (No. 1985). It is important to note the presence of crocodile's species as *C. acutus* species belonging to the family Crocodylidae (M. Rivera-Rodriguez pers. comm.). Currently the PVE supports artisanal fisheries and ecotourism activities, which also gives an important socio-economic regional value for the goods and services offered to the population.

18.1.2 Socio-environmental Issues

Despite its socio-economic importance at regional level, Cuyutlán Lagoon which is part of the PVE, is under strong anthropogenic pressure resulting from the different uses and infrastructure that surrounds it. Specifically for the PVE, there are plenty agricultural downloads from the adjacent areas through their drains, mainly in the rainy season (Trani-García et al. 2013). Added to this, the PVE has records of changes in the trophic state, derived from agricultural activities in adjacent areas and continental drag of agrochemicals from this area causing changes in the trophic status, as described in Anguiano-Cuevas et al. (2015).

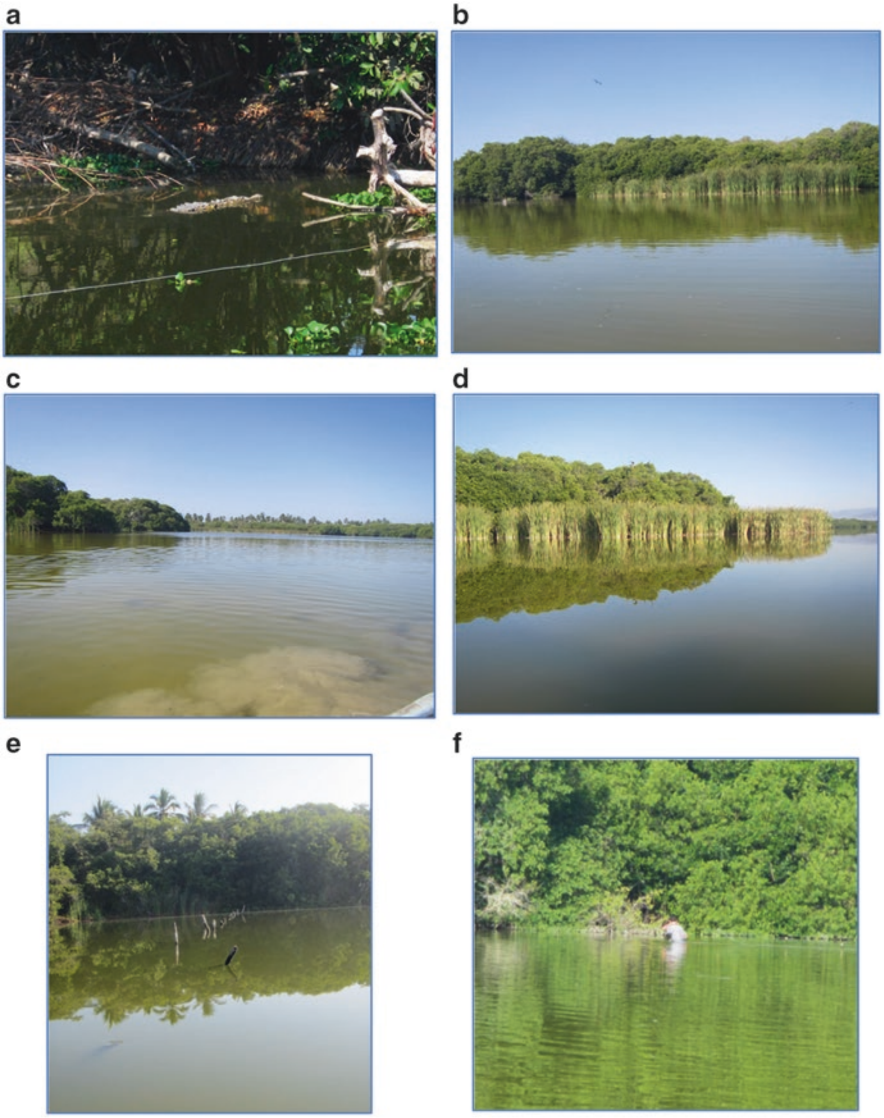

Fig. 18.2 *Palo Verde* Estuary (PVE) surroundings; mangroves, water, crocodiles (*C. acutus*), artisanal fishings and lands flooding

These damages are identified as a result of a series of failed agrochemicals subsidy programs derived from a lack of technical and environmental assistance for their implementation (Magaña-Magaña and Leyva-Morales 2008; Pérez-Espejo et al. 2012; Anguiano-Cuevas et al. 2015). In addition to the aforementioned problems, there are documentations of recent developments of human-crocodile interactions in various local media (Ochoa-Anguiano and González-Rincón 2013; Notimex

2014; Tadeo 2014). It is possible that these events are due to the invasion of this species habitat and their ability to survive in man-modified areas or due to the resource competition between them and fishermen; which has led to direct hunting of the species or indirect (bycatch) which in some cases has decimated their populations as reported in several lagoon systems (McGregor 2005; Hernandez-Hurtado et al. 2006; Balaguera-Reina and González-Maya 2010; Dunham et al. 2010). This can be considered as an additional threat to that generated by modifying the physicochemical conditions of water (UNCTAD 2014) for this species, despite their ability of adaptation to altered sites. Despite this alteration scheme, human activities related to the resources can be grouped into two opposite but complementary categories, *the use of a protected resource with production and extraction activities*. In Mexico, the first are intended to be regulated by ecological systems and the later with the protection area (PA) (Pereira-Corona et al. 2015). In this sense, it can be said that the institutional context and co-management are the pillars on which sustainable management is held.

18.2 Environmental Regulations

The PVE is linked to two environmental policy instruments (EPA): (1) Land Management Program for Colima State (POETEC), and (2) the Regional Land Management Program of the sub-basin of Laguna Cuyutlán (PROETSLC). From this, other provisions are established such as those found in Article 20 Bis 2 of the LGEEPA, and in various laws of the State of Colima. The common aim of these EPA is to assess and schedule, from an environmental perspective, the land use, the use of natural resources, the productive activities and their relationship with the urban and rural development, in order to reconcile all these components (environmental, urban, rural and economic), and so they become the baseline for the development of programs and projects that are intended to be executed, from their analysis of deterioration and potential uses (POEC 2007; Official Gazette of the Federation DOF 2015a). It is very important to note that, in order to achieve a greater protection for the PVE area, it would be appropriate that in the POETEC of the local territory was issued, in which specific regulations appear to allow greater protection to this area.

In the POETEC the PVE is considered under Protection Policy, while the adjacent agricultural area is within the Land Management Area (LMA) No. 88 with a Sustainable Use Policy (Table 18.1) (POEC 2012, 2013). In the PROETSLC, the PVE corresponds to the LMA No. 65 under Protection Policy, while the agricultural area forms a part of the LMA No. 61 under the Agricultural Use Policy (Table 18.1). Both LMA's are adjacent to each other with a series of ecological criteria set out in different zonings (POEC 2007, 2012, 2014). In the PROETSLC it is emphasized on the overall conservation of Basins III and IV of Cuyutlan Lagoon, by declaring the corresponding protected natural areas which have not yet been executed.

Table 18.1 Land Management Area (LMA) number and environmental policy where the PVE and common lands (named *ejidos*) are located according to PROETSLC (POEC 2007) and POETEC (POEC 2012)

	POETEC		PROETSLC	
	LMA no.	Environmental policy	LMA no.	Environmental policy
PVE	33	Protection	65	Protection
Common lands (ejidos)	88	Sustainable use	61	Sustainable use

Among the various ecological criteria defined in the Agricultural Use Policy established in PROETSLC, actions to prevent chemical discharges into the PVE are expected which, according to the environmental problems described above, it is evident that these actions have not been executed yet and this situation may worsen in the future due to economic and population growth, and also because of the activities planned for the coastal lagoon.

This reflects the need of implementing the third generation strategic environmental assessment consistent with the use and integration of these instruments to give greater certainty to environmental sustainability as quoted in Ahumada-Cervantes et al. (2012).

18.2.1 Policy Instruments That Support Sustainable Use of C. acutus

At a national status, the *C. acutus* is under Mexican Environmental Standard NOM-059-SEMARNAT-2010 (NOM-059). That is, it is categorized under those that could potentially be threatened by factors that adversely affect their viability, so the need to foster recovery and conservation, or recovery and conservation of associated species populations (SEMARNAT 2014; Reyes Herrera et al. 2015). However, the same NOM-059 in the criterion 6.3.6 states that the exclusion of a species (e.g., *C. acutus*) from the list is feasible when that category is subject to special protection, and it determines that protective measures have been and continue to be appropriate and sufficient to stop the pressures that were forced on such species, and therefore its viability can be ensured (DOF 2010).

About this fact, in early 2014 the SEMARNAT through its website issued a notice of updating the NOM-059-SEMARNAT-2010 to submit proposals for inclusion, exclusion or change of status of the species list risk of Annex III of the norm itself. This means that if the study of populations of *C. acutus* in the PVE demonstrates the feasibility of their harvesting, this would be a first step in the national context to support the exclusion of *C. acutus* from the special protection category where it is currently registered in the norm.

18.2.2 Institutional Supporting Wildlife Management Units

From the provisions of Mexican law, the WMU are production or display units in a clearly defined area under any ownership (private, common lands (ejidos), communal, federal, etc.), for the use of specimens, products and sub-products of wildlife resources and require specific management for its operation in order to contribute to biodiversity conservation with the requirements of production and economic development of Mexico (CONABIO 2009; DOF 2015b). Basically the establishment of the WMU is based on the General Law on Wildlife (LGVS) in its Title VII, of the Sustainable Use of Wildlife, Chapter I concerning the Extractive Use (DOF 2015b). Under this legal framework, the use of specimens, parts and sub-products of wildlife are allowed through the WMU, because they work as spaces that promote alternative schemes of production compatible with environmental care through the rational, ordered and planned of the renewable natural resources that are contained, and which in turn slow down or reverse the processes of environmental degradation. Thus, the WMU is subjected to compliance of standards established in the LGVS, their Regulations and the own management program established for it, whose implementation is within the competence of the SEMARNAT through the General Direction of Wildlife (SEMARNAT 2014).

18.2.3 Co-management/Participatory Management

Coastal zone communities differ from other non-coastal human settlements, since the diversification of economic activities, the magnitude, diversity and importance of natural resources associated with the coastal zone help to shape these communities that have diverse conditions (natural, cultural and socio-economic), which in turn have a complex and a dynamic nature.

The content of this work is framed from the perspective of co-management or participatory management, based on identified key factors having relationship with PVE as a basis for management, harvesting and preservation of *C. acutus*. The key factors identified in the area of interest are the government sector, the Ecological Center Tortugario of Cuyutlan (ECTC) and the concerned community (farmers, fishermen) which would play a crucial role in the functioning of the ECTC, operated in a participatory manner by the interested parties as partners to achieve that its use yields a higher socioeconomic benefit with technical and scientific support from the ECTC. Thus, proper management will depend on the sector's contribution interested in these productive activities.

Under this projection, the role of the ECTC would be essential due to the fact that they occupy a position at a primary level in the operation and functioning of the WMU: (1) from here, it would serve as an advisor or directive of the training on technical and scientific management concerning the extractive crocodile WMU, the concerned community, and (2) serve as a commercial manager of products and

Table 18.2 Conditions for sustainable and extractive use provided in the Article 84 of LGVS (2015b)

(a) That the requested rates are lower than the natural renewal of stocks subject to harvesting, in the case of specimens of wild species
(b) That they are the product of controlled reproduction, in the case of specimens of wildlife in confinement
(c) That this will not have negative effects on the population and it will not modify the life cycle of the specimen in the case of use of parts of the specimens
(d) That this will not have negative effects on populations or manipulation that permanently damages the organism in the case of derivative products of the specimen

sub-products obtained from *C. acutus*. Thus it is possible that the ECTC would have a better monitoring and greater control over the management of *C. acutus* for its integral and sustainable use based on the specifications of Management Plans and Planning (TMP) recently published by SEMARNAT (2014). This without forgetting the coordination with the concerned community group (as proposed by Bocco et al. 2000) so the work does not have a bias towards a purely academic perspective, and that it exists interest in developing alternative projects, and that it will lead them to financial results in the short term if possible.

Thus, from this perspective, the ECTC can process at SEMARNAT the corresponding collection of permits for the extraction of breeding specimens from the same habitat, with previous studies of populations established in the LGVS (Table 18.2), which would facilitate the adaptation of *C. acutus* for its intensive management as it would belong to the same environment where the reintroduction, repopulation and/or controlled reproduction could later occur. These activities would be within the functions which the ECTC would support in the extractive WMU.

The ECTC would relate directly with nearby communities conducting fishing activities and/or agriculture, and so involve and make them participatory of the extractive management, providing technical training or direct them to the corresponding authorities for advice regarding the mechanisms for the marketing of manufactured products and sub-products (obtainment of sale and export permits). Relations between ECTC functions and specifications to be met in the natural resource management by the interested parties and the WMU are shown in Fig. 18.3.

About population studies of *C. acutus*, there are some done in the Basin III of the LC, whose results show an increase in sights of crocodiles from 3.9 org/km^2 in 2010 to 5.5 org/km^2 at 2012 (Gaviño-Rodríguez et al. 2011; Reyes Herrera et al. 2015). This quantity of organisms represents a density ranging from low to medium density according to Aguilar-Olguin (pers. Comm.), based on the density categories established by Thorbjarnarson 1989 and Buitrago 2003. This author has registered average relative densities of 5.31 org/km specifically in the PVE corresponding to the category of medium density. However, it assumes that it is likely that the population of *C. acutus* is not increasing but it only sustains itself and that the human impacts may negatively be manifested in the size and distribution structure as a result of their habitat reduction. This could possibly be further related to recent

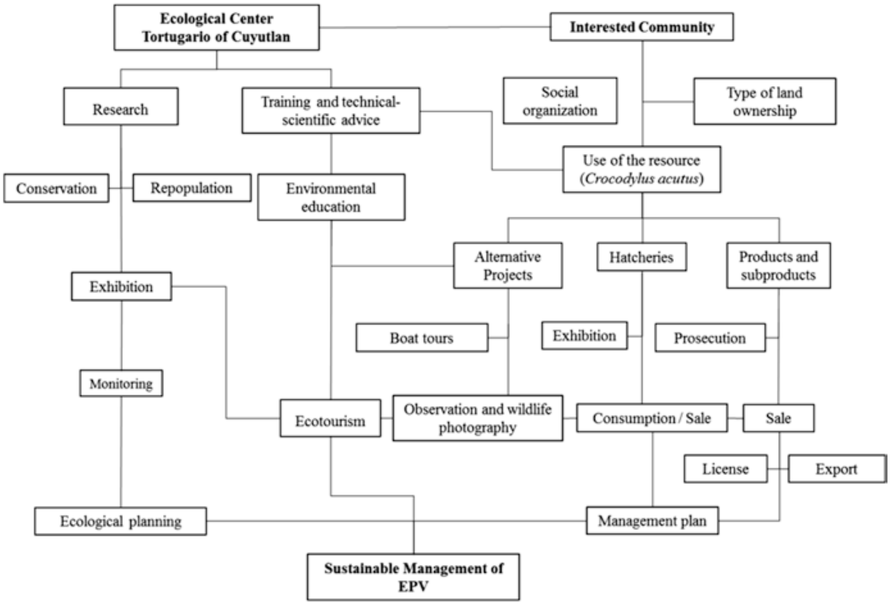

Fig. 18.3 Role of the ECTC and specifications to be met by the interested parties and the WMU in natural resource management (*C. acutus*) of the PVE (Modified by Bocco et al. 2000)

human-crocodile incidents reported in the diverse aforementioned media (Notimex 2014).

Both events show the need for action involving interested parties related to the PVE, as allies for species protection through their sustainable use: (1) justify the preservation of natural resources, providing it gives employment to the community, and (2) ensure the conservation of the species with repopulation support, on the assumption that it would be naturally inhibited by environmental and/or anthropogenic factors as mentioned by Gaviño-Rodríguez et al. (2011) and Reyes Herrera et al. (2015).

An additional advantage that opens up possibilities for marketing *C. acutus* products and sub-products in the WMU is the fact that the crocodile farms are an attraction for eco-tourism, which would generate additional revenue for the ECTC coupled with that generated by boat rides and guided tours (SEMARNAP 1999, 2014a) favored by the PVE location: 5 min from the town of Cuyutlán with land communication via Highway, just 40 min from the city and port of Manzanillo and Colima capital state, where a considerable number of tourists arrive via cruise ship (Table 18.3), which in average represents more than 30,000 potential tourists in a year, according to data from INEGI (2013) for the period 2000–2012, excluding national tourist from nearby cities as Tecomán, Colima, Guadalajara and Morelia, mostly in holiday periods; without forgetting that all visitors would take lectures on environmental education of resources protected in the WMU, reinforcing the services provided by the ECTC.

Table 18.3 Number of passengers and cruise arrivals to the port of Manzanillo in the period 2000–2012

	2000		2011		2012	
Port	Passengers	Arrivals	Passengers	Arrivals	Passengers	Arrivals
Manzanillo, Col.	10,351	10	50,551	35	23,583	14

INEGI (2013)

This opens the possibility that the market ceases to be predominantly national, which could contribute to a desirable effect, previous agreements with travel agencies in the tourism industry offering packages to the PVE, including ecotourism traveling and products marketing, leaving a part of the economic flow generated and economic benefits to the involved sector, including the local population.

A similar case about the start of an intensive WMU can be found in the Southeast of México in Chiapas State in the WMU *Cocodrilario el Boquerón* in a location known as "La Lagunita" disposed as Community sewer for 30 years (CBCH 2008). This figure of WMU was finalized after intensive meetings with different departments (CONANP, CONAFOR, SAGARPA, SEPESCA, CONAPESCA, PROFEPA, SEMARNAT) since 2004, starting in 2007 with SEMARNAT formalities, obtaining the same year the license for the establishment of the WMU *Cocodrilario el Boquerón* with an intense modality with code SEMARNAT-UMA-IN-0047-CHIS/07. The WMU members consider that it is actually a project which has a multidirectional potential and pretends to be a unique prototype in the State as a key factor for the integration of social participation, productive development, ecotourism and conservation of the river crocodile with a profit with local, state, national and international areas of influence (CBCH 2008).

Some additional examples of the sustainable use of crocodiles are the national event organized by the Secretariat of Agriculture, Livestock, Rural Development, Fisheries and Food (SAGARPA 2012), where intensive crocodile farm "Wotoch Aayin" in the community of Isla Arena in Calkiní, Campeche, was one of the chosen projects with greater economic, social and environmental impact at the National Meeting of Projects Exchange of Successful Experiences. Another example is reported by the company Business Bloomberg (2013), in a note from the Mexican newspaper "El Financiero" in its digital version, where the average price of a bag oscillates around $800 USD (http://us.louisvuitton.com/eng-us/women/rare-and-exceptional), and it is believed that the demand for crocodile skins is large and is one of the trends that came here to stay.

Meanwhile, in the Panamanian newspaper "La Prensa", Berrocal (2008), discloses Mexico as part of the Kubota SA company crocodile market, whose owner says that the demand of crocodile skin boost the exportation, which is estimated around 24 thousand units a year, each valued between 24 and 32 dollars. This under the authorization of the National Environmental Authority (ANAM) and the support of CITES. These companies accept that like any other activity, sustainable use of crocodiles is not an easy task and much less dangerous; however, they acknowledge that brings economic benefits and new sources of local employment, as set in the

PVE. This coincides with the statement of Retes-López et al. (2010), who stated textually that "*international trade of wild flora and fauna species produces billions of dollars annually and consists of hundreds of thousands of species of living organisms, parts, products and their derivatives*".

18.3 Discussions

18.3.1 Establishing the Extractive WMU

The establishing the WMU could help ensure its preservation in the PVE, reducing human-crocodile interactions, where the worst is bear by crocodiles, as even among the villagers the illegal hunting is recognized. In this regard it should be remembered that according to the SEMARNAP (1999), it is shown that conservation itself does not guarantee the permanence of species, that to achieve this conservation should be linked to measures designed to meet the requirements of social welfare and only sustainable development is capable of breaking the vicious cycle of poverty that contributes to ecological deterioration.

Moreover, the WMU are considered as the main pillar of sustainable development since their establishment and development is part for the conservation, management and sustainable use of these wildlife species. There are considered, among the actions of the Conservation and Sustainable Use strategies, the proposals for protected natural areas that are essential to the conservation of these organisms (among others) supported with environmental education and aimed to different society sectors (SEMARNAP 1999).

Thus, The installation of the extractive WMU lay the foundation and would help in the declaration of Natural Protected Area (PNA) to the PVE with its type and characteristics of a flora and fauna protected Area (Annex IV), according to Article 54 established in LGEEPA 2015a), Section II, based on Article 47 and 47 BIS, fraction II "buffer zone" subsections c) "sustainable use of natural resources" and/or e) "of special use" due to the fact that this type of flora and fauna protected areas may allow the exploitation of natural resources by resident communities.

However, it must be ensured that establishing a PNA would promote the community development and not to become an obstacle to achieve this purpose, due to the roots of people to their place of origin, as they are difficult to be persuaded as well mention by Riemann et al. (2011).

18.3.2 Analysis from the Perspective of Driving Forces-Pressures-States–Impacts and Responses Model

Analyzing the set of interrelated factors in the PVE from the perspective of the Driving Forces-Pressure-State-Impact-Response (DPSIR) model, it can be said that the environmental problems described in the PVE by Anguiano-Cuevas et al. (2015) have been the result of agri-environmental policies (e.g. subsidies for the purchase of chemicals for agriculture, lack of training for farmers, lack of support in the programs) poorly designed that promote activities that put pressure on the PVE (e.g. agriculture, use of agrochemicals, continental contribution of pollutants) and alter their state (PVE trophic conditions) causing impacts on the ecosystem services (e.g. blooms of toxic phytoplankton, mortality of fish and birds, anoxia in the ecosystem, succession of species, changes in the overall functionality of the PVE, etc.).

This contrasts with the purpose of environmental policies established by the LMA for the study area, where in its designation of uses between LMAs (No. 61 and 65), the full functionality of the ecosystem was not considered in relation to the influence of the micro basin where these LMAs are located. It can be perceived that in reality there is an incompatibility of uses between both LMAs, given their proximity, due to the fact that in a natural manner the effects derived from activities in the upper part of the basin are manifested in the lower part and these effects are contrary to the objective of the Protection Policy established in the POETEC (POEC 2012). This has been previously described by SEMARNAT (2006), who points out that altering the basic environmental structure and functions in one of the ecosystem that forms the environmental system due to a project or a poorly design activity, can produce environmental impacts on the rest of the coastal ecosystems with which ecologically is linked, affecting the whole environmental system and the economic activities that take place inside.

From the perspective of this referred scenario, and given that the PVE is not an isolated system, but it is in continue interaction with other ecosystems, it is desirable that the protection policies established in the PROETSLC to be materialized, e.g. the statement of ANP (Ortiz-Nielsen 2014), which is essential to provide legal certainty in the protection of ecosystems. As noted above, economic development can provide protection for the environment and replace human activities that could threat the environment with other more sustainable, that confer benefits both the social and the coastal system (PVE) (OECD 2001; Yáñez-Arancibia et al. 2013, 2014).

That is, if the current trend continues, it could lead to the deterioration of the PVE making the restoration or rehabilitation unaffordable and irreversible. In this sense, the response would be the participative management to integrate the economic aspect (WMU) to the ecosystem and biodiversity protection in the decision-making process. In this order of ideas Fig. 18.4 graphically represent the information analyzed from the DPSIR perspective.

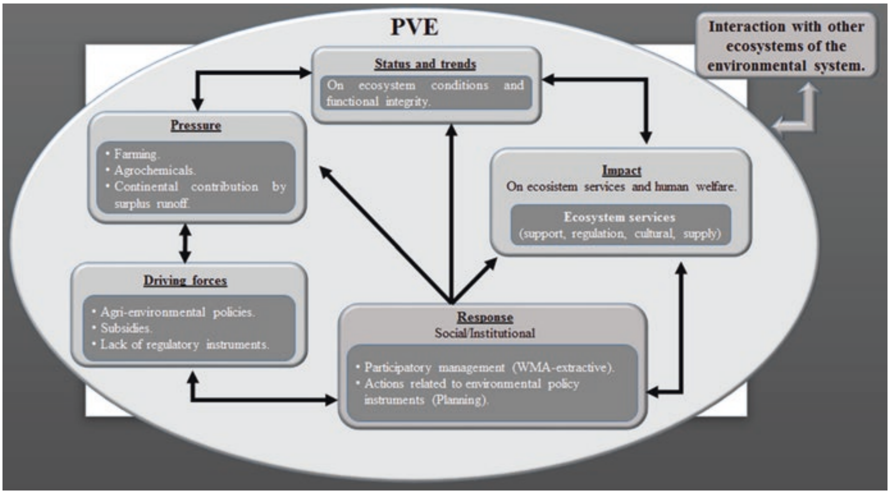

Fig. 18.4 Graphical representation that sets the environment-society relations from the DPSIR approach that causes the predominant problem in the PVE (Modified from Santos-Martín et al. (2015)

In this sense, the implementation of actions for sustainable development in the PVE is possible and, through its implementation, they are a guarantee for the preservation and long-term preservation of its functional integrity and therefore the habitat of many species from aquatic to terrestrial, as it has been recognized (CIMARES 2012).

These factors would give legal certainty to the approach presented in this paper, from the representation of the cyclical nature of the Integrated Coastal Zone Management for identification within the theory of the time cycle of public policy, plus the asynchrony between government periods that complicates decision-making and hinder the conciliation of viable long term projects, as mentioned by several authors (Azuz-Adeath and Rivera-Arriaga 2004; Santos-Martín et al. 2015). The analysis from this perspective can be the general methodological guide to understand the ecological, socioeconomic, cultural and political events taking place as noted by Caviedes et al. (2014), giving the possibility that the success on the conservation of natural resources becomes more efficiently. This, contrary to what described by Santos-Martín et al. (2015), which define it as a so far limited success despite the efforts in the management processes and points out as one of the possible causes the disconnection between management scale and the scales where the ecological processes occur, plus the low specificity and incompatibility of the social, economic and environmental objectives.

18.4 Conclusions

The WMU is a policy instrument that needs to be continuously strengthened and adapted to social changes to show that the provisions of the LGEEPA and its environmental policy instruments have achieved their goal as promoters and support for the conservation and community development in the socio-environmental area in a win-win scheme based on the functioning of ecosystems (ecosystem-based management).

The expansion of the WMU is an area of opportunity to be implemented as an alternative project for the sustainable use of *C. acutus* present in the PVE and it can be used as a strategy for developing rural tourism in the PVE, and be replicated in other Mexican coastal systems. Additionally, in itself it represents an opportunity to reconcile human activities with the environment, as it ensures sustainability conditions in the short, medium and long term and is the guarantor of the environment preservation.

The legal status of the susceptible species to be commercially exploited must be modified and applied, since the current economic situation and employment demands it and, it has been recognized and integrated into the different schemes of public policy but timidly executed. Only with proactive and reliable implementation of these policies, it will be adopted consequently as an option for community development. Other productive projects (observation and wildlife photography) as sustainable development alternatives could be explored in the short and long term, with a preliminary analysis of the inherent factors to ensure the functional integrity of the ecosystem.

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Chapter 19
Mangrove Inventory, Monitoring, and Health Assessment

Ajai and H.B. Chauhan

Abstract Mangroves occur world-wide in the tropics and subtropics, mainly between 32°N and 38°S latitudes. These are assemblages of salt tolerant shrubs and trees that grow along the inter-tidal regions in the form of narrow strips or as extensive patches in estuarine habitats and river deltas. Mangroves or Mangal perform many important functions in the coastal and marine environment as well as provide vital and unique ecosystem goods and services including livelihood of coastal people in terms of forest produce and fishery resources. In spite of its ecological and economic significance, mangroves are under threat from human activities in addition to natural causes. The anthropogenic threats to this ecosystem include reclamation of mangrove areas for human habitation, aquaculture, agriculture, port and industrial development. Conservation and management of the mangrove ecosystem requires spatial inventory of mangrove cover, its type and canopy density as well as its monitoring with time. The physical conditions and harsh environment in and around the mangrove forest make the conventional methods of data collection extremely difficult. Under such conditions, data from remote sensing satellites provide a viable option for mapping and monitoring of mangroves.

Mangroves, their importance and threats to this vital ecosystem are discussed here. Remote sensing techniques and its application in the context of mangroves have been briefly discussed. Use of satellite data for mapping and monitoring of mangrove forests has also been described in this chapter. Methodology for spatial inventory of mangroves at community level along with canopy density is also presented in detail. Case studies on the inventory and monitoring of mangroves have also been discussed. A model for mangrove health assessment, based on the satellite data and conventional inputs, have been developed and presented here. This multiparameter health assessment model has been discussed and demonstrated through a case study from India.

Keywords Mangroves • Inventory and monitoring • Remote sensing • Earth observations • Mangrove health model • Coastal ecosystems

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19.1 Introduction

Mangroves are assemblages of salt tolerant shrubs and trees (also called hydrophytes) that grow along the inter-tidal regions in the form of narrow strips or as extensive patches in estuarine habitats and river deltas of tropical and sub-tropical coastal areas. The term "mangrove" originated from the Portuguese and English words 'Mangue' and 'Grove'. These two words mean 'mangrove plants' and 'group of trees' respectively. The term 'mangrove' is, thus representative of a single plant as well as a plant community. MacNae (1968) proposed 'mangal' as a term for the community, leaving "mangrove" for the constituent plant species.

The term 'mangrove' describes both the ecosystem and the plant families that have developed specialized adaptations to live in the tidal environment. Majority of plants are evergreen trees, although deciduous trees, perennial and evergreen shrubs, parasites and climbers, perennial grasses and perennial ferns are also common constituents of this ecosystem together with algae and fungi (Tomlinson 1986).

Recently, Mukherjee et al. (2014) have arrived at the following definition of mangrove through consensus: Mangroves are woody plants that grow normally in the tropical and sub-tropical latitudes along the land- sea interface, bays, estuaries, lagoons and back waters. These plants and their associated organisms constitute the "mangrove forest" or "mangal". The mangal and its associated abiotic factors constitute the "mangrove ecosystem".

Mangroves occur world-wide in the tropics and subtropics, mainly between 32°N and 38°S latitudes. Sheltered environment with brackish water influx, estuarine and deltaic muddy soil, good rainfall (1000–3000 mm) and temperature between 26 and 35° C are considered as ideal habitat for luxuriant growth of mangroves.

In comparison to terrestrial forests, the structure of the mangrove forests appears to be relatively simple as the forest canopy cover, usually comprises of the main branches and leaves of the trees, and there are few smaller plants on the floor of the forest (Kathiresan and Bingham 2001). The vines and shrubs common to the most of the tropical terrestrial forests are missing in case of mangrove forests, though there is often a carpet of mangrove seedlings present. The missing understory, in case of mangrove forests, is basically due to the prevailing harsh conditions and the lack of presence of light.

To survive in harsh coastal conditions, mangrove plants exhibit a wide variety of adaptations in terms of morphology, physiology, anatomy, physiognomy, development of seed and seedlings, sclerophyllous leaves with sunken stomata, and vivipary are some of the common adaptations. The roots possess pneumatopores or lenticels for the purpose of gaseous exchange. Exclusion of salt through rhizo-filtration, excretion of salts through salt glands, and the ability to deposit salt in older leaves, barks and pneumatophores help the plants to acquire high level of salt tolerance (Tomlinson 1986). The control of tissue water potential is done through specialized leaves and stems.

Mangrove communities, with their unique adoptive features form different zonations, according to physiography, topography, tidal inundations, salinity regimes etc. Each species of mangroves has its own solutions to cope with the problems of anoxia, high salinity and frequent tidal inundation. This may also be the reason why on some coastlines mangrove tree species show distinct zonations. Small variations within a mangal may lead to highly varying methods for coping with the environment. Therefore, the mix of species is partly determined by the tolerance of the individual species to the prevailing physical conditions, such as salinity, tidal inundation, but may also be influenced by other factors such as predation of plant seedlings by crabs.

19.1.1 Importance of Mangroves

Mangroves perform many important functions in the context of the coastal and marine environment as well as provide vital and unique ecosystem goods and services to the human society. It helps in reduction of coastal erosion, stabilizing the shore-lines and protecting the coast by acting as wind breakers and barriers against storm surges and heavy tides (Bahuguna et al. 2008; Giri et al. 2007; Kathiresan and Rajendran 2006; Dahdouh-Guebas et al. 2005). Its massive root systems are efficient at dissipating wave energy. Mangrove forest play important role in transferring organic matter and energy from the land to marine ecosystem through the detritus from the fallen leaves and branches. Bacteria breaks down the detritus, releasing useful nutrients in to water and thus forms the base of the highly productive marine food chain. The dense root system forms a home for fish, crabs, shrimps and mollusc. They also serve as nursery grounds for the larvae and juveniles of marine inhabitants. In addition, mangrove forests are nesting and migratory sites for a large number of bird's species as well as home to a variety of reptile, amphibian and mammal species. For example, the Sunderbans mangroves are the home to Bengal tigers, spotted deer, saltwater crocodiles, fishing cats and several dolphin species.

Mangroves also contribute to the livelihood of coastal people in terms of forest produce and fishery resources. A balance exists between the complex biological system of mangrove forests and the local people who exploit the system without destroying it. In addition to serving as a habitat to support the faunal biodiversity in the coastal areas, mangrove wetlands also serve as potential recreation site for fishing, boating, bird watching, sightseeing and photography. Mangroves are self-generating, self-perpetuating and highly resilient littoral formations, playing a major role in the global cycle of carbon, nitrogen and sulphur. They act as sink for detritus and sediments draining from coastal catchments and help in territory assimilation of waste.

The exact number of mangrove species is still a matter of debate and ranges from 50 to 70 according to different classification systems (Spalding et al. 2010; FAO 2007; Aksornkoae et al. 1993; Tomlinson 1986; Saenger et al. 1983; Lugo and Snedaker 1974), with the highest species diversity found in Asia, followed by eastern Africa. Mangrove's three genera, namely, Acrostichum, Avicennia and Rhizophora have the global presence (Spalding et al. 2010). More than 50 mangrove species grow along the coast of Asia and that is the highest among all the continents. Oceania region also supports more than 50 species of mangroves. Australia has about 30 species. Africa has about 14 species of mangroves. South America as well as North and Central America have only ten mangrove species (FAO 2007).

19.1.2 Threats to Mangroves

Despite the ecological and economic significance of mangroves they are considerably under pressure from human activities. Its area has declined by 30–50% in the past 50 years, a higher rate than most other biomes (Balmford et al. 2002). It has been estimated that the loss of world's mangrove forest may be as high as 60% by 2030 (UNEP-WCMC 2006; Alonge 2002; Valiela et al. 2001). The major threats to mangroves are:

Plantations and young plants are being damaged by the attachment of barnacles to their stem. Insect attack also causes considerable damage to the leaves of various species of mangroves like *Rhizophora mucronata, Avicennia alba* and *Sonneratia alba.* Other species have also been found to be attacked by fungus.

Reduction in the quantity and periodicity of fresh water flowing into mangrove wetlands affects the density as well as diversity of the mangroves through increasing water and soil salinity. This problem, mainly, affects the highly saline sensitive species like *Sonneratia apetala*, *Xylocarpus granatum, Heritiera fomes* and *Nypa fruticans.*

Though extensive areas of mangrove forest occur on sedimentary shoreline, where large rivers discharge on to the low gradient coast, but excess input of sediments to mangroves can cause its degradation and death.

The other major threats to this ecosystem are reclamation of mangrove areas for human habitation, aquaculture, agriculture, port and industrial and other developmental. One of the major human induced threats to mangrove ecosystem is its reclamation for shrimp aquaculture ponds. It accounts for the loss of about 20–50% mangrove areas globally (Primavera 1997). Unauthorized felling and cutting of branches of mangroves for fuel and fodder purpose often causes damage to mangrove ecosystem, however, the damage caused by this practice is not much significant. Dumping of sewage, solid and toxic wastes are the other kind of anthropogenic threat to this vital coastal ecosystem.

Barge movement in rivers, resulting in the strong wave action, due to which young regeneration/saplings get uprooted. This also leads to erosion on the river banks.

Inadequate infrastructure for protection also poses some difficulties in mangrove conservation but the threat to mangrove wetlands because of this problem is very limited.

19.2 Global Mangrove Cover Distribution

As mentioned earlier, mangrove forests are found in tropics and subtropics, between 32°N and 38°S latitudes. Global area under mangrove cover has been estimated to vary between 11 and 24 million hectares (Giri et al. 2010; FAO 2007; Wilkie and Fortuner 2003; Spalding et al. 1997; Fisher and Spalding 1993; Finlayson and Moser 1991). The first attempt to estimate world's total mangrove cover was done as part of the FAO and United Nation Environmental Program (UNEP) called 'Tropical Forest Resources Assessment' in 1981(FAO & UNEP 1981). As per this study, the world's total mangrove area was estimated as 15.6 million hectares. However, according to FAO (2007), the world's mangrove area estimated in 2005 was 15.2 million hectares. This study reports that about 20% of the mangrove area or 3.6 million hectares has been lost since 1980. However, more recently the rate of net loss in the mangrove area has been slowed down. According to Finlayson and Moser (1991) the total mangrove area of the world is about 14 million hectares: out of this the old world tropical mangrove, i.e., the Indo-Pacific tropical zones and tropical Australia have the most dominated and important mangroves in respect of species diversity, richness, abundance and succession. World Mangrove Atlas (Spalding et al. 1997) provides the global mangrove cover area as 18.1 million hectares. The most dominant and single largest mangrove patch of the world is in the Sunderbans, the Ganga–Brahmaputra-Meghna deltaic regions or estuarine mouths of both India and Bangladesh (Naskar and Mandal 1999).

The most recent estimates, by Giri et al. (2010), provide the global area under mangrove cover as 13.77 million hectares, spanning in 118 countries. They have used GLS data set of 2000. Their estimate was 12.3% lesser than the estimate provided by FAO (2007). GLS is a global data set of Landsat TM images (30 m spatial resolution, pertaining to 1997–2000 time frame), prepared in partnership between USGS, NASA, GEO and US Climate Change Science Program (CCSP) and NASA's Land Cover and Land Use Program, LCLUC (Gutman et al. 2008). As per the their mapping, approximately 75% of worlds mangroves are found in the following 15 countries: Indonesia, Australia, Brazil, Mexico, Nigeria, Malaysia, Myanmar, Papua New, Bangladesh, China, India, Guinea Bissau, Mozambique, Madagascar and Philippines (in the descending orders of mangrove area). Only 6.9% of the world's mangroves are protected under the existing Protected Area Network (IUCNI-IV). As per the Giri's study (Giri et al. 2010), largest extent of mangroves was found in Asia (42% of the global mangroves), followed by Africa (20%), North and central America – 15%, Oceania – 12% and South America – 11%.Indonesia with largest mangrove area, accounted for 22.6% of the global mangroves. Australia had the second largest area under mangrove cover (7.1%). India stood at 11th position, accounting for 3.1% of the global mangrove cover.

19.3 Classification of Mangrove Types

Mangroves get tightly bound to the coastal environment in which they occur. There are the following six types of mangrove forests (Colin 1995):

1. Over wash mangrove forests: These are small mangrove islands that are frequently washed by tides. Here the dominant species is Rhizophora.
2. Fringing mangrove forests: These are strips of mangroves found along the borders of the protected shorelines and islands influenced by daily tide range. They are sensitive to erosions. Here also the dominant species is Rhizophora.
3. Riverine mangrove forests: These are luxuriant stands of mangroves found along the rivers and creeks which get inundated by daily tides. They are influenced by large amount of fresh water inputs along with the fluvial nutrients. These are often composed of Avicennia, Rhizophora and Laguncularia.
4. Basin mangrove forests: these are stunted mangroves located along the interior side of swamps. These are often dominated by Avicennia.
5. Hammock mangrove forests: These are similar to the basin mangrove types except that they are found in more elevated areas than the above four types. Here, dominated species is Rhizophora.
6. Shrub mangrove forests: These are dwarfed mangroves occurring along the flat coastal fringes.

As the above classification of mangrove functional types has the limitations in providing information on the physical processes that takes place in all types of mangrove forests, a new classification has been suggested (Kathiresan 2005). These include, (i) river dominated, (ii) tide dominated and (iii) interior mangrove forests. Based on the substrate, tidal range and sedimentation, the following six more broad classes of mangrove settings have been suggested by Thom (1984) and Galloway (1982):

1. Large deltaic system, occurring in low tidal range and with very fine allochtonous sediments (examples are the mangroves of Sunderbans and Borneo)
2. Tidal plains, where alluvial sediments are reworked by the tides and there is presence of large mudflats.
3. Composite plains (under the influence of both, tidal and alluvial conditions). Examples: lagoons formed behind wave built barriers where mangroves grow.
4. Drowned bedrock valleys (e.g. mangroves of Northern Vietnam or Eastern Malaysia).
5. Fringing barriers with lagoons, high wave energy conditions with allochtonous sediment of fine sand and mud. Examples: mangroves of Philippines.
6. Coral Coasts: here mangroves grow at the bottom of coral sands or in platform reef. Examples: mangroves of Indonesia, India and Singapore.

The first five types of mangrove wetlands can be found on coast dominated by terrigenous sediments (shallow marine sediments consisting of material derived from the land surface) whereas the sixth one can be found in the oceanic islands, coral reefs and carbonate banks. Mangrove forests are usually characterized by uniform type of trees and shrubs and the species diversity decreases with increasing latitudes.

19.4 Remote Sensing

Remote sensing is the science of acquiring information about objects from measurements made by scientific devices called "sensors" without there being any physical contact between the target and the sensing device. Any force field i.e. gravity, magnetic or electromagnetic could be used for remote sensing measurements. However, in the modern context, the term remote sensing refers to the identification of earth features by detecting the characteristic electromagnetic radiation that is reflected or emitted by the earth surface. The whole edifice of remote sensing is built on the premise that all objects reflect/emit/scatter a portion of the electromagnetic energy incident on it depending upon its physical characteristics. For example, a red rose flower absorbs all the solar radiation falling on it except the red portion of the EM radiation and thus it appears red in colour. Similarly, leaves of the plant appear green to human eyes because it absorbs all the sun light falling on it except the light in green wavelength. In addition, the objects on the earth emit radiation depending on their temperature and other characteristics. If we study the reflectance/emittance of any particular object as a function of wavelength, we get a typical reflectance/emittance response pattern of the object as a function of wavelength, which is unique for that object. This reflectance/emittance pattern is known as 'spectral signature' and is characteristic of the object (Joseph 2007). Thus, proper interpretation of the spectral signature enables identification of the objects. This could be achieved by aerial as well as space-based remote sensing for large area coverage.

The main thrust of modern satellite-based remote sensing, apart from providing vantage point in space for viewing large areas on ground, has been to extend human eye's visibility range. This is done by converting interaction of non-visible part of electromagnetic radiations like infrared, short wave Infra-red (SWIR), thermal infrared, microwave etc. with matter into interpretable information. There are sensors which can detect the electromagnetic radiations in the above mentioned non-visible portion of the electromagnetic spectrum. These sensors can be mounted on the aerial or space based platforms for collecting the reflected or emitted solar radiations. Sensors for detecting electromagnetic radiations are of two types: (i) passive sensors and (ii) active sensors. Passive sensors do not have their own source of electromagnetic radiations for illuminating the target/objects, they measure the radiations reflected/emitted by the object. Active sensors have their own source of electromagnetic radiation for illuminating the target. RADARS and LIDARS are examples of active sensors. By analysing the radiations measured by the sensors, it is possible to detect, identify and classify various objects/phenomena.

Space borne remote sensing of the earth resources began with the successful launch of the Earth Resources Technology satellites ERTS-1 (later renamed as

LANDSAT) by United States of America in 1972. Since then, remote sensing has made rapid advances in the last four and a half decades, in terms of the increased spatial, spectral, radiometric and temporal resolutions. Spatial, spectral and, radiometric resolutions are the key sensor parameters which defines the quality of images captured by these sensors. These three types of resolutions are responsible for discrimination and identification of different objects on the image. Spatial resolution is defined by the pixel size and is governed by the instantaneous field of view (IFOV) of the collecting optics of the sensor system. Basically, it is the capability of the sensor to discriminate and identify the smallest object on the ground (Joseph 2007). Spectral resolution is basically the spectral bands in which images are taken by a particular sensor. Radiometric resolution is the capability of the sensor to discriminate two targets based on their reflectance/emittance difference. It is measured in terms of the smallest radiance difference, emitted or reflected by two targets that can be detected. Higher the radiometric resolution, smaller the radiance difference (reflected/emitted by the two targets), that can be detected. The fourth one, the temporal resolution, is the capability of sensors to view the same ground area at frequent interval of time.

During the last one and a half decades, Earth Observation (EO) satellites have been providing data with sub meter spatial resolutions in panchromatic (e.g. IKONOS, QuickBird, Worldview-3, Cartosat-2). Multispectral and stereo data are available with 2.5–30 m spatial resolutions from remote sensing satellites (e.g. Resourcesat-2, SPOT-7, LANDSAT-8 Thematic Mapper, ASTER, Cartosat-1). On the other hand EO satellites also provide medium spatial resolution data at a very high repeat cycle (MODIS, AWiFS on Resourcesat-2, Oceansat-2). Above mentioned data, available from EO satellites, have been widely used world over, for inventory, monitoring and management of natural resources, area development planning, urban and infrastructure development, environment monitoring and assessment as well as for disaster mitigations.

Remote sensing data collected from space platforms have also been extensively used in mapping and monitoring of coastal zones, its environment, coastal ecosystems and ecologically sensitive areas (ESA) in addition to assessing the ocean biological resources. Various coastal features, landforms, vital habitats and ecologically sensitive areas are very well discernible on the images acquired by remote sensing or Earth Observation (EO) satellites.

As an example, Fig. 19.1 shows IRS LISS-3 FCC (False Colour Composite) image of the part of gulf of Kachchh, western coast of India. In FCC images, vegetation is seen in red colour, water in blue, barren land in light grey, and sandy area and salt effected land are seen in white. Tonal variations in red colour (light to dark red) indicate the increasing vigour or density of the vegetation. Similarly, dark blue tone represents deep water and light blue represents either the shallow water or turbid water. Information extraction from these images, through visual interpretation, is done on the basis of colour, tone, texture, association, shape, size and location of the features appearing on the image. For example, mangroves can be discriminated from the other coastal vegetation as well as terrestrial vegetation based on its tone (red/pale red), association (in warm waters, on mudflat substrate and low energy

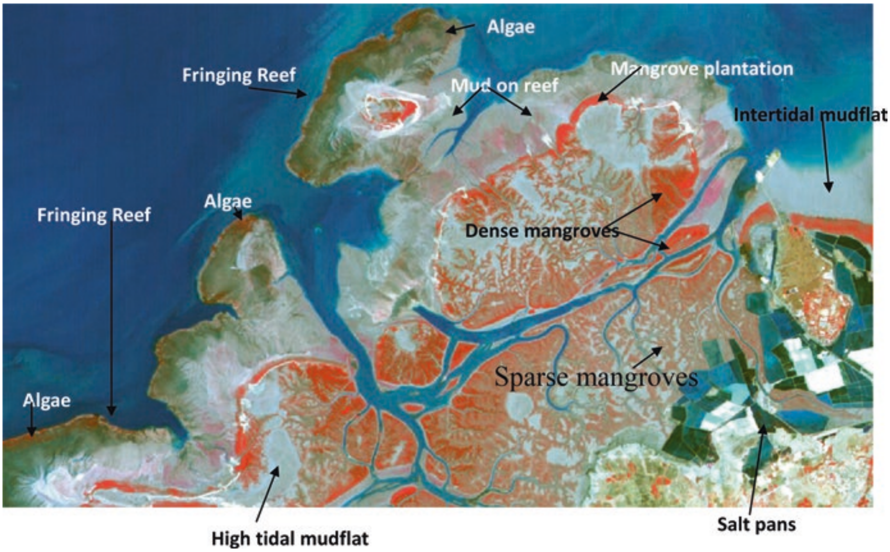


Fig. 19.1 IRS LISS-3 image of part of Gulf of Kachchh, western coast of India, showing various coastal features and landforms as well as vital coastal ecosystems and habitats such as coral reefs, inter-tidal mud flats and mangroves

coast), location (in intertidal area and near high tidal area), etc. The image in Fig. 19.1 shows various coastal features, coastal land forms, vital coastal ecosystems and habitats, including mangroves and its density classes (dense and sparse), coral reefs, algae, mud flats, salt pans etc.

As discussed earlier, the quality of the images, acquired by earth observation satellites, are governed by sensor parameters such as spatial, spectral and radiometric resolutions. Spatial resolution is the most important among these sensor parameters from the perspective of the general users. Thus it will be appropriate to discuss a little bit more about the importance of spatial resolution in the context of the spatial information extraction from the satellite images. Spatial resolution is a measure of the sensor's ability to image the two closely spaced objects on the ground, distinctly so that they are distinguishable as separate objects. Thus a sensor with one meter spatial resolution can reproduce image with "finer details" as compared to a sensor with say, 30 m resolution. However, as we increase the spatial resolution (say from 30 m to 1 m) the area coverage (swath) of the image taken by the sensor decreases. So, if we want to cover a large area on the ground in one frame of the image (swath) we cannot achieve very high spatial resolution. Swath and temporal resolution are also related. Larger the swath coverage higher is the temporal resolution i.e. decrease in the satellite revisit time (satellite can revisit the same ground area in less time). This helps in frequent monitoring of the same area on the ground. Thus depending upon the desired purpose/use of the images, the spatial and temporal resolutions and swath are decided. If we want to use image for a large scale detailed level mapping, we need to have very high spatial resolution but will have to

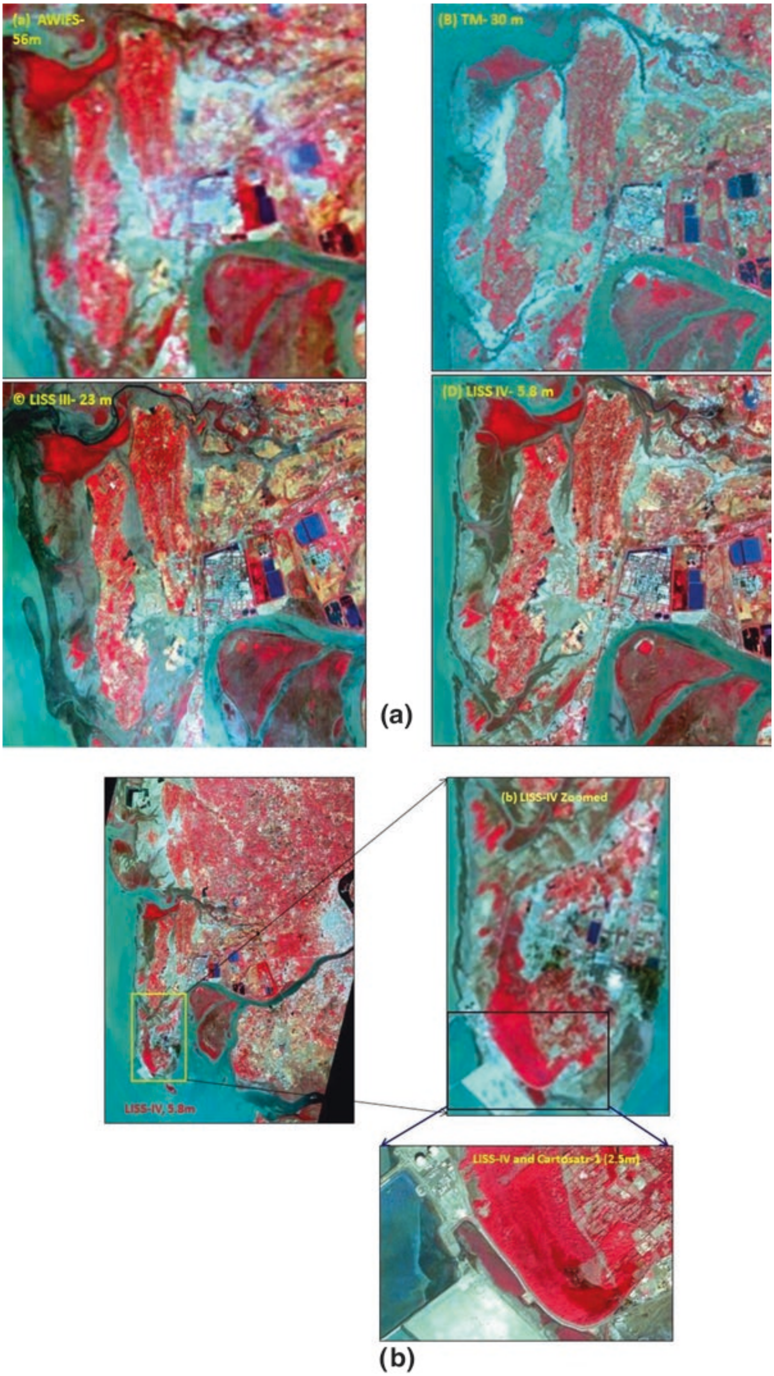


Fig. 19.2 (**a**) Impact of change in spatial resolution on the information content of the satellite image. As the spatial resolution improves from 56 m (AWiFS) – 30 m (Landsat TM) – 23 m (IRS LISS-III) – 5.8 m (IRS LISS IV), the sharpness of the image increases and more and more finer details are seen (**b**) shows LISS-IV image and zoomed LISS-IV image of a small part (shown as *yellow rectangular box*) and the corresponding image of (Cartosat-1 + LISS IV) having effective

sacrifice the swath coverage. If we wish to monitor certain dynamic events on the ground (e.g. natural disasters), we need to have high revisit frequency and thus have to opt for moderate spatial resolution sensors. For example IRS AWiFS sensor has coarse resolution of 56 m but has a large swath (370 km) and a revisit time of 3–5 days. IKONOS has a very high spatial resolution of 1 m but it has its swath restricted to only 11 km.

Figure 19.2 explains the impact of the spatial resolution on the quality of the image. Figure 19.2a shows the images of the same ground area, taken by sensors of different spatial resolutions (56–5.8 m). The area shown in the image is the coastal town of Hazira, Surat district of Gujarat state, western coast of India. The images shown in Fig. 19.2a are the AWIFS image (56 m), Landsat TM (30 m), LISS-III (23 m) and LISS-IV (5.8 m). We can see as the spatial resolution increases from 56 to 5.8 m, the sharpness of the image increases and we can see finer details. We cannot zoom a given image beyond a certain limit in order to see much more details, instead we need to use images with higher spatial resolution. Figure 19.2b shows the LISS-IV image (same as in 19.2a) and the zoomed LISS-IV image of its small portion and Cartosat-1+ LISS-IV merged image (2.5 m effective resolution) of the corresponding area. One can see that the zooming of the LIS-IV image has led to blurring (Fig. 19.2b) and does not help in extracting further details from the image. Whereas, much larger details are seen in the corresponding 2.5 m (cartsat-1 + LISS-IV merged) image. Jetties and the industrial installations/constructions can be clearly seen at the coast, adjacent to the red mangrove patch in the 2.5 m image which is not clearly seen in LISS-IV (5.8 m) image.

The level of information extraction (in terms of the details) and the scale of maps which can be prepared from the satellite images depend on the spatial resolution (Ajai 2004). Global/Regional level maps (1:1–5 million scale) on various themes such as land use/land cover, geomorphology, vegetation, water, mangroves etc. can be prepared from satellite images having 200 m–1 km spatial resolution. National/state level mapping (1:250,000 scale) will require satellite images with 70–100 m resolution. Maps on 1:50,000/25,000 scale can be prepared using satellite images of (20–40 m)/(10–15 m) spatial resolution. Whereas detailed level thematic mapping on 1:4,000–10,000 scale will require satellite data of 0.5–5 m spatial resolution (Ajai 2004).

19.5 Mangrove Inventory

One of the basic information required for conservation and management of mangrove ecosystem is the spatial inventory of mangrove cover, its type and canopy density. The physical conditions and harsh environment prevailing in and around the

Fig. 19.2 (continued) resolution of 2.5 m. Zooming of LISS IV image has led to blurring and does not provide any further. Much more details are seen in the corresponding 2.5 m image. Jetties and industrial installations/built up area are clearly seen near the coast, adjacent to the *red* mangrove patch mangrove area make the conventional methods of data collection (through field survey and measurements) extremely difficult. Under such conditions, data from earth observation satellites provide a viable and cost effective alternative for creating spatial inventory of area under mangrove cover as well as its canopy density. On satellite image, mangroves can be differentiated from the other coastal vegetation based on tone, association, locations etc. (Nayak and Bahuguna 2001). Impact of spatial and spectral resolutions on the accuracy of mapping of mangrove has been studied by Gao (1999) in the mangrove forest of Waite Mata harbour, Auckland, New Zealand. Mangroves were mapped in to lush green and stunted categories from SPOT HRA and Landsat TM images at 10, 20, 30 m spatial resolutions using MXL (maximum likely hood) classification technique. Remote sensing satellite data has been extensively used in mapping and monitoring of the mangrove forests at global, national and local scales (Kumar et al. 2012a, b, 2016; Ajai et al. 2012, 2013; Kuenzer et al. 2011; Giri et al. 2007, 2010; Ramasubramanian et al. 2006; Nayak and Bahuguna 2001; Azipuru et al. 2000; Blasco et al. 1986, 1998; Green et al. 1998; Chaudhary 1990; Dutrieux et al. 1990; Untawale et al. 1982). Satellite data acquired during the low tide period are used for mapping of mangrove as during low tide maximum coastal zone gets exposed.

Giri et al. (2010) has carried out inventory of the mangroves of the world by using Landsat-TM data of 1997–2000 time frame. Details of their findings are discussed earlier (in Sect. 19.2). Giri et al. (2015) have studied distribution and dynamics of mangrove forests of south Asia. Nayak and Bahuguna (2001) have mapped the mangrove cover along with its crown density for India using Indian Remote Sensing Satellite data. Green et al. (1998) have used SPOT- XS and Landsat- TM data for mapping of mangroves in the Eastern Caribbean Islands. Blasco et al. (1986) and Gang and Agatsiva (1992) have used remote sensing data for mapping of mangroves of Kenya coast. Dutrieux et al. (1990) have made inventory of mangrove forest of Mahakam Delta, East Kalimantan, Indonesia using SPOT images. Everitt et al. (1996) have used remote sensing data for discriminating and mapping of black mangroves on Texas coast. Satyanarayana et al. (2011) have used very high spatial resolution multispectral data (2.4 m) from QuickBird along with ground truth measurements for assessment of mangrove vegetation at Tumpat, Kelantan river delta, Malaysia.

In recent past, studies have been carried out to understand the spatial relationship between mangroves and their immediate environment, viz., 'community zones' (Tomlinson 1994; Hogarth 1999), using advanced digital image processing techniques (Nayak and Bahuguna 2001; Blasco et al. 1998). These mangrove zones exhibit unique spectral signatures. Study by Nayak and Bahuguna (2001) has revealed the potential of remote sensing data in discerning and mapping mangrove communities. Earth Observation Satellite data has been extensively used for inventory, mapping and monitoring of mangroves and other coastal vegetation at global, national and local scales (Giri et al. 2010, 2015; Upadhay et al. 2015; Patel et al. 2014; Ajai et al. 2013; Kumar et al. 2012a, b; Heumann 2011; Satapathy et al. 2007; Kovcas et al. 2005; Nayak and Bahuguna 2001; Green et al. 1998; Gang and Agatsiva 1992).

Moderate to high resolution multispectral data from earth observation satellites such as IRS LISS-III/LISS-IV, Landsat-8 Thematic Mapper, SPOT-7and ASTER are considered as the primary data source for mangrove inventory, mapping and

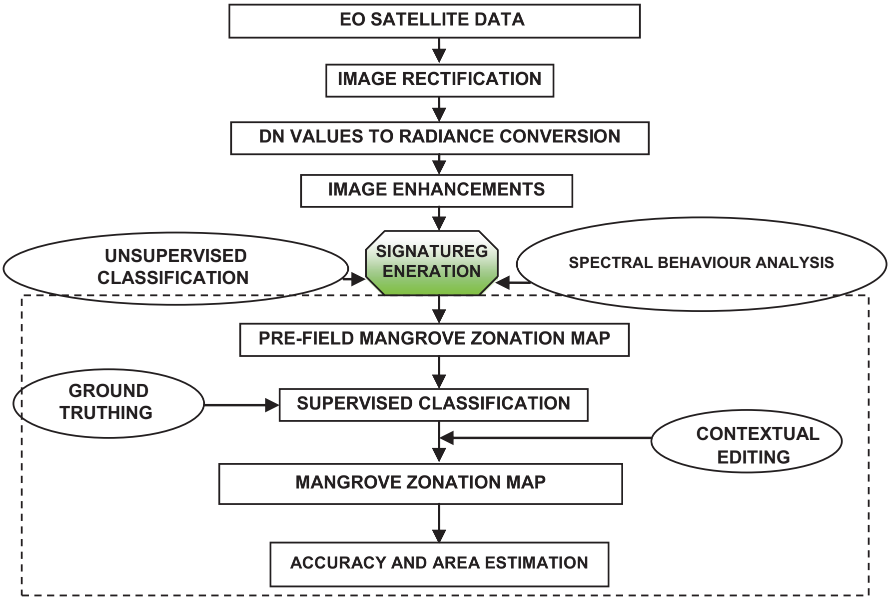


Fig. 19.3 Methodology for mangrove community mapping (Ajai et al. 2012)

monitoring. As the mangrove forests are tide inundated, care should be taken to use the satellite data acquired during the period of low tide. For delineating fringe mangroves along narrow creeks and young/new plantations, very high resolution (about one meter) satellite data may also be required as supporting data for better interpretations. Satellite data in short wave infra-red band is very effective for differentiating mangroves from the adjacent other vegetative covers.

The basic premise on which the zonations of mangrove communities using spectral data are done, is based on the fact that each of the 'mangrove community classes' have different reflectance or radiance/DN numbers in different spectral bands of the multispectral data sets acquired by the EO satellites (Nayak and Bahuguna 2001). Thus these classes are separable in multi-dimensional spectral feature space.

Details of the methodology for mangrove community zonation mapping are given in Fig. 19.3. Broadly, the methodology is based on the hybrid of supervised and unsupervised digital image classifications followed by contextual editing to improve the mapping accuracy. The first step in digital analysis of the satellite data is geo-referencing/rectification of the satellite images using ground control points (GCPs) from topographical maps or collected through field survey using GPS. Geometric correction is performed to improve the geo-location to a root mean square error of half pixel, an accuracy needed for subsequent change analysis and monitoring of mangroves. Details on the above image rectification and classification are given in Joseph (2007). Digital values for each image pixels are converted to spectral radiances using calibration equations given as under:

$$L_{rad} = \{[DN / \max \text{grey}] \times [L_{\max} - L_{\min}]\} + L_{\min} \quad (19.1)$$

Where, DN = Digital number of each pixel; max grey = maximum grey values (e.g., 255 for LISS-III). L_{max} and L_{min}, the maximum and minimum radiance values, can be obtained from sensor calibration details provided in the metadata.

After radiance conversion, unsupervised classification is performed using Iterative Self- Organizing Data Analysis (ISODATA) classifier on the geo-referenced images. The ISODATA method uses minimum spectral distance to assign a cluster for each candidate pixel. Depending upon the scene characteristics, arbitrary clusters are specified with a convergence threshold of 0.99. The clusters thus obtained were colour coded for better visual discriminations and then pre-field classification maps are prepared.

It has been established through the field observations that the land topography, level of salinity and duration of tidal inundation are the predominant factors influencing the species composition of the mangrove communities. The false colour composite (FCC), prepared from digital multispectral data is able to discern the major mangrove communities, to a considerable extent. To identify the subtle differences in species composition, maps generated from ISODATA classifier (as described above) are used in ground-truthing (GT). Ground truth points are selected in such a way that each of the digitally discernible mangrove communities is well represented and covered for ground truth data collection. Geographical coordinates, information on species composition and dominance, influence of tidal inundation and ground photographs for each the 'ground truth points' are collected during the field/ground truth visits. Based on the ground truth information, supervised classification of the multispectral data is performed with sufficient training sets for each of the mangrove communities and other classes present in the study area. Ground-truth information is used to identify the training sites on the image for each of the mangrove communities and other classes present in the study area. The radiances/DN numbers and their associated statistics (mean and variance-covariance) computed for each of the training sites, are used to train the classifier. Entire image is then classified in to various classes. Mangrove community classes, thus obtained, are evaluated critically with the ground information and expert knowledge. Contextual editing is also performed (wherever necessary) to improve the classification accuracy. Area statistics is drawn based on final classification. Contextual editing is, basically, the use of non-spectral information such as: geography, location, and association etc. to improve the discrimination of the spectrally overlapping classes. For example, the pixels which are classified as mangroves based on the spectral information, but located out-side (landward side) the highest of the high tide line (HTL), has to be recorded as the terrestrial vegetation and not mangroves. Contextual information are added as a series of explicit decision rules that are applied to the entire image while classification is performed. Contextual editing has been extensively used in the coastal habitat mapping based on the multispectral data analysis and it helps in improving the mapping accuracy (Mumby et al. 1998). Finally, the map outputs are generated at the desired scale and area statistics computed. Accuracy assessment of the mangrove community maps are carried out using the field/ground truth data. Usually, the ground truth/field information collected for the 50% of the total number of the GT (ground truth) points, are used in training the classifier and the remaining 50% GT points are used for estimating the classification/mapping accuracy.

19.5.1 Case Study for India

Inventory of the mangrove forest cover in India is regularly done (once in a 2 years) since 1987. Spatial inventory of mangrove forest cover through the analysis of the multispectral LISS-III data from the Indian Remote Sensing Satellites (IRS) is carried out regularly by Forest Survey of India (FSI). The spatial inventory provides the information on the area and crown density (three classes: very dense, moderately dense and open) at national, state and district levels (FSI 2013).

However the above mapping of mangroves by FSI does not provide information on mangrove community. Methodology as discussed above has been used to generate spatial inventory of the mangroves at community level for India (Ajai et al. 2012, 2013). Before going in to the details of the case study on community level mapping of mangroves, it will be appropriate to discuss the status of the mangrove forest cover, its geographic distributions and types of mangrove in India as well as the threat to this important ecosystem.

19.5.1.1 Mangrove Forests of India

India has a long coastline of 7517 km. It has about 4628 km^{2}area under mangrove cover (FSI 2013) which is about 0.14% of the geographic area of the country. In terms of the mangrove area, India stands at the 11th position in the world. In India, mangrove forests are found on both eastern and western coasts, mostly within the sheltered inter tidal flat deltaic lands, funnel shaped bays, broad estuarine mouths and shallow or frequently tidal inundated coastlines. Each of these habitats has a unique environmental setting created by factors like geomorphology of the coast, climate, tidal amplitude, and duration and quantity of freshwater inflow and the salinity quotient caused by mixing up of the freshwater with the sea water. Though mangrove forests are found in all the nine maritime states of India and the Andaman and Nicobar Islands, but major patches are confined to only seven states and the Andaman and Nicobar Islands (shown in Fig. 19.4). Basically, mangrove forests of India are located in three zones: (i) East coast, (ii) West coast and (iii) Andaman and Nicobar Islands. In most part of the west coast, hilly terrain starts after a narrow strip of flat land. Whereas, east coast comprises of vast flat terrain and many deltas. In east coast, major formations of mangrove forests are: Sunderbans (West Bengal), Bhitarkanika and Mahanadi (Odhisa), and Pichhavaram and Muthupet (Tamilnadu). The major mangrove areas in west coast are: Vembanad (Kerala), Coondapur

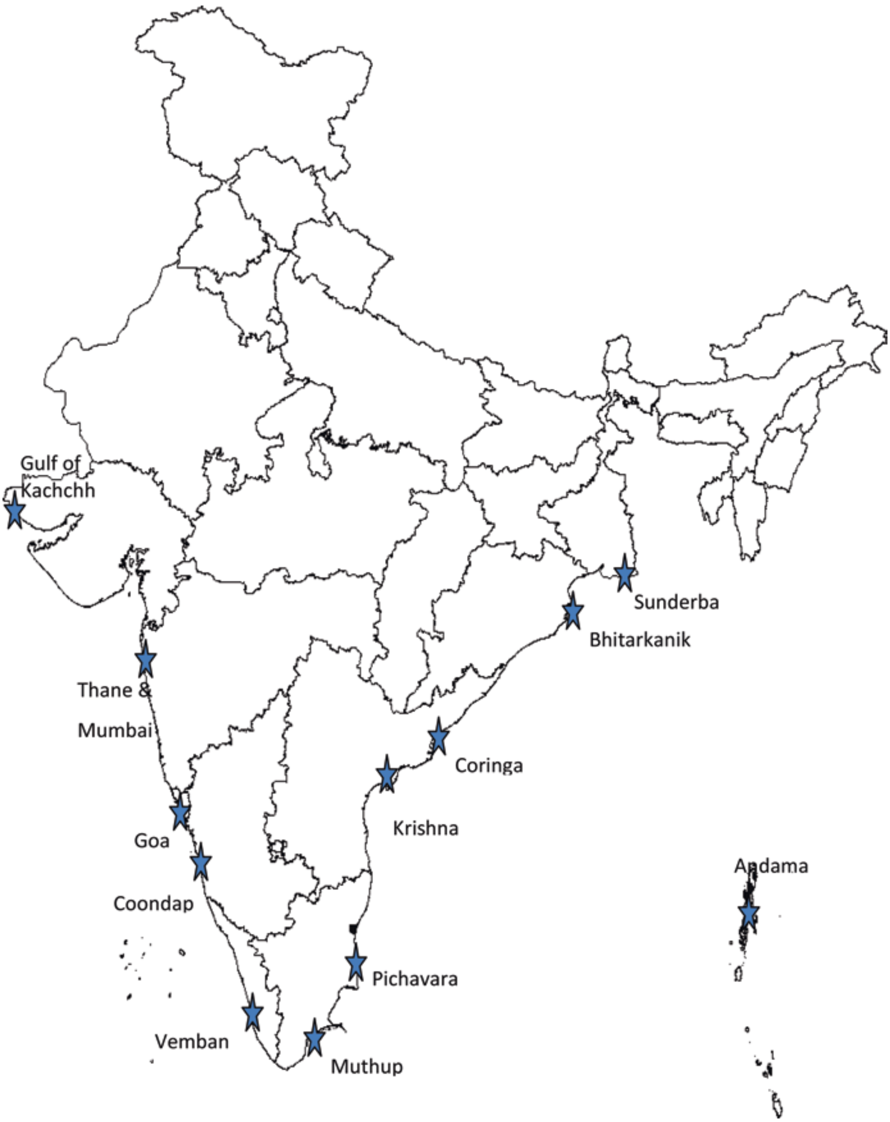


Fig. 19.4 Geographic distributions of major mangrove forests in India

(Karnataka), Goa, Thane and Mumbai (Maharastra) and Gulf of Kachchh (Gujarat). Rivers are the major source of fresh water to the east coast mangroves of India. The rivers Ganga and Brahmaputra are main source of fresh water to Sunderbans. Mangroves of the Mahanadi and Krishna deltas are being fed with their respective rivers. Cauvery river is the main source of fresh water for the mangroves of Pichhavaram and Muthupat. There are no major river systems in the west coast and

Andaman and Nicobar Islands. Mangroves in the west coast, except a few large patches, are mainly confined to the fringes of rivers and creeks.

The mangroves of the Andaman and Nicobar Islands and Bhitarkanika on one side were amongst the most diverse while the Mangroves of Gujarat on the other side had a low diversity but were thriving in very difficult environmental conditions.

On the macro-scale, geomorphic setting of the mangrove wetlands of the east coast of India is different from those of the west coast. Steep slopes, rises, promontories and drowned estuaries characterize the west coast of India, while the east coast shows a sequence of delta formations (Ahmad 1972). This vast expanse of deltaic soil and mud-flats favours luxuriant growth and diversity of species. According to Jagtap et al. (1994), the mangrove flora of India consists of 50 species, of which 37 species are reported from east coast and 20 from the west coast. Species such as *Pemphisacidula* is endemic to islands of Gulf of Mannar of Tamil Nadu, *Scyphyphora hydrophyllacea* is endemic to Godavari mangroves of Andhra Pradesh and *Nypa fructicans* is present only in Sunderbans.

Along the east coast the tidal amplitude, volume and intensity of fresh water inflow decreases from north to south affecting the soil characteristics, mainly, salinity, moisture content, chemical composition, texture and nutrient availability. Correspondingly, the species diversity and extent of mangroves also decreases. This is clearly evident as Sunderbans of West Bengal harbours around 30 true mangrove species while Pichavaram and Muthupet of Tamil Nadu records only 13 species (Naskar and Mandal 1999; Mandal and Naskar 2008).

Sunderban lies in the delta of two major rivers Ganga and Brahmputra of Indian sub-continent. The Indian boundary of the Sunderban, confines through Hoogly and Raimangal on the west and the east. The name sunderban is derived either from local name “Sundri” of the once prevalent mangrove species *Heritiera* or from words Sunder ban (beautiful forest). Topographically, Sunderban area exhibits number of anastomosing distributaries. The river Bartala, Muri Ganga, Saptamukhi, Icchamati, Kali nadi, Bidya and Gosaba carry fresh water from the upper reaches. A number of the islands are formed at the bay by these rivers. Mangrove is also described as the flora of formative islands and the flora of riverbank and swamps forest. The formative island species are mainly tree type of the *Avicennia spp., Excoecaria agallocha*, and *Sonneratia apetala.* The major species of the dense mangroves forest includes *Avicennia spp., Excoecaria agallocha, Sonneratia apetala, S. Caseolaris Heritiera fomes*, *Phonix paludosa, Rizophora apiculate, R. mucronata and Nypa fruticans* almost cover all the tidal zones of Sunderbans (Naskar and Mandal 1999).

Bhitarkanika national park, a unique habitat of mangrove forests in the east coast, is located in the Kendrapara district of Odhisa state. Bhitarkanika has very high mangrove species diversity. Mangrove and mangrove associates found here are: *Avicennia Alba, Excoecaria, Sonneratia, Heritiera*, *Phonix* and *Salvadora.*

Coringa Wildlife Sanctuary, in the east coast, is located south of Kakinada bay flanked by the shallow bar-built Bay towards the north and extensive network of estuarine creeks and canals emanating from river Godavari in the south. Coringa wildlife sanctuary covers mangroves of east Godavari district, Coringa reserved forest, Coring RF extension and Bhairavpalem RF. *Avicennia Alba* and *Sonneratia* are the major types of mangrove that exists and forms large colonies. *Avicennia officinalis* are also found in many landward side areas.

Pichavaram mangrove block is located between Vellar and Coleroon estuaries. *Avicennia* and *Rizophora are* distinctly separable in Pichavaram. The *Rizophora* is found along the fringes of tidal creeks and channels, whereas *Avicennia* is found more seaward side which are more saline. Muthupet is at the southern most end of Cauveri delta. Various tributaries of Cauveri flow through Muthupet and nearby villages. Muthupet mangroves are of *Avicennia*, mainly *Avicennia marina* and *A. Suaeda.*

Andaman and Nicobar Islands of India accounts for about 13% of the total mangrove forest area of India (FSI 2013). Mangroves of Andaman and Nicobar Islands are recognized as the best in the country in terms of density (about 76.5%) and growth (Dageretal 1991). Recently, 25 true mangrove species, distributed among 10 families and 14 genera, have been recorded in Andaman and Nicobar as compared to the earlier reported number of species, ranging from 17 to 36 (Gautham-Bharati et al. 2014; Dam Roy et al. 2009).

Vembanad in Kerala state (West coast) has mangrove forest comprising of mainly Excoecaria and *Rizophora.* Coondapur *in Karnataka state has Rizophora*as major species along the river having moderate salinity. Dominant mangrove species in Goa are the *Avicennia, Excoecaria, Rizophora* and *Sonneratia*. In the state of Mahrastra, major mangrove patches are found around Mumbai, Raigad and Thane. Mangrove along the Manori and Malad, Vashi in new Mumbai are mainly *Avicennia marina* and *Avicennia Alba*. Mangroves around Thane creek are in good condition however they are under anthropogenic pressure.

The largest mangrove formation in the west coast is found on the Gujarat coast, mainly confined to south Gujarat, Gulf of Kachchh and Kori creek (Indus deltaic region). As this area is not easily accessible, the mangrove condition in this area is very good. In this region, apart from few patches of *Rizophora* and *Sonneratia*, the dominant species is the *Avicennia* (Singh 2000).

Furthermore, within the sheltered habitats, the species form distinct communities due to their strong dependence on specific environmental gradients like temperature, tidal frequency, sedimentation processes, freshwater availability, wave and storm surges, and physico-chemical soil properties. The distinct zonation pattern, in which each zone comprising of one or two dominant species or an assemblage of a particular group of species is clearly noticeable within short transects from the coast towards the landward side.

19.5.1.2 Community Level Mapping for India

Mangrove community level mapping for the entire Indian coast has been carried using Resourcesat-1 LISS-III/LISS-IV data on 1:25,000 scale. The methodology, as discussed earlier, has been applied to classify and map the mangrove communities for all the maritime states and union territories of India harbouring mangroves,

Table 19.1 Key parameters of Resourcesat-1 LISS-III and LISS-IV sensors

		LISS-IV		
		Mono mode	MX mode	LISS-III
Spatial resolution	Band 2 (green)	5.8 m	5.8 m	23.5 m
	Band 3 (red)			
	Band 4 (NIR)			
	Band 5 (SWIR)			
Swath-width	All bands	70 km	23.9 km	140 km
Radiometric resolution/ quantization	All bands	7 bit rescaled	7 bit	7 bit
Spectral	Band 2 (green)		520–590 nm	520–590 nm
Coverage	Band 3 (red)	620–680 nm	620–680 nm	620–680 nm
	Band 4 (NIR)		770–860 nm	770–860 nm
	Band 5 (SWIR)			1550–1700 nm

namely, Gujarat, Maharashtra, Goa, Karnataka, Kerala, Tamil Nadu, Puducherry, Andaman and Nicobar Islands, Andhra Pradesh, Odhisa and West Bengal.

Details of the Resourcesat-1 LIS-III and LISS-IV sensors are given in the Table 19.1.

Classification System

A hybrid classification system involving both geomorphological and ecological characteristics of the habitat is ideal for mangrove community mapping as both the characteristics strongly influence the radiance recorded by the electro-optical sensors on board EO satellites. Such a classification system was evolved for mapping of Mangrove vegetation of Indian coast (Ajai et al. 2012; Nayak and Bahuguna 2001). The classification system for mangrove community mapping, based on the geomorphological zones and ecological classes, is given in Table 19.2. Though a direct geomorphological classification is not done, mangrove community zonation is quite visible based on definite environmental gradients. Depending on the dominance of species present in these zones and its crown cover density, mangrove community classes are delineated and assigned class names in the order of dominance. Mangrove communities with more than three mangrove species are classified as mixed mangrove communities.

Mangrove community classes are delineated based on the dominance of the genus present in these zones and its crown density. Classes names are assigned in

Table 19.2 Geomorphological zones and ecological classes of mangroves

Geomorphological zones		Ecological classes	
Onshore areas	Beach	Muddy sandy	Fringe tidal mangroves
Deltaic complex	Estuarine mouth	Inter-tidal	*Avicennia* pure (three species)
		Mudflat	*Aegialitis* (only in Sunderbans and Mahanadi)
	Mid estuary (creeks and canals more)	Inter-tidal	*Rhizophora, Bruguiera, Ceriops, Son neratia, Aegiceras, Xylocarpus*
		Mudflat	
	Inner estuary	Inter-tidal	*Rhizophora, Bruguiera, Heritiera, Carberra*, *Cynometra, Excoecaria*,
		Mudflat	
	Outer estuary	High-tidal	*Dalbergia, Derris*,
		Mudflat	*Excoecaria, Acrostichum, Pongania*
			Marsh vegetation
			Saline blanks
Bay complex	Mouth	Inter-tidal	*Rhizophora, Bruguiera*,
		Mudflats	*Ceriops, Sonneratia, Xylocarpus*
	Middle zone	High-tidal	*Avicennia, Phoenix*,
		Mudflats	*Lumnitzeralittoralis, Heritieralittoralis, Nypa*
	Inner zone	High-tidal	*Avicennia*, *Nypa, Acrostichum, Thespesia, Derris*
		Mudflats	Marsh vegetation
			Saline banks
Gulf complex	Seaward zone	Sub-tidal	Algae
		Mudflat	
	Inner zone	Inter-tidal	*Rhizophora, Sonneratia, Avicennia, Ceriops, Bruguiera*
		Mudflat	
	Outer zone	High-tidal	*Avicennia*,
		Mudflat	Salt marsh vegetation
			Grass/acanthus
			Saline blanks
Offshore area	Continental shelf		Algae/seaweeds
	Islands		Mangroves sand vegetation
	Coral reefs		Algae/seaweeds/seagrass

Ajai et al. (2012)

order of dominance. Mangrove communities with more than three mangrove genus are classified as mixed mangrove communities. In some cases the term "Sparse" has also been used to represent the mangroves which have density between 10 and 20%. Zoning also includes condition assessment in the form of density-wise classification of the communities. Mangroves have been classified in four density classes, viz., very dense (>70% cover), moderately dense (40–70% cover), dense (10–40%) and degraded (<10% cover).

As per this spatial inventory on 1:25,000 scale, using Resourcesat-1 data of 2005–2007 timeframe, the area under mangrove cover for Indian coast is 495,620 ha

Table 19.3 State wise area and major communities of mangroves in India

State/union territory	1990–1993	2006–2007	Dominating communities
	Area in km^2		
Gujarat	1014.6	890.69	*Avicennia* (+4)
Maharashtra	222.6	270.92	*Avicennia* (+2)
Goa	6.7	34.63	*Avicennia* (+3)
Karnataka	8.7	6.05	Mixed (+2)
Kerala	10	6.63	Mixed (+4)
Tamil Nadu (TN) including Puducherry (P)	23.6	57.35	*Avicennia* (+4)
Andhra Pradesh	380	351.27	*Avicennia* (+3)
Orissa	187	221.07	*Avicennia* (+10)
West Bengal	1838	2529.27	*Avicennia* (+5)
Andaman	679	566.58	*Rhizophora* (+17)
Nicobar	70.9	21.74	*Rhizophora-Bruguiera* (+2)
Total	4441.1	4956.2	

(Ajai et al. 2012, 2013). The state-wise details, in terms of the mangrove cover and dominant communities are given in Table 19.3.

For comparison purpose, the area under mangrove cover as mapped using data of 1990–1993 timeframe is also given in the table. There is an overall increase in the mangrove area of India during the above mentioned period. State of West Bengal has the highest area under mangrove cover among all the Indian coastal states. In this study, 37 true mangrove and 32 mangrove associate species have been recorded in West Bengal (mainly Sunderbans). There has been considerable decrease in mangrove vegetation of Gujarat, Karnataka, Kerala, Andhra Pradesh, and Andaman and Nicobar islands during the above period. Significant increase in the mangrove area has been found for the states of Maharashtra, Goa, Tamil Nadu and West Bengal. The most dominant mangrove community found in most of the Indian states is *Avicennia*. The other dominant mangrove community found is *Rhizophora.*

Examples of the mangrove community zonation maps, prepared using the methodology given in Fig. 19.3 and IRS LISS-IV/III, are given in Fig. 19.5 for Bhitarkanika mangrove forest and in Fig. 19.6 for a part of Sunderbans. IRS images (FCC) for the respective areas are also given in the Figs. 19.5 and 19.6.

The above examples of community zonation mapping are based on the IRS LISS IV (5.8 m resolution) or LISS- III data (23 m resolution). However, if we intend to do the community zonation mapping for smaller study areas, e.g., a small islands spanning a few tens to hundred meters, we need to use satellite data having very high spatial resolution (say about 1 m). Mapping can be done on a larger scale of 1: 10,000 scale using satellite data of 1 m spatial resolution. Example of community zonation mapping, using very high spatial resolution satellite data (IKONOS data with 1 m resolution) for the Maipura and Twin islands situated in the east of Bhitarkanika reserve forest is given in Fig. 19.7. FCC of the area is also given as reference.

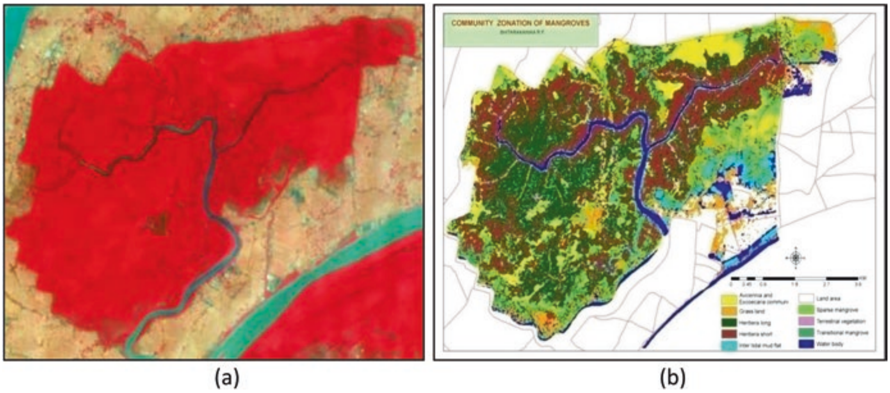


Fig. 19.5 LISS-III FCC image (**a**) and Mangrove community zonation map (**b**) of Bhitarkanika Reserve Forest, Odhisa state, East Coast of India

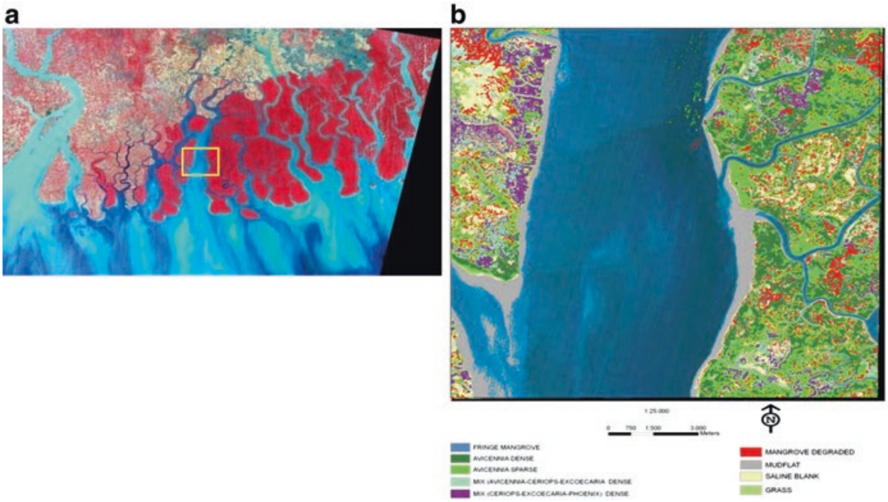


Fig. 19.6 (**a**) IRS LISS III Image of Sunderbans mangrove forest (**b**) Mangrove community zonation map of parts of Sunderbans (*yellow box*) in South 24 Parganas district of West Bengal state, East Coast, India

19.6 Monitoring of Mangroves

Monitoring of the coastal zones and its landuse/landforms as well as vital habitats has become simpler and cost effective with the availability of high spatial resolution data from earth observation satellites during the past three decades. Many studies on monitoring of mangrove ecosystems have been carried out using multi date satellite data (Kanniah et al. 2015; Bhavsar et al. 2014; Kumar et al. 2012a, b; Aizpuru et al.

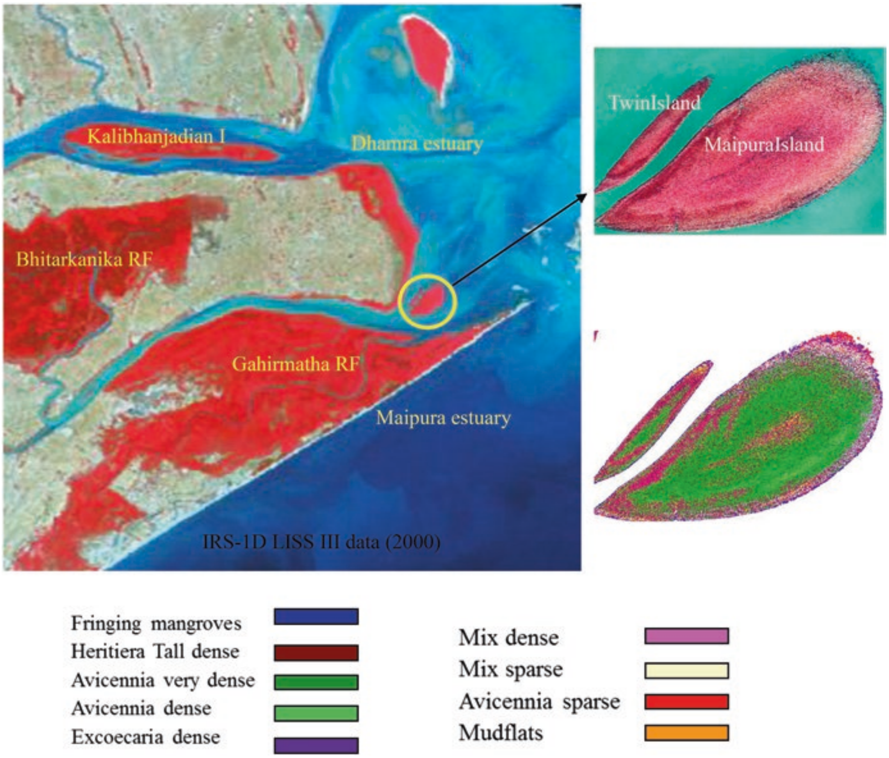
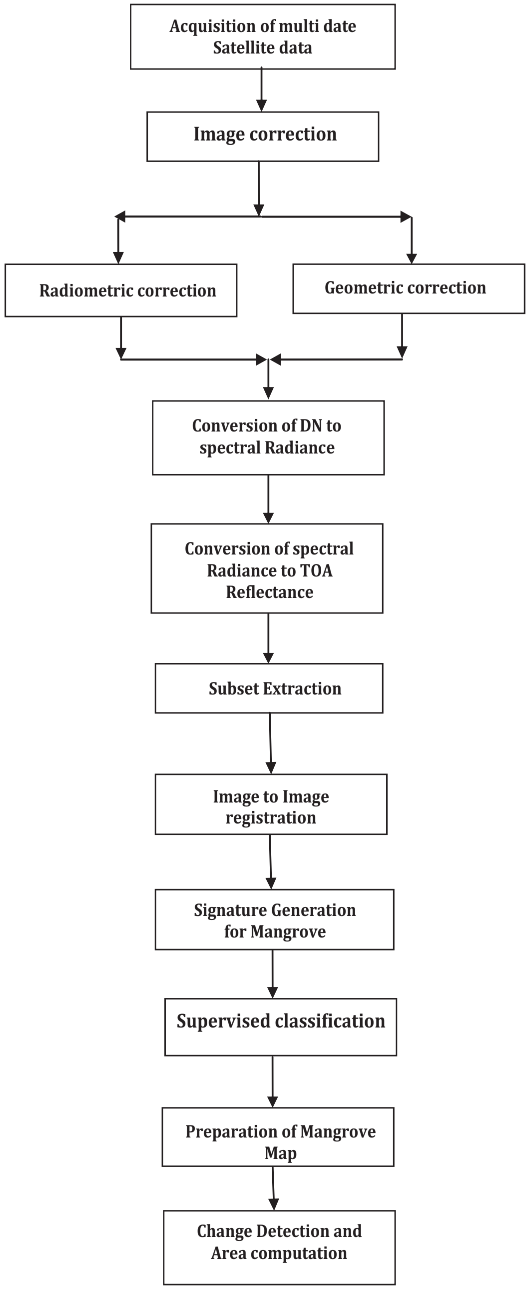


Fig. 19.7 Very high resolution (1 m) image and Mangrove community zonation map for Maipura and Twin islands, west of Bhitarkanika national park, Kendrapara district of Odhisa, east coast, India

2000). Kanniah et al. (2015) have monitored mangrove status change over a period of 25 years in Iskandar Malaysia using Landsat- TM, ETM and OLI (Operational Land Imager) data. Giri et al. (2007) have used multi-temporal satellite data of 1973–2000 to monitor the status of mangroves of Sunderbans in Bangladesh and India. Aizpuru et al. (2000) have carried out global assessment of the cover change of mangrove forest using satellite images at medium to high resolutions. Reddy et al. (2007) have used remote sensing and GIS to monitor mangroves of Bhitarkanika wild life sanctuary, Odhisa state, east coast, India. Mangrove regeneration has been monitored using multi- spectral IRS- LISS III data in Gulf of Kachchh, Gujarat state in western India by Upadhyay et al. (2015).

Monitoring is done by acquiring EO satellite data of desired spatial resolutions at the required interval of time. Multi-date images are geo-referenced and co-registered to find out the spatial changes occurring in a particular landcover or landuse class. Details of the methodology for processing and analysis of multi-temporal satellite data for monitoring of mangrove forest is given in Fig. 19.8. Multi date satellite data are subjected to radiometric normalization and geometric correction.

Fig. 19.8 Methodology for monitoring of the status of mangroves using satellite data



For radiometric normalization the DN values of the images are converted to top-of-the-atmosphere reflectance values using the standard method (Nayak et al. 2003a, b). The geometric correction is carried out by geo-referencing the images using ground control points and all the images are assigned to a common projection system so that they could be mutually compared and analysed for change detection studies. The study area is then extracted from the satellite images. Mangrove mapping for each of the dates of the data acquisitions are done by analysing the satellite images using supervised classification technique (Joseph 2007; Nayak et al. 2003a). The images are classified into various classes present in the study area, such as, Mangrove Dense, Mangrove Sparse, Intertidal Mudflat, High-tidal Mudflat, Algae, Sand, Salt pan etc.

A few examples on monitoring of the status/condition of mangroves using EO satellite data are discussed below.

19.6.1 Marine National Park, West Coast of India

The Marine National Park and Sanctuary (MNP&S) is located along the southern shore of Gulf of Kachchh in the Jamnagar district of Gujarat, west coast of India (between 22° 15′N to 23° 00′N latitudes and 69° 00′ E to 70° 30′E longitudes) (Fig. 19.4). MNP&S is endowed with ecologically sensitive habitats such as mangroves, mudflats, coral reefs, sea grasses and sand dunes. It was established by a set of Government notifications during the period 1980–1982. The entire notified protected area comprises of 457.92 km^2 of Marine Sanctuary and 162.89 km^2 of Marine National Park. The MNP&S supports 215 species of molluscs including oysters, 3 species of sea turtles, 3 species of marine mammals (Dolphin, Porpoise and Dugong), 144 different varieties of fishes, 27 species of commercially important Prawns, 49 species of hard corals, 10 species of soft corals, 100 species of algae, 6 species of sea grasses along with few mangrove species comprise the biological wealth of this eco-region (Singh et al. 2006). Since 1991, mangroves and coral reefs have been provided extra protection under the 1991 Coastal Regulation Zone (CRZ) notification of the Government of India.

Mangroves of the part of the Marine National Park and Sanctuary, Gulf of Kachchh, Gujarat (Fig. 19.9) have been mapped and monitored using IRS P6 LISS-III (Linear Imaging Self Scanning-III) data of 02 March 2006, 28 December 2009 and 11 January 2011 (Fig. 19.10). In addition, ecological changes in mangroves and coral reefs have also been studied (Kumar et al. 2012b). Using the above multi-temporal satellite data and the methodology as given in Fig. 19.8 mangrove forests were mapped for each of the three dates for the study area (core Marine National Park, includes islands such as Pirotan, Jindra-Chhad, Mundeka Bet and Dideka Bet) extending from 69° 49′ to 70° 02′ E longitude and from 22° 37′ to 22° 29′ N latitude (Fig. 19.11).

The result, in terms of the classified output showing mangrove forests as well as other vital coastal habitats are given in Fig. 19.11a–c. Mangrove forests (dense and sparse) are shown in green colour. In a previous study, mangrove forests of the core marine national park were also mapped and monitored for the period 1975–2001,

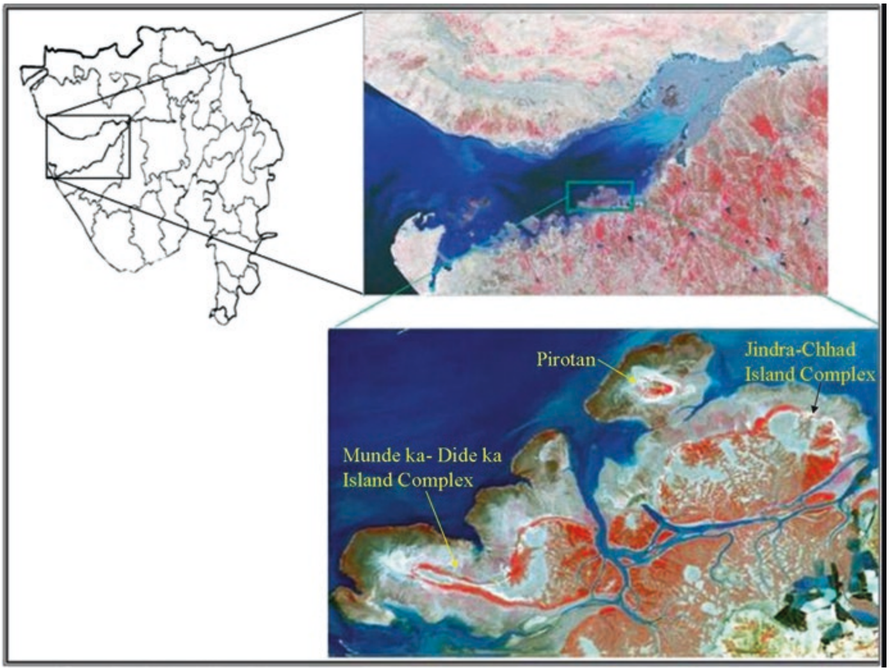


Fig. 19.9 Marine National Park and Sanctuary, Jamnagar, west coast, India

using multi-temporal satellite data and same methodology (Nayak and Bahuguna 2001; Bahuguna et al. 2007). Integrated output of the above two studies is shown in Fig. 19.12. It provides the area under mangrove forest cover during 11 different years, over a span of 36 years (1975–2011). It also provided the status of the mangrove crown density (dense and sparse) during each of the above 11 years. Significant changes in the mangrove forest area were observed during the period 1975–2011. Figure 19.12 shows that the degradation of mangrove forest continued from 1975 to 1985. This was, mainly, due to cutting of mangroves by local, people for fuel and fodder. Once the area was declared as marine national park in 1985, extensive conservation measures were taken by the state forest department. This had resulted in reversing the trend of degradation of the mangrove ecosystem after 1985 and helped towards restoring the environment till 1993. Degradation of mangroves after 1993 has increased due to anthropogenic pressure and oil spills. There has been sharp decrease in the area of, both dense and sparse mangroves, during 1975–1982. This reduction was attributed to the cutting of mangroves for fuel, grazing, mining and dredging, creation of salt pans on the coast, coastal development, oil and other chemical hazards, creation of harbours and ship breaking yards and waste disposal by steamers (Bahuguna et al. 2007). During 1985–1993, the area under mangroves increased and density also improved due to conservation measures taken after the notification of the area as marine national park. The condition of the mangroves deteriorated due to oil spill in October 1998 and the decline in the mangrove area

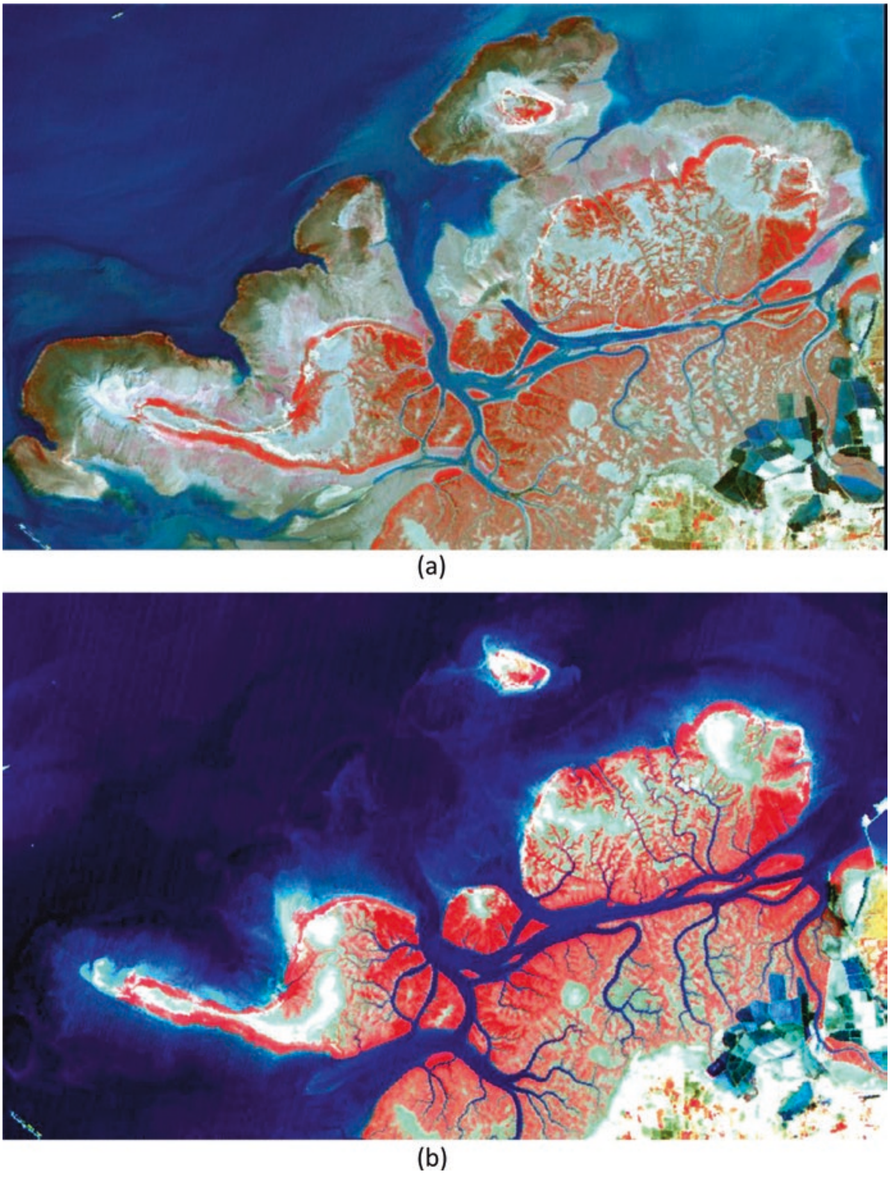


Fig. 19.10 IRS P6 LISS-III images of Marine National Park, west coast, India of (**a**) 02 March 2006, (**b**) 28 December 2009 and (**c**) 11 January 2011. Mangrove forests are seen in *red* colour

(c)

Fig. 19.10 (continued)

could be seen up to 2000. The condition of the mangrove could improve only after the year 2000. From the year 2001, the mangrove cover has been increasing (Fig. 19.12).

19.6.2 Anthropogenic Impact on Mangrove Forest: Coringa Wildlife Sanctuary, East Coast, India

Coringa Wildlife Sanctuary (CWS) is located in the reserved forest of Godavari delta, in the east coast of India (Fig. 19.4). This was declared as a wild life sanctuary in 1978. Fresh water inters the mangrove complex from the Godavari river, and its distributaries mainly Coringa and Gaderu. The mangrove formation at Coringa covers a total area of about 333 km^2 and comprises 32 species, of which ‘true mangrove’ (major and minor components) represent 15 species. Apart from this 11 species of mangrove associates and 6 species of salt marsh are also found. Monitoring of the Coringa mangrove reserve forest was carried out using satellite data of the years 1977 (Landsat MSS having 80 m spatial resolution), 1998 (IRS LISS-II, 36 m spatial resolution) and 2007 (IRS LISS-III, 23 m spatial resolution), 2011 (LISS III) and 2016 (LISS III). Mangroves seen in the 1977 image (Fig. 19.13a) at the periphery of the southern boundary of the reserved forest (boundary seen in green colour) have been cleared as seen in 1998 LISS II image (Fig. 19.13b). IRS LISS III image

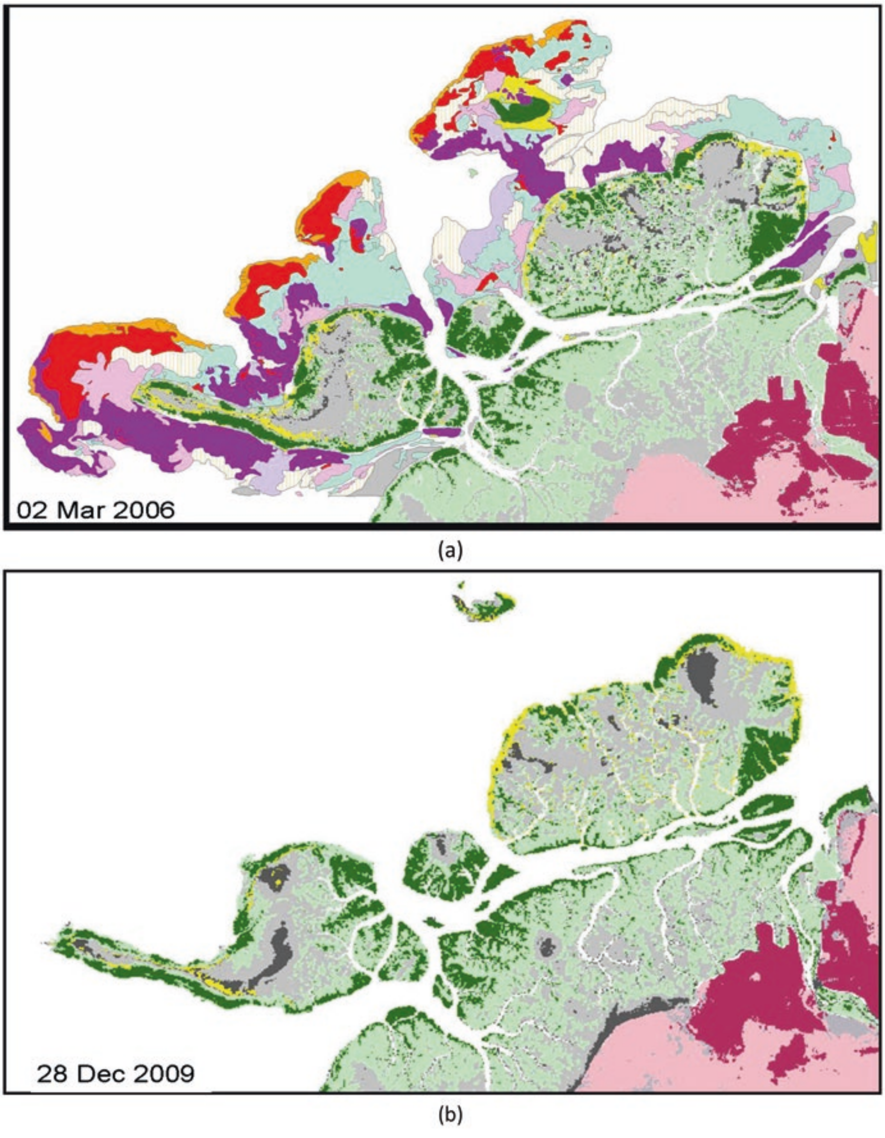
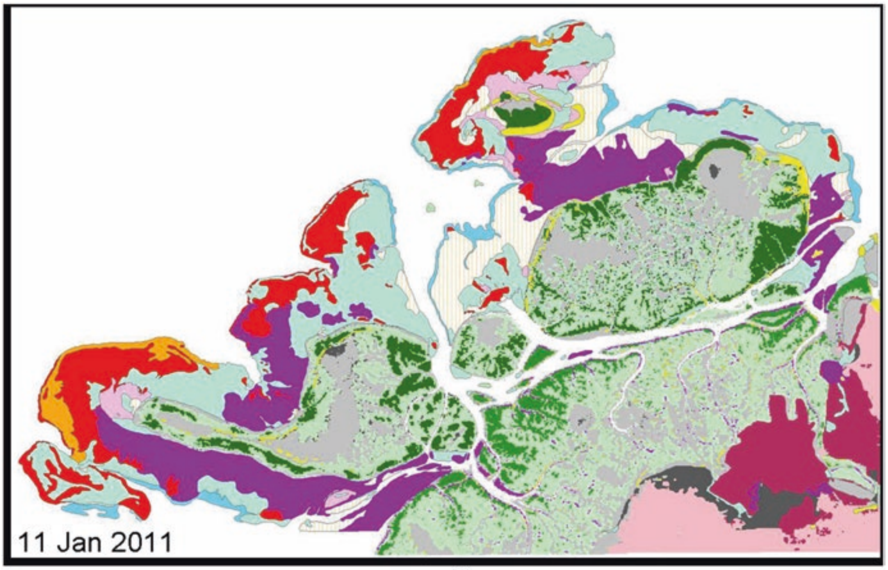


Fig. 19.11 Mangrove cover maps of core MNP, Jamnagar, west coast, India for (**a**) 2006, (**b**) 2009 and (**c**) 2011



(c)

Fig. 19.11 (continued)

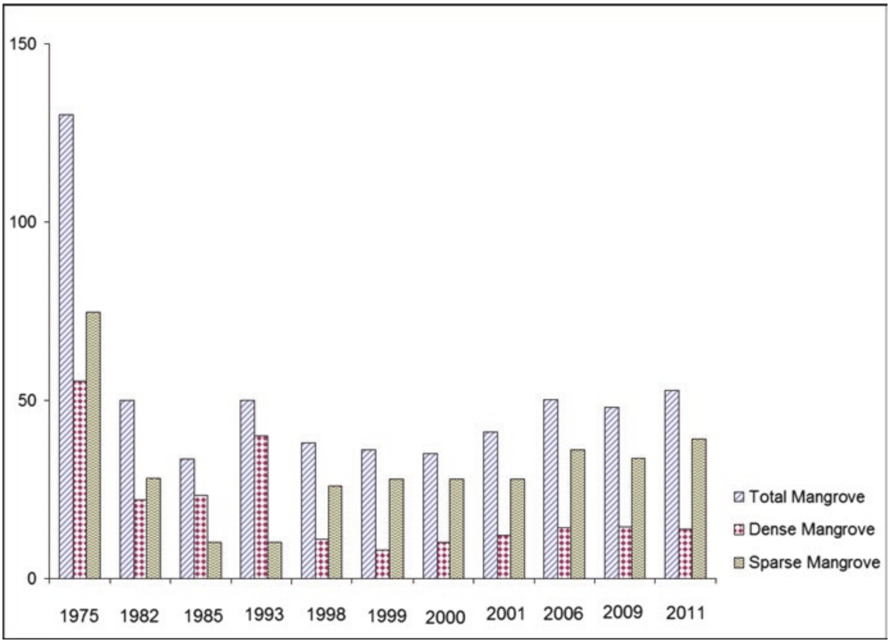


Fig. 19.12 Variation of mangrove cover including crown density (dense and sparse) in the core marine national park, Jamnagar, west coast, India during the period 1975–2011

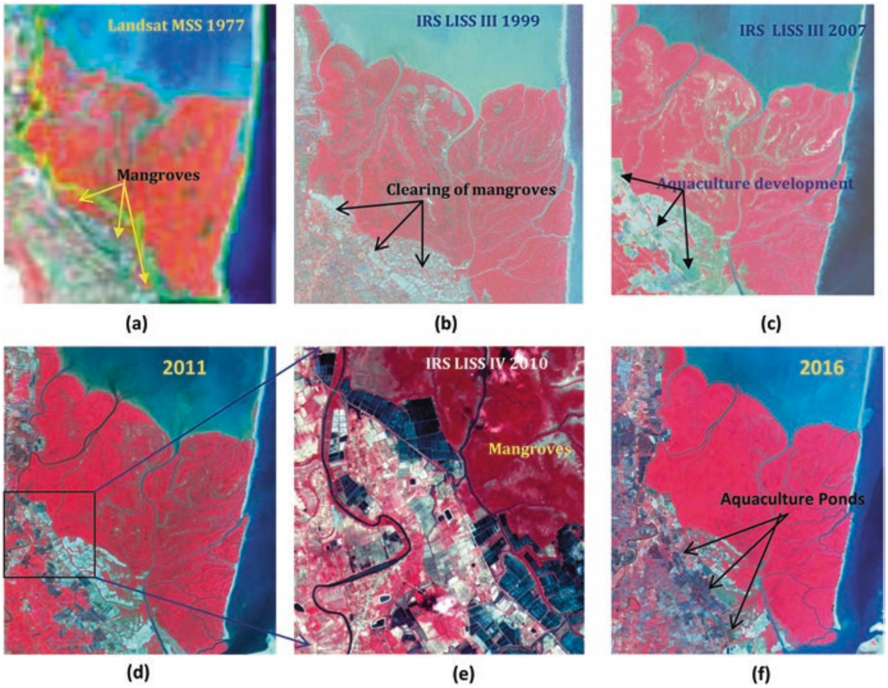


Fig. 19.13 Multi-date satellite images of Coringa Wildlife Sanctuary, East Coast, India, showing impact of aquaculture on mangroves

of 2007 (c) clearly shows the aquaculture ponds are being made in the reclaimed area. Subsequent LISS III images of 2011 (d) and 2016 (f) show further increase in the aquaculture ponds. Details of the aquaculture activities are seen in the high resolution LISS IV image of 2010 (Fig. 19.13e). This is an example of the degradation of the mangrove forest because of the anthropogenic interventions.

19.6.3 Impact of Sediment Deposition on Mangroves in Indus Delta

As discussed earlier, excess sediment input to mangroves may lead to its degradation. We will discuss one specific example on monitoring the degradation of mangrove forest due to excess sand deposition. This example is from the Indus river delta. Multi-temporal satellite data have been used to monitor the status of mangrove forests. Figure 19.14 shows the Indus delta as captured on IRS AWiFS image as well as the study area. The study area is a part of the lower Indus delta. The region of lower Indus Deltaic plain, situated on the west of Great Rann of Kachchh, and north-west of Gulf of Kachchh in Gujarat state of India, is known as Kori Creek.

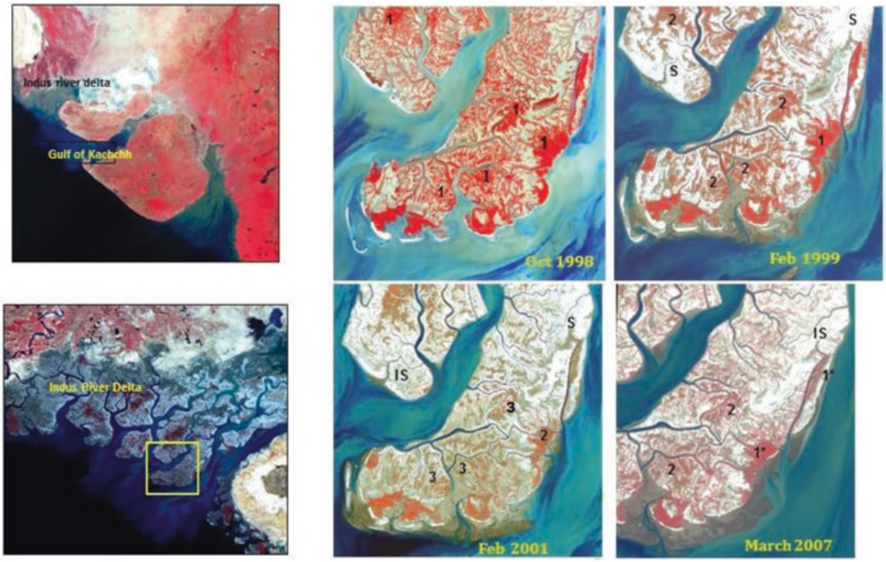


Fig. 19.14 Monitoring the degradation of Mangroves due to sand deposition in Kori Creek, Indus river delta, Gulf of Kachchh, west coast of India

Indus is 21st largest river of the world in terms of annual flow. It covers a distance of 2880 km before draining in to the Arabian Sea. It forms a huge triangular fan-shaped delta covering an area of about 6000 km^2 and comprises of 17 major creeks and many minor creeks and vast expanses of mudflats and large number of patches of fringing mangroves. The mangrove ecosystem of Indus delta is unique because it constitutes the largest area of arid climate mangroves in the world. Mangroves, in this delta grow in an environment characterized by low rainfall, highly variable seasonal temperature and high rate of evapotranspiration. The diversity of mangroves in this region is low, comprising mostly of *Avicennia marina*, with smaller patches of *Rhizophora mucronata, Rhizophora apiculata, Acanthus ilicifolius* and *Ceriops tagal* (Nayak and Bahuguna 2001). Due to very less rainfall, mangroves in this region depend, mostly, on the Indus River for fresh water supplies.

One of the major natural problems which often lead to the degradation of mangroves around Kori creek is the sedimentation and erosion. Sand gets deposited in the mangrove area, leading to degradation of mangroves.

Mangroves flourish on sedimentary shorelines where large rivers discharge on to low gradient coast. Despite this, excess inflow of sediments to mangrove areas may cause its degradation. Aerial roots are major adaptation of the majority of mangrove species to the environmental stresses in the intertidal zones, allowing root respiration in anaerobic waterlogged soils. These roots may differ in architecture between species, from tall proproots in *Rhizophora* to lower pneumatophores in *Avecinnia* and *Sonneratia*, knee roots in *Bruguier*a and *Xylocarpus mekongensis*, buttresses in *Ceriops* and plank roots in *Heritiera* and *Xylocarpus granatum*. The importance of aerial root architecture in determining tolerance thresholds of mangrove species to sediment burial has been studied and discussed in detail by Ellison (1998). As per his study, the impact of sedimentation ranged from reduced vigour to death of mangrove trees, depending on the amount and type of sediments and the species of mangroves involved.

When pneumatophores of *Avecinnia* or *the knee roots of Bruguier*a are completely buried, the mangrove trees usually die. When the sediment deposition is less than the height of the aerial roots of these mangrove species, the tress can still be adversely affected or die (Allingham and Neil 1995; Lee et al. 1996; Ellison 1998).

The ability of mangrove species to cope with the root burial (due to sediment deposit) varies between species as a function of their root architectures. The pneumatophores of *Avecinnia* and *Sonneratia* and knee roots of *Bruguier*a and *Xylocarpus mekongensis* may be able to extend upward (Hutchings and Saenger 1987). However, these responses take time, but mangrove tree can adjust to gradual burial.

Response of mangrove species to root burial is not uniform. It depends on various factors such as, sediment texture, the presence of soil fauna and the tidal range. Burial by sand does not appear to cause death of trees as readily as by silt or clay. This could be because soil aeration is better through sand (Ellison 1998).

IRS LISS-III images of October 1998, February 1999, February 2001 and March 2007 pertaining to Kori creek are given in Fig. 19.14. Excess deposition of sand over the mangrove area (due to some very specific coastal process) started sometime in the beginning of November 1998. IRS LISS-III image of October 1998 (prior to sand deposition) shows patches of dense mangroves (indicated as 1, on the image) and sparse mangroves (indicated as- 2). LIS-III image of February 1999 (post event) clearly shows the sand deposition (seen as white patches on the image, also marked as 'S') and mangrove degradation has also taken place: dense mangrove patches (marked as 1 in Oct 1998 image) has changed to sparse mangrove class ('2', in Feb 1999 image). Degradation of mangroves continued beyond Feb. 1999 due to sand deposition. It is clearly seen on the LISS-III image of February 2001 (Fig. 19.14) that the mangroves have further degraded. The patches of sparse mangroves ('2' on Feb. 1999 image) has changed to the degraded mangrove category ('3') and sand cover has further increased in Feb 2001 image (indicated as 'IS' – increased sand). Mangrove-1 in Feb 1999 image has changed to mangrove-2 category in Feb 2001 image. IRS KLISS-III image acquired after another 6 years (March 2007) indicates improvement in the mangrove cover density (indicated as '1*') at few places, as compared to Feb 2001. Change in mangrove-3 to mangrove-2 class has also been observed in the March 2007 image.

19.7 Mangrove Health Assessment

We need to understand the meaning and concept of the "Ecosystem Health", before getting in to the real issues related to the health assessment of mangrove ecosystem. Several definitions of 'ecosystem health' exist as of today. Among these, one of the most popular working definitions of ecosystem health (Costanza et al. 1992) is "An ecological system is healthy and free from distress syndrome if it is stable and sustainable – that is, if it is active and able to maintain its autonomy and organization over time and is resilient to any kind of stress". Ecosystem health is linked to the concept of sustainability, which is seen to be a comprehensive, multi-scale, dynamic measure of system resilience, organization and vigour. According to this definition, an unhealthy system is one that is not sustainable and will eventually cease to exist with time, unless appropriate corrective measures are taken well in time.

Health is a measure of overall performance of a complex ecosystem that is built up from the behaviour of its various components. Such measures of system health imply a weighted summation or a more complex operation over the component parts, where the weighting factors incorporate an assessment of the relative importance of each component to the functioning of the whole system. This assessment of relative importance incorporates "values", which can range from subjective and qualitative to objectives and quantitative as we gain more knowledge about the system under study.

This diversity of definitions arises from the fact that different workers have given importance to different factors that impact an ecosystem health. These definitions can be grouped into the following major categories:

Health as homeostasis
Health as the absence of disease
Health as diversity or complexity
Health as stability or resilience
Health as vigour or scope of growth
Health as balance between system components

Same issues arise when we try to arrive at a comprehensive definition of "mangrove health". Each of the aspects has to be considered to arrive at a definition that meets the end user's requirement. To develop a "mangrove health assessment model", the first step is to identify environmental indicators of the mangrove ecosystem health that need to be included in the multi-parametric health model. The choice of these indicators and the weighting factors to be assigned to each of them, may differ from one mangrove zone to the other depending on the biological, ecological, environmental and geomorphic setup. As mentioned earlier, the choices of indicators and their weighting factors could be subjective as it may depend on the researcher's perceptions, understanding and knowledge about the particular mangrove forests, including its biological, ecological, environmental and geomorphic settings.

Monitoring the health of mangrove ecosystem has also been attempted using satellite derived vegetation indices. One of the most popular and widely used vegetation indices is NDVI (Normalized Difference Vegetation Index). It is defined as the normalized ratio of the spectral reflectance measured in red and near infra-red (NIR) wavelength regions {NDVI = (NIR-RED)/(NIR + RED)}.NDVI is, basically, determined by the degree of absorption by leaf chlorophyll in the red wavelength, which is proportional to leaf chlorophyll density, and by the reflectance in NIR wavelength region, which is proportional to green leaf density. NDVI has been found to be sensitive to the green leaf area and green leaf biomass. The vegetation indices such as NDVI, derived from the multispectral satellite data, represent the vegetation vigour/canopy density and thus can be used as one of the indicator for the ecosystem health. Chellamani et al. (2014) have used NDVI, derived from SPOT-VGT products, to characterize and monitor the health of Indian mangroves. Kovcas et al. (2005 have assessed and mapped leaf area index of mangroves at the species level using IKONOS satellite data. As leaf area index is related to canopy density and green biomass, it can also be used as one of the indicator of the mangrove health.

The above health models based on a single parameter such as the vegetation vigour/canopy density (represented by NDVI) are not robust and may not work in certain conditions, specially, when mangroves are stressed due to anthropogenic or natural causes. On the other hand multi parametric health models, accounting for the mangrove vigour/canopy density, the weather, hydrology, stress factors, environment etc. may be more robust. Such multi parametric 'Mangrove health assessment model' has been developed for the selected mangrove zones of India (Ajai et al. 2012). This model uses a number of environmental indicators which are generated using both, satellite data and field survey/measurements. In this study, selection of the environmental indicators and their weighting factors (for the model) was done through brain storming discussions, organized by Space Applications Centre, Ahmedabad, India. Experts from all over India participated in the brain storming discussions. In addition to identifying the environmental indicators of the mangrove habitat health, the 'frame work' for the model was also finalized to assess the health and mangroves-at-Risk using remote sensing and Geographic Information System (GIS) techniques. Based on the detailed deliberations by the experts, the environmental indicators and their relative weightages, as finalized by the experts, are given in the Table 19.4. As the health status of mangroves is to be given in a spatial domain, these environmental indicators (for the mangrove forest) are to be generated in the form of spatial thematic layers.

Table 19.4 Environmental indicators and their weightages

Sr. No.	Parameters	Weightage
1	Canopy cover	12
2	Floral diversity	11
3	Obstruction to natural flow	10
4	Drainage density	9
5	Natural regeneration	8
6	Anthropogenic stress	7
7	Stand size/fragmentation	7
8	Change in vegetation pattern	6
9	Tree height	5
10	Defoliation	4
11	Erosion/accretion	3
12	Sedimentation pattern in water ways	2

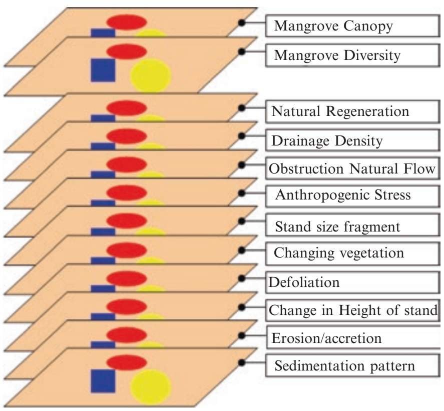
The Model Framework

The conceptual frame work of the multi-parameter mangrove health model is given in Fig. 19.15. Spatial thematic layers for each of the above mentioned 12 environmental indicators (Table 19.4), are to be created for the study area and integrated in GIS environment by giving proper weightages. The model (Fig. 19.15) followed the cell-based grid analysis for assessing mangrove health. The entire study area is divided into grids of 1 ha (100 × 100 m) size. This forms the master grid. Master grid is then overlaid (intersected) with each of the above mentioned input thematic layers. The information in each grid pertaining to all the themes are transferred into a column in the master grid.

For each of the input thematic layers, different classes in that theme are given a value (rank) in the range 0–100 based on its importance in determining mangrove health. As mentioned earlier, each input thematic maps are assigned a weightage (Table 19.4). The values in each column (representing each theme) are then used to generate the final Mangrove Health Index. For each grid, the value of the class (rank) in a particular theme is multiplied by the weightages for that theme. Product of rank and weightages for each of the 12 themes are summed and the final value is then divided by the sum of all the weightages (i.e. 84, Table 19.4) to generate mangrove health index value for that particular grid.

The criteria given in the Table 19.5 are applied to the computed values of 'Mangrove Health Index' (MHI) to provide the health status of the mangroves.

Mangrove Health Assessment Model



Model is based on the indicators-approach.
Twelve indicators of Mangrove health have been used in the model.

SN	Parameters	Weightage
1	Canopy	12
2	Floral diversity	11
3	Obstruction to natural flow	10
4	Drainage density	9
5	Natural regeneration	8
6	Anthropogenic stress	7
7	Stand size/fragmentation	7
8	Change in vegetation pattern	6
9	Height	5
10	Defoliation	4
11	Erosion/accretion	3
12	Sedimentation	2

Map of each parameter (Rank) x weightages / sum of weightages (84)* = MH
The sum of weightages is done here to bring the values in the range of 1-100

MH =

Mangroves in pristine health	**80.1-100**
Mangroves vulnerable to degradation	**40.1-80**
Mangrove degrading	**10.1-40**
Mangrove degraded	**1-10**

Fig. 19.15 Conceptual frame work of Mangrove Health Assessment Model (Ajai et al. 2012)

Table 19.5 Criteria for mangrove health assessment

Sr. No.	Mangrove health category	MHI values
1	Very good/in pristine health	80.1–100
2	Good health	40.1–80
3	Vulnerable to degradation	10.1–40
4	Degraded	01–10

The details of the values (ranks) for each of the classes of the 12 themes (Environmental Indicators) that are used in computing the Mangrove Health Index (MHI) are discussed below.

Theme 1: Canopy Cover

Canopy cover density layer is generated from the mangrove density map. This thematic map is intersected with the master grid map (100 × 100 m) and the corresponding values are transferred to the grid under a new field (column). For all the other categories (non mangrove classes) the value of the grid for this theme is give zero value. The values for different classes are assigned as per details given as under:

Sr. No	Category	Value
1.	Very dense mangroves	100
2.	Dense mangrove	75
3.	Sparse mangrove	40
4.	Degraded mangrove	10
5.	All other categories	0

Theme 2: Floral Diversity

The mangrove community zonation map serves as an input for this theme. Sometimes it is not possible to identify all species in satellite data, so field information collected is also used. Grids are identified using GPS locations. The values for the theme are based on the following criteria.

Sr. No	Category	Value
1.	More than two mangrove species	100
2.	Two mangrove species	80
3.	Less than two mangrove species	60
4.	Less than two other species (associated species)	40
5.	Two other species (associated species)	20
6.	More than two other species (associated species)	10

Theme 3: Tree Height

In the case of mangroves in India, the average height of major mangroves of Godavari (4.5 m), Krishna (4.5 m), Bhitarkanika (6 m), Mahanadi (6 m), Andaman and Nicobar (7 m), Muthupet and Pichhavaram (5–6 m), and Gujarat (2–3 m) are considered as the height of healthy mangroves in India. The following table indicates the health of the mangroves trees related to their height. They have been assigned the respective values. However, the values assigned to different height classes will vary from one mangrove zone to the other depending upon the mangrove the ground conditions. For this theme, the mangrove density map and field data have been used as an input. Extrapolation of field data is done on a grid to generate this theme. The values assigned to different height classes are as follows:

Sr. No.	Category	Tree height	Value
1.	Healthy	>3	100
2.	Moderately healthy	1.5–3.0 m	75
3.	Moderately unhealthy	1.0–1.5 m	50
4.	Unhealthy	<=1 m	25

Theme 4: Defoliation

Leaf shedding, except for *E. agallocha* and *Xylocarpus* species in summer (or during the leaf shedding season) is considered as unhealthy. As the above two species are not found in the study area, any kind of defoliation is to be considered unhealthy. The values assigned to different classes of this theme areas under:

Sr. No.	% Unseasonal defoliation	Category	Value
1.	0	Healthy	100
2.	1–10	Moderately unhealthy	60
3.	>10	Unhealthy	10

In case no defoliation is found in a grid, that particular grid is considered as healthy and the theme value is given a value 100.

Theme 5: Obstruction to Natural Flow of Tidal and Freshwater

The mangrove density map serves as an input for this theme. Formation of sandbars across river mouths or estuaries/formation of sandbars or deposition across rivers or creeks and canals as well as abrupt change in water colour from black or dark blue to light blue (barrages) are all considered unhealthy. Based on the area of mangroves that they influence and the time duration for which they are found, the following values are assigned:

Sr. No.	Criteria	Value
1.	Obstruction in 1 ha area or more for more than or equal to 6 months	20
2.	Obstruction in 1 ha area or more for less than 6 months	40
3.	Obstruction in less than 1 ha area for more than or equal to 6 months	60
4.	Obstruction in less than 1 ha area for less than 6 months	80
5.	No obstruction	100

This theme is calculated by assigning the above values to each polygon of the canopy map by visually observing the obstruction and the extent of influence of such obstruction on the vegetation.

Theme 6: Drainage Density

The creek network map serves as an input for this theme. This map can be very well prepared from the high resolution satellite data (e.g.: LISS-IV, SPOT). Based on the percentage area covered by the creeks/drainages in a grid, the values are assigned as given in the following table:

Sr. No.	% Creek area	Value
1.	More than 20%	100
2.	10–20%	80
3.	5–9.9%	60
4.	1–5%	40
5.	Less than 1%	20

Theme 7: Sedimentation Pattern in Waterways

Very high turbidity has been considered to be unhealthy for the mangroves. Very less or no turbidity is considered as healthy. The values assigned to each of the classes under this theme is given as under:

Sr. No.	Criteria	Value
1.	Highly turbid	20
2.	Moderately turbid	70
3.	Less turbid	100

Satellite images are used to create a map of water bodies which are then classified based on their turbidity. This spatial layer is the intersected with the grid map and finally the 'values' are assigned to each grid using above table to generate the above mentioned sedimentation pattern map.

Theme 8: Natural Regeneration

Presence of 1 seedling per square metre (10,000 seedlings per ha) immediately after the monsoon season can be considered as 100% success in natural regeneration. This is calculated based on literature which indicate presence of 11,000–17,000 seedlings of various species per ha in healthy mangrove ecosystem. The values for these themes are assigned based on the post monsoon field observations. These values have been indicated in the following table. If natural regeneration is observed in a grid, the value assigned to the grid is 100 otherwise the value is 10.

Sr. No.	Criteria	Health	Value
1.	Areas with natural regeneration	Healthy	100
2.	Areas with no natural regeneration	Unhealthy	10

Theme 9: Anthropogenic Stress

There are a variety of human actions which may create stress and affect the mangrove health, directly or indirectly. The major anthropogenic actions which may create stress to mangrove health are given below in the table. Multi- temporal satellite data can be used to identify and map these anthropogenic activities and their impact on mangroves. Weightage for these stresses are calculated according the criteria provided as under:

	Anthropogenic stresses:	Weightages
a.	Pollution – discharge/oil spill	5
b.	Dredging	4
c.	Embankment	10
d.	Grazing in the mangrove patch	8
e.	Grazing near the mangrove patch	6
f.	Direct cutting of mangroves	10
g.	Reclamation of mangrove forest	10
	(Aquaculture/salt pans/residential/any other developmental activities)	

Classification Criteria

1. Distance from the stress (More the distance from the mangrove stand less is the stress)

The values, based on the distance of the ‘stress’ (includes habitation) from the mangrove stands are calculated as per the following criteria:

100 m or less	10
101–500 m	20
501–1 km	40
1–2.5 km	60
2.5–5 km	80
>5 km	100

2. Population: Values assigned to different population density classes are given below:

<100	100
101–500	80
501–1000	60
1001–5000	40
5001–10,000	20
>10,000	10

Average of [Map of individual anthropogenic parameter (based on the distance from the stress) + Map of Distance from the habitation + Map of Population density] × weightage of each anthropogenic parameter

(a) **Pollution**

No pollution	100
500 m or more is the distance of the source from the mangrove stand	60
100–500 m or more is the distance from the mangrove stands	30
Less than 100 m or more is the distance from the mangrove stands	10

(b) **Reclamation of mangrove forest** (aquaculture/salt pans/residential/any other developmental activities)

Presence/Absence. Reclamation is High stress (10). No reclamation means no stress (100)

(c) **Embankment**- Presence of embankment in a creek which is the only source of freshwater for a patch will be unhealthy – 10

Presence/Absence. Presence is High stress (10). Absence is no stress (100)

(d) **Grazing in the mangrove patch**

Presence/Absence. Presence is High stress (10). Absence is no stress (100)

(e) **Grazing near the mangrove patch**

Presence/Absence. Presence is High stress (10). Absence is no stress (100)

(f) **Direct cutting of mangroves**

Presence/Absence. Presence is High stress (10). Absence is no stress (100)
Single value based on above (can be entered as a field in the 1 ha grid shape file)

(g) **Dredging**

Dredging is similar to embankment. Presence of dredging in a creek will be unhealthy – 10

Theme10: Stand Size/Fragmentation of the Stand
It includes the following two components which need to be separately calculated and the put as an input into the total theme value.

(a) Change in the size of the stand
(b) Percentage of area occupied by saline blanks

Stand Size
It is based on the in mangrove stand size between two time frames. It can be computed using satellite data of two time frame. Values are assigned based on the following criteria:

Change in the size of the stand	
No change	100
1–5%	80
5.1–20%	60
20–40%	40
More than 40%	20

Saline Blank
Saline blanks can be mapped using high resolution satellite images supported with ground truth. The values for this theme are computed on the basis of percent of saline blanks in a grid, using the following criteria:

Percentage of area occupied by saline blanks	
No saline blanks	100
1–5%	80
5.1–20%	60
20–40%	40
More than 40%	20

Theme 11: Changing Vegetation Pattern
The input remains similar to the stand size and the following are the values for the classes under this theme:

(i) Mangrove to non-mangrove – UH	10
(ii) Mangrove to halophyte – MUH	40
(iii) Halophyte to mangrove – MH	60
(iv) Non-mangrove to mangrove – H	100
(v) No change in mangroves H	100

Theme 12: Erosion/Accretion

Erosion/accretion of mangrove stands along shore/river banks can be mapped using mult- date satellite images. Erosion of mangroves (tide dominated mangroves) over a period is treated as unhealthy and accretion of mangroves along the shore is healthy. Accretion of sand inside mangrove stand is unhealthy. The values for this theme are calculated as per the details given as under:

Erosion	10
Accretion inside mangrove stands	20
Accretion along shore	80
No erosion	100

All the above 12 themes are integrated in GIS environment (Fig. 19.15) to compute Mangrove Health Index (MHI) and to finally create the Mangrove Health Maps.

19.7.1 Case Study: Mangrove Health Assessment in Pichhavaram Reserve Forest, India

Pichhavaram mangrove forest, situated at the east coast of India, is one of the two major mangrove wetlands having richest mangrove diversity in Tamil Nadu state, southern India. True mangrove species such as *Rhizophora apiculata, Rhizophora mucronata, Avicennia marina, Avicennia officinalis, Exocecaria agallocha* and associate mangrove species like *Sueadamaritima, Salicornia and Sueadamonoica* are largely found in Pichavaram wetlands. The wetlands act as the nursing ground for a variety of fishes and marine organisms such as mollusks, crabs, prawn, bivalves etc. The wetland lies in the centre of the Coleroon-Vellar estuary in the east coast and hence fed by sufficient amount of fresh water in monsoon season. In summer, tidal influence is more than the freshwater flow. Many natural creeks and canals that have tidal diurnal flow inundating the mangrove vegetation also feed the wetland. There are about 12 hamlets surrounding the wetlands where people depend on mangroves for their livelihood such as fishing, firewood, grazing and fencing. In 1990s the restoration of degraded mangroves of Pichhavaram carried out by MS Swaminathan Research Foundation (MSSRF), Chennai and State Forest Department led to the increase in mangrove area from 280 ha in 1987 to 820 ha in 2006. The devastating tsunami which hit on Tamil Nadu coast on December 26, 2004 made the local community to realize the importance of mangroves in protecting them from the natural hazards. After that, the dependence on green mangroves for grazing and firewood collection has reduced owing to the participation of community in restoration of mangroves.

Multi- temporal data from IRS satellite have been used in assessing the health of mangroves. The oldest satellite image available for this study area is of 1990. Post

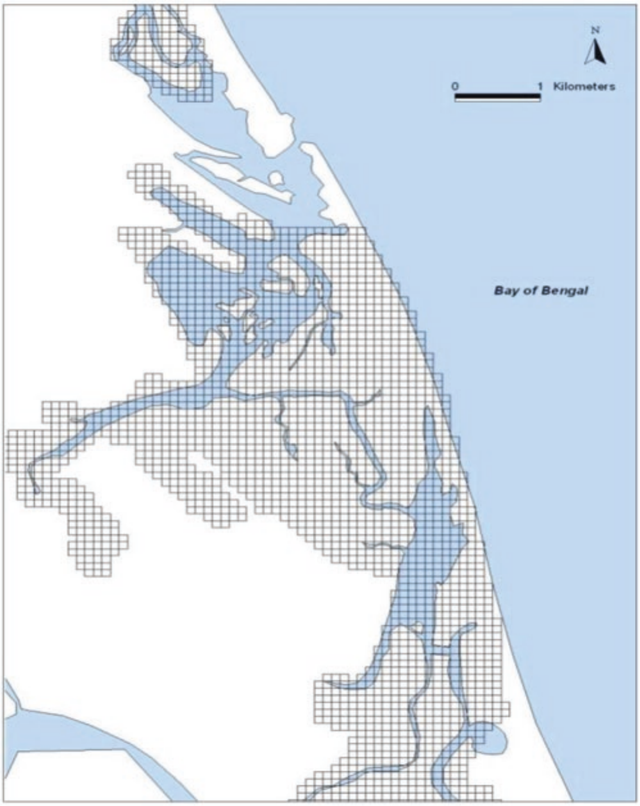


Fig. 19.16 Shoreline map of Pichhavaram reserve mangrove forest, east coast of India, overlaid with 1 ha grids

monsoon and pre monsoon data of 2005, 2006, 2007 and 2008 have been used to assess the mangrove health. Primary data such as vegetation field survey in mangrove forests using sample survey and track survey, presence of anthropogenic pressure (pollution, dredging, grazing, reclamation etc) using reconnaissance survey and the location of user villages of the mangroves were collected and used (Ajai et al. 2012).

The methodology as discussed above has been used to generate the spatial thematic layers for each of the 12 environmental indicators (as given in Table 19.4) for the study area. The details on the computation for each of the12 themes, for Pichhavaram reserve forest are summarized below. These theme maps were prepared using multi date satellite data as well as field surveys/measurements (Ajai et al. 2012).

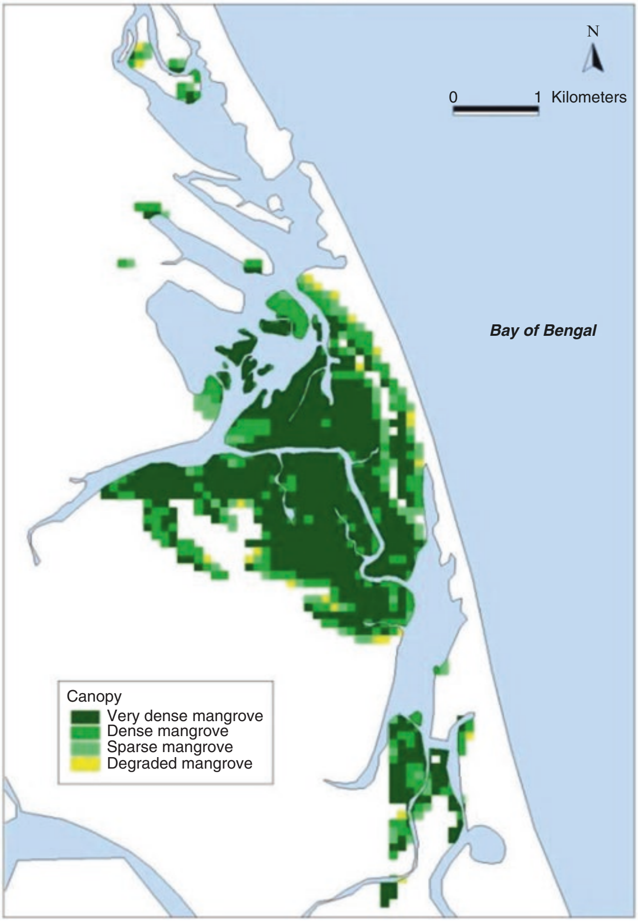


Fig. 19.17 Mangrove canopy map of Pichhavaram reserve mangrove forest, east coast of India

(i) **Canopy cover**: It is mapped using standard methods of analyzing remote sensing data of pre and post-monsoon
(ii) **Floral diversity:** Mangrove community zonation has been done using remote sensing data and is supported by species count by quadrant or PCQ method in single observation
(iii) **Obstruction to natural flow of tidal and freshwater:** This is mapped using pre and post monsoon satellite data for the presence of any obstruction across waterways

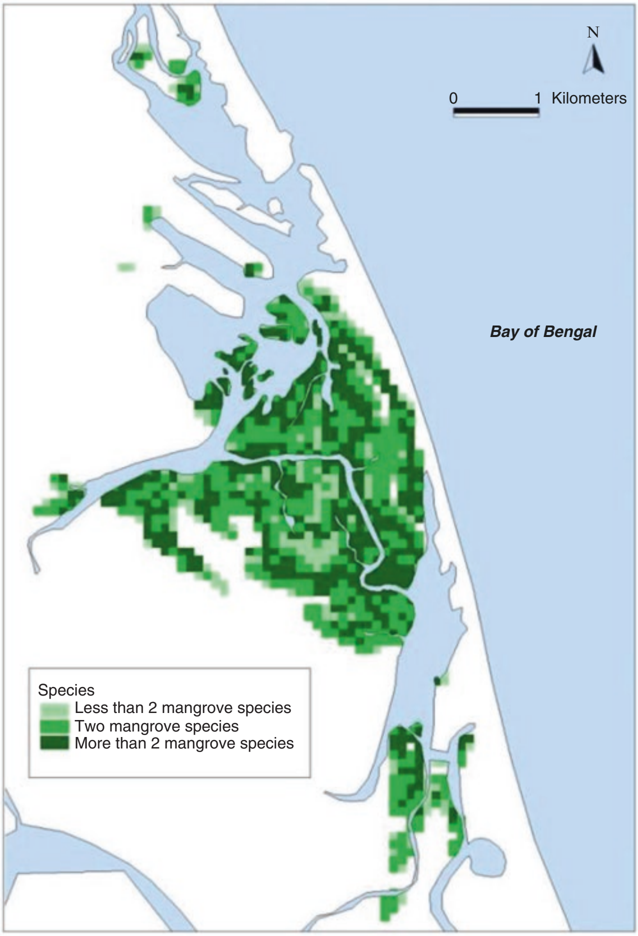


Fig. 19.18 Mangrove community zone map of Pichhavaram reserve mangrove forest, east coast of India

(iv) **Drainage density/Running water spread**: Percentage area of running water near mangroves is calculated using the area covered by creeks and canals network based on satellite images and field measurements.

(v) **Natural regeneration**: Field observations in the post monsoon season conducted to map the natural regeneration in the wetland

(vi) **Anthropogenic stress**: Information on pollution, dredging, grazing, reclamation, population pressure, etc. have been obtained through field data collection

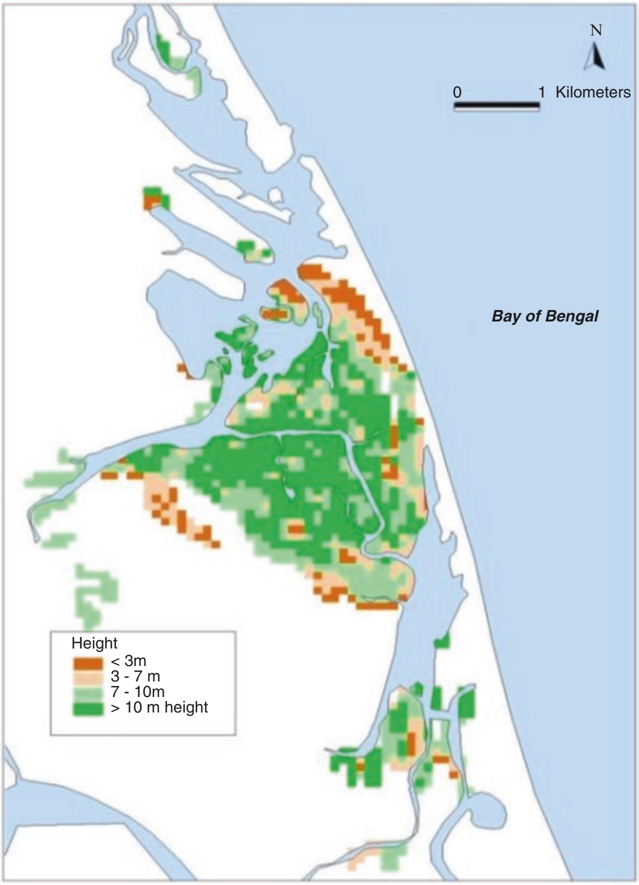


Fig. 19.19 Height of mangrove trees in Pichhavaram reserve mangrove forest, east coast of India

(vii) **Stand size/ fragmentation**: Change in stand size/fragmentation in mangrove vegetation have been mapped using multi-temporal temporal satellite data

(viii) **Change in vegetation pattern**: Changes such as mangrove to mangrove, mangrove to halophyte, mangrove to non-mangroves were mapped using multi-temporal satellite data.

(ix) **Tree height**: The average vegetation stand height is measured through field observation using site-specific standard methods.

(x) **Defoliation**: Pre and Post-monsoon satellite data have been used to map the defoliation including unseasonal shedding of *Excoecaria agallocha* and *Xylocarpus* species

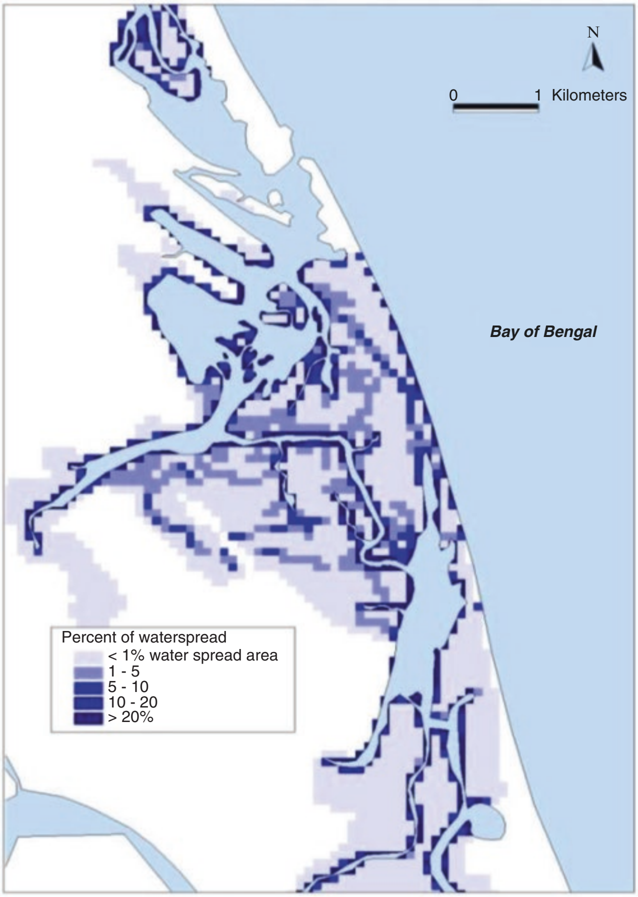


Fig. 19.20 Drainage and water spread map of Pichhavaram reserve mangrove forest, east coast of India

(xi) **Erosion/r accretion**: Multi- temporal high resolution satellite data have been used to map erosion/accretion

(xii) **Sedimentation pattern in waterways**: Pre and post monsoon satellite data and field observations have been used to map the sedimentation along the rivers, lagoons, creeks and channels and other open water bodies.

As Pichhavaram mangrove is a protected area, it has no influence of anthropogenic pressure, sedimentation and dredging inside the wetland area, the indicator values are considered as 100 throughout the wetland grids. Similarly, natural regeneration is noticed in all mangroves area after monsoon period. So this indicator value is also considered as 100 for entire wetland.

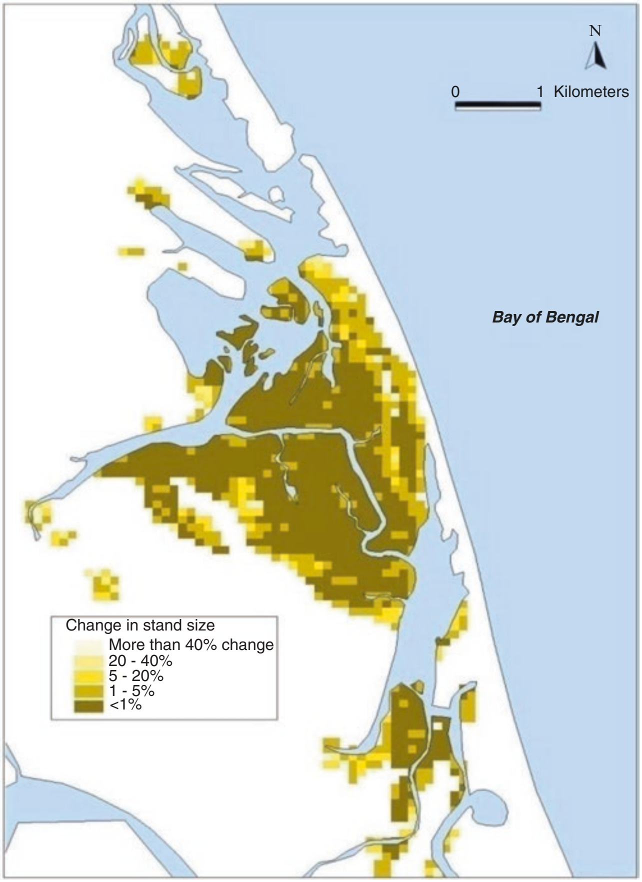


Fig. 19.21 Map of change in -stand size of Pichhavaram reserve mangrove forest, east coast of India

Thematic maps representing each of the indicators (themes) for the Pichhavaram mangrove forest were generated using the methodology described above. These maps were then overlaid/intersected with the master grid of size 1 ha. The computation of the grid values for each of the indicators is done by area weighted method instead of taking a simple average of values assigned to all the classes present in that particular grid. The weightages for each of the themes (Table 19.4) were used along with the rank 'values' as described earlier and integrated in GIS environment to compute the grid-wise "Mangrove Health Index (MHI)" for the study area.

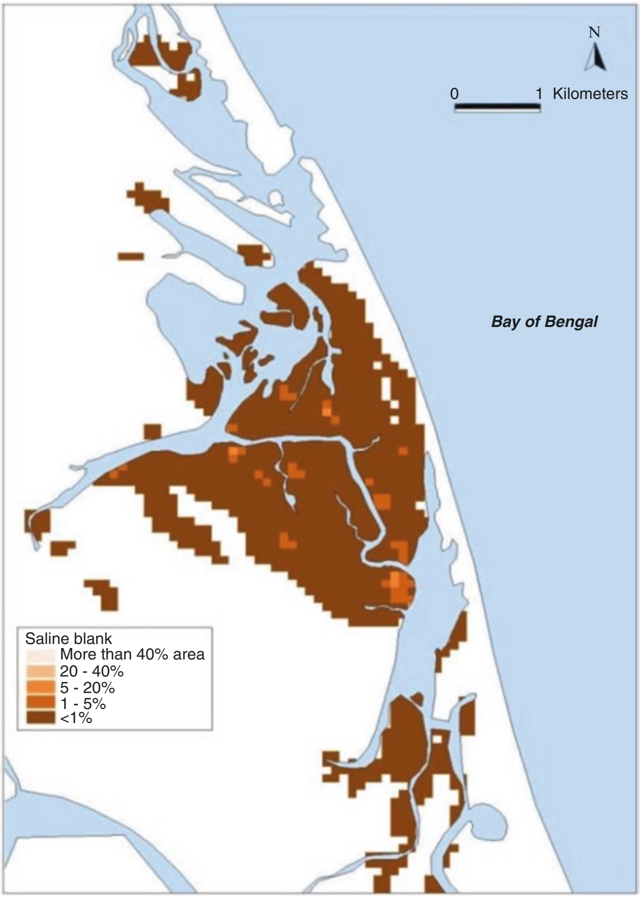


Fig. 19.22 Presence of saline blanks of Pichhavaram reserve mangrove forest, east coast of India

The indicator/theme maps (with grid size of 1 ha) for the Pichhavaram reserve forest are given in Figs. 19.16, 19.17, 19.18, 19.19, 19.20, 19.21, 19.22, 19.23, 19.24, and 19.25.

As described earlier, the mangrove health index (MHI) is calculated for each grid and the MHI map for the Pichhavaram mangrove forest is shown in Fig. 19.26.

"Mangrove Health Index (MHI)" values in the range 1–10, 10–40, 40–80 and 80–100, represent mangroves in degraded, vulnerable to degradation, good health and pristine/very good health conditions respectively (Table 19.5). In Fig. 19.26,

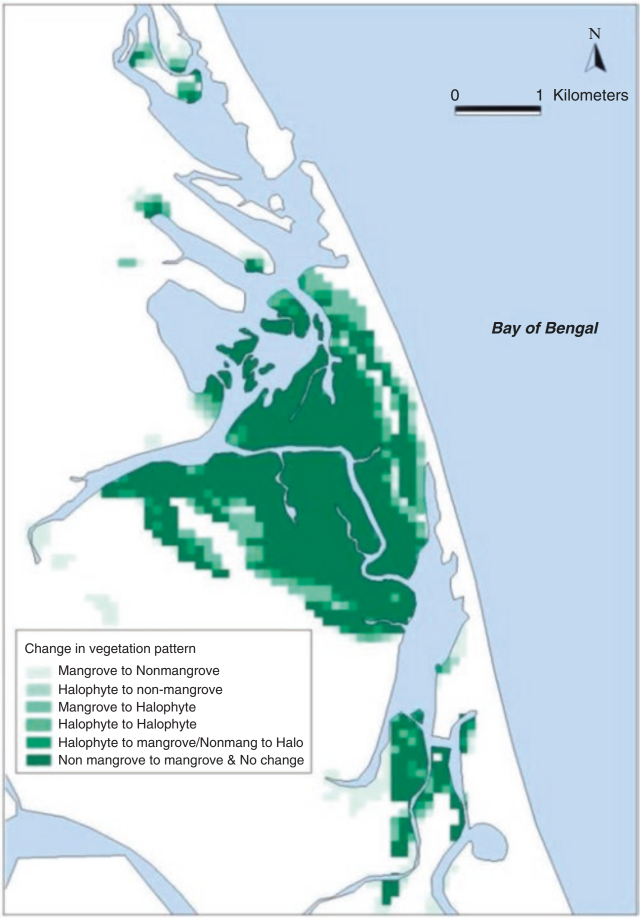


Fig. 19.23 Change in Vegetation pattern of Pichhavaram reserve mangrove forest, east coast of India

MHI (computed on 1 ha grids) are shown in four different colours, representing the above mentioned four health classes.

It is clear from the "Mangrove Health Index" map of Pichhavaram reserve forest (Fig. 19.26) that the mangroves of this reserve forest falls in 'good' and 'pristine/very good' health categories. A very large area of mangrove in the central portion of the reserve forest, are in pristine health conditions (Fig. 19.26). There are no degraded mangroves (MHI <40). It may be because of the protection provided to this wetland by the state forest department and MSSRF. The mangrove health map has been verified through the ground visits of the wetland.

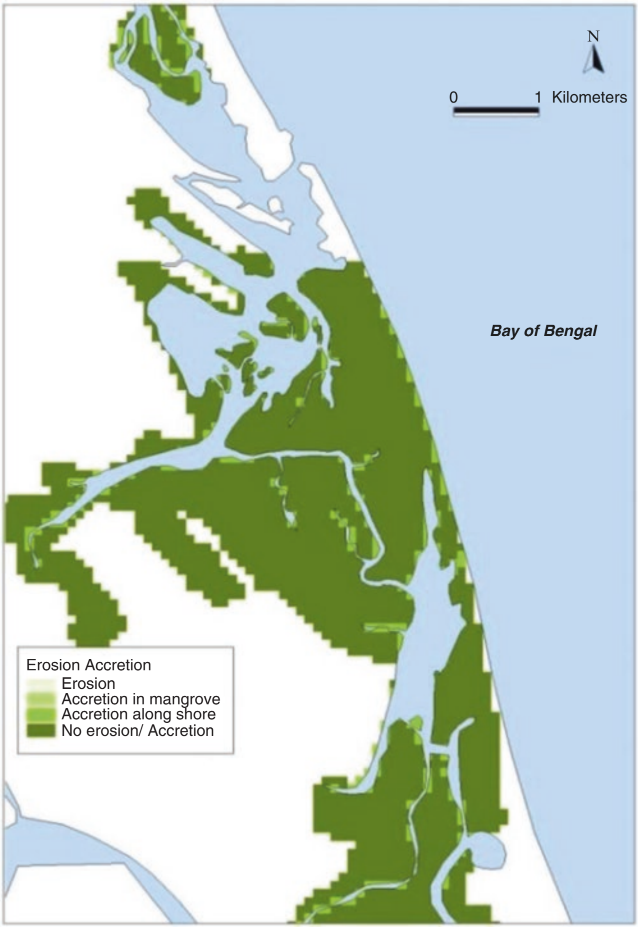


Fig. 19.24 Erosion/Accretion map of Pichhavaram reserve mangrove forest, east coast of India

19.8 Conclusions

Mangroves and its characteristics, geographic distributions and importance of this ecosystem has been discussed. The possible threats, both anthropogenic and natural, have also been discussed. Remote sensing techniques and its applications in inventory and monitoring of mangroves has been briefly presented. Identification of mangroves and other coastal ESAs (Ecologically Sensitive Areas) such as mudflats, coral reefs etc. on the satellite image have also been described. Details of the methodology for mapping of mangroves at community level using satellite data has

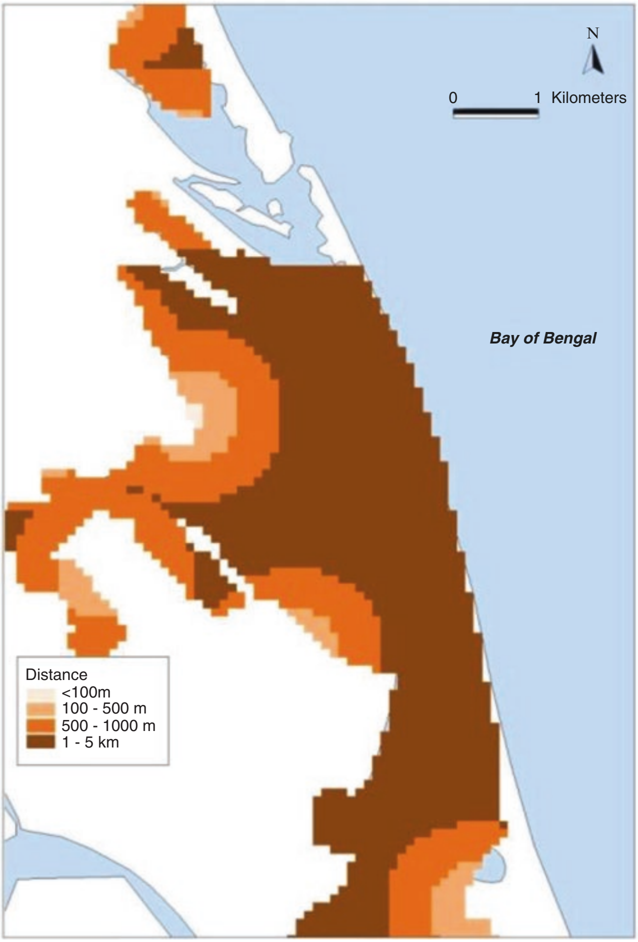


Fig. 19.25 Location of villages along with buffer of Pichhavaram reserve mangrove forest, east coast of India

been given along with case studies. Techniques for remote sensing based monitoring of the mangrove forests along with other coastal ESAs have also been described and explained along with the demonstrative case studies from India. Mangrove health assessment model based on satellite data has been developed and implemented. The case study demonstrating the implementation of the mangrove health model has also been presented. Remote sensing has been found to be very useful in mapping and monitoring of the mangroves as they normally occur in the inaccessible and difficult areas.

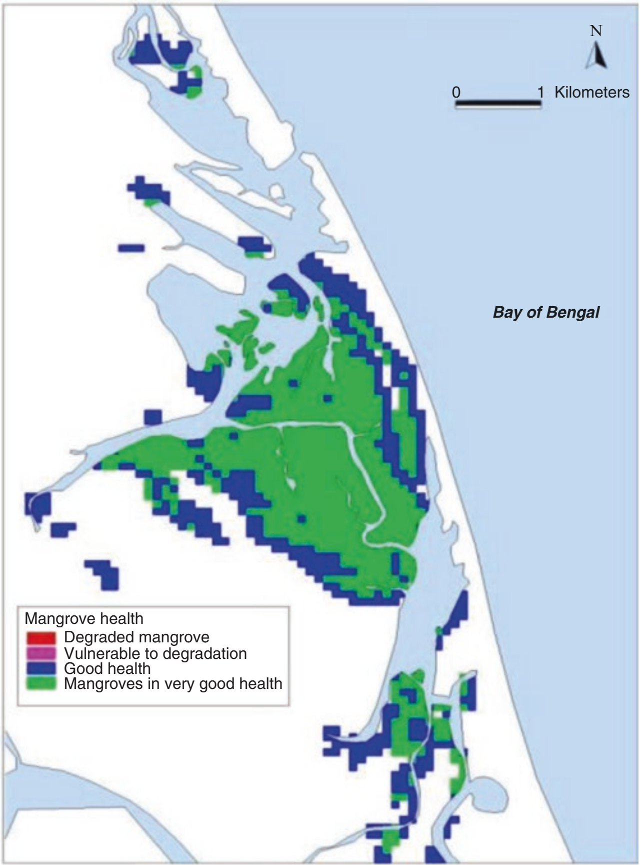


Fig. 19.26 Mangrove Health Index map of Pichhavaram reserve mangrove forest, east coast of India

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Chapter 20
How Can Accurate Landing Stats Help in Designing Better Fisheries and Environmental Management for Western Atlantic Estuaries?

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Abstract Fisheries in Brazil, the country with most territory in South America, is not comprehensive at all. The lack of precise historical data is the main concern faced by perspectives of fishery management. The majority of data are simplistic records of extrapolated biomass published by federal entities regardless the habitat from where fish resources were harvested, how they were captured and lengths of capture. Little attention was given to the social, economic and cultural aspects of traditional communities and their livelihood. Worst is the case of estuarine systems of the Western Atlantic, where artisanal fishery rules the landings and the absence of proper monitoring, control and surveillance leads to poor managerial actions. As a result, fishery studies conclude that most fish stocks of Western Atlantic estuaries show signs of over-exploitation. Palliative measures such as closed periods for capture of common resources has emerged as urgent option aiming to reduce the impacts of overfishing. Bycatch reduction devices are examples of emerging

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options. Fishery management of Brazilian estuaries urges accelerated actions: introduce rights and duties-based fishery management to guarantee the declaration of every fisherman activities; enable fishers to organize themselves through the idea of ecological sustainability and economic efficiency; and acquiring daily reports of fish landings through stakeholder approach and co-management. Fishery pressure is not the unique responsible for reduced estuarine production. Bycatch due to small-mesh nets, oxygen-consuming effluents, emerging pollutants, solid wastes, deforestation of mangrove forest for human purposes and human-driven changes in river flow and estuarine morphology are rapidly changing the nature of a nursery environment. Co-management, long-term data and daily reports on production can help to design stock assessment models, understand variations in biomass over time, detect problems of uncontrolled fishing effort, point periods of seasonal habits for each fishery resource, and, most importantly, guarantee that enough juveniles of each living resource can be recruited to adult stocks. The compliance of ecological data and biological research, robust data for landing stats and the social profile of the fishery community seems to be the ideal approach to build proper rules of co-management in Western Atlantic estuaries.

Keywords South American estuaries • Estuarine conservation • Co-management • Fishery impacts • Fish resources overexploitation • Coastal management

20.1 Introduction

Fisheries management aims to focus on actions and plans to promote sustainable ecological, biological and socio-economic benefits from the use of renewable aquatic resources. Usually, this should result in annual surplus that can be harvested without irreversible decrease in biological productivity (Beck et al. 2001; King 2007; Berkes 2009; Cochrane and Garcia 2009; Eide 2009; Isaac et al. 2010; Longhurst 2010). Such measures should also result in environmental conservation – a positive externality, since sustainable biological yields in wild populations can only be achieved if they live in relatively good quality settings. Worldwide harvests include at least 4 thousand species of aquatic animals, with a production of 120 million metric tons per year (Lackey 2005; FAO 2011). Traditionally the word “fish” used in fisheries management includes not solely finfish, but a wide range of aquatic organisms that are captured for income or subsistence (e.g., mackerel, tuna, guppies, sea turtles, seals, sea urchins, squid, clams, crabs and lobsters) (Blaber 2002; Guebert-Bartholo et al. 2011). Although captures include many vertebrates that nowadays are not the main focus of fishing, rather are groups and/or species of conservational concern such as marine turtles, sharks and marine mammals, estuarine fisheries are mainly concentrated on finfishes and invertebrates (Beck et al. 2001; Barletta and Costa 2009; Silva-Cavalcanti and Costa 2009; Blaber 2013; Guebert et al. 2013).

The benefits provided by fishing activities are difficult to measure and quantify. They are especially measured as wholesale or retail value more easily calculated for commercial fisheries because the products are usually sold, generating figures that can be converted, compiled and compared across time and space (Lackey 2005). However, measures of catch in weight, number, length, and socioeconomic value, can only partially be accessed to help estimating the benefits provided to fishers and their families, communities and to society. These may also reflect and represent quali-quantitatively the importance of anthropogenic interventions (including fishing) at a given location and associations with cultural aspects (King 2007; Barletta and Costa 2009; Guebert-Bartholo et al. 2011; Barletta et al. 2016).

The worldwide fishery crisis, lead to questioning of the simplistic paradigms about its management objectives, especially since the year 2000, due to the rise of a wider and integrated understanding of marine systems. Since realising that fisheries collapse is not exclusively caused by overexploitation of living resources, but also results from changes in climate and the marine environment, the aim now is to implement the ecological and social sustainability of fisheries proposed by the Code of Conduct for Responsible Fisheries (FAO 1995; Charles 2001; Berkes 2009; Eide 2009; Barletta and Saint-Paul 2010; Saint-Paul and Barletta 2010). The document highlights the need to assess and manage fisheries through an integrated focus on biological, technological, economic, social and political aspects. It is acknowledged since the 1970s (Kesteven 1973; Csirke 1984), but consistent suggestions exist since the end of 1980s, and practical strength has been achieved only during in this century (Berkes et al. 2001; Castilla and Defeo 2005; Berkes 2009; Blaber and Barletta 2016; Costa and Barletta 2016).

Ecological issues dealt with in treaties, laws, and government policies are now more substantially guiding fisheries management (de Mitcheson 2009) at different spatial scales. The United Nation Convention on the Law of the Sea (UNCLOS 1982) and The Convention on Biological Diversity (CBD 1992) obligates signatory nations to preserve their biological diversity and protect species at risk of extinction to the maximum extent possible. Whenever these legal instruments provide constraints on the scope, type and intensity of the permitted fishing in order to maintain ecological services, they are able to diminish changes to the environment and consequently their damages to aquatic resources.

According to FAO Technical Guidelines for Responsible Fisheries, "any marine geographical area that is afforded greater protection than the surrounding waters for biodiversity conservation or fisheries management purposes will be considered a Marine Protected Area (MPA)" (FAO 2011). Especial attention has been given to coastal ecosystems as estuaries, wetlands, coral reefs, seagrass beds and other spawning and nursery areas (e.g. coastal lagoons, flooded forests). The United Nations Conference on Environment and Development (Agenda 21; Chapter 17; pp. 167–195) urged coastal states to maintain biological diversity and productivity of marine species and habitats under national jurisdiction (UNCED 1992). According to Agenda 21, these states are responsible for the identification of marine ecosystems with high levels of biodiversity and productivity and other critical areas, as well as providing limitations on use of resources in these areas through the implementation of protected areas (UNCED 1992).

Coastal ecosystems are especially important for being temporary grounds for commercial and subsistence species that use these spaces and their resources as settlement, feeding and nursery (Whitfield 1999; Kjerfve 1994; Able 2005; Dantas et al. 2010; Ramos et al. 2016; Lima and Barletta 2016; Potter et al. 2013). Many fish species spawn in estuaries at times that ensure protection and food availability for their eggs and larvae (North and Houde 2003; Martino and Houde 2010; Lima et al. 2015). Seasonal variations on environmental variables (e.g. salinity, temperature, dissolved oxygen, turbidity and food availability) are the main factors influencing the spatio-temporal distribution of fishes in estuaries around the world (Blaber 1997; Barletta-Bergan et al. 2002a, b; Barletta et al. 2005; Hoffmeyer et al. 2009; Ooi and Chong 2011; Williams et al. 2012).

Estuaries support a high biological production and also have high recovering capacity. However, ecological services, including fisheries, can be reduced by human interventions (Barletta et al. 2016; Dyke and Weaver 2013; Elliott and Quintino 2007), especially unplanned or unpredicted ones. In the last 500 years, the majority of tropical estuaries underwent human colonization and exploitation of their natural resources and services based on the European model. This colonial model has no regard to nature and/or resources conservation. In addition, the high diversity and performance of ecological services provided by estuarine areas favoured poorly planned urban and industrial settlements. Face the large scale of occupation of coastal ecosystems as these, problems of basic sanitation became one of the most important concerns regarding modification of water quality (Costa and Barletta 2016) in the twenty-first century since effluents, solid wastes and port facilities have surpassed all carrying capacity of the local-regional environment. Urban settlements tends to be more widely spread along the coasts, meaning that coastal habitats are likely to be altered or polluted, sometimes to the point of impairing its services and resources maintenance. Therefore, tropical estuaries were often severely abused and are now in a precarious state of natural conservation, overexploited or heavily polluted by sewage, agricultural run-off, industrial effluents and solid wastes (Blaber 2000; Lima et al. 2014; Costa and Barletta 2016), not to mention that often they are encircled by settlements of sub-human conditions that also include traditional fishers populations.

South American estuaries are among those that have been altered by direct and indirect human actions such as dredging, aggradation, damming, road building, pollution by wastewater and solid wastes, and their potential for producing sustained fish harvests has been reduced or permanently impaired (Barletta et al. 2010, 2016; Blaber and Barletta 2016; Costa and Barletta 2016; Reis et al. 2016). One of the most urgent challenges in managing coastal fisheries is to stop and revert the loss of coastal wetlands. These wetlands provide habitat for many adult fish, shellfish and crabs, and are essential breeding and rearing areas, and provide a number of geochemical services essential to water quality. Therefore, efforts are arising to protect these ecosystems.

Science-based knowledge is gaining strength regarding the understanding of estuaries as important areas for both continental and marine biodiversity by pointing out how/how much anthropogenic modifications are changing these ecosystems (Blaber 2000; Costa and Barletta 2016). By following the agreements proposed by the United Nations for responsible fisheries, hundreds of articles have been produced as reference for the establishment of fishery management. However, fishery agencies fail in producing consistent data for the fisheries stats needed to fill in the proposed models. In Brazil, the country with most territorial responsibility in South America, the Fishery and Aquaculture Ministry, recently downgraded to a State Secretary, has not produced fishery stats since 2011 (MPA 2011). Due to limited and unreliable marine fisheries data it is not possible now to predict how many fishes are produced by the artisanal or industrial fleets for each region. Between 2010 and 2011, the marine extractive fishery contributed to 42.4% and 38.6% of total fishing production in the country, respectively (MPA 2011). Whitemouth croaker (*Micropogonias furnieri*), acoupa weakfish (*Cynoscion acoupa*), Brazilian sardinella (*Sardinella brasiliensis*), mullets (*Mugil* spp.), skipjack tuna (*Katsuwonus pelamis*), and shrimps (*Farfantepenaeus subtilis*, *Xiphopenaeus kroyeri*) are among the most important catches (MPA 2011). However, the production of estuarine fishery cannot be assessed because there is no discriminant information of which marine environment these information derived from (MPA 2011).

In general, coastal resources have been heavily fished and often show signs of overexploitation (Guebert-Bartholo et al. 2011; Barletta and Costa 2009). An important point for fishery management in estuaries is that these systems, and their adjacent habitats (e.g. beaches and coastal reefs – coral, algae or beachrocks), are especially important for the settlement of high densities of juvenile stages of many marine fishes (Beck et al. 2001; Able 2005; Sheaves et al. 2014). Whereas estuaries are exporters of juveniles to coastal waters and high seas, intense fishing pressure on early length classes, decreases the recruitment of exploited fish stocks (Barletta-Bergan et al. 2002a, b; Barletta et al. 2005, 2008; Dantas et al. 2010, 2015; Ramos et al. 2012, 2016; Ferreira et al. 2016).

Estuarine artisanal fisheries play an important role in the economic and social context of traditional communities, being essential to the livelihood of fishers, mussel pickers, crab pickers, and their families (Souza and Neumann-Leitão 2000; Silva et al. 2007; Barletta and Costa 2009; Silva-Cavalcanti and Costa 2009; Guebert-Bartholo et al. 2011). However, the lack of knowledge on ecological issues by both harvesters and managers, together with long-term failures in registering and processing coastal landing stats, are the main concerns when fishery management models are put to question. The advancement of knowledge and scientific research on the ecology of estuarine fishery resources note that although these ecosystems may be highly resilient, they are also limited, and their use must be ordered aiming at co-management and sustainability (Beck et al. 2001; Mendonça 2006; King 2007).

Here, we synthetize what has been achieved in the last years regarding fisheries in the Parnaíba Delta (02°53′40.89″ S; 42°04′36.95″ W), Goiana Estuary (07°33′44.04″ S; 34°52′55.81″ W), São Francisco Bay (10°25′10.32″ S; 36°32′26.26″ W) and Laguna Estuarine Complex (28°23′48.83″ S; 48°49′20.91″

W), all located along the Western Atlantic coast of South America. The aim of this chapter is to relate key environmental factors and man-driven changes for each estuary with the variation in the catches of the main landed fishery resources. Finally, we propose improvements to managerial plans by linking science-based ecological information to possible future accurate landing stats.

20.2 Study Areas and Natural Featurings

20.2.1 Parnaíba River Delta

The Parnaíba River Delta complex (Maranhão and Piauí States) is located on eastern South America (Northeast Brazil) (Fig. 20.1). It lies on the border of two large South American biomes, the semi-arid northeast with little water supply and covered by the caatinga (type of savanna), and the Amazon rain forest, with much larger rainfall rates and humidity-dependant forest ecosystems. The Parnaíba Delta covers an area of 275,000 ha (aprox.) and consists of an estuarine system comprising five river mouths (Tutóia, Melancieira, Caju, Canarias and Igaraçu River Estuaries) and 90 islands (Mai and Loebmann 2010) (Fig. 20.1). It is recognized as the third largest deltaic formation worldwide and the sole of this kind in the Americas to debouche directly into the Atlantic Ocean (Guzzi 2012). The main channels of estuaries in the deltaic complex of Parnaíba River have low depths (7–15 m) and are surrounded by flooded mangrove forests, several islands having dunes and lagoons of freshwater (Andrade 2012).

Water temperature at Equatorial coasts is little variable, ranging from 27 to 28.5 °C. Therefore, rainfall rates are high, and responsible for seasonal changes in the salinity gradient, which have a greater influence in the environment. The period

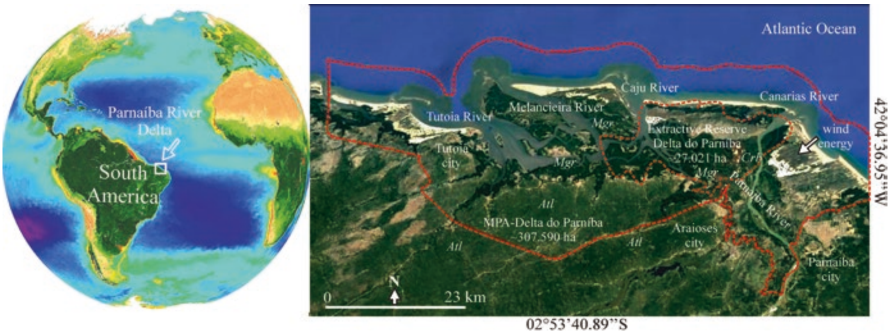

Fig. 20.1 Parnaíba River delta estuaries. *Atl* Atlantic Rain Forest, *Crb* carnaúba plantation, *Mgr* mangrove flooded forest. *Red dashed line* (----) limits the marine protected areas "APA-Delta do Parnaíba" and extractive reserve Delta do Parnaíba (Source: Google Earth)

of highest rainfall occurs between January and May (monthly average: 50–350 mm, total: ~1000 mm), while the driest season occurs between June and December (~50 mm) (Farias et al. 2015) (Fig. 20.2a). Salinity ranges from 0 in the areas next to Parnaíba River to 35–41 seaward, at least during the driest season (Lima 2012). The salt wedge forms 20 km upstream, favoured by the intensification of winds (Lima 2012; Silva et al. 2015). During the rainy season, salinity drops to 12–18 seaward, when river flow increases (Lima 2012; Silva et al. 2015). The mouths of Tutóia, Melancieira and Caju estuaries are mainly influenced by marine coastal water throughout the year, while Poldros and Igaraçu estuarine mouths receives greater freshwater influence from the main channel of the Parnaíba River (Lopes 2015). This pattern favours the use of the three later estuaries by marine and estuarine fishes, while the occurrence of freshwater fishes is only recorded in the other two (Lopes 2015).

The Parnaíba Delta is inside an MPA of the type sustainable use [“MPA – Delta do Parnaíba” (~307.590 ha)], coordinated by the Chico Mendes Institute for Biodiversity Conservation (ICMBio) since 1996, which also includes an MPA of the type extractive reserve (~27,021 ha) (Fig. 20.1). Although there is no management plan for the extractive reserve, the main propose of this MPA is to regulate the sustainable use of natural resources, especially aquatic organisms such as fin fish

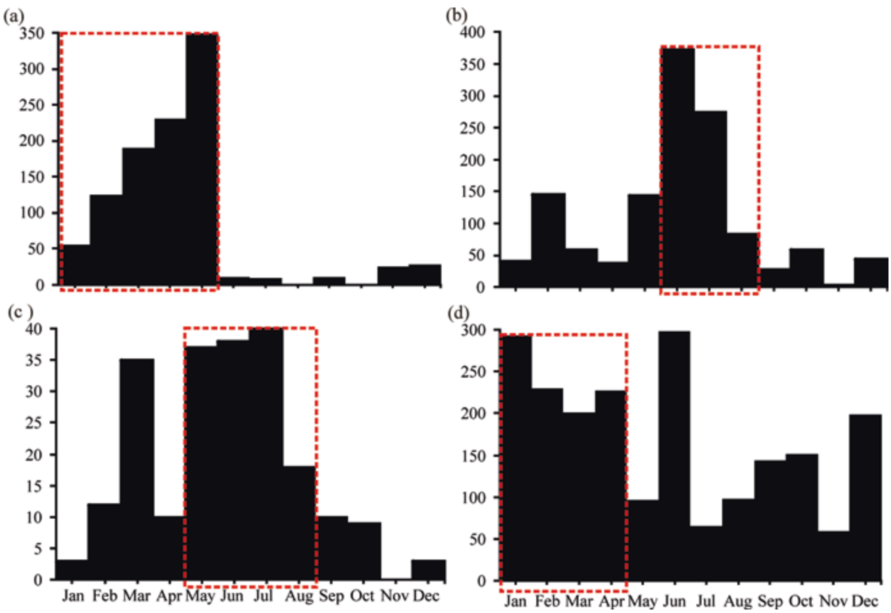

Fig. 20.2 Climatic average for total monthly rainfall (mm) patterns in Western Atlantic estuaries. (**a**) Parnaíba River Delta Estuaries; (**b**) Goiana River Estuary; (**c**) São Francisco River Estuary; (**d**) Laguna Estuarine System. *Red dashed boxes* (----) indicates the period of rainy season adopted in this text

and shellfish, which are commonly caught by several extractive activities of traditional communities inhabiting the islands of the Delta.

20.2.2 Goiana River Estuary

The Goiana River Estuary (Pernambuco and Paraíba States) is also located in the eastern coast of South America (Northeast Brazil). The Atlantic Rain forest extends only a couple hundred kilometres landwards, and separates the humid coast from the wide semi-arid continental mass (Fig. 20.3) (for more information see Barletta et al. 2017a in this book). The river basin is formed by the confluence of smaller rivers, originating the Goiana River, whose estuarine system has a total area of ~4700 ha. Air temperature changes relatively little along the year. The average air temperature is 25 °C, and oscillates between 27 °C in summer and 24 °C in winter months (Barletta and Costa 2009). Its climate is tropical, with well defined dry and rainy seasons (Barletta et al. 2010). The driest season extends from September to February and the rainy season from March to August, however the highest rainfall rates are observed between July and August (Fig. 20.2b) (Barletta and Costa 2009). Vegetation is predominantly mangrove flooded forest under the influence of tides (0–2.5 m). This estuary encompasses a large diversity of habitats, including main channel, flood plains, tidal creeks and surrounding mangrove forest (Barletta and Costa 2009; Lima et al. 2016).

In Goiana Estuary, the freshwater flow into the systems during the dry season is very small, and the environmental quality strongly depends on short and uncertain periods of rainfall (Barletta et al. 2010; Lima et al. 2015). Therefore, freshwater fish assemblages are not as diverse as marine fish assemblages in this system (Lima et al. 2015). Estuarine morphology allows coastal waters to influence even the upper estuary, and marine demersal fishes can reach areas next to the river (Lima et al.

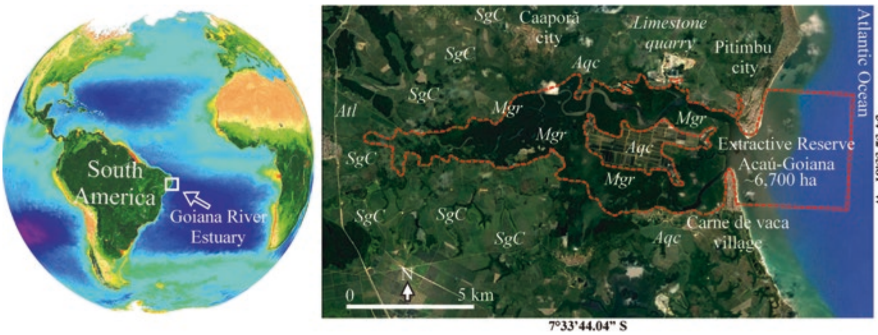

Fig. 20.3 Goiana River Estuary. *Atl* Atlantic Rain Forest, *Aqc* aquiculture, *Mgr* mangrove flooded forest, *SgC* sugarcane plantation. *Red dashed line* (----) limits the extractive reserve "Resex Acaú-Goiana" (Source: Google Earth)

2015). Since the estuary is relatively shallow, there is no vertical stratification of salinity, water temperature and dissolved oxygen most of the year (Dantas et al. 2010; Lima et al. 2015). Fish are distributed along spatial gradients determined by temporal factors. Especially the seasonal fluctuation of salinity form an ecocline (Dantas et al. 2010; Lima et al. 2015; Ferreira et al. 2016; Ramos et al. 2016). During the late rainy season, when continental runoff increases seaward, marine fish migrate to the lower portions of the system to inhabit waters of higher salinity (Dantas et al. 2010; Ferreira et al. 2016; Ramos et al. 2016). This season is also acknowledged as the time when most fishes and invertebrates that grow within the estuarine habitats are exported out to the sea (Dantas et al. 2010; Lima et al. 2014; Ferreira et al. 2016).

This area is responsible for supporting a rich fauna of fishes, crustaceans and mollusks, which harvest ensures the livelihood of the traditional populations surrounding the city centre and smaller villages as Goiana, Tejucopapo, São Lourenço and Carne de Vaca (in Pernambuco State), and Caaporã and Pitimbú (in Paraíba State) (Fidem 1987). For this reason, the MPA-extractive reserve Acaú-Goiana (~6700 ha) was created in 2007 to guarantee the sustainable use of fisheries resources, such as finfishes and the Bivalvia *Anomalocardia brasiliana*, as well as to preserve the system for livelihood (Barletta and Costa 2009; Guebert et al. 2013) (Fig. 20.3). Consolidated management plans for extractive reserve Acaú-Goiana still do not exist and only recently fishers are in the process of recognized/realized what an extractive reserve really is (Guebert et al. 2013).

20.2.3 São Francisco River Estuary

The São Francisco River basin has 63,478,100 ha of drainage area. The São Francisco River itself extends 2700 km from source to the estuary (CBHSF 2004), and has significant environmental, economic, social and cultural importance for Brazil. Its status is based on the large volume of water that it carries (average: 2846 $m^3 s^{-1}$), coastal fertilization potential, hydroelectric power generation, waterway, fishing and other human uses as water supply. Due to its length and different environments, the ecocline along the hydrographic basin can be divided into four major physiographic units: upper, middle, sub-middle and lower São Francisco River (ANA 2002). The last, and shortest, stretch of 274 km between Paulo Afonso (Bahia State) and the river mouth includes the São Francisco Estuary (Alagoas and Sergipe States) (Fig. 20.4) (Godinho and Godinho 2003).

The estuary is located on the east coast of South America (Northeast Brazil). The average annual air temperature for the São Francisco Estuary is 25 °C, evaporation is 2300 mm per year and the average annual rainfall varies from 800 to 1300 mm (Burger 2008). The climate of the region surrounding the river mouth is tropical semi-humid (Burger 2008), with constant strong winds, leading to low precipitation rates. The seasons of the region have two distinct periods, a dry season (September to February) and a rainy season (March to August) (Fig. 20.2c). However, the con-

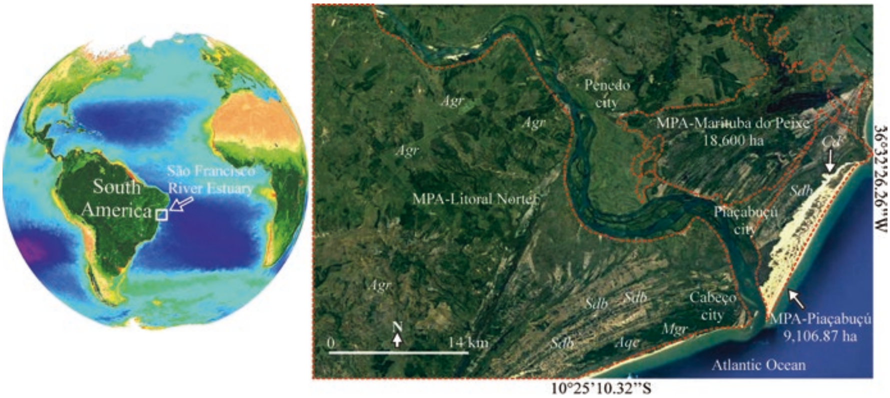

Fig. 20.4 São Francisco River Estuary. *Agr* agriculture, *Aqc* aquiculture, *Cd* coastal dunes, *Mg* mangrove flooded forest, *Sdb* sandbank vegetation. Georeferenced limits for the marine protected area "APA-Litoral Norte" are not available (Source: Google Earth)

ventional climate patterns became less evident recently and appear to be changing after the construction of large hydroelectric power plants (Burger 2008). The estuary is dominated by meso-tides of semi-diurnal type with spring tides reaching 2.6 m. The wave regime is of high energy, predominantly E and NE in January to May and from September to November; SE waves occur from March to August (Dominguez 1996; Medeiros et al. 2008).

The São Francisco Estuary presents a well-defined salinity gradient. The intrusion of marine waters forms a salt-wedge towards the inner estuary and is heavily influenced by the tide, with less influence of the river flow and winds (Knoppers et al. 2007). Considering that the main forcing of the salt water entering the estuary is tidal, higher penetrations of saline water bodies occur during spring tides (Oliveira et al. 2008). The coastal environment is characterized by the presence of riparian vegetation with predominance of mangrove flooded forests. The mouth of estuary covers a complex of coastal plains, composed of a number of interconnected channels, lagoons and floodplains, surrounding the main channel that flows into the Atlantic ocean (10° 36′S and 36° 23′W) (Souza 2007; Medeiros et al. 2011). In the Alagoas State, since 1983, the coastal dunes adjacent to the São Francisco Estuary was included in a MPA-sustainable use, the "MPA-Piaçabuçú" (~9106.87 ha), whose management plan was implemented late in 2010 (ICMBio 2010) (Fig. 20.4). In 1988, it was created another MPA, the "MPA-Marituba do Peixe" (~18,600 ha) with a proposed management plan in 2006 (IMA 2006) (Fig. 20.4). These two MPAs cover the left margin of the seaward portion of the estuary (Fig. 20.4). In the Sergipe State, there is the MPA "MPA-Litoral Norte", created in 2004, covering the right margin of the São Francisco Estuary (Fig. 20.4). These protected areas aim at the sustainable development of socio-economic activities by creating rules to protect and conserve the different habitats and biodiversity within its territory (IMA 2006; ICMBio 2010; Santos et al. 2014).

20.2.4 Laguna Estuarine Complex

The Laguna Estuarine Complex is located in south Brazil, being divided in two sectors (Fig. 20.5). The first sector is formed by the Santa Marta and Camacho lagoons (3100 ha) (Fig. 20.5). The second sector is formed by the Santo Antonio dos Anjos, Imaruí and Mirim lagoons (18,400 ha) (Fig. 20.5). Each lagoon is linked to the other by small channels. Water flows seaward through a single channel, the estuary mouth. The coastal areas adjacent to the system is part of a MPA ("MPA-Baleia Franca") created by federal decree in September 2000 with ~154,866.27 ha expanding seawards aiming at the protection of the southern right whale (*Eubalaena australis*) population that uses the area (Fig. 20.5).

The estuary is located in a strip of coastal plains with an extended sedimentary field bounded on the west by the mountains and on the east by the sea. It is a typical choked lagoon, connected to the sea by a single entrance channel (Kjerfve 1994). The salinity ecocline is formed by the connection between the sea and the Santo Antonio dos Anjos Lagoon (Fig. 20.5). The system receives freshwater from Tubarão River basin which discharges directly in the low estuary (Santo Antonio dos Anjos lagoon). There, the influence of coastal waters contributes to create mesohaline to polyhaline conditions (salinity 5–25) depending on rainfall rates (Fig. 20.2d). Mirim and Imaruí lagoons receive freshwater discharges from D'Una River, contributing to reduce the local salinity, especially during the rainy season (Fig. 20.5) (SDM 2002). Therefore, at its north portion, the estuary has limnetic conditions (salinity <0.5) during the entire year. The middle estuary presents limnetic to oligohaline conditions (salinity 0.5–5), with seasonal influence of rainfall patterns (MacLusky and Elliott 2004) (Fig. 20.5).

Atmospheric conditions are controlled by the intertropical air mass, especially between September and March. Polar air masses are more frequent through April to

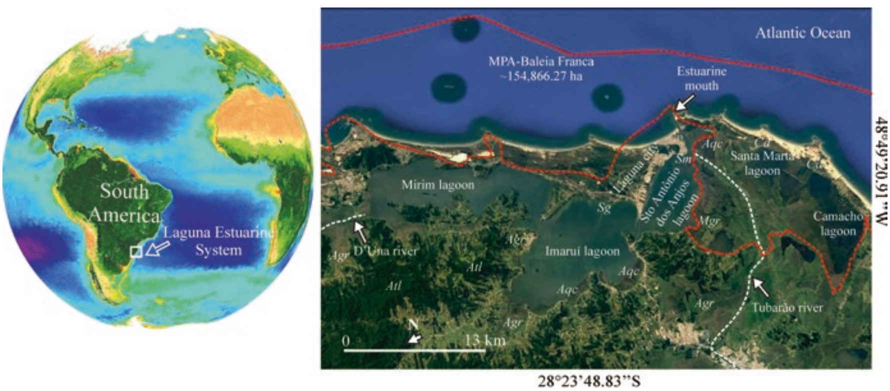

Fig. 20.5 Laguna Estuarine System. *Agr* agriculture, *Atl* Atlantic Rain Forest, *Aqc* aquiculture, *Cd* coastal dunes, *Mgr* mangrove flooded forest, *Sg* seagrass bed, *Sm* salt marshes. *Red dashed line* (----) limits the marine protected area "APA-Baleia Franca"(Source: Google Earth)

August, being responsible for maintain the mesothermal climate gradients. The contact between the intertropical and polar air masses forms the Atlantic Polar Front, responsible for the rainfall distribution along the year. The annual rainfall in the region ranges between 1460 and 1820 mm, with 129–144 days of rain. At the estuary itself, these values are between 1000 and 1250 mm (monthly average: 100–300 mm), concentrated between January and March (Fig. 20.2d) (EPAGRI 2014).

20.3 Ecological Services, Land Use and Anthropogenic Impacts

20.3.1 Parnaíba River Delta

Parnaíba city is 14 km upstream from the Parnaíba Delta, but urban settlement in the region accompanied extractive activities of forest products, as maniçoba rubber (*Manihot esculenta*), babaçú oil (*Orbignya phalerata*) and carnaúba wax (*Copernicia prunifera*) (Fig. 20.6e and Table 20.1). These products and their industrialization dominated the economic cycles until the first half of the twentieth century (Mendes 2003). Carnauba wax was then the main product exported to Europe. It justified infrastructure investments, such as railway lines and harbour equipment. Parnaíba is now a commercial reference in fruits production, rice and general trade, all contributing to populational increases that is currently at 170 thousand people (Fig. 20.6) (Carvalho and Gomes 2006).

Most of the environmental impacts are associated to non-sustainable fisheries using gillnets of small mesh size (2 cm between knots). These nets catch fish in young and sub-adult phases, most of them commercially important as lane snapper *Lutjanus synagris*, dog snapper *L. jocu* and common snook *Centropomus undecimalis* (Table 20.1).

Although shrimp trawlers operate unattended just outside the MPA, they are affecting fish stocks by capturing several young fishes near the coast (3–4 nautical miles). Crab fishing remains the main economic activity of the sector, representing approximately 90% of local landings (Farias et al. 2015). The main fishing pressure is related to the uçá-crab (*Ucides cordatus*) stocks, which for decades is caught from the mangrove forests above their catch limit, with no quota per person or minimum size and sex control. Uçá-crab produces around 1000 tons per year (Table 20.1).

The rainy season marks the occurrence of megafauna as the marine manatee *Trichechus manatus manatus* (Souza 2015) and sea turtles (Silva et al. 2015) (Table 20.1). The Delta is also inhabited by several species of birds, such as guará (*Eudocimus ruber*), whimbrel (*Numenius hudsonicus*), the osprey (*Pandion haliaetus*) and the magnificent frigate bird (*Fregata magnificens*) (Campos et al. 2015) (Table 20.1) part of marine food webs.

Recent anthropogenic impacts at the Parnaíba Delta include the occupation of the margins of lagoons by wind-energy turbines, which downgrades the landscape;

Fig. 20.6 Anthropogenic impacts affecting the Parnaíba River Delta Estuaries. (**a**) loss of landscape value near lagoons for wind-energy; (**b**) mass tourism; (**c**) solid wastes pollution (e.g. plastics); (**d**) unplanned urban area (Parnaíba city); (**e**) carnaúba farms in wetlands; (**f**) rice farms on the river margins

also, the uncontrolled mass tourism and; urban settlements, producing solid wastes that are not properly disposed off (Fig. 20.6a–d). However, by the reason of this ecosystem is a Federal Protected Unit, the anthropogenic effects caused by the urban settlement are somewhat controlled by IBAMA and ICMBio, and certain activities are subject to environmental licencing (Fig. 20.1).

Table 20.1 Main forms of land use, ecological services, living resources and anthropogenic impacts at Western Atlantic estuaries

Estuary	Land uses	Main ecological services	Main living resources	Main anthropogenic impacts
Parnaíba River delta estuaries	**Fisheries:**	**Nursery:**	**Fisheries resources:**	**Fisheries impacts:**
	Artisanal	Fishes and invertebrates	Uçá-crab	Small mesh-size fishery
	Recreational	**Seasonal users:**	Common snook	Intensive bycatch
		Manatee	Mullets	**Other human impacts:**
	Subsistence	Sea turtles	Lanne snapper	Harbour moorings
	Other uses:	Birds	Dog snapper	Mangrove deforestation for agriculture
	Conservation	**Other services:**	Acoupa weakfish	Mangrove deforestation for wind-energy generation
	Electric power generation	Coastal fertilization	**Other resources:**	Railway lines
	Holiday homes	Coastal protection	Carnaúba wax	Wetlands drainage
	Tourism	Water supply	Maniçoba rubber	Urban settlements
			Babaçu oil	Domestic/agriculture effluents
Goiana River Estuary	**Fisheries:**	**Nursery:**	**Fisheries resources**:	**Fisheries impacts:**
	Artisanal	Fishes and invertebrates	White mullet	Small mesh-size fishery
	Recreational	**Protection of endangered species**:	Longnose stingray ray	Overexploitation of lobsters
	Subsistence	Atlantic goliath grouper	Uçá-crab	Interaction biota-plastic debris
	Other uses:	Manatee	Caribbean venerid clam	Microplastic pollution
	Conservation	Sea turtles	Stout mussel	Microplastic ingestion by fishes
	Holiday homes	**Other services**:	Lobsters	Intensive bycatch
	Tourism	Coastal fertilization	Acoupa weakfish	Illegal fisheries of crabs
	Aquaculture	Effluents dilution	Snooks	**Other human impacts:**
		Water supply	Snappers	Mercury contamination
			Cutlassfish	Mangrove deforestation for sugarcane plantations, milling and mining
			Mojarras	
			Pink and white shrimps	River drainage
				Domestic/aquaculture/agriculture effluents
				Dredging main channel for sand
				Unplanned urban settlements

São Francisco River Estuary	**Fisheries:**	**Nursery:**	**Fisheries resources**:	**Fisheries impacts:**
	Artisanal	Fishes and invertebrates	Seabob shrimp	Small mesh-size fishery
	Recreational	**Other services:**	Vaillant's anchovy	Intensive bycatch
	Subsistence	Waterway	Mojarras	**Other human impacts:**
	Other uses:	Water supply	Snooks	Domestic/aquaculture/agriculture effluents
	Holiday homes		Mullets	Dredging main channel for sand
	Tourism		Jacks	Damming for hydroelectric power generation
	Electric power generation		Shark and Rays	River dranaige
			Sand crabs	Water transposition to other basins
				Impoundments
				Mangrove and Atlantic Rainforest deforestation for sugarcane and coconut plantation
				Urban settlements
Laguna Estuarine System	**Fisheries:**	**Nursery:**	**Fisheries resources:**	**Fisheries impacts:**
	Artisanal	Fishes and invertebrates	Pink shrimp	Small mesh-size fishery
	Recreational	**Protection of endangered species:**	Lebranche mullet	Ghost fishing
	Subsistence	White sea catfish	Whitemouth croaker	Illegal fisheries
	Industrial	Black drum	Bluefish	Intensive bycatch
	Other uses:	**Other services:**		**Other human impacts**:
	Conservation	Coastal fertilization		Rivers drainage
	Holiday homes	Waterway		Ferry boat transport
		Water supply		Atlantic Rainforest deforestation for rice culture
				Urban settlement
				Domestic/aquiculture/agriculture effluents

20.3.2 Goiana River Estuary

Possibly 450–1000 fishers' traditional families exploit natural living resources from the Goiana Estuary for subsistence and income generation (Barletta and Costa 2009). The main subsistence catches are ariid catfishes (*Cathorops spixii*, *C. agassizii* and *Sciades herzbergii*); commercial targets are sciaenids, gerreids, centropomids, lutjanids and mugilids (Fig. 20.7a–f); and crabs, shrimps, oyster and shellfish (bivalves *Anomalocardia brasiliana* and *Tagelus plebeius*) serve both demands (Figs. 20.8a–d, 20.9a, b, 20.10a–d, and Table 20.1) (Barletta and Costa 2009; Silva-Cavalcanti and Costa 2009). However, overexploitation of stocks is a detectable problem, especially for the most profitable catch: lobster (Table 20.1) (Guebert-Bartholo et al. 2011).

The Goiana Estuary provides a wide variety of ecological services to the local population and surrounding areas. It also includes important habitats for different

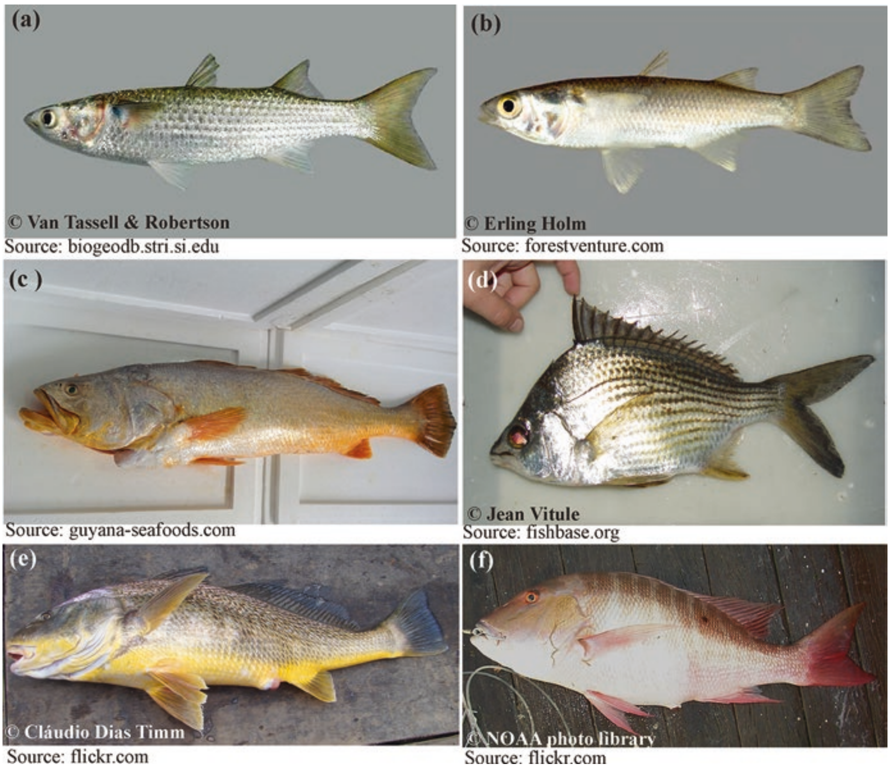

Fig. 20.7 Examples of important catches of finfish form the Western Atlantic estuaries and adjacent coastal areas. (**a**) lebranche mullet (*Mugil liza* – Mugilidae); (**b**) white mullet (*Mugil curema* – Mugilidae); acoupa weakfish (*Cynoscion acoupa* – Sciaenidae); (**d**) Brazilian mojarra (*Eugerres brasilianus* – Gerreidae); (**e**) white mouth croaker (*Micropogonias furnieri* – Sciaenidae); (**f**) mutton snapper (*Lutjanus analis* – Lutjanidae)

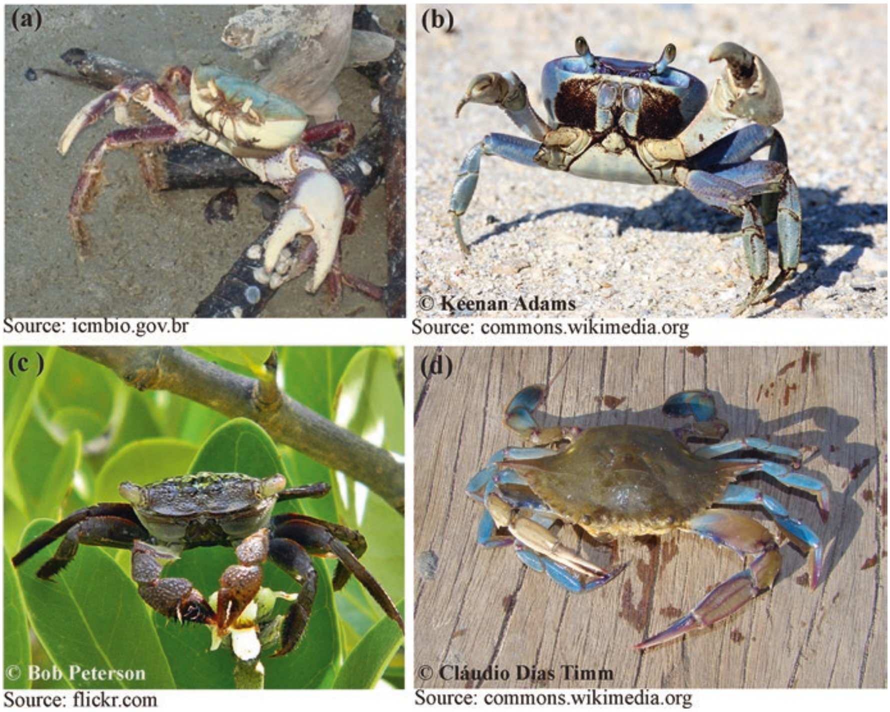

Fig. 20.8 Examples of important crab species from Western Atlantic estuaries and adjacent coastal areas. (**a**) Uçá-crab (*Ucides cordatus*); (**b**) Blue crab (*Cardisoma guanhumi*); (**c**) mangrove tree crab (*Aratus pisoni*); (**d**) swimming crab (*Callinectes danae*)

uses and ontogenetic phases of endangered species as *Epinephelus itajara* (Atlantic goliath grouper), *Trichechus manatus manatus* (manatee), *Chelonia mydas* and *Lepidochelys olivacea* (sea turtles) (Table 20.1) (Barletta and Costa 2009). Fishing with gillnets and interaction with plastic debris were considered the most important threats to the sea turtle populations that feed, mate and nest in the area (Table 20.1) (Guebert et al. 2013).

Economic activities such as sugarcane plantations and milling, aquaculture, limestone mining for cement production, dredging for sand and land development are among the modifications that influence the estuarine ecosystem and water quality (Figs. 20.11a–f, 20.12a–d, 20.13, and Table 20.1) (Barletta and Costa 2009). The Goiana Estuary receives wastewater effluents from the sugarcane planting and milling industry and sewage from urban settlements (Costa et al. 2009). These are known as potential sources of, for example, mercury to subsistence and commercial fishery resources, such as cutlassfish *Trichiurus lepturus* and the shellfish *A. brasiliana* (Barletta et al. 2012; Costa et al. 2009; Silva-Cavalcanti et al. 2016) (Table 20.1). Although tissue contamination are below World Health Organization critical limits for human consumption, further contamination can increase mercury levels, making this fisheries resources unsafe for frequent consumption, especially by pregnant women and children (Costa et al. 2009).

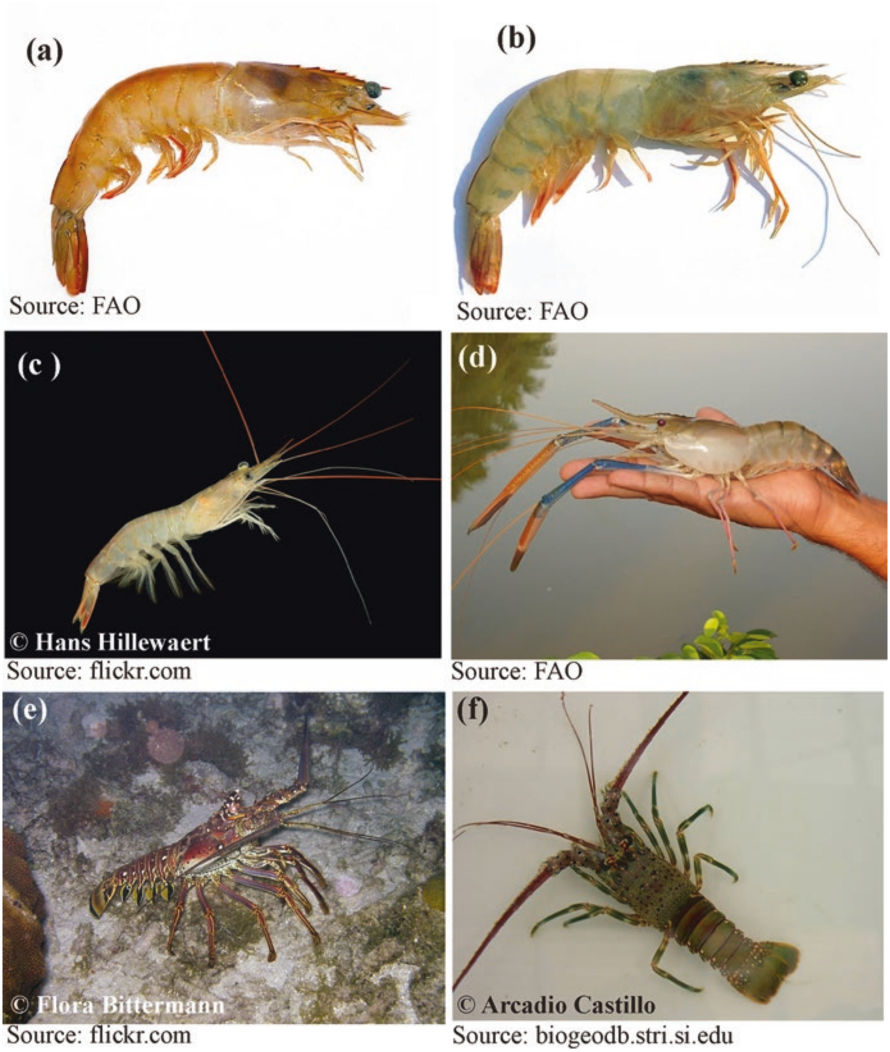

Fig. 20.9 Examples of important prawns and lobsters species inform the Western Atlantic estuaries and adjacent coastal areas. (**a**) pink shrimp (*Farfantepenaeus subtilis*); (**b**) white shrimp (*Litopenaeus schmitii*); (**c**) Atlantic seabob (*Xiphopenaeus kroyeri*); (**d**) freshwater shrimp (*Macrobrachium* sp.); (**e**) red lobster (*Panulirus argus*); (**f**) green lobster (*Panulirus laevicauda*)

Contamination by plastic debris and microplastics (<5 mm) is a concern in the main channel and tidal creeks of the mangrove flooded forest in the Goiana Estuary (Table 20.1) (Ivar do Sul et al. 2014; Lima et al. 2014, 2015, 2016). These solid wastes have multiple and complex sources, being related to urban settlements, tourism, continental sources, land runoff and especially fisheries. In addition, when precipitation increases the estuary is a powerful exporter of this contaminant to the marine environment (Lima et al. 2014). For example, microplastics density were

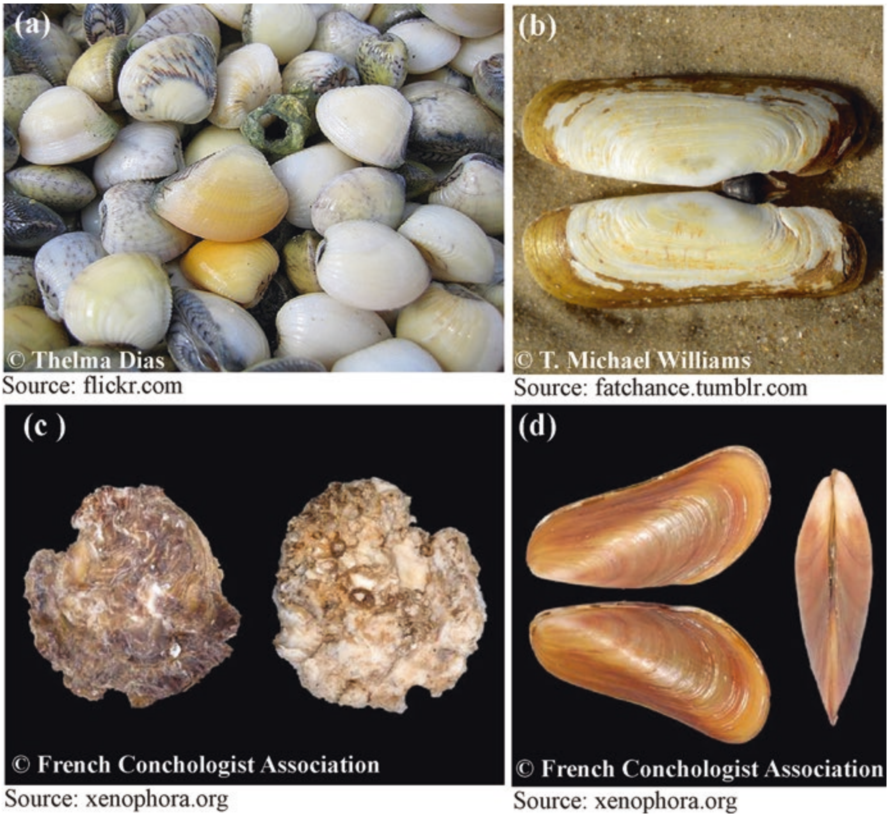

Fig. 20.10 Examples of important molluks inform the Western Atlantic estuaries and adjacent coastal areas. (**a**) Caribbean venerid clam (*Anomalocardia brasiliana*); (**b**) stout mussel (Tagelus plebeius); (**c**) mangrove oyster (*Crassotrea rhizophorae*); (**d**) Charru mussel (*Mytella* sp.)

comparable to half of the total fish larvae and had density equivalent to total fish eggs along the main channel (Lima et al. 2014).

The ingestion of plastic threads by subsistence and commercial demersal fishes was observed in 20% of ariid catfishes (*Cathorops spixii*, *C. agassizii, Sciades herzbergii*) (Possatto et al. 2011) and 13% of Gerreids (*Diapterus rhombeus*, *Eugerres brasilianus*, *Eucinostomus melanopterus*) (Ramos et al. 2012). In sciaenids, it was registered a contamination of 8% of *Stellifer* (*S. brasiliensis*, *S. stellifer*) (Dantas et al. 2012) and 100% in adult *Cynoscion acoupa* (Ferreira et al. 2016) (Table 20.1).

20.3.3 São Francisco River Estuary

The importance of the São Francisco River is not only related to the amount of water carried across a semi-arid region, but mainly to its historical and economic contribution to the establishment of communities on a basin-wide scale and the

Fig. 20.11 Anthropogenic impacts affecting the Goiana River Estuary. (**a**) deactivated water pumping system at prawn farm; (**b**) domestic effluents (sewage); (**c**) mangrove deforestation for limestone quarry; (**d**) artisanal fishing harbour (Pitimbú city); (**e**) sugarcane plantation; (**f**) unprotected soil after sugarcane burning and harvest.

creation of cities around it (Fig. 20.14a, b). In addition, the use of water potentials is an important issue in future aquaculture and irrigation projects due to excellent soils on the river margins (Pruski et al. 2005).

Although agricultural areas have been developed along the San Francisco Estuary, the margins and sub-basins have a better conservation status, with less environmental degradation at the middle and sub-middle reaches, especially due to the presence of the MPAs-sustainable use in the region (Silva et al. 2010) (Fig. 20.4). Rice was the most intense economic activity (Gois et al. 1992), which employed most of the labour, and participated in boosting the regional economy until the

Source: LEGECE

Fig. 20.12 Dredging sand in the upper portion of the Goiana River Estuary. (**a**) dredging; (**b**) dredger; (**c**) pumping of sand; (**d**) transportation to building sites

1970s (Fig. 20.14c). The polyculture of food completed the supply for the local population, and supplied numerous municipal fairs in the region (Rieper 2000). Other cultures, due to the low productivity achieved, were less significant (e.g. corn, beans, cotton and manioc) (Table 20.1). Despite the decline in recent years, coconut culture still ranks as an important part in the local economy. Few cultures remain near the estuary, among them coconut, rice and manioc, especially in Sergipe State (Fig. 20.14c and Table 20.1).

Fishing in São Francisco River has lower yields than expected for a river this size due to the oligotrophic and depleted waters, a consequence form anthropogenic changes made upstream (Knoppers et al. 2007, Oliveira et al. 2008). The lower river and the estuarine system are the most productive areas (Araújo and Sá 2008). Fishes migrate upstream to find suitable places to spawn, feed and protect larvae and young, especially in wetlands and among aquatic macrophytes (Araújo and Sá 2008; Pompêo 2008). The estuarine area has a rich diversity of species, functioning as nursery and shelter, promoting the recovery of the stock of several commercially important species of fish, crustaceans and mollusks (Table 20.1). The fishing fleet operates primarily in the estuarine region and adjacent coastal areas. The Families Carangidae, Centropomidae, Gerreidae, Lutjanidae, Mugilidae and Sciaenids are of economic importance, and use the estuary during, at least, one stage of their onto-

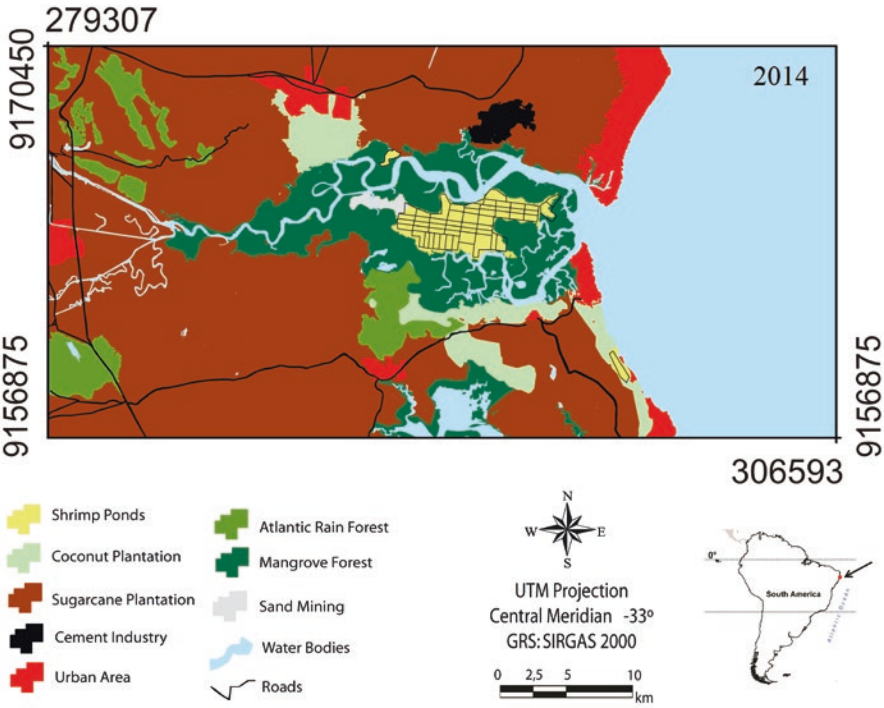

Fig. 20.13 Current impacts around the extractive reserve "Resex Acaú-Goiana" (Goiana River Estuary). Sugarcane plantation is the most pronounced impact due to deforestation of mangrove and Atlantic Rain forests leaving the margins unprotected against erosion and impairing the local hydrological cycle

genetic development, but especially when juveniles (Figs. 20.7a–f and 20.15a–f). The use of the river mouth area as nursery for commercial species as mullets *Mugil* spp. and Brazilian mojarra *Eugerres brasilianus* is observed in the south coast of Alagoas State (Paiva et al. 2013).

The São Francisco River has been severely degraded by dredging of the main channel for sand, improper wastewater disposal, building and operation of artisanal fishing harbours (Fig. 20.14d–f). However, the main impacts is damming for developing systems of hydroelectric power plants along the entire river's course (Fig. 20.14a) (MMA 2004a). Since 1950, interventions in the main channel triggered a series of impacts on the water regime of the river. The change in water potential energy caused by successive impoundments, withdrawal of water for irrigation, deforestation, intensification of erosion, soil sealing, among others, resulted in changes in water and sediments transport and deposition (Lima et al. 2001; Felippe et al. 2009; Ribeiro et al. 2010) (Table 20.1). The damming of major rivers to build hydroelectric power plants causes a series of changes in limnological characteristics in dammed areas and stretches downstream, as well as the decline in

Fig. 20.14 Anthropogenic impacts affecting the São Francisco River Estuary. (**a**) Xingó hydroelectric power plant in the lower river; (**b**) urban settlement (Penedo city); (**c**) rice farm on the river margin; (**d**) sand mining at the lower river; (**e**) improper wastewater disposal; (**f**) artisanal fishing harbour (Cabeço city)

biodiversity of native fish populations and reduction of biological production, including fisheries yields (Costa 2003; Silva 2009).

Anthropogenic modifications in the lower São Francisco River decreased the magnitude of its flow, which current is only 800 $m^3 s^{-1}$ (natural average river flow 2.846 $m^3 s^{-1}$). Together with the loss of seasonal and interannual pulses, the flow is now stable. Therefore, an increase of the saline intrusion in the stagnant estuary that changes the original conditions is observed (Medeiros et al. 2008). The construction of dams and their reservoirs across the entire drainage basin are also changing the river water quality and the composition of biogenic matter of coastal zones, affecting productivity, trophic interactions and exportation of materials to the ocean (Knoppers et al. 2007). The decrease in the river sediment supply changed the sedi-

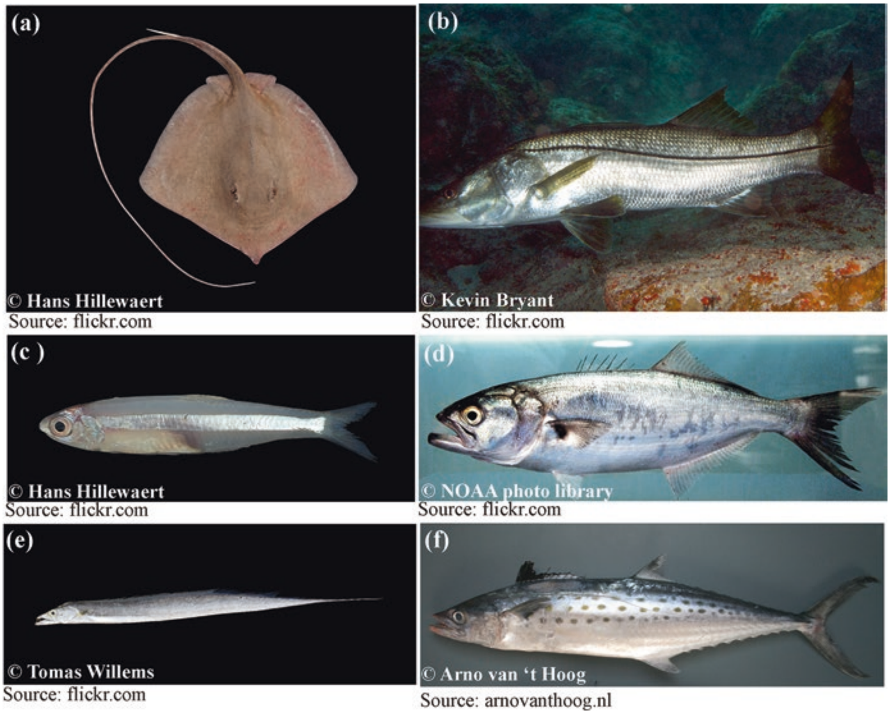

Fig. 20.15 Examples of important finfish inform the Western Atlantic estuaries and adjacent coastal areas. (**a**) longnose stingray (*Dasyatis guttata* – Dasyatidae); (**b**) common snook (*Contropomus undecimalis* – Centropomidae); (**c**) broadband anchovy (*Anchoviella lepidentostole* – Engraulidae); (**d**) bluefish (*Pomatomus saltatrix* – Pomatomidae); (**e**) cutlass fish (*Trichiurus lepturus* – Trichiuridae); (**f**) serra mackerel (*Scomberomorus brasiliensis* – Scombridae)

mentary balance of the estuary, triggered processes of erosion of the river shore (which contributes to the silting of river sections) and accelerated coastal erosion (Bittencourt et al. 2007).

The estuarine mouth of São Francisco River and adjacent areas are significantly degraded, mainly due to dams and water transposition/subtraction, directly affecting the river and indirectly changing socioeconomic and cultural aspects of riverside communities. These careless interventions are exhausting the river, and closing to breaking its recovery capacity. The idea of transposing the waters of São Francisco River exists since the eighteenth century (Andrade 2012; Henkes 2014). The transposition project aims to capture water at two points of the São Francisco and take it deeper into the semi-arid to provide water for the population. It would ensure agricultural crops, industrial activities and tourism, and finally secure the settlements of rural populations in the region. From the colonial point of view, it is imperative to promote the growth of productive activities in the region, to decrease public spending on emergency measures during the frequent and prolonged droughts and ensure water for distribution and reservation (dams, rivers, wells, cisterns and water mains) (Henkes 2014). However, the water transposition in this basin has been widely criticized for its potentially severe environmental impacts (Lima 2013). In addition, studies demonstrated that the north axis project was designed to benefit economic development (irrigation of intensive fruit crops) not taking into account sustainable development (e.g. causing soil degradation by salinization). The execution of the project has already caused environmental damage and negative social impacts (Henkes 2014), even before its completion.

Forty-four environmental impacts were identified, being 23 considered more relevant, of which 11 are positive and 12 negative (Stolf et al. 2012). Positive impacts include increased water supply and security for urban and rural populations, employment and income in regional economy, reduction of emergency during drought, boosting agriculture and incorporation of new areas into the production process, improving water quality in the receiving basins, reduction of rural exodus, migration, diseases and deaths (Stolf et al. 2012). Negative impacts in the receiving basins include loss of jobs and income due to land expropriations, changes in composition and biodiversity loss of native aquatic communities, social risks during the construction phase, interference with indigenous populations and cultural heritage, pressure on urban infrastructure, loss and fragmentation of ~430 ha of native and terrestrial wildlife habitats, introduction of non-native species (both aquatic and terrestrial), and modification of the river system (Lima 2013).

20.3.4 Laguna Estuarine Complex

The main anthropogenic impacts in the Laguna Estuarine Complex are related to basin deforestation and wastewater from agriculture and urban expansions (Fig. 20.16a–d), which have increased significantly in the recent decades generating effluents with high organic loads and consequently oxygen consumption (Fig. 20.16c). The system supports intense seasonal tourism (summer vacations), artisanal and recreational fishing and agricultural activities. Moreover, urban expansion at coastal areas bordering the lagoon grew aggressively over coastal ecosystems as dunes and wetlands in the last decades (Figs. 20.16b, d, and 20.17 and Table 20.1).

According the Macrodiagnostic for the Coastal Zones, realized by the Coastal Management Group of Santa Catarina State (SPG 2010), the urban area around the system covers 3.5% of the total area (1165.69 km^2); agriculture (especially rice and corn) cover 9.5%; pastures 27.2%; reforestation 0.98% and mining 0.04%. The landscape around the system considerably changed in quality and expended in area over the last 30 years (Fig. 20.17). In 1985, the urban area covered 30.19 km2, increasing to 63.08 km^2 in 2010 (Fig. 20.17). Agriculture increased from 71.88 km^2 in 1985 to 227.01 km^2 in 2010, while aquaculture area increased from 0.43 km^2 in 1997 to 9.52 km^2 in 2010.

Fig. 20.16 Anthropogenic impacts affecting the Laguna Estuarine System. (**a**) fishing harbour; (**b**) urban settlement (Laguna city); (**c**) illegal wastewater disposal; (**d**) riverside communities

Remnants of the Atlantic Rain Forest, rice culture, urban areas and holiday homes compose the landscape next to the north region of the Laguna Estuarine Complex (Table 20.1). There, the main economic activity is agriculture that ranges from family/subsistence to the major commercial and mechanized scale. The main cultures are rice, corn, forage for cattle and manioc. In the south region, there is a dominance of the fisheries industry (fishing harbour) (Fig. 20.16a) and ferryboat transport (Table 20.1). The fishing activity is very intense in the system, with artisanal and sportive fisheries present into both the lagoons' complex and coastal waters.

The complexity of multiple uses presents many opportunities and risks to ecological and socio-economic services. It includes nursery for fishes (e.g. croakers, mullets, snooks) and invertebrates (crabs and shrimps); protection for endangered species (e.g. the white sea catfish *Genidens barbus* and the black drum *Pogonias cromis*) (Table 20.1); and support for artisanal fisheries that depends on the living resources from the lagoons. However, dealing with effluents and solid wastes, as well as protecting the margins to a reasonable distance from the water remain a challenge.

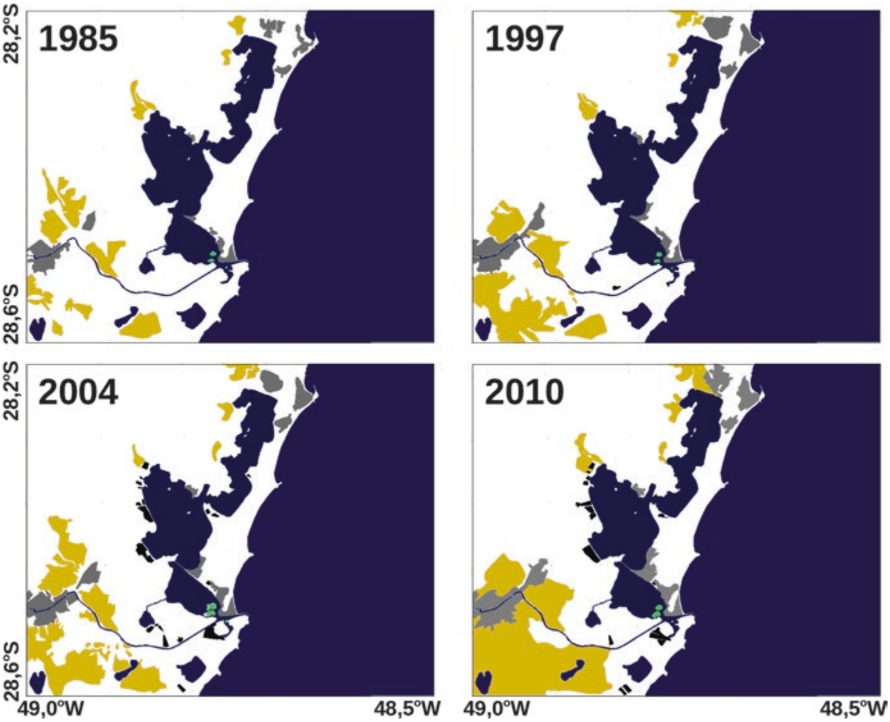

Fig. 20.17 Multitemporal analysis using Landsat 5 Thematic Mapper (TM) bands for land use classification in the Laguna Estuarine System (Method: MAXVER supervised classification). Colours legend: (□) natural landscape (■) agriculture; (■) aquaculture; (■) urban areas; (■) salt marsh; (■) coastal waters (lagoons and coast)

20.4 Fisheries and Fishes Resources

20.4.1 Parnaíba River Delta

Although small-scale industrial fishery is observed in the coastal adjacent areas, fishing activities in the Parnaíba Delta itself are almost exclusively artisanal. The mangrove uçá-crab and the common snook represent approximately 90% of total fishing products (Farias et al. 2015) (Figs. 20.15b and 20.18a–d). The Delta fishery production in 2006 was ~3500 tons (IBAMA 2008a). Finfishes represented 60% of the total production, followed by shrimps (20%), crabs (17%) and oysters (3%) (IBAMA 2008a). A compilation of total finfish production in the Delta from 1999 to 2006 shows that highwaterman catfish (*Hypophthalamus edentatus*), longnose stingray ray (*Dasyatis guttata*) (Fig. 20.15a), mullets (*Mugil* spp.) (Fig. 20.7ab,) and serra mackerel *Scomberomorus brasiliensis* (Fig. 20.15f) production exceeded 100 tons each during the period. While whitemouth croaker (*Micropogonias furnieri*) (Fig. 20.7e) and crucifix sea catfish (*Sciades proops*) productions exceeded

Fig. 20.18 Uçá-crab (*Ucides cordatus*) exploitation in the Parnaíba River Delta Estuaries). (**a**) Boat used in the mangrove forest; (**b**) crab picking; (**c**) commercialization at the port; (**d**) truck-loads of crabs arriving in town

400 tons each (Farias et al. 2015). According to ESTATPESCA data, Piauí State produced more than 6000 tons of fisheries per year between 1999 and 2010, with a maximum of 10,456 tons in 2008 (ESTATPESCA 2010). The most important fisheries resources are the uçá-crab, lane snapper (*Lutjanus synagris*), serra mackerel, king mackerel (*Scomberomorus cavalla*), blackfin tuna (*Thunnus atlanticus*) and acoupa weakfish (*Cynoscion acoupa*) (Fig. 20.7c and Table 20.1).

The period of highest rainfall is recognized as also being the time of greater fishing productivity in the region. The contribution of a large volumes of freshwater into the estuaries increases nutrient loads and productivity, therefore increasing the abundance and diversity of species (Mai and Leobmann 2010; Melo 2012; Lima 2012; Melo et al. 2015). It functions as spawning, growth, protection and feeding grounds of many fish species (Fernandes 2015). The common snook, caught in mesh size of 12 cm between knots, showed highest productivities between February and July 2014, coinciding with the rainy season in the Parnaíba Delta (Farias et al. 2015). Mullets, being omnivorous and pelagic, showed higher production in the months of lower turbidity and higher primary productivity (Farias et al. 2015). However, no significant differences in catches of freshwater shrimps between dry and rainy seasons were observed.

Fisheries at night are limited by the absence of adequate navigation equipment or land-based references, even inside the estuarine system. When possible, they target mainly mullets *M. curema* and *M. rubrioculus* (mullets). Another important fishing resource is the broadband anchovy *Anchoviella lepidonstole* captured with artisanal purse seine nets (Fig. 20.15c). The fishing of mullets and anchovies occur mainly in the dry season, between July and November.

The main fishing gear used along the Delta coastline is the gillnet, which operates drifting or fixed, from the surface to bottom according to the target species, pelagic or demersal (Fig. 20.19a). Fishers start activities with gillnets at dawn and finish at dusk. Activities include 4 to 16 fishing sets per day (depending on production), with drifts of 25 min to 1 h in each set. At Pedra do Sal beach, there is an iconic and important traditional fishing community of Piauí State. They usually catch common snook, acoupa weakfish and king weakfish with gillnets. Reef fishes, such as lane snapper and the tarpon *Megalops atlanticus* are also captured, using hand line.

Vessels are usually small with power engine (12 HP), sailing or rowing. Crew accommodates up to three men, and trips take one to two days at sea (Fig. 20.20a). Boats (90%) are made of wood and measure between 3 and 9 m in length. Larger boats (n = 12) have power engines of up to 60 HP, operate gillnets, store up to 2000 kg per trip of 3–4 days (Fig. 20.20d). Most boats are based at the Canaries, and fish for common snook and acoupa weakfish. These species are usually caught in the rainy season, between December and May.

Currently, there are about 450 fishing boats distributed among villages of the main islands and fishing colonies at coastal beaches. It includes the industrial fleet and ~200 boats form a small artisanal fleet. Approximately 4800 families are involved in this activity. Despite the well described and recognized ecological importance of the Delta, uncontrolled exploitation of natural resources, especially of fisheries, is widespread. Close surveillance and planning need to begin as soon as possible (Fig. 20.1).

20.4.2 Goiana River Estuary

The coastal zone of Pernambuco State has 187 km, with a short continental platform characterized by the presence of beachrocks parallel to the coast (Lana et al. 1996). Along this coast, there are 14 estuaries simply lined to harbouring extensive mangroves, including the Goiana Estuary at its north border with Paraíba State (Braga 2000; Barletta and Costa 2009). Pernambuco is the fifth fish producer in the Brazilian Northeast, with landings estimated at 14,000 tons in 2006. It represented 9.2% of the total landings in the region (IBAMA 2007). Approximately 97.7% of the Pernambuco fisheries production is artisanal, with more than 25,000 fishers involved and forming 34 fisheries colonies (GAP 1998; IBAMA 2001; Lucena et al. 2013).

The Goiana Estuary and its adjacent coastal areas are acknowledged as an important environmental asset not only for the fishery productivity of the state, but also for the presence of traditional fishers communities (Barletta and Costa 2009; IBAMA 2004, 2007). The municipality of Goiana was responsible for ~25% (1500 tons) of fish production in Pernambuco (IBAMA 2004). The main types of fishing boats are canoe, raft, lobster boats and fishing weirs (Figs. 20.19c and 20.20a–e) (Isaac et al. 2006). Inside the estuary, they use small wooden canoes and

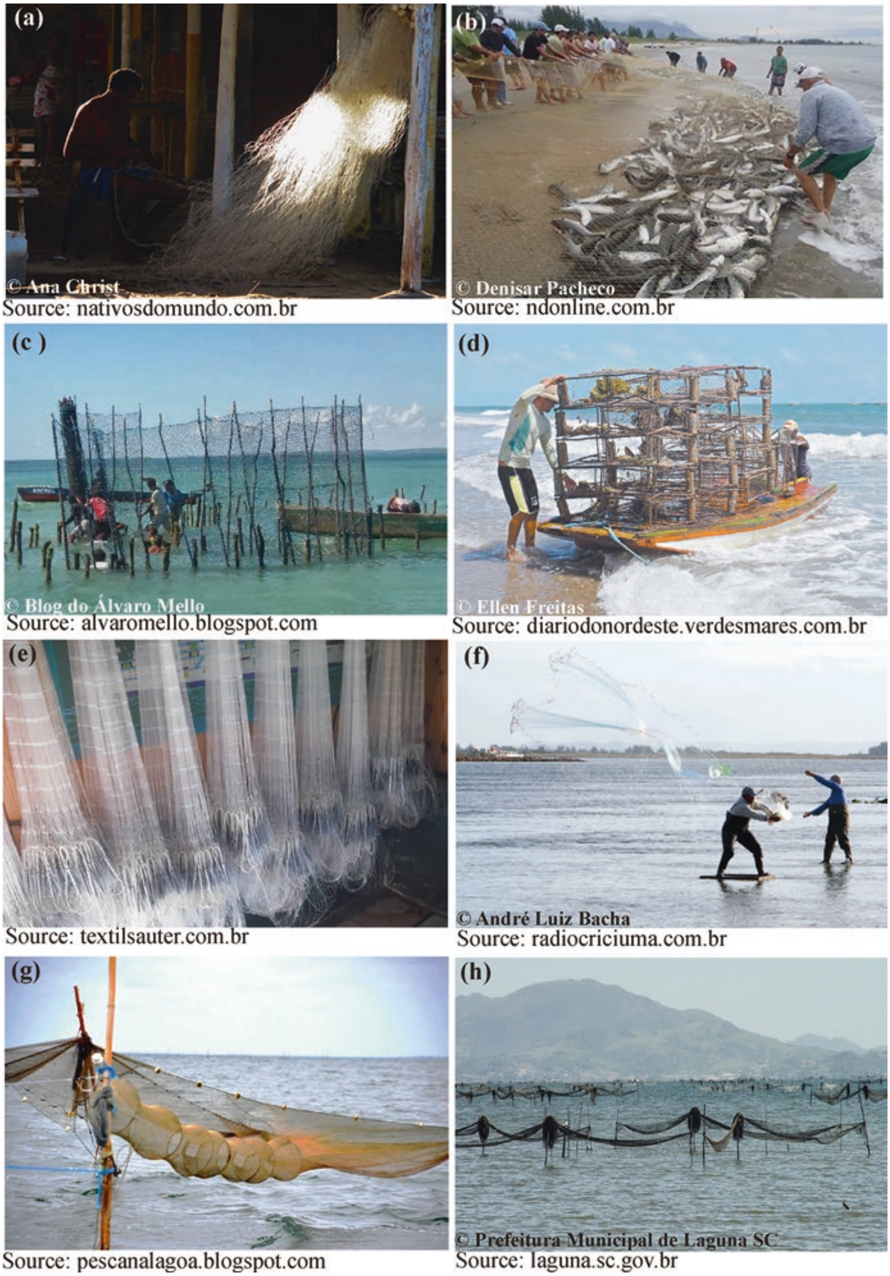

Fig. 20.19 Examples of the main fishing gears used in Western Atlantic estuaries and adjacent coastal areas. (**a**, **b**) small mesh-size gillnets; (**b**) fishing weir (Ponta de Pedras estuarine beach); (**d**) lobster traps; (**d**, **e**) cast nets (mullets capture at Laguna Estuarine System); (**g**) fyke net or "aviãozinho" in the Laguna Estuarine System, used for pink shrimp captures and bycatches (**h**) "aviãozinho" nets suspended after fishery

Fig. 20.20 Examples of the main vessels used in Western Atlantic estuaries and adjacent coastal areas. (**a**) wooden canoe with power engine (**b**) wooden canoe without power engine (**c**) lobster boat; (**d**) small vessel used in coastal areas of Laguna Estuarine System ("baleeira"); (**e**) most common vessel for coastal and estuarine fishery in the Goiana River Estuary; (**f**) seine vessels of the industrial fishery of the Laguna Estuarine System, for mullets fishing

rafts propelled by rows or singe triangular sails (Barletta and Costa 2009). In adjacent coastal waters, reefs and inner shelf fisheries requires more investment in boats that include engine, gear and crew (Barletta and Costa 2009).

The fishing weir technique began in the north coast of Pernambuco in 1694 (Silva 2001) (Fig. 20.19c). In Goiana there were once 29 weirs distributed along its estuarine beaches (Pitimbu, Acaú, Ponta de Pedra, Catuama and Barra de Catuama).

These beaches are shallow areas protected by sand banks, with maximum depth of 1.5 m during spring tides (Lucena et al. 2013). In the early 1990s, fishing with weirs was responsible for 11.2% of the total production of Pernambuco. The production increased until 1996, when 268.3 tons were registered. This figure was maintained around 234 tons in the three following years. From 1999 began a decreasing trend until 2004, when a production of 105 tonnes was recorded. Despite the increased production in 2006 (159 tons), the main factor that reduced this activity was the Environmental Crimes Law of 1998, which prohibited the cutting and use of the mangrove wood rods necessary to build the weirs. Therefore, fishers needed to nearly stop using this gear because alternative woods were more expensive than the product. The result was that fishing weirs lost importance when compared to other fishing activities.

Between 1999 and 2006 the number of canoes operating at the harvest at fishing weirs, decreased 65.6%, from 128 to 44 in 2004. In the two last years (2005 and 2006), 73 canoes were registered as operating at the weirs, however, this increase did not recover the production levels observed in the early 1990s (IBAMA 2005, 2006, 2007; Lucena et al. 2013). In 2006, weirs production represented only 1.14% of the total production of Pernambuco (IBAMA 2007; Lucena et al. 2013).

Fisheries at the Goiana Estuary are artisanal. Gillnet with small mesh size (<60 mm) is the most common gear (Fig. 20.19a, b), although others as trap barriers (seasonally), longline, lobster trap (Fig. 20.19d), spear diving, hook and line are also used (Table 20.2) (Guebert-Bartholo et al. 2011). Only men capture fish. Techniques include diving/spear in reef areas for fishes; otter trawl nets in coastal area for prawns; and cast nets and gill nets in the estuarine and coastal areas for fish and prawns captures (Barletta and Costa 2009).

Fishers from Goiana Estuary are distributed in communities at Goaina (Baldo do Rio, São Lourenço and Carne de Vaca), Caaporã and Pitimbu (Acaú) cities (Barletta and Costa 2009; Silveira et al. 2013). Fishers from Baldo do Rio and Caaporã move seaward in small wooden canoes, called "caícos" (Fig. 20.20b), aiming at fishing estuarine species with a rectangular trap net deployed at the entrance of mangrove tidal creeks, or use bottom gill nets ("caçoeira") to capture demersal fishes crossing the main channel (Guebert-Bartholo et al. 2011; Silveira et al. 2013). These communities are also involved in the capture of the uça-crab, using small canoes to reach the mangrove flooded forest (Barletta and Costa 2009; Silveira et al. 2013). The main landing port at the middle reaches of the estuary is the Gongaçari community. Shrimp fishing with trap and fyke nets was once important for these two communities. Currently, estuarine shrimps are less abundant and their captures are restricted to beaches and other coastal areas using seine nets (Silveira et al. 2013). São Lourenço is the municipality closest to the estuarine mouth, on the margins of Megaó River, a tidal creek that drains to Goiana Estuary (Silveira et al. 2013). The community there is the poorest, with a high proportion of crab, oyster and mussel pickers among its population. Carne de Vaca (Pernambuco) and Acaú (Paraiba) are located at the mouth of the Goiana Estuary, on estuarine beaches (Barletta and Costa 2009; Silveira et al. 2013). Mussels, crabs, shrimps, lobsters and finfish exploitation is common for these both fishing communities.

Table 20.2 Finfish production of Goiana River Estuary artisanal fleets between 2013 and 2015 within the extractive reserve Acaú-Goiana" and adjacent coastal areas

				2013		2014		2015	
Common name	Species	Total (kg)	(%)	Dry season	Rainy season	Dry season	Rainy season	Dry season	Rainy season
White mullet	*Mugil curema*	11,970.1	24.07	1672.0	253.4	3276.6	1923.3	4232.9	611.9
Longnose stingray	*Dasyatis guttata*	10,114.0	44.42	66.0	52.0	2414.5	5790.0	1183.0	608.5
Cobia	*Rachycentron canadum*	3562.8	51.58	93.0	20.0	598.3	961.5	1232.5	657.5
Common snook	*Centropomus undecimalis*	3608.4	58.84	320.0	967.9	715.3	707.0	628.5	269.7
Brazilian mojarra	*Eugerres brasilianus*	2310.4	63.49	431.4	620.7	405.6	584.8	222.8	45.0
Mutton snapper	*Lutijanus analis*	1425.7	66.35	139.8	45.7	346.1	482.4	296.0	115.7
Acoupa weakfish	*Cynoscion acoupa*	1305.8	68.98	213.4	270.6	358.2	318.2	105.0	40.4
Mackerel	*Scomberomorus japonicus*	1223.7	71.44	46.5	31.9	510.9	362.9	160.0	111.5
Lebranche mullet	*Mugil liza*	1183.9	73.82	67.5	428.5	103.5	452.2	100.7	31.5
Crucifix sea catfish	*Arius proops*	1192.7	76.22	190.0	121.6	206.1	288.3	360.2	26.5
Florida pompano	*Trachinotus carolineus*	1183.7	78.60	196.2	421.0	211.9	248.4	84.6	21.6
Lane snapper	*Lutjanus synagris*	1051.4	80.72	52.4	154.5	293.0	403.5	148.0	0.0
Dog snapper	*Lutjanus jocu*	993.5	82.71	75.0	2.0	215.8	404.0	239.7	57.0
Sailor's grunt	*Haemulon parra*	817.5	84.36	99.0	106.0	49.5	403.0	160.0	0.0
Sheepshead	*Archosargus probatocephalus*	880.8	86.13	63.5	200.9	257.7	324.5	32.0	2.2
Crevalle jack	*Caranx hippos*	728.1	87.59	9.5	0.0	179.1	484.0	44.5	11.0

(continued)

Table 20.2 (continued)

Common name	Species	Total (kg)	(%)	2013 Dry season	2013 Rainy season	2014 Dry season	2014 Rainy season	2015 Dry season	2015 Rainy season
Atlantic barracuda	*Sphyraena marina*	619.7	88.84	12.0	23.0	126.2	333.0	81.5	44.0
Nurse shark	*Ginglymostoma cirratum*	609.5	90.07	204.0	0.0	59.0	201.5	132.5	12.5
Stoplight parrotfish	*Sparisoma viride*	580.4	91.23	0.0	0.0	80.5	190.5	240.4	69.0
Blue runner	*Caranx crysos*	559.0	92.36	0.0	70.5	209.0	216.0	63.5	0.0
Tarpon	*Megalops atlanticus*	387.0	93.14	0.0	0.0	101.0	0.0	286.0	0.0
Atlantic thread herring	*Opisthonema oglinum*	364.0	93.87	70.0	0.0	86.0	42.0	166.0	0.0
Black grouper	*Mycteroperca bonaci*	347.0	94.57	48.8	4.2	158.5	129.0	2.0	4.5
Common dolphinfish	*Corypheana hippurus*	277.9	95.12	24.0	0.0	211.5	21.4	21.0	0.0
Roughneck grunt	*Pomadasys corvaeniformis*	284.0	95.70	82.0	98.5	36.0	60.5	7.0	0.0
Atlantic moonfish	*Selene stapinnis*	243.0	96.18	1.0	31.3	68.6	104.5	28.0	9.6
Little tunny	*Euthynnus alletteratus*	223.5	96.63	12.0	0.0	100.0	26.5	85.0	0.0
Torroto grunt	*Genyatremus luteus*	218.9	97.07	76.4	42.9	31.6	22.0	31.5	14.5
Atlantic spadefish	*Chaetodipterus faber*	178.0	97.43	0.0	0.0	3.0	28.0	100.5	46.5
Agujon needlefish	*Tylosurus acus*	155.5	97.74	0.0	0.0	140.5	15.0	0.0	0.0
Horse-eye jack	*Caranx latus*	128.8	98.00	4.0	0.0	45.8	55.0	21.5	2.5
Yellowfin tuna	*Thunnus albacares*	115.0	98.24	44.5	0.0	46.5	12.0	12.0	0.0

Redband parrotfish	*Sparisoma aurofrenatum*	114.0	98.46	0.0	7.0	24.0	83.0	0.0	0.0
Spotted goatfish	*Pseudupeneus maculatus*	105.5	98.68	0.0	0.0	29.0	42.0	23.0	11.5
White grunt	*Haemulon plumierii*	99.0	98.88	0.0	0.0	29.5	35.0	29.5	5.0
Slender halfbeak	*Hemirhamphus roberti*	84.9	99.05	3.8	19.0	6.1	15.0	21.5	19.5
Black margate	*Anisotremus surinamensis*	75.9	99.20	32.1	0.0	11.0	1.5	17.8	13.5
Brazilian sardinella	*Sardinella brasiliensis*	52.5	99.30	0.0	0.0	8.5	44.0	0.0	0.0
Whitemouth croaker	*Micropogonias furnieri*	56.0	99.42	0.0	6.5	27.5	22.0	0.0	0.0
Cubera snapper	*Lutjanus cyanopterus*	45.0	99.51	0.0	0.0	5.5	29.0	6.0	4.5
Largehead hairtail	*Trichiurus lepturus*	42.5	99.59	0.0	0.0	10.0	20.0	12.5	0.0
Barbu	*Polydactylus virginicus*	31.5	99.66	17.0	4.5	0.0	10.0	0.0	0.0
Tomtate grunt	*Haemulon aurlineatum*	31.0	99.72	0.0	14.5	0.0	16.5	0.0	0.0
Jamaica weakfish	*Cynoscion jamaicensis*	19.2	99.76	0.0	0.0	0.0	19.2	0.0	0.0
Shorthead drum	*Larimus breviceps*	19.0	99.80	3.5	0.0	15.5	0.0	0.0	0.0
Tripletail	*Lobotes surinamensis*	16.4	99.83	4.0	3.0	4.4	0.0	5.0	0.0
Sand drum	*Umbrina coroides*	15.5	99.86	4.0	0.0	0.0	11.5	0.0	0.0
Chiraa grunt	*Haemulon squamipinna*	14.0	99.89	0.0	0.0	0.0	14.0	0.0	0.0

(continued)

Table 20.2 (continued)

Common name	Species	Total (kg)	(%)	2013		2014		2015	
				Dry season	Rainy season	Dry season	Rainy season	Dry season	Rainy season
Atlantic goliath grouper	*Epinephelus itajara*	14.0	99.92	0.0	0.0	0.0	0.0	14.0	0.0
Largescale fat snook	*Centropomus mexicanus*	12.5	99.94	0.0	12.5	0.0	0.0	0.0	0.0
Atlantic bumper	*Chloroscombrus chrysurus*	8.5	99.96	0.0	0.0	0.0	8.5	0.0	0.0
Ground croaker	*Bairdiella ronchus*	6.0	99.97	0.0	6.0	0.0	0.0	0.0	0.0
Guiana longfin herring	*Odontognathus micronatus*	4.5	99.98	4.5	0.0	0.0	0.0	0.0	0.0
Spotted eagle ray	*Aetobatus narinari*	4.0	99.988	4.0	0.0	0.0	0.0	0.0	0.0
Rainbow runner	*Elagatis bipinnulata*	3.0	99.994	0.0	0.0	0.0	3.0	0.0	0.0
Burro grunt	*Pomadasys crocro*	1.0	99.996	0.0	1.0	0.0	0.0	0.0	0.0
Bluefish	*Pomatomus saltator*	1.0	99.998	1.0	0.0	0.0	0.0	0.0	0.0
Permit	*Trchinotus falcatus*	1.0	100	0.0	0.0	1.0	0.0	0.0	0.0
	Total	49,721.0		4387.8	4041.1	11,817.8	15,969.1	10,638.1	2867.1

Mollusks are regarded as a subsistence resource by only 4.2% of the families around the Goiana Estuary. However, a study with 360 subjects estimated that 69.7% of mussel pickers in the extractive reserve, and 88.2% in the surrounding non-protected areas report mussel picking as their main source of subsistence and income generation (Barletta and Costa 2009; Silva-Cavalcanti and Costa 2009). Oysters (*Crassotrea rhizophorae*) are collected by men, diving or removing from red mangrove roots (Fig. 20.10c). Charru mussel (*Mytella* spp.) is also collected at sub-tidal areas by a small group of men at the lower estuary, near Acaú (Fig. 20.10d). In total, approximately 450 families (5 person per family resulting in 2250 people) are shellfish pickers focused on Caribbean venerid clam (*A. brasiliana*) and stout mussel (*T. plebeius*) (Figs. 20.10a, b and 20.21a–c). This activities are performed during at least the 4 h of the day period low tide at extensive sand banks of the lower estuary and estuarine beaches (Barletta and Costa 2009; Silva-Cavalcanti and Costa

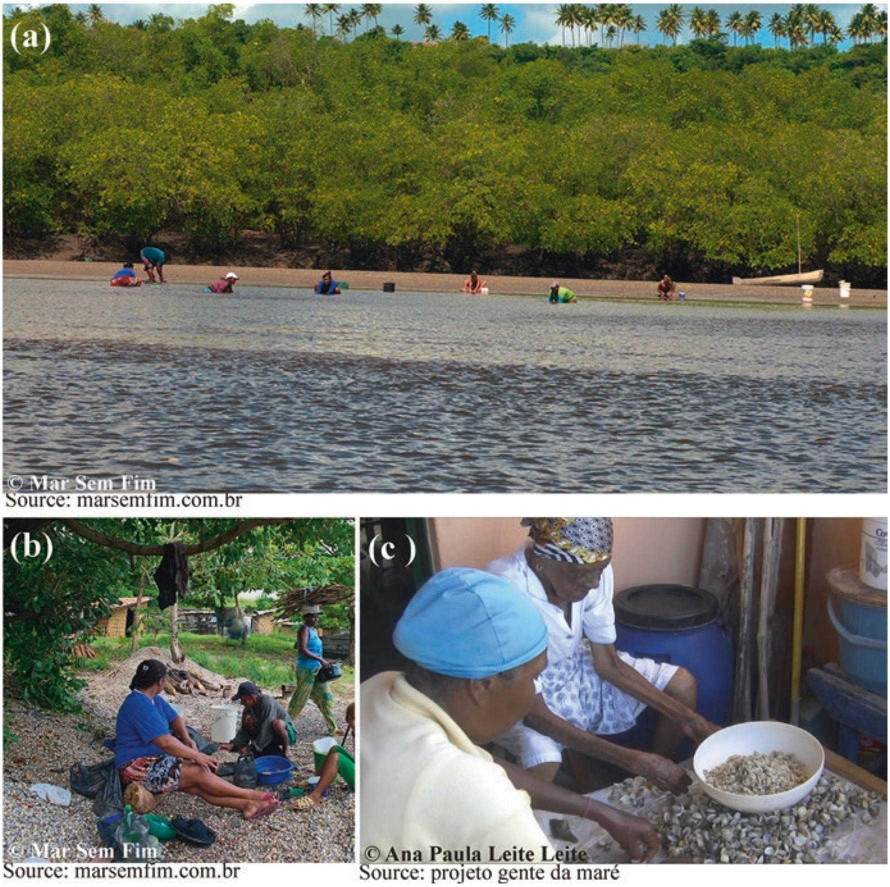

Fig. 20.21 Picking of Caribbean venerid clam (*Anomalocardia brasiliana*) at the extractive reserve "Resex-Acaú-Goiana". (**a**) Shellfish pickers; (**b**, **c**) preparation of the product for sale

2009). Women (82.4%) are dedicated to mussel picking within the extractive reserve, while in the non-protected area this activity is divided with men (50%) (Silva-Cavalcanti and Costa 2009). The CPUE varies from 1.15 to 2.2 kg of clean meat per picker per hour (Silva-Cavalcanti and Costa 2009).

Traditional populations commonly exploit crabs. Picking (or traping) of land mangrove crabs is made by men, while all family members might fish swimming crabs using line and hook or nets with baits (Barletta and Costa 2009). In the 1990s, coinciding with the introduction of shrimp farming projects in the northeast Brazilian coast, a major viral epidemic decimated the population of uçá-crabs in a wide area, including Pernambuco. The crab pickers began to explore other crustaceans as wimming crab (*Callinectes* spp.), blue-crabs (*Cardisoma guanhumi*) and mangrove tree crab (*Aratus pisoni*) (Fig. 20.8a–d). In 2010, the land crab population began to be restored in the Goiana Estuary. However, the introduction of a small predatory net for land crab capture began to reduce the already fragile populations for the second time (Silveira et al. 2013). This method uses of a tangle of thin nylon or plastic threads usually obtained from ice bags. At low tide, these traps are placed at the entrances of crab burrows, as a cork. Crabs entrap in these bundles of filaments when they try to leave their burrows, or wish to return to them (Silveira et al. 2013). So they are easily captured even in non-commercial sizes, in which case most are not returned to the environment. The uçá-crab picking used to be intense during the first trimester of the year when the crabs leave their burrows to reproduce and are more vulnerable to capture (Silveira et al. 2013). Since 2003, crab capture is a closed for during part of summer months. Since there is no efficient enforcement through close surveillance, traditional communities may take advantage of the animals (Silveira et al. 2013).

Commercial fisheries for shrimps (*F. subtilis*, *Litopenaeus schimidt*, *Xiphopnaeus kroyeri*) take place during the rainy season (March to August) at the estuarine beaches and adjacent coastal waters, especially with seine nets (Fig. 20.9a, b). However, lobster (*Panulirus argus* and *P. laevicauda*) is the most profitable catch and 53% of fishers are dedicated to its capture (Guebert-Bartholo et al. 2011) (Fig. 20.9e, f). Lobsters are captured in depths of >30 m with nets and a number of different models of traps at the coastal waters of the continental shelf (Barletta and Costa 2009). Although diving with on board-based air compressors for lobster capture is illegal, this activity is clearly common (Guebert-Bartholo et al. 2011). Lobster exports generates incomes in US dollars. For this reason, non-traditional stakeholders encourage the competitive capture and commercialization of lobsters, recruiting young men for trips that last ~30 days at sea, in boats with very precarious conditions, and whose payment is agreed in bycatch yield above boat costs (Fig. 20.20c). Therefore, both resources and fishers are over-exploited (Table 20.1). Seasonal closure for lobster populations recovery is enforced by law from December to March (late dry and early rainy seasons). However, fishers admit to fish off-season (Guebert-Bartholo et al. 2011). In Acaú the lobster fleet is the most profitable economic activity (Barletta and Costa 2009; Silveira et al. 2013).

Although subsistence fisheries occur along the whole estuary, they are restricted to dryer months because during the late rainy season (June to August) freshwater moves seaward down to the lower estuary and carry fishes out to coastal waters (Barletta and Costa 2009). Estuarine fish species, such as catfishes (*C. spixii*, *C. agassizi* and *S. hertzbergii*) are mainly captured for subsistence since they are highly abundant in the estuary (Dantas et al. 2010). Commercial fisheries target different species, such as acoupa weakfish, snooks (*C. undecimalis*, *C. pectinatus*, *C. mexicanus*), snappers, Brazilian mojarra, halfbeaks (*Hemiramphus* spp.), mullets and cutlassfish (*T. lepturus*) (Fig. 20.7a–f and 20.15b, e). Estuarine fishing is common for all communities living along its margins, however, the most important activity in Carne de Vaca is coastal fishing (Barletta and Costa 2009; Silveira et al. 2013).

Current data on fishery landing counts on an extensive and ever growing data bank about the Goiana Estuary (see grant n° 405818/2012-2 /COAGR/CNPq). A report analyzed the performance of the artisanal fleet between 2013 and 2015 within the extractive reserve Acaú-Goiana and surrounding coastal areas to determine fishery productivity related to the estuary (Table 20.2). Fishing occurs inside the estuary and along 47 km of coastal area, and up to 35 km offshore (Fig. 20.22). Nearshore

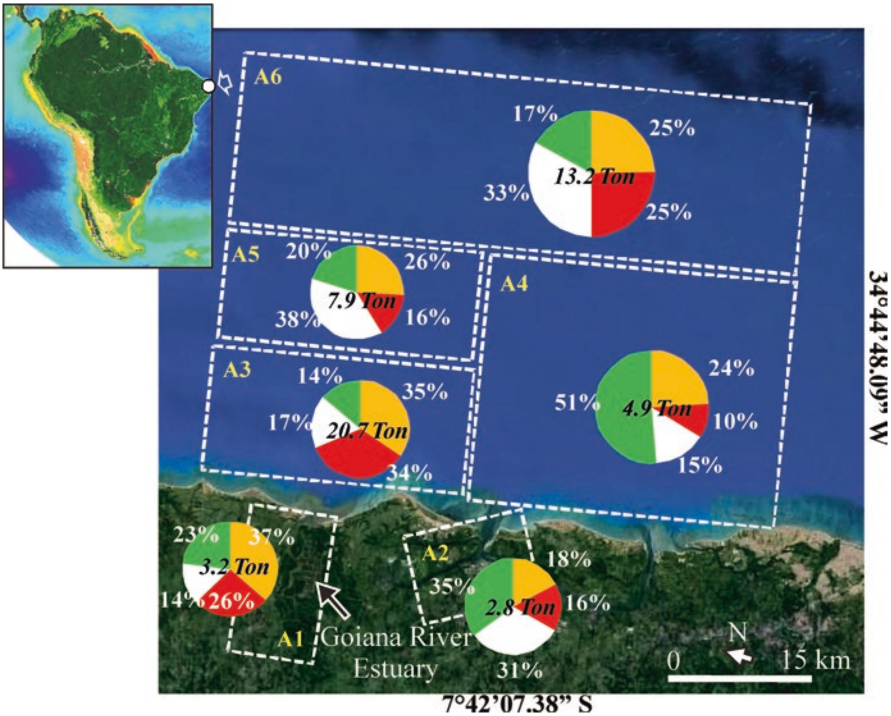

Fig. 20.22 Distribution of the artisanal fishery fleet activity and production at the extractive reserve "Resex-Acaú-Goiana" and adjacent coastal areas relative to space. (*A1*) Goiana River Estuary; (*A2*) adjacent estuaries; (*A3*) nearshore coastal waters – northbound (9.5 km²); A4: nearshore coastal waters – southbound (37.5 km²); (*A5*) nearshore area farthest from the estuarine mouth; (*A6*) offshore. Colours legend: (□) early rainy season; (■) late rainy season; (■) early dry season; (■) late dry season

fishing occurs from the estuarine mouth northwards for 9.5 km (Pitimbú beach), and 37.5 km in the south direction, covering five estuarine beaches (Fig. 20.22).

The monitoring of fishing landings began in 2012 at the main landing sites. Daily production of the artisanal fishing fleet, based at Acaú and Pitimbú was recorded and analysed. From January 2013, two observers native of traditional communities carried out the daily monitoring of landings. Although information requires more in depth analysis, some important conclusions can already be drawn, which justifies the continuation of this important study for the management of artisanal fisheries in the extractive reserve.

During 36 months, ~50 tons of fish were captured, which includes 58 target species of the local fleet (Table 20.2). Of these, 12 species accounted for 80% of the catch (e.g. cobia, common snook, mojarra, mutton snapper, acoupa weakfish) (Table 20.2). The most important fishes were white mullet (*M. curema*, ~12 tons) (Fig. 20.7b) and longnose stingray ray (*Dasyatis guttata*, ~10 tons) (Fig. 20.15a) (Table 20.1). Moreover, it was possible to detect seasonal trends in capture (Fig. 20.22). Among the areas used by the fishing fleet the nearest coastal area to the Goiana Estuary was the one with the highest catch (~21 tons) (Fig. 20.22 A3). Seventy percent of the capture took place during the dry season (September to February) when species are grouped to reproduce near the estuary entrance (Fig. 20.22A3). During the end of the rainy season (June–August), the increased precipitation in the region increases the river flow and salinity at coastal regions decreases. Consequently, fishery production also decreases. At the same time, the fishing activities in the coastal region further south of the estuary mouth increases (Fig. 20.22A4).

Seasonality in fish production was also detected for average daily landing, calculated from the catch per unit effort (kg per fisher per day) (Fig. 20.23). Each year had a fishing seasonal trend, following rainfall. On average, days with higher rainfall rates had the lowest CPUE. Importantly, precipitation patterns were not similar among years, which significantly influenced landings and fish production (Fig. 20.23). These results indicate that the region is being influenced by climatic cycles longer than 12 months. Thus, it is necessary to continue recording daily data for at least another 36 months in order to infer fisheries productivity and propose more consistent management options for the extractive reserve.

20.4.3 São Francisco River Estuary

The estuarine region of the São Francisco River is dominated by small-scale fishing, as observed in the entire northeast coast of Brazil. However, it is a significant activity to local communities (Rangely et al. 2010). There are about 15 boats regularly landing at Piaçabuçu and 70 at Pontal do Peba beach, in the same city. Fishing activities are observed across the continental shelf between 10°20′ S and 10°50′ S, including the north and south portions of the river mouth, especially in shallow waters between the 15 and 20 m (Coelho and Santos 1995).

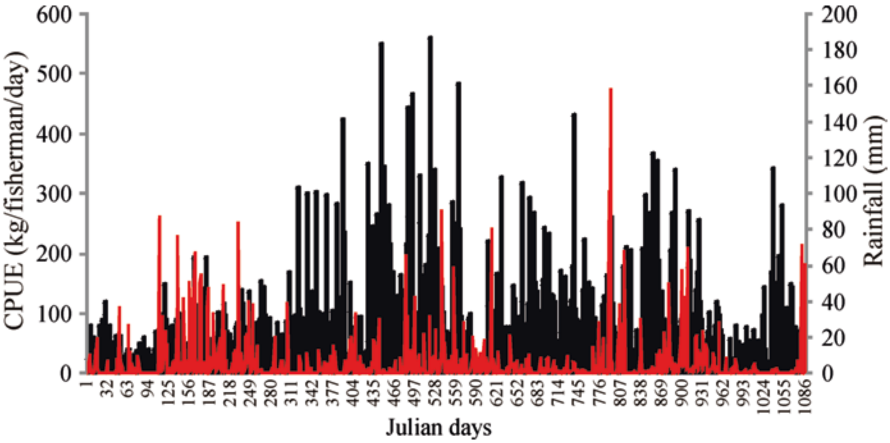

Fig. 20.23 (■) Average daily production (kg) per fisher at the extractive reserve "Acaú-Goiana" and adjacent coastal areas; and (■) daily precipitation between January 2013 and December 2015. Counts in Julian days

Of the many fishing gear used in the region, gillnets are the most important (Fig. 20.19a–b). Also present are handlines, trawls (near the coast), falling nets, artisanal longline ("groseira"), traps ("covos") (Fig. 20.19c), among others. Artisanal fishers combine small business for income and subsistence, employing small to medium-sizedwood boats, acquired from small shipyards, with power engine or not (Oliveira 2012), as well as boats built by themselves using natural materials (Fig. 20.20a, b) (Santos et al. 2005).

Motorized fishing in northeast states of Brazil is focused on shrimp because this region has one of the largest shrimp banks (Tonial 2011). The shrimp fishery operates along the coast, where the estuarine main channel provides organic matter for the development of these organisms, usually where finer sediments deposit to form wide muddy bottoms. Evaluating the main catches in the estuary and adjacent areas during 2014 and 2015, there is a greater representation of penaeid shrimp, highlighting the Atlantic seabob shrimp (*Xiphopenaeus kroyeri),* with 45% (649.3 tons) of landed biomass. The Atlantic sea bob shrimp had higher catches in the dry period (August–November) (Fig. 20.24a and Table 20.3).

Whereas less representative, 71.68 tons (5%) landed at Piaçabuçu between 2014 and 2015, the white shrimp (*L. schmitti*) is the most profitable fishery resource per kg (Fig. 20.24b and Table 20.3). Their highest production occurred in May–July in 2014 (14.61 tons) and July–September in 2015 (18.91 tons), presenting higher catches during the rainy season (Fig. 20.24b). February was the less productive month in both years for both species, sea bob and white shrimps, corroborating the previous results (Coelho and Santos 1995).

Freshwater shrimps (*Macrobrachium* spp.) represent ~2% of the landings in the region (Fig. 20.24c and Table 20.3). They are caught using mobile traps called

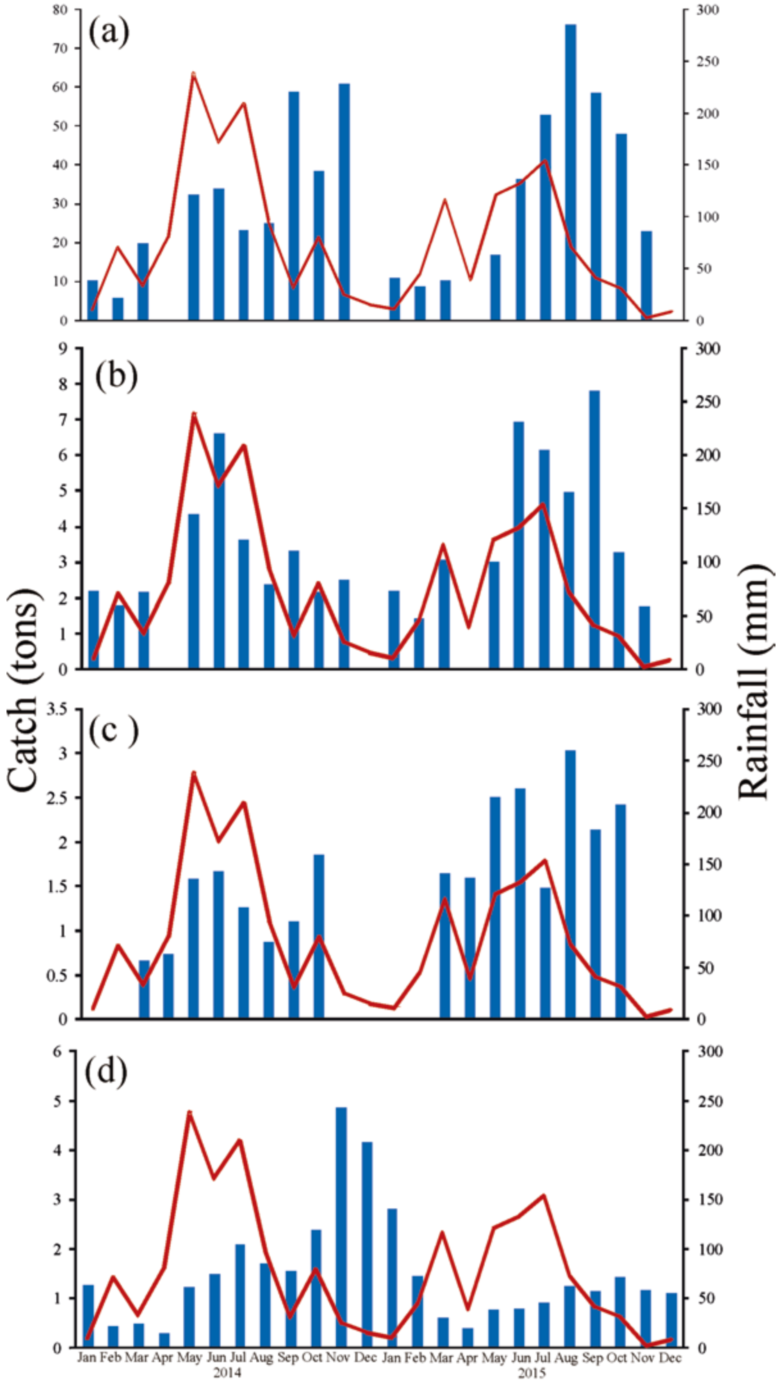

Fig. 20.24 (■) Monthly total catches of (**a**) Atlantic seabob shrimps; (**b**) white shrimps; (**c**) freshwater shrimps; (**d**) sand crabs; and (■) rainfall during 2014 and 2015 (Source: CODEVASF and field data)

Table 20.3 Fisheries production of São Francisco River Estuary and adjacent coastal areas landed at Piaçabuçu port during 2014 and 2015

Common name	Species or family	2014	2015	Total
Atlantic seabob shrimp	*Xiphopenaeus kroyeri*	307.95	341.35	649.3
Anchovys	Engraulidae	123.22	107.94	231.16
Shrimp trawl bycatch	Sciaenidae, Gerreidae, Haemulidae, Dasyatidae, Trichiuridae, Ariidae, Bothidae, Serranidae, Aethridae, Pristigasteridae, Cynoglossidae, etc.	53.41	81.96	135.37
Mojarras	Gerreidae	25.71	50.82	76.53
Jacks	*Carnx* spp.	36.57	36.59	73.16
White shrimps	*Litopenaeus schmitti*	31.13	40.55	71.68
Snooks	*Centropomus* spp.	22.5	26.4	48.9
Sand crabs	Portunidae	21.9	11.59	33.49
Sharks	Carcharhinidae, Sphyrnidae	13.87	19.1	32.97
Mullets	*Mugil* spp.	12.5	19.8	32.3
Rays	Dasyatidae	11.44	17.4	28.84
Freshwater shrimps	*Macrobrachium* spp.	9.76	17.45	27.21

creels. Increased catches (from 4.52 to 7.60 tons), especially in the period between May and July 2014 and from August to October in 2015, were observed (Fig. 20.24c). Captures are also associated to higher rainfall (Fig. 20.24c). Shrimp trawling bycatch is responsible for the capture of 135.37 tons (10%) of a variety of organisms at different life stages. It includes high captures of Teleostei, Elasmobranchii, and crustaceans (Table 20.3).

For crustacean production, sand crabs showed monthly production from 0.39 to 4.87 tons in the studied period (Fig. 20.24d and Table 20.3). Between October 2014 and January 2015 (dry season), it showed higher catches (Fig. 20.24d). They are caught using "pituqueiras", a trap made of a metal ring with a piece of gillnet in the center and a multifilament polyethylene cord to launch and collect the crabs.

Anchovies (Engraulidae) accounted for 16% (231.16 tons) of the catches, representing the second largest biomass landed in the estuary (Table 20.3). Anchovies showed a monthly average production of 9.63 tons between 2014 and 2015, and little variation of monthly biomass between rainy and dry seasons. However, from September to December 2015 lower catch rates were observed (Fig. 20.25a), related to successive river water flow reductions in this period that further modified the river and adjacent regions. Anchovies fishing uses surface gillnets in the river and estuary and occurs daily, from dusk to dawn. There is no pre-established closed season for the capture of anchovies, mainly due to their biological characteristics (high recruitment and rapid maturation) contributing to apparently sustainable high catches. However, there is a great effort (number of fishers per day) put to their capture. Sampaio et al. (2015) stated that the vaillant's anchovy *Anchoviella vaillanti* was the most abundant engraulid species, with average CPUE of 8 kg per fisher per day.

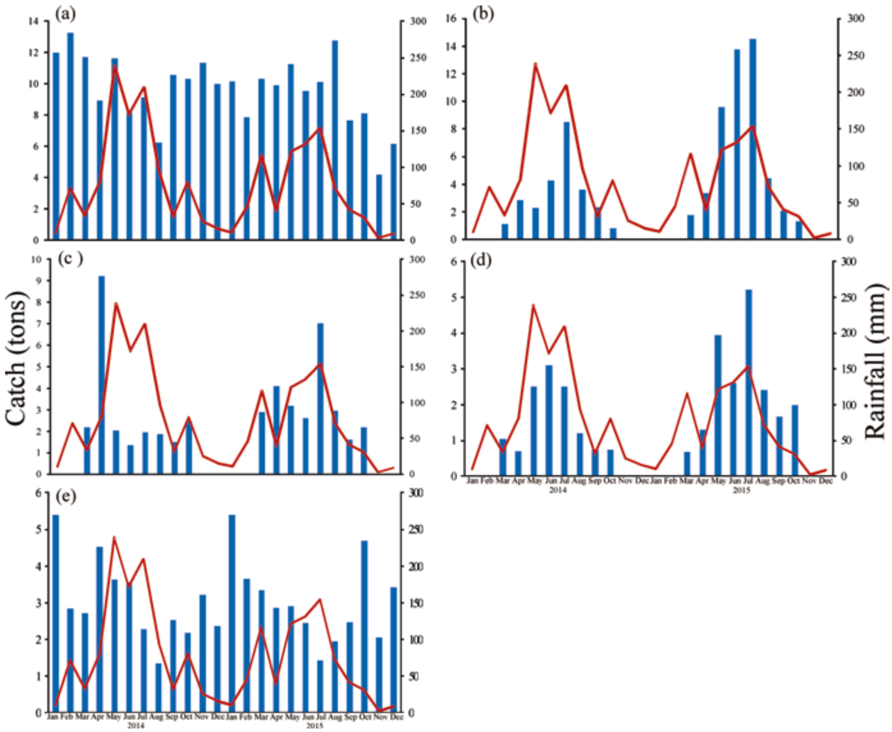

Fig. 20.25 (■) Monthly total catches of (**a**) anchovies (Engraulidae); (**b**) mojarras (Gerreidae); (**c**) snooks (Centropomidae); (**d**) mullets (Mugilidae); (**e**) jacks (Carangidae); and (■) rainfall during 2014 and 2015 (Source: CODEVASF and field data)

Mojarras (Gerreidae) are captured in the river and estuary with midwater gillnets, and yield a total production of 76.53 tons (5%) (Table 20.3). Higher catches are observed in June and July in both years studied, 2014–2015 (Fig. 20.25b). Snooks (Centropomus spp.) are captured with mono and multifilament nets and represents 2% of the captures (48.9 tons) (Table 20.3). They are the most profitable finfish, being much appreciated in the whole region. During the study period, the highest catches occurred in April 2014 and July 2015, when 9.21 and 7.01 tons were landed in the region, respectively (Fig. 20.25c). Mullets (Mugil spp.) are captured with midwater gillnets producing 32.3 tons (Table 20.3). They are also economically important. They had higher catches from May to July in both years 2014 and 2015 (Fig. 20.25d). Mojarras, snooks and mullets have their highest captures associated to the rainy season (Fig. 20.25a–c). However, for the jacks (Caranx spp.), August 2014 and July 2015 were the months with lower catches (1.33 and 1.42 tons, respectively) (Fig. 20.25e). Although higher catches are observed in rainy months, the highest catches are associated to the dry season, especially in January

(Fig. 20.25e). Jacks are captured with multifilament gillnets and hand lines. During the analyzed period (2014–2015) 73.16 tons (5%) of mullets were landed at the Piaçabuçu (Table 20.3).

In 2015, there was an increase of 27.4 and 35.9% in the catch of sharks and rays, respectively, compared to 2014 (Table 20.3). This increase can be related to a higher fishing effort along the coast before the closed period of fishing in the river (November to February), and fishers directing efforts towards resources as elasmobranches. Capture of sharks and rays in the river result from intrusion of saline water in the estuarine environment, caused by changes in river flow. Sharks are caught by artisanal longline ("groseira"), gillnets and hand lines. They are landed at Piaçabuçu on an average of 1.37 tons per month. February 2015 was the month with the highest production (3.6 tons) (Fig. 20.26a). Rays are mainly as shrimp trawling bycatch, or with bottom handlines. During the study period, the average production was 1.20 tons per month. In January 2015 reached 3.55 tons (Fig. 20.26b). The periods in which catches of sharks and rays were higher are also the driest months,

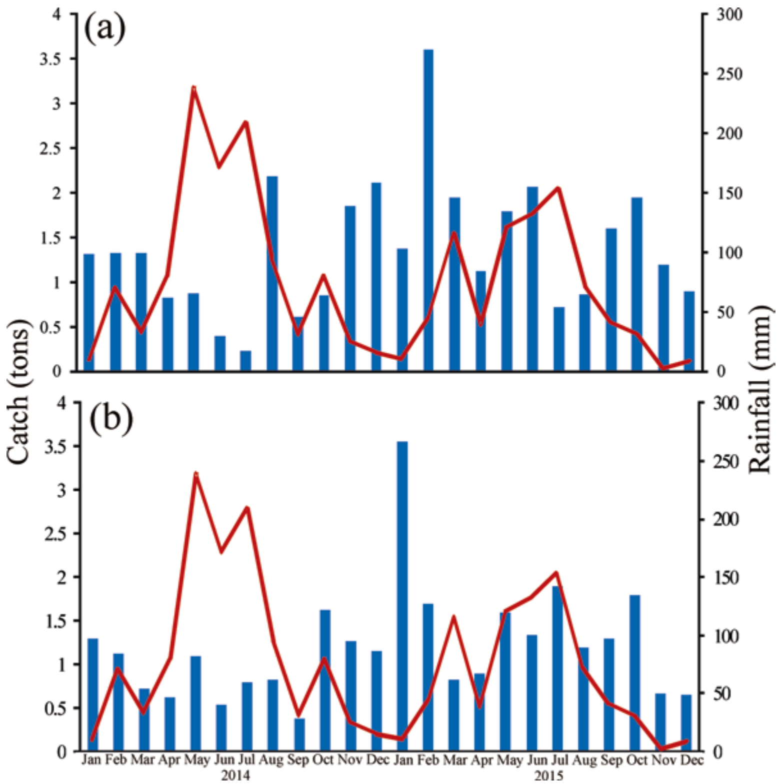

Fig. 20.26 (■) Monthly total catches of (**a**) sharks (Carcharhinidae, Sphyrnidae); (**b**) rays (Dasyatidae); and (■) rainfall during 2014 and 2015 (Source: CODEVASF and field data)

between October 2014 and February 2015 (Fig. 20.26a,b), corroborating the hypothesis of saline intrusion leading to fish community changes.

20.4.4 Laguna Estuarine Complex

The main living resources exploited by the fishing activity in the area are teleost fishes and crustaceans (Table 20.1). Artisanal fisheries are important to Santa Catarina State, with 25 thousand fishers distributed among 186 fisheries colonies accounting for 30% of the fishery production in the state (EPAGRI/CEPA 2010). In the lagunar system, 6223 fishers are directly involved with artisanal fisheries, representing around 16% of all artisanal fishers registered in the state (EPAGRI/CEPA 2010). Usually, local fishers use artisanal boats to assist in their activities. Wooden canoes with outboard (or central) power engine are the most used (Fig. 20.20a). These artisanal fishing boats have a rustic design and are not used to navigate open waters. The fishing boats adopted for fishing in the coastal marine zones are longer, with more powerful engines and higher bow height when compared to the boats used in the lagoons (Fig. 20.20d, e). The main boats adopted by this fleet are “baleeira” (Fig. 20.20d) and “botes” (Fig. 20.20b).

The artisanal fishing in the study region is developed in the coastal marine zone and inside the lagoons and channels. According to Nédélec and Prado (1990) classification, it is possible to identify 11 main artisanal fishing methods in these two regions. In the coastal marine zone, gillnet (Fig. 20.19a, b), driftnets, bottom longlines, purse and beach seines, cast nets (Fig. 20.19e, f) and fishing rods are the most common gears. Inside the estuarine system the most important fishing techniques are fyke nets (Fig. 20.19g, h), traps, gillnets, bottom longlines, trammel nets, fishing rods and manual trawl nets (called “gerival”). In the south region (approaching the estuarine mouth), recreational fisheries with rods and reels and artificial or natural lures are also usual along the year.

Notably, crustaceans and finfishes are the two major groups caught by the artisanal fishing (Fig. 20.9a–d and Table 20.1). However, fishing gears normally catch not only a specific target species, but also a large range of species. Crabs are also a significant bycatch. Due to their commercial importance, there are also specific methods for their capture. The two most important are longlines and traps (Medeiros 2004). In coastal marine fisheries, most of the crabs captured as bycatch are discarded at sea and a smaller portion brought to the family diet. On the other hand, for fishers working inside the lagoons, crabs are one of the main sources of financial income.

The main fishing technologies adopted in by this group are fyke nets designed for shrimp catches and known in the region as “aviãozinho” (i.e. *F. paulensis*, *F. brasiliensis*, *L. schmitti*, *X. kroyeri* and *Artemesia longinaris*) (Medeiros 2004) (Fig. 20.19g, h). “Aviãozinho” is a set of funnel-like nets (five to seven in star-like arrangement – 25 mm mesh size), forming an “Y” with three rings to maintain the net open (Fig. 20.19g–h). Poles fix the net to the bottom at shallow areas. A luminous attractor is located at the centre of the net. Fishing then occurs at night for ~12 h (Vieira et al. 1996). The artisanal fishery of the pink shrimp (Fig. 20.9a) is the most intense activity in the Laguna Estuarine Complex, representing 99% of the captures of shrimps, always using "aviãozinho" (Sunye et al. 2014). This is a non-selective fishing technique that produces a wide diversity of bycatch, including fishes in early life stages and swimming crabs (*Callinectes sapidus*, *C. danae*, *C. ornatus*) (Sunye et al. 2014) (Fig. 20.8d and Table 20.1). In July 2011 the shrimp production by this method was ~85 tons in Santa Catarina. In July 2012, the production increased to 210 tons. However, the importance of Laguna for pink shrimp production is not fully understood, because historical data on this fishery resource are not available. The sole information is that before the creation of a closed period in 2005, only 55 tons of pink shrimps were harvested between 2004 and 2005, indicating a reduction of 90% in the production (Sunye et al. 2014).

Gill nets are mainly used for finfish captures (Medeiros 2004). Nearly 35 species are commercially important for the artisanal fishing at Laguna. The main teleost species caught by the artisanal fishing are lebranche mullet (*M. liza*) (Fig. 20.7a), bluefish (Pomatomus saltatrix) (Fig. 20.15d), and withmouth croaker (M. furnieri) (Fig. 20.7e). It is possible to observe three important fisheries seasons in the region. The first season starts in late autumn with a focus on the lebranche mullet; during the winter, the main fishery resource is the bluefish, and finally during the end of spring and summer, the catches are focused on the withmouth croaker (Bannwart 2014).

At the sea-lagoon interface, the cooperative fishery between the bottlenose dolphin *Tursiops truncatus* (known as "boto-da-tainha" in the region) and traditional fishers for the capture of mullets (*Mugil* spp.) is a very interesting heritage that lasts for decades (Daura-Jorge 2011) (Fig. 20.27a). Additionally, local fishers use canoes and cast nets for the catch of lebranche mullet (Fig. 20.27b, c). The main fishing activity targeting mullets occurs annually during the autumn reproductive migration (April to July) taking place along the southeast and south coasts of Brazil (OCEANA 2015). It aims at aggregations, which attracts simultaneous efforts of artisanal (coastal areas) and industrial fisheries (internal platform) stakeholders (Lemos et al. 2014).

The harvest of mullets in 2015 resulted in a total catch of about 4000 tons. Artisanal fisheries accounted for 46.6% (~1832 tons), while industrial fisheries produced over 2100 tons of mullet this season (53,4%) (OCEANA 2015). Artisanal capture is associated to an effort of 12,976,322 fishers per day, through 29 different fishing methods. Industrial fishing included 49 seine vessels responsible for mullets landed between June and July 2015 in the State of Santa Catarina (OCEANA 2015) (Fig. 20.20f). Artisanal fishery produced 3578 tons in the current harvest (2016) of mullets in the Santa Catarina state, exceeding the 1832 tons produced in 2015. The State Federation of Fishermen of Santa Catarina acknowledged this as the best harvest in the last 15 years. Only in May 2016, about 40 tons of mullets were captured on a Saturday morning, at the south region of the Laguna Estuarine Complex alone. However, data on totals or fine details (biological data as size, weight, maturation state etc.) of the production are not available for the region.

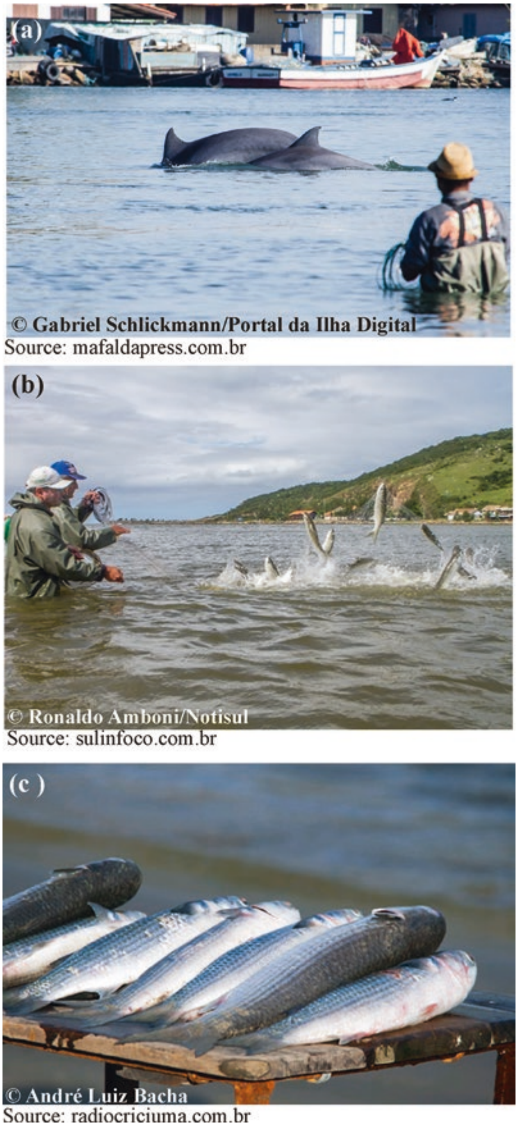

Fig. 20.27 Cooperative fishery between the bottlenose dolphin (Tursiops truncates) and traditional fisher for the capture of mullets (Mugil spp.) in the sea-lagoon interface (Laguna Estuarine System). (**a**) interaction between the dolphin and fishermen; (**b**) cast nets used for the captures; (**c**) vendor display. In this fishery, the fishers throw the cast net next to the dolphins, and when mullets try to escape, they are raped by the dolphins and are forced to go back onto the net

20.5 Discussion

20.5.1 The Importance of Accurate Landing Stats

As common feature, most estuarine complexes in the Western Atlantic coast have a simplistic record of the production in total biomass for a given location as fishery data, without descriptions of capture per unit effort, lengths of capture, bycatches or habitats of harvest. The FAO Technical Guidelines on Fisheries Management series defines fisheries management as the "integrated process of information gathering, analysis, planning, consultation, decision-making, allocation of resources and formulation and implementation, with enforcement as necessary, of regulations or rules which govern fisheries activities in order to ensure the continued productivity of the resources and accomplishment of other fisheries objectives" (FAO 1997). Therefore, accurate and detailed landing stats of artisanal fisheries need to inform which species are being fished, what is captured and discarded, and in which season the species are being captured (King 2007; Isaac et al. 2010). Where, how and what quantities? Lengths and weights of different developmental stages? Production is increasing or dropping? There is no way to build estuarine fishery management without good quality and representative information (King 2007; Longhurst 2010).

The problem is even more complex when trying to access fisheries stats on a national scale. Until 2006 these were designed and carried out by IBAMA, with a record for each municipality, federation units and geographical regions of Brazil. Since then it became a responsibility of the Fishery and Agriculture Ministry (IBAMA 2007). Currently, this ministry is extinct and comprehensive annual fisheries statistical bulletins have not been issued since 2011 (MPA 2011). Agencies fail in providing consistent data, and due to this limitation, it is no longer possible to estimate production by the artisanal and/or industrial fleets per location and time in this country (MPA 2011). In addition, the production regarding estuarine fishery and their yields cannot be assessed because there is no discriminant information relative to the habitat where harvests actually occur (MPA 2011). The example of Brazil in terms of fragility of institutions responsible for fisheries management is no exception. Unfortunately, it also extends to environmental agencies, and the integration of the actions becomes an even more difficult task if coastal and marine fisheries is to be sustainable or resilient.

Although estuaries along the Western Atlantic are acknowledged to be responsible for the current fishing status of Brazilian coasts (Blaber and Barletta 2016), until now it is not possible to predict what is really going on regarding estuarine fisheries. To guarantee the continuation of fisheries production of estuarine resources, Western Atlantic coastal wetlands requires accurate, complete and detailed landing stats. Monitoring of fishing activities by proper sampling of data and specimens must provide better description and assessment of exploited resources.

As a first step, it is necessary to introduce a rights and duties-based fishery management model in each estuarine complex in order to guarantee the declaration of every fisherman, their activities, fishing gear used and boat types to agencies of the different federal entities (Berkes et al. 2001).

The second step would be to empower traditional communities through the introduction of the idea of ecological sustainability and economic efficiency of fisheries resources, especially attempting to reduce extreme poverty (de Mitcheson 2009). This was demonstrated to be an efficient way to enable fishers to organize themselves, establishing a bottom-up movement of co-management and informed choices.

The last step of rapid and efficiently acquiring information of fishery production using the cheapest option in fisheries management, such as the emergence of the stakeholder approach and co-management (Hall and Close 2007; Schafer and Reis 2008) would then be more easily climbed.

Any citizen, or group, affected by or having an interest in programs, actions and decisions concerning a given territory is a stakeholder (Lackey 2005; Hall and Close 2007; Schafer and Reis 2008). Stakeholders are included in the process of defining goals and management measures for the territory they inhabit/use, its resources and services. The involvement and articulation among stakeholders is primordial for a higher chance of guaranteeing the widespread support of society since most traditional and non-traditional populations does not dominate science-based techniques of fisheries management (Lackey 2005; Schafer and Reis 2008). In this sense, co-management represents a further development in relation to conventional fisheries management, in which some government authority manages fishing simply through the issuing of regulations and local institutions (fishers associations, NGOs) (Lackey 2005; Schafer and Reis 2008).

Close and detailed environmental and biological monitoring, scientific support through consistently designed experiments to detect population changes, and co-management observing community and resources needs are receiving more attention in the last decades since there are many fish resources that recovered their stocks based on the employment of one, two or all of these concepts (González et al. 2006; Frangoudes et al. 2008). In Galicia (Spain), co-management, investments in training, improvement of local organizations, use of seeding to regenerate degraded areas, understanding and respecting the social dimension of shellfish gathering were decisive strategies to a more successful exploitation of the grooved carpet shell (*Venerupis decussate*), carpet shell (*Venerupis pullastra*), and cockles (*Cardium edule*) (Frangoudes et al. 2008). Fishers organizations, with the support of the government, have combated and reversed overexploitation and much better regulated the shellfish market through women empowerment (Frangoudes et al. 2008). The activity, traditionally developed by women, in an open access regime (commons), is now actively co-managed through a license system (Frangoudes et al. 2008).

Another case of improvement in the relationship fishers-resources related to mollusks is observed in Chile. There, coastal benthic fisheries of loco shellfish (*Concholepas concholepas*) was the most significant local resource (González et al. 2006). The closure of the loco fishery between 1988 and 1992 was the result of an overfishing crisis. It lead to the implementation of the Territorial Use Rights model (TURFs) into fisheries legislation. Currently, the abundance of shellfish within the TURFs increased and legal fishing is stabilized (González et al. 2006). The loco fishery was ordered, and fishers perception about the TURFs is mostly positive. However, outside the TURFs at least half of the captures remain illegal (González et al. 2006). The challenge now lies at expanding the experimental gains to wider scales to diminish illegal catches, increase fishers participation at other localities and provide feedback to decision makers for the completion of the adaptive management through accurate biological, landing and market data.

Wherever monitoring, control and surveillance of fishery is absent, incentives such as high price, low cost and easy access will disturb fish stocks. This is the case of the illegal fishery of the endemic South African mollusk abalone (*Haliotis midae*) in the Eastern Cape of South Africa (Raemaekers and Britz 2009). Since 1997, the illicit abalone trade and uncontrolled fishing effort has decreased the average size and densities. By 2005, a fleet of 30 boats was responsible for the existed, harvesting of 1000–2000 tons of abalone per year, with export value of 35–70 million US$ (Raemaekers and Britz 2009). A 'Management Plan for the Eastern Cape Abalone Resource' was proposed in 2002, but never implemented (Raemaekers and Britz 2009). The absence of fishing rights created the conditions for the illegal operations, a serious deficiencies in abalone fishery management.

The co-management appears to be able to promote fisheries conservation in Brazil also, by integrating the human element into fisheries management (Berkes et al. 2001). Participatory management experiments are taking place across the country, however with little representatives in estuarine regions. In the flooded forests of the continental Amazon, for example, the participation of fishers in the evaluation of arapaima (*Arapaima gigas*) stocks, the determination of fishing quotas, and the surveillance of management rules helped to reverse the status of over-exploitation of the species (Viana et al. 2007). With the implementation of co-management systems in the Sustainable Development Reserve of Mammirauá, fishing of arapaima increased from 5.5 to 132.9 tons between 2002 and 2010, even with a significant increase from 50 to 551 fishers (Amaral et al. 2011).

Co-management is, so far, the best option available for living resources conservation and sustainability in Western Atlantic wetlands (Lackey 2005; González et al. 2006; Frangoudes et al. 2008; Schafer and Reis 2008). Together with capacity building within and beyond traditional communities, good quality and long-term data are the basis for this model. To properly build accurate landing stats, it is necessary to invest time and resources in training fishers and their families in describing their catches and environmental settings where/when capture took place. Also, perseverance in producing daily reports is vital. These reports need including correct identification of taxa, total weight of captures, lengths and weights for individual specimens in a representative sample, including bycatches; location of habitat where fishing took place (mangrove creeks, main channel, mangrove forest, estuarine beaches, coasts); effort in terms of time and number of fishers and; details of gears and boats used. However, artisanal fishery is far more difficult to predict when compared to industrial fishery. In the last case, the fishery company needs licensing and are obligated to provide data to manage the landed product (King 2007). This is the reason why control and surveillance in the artisanal fishery are of extreme importance. Legal terms and conditions for the exploitation of resources in traditional communities is an urgent approach to avoid illegal, unreported and unregulated fisheries (Pitcher et al. 2002) that impair conservation and compromises food safety.

The gathering of landing data generates information on capture rates (fishing mortality) and changes in the biomass of a stock and bycatch over time (King 2007). To understand patterns and propose feedback actions it is necessary the compliance of long-term data. However, past information is scarce and consistency is a hesitation, especially for changeable and heterogeneous environments as estuaries. Reports of estuarine landings is a difficult task at short-term. However, it must be initiated as soon as possible, since estuaries are acknowledged as one of the most productive marine environments and, at the same time probably the most degraded coastal settings (Beck et al. 2001; Blaber 2002; Blaber and Barletta 2016; Costa and Barletta 2016). If proper use should guarantee their sustainability, working with fisheries ecology and management, a type of use that will demand better levels of water and surrounding land quality, could be an interesting strategy towards quickly achieving larger nature conservation goals.

Long-term data on daily production of estuarine resources by habitat would be important to accurately design analytical assessment models. These models are necessary to understand variations in fish production using coefficients of fishing mortality rates and initial age/length of captures in order to indicate the moment a resource is available for capture, or when it should be spared (King 2007; Castello et al. 2009). Precocious captures (juvenile stages) and intense exploitation difficult a sufficient number of individual to reach an ideal weight to be recruited into adult stocks and reproduce (King 2007). The theoretical maximum sustainable production is the intersection between optimal values of fishing captures and initial age/length of capture, and help determining the legal possible length of capture close to the average length of first maturity (L_{50}). This could be sufficient to consider a fish stock legally protected by regulatory measures (King 2007; Longhurst 2010). The main idea is to guarantee that each individual can have the chance to reproduce at least once, hopefully generating new recruits to adult stocks.

Most studies are focused on a total number of fishes without concerning to the importance of estuarine habitats for their different developmental stages (larvae, juvenile, sub-adults and adults), and this is also true for fisheries stats (Dantas et al. 2010; Lima et al. 2015, 2016; MPA 2011). Ontogenetic approaches are important to understanding the nursery function of estuaries and their habitats for the early stages of fishing resources that depend on estuaries, partial or totally, to complete their life cycle (Beck et al. 2001; Able 2005; Ferreira et al. 2016). The natural and expected environmental variability of estuaries provides specific seasonal abiotic settings at each habitat, guaranteeing that specific water physico-chemical conditions available for different life stages at the right time and space. Failure to determine patterns of habitat uses by different resources makes it challenging to assess the likely consequences of fisheries management actions, such as adjustments on fishing harvest, changes in gear uses and regulations and the establishment of moratoria on fishing (Lackey 2005).

Studies about commercial and subsistence fish species which use estuaries and adjacent coastal waters of Brazilian coasts have provided information on the L_{50} for common snook (Dantas and Barletta 2016), acoupa weakfish (Ferreira et al. 2016), mojarras (Ramos et al. 2016), drums (*Stellifer brasiliensis* and *S. stellifer*) (Dantas et al. 2015), and catfishes (Dantas et al. 2010). However, selectivity of fishing gears, if any, varies widely among estuaries. Therefore, fishes that have not reached legal length (or biological maturity and reproduction stage) are frequently captured as bycatch, affecting recruitment and resulting in a reduction of stocks (Rueda and Defeo 2003).

20.5.2 Fishery technology

Actually, nearly 7 million tons of incidental catches are annually produced around the world, which corresponds to approximately 8% of total marine catches (Eayrs 2007; FAO 2008). In this context, usually, the incidental catches are recorded as bycatch. These species are not the target of the fishery activity but are opportunely caught, and can be used to generate extra income for fishers or simply are discarded back into the environment (Alverson et al. 1994; Graça-Lopes et al. 2002; Branco and Verani 2006). In most commercial fisheries, bycatch generation has emerged as a serious concern. Most non-target species caught returned to the sea already dead (Gillett 2008), and therefore incapable of continuing replenishing their particular stock. This scenario can be worst if we consider that most of the bycatches consist of juvenile fishes (Haimovici and Mendonça 1996; Kelleher 2005; Branco and Verani 2006), or fish that are important food resources to larger and commercially important species.

Faced with this problem, many countries have been adopted several environmental regulation to minimize bycatch. For that reason, in the last two decades, technological modifications in fishing gears have emerged as a complementary fisheries management strategy to help overcome this problems in the continuation of this activity. These modifications are known worldwide as bycatch reduction devices (BRDs) (Broadhurst et al. 2006, 2012). The BRDs are technological modifications designed primarily to the bycatch exclusion. In this approach, the researchers, based on local knowledge of the fishery, fishing method and behavior of local catches, propose some technological alterations to increase the selectivity of the fishing gear. BRDs have been shown to be an important management strategy in several parts of the world, due to its lower cost compared to traditional, coercive, fishing rights monitoring methods adopted by the governmental and non-governmental organizations (Campbell and Cornwell 2008). Such options are also common in attempts to guarantee reduction of bycatch of species of conservation concern, as marine turtles [exists at the cod end of trawl nets – TED (turtle excluder device)] and marine birds (circular hooks) (Werner et al. 2006).

In South America, experiments with technological changes in fishing gear are still incipient. In the southern Brazilian region, the Fisheries Science and Technology Group (TECPESCA) at the Santa Catarina State University (Brazil) have evaluated some alternative solutions for shrimp trawls. In this region, the bottom trawls are economically relevant although they can contribute to environmental impacts in coastal regions. According to Branco and Verani (2006), for each kg of shrimp, this fishing method caught 19 kg of bycatch. To minimize bycatch generation, one of the technological approaches proposed by the group was the *fisheye* at the cod-end of the trawl. The *fisheye* is a device that provides the flatfish escape through an elliptical opening. Additionally, this BRD aims at the escape of pelagic juveniles through the square meshes. Preliminary results show that this BRD reduces the 1:19 kg proportion to 1:9 kg, with an implementation cost of just 22 US$ per unit.

When fishery is regulated, fish populations, generally, respond with increased production (King 2007). While fishing increases, mortality also increases, biomass decreases and the low competition for resources among survivals prompts the population to respond with high population growth rates. If capture rate increases, the biomass decreases until the fishing mortality stipulates the maximum sustainable yield of a fish stock. If captures of smaller individuals continue, the trend is the extinction of that stock. Estuarine landing data are thus necessary to identify resources with greater vulnerability, higher resilience and more affordable sustainable yields.

20.5.3 Closure of Captures of Specific Stocks

By using scientific literature regarding the life cycle of fisheries resources, periods of migrations and seasonality of abundance, government agencies (e.g. IBAMA, ICMBio) is rich in rules for the sustainable exploitation of fish stocks in estuaries and coastal adjacent water by the use closed periods (Castro et al. 2004; Barletta et al. 2005, 2008; Silvano et al. 2006). This fisheries management strategy restricts access of fishers to a specific location or times of the year. Closed periods are receiving attention and strength as an important short-term, emergency, management option that help the recovery of fishery resources in Western Atlantic estuaries. However, these will always remain palliatives unless more consistent actions don't follow and fail to restore environmental conditions (both biological and abiotic) to guarantee support to stocks resilience. In addition, some other important measures to guarantee the sustainable use of estuarine resources include no-take zones (associated to the spill-over principle), family or individual quotas for the capture of common resources and efforts to reduce/eliminate untreated wastewater disposal and to improve river flow quantity and quality. Closure is seldom applied to all stocks at a time (complete exclusion zone), and difficult to inforce.

Considering the most captured fishery resource, the uça-crab, a study in the Parnaíba Delta recorded that the capture per unit effort (CPUE) varied between 14.6 and 22.6 crab per picker per day from 1999 to 2002 (Legat et al. 2005). The total monthly capture can exceed 2 million crabs when the CPUE is at its maximum, producing up to 21 million crabs during the period of one year (Legat et al. 2005) (Fig. 20.18). This is a strong evidence of over-exploitation and possible near collapse of uça-crab fishing if management options are not available quickly (Fig. 20.18). Only recently, in 2014, it was decreed a closed period for the uçá-crab in cycles of five days of interdiction and ten days of free catch from January to March (BIOMADE 2015). Two federal environmental agencies are responsible for the monitoring and surveillance, which occurs at Tatus port, the main access to the

Parnaíba Delta and the surrounding mangrove forest (BIOMADE 2015). Pickers that have captured the crabs outside the closed period are obligated to report their yields before commercialization (BIOMADE 2015). However, current and descriptive landing stats are not available for comparison because management plans for the "MPA-Delta do Parnaíba" have never been drawn and issued, and therefore there was no concern in producing accurate data for fishery management until collapse was too close to be avoided. In addition, whereas the Parnaíba Delta consist of five estuaries, it is not possible to know from which system a fishery resource is coming without previous genetic research establishes traits of sub-populations.

For the São Francisco Estuary the problem is more difficult to equate and solve. Whereas the estuarine portion of river represents the smallest portion of a long river basin, most ecological studies and lists of species are focused on freshwater fishes and fisheries. Since 2007, efforts aiming to renew fish stocks in the river basin includes a closed period from November to March, but only for riverine fishes. This measure, known as "Piracema", is a period of summer rains in which fishes move downstream and upstream to relocate and reproduce (IBAMA 2007). Therefore, the importance of the estuarine system for fishes that depends on different habitats to complete their life cycles has been neglected so far. Less importance was given to the estuarine fishery resources, and previous data on fishery productivity for the estuarine and coastal regions of the São Francisco River are rare. Only recently, in 2009, monitoring and surveillance was intensified in the Lower São Francisco River. The operation aims at guaranteeing the enforcement of closed periods and prohibit the use of illegal fishing gears. For some fisheries resources, such as sharks, rays, jacks, anchovies and sand crab, there is no closed period. However, captures of mullets, snooks, mojarras and the freshwater shrimp are prohibited from November to February, while for the white and the Atlantic sea bob shrimps the closed periods are from April and December.

Whereas the Santa Catarina state is the most important industrial marine-extractive fishery producer in the country, representing more than 22% of the total national production, data on artisanal fisheries in estuaries has also been neglected (Andrade 1998; MPA 2011). There is no ecological study on estuarine fish assemblages in the Laguna Estuarine Complex, and little concern developed over invertebrate resources (Sunye et al. 2014). Although the fishing activity have prominence, there is no information about the volume and diversity of caught species once there is no fisheries monitoring plan. The latest information on industrial fishing date back to 2012 (UNIVALI/CTTMar 2013), and brings very general data, with no information on the artisanal fisheries landings. This is a tragic scenario due to the observed exponential increase in the fishing effort and the possibility of collapse for some commercial species.

Fixed fyke nets ("aviãozinho") were introduced to Laguna in 1970, and the exploitation of pink shrimps increased, reaching 5000 active fishers in the early 1980s (Fig. 20.19g, h). The over-exploitation reduced the stocks and the number of fishers decreased to 2500 at the end of the 1990s. The low selectivity of "aviãozinho" is responsible for high levels of bycatch in the region. This gear captures almost the same biomass of pink shrimp and crabs, and ~700 g of commercially important juvenile fishes per each kg of shrimp (Sunye et al. 2014). Since 2005, the federal government proposed a closed season for pink shrimp during four months (15th July to 15th November) (MMA 2004b). This management action is still in place. Before the closed period (2004/2005), only 55 tons of pink shrimps were being harvested, indicating a reduction of 90% in the previously declared production. It means that the recovery capacity of pink shrimp stocks might be heavily compromised.

At the same estuary, mullet's fishery is traditionally acknowledged as one of the most important in the region. However, the management of mullet's fishery has historically been driven by ad hoc measures and limited by poor knowledge of the exploitation potential of the stocks. The migratory character and seasonal regime of exploitation, established over a large geographical area and for different fisheries are the main issues for the serious limitations of any monitoring processes. Surveillance must be intense and focused enough to detect the variability of biomass during the short fishing season. Only recently in 2008, the closed period for mullets (*M. platanus* and *M. liza*) in estuarine-lagoon mouths along the Brazilian coast at the southeast and south regions (including the Laguna) was issued, and extends from 15th March to 15th August, except for casting nets (IBAMA 2008b). In this document, estuarine mouths are defined as 1000 m from the estuary or lagoon seawards; 200 m upstream from the mouth into the river; and 1000 m long on the coastal banks adjacent to the mouths of the estuaries/lagoons. Fishing on the continental platform remains permitted.

20.5.4 The Goiana River Estuary: Case Study

The more encouraging scenario in terms of fisheries management and data generation, among the studied estuaries in this chapter, is the Goiana Estuary. For this system, the fish fauna, abiotic variables and anthropogenic contamination (plastic and mercury) were well characterized along the ecocline (environmental gradients) and during year-round cycles (Barletta and Costa 2009; Costa et al. 2009; Dantas et al. 2010; Lima et al. 2014, 2015; Ferreira et al. 2016; Silva-Cavalcanti et al. 2016). Thus, science-based information to support managerial plans is already available to be communicated and used. The problem is that the conventional mechanism through which scientists generate and provide consistent data on the biology and population dynamics of fishery resources; pass information to the management agencies; and agencies design management rules and actions together with the community is still not in place (Castello et al. 2009). It follows the assumptions of Europe and North America, where there are human and financial resources to make it work the way it was designed (Castello et al. 2009), and regardless local limitations. In Brazil, as in other countries of South America that share institutional and environmental profiles, financial resources for fisheries studies and management are scarce, and the conventional model can no work with a minimum of effectiveness. The situation practically excludes any hope of sustainability for estuarine systems across the whole continent. More than half of the most productive fish stocks in Brazil, which are managed by conventional approaches, are actually overexploited (Paiva 1997).

The use of effective marine and coastal protected areas mechanisms to reduce fishing effort and trophic chain models to improve the knowledge of aquatic systems are among the most sought advances in fishery science (Pauly et al. 2002). However, these models have limited utility without good quality and long-term data to feed them. Management agencies with more resources in the world were unable to ensure compliance with simple management rules such as quotas (Pauly and Maclean 2003). In Brazil, many management rules exist that need to be enforced, even in the case of important fisheries such as the lobster fleet in the northeast or the sardine in the southeast regions (Paiva 1997; Isaac et al. 2006). A simple, but common, example is the Marine Extractive Reserve Acaú-Goiana (6700 ha), created by federal decree in 2007, after a legitimate community demand, to guarantee the protection of livelihoods and resources sustainability as *A. brasiliana*, in the Goiana Estuary. It still does not have a management plan issued in 2016, stakeholder decisions made at its deliberative council are ineffective, and overexploitation of fish stocks and other environmental concerns are real problems (Barletta and Costa 2009; Guebert-Bartholo et al. 2011; Guebert et al. 2013). Although a closed period for uçá-crab captures exists since 2003, traditional communities take advantages of the lack of surveillance and crab picking is uncontrolled (Silveira et al. 2013).

According to the official records, in 2005 ~380 tons of commercial fishes were landed at the three municipalities surrounding the Goiana Estuary, including five estuarine-dependant species (*Centropomus* sp., *Cynoscion acoupa*, *Lutjanus* sp., *Eugerres brasilianus* and *Hiporhamphus* sp.). Current data generated by reaserch projects on fishery landings at the Goiana Estuary during 36 months (2013–2015), recorded ~50 tons of fish, which includes 58 target species of the local fleets (Table 20.2). These two different sources have a variation of 330 tons along ten years of non-reported landings. However, it is not possible to safely conclude that fish stocks have collapsed due to limitation and inconsistency of past data.

Such considerable discrepancy between the official data and those obtained from scientific surveys was also observed in the Bragantine peninsula, at the State of Pará, North Brazil (Isaac et al. 2010). The official statistics reported a total fishery production ranging from 10,000 to 20,000 tons per year between 1995 and 2004. However, the total production at the Bragança port for all types of catch was ~6000 tons, taking into consideration all boats arriving in a 13 months period from 2000 to 2001 (Isaac et al. 2010). The authors also attributed this differences to insufficient efforts from government agents to collect data to feed models and correctly fulfil the needs of the statistical methods chosen (sampling system with strict monitoring), which apparently use extrapolations for calculating the total production (Isaac et al. 2010). However, the large diversity and considerable dispersion of the fishery modalities in the region do not allow extrapolations, which makes the official statistics inaccurate (Isaac et al. 2010), posing a serious risk of poor decision being made for lack of data, despite the will to protect both the activity/fishers and the environment.

20.5.5 Human impacts

Another problem for fishery science is that human impacts that lead to environmental degradation were, and still are, neglected as possible causes of fish stocks collapse. Human impacts derived from unplanned urban settlements, unsustainable land use and industries/domestic effluents are the main causes of habitat loss and/or degradation (Figs. 20.6, 20.11, 20.12, 20.13, 20.14, 20.16, and Table 20.1). This is especially pronounced in areas around estuaries, where shipping ports, cities, agriculture and aquaculture cause damages to fisheries production and therefore the exportation of biomass from regional seas (Lotze et al. 2006; Dallas and Barnard 2009; Barletta et al. 2016).

Mangrove deforestation for whatever reasons has severely reduced habitat for the development of estuarine resources, what reduces fish stocks for local communities and beyond, since estuarine fish are sold elsewhere to generate income and supply distant markets. In the Cayapas-Mataje delta in Esmeraldas Province (Ecuador/Colombia border), at least 6000 inhabitants directly rely on mangrove resources for living (Ocampo-Thomason 2006). Artisanal fishing and arc-shells picking (*Anadara tuberculosa, A. similis* and *A. grandis*) are the main economic activities (Ocampo-Thomason 2006). However, changes such as African palm culture and commercial shrimp farming are having an impact on the mangrove ecosystem. Construction of shrimp farms has led to the destruction of arc-shells recruiting and living grounds and damage to agricultural land (Ocampo-Thomason 2006).

Traditional communities responded to these changes by creating new management strategies agreed among political, ecological and social organizations. Two movements came together in the north of Esmeraldas to defend the mangrove areas, the Fundación para la Defensa Ecológica in 1989 and the Black Communities Process. After 5 years of mobilizations and campaigns, the Ecuadorian government granted the area status of ecological reserve, the Ecological Mangrove Reserve Cayapas-Mataje (Ocampo-Thomason 2006). It led to the implementation of a novel practice called 'custodias', a piece of land protected by recognized community leader who retains traditional knowledge in resource management. It has been calculated that 98% of illegal mangrove removal has stopped, confirming the potential success of this aproach (Ocampo-Thomason 2006).

Another issue for estuarine ecological processes is the lack of basic sanitation (sewage treatment), often the worst concern in estuarine pollution and degradation (Costa and Barletta 2016). High loads of solid wastes are also contaminating estuaries increasing the risk of higher plastic debris ingestion (macro or micro) by estuarine fauna (Lima et al. 2014; Ferreira et al. 2016) (Table 20.1). Wastewater from household, industries and farms generate high amounts of pesticides (POPs) and heavy metal that directly affect water and aquatic biomass quality (Costa et al. 2009; Barletta et al. 2012; Silva-Cavalcanti et al. 2016). Shellfish species, for example, depend on good water and suspended particulate organic matter quality to generate meat safe for consumption. Efforts to better use of surrounding land and treatment of effluents are important in determining shellfish survival, meat quality and livelihoods (Barletta and Costa 2009).

20.5.6 In Search of Best Options for Fishery Management of Wester Atlantic Estuaries

The ideal approach for fisheries management in Western Atlantic estuaries would be to compile data originated from ecological and biological research, robust data for landing stats and the social profile of the fishery community for model estuarine regions in order to build proper rules of co-management (Castello et al. 2009; Schafer and Reis 2008; Gasalla et al. 2010). Ecological and biological research are necessary as references for the functioning of estuarine habitats for biological species and communities; how they are distributed according to seasonal patterns on the estuarine ecocline (heterogeneous gradients) and which impacts are causing changes to the environment (Barletta et al. 2016; Ferreira et al. 2016) (for more information see Barletta et al. 2017 in this book).

The scientific literature on estuarine ecology is gaining more quality in the last decades, but it still needs to be based on better designed sampling and statistical treatments. Many studies assess annual or interannual patterns in biological communities based on expensive and detailed scientific surveys that cannot extend for many years (Blaber 2000; Lotze et al. 2006; Elliott and Quintino 2007; Costa and Barletta 2016; Blaber and Barletta 2016; Reis et al. 2016). Despite financial concerns, these studies are indeed able to detect that biological species, including fishing resources and their prey, may either naturally or driven by human intervention vary in abundance, size and biomass among years (Berasategui et al. 2004; Lorenzo et al. 2011). In some periods, there may be wondrous spawning success, as well as little or no reproduction/recruitment. Long-term climatic changes, as well as episodic events and catastrophes, also alter the productive capacity of estuaries. Therefore, changes in fishery resources abundance over decades and centuries may not be apparent without long-term data sets (Garcia et al. 2003; Ríos-Pulgarín et al. 2016). Droughts, usually much less visible at estuaries than at most of the river basins, are also an important issue, since the period of no rainfall implies that estuarine biota, nutrients and pollutants loads will not be flushed out of estuaries to coastal marine ecosystems and beyond (Lima et al. 2014; Costa and Barletta 2016).

The Scientific literature on ecology and uses of estuaries is increasing in quality, including regions of the Globe once deprived of such privilege as South America and Africa (Blaber 2000; Lotze et al. 2006; Elliott and Quintino 2007; Costa and Barletta 2016; Blaber and Barletta 2016; Reis et al. 2016). But, to compile relevant information relating ecological issues and fishery landings remains a challenge for Western Atlantic fishery management because the literature still needs to make this final leap of closing the links between subjects; be communicated to management technicians and the public directly interested in the data. There is a long gap between the references generated by the scientific community and the implementation of fishery management plans.

The task is further impaired when the lack of historical fish landings data is taken into consideration. At this time, it is not possible to draw a clear picture of fisheries interference with estuarine functioning based on the existing information regarding fish stocks in Brazil. There is an urgent need to gather accurate landing information in to be crossed with biological and ecological knowledge in order to choose consistent fishery management priorities for each estuarine ecosystem. First, the approach needs to be used in estuaries as singular systems, but with time, there will be the need of electing model systems to be fully monitored and from which decisions can be extrapolated to other similar settings. Fish stocks are exploited at different levels and it depends mainly/roughly on the latitude of the estuarine complex. Characteristics of the river basin, fauna, social uses and other descriptors are common to relatively wide bands along the Western Atlantic coast. Therefore, applicable measures should be possible to be reproduced from one estuary to the other with minimum data sampling and re-evaluation of the fisheries management model. Co-management and daily reports on production can help to design stock assessment models, understand variations in biomass over time, detect problems of uncontrolled fishing effort, point periods of seasonal habits for each fishery resource, and, most importantly, guarantee that enough juveniles of each living resource can be recruited to adult stocks accordingly.

The first Marine Protected Area (MPA) in a Brazilian estuary was created only recently, in the early 1980s (Diegues 2008). Despite the number and variety of coastal ecosystems, including estuaries, the number of officially and effectively protected areas is still insufficient, and should not comply with the assumed commitment of protecting at least 10% of coastal and marine areas by 2020. Ideally, conservation measures should also guarantee the conservation of much wider coastal areas than MPAs. Recently, federal agencies are working on fishing restrictions and monitoring strategies, closed periods and areas, and quota for the catches aiming at implementation at marine protected areas only. If it works, more consistent management practices will be devised in order to protect local and regional resources from all human and particularly fishery impacts, by improving the performance of estuarine ecological services.

Whereas estuaries have highly dynamic and heterogeneous habitats, working in favor of the estuarine ecocline (environmental gradients) is also important to the continued production of socio-economically relevant estuarine-dependent fish resources (Barletta et al. 2005, 2008; Barletta and Costa 2009; Ferreira et al. 2016). For each fishery resource, ontogenetic stages use different strategies and habitats to complete their life cycles according to the spatio-temporal variability of abiotic compartments across the estuarine gradient. Therefore, reliable ecological and fish landing data are fundamental in managerial actions because ecologically planned data acquisition can consistently explain how environmental gradients influence the distributional patterns of communities due to their biological demands on trophic webs, reproductive timing and use of temporary nursery habitats, including their seasonal peaks of abundance along the river-estuary-coastal area-sea continuum.

The implementation of measures aiming at mitigating impacts to fish stocks needs to be achieved in accordance with social subjects and organizations. In this sense, it is extremely important, continuity and expansion of the monitoring network of fisheries production, with coverage of relevant landings of estuarine, coastal and marine productions by training and using the valuable human resources within each community. Adequately trained local human resources should benefit accuracy based on will of subjects to inform their catch and comprehension of the working dynamics, also it should facilitate the disclosure of new insights into the systems functioning. If the lack of precise information about the artisanal fishery landings keeps its present course, the incompatibility with the sustainability paradigm will continuous to compromise the myriad of ecological services and living resources in the Western Atlantic estuarine complexes.

The historical negligence of authorities in building reliable datasets for the fishery statistics in Brazil is denounced in studies regarding global (Worm et al. 2009) and South American fisheries (Barletta et al. 2010; Reis et al. 2016). In half of all studied ecosystems, the average exploitation of marine resources has declined globally (Worm et al. 2009). Now, figures are near or below the rate of maximum sustainable yield (Worm et al. 2009). However, the global scenario lack the input of information on the Western South Atlantic fishery. The absence of data for Brazilian coastal ecosystems in global compilations and analysis points to two main concerns in national politics. The first is that, apparently, the fast growing stack of papers containing scientific information is only a short-term tactics to provide numbers in United Nation conferences and agreements. The second is that it reveals the inefficient communication between the academy and institutions responsible for structuring the fisheries and environmental conservation in Brazil, especially the development of strategies to better use of estuarine and coastal marine resources.

20.6 Conclusion

Estuaries and other coastal environment of the Western Atlantic are extremely important for providing living and non-living resources, as well as numerous ecological services, as livelihood for traditional communities. Although these services and resources are collapsing through the years, there is strong expectations aiming at the recovery of fish stocks and habitat restoration, especially by the scientific community and governmental sectors. However, the majority of fishery studies point the lack of communication between managers, governmental agencies and scientists for building models for the management of fisheries and enviornmental conservation as their main concerns. It is agreed that urgent attention needs to be payed to estuaries. They provide large amounts of fishery resources (e.g. dog snapper, acoupa weakfish, mullets) to adjacent coasts and high seas in addition to their own internal fisheries production. The semi closed characteristic of estuaries, guarantee access to high densities and biomass of important subsistence species (e.g. prawns) and migratory fishes of commercial importance (e.g. common snook), which are easily captured. Still, most tropical estuaries are surrounded by mangrove forests with a diversity of habitats, and less privileged local communities target the capture of the flooded forest fauna (e.g. uçá-crab).

Therefore, there are two main points for Western Atlantic coastal wetlands managements to take into consideration. The first is to guarantee the sustainable use of estuarine resources, and the second regards to the continuum of estuarine production for the recovery of fish stocks. These issues lead to urgent points, such as accurate records of landing data for fishery assessments in estuaries, rapid management measures for the collapsed resources, and habitat loss control. Co-management has been pointed as the best available option for guarantee information on landed living resources along estuaries and adjacent coasts. This is particularly important in systems, where the availability of fish resources changes dramatically according to the seasonal fluctuation of the estuarine ecocline or due to some large human intervention.

Short-term fishery management options need to be performed in every wetland. It includes increasing numbers of no-take zones, closed periods for the capture of fish resources, the consolidation of better fishery technologies and rules to avoid precocious capture of non-rentable bycatch and/or vulnerable species. It is also important to guarantee that systems maintain their environmental importance to species that depend upon estuarine habitats to complete their life cycles or for the living of their food resources. Organization of fish markets, empowerment of traditional activities and human resources, and better tourism practices are among the other options to earn money and diminish the direct exploitation of estuarine living resources. All these measures need to take into account the social and cultural characteristics of the traditional communities and their fisheries through the implementation of educational and socioenvironmental assessments.

Estuaries along the Brazilian coast are intensively suffering pressures besides overexploitation of fish resources. It includes accelerated changes in water flow and quality, and modifications for human purposes. The most pronounced impacts are expanded hectares of sugarcane plantation around estuaries (e.g. Goiana Estuary), uncontrolled agriculture and aquaculture practices, intensive capture of bycatch (e.g. Laguna Estuarine System), regulation of river flow by hydroelectric power plants and dredging leading to marine water intrusion and mischaracterization of the estuarine ecocline (e.g. São Francisco Estuary), and uncontrolled mass tourism (e.g. Parnaíba Delta). For this reason, coastal wetlands urges managerial options for extractive reserves (i.e. Acaú-Goiana) whose plans are not yet drawn; sustainable use practices within or outside formal MPAs for intense estuarine fisheries (i.e. Laguna); and monitoring, control and surveillance of estuarine regions where management plans are already available (i.e. MPAs Delta do Parnaíba, Piaçabuçú, Litoral Norte and Marituba do Peixe). All these steps must include the participation of stakeholders aiming at understanding local needs and generation of data for the formulation of consistent models.

Gathering data for estuarine fisheries management is not a new demand for neither public nor private sectors of the economy. However, building a relevant data collection for the coupling of economic and conservation purposes might be a new challenge to be faced by scientist, policy makers and the local communities together. The characteristics of the data collection and processing actions will need to be in agreement with the assets and limitations of each environment.

The choice of model estuaries to exemplify the type of data and information needed made in this chapter conveys two main messages: one that the estuaries on the Western Atlantic may be facing similar threats at environmental, social and political levels and; a second one that they are still in conditions of regain control of their resilience patterns and recover from decades of disregard. As common threats to be tackled would be basic sanitation, surrounding land use, flow regulation and empowerment of protected areas management. All these actions are somewhat displaced from the reach of fisheries management, but would greatly improve environmental quality and catch possibilities. The fact that the four systems appear to remain productive, despite the lack of reliable and recent fisheries data, suggests that conventional adjustments in fisheries practices, including full landing records might already cause some improvement in managerial options that favour local communities and might benefit commercial enterprise in the end.

Among the necessary actions that can be made at harbours of each estuarine system would be fleet registration (boats, gears, crews and docking patterns), daily record of total catch per day, details about most profitable catch (species, local of capture, individual size and weight, maturation) and market pricing.

The financial and human resources investments to proceed with these collections would be recompensed in fisheries productivity in a few years, if data could be treated and management suggestions followed. Although some restrictions would have to be implemented (no-take zones, gear regulation, size limits and seasonal closures) at different time scales, only data gathering and treatment would base decision on whether each environment is improving its fisheries production and related general environmental quality.

Finally, fisheries at the Western Atlantic coast will never be ideal without continuous investments in scientific studies to conduct long-term fish stock assessments and robust ecological information. The historical miscommunication between scientists and managers, and little efforts to onset the management of coastal wetlands is proving an inconceivable burden for these environments, and artisanal and industrial fisheries will fall become non-profitability due to the collapse of overexploited fish stocks. A large part of resources used worldwide is strictly from estuarine production. However, although environmental management can be consistently established, the current available information regarding estuarine fisheries do not allow building fishery managements models that work on any timescale. What will be the future impacts of fisheries and habitat loss on nursery services along the estuarine ecocline? How resilient estuaries can be withstanding human modifications and climate changes? Will Western Atlantic fish stocks recover and overcome these pressures, or will they collapse? Without landing data that reflects estuarine ecology and conditions, we simply don't know.

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Chapter 21
Returning the Tide to Dikelands in a Macrotidal and Ice-Influenced Environment: Challenges and Lessons Learned

Laura K. Boone, Jeff Ollerhead, Myriam A. Barbeau, Allen D. Beck, Brian G. Sanderson, and Nic R. McLellan

Abstract The objectives of this chapter are to (1) document lessons learned from the design, implementation and monitoring of a salt marsh restoration in the upper Bay of Fundy, Canada, and (2) consider how the lessons can be applied to future restoration projects. The Fort Beauséjour salt marsh restoration sites are exposed to very large tides (up to 14 m), waves, and snow and ice in winter. This project involved a managed re-alignment, with two restoration cells and two reference sites. Before breaching, design criteria were established (e.g., the restoration cells must fully flood at high tide and drain slowly) and a hydrodynamic model was used to test breaching options. Pre-restoration monitoring was completed in 2009–2010, the old dike was breached in October 2010, and post-breach monitoring commenced thereafter. Measurements of water level, velocities, and discharge at one breach, compared very well to model predictions. Likewise, patterns of sediment deposition were as predicted, and sedimentation rates were as expected based on empirical studies done in the area. The bioengineering species saltwater cordgrass (*Spartina alterniflora*) took 2 years to colonize the cells; it initially spread vegetatively and then by seeds. Plant cover became extensive in year 5 post-breach. Invertebrate and salt pool biological communities are lagging behind. Lessons learned include: (1) plan for future conditions and provide adequate accommodation space for development of a new marsh; (2) multi-level partnerships are critical to the success of such

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projects; (3) monitoring with a research focus ensures observation and quantification of unexpected phenomena; and (4) the design process used, including the hydrodynamic model, was successful and can be used again for similar situations.

Keywords Bay of Fundy • Ice • Macrotidal • Restoration • Salt marsh • Salt pool • Sedimentation • *Spartina*

21.1 Introduction

21.1.1 Environmental Context

The Atlantic coast of North America is an old coast characterized by highly eroded mountain ranges and, at lower latitudes, extensive sedimentary shorelines (Bally and Palmer 1989; Bertness 2007). Salt marshes are a dominant habitat south of New England (Bertness 2007). At higher latitudes, past glacial scour moved much of the sediments offshore, leaving rocky shores as the dominant coastal habitat (Maine, Atlantic coast of Nova Scotia, Newfoundland). An exception is the Bay of Fundy (Fig. 21.1). In the upper Bay of Fundy (along both arms, Chignecto Bay and Minas Basin), tidal flats and salt marshes are the dominant habitat. For example, on the New Brunswick coast of Chignecto Bay, tidal flats and salt marsh comprise 92% of the coastline area (58% and 34%, respectively; New Brunswick Department of Natural Resources unpublished data).

The soft-sediment habitats in the Bay of Fundy are the result of very large, semidiurnal tides (some of the highest in the world, frequently over 14 m amplitude in the upper Bay) and consequent strong eroding tidal currents on surrounding cliffs of friable sedimentary rock, primarily shale and siltstone in Chignecto Bay (Desplanque and Mossman 2004). The upper Bay also has very high suspended sediment concentration in the water, typically 300 mg/L or higher (Amos and Tee 1989).

The Bay of Fundy, being in the north temperate zone, experiences winter disturbance including annual freezing and ice scour along the coastline, but not in the open water (Desplanque and Mossman 2004; Drolet et al. 2013; Gordon and Desplanque 1983). In salt marshes, the strong seasonality means that plants become senescent and die in the fall. Dead grass stems and blades either get sheared in wintertime, forming wrack (mats of dead plant matter), or get crushed and eventually buried forming peat (Baerlocher and Moulton 1999; Redfield 1972). Winter ice not only scours the surface of the salt marsh (Ewanchuk and Bertness 2003), but also damages its seaward edge, leaving it in a constant state of recovery. Furthermore, ice blocks move sediment and marsh turf around, which depending on currents and wind are often deposited high on the shoreline, including in the high marsh (Argow et al. 2011; Ollerhead et al. 1999; Redfield 1972).

Because of cool summer air temperatures and intense water movement, the saltwater cordgrass *Spartina alterniflora*, the bioengineer species of salt marshes along

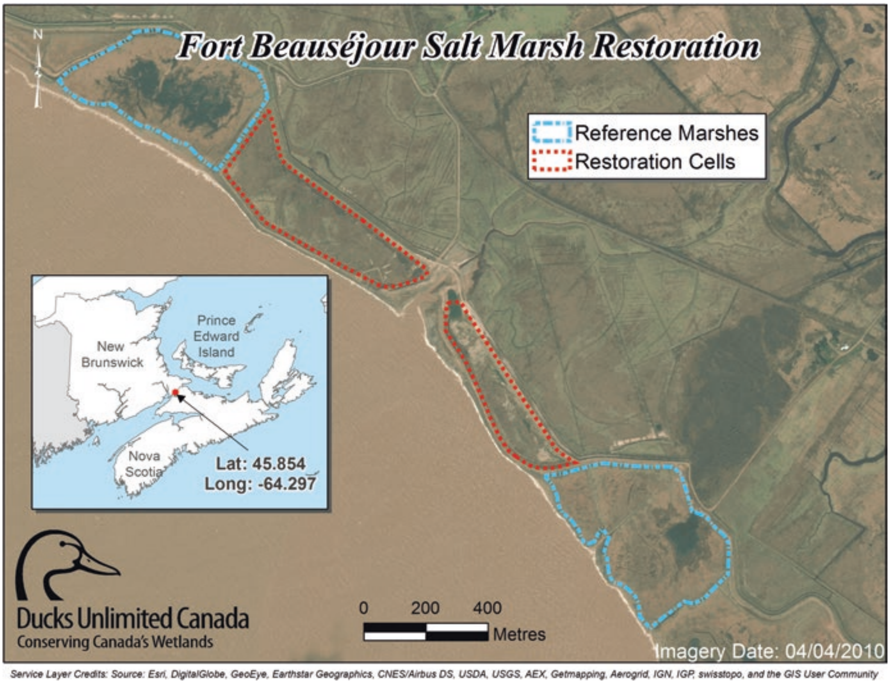

Fig. 21.1 Location of the restoration cells (Northwest or NW cell, and Southeast or SE cell) and the reference salt marshes in Aulac, New Brunswick, Canada. The restoration will ideally provide 18.4 ha of new salt marsh (12.6 ha from the NW cell and 5.8 ha from the SE cell)

the Atlantic coast of North America (Redfield 1972; Bertness 2007), grows to a relatively small size in the Bay of Fundy. *Spartina alterniflora* plants on the exposed marsh edge grow to 30 cm in height by August/September. This is much less than those growing along protected creeks, which can reach 90 cm plant height, or in salt marshes along the nearby Northumberland Strait, where tidal amplitudes are only 1–2 m. In a study that specifically compared *S. alterniflora* growing in the Bay of Fundy and in the Northumberland Strait, Baerlocher and Moulton (1999) suggested mechanical fragmentation, leading to mass loss in apical sections of leaves as well as whole leaves, as an underlying mechanism for smaller plants. This effect on plants in the Bay of Fundy is mentioned as a relevant example of the various and often cryptic effects of the macrotidal environment.

21.1.2 Historical Context

Salt marshes in the Bay of Fundy have a long history of human use and manipulation (Hatvany 2003; Mount Allison University Archives 2004). Indigenous peoples harvested plants and animals from salt marshes for at least 5000 years prior to

European contact. Oral histories allude to salt marshes being used seasonally by the Mi'kmaq people with seasonal encampments along marsh edges.

European exploration of the region began prior to 1600, with French colonists and their descendants (the Acadians) settling in the Minas Basin region starting in the mid-1600s (Butzer 2002; Nova Scotia Department of Agriculture and Marketing 1987). The colonists brought with them marsh diking technology utilized in France. Coastal marshes are extremely fertile areas and were seen as excellent potential farmland; so, the Acadians began to build dike systems complete with "aboiteaux", equivalent to modern-day sluice gates. The aboiteaux were important in enabling water on the land to drain, without letting seawater in, thus allowing soil salinity to diminish and traditional crops to grow successfully. Rapid reclamation of marshlands had begun with 24 aboiteaux by 1650, an extensive dike system by 1671 (Butzer 2002), and new settlements and additional dike networks starting in the Chignecto Bay region in 1672 (Mount Allison University Archives 2004). English and American colonists in the latter half of the eighteenth century also influenced the history of the dikelands (Mount Allison University Archives 2004; Nova Scotia Department of Agriculture and Marketing 1987).

During the "hay day" period for the marshes (early twentieth century), new technologies in ditching, diking and drainage were developed, leading to additional reclamation of marsh habitat for agriculture at the expense of pristine salt marsh complexes (Mount Allison University Archives 2004; Nova Scotia Department of Agriculture and Marketing 1987). The increase in automobile use in the 1920s and 1930s, led to the decrease in demand for hay and a lowering of hay prices from $28 to $7/ton. During this time period, agricultural areas of reclaimed salt marsh were abandoned entirely or reutilized as cattle pasture. During the Great Depression, minimal funds were allocated to the maintenance of the dike systems, which led to several cases of passive restoration. Post WWII, industrial advancements paired with increased value of coastal land led to revamping of dike systems to reclaim large sections of coastal habitat for other uses such as transportation and infrastructure.

21.1.3 Salt Marsh Restoration

As much as 85% of the original salt marsh has been lost in the Bay of Fundy since the arrival of early European colonists (Hanson and Calkins 1996; Thomas 1983). Due to the substantial historical loss and the numerous ecosystem services that salt marshes provide (Gedan et al. 2009; Chmura et al. 2012), there has been growing interest in restoring these systems. Over the last decade, there have been several coastal wetland restoration projects in Atlantic Canada (Bowron et al. 2012), most of which were in the Bay of Fundy. These projects were largely in response to compensation needs in support of wetland policies (van Proosdij et al. 2014) or to offset fish habitat loss. Many of these restoration projects were designed and actively implemented with the main goal of restoring wetland habitat. However, some had an additional functional goal of providing coastal protection from increased risk of erosion from climate change and sea level rise (van Proosdij et al. 2014). These projects all involved the partial or complete removal of dikes or the removal/upscaling of culverts to increase tidal access to inland areas. Some also included excavation to open former creeks and to create salt pools (Bowron et al. 2012). Many of these planned projects included a monitoring regime focused on sediment accumulation and ecosystem recovery/change (e.g., soils and sediments, geomorphology, vegetation, fish, invertebrates; Bowron et al. 2012).

In comparison to most other planned salt marsh restoration projects, the Fort Beauséjour salt marsh restoration discussed in this chapter is unique. First, it is located at the head of the Bay of Fundy and is completely exposed to wave action and an extremely large tidal range. Other restoration projects on the Bay, documented by Bowron et al. (2012), have been implemented in protected areas not exposed to significant wave energy (e.g., Cheverie Creek and Walton, Nova Scotia). Additionally, our project is one of a few in Atlantic Canada that has as a primary objective to create coastal protection, and it is likely the first where there is risk to coastal infrastructure if the project is unsuccessful (van Proosdij et al. 2014).

Prior to planned salt marsh restoration projects, there have been documented examples of passive restoration projects in the region. These sites have provided researchers opportunities to gain some understanding of long-term projections, expectations, and recovery rates of salt marshes (Byers and Chmura 2007; Flanary and Chmura 2007; MacDonald et al. 2010), as well as historical information about these areas before, during and after diking (Noël et al. 2005). Examples of these sites include the largest existing salt marsh in the Bay of Fundy, the John Lusby saltmarsh complex (600 ha of formerly diked farmland in Amherst, Nova Scotia). Its restoration and transition to salt marsh began in the 1940s when the dikes began to breach due to neglect of the existing dike system (Byers and Chmura 2007; Flanary and Chmura 2007). This salt marsh, located at the upper reaches of the Bay of Fundy, is now managed by the Canadian Wildlife Service (Environment Canada). Another example of a passively restored salt marsh is the 95 ha Saint's Rest marsh in Saint John, New Brunswick. Similar to John Lusby, it is former agricultural land and the dike system was not maintained, which led to breaching in the 1950s (Noël et al. 2005).

21.2 Objective of the Chapter

The objective of this chapter is to take lessons learned from the design, implementation and monitoring over 6 years (1 year pre-breaching of the dike and 5 years post) of the Fort Beauséjour salt marsh restoration in the upper Bay of Fundy, and consider how they can be applied to future restoration projects in a similar environment.

21.3 Study Site and Project Development

21.3.1 Study Site Characteristics

The Fort Beauséjour salt marsh restoration project is located in Aulac, New Brunswick (45.854° N, 64.297° W; Fig. 21.1) at the head of Cumberland Basin in the upper Bay of Fundy. Cumberland Basin is a 118 km^2 estuary with semi-diurnal tides that are typically 10–14 m in amplitude, and so is well-flushed and the water column is well-mixed (Amos and Tee 1989; Desplanque and Mossman 2004; Fisheries and Oceans Canada 2016). The water has a very high load of suspended sediments, typically 300 mg/L, but values up to 3500 mg/L have been recorded in the area (station 6 in Amos and Tee 1989). van Proosdij et al. (1999) determined the composition of the suspended sediment to be 95% coarse silt, 2.5% clay and 1.5% sand, and to have a mean grain size of 0.036 mm. The Fort Beauséjour site is exposed to wind and waves, with a fetch ranging from 5 to 20 km (Ollerhead et al. 2010). The mouth of Cumberland Basin is influenced by ocean waves [maximum significant wave height (H_s) = 2 m; maximum period = 8 s; Amos and Asprey 1981 in Amos and Tee 1989]; maximum H_s diminishes to ~1.5 m by the time waves reach our study site at the head of the Basin.

The restoration project comprises four separate sites (Fig. 21.1): two cells that are undergoing restoration (henceforth Northwest or NW cell, and Southeast or SE cell) and two established salt marshes that serve as reference sites (West and East reference marshes). The restoration cells were farmland protected by dike before the restoration project began.

21.3.2 The History of the Fort Beauséjour Project Partnership

The potential for a salt marsh restoration was first assessed in 2006 following the re-alignment of the agricultural dike by the New Brunswick (NB) Department of Agriculture (Millard et al. 2013). A new dike was constructed inland as the seaward dike was eroding more quickly than was feasible to maintain. The foundation that allowed this project to occur stems directly from partnerships that were formed among various groups (government, academic and private; Fig. 21.2a) who saw the value and benefits of this work. It is important to stress that without these partnerships, this work (and the knowledge gained from it) would not have been possible.

In spring 2009, Ducks Unlimited Canada (DUC) approached the NB Department of Agriculture with a conceptual plan to complete a salt marsh restoration project in the area between the new and old agricultural dikes. For DUC, this would provide the benefit of additional salt marsh habitat, while the Department of Agriculture would benefit from a buffer from wave action and erosion for the newly constructed dike (see also French 2006). Discussions began into potential funding sources for

Fig. 21.2 Photographs of the initiation and implementation of the restoration project. (**a**) Representatives of project partners, namely Ducks Unlimited Canada (DUC), Fisheries and Oceans Canada, New Brunswick Department of Agriculture, Aquaculture and Fisheries, New Brunswick Department of Transportation and Infrastructure, and Mount Allison University, initiate dike breaching at initiation ceremony. (**b**) Construction of the opening in the NW cell (the westernmost breach). Note the cut in the middle (similar to the concept tested in Fig. 21.3) and the armor stone on the breach sides. (**c**) Finished breach at the southern end of the SE cell (i.e., the eastern opening); photograph taken looking to the northwest. (**d**) Sediment-laden salt water entering the restoration site after the breaching (Photographs taken in October 2010 by Jeff Ollerhead, except **a** by DUC)

this project. The province of New Brunswick has policies that aim to protect wetlands (Government of New Brunswick 2002), which include a compensation process requiring proponents to pay for the replacement of lost or altered wetlands. The NB Department of Transportation and Infrastructure and Fisheries and Oceans Canada both have wetland compensation and fish habitat creation needs and thus were interested in forming partnerships with this project too. Actual project discussions began in May 2009 and resulted in funding sources to support infrastructure improvements to the new dike system (armor rocking, machinery time, etc.) and implementation and monitoring of the restoration project. The initial research team (including Mount Allison University, Acadia University, and DUC) began baseline monitoring of the site and developed a design for the restoration project and long-term monitoring program. This initial work included a plant inventory, analyses of site elevation and tidal water level measurements. In summer 2010, prior to the mechanical breaching of the sites, research and monitoring were expanded to include birds, nekton, invertebrates and salt pools, and the addition of the University

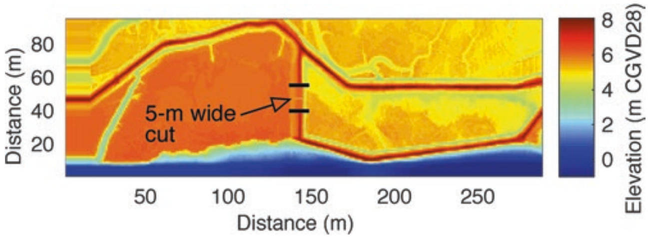

Fig. 21.3 Example of a hypothetical design for breaching the dike. The grid has a 5-m resolution and is rotated so that the shoreline is parallel to the x-axis. The domain was extended southwards and westwards. The West reference marsh was resolved. A 75-m wide entrance was opened to a level of 6 m (CGVD28), and a 5-m wide cut was made near the middle of the entrance to a level of 5.5 m

of New Brunswick to the research team. The long-term monitoring continues to be supported by the NB Environmental Trust Fund, DUC, MITACS, University of New Brunswick, and Mount Allison University.

21.3.3 Site Design

The team charged with designing the dike openings followed a process similar to that outlined by Coats et al. (1989). The design had to satisfy a number of competing criteria. Specifically, the restoration cells were to be designed to:

- fully flood at high tide (i.e., the water level in the Bay and cells should be equal);
- drain slowly to maximize the opportunity for sediment deposition; and
- minimize wave impact on the new agricultural dike.

It was also necessary to find a design that did not result in water flow speeds through the openings that would exceed the threshold for erosion. It was important that the openings maintain their geometry for as long as possible (to meet the other criteria outlined above). The main goal here was to maximize sediment deposition to build up the elevation in the cells so that salt marsh could begin to re-establish; in other words, a geomorphic foundation needed to be laid for the restoration of ecosystem services. Before the openings were excavated, the ground within the restoration cells was below that of the East reference marsh by up to 2.1 m.

To prevent the blue clay floors of the dike openings from eroding, as a result of tidal flow through the openings, flow speeds needed to be kept below 1.5 m/s, which is the approximate threshold for erosion at the clay/silt boundary (see the Hjulström diagram; e.g., Figure 10.12 in Trenhaile 2016). Blue clay is a relatively uniform clay with very low permeability and occasional sand pockets and/or cobbles

(Terzaghi et al. 1996). Given that the fill materials comprising the dikes themselves were more erodible, the decision was made to armor the sides of the dike openings with rock repurposed from the sites.

A variety of breach design possibilities were explored to find the best way to minimize wave impact on the new dike. The openings had to prevent waves from the Bay impinging on the new dike to the extent possible. The new dike needed to survive with minimal protection until a "healthy" marsh established in front of it. Some other design elements were considered, such as grading the site, excavating creeks and/or pools, and creating mounds within the cells to possibly speed up the establishment of vegetation; however, these ideas were abandoned when it became evident that the project budget could not possibly accommodate such features.

Site design began with a review of a previously conducted regional survey of potential restoration sites (Millard 2008). This work provided a solid foundation for planning the project and was an invaluable starting point. Millard (2008) considered the viability of restoring salt marshes in several locations near Aulac, NB, including the site restored in this project. The possibility of raising the ground elevation in the potential restoration sites via sediment deposition by tiding (i.e., a breach in the dike that could be opened and closed) and by creating a full breach was analyzed. Millard concluded that the only way to get adequate sediment deposition (to raise the ground to a level where vegetation could establish) was through tiding, which was not possible due to budgetary constraints. This conclusion assumed that sediment settling speed could be estimated using Stokes' Law (which is about the frictional force exerted on small spherical objects in a viscous fluid). However, silt and clay particles are often deposited in flocculated forms, which have a higher settling speed than the individual constituent particles (Christiansen et al. 2000).

Given that our task was to develop a design for this site that could not involve tiding or adding fill (too expensive), Millard (2008)'s conclusions were carefully considered. We ultimately disagreed with some of her conclusions for two reason: (i) we had concerns that the volume to be filled by sedimentation had been overestimated and, most crucially, (ii) we are convinced, based on data from field experiments, that flocculation plays a significant role in sediment settling in this environment and that Stokes' Law is not a good predictor of actual sediment settling rate.

Other cautions offered in Millard's assessment we fully shared. For example, our site "experiences high relative exposure [to waves] and seaward dykes are currently being eroded. Even if the elevation of the land was raised by sediment loading, it may not be possible to ever completely remove the seaward dykes without further eroding the land behind, including the newly deposited material" (Millard 2008, p. 133). As discussed in this chapter, attempting to design a project whereby viable salt marsh would develop before the seaward dike eroded completely, leaving the whole site vulnerable to erosion by waves, was, and remains, a major challenge.

21.4 Methods, Results and Discussion

21.4.1 How Did We Meet Our Criteria for Planning the Breaching of the Dikes?

As outlined in the previous section, our task was to design dike breaches that would allow the restoration cells to fully flood, result in maximum sediment deposition, and minimize wave impact on the new dike landward of the failing dike, etc. To do this, a data set was assembled as follows.

- K.E. Millard provided us with the 2006 LiDAR-derived digital elevation model that she had created for the sites for her M.Sc. thesis work (Millard 2008).
- Water level data were recorded at the sites for multiple spring-neap tidal cycles using a self-logging pressure sensor (Onset Computer Corporation). Data were relative to the Canadian Geodetic Vertical Datum of 1928 (CGVD28).
- Tide height predictions for Canadian Hydrographic Service (CHS) station 215 (Joggins, NS) relative to Chart Datum were obtained for the period coinciding with the recorded water level data.
- Aerial photography was acquired for the sites in August 2009 from Nortek Resource Solutions Inc. with 15 cm pixel resolution.
- Field observations and ground-level photographs were collected in the summer of 2009.

The measured and predicted water level data were then analyzed to develop a tidal transformation equation that allowed us to convert predicted tide height at CHS station 215 to predicted water level at the sites, relative to the restoration cell surfaces and reference marshes, for any time. The resulting data allowed us to calculate flooding frequencies at various water depths for the proposed restoration cells for a multi-year period. Maximum flooding depth was over 2 m at peak spring tide.

Sanderson created a two-dimensional wetting-drying model for the site using standard approaches (similar to Sanderson 2011; reports by Sanderson in Ollerhead et al. 2010). His hydrodynamic model solved the shallow water equations and included an algorithm for wetting and drying surfaces as the tide rose and fell. Field accelerations were computed using a semi-Lagrangian method (McGregor 1993) with bottom drag calculated implicitly (the drag coefficient parameterizes the magnitude of friction between the water column and the underlying bed). The barotropic mode was calculated using iterative under-relaxation. A small number of iterations corresponded to treating all but the shortest waves as being explicit.

Possible openings in the old dike were then simulated and run in the model. Possibilities explored included different numbers of openings, different widths, openings with and without notches, and various placements around the restoration cells. See Fig. 21.3 for an example of a design that was tested but not used. All outputs were evaluated against the design criteria (listed above). If one or more of the criteria were violated, then the design was discarded. For example, a narrow opening

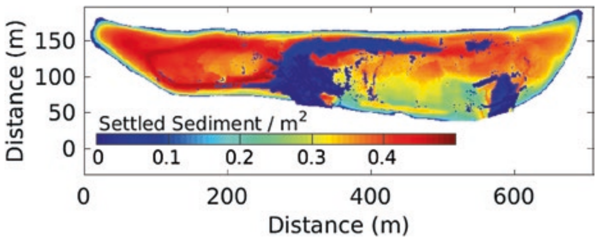

Fig. 21.4 Example of model output showing the expected pattern of sediment deposition in the SE cell. The openings are as-built and the units (settled sediment per m^2) are arbitrary

that resulted in predicted flows on the flood tide of 2.5 m/s was discarded. Discussing the modeling in detail is beyond the scope of this chapter, but the following was considered: (i) flow velocities through the hypothetical openings and in the restoration cells, (ii) likely patterns of sediment deposition, and (iii) possible wave impact on the new dike. The ideal designs were ones that minimized (i) and (iii) and maximized (ii). The final design selected balanced optimal modeled outcomes with construction cost and complexity.

Given that building a sediment platform for a new salt marsh was of paramount importance, particular attention was paid to sediment deposition (ii). An example of model output for sediment deposition is included as Fig. 21.4. Note that the units (settled sediment per m^2) are arbitrary; this is because there is no reliable method for estimating the rate of sediment accumulation that accounts for flocculation (Christiansen et al. 2000). However, the model was helpful in predicting the pattern of relative sediment accumulation. Also, since Millard (2008) had calculated a maximum sediment deposition rate of 51.3 cm/year for this general site (based on the empirical work of van Proosdij et al. 2000, 2006a, b), we were confident that sufficient sediment would be deposited to provide a foundation for a successful salt marsh restoration, at least for the first 5 years.

After final design approval, construction commenced in the early autumn of 2010. One opening was excavated for the NW cell and two openings were excavated for the SE cell (Figs. 21.2b, c and 21.5). The intention was that the floor of each opening would be at approximately 5.2–5.4 m relative to mean sea-level (MSL) as defined by CGVD28. The floor of each opening was composed of blue clay and, as noted previously, the sides were armored with rock (Fig. 21.2b, c). Unfortunately, the middle opening (in the SE cell) was cut down too far during construction and ended up closer to 5.0 m relative to MSL (CGVD28). There was no way to reverse this excavation once completed. This meant that water flow in and out of the SE cell using the middle opening may be expected to be greater than planned. With the openings cut, flooding of the site with salt water commenced (Fig. 21.2d).

One reason for making the openings relatively wide was to allow for the possibility of ice blocks entering the restoration cells (see Fig. 21.6 for examples of winter conditions at the sites). In December 2012, the cells were covered by some snow and ice, but the channels were open and there were no ice blocks (Fig. 21.6a). By

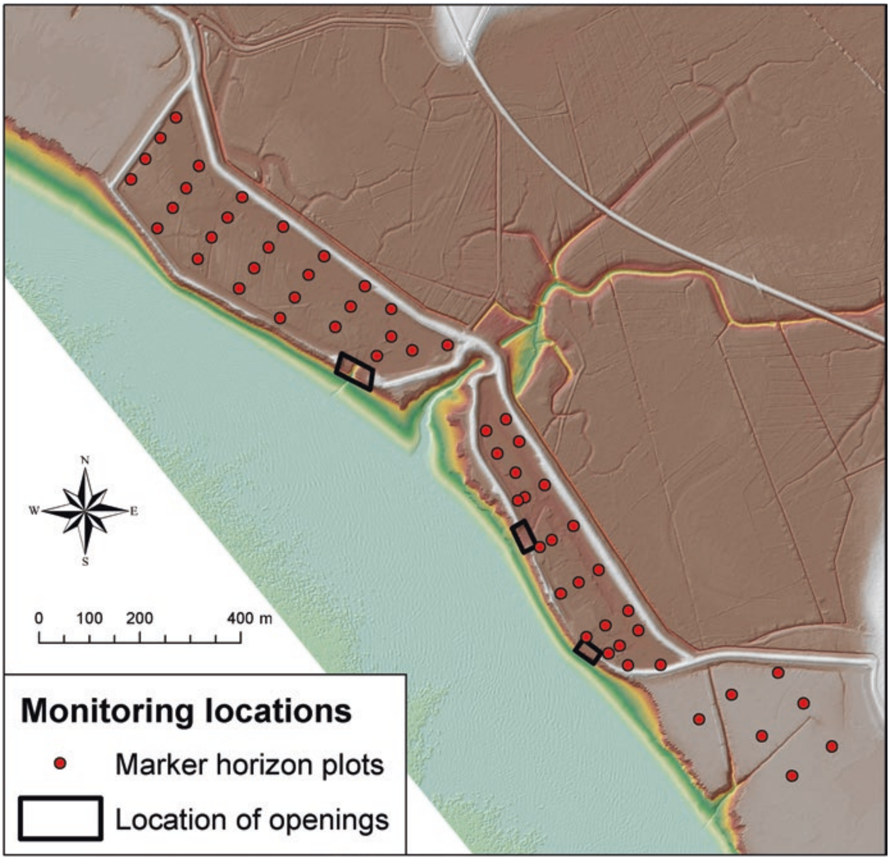

Fig. 21.5 The restoration cells showing the breaches in the old dike (*black-lined rectangles*) and the locations of the marker horizon plots (*red dots*) to study sediment dynamics. Note the seven marker horizon plots in the East reference marsh

February 2013, there were many ice blocks in the bay by the sites, including blocks exceeding 30 m^3 in size (Fig. 21.6b). Areas of the cells close to the openings were littered with ice blocks and many of the channels were plugged with ice and ice blocks (Fig. 21.6c, d). The openings themselves were plugged with ice blocks (e.g., the middle opening in Fig. 21.6e) and flow into and out of the cells was likely being restricted by this ice.

The amount of ice in the restoration cells in late February 2013 was impressive, but varied throughout. In some locations, the cell's surface could not be seen because of the amount of ice and snow (e.g., Fig. 21.6f). Most of the ice carried sediment (see also Macfarlane et al. 2011; van Proosdij et al. 2006b). Any ice blocks that melted within the cells likely deposited sediment. At some higher-elevation locations in the cells, the contribution of sediment from ice blocks to the overall sediment budget

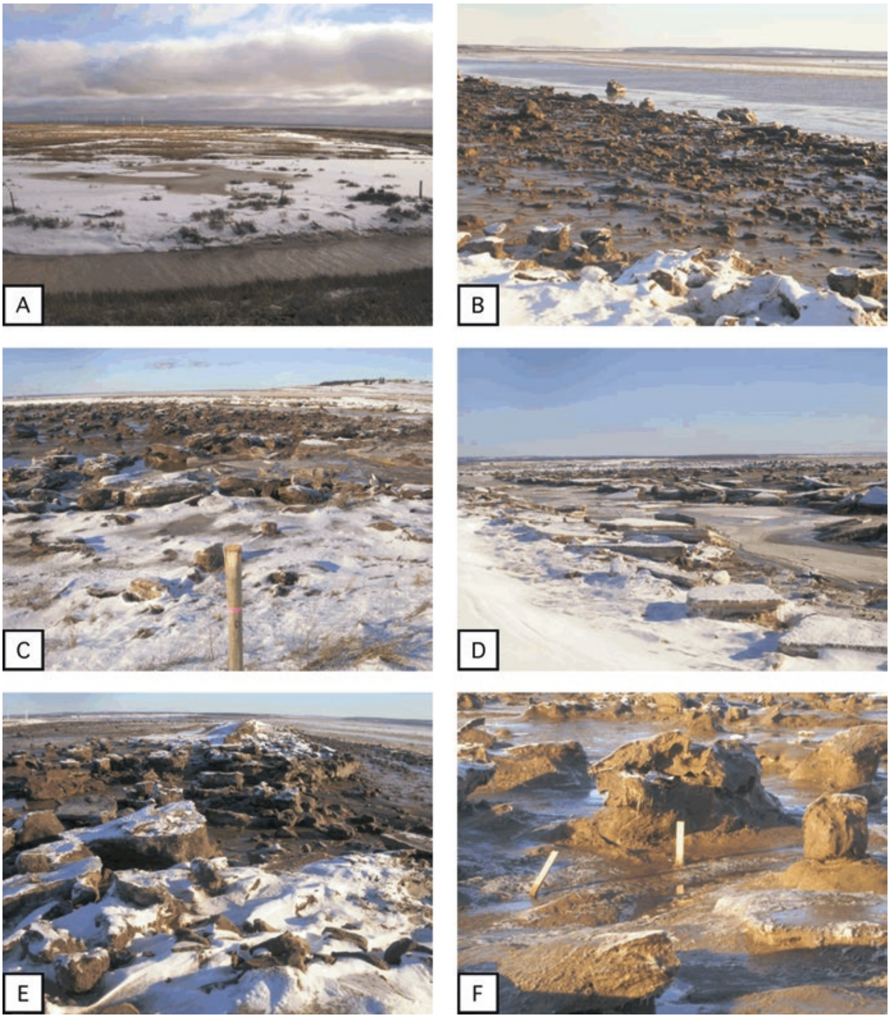

Fig. 21.6 Photographs illustrating winter conditions in the restoration cells in December 2012–February 2013. Panels **a**–**f** are described in the accompanying text (Photographs by Jeff Ollerhead)

may be significant. This has been observed in some salt marshes in eastern North America (Argow et al. 2011; Ollerhead et al. 1999).

Given the amount of ice that can occur within the cells, it is not surprising that the stakes installed in 2010 to indicate marker horizon plots (Fig. 21.5; discussed below) are almost all gone and that a number of marker horizon plots have been lost (eroded away). The ice erodes the surface and can change the local hydrology while it is present. Based on winter observations (e.g., Fig. 21.6), the goal of allowing ice into the restoration cells to contribute to the natural geomorphic evolution of the restored salt marshes was achieved.

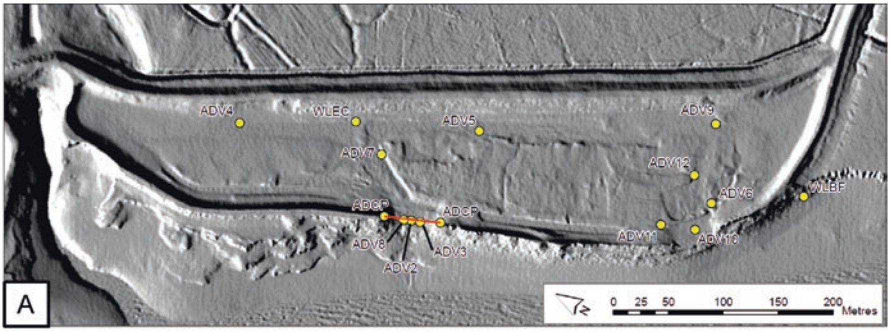

Fig. 21.7 (**a**) Locations of instruments installed to measure water flow over high tide cycles in the SE cell in autumn of 2012. The ADCP (Acoustic Doppler Current Profiler, specifically a RDI StreamPro) was towed across the middle opening (*red line*). Note the channel in the LiDAR image, running north from the middle opening, that directs water in the +u direction on the ebb tide. *ADV* Acoustic Doppler Velocimeter (SonTek Triton) and *WL* water level logger (Onset Hobo). (**b**) Photograph of the ADCP being towed across the middle opening (in SE cell) (Photograph by Jeff Ollerhead)

Qualitative and quantitative observations made post-construction of the dike openings suggested that the final approved design was functioning as planned. The restoration cells were fully filling with sea water and draining slowly, giving sediment time to deposit. By the end of the first year, substantial sediment had been deposited in both cells and there was no obvious erosion of the openings; these results will be discussed more fully in a subsequent section. Of interest here is evaluating the success of our design in terms of model predictions and hydrodynamic processes. In autumn of 2012, water flows through the middle opening were measured; this opening was selected because if the design criterion of 1.5 m/s maximum flow were to be violated anywhere, it would be at the middle opening given the overcut to the floor. Using measured water levels, breach geometry and site

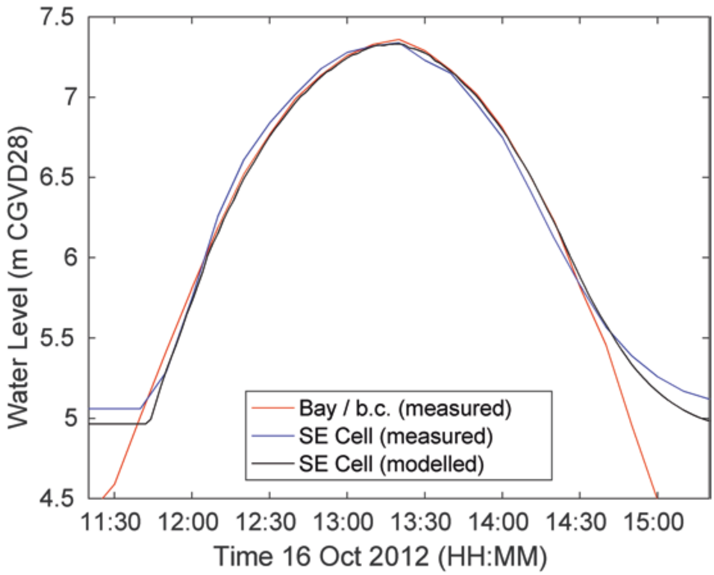

Fig. 21.8 Comparison of water levels on 16 October 2012 in the SE cell. The *red line* shows water level as measured in the Bay and as used at the open boundary of the model (i.e., b.c. = boundary condition used to drive model). The *blue line* shows water level measured inside the SE cell. The *black line* shows modeled water level inside the SE cell at the same position as measured. Conditions set in the model were high bottom friction and filtered forcing

topography, a slightly modified version of the two-dimensional wetting-drying model used to design the breaches was rerun for tide cycles for which flow data were recorded using an Acoustic Doppler Velocimeter (ADV) and/or Acoustic Doppler Current Profiler (ADCP) (Fig. 21.7a). The ADCP was pulled across the middle breach using a cable (Fig. 21.7b). The ADV was mounted in various locations in the SE cell (Fig. 21.7a) on a post driven into the ground.

There was very good agreement among water levels in the Bay, measured in the SE cell, and modeled (Fig. 21.8). As well, the desired lag in water level drop after the high tide was achieved by the design. Results for both cells were equally satisfactory.

Measured and modeled water velocities were also compared for all locations shown in Fig. 21.7a, using both a high and low bottom drag coefficient due to uncertainty with respect to setting this parameter for the site. In general, there was very good agreement for both alongshore and across-shore water velocities (see Fig. 21.9 as an example and Table 21.1). For most of the comparisons, the model indicated an inflowing current beginning about 10 min earlier than measured by the ADV (positive v current, Fig. 21.9). The modeled outflow (negative v current) also persisted longer than the ADV measurements. This is because the ADV was measuring flow at 10–15 cm above the bottom, and so no data could be recorded until the water was

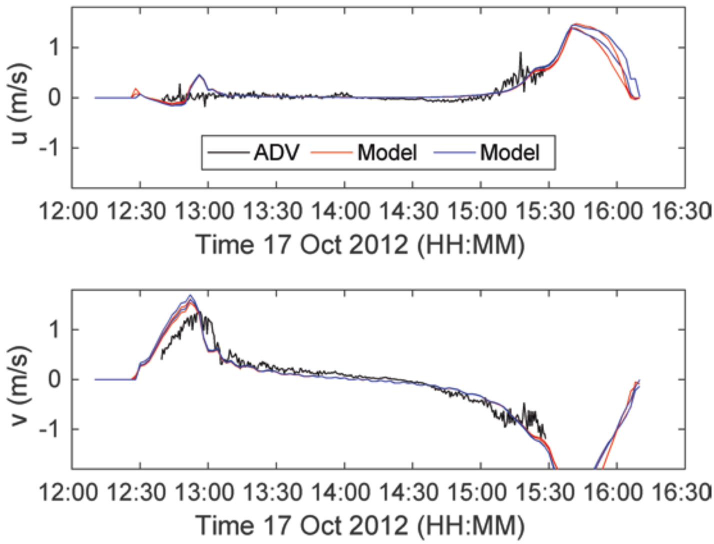

Fig. 21.9 Comparison of modeled currents (*colored lines*) with ADV measurements (*black line*) in the SE cell on 17 October 2012 at position 8 (middle opening; see Fig. 21.7a). Positive u is alongshore current to the right (i.e., southeast direction) and positive v is across-shore current into the cell. The two model runs (i.e., *red lines* vs *blue lines*) had different bottom friction and filtering conditions, and discussing the differences is beyond the scope of the chapter

at least 20 cm deep. Vector correlation analysis (coefficients *R*, Table 21.1) for 19 data sets of water velocities also showed good correspondence. Note that in most instances, using a high bottom drag coefficient produced modeling results that better matched measurements. The vector correlations for a few locations near the eastern opening were in less agreement (notably positions 11 and 12, Fig. 21.7a and Table 21.1), probably because of a complex topography that was poorly resolved by both measurements and the computational model.

Finally, water discharge measured through the middle opening was compared to model output. Model output was broadly consistent with ADCP measurements (see Fig. 21.10 as an example). Also, similar to the ADV data, the ADCP-derived discharge tended to be more consistent with model simulations when using a higher bottom drag coefficient (although model performance was not particularly sensitive to the bottom drag coefficient selected). Note that the modeled discharge had more variability than the measured discharge, which was probably due to the averaging inherent in calculating discharge from a single ADCP that traversed the entrance (Fig. 21.7b). Overall, the results were very reasonable and robust.

Table 21.1 Vector correlation coefficients *R* obtained by comparing model output to ADV measurements made in 2012 (see Fig. 21.7a for the positions in the SE restoration cell). *R* can vary between 0 corresponding to no vector correlation and 2 corresponding to a perfect vector correlation; this analysis is a generalization on the usual method of correlating one-dimensional fields to a two-dimensional space (Crosby et al. 1993). Two types of model simulations were run, one with high bottom drag and a second with low bottom drag

Time	Position	*R* (with high bottom drag)	*R* (with low bottom drag)
21 Sep 1642	2	1.278	1.313[a]
22 Sep 0514	2	1.588	1.159
22 Sep 1740	3	1.073	0.974
17 Oct 1355	8	1.460	1.340
16 Oct 1311	4	0.867	0.583
17 Oct 0136	5	0.914	0.879
21 Oct 1719	7	1.046	0.891
20 Oct 0400	9	0.948	0.819
20 Oct 1622	9	0.965	0.813
21 Oct 0455	9	0.976	0.896
18 Oct 0222	6	1.052	0.629
18 Oct 1441	6	1.201	0.473
19 Oct 0309	6	1.363	0.700
19 Oct 1530	6	1.093	1.012
14 Nov 0031	10	0.990	1.194[a]
14 Nov 1250	10	1.265	0.801
15 Nov 1336	11	0.477	0.391
16 Nov 0205	12	0.532	0.676[a]
16 Nov 1424	12	0.591	0.738[a]

[a]Indicates the few cases where the low drag calculation was better correlated to ADV measurements

In sum, the modeling approaches taken to design the breaching of the old dike were sound and successful. Sanderson's two-dimensional wetting-drying model allowed us to a-priori test various breach designs, evaluated against clearly thought-out design criteria. Water flow, as measured through the middle breach (on 17 October 2012, Fig. 21.9), did not exceed 1.5 m/s even at maximum spring tide. Nothing unexpected happened, and even with the over-excavation of the middle breach, the design criteria were not violated.

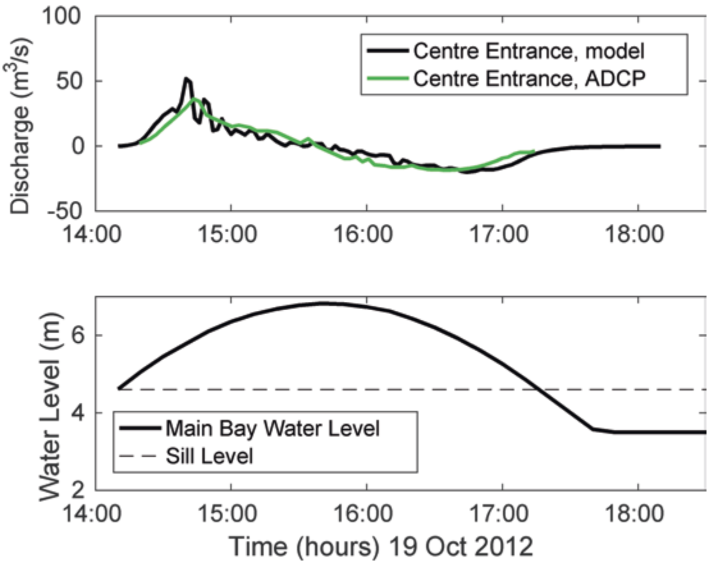

Fig. 21.10 Comparison of measured and modeled discharge of water through the middle opening (breach) in the SE cell. Model calculations are compared with ADCP measurements made on 19 October 2012. Note the very reasonable agreement between model output (*black line*) and the corresponding ADCP measurements (*green line*). The lower panel shows the water level used to drive the model at its offshore boundary. This simulation used a high bottom drag coefficient and included Rayleigh damping

21.4.2 Were the Pattern and Amount of Sedimentation as Expected?

To measure the rate and amount of sediment deposition in the restoration cells, 50 plots with a marker horizon were installed (Fig. 21.5) in early October 2010 (before the breaching of the old dike). An additional seven marker horizons were installed in the East reference marsh (Fig. 21.5). The method used was similar to that described in Bowron et al. (2009). Briefly, each plot was 0.75 m × 0.75 m in size and a 5 cm × 5 cm wooden post was used to mark it (Fig. 21.11a). The vegetation was trimmed where necessary and a layer of feldspar clay deposited on the surface (about 5 mm thick). The location of each plot was measured with Real-Time Kinematic and Differential GPS (RTK DGPS; Leica 1200 with three-dimensional accuracy of 2 cm or less). Each subsequent October, the marker horizons were sampled (if possible) using cryogenic coring (Fig. 21.11b, c). A plot was typically relocated using RTK DGPS, since many of the posts installed in 2010 were damaged (Fig. 21.11b), missing, or buried within the first year. If a marker horizon could not be located after three cores, it was deemed missing or compromised (eroded). Total accretion since the marker horizon was installed was estimated by measuring the amount of material above it (Fig. 21.11c). By measuring at regular temporal

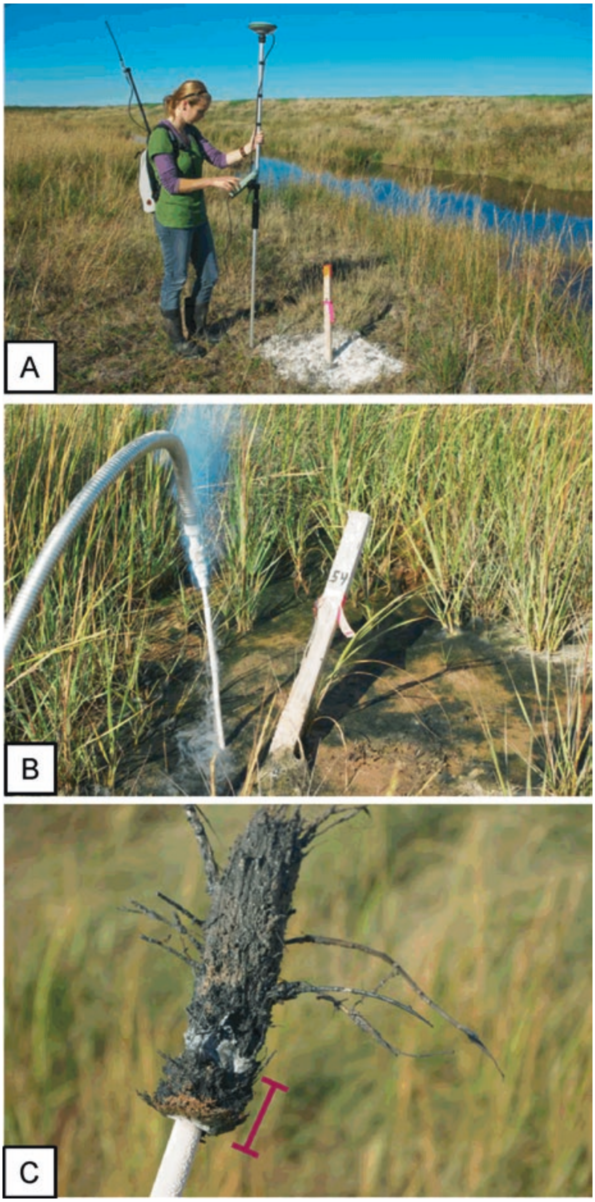

Fig. 21.11 (**a**) Feldspar marker horizon immediately after installation in October 2010; location recorded with a Leica 1200 Real-Time Kinematic and Differential GPS (RTK DGPS). (**b**) Cryocoring a plot in a reference salt marsh in a subsequent year. (**c**) Sediment deposited above a marker horizon (indicated with the *purple line*) in a subsequent year (Photographs by CBWES and Jeff Ollerhead)

intervals, rates of accretion can be determined. This method does not account for other changes such as sediment compaction. The only way to measure absolute change in surface elevation is with a complementary technology like Rod Sediment Elevation Tables (RSET) or RTK DGPS. In our case, because significant changes in surface elevation were expected (i.e., cm to dm), and because ice would have

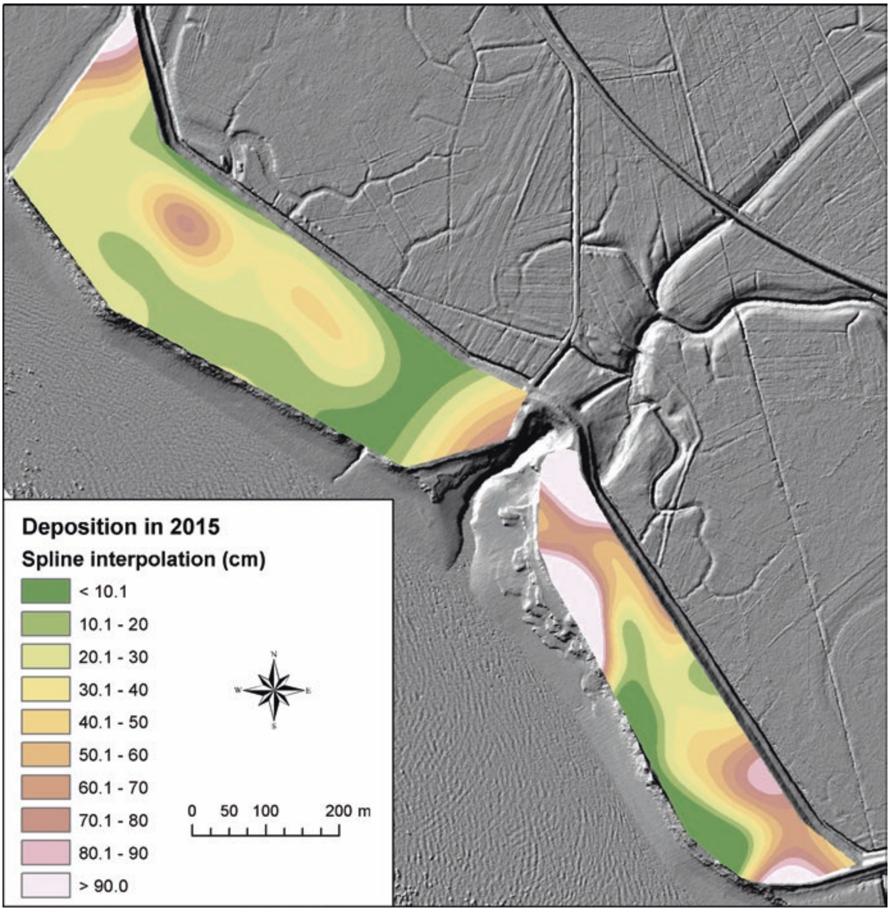

Fig. 21.12 Total deposition of sediment in both restoration cells from October 2010 to October 2015 (5 years). The interpolated surface was constructed using ESRI's ArcGIS software, the data combined from the cryocoring and RTK DGPS measurements for the 50 points (Fig. 21.5), and a "regularized" spline interpolation (constrained to the cell boundaries)

destroyed any RSET stations, we also used RTK DGPS to take a measurement of surface elevation at each plot each year.

It quickly became apparent that both types of measurement (cryocoring and RTK DGPS) needed to be used to determine amount of sediment deposited, because of standing water and/or the thickness of mud deposited over some locations. It was only possible to core through 30–45 cm of mud with the equipment available, and at some locations more than 50 cm of mud was deposited in the first year. Furthermore, some locations could not be measured in the first year because the mud surface was too unconsolidated and unstable to walk on. In 2012 (2 years post-breach) and onwards, all plots were visited. The correlation between the thickness of sediment deposited as measured by the two methods was assessed for each year

Table 21.2 Sediment deposition in the restoration cells and a reference marsh for the period October 2010 to October 2015, estimated from cryocoring and RTK DGPS measurements at marker horizon plots

	Northwest cell	Southeast cell	Cells combined	East reference marsh
Number of plots	28	22	50	7
Mean deposition (cm)	27.1	45.3	35.1	3.5
Standard deviation (cm)	15.3	36.4	27.9	0.8
Maximum (cm)	72.0	96.0	96.0	5.0
Mean ÷ 5 (cm year^{-1})	5.4	9.1	7.0	0.7

Three negative values (−8, −12, −30 cm) resulting from surface erosion were included in the calculation for the Southeast Cell

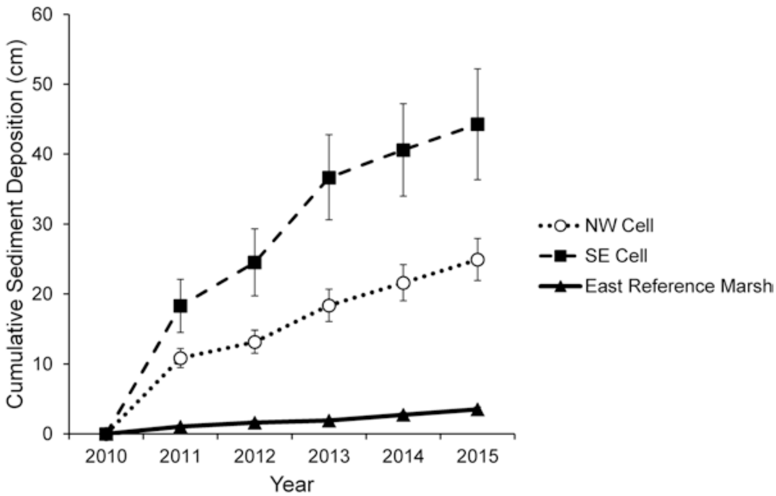

Fig. 21.13 Mean cumulative sediment deposition (with some compaction) for the two restoration cells and the East reference marsh from 2010 to 2015. Error bars are 1 standard error (for the reference marsh, they are smaller than the symbols); $n = 27–28$, 20–22, and 7 for the NW cell, SE cell and reference marsh, respectively

in the restoration cells, and the measurements compared well (Pearson correlation, $r^2 > 0.93$, $n = 32–39$, $p < 0.001$ for each year). Thus, it was concluded that there was no systematic error between the methods. For analysis, the coring measurements were used unless not available. As noted above, the RTK DGPS error is on the order of 1–2 cm, which for 5 cm of deposition would be substantial, but for 50 cm of deposition, it is not.

The amount of sediment deposition recorded in the restoration cells since October 2010 was variable and generally high (Fig. 21.12). In both the NW cell and SE cell, the greatest deposition occurred at the ends of the cell and close to the new dike where the surface was topographically low prior to restoration; this is as expected. Some of the interpolated values of >90 cm in parts of the SE cell are likely overestimates but not grossly so (the maximum value measured in the cell was 96 cm after 5 years). Overall, mean deposition (calculated from the point data) for the reference marsh was 3.5 cm (0.7 cm per year; Table 21.2); this value compares well to work done in the upper Bay of Fundy in the past (Chmura et al. 2001; Ollerhead et al. 2003) and to the 2012, 2013, and 2014 values of 0.8, 0.6, and 0.7 cm per year, respectively, for this reference marsh. Mean deposition in the restoration cells in 2015 (since 2010) was 35.1 cm or 7.0 cm per year. This compares to 10.0, 9.0, 8.8 and 7.7 cm per year for measurements done in 2011, 2012, 2013, and 2014, respectively (Fig. 21.13). Thus, there is decreasing mean annual deposition, which makes sense given that there is less frequent tidal flooding of the restoration marsh surface as elevation increases with deposition. In other words, the rate of deposition should be slowing down and it is. It is also likely that slight compaction of the sediments is contributing to the apparent decrease in mean annual deposition rate, but this cannot be resolved with the RTK DGPS measurements.

Thus, the data collected demonstrate that the mean amount of sediment deposition in the restoration cells is still an order of magnitude greater than in the adjacent reference salt marsh. This outcome is on the order of what was expected based on empirical work measuring sediment deposition over single tidal cycles and longer temporal scales on a nearby salt marsh (van Proosdij et al. 2000, 2006a, b). The deposition in the cells also remains spatially variable (Fig. 21.12). There is now more than 90 cm of deposition in some low areas and essentially none at some high areas. There has been little deposition near the breaches due to higher water flows out of the cells on the ebb tide (i.e., no opportunity for deposition). Furthermore, the rate of deposition in the SE cell remains close to double that in the NW cell. This makes sense given some of the lowest topographic areas that existed in the SE cell prior to restoration (i.e., the greatest depth of water can pool in the lowest areas, providing the greatest amount of suspended sediment to settle out of the water column).

21.4.3 What Happened to the Dikes?

The impetus for the restoration project was erosion of the "old" (seaward) dike, and it has continued to erode. In 2010 and 2011, the three breaches in the old dike were mapped using ground-based laser scanning (LiDAR). Details of this work are not included here, but the key conclusion from that work was that significant erosion on the seaward face of the old dike was occurring beside the openings and that it was causing armor stone on the sides of the breaches to shift and collapse.

The change in shape of the breaches and the ongoing erosion of the whole old dike was studied in 2012. Three digital elevation models (DEMs) were derived from airborne LiDAR data from 2006, 2009/2010, and 2012 using ESRI's ArcGIS software. The 2006 LiDAR mission was flown before the dikes were breached, the 2010 LiDAR mission about 2 months post breach, and the 2012 LiDAR mission about 1.5 years post breach. Thus, these data offered an opportunity to assess changes from

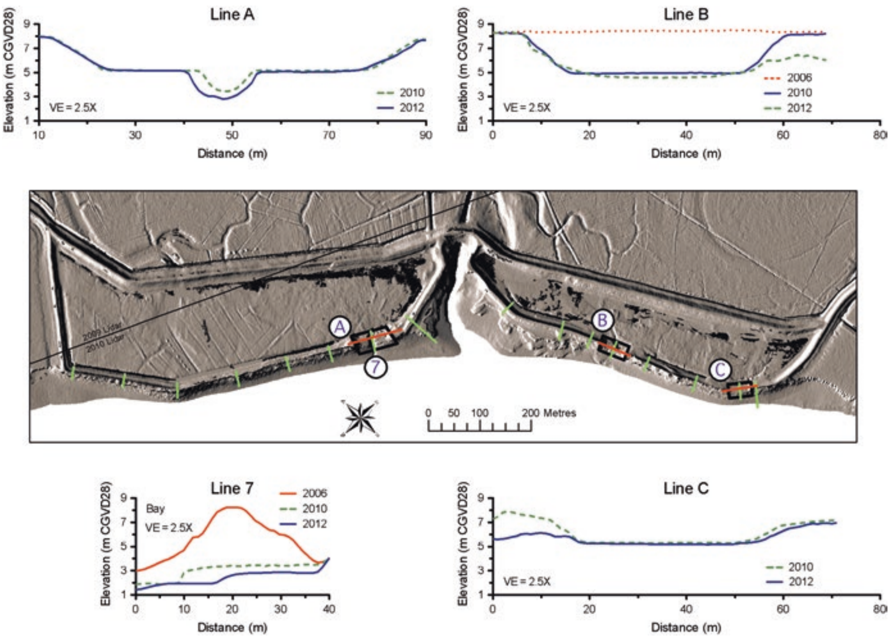

Fig. 21.14 Profiles of the breaches in the old dike from 2006, 2010 and 2012 in the restoration cells. Lines A, B and C are the western, middle and eastern openings, respectively, and Line 7 is through the western opening

pre-restoration to "as built" to post-restoration. A number of lines were drawn across the surfaces in each DEM to extract profiles across the openings and over the old dike adjacent to the openings (Figs. 21.14 and 21.15).

The three openings in the old dike experienced different degrees of erosion over the 2010–2012 period (Fig. 21.14). The cross-section of the western opening (Line A in the NW cell) changed little over the first 2 years. The armor stone was still largely in place and the bed of the channel eroded little (keeping in mind that differences of ±20 cm are generally considered inconsequential due to cumulative errors in the LiDAR measurements and uncertainty in relative accuracy between data sets). The only significant change was for the channel that runs down the middle of the opening (Line 7). This channel was wider and deeper in 2012 than when it was constructed; this was expected given the concentration of ebb flow in this channel on the falling tide. The profile for Line 7 also shows that some of the deepening occurred by headward erosion of the channel bed (compare the 2010 and 2012 profiles). The cross-section of the middle opening (Line B in the SE cell) changed little on the northwest side, but the armor stone had slumped and rolled away and the bank had eroded on the southeast side in the first 2 years. The bed of the opening had eroded little. The pattern was similar at the eastern opening (Line C in the SE cell) where the bed had eroded very little, but at the banks the armor stone had slumped and rolled away and the banks had eroded. One might be tempted to conclude that the

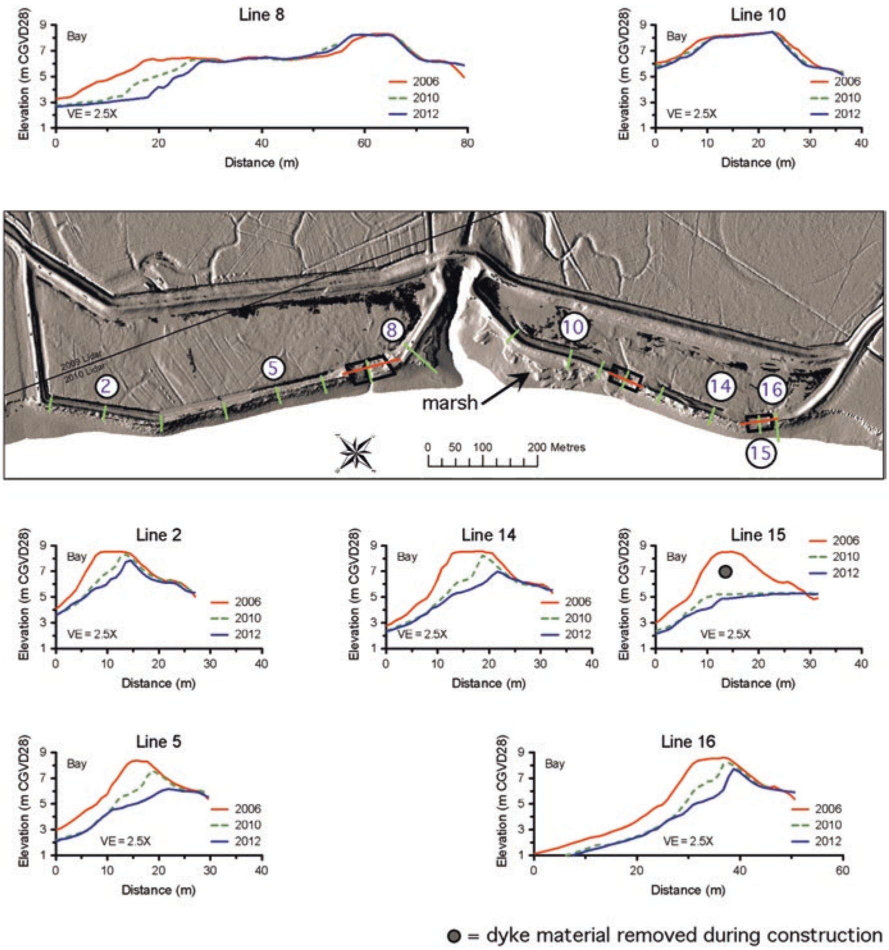

Fig. 21.15 Profiles across the old dike from 2006, 2010 and 2012 for selected areas in the restoration cells

loss of the armor stone had contributed to erosion of the banks. However, consideration of the profiles and observations presented in Figs. 21.14 and 21.15 suggest otherwise.

The profiles show the old dike losing width and elevation in most places as the seaward face eroded (Fig. 21.15, Lines, 2, 5, 14 and 16). At Line 2 (in the NW cell), the old dike had eroded sufficiently that the core of the dike (or some previous iteration of the dike) was exposed well in front of the dike face (Fig. 21.16a). In locations like Line 5, the dike was no longer high enough to hold back seawater from the Bay at spring high tide. Effectively, by 2012 there were more openings in the old dike than the three constructed as part of the restoration project (e.g., Fig. 21.16b). At locations such as Line 14 (in the SE cell), the old dike was so narrow and low that

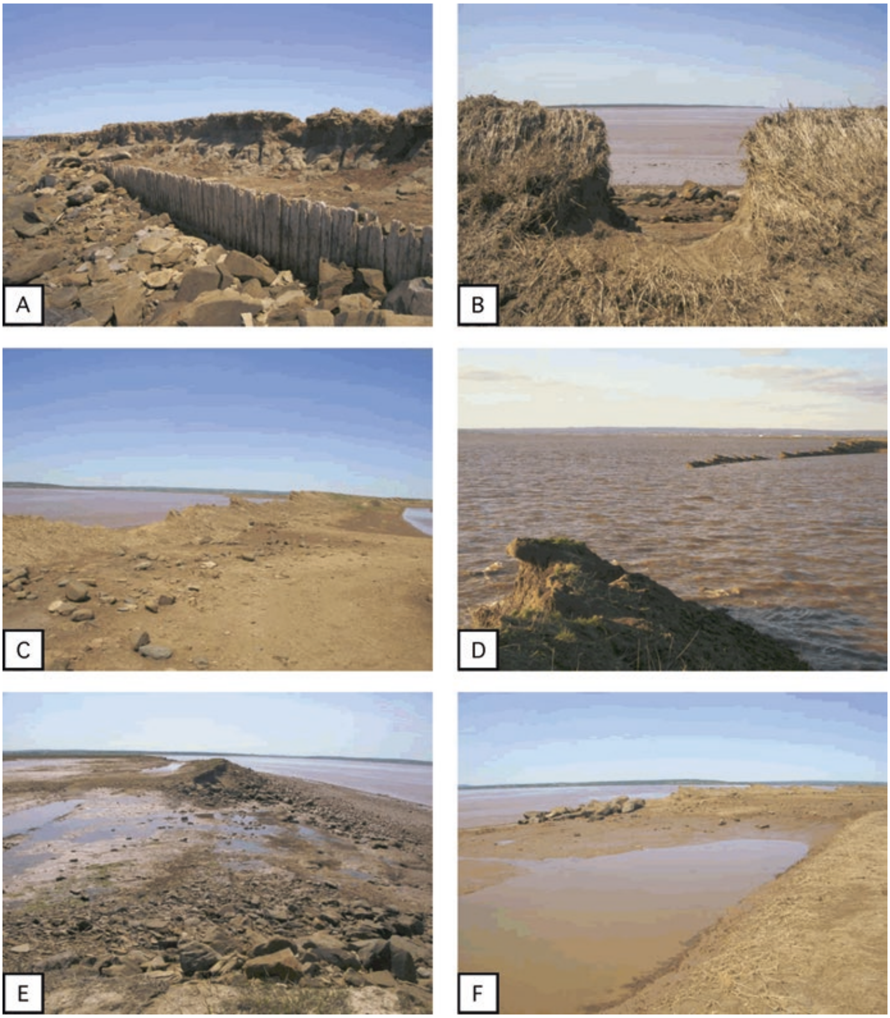

Fig. 21.16 Photographs illustrating erosion of the old dike from October 2012. Panels **a**–**f** are described in the accompanying text (Photographs by Jeff Ollerhead)

it was only going to be some matter of months to years before the effective width of the eastern opening (Line C in Fig. 21.14) doubled or even tripled (Fig. 21.16c, d). By October 2015, the eastern opening as constructed was, in fact, eroded away and essentially unrecognizable (Fig. 21.17; compare with Fig. 21.2c).

Furthermore, the dike eroded out from under and around the armor stone and the stone was left in a pile. Figure 21.16e shows how the armor stone slumped and rolled away and the bank eroded on the southeast side of the middle opening (Line B in Fig. 21.14). Figure 21.16f shows the state of the armor stone on the northwest bank of the eastern opening (Line C in Fig. 21.14) in 2012. There was and is no evidence

Fig. 21.17 Photograph of the eastern opening (in the SE cell) taken in October 2015. It was taken from a location near where the photograph in Fig. 21.2c was taken (Photograph by Jeff Ollerhead)

that water flow through the openings caused the erosion. Erosion of the old dike had been ongoing and persistent for years. The profiles for Line 15 (Fig. 21.15) show where the dike was in 2006, where the opening floor was in 2010, and that the only change to the opening floor from 2010 to 2012 was some headward erosion caused, most likely, by water draining over the sill and back into the Bay on the ebb tide.

The other relevant observation that may be drawn from Fig. 21.15 is that where marsh still existed in front of the old dike, the dike itself was not eroding. The profiles for Line 8 (in the NW cell) show that the margin of the marsh in front of the dike had been (and is) eroding over time (as is the case for the reference marshes that flank the restoration cells), but that the dike profile itself has changed little since 2006. Material that appeared to have been added to the dike face in this location in 2010 during construction of the openings remained in place. The profiles for Line 10 (in the SE cell) illustrate the same situation – where marsh is present in front of the dike, the dike is protected from erosion.

Qualitative observations since 2012 confirm that erosion of the seaward edge of both the old dike and the reference salt marshes that flank the restoration cells is continuing. In many locations, the old dike is no longer high enough to hold back seawater from the Bay at spring high tide. There are now many more openings in the old dike than those constructed as part of the project, including new openings at the

Fig. 21.18 Methods for sampling biological communities. (**a**) Quadrat (0.5 m × 0.5 m) for plants and invertebrates on the emergent marsh. (**b**) A minnow trap and an invertebrate activity trap being deployed to be just below the water surface for one overnight high tide. The minnow trap is 40 cm long with two openings (2.0–2.8 cm in diameter) at either end, and 0.7 cm mesh. It was baited with two saltine crackers. The invertebrate activity trap was made of a 2-L clear pop bottle with the spout cut, the opening of the spout covered with mesh (0.3 cm by 0.4 cm), and the spout taped back inverted into the bottle. (**c**) A minnow trap being processed on location the morning after an evening deployment. (**d**) Lift net for a snapshot measure of aquatic animals in an undisturbed column of water in salt pools in the morning. Net dimensions are 16.75 cm × 25 cm, and mesh size is 180 μm. Water depth was measured at the location to get the dimensions of the water column sampled. (**e**) Fyke net deployed in a marsh creek for three consecutive spring high tides (monitored after every high tide). The main hoop diameter is 0.6 m and wings are 2.4 m long. (**f**) Sampling of birds at predetermined fixed locations in the morning, using point counts of 10 min. A set of point counts at one location consisted of five 1-min counts and one 5-min count, during which all birds observed and heard were recorded. Additionally, key salt pools within the sites were visually surveyed for waterbirds while moving among point count locations. Binoculars were used for bird identification and counting (Photograph A taken in a restoration cell and other photographs taken in a reference marsh, by Laura Boone, Allison Dykstra, Myriam Barbeau, Spencer Virgin, Mike Brylinski and DUC, respectively)

south and north ends of the SE cell. Thus, the original openings have become essentially irrelevant in terms of the hydrodynamics in the restoration cells.

21.4.4 Did the Biological Communities Establish as Expected?

A number of biological communities were regularly sampled. For plants and invertebrates on the emergent marsh (area exposed to air at low tide), three permanent transects were placed in each restoration cell and reference salt marsh perpendicular to the old dike. These were sampled in a random stratified manner by randomly placing three quadrats in each of five equal sections per transect (15 quadrats per transect, 45 quadrats per site; Fig. 21.18a). All plant stems and invertebrates were identified and counted in each quadrat. This sampling was conducted monthly from June to August in 2010–2015 to quantify community structure 1 year pre-breach and 5 years post-breach. These data are presented as averages over months for each transect for a given site and year. Animals in the water column of the salt pools were sampled using minnow traps and invertebrate activity traps (Fig. 21.18b, c), which were deployed in the evening, left over a high tide period and recovered the following morning. This was complemented by a lift net sample (Fig. 21.18d) also done in the morning. Animals were identified, counted and returned to their original location. This sampling was done monthly in the summer, but we present only June of each year post-breach (2011–2015) for the large salt pool (>1000 m^2 in area) located in each site. The large pools were sampled at five locations. The data from each capture method was combined, and transformed to presence-absence for analysis. The benthos of the salt pools was sampled using a benthic grab (81.7 $cm^2 \times 5$ cm deep) once in the summer (between mid-June and mid-July). The sediment grabs (five per large salt pool) were sieved using a 250-μm mesh, and the contents were preserved in 95% ethanol until they could be processed for epi- and infauna under a dissecting microscope (i.e. identified and counted). The composition of each type of community (on the emergent marsh and in salt pools) was analysed using non-parametric multivariate analysis of variance (not presented here) and visualized using non-parametric multidimensional scaling (nMDS) (Clarke 1993).

The plant community is the habitat-forming community of salt marshes (Redfield 1972; Burdick et al. 1997; Bertness 2007) and so is discussed first. The plant recovery pattern in our restorations cells showed a smooth temporal trajectory since the breaching (Fig. 21.19), and is nicely represented by the nMDS graph (Fig. 21.20a). The reference marshes have a distinctive community composition fairly typical of a mature salt marsh in this region (Redfield 1972; Chmura et al. 1997; Bertness 2007). Specifically, there is a large high marsh zone dominated by salt marsh hay *Spartina patens*, and a narrow low marsh zone of saltwater cordgrass *S. alterniflora*. The plant community in the reference marshes is represented by a distinct grouping of data points on the nMDS graph, and remained similar from one year to the next. The pre-breach restoration cells also showed a distinct community reflecting mostly terrestrial plants (e.g., Timothy grass *Phleum pretense*, *Aster* sp., wild morning glory *Calystegia*

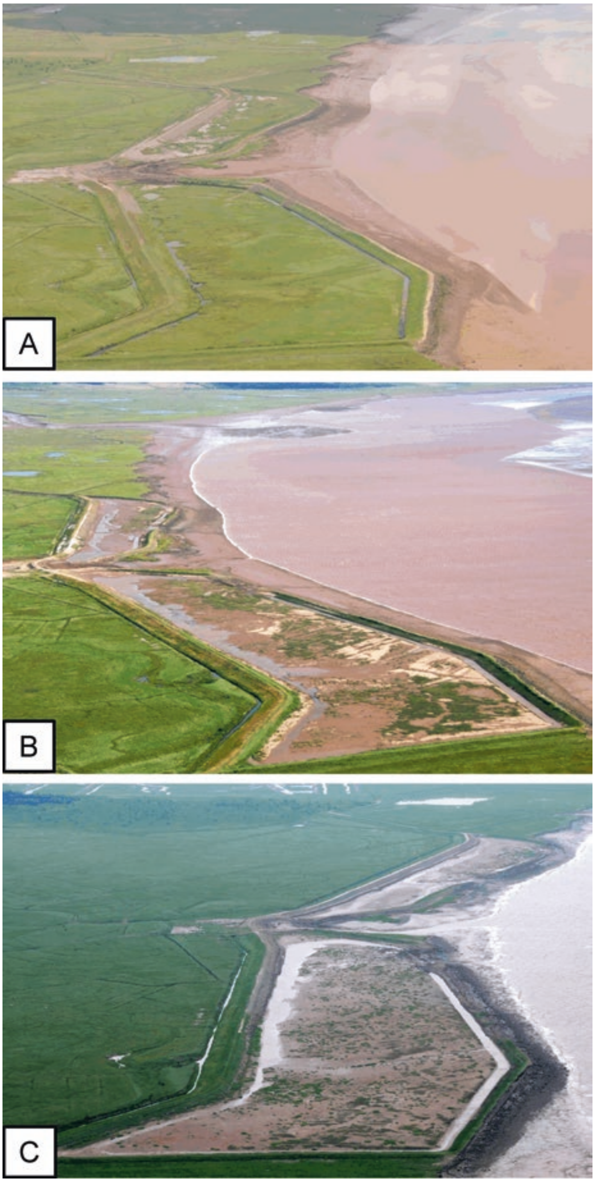

Fig. 21.19 Aerial photographs of the two restoration cells showing the degree of change in the plant community throughout the project. (**a**) 1 August 2010, with the terrestrial plant community before the breaching the old dike; (**b**) 24 August 2011, year 1 after the breaching, with *Spartina pectinata* (freshwater cordgrass) surviving in the higher elevation areas in the cells; and (**c**) 7 July 2014, year 4 after the breaching, with patches of *S. alterniflora* (saltwater cordgrass) being apparent (See also Fig. 21.22b for a closer view of the NW cell 1.5 months later) (Photographs by Eastern Eyes Photography (DUC))

sepium, yarrow *Achillea millefolium*, wild strawberry *Fragaria virginiana*), and so also grouped separately on the nMDS graph. The spring following the breach (2011), the restoration cells looked like mudflats with plants appearing later that summer (Fig. 21.19b). These plants were freshwater cordgrass *Spartina pectinata* (observed in our pre-breach sampling; Fig. 21.21) as well as a new colonizer, sea blite *Suaeda* spp., an annual succulent species. The surviving *S. pectinata* plants were in a stressed

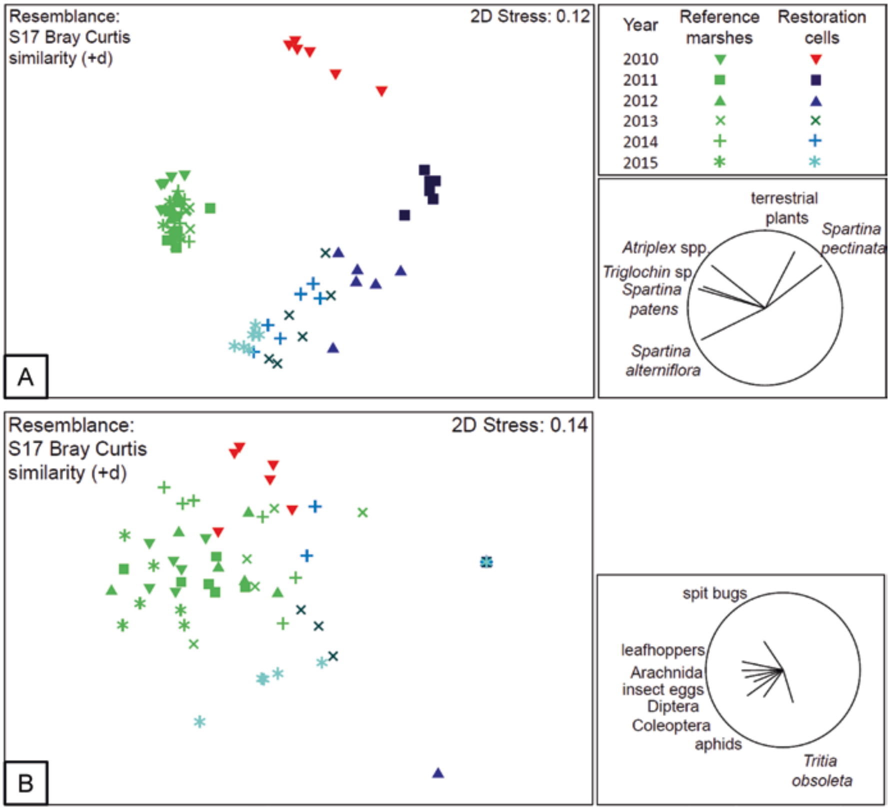

Fig. 21.20 Non-metric multidimensional scaling (nMDS) graphs for community composition for emergent (**a**) plants and (**b**) invertebrates in the two restoration cells and reference salt marshes. Each point on the graph represents a transect (data pooled over 45 quadrats sampled over three summer months) within each site for a given year. Clustering of points indicates similarity in community composition. The reference marshes for all years (2010–2015) are in green, restoration cells before the breach (2010) are in red, and restoration cells post-breach (2011–2014) are in blue. The vectors to the right reflect the correlations between main taxa contributing to the pattern and the nMDS axes. The vector for a given taxon shows the direction of increased density across the nMDS graph. The circle indicates the maximum vector length ($r = 1$ if parallel to one of the nMDS axes); only correlations with $r > 0.7$ (for plants) and 0.3 (for invertebrates) with at least one nMDS axis are shown. Plant densities were fourth root transformed before analysis. Main contributing terrestrial plants were *Calystegia sepium*, *Aster* sp., *Carex* sp., and generally Poaceae; Arachnida were spiders and mites; leafhoppers, spit bugs and aphids are Cicadellidae, Cercopoidea and Aphidoidea, respectively; and *Tritia obsoleta* is the eastern mudsnail

condition, reflected by stunted growth. On the nMDS graph in 2011, the restoration cells showed little variation in community composition (the data points group tightly together), reflecting a mostly muddy environment with low densities of *S. pectinata* and *Suaeda* spp. (Fig. 21.20a). In subsequent years (2012, 2013), small patches of *S. alterniflora* appeared, which grew via rhizomes (Figs. 21.19c, 21.21, and 21.23a).

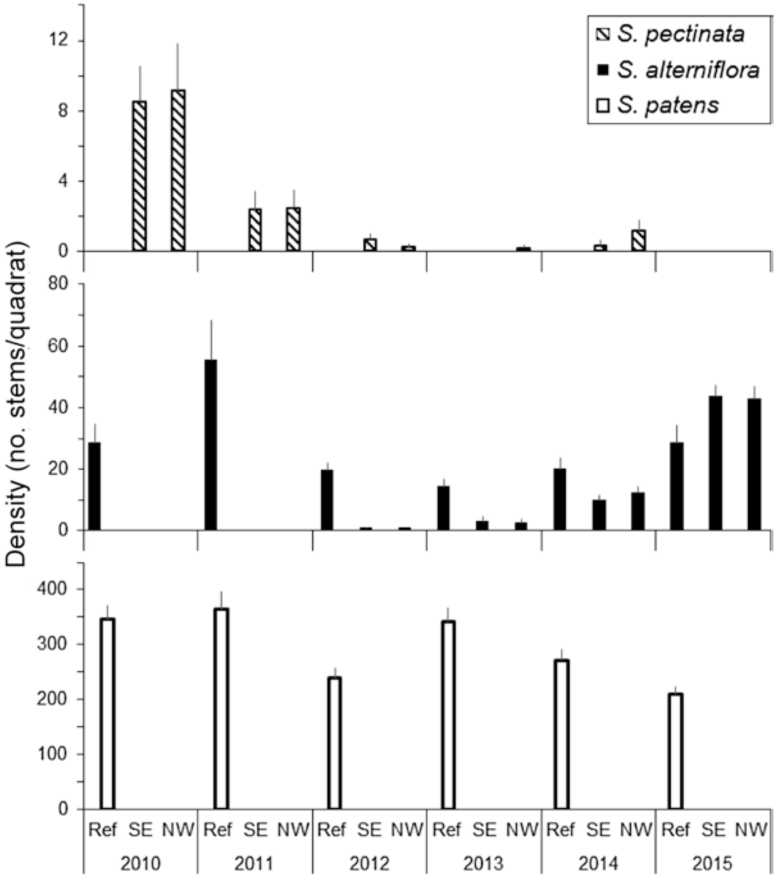

Fig. 21.21 Mean density of the *Spartina* grasses in each of the restoration cells (n = 45 quadrats) and in the two reference marshes combined (n = 90 quadrats), for 1 year bre-breach (2010) and 5 years post-breach (2011–2015). Error bars are 1 standard error. The quadrat was 0.25 m^2

This led to high patchiness in plant species within the restoration cells, which is apparent in the wider scatter of data points for these years on the nMDS graph (Fig. 21.20a). The community trajectory since 2011 has moved towards the reference condition. In 2013, the patches of *S. alterniflora* were in excellent condition with tall and vibrant plants. In the spring of 2014, a large number of *S. alterniflora* seedlings were observed throughout the restoration cells (Fig. 21.23). This sexual reproduction along with continued rhizomal (asexual) spread lead to a near continuous coverage of *S. alterniflora* in the restoration cells in 2015 (particularly in the NW cell; Fig. 21.22c). Densities of *S. pectinata* within the restoration cells also reduced drastically in 2014 and disappeared in 2015 (Fig. 21.21), due to abiotic stress and not interspecific competition. Indeed, *S. pectinata*, considered a brackish species, cannot tolerate salinities above 200 μM NaCl for long periods of time (Warren et al. 1985). These dynamics

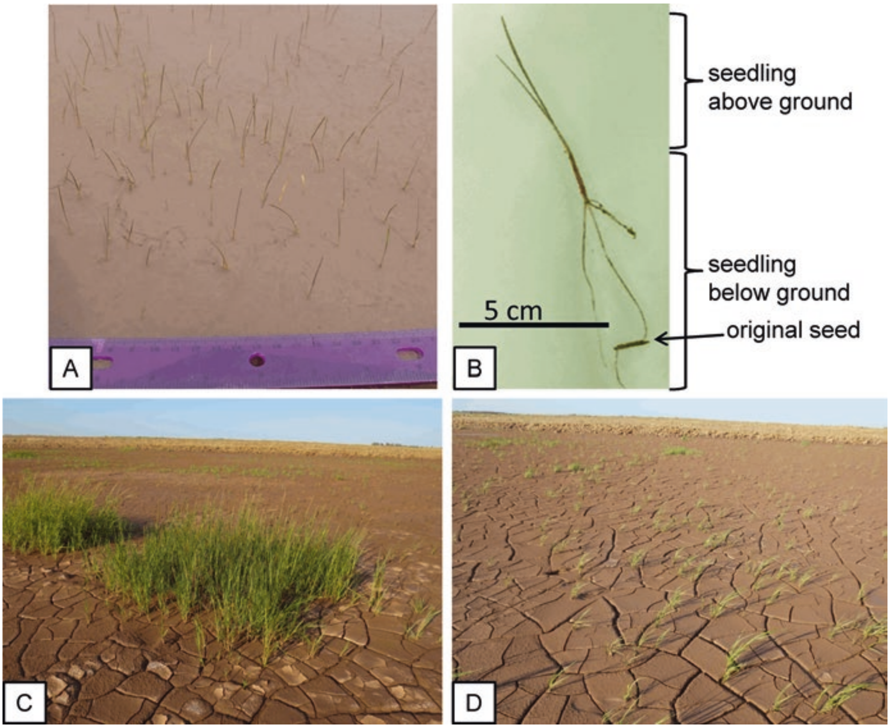

Fig. 21.23 (**a**) *Spartina alterniflora* seedlings in a restoration cell on 9 June 2014. (**b**) *Spartina alterniflora* seedling collected on 18 June 2015. (**c** and **d**) *Spartina alterniflora* patches (with plants that are ≥2 years old) and individual YOY plants (that were seedlings in early summer) on 26 August 2014. The tallest plants in the patches average 85 ± 19 (standard deviation) cm in height, and the YOY plants average 54 ± 12 cm in height (Photographs by Myriam Barbeau, except by Meagan Hicks for **b**)

are apparent on the nMDS graph as the group of data points becomes tight again in 2015 due to the lowered variation (Fig. 21.20a). The spread of points on the nMDS graph also changes between 2014 and 2015 due to the disappearance of the last "terrestrial" plant species (*S. pectinata*) within the restoration cells proper. Note that *S. pectinata* now grows vibrantly along the mid to higher edges of the dike. The lack of overlap between reference and restoration sites in 2015 is due to the continued absence of *S. patens* within the restoration cells (Fig. 21.21). *S. patens* and the various other plants found in the high marsh zone of a mature marsh (e.g., seaside plantain *Plantago maritima*, sea lavender *Limonium carolinianum*, orach *Atriplex* spp., arrowgrass *Triglochin maritima*) currently occur on the seaward face of the new dike. These species are expected to start spreading into the restoration cells proper once the marsh surface has accreted sufficiently. The elevation of the surface of the restoration cells is still about 1 m below the surface of the adjacent reference salt marshes.

The contrasting type of *S. alterniflora* spread between the early years (2012 and 2013) and the most recent years is intriguing. It is not clear how the first *S. alterni-*

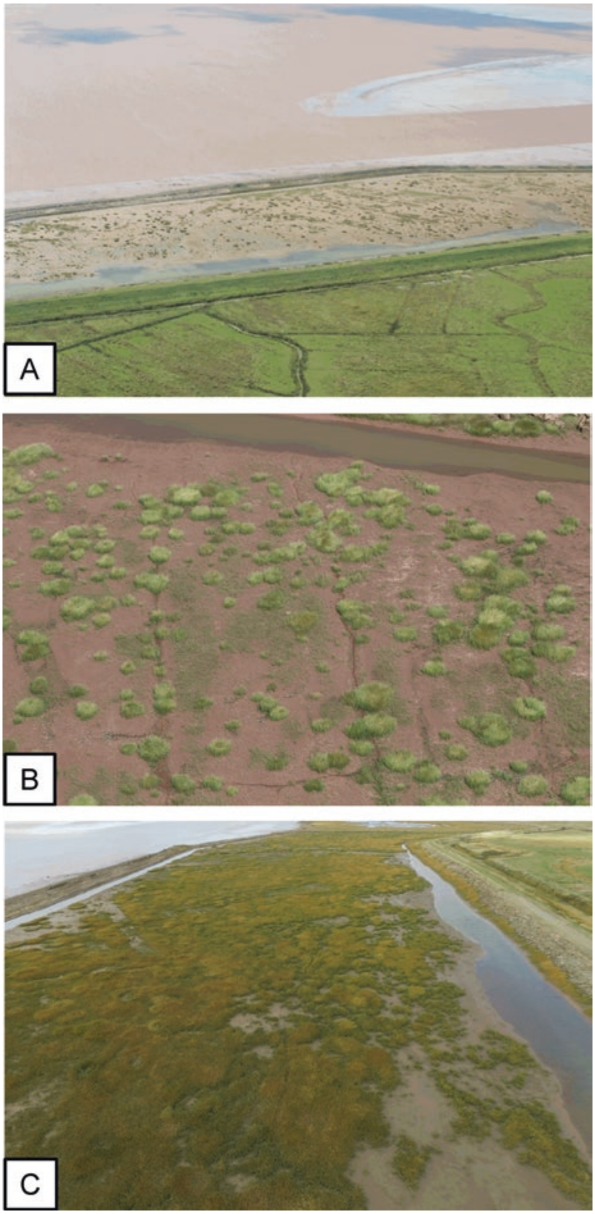

Fig. 21.22 Aerial photographs of the Northwest (NW) restoration cell taken on (**a**) 15 September 2013 showing small *Spartina alterniflora* patches and surviving *Spartina pectinata* plants in the higher elevation areas in the cell; (**b**) 25 August 2014 showing *S. alterniflora* patches and *S. alterniflora* young-of-the-year (YOY) growing in between the patches; and (**c**) 19 October 2015 showing near full coverage of the cell by *S. alterniflora*. In **c**, one can still differentiate the patches (now with plants ≥3 years old) from the 2-year-old and YOY plants (Photographs by Beth MacDonald, Julie Paquet and Sebastian Richard, respectively)

flora plants colonized our restoration cells. Transport of rhizomes by ice is suspected as a contributing mechanism; plant material has been observed in ice blocks (Redfield 1972; van Proosdij et al. 2006b). In any case, once established, patches of *S. alterniflora* would triple or quadruple in area over one summer (ME Hicks, DW Schneider, SDS Virgin and MA Barbeau, unpublished data). Then, the sudden large increase in *S. alterniflora* density observed in 2014 and 2015 was primarily due to the appearance of *S. alterniflora* seedlings (Figs. 21.22 and 21.23). These grew over their first summer and started to produce clones in their second summer (similar to as described in Redfield 1972). Our observed pattern for *S. alterniflora* dynamics was opposite to what is typically described in the literature, summarized by Smith and Warren (2012, p. 193): "Colonization of new areas by *S. alterniflora* initially depends upon seeds, but thereafter vegetative growth becomes important in the expansion of these populations (Metcalf et al. 1986)". As also noted by Smith and Warren (2012) and others, the importance of these two types of reproduction (vegetatively by rhizomes and sexually by seeds) depends on circumstances. Vegetative spread is generally more important in already established or natural salt marshes. Spread by seeds is successful in "disturbed" or unvegetated areas (Metcalfe et al. 1986), although germination of these seeds requires fairly narrow abiotic conditions (Biber and Caldwell 2008; Mooring et al. 1971; Redfield 1972). The extensive appearance of *S. alterniflora* in our restoration cells in 2014 may be due to soil conditions ameliorating sufficiently to enable seeds to geminate. Seeds of *S. alterniflora* can survive floating for up to 25 days in seawater (Elsey-Quirk et al. 2009), which would allow their dispersal into our restoration cells from other salt marshes surrounding the Cumberland Basin and other regions of the upper Bay of Fundy. We think, however, that the large cohorts of seedlings are offspring of the plants that had colonized in 2012 and 2013 and grown in our sites. Tall vibrant plants full of seeds were observed in the autumn of 2013 (which is before the seedlings appeared en masse). Genetic analyses of the *S. alterniflora* plants are planned for the future (specimens are already collected and preserved) to determine parentage. In any case, with the establishment of seedlings, a very rapid increase in plant cover throughout the restoration cells was observed.

Most other biological communities that we monitored are lagging behind the plant community, and this has been observed in other salt marsh restoration projects (Roman and Burdick 2012). The invertebrates on the emergent surface of the restoration cells are just starting (after 5 years) to approach the assemblage in the reference marshes (consisting of herbivorous insects and arachnids). On the relevant nMDS graph (Fig. 21.20b), the data points for the restoration cells after the breaching initially showed a very tight and distinct cluster of extremely low densities as well as the occasional presence of intertidal mudsnails. In 2013, 2014 and particularly 2015, the data points approached those of the reference marshes. The trajectory is somewhat reminiscent of the plant dynamics (though not as smooth).

The salt pools have not yet fully stabilized since sedimentation is still quite high (Fig. 21.13, Table 21.2) and plants have just recently extensively covered the restoration cells. Small salt pools (5–53 m^2 in area) are beginning to form in the restoration cells (Dykstra 2015). The minnow and aquatic invertebrate community in the large salt pools is still different (Fig. 21.24a), with fewer or no sticklebacks (*Gasterosteus aculeatus* and *Pungitius pungitus*), mummichog (*Fundulus heteroclitus*), and water boatmen (*Corixa* spp.) in the restoration cells compared to the reference marshes. This community in the restoration cells is more variable and sometimes reflective of oceanic animals appearing there (such as ctenophores).

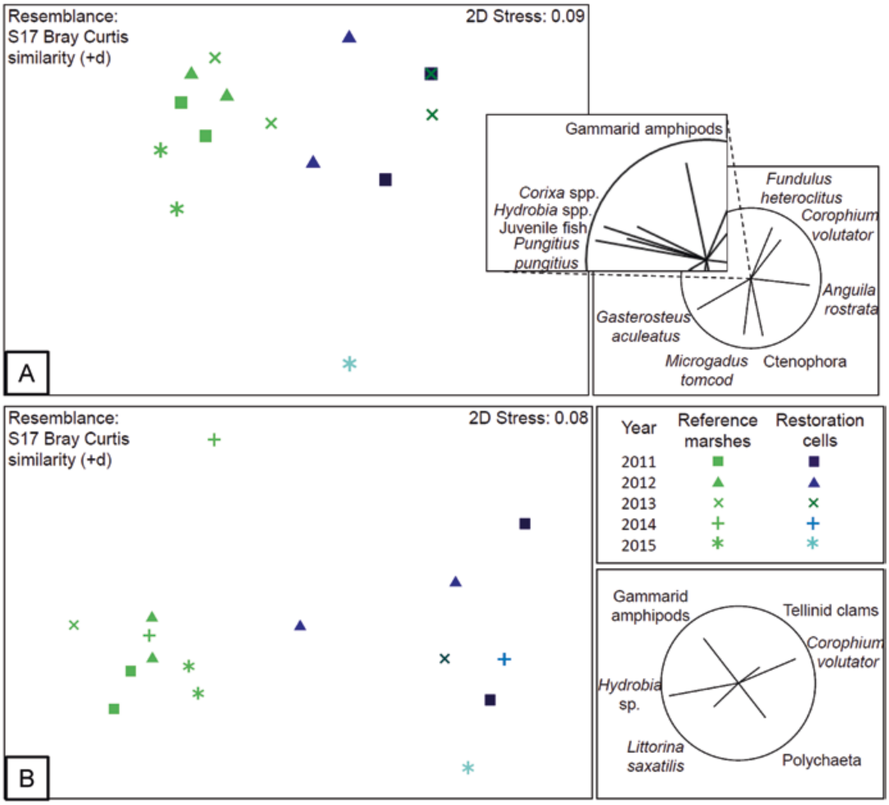

Fig. 21.24 Non-metric multidimensional scaling (nMDS) graphs for community composition for salt pool animals in the (**a**) water column (sampled via minnow traps, invertebrate activity traps and lift net) and (**b**) on/in the benthos (sampled via benthic grabs) in the two restoration cells and reference salt marshes. Each point of the graph represents a large pool (averaged over 5 sampling locations) sampled once in mid-June to mid-July within each site for a given year. See Fig. 21.20 caption for more details on this type of graph. For the taxa vectors, only correlations with $r > 0.6$ (for water column) and 0.3 (for benthos) with at least one nMDS axis are shown. Data for water column animals were transformed to presence/absence, and for benthic grabs were square-root transformed prior to analysis. Minnows included mummichog, 3- and 9-spine sticklebacks, tomcod, and eel; aquatic and benthic invertebrates included water boatman, gammarid and corophiid amphipods, *Hydrobia* snails, periwinkles, comb jellies, tellinid clams and polychaete worms

Indeed, the water characteristics of the salt pools in the restoration cells reflect the lower surrounding elevation and more frequent flushing by the ocean, with salinities, pH and dissolved oxygen being closer to oceanic values as well as less temporally variable than salt pools in the reference marshes (Table 21.3). Also, the pool vegetation (green algae and ditch grass *Ruppia maritima*) is still very sparse in the restoration cells, thus providing little microhabitat compared to the salt pools in the restoration marshes (Dykstra 2015; see also Fig. 21.18b–d). The bottom of the salt pools in the restoration cells has less compact sediments (as measured by sediment

Table 21.3 Mean ± standard deviation of water variables measured in the large salt pools (>1000 m^2 in area) in the Northwest restoration cell and both reference salt marshes in summer (June–August) of 2015 (Dykstra 2015). Variables were measured in the evening just prior to deploying traps (Fig. 21.18b) and before an overnight high tide

Water variable (units)	NW restoration cell (n = 15)	Reference sites (n = 28–30)
Salinity (ppt)	25.4 ± 0.7	23.0 ± 6.3
pH	7.7 ± 0.1	9.4 ± 0.6
Dissolved oxygen (mg/L)	9.9 ± 0.7	17.3 ± 6.0

Measurements were taken in previous years (when there was a large salt pool in both restoration cells) and showed similar patterns

penetrability; Dykstra 2015) and a community similar to that of adjacent mudflats which is dominated by the amphipod *Corophium volutator* (Fig. 21.24b; Gerwing et al. 2015). The dominant animal on the bottom of the salt pools in the reference marsh is the small *Hydrobia* snail, and has only very occasionally appeared in the restoration cells to date.

Two other communities, the nekton along the creeks and the birds, were sampled in the early years (2010–2012) by Dr. Mike Brylinski from Acadia University and by DUC, respectively. The large and mobile nekton entering the restoration cells and the reference marshes and sampled during three consecutive spring tides in each of spring, mid-summer and late summer (Fig. 21.18e), was similar even 1 and 2 years post breaching. It consisted mainly of tomcod *Microgadus tomcod*, American eel *Anguilla rostrate* and mummichog, and moderate abundances of three-spine stickleback, Atlantic silverside *Menidia menidia* and nine-spine stickleback. Indeed, Burdick et al. (1997) and Roman et al. (2002) reported that fish use of restoration sites can rebound quickly (after one season). However this does not imply that a fish community has been reestablished (the latter takes longer; Burdick et al. 1997; Raposa and Talley 2012).

The birds were sampled along 14 locations (point counts) in the restoration cells and reference marshes, weekly or biweekly from late spring to fall (2010–2012), on mornings with little wind (<5 Beaufort scale) and good visibility (Fig. 21.18f). Over 40 different bird species were recorded during the surveys, including songbirds (main ones: Nelson's sparrow *Ammodramus nelsoni*, savannah sparrow *Passerculus sandwichensis*, song sparrow *Melospiza melodia*, swamp sparrow *Melospiza georgiana*, common yellowthroat *Geothlypis trichas*, red-winged blackbird *Agelaius phoeniceus*), shorebirds and wading birds (main ones: Willet *Tringa semipalmata*, great blue heron *Ardea herodias*, killdeer *Charadrius vociferous*, spotted sandpiper *Actitis macularius*, semipalmated sandpiper *Calidris pusilla*) and water fowl (main ones: Canada goose *Branta canadensis*, American black duck *Anas rubripes*, mallard *Anas platyrhynchos*). More shorebirds have been observed in the restoration cells, and songbirds have not yet been observed to breed in them, which is not surprising since the plants have just started to cover the area. These patterns are similar to those reported previously (Warren et al. 2002; Shriver and Greenberg 2012). Our plan is to restart monitoring the bird community once the plant community is more restored.

In our biological studies, the two restoration cells have been treated as two replicates, but differences between them are informative. The SE cell started with a lower overall elevation than the NW cell, and so has been experiencing more sedimentation (Fig. 21.13). In the areas with the greatest amount of sediment deposition in the SE cell (in its northwest section and its southeast tip), overall plant coverage was still much less in 2015 than elsewhere in that cell or than throughout the NW cell. Note that a significant difference in *S. alterniflora* density was not detected between restoration cells in 2015 (Fig. 21.21), partly because a substantial portion of the SE cell has nearly full plant cover and partly because of where our permanent transects are located (which did not include the northwest section of that cell). In the first 2 years, the northwest section of the SE cell was not accessible by walking, except when frozen, because the mud was very soft and fluid. As the mud solidified, *S. alterniflora* seedlings became established in this section in 2014 and 2015, and small patches were observed in 2015 and 2016 that resembled the pattern of patches seen in Fig. 21.22a (i.e., observed throughout the NW cell in 2013). The fact that overall plant cover is less in the SE cell than in the NW cell is of concern, because the SE cell is narrower (75–100 m wide) than the NW cell (125–200 m wide). It is not clear whether a functioning salt marsh will develop in the SE cell before the old dike degrades so much that it no longer offers protection from waves and sediment resuspension (e.g., French et al. 2000). It may also be too narrow to provide sufficient accommodation space for the salt marsh to adjust to the local rate of relative sea level rise. Simply put, healthy vegetation cover is crucial to a salt marsh, both in terms of ecosystem services as well as in trapping sediment and baffling wave energy, and the low amount of vegetation in sections of the SE cell reduces the prospect of its successful restoration.

Another difference between the restoration cells is the almost complete disappearance of the large salt pool in the SE cell, starting in 2014, partly because of sedimentation but also because of the extra armoring of the new dike placed in that cell in the autumn of 2013. The added rock and fill material placed adjacent to the new dike locally displaced the marsh surface upward, causing the large salt pool to partially drain. Other than plant coverage and the recent disappearance of the large salt pool, the various biological communities have had similar composition and dynamics in both cells, as represented by the clustering of points from each year in the nMDS graphs (Figs. 21.20 and 21.24).

In summary, the biological community dynamics that we observed to date are within the range reported for other salt marsh restoration projects in New England and Maritime Canada (Burdick et al. 1997; Roman and Burdick 2012; Warren et al. 2002). Of the intentional restoration projects undertaken to date in the Bay of Fundy (Bowron et al. 2012), those at Walton River (Neatt et al. 2013; van Proosdij et al. 2010) and Cogmagun River (Bowron et al. 2015), in Minas Basin, Nova Scotia, are most similar to our project in that they involved breaching a dike and salt marsh vegetation was essentially absent from the sites to be restored. Although both projects are located in a macrotidal environment, they differed from ours in being relatively recently diked (1989 and 1991, respectively) to create a freshwater impoundment, and in being located 2–3 km upriver and so are protected from waves.

The establishment of *S. alterniflora* was faster in those projects than in ours, colonizing in the first year post breach (mostly via seeds) and having spread throughout the site by year 3 post (compared to 2 and 5 years post, respectively, for our project). The presence of seedlings early on in those two projects is not surprising since the water entering the restoration sites would have been moving over salt marsh beforehand and so be carrying seeds with it (compared to our sites which are open to the Bay). As well, conditions may have been immediately suitable for seed germination in those two projects, with the marsh elevation not being too low (the sites were diked for less than two decades) and the soil having appropriate edaphic conditions (since the sites were previously freshwater impoundments). Our project is thus unique given how exposed our sites are to wave energy, in addition to the high tidal amplitude; the time-scales of biological community establishment observed seem reasonable for our latitude and hydrodynamic conditions.

21.5 Implications for Salt Marsh Restoration in Macrotidal and Ice-Influenced Environments

21.5.1 Lessons Learned

Several lessons have been learned through the design, implementation and monitoring of this project that can be utilized for future salt marsh restorations in the region.

1. Planning for future (and not current) environmental conditions is paramount. As the local rate of relative sea level rise is projected to increase in coming decades, any dike re-alignment must provide sufficient accommodation space (i.e., the distance between the old and new dikes) to give newly created salt marshes enough time to accrete and possibly migrate landward in step with changes in sea level. Creating wide salt marshes (>200 m) will also allow people living in adjacent settlements to take full advantage of all the ecosystem services that salt marshes provide, including buffering against erosion.
2. Partnerships and a proactive approach are very important. The partnerships developed as part of this project have allowed for the success of the project and will prove invaluable for the implementation of future projects. By having a network of contacts in place, future projects will benefit from a more rapid mobilization of resources, which is important. The current restoration project would have benefitted from more timely planning and construction, as the rapid deterioration of the old dike has resulted in increased wave action on the developing marshes. We suspect that this may have had a deleterious effect on the developing plant communities.
3. Monitoring with an investigative or research focus, rather than following a rigidly prescribed protocol, allowed us to observe and quantify several unexpected phenomena (e.g., the importance of *S. alterniflora* seedling to plant spread).

These phenomena were detected early and our monitoring program was modified as a result. Thus, having a monthly monitoring program (during the summers) of the biotic communities currently in place was key to some of our discoveries.

4. The hydrodynamic model used for this project allowed us to implement a successful design that met the required criteria. The openings functioned as desired and sediment was deposited to the greatest degree where expected. Our estimates of how much sediment would be deposited, based on the empirical work of van Proosdij et al. (2000, 2006a, b), were demonstrated to be reasonable. We are confident that the same approaches could be used to design another restoration project in a similar environment.

21.5.2 Thoughts Related to Climate Change Adaptation Plans (for Communities and Municipalities)

Along the coastlines of New Brunswick and Nova Scotia, there are 364 km of dikes which protect more than 32,000 ha of agricultural land (van Proosdij and Page 2012). These dikes also protect other important infrastructure (including roads, rail lines, businesses and homes) – the loss of which would have major implications for public health and safety as well as consequential economic ramifications. Regional studies are being conducted to evaluate areas most susceptible to the greatest flooding risk (Daigle 2014; van Proosdij and Page 2012; Webster et al. 2011, 2012) as well as economic risks for these areas (Wilson et al. 2012). An added concern is increasing costs associated with maintaining dikes and their associated infrastructure that protect these coastal resources, which will continue to increase with sea level rise (Singh et al. 2007; van Proosdij and Page 2012). Thus, a reasonable response (adaptation) to climate change and sea level rise in our area is to move dikes inland and restore the interdike areas to salt marsh to help minimize erosion and flooding (French 2006, Lindham and Nicholls 2010) and therefore reduce the lifecycle costs of maintaining dikes.

The success of our project will hopefully inform community-based adaptation planning that is already underway in our area. Communities most vulnerable to salt water flooding are already building partnerships with non-government organizations (like DUC), various departments of municipal, provincial and federal governments, and universities (Marlin 2015; Marlin et al. 2016). This community-level approach has resulted in reports, mapping tools (e.g., http://arcgis.mta.ca/toolkit/), and public information sessions (Marlin 2013; Lieske et al. 2014; Lieske 2015). In some cases, salt marsh restoration has been identified and agreed upon, based on discussions with many stakeholders, as one of the tools that will be used to adapt to climate change and sea level rise-related flooding risk (Marlin 2013). Although resistance to salt marsh restoration remains, because of desires by some people to keep land for its current use (Marlin et al. 2007), changes in land use are likely inevitable. Given that any potential project should go through a cost/benefit review involving key stakeholders to develop an agreed-upon approach, successful projects in our area (e.g., Bowron et al. 2012; this study) demonstrate that salt marsh restoration is a viable option that deserves serious consideration.

21.6 Conclusions

The project is still in a race against time. The old dike continues to erode at a relatively rapid pace. To protect the new dike, a healthy salt marsh needs to develop in the restoration cells. This seems to be happening in the NW cell, where percent cover by salt marsh vegetation is extensive and the surface is continuing to accrete. In the SE cell, the situation is less certain; plant spread has been slower and overall plant percent cover is less, most likely because of a lower initial surface elevation and high rate of sediment deposition resulting in a soft, unconsolidated surface. This said, sediment continues to be deposited; so, it is a matter of waiting to see if salt marsh development continues in the SE cell in the face of increased exposure to waves and ice now that the old dike is largely eroded and breached in numerous places. Salt marsh recovery is not inevitable, and there are cases in other geographic locations of sites becoming mudflat (French et al. 2000).

There is not much that can be done now but continue to monitor the project and learn as many lessons as possible. We will continue to employ an investigative or research approach to monitoring, to ensure that we do not miss important changes or events. The failure of the old dike was inevitable and now it is a matter of waiting to see what type of salt marsh develops in front of the new dike on the foundation that has been deposited over the past 5 years and learning from the ecosystem differences that are appearing between the SE and NW cells.

This said, monitoring the project has improved our knowledge of the ecology of Maritime Canadian salt marshes and added to the growing literature on salt marsh restoration worldwide. Restorations in our region have historically been studied passively on abandoned agricultural sites (e.g., Byer and Chmura 2007). More recently, the focus has shifted to monitoring deliberate, managed projects (Bowron et al. 2009, 2012; van Proosdji et al. 2010). The Bay of Fundy provides an excellent area for future salt marsh restoration projects due to its high suspended sediment content, extensive tidal flushing, and historical land use patterns. Worldwide historical salt marsh habitat loss has been estimated at 65% (Bromberg Gedan and Silliman 2009). Barring expensive interventions, it is inevitable that rising seas will flood much that was previously reclaimed. Salt marsh restoration is a means to make the most of the reality that confronts us by restoring the natural aesthetic and returning valuable ecosystem services to our coasts while protecting human infrastructure at the same time.

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Chapter 22
Macrobenthic Assemblage in the Rupsha-Pasur River System of the Sundarbans Ecosystem (Bangladesh) for the Sustainable Management of Coastal Wetlands

Salma Begum

Abstract Information on benthic fauna in the coastal region of Bangladesh is scarce. Owing to its geographical location, land characteristics, and multiplicity of rivers, the south western region of Bangladesh contains the world's largest mangrove forest, the Sundarbans, containing rich floral and faunal biodiversity. Non-forestry product of the Sundarbans e.g. aquatic biotic community becomes more plentiful with presence of benthic invertebrates in the system since macrobenthos perform various ecological roles in mangrove functioning. To obtain insight into macrobenthos assemblage patterns along the Rupsha-Pasur river system, along the Sundarbans, Mongla and Koromjal were the sites of investigation. The macrobenthos assemblage patterns with environmental parameter from 10 m depth during low tide were observed from those sites. Among the major groups found in both sites, Annelida (including echiura, oligochaeta and polychaeta) represented highest diversity with 22 species, followed by 21 species of arthropoda (Crustacea-decapoda), 20 species of Mollusk (gastropoda), 9 species of arthropoda-bivalvia, few crustacea-isopoda, crustaceae-tanaidaceae and nemartina taxa. The relative abundance was not significantly different in Mongla and Koromjal. A significant seasonal effect on the relative abundance was found by performing the Kolmogorov-Smirnov test, at $p < 0.05$. The monsoon gave rise to more diverse macrobenthic species than the dry season. To study the relations between the macrobenthic species, relative abundances, and the environmental parameters, redundancy multivariate analysis was applied. The overall results indicate relative species abundance is influenced by the combined effects of environmental parameters and biological parameters. Further research is necessary to understand the species and system dynamics. The research of non-forestry product of the Sundarbans is necessary for the future management of the Sundarbans in a sustainale manner.

Keywords Macrobenthos • Assemblage • Sundarbans • Ecosystem • Rupsha-Pasur river

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22.1 Introduction

22.1.1 Benthos

Compared to the higher latitudes, benthic studies are very scarce in the tropics. The exploration of tropical continental shelves first began off of the western edge of Africa in the 1950s (Buchanan 1957), but apparently there was not that much increase in such studies until the mid-1970s (Ansari 1977; Parulekar et al. 1982). A few studies on the bottom fauna were conducted in 1950s in India (Seshappa 1953; Kurian 1953) off of the Madras, Malabar and Travancore coasts, and afterwards more benthic studies were done in India (Radhakrishna and Ganapati 1969; Damodaran 1973; Ansari 1977; Parulekar et al. 1982; Raut et al. 2005). Some innovative research on invertebrates in the Sundarbans ecosystem e.g. the giant honey bee, mud crab, and various shrimp species, but detailed work of benthic invertebrates is scarce. A total of 301 species of mollusks and over 50 species of commercially important crustaceans have been recorded so far in the coastal zone of the Bay of Bengal, though details regarding their diversity pattern and their underlying mechanisms are yet to be investigated (Quader 1981; Maruf 2004). Albeit with little or no concerted effort to unravel the question of faunal associations at a community level of organization, detailed benthic study in the coastal tropical mangroves especially in Bangladesh, on microbenthic assemblages are yet to be done.

22.1.2 Mangroves

Mangrove ecosystems comprise a taxonomically diverse group of tropical and subtropical trees and shrubs, are indigenous to their habitat, and are a major contributor to the intertidal coastal and marine environment. Mangroves grow between the level of high water of spring tides and a level close to but above mean sea level that is rich in organic matter and nutrients, which support a large biomass of flora and fauna (Macnae 1968; Chapman 1976; Prabha 1994). It acts as a buffer in protecting inhabitants of numerous human settlements against cyclones, rising sea tides and other hazardous weather events. In addition to its ecological functions, it supports livelihoods of about eight million people living in the vicinity of the forest directly and indirectly (Uddin et al. 2013). Some mangrove trees produce excellent quality honey and wax, grasses and Hental (*Phoenix paludosa*) leaves also collected by collectors. The non-forestry products are different varieties of fish, prawns, and mollusks. In Dubla island, fishing camp is operates from November to February each year, catch fish from the Bay of Bengal and dry fish for export.

The mangrove type of vegetation is characteristically present along river deltas, estuaries, and sea coasts and has the ability to grow where other trees cannot grow.

They thus make a significant contribution in benefitting the environment. Their coverage in the coastal system provides many diverse species of fish, crustacean, birds and mammals shelter. The important components of this ecosystem are water, soil, sediment and the biota – an admixture of *euryhaline* fauna and flora (Chapman 1976). The mangrove ecosystem foundation provides a complex food chain and a detrital food cycle (Odum and Mclvor 1969). Mangrove leaves and wood are made mainly of lignocellulose components that are degradable by microorganisms (Alongi et al. 1989). When mangrove leaves drop into tidal waters they are colonized within a few hours by marine bacteria and fungi and residing in the sediment, take 2–6 months to decompose, or more for degradation (Newell et al. 1984). The heterotrophic bacteria in mangroves consist of microorganisms with cellulolytic, pectinolytic, amylolytic, and proteolytic activity (Matondkar et al. 1981). The fungi that decompose mangroves have pectinase, protease, and amylase activities and the capacity to degrade lignocellulosic compounds (Findlay et al. 1986). In this way they convert difficult-to-digest carbon compounds into nitrogen rich (organic) detritus material in the active process of decomposition. Thus, these energy rich large active microbial populations are both attached and living free (Odum and Heald 1975a). Other organisms, in addition to bacteria and fungi, may also colonize vegetative material and contribute to detritus formation (D'Croz et al. 1989). Therefore the decomposing process of mangrove leaves reveals a complex community composed of fungi, bacteria, protozoa, and microalgae (Odum and Heald 1975b). The resulting pieces covered with microorganisms become food for about one third of the species (about 120 sp) living in the mangrove ecosystem. The detritus feeders included crustaceans, mollusks, insect larvae, nematodes, polychaeta, and some fish species (Odum and Heald 1975b). Species that consume detritus have commercial importance such as shrimp, fish, mollusks, oysters, and mussels. With other small benthic detritivores, these organisms are an important food source for economically important fish, especially juvenile snappers, snook, and croakers or drums. It is reported that 85% of the total fish catch is mangrove-dependent (e.g. Flores-Verdugo et al. 1987). Although many mangrove-associated organisms do not consume detritus themselves, they benefit indirectly by feeding on detritus feeders. Since the detritus eaters are food for carnivores including fish and crabs, subsequently birds and large fish follow the food chain, concluding with human beings. Therefore the mangrove ecosystem is considered as a unique and irreplaceable ecosystem owing to the detritus food cycle enabled by the benthic organisms (Alongi 2002). Hence, the detritus is the base of an extensive food web and serves as a nutrient source in which organisms of commercial importance participate. The formation of mangrove ecosystems along the Bay of Bengal is controlled by marine and terrestrial factors such as local climate, salinity and other edaphic characteristics. These, together with the distance from the sea, the frequency and duration of inundation, and tidal dynamics, govern to a great extent the local distribution of benthic species and their succession (FAO 1999).

22.1.3 Distribution of Bangladesh Mangroves

Owing to its geographical location, land characteristics, and multiplicity of rivers, the south western region of Bangladesh comprises the world's largest mangrove Sundarbans, with its rich floral and faunal biodiversity. The Sundarbans is home to an estimated about 505 species of wildlife, including 355 species of birds, 49 species of mammals, 87 species of reptiles, 14 species of amphibians as well as emblematic species such as the Bengal Tiger (Hussain 2014). Additionally, there are about 234 species of flora and more than 300 species fish, which includes 237 species of finfish, 38 species of shellfish and 34 species of mollusks also enrich the ecosystem. The Sundarbans also supports 8 species of amphibians, 53 species of reptiles, 315 species of birds, 49 species of mammals (FAO 1982; Gopal and Chauhan 2006). Flagship species such as the Royal Bengal tiger, saltwater crocodile, spotted deer and freshwater dolphin of this system has attracted scientists for ecological research and tourists for aesthetic values. Because of their diversity and integrity, the Sundarbans was declared as a Natural World Heritage site in 1997 by UNESCO and a Ramsar site of international importance in 1992. The Sundarbans lies between latitudes 21′39 and 22′30N and longitudes 89′01 and 89′52E in the southern most parts of the administrative districts of Bagerhat, Khulna and Satkhira of Bangladesh. It stretches from the Baleswar river in the east to Harinbhanga and Raimangal rivers along the Indian border in the west. At present, the total area of the entire Sundarbans is about one million ha, about 60% of which belongs to Bangladesh and 40% to India. The Raimangal river separates the two countries. The initial area nominated for west wildlife sanctuary of the Sundarbans was, a 71,500 ha area adjacent to the border of the Indian Sundarbans. In response to the recommendation of bureau of Bangladesh, the Bangladesh government agreed to also include the Sundarbans south (37,000 ha) and Sundarbans east (31,000 ha) sanctuaries. The sanctuaries are intersected by a complex network of tidal waterways, mud flats and small islands of salt tolerant mangrove forests. The area is flooded with brackish water during high tides, which mix with freshwater from inland rivers. Thus, it is a region of transition between the freshwater of the rivers originating from the Ganges and the saline water of the Bay of Bengal. Due to natural processes, the Sundarbans estuary is still in the process of formation in some areas. The Sundarbans was raised by the deposition of sediments as a result of soil erosion in the Himalayas. The process of sedimentation was accelerated by the tidal system of the sea. The substratum consists mainly of quaternary era sediments, sand and silt mixed with marine salt deposits and clay. Therefore, as per the natural process, silt carried from the catchment of the rivers draining into the estuary, get deposited and islands are built up (ODA 1985; IWM 2013). The neo-tectonic movements during the tenth to twelfth century AD have caused the Bengal basin to tilt eastward. Evidence from Borehole studies were indicated that while the western side of the Sundarbans is relatively stable, the south-eastern corner is an active sedimentary area that is subsiding (Gopal and Chauhan 2006). In the Bangladesh portion of the Sundarbans, the formation of the pro-gradational sequence began prior to 4000 years BC and ended about 1800 calibrated years BC (Allison 1998; Allison et al. 2003). Thus the Sundarbans transforms into a series of low elevated isolated land masses (e.g. islands) under strong tidal influences. Geological and tectonic activities, along with past and present drainage patterns, have defined the present geomorphology of the Sundarbans. In general, four morphometric categories can be found: (i) river, (ii) mudflats (iii) ridges and (iv) back swamp basins (EGIS 2001.). Vegetation along the gradient landform reveals that species assemblage patterns change along with the shifting pattern of landform and elevation from sea level. The climate in the Sundarbans has a marked seasonal difference between the monsoon and dry season. The wet monsoon season runs from May to September and brings heavy rains under the influence of the south-west trade winds, with 75% of the annual precipitation occurring during this period (JOEC 2002). The dry season is from October to April and brings infrequent rainfall, under the influence of the north and northeast winds. The temperature in the dry season can rise above 35 °C during the months of April and May. Due to the location, the Sundarbans is the immediate pathway of cyclonic storms generated over the sea and those could be accompanied by heavy tropical storms, floods, and tidal bores, as a result, many disasters strike fast and unexpectedly. Nearly 80% of disasters occur during the dry season. The Sundarban's highly productive mangrove ecosystem acts as a natural fish nursery, a natural shield against the fury of cyclonic storm, prevents erosion due to tidal action, as well as regulates atmospheric pollution. The population surrounding of the Sundarbans is very dense, about eight million of people living in close proximity to (within 10 km) significant mangrove areas (>100 ha) (Uddin et al. 2013).

22.1.4 Benthic Diversity in the Mangrove Sundarbans, Bangladesh

The intertidal ecosystem, the Sundarbans freshwater comes from the upstream Ganges, runoff from the catchments of various distributaries, and marine water available from the tidal actions of the sea. Thus, the Sundarbans wetland is a transitional and unique habitat for aquatic organisms. Rivers, estuaries and regular flooded lands are the main habitats for fish, crustacea and other aquatic organisms in this ecosystem. The mangrove ecosystem serves as shelter, feeding, and breeding zones for crustaceans, mollusks, fish of commercial importance, and resident and migratory birds (FAO 1999). Aquatic energy flow and the food chain are channeled from phytoplankton through pelagic and benthic components of the food web. Since the relative importance of links between specific trophic levels varies considerably from place to place in the coastal areas (Allen 1971), efforts have been taken to understand the coastal marine distribution pattern for major ecosystem components of subarctic, temperate and some tropical regions (e.g. Thorson 1971). Nevertheless, regional and local data is important in order to get a detailed picture of the distribution as well as productivity of such systems (e.g. Begum et al. 2010). Quantitative relationships between the abundance of coastal marine fish and the availability of their food sources are important (e.g. Begum et al. 2009). So far invertebrate biodiversity in Asian rivers and coastal areas has not been studied thoroughly (FAO 1999). The general composition of the benthos of large Asian rivers appears similar to that of related habitats around the world but aquatic resource and tropical ecology contains relatively little information about Asian coasts and seas. There is little published information on wildlife (Hendrichs 1975), for example, vertebrate and invertebrate and other fauna surveys include those of Gittins (1981) and Khan (1986) for rhesus macaque, spotted deer, and birds (Sarker 1985a, b, Sarker and Sarker 1986). The gap in our knowledge is due in part (Lowe-McConnell 1987) to a lack of primary research on the distribution pattern of invertebrates, biology and morphology but is also a reflection of a scattered, highly fragmented literature, some of which is either inaccessible or locally published. Therefore time and effort should be given to studies on resource management. In this way, data accumulation and information gaps might be filled. In Bangladesh about eight lac people live around the coastal and marine zones which have the economic as well as sociocultural importance. In terms of being a coastal resource, the benthic mollusk and its socioecological significance around the Sundarbans ecosystem. In the coastal regions, the mollusks shells are collected and are used for many different purposes, e.g. for food, for making poultry and fish feed, for lime production, ornamental usage, paint making etc. pearls are now collected from bivalves (Sarker 1994). The main harvest seasons for mollusks are from the month November to March. Indiscriminate exploitation rates may cause damage to the trophic structure of ecosystem and also affect the ecosystem functioning. However, the general assemblage pattern e.g. diversity, distribution pattern in and around the Sundarbans of these coastal benthic organisms are not available. The detailed knowledge about the existing benthic distribution pattern, species composition, harvest pattern and season, morphological documentation is still unrealized and inconsistent. Moreover, no systematic research on the marine and coastal benthic ecosystems has been conducted (Quader 1981; Maruf 2004). Therefore it is necessary to understand the coastal benthic distribution and underlying mechanism especially for the unique intertidal mangrove ecosystem. To recognize this, priority must be given to understanding the benthic species assemblage and their diversity pattern. Therefore in this chapter we explore the macrobenthic faunal assemblages, diversity of the benthic fauna, and the driving force of the gateway to the coastal intertidal Sundarbans mangrove ecosystem of Bangladesh.

The overall goal is to understand the macrobenthic assemblage in the Rupsha-Pasur River System of the Sundarbans Ecosystem, Bangladesh.

Specific objectives are:

(i) To investigate the assemblage pattern of the macrobenthos in the Rupsa Pashur river
(ii) To explore how the assemblage pattern is linked to environmental parameters

22.2 Methodology

22.2.1 Study Area

Two sites were selected to get a preliminary picture of the microbenthic distribution within the Sundarbans ecosystem. The sites are Mongla and Koromjal respectively (Fig. 22.1). The information on the particular benthos hotspots in the Sundarbans is not known. The selected two sample sites are considered as the gateway to the Sundarbans and this is the first step to know the ecosystem for searching the good representative benthos samples of Sundarbans ecosystem. Data was collected from November 2014 to December 2015. At the present, the area has a tropical monsoon climate characterized by a reversal of the wind and marine current regimes and river dynamics (Writky 1961; Durand and Petit 1995) with an 8-month rainy season and 4 months of a dry season. The rate of precipitation varies from 1800 to 2790 mm. Most of the rain occurs in May–September and some is received in October and rest of the dry season (Gopal and Chauhan 2006). It is also influenced by the climate anomaly known as El Niňo Southern Oscillation (ENSO), with its two extreme faces, El Niňo and La Niňa that can respectively decrease and increase precipitation rates over the continent. These phenomena cause heavy rainfall during the NW monsoon between October December and March and may cause severe droughts occurring during the SE monsoon season between May and October (e.g. Aldrian and Susanto 2003). The summer monsoon is relatively weak compared to the winter monsoon (Liu and Xie 1999).

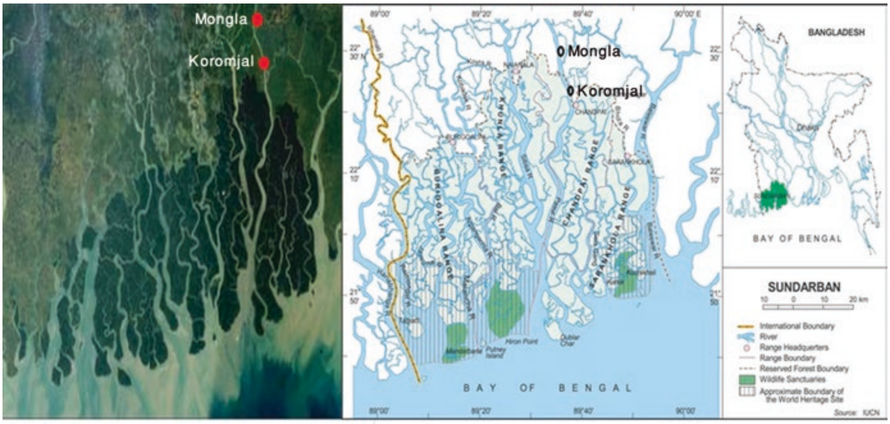

Fig. 22.1 Study site Koromjal and Mongla (delivered from the Google map) and position of the study site Mongla, koromjal

22.2.1.1 Mongla

The main entry point to the Sundarbans from the east side is Mongla upazila (in Bagherhat district), the second sea port of Bangladesh, where the Rupsa and Pashur rivers converge approximately 71 nautical miles (131 km) upstream of the Bay of Bengal. The population in Mongla is 137,947, 54.73% of whom are male and 45.27% female. The Mongla sea port is protected by the Sundarbans mangrove ecosystem. Tourist's boat and vessels are going through Mongla and about 100,000 persons are annually visited the Koromjal through the Mongla (Begum et al. 2015). Therefore, Mongla is also a popular tourist quarry stop for those travelling to the Sundarbans mangrove system. The sediment samples were collected from the Rupsa river system near the Coastguard station (22°35′86″N 89°34′44″E Fig. 22.1).

22.2.1.2 Koromjal

Next to Mongla, Koromjal is the gateway point of the Sundarbans from east side and located in the *Chadpai* Range, Sundarbans. It belongs to Bagerhat district under Chitalmari upazila. The Bangladesh forest department procured an initiative to protect different kind of wild animals in a semi-natural environment in Koromjal. For example a deer and crocodile rearing station is situated here. Therefore this site is popular as a one-day tour stop. The sediment samples were collected from the Rupsa-Pashur river system in Koromjal canal system (N 21°59′26″ E 89°35′21″ Fig. 22.1).

22.2.1.3 Tidal Range

Both, Koromjal and Mongla sites are situated in proximity to the sea and thus tidal flux occurs twice a day and the highest amplitude recorded 5 m and during the tidal range in low tide is about 2–3.5 m (Choudhury et al. 1984).

22.2.1.4 Season

The climate of this area is dominated by two major seasons e.g. the monsoon runs from May to October and the dry season runs from November to March (see also ASEAN-Australia Marine Science Project 1992).

22.2.1.5 Salinity

Salinity gradient in the study area along the Rupsha-Pashur river system reveals mesohaline (±5 – ±18 ppt) and usually Mongla falls under the lower range and the Koromjal falls into the higher range of this gradient, though it depends on the local or regional river dynamics and tidal events.

22.2.2 Sampling

Samples of water and sediment were collected from 2 different sites of the Rupsha-Passur river system, Mongla (N 22°35′86″ E 89°34′44″ Fig. 22.1) and Koromjal (N 21°59′26″ E 89°35′21″) throughout the year in the monsoon and dry season e.g. (monsoon: May, June, Physico-chemical parameters of water, e.g. temperature, pH, conductivity, dissolved oxygen, salinity, were determined fortnightly for each site throughout the study time July, August, September and dry season: October, November, December, January, February). Sediment samples were taken from a 10 m depth during the low tide. The Ekman bottom grab sampler (sample 0.023 m^2) was using to sample the sediment from the benthic habitat. This approach of Ekman bottom grab sampler is well documented in the literature of Helgen (2001). At each sample site, three Ekman grabs (three replicates) were taken for statistical confidence. Collected samples were sieved through 0.5 mm mesh size and fixed with 4% formaldehyde buffered with hexamethelentetramine (e.g. Gerdes et al. 1992). The laboratory animals were then sorted and identified with different taxonomical groups (Day 1967; Gosner 1971). Identification was performed with the help of a stereo zoom biological trinocular LED microscope following available identification keys (Day 1967; Gosner 1971). For identification and counting observation we used numerical abundance of each species was recorded and primarily expressed as no.m^{-2} (Welch 1948). This data was then converted to relative abundance in order to standardize it. Physico-chemical parameters of water, e.g. temperature, pH, conductivity, dissolved oxygen, salinity, were determined fortnightly for each site throughout the study time (APHA 1998).

22.2.3 Statistical Analysis

To study the relations between the macrobenthic species, relative abundances, and environmental parameters, redundancy multivariate analysis (RDA, Rao 1964) was applied by using the CANOCO 5 software package (Ter Braak and Šmilauer 2002; Šmilauer and Lepš 2014). All taxonomic data were standardized and logarithmically transformed. The length of the variance gradient was estimated by means of a preliminary principal component analysis (PCA, Hill and Gauch 1980). In order to evaluate the significance of the RDA axes, using the nonparametric Monte Carlo permutation test (Manly 1992), which expecting the null hypothesis that there is no relations exist between the variation in relative abundance of the macrobentghic species and environmental parameters. The benthic abundance and seasonal effect was evaluated by the Kolmogorov-Smirnov test with STATISTICA 13 package software (Nisbet et al. 2009, StatSoft Inc. 2013).

22.3 Results

22.3.1 Physico-Chemical Parameters of Study Sites

The mean values of physico-chemical parameters of water at 10 m depth during the low tide, estimated fortnightly over a period of 10 months (Nov, Dec, Jan, Feb, May, June, July, Aug, Sep and Oct) are summarized in Table 22.1 as mean value. Maximum mean water temperature (28.99 °C) was recorded in Mongla site while highest amount of pH (7.94), DO (7.92 mg/l), salinity (6.84 ppt) and Conductivity (1.58 mS/cm) were recorded in the Koromjal site.

Among the major groups found in the both sites, Annelida (including echiura, oligochaeta and polychaeta) represented highest diversity of 22 species, followed by arthropoda (Crustacea-decapoda) 21 species, 20 species of Mollusk (gastropoda), 9 species of arthropoda-bivalvia, few crustacea-isopoda, crustaceae-tanaidaceae and nemartina taxa (Table 22.2). The relative abundance was not significantly different in these two sites. A significant seasonal effect on the relative abundance was found (Fig. 22.3. Kolmogorov-Smirnov test, at $p < 0.05$) value $P < 0.001$, Stdev = 1.193 (Monsoon: May, June, July, August, September), and 0.0799 (dry season: October, November, December, January, February), N = 820. Monsoon is more diverse in macrobenthic species than dry season.

Environmental parameters were not significantly differing in the macrobentic distribution but season has a significant influence ($p < 0.001$, Fig. 22.2).

Table 22.1 Physico-chemical parameters [mean ± SD (range)] of water in two different sites (Koromjal and Mongla)

Parameters	Koromjal	Mongla
Temperature (°C)	27.80 ± 4.74 (21.90–33.20)	28.99 ± 5.14(21.60–33.80)
Dissolved oxygen (mg/1)	7.92 ± 0.35 (7.30–8.40)	7.83 ± 0.47 (7.25–8.90)
pH	7.94 ± 0.33 (7.50–8.40)	6.93 ± 0.42 (6.2–7.6)
Salinity (ppt)	6.84 ± 3.07 (2.89–11.01)	4.64 ± 2.41 (1.15–8.40)
Electric conductivity (mS/cm)	1.58 ± 0.54(1.10–2.42)	0.62 ± 0.083(0.52–0.74)

Table 22.2 Major taxa group with species list of study site Koromjal and Mongla

Taxa	Species	Taxa	Species
Annelida-Echiura	Unknown sp 1	Arthropoda-Insecta	*Penaeus canaliculatus*
Annelida-oligochaeta	*Bothrioneurum iris*	Cnidaria	Unknown sp 2
Annelida-oligochaeta	*Limnodrilus hoffmeisteri*	Crustaceae -decapod	*Metapenaeusensis*
Annelida-Polychaeta	*Dendronereides heteropoda*	Crustaceae -decapod	*Metapenaeus monocerus*
Annelida-Polychaeta	*Dendronereides* sp.	Crustaceae -decapod	*Palaemon* sp.
Annelida-Polychaeta	*Dendronereis aestuarina*	Crustaceae -decapod	*Paracymus evanescens*
Annelida-Polychaeta	*Dendronereis arborifera*	Crustaceae -decapod	*Pelecyora trigona*
Annelida-Polychaeta	*Glycera* sp.	Crustaceae -decapod	*Penaeus penicillatus*
Annelida-Polychaeta	*Lumbrineris* sp.	Crustaceae-tanaidaceae	Unknown sp 3
Annelida-Polychaeta	*Lycastis indica*	Crustaceae-decapode	Unknown sp 4
Annelida-Polychaeta	*Lycastonereis indica*	Crustaceae-Isopode	Unknown sp 5
Annelida-Polychaeta	*Namalycastis fauveli*	Mollusca-Bivalvia	*Anadara antiquate*
Annelida-Polychaeta	*Namanereis quadraticeps*	Mollusca-Bivalvia	*Anadara granosa*
Annelida-Polychaeta	*Nereis caudata*	Mollusca-Bivalvia	*Donaxi incarnates*
Annelida-Polychaeta	*Nereis falcaria*	Mollusca-Bivalvia	*Meretrix meretrix*
Annelida-Polychaeta	*Nereis lamellosa*	Mollusca-Bivalvia	*Modiolus striatulus*
Annelida-Polychaeta	*Nereis mossambica*	Mollusca-Bivalvia	*Penaeus japonicas*
Annelida-Polychaeta	*Nereis operta*	Mollusca-Bivalvia	*Perna indicus*
Annelida-Polychaeta	*Perinereis nuntia*	Mollusca-Bivalvia	*Perna viridis*
Annelida-Polychaeta	*Talehsapia annandalei*	Mollusca-Bivalvia	*Polymesoda bengalensis*
Annelida-Polychaeta	*Tylonereis bogoyawlenskyi*	Mollusca-gastropoda	*Assiminea beddomeana*
Annelida-Polychaeta	*Tylonereis fauveli*	Mollusca-gastropoda	*Assiminea brevicula*
Arthropoda crustaceae Decapods	*Ampelisca* sp.	Mollusca-gastropoda	*Cerithidea alata*
Arthropoda crustaceae-Decapods	*Balanus amphitrite*	Mollusca-gastropoda	*Cerithidea obtuse*
Arthropoda crustaceae-Decapods	*Macrobrachium Rosenbergii*	Mollusca-gastropoda	*Conus striatus*
Arthropoda crustaceae-Decapods	*Metaplax dentipes*	Mollusca-gastropoda	*Conus textile*
Arthropoda crustaceae-Decapods	*Metaplax intermedia*	Mollusca-gastropoda	*Littoraria undulate*
Arthropoda crustaceae-Decapods	*Myomenippe hardwickii*	Mollusca-gastropoda	*Littoraria scabra*
Arthropoda crustaceae-Decapods	*Penulirus polyphagus*	Mollusca-gastropoda	*Nassarius stolatus*
Arthropoda crustaceae-Decapods	*Portunus pelagicus*	Mollusca-gastropoda	*Natica tigrina*

(continued)

Table 22.2 (continued)

Taxa	Species	Taxa	Species
Arthropoda crustaceae-decapod	*Portunus sanguinolentus*	Mollusca-gastropoda	*Neritina violacea*
Arthropoda crustaceae-Decapods	*Pseudosesarma edwardsi*	Mollusca-gastropoda	*Onchidium tigrina*
Arthropoda crustaceae-Decapods	*Scylla serrata*	Mollusca-gastropoda	*Pila globosa*
Arthropoda crustaceae-Decapods	*Scylla tranquibarica*	Mollusca-gastropoda	*Stenothyra bhlanfordiana*
Arthropoda crustaceae-Decapods	*Varuna litterata*	Mollusca-gastropoda	*Stenothyra deltae*
Arthropoda crustaceae-decapod	*Uca rosea*	Mollusca-gastropoda	*Telescopium telescopium*
Arthropoda-Insecta	*Canthydrus morsbachi*	Mollusca-gastropoda	*Theodaxus oualaniensis*
Arthropoda-Insecta	*Guignotus flammulatus*	Mollusca-gastropoda	*Trochus niloticus*
Arthropoda-Insecta	*Guignotus inconstans*	Mollusca-gastropoda	*Turbo marmoratus*
Arthropoda-Insecta	*Heterocerus* sp.	Nemartina	Unknown sp 6

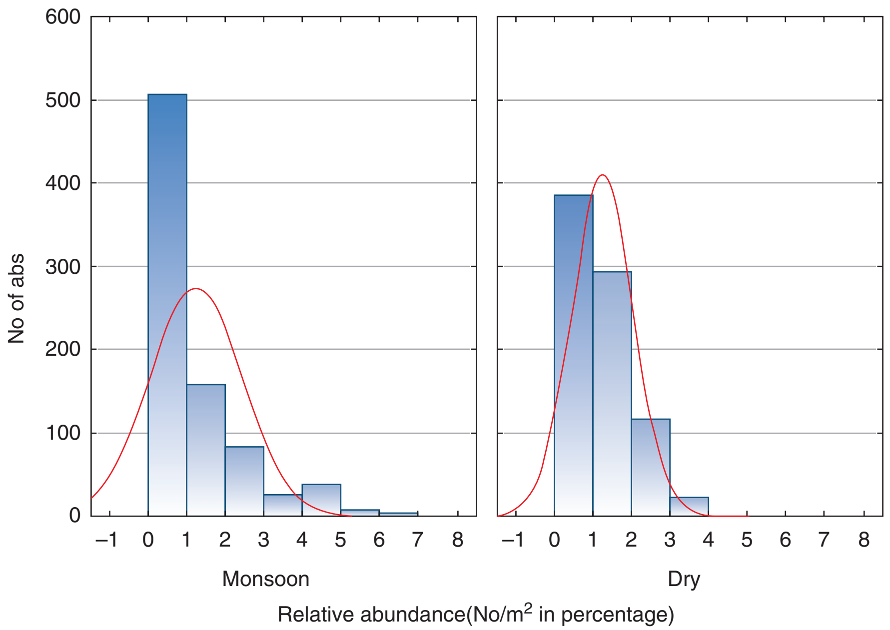

Fig. 22.2 Macrobenthic species relative abundance and seasonal effect ($p < 0.001$, at 5% Kolmogorov-Smirnov test, Stdev = 1.193 for Monsoon (May, June, July, Aug, Sep), and 0.0799 for Dry season (Oct, Nov, Dec, Jan, Feb), N = 820)

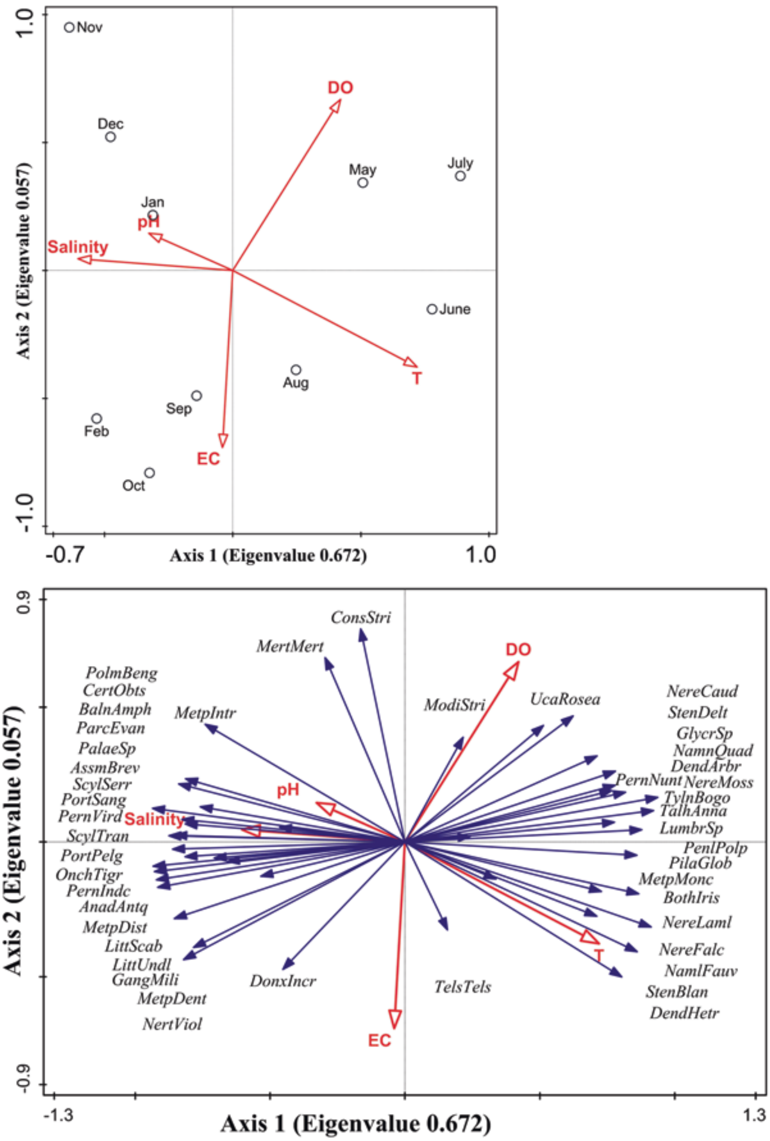

Fig. 22.3 Results of RDA analysis illustrating macrobenthic species in relation to months/season (*above*) and environmental variables (*below*) for Koromjal study site. Environmental parameters are abbreviated as indicated in text. Please see the Annex 1 for the species types are indicated

22.3.2 Multivariate Analysis

Principal component analysis (PCA) has revealed data are compositional and have a gradient 0.2 standard deviation (SD), redundancy analysis (RDA) was applied as recommended by Šmilauer and Leps (2014). The first two dimensions of the RDA (Fig. 22.3 for Koromjal (site 1) and Fig. 22.4 Mongla (site 2) account for 67% and

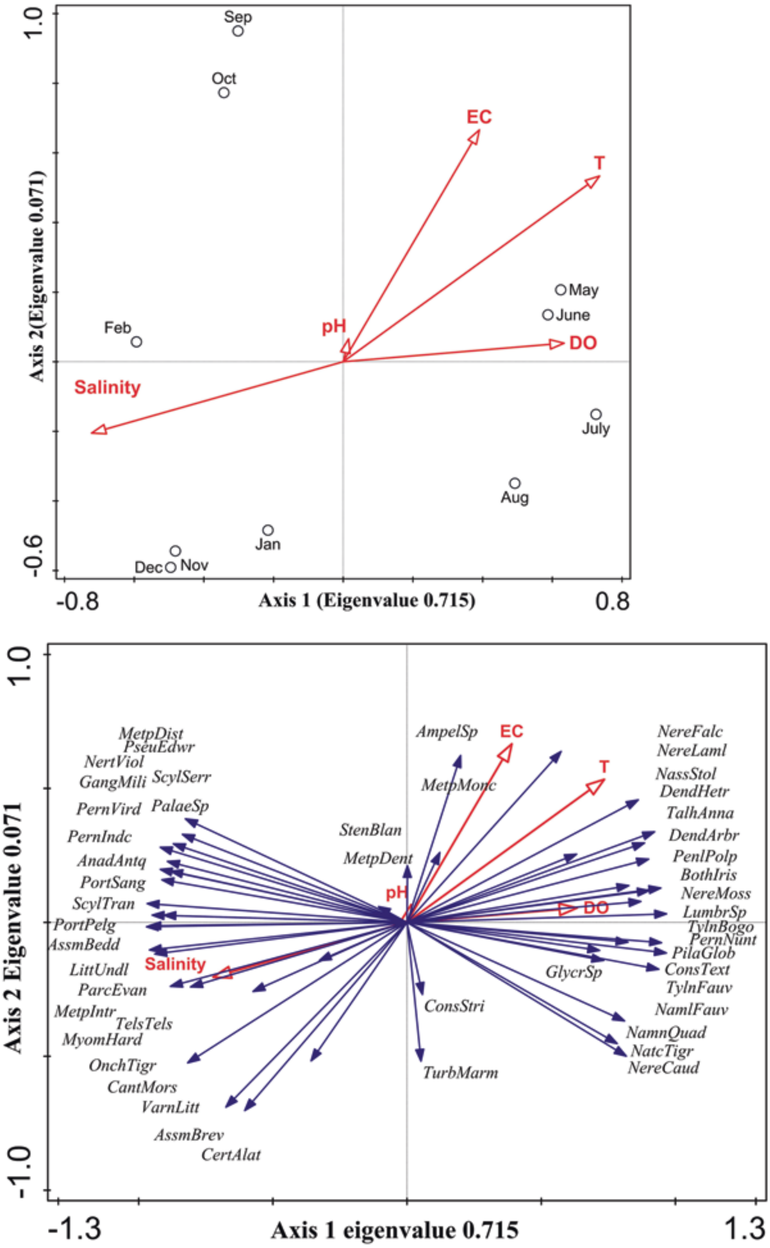

Fig. 22.4 Results of RDA analysis illustrating macrobenthic species in relation to months/season (*above*) and environmental variables (*below*) for Mongla study site. Environmental parameters are abbreviated as indicated in text. Please see the Annex 2 for the species types are indicated

6% and 71% and 7% of the total variance of microbenthic species assemblage and environmental parameters of the ecosystem respectively. The total variation present in the species data is called total inertia and is given by the sum of all constrained eigenvalues. For Koromjal site, test of significance of all canonical axes trace is 0.82 (site 1) and 0.88 (site 2), F ratio 3.65 (site 1) and 5.78 (site 2) respectively. P value showed 0.03 (site 1) and 0.004 (site 2) respectively. The ratio $\lambda 1 + \lambda 2$/total variance, a measure of the goodness of fit equivalent to R^2 (Jongman et al. 1987), is 0.82 and 0.88 for Koromjal and Mongla site 2 respectively. The relative distance between samples explains the differences in microbenthic species composition. Results of RDA analysis of macrobenthic species distribution in Koromjal (Fig. 22.3) in relation to its environmental parameters directed the relative abundance of the species as salinity in one cluster and the other parameters (such as temperature, dissolved oxygen etc.) in opposite direction cluster. The RDA analysis of months (Fig. 22.3) and environmental parameters also directed as the monsoon (May, June, July, August and September) and dry season (October, November, December, January and February) this also coincides (Fig. 22.4) with Mongla site analysis. Overall, the species composition was coincides with monsoon as one direction and dry season as another direction. However, pH did not dominate that much in overall distribution and showed a tendency to the salinity direction in both sites.

22.3.3 Macrobenthic Species Assemblage Pattern

Total 82 species including few (six) unidentified species belonging to eleven groups e.g. Cnidaria, Nemartina, Annelida-Echiura, Annelida-oligochaeta, Annelida-Polychaeta, Arthropoda-crustaceae -decapoda, Arthropoda-Insecta, Arthropoda-crustaceae- tanaidaceae, Mollusca-bivalvia, Mollusca-gastropoda were found. Among them relative abundance of annelida was shown higher in both sites followed by crustacea decapoda, gastropoda, bivalvia and insectecea.

RDA analysis in Koromjal site illustrated macrobenthic species assemblage in relation to its environmental parameters in different months or seasons. The analysis revealed, species assemblage in dry season coincides with salinity direction and clustered as follows: *Polymesoda bengalensis, Cerithidea obtuse, Balanus amphitrite, Paracymus evanescens, Palaemon* sp.*, Assiminea brevicula, Scylla serrata, Portunus sanguinolentus, Perna viridis, Scylla tranquibarica, Portunus pelagicus, Onchidium tigrina, Perna indicus, Anadara antiquate, Metaplax distincta, Littoraria scabra, Littoraria undulata, Gangetia miliacea, Metaplax dentipes, Neritina violacea, Donax incarnates, Metaplax, intermedia, Telescopium telescopium Meretrix meretrix and Conus striatus*. Temperature, Electrical conductivity and dissolved oxygen has coincides with monsoon and species were clustered as follows: *Dendronereides heteropoda, Stenothyra blanfordiana, Namalycastis fauveli, Nereis falcaria, Nereis lamellosa, Bothrioneurum iris, Pila globosa, Penulirus polyphagus, Lumbrineris* sp.*, Talehsapia annandalei, Tylonereis bogoyawlenskyi, Perinereis* *nuntia, Nereis mossambica, Dendronereis arborifera, Namanereis quadraticeps, Glycera* sp.*, Stenothyra deltae, Nereis caudate and Uca rosea* (Annexure 1).

Similarly RDA analysis in Mongla revealed a slight different combination (Fig. 22.4) but majority in same direction such as salinity directed to dry seasonal species assemblage were as follows: *Metaplax distincta, Pseudosesarma edwardsi, Neritina violacea, Gangetia miliacea, Scylla serrata, Perna viridis, Palaemon* sp.*, Perna indicus, Anadara antiquate, Portunus sanguinolentus, Scylla tranquibarica, Portunus pelagicus, Assiminea beddomeana, Littoraria undulata, Paracymus evanescens, Metaplax intermedia, Telescopium Telescopium, Myomenippe hardwickii, Onchidium tigrina,Canthydrus morsbach, Stenothyra blanfordiana and Metaplax dentipes.* Temperature, Electrical conductivity and dissolved oxygen has coincides with monsoon and species were assemblaged as: *Varuna litterata, Assiminea brevicula, Cerithidea alata, Conus striatus, Turbo marmoratus, Glycera* sp.*, Nereis caudata, Natica tigrina, Namanereis quadraticeps, Namalycastis fauveli, Tylonereis fauveli, Conus textile, Pila globose, Perinereis nuntia, Tylonereis bogoyawlenskyi, Lumbrineris* sp.*, Nereis mossambica, Bothrioneurum iris, Penulirus polyphagus, Dendronereis arborifera, Talehsapia annandalei, Dendronereides heteropoda, Nassarius stolatus, Nereis lamellose, Nereis falcaria, Metapenaeus monocerus and Ampelisca* sp. (Annexure 2).

There was no significant effect of single environmental parameters observed but common patterns show (RDA Figs. 22.3 and 22.4) a number of species assemblages due to the salinity force e.g., *Palaemon* sp.*, Gangetia miliacea, Neritina violacea*, *Scylla serrata, Metaplax distincta, Anadara antiquate, Littoraria undulate, Onchidium tigrina, Assiminea brevicula* and the other environmental forces such as dissolved oxygen, temperature and electrical conductivity. Species assemblages under these patterns are such as, *Nereis falcaria, Nereis lamellosa, Bothrioneurum iris, Namalycastis fauveli, Nereis mossambica, Nereis caudata Pila globosa.* Multivariate analysis showed there is no significant effect of single environmental parameters but seasonal effect was significant ($p < 0.001$). Relative abundance of macrobenthos species showed higher in monsoon.

22.4 Discussion

22.4.1 Climate Dynamics of the Study Area

Mongla and Koromjal sites are situated in a proximity to the sea and thus tidal flux occurs twice a day and the highest amplitude recorded 5 m and during the tidal range in low tide is about 2–43.5 m. Therefore it is expected that within these two sites environmental parameters need long term monitoring for a detailed picture of its dynamics (e.g. Choudhury et al. 2015). The wet monsoon season runs from May to September and brings heavy rains under the influences of south-west trade winds, with 75% of the annual precipitation occurring during this period (Gopal and Chauhan 2006; IWM 2013). The dry season is from October to April and brings infrequent rainfall, under the influences of the north and northeast winds. However, the Sundarbans saline water intrusion is highly seasonal but the concentration of different salt in different months of the year may vary in different sampling depth (Smith et al. 2008). The gateway point to the Sundarbans, two study sites Koromjal and Mongla, have a tropical monsoon climate with a prolonged rainy season and 4–5 months of a drier season (Gopal and Chauhan 2006; Aldrian and Susanto 2003). In this research, detailed data such as analysis of marine currents system, composition of bottom sediments and detailed physico-chemical analysis of water and sediment of the two sites were not possible to measure within the same time and effort, which will be done in future task. Though it is not enough for this deltaic dynamic system for drawing a conclusion about the environmental parameters in this research with 1 year parameters, multivariate result showed two sites does not differ significantly in terms of environmental parameters (Table 22.1). It was also found that mangrove ecosystems have traditionally remained as important sentinels of environmental change with more sensitivity towards combination of stressors rather than a single stressor (Cabecinha et al. 2009; Sun et al. 2011). Recent environmental parameter change phenomena also showed that patterns of change are not similar in the following years (such as 2010, 2011) and some years are even with extreme events of flood and storms that strikes beyond prediction (Choudhury et al. 2015). This has effect on physico-chemical properties of habitat and eventually shaped the organisms in bottom-up regulation of food webs. Nevertheless, it was expected that within this both sites environmental parameters need a long term monitoring on sediment composition and physico-chemical and biological properties to have a detailed picture on its dynamics (Choudhury et al. 1994).

22.4.2 Seasonal Effect

The research found that salinity and dry season is correlated (Figs. 22.3 and 22.4) to the relative abundance of the habitat preference, sediment type and nutrient availability (Lu 2005; Roy et al. 2014) might depends on season and therefore season showed a significant high relative abundance in monsoon (Fig. 22.3, $p < 0.001$). RDA analysis revealed a large number of Polychaeta are clustered in monsoon in both sites this might be related with the habitat chemistry and nutrient availability (e.g. Rahaman et al. 2014). Seasonal distribution of species in different months (Figs. 22.3 and 22.4) indicated the possibility of unknown combined effect such as fresh water flow, sediment transport and nutrient dynamics that will lead to changes in ecosystem structure and functioning (Walsh et al. 1995; Cloern 1996).

22.4.3 Relative Abundance and Environmental Variable

About 82 species were found in both the Koromjal and Mongla sites, those apparently situated about 20 km range but there might be very micro level local influence of the river Rupsha- Pasur system which was not detected in this research. However, neither site nor environmental parameters have significant effect of the relative abundance, whereas the season has a significant influence ($p < 0.001$, Fig. 22.2). The seasonal effect is also observed by the results of Roy et al. (2014) and Lu (2005). It could be stated that the combined effect of environment for both sites were significant ($p = 0.03$ and $p = 0.004$, Figs. 22.3 and 22.4), the p values obtained from a permutation test is similar to p values from standard statistical tests e.g. for a multiple regression. Nevertheless, it refers that the macrobenthic species were observing occurred without relation to the five environmental descriptors we have measured. And the p value is low (<0.05) for our dataset, it could be said that at least one of the five variables has a significant effect (Šmilauer and Lepš 2014). Importantly multiple regressions attempts to predict values of not a single variable, but many variables, each benthic species being one. Consequently, the results from the permutation test may not necessarily apply to all the species: some might respond and some do not, some even respond in different ways than other (Figs. 22.3 and 22.4) and general linear model show at least 55 species in Koromjal and at least 69 species in Mongla have significant correlation with at least one of the environmental parameters.

22.4.4 Multivariate Analysis and Major Species Cluster

It is shown in the RDA analysis that species mainly clusters towards salinity and distribution pattern divided among different months (Figs. 22.3 and 22.4) and pointed in seasons as dry season and monsoon. However, annelida-polychaeta species were common in monsoon in both sites and this coincides with the result of similar research carried out in east and west coast of India (Prabha 1994; Sunilkumar 1995). This is may be due to the fact that Polychaetes have a mucos secreting device which is used to protect themselves in adverse conditions in the dynamic coastal system (Sadhana 1993). During the monsoon season, main rivers (Ganges -Bhramaputra- Meghna) discharge about 80% of the annual fresh water flow and other parameters like salinity, temperature, mean particle diameter, sand and depth (Ganesh and Raman 2007) that might favor the annelida specially polychaete species distribution. Among the polychaeta, common species are *Namalycastis fauveli, Nereis falcaria, Nereis lamellosa, Bothrioneurum iris, Lumbrineris* sp., *Talehsapia annandalei, Tylonereis bogoyawlenskyi, Perinereis nuntia, Nereis mossambica, Dendronereis arborifera, Namanereis quadraticeps, Nereis caudate etc. followed by gastropoda pila globosa*. In dry season due to higher salinity and less fresh water flashing (e.g. Dittmann 2002) Crustacea -decapoda such as *Scylla serrata, Palaemon* sp., *Scylla tranquibarica, Portunus pelagicus* were dominating which differ from the results obtained by the similar research by Anbuchezhian et al. (2009) in the south east coast of India . The reason might be the dynamics of different river system and local physico- chemical and biological parameters of the system. The RDA analysis also revealed, gastropoda was the third dominating group and relative species abundance was followed by e.g. *Assiminea beddomeana, Littoraria undulata,* *Telescopium telescopium, Neritina violacea, Gangetia miliacea* and bivalve *Perna viridis*. Importantly relative abundance of *Telescopium* was distributed during winter in Mongla and during late monsoon in Koromjal this may open the window of complexity of the whole system, both in natural and anthropogenic effects that could shift the species assemblage pattern and functioning of the system like other coastal ecosystems around the world (Comin and Valiela (1993). Species with relative abundance less than five was excluded only from the Figs. 22.3 and 22.4 but not from the analysis.

22.4.5 Anthropogenic Activities

It is reported that the demographic trends of population densities in coastal areas are increasing throughout the world and about 67% estimated global population lives on the coast or within 60 km of the coast and the percentage might reached at double within 30 years (Norse 1995). These growing populations intensify pressures on utilization of resources in coastal areas and therefore lead to habitat degradation, fragmentation and destruction. This could be a special problem in Bangladesh where the highest marine diversity is found near to centres of high human population growth. Resulting threats to coastal systems are frequently interlinked (Sebens 1994; Norse 1995; Begum et al. 2015). Those threats includes habitat loss: which is the most important and critical issue, overexploitation and other effects of fishing; pollution including direct and indirect effects of inorganic and organic chemicals; eutrophication, species invasions, water-shed alteration and physical alterations of coasts, tourism, marine litter, and the fact that humans have little perception of the oceans and their marine life. Importantly the threats from commercial fishing on biodiversity of coastal areas have been neglected.

It is important that freshwater flow is regular with the tidal influence because the principal source of freshwater in the Sunderbans is the discharge from the Ganges and runoff from various distributaries. Most of these distributaries have lost their connection with the Ganges due to geological processes or human interventions. Significant interventions that altered the river flow are the Farakka barrage in up India, the Mongla–Gashiakhali cut, conversion of mangrove to agriculture and aquaculture, shrimp fry collection through the coast, oil spill litter from traditional tourism practice. In the last December 2014 The oil tanker, anchored in the river at Joymoni Ghola of the east range of Sundarbans at night, A cargo trawler hit the oil tanker from behind when it was starting its voyage in early morning and the spilled oil was spreaded around 20 km^2 areas including Joymoni, Nandobala, Andharmanik and Mrigomari in the Sundarbans ecosystem (Fig. 22.5). However the benthic system needs to be checked in terms of the ecosystem food chain and health. Therefore to get the ecosystem services from the mangrove ecosystem, the unique detritus food cycle should be unimpaired and it is time to check the detail species diversity and underlying mechanism in each location of the Sundarbans for the sustainable management of the ecosystem.

Fig. 22.5 The Sundarbans before the oil spill and after oil spill (*black* band in the vegetation), photo: field survey 2014

22.5 Conclusion

Owing to the low lying, very flat and dynamic nature, the physical and socioecological systems in the coastal area of Bangladesh differ from the rest of the country. This unique ecosystem has developed under the influence of the intertidal brackish water and sweet water interaction. Therefore its ecosystem is vulnerable to any change in its biophysical conditions. Multivariate statistical method of principal component analysis (PCA) was used to conclusively establish the relationships between macrobenthic species (dependent variables) and environmental variables (explanatory variables). Accordingly RDA helped to visualise the ecological patterns by coupling biotic variables and environmental datasets in a single two-dimensional plot. It was established that the study area is highly influenced by seasonal and complex dynamic functioning patterns. Anthropogenic activities might add factors which are more localized and temporal in the mangrove ecosystems. Since, the Sundarbans is intersected by a large network of rivers channels and creeks varying in width from a few feet to several miles. The larger of these rivers are the remains of former beds of Ganges, which has gradually shifted eastward and are no longer directly connected to the main river. The Baleswar, which passes along the eastern boundary of the forest, is only the river is connected with the Ganges and receives direct fresh water effluence from the river. A large number of channels and creeks flow into the Sundarbans, these, in addition to flooding the forest floor, make most of the forest ecosystem accessible y country boats during high tides and make the ecosystem extraction activities relatively easy and play important roles in navigation within the forest. The Sundarbans acts as a buffer between the densely populated agricultural land and sea and protects the hinterland from major damages. It is obvious that the Sundarbans play a very vital role in the economy of the region and generate a lot of economic activities and provide livelihood to a very large number of people in the region.

Thus, it can be concluded that the Sundarbans mangrove ecosystem is dynamic in nature where the environmental parameters forced the macrobenthic species which seemed to be influenced by the combined effects of environmental parameter and biological parameters, not as a single force. However there is no management practice for the sustainable management of the non-forestry and other minor products. More research must be undertaken in order to get more data sets along the mangrove coverage and the river systems. Furthermore, detailed research is necessary to understand the species and the system dynamics which seemed to be influenced by the combined effects of environmental parameter and biological parameters.

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Annexures

Annexure 1

Results of the RDA analysis (Koromjal) macrobenthic species types are indicated as follows: PlomBeng, *Polymesoda bengalensis*; CertObst, *Cerithidea obtuse*; BalnAmph, *Balanus amphitrite*; ParcEvan, *Paracymus evanescens*; PalaeSp, *Palaemon* sp.; AssemBrev, *Assiminea brevicula*; Scylserr, *Scylla serrata*; PortSang, *Portunus sanguinolentus*; PernVird, *Perna viridis*; ScylTran, *Scylla tranquibarica*; PortPleg, *Portunus pelagicus*; OnchTigr, *Onchidium tigrina*; PernIndc, *Perna indicus*; AnadAntq, *Anadara antiquate*; MetaDist, *Metaplax distincta*; LittScab, *Littoraria scabra*; LittUndl, *Littoraria undulata*; GangMili, *Gangetia miliacea*; MetaDent, *Metaplax dentipes*; NertViol, *Neritina violacea*; DonxIncr, *Donax incarnatus*; MetpIntr, *Metaplax*; intermedia,TelsTels, *Telescopium telescopium*; Dendhert, *Dendronereides heteropoda*; StenBlan, *Stenothyra blanfordiana*; NamlFauv, *Namalycastis fauveli*; NereFalc, *Nereis falcaria*; NereLaml, *Nereis lamellosa*; BothIris, *Bothrioneurum iris*; PilaGlob, *Pila globosa*; PenlPolp, *Penulirus polyphagus*; LumbrSp, *Lumbrineris* sp.;TalhAnna, *Talehsapia annandalei*; TylnBogo, *Tylonereis bogoyawlenskyi*; PernNunt, *Perinereis nuntia*; NereMoss, *Nereis mossambica*; DendArbr, *Dendronereis arborifera*, NamnQuad, *Namanereis quadraticeps*; GlysrSp, *Glycera* sp.; StenDelt, *Stenothyra deltae*; NereCuad, *Nereis caudata*; UcaRosea, *Uca rosea; ConsStri*, *Conus striatus*; MertMert; Meretrix meretrix; and Nov, *November*; Dec, *December*; Jan, *January*; Feb, *February*; may, *May*; June, *June*; July, *July*; Aug, *August*; Sep, *September*; Oct, *October*

Annexure 2

Results of RDA analysis (Mongla) macrobenthic species types are indicated as follows: MetpDist, *Metaplax distincta*; PseudEdwr, *Pseudosesarma edwardsi*; NertViol, *Neritina violacea*; GangMili, *Gangetia miliacea*; ScylSerr, *Scylla serrata*; PernVird, *Perna viridis*; PalaeSp, *Palaemon* sp.; PernIndc, *Perna indicus*; AnadAntq, *Anadara antiquate*; PortSang, *Portunus sanguinolentus*; ScylTran, *Scylla tranquibarica*; PortPelg, *Portunus pelagicus*; AssmBedd, *Assiminea beddomeana*; LittUndl, *Littoraria undulata*; ParcEvan, *Paracymus evanescens*; MetpIntr, *Metaplax intermedia*; TelsTels, *Telescopium Telescopium*; MyomHard, *Myomenippe hardwickii*; OnchTigr, *Onchidium tigrina*; CantMors, *Canthydrus morsbachi*; VarnLitt, *Varuna litterata*; AssmBrev, *Assiminea brevicula*; CertAlat, *Cerithidea alata*; ConsStri, *Conus striatus*; TurbMarm, *Turbo marmoratus*; GlycrSp, *Glycera* sp.; NereCaud, *Nereis caudata*; NatcTigr, *Natica tigrina*; NamnQuad, *Namanereis quadraticeps*; NamlFauv, *Namalycastis fauveli*; TylnFauv, *Tylonereis fauveli*; ConsText, *Conus textile*; PilaGlob, *Pila globose*; PernNunt, *Perinereis nuntia*; TylnBogo, *Tylonereis bogoyawlenskyi*; LumbrSp, *Lumbrineris* sp.;NereMss, *Nereis mossambica*; BothIris, *Bothrioneurum iris*; PenlPolp, *Penulirus polyphagus*; DendArbr, *Dendronereis arborifera*; TalhAnna, *Talehsapia annandalei*; DendHert, *Dendronereides heteropoda*; NassStol, *Nassarius stolatus*; NereLaml, *Nereis lamellose*; NereFalc, *Nereis falcaria*; MetpMonc, *Metapenaeus monocerus*; AmpelSp, *Ampelisca* sp.; StenBlan, *Stenothyra blanfordiana*; MetpDent, Metaplax dentipes; and Nov, *November*; Dec, *December*; Jan, *January*; Feb, *February*; May, *May*; June, *June*; July, *July*; Aug, *August*; Sep, *September*; Oct, *October*

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Chapter 23
Ecological Services of Intertidal Benthic Fauna and the Sustenance of Coastal Wetlands Along the Midnapore (East) Coast, West Bengal, India

Susanta Kumar Chakraborty

Abstract Human existence is entirely dependent on the products and services of biodiversity for food, medicines, shelter, clothing materials, aesthetics etc. Ecological services on the other hand denote the contribution of nature to a variety of "goods and services" to mankind in respect of economics and ecology. Biodiversity being an important component of the mother earth renders valuable ecological services to all the compartments of the environment including coastal zone which is the interface of the land and sea and represents an eco-potential ecosystem along with its different geo-morphological components like estuaries, mangroves, dunes, deltas, lagoons, intertidal zones, etc. The present article focuses on the functional contribution of benthic biodiversity towards sustenance of a short but geo-morphologically diversified intertidal zones of coastal Midnapore (East) District, West Bengal, India which is in continuation of Sundarbans mangrove estuarine complex of India These benthic fauna, both macrobenthos (brachyuran crabs, molluscs, polycaetes, actiniarians etc.) and meiobenthos(nematodes, foraminifera, copepods, polychaetes etc.) render valuable ecological services by making sediments loaded with living organisms by bioturbation, releasing millions of benthic larvae (meroplankton) to the aquatic system as the food of fishes, converting mangrove leaves into detritus,releasing nutrients, ploughing sediments to maintain textural composition, acting as food for demerseal fishes, bioaccumulating pollutants,serving as bioindicator, and providing aesthetics.

Keywords Midnapore coast • Intertidal benthos • Ecological services • Biodiversity

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23.1 Introduction

23.1.1 Coastal Environment and Biodiversity

Coastal zone representing the junction of terrestrial with marine ecosystems, harbours diversified Flora, Fauna, and Microbes in the form of mangroves and its associates, benthos (macro, meio and micro benthos) nekton and plankton in different geo-morphological units like estuaries, creeks, intertidal and sub-tidal zones, mangroves, delta etc. All these faunal and floral inhabitants have been found to display varied patterns of succession, distribution, and eco-dynamics in tune with the changing ecological gradients and also by enjoying definite ecological niche.

'Biodiversity' is defined as 'variability among living organisms from all sources including inter phyla terrestrial, marine and other aquatic ecosystems and ecological complexes of which they are part: that includes diversity within species, between species and of ecosystems (UNCED 1992)'. Interactions within, between and among various levels of biodiversity is the main intrinsic mechanism to maintain the self sustaining structural and functional attributes of biodiversity within the scale of time and space (Chakraborty 2003). Biodiversity of coastal zones and their hydrologically linked coastal areas have come under tremendous environmental pressures during recent decades. The resilience of coastal ecosystem in order to cope up with ongoing environmental stresses is being threatened by pollution and over exploitation of resources. A management strategy based on the sustainable utilization of coastal resources should have objective of ecosystem integrity maintenance that is the maintenance of system components, interactions among them and the resultant behaviour or dynamic of the system (Mukherjee and Bakshi 1998). Biodiversity has become a password all over the globe especially after the Rio-Conference, in 1992 where 167 countries signed the Convention on Biological Diversity (CBD) to ensure the conservation and sustainable use of biodiversity and the equitable sharing of the benefits from utilizing genetic resources (CBD 2010a).

After the collective failure to achieve the Convention on Biological Diversity's (CBD's) 2010 target to substantially reduce biodiversity losses, 10th Conference of the Parties (COP) of the CBD took place in October 2010 in Nagoya, Japan, to determine next steps following the failure to achieve the 2010 target of substantially reducing the loss of biodiversity (Butchart et al. 2010; CBD 2010a, b). The proposed Convention on Biological Diversity's in the year 2020 is expected to give more emphasis on ecosystem services approach using three categories – "red" (urgent threats), "green" (conservation and sustainable use), and "blue" (socioeconomic drivers) (Fig. 23.1). Targets should be supported by indicators which are to be used to estimate the value of ecosytem services and should reflect the urgency of the threats as because targets are interdependent (Perrings et al. 2011).

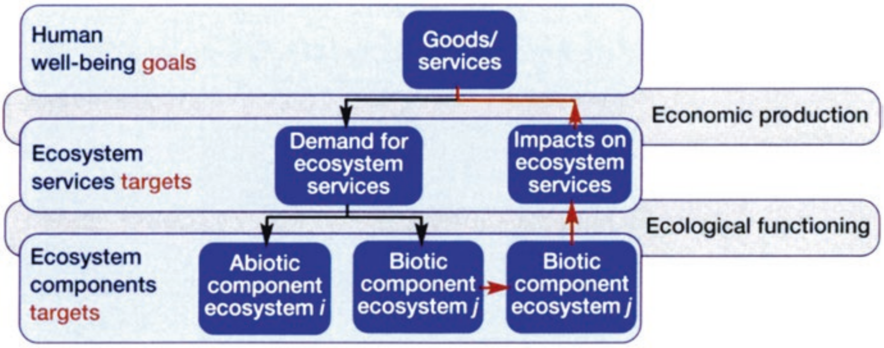


Fig. 23.1 Relationship between demand for ecosystem services and biodiversity conservation; *Black arrows* indicate the demand for services while *red arrows* show impact (Perrings et al. 2011)

23.1.2 Ecological Services: Concept and Types

Ecological services denote the contribution of nature to a variety of "goods and services," which in economics would normally be classified under three different categories (Barbier 2007): (1) "**goods**" representing bioresources, water, and genetic materials, (2) "**services**" in the form aesthetic (recreational and tourism) benefits or certain ecological regulatory and habitat functions, such as water purification, climate regulation, erosion control, and habitat maintenance, and (3) **cultural benefits** mostly in the form of spiritual and religious beliefs, heritage values. The Millenium Ecosystem Assessment (2005) defines Ecosystem Services as "benefits people obtain from ecosystems". These benefits may be derived directly or indirectly (Costanza and Fiber 2002; Costanza et al. 1997) and can cover a wide variety of services which may include regulatory or supporting services (Beaumont et al. 2007).

23.1.3 Interdependencies Between Ecosystem Services in Respect of Biodiversity Targets

As explained by Boyd and Banzhaf (2007), "as end products of nature, final ecosystem services are not benefits nor are they necessarily the final product consumed. For example, recreation is often called an ecosystem service. It is more appropriately considered a benefit produced using both ecological services and conventional goods and services (Fig. 23.1).

Biodiversity is frequently treated as being synonymous with taxonomic diversity, which is usually tabulated as the number of species observed in an ecosystem. However, while individual species do play a major role towards the provision of a particular ecosystem services, the biodiversity that supports these services is generally functional diversity, not species richness (Kremen and Ostfeld 2005).

Ecosystem services generally depend on the maintenance of functional diversity. The taxonomy of species present in a given ecosystem is less relevant to the functioning of that ecosystem than the functional traits those species possess. For this reason, the study of biodiversity and ecosystem functioning has recently focused more on functional rather than on taxonomic diversity and several important studies have shown how traits can be used to understand the relationship between biodiversity and ecosystem functioning (Barrett 1994; Solan et al. 2004; Butchart et al. 2010; Kattge et al. 2011).

How much diversity is needed in ensuring ecosystem services depends on the range of environmental conditions expected. The greater the expected variation in those conditions, the greater the required diversity within functional groups will be (Elmqvist et al. 2003).

The Millennium Ecosystem Assessment (2005) defined ecosystem services as the benefits that people obtain from the functioning of ecosystems. Ecosystems that are managed for a single service – such as the production of food, fuel, or fibre, or the control of particular pests or pathogens – frequently lose the services provided by the species removed in the process (Naeem et al. 2009). One noteworthy advance in the 2020 targets over the 2010 targets is the recognition that the failure to address interrelationships and interdependencies between the services produced by particular ecosystems having unique social origins (Barrett 1994, 2003).

Coastal-estuarine- mangrove ecosystem services include supporting and regulating services, provisioning services, and cultural services, as defined in the Millennium Ecosystem Assessment (2005). It is sometimes difficult to recognise ecosystem services and to quantify them accurately, partly because they often provide indirect benefits, meaning that they remain poorly understood in relation to their importance (Myers 1996). Constanza et al. 1997 estimated the global value of biodiversity to be roughly $38 trillion, although this remains a highly controversial figure. Using a careful analysis of existing case studies, Balmford et al. (2002) found that the benefits of conversion of land (and subsequent loss of ecosystem services) were always outweighed by the costs. In each case, private benefits were accrued at the cost of social (community) benefits (Figs. 23.2 and 23.3).

23.1.4 Marine- Coastal-Estuarine-Mangrove Ecosystem and Their Ecosystem Services

Marine- coastal- estuarine- mangrove ecosystem represents the most productive and dynamic ecosystem of the world. It supports innumerable number of flora and fauna in its diversified habitats and ecological niches. However, the tremendous pace of development (industrialization, urbanization etc.) has contributed a lot towards the loss of biodiversity in several sensitive, vulnerable and productive ecosystems of the world including the coastal one. Estuaries are ecologically important coastal environments situated between fresh water rivers and the sea, characterized by highly

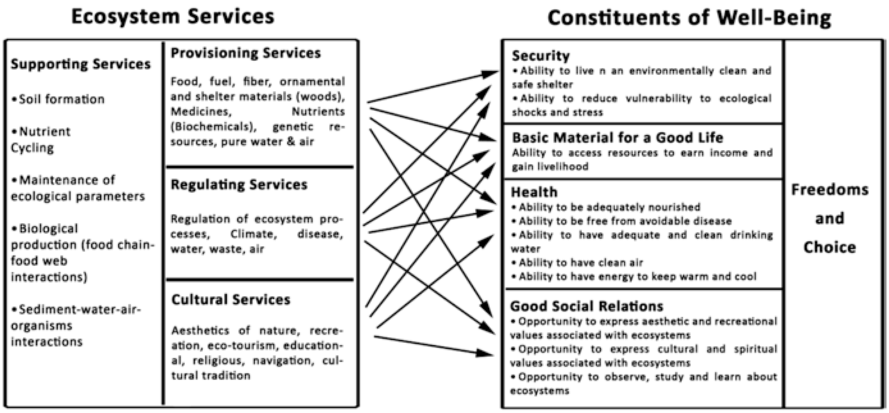


Fig. 23.2 Millennium Ecosystem Assessment (Zakri 2003)

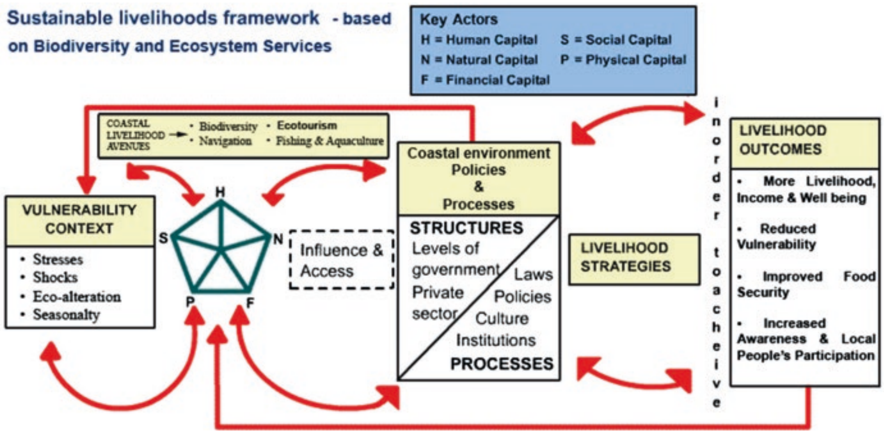


Fig. 23.3 Linkages among governance, biodiversity and ecological services for sustainable livelihood generation

varying physicochemical, morphological and hydrological conditions (Carter 1988; Ysebaert and Hermen 2002), which exhibits some of the most biologically productive habitats on earth (Kennish 2002; McLusky and Elliott 2004).

Marine ecosystems collectively form the largest ecosystem on Earth and yield some of the most critical ecosystem services. One well-known service is the provision of food in the form of fish, and the relationship between harvested marine diversity and ecosystem services is well documented (Jackson et al. 2001; Worm et al. 2006). But the oceans provide a wide range of ecosystem services that are less visible, though no less important. For example, the regulation of climate through marine biogeochemical pathways is roughly equivalent to terrestrial contributions (e.g. an estimated 92.2 Gt. of carbon per year entering the ocean from the atmosphere and 90.6 Gt of carbon per year entering the ocean to the atmosphere (Denman et al. 2007).

Another important, but often overlooked, service is the production of oxygen: it is estimated that one out of every two breaths' worth of oxygen that we take is produced directly by marine phytoplankton (Behrenfeld et al. 2006). Marine microorganisms also degrade or purify very large amount of wastes that have been intentionally dumped into the sea for decades, such as the >10 trillion liters of domestic sewage released annually in the US alone (NRC 1993). Marine biodiversity is affected by warming, acidification, altered upwelling and stratification patterns, and increased variability (Worm and Lotze 2009; Koch et al. 2006).

23.1.5 Intertidal Macrobenthic Fauna: Functional Relationships

Benthic fauna represents an important and essential structural component of any marine and estuarine ecosystem which includes some of the high biodiversity areas in its several specialized habitats like salt marshes, coral reefs and mangroves and each habitat has specific benthic community (Chakraborty 2009, 2011; Chakraborty et al. 2012). The assemblages of intertidal benthic fauna are structured by arrays of ecological parameters, both living and nonliving ones which operate in an intricate manner resulting in multidimentional relationships.The biotic relationship includes inter- and intraspecific competition, feeding and predator-pray interactions, the production of biomass, and the production and delivery of recruiting stages (Gray and Elliot 2009). Functional studies of ecosystems after being initiated by Lindeman (1942), has gained momentum during last few decades which dealt mostly with the trophic inter relationships and flow of energy in the form of primary production, secondary production etc. (Elliot and Taylor 1989). Production represents the increase of biomass (organic matter) by organisms whether that organic matter is accumulated for growth and reproduction leading to the development or eggs, sperms and larvae. The larger and more abundant species are important for human consumption while the incredible variety of small species contribute for ensuring complexity and functioning of tropical ecosystems (Hendrickx 1995). A closer examination of almost any coastal beach, marine and estuarine, shows the signs of a galaxy of life that exists beneath its surface, the evidence of which is laid down by their highly dramatic activities, with a mass of changing contours caused by various organisms. Traditionally, the fauna of marine sediments is classified into meio- and macrofauna by the use of defined sieve sizes, whereby organisms retained on a 0.5 or 1 mm mesh sieve are referred to as macrofauna (McIntyre 1969; Rees 1984; Bachelet 1990; Ditmann 2000).

The study of intertidal macrobenthic fauna has been well attempted and understood in recent past as a basic component of the trophic interactions. The benthic fauna produce millions of larvae in the form of meroplankters which not only support fish populations but also maintain the ecological equilibrium. Nearly half of the world's commercial fish catch come from the sea and estuary which supports the lives of a large number of shell fishes and demersal fishes whose main food items are derived from the benthic animals. Mangrove crabs play an essential role for leaf litter degradation in these systems (Robertson 1986; Micheli 1993a, b; Alongi 2002, 2008; Chatterjee and Chakraborty 2014; Chatterjee et al. 2014). The diversity and abundance of benthic fauna has positive relationships with demersal fisheries and aquaculture (Chakraborty 1996). Similarly, the intertidal and benthic macrofauna are very sensitive to environmental stress hence water quality biologists use them to study the environmental changes (Washington 1984; Stark 1998).

The assemblage of different organisms of different sorts is commonly known as *community*. Density and diversity of species are the two important criteria in a community, which could probably assess not only the nature of habitat but also indirectly reflect the mode of interactions between various abiotic and biotic components of the environment. The concept of ecological niche pervades all of ecology (Giller 1984). It arose as an attempt to describe the total role of a species in a community, defining all the bonds between population, community and ecosystem. Grinnell (1971) used the term niche as a habitat concept, defining the ultimate distributional unit of a species, Odum (1971) defined that the *habitat is the dwelling place* of a population and *niche as its profession.* Hutchinson (1958) on the other hand, considered the niche to be defined by the total range of environmental variables to which a species must be adapted (physical, chemical and biological), and under which a species population lies, and replaces itself indefinitely. Every pertinent environmental variable thus can be considered as a gradient along which the species has an activity or tolerance range.

This multidimensional approach provides a means of conceiving how species relate to one another and has thus enhanced the interpretation of community organization. Most organisms do not inhabit their potential fundamental niche, but, due to interaction with other organisms, occupy a reduced, realized niche. The major interactions are considered to be predation and competition. Species tends to share parts of each other's fundamental niches, resulting in simultaneous demands upon some resources by two or more species populations. In Hutchinson's terminology, the niche hyper volumes of species include parts of others, thus overlap. The natural effects of inter specific competition include competitive exclusion, niche shift (O'Conner et al. 1975; Odum and Barrett 2005), character displacement (Grant 1975; Martinez 2004) and changes in resource level (Rusterholz 1981).

23.1.6 Benthos-Sediment Interaction: Bioturbation (Fig. 23.4)

Sedimentary environments are dynamic habitats where the sediments, the fundamental building blocks of the habitats, are continually structured by the local physical regime. In fine sands to mud, biotic structuring of the habitat by burrowing, tube building, defecation and secretion often is significant (Rhoads and Boyer 1982; Woodin

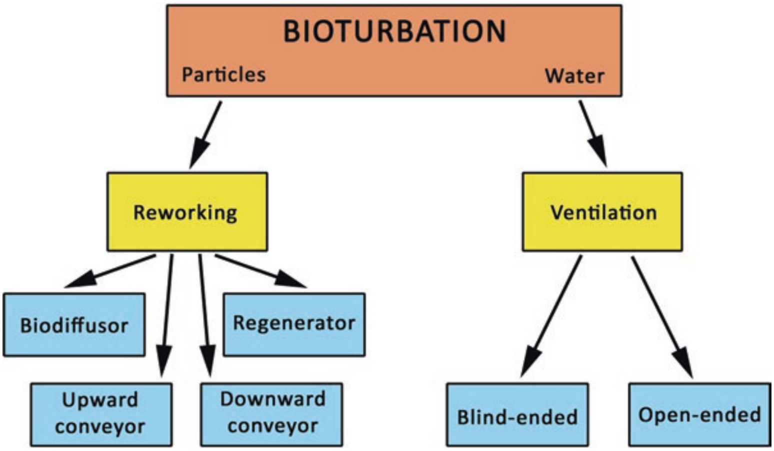


Fig. 23.4 Diagrammatic representation of 'Bioturbation' and its different components (Kristensen et al. 2012)

1999). This biogenic process collectively termed as 'bioturbation', is of special importance in the cycling of trace metals, nutrients and pollutants, including radionuclides, between shallow sea sediments and the water column (Zeitzschel 1980; Kremling 1983; Nolting 1986; Swift 1993; Wilkinson et al. 2007). Bioturbators include a wide array of benthic fauna like crustaceans (such as burrowing crabs and shrimps), polychaete worms, molluscs, echinoderms, brachyopods, sipunculans, cnidarians, priapulida and other meiobenthic faunal organisms (Woodin 1974, 1978, 1981).

Deposit- feeders are the most prominent group of bioturbators as they constantly process sediment for food, resulting in horizontal and vertical movement of particles in the sediment. In some habitats, biogenic alterations affect the erodibility of the sediments (Luckenbach 1986; Meadows and Tait 1989; Kihslinger and Woodin 2000), the distribution of grain sizes (Rhoads and Stanley 1965; Cadee 2001; Lim 2006), and the concentrations of pore water constituents (Aller 1982). Burrowing and defecation can affect infaunal recruitment patterns (Posey 1986); infaunal secretions can alter the chemical constituents of the sediment (King 1986; Woodin 1981). Biogenic structures can alter the ability of individuals to burrow through the sediment (Brenchley 1982). Biogenic modifications therefore appear to affect the distribution and abundance patterns of infauna. Defecation by large infauna has a negative effect on other infauna, causing reduced growth rates, reduced recruitment and increased emigration (Brenchley 1976). The patterns of distribution and abundance of organisms are frequently correlated with the condition of habitat heterogeneity changes which in turn lead to changes in the abundance of physical and biogenic structures (Martinez 1992; Lee 2008).

The present chapter is aimed at reviewing and highlighting the ecological services which are being rendered by virtue of ecological interactions of intertidal benthic biodiversity of the coastal environment of a very short but eco-bio-potential coast in the Midnapore (East), District, West Bengal, India.

23.2 Benthic Fauna of Midnapore (East) Coast and Ecological Services

Intertidal benthic fauna, both macro and meiobenthos represent a major biodiversity component of costal environment and renders valuable ecological services by virtue of their sheer diversity, biomass, community interactions, synchronising their survival strategies within changing and unstable habitats. Different aspects on these points are being discussed below.

23.2.1 Uniqueness of Midnapore (East) Coast: Global Perspective

The world's coast line is about 440,000 km long (Pethic 1984, 1992) and 66% of the world's population resides within a very close vicinity of the coastal environment. By virtue of possession of so many geo-morphological units like continental self, intertidal belts, dunes, deltaic islands with mangroves, back water and brackish water zones, lagoons and estuaries, the coastal zone governs regional climatic conditions, food production, harvesting of energy resources, human settlements, industrialization etc.

Coastal zone includes both the area of land subject to marine influences and the area of the sea subject to land influences. Coastlines are formed by morphological changes governed by climatic and geological processes. They constitute a transition zone where land and freshwater meet saline water, and across which the effects of land on the ocean, and vice versa, are transferred and modified. Among the several definitions on coastal zones, the following is one of the most accepted one "the inland extent of coastal ecosystem is defined as the line where land-based influences dominate up to a maximum of 100 km from the coastline or 50-m elevation (whichever is closer to the sea), and with the outward extent as the 50-m depth contour. Marine ecosystem begins at the low water mark and encompass the high seas and deepwater habitats" (Millennium Ecosystem Assessment 2005).

Coasts are of great ecological and socioeconomic importance. They sustain economies and provide livelihood through fisheries, ports, tourism, and other industries. They also provide ecosystem services such as regulating atmospheric composition, cycling of nutrients and water, and waste removal (Jennerjahn et al. 2009). Coastal ecosystems are repositories of biological diversity and provide a wide range of goods and services (Kautsky 1981). The major habitats of the coastal zone are coral reefs; sea grass beds/meadows; coastal or barrier islands; rocky coasts; cliffs; intertidal rocky, mud, or salt flats; rock pools; sandy, pebble, or rocky beaches; dune systems; saline, brackish, and freshwater lagoons; estuaries and coastal river floodplains; salt marshes; mangrove forests and other unique coastal vegetation such as sand dune flora -all of which have been highly modified over millennia by human activities (Carter 1990).

Pethick (1992) stated that the environment for a given coastal landform consists of the energy inputs into the coastal zone and the inorganic and biological materials from which it is formed. Thus, the present form is the result of continuous adjustments to the environmental changes over the geological and historical periods in the regional physiographic and environmental settings. This indicates that the stability in coastal changes is a dynamic equilibrium, which takes place in temporal and spatial scales. Tidal current ridges, channels, offshore bars, sand banks and also the islands over the shallow seas are the major topographic features in the coastal zone (Hughes et al. 1998; Paul 2002).

Different sub zones within the boundaries of coastal zone are significant for their acceptance of various energy inputs and therefore show the reflection of various topographic forms. Coastal interface or the direct interacting zones between materials and energy inputs, occupy a large space in West Bengal for its macro tidal environment, enormous water course and the flatness and lowland nature of tidal flats. However, on the inland tidal basins within estuarine sections and on the tidal flats particularly in the macro tidal environment, the extension of near shore zone includes intertidal zone and subtidal zone under the rise and fall of tidal levels twice daily (Paul 2002; Bird 2008).

Indian coast has a land frontier of 15,200 km. Coast line stretches about 5700 km on the main land and about 7500 km including the two island territories and exhibits most of the known geomorphological features of coastal zones. Presently Indian coastline is facing increasing human pressures which have resulted in substantial damage to its ecosystems (Qasim et al. 1988).

The coastal area of West Bengal extends over 0.28 million hectors and 220 KM of coastal line. The coastal belt of Midnapore district, West Bengal, India sharing 27% (60 KM) of coastal tract of West Bengal, India (Fig. 23.5) is a contiguous part of deltaic Sundarbans Mangrove Ecosystem – a world Heritage site, extending along the Hooghly estuary from New Digha (at the confluence of Subarnarekha with Hooghly) to the extreme south-west point of Midnapore district and then curving around Junput, Rosulpur, Khejuri and Haldia on the east to further north east upto Tamluk or even Kolaghat on the bank of Rupnarayan (Chakraborty 2010).

The coastal plain of West Bengal reveals a long history of deposition and seaward advancement over the Bengal Basin tectonic setting by neotectonics and concomitant sea level changes in this physical domain (Paul 2002). The coastline of West Bengal is at present being eroded with the site of sedimentation shifting back to inland and the coastal plain streams have started filling their own valleys with sandy sediments perhaps as a response to a recent, slow and small rise in sea level in the region. Everywhere along the coast, sand cliffs and clipped clay banks characterize the back shore and beaches experience an alarming rate of erosion (Paul 2002).

The inner boundaries of the coastal plains of Midnapore (East) littoral tract represent the corresponding shorelines of Post-Pleistocene high stand of restored sea level (Paul 2002). The coastal tract of Midnapore is characterized by sand dunes, long shore currents, high salinity, less turbidity and less vegetational coverage (Mukherjee and Chatterjee 1997; Chakraborty 2010) in comparison to its

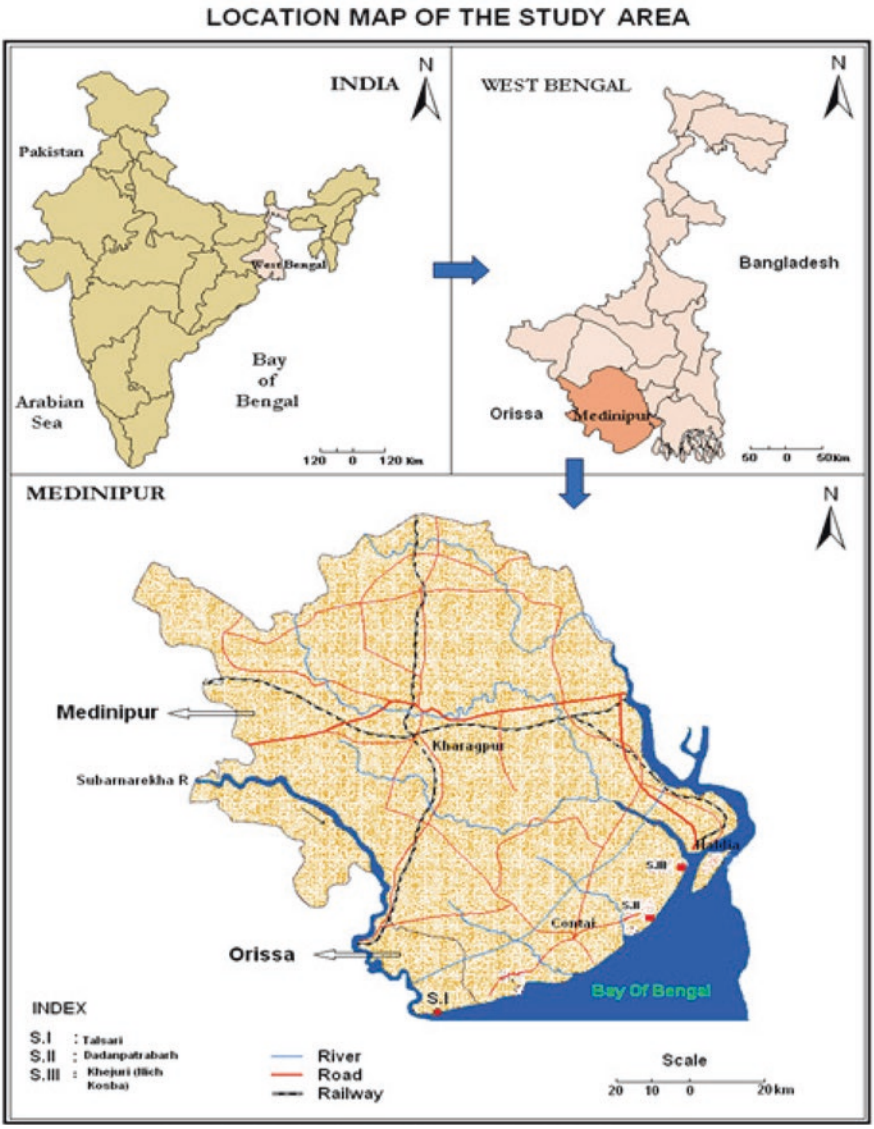


Fig. 23.5 Location map of Midanpore (East) Coast, West Bengal, India

counterpart, the South 24-Pargana district, supported by Sundarbans Mangrove ecosystem (Mukherjee and Bakshi 1998).

This coast is dominated by a high energy macro-tidal environment with waves produced by a long fetch across the Bay of Bengal, coupled with predominant southwesterly monsoon winds blowing onshore and also easterly and southerly winds encouraged by the visiting cyclones over the Bay-head coast in the summer months. The sandy coasts of Midnapur littoral tract have been confronted with a number of coastal issues (Sea Level Rise, Coastal Erosion Control Measures, Storm Hazard Mitigation, Environmental Refugees, Coastal pollution, Industrial Aquaculture, Navigability of the Hooghly Estuary, Coastal Recreational Exploitation etc.) which have created conflicts between various resource users and interest groups, between developers and ecologists, engineers and geoscientists and landowners and economists in West Bengal and in parts of northern Orissa (Paul 2002).

23.2.2 Habitat Diversity Vis-à-Vis Biodiversity in Midnapore Coast

This ecotone includes diversified habitats having unique geomorphology, varied and productive biodiversities, vulnerable and unstable ecological settings; experiences a lot of environmental hazards (cyclone, tidal waves), and environmental perturbations (global warming, sea level rise, eutrophication); acts as a sink of sewage, oil, pesticides, heavy metals etc. Midnapore (East) coast sharing 27% of total coastal tract of West-Bengal (60 Km) and locating in the East coast of India at the confluence of Hooghly and Subarnarekha estuaries with the Bay of Bengal, supports the lives of diversified pelagic and benthic fauna along with very specialized halophytic flora in an array of habitats. This coast is unique and differs from the coastal belts of South 24 Parganas District in West-Bengal because of the presence of chains of sand dunes, littoral drifts, long shore currents, higher turbidity and salinity (Annon 2005).

23.2.3 Mangroves of Midnapore (East) Coast: Diversity and Threats

Mangrove ecosystem represents one of the most productive natural wetlands found in the intertidal zone of tropical and subtropical regions of the world (Macnae 1969; Chaudhuri and Choudhury 1994). This specialized ecosystem, dominated by intertidal salt tolerant halophytic vegetation and enjoying the influence of two high and two low tides a day, offers a dynamic ecosystem for bioresource development on one hand and maintenance of ecological balance through the protection of coastal line on the other (Chakraborty 2011). This detritus based coastal ecosystem is highly productive having an average productivity of 2500 mg cm^2 per day and plays an important role in the biogeochemical cycles of the coastal environment (Jennerjahn and Ittekkot 2002). The importance of mangrove ecosystem for its potential for fisheries and aquaculture development has received wide acceptance all over the globe mainly because of two reasons: Firstly, large quantities of energy, in the form of mangrove plant contributed detritus, is exported from the mangrove forest to open water bodies (Odum and Heald 1975) and positive correlation exists between

Fig. 23.6 Bioresource based ecological services and livelihood sustenance at intertidal belt of Midnapore coast

the extent of mangroves and total fisheries yield from adjacent water (Macnae 1974; Lee 1995), Secondly, profitable regional and international markets for high quality aquaculture products are available which sustain the livelihoods of considerable number of local people (Fig. 23.6). Owing to difficulties in accessing rigorous mangrove ecosystems, indepth ecological researches are limited in comparison to such studies on some other coastal ecosystems like coral reefs. However, their geographical distribution, floral and faunal surveys and potential for fisheries has prompted more research on mangroves in the last couple of decades (Ellison et al. 1999).

Mangrove areas have considerable environmental and ecological value, as they prevent erosion of coastlines, protect adjacent coral reefs, tropical cylones and tidal waves and sea grass beds from the input of terrestrial sediments and serve as nursery sites, feeding grounds and protection areas for many fish species, invertebrates, mammals and birds (Danielsen et al. 2005).

This specialized ecosystem is very productive and forms one of the bases of food chain in the coastal areas. This ecosystem also protects coastal population and supports coastal fisheries (Boesch and Turner 1984; Kathiresan and Qasim 2005). As mangrove ecosystem is structured by the combination of two sub systems-mangrove subsystem and its adjoining estuarine aquatic subsystem which always interact with one another by a unique physical process- the tide, the interactions always go on among different living and non living components resulting in continuous nutrient cycling (Chakraborty 2011). Mangroves grow better in areas of low wave-energy shorelines, river deltas, and floodplains where fine sediments, muds, and clays accumulate and peats will form (Odum et al. 1982) and can grow in waters from high-to-low nutrient concentrations. In removing nutrients from surface water, mangrove forests can be important nutrient sinks for an estuary. Fluctuating tidal water are important for transporting nutrients, controlling soil salinities, and dispersing propagules.

Historically and geographically the Midnapore (East) coast is a contiguous part of deltaic Sundarbans of global importance, limiting the Hoogli estuary on the western front (Fig. 23.5). Naturaly, this coastal tract was once covered by dense mangrove forests, the relics of which are still found which have been surviving after sustaining the century old anthropogenic pressures, especially in developing this coast as the main hub for coastal tourisms and also commercial fishing. Fifty seven species of mangroves and their associated plants, mostly comprised of *Avicennia officinalis, A. alba, Exocoeria agallocha, Acanthus ilicifolius, Sueda maritima, Salicornia brachiata, Rizophora mucronata and Ipomea pescaprae.* Dune growing plants such as Ipomea, Spinifix, Pandanus etc. play a major role to stabilise dunes, and act as buffer against erosion (Chakraborty 2010).

23.2.4 Mangroves: Ecological Services and Threats

Mangroves are coastal forests that include saline tidal areas along sheltered bays, estuaries, and inlets in the tropics and subtropics throughout the world. Around 50–75 woody species are designated as "mangrove," which is a term that describes both the ecosystem and the plant families (Ellison and Farnsworth 2001). In the 1970s, mangroves may have covered as much as 200,000 km^2, or 75% of the world's coastlines (Spalding 1997). But since then, at least 35% of global mangrove area has been lost, and mangroves are currently disappearing at the rate of 1–2% annually (Valiela et al. 2001; Alongi 2002).

The worldwide destruction of mangroves is of concern because they provide a number of highly valued ecosystems services, including raw materials and food, coastal protection, erosion control, water purification, maintenance of fisheries, carbon sequestration, tourism and recreation. For many coastal communities, their traditional use of mangrove resources is often closely connected with the health and functioning of the system, and thus this use is often intimately tied to local culture, heritage, and traditional knowledge.

Of the ecosystem services listed, three have received most attention in terms of determining their values to coastal populations. These include.

1. Their use by local coastal communities for a variety of products, such as fuel wood, timber, raw materials, honey and resins, fishes and shellfishes (crabs, molluscs etc.);
2. Their role as nursery and breeding habitats for offshore fisheries; and.
3. Their propensity to serve as natural "coastal storm barriers" to periodic wind and wave or storm surge events, such as tropical storms, coastal floods, typhoons, and tsunamis.

Although many factors contribute to global mangrove deforestation, a major cause is aquaculture expansion in coastal areas, especially the establishment of shrimp farms (Barbier and Cox 2003). Aquaculture accounts for 52% of mangrove loss globally, with shrimp farming alone accounting for 38%. Forest use, mainly from industrial lumber and woodchip operations, causes 26% of mangrove loss globally. Freshwater diversion ac counts for 11% of deforestation, and reclamation of land for other uses causes 5% of decline. The remaining sources of mangrove deforestation consist of herbicide impacts, agriculture, salt ponds, and other coastal developments (Valiela et al. 2001). The extensive and rapid loss of mangroves globally reinforces the importance of measuring the values of such ecological services, and employing these values appropriately in coastal management and planning.

In order to assign a value to ecosystem services rendered by mangrove ecosystems, a study has been conducted for Thailand by Barbier (2007), who compared the net economic returns per hectare to shrimp farming, the costs of mangrove rehabilitation, and the value of mangrove services. All land uses after being investigated over 9-years period (1996–2004), and the net present value (NPV) of each land use or ecosystem service was estimated in 1996 as US $ per hectare. The NPV arising from the net income to local communities from collected forest and other products and shellfish was $484 to $584 per hactare. In addition, the NPV of mangroves as breeding and nursery habitat in support of offshore artisanal fisheries ranged from $708 to $987 per hactre. More economic evidences of the protective role of mangroves were recorded against different natural disasters such as tropical storms,coastal cyclones etc. (Das and Vincent 2009). The ability of mangroves to stabilize sediment and retain soil in their root structure reduces shoreline erosion and deterioration (Wolanski 2007). Mangroves also serve as barriers in the other direction; their water purification functions protect coral reefs, seagrass beds, and important navigation waters against siltation and pollution (Wolanski 2007; Wolanski et al. 2009).

23.2.5 Diversity, Distribution and Adaptability of Intertidal Benthic fauna

Benthic component is the most wide spread habitat on earth and support high biodiversity which serve as key to ecosystem (Anvar Batcha 1997). The benthos refers collectively to all aquatic organisms, which dwell in, on or near the bottom of water bodies. The benthos encompasses a huge array of life with many phyla. Their distribution spans from the utmost depth of the ocean to the high tide level and from fresh water through estuaries to the hyper saline tropical coastal lagoons. Generally benthic communities are much more diverse in terms of species richness than those of pelagic realm. Traditionally the fauna of marine sediments is classified into microfauna, meiofauna and macrofauna by the use of defined sieve sizes. Three functional groups of benthos can also be recognized. They are the infauna, epifauna and hyperfauna i.e. organisms living within the substratum, on the surface of the substratum and just above it respectively. The benthic fauna being very significant faunal component of shallow water estuarine and coastal marine ecosystems in respect of diversity, density and biomass ensure longterm maintenance of ecological health and funtioning of such aquatic systems (Chaudhuri and Choudhury 1994; Chakraborty 2011). No aquatic system will function long without a healthy benthic community (Chaudhuri and Choudhury 1994; Chakraborty 2011).

The benthic faunal components in a mangrove ecosystem comprising of terrestrial, estuarine or marine groups, are adapted for stressful situation of widely fluctuating environmental parameters. Animals in such habitats acquire a certain degree of euryhalinity as an insurance against the fluctuating environmental condition especially the salinity. Of all the factors, salinity, perhaps, is the most variable components of the ecosystem which really exerts perceptible impact on the behaviour of the littoral mudflat community. The faunal groups are generally confined to hard substrate offered by mangrove vegetation and comprising of sessile (porifera, cnidarians), wandering species (horse shoe crabs), muddy substrate with burrowing forms (in-fauna consisting of polychaetes, brachyuran crabs, wood boring animals, mud burrowing bivalves and gobiid fishes) and forms errant on substratum (epifauna exemplified by oysters, barnacles).

Altogether 3122 species of fauna consisting of 711 insects, 546 fin-fish, 435 birds, 311 molluscs, 139 crabs, 85 reptiles, 71 mammals, 55 prawns, 13 amphibians and 7 fish parasites are on record from coastal region of India (Alfred and Ramakrishna, 2004). East coast is represented by maximum number of animal species (2,075) followed by Andaman and Nicobar Islands with 827 species and by west coast with 728 species. The reasons for the species richness in the east coast of India may be due to the presence of luxuriant mangrove forests and to the proper surveys conducted there, as compared to other coastal regions (Kathiresan 2004). In terms of species richness, Sundarban Mangroves of West Bengal ranks first with 1434 species representing 20 phyla followed by 914 species of Andaman and Nicobar Islands and 801 species of Tamil Nadu (Annon 2003; Alfred and Ramakrishna 2004). Previous observations revealed that Hooghly-Matla estuarine complex harboured a total of nine species of actiniaria (Bairagi 1998), 69 species of polychaetes (Misra 1995), 26 species of brachyuran crabs (Chakraborty et al. 1986) and 120 species of molluscs (Subba Rao et al. 1992).

23.2.6 Benthic Faunal Diversity of Midnapore (East) Coastal Belt

A galaxy of biodiversity components such as mangroves and its associated flora, pelagic fauna- fin fishes and shell fishes, zooplankton, benthic fauna- macrobenthos (actiniarians, brachyuran crabs, brachyopoda, polychaets and molluscs) and meiobenthos (gastroticha, kynorhinca, nematoda, herpacticoid copepoda, foraminifera etc.) are the happy residents of coastal tract of Midnapore (East).

Three hundred twenty-two species of fin fishes under 78 families and 11 orders have been reported forom the Midnapore(East) coastal belt that includes both pelagic and demerseal fishery resources (Yennawar et al. 2015), 13 species of prawns and shrimps (Annon 2005), 17 species of zooplankton (Manna et al. 2008), 48 species of mollusc under 3 classes, 15 orders and 36 families, (Khalua et al. 2003), 12 actiniarian species belonging to 2 classes, 3 orders and 6 families, 22 species of polychaetes under 16 genera and 6 families (Chandra et al. 2003), 55 species of plant dependant insects mostly under the orders of lepidoptera, coleotera, hemiptera, orthoptera etc. (Jana et al. 2013, 2014), have been found to inhabit in different forms of ecological habitats. Besides, two species of horse-shoe crabs viz. *Carcinocorpius rotandicaudata* and *Tachypleus gigas;* one species of holothruoid, one species of brachyopoda (Samanta et al. 2014a, b, c and 2015a, b) and one species of nemertean (Ghorai et al. 2014, 2015) have been recorded. All these faunal and floral inhabitants have been found to display varied patterns of succession, distribution, eco-dynamics in tune with the changing ecological gradients and also by enjoying definite ecological niches. Different life-forms by virtue of their survival strategies, behavioral manifestations and ecological interactions have tended to contribute remarkable ecological services towards sustainability of the entire ecosystem in general and biodiversity development in particular. (Chandra et al. 2003; Chandra and Chakraborty 2008; Khalua et al. 2003; Dey et al. 2007, 2008, 2010; Jana et al. 2013, 2014; Chakraborty et al. 2010) (Tables 23.1, 23.2, and 23.3; Figs. 23.7, 23.8, 23.9, 23.10, and 23.11).

The nature of diversity and abundance of different species through different seasons is mainly due to a combination of various factors like, marked salinity fluctuation, south-west monsoon accompanied by heavy rains and also the structural modifications of sediments (Chakraborty and Choudhury 1992; Chaudhuri and Choudhury 1994; Chakraborty 2011). In addition, effects of physiological stresses in the form of high sediment and water temperature, high turbidity coupled with poor light penetration, daily tidal variation leading to inundation and exposure, periodical desiccation and the availability of food cannot be overruled.

23.2.7 Ecology of Benthic Fauna: Community Interractions

In the estuarine ecosystem, benthos is one of the important structural elements of the food web and it plays a significant role in the ecosystem dynamics (Herman et al. 1999; Ysebaert and Herman 2002; Fischer and Sheaves 2003; Ellison 2008). Community concept in benthic ecology evoked much debate which centred on whether animal communities should be described on the basis of the substratum they occupy, i.e. the biotopes, or the hydrobiological parameters to which they are exposed. Ecological communities vary across different spatial and temporal scales (Andrew and Mapstone 1987; Ysebaert and Herman 2002).

The stable community structure and species composition at the intertidal and shallow water sites indicated that greater environmental rigorousness does necessarily imply less faunal stability. Differences in diversity among these sites were interpreted as evidence that biological accommodation was responsible for a reduction

Table 23.1 Occurrence and distribution of molluscan species in the coastal tract of Midnapore (East), West Bengal, India

Sl. No.	Species	Subarnarekha mouth	Shankarpur	Junput
	Class – Gastropoda			
	Order – Mesogastropoda			
	Family – Assimineidae			
1	*Assiminea brevicula*	++	–	+++
2	**A. francesiae (wood)**	–	–	–
	Family – Littorinidae			
3	*Littorina (Littoraria) melanostoma* Gray	+++	++	–
4	*Littorina (Littoraria) undulata Gray*	–	–	–
	Family – Potamididae			
5	*Telescopium telescopium*	++	+	++
6	*Cerithidea (Cerithideopsis) cingulata (Gmelin)*	+++	++	+++
7	*Cerithidea obtusa* (Lamarck)	++	_	+
	Family Epitoniidae			
8	*Acrilla acuminata* (Sowerby)	–	_	–
	Family Naticidae			
9	*Natica tigrina* (Roeding)	+	–	–
10	*Natica vitellus* (Linnaeus)	–	–	–
11	*Polinices didyma (Roeding)*	–	+	–
12	*Polinices tumidus (Swainson)*	+	–	–
	Family Tonnidae			
13	*Tonna dolium (Linnaeus)*	–	–	+
	Family viviparidae			
14	*Bellamya bengalensis f. typica* (Lamarck)	++	–	–
	Family Thiaridae			
15	*Thiara (Tarebia) lineata* (Gray)	–	–	–
	Order Archaeogastropoda			
	Family Trochidae			
16	*Umbonium vestiarium (Linnaeus)*	–	–	–
	Family Neritidae			
17	*Neritina(Dostia) violacea* (Gmelin)	–	–	–
18	*Neritina smithi Wood*	–	–	–
	Order Neogastropoda			
	Family Muricidae			
19	*Murex tribulus* (Linnaeus)	–	+	–
	Family – Nassariidae			
20	*Nassarius (Hima) stolatus* (Gmelin)	+	–	–
21	*Nassarius faveolatus* (**Mss. Dunker Reeve**)	–	–	–
	Family – Olividae			
22	*Amalda ampla* (**Gmelin**)	+	–	–
23	*Olivancillaria gibbosa* (Born)	+	–	–

(continued)

Table 23.1 (continued)

Sl. No.	Species	Subarnarekha mouth	Shankarpur	Junput
	Order – Soleolifera			
	Family – Onchidiidae			
24	*Onchidium tigrinum* (Stoliczka)	+	–	–
25	*Onchidium tenerum* (Stoliczka)	+	–	–
	Order – Nudibranchia			
	Suborder – Arminacea			
	Family – Arminidae			
26	*Armina sp.*	+	–	–
	Order – Basommataphora			
	Family – Lymnaeidae			
27	*Lymnaea (Pseudosuccinea) luteola* f. ovalis Gray	–	–	–
28	*Ellobium (Auricula) gangeticum* (Pfeiffer)	–	–	+
	Order – Entomotaeniata			
	Family – Atyidae			
29	*Haminea crocata* Reeve	–	+	–
	Class – Bivalvia			
	Order – Arcoida			
	Family – Arcidae			
30	*Anadara granosa* (**Linnaeus**)	+	+	–
	Order – Mytiloida			
	Family – Mytilidae			
31	*Perna viridis* (Linnaeus)	+	+	–
32	*Modiolus undulatus* (Dunker)	+	+	–
33	*Modiolus striatulus* (Hanley)	++	–	–
	Order – Veneroida			
	Family – Donacidae			
34	*Donax (Hecuba) scortum* (Linnaeus)	–	+	–
35	*Donax(Latona) incarnatus* Gmelin	–	+	–
	Family – Psammobiidae			
36	*Sanguinolaria (Soletellina) acuminata* (Deshayes)	–	+	–
	Family – Veneridae			
37	*Meretrix meretrix* (Linnaeus)	+++	+	–
	Family – Corbiculidae			
38	*Corbicula striatella* Deshayes	–	–	–
	Family – Solenidae			
39	*Solen brevis* Gray	–	–	–
	Family – Cultellidae			
40	*Neosolen aquaedulcioris* Ghosh	–	+	–
	Family – Tellinidae			

(continued)

Table 23.1 (continued)

Sl. No.	Species	Subarnarekha mouth	Shankarpur	Junput
41	*Macoma birmanica* (Philippi)	–	+	–
	Order – Myioda			
	Family – Pholadidae			
42	*Barnea condida* (Linnaeus)	–	+	–
	Order – Pterioida			
	Family – Ostreidae			
43	*Saccostrea cucullata* (Born)	–	++	–

in diversity from values predicted by the neutral model. In estuarine and marine benthos, biological accommodation has been inferred from high levels of diversity in unstressed communities (Gray 1981). But studies, both in estuarine and coastal environments, have documented wide variations in the density of benthic populations, from season to season and year to year (Poors and Rainer 1979; Levings 1975; Maurer et al. 1978; Koch et al. 2005; Uysal et al. 2002; Alongi 1988a). Estuarine communities have been characterised in having low diversity, lacking persistence and resiliency (Boesch 1977) because of a combination of low environmental constancy and predictability.

Several aspects of the ecology of the coastal and estuarine benthos and the interactions among them are already known from the contributions of (Alongi 1987; Underwood et al. 2000; Dye 2006; Macintosh et al. 2002; Lee 2008). The intertidal benthos are subjected to and interact with a number of ecological parameters such as sediment texture, salinity, pH, dissolved oxygen, nutrients etc. But these factors hardly act in isolation because of climatic influences on sea level which in turn govern the changes in vegetation structure, settlement of benthos and their food habits and so forth (Chakraborty and Choudhury 1989, 1992, 1993, 1994; Chakraborty 2011; Chakraborty 2013; Chakraborty et al. 2009, 2012).

The concepts and practices in the realm of ecology have been steadily evolving over the years in respect of the scales and variabilities of the habitats. The abiotic components of the habitats initially form the template and framework on which species might settle, compete and survive successfully leading to Clement's original monoclimax theory of successional change (Townsend et al. 2008).This paradigm has now changed with the development of concept that the organisms inhabiting an ecosystem have a range of effects on physical structure and dynamics of the system and thereby modify and shape the system in tune with their own requirements for sustenance (Hansell 2005). This represents the aquatic depositional system where organisms burrow, restructure and process the materials of their surroundings in a process known as bioturbation (Reise 2002). Such beneficial functions or processes rendered by the biota are being recognised as valuable and critical services, more precisely known as ecosystem services (Chapin et al. 1997).The biotic component of the system is often reported as some measure of the variety of species that contribute to the process, under the general term of "biodiversity".

This has led to the question-How does biodiversity affect ecosystem function?

Table 23.2 Distribution of polychaetes in the coastal tract of Midnapore (East) District, West Bengal, India

Sl. No.	Species	Subarnarekha mouth	Sankarpur	Junput
	Class: Polychaeta			
	Sub class: Errantia			
I	**Family – Phyllodocidae**			
1	*Eteone oranata* Grube	+++	–	+
2	*Eteone barantollae* Fauvel	+++	–	–
II	**Family – Glyceridae**			
1	*Glycera rouxii* Audouin and Edwards	++	+	–
2	*Glycera tesselata* Grube	+	–	–
3	*Glycera alba* Rathke	++	–	–
4	*Scoloplos sagarensis* Misra	+	+	+
III	**Family – Nereidae**			
1	*Perinereis cultrifera* Grube	–	–	+ ++
2	*Perinereis nuntia* Saviguy	–	–	+++
3	*Neanthes chingrighattensis* Fauvel	+	–	++
4	*Namalycastis indica* Southern	–	–	–
5	*Namalycastis fauveli* Rao	–	–	–
6	*Lycastoneries indica* Rao	–	–	+
7	*Neries glandicincta* Southern	++	–	–
IV	**Family – Hesionidae**			
1	*Talehsapia annandalei* Fauvel	–	–	++
V	**Family – Lumbrinereidae**			
1	*Lumbrinereis polydesma* Southern	++	–	–
VI	**Family – Eunicidae**			
1	*Diopatra cuprea* Bosc	+++	+	–
	Sub class: Sedentaria			
VII	**Family – Capitellidae**			
1	*Mastobranchus indicus* Southern	–	+++	+
2	*Paraheteromastus tenuis* Monro	–	++	–
VIII	**Family – Spionidae**			
1	*Polydora ciliata* Johnston	–	++	–
IX	**Family – Maldanidae**			
1	*Axiothella obockensis* Gravier	+++	–	–
2	*Maldane sarsi* Malmgren	+++	–	–
X	**Family – Oweniidae**			
1	*Owenia fusiformis* Delle Chiaje	++	–	–

The biodiversity-ecosystem function debate is therefore a highly active area of research and shedding further light on the dynamics and functional role of depositional systems (Solan et al. 2006).

Intertidal depositional system being the most physically challenged and stressful environments (Kaiser et al. 2005) for the residing/inhabiting biota as they are always subjected to surging waves and tidal fluctuations leading to highly variable ecologi-

Table 23.3 Cniderian diversity of Midnapore (East) Coast

Sl. No.	Species	Subarnarekha mouth	Sankarpur	Junput
	Phylum: Cnidaria			
	Class: Scyophozoa			
	Order: Rhizostomeae			
	Family: Catostyiodae			
1	*Acromitus flagellarus*(Haeckel)	–	+	–
2	*Acromitus rabanchatu*(Annandale)	–	+	–
	Phylum: Anthozoa			
	Class: Octocorally			
	Order: Pennatularia			
	Family: Virgulariidae			
3	*Virgularia elegans* (Gray)	+	–	–
	Family: Veretillidae			
4	*Cavernularia elegans* (Herklots)	+	_	–
5	*Cavernularia obesa* (Valenciennes)	+	+	_
	Sub class: Hexacorallia			
	Order: Actiniaria			
	Family: Edwardsiidae			
6	*Edwardsia jonesii* Seshaiya and Cuttress	+	+	+
7	*Edwardsia tinctrix* (Annandale)	+	+	+
	Family – Actiniidae			
8	*Paracondylacytis indicus* Dave	+	+	–
	Family – Diadumenidae			
9	*Diadumene Schilleriana* (Stoliczka)	+	–	–
	Family – Haliactiidae			
10	*Phytocoeteopsis ramunnii* (Panikkar)	+	+	–
11	*Palocoetes exul* (Annandale)	+	–	+

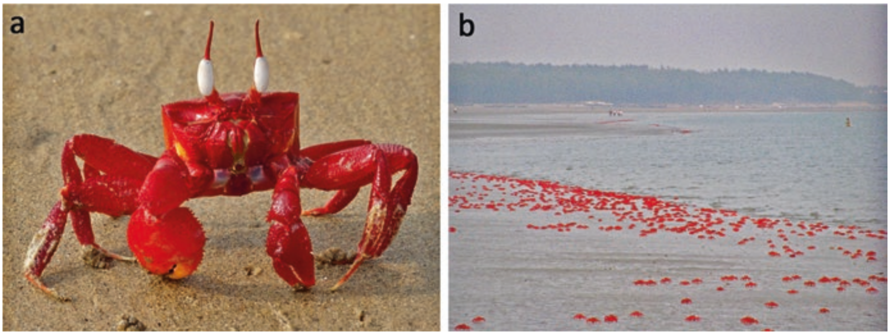


Fig. 23.7 Agglomeration of several individuals of ghost crab, *Ocypoda macrocera* and their onward movement towards tidal water margin for releasing larvae; the same crab in larger dimension

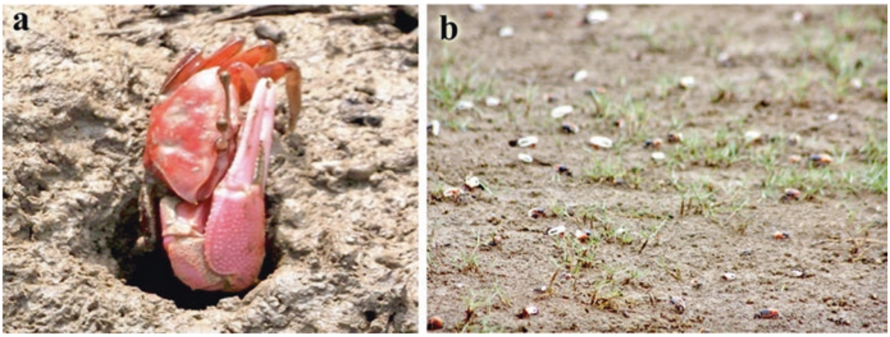


Fig. 23.8 Intertidal belt with saline grass and sandy-clayey sediment, packed up with fiddler crab, *Uca annulipes annulipes*; another species of fiddler crab, *Uca acuta acuta* in larger dimension

cal changes in terms of physicochemical parameters and making the living conditions very harsh for the organisms (Alongi 1990; Chakraborty 2011) Under such condition where resources are available but the habitat is complex and under the influence of severe physical factors, the coexistence of species becomes difficult leading to the restriction of variation of available niche space and thereby fewer species can compete for resources (Ricklefs and Miller 1999). The open structure of depositional habitats and the life style of the burrowing organisms are considered to further reduce the influence of interference and competition between species (Nybakken and Bertness 2005).

Each of the sandy and muddy depositional ecosystems has their own characteristic attributes that influence the metabolic processes and transformations that occur within them. The transformations of matter or energy driven by biotic assemblages that occur within natural ecosystems are termed "Ecosystem functions" while those functions that are rather subjectively, deemed to be important to humans are termed" ecosystem services" (Chapin et al. 1997).

Intertidal benthic animals in mangrove-estuarine system have been studied mostly for their trophic and bioturbating roles and their capacity for commercial fishing production. Different macrobenthic faunal groups viz. brachyuran crabs, polychaetes, molluscs in mangrove estuarine ecosystem of West Bengal displayed varying fluctuation trend with regard to their seasonal population density and community structure. The fiddlers crab's population density displayed its maximum during premonsoon followed by postmonsoon while many species of polychaetes have been found to occur in maximum density during monsoon. The species diversity indices taking into consideration of total benthic faunal groups were maximum during premonsoon and minimum during monsoon. However, such indices with polychaetous and molluscan faunal components registered maximum values during monsoon (Chakraborty et al. 1992a, b; Chakraborty and Chaudhury 1993, 1994; Chandra et al. 2003; Chandra and Chakraborty 2008; Khalua et al. 2003, 2008) (Fig. 23.12a, b).

The occurrence of macro and meiofaunal components is related to sediment properties, especially grain size attributes (Tietjen 1991). The diversity, individual size and biomass of macrofaunal organisms are often lower with higher mud con-

Fig. 23.9 Different species of brachyuran crabs. (**a**) *Ocypoda macrocera;* (**b**) *Uca acuta acuta* (male); (**c**) *Uca acuta acuta* (female); (**d**) *Uca lactea annulipes* (male); (**e**) *Uca triangularis bengali* (male); (**f**) *Uca triangularis bengali* (female); (**g**) *Sesarma (Chiromantes) bidens;* (**h**) *Metopograpsus maculatus*

tents (Dittmann 2000) but dominance can be high for few taxa (Polychaeta, Brachyopoda, Actiniarians) tolerant to anoxic conditions and high hydrogen sulphide concentrations (Grassle and Grassle 1974). Sediment properties can also affect the burrowing behaviour and distribution pattern of crabs (Rossi and Chapman 2003; Chatterjee and Chakraborty 2014; Chatterjee et al. 2014).

Benthic distribution patterns are in fact not only determined by abiotic factors, but also biotic factors, and both vary with spatial and temporal scales (Alongi 1987;

Fig. 23.10 Diferent benthic forms- (**a**) Brachiopod; (**b**) Polychaete; (**c**) Actinarian; (**d**) Horse-shoe crab

Fischer and Sheaves 2003).The occurrence and activities of benthic organisms can modify, stabilise or destabilise the sediment (Reise 2002). Animal-sediment interactions are often linked to the feeding modes of organisms. For example, deposit feeders with a high bioturbating activity can exclude suspension feeders, which rely on more stable substrates (Rhoads and Young 1970). Interactions between trophic groups and sediment properties do not only affect spatial patterns, but ultimately the dominance of certain feeding modes in an area.

Benthic community structures are determined by the direct and indirect species interactions (Thrush et al. 1992). Species interactions are complex, as organisms can affect other organisms in a variety of ways: predation, competition, ammensalism, amelioration etc. Different bioturbatory structures such as burrows, worm tubes, mounds etc. create small-scale habitat heterogeneity below and above the sediment surface which are being used by other faunal components based on sediment amelioration representing positive ecological processes.

Several researchers have put forward their views regarding the influence of different ecological parameters on such spatial and temporal variation of density and community of macrobenthic species. Teal (1958) and Kinne (1963) advocated the role of salinity and temperature affecting the animal distribution in the brachish water zone. Abele (1974) mentioned that substrates are important in determining the species composition of the various habitats. A species can use one substrate as a shelter, another as a feeding site and other as a source of nutrition, thus reducing

Fig. 23.11 Nemertean species in its fully extended form

competitive interactions for each one resulting in appositive co action in community existence (Thrush 1991). Thus greater number of species can inhabit in a narrow intertidal belt, resulting in the change of species diversity. Sanders (1968) postulated the role of sediments and opined that most diverse communities will always be found in the tropics and deep sea because of the constancy of their soil environments. Nutrient enrichment in the littoral zones of the mangrove ecosystem play vital role in determining the settlements, growth, population fluctuation and community interactions of fauna (Heald and Odum 1970). Owing to the highly buffered nature of sea water, pH has no limiting effect on marine organisms (Michael 1984).

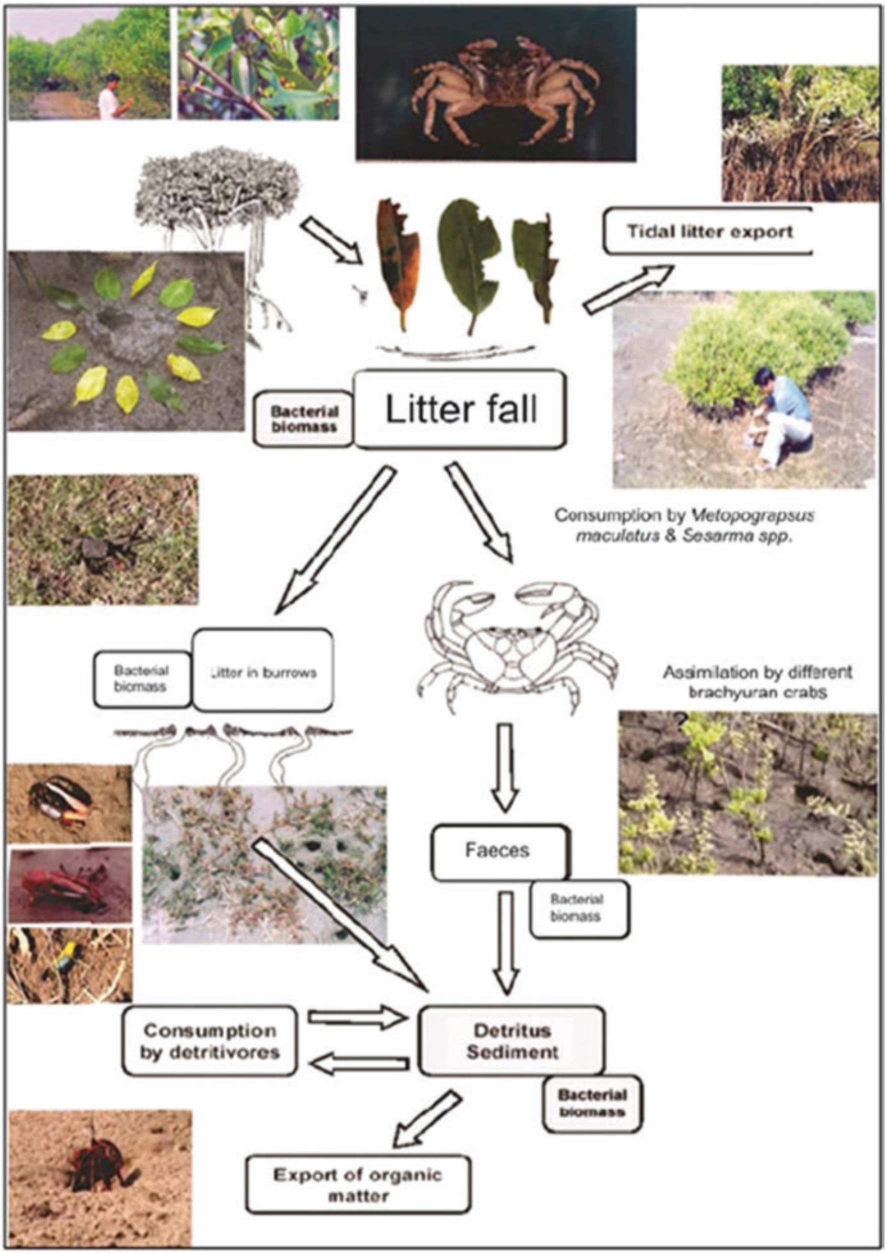


Fig. 23.12 (**a**) Schematic representation of the role of brachyuran crab in litter decomposition process; (**b**) schematic representation of mangroves leaves, its fate through the action of brachyuran crabs in the coastal tract of Midnapore; (**b**) schematic representation of biotransformation of Lipids through different trophic levels by a brachiopod

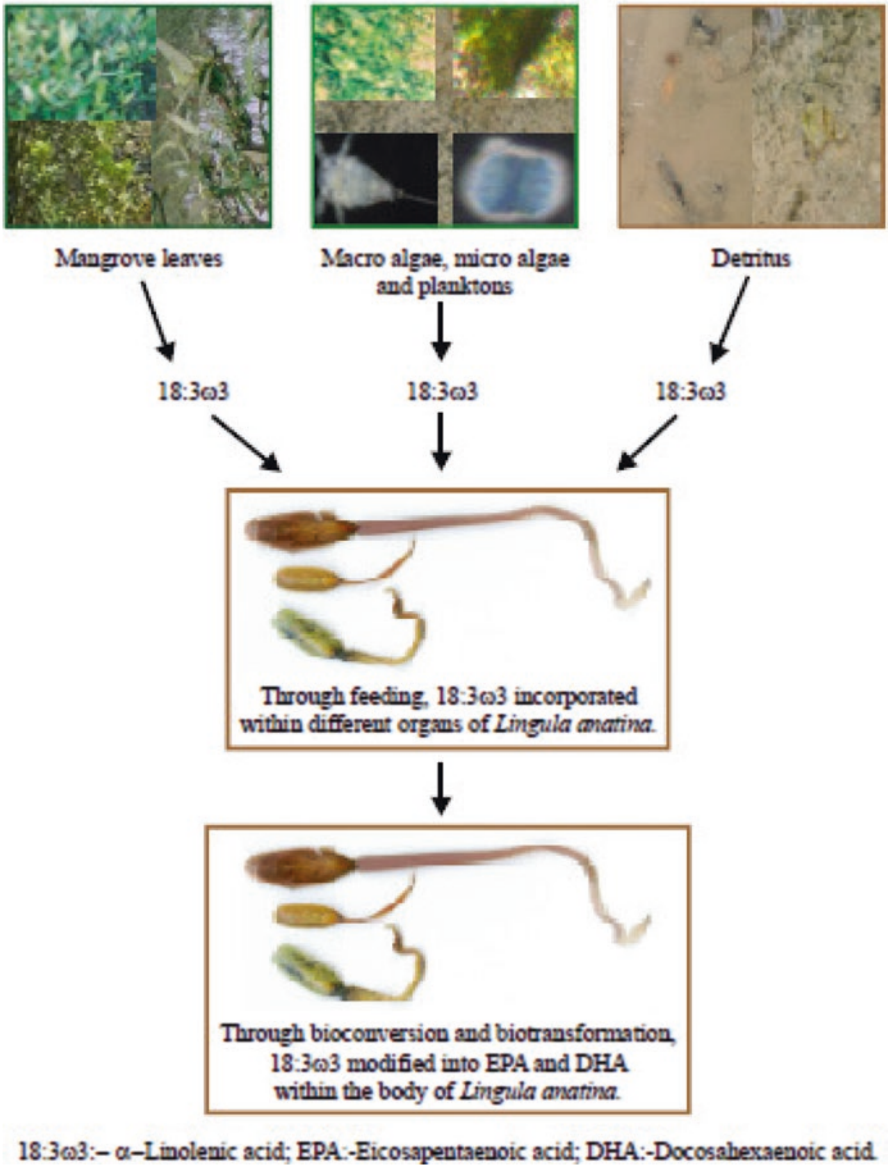


Fig. 23.12 (continued)

23.2.8 Meiobenthic Fauna: Diversity, Distribution and Ecological Services

About 60 species of meiobenthic fauna, mostly comprised of nematoda, foraminifera, polychaeta, herpacticoid copepod etc. have been recorded from the Midnapore coastal tracts which have been found to display distinct seasonal variations with respect to their diversity and density which also synchronises with fishery production vis-à-vis fish landings of the studied zones (Datta and Chakraborty 2015) (Tables 23.4, 23.5, and 23.6). These faunal components can be used to hypothesise the mutualistic dependence of meiobenthic fauna with fishery components and how such relationships may be used to decipher the underlying principles for developing tools for fisheries management (Heald 1969; Wolanski et al. 2009).

A total number of 51 soil microarthropods belonging to different insect's orders viz. Collembolan, hymenoptera, diptera and isopteran have been recorded from the different parts of this coast and they were found to play considerable roles in estuarine-mangrove nutrient cycling (Dey et al. 2007, 2008, 2010). Thirty seven species of meiobenthic foraminifera (Ghosh et al. 2014) and more than 20 species of nematodes (Datta and Chakraborty 2015) along with a number of oligochaetes, polychaetes, harpecticoid copepods, kinorhyncha etc. have been found to be present in this coastal belt (Datta and Chakraborty 2015; Datta et al. 2016) (Table 23.7; Figs. 23.13, 23.14, and 23.15).

In shallow marine and estuarine sediments, the coupling between the processes in the water column and the dynamics in the sediments is well documented. The benthic system comprehends a highly diverse community, composed of bacteria, micro-, meio- and macrobenthos, with the classification of benthic organisms generally relying on the organism size. The term "meiofauna" is actually derived from the Greek word *meio* meaning "smaller". Research on meiobenthic fauna have been known since the eighteenth century. The term "meiobenthos" was introduced and defined in 1942 by Mare in her account of the benthos of muddy substrates off Plymouth, England (Higgins and Thiel 1988) to indicate those benthic metazoans smaller than the 'macrobenthos', but larger than 'microbenthos'. In practice meiobenthic organisms consist of animals with size ranging from 63 to 500 μ (Coull 1999). The meiofauna are by no means a homogenous ecological group. There is a wide diversity of habitats in which meiofauna live. Meiofauna occur in both freshwater and marine habitats, from high on the beach to the deepest depths of the water body. Sediments of all kinds from the softest of muds to the coarsest shell gravels and all those in between harbor meiofauna. Meiofauna plays an important role in maintaining ecological balance by pray predator relationships within their community.

Meiofauna, mobile or hapto-sessile benthic invertebrates distinguished from macro benthos by their small sizes, shares considerable portion of total benthic biomass in marine habitats. These are exclusively important within any estuarine and marine systems since they facilitate biomineralization, support various higher trophic levels and show a high sensitivity to anthropogenic actions, thereby making

Table 23.4 Meiobenthic foraminiferan diversity in Midnapore (East), Coast, West Bengal, India

Species	Subarnarekha mouth	Sankarpur	Junput
Miliammina fusca	+	–	+
Reophax dentaliniformes	+	–	–
Cribrostomoides jeffreysi	–	+	+
Haplophragmoides canariensis	+	–	+
Ammobaculites agglutinans	+	+	+
Ammobaculites exiguus	+	+	+
Ammobaculites americanus	+	+	–
Arenoparrella mexicana	+	–	–
Trochammina inflata	+	+	–
Textularia agglutinans	+	–	–
Spiroloculina depressa	–	+	+
Spiroloculina indica	–	+	+
Quinqueloculina lamarckiana	+	+	–
Quinqueloculina seminulum	+	+	+
Quinqueloculina vulgaris	–	+	–
Quinqueloculina venusta	–	+	–
Triloculina brevidentata	+	+	+
Triloculina insignis	–	+	–
Triloculina trigonula	–	+	+
Lagena perlucida	+	+	–
Lagena interrupta	+	+	–
Glandulina laevigata	–	+	–
Bolivina seminuda	–	+	–
Bolivina striatula	–	+	+
Cancris auriculus	–	+	–
Cibicidoides wuellerstorfi	–	+	–
Nonion boueanum	–	–	+
Nonion scaphum	–	+	+
Nonionella labradorica	–	+	–
Nonionella turgida	+	+	–
Nonionella grateloupi	–	+	–
Hanzawaia concentrica	–	+	+
Ammonia beccarii	+	+	+
Ammonia tepida	+	+	+
Asterorotalia dentata	–	+	–
Asterorotalia inflata	–	+	–
Asterorotalia trispinosa	+	+	+
Asterorotalia multispinosa	+	+	+

(continued)

Table 23.4 (continued)

Species	Subarnarekha mouth	Sankarpur	Junput
Elphidium advenum	–	+	–
Elphidium crispum	+	+	+
Elphidium discoidale var. multiloculum	–	+	–
Elphidium hispidulum	+	+	+
Elphidium incertum	+	+	–
Elphidium somaense	+	+	–

Table 23.5 List of some meiobenthic nematofauna from the coastal area of Midnapore (East), West Bengal, India

Sl. no.	Taxa	Order	Family
1	*Bathylaimus*	Enoplida	Tripyloididae
2	*Mesacanthion*	Enoplida	Thoracostomopsidae
3	*Oncholaimellus brevicauda*	Enoplida	Oncholaimidae
4	*Onchlaimus*	Enoplida	Oncholaimidae
5	*Viscosia*	Enoplida	Oncholaimidae
6	*Halalaimus*	Enoplida	Oxystominidae
7	*Rhynchonema deghaensis*	Monhysterida	Xyalidae
8	*Daptonema*	Monhysterida	Xyalidae
9	*Theristus*	Monhysterida	Xyalidae
10	*Eleutherolaimus*	Monhysterida	Linhomoidae
11	*Terschellingia*	Monhysterida	Linhomoidae
12	*Megodontolaimus coxbazari*	Chromadorida	Chromadoridae
13	*Cyathoshiva amaleshi*	Chromadorida	Cyatholaimidae
14	*Dichromadora*	Chromadorida	Cyatholaimidae
15	*Odontophora*	Araeolaimida	Axonolaimidae

Table 23.6 According to Higgins and Thiel 1988 there are at least 22 phyla, out of 33 metazoan phyla recorded as meiofaunal composition worldwide

Sarcomastigophora	Oligichaeta	Cumacea
Ciliophora	Sipuncula	Halacarida
Cnidaria	Tardigrada	Pycnogonida
Turbellaria	Cladocera	Palpigradida
Nemartina	Ostracoda	Insecta
Nematoda	Mystacocarida	Bryozoa
Gastrotricha	Copepoda	Entoprocta
Rotifera	Syncarida	Brachiopoda
Loricifera	Thermosbaenacea	Aplacophora
Priapulida	Isopoda	Gastropoda and Bivalvia
Kinoryncha	Tanaidacea	Holothuroidea
Polychaeta	Amphipoda	Tunicata

Table 23.7 Species composition of microarthropods and their relative abundance (%) at Nayachar Island, Midnapore (East) Coast

S. no.	Soil microarthropods	Relative abundance (%)
Group	Acarina	36.3%
1	*Scheloribates thermophilus*	5.91
2	*Scheloribates parvus*	4.7
3	*Scheloribates praeincisus*	4.23
4	*Oppia sp*	2.97
5	*Multioppia sp*	2.27
6	*Tectocepheus velatus*	2.25
7	*Tectocepheus sp*	1.63
8	*Xylobates seminudus*	4.36
9	*Galumna flabellifera*	3.18
10	*Allonothrus sp*	1.23
11	*Masthermannia sp*	1.59
12	*Metabelba obtusus*	1.03
Group	**Collembola**	27.3%
13	*Hypozetes aculeifer*	0.95
14	*Isotomurus balteatus*	3.51
15	*Isotomiella minor*	2.46
16	*Isotoma sp*	0.83
17	*Proisotoma sp*	1.35
18	*Entomobrya sp*	2.55
19	*Sinella sp*	2.83
20	*Lepidocyrtus sp*	1.44
21	*Calx sp*	1.95
22	*Lepdocyrtus medis*	1.63
23	*Cyphoderus sp*	1.31
24	*Sminthurides appendiculatus*	3.39
25	*Sminthurides sp*	1.23
26	*Sminthurides aquaticus*	0.7
27	*Mesaphorura choudhuri*	1.07
28	*Cryptopygus sp*	0.95
Group	**Coleoptera**	10%
29	Family: Carabidae	4.26
30	Family Staphylinidae	4.35
31	Family Dytiscidae	1.39
Group	**Diptera**	5.7
32	Family: Mycetophilidae	3.47
33	Family: Tipulidae	1.53
Group	**Isopoda**	5.1%
34	*Philoscin sp*	3.79
35	*Procellionides sp*	1.31
Group	**Hymenoptera**	5%

(continued)

Table 23.7 (continued)

S. no.	Soil microarthropods	Relative abundance (%)
36	*Monomorium destructor*	1.94
37	*Monomorium floricola*	2.27
38	*Monomorium latinode*	0.79
39	*Pheidola roberti*	0.59
Group	**Other arthropods**	10.6%
40	*Marpissa sp*	0.43
41	*Artema sp*	0.47
42	*Uroctea sp*	0.51
43	**Centiped**	2.12
44	**Milliped**	1.11

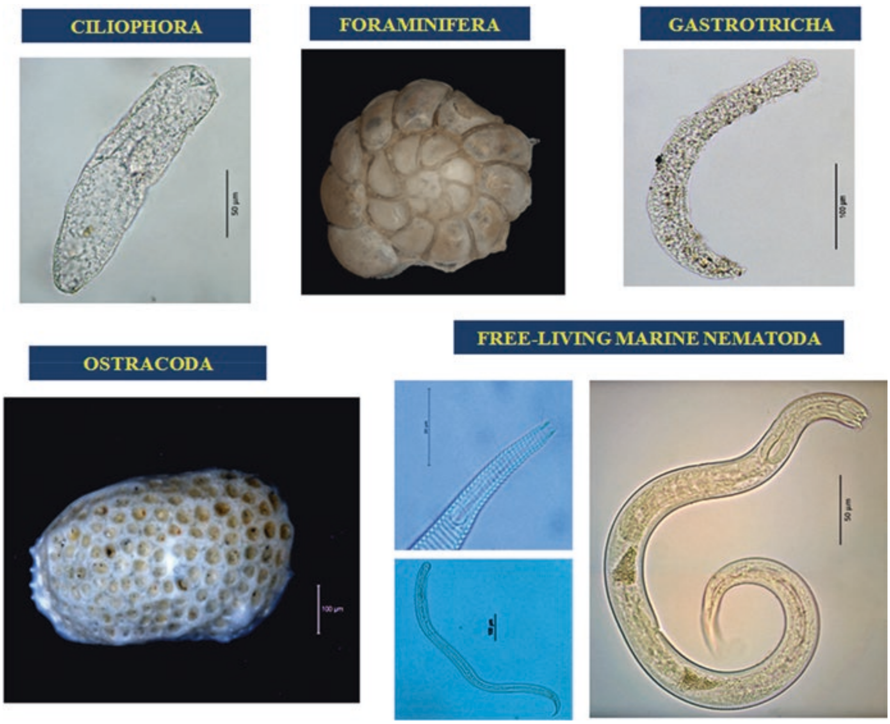


Fig. 23.13 Different reprenstatives of meiobenthic fauna

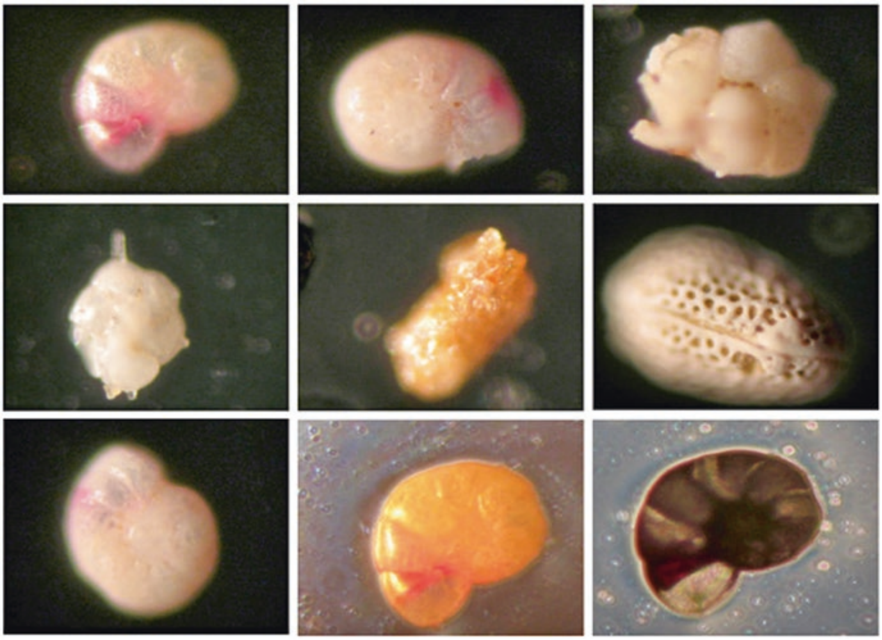


Fig. 23.14 Different representatives of Foraminifera

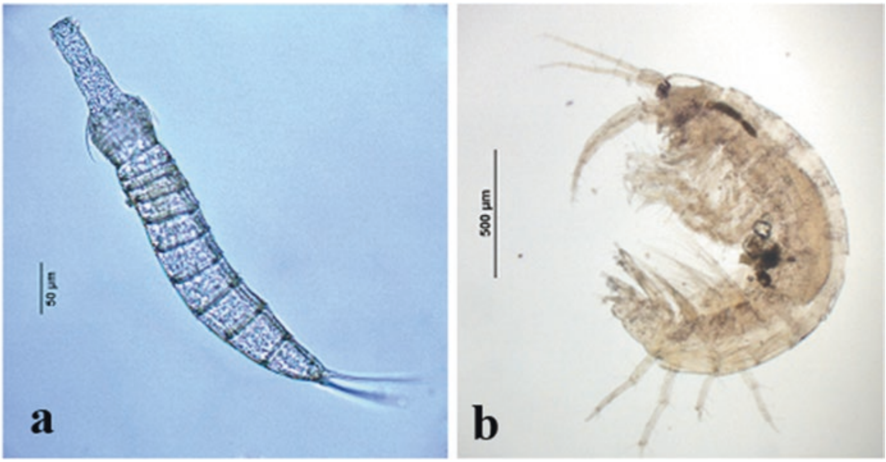


Fig. 23.15 Two meiobenthhic faunal group (**a**) Kinoryncha; (**b**) Amphipoda

them excellent organisms for pollution bio-monitoring. Besides, their large abundance attracts a considerable number of fin-fishes and shell fishes which visit the coastal belts in order to foreage in the benthic habitats, mostly in the intertidal and subtidal zones.

Intertidal meiofauna occuping the interstitial spaces between the sandgrains of sedimentary habitats represent the most diverse faunal component of marine benthos (Coull 1999). According to Higgins and Thiel 1988, there are at least 22 phyla out of 33 metazoans recorded as meiofaunal composition worldwide. The taxa belonging to meiobenthic fauna are Sarcomastigophora, Ciliophora, Cnidaria, Turbellaria, Nemartina, Nematoda, Gastrotricha, Rotifera, Loricifera, Priapulida, Kinoryncha, Polychaeta, Oligichaeta, Sipuncula, Tardigrada, Cladocera, Ostracoda, Mystacocarida, Copepoda, Syncarida, Thermosbaenacea, Isopoda, Tanaidacea, Amphipoda, Cumacea, Halacarida, Pycnogonida, Palpigradida, Insecta, Bryozoa, Entoprocta, Brachiopoda, Aplacophora, Gastropoda and Bivalvia, Holothuroidea and Tunicata (Table 23.6). A complicating factor in the taxonomy of meiofauna is not only by their small size, often associated with structural simplification, but also high percentage of morphologically similar or even identical species within related groups (Westheide 1991; Giere 1993).

Meiofaunal organisms can graze on bacteria and microalgae (Alongi 1988a, b) or pray on other meiofauna and juvenile stages of macrofauna (Watzin 1985). In turn, they are being used as novel sources of food for macrofauna as well as demersal fishes (Marinelli and Coull 1987) but top-down control of meiofaunal populations appears unlikely (Coull 1999).

There is also evidence that meiofauna play an important role in making detritus available to macroconsumers (Tenore et al. 1977). Meiobenthic fauna also serves as live food for higher trophic levels. A potential meiobenthic faunal food web includes macrofauna, meiofauna, swimming predators and meiofaunal food. Meiofaunal food is supposed to include diatoms, bacteria, detritus and perhaps protozoans and dissolved organic matter (Higgins and Thiel 1988). Meiobenthic fauna are very important within estuarine and marine systems since they facilitate biomineralization, support various higher trophic levels and show a high sensitivity to anthropogenic actions, making them excellent organisms for estuarine pollution bio-monitoring (Coull 1999). Meiobenthic organisms exploit the interstitial matrix of marine soft sediments and their small size and low mobility make them very sensitive to the biogeochemical pathways of their habitats. Since these meiobenthos shows high species diversity, short generation times and direct benthic development, they are considered a very good biological tool to monitor changes in the benthic environment (Kennedy and Jacoby 1999). As work on meiobenthic fauna in the East coast of India is very scattered, insufficient and inadequate, the present write up has been taken to document meiofaunal availability and diversity around the coast as well as to highlight the trend of their changes in respect of their diversity and distribution in the Midnapore (East), coast determines the changes with present scenario.

Prospects of coastal aquaculture as well as capture fisheries in a stretch of 7000 km. coastal tracts in India are primarily dependant on the natural supply of fish foods in the form of planktons, both phytoplankton and zooplankton, as well as different forms of benthos, the diversity and density of which are governed by the bio-geo-chemical cycling pertaining to mangrove-estuarine-marine ecosystems. By virtue of their roles as major contributors of coastal biological productivity, this interesting group of faunal component holds great promise in sustaining the coastal aquaculture and fisheries as many commercial fishes such as mullets, flatfishes, gobiids etc. predate on meiobenthic fauna, such as harpacticoid copepod, gastrotricha, nematoda, polychaeta etc. for their survival and growth. Selective feeding preferences on some groups of meiobenthic fauna by a particular fish species have been noted. The presence of considerable number of free-living marine nematodes in the gut of prawns reflects the food preference of such commercially important shell fishes (Datta and Chakraborty 2015).

23.2.9 Benthic Diversity and Assessment of Environmental Health

Taxonomy, the method of classification, is a language of communication for different biological fields by assigning internationally acceptable nomenclature to each taxa. In modern context, incorporating traditional taxonomic practices with advanced molecular techniques, the ultimate evolutionary relationship within the members of a clade can only be achieved. Not only in phylogentic study but also in the development of environmental and pharmaceutical biotechnological tools.

Traditionally macrofauna is used in the assessment of marine environmental health while meiofauna, the microscopic interstitial fauna are also being acted to give signal on environmental changes as bioindicators (Pradhan and Chakraborty 2008; Datta and Chakraborty 2015). This type of organism is potentially a crucial link as the interaction between pollutants and the biosphere occurs at low level of phylogenetic organization, carefully neglected from the biological spectrum. Marine benthic free-living nematoda being the most abundant metazoan meiofauna was proposed as an indicator for assessing the ecological quality of marine ecosystems according to the European Water Framework Directive (WFD). Not only in pollution monitoring, meiofauna also plays important roles in benthic community processes such as bioturbation (organic decomposition, nutrient cycling, redistribution of organic material, oxygenation of the sediment) and an effective link in food web. These organisms also act as indicator for global climate change. Meiofauna stimulate bacterial growth by mechanically breaking down the detrital particles, excrete nutrients or by producing slime trails through the secretion of mucus. The significant top-down control of meiofauna in microbial mineralization of polycyclic aromatic hydrocarbon such as naphthalene has already been proved using molecular tools like RFLP etc. High concentration of Sodium Channel Blockers (SCB), a group of neurotoxin such as tetradotoxin (TTX) and saxitoxin (STX), in free-living marine nematodes have already been confirmed using a tissue culture bioassay and their role in accumulation and transfer in marine environment has been proved significant. The analysis of mitochondrial cytochrome oxidase subunit 1 (COI) gene, nuclear rDNA, rRNA etc. are used generally to reveal the cryptic diversity, intra-genomic variation as well as identification of the meiofaunal groups and new procedures are still waiting to add the accuracy in phylogenetic anlysis. Different laboratory cultural procedures were developed for meiofauna, depending on their feeding and behavioural ecology. Therefore, smaller marine meiofaunal organisms like free-living nematoda, gastrotricha, ostracoda, kinorhynca, foraminifera, oligochaeta, nemaertea etc. can be effectively utilized in translational and regenerative biological research.

The benthic diversity regulates the physical, chemical, and biological environment of the estuary and link the sediment to aquatic food web, through their burrowing and feeding activities (Camilleri 1992). Filter feeders in the benthic community pump large amount of water through their bodies. As they filter this water for food, they clean the water by removing sediments and organic matter. Organic matter that is not used within the water-column is deposited on the bottom of the estuary. Deposit feeders then remineralize it into nutrients, which are then given back into the water column. This remineralization of organic matter is an important source of nutrients to the aquatic environment and is critical in maintaining the high primary production rates of estuaries (Giere 1993).

Many of the benthic organisms have pelagic larvae, a component of planktonic community and influence considerably in the planktonic food web. It is a well established fact that there exists always a nexus between the benthic standing crop and the production of exploited demersal fishes and crustaceans (Parulekar et al. 1980; Heald and Odum 1970; Zimmerman et al. 2000).

The response of estuarine-coastal ecosystem to the threats of ongoing environmental perturbations mainly because of pollution and associated environmental impacts are appeared to be complex and diverse. In order to assess the intensity of such impacts, only chemical and physical mode of assessment appear to be inadequate necessitating the adoption of biological methods to derive proper assessment results. The use of faunal diversity as an indicator of health is the most advantageous and cost-effective approach (Dauvin et al. 2003). Biomonitoring refers to the use of biological responses to assess changes in the environment that are often due to anthropogenic activities. The benthic fauna is the most amenable and suitable group to focus on this purpose (Warwick 1988; Warwick et al. 1987). These integrate many small negative effects and it is also an indicator of past transient events that may be missed by water quality monitoring programs (Maiti Dutta et al. 2014; Sanyal et al. 2015).

Benthic infaunal monitoring is widely accepted as the fundamental step to most recent interdisciplinary studies of contaminant effects on ecosystems. From the monitoring perspective, benthos offer mainly three positive attributes: (1) They are *relatively* sedentary and long-lived, the infauna cannot avoid exposure to contaminated sediments (2) They occupy an important intermediate trophic position, and (3) Infaunal communities are composed of a diverse array of species which respond differentially to varying environmental conditions, like high mortality of pollution-sensitive species and increasing abundance or frequency of pollution-tolerant species (Bilyard 1987; Weisberg et al. 1993). Responses of the infauna are representative of overall ecosystem status, because the infauna generally depends upon and interact with biological process in the water column (Warwick et al. 1987).

Use of an acceptable species checklist and taxonomic literatures, maintenance of voucher specimens, use of comprehensive sample storage procedure, expert check-ing of difficult benthic taxa etc. are considered to be the important prerequisites (Gray and Elliot 2009).

However, the measure of ecological change is never found to be exact but it can be precise and accurate within certain limits and the measurements are made for a reason.It is therefore imperative to reinforce the difference between precision and accuracy in recording marine benthic analyses (Gray and Elliot 2009). Moreover, the taxonomic sufficiency has occasionally been debated i.e. what level of taxonomic identification in such assessment by benthos is required to answer the questions being posed. The detection of large populations of opportunistic species (polychaetes) does not require high-level taxonomic separation (to the species level)-family- or order level identification can provide sufficient information (Warwick 1988; Gray and Elliot 2009). Dauvin et al. (2003) and Diaz et al. (2004) have advocated in favour of using lower level of taxonomic separation e.g. to order or family as sufficient for benthic impact assessment.

In addition to the knowledge base pertaining to the environment-biology relationships along with biology-biology relationships, it is imperative to have knowledge of the way in which the benthic biota modify and structure the sediments (the biology-environment relationships) which is needed to increase both conceptually and quantitatively (Gray and Elliot 2009).

The Phylum, Nematoda has been considered to be an indicator for assessing the ecological quality of marine ecosystems according to the European Water Framework Directive (WFD), Directive 2000/60/EC (Moreno et al. 2011; Semprucci et al. 2014).

23.2.10 Macrobenthic Fauna -Brachyuran Crabs: Their Roles as Ecosystem Engineers

Unlike most intertidal organisms, brachyuran crabs are semi-terrestrial and very active at low tide, returning to their burrows at high tide. They play a significant positive role in maintaining the steady state of the ecosystem and enhance its biological potentiality. Brachyuran crabs constituting an important faunal component in the food web of the coastal belt, facilitate and accelerate the decomposition cycle as a macrodecomposers and thereby govern the ecosystem functioning to a large extent after being morphologically, physiologically and behaviourally well adapted to their their respective habitats (Kristensen 2008).

23.2.11 Ecological Services Through Excavation of Burrows

The burrow is a very important resource for the crabs (Crane 1975; Zeil and Layne 2002). It offers protection from aquatic predators during high tide and from aerial and terrestrial predators during low tide, when the crabs are active on the surface. It provides a safe refuge for moulting animals and for females while incubating their eggs. The burrow protects the crabs from desiccation during their activity on the surface by offering them access to water, which is needed for respiration and feeding. The burrow walls are important sites for nitrification and de-nitrification processes in the sediment. Brachyuran crabs excavate and maintain semi-permanent open burrows, and remove large amounts of sediment during feeding and burrow maintenance forming a surface mound around burrows (Iribarne et al. 1997; Botto and Iribarne 2000). Their burrowing habit assists in oxidizing the sulphides that build up, due to the high rates of organic decomposition. It is well acceptable that coastal and estuarine food chains are based to a significant extent on detritus and dissolved organic matters produced from the breakdown of rooted and attached macrophytes of intertidal and shallow subtidal habitats. Burrow construction and maintenance, ventilation and import of organic matters play a significant role in energy flow and nutrient cycling in coastal system (Alongi 1990; Dworschak et al. 1993).

Burrowing fiddler crabs (Ocypodidae) and sesarmid (Grapsidae) are the most important macro invertebrates in many salt marshes (Emmerson 1994; Kristensen and Kostka 2005). They are often present in large numbers and their burrowing activities can directly break and transport sediments, decrease the hardness of the soil (Botto et al. 2005), modify microtopography, and increase the density of coarse particles on the soil surface (Warren and Underwood 1986). Crab burrowing also affects soil chemistry and associated microbial processes, increases soil oxygenation, and alters pore water salinity (Fanjul et al. 2007). Burrowing crabs significantly affect below ground processes that can impact marsh plants in at least three ways.

First, crab burrowing increases the passage of liquid and gas between the soil and environment (increasing drainage), increasing soil oxidation (Daleo and Iribarne 2009) and the decomposition rate of organic debris (Fanjul et al. 2007).

Second, crab burrows can selectively trap sediments that have high organic matter concentrations, finer grain size and low density through the interactions of the burrow opening with tidal water, which can facilitate organic matter decomposition such prosesses which can in turn increase nutrient availability and thus, promote their growth (Iribarne et al. 1997; Botto et al. 2006).

Third, crab excavation transports soil and nutrients from deep layers to the marsh surface (Robertson and Daniel 1989; Fanjul et al. 2008), which might accelerate the turnover of soil and nutrients. In order to avoid interspecies and intraspecies competitions, the brachyuran crabs displayed distinct horizontal and vertical distributional pattern reflecting their adaptation to different degrees of terrestriality in respect of inundation and exposure, sediments, salinity, temperature etc (Lugo 1980; Chakraborty and Choudhury 1989).

Fiddler crabs (genus *Uca*) are a common and conspicuous element of the fauna of intertidal mudflats and mangroves in tropical and subtropical regions (Crane 1975). They are also highly diverse with about 100 species having been described (Crane 1975; Thurman 1984; Beinlich and von Hagen 2006; Ng et al. 2008). *Uca* species have received a considerable amount of scientific attention with a broad range of studies investigating sexual selection, reproductive isolation, visual and acoustic display, combat, foraging, claw asymmetry, allometric growth, limb regeneration, claw mechanics, morphometrics, circadian rhythms, color change histology, osmoregulation, heat tolerance, visual neurology, toxicity, environmental monitoring, and pollution (How et al. 2008, 2009; Rosenberg 2001; Crane 1975; Kalz 1980; Zucker 1981; Kim et al. 2004a, b, 2006) published an extensive and detailed monograph of the genus, and proposed a number of new sub-generic groupings.

The crab communities of mangrove forests with *Avicennia marina* were dominated by microphagus species mostly belonging to the family Ocypodidae (species that prefers to feed detritus) rather than the leaf-eating Sesarmid crabs as found in the other forest types (Robertson and Daniel 1989). Decomposition within the mangrove forest accounts for 20–70% of litter fall, depending on frequency of tidal inundation. Some of the litter are consumed by crabs on the forest floor while much more are removed after being carried to their burrows (Micheli 1993a, b). Crab consumed greater than 78% of the buried litter within 6 h. However, crabs are often referred to as 'sloppy feeders' and it has been estimated that 20% of leaf materials is lost from the mandibles on to the floor of the burrow (Camilleri 1989).

Ocypodids and sesarmids are generally considered to be deposit feeders. Ocypodids primarily eat bacteria (Dye and Lasiak 1987) or microalgae (France 1998), while the consumption of leaf litter by some sesarmids hasa distinct effect on litter dynamics in mangrove systems (Twilley et al. 1997; Lee 1998). Crabs are omnivores, feeding primarily on algae (Woods 1993) and taking any other food, including molluscs, worms, other crustaceans, fungi, bacteria and detritus, depending on their availability and the crab species. For many crabs, a mixed diet of plant and animal matter results in the fastest growth and greatest fitness (Buck et al. 2003).

23.2.12 Bioturbation and Intertidal Macrobenthos

Sedimentary environments are dynamic habitats where the sediments, the fundamental building blocks of the habitats, are continually structured both by the local physical regime and by the biotic forces. Different biogeophysicochemical activities, mostly rendered by the intertidal biota result conversion of sands to mud, biotic structuring of the habitat by settlement, growth and development, burrowing, tube building, defecation and secretion often is significant (Rhoads and Boyer 1982; Woodin 1999; Gilbert et al. 2007).

This biogenic process collectively termed as '**Bioturbation'**, is of special importance in the cycling of trace metals, nutrients and pollutants, including radionuclides, between shallow sea sediments and the water column (Zeitzschel 1980; Kremling 1983; Nolting 1986; Swift 1993; Pandya and Vachhrajani 2010, 2011). The term 'Bioturbation' originates from ichnology, to describe traces of life in fossil and modern sediments, and has subsequently been adopted in soil and aquatic sciences (Meysman et al. 2006a; Huhta 2007; Wilkinson et al. 2009) (Fig. 23.16).

In many cases, the original meaning from ichnology is strictly followed, and the term refers solely to the redistribution of particles and the formation of biogenic structures by burrowing animals. In other cases, it is used in the context of all physi-

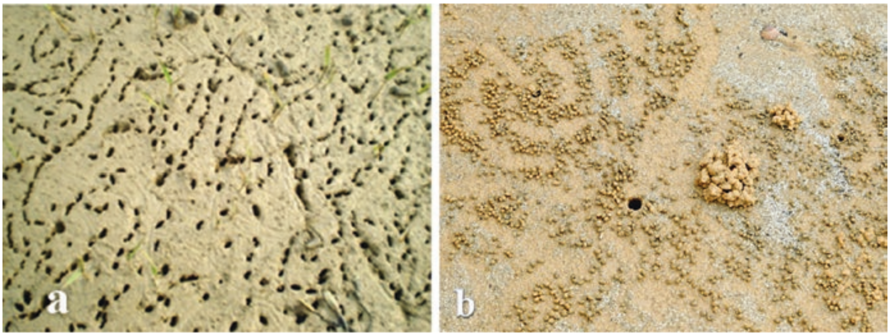


Fig. 23.16 Bioturbitory signatures (**a**) by a Brachyopd, *Lingula anatina* and (**b**) by a tiny crab, *Dotilla blanfordi*

cal disturbances caused by animals on the substratum, including particle (reworking) and water (ventilation) movements. Both in terrestrial and aquatic environment, animal bioturbation results from comparable activites, including burrow and mound construction, the lateral 'ploughing' of the surface (e.g. by moles or heart urchins), particle ingestion and egestion during foraging (e.g. deposit feeding, geophagy or lithophagy), food caching and prey excavation, wallowing and trampling, and the infilling of abandoned burrow structure (Meysman et al. 2006a).

Bioturbators include a wide array of benthic fauna like crustaceans (such as burrowing crabs and shrimps), polychaete worms, molluscs, echinoderms, sipunculans, cnidarians, priapulida and other meiofaunal organisms. Both in terrestrial and aquatic environments, bioturbation results from comparable activities, including burrow and mound construction, particle ingestion and egestion during foraging, food caching, prey excavation, wallowing etc (Fager 1964; Aller 1988; Aller and Aller 1998; Cadee 2001).

Deposit- feeders are the most prominent group of bioturbators as they constantly process sediment for food, resulting in horizontal and vertical movement of particles in the sediment. In some habitats, biogenic alterations affect the erodibility of the sediments (Kihslinger and Woodin 2000; Wilkinson et al. 2009), the distribution of grain sizes (Lim 2006), and the concentrations of pore water constituents (Aller 2001). Burrowing and defecation can affect infaunal recruitment patterns and infaunal secretions can alter the chemical constituents of the sediment (Woodin 1981). Biogenic structures also alter the ability of individuals to burrow through the sediment. Biogenic modifications therefore appear to affect the distribution and abundance patterns of infauna. Defecation by large infauna has a negative effect on other infauna, causing reduced growth rates, decreased recruitment and increased emigration. The patterns of distribution and abundance of organisms are frequently correlated with different grades of habitat heterogeneity. Changes in habitat heterogeneity may be attributable to changes in the abundance of physical and biogenic structures.

Bioturbation, that is, the disturbance of sediment layers by biological activity, is a significant process on the ocean floor. In that environment, numerous animals such as worms exist by consuming organic matter trapped between sediment grains. Macro invertebrates directly mediate key processes in the biogeochemical cycle of sediment, by releasing nutrients and consuming oxygen and organic matter. They indirectly mediate processes by redistributing particulate and dissolved substances in the sediment. In modern ecological point of view, bioturbation is now recognized as an archetypal example of 'ecosystem engineering' (Meysman et al. 2006a) and act as major modulators of microbial activites and biogeochemical processes in aquatic environments. Going beyond these relatively straightforward transactions, they alter the nature of the sediment in ways that affect organisms other than their direct competitors, predators, or prey. Ecologists call such species 'ecosystem engineers' because they create new habitats and change the availability of nutrients to other species (Kristensen 2008).

In the present context, bioturbation is such a denominator and acts as an 'umbrella' term that covers all faunal transport activities physically disturbing the substratum. It is separated into activities by animals that directly move and mix particles by the process of reworking (Solan and Wigham 2005; Mermillod-Blondin 2011) and directly move water through burrows by the process of ventilation (Kristensen and Kostka 2005; Meysman et al. 2006a). A common phenomenon in the sedimentary environments is biogenic alteration of the sediments by infauna, including defecation, burrowing, tube building, feeding and irrigation. In some habitats biogenic alterations affect the erodibility of the sediments (Luckenbach 1986). Sediment biogeochemical processes play important roles in the metabolism and nutrient cycling of salt marshes (Webb and Eyre 2004).

The most important nutrients in the coastal ecosystems are dissolved inorganic nitrogen and phosphorus compounds (Conley 2000). According to Clough 1982, the considerable size of the root biomass of mangroves suggests they are important in nutrient cycling of organic and inorganic materials. Macrophyte detritus in the form of mangrove trees and saltmarsh grasses, typically has high C: N and C: P ratio (Nielsen and Andersen 2003). Litter fall plays a crucial role in the nutrient cycling of mangrove systems as because a large amount of organic matter returned to the aquatic system through leaf senescence. A volume of data on litter production had already been reported for several studies (Rao et al. 1994; Lacerda et al. 1995; Reise 2002). The abscised leaves release substantial amounts of inorganic nutrients and Dissolved Organic Materials (DOM), which contribute sugars, proteins, and polyphenols to the surrounding water environment within a relatively short time period . Sugars and proteins are very susceptible to microbial degradation, and thus can be quickly incorporated into food webs. However, tannins, a class of polyphenols, are known to suppress nutrient utilization for microorganisms through complexation and precipitation of N-containing compounds. Since mangrove leaves contain a large amount of condensed tannins (CT), it is expected that CT leached from mangrove leaves may sequester pertinacious materials, preventing them from rapid loss from the mangrove forest (Nagamitsu et al. 2006).

Approximately 20% leaf materials after being removed from the burrows, enters the nutrient cycle as organic matter and 68% enters as ammonium (faeces) via ammonification (Camilleri 1989). The availability of nitrogen in mangrove ecosystems depends on a complex pattern of bacterial activity within the anoxic mangrove mud, the thin oxic (oxygen-containing) zone at the visible surface of the mud, and the inner oxic linings of animal burrows. Bacteria transform nitrogen in organic material into free ammonium, nitrate, or gaseous nitrogen through three processes: ammonification, nitrification, and denitrification .These processes are closely linked with each other wherever oxic sediment meets anoxic sediment; such conditions occur at some depth below the visible surface of the mangrove mud and around the burrows of mud crabs. The burrows of the tunneling mud crabs are the primary sites for both the export of ammonium and the removal of nitrogen from the mangrove ecosystem. As the bacteria in the burrow wall rapidly consume the oxygen entering from the burrow water, the oxic mud layer around the burrow is usually much thinner than the oxic layer at the visible mud surface. Consequently, the diffusion path for ammonium from the anoxic zone across the oxic wall into the burrow water is very short and ammonium generated in the anoxic mud around the burrow can more easily enter the burrow water and be flushed away by tidal currents. Therefore, burrows of crabs can be considered as the primary site for the release of ammonium into the surrounding mangrove ecosystems. However, the burrow linings seem to offer ideal conditions for both ammonium-oxidizing (nitrifying) bacteria and nitrate-reducing (denitrifying) bacteria and thus allow for a tight coupling of nitrification and de-nitrification. Through this interaction, crab burrows can effectively remove nitrogen from the aquatic ecosystem in the form of gaseous nitrogen (Holmboe et al. 2001). In this process of nutrient cycling, a huge amount of mangrove leaves contributed detritus are supplied to the adjoining aquatic environment which are instrumental for the higher biological productivity in general and fishery production in particular. The considerable size of the root and leaves of mangroves suggests they are important in nutrient cycling of organic and inorganic materials (Fig. 23.12a).

Mangroves leaves are grazed upon predominantly by insects, but they are also a food source for different mangrove crabs. It has been reported that organic carbon constituting an important nutrient is accumulated in higher proportion in the intertidal belt of tropical coast endowed with mangrove vegetation in comparison to the sub tidal zone (Aller 1998; Alongi 1988a, b). On the other hand, the rate of decomposition by benthic fauna are more rapid in the sub tidal zone in comparison to intertidal flat (Alongi 1992), but a large proportion of deposited carbon appears to be buried as refractory bit like materials (Brunskill et al. 1998). The intertidal belts along with mangroves function like a sponge with complex network of biogenic structures which foster to and fro movement of interstial water, nutrients and gases both vertically and laterally coupled with tidal advection and drainage. Leaf breakdown is defined as weight loss due to physical fragmentation, animal feeding, microbial activity and leaching. The chemistry of mangrove detritus changes profoundly during decomposition. Decomposition involves three processes like fragmentation, leaching and decay (Robertson et al. 1992). Generally, plant litter decomposition rates of terrestrial and aquatic plants have been shown to decrease with increasing biochemical complexity and increase with increasing element concentrations, particularly nitrogen. Decomposition and subsequent remineralization of mangrove detritus is important in nutrient dynamics within the forest as well as in off shore systems (Twilley et al. 1997). Rate of decomposition of organic matters are also influenced by several factors such as temperature, grain size, bioturbation and physical disturbance, seasonal parameters like precipitation, wind flow but the dominating regulatory factors appear to be the quantity and reactivity of organic matter. It has been noted that there exists characteristic differences between C and N concentration, C and N ratio, pH, etc. in between mangrove dominated and mangrove free intertidal belt (Alongi et al. 1999). The intertidal belt of Midnapore coast supports the macrobenthic fauna of which a major population part is being shared by the brachyuran crabs, a bioenergetically significant macrobenthic faunal group of this specialized environment (Chakraborty and Choudury 1985; Chakraborty et al. 1986; Chandra et al. 2003; Khalua et al. 2003; Chakraborty 2009, 2010).

Fiddler crabs are common detritivorous macrofauna of salt marshes (Bertness 1985; Thurman 1984). The detritus that fiddler crabs consume is derived from decayed *Spartina alterniflora*, although their main food source is actually the microorganisms and bacteria that grow on the decomposing *S. alterniflora* (Genoni 1991). Genoni (1985) states that fiddler crabs are food-limited. Reasons for this view include the lag time for *S. alterniflora* to degrade into detritus, the limited time available for feeding due to tidal cycles, and the quality of their food source (Genoni 1985). Burrowing by fiddler crabs brings buried organic matter to the sediment surface, where it can be used as an additional food source (Genoni 1991) and aid in *S. alterniflora* growth by providing more available nutrients (Genoni 1991).

The presence of crab burrows also modifies the hydraulic parameters of the sediment. Hydraulic conductivity can be increased by one or two orders of magnitude resulting in a value of 0.1–1.0 m/day (Hughes et al. 1998). Overall surface infiltration is increased also, and the crab burrows themselves cause extremely large infiltration rates having an average of 11 m/day (Hughes et al. 1998). Fiddler crab burrows are believed to have important effects on sediment chemistry and other sediment characteristics (Katz 1980; Montague 1980; Bertness and Miller 1984; Bertness 1985; Genoni 1991; Nomann and Pennings 1998). The burrows may act as oxygen inlet tubes, increasing sediment oxygen levels (Katz 1980; Bertness 1985). Burrows are also supposed to act as toxin outlet tubes, reducing the accumulation of metabolic products, such as sulfide.

Burrowing crabs significantly affect belowground processes that can impact marsh plants and associated biodiversity components (Bertness 1985; Lee 1998, 1999, 2008; Smith et al. 2009) in at least three ways. First, crab burrowing increases the passage of liquid and gas between the soil and overlying environment, increasing soil oxidation (Katz 1980; Daleo and Iribarne 2009; Weissberger et al. 2009) and the decomposition rate of organic debris (Fanjul et al. 2007). Second, crab burrows can selectively trap sediments that have high organic matter concentrations, finer grain size and low density through the interactions of the burrow opening with tidal water, which can facilitate organic matter decomposition, which can in turn increase nutrient availability and thus, promote their growth (Botto and Iribarne 2000; Botto et al. 2005, 2006). Third, crab excavation transports soil and nutrients from deep layers to the marsh surface (Fanjul et al. 2007, 2008), which might accelerate the turnover of soil and nutrients. Soil properties and plant assemblage characteristics influenced by crab excavation and burrow deposition can in turn affect burrowing processes (Neira et al. 2006). Sesarmid crabs play a key role as a major link between primary and secondary production through the degradation of mangrove leaf litter. The leaves of *Avicennia marina* contain rich nutrients more palatable compared to other mangroves leaves (Ravichandran and Kannupandi 2004).

23.2.13 Bioturbation and Nutrient Cycling

Bioturbation, the disturbance through the stiring or mixing of sediments layers by biological activities viz. mobility, feeding, burrowing etc. of benthic animals, affects the geochemistry of sediments and their interstitial water (Woodin et al. 1998). Brachyuran crabs, polychaetes, and globid fishes represent the major agents of bioturbation of Sundarbans and its adjoining coastal environments. Burrows and other bioturbatory structures like pseudo-pelletes, sand balls, mud balls, sand pyramids, semidomes, chimneies, hoods etc. have been documented alongwith seasonal variations on the rate of sediment distabilisation, intensity of soil excavation, pumping of water and oxygen within soil, etc. All such activities have been hypothesized as adaptive strategies in respect of sexual selection, aggression, predation and foraging, cortship and mating, terretoriality and mostly to exihit the strength and energy (Chatterjee et al. 2008).

One of the major bioturbatory activities is on the microbial degradation rate of sediment organic materials by microbial and benthic community. In mangrove ecosystem, macrobenthos along meiobenthic fauna and several microarthropods (Dey et al. 2010) primarily acted upon mangrove litters for their ready conversion into detritus by microbial activities. Detrital export from mangrove forests is a source of nutrients and energy to nearby ecosystems as evident from the Biscayne Bay, FL (Fleming et al. 1990). Bosire et al. 2005 have highlighted litter degradation and CN dynamics in reforested mangrove forest in Kenya. Raulerson (2004) studied the leaf litter processing by macro detritivores in natural and restored Neotropical mangrove forests.

Many species of mangrove plants produce viviparous propagules that develop to substantial sizes before dispersal, a behaviour that has been hypothesized as an adaptation to their intertidal habit. The large, energy-rich propagules attract grazers both before insects, (Murphy 1990) and after grapsid crabs abscission (Smith 1987).

The microbial activities in the mangrove forest for litter decomposition depend on temperature and moisture of soil (Gupta 2002). However, during monsoon excessive rainwater and water runoff create water logging condition and thereby preventing the penetration of atmospheric oxygen and lowers the activities of microbes and other soil inhabiting detritivores. During premonsoon, higher air temperature enhances the diffusion of oxygen in soil which in turn increases the redox potential, a measure of the electron pressure or availability in a solution (Mandal et al. 2009). Oxidation occurs not only during the uptake of oxygen but also if hydrogen is removed or more generally, a chemical gives up an electron. Reduction is the opposite process of giving up oxygen gaining hydrogen or gaining an electron (Mitsch and Gosselink 1986).

23.2.14 Bioturbatory Structures of Different Brachyuran Crabs in Different Study Sites

Different brachyuran crabs formed different types of burrows and bioturbatory structures; almost all intertidal brachyuran crabs construct its own burrows for taking shelter, to prevent from the predator's attack, courtship, feeding etc (Christy 1982; Chrisy and Salmon 1984). Plaster casts of brachyuran crab's burrows were successfully recovered over the full lengths especially at High Tide Level (HTL) and Mid Tide Level (MTL) while at the Low Tide Level (LTL) such success was partially achieved because of clayey soil and accumulation of water at the bottom of the burrow.

23.2.14.1 Burrow Depth and Diameter: Manifestation of Bioturbatory Activities

Ocypoda macrocera

The shape and structure of burrows of *Ocypoda macrocera* varied in accordance with different size classes of this crab. Smaller crabs having mean carapace length of upto 11 mm were found to form shallow J- shaped burrows, which inclined vertically into the substratum. Larger crabs having (mean carapace length of 21.5–24.5 mm) used to construct Y- shaped and spiral burrows. These Y- shaped burrows have a primary arm, anterior portion of which extends to the surface forming the burrow opening, while two secondary arms, at the posterior end terminate in a blind spherical structure.The two arms join in a single shaft and end to a chamber at the base. The secondary arms and chambers are supposed to be used for mating or as a refuge from predation The maximum depth of the burrow was recorded (138.60 ± 0.40 cm) in June, 2009 at HTL (High Tide Level) and the minimum depth was observed (50.80 ± 1.04 cm) in July, 2009 at LTL (Low Tide Level) (Chatterjee and Chakraborty 2014; Chatterjee et al. 2008, 2014). The maximum burrow diameter was recorded (5.1 ± 0.24 cm) during pre-monsoon 2008 at HTL and that of minimum (1.6 ± 0.14 cm) was found during post-monsoon, 2009–2010 at LTL. (Chatterjee et al. 2008, 2014).

Dotilla blanfordi

The smallest member of the family Ocypodidae is *Dotilla blanfordi* which constructs straight and narrow burrow. The maximum depth of the burrow was recorded (50.8 ± 0.14 cm) in November, 2008 at HTL and the minimum depth was (25.00 ± 1.56 cm) found in March, 2008 at LTL. The maximum burrow diameter was recorded (1.6 ± 0.12 cm) during pre-monsoon 2009 at LTL and that of minimum was observed (0.4 ± 0.22 cm) during post-monsoon, 2008–2009 at MTL.

Dotillopsis brevitarsis

Maximum and minimum values of burrow depth of *Dotillopsis brevitarsis* were revealed as (48.4 ± 1.1 cm) in May, 2008 at MTL (Mid Tide Level) and (22.9 ± 1.10 cm) in August, 2009 at LTL respectively (Chatterjee and Chakraborty 2014; Chatterjee et al. 2014). The maximum burrow diameter was recorded (1.8 ± 0.40 cm) during pre-monsoon 2009 at LTL and that of minimum (0.7 ± 0.20 cm) was found during post-monsoon 2009–2010 at MTL (Chatterjee and Chakraborty 2014; Chatterjee et al. 2014).

Uca acuta acuta

The burrow of *Ucaacuta acuta* always displayed uniformity with regard to shape. The upper part of their burrows remained straight up to 15 cm in length, after which it took a bend of nearly 45° angle towrds descending part of the burrow and ultimately ended up forming a chamber. During high tide, the burrow entrance remained totally closed with mud. The maximum depth of their burrows were observed (112.2 ± 1.04 cm) in May, 2009 at HTL and the minimum depth were recorded (34.60 ± 0.14 cm) in July, 2008 at LTL (Chatterjee and Chakraborty 2014; Chatterjee et al. 2008, 2014). The maximum burrow diameter was found (4.8 ± 1.14 cm) during monsoon 2009 at MTL and that of minimum was noticed (1.0 ± 0.52 cm) during post-monsoon, 2009–2010 at HTL.

Uca lactea annulipes

Out of three species of fiddler crabs recorded from mangrove estuarine complex (Chakraborty et al. 1986), *Uca lactea annulipes* is the most abundant in the Midnapore coastal belt and its burrow initially remained straight up to 10 cm after which taking a bend it ended in a chamber depicting J-shaped configeration. All burrows were having air in the upper two-third parts and water only in the lowest one third part used to build J- shaped burrows. During high tide, the burrow entrances remained totally closed with mud. The maximum depth of the burrow was exhibited (101.6 ± 0.58 cm) in May, 2009 at HTL and the minimum depth was recorded (30.5 ± 1.2 cm) during July, 2008 at LTL. The maximum burrow diameter was revealed (2.75 ± 0.30 cm) during monsoon 2009 at LTL and that of minimum was found (0.9 ± 0.01 cm) during post-monsoon, 2009–2010 at HTL (Chatterjee and Chakraborty 2014; Chatterjee et al. 2008, 2014).

Uca triangularis bengali

The maximum depth of the burrow of *Uca triangularis bengali* was recorded (98 ± 0.28 cm) in June, 2009 at HTL and the minimum depth was observed (29.6 ± 0.42 cm) in December, 2009 at LTL. The maximum burrow diameter was recorded (2.9

± 0.04 cm) during monsoon 2009 at MTL and that of minimum was observed (0.6 ± 0.02 cm) during post-monsoon, 2008–2009 at HTL (Chatterjee and Chakraborty 2014; Chatterjee et al. 2014).

Sesarma (Chiromantes) bidens

The upper part of burrows of *Sesarma (Chiromantes) bidens*, the most dominant member of the family grapsidae in respect of their biomass and activities remained straight up to 35 cm in length and then curved slightly, which was not observed at LTL. Maximum depth of the burrow constructed by *Sesarma (Chiromantes) bidens* was recorded (135.8 ± 1.2 cm) in June, 2009 at HTL and that of minimum was noted (53.3 ± 0.46 cm) in November, 2009 at MTL. The maximum burrow diameter was recorded (4.4 ± 0.08 cm) during monsoon 2009 at MTL and that of minimum was found (1.8 ± 0.42 cm) during post-monsoon, 2009–2010 at HTL. (Chatterjee et al. 2008, 2014)

Sesarma taeniolatum

The upper part of the burrows remained straight up to 28 cm in length and burrows were completely absent at LTL. The Maximum depth of the burrow was recorded (103.5 ± 0.22 cm) in July, 2009 at HTL and that of minimum was observed (52.7 ± 0.27 cm) in January, 2009 at MTL. The maximum burrow diameter was revealed (1.8 ± 0.22 cm) during post-monsoon, 2009–2010 at MTL and that of minimum was found (0.6 ± 0.03 cm) during pre-monsoon, 2008 at HTL (Chatterjee et al. 2014; Chatterjee and Chakraborty 2014).

Metopograpsus maculatus

Metopograpsus maculatus, another member of the family grapsidae was found to inhabit at LTL and MTL. Maximum and minimum depths of their burrows were recorded as (45.8 ± 0.35 cm) in May, 2009 at MTL and that of minimum was observed (25 ± 0.12 cm) in March, 2008 at LTL. The maximum burrow diameter was exhibited (4.2 ± 0.24 cm) during post-monsoon 2009 at LTL and that of minimum was found (1.6 ± 0.5 cm) during monsoon, 2009 at MTL (Chatterjee and Chakraborty 2014; Chatterjee et al. 2014).

Metaplax intermedia

Metaplax intermedia was found to construct burrows with two entrances which are provided with small hood of soft mud substratum. The maximum depth of their burrows were noticed (52.2 ± 0.08 cm) in June, 2008 at HTL and that of minimum was

(30.5 ± 0.62 cm) in July, 2008 at LTL. The maximum burrow diameter was recorded (1.9 ± 0.40 cm) during post-monsoon 2008–2009 at LTL and that of minimum was observed (0.6 ± 0.22 cm) during monsoon, 2009 at LTL. (Chatterjee et al. 2014; Chatterjee and Chakraborty 2014).

23.2.14.2 Bioturbatory Structures (Fig. 23.17)

Crabs are mostly active animals with complex behaviour patterns (Zucker 1974). They can communicate by drumming or waving their pincers. Crabs tend to be aggressive towards one another and males often fight to gain access to females. The brachyuran crabs are well known for the behaviour of building various structures from moist mud sand at the entrance of their burrows. Some crabs and other crustaceans like thalasinia build different bioturbatory structures (pillars, hoods, chimneys, semidomes, mudballs, sand balls etc.) next to their burrows (Powers and Bliss 1983; Chaudhuri and Choudhury 1994; Fanjul et al. 2007, 2008). Some of these

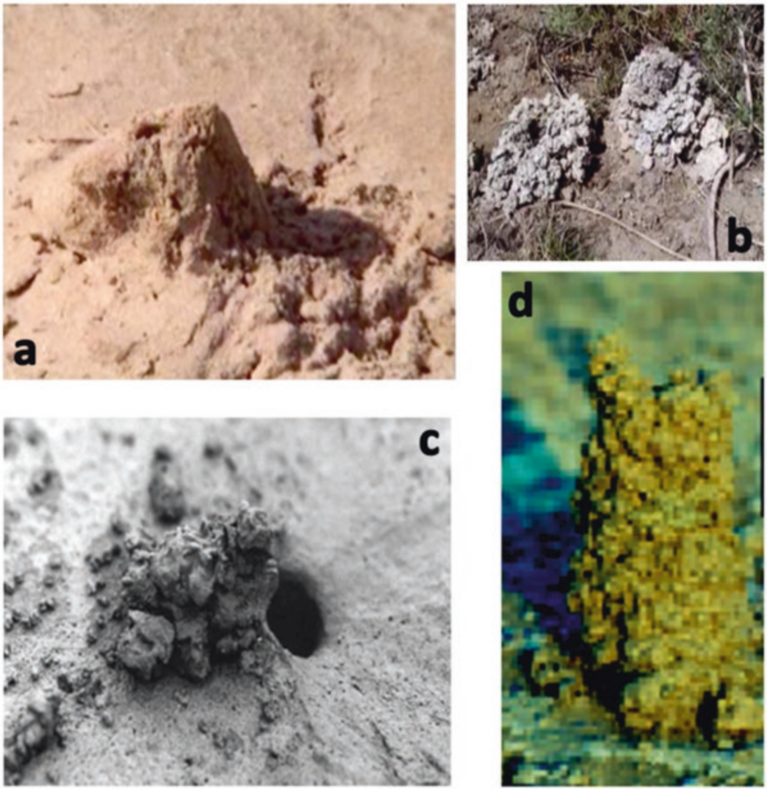


Fig. 23.17 Different bioturbitory structures (**a**) Sand Pyramids by *O. macrocera*; (**b**) Hood by *Sesarma sp.*; (**b**) Semidome by *Uca annulepis annulepis* (female); (**d**) Chimny by *Uca acuta acuta*

structures have been shown to function for sexual attraction, e.g., pillars (*Uca*: Christy 1988), hoods (*Uca*: Zucker 1981), mudballs (*Uca*: Oliveira et al. 1998), and pyramids (*Ocypode*: Hughes 1973); some are related to aggression (fighting or invasion to other burrows), e.g., hoods (*Uca*: Zucker 1974 and *Cleistostoma*: Clayton 1988), chimneys (*Uca*: Wada and Murata 2000), and barricades (*Ilyoplax*: Wada 1983); and some are related to environmental regulation, e.g., hoods (*Uca*: Powers and Cole 1976), chimneys (*Uca*: Crane 1975; Thurman 1984), and igloos (*Dotilla*: Takeda et al. 1996). Such biogenic structures having great functional roles in the ecosystem through facilitating bio geochemical cycling in one hand and ensuring their survival strategies vis-a-vis feeding, territory protection, aggression, female attraction and courtship signaling on the other (Flores and Paula 2002; Chatterjee et al. 2008, 2014).

Sandball, Pseudo Pellet and Mudball (Figs. 23.16, 23.17, and 23.18)

The number of sandballs around the burrow openings of *Ocypoda macrocera* varied from 25.24 ± 0.62 at MTL during pre-monsoon, 2008 to 70.28 ± 2.37 at HTL during monsoon, 2008. The male crabs were tended to place sand balls mostly in a circular pattern around the burrow openings at high tide zones.

Dotilla blanfordi has been found to make numerous sand balls in the form of pellets. Intensity of burrow diggings was quantified on the basis of the number of pellets excavated per 30 min duration per crab. The number of sand balls around burrow openings fluctuated from 21.56 ± 0.15 at MTL during monsoon, 2008 to 166.58 ± 0.79 at LTL during pre-monsoon, 2009.

Fig. 23.18 Preparation of burrow configuration with plaster of paris

The number of slightly rounded excavated mud of *Uca acuta acuta*, around the burrow openings varied from 12.2 ± 0.79 at HTL during post-monsoon, 2009–2010 to 65.92 ± 2.30 at LTL during monsoon, 2008–2009. The number of oval shaped excavated mud around the burrow openings of *Uca lactea annulipes* ranged from 9.45 ± 0.05 at LTL during pre-monsoon, 2008 to 22.92 ± 5.93 at HTL during post-monsoon, 2009–2010.

Although both sexes of *Uca triangularis bengali* could make mud balls, there were significant differences between them with regard to size and placement. Males made more mud balls of larger diameter and placed them comparatively longer distant places away from their burrow entrances than females which reflects the possession of more energy in males. The number of mud balls around the burrow openings fluctuated from 1.96 ± 0.39 at LTL during post-monsoon, 2008–2009 to 11.08 ± 0.87 at MTL during monsoon, 2009.

The amount of excavated mud around the burrow openings in *Sesarma (Chiromantes) bidens* varied from 31.36 ± 0.54 at HTL in pre-monsoon, 2008 to 71.50 ± 1.25 at MTL during post-monsoon, 2009–2010.

Four brachyuran species viz. *Dotillopsis brevitarsis, Sesarma taeniolatum, Metopograpsus maculatus*and *Metaplax intermedia* have been unable to make any sand or mud balls (in terms of number) among the ten studied brachyurans (Chatterjee 2008; Chatterjee et al. 2014).

Sand Pyramids : (Figs. 23.16 and 23.17)

Among different studied crabs, only larger male *Ocypoda macrocera* (mean carapace length 21.5–30.5 mm) used to build pyramid like bioturbatory structure. A pyramid is any three- dimensional polyhedron where the faces other than the base are triangular and converge on one point, called the apex. The maximum height of pyramid was recorded as (8.03 ± 0.24 cm) at HTL during pre-monsoon, 2009 and that of minimum was found as (2.54 ± 0.15 cm) at MTL during monsoon, 2008 (Chatterjee 2008; Chatterjee et al. 2014).

Pellets: (Fig. 23.16)

Pellet is a generic term used for a small particle or grain. Sand Pellets are the residual parts of sediment after being scooped by the mandible-maxilla of respective brachyuran crabs for extracting their food materials. These earth materials are being prepared by different species of brachyuran crabs, the structural entities of which conform to the behavior patterns of each species. Pseudo-faecal pellets made by *Dotilla blanfordi* for feeding activities were found densely placed around the burrow in a circular pattern. The intensity of digging was more pronounced just after emergence at the onset of low tide and before sealing of their burrows just before high tide. After being threatened, each individual was found to prepare new burrow quickly especially at low tidal level. These crabs accumulate sand particles in the dorsal buccal region after being collected with chelae from where they are periodically removed by the chelae, swept underneath the body and kicked to the nearer place. The diameter of pellets was found to range from 0.30 ± 0.02 cm at MTL during pre-monsoon, 2008 to 0.89 ± 0.02 cm at LTL during monsoon, 2009 made by *Dotilla blanfordi* (Chatterjee 2008; Chatterjee et al. 2014).

Chimney: (Fig. 23.17)

Literally chimney is a vertical shaft that provides a path through which smoke from a fire is carried away through the wall or roof of a building. Brachyuran crabs also made chimney which was composed of large wet soil pellets i.e., mud balls. The soil for the chimney was carried up from within the burrow, not from the surface substrate near the burrow as with the chimney building of *U*ca *acuta acuta***.** Therefore, the colour of the chimney and the substrates composing these structures generally differed. Tower like cylindrical structure, chimney was made by only male *Uca acuta acuta* and the height of chimney ranged from 6.43 ± 0.20 cm at LTL during monsoon, 2008 to 0.58 ± 0.30 cm at HTL during post-monsoon, 2009–2010.

Semidome

Semidome is a roof covering a semicircular space i.e. half a dome. Only male *Uca lactea annulipes* could construct semidomes by scrapping up bits of sediment from the surface with their first four walking legs on the side with the major claw, and depositing the sediment materials around the burrow entrance in triangular shaped semidomes more frequently during monsoon. The maximum height of semidome (4.98 ± 0.07 cm) was recorded during monsoon, 2008 at LTL and the minimum height (0.39 ± 0.02 cm) was exhibited during pre-monsoon, 2008 at HTL.

Mudball: (Fig. 23.17)

Males of *Uca triangularis bengali* also made mudball which was composed of large wet soil pellets and the diameter of mud balls were ranged from 1.60 ± 0.90 cm at MTL during post-monsoon, 2009–2010 to 0.19 ± 0.04 cm at HTL during monsoon, 2008 (Chatterjee 2008; Chatterjee et al. 2014).

Hood: (Fig. 23.17)

Literally hood means a soft covering for the head, worn by women, which leaves only the face showed. Both sexes of *Sesarma (Chiromantes) bidens* used to build slightly flat hood around their burrow opening.The height of hood was ranged from 5.84 ± 0.29 cm at HTL during monsoon, 2008 to 11.89 ± 0.04 cm at MTL during post-monsoon, 2009–2010. At LTL, the hood was not found during the study period (Chatterjee 2008; Chatterjee et al. 2014).

23.2.14.3 Seasonal Variation in Quantity (Weight) of Excavated Sand/Mud (in Terms of Weight in gm)

The weight of the sand ball of *Ocypoda macrocera* varied from 118.42 ± 2.79 g at MTL during pre-monsoon, 2009 to 340.00 ± 4.85 g at HTL during monsoon, 2009. The weight of the pellet of *Dotilla blanfordi* ranged from 26.58 ± 0.17 g at HTL during post-monsoon, 2009–2010 to 115.84 ± 0.68 g at LTL during Pre-monsoon, 2008. In *Dotillopsis brevitarsis*, it varied from 50 ± 0.48 g at MTL during monsoon, 2008 to 110 ± 0.54 g at LTL during pre-monsoon, 2008. The weight of the excavated mud of *Uca acuta acuta* ranged from a minimum of 101.75 ± 1.09 g at HTL during post-monsoon, 2009–2010 to a maximum of 254.83 ± 0.58 g at LTL during monsoon, 2009. In *Uca lactea annulipes*, the weight of the excavated mud varied from a minimum of 62.64 ± 0.89 g at MTL during pre-monsoon, 2008 to a maximum of 186.58 ± 1.10 g at LTL during monsoon, 2009. The weight of the excavated mud of *Uca triangularis bengali* exhibited a range from to 55.67 ± 0.86 g at MTL during post-monsoon, 2008–2009 to 166.50 ± 7.42 g at MTL during monsoon, 2008. The weight of the excavated compact mud of *Sesarma (Chiromantes) bidens* around the burrow opening varied from 271.64 ± 0.90 g at MTL during pre-monsoon, 2008 to 441.17 ± 2.53 g at MTL during post-monsoon, 2009–2010. *Sesarma taeniolatum* was not found to excavate mud during the study period. Weight of the excavated mud of *Metopograpsus maculatus* exhibited a range of 64 ± 0.27 g at LTL during monsoon, 2008 to 115 ± 0.80 g at MTL during post-monsoon, 2008–2009. The minimum and maximum weight of excavated mud of *Metaplax intermedia* were 42.75 ± 1.07 g at HTL during monsoon, 2009 and 92.33 ± 0.48 g at LTL during post-monsoon, 2009–2010 respectively (Table 23.8).

Table 23.8 Seasonal variation of weight (gm) of excavated sand/mud balls

Crabs species	Minimum	Maximum
Ocypoda macrocera	118.42 ± 2.79 at MTL in pre monsoon, 2005	340. 00 ± 4.85 at HTL in monsoon, 2005
Dotilla blanfordi	26.58 ± 0.17 at HTL in post monsoon, 2005–2006	112.50 ± 0.75 at LTL during Pre-monsoon, 2005
Uca lactea annulipes	65.58 ± 0.93 at MTL in pre monsoon, 2005	186.58 ± 1.10 at LTL in monsoon, 2005
Uca acuta acuta	105.00 ± 0.75 at MTL in post monsoon, 2005–2006	254.83 ± 0.58 at LTL during monsoon, 2005
Uca triangularis bengali	76.67 ± 0.08 at HTL during pre-monsoon, 2005	166.83 ± 1.69 at MTL during monsoon, 2005
Sesarma chiromantes bidens	279.92 ± 0.87 at MTL during pre monsoon, 2005	441.25 ± 2.50 at HTL in post monsoon, 2005–2006

23.2.14.4 Seasonal Variation in Distance (cm) of Placement of Excavated Matter (Sand/Mud)

The seasonal variation of the distance of placement of excavated matters from their burrow opening varied widely in different selected brachyuran crabs. The range of placed excavated sand of *Ocypoda macrocera* was 41.17 ± 1.15 cm at LTL during monsoon, 2008 to 102.50 ± 1.15 cm at HTL during post-monsoon, 2009–2010. Such distance of pellet of *Dotilla blanfordi* varied from 6.87 ± 0.28 cm at MTL during monsoon, 2008 to 30.83 ± 0.79 cm at HTL during post-monsoon, 2009–2010. In *Dotillopsis brevitarsis*, it exhibited a range from 3.22 ± 1.20 cm at LTL during post-monsoon, 2008–2009 to (8.26 ± 1.02 cm) at MTL during post-monsoon, 2009–2010. The mud balls placement of *U. acuta acuta* showed a distance of 7.08 ± 0.22 cm at HTL during post-monsoon, 2009–2010 to 24.75 ± 0.38 cm at LTL during monsoon, 2009 from their respective burrows. The distance of placement of mud balls of *U. lactea annulipes* varied from 5.69 ± 0.05 cm at HTL during pre-monsoon, 2008 to 18.83 ± 0.36 cm at LTL during monsoon, 2009. The mud balls prepared by *U. triangularis Bengali* were found to place at a distance of 3.96 ± 0.32 cm at HTL during post-monsoon, 2008–2010 to 10.17 ± 0.08 cm at LTL during pre-monsoon,2009. *Sesarma (Chiromantes) bidens*, *Sesarma taeniolatum, Metopograpsus maculatus* and *Metaplax intermedia* did not kick off their excavated matters from their respective burrows during the study period (Table 23.9).

23.2.14.5 Seasonal Variation in Sequence of Placement of Sand or Mud Balls/30 Min

Seasonal variation in the sequence of placement of sand balls of *Ocypoda macrocera*, revealed its maximum value (58.75 ± 1.28 times) at HTL during monsoon2009 and minimum value (13.58 ± 1.02 times) at MTL during pre-monsoon, 2008. In *Dotilla blanfordi*, it was maximum (63.52 ± 0.61 times) at LTL during pre-monsoon, 2008 and minimum (7.53 ± 0.58 times) at HTL during post-monsoon, 2008–2009. In *Dotillopsis brevitarsis*, it was found that maximum value was found (55.32 ± 0.61 times) at LTL during pre-monsoon, 2008 and that of minimum (8.24 ± 0.28 times) was recorded at MTL during monsoon, 2009. *Uca acuta acuta* displayed its highest value (47.42 ± 0.06 times) at LTL during monsoon, 2009 and minimum (11.93 ± 1.32 times) at MTL during pre-monsoon, 2008. *Uca lactea annulipes* highlighted its highest value (39.92 ± 0.94 times) at LTL during monsoon, 2009 and minimum value (1.33 ± 0.44 times) at HTL during post-monsoon, 2009–2010**.** In *Uca triangularis bengali* , the placement of mudball was maximum (31.67 ± 0.30 times) at LTL during monsoon, 2009 and minimum (7.00 ± 0.14 times) at HTL during post-monsoon, 2009–2010. *Sesarma (Chiromantes) bidens*, *Sesarma taeniolatum, Metopograpsus maculatus* and *Metaplax intermedia* were unable to place their excavated matters sequentially from their respective burrows during the study period (Table 23.10).

Table 23.9 Seasonal variation of distance (cm) of placement of excavated sand/mud balls

Crabs species	Minimum	Maximum
Ocypoda macrocera	42.25 ± 1.26 at LTL during monsoon, 2005	102.50 ± 1.15 at HTL during post monsoon, 2005–2006
Dotilla blanfordi	7.33 ± 0.17 at LTL during monsoon, 2005	30.83 ± 0.79 at HTL during post monsoon, 2005–2006
Uca lactea annulipes	5.83 ± 0.08 at HTL during pre monsoon, 2005	18.83 ± 0.36 at LTL during monsoon, 2005
Uca acuta acuta	7.08 ± 0.22 at HTL during post-monsoon, 2005–2006	24.75 ± 0.38 at LTL in monsoon, 2005
Uca triangularis bengali	5.67 ± 0.30 at HTL during post monsoon, 2005–2006	9.75 ± 0.38 at LTL in monsoon, 2005

Table 23.10 Seasonal variation in frequency of sand balls/mud balls placement for a period of 10 min

Crabs species	Minimum	Maximum
Ocypoda macrocera	(14.17 ± 0.96) at MTL during pre monsoon, 2005	(58.75 ± 1.28) at HTL during monsoon, 2005
Dotilla blanfordi	(7.33 ± 0.17) at LTL during monsoon, 2005	(30.83 ± 0.79) at HTL during post monsoon, 2005–2006
Uca lactea annulipes	(1.33 ± 0.44) at HTL during post monsoon, 2005–2006	(39.92 ± 0.94) at LTL during monsoon, 2005
Uca acuta acuta	(12.42 ± 0.30) at HTL during post-monsoon, 2005–2006	(47.42 ± 1.06) at LTL during monsoon, 2005
Uca triangularis bengali	(7.00 ± 0.14) at HTL during post-monsoon, 2005–2006	(31.67 ± 0.30) at LTL during monsoon, 2005
Sesarma chiromantes bidens	(25.50 ± 0.88) at HTL during pre monsoon, 2005	(69.33 ± 1.24) at MTL during post monsoon, 2005–2006

23.2.14.6 Bioturbatory Scores of Major Macrobenthic Fauna (Table - 23.11)

Bioturbatory scores are calculated in order to develop a benchmark to assess the bioturbation potential as well as to interprete the outcomes of the bioturbitory activities of several macrobenthic fauna. From these scores it is possible to compare not only the functional contribution among different benthic groups but also among different species of certain taxa.

Bioturbatory Scores of Different Brachyuran Crabs

Bioturbatory scores of ten intertidal brachyuran crabs belonging to two major families' viz. Ocypodidae (six species) and Grapsidae (four species) after being calculated, have shown maximum score by *Ocypoda macrocera* (8) followed by *Dotilla blanfordi* (**7**), *Uca lactea annulipes* (7), *Sesarma (Chiromantes) bidens* (7). The bioturbation scores were shown by *Sesarma taeniolatum* (6), *Metopograpsus maculatus* (6), *Uca acuta acuta* (6) followed by *Uca triangularis bengali* (4), *Dotillopsis brevitarsis* (3) and *Metaplax intermedia* (3).

Bioturbatory Scores by Different Different Polychaetes: (Table 23.12)

Bioturbatory scores of 18 different polychaete species belonging to six major families have revealed almost equal bioturbitory potentialities ranging from 3 to 6 where only the species *Eteone ormata* displayed the maximum scores i.e. 6.

Bioturbatory Scores by a Brachypod: (Table 23.13)

Only one species *Lingula anatina* under the phylum Brachiopoda displayed moderate bioturbitory activities showing maximum bioturbitory scores of 6.

23.2.14.7 Diversity and Mode of Decomposition by Microarthropods

Diversity and Succession of Microarthropods

A long term study, dealing with the role of microarthropods in nutrient cycling of an mangrove ecosystem, in the Nayachara Island located at the south-eastern part of Midnapore(East) coast has highlighted their gradual occurrence and succession in respect of decomposition of the organic plant litter (Figs. 23.19 and 23.20). The succession was initiated with the occurrence of collembola followed gradually by acarina, coleoptra, diptera, isopoda, hymenoptra and others, resulting in the generation of maximum nutrients after 6 months of decomposition of mangrove litters (Dey et al. 2010).

Ecological Services by microarthropods by way of mangrove litter decomposition

The succession in the occurrence of different microarthropod faunal components in the process of decomposition of a selected mangrove plant species, *Avicennia officinalis* (L) revealed 45.45% relative abundance of collembola after 3 months of decomposition followed by acarina (36.66%) and coleoptera (9.09%). After 6 months of decomposition, acarina population contributed 37.5% of total microarthropod population followed by collembola (33.33%), coleoptera (8.33%), diptera (8.3%) and isopoda (4.16%). After 9 months, collembola population was recorded as 19.09% followed by isopoda (17.47%), acarina (14.28%), coleoptera (14.28%)

Table 23.11 Bioturbatory scores by different species of brachyuran crabs at Midnapore(East), Coast

Sl. No.	Brachyuran crabs	Scores			
		Mobility	Feeding	Burrowing	Total
Family – Ocypodidae					
1.	*Ocypoda macrocera* Edwards	3	2	4	8
2.	*Dotilla blanfordi* Alcock	2	2	3	7
3.	*Dotillopsis brevitarsis* (De Haan) Kemp	1	1	1	3
4.	*Uca acuta acuta* (Simpson)	2	2	2	6
5.	*U. lactea annulipes* (H. Milne Edwards)	3	2	2	7
6.	*U. triangularis bengali* (Nobili)	1	1	2	4
Family – Grapsididae					
7.	*Sesarma chiromantes bidens* (de Haan,1835)	2	1	3	6
8.	*Metopograpsus maculates* Milne Edwards	1	1	1	3
9.	*Metaplax intermedia* de Man	1	1	1	3

Table 23.12 Bioturbatory scores by different species of Polychaetes at Midnapore (East), Coast

Sl. No.	Name of the polychaete	Mobility	Feeding	Burrowing	Total
1	*Eteone barantollae*	2	1	1	4
2	*Eteone ornata*	3	2	1	6
3	*Glycera rouxii*	2	1	1	4
4	*Glycera tesselate*	2	2	1	5
5	*Glycera alba*	2	2	1	5
6	*Perineries cultrifera*	0	1	2	3
7	*Perineries nuntia*	0	1	3	4
8	*Neanthes chingrighattensis*	2	1	1	4
9	*Lycastonereis indica*	1	1	2	4
10	*Nereis glandicincta*	0	2	3	5
11	*Mastobranchus indicus*	1	2	2	5
12	*Paraheteromastus tenuis*	1	2	1	4
13	*Talehsapia annandalei*	0	2	1	3
14	*Axiothella obockensis*	0	1	2	3
15	*Maldane sarsi*	0	2	2	4
16	*Owenia fusiformis*	0	1	2	3
17	*Lumbrinereis polydesma*	1	2	1	4
18	*Diopatra cuprea*	0	1	2	3

Table 23.13 Different bioturbatory scores of a brachyopod species, *Lingula anatina* at three different sites at three study sites of Midnapore (East) coast

Study site	Mobility	Feeding	Burrowing	Total
SI	3	2	1	6
SII	0	2	1	3
SIII	1	2	1	4

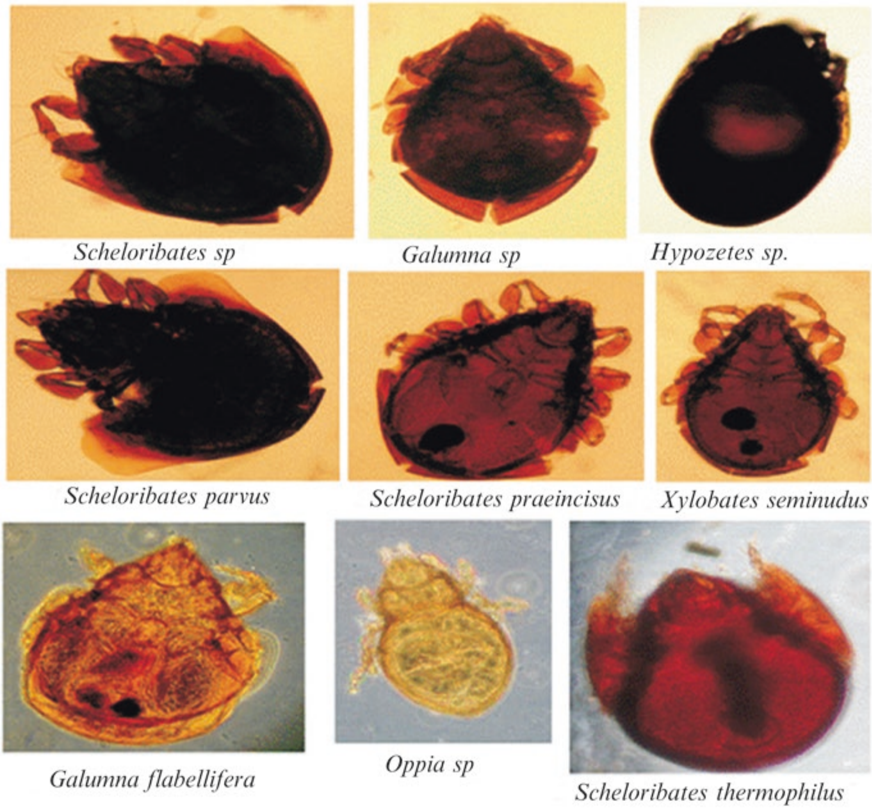


Fig. 23.19 Microarthropod faunal components

and others (9.22%). After 12 months, 19.50% was represented by isopoda followed by hymenoptera (18.29%), coleoptera (18.29%), acarina (14.63%), collembola (10.97%) and other microarthropods (7.31%) (Table 23.3 and Figs. 23.21 and 23.22).

The rate of decomposition of *Avicennia officinalis* (L) after 3 months was 34%, whereas after 6 months, it reached to 57%. After 9 and 12 months, these were 62% and 70%, respectively. After 3 months of decomposition, different physicochemical parameters were recorded such as pH (7.3), organic carbon (4.3%), total N (1,220 ppm) and available K (120 ppm). After 6 months, the recorded parameters were as pH (7.1), organic carbon (5.8%), total N (1,230 ppm), total P (96 ppm) and available K (120 ppm). After 9 months, the estimated parameters were as pH (6.5), organic carbon (6%), total N (1,230 ppm), total P (84 ppm) and available K (160 ppm). After 12 months, different parameters were observed as pH (6.6), organic carbon (6%), total N (1,220 ppm), total P (84 ppm) and available K (120 ppm).

Fig. 23.20 Microarthropod faunal components.

During the first phase (initial 3 months), the collembola population was found to be maximum followed by acarina, coleoptera and other microarthropods. During second phase of decomposition (3–6 months), diptera marked its first appearance in the decomposing litter while population of different groups of microarthropods *viz.* acarina, collembola and other microarthropods gradually increased. In the third phase (6–9 months), the population density of acarina, collembola and coleoptera of different species showed decreasing trend whereas the population density of hymenoptera revealed an increasing trend. In the last phase (9–12 months), the population density of acarina, collembola and coleoptera started decreasing whereas population density of isopoda, hymenoptera and other microarthropods were increased (Dey et al. 2007, 2008).

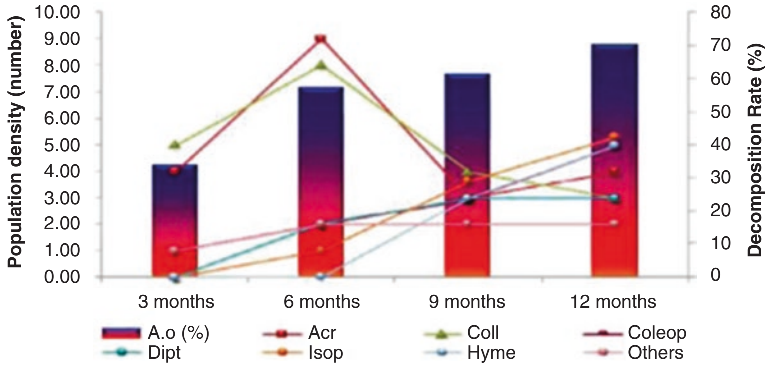


Fig. 23.21 Density of soil microarthropods' population in relation to physichochemical paramentrs during different phases of litter decomposition of *Avicennia officinalis*

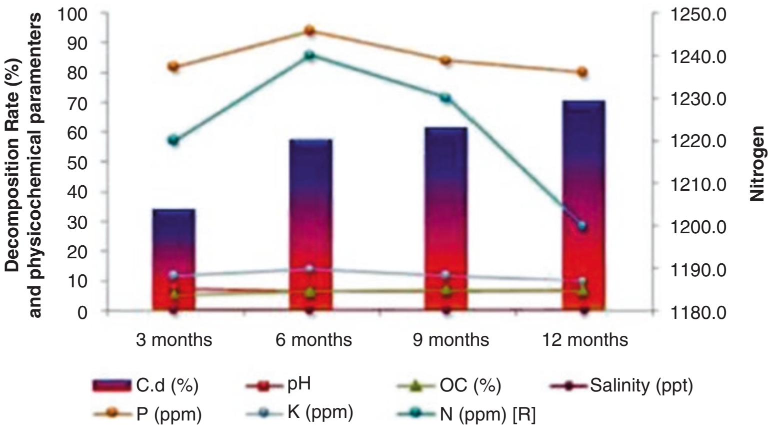


Fig. 23.22 Different decomposition rate and changes of physicochemical parameters during different phases of litter decomposition of *Avicennia officinalis*

23.2.15 Biochemical Ecology: Ecological Services by Way of Biotransformation of Nutrients

In mangrove–estuarine-coastal ecosystem, the open pathways of nutrient transports are driven by physical processes, such as, tides, run off, matereological parameters like precipitation and biological factors especially litter fall, decomposition and mineralization. The primary food source for aquatic and intertidal mangrove dwelling faunal components is derived from vascular plant detritus mostly from mangrove leaves. The breakdown of mangrove leaves is brought about by the activities of microorganisms, such as, fungi, bacteria and protozoa (Chakraborty 2011).

It is interesting to note the pathways of conversion of different biochemical constituents through the ecosystem functioning in mangrove-estuarine system. Lipids being an important biochemical component in the living organisms play vital physiological functionings (Ackman 1989) and also contribute profusely in the flow organic materials through food chains and food webs operating in the ecosystem. Studies have been conducted in the coastal Midnapore (East) district to understand the mode of bioconversion and biotransformation of lipids in the following pathway:

Mangroves → Soil → Water → Macrodecomposers → Microdecomposers → Detritus → Detritivores (Benthos) → Detritus etc.

These studies on three different macro benthos (fiddler crab, brachiopod and nemertean) have revealed mode of ocurrences and characteristics of different fractional components of fatty acids in different parts of studied fauna and the made of bioconversion of such biochemical entities. The ultimate outcome was the enrichment of the system with lot of decomposition products through such biogeo chemical processes leading to the enrichment of nutrients so that all the biodiversity components get the benefit through trophic relationships (Chakraborty 2013).

Lipids of plant and animal origins can be divided into three main classes, viz., neutral-, glycol- and phospholipids (Gurr and James 1980). The various main classes in the neutral lipids are, hydrocarbons, wax esters, steryl esters, triacylglycerols (or simply, triglycerides) and sterols. Besides, vitamins and terpenoids, included in the lipids, may also be present in naturally occurring lipids. Lipids belong to a heterogeneous class of predominantly non-polar compounds, mostly insoluble in water, but soluble in non-polar organic solvents such as chloroform, benzene etc. They are either esters, mostly of fatty acids, with different alcohols, and contains also in many cases other constituents also. They are compounds of mainly C (carbon), H (hydrogen), and O (oxygen), but carry in some cases P (phosphorus), N (nitrogen) and S (sulfur). Although the term lipid is sometimes used as a synonym for fats, but there are sub-group of lipids, called triglycerides. Lipids also encompass molecules such as fatty acids and their derivatives (including tri-, di-, monoglycerides, and phospholipids), as well as other sterol-containing metabolites such as cholesterol (Michelle et al. 1993). Although humans and other mammals use various biosynthetic pathways to both break down and synthesize of lipids, some essential lipids cannot be made this way.

23.2.15.1 Bioconversion and Biotransformation of Fatty Acids by an Intertidal Macrobenthos (Brachyopod), *Lingula anatina*

Fatty acids have been used as qualitative markers to trace or confirm predator prey relationships in the marine environment for more than 30 years (Prato et al. 2012). Fatty acids are useful tool to study trophic ecology and determine food web connections, contrary to more traditional gut content analysis which provide information on dietary intake and food constituents leading to the sequestration of lipid reserves over a longer period of time (Auel et al. 2002).

Table 23.14 Percentage of Total Lipid (TL) obtained from various body parts of *Lingula anatina* and its primary food sources

Sample	Amount taken	Total lipid obtained	Percentage of total lipid (w/w)
Lophophore	1.08 g	8.5 mg	0.79
Pedicle	3.20 g	11.2 mg	0.35
Muscle	4.27 g	126.1 mg	2.95
Gut content	0.56 g	11.7 mg	2.09
Plankton	2.80 g	23.6 mg	0.84
Mangrove leaves	16.3 g	28.1 mg	0.17
Detritus	9.63 g	14.2 mg	0.15

Table 23.15 Percentage of Neutral Lipid (NL), Glycolipid (GL), and Phospholipid (PL) obtained from Total Lipid (TL) of muscle sample of *Lingula anatina*

Samples	Percentage of lipid (w/w)
Neutral lipid (NL)	34.55
Glycolopid (GL)	11.41
Phospholipid (PL)	54.03

In a marine ecosystem, generally qualitative similarities are observed in the fatty acid composition of the organisms which occupy different trophic levels. The first link of the food chain i.e. phytoplankton are able to synthesize all the fatty acids de novo and composition of fatty acids changed significantly in the decomposing leaves of mangroves (Bligh and Dyer 1959; Moreno et al. 1979; Alikunhi et al. 2010). High concentration of ω3 fatty acids which are generally considered as to be fatty acids of marine life, were found to have been mainly contributed by some phytoplanktonic species (diatoms, dianoflagellates etc.) to the marine ecosystem (Sargent et al. 1976). Examination on various parts of *Lingula anatina*, an intertidal detritivorous macrobenthic animal of the studied areas revealed the presence of appreciably high amount of 20:5ω3 fatty acids (Tables 23.14, 23.15, 23.16, and 23.17) associated with other polyunsaturated fatty acids of the –ω3 series.

It is thus envisaged that intake of higher amount of the precursor acid (18:3ω3), through primary food sources and their subsequent chain elongations and desaturation processes de novo would lead to the formation of –ω3 unsaturated fatty acids in higher levels. Intake of these diets enriched with –ω3 acids may explain the mode of accumulation of these polyunsaturates in considerable levels in the different parts of lingulid brachiopod. The α-linolenic acid (ALA 18: 3ω3), the primary precursor molecule for the –ω3 family of fatty acids in animal tissues must come from diet. The gut content analysis revealed that food contents of *L. anatina* have been found to include fragmented mangrove leaves, detritus and planktonic components which is in tune with the earlier findings conducted by Emig (1997). The presence of EPA and DHA within different body parts of *L. anatina* has strengthened the fact that these pharmaceutically important fatty acids are thought to have been derived from ALA, present in dietary food sources of *L. anatina* through biotransformation pro-

Table 23.16 Fatty acid compositions of TL of different body parts of *Lingula anatina* and its primary food sources (Plankta, mangrove leaves and detritus) as determined by GLC of methyl esters

Components[a]	Lophophore	Pedicle	Muscle	Gut	Plankton	Mangrove leaves	Detritus
14:0	1.4	0.9	4.5	3.1	9.0	2.1	6.5
15:0	0.9	1.2	1.3	1.3	1.3	4.2	2.4
16:0	23.0	23.4	14.7	23.2	20.4	87.0	38.2
17:0	3.3	2.9	1.8	1.9	1.1	2.1	0.6
18:0	21.3	19.8	11.7	10.1	9.5	13.5	4.4
22:0	0.7	0.8	0.5	1.0	0.3	0.2	0.4
24:0	2.7	2.8	1.2	0.6	1.3	0.4	0.7
Total SAFA	**53.3**	**51.8**	**35.7**	**41.2**	**42.9**	**109.5**	**53.2**
14:1					0.3		2.7
15:1		1.0			0.1	0.8	0.6
16:1	4.5	1.5	8.3	8.7	10.8	4.6	11.0
17:1	0.7	5.2	1.3	1.3	1.7	0.7	0.3
18:1ω9	4.2	3.4	8.5	7.0	5.3	29.8	13.20
22:1	0.5	0.7	1.1	0.2	0.2		0.1
24:1	0.6	0.9	0.4		0.2		
Total MUFA	**10.5**	**12.7**	**19.6**	**17.2**	**18.6**	**35.9**	**27.9**
16:2	0.1	0.6	0.1	0.2	0.2	0.4	1.1
17:2	2.3	0.5	0.1	0.1	2.2		
18:2ω6	1.2	1.2	1.8	1.9	1.9	49.50	9.0
18:3ω6	0.1			0.2	0.3	1.4	1.5
18:3ω3	**1.0**	**0.7**	**3.5**	**2.5**	**3.1**	**100.70**	**2.7**
20:3ω3	0.1	0.5	0.1	0.1	0.1		0.3
20:4ω6	10.4	9.6	7.6	5.3	2.6		0.2
20:4ω3	2.4	2.3	2.0	2.1	1.9	0.8	1.3
22:4ω6	0.5		0.8	0.8	0.6	0.18	
20:5ω3	**10.1**	**12.7**	**14.4**	**15.8**	**12.5**		**1.2**
21:5ω3	0.2		0.04		0.1		
22:5ω6	0.7	0.9	0.6	2.1	0.2		0.04
22:5ω3	1.3	1.6	2.4	1.6	1.0		0.02
22:6ω3	**5.2**	**4.5**	**10.3**	**7.9**	**11.1**		**0.2**
Total PUFA	**35.6**	**35.1**	**43.74**	**40.6**	**37.8**	**152.98**	**17.56**
Total –ω3	**20.1**	**23.1**	**32.7**	**30.0**	**29.7**	**101.5**	**5.72**
Total –ω6	**12.9**	**11.7**	**10.8**	**10.3**	**5.6**	**51.08**	**10.74**
PUFA/SAFA	**0.67**	**0.67**	**1.23**	**0.98**	**0.88**	**1.39**	**0.33**

[a]First and second figures represent, carbon chain length: number of double bonds. The −ω values represent the methyl end chain from the center of double bond furthest removed from the carboxyl end

% w/w of each component in total fatty acids

Table 23.17 Fatty acid compositions of Neutral Lipids (NL), Glycolipids (GL), and Phospholipids (PL) obtained from Total lipids (TL) of muscles of *Lingula anatina* as determined by GLC of methyl esters

Components[a]	Muscle/NL	Muscle/GL	Muscle/PL
14:0	8.5	4.9	1.0
15:0	0.3		0.4
16:0	19.9	29.5	21.5
17:0	1.5	2.6	1.8
18:0	9.3	16.8	18.0
22:0	1.7	0.8	0.4
24:0	1.9	2.2	4.5
Total SAFA	**43.1**	**56.8**	**47.6**
15:1		0.1	0.4
16:1	9.0	6.0	2.3
17:1	0.1	0.3	1.6
18:1ω9	7.2	5.9	3.0
22:1	0.3	0.1	0.3
24:1	0.7	0.2	0.9
Total MUFA	**17.3**	**12.6**	**8.5**
16:2	1.3	0.1	0.7
17:2	0.4	0.1	0.3
18:2ω6	1.2	1.3	0.9
18:3ω6	0.1	0.1	
18:3ω3	2.2	2.0	**1.3**
20:3ω3	**0.1**	**0.2**	0.2
20:4ω6	10.3	4.8	10.8
20:4ω3	1.3	2.0	2.6
22:4ω6	0.7	0.5	0.4
20:5ω3	11.5	9.9	**13.9**
21:5ω3	**0.1**	**0.1**	0.03
22:5ω6	0.9	0.6	1.2
22:5ω3	1.2	1.4	2.0
22:6ω3	6.2	3.8	**7.7**
Total PUFA	**37.5**	**26.9**	**42.03**
Total –ω3	**22.6**	**19.4**	**27.73**
Total –ω6	**13.2**	**7.3**	**13.3**
PUFA/SAFA	**0.87**	**0.47**	**0.88**

[a]First and second figures represent, carbon chain length: number of double bonds. The −ω values represent the methyl end chain from the center of double bond furthest removed from the carboxyl end

% w/w of each component in total fatty acids

cesses within the body of this rare benthic brachiopodan faunal component. It was recorded that phospholipids obtained from muscles of *L. anatina* were the major classes of lipids which form structural and functional components of cell membranes. Presence of moderate to high levels of EPA and DHA, within different body parts of studied species, derived through bioconversion of ALA from food sources indicated that they have been the good sources of EPA and DHA which are being considered as the precursors of several metabolites that are potent lipid mediators. Many investigators recognize them as to be the beneficial components for human being as in the prevention or treatment of several diseases (Serhen et al. 2008; Samanta et al. 2014a, b, c; Samanta et al. 2015b). Recent investigation has revealed that muscles of the studied animal, *L. anatina* stored major amount of all fatty acids**.** Presence of EPA and DHA in plankton samples indicated that *L. anatina* obtains these fatty acids from the planktons as food source. Major mangrove plant leaves of the studied area have been found to possess moderate to high amount of α-linolenic acid (18:3ω3) which is the precursor of long chain PUFAs viz. EPA (20:5ω3) and DHA (22:6ω3). Presence of qualitatively similar type of fatty acids in *L. anatina*, inhabiting in three contrasting study sites of the Subarnarekha mangrove estuarine complex has enabled to arrive at a conclusion that the occurrence of different morphotypic forms of *L. anatina* as observed during present study belong to same genus and species (Samanta et al. 2014a).

Presence of high levels of carnivorous markers of the studied species i.e. oleic acid (18:1ω9, derived from animal sources because of the consumption of zooplankton, animal detritus etc., occurred in the intertidal belts) in different body parts of *L. anatina* have indicated that all those fauna are the co-inhabitants of the habitat. Presence of high amount of 22:6ω3 and 20:4ω6 in different body parts of *L. anatina* has established the facts that diatoms, dianoflagellates and macroalgae tended to constitute the basal portion of food pyramid of this complex estuarine ecosystem (Samanta et al. 2015a, b). (Figs. 23.23, 23.24, and 23.25).

23.2.15.2 Bioconversion and Biotransformation of Fatty Acids by an Intertidal Macrobenthos (Fiddler Crab), *Uca acuta acuta*: (Tables 23.18 and 23.19)

The *Uca* species of the present study grazes on the leaf detritus (Heald 1969; Heald and Odum 1975; Chatterjee et al. 2014) and thus consumes considerable amount of ALA (18:3ω3). Analysing of fatty acid compositions of some organs viz. body flesh, big chela flesh, *Hepatopancreas* and gut content of an abundant fiddler crab species, *Uca acuta acuta*, feeding mostly on the detritus, have revealed the occurrence of different fractional fatty acid components, as shown in the Table 3 which are mostly derived from the decomposition product of mangrove leaves.

Fig. 23.23 Percentage of Total Lipid (TL) obtained from various body parts of *Lingula anatina*

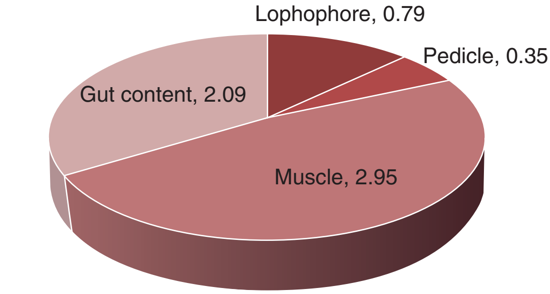


Fig. 23.24 Percentage of Total Lipid (TL) obtained from various primary food sources of *Lingula anatina*

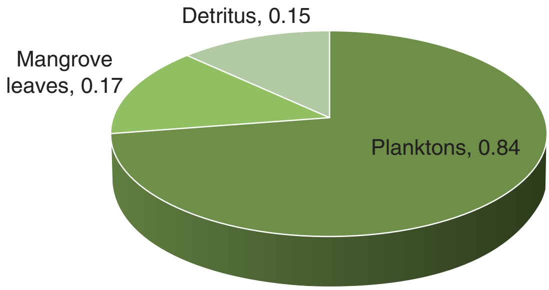
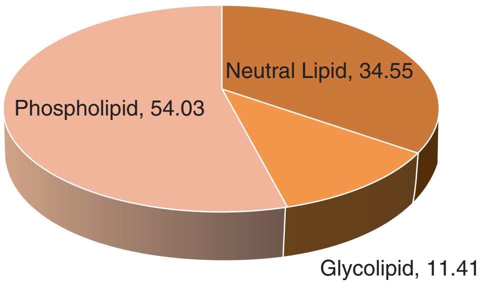


Fig. 23.25 Percentage of Neutral Lipid (NL), Glycolipid (GL), and Phospholipid (PL) obtained from Total Lipid (TL) of Muscle sample of *Lingula anatina*



The most common constituent of the soluble lipids of the plant cuticle are wax esters, defined as fatty acid esters of fatty alcohols (Alikunhi et al. 2010). Generally the unsaturated fatty acids of plant leaf surface waxes of mangrove plant leaves composed of mono-, di- and trienoic moieties containing 18-carbon chains (Misra 1987). This observation has been also confirmed in the present study. Besides, special mention can be made on α-linolenic acid (ALA, 18:3ω3) which is the precursor of the long chain polyunsaturated fatty acids (PUFA's) of –ω3 series, viz., eicosapentaenoic acid (EPA, 20:5ω3) and docosahexaenoic acid (DHA, 22:6ω3. The *Uca* species of the present study grazes on the leaf detritus (Grimes et al. 1989; Ringold 1979) and thus consumes considerable amount of ALA (18:3ω3). Most of the unsaturated fatty acids have been utilized by various organisms including *Uca.*, whereas a part of those washed out by the tidal water (Heald 1969; Heald and Odum 1975). Fatty acid compositions of the different organs viz. Body muscle, Largest

Table 23.18 Fatty acid compositions of Total Lipid (TL) from various parts of *Uca acuta acuta* (Fiddler crab) as determined by GLC of methyl esters

Components[a]	Body flesh	Big Chela flesh	Gut content	Hepato pancreas
13:0		1.0		0.2
14:0	2.2	4.3	4.0	6.9
14:1				0.1
15:0	2.3	4.5	1.8	0.4
15:1	0.7		0.3	0.3
16:0	21.6	18.7	25.5	13.6
16:1	9.9	8.0	17.4	10.3
16:2	0.2		2.0	2.5
17:0	2.5	1.3	2.6	3.0
17:1	0.6	2.6	1.1	3.9
18:0	9.4	12.4	10.2	10.7
18:1ω9	6.3	7.2	7.3	9.6
18:2ω6	2.9	4.7	3.9	4.4
18:3ω6	0.5	0.4	0.5	1.2
18:3ω3	0.3	0.3	0.4	1.0
20:3ω3	0.2	1.5	0.3	0.1
20:4ω6	0.1		0.1	0.7
22:0	0.4	0.1	0.6	0.3
22:1	0.1	1.0	0.2	0.3
20:4ω3	6.2	8.4	7.6	9.8
22:4ω6	0.1	0.1	0.1	0.2
20:5ω3	22.3	14.3	10.3	13.3
21:5ω3	0.1			0.2
22:5ω6				0.007
24:0	0.7	0.5	0.3	1.0
24:1	0.4	1.0	0.6	0.7
22:5ω3	0.1	0.2	0.1	0.2
22:6ω3	9.3	7.0	2.5	4.8

[a]First and second figures represent, carbon chain length: number of double bonds. The −ω values represent the methyl end chain from the center of double bond furthest removed from the carboxyl end

% w/w of each component in total fatty acids

chella flesh, Hepatopancreas and Gut content of the studied crab, *Uca acuta acuta*, which feeds on the detritus are presented in the Table 23.3.

Different organs of *Uca acuta acuta* showed the presence of α-linolenic acid (18:3ω3) in considerable amounts (about 30–40%) which constitute the leaf lipids of major plants of this area. Alfa linolenic acid (ALA, 18:3ω3) is the primary precursor molecule for the –ω3 family of fatty acids in animal tissues, must come from the diet i.e., from the plant leaves. On the other hand, the detritus contained only 2.7% of ALA. This signifies that the leaf litters formed out of breakdown of mangrove leaves and after being exposed to tidal waters, and microbial activities undergo decomposition process and detritus is formed (Heald 1969; Odum 1975). Since

Table 23.19 Fatty acid compositions of Neutral Lipid (NL), Glycolipid (GL) and Phospholipid (PL) from body flesh of *Uca acuta acuta* (Fiddler crab) as determined by GLC of methyl esters

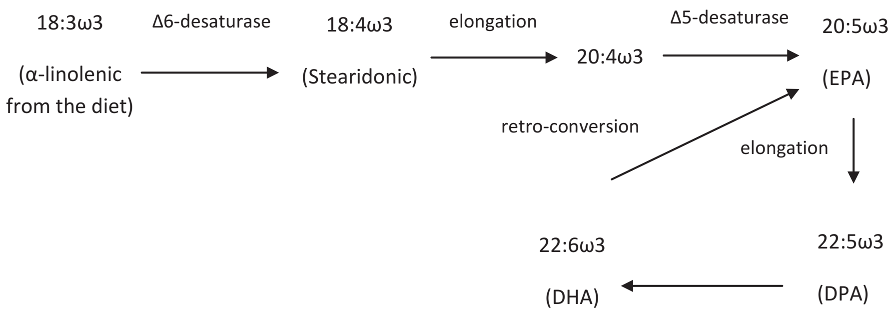
Components[a]	Body flesh/NL	Body flesh/GL	Body flesh/PL
14:0	10.4	3.6	0.4
15:0	0.5	0.9	0.2
15:1		1.5	0.2
16:0	48.9	32.6	20.0
16:1	19.0	7.9	7.4
16:2	0.9		0.1
17:0	0.6	3.0	0.5
17:1	2.2	2.6	0.1
17:2		0.1	
18:0	4.5	25.0	13.2
18:1ω9	3.2	4.4	7.2
18:2ω6	1.7	2.2	3.3
18:3ω6	0.6	0.3	0.7
18:3ω3	0.3	3.6	0.3
20:3ω3	0.04		0.3
20:4ω6	0.6		0.1
22:0	0.1		0.6
22:1		0.6	0.4
20:4ω3	1.2	4.1	8.2
22:4ω6	0.1	0.6	0.1
20:5ω3	3.6	1.1	22.0
21:5ω3	0.009	1.4	0.2
22:5ω6	0.1		
24:0	0.05	0.2	1.0
24:1	0.2	2.5	0.7
22:5ω3	0.01	0.4	0.3
22:6ω3	0.5	0.6	11.9

[a]First and second figures represent, carbon chain length: number of double bonds. The −ω values represent the methyl end chain from the center of double bond furthest removed from the carboxyl end

% w/w of each component in total fatty acids

only a small portion (5%) of the leaf material was found to have been removed by grazing insects before leaf abscission, most of the material become widely dispersed by seasonal currents. A key group of small animals, comprising only a few species but very large number of individuals, (in the present study *Uca acuta acuta* being one) ingest large quantities of the vascular plant detritus, and thus participates in the food chain. It is clear that the *Uca acuta acuta* species consumes considerable amounts of ALA (18:3ω3), by grazing on the detritus.

The α-linolenic acid (ALA, 18:3ω3) is the primary precursor molecule for the –ω3 family of fatty acids (Gunstone et al. 2002) in animal tissues, must come from the diet i.e., plant leaf detritus, as in the present study. The main pathways to the formation of eicosapentaenoic acid (EPA, 20:5ω3) and docosahexaenoic acid (DHA, 22:6ω3) requires a sequence of chain elongation and desaturation steps (Δ5 and Δ6 desaturases) with acyl-coenzymeA esters as substrates (Gurr et al. 2002). Thus α-linolenic acid is elongated and desaturated with double bonds being inserted between existing double bonds and carboxyl group. Also, retro-conversion of 22:6ω3 to 20:5ω3 may take place in the peroxisomes of the cells by removal of the first two carbon atoms by a process of β-oxidation under circumstanses, as shown below:



Only fragmentary studies have so far been made on the transmission of plant biochemicals to detritivore animals and their bio-converion thereupon. Of the studies so far done, in the Sunderbans mangrove forest, mention can be made of the fatty acids and sterols of mangrove leaves (Misra et al. 1984, 1986), triterpenoids and sterols of mangrove plant leaves (Ghosh et al. 1985) and fatty acids of the detritivores, *Boleophthalmus boddarti* (Banerjee et al. 1997) and biotransformation of oleanolic acid to oleanonic acid of mangrove leaves (Misra et al. 1986).

Based on the investigation on the role of a dominant fiddler crab species, *Uca acuta acuta* on the biotransformation of plant biochemicals especially of lipid and fatty acid components from plants to detritus and there from to benthic fauna such as fiddler crab, of Midnapore (East) coastal tract, West Bengal, India the following inferences can be drawn:

1. The mangrove plant leaves of the study area contains high levels of α-linolenic acid (18:3ω3), which is a precursor of long chain PUFA's, viz., EPA (20:5ω3) and DHA (22:6ω3).
2. *Uca acuta acuta* is an important bio-energetically significant animal on the leaf detritus, and thus consumes considerable amount of α-linolenic acid (ALA, 18:3ω3).
3. The ALA thus consumed by this abundant grazing animal in the estuarine mud flat in this area, are converted *in vivo* to long chain polyunsaturated fatty acids of –ω3 series, viz., EPA and DHA.
4. Considerably high levels of EPA and DHA have been found in the various samples of *Uca acuta acuta*, particularly in the body flesh of the animal.

5. Through this study, it has been established that, *Uca acuta acuta* are capable of biosynthesizing long chain PUFA's efficiently (Das et al. 2014a, b).

Of the studies so far done in the Sunderbans mangrove forest, mention can be made of the fatty acids and sterols of mangrove leaves (Misra et al. 1984, 1986), triterpenoids and sterols of mangrove plant leaves (Ghosh et al. 1985) and fatty acids of the detritivores, *Boleophthalmus boddarti* (Banerjee et al. 1997) and biotransformation of oleanonic acid of mangrove leaves (Misra et al.1986).

23.2.15.3 Bioconversion and Biotransformation of Total Lipids of Muscles of an Intertidal Benthic Nemertean, *Cerebratulus sp. alongwith* Its Primary Food Sources

Fractional components of fatty acids as represented in Tables 23.20 and 23.21 showed that food components of *Cerebratulus sp.* included appreciable amount of α- linolenic acid (ALA). Muscles of *Cerebratulus sp.* have shown to contain 9 different types of MUFAs and 12 different types of PUFAs. Among PUFAs arachidonic acid (AA, 20:4ω6) registered highest amount (5.00%) followed by eicosatrienoic acid (ETE, 20:3ω3) 4.48%, docosahexaenoic acid (DHA, 22:6ω3) 2.89% and so on. However, mangrove leaves did not reveal the occurrence of AA, ETE and DHA. The quantity of linoleic acid (LA, 18:2ω6) exhibited moderate to high amount (1.84–49.50%) in different studied samples.

Fatty acids have been used as qualitative markers to trace or confirm predator prey relationships in the marine environment for more than 30 years (Prato et al. 2012). More recently, they have also been used to identify key processes impacting the dynamics of some of the world's major ecosystems. The fatty acid trophic marker concept is based on the observations that marine primary producers lay down certain fatty acids which may, be transferred conservatively to, and hence can be recognized in, primary consumers (Dalsgaard et al. 2003).

In a marine ecosystem, generally qualitative similarities are observed in the fatty acid composition of the organisms which occupy different trophic levels. The first link of the food chain i.e. phytoplankton are able to synthesize all the fatty acids de novo and composition of fatty acids changed significantly in the decomposing leaves of mangroves (Moreno et al. 1979; Alikunhi et al. 2010). High concentration of −ω3 fatty acids which are generally considered as to be fatty acids of marine life, were found to have been mainly contributed by some phytoplanktonic species (diatoms, dianoflagellates etc.) to the marine ecosystem (Sargent et al. 1976). Examination on muscles of *Cerebratulus sp.*, an intertidal detritivorous macrobenthic nemertine of the studied areas revealed the presence of appreciably high amount of 20:5ω3 fatty acids associated with other polyunsaturated fatty acids of the –ω3 series. It is thus envisaged that intake of higher ammount of the precursor acid (18:3ω3), through primary food sources and their subsequent chain elongations and desaturation processess de novo would lead to the formation of –ω3 unsaturated fatty acids in higher levels. Intake of these diets enriched with –ω3 acids may explain the mode of accumulation of these PUFAs in variable amounts in the body parts (muscles) of studied macrobenthic estuarine fauna.

Low values of the PUFA/SAFA ratio as determined in the present research investigation are because of the presence of higher levels of palmitic acid (16:0), suggesting a contribution of vegetal detritus in the diet of *Cerebratulus sp.* PUFAs in green algae predominantly comprised of 18:2ω6 and 18:3ω3 and these fatty acid compositions are similar to those of terrestrial (Vascular) plants since they have common ancestors. In the present study, an appreciable amount of 18:2ω6 and 18:3ω3 have been estimated indicating that the species under study used to consume considerable amount of green algae (phytoplankton) occurring over the surface of the soil and also from the supply of neighboring mangrove vegetations. Terrestrial organic matters can also be associated with bacteria or fungi and constitutes an attractive and energetically utilizable food sources for invertebrates (Barlocher and Corkum 2003). In the present study, the odd branched fatty acids have been recorded as an indicator of bacterial derivative which highlights a source of food supply for *Cerebratulus* from decaying organic matters. Presence of highest amount of AA (20:4ω6, Table 23.2) in the muscles of studied species further indicated that detritus serves as one part of food of *Cerebratulus*. Presence of moderate to high levels of EPA and DHA, within muscles of studied species, derived through bioconversion of ALA from food sources indicated that they have been the good sources of EPA and DHA which are being considered as the precursors of several metabolites that are potent lipid mediators. Many investigators recognize them as to be the beneficial components for human being as in the prevention or treatment of several diseases (Serhan et al. 2008).

Lipid has been recognized as essential component in animal nutrition as well as aquaculture feed. Therefore, deposition of lipid (Fatty acid) which was found as a major constituent in methanolic extract of C*erebratulus sp*. might be obtained from their food. These compounds help defend *Cerebratulus sp.* against predators. The invented chemical defenses are thought to provide ecological advantages and may function as a driving force in the evolution of this group (Cimino and Ghiselin 1998; Prachi et al. 2012). The research investigation in the costal tract of Midnapore (East) has revealed that muscles of the studied animal stored major amount of all fatty acids and presence of EPA and DHA in plankton samples indicated that *Cerebratulus* obtains these fatty acids from the planktons as food source. Major mangrove plant leaves of the studied area have been found to possess moderate to high amount of α-linolenic acid (18:3ω3) which is the precursor of long chain PUFAs viz. EPA (20:5ω3) and DHA (22:6ω3). Presence of high levels of carnivorous markers of the studied species i.e. oleic acid (18:1ω9, derived from animal sources because of the consumption of zooplankton, animal detritus etc., occurred in the intertidal belts) in muscles of *Cerebratulus* have indicated that they are the inhabitants of the studied ecotone. Presence of high amount of 22:6ω3 and 20:4ω6 in animal samples has established the facts that diatoms, dianoflagellates and macroalgae constitute the basal portion of food pyramid of this complex estuarine ecosystem (Ghorai et al. 2015).

The Eicosatrienoic acid has been reported to induce spawning in the male lugworm, *Arenicola marina* (Pacey and Bentley 1992). The heighest amount ETE

Table 23.20 Percentage of Total Lipid (TL) obtained from muscles of *Cerebratulus sp.* and its primary food sources

Sample	Amount taken	Total lipid obtained	Percentage of total lipid (w/w)
Muscles	5.23 g	125.34 mg	2.39
Planktons	3.81 g	24.73 mg	0.64
Mangrove leaves	15.22 g	29.34 mg	0.19
Detritus	8.92 g	16.39 mg	0.18

among all detected fatty acids during present investigation indicated the fact that this particular FA is thought to play important role in their reproductive strategy during breeding period. The ability of PUFAs, particularly GLA (γ-linolenic acid), recorded during present FA analysis has tended to enhance free radical generation and lipid peroxidation process specifically in tumor cells which is supposed to be because of their tumoricidal actions (Kirubakaran et al. 2011).

An investigation on the mode of occurrence of different classes of fatty acids in body muscles of a mangrove- estuarine benthic macrofauna, *Cerebratulus sp.* inhabiting in an ecotone, at the confluence of an estuary (Subarnarekha) with Bay of Bengal (longitude 87°5′E to 88°5′E and latitude 20°30′N to 22°2′N) in the North-East coast of India, has shown the variabilities in the amount of different fatty acid components in the muscles of studied species and its food sources (Tables 23.20 and 23.21; Fig. 23.26) which have prompted to arrive at a conclusion on the mode of biotransformations and bioconversions of these bioactive substances.

23.3 Discussion

Humans have always depended on nature for environmental assets like clean water, nutrient cycling and soil formation. Human domination of the biosphere has been rapidly altering the composition, structure and functions of ecosystems (Vitousek et al. 1997), often eroding their capacity to provide services critical to human survival (Palmer et al. 2004). A recent classification of ecosystem services divides them into four categories: Provisioning services (food, fuel, fibre, timber etc.); Regulating services (climate, flood control etc.); supporting services (pollination, soil formation and other ecological properties and processes on which biodiversity and other ecosystem services depend) and Cultural services (recreational, spiritual and aesthetic values) (Millenium Ecosystem Assessment 2003). However, ecological understanding is being used to chalk out strategies for conservation and management (Balmford et al. 2003). A recent global analysis by the Millenium Ecosystem Assessment (2005) has concluded that well over half of the world's ecosystem services are being degraded or used unsustainably.

Economists and ecologists are just beginning to credibly assign economic value to ecosystem processes, transforming these processes into tradable, marketable services. This approach has triggered the intersest to assess the relative importance of

Table 23.21 Fatty acid compositions of Total Lipids (TL) of muscles of *Cerebratulus sp.* and its primary food sources (Plankta, mangrove leaves and detritus) as determined by GLC of methyl esters

Components[a]	Muscles	Planktons	Mangrove leaves	Detritus
14:0	1.00	9.0	2.1	6.5
15:0	1.26	1.3	4.2	2.4
16:0	7.96	20.4	87.0	38.2
17:0	1.71	1.1	2.1	0.6
18:0	5.32	9.5	13.5	4.4
20:0	2.01			
21:0	0.27			
22:0	0.62	0.3	0.2	0.4
23:0	0.21			
24:0	0.13	1.3	0.4	0.7
Total SAFA	**20.49**	**42.9**	**109.5**	**53.2**
14:1	0.86	0.3		2.7
15:1	0.20	0.1	0.8	0.6
16:1	1.66	10.8	4.6	11.0
17:1	0.80	1.7	0.7	0.3
20:1	0.29			
18:1ω9	3.02	5.3	29.8	13.20
22:1	0.57	0.2		0.1
22:1ω9	0.57			
24:1	0.66	0.2		
Total MUFA	**8.63**	**18.6**	**35.9**	**27.9**
16:2		0.2	0.4	1.1
17:2		2.2		
20:2	0.30			
22:2	0.43			
18:2ω6	1.84	1.9	49.50	9.0
18:3ω6	0.46	0.3	1.4	1.5
18:3ω3	**1.36**	**3.1**	**100.70**	**2.7**
20:3ω3	4.48	0.1		0.3
20:3ω6	0.00			
20:4ω6	5.00	2.6		0.2
20:5ω3	**0.52**	12.5		1.2
22:6ω3	**2.89**	11.1		0.2
Total PUFA	**17.28**	**34**	**152**	**16.2**
Total –ω3	**9.25**	**26.8**	**100.7**	**4.4**
Total –ω6	**7.3**	**4.8**	**50.9**	**10.7**
PUFA/SAFA	**0.84**	**0.79**	**1.39**	**0.30**

[a]First and second figures represent, carbon chain length: number of double bonds. The –ω values represent the methyl end chain from the center of double bond furthest removed from the carboxyl end

% w/w of each component in total fatty acids

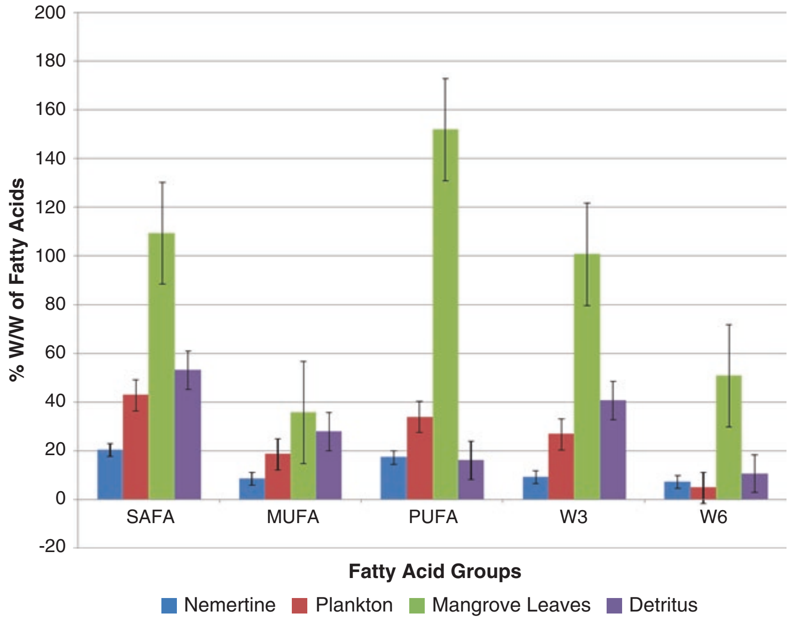


Fig. 23.26 Fatty acid composition in Nemerteans and their food materials

ecosystem processes and other forms of capital (physical, social, cultural, intellectual). Economic valuation need not cover all values of ecosystems; progress is made simply by capturing values that are presently egregiously overlooked. As a society, the modern world routinely uses cost–benefit analyses to judge between alternative pathways.

The valuation of ecosystem services facilitates difficult decisions about how to proceed with development to keep up with ever-expanding human populations (Chakraborty et al. 2014).

Several attempts have been made to classify and exhaustively list ecosystem services. There is no present consensus on a useful taxonomy, though the Millenium Ecosystem Assessment (2005) has established groupings that align ecosystem functions with goods and services. For example, the Millenium Ecosystem Assessment (2005) has identified functions as provisioning (food, water, fuel, fibre), regulating (prevention of soil erosion, flood control), cultural (recreation, spiritual value, sense of place) or supporting (soil formation, nutrient cycling, oxygen from photosynthesis).

The paradigm of ecosystem services is not accepted by everyone, and even among those who champion this new way of thinking about nature, differences of opinion can be dramatic. For example, some scientists suggest a virtual one-to-one mapping between ecosystem services and biodiversity. Others point out that there are many cases where low diversity systems can provide tremendous ecosystem services, and that modest losses of biodiversity may not substantially undermine ecosystem services. The point of economic valuation is to estimate the importance of changes in ecosystem services to human welfare. Many of these services have already been targets of valuation studies, including finfish and non finfish commercial and subsistence fisheries, timber and firewood, dyes, tourism, medicinal remedies, education, physical coastal protection, carbon storage, and even existence and option values.

Estuarine and coastal ecosystems (ECEs) are some of the most heavily used and threatened natural systems globally (Worm et al. 2006; Halpern et al. 2008). Their deterioration due to human activities is intense and increasing; 50% of salt marshes, 35% of mangroves, 30% of coral reefs, and 29% of seagrasses are either lost or degraded worldwide (Valiela et al. 2001; MEA 2005; UNEP 2006). This global decrease in ECEs is known to affect at least three critical ecosystem services (Worm et al. 2006): the number of viable (non collapsed) fisheries (33% decline); the provision of nursery habitats such as oyster reefs, seagrass beds, and wetlands (69% decline); and filtering and detoxification services provided by suspension feeders, sub merged vegetation, and wetlands (63% decline). The loss of biodiversity, ecosystem functions, and coastal vegetation in ECEs may have contributed to biological invasions, declining water quality, and decreased coastal protection from flooding and storm events (Braatz et al. 2007; Cochard et al. 2008; Koch et al. 2009).

An important issue for valuing certain ECE services, such as coastal protection and habitat-fishery linkages, is that the ecological functions underlying these services vary spatially and temporally. Allowing for the connectivity between ECE habitats also may have important implications for assessing the ecological functions underlying key ecosystems services, such coastal protection, control of erosion, and habitat-fishery linkages.

Based on such observations, an action plan for protecting and enhancing the immediate and longer-term values of ECE services has to be developed as because the connectivity of ECEs across land-sea gradients also influences the provision of certain ecosystem services, and management of the entire seascape will be necessary to preserve such synergistic effects. Other key elements of an action plan include further ecological and economic collaborative research on valuing ECE services, improving institutional and legal frameworks for management, controlling and regulating destructive economic activities, and developing ecological restoration options.

Ecosystem management is based on understanding of how natural systems work and how human activities may influence these systems. Management is also about identifying the values we wish to protect and the economic costs or gains of preservation. This suggests that the evaluation of ecosystem goods and services, from both economic and ecological perspectives, is a necessary ingredient in practical policy. Biodiversity has a fundamental role in providing the basis for all ecosystem goods and services, although detailed understand ing of the complex underlying mechanisms is still limited. Some general aspects of biodiversity do, however, link directly to goods and services. People's perception spanning several decades are now being utilised not only to understand the trend of change of biodiversity but also to ensure proper conservation strategies (Patra et al. 2005; Mishra et al. 2009).

Intertidal benthic faunal diversity dominate coasts and estuaries, and the resident flora and fauna play important roles that influence ecosystem services such as nutrient and sediment transport and primary and secondary productivity (Herman et al. 1999; Levin et al. 2001). These habitats are subject to a wide range of stresses associated with changing patterns of resource use and direct exploitation (Chakraborty 1998; Gray 1997; Qasim et al. 1988). Consequently, there is a pressing need to gain a better understanding of the relationships between diversity and ecosystem processes in shallow coastal habitats (Solan et al. 2004).

To understand the functional relationships between organisms and fluxes of energy or matter in soft sediments, the demand of the time is to bridge the gap between the spatial and temporal variability in the distribution of species and their behaviour by gathering information on the transport and transformation of chemicals within the sediment and across the sediment-water interface (Dye and Lasiak 1986). Interactions between bioturbation and mineralisation processes in sediments are highly nonlinear and are characterized by the presence of strong feedback loops among macrofauna, their food, and their chemical environment (Huhta 2007; Gray and Elliott 2009).

In such context, the functional role of benthic fauna is not only to sustain the balance in the ecosystem functioning but also to ensure livelihood generation by way of ecological services.

23.4 Concluding Remarks

The intertidal benthic fauna in the Midnapore (East) coast, West Bengal have been found to render valuable ecological services by virtue of their position and functional roles in the food chains and food webs of the mangrove estuarine coastal wetlands. These services include stabilisation of the sediments alongside enriching the biodiversity by facilitating the development of conducive ecological condition for the settlement and flourishing of biota such as algae, microbes (bacteria and fungi), meiobenthos etc. because of the pumping of sufficient air, and water through the formation of innumerable pores, tubes, burrows etc., reshuffling of nutrients and sediment texture by bioturbation activities, releasing of millions of larvae in the form meroplanktons in the aquatic subsystem and thereby providing live food to the fishes, and biotransfomation of organic chemicals into inorganic and vice-versa. All of these have contributed in a holistic manner to the sustenance of mangroves and their associated biodiversity components which in turn serve a number of ecological services. Alongside providing clean air and water, other ecological services rendered by mangrove ecosystem include protection from flood, cyclones and others natural disasters, reduction of salt water intrusion to agricultural lands and ground freshwater supplies, arresting soil erosion, recycling nutrients, filtering pollutants, regulating water flows and supplies, maintaining biodiversity and contributing to carbon sequestration. Mangrove forest can sequester far more carbon per hectare than tropical rainforests or marshes with the ability to store huge amount of carbon (about 1000 tons per hectare) over thousand of years (UNEP 2014).

It has been estimated that with the current rate of loss, people are expected to be deprived of the various values obtained from mangrove ecosystem services within the next 100 years (UNEP 2014). Thus it is of major importance to conserve these irreplaceable coastal wetland ecosystems in order to avoid the adverse consequences caused by their degradation (UNEP 2014).

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Erratum to: Land Use and Occupation of Coastal Tropical Wetlands: Whale Coast, Bahia, Brazil

Sirius O. Souza, Cláudia C. Vale, and Regina C. Oliveira

Erratum to:
Chapter 5 in: C.W. Finkl, C. Makowski (eds.), *Coastal Wetlands: Alteration and Remediation*, Coastal Research Library 21, https://doi.org/10.1007/978-3-319-56179-0_5

In chapter 5 titled **"Land Use and Occupation of Coastal Tropical Wetlands: Whale Coast, Bahia, Brazil"**, the first name of the second co-author was presented as "Câmara" C. Vale, by oversight. The correct first name is "Cláudia" and the name has been updated as Cláudia C. Vale in the chapter header and opener, Table of Contents, and Contributors list.

Figure 5.15 on page 178 has been cut off at the bottom and it has been repositioned.

The updated online version of this chapter can be found at https://doi.org/10.1007/978-3-319-56179-0_5

C.W. Finkl, C. Makowski (eds.), *Coastal Wetlands: Alteration and Remediation*,
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