Coastal Research Library 8

Charles W. Finkl
Christopher Makowski *Editors*

Environmental Management and Governance

Advances in Coastal and Marine Resources

Springer

Environmental Management and Governance

Coastal Research Library

VOLUME 8

Series Editor:

Charles W. Finkl
Department of Geosciences
Florida Atlantic University
Boca Raton, FL 33431
USA

The aim of this book series is to disseminate information to the coastal research community. The Series covers all aspects of coastal research including but not limited to relevant aspects of geological sciences, biology (incl. ecology and coastal marine ecosystems), geomorphology (physical geography), climate, littoral oceanography, coastal hydraulics, environmental (resource) management, engineering, and remote sensing. Policy, coastal law, and relevant issues such as conflict resolution and risk management would also be covered by the Series. The scope of the Series is broad and with a unique crossdisciplinary nature. The Series would tend to focus on topics that are of current interest and which carry someimport as opposed to traditional titles that are esoteric and non-controversial. Monographs as well as contributed volumes are welcomed

For further volumes:
http://www.springer.com/series/8795

Charles W. Finkl • Christopher Makowski
Editors

Environmental Management and Governance

Advances in Coastal and Marine Resources

Springer

Editors
Charles W. Finkl
Florida Atlantic University
Boca Raton, FL, USA

Coastal Education and Research Foundation (CERF)
Coconut Creek, FL, USA

Christopher Makowski
Florida Atlantic University
Boca Raton, FL, USA

Coastal Education and Research Foundation (CERF)
Coconut Creek, FL, USA

ISSN 2211-0577 ISSN 2211-0585 (electronic)
ISBN 978-3-319-06304-1 ISBN 978-3-319-06305-8 (eBook)
DOI 10.1007/978-3-319-06305-8
Springer Cham Heidelberg New York Dordrecht London

Library of Congress Control Number: 2014945759

Printed on acid-free paper

Springer is part of Springer Science+Business Media (www.springer.com)

Preface

This volume in the Coastal Research Library (CRL) considers various aspects of coastal environmental management and governance. As the world population grows, more and more people move to the coastal zone. There are many reasons for this drang to the shore, not the least of which are increased opportunities for employment and relaxation in a salubrious environment. But, as population densities increase beyond the carrying capacity of fragile coastal zones and sustainability seems ever more elusive, more than remedial measures seem required. Because governance in the coastal zone has generally failed the world over, it is perhaps time to reconsider what we are doing and how we are doing it. Depopulation of many coastal zones would be a laudable goal, but just how this might be accomplished in a socially acceptable manner is presently unknown. Perhaps some socioeconomic incentives can be devised to lure people back towards hinterlands, but until such goals or efforts are implemented there seems little choice other than trying to make things work with the present state of affairs.

This volume thus considers a range of selected advances that highlight present thought on a complex subject that invariably, one way or the other, involves consideration of coastal natural resources. Whether it is coastal hazards, sustainability of fishers and aquaculture, resolution of environmental conflicts, waste disposal, or appreciation of biophysical frameworks such as coastal karst or impactors such as fluctuating sea levels, more advanced *out of the box* thinking is required to solve today's problems. Approaches to potential solutions are sometimes based on models or perhaps more commonly on an individual's ratiocinative powers where one can deduce logical outcomes. It is unfortunate that in many cases governmental approaches to solutions are lethargic and ineffective, making it all the more imperative to suggest advanced approaches to old problems that linger on. This book thus attempts to highlight some examples of advancements in thought processes, observation, comprehension and appreciation, and better management of coastal resources.

Environmental Management and Governance: Advances in Coastal and Marine Resources is subdivided into five parts: Part I, Coastal Hazards and Beach Management-Certification Schemes; Part II, Ocean Governance, Fisheries and

Aquaculture: Advances in the Production of Marine Resources; Part III, Exploration and Management of Coastal Karst; Part IV, Coastal Marine Environmental Conflicts: Advances in Conflict Resolution; and Part V, Examples of Advances in Environmental Management: Analyses and Applications that collectively contain 17 chapters. These subdivision are, of course, artificial and meant only to help organize the material into convenient study groups. Chapters in each part are briefly described in what follows.

Part I contains three chapters that deal with coastal hazards and beach management. In Chap. 1 ("Geological Recognition of Onshore Tsunamis Deposits"), Costa, Andrade, and Dawson discuss enhancements of our abilities to recognize (paleo) tsunami specific signatures in coastal sediments through the application of diverse sedimentological techniques. They show in this chapter how it is possible, through the use of diverse sedimentological proxies, to obtain information about the presence or absence of tsunami indicators, establish their likely source, and collect valuable information about tsunami run-up, backwash or wave penetration inland. Botero, Williams, and Cabrera, in Chap. 2 ("Advances in Beach Management in Latin America: Overview from Certification Schemes"), analyze beach certification schemes as part of beach management in Latin America. These authors highlight advances in beach management in Latin America by pointing out main conceptual, methodological, and practical challenges to be achieved for scientific and decision makers of the continent. Chapter 3 ("New Methods to Assess Fecal Contamination in Beach Water Quality") by Sarva Mangala Praveena, Kwan Soo Chen, and Sharifah Norkhadijah Syed Ismail deals with an emerging paradigm for assessment of recreational water quality impacted by microbial contamination. Advances in this topic are important because recreational water is susceptible to fecal contamination, which may increase health risk associated with swimming in polluted water.

Part II also contains two chapters, but these efforts focus on broader issues of advances in ocean governance that involve new developments in coastal marine management and fisheries and aquaculture production. Chapter 4 ("New Approaches in Coastal and Marine Management: Developing Frameworks of Ocean Services in Governance") by Paramio, Alves, and Vieira delves into aspects of "Modern" and "post-Modern" views of ocean uses as a source of resources and space; for example, how economic development is now supplemented by functions the marine environment provides, such as human life and well-being. Ocean governance remains a current focus of discussion for policymakers aiming to address sustainability principles and perspectives in a more effective way. Chapter 5 ("Interaction of Fisheries and Aquaculture in the Production of Marine Resources: Advances and Perspectives in Mexico"), by the Pérez-Castañeda team (Roberto Pérez-Castañeda, Jesús Genaro Sánchez-Martínez, Gabriel Aguirrte-Guzmán, Jaime Luis Rábago-Castro, and Maria de la Luz Vázquez-Sauceda) indicates advances that are indicative of the potential value of aquaculture as a complementary productive activity that will meet the growing human demand for food from the sea. This advanced understanding is critical because, in terms of global fisheries production, the maximum fisheries catch potential from the oceans around the world has apparently been reached.

Part III contains Chap. 6 (“Advances in the Exploration and Management of Coastal Karst in the Caribbean”) by Michael J. Lace. This chapter is important because it explains that significant karst areas remain to be explored while illustrating associated landform vulnerabilities, anthropogenic effects, and range of coastal resource management and preservation initiatives that should be applied. These advances highlight unreported field research in selected island settings that support an emerging view of complex karst development.

Four chapters that deal with advances in coastal resources conflict resolution comprise Part IV. Chapter 7 (“Mud Crab Culture as an Adaptive Measure for the Climatically Stressed Coastal Fisher-Folks of Bangladesh”) by Khandaker Anisul Huq, S. M. Bazlur Rahaman, and A. F. M. Hasanuzzaman is an example of new adaptive measures for ensuring the security of food and livelihood of coastal poor people. Highlighted here is on-farm adaptive research on crab fattening/culture as a livelihood option for the fisher folks. This chapter shows how to recommend and carry out comprehensive crab culture extension programs for building capacity and improving economic conditions in climatically stressed coastal communities. Chapter 8 (“The Guadalquivir Estuary: A Hot Spot for Environmental and Human Conflicts) by the Ruiz team (Javier Ruiz, Mª José Polo, Manuel Díez-Minguito, Gabriel Navarro, Edward P. Morris, Emma Huertas, Isabel Caballero, Eva Contreras, and Miguel A. Losada) demonstrates how the application of robust and cost-efficient technology to estuarine monitoring can generate the scientific foundations necessary to meet societal and legal demands while providing a suitable tool by which the cost-effectiveness of remedial solutions can quickly be evaluated. A holistic approach to understanding the estuarine ecosystem, including its physical and biogeochemical dynamics and how these control biodiversity, is identified as the first step towards making knowledge-based decisions for sustainable use. Chapter 9 (“Shrimp Farming as a Coastal Zone Challenge in Sergipe State, Brazil: Balancing Goals of Conservation and Social Justice”) by Juliana Schober Gonçalves Lima and Conner Bailey discusses marine shrimp farming in Brazil from the perspective of both social justice and environmental conservation. Conflicts arose here because the rearing of marine shrimp became an important local economic activity that increasingly occupied large areas on the coast. Shrimp farming is practiced mainly through extensive family-based production systems in mangrove areas that were subsequently declared Permanent Preservation Areas by Brazilian law. As a result, these family shrimp farms are considered illegal, but the farms themselves long predate promulgation of the law and represent an important source of livelihood for hundreds of families. Chapter 10 (“Regional Environmental Assessment of Marine Aggregate Dredging Effects: The UK Approach”) by Dafydd Lloyd Jones, Joni Backstrom, and Ian Reach describes the MAREA (Aggregate Regional Environmental Assessment) methodology, and shows how similar regional assessment exercises could contextualize the effects and impacts of multiple marine dredging activities in other parts of the world. Each MAREA assesses the cumulative impacts of marine dredging activities using regional-scale hydrodynamic and sediment transport models linked to regional-scale mapping of sensitive receptors.

Part V contains seven chapters that consider various aspects of advances in environmental management based on examples of analyses and applications. Chapter 11 ("Advances in Large-Scale Mudflat Surveying: The Roebuck Bay and Eighty Mile Beach, Western Australia) by Robert J. Hickey, Grant B. Pearson, and Theunis Piersma deals with advances in mudflat surveying using the example of shores along Roebuck Bay and Eighty Mile Beach in northwestern Australia, the richest known intertidal mudflats in the world. Chapter 12 ("Sea-Level Indicators") by Niki Evelpidou and Paolo A. Pirazzoli illustrates how the study of relative sea-level changes is an essential element of ocean observation and technological advances that are necessary to improve the determination of levels (elevation or depth), chronological estimations, and the identification of appropriate sea-level indicators. Although levels are determined with satellites, oceanographic vessels, geophysical equipments, leveling techniques, tide-gauge devices, or even direct measurement by an observer, chronological estimations may result from radiometric analysis of samples, comparison with stratigraphic sequences, archaeological or historical data, assumptions on erosion or deposition processes, or even from glacio-isostatic or climate modeling. Indicators of fossil or present-day sea-level positions are nevertheless the most important elements for a sea-level reconstruction, because they provide information not only on the former level but also on the accuracy of the reconstruction. In Chap. 13 ("Advancement of Technology for Detecting Shoreline Changes in East Coast of India and Comparison with Prototype Behavior) by R. Manivanan, various aspects of intake/outfall of nuclear power plant on the coast, especially the dispersion of warm water discharges under different environmental conditions, is simulated using mathematical modeling techniques and suitable locations of intake and outfall with the minimum recirculation. This chapters discusses advances for optimizing the efficiency of power plants by locating the intake/outfall so there is minimum recirculation of warm water in the intake under the prevailing coastal environmental conditions. Chapter 14 ("Coastal Dunes: Changes of Their Perception and Environmental Management") by Tomasz A. Łabuz outlines coastal dune types and conditions for their development, while considering functions and practical use of coastal dunes. Of special interest here are advancing and changing attitudes to environmental management of coastal dunes that include various new approaches to use and perception of dunes that result from cultural and societal development. Chapter 15 ("Advances in Brine Disposal and Dispersion in the Coastal Ecosystem from Desalination Plants") by R. Manivanan observes brine water plume behavior in the vicinity of coastal areas with different outfall locations. This study indicates that higher velocity and larger port diameter enhances dispersion rates and minimizes adverse effects on the marine ecosystem. Chapter 16 ("Estuaries Ecosystems Health Status – Profiling the Advancements in Metal Analysis") by Ahmad Zaharin Aris and Looi Ley Juen demonstrates advanced analytical methods and detection techniques available for metals analyses. Environmental forensic approaches and application of various metal pollution indicators, indices, modeling, and statistical analysis are used to assess estuarine ecosystem health status. Chapter 17 ("Floating Offshore Wind Farms and Their

Application in Galicia (NW Spain)") by Laura Castro-Santos and Vicente Diaz-Casas provides a methodology for calculating the life-cycle costs of developing a floating offshore wind farm. This example was developed for a semisubmersible floating offshore wind platform and a general offshore wind turbine of 5 MW. The farm will be composed of 21 offshore wind turbines, with a total power of 107 MW.

While it is understood this volume does not include all advancements in the management and governance of environmental systems, a thorough selection of topics have been addressed. From coastal hazards, to ocean services, to aquaculture, this book presents a diverse cross-section of studies that provide innovative environmental stewardship on an international scale. However, these studies are only the beginning. From these new ideas spring forth new ways of thinking to effectively protect, manage, and govern fragile coastal ecosystems found around the world. By delving into original, pioneering methods and practices, as illustrated throughout this volume, true advancements are then achieved.

Coconut Creek, FL, USA Charles W. Finkl
Boca Raton, FL, USA Christopher Makowski

Contents

Contributors

Gabriel Aguirre-Guzmán Facultad de Medicina Veterinaria y Zootecnia, Universidad Autónoma de Tamaulipas, Tamaulipas, Mexico

Fátima Lopes Alves CESAM – Centre for Environmental and Marine Studies, Department of Environment and Planning, University of Aveiro, Aveiro, Portugal

César Andrade IDL, Centro and Departamento de Geologia, Faculdade de Ciências, Universidade de Lisboa, Lisbon, Portugal

Ahmad Zaharin Aris Environmental Forensics Research Centre, Faculty of Environmental Studies, Universiti Putra Malaysia, Serdang, Selangor, Malaysia

Joni Backstrom Fugro EMU Ltd, Southampton, UK

Conner Bailey Department of Agricultural Economics & Rural Sociology, Auburn University, Auburn, AL, USA

S.M. Bazlur Rahaman Fisheries and Marine Resource Technology Discipline, Khulna University, Khulna, Bangladesh

Camilo-Mateo Botero Grupo Joaquín Aaron Manjarres, Universidad Sergio Arboleda, Santa Marta, Colombia

Isabel Caballero Department of Ecology and Coastal Management, Instituto de Ciencias Marinas de Andalucía ICMAN-CSIC, Puerto Real (Cádiz), Spain

Juan Alfredo Cabrera Grupo Costatenas, Universidad de Matanzas, Matanzas, Cuba

Laura Castro-Santos Department of Naval and Oceanic Engineering, Integrated Group for Engineering Research (GII), University of A Coruña, A Coruña, Spain

Kwan Soo Chen Department of Environmental and Occupational Health, Faculty of Medicine and Health Sciences, Universiti Putra Malaysia (UPM), Serdang, Selangor Darul Ehsan, Malaysia

Eva Contreras Fluvial Dynamics and Hydrology Research Group, Interuniversity Research Institute of Earth System in Andalusia, University of Córdoba, Córdoba, Spain

Pedro J.M. Costa IDL, Centro and Departamento de Geologia, Faculdade de Ciências, Universidade de Lisboa, Lisbon, Portugal

Sue Dawson Department of Geography, School of the Environment, University of Dundee, Dundee, Scotland, UK

Vicente Diaz-Casas Department of Naval and Oceanic Engineering, Integrated Group for Engineering Research (GII), University of A Coruña, A Coruña, Spain

Manuel Díez-Minguito Environmental Fluid Dynamics Group, Andalusian Institute for Earth System Research, University of Granada, Granada, Spain

Niki Evelpidou CNRS – Laboratoire de Géographie Physique, Meudon, France

Faculty of Geology and Geoenvironment, National and Kapodistrian University of Athens, Athens, Greece

A.F.M. Hasanuzzaman Fisheries and Marine Resource Technology Discipline, Khulna University, Khulna, Bangladesh

Robert J. Hickey Department of Geography, Central Washington University, Ellensburg, WA, USA

Emma Huertas Department of Ecology and Coastal Management, Instituto de Ciencias Marinas de Andalucía ICMAN-CSIC, Puerto Real (Cádiz), Spain

Khandaker Anisul Huq Fisheries and Marine Resource Technology Discipline, Khulna University, Khulna, Bangladesh

Sharifah Norkhadijah Syed Ismail Department of Environmental and Occupational Health, Faculty of Medicine and Health Sciences, Universiti Putra Malaysia (UPM), Serdang, Selangor Darul Ehsan, Malaysia

Tomasz A. Łabuz Institute of Marine and Coastal Sciences, University of Szczecin, Szczecin, Poland

Michael J. Lace Coastal Cave Survey, West Branch, IA, USA

Juliana Schober Gonçalves Lima Department of Fisheries and Aquaculture Engineering (Núcleo de Engenharia de Pesca), NEP, Federal University of Sergipe, São Cristovão, Sergipe, Brazil

Dafydd Lloyd Jones MarineSpace Ltd, Southampton, UK

Ley Juen Looi Environmental Forensics Research Centre, Faculty of Environmental Studies, Universiti Putra Malaysia, Serdang, Selangor, Malaysia

Miguel A. Losada Environmental Fluid Dynamics Group, Andalusian Institute for Earth System Research, University of Granada, Granada, Spain

Ramasamy Manivanan Mathematical Modeling for Coastal Engineering (MMCE), Central Water and Power Research Station, Pune, India

Edward P. Morris Department of Ecology and Coastal Management, Instituto de Ciencias Marinas de Andalucía ICMAN-CSIC, Puerto Real (Cádiz), Spain

Gabriel Navarro Department of Ecology and Coastal Management, Instituto de Ciencias Marinas de Andalucía ICMAN-CSIC, Puerto Real (Cádiz), Spain

Luz Paramio CEEAplA – Centre of Applied Economics Studies of the Atlantic, University of Azores, Ponta Delgada, Portugal

Grant B. Pearson Bennelongia Environmental Consultants, Jolimont, WA, Australia

Roberto Pérez-Castañeda Facultad de Medicina Veterinaria y Zootecnia, Universidad Autónoma de Tamaulipas, Tamaulipas, Mexico

Theunis Piersma Department of Marine Ecology, NIOZ Royal Netherlands Institute for Sea Research, Den Burg, Texel, The Netherlands

Animal Ecology Group, Centre for Ecological and Evolutionary Studies, University of Groningen, Groningen, The Netherlands

Paolo A. Pirazzoli CNRS – Laboratoire de Géographie Physique, Meudon, France

Mª José Polo Fluvial Dynamics and Hydrology Research Group, Interuniversity Research Institute of Earth System in Andalusia, University of Córdoba, Córdoba, Spain

Sarva Mangala Praveena Department of Environmental and Occupational Health, Faculty of Medicine and Health Sciences, Universiti Putra Malaysia (UPM), Serdang, Selangor Darul Ehsan, Malaysia

Jaime Luis Rábago-Castro Facultad de Medicina Veterinaria y Zootecnia, Universidad Autónoma de Tamaulipas, Tamaulipas, Mexico

Ian Reach MarineSpace Ltd, Southampton, UK

Javier Ruiz Department of Ecology and Coastal Management, Instituto de Ciencias Marinas de Andalucía ICMAN-CSIC, Puerto Real (Cádiz), Spain

Jesús Genaro Sánchez-Martínez Facultad de Medicina Veterinaria y Zootecnia, Universidad Autónoma de Tamaulipas, Tamaulipas, Mexico

María de la Luz Vázquez-Sauceda Facultad de Medicina Veterinaria y Zootecnia, Universidad Autónoma de Tamaulipas, Tamaulipas, Mexico

José António Cabral Vieira CEEAplA – Centre of Applied Economics Studies of the Atlantic, University of Azores, Ponta Delgada, Portugal

Allan T. Williams Built Environment, Swansea Metropolitan University, Swansea, Wales, UK

Part I
Coastal Hazards and Beach Management-Certification Schemes

Chapter 1
Geological Recognition of Onshore Tsunami Deposits

Pedro J.M. Costa, César Andrade, and Sue Dawson

Abstract The study and understanding of coastal hazards is a fundamental aspect for most modern societies. The consequences of extreme events such as tsunamis are being regarded as major threats for coastal regions. The sedimentological record provides a database useful to characterize and evaluate recurrence of tsunamis, which contributes to assessing the vulnerability of any coastal area to this natural hazard. Thus, the enhancement of our ability to recognize (palaeo) tsunami specific signatures in coastal sediments, through the application of diverse sedimentological techniques, is of unquestionable interest.

This work reviews and discusses contributions provided by developments in the study of onshore tsunami deposits based on a group of sedimentological attributes\ characteristics.

1.1 Introduction

In addition to the long-term processes operating in a region, catastrophic inundation events such as tsunamis (and storms) can contribute significantly to the stratigraphy of any given area. In contrast to contemporary tsunami events, for which eyewitness descriptions and instrumental and field measurements of both erosional and depositional effects are utilized in modelling studies (e.g. Finkl et al. 2012), (palaeo) tsunami recognition depends on the identification of ancient tsunami deposits

P.J.M. Costa (✉) • C. Andrade
IDL, Centro and Departamento de Geologia, Faculdade de Ciências, Universidade de Lisboa, Edifício C6, Campo Grande, 1749-016 Lisbon, Portugal
e-mail: ppcosta@fc.ul.pt; candrade@fc.ul.pt

S. Dawson
Department of Geography, School of the Environment, University of Dundee, Nethergate, Dundee DD1 4HN, Scotland, UK
e-mail: s.dawson@dundee.ac.uk

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8, DOI 10.1007/978-3-319-06305-8_1, © Springer International Publishing Switzerland 2015

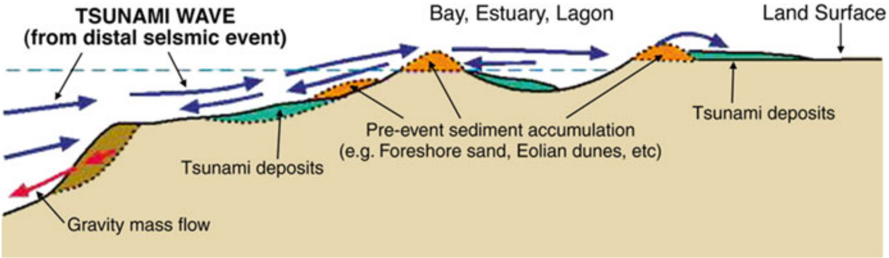


Fig. 1.1 Schematic illustration of principal pathways of tsunami sediment transport and deposition (Dawson and Stewart 2007 after Einsele et al. 1996)

(e.g. Bourgeois et al. 1988; Long et al. 1989; Smit et al. 1992; Bondevik et al. 1997; Clague et al. 2000; Dawson and Stewart 2007; Morton et al. 2011; Chagué-Goff et al. 2011; Goff et al. 2012; Goto et al. 2011a).

Tsunami deposition is usually characterized by the re-deposition of coarse shallow marine or coastal sediments in terrestrial and/or transitional (e.g. lagoonal, estuarine) environments (Fig. 1.1). Recognition of these deposits is the primary method for reconstructing tsunami minimum inundation distance and run-up, although patterns of erosion and deposition by both landward- and seaward-directed flows are complex, these patterns being further complicated by the existence of more than one wave associated with the same tsunami (Moore and Moore 1984; Synolakis et al. 1995; Bondevik et al. 1997; Le Roux and Vargas 2005; Nanayama and Shigeno 2006; Paris et al. 2007), thus introducing uncertainties in those reconstructions. In particular, because the maximum altitude at which tsunami sediments are deposited in the coastal zone is nearly always lower than the height reached by the tsunami. In fact, the upper sediment limit is generally regarded as a minimum level reached by the tsunami waves (this assumption is of crucial importance for hazard and physical and numerical modelling because sediment evidence might underestimate the maximum inland flooding penetration).

The nature of tsunami deposits varies greatly with coastal and nearshore morphology, the height of tsunami waves at the coast and run-up, and with the nature and amount of existing sediment in any coastal setting when affected by such an event. Consequently, the possible variations in sedimentary processes and products during these complex events remains poorly understood but in general a tsunami deposit will only be produced if there is a suitable supply of sediment and accommodation space in the coastal zone. More recently, the subsequent backwash has been regarded as a process of significant geomorphic and sedimentologic consequences (e.g. Hindson and Andrade 1999; Le Roux and Vargas 2005; Paris et al. 2010b), though the spatial extension of the correspondent signature is usually more restricted due to channelling effects. However, recent studies conducted in the nearshore area demonstrate the importance of the backwash process within tsunamigenic sediment transport (e.g. Goff et al. 2012). The geomorphological consequences and difficulty in differentiating tsunamis and storms in coastal dunes or barrier islands have also been addressed (e.g. Andrade 1990; Andrade et al. 2004; Regnauld et al. 2008; Goff et al. 2010b).

Due to their specific physics and particular sediment transport processes tsunami (and extreme storms) tend to leave their sediment imprint in a wide range of environments (e.g. alluvial plains, estuaries, coastal lagoons, embayments, nearshore and offshore areas) although storms usually exhibit a smaller amount of inland penetration. However, many of these environments display a low preservation potential for event deposits (Einsele et al. 1996) and the recognition of tsunami and storm deposits is constrained by the poor preservation of those deposits (or absence) in the stratigraphic record. In many cases, subsequent anthropogenic activity, the erosion characteristics of the event, the relative changes in sea level in a millennium timescale and the absence of lithological differentiation makes palaetsunami deposits difficult to identify and therefore also makes it difficult to make inferences regarding the return intervals of such events (Szczucinski 2012; Yawsangratt et al. 2012).

1.2 Nearshore and Offshore Deposits

Nearshore and offshore deposits have been described essentially in association with several specific tsunami events worldwide (Smit et al. 1992; Cita et al. 1996; Fujiwara et al. 2000; van den Bergh et al. 2003; Terrinha et al. 2003; Abrantes et al. 2005, 2008; Noda et al. 2007; Gracia et al. 2010) and were considered by Dawson and Stewart (2008) a "very much neglected research area" within tsunami sedimentary recognition. Weiss and Bahlburg (2006) considered that offshore tsunami deposition in deep marine environments well below the wave base of severe storms are theoretically much more likely to preserve tsunami deposits than shallow settings. Despite of that fact, these authors noted that there are only a few descriptions in the literature of marine, and particularly subtidal, tsunami deposits (Pratt 2001, 2002; Bussert and Aberhan 2004; Cantalamessa and Di Celma 2005; Schnyder et al. 2005).

In the offshore area, the term "deep-sea homogenite" has been used to define a massive, poorly sorted, grain-supported unit that contains large reworked shallow-marine fossils and occasional large intraclasts that have been described in association with the Bronze Age Santorini tsunami event (Cita et al. 1996). Other tsunamigenic deposits were discussed in an offshore sedimentary context and related with events such as the K/T meteoric tsunami (e.g. Smit et al. 1992; Albertão and Martins 1996), the AD 1755 tsunami (Terrinha et al. 2003; Abrantes et al. 2005, 2008; Gracia et al. 2010), the 2003 Tokashioki earthquake (Noda et al. 2007) or to try and match earthquake-triggered turbidites with tsunamigenic events from the Saguenay (Eastern Canada) and Reloncavi (Chilean margin) (St Onge et al. 2012). In fact, another peculiar note in terms of offshore tsunami deposits is that some have been specifically attributed to processes of tsunami backwash and the generation of gravity-driven flows of turbid water from nearshore to deep water (e.g. Abrantes et al. 2008; Paris et al. 2007).

1.3 Onshore Boulder Deposits

There are two main types of onshore sedimentary evidences associated with tsunami and storms: one consisting in deposits of large boulders and the other in the deposition of finer (typically sand-sized) sediments in coastal areas.

To facilitate the classification of larger particles Blair and McPherson (1999) revised the Udden-Wentworth scale (Wentworth 1922) to describe in greater detail the size of boulders and other larger particles. The grain size of fine, medium, coarse, and very coarse boulders range from 25.6 to 51.2 cm, 51.2–102.4 cm, 102.4–204.8 cm, and 204.8–409.6 cm, respectively. Larger rocks or megaclasts, include fine (4.1–8.2 m) and medium (8.2–16.4 m) blocks.

In the case of larger particles, the differentiation between tsunami and storm deposits is firstly based on the identification of boulders that have been transported inland and\or upward from or within the coastal zone, and against gravity. In some cases, these boulders appear simply overturned a few m inland from their original source area. The recognition of boulder deposits associated with both tsunami and storms has been intensely debated in the literature (e.g. Bryant et al. 1992; Young et al. 1996; Nott 1997; Bryant and Nott 2001; Noormets et al. 2002; Goff et al. 2004, 2006, 2007; Williams and Hall. 2004; Scheffers and Kelletat 2005; Hall et al. 2006; Bourrouilh-Le Jan et al. 2007; Scheffers and Scheffers 2007; Kelletat 2008; Paris et al. 2010b; Scheffers 2008; Scheffers et al. 2008; Etienne and Paris 2010; Fichaut and Suanez 2010; Goto et al. 2010a, b, 2011b; Nandasena et al. 2011). From the many examples in the literature a few deserve special notice because of their specific lithological, geological, geomorphological or oceanographic significance.

In terms of boulder deposits there are many examples worldwide attributed to deposition by storms and tsunami (compiled by e.g. Sheffers and Kelletat 2003; Scheffers 2008). They range from over 10 up to 1,000 m^3 and, depending on the bulk rock density their mass can exceed 2,000 t (Scheffers and Kelletat 2003). They have been found at various elevations from the intertidal zone to a few tens of meters above the present sea level. Shi et al. (1995) reported that hundreds of boulders were deposited as far as 200 m inland by the December 12, 1992 tsunami in Flores (Indonesia), especially in the area of Riangkroko where waves reached 26 m. The deposition of boulders in association with tsunamigenic events were discussed essentially after the 1990s (e.g. Paskoff 1991; Dawson 1994; Hindson and Andrade 1999). For instance, Hindson and Andrade (1999) noted that at several locations on the Algarve coastline the AD 1755 tsunami was associated with the deposition of both continuous and discontinuous sand sheets, some of which contain boulders. The individual boulders were frequently pitted and sculptured by bioerosion and in hollows marine endolithic mollusca were found and used to indicate the marine provenance of the boulders (Fig. 1.2 for example).

The imbrication of boulders (at certain altitudes and distances from the coastal edge), coupled with the presence of shell and debris inclusions, were used as a diagnostic criteria of tsunami deposits (Bryant and Nott 2001). Hall et al. (2006) focused exclusively in storm wave impacts on boulders sitting at the top of cliffs in Aran and

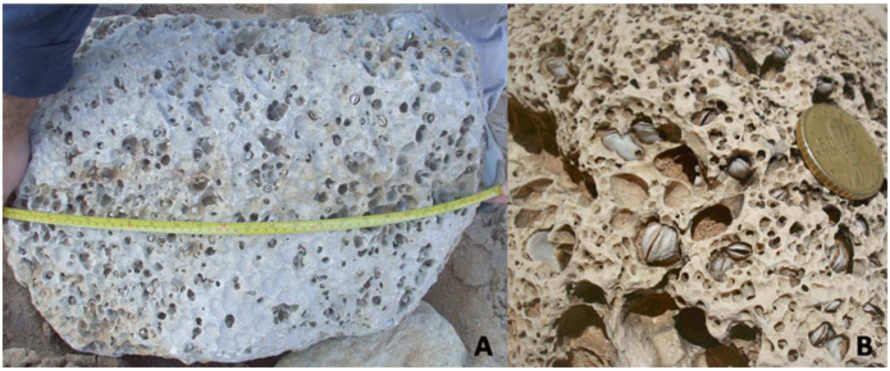


Fig. 1.2 Boulders exhibiting endolithic shells (Praia do Barranco, Portugal). *Left image* – Boulder measuring approximately 0.5 m (long axis – A) on *top* of other boulders. *Right images* – Detailed view of in situ shells, within the borings (Costa et al. 2011)

Shetland Islands (North Sea), and identified inverted boulders exclusively transported by storms. Saltation of these boulders during transport was implied by the presence of shatter marks on the upper limestone ramps on Aran (Williams and Hall 2004) and by trails of impact marks and chipped edges visible on otherwise weathered and lichen-covered surfaces. Hansom et al. (2008) provide modelled solutions for the forces of wave impact and subsequent lift at those sites. According to Hall et al. (2006), the characteristics and distribution of cliff top storm deposits allows the definition of wave properties that could generate those boulder accumulations. According to these authors, cliff top storm deposits require full exposure to storm waves and limited nearshore attenuation. Switzer and Burston (2010) stated that the imbrication, mixed lithology and sedimentary characteristics of boulder deposits at Little Beecroft Head and Greenfields Beach (Australia) provided compelling evidence for large-scale movement attributed to washover by single or multiple events. If the deposits were late-Holocene in age then hypothesise of higher Holocene sea level must be discarded and it is likely that storms and tsunami may have both played a role in the development of the high elevation boulder deposits. However, as in many other sites where boulder deposits transported against gravity have been found, it remains unclear which (i.e. tsunami or storms) was the exact mechanism of emplacement.

1.4 Onshore Cobble and Gravel Deposits

Different size-ranged clasts associated with tsunamis and storms are also described in the literature. In terms of cobble and peeble deposits (2–256 mm diameter – Krumbein and Sloss 1963) a few studies have been conducted over recent years. For example, Morton et al. (2008) analysed coastal gravel-ridge complexes deposited either by tsunamis or hurricanes on islands in the Caribbean Sea. The ridge complexes of Bonaire, Jamaica, Puerto Rico (Isla de Mona) and Guadeloupe consisted of clasts ranging in size from sand to coarse boulders derived from the adjacent coral reefs or subjacent rock platforms. The authors observed that the ridge complexes were internally organized, displayed textural sorting and a broad range of ages indicative of several historical events. Some of the cobble deposits displayed seaward-dipping beds and ridge-and-swale topography, whereas other terminated in fans or steep avalanche slopes. Together, the morphologic, sedimentologic, lithostratigraphic, and chronostratigraphic evidence indicated that ridge complexes were not entirely the result of one or a few tsunamis as previously reported (e.g. Scheffers and Kelletat 2003) but resulted from several events including not only tsunamigenic but also storm/hurricane events. Furthermore, in a nearby region (French West Indies) Caron (2011) used samples from beachrock and non-cemented coarse-grained coastal deposits and applied quantitative textural and taphonomic analysis to discriminate different depositional processes associated with storm and tsunami waves.

Research in Hawaii identified three distinct coarse-clastic depositional assemblages that could be recognized based on clast size, composition, angularity, orientation, packing, elevation and inland distance of each accumulation (Richmond et al. 2011). These deposits were characterized as:

1. Gravel fields of isolated clasts, primarily boulder-sized, and scattered pockets of sand and gravel in topographic lows.
2. Shore-parallel and cuspate ridges composed mostly of rounded basalt gravel and sand with small amounts of shell or other biogenic carbonate. The ridges ranged in height from about 1–3 m.
3. Cliff-top deposits of scattered angular and sub-angular (cobble and gravel) clasts along sea cliffs that were generally greater than 5 m elevation.

The authors concluded that the gravel fields were primarily of tsunami origin from either the 1975 Kalapana event, or a combination of tsunamis during 1868 and 1975. The ridge deposits were presently active and sediment continues to be added during high wave events. The cliff-top deposits contained evidences of deposition by both tsunami and storm processes.

Costa et al. (2011) observed spreads of cobbles and boulders (typically with an A-axis of *ca*. 0.30 m but some with smaller dimensions) that extended several hundred meters inland and well beyond the present landward limit of storm activity in a low-lying area of the Algarve (Portugal). The marine origin of the boulders was demonstrated by well-developed macro-bioerosion sculpturing and in situ skeletal remains of endolithic shallow marine bivalves. The authors associated (using radiocarbon age-estimation of *Petricola lithophaga* whole shells) the transport of these boulders with the desctructive Lisbon tsunami of AD 1755.

1.5 Sedimentological Characteristics of Onshore (Sand-Sized) (Palaeo) Tsunami Deposits

In this sub-chapter, the criteria used to recognize and differentiate tsunami deposits consisting of the finer fraction (i.e. typically sand) is presented. These features/criteria reflect the characteristics of tsunami waves, transport peculiarities, preservation potential and sedimentary sources. The first studies to use geological record to detect prehistoric tsunamis were conducted by Atwater (1987) and Dawson et al. (1988). Since then many papers have been published discussing several aspects concerning features associated with tsunami deposits. The study of modern deposits carried out during immediate post tsunami surveys provided the opportunity to refine palaeotsunami diagnostic criteria, without the uncertainty of the generating event and preservation issues due to natural and anthropogenic disturbance. Understandably, the number of studies on tsunami sedimentation increased exponentially since the 2004 Indian Ocean event, but care should be taken in adopting as of unquestionable universal applicability inferences derived from research on this particular event. Typically, tsunamis can leave sedimentary imprints on shores far from the event source, and usually less than a kilometre from the coastline. Tsunami deposits are usually thicker in topographic lows (areas of spatial deceleration of flows) and thin over topographic highs (areas of spatial acceleration of flows) (Gelfenbaum et al. 2007). In fact tsunami sediments can also be eroded during phases of backwash and have also been linked to new phases of sedimentation during backwash. The preservation of tsunami deposit is a fundamental factor in any sedimentological study focusing in recurrence intervals of such events.

Tappin (2007) discussed sedimentary features associated with tsunami, stressing that the development of realistic scenarios of risk requires reliable data on tsunami frequency, which is obviously constrained by the sporadic absence of deposit, to which we could add the eventual unability to recognize a particular tsunami-deposited layer as such in a given sedimentary sequence. In fact, Szczucinski (2012) conducted five yearly surveys after the 2004 Indian Ocean tsunami and concluded that the post-tsunami recovery of coastal zones was generally in the order of a few months to a few years. The study by Nichol and Kench (2008) in the Maldives found that within 2 years, significant reworking and bioturbation of the tsunami deposit occurred. The major macroscopic change observed was the fast removal of the thin layer of very fine sediments usually representing the top of the tsunami deposits, though such a thin layer consisting of very fine-grained material has not been reported as ubiquitous in other places worldwide and surveyed shortly after tsunami inundations. Szczucinski (2012) observed that almost all the near-surface structures of the tsunami deposits were removed with time (i.e. after at least one rainy season). Tsunami deposits thinner than 10 cm usually acquired a massive appearance after 1 or 2 years; the only remnants of the primary structures, for instance fining upward, having vanished out. This was attributed to bioturbation by growing roots and burrowing animals like crabs and rodents. A few years earlier Szczucinski et al. (2007) detected that tsunami deposits thinner than 1 cm were occasionally washed away, the depositional relief was flattened and deposits at the slopes were partially eroded; and yet, in other locations, the sedimentary bodies, including thin sand *laminae*, and sedimentary structures, such as lamination and size grading, persisted at century-long timescales (e.g. Washington State, Boca do Rio, Martinhal) following deposition. According to Yawsangratt et al. (2012) micropalaeontological evidences (i.e. carbonate foraminifera) may be subjected to significant dissolution 4.5 years after tsunami emplacement; again, this post depositional disturbance is not exclusive of tsunami deposits and rapid intrasediment dissolution or downwearing of carbonate foraminifera tests, ostracoda valves or diatom frustules is a common drawback micropaleontologists working in Holocene sediments of various facies are used to and aware of, and not exclusive of high-energy events of abrupt marine inundation. In this context, it is somewhat surprising that Lowe and de Lange (2000) suggested, based on a study from New Zealand, that a tsunami needs to raise a height of at least 5 m in order to leave any long-term, recognisable sedimentary signature (cf. Goff et al. 2010a). This statement disregards the effect of sediment concentration in tsunami waves in determining the size and preservation potential of the depositional signature, just as a number of other relevant variables, such as the presence of a compatible source, accommodation space, rapid capping of the inundation deposit, among other, which are completely independent of the tsunami amplitude.

During the last two decades, several authors (e.g. Shi et al. 1995; Goff et al. 1998; Gelfenbaum and Jaffe 2003; Dawson and Stewart 2007; Huntington et al. 2007; Shiki et al. 2008; Switzer and Jones 2008; Chagué-Goff et al. 2011) have postulated criteria to distinguish (palaeo) tsunami deposits. These are described below and summarized in Table 1.1. Dawson and Stewart (2007) discussed the processes of tsunami deposition, identifying the three main aspects that make the depositional process unique, tsunami source, propagation and inundation. The establishment of source material has been widely used (e.g. Moore and Moore 1986; Atwater and Moore 1992; Dawson et al. 1996a; Minoura et al. 1997; Bourgeois et al. 1999; Hindson and Andrade 1999; Gelfenbaum and Jaffe 2003; Switzer et al. 2005; Szczucinski et al. 2006; Babu et al. 2007; Morton et al. 2007, 2008; Narayana et al. 2007; Dahanayake and Kulasena 2008; Higman and Bourgeois 2008; Switzer and Jones 2008; Jagodzinski et al. 2009; Costa et al. 2009; Paris et al. 2009; Mahaney and Dohm 2011) because it allows one to reconstruct the origin and pathway of former tsunami waves. However, it has been commonly reported that tsunami waves transport essentially sediment that is available in the coastal fringe landward of the boundary defined by the seasonal depth of closure of the beach (and coastal) profile (e.g. Atwater and Moore 1992; Clague and Bobrowsky 1994; Dawson 1994, 2004; Moore et al. 1994; Hindson et al. 1996; Kortekaas and Dawson 2007; Paris et al. 2010b; Goff et al. 2010a; Costa et al. 2012a, b). In contrast with this, micropalaeontological evidences have indicated either relevant changes in the population of Nannoplankton, Foraminifera, Diatoms and Ostracods or that marine species from offshore/nearshore have been transported inland and deposited by tsunami (e.g. Hemphill-Haley 1996; Hindson et al. 1996; Patterson and Fowler 1996; Shennan et al. 1996; Clague et al. 1999; Dominey-Howes et al. 2000; Chagué-Goff et al. 2002; Dawson and Smith 2002; Abrantes et al. 2005; Dawson 2007; Kortekaas and

Table 1.1 Table summarizing the criteria to identify and differentiate tsunami deposits

Criteria	Features (from selected references)
Physics	Very long wave length
	Very high velocity and speed current
	Few waves but with backwash
	Swift inundation with high shear stress and erosion
Sedimentary structures (requires stratigraphic context/analysis)	Erosional/abrupt/sharp/unconformity basal contact
	Massive/chaotic unit
	Normal grading (in places repeated)
	Unit with several laminae
	Cross-stratification
	Soft-sediment deformation
	Loading structures
	Parallel lamination or cross-lamination
	Convolutions
	Ripple-marks
	Mud drapes
	Rip-up clasts
	Broken shells
Sediment source (requires multiple source analysis)	Typically reflects the material available in the coastal fringe (i.e. beaches and berms, aeolian, inner shelf landward of closure depth)
	Grain size range from mud to boulders
	Multi-modal grain-size distribution indicating multiple sources
	Increase of heavy mineral concentration in the base of the deposit
	Increase of platy minerals (i.e. micas) in the top of the deposit
	SEM microtextural imprints suggest increased presence of percussion/mechanic marks
Palaeontological features (requires stratigraphical, palaeoecological and source analysis)	Marked changes in diatoms, foraminifera, ostracods, nannoplankton, pollen (usually presenting either a chaotic assemblage or a wider range of species and, in cases, offshore/nearshore species)
Geochemical signatures (requires source analysis)	Increase inf Cl, Na, Mg, Ca, K, SiO_2, CaO, Cr, MgO, I, Fe, S
	Increases in the ratios of SiO_2/Al_2O_3, CaO/Al_2O_3
	Increase in carbonate content (shell)
	Subtle variations in source-sensitive elements: K/Rb, La/Sm and Hf/Ta
	Enrichment in Cu, Pb, Zn or, in contrast, dilution of anthropogenic elements
Geomorphological aspects (requires regional context)	Multiple breaching of dune systems or individual overwash fans
	Dune ridges and sand dune pedestals
	Landward sand sheets
	Hummocky topography
	Parabolic dunes

Dawson 2007; Mamo et al. 2009; Sawai et al. 2009; Paris et al. 2010a; Ruiz et al. 2010). Although a site-specific component might be a central feature of any tsunami deposits some generalizations are possible interrelated with sedimentary structures, sediment source, palaeontological, geochemical and geomorphological signatures (Table 1.1).

1.5.1 Sedimentary Structures

In terms of sedimentological structures, an erosive/sharp/abrupt basal contact is a common feature and is symptomatic of the energy involved in the emplacement of tsunami deposits. However, this criterion was also recognized in storm deposits (Switzer 2008). The sharp/abrupt/erosive contact of tsunamigenic layers were firstly described by Dawson et al. (1988) and Minoura and Nakaya (1991). Bondevik et al. (1997), analysing evidences lay down by the Storegga tsunami in Norway, detected that the tsunami deposit rest on an erosional unconformity which in cases has removed more than 1 m of underlying sediment. Moreover, Nanayama et al. (2000) observed deposits that resulted from the 1993 Hokkaido-nansei-oki (Japan) tsunami and identified distinctive sharp erosional bases in the tsunamigenic unit. Gelfenbaum and Jaffe (2003) analysed the erosion and sedimentation associated with the 1998 Papua New Guinea tsunami and observed that the beach face and berm showed no evidence of deposition from the tsunami. However, on the berm, exposed roots and scour at the base of some palm trees indicated erosion of approx. 20–30 cm of back-beach sand and they observed that only erosional signatures had been left by this tsunami to the landward side of the berm, up to about 50 m from the shoreline. Chandrasekar (2005) described erosion of up to 2 m over large tracts of beach associated with the return flow of the 2004 Indian Ocean tsunami. In Thailand, Szczucinski et al. (2005) and (2006), Hori et al. (2007) and Fujino et al. (2009) observed an erosive sharp basal contact between the tsunamigenic and the underlying layers. Choowong et al. (2009) also noted that during the same event, erosion and deposition occurred mainly during two periods of inflow and that the return flow was mainly erosive. Paris et al. (2009) described erosion associated with the 2004 Indian Ocean Tsunami in Banda Aceh (Indonesia) that extended up to 500 m inland. These authors quantified the overall coastal retreat from Lampuuk to Leupung as of the order of 60 m (*ca.* 550,000 m^2) and locally in excess of 150 m. The erosional impact of tsunamis is still controversial, not only the recognition of associated patterns in the sedimentary record, other than the erosive base and quantification of the amount of sediment removed but also the mechanisms and processes associated and responsible for the erosional/depositional balance during a tsunami. In fact, Bahlburg and Spiske (2011) analysing the sedimentary record of the February 2010 tsunami at Isla Mocha (Chile) observed that the tsunamigenic unit was produced essentially (i.e. >90 %) by the backflow. These authors suggest that due to the lack of sedimentary structures, many previous studies of modern tsunami sediments assumed that most of the detritus were deposited during inflow and an uncritical use of this assumption may lead to erroneous interpretations of palaeotsunami magnitudes and sedimentary processes if unknowingly applied to backflow deposits. Typically tsunami deposits present sediment size that can vary from mud to boulders and, in many cases, grain-size variation in tsunami deposits is controlled by the size of sediment available for transport, rather than by flow capacity (Bourgeois 2009) or direction.

The detection of sedimentary structures is limited by sampling methods because coring (which is frequently used), in contrast to trench excavation, is in general destructive. Sedimentary structures are also difficult to identify in tsunami deposits due to the common deposition as massive deposit (e.g. Dawson et al. 1995; Dahanayake and Kulasena 2008). However, there has been several tsunami deposits where sedimentary structures including *laminae* (e.g. Reinhart 1991; Bondevik 2003), rip-up clasts (e.g. Dawson 1994; Shi et al. 1995; Hindson and Andrade 1999; Bondevik et al. 2003; Gelfenbaum and Jaffe 2003; Goff et al. 2004; Morton et al. 2007; Paris et al. 2009), cross-stratification (e.g. Choowong et al. 2008) and soft-sediment deformation (Matsumoto et al. 2008) have been observed. Muddy laminae or organic layers can represent evidence for multiple waves of the tsunami wave train (e.g. Reinhart 1991; Bondevik 2003). Furthermore, loading structures at the base of the deposit have been reported in literature (e.g. Dawson et al. 1991; Minoura and Nakaya 1991; Costa 2006; Martin and Bourgeois 2012).

Another peculiar feature observed in many tsunamigenic deposits worldwide is the enrichment in bioclasts or shells (many of them broken) when compared with the under and overlying layers (e.g. Bryant et al. 1992; Albertão and Martins 1996; Imamura et al. 1997; Clague et al. 1999; Donato et al. 2008) and in cases platy or prolate shell fragments occur aligned suggesting a ghosty lamination (e.g. Dawson et al. 1995; Hindson et al. 1996; Hindson and Andrade 1999). For example, Clague and Bobrowsky (1994) observed that tsunami sand deposits commonly include fragments of bark, twigs, branches, logs and other plant material. Moreover, Donato et al. (2008) showed that shell features could be used as useful indicators of tsunamigenic deposit due to their vertical and lateral extent, to the allochthonous mixing of articulated bivalve species (e.g. lagoonal and nearshore) out of life position, and to the high amount of fragmented valves, with angular breaks and stress fractures. The authors suggested that the taphonomic uniqueness of tsunami deposits should be considered as a valid tool for tsunamigenic recognition in the geological record.

The sedimentological fingerprint of currents associated with tsunami events have also been observed in the form of parallel lamination, cross-lamination, convolutions and ripple-marks (e.g. Shiki et al. 2008). Moreover, Morton et al. (2007) detected palaeocurrent indicators in tsunami deposits indicating seaward return flow. In deposits laid down by the 2004 Indian Ocean tsunami, in Thailand, Choowong et al. (2008) observed capping bedforms and parallel *laminae*, cross-lamination, rip-up mud and sand clasts. The authors also observed normal grading but some reverse grading was locally recognized. According to Choowong et al. (2008) reverse grading in tsunami deposits indicates a very high grain concentration within the tsunami flow, and was possibly formed at the initial stages of inundation in shallow water. Cross-bedding was seen as restricted to return-flow sediments

(Nanayama et al. 2000). Individual deposits are generally well sorted (many are massive) and characterised by sets of fining-upwards sediment sequences that were interpreted by Shi (1995) as indicative of deposition by individual tsunami waves. Dawson and Smith (2000) characterised a tsunami sequence in Scotland by several fining upward sequences indicative of a series of tsunami waves and episodes of backwash. Furthermore, run-up and return flow deposits were also differentiated by Dawson et al. (1996a), Nanayama et al. (2000) and Goff et al. (2001). In the case of the Indian Ocean tsunami, Paris et al. (2007) observed a landward sequence thinning, fining and sorting. Normally-graded couplets or triplets of layers were used to identify the run-up of each wave. The topmost layers, interpreted as the backwash deposition, describe a seaward sequence of decreasing mean grain-size.

The time lag separating tsunami wave-trains is occasionally marked between different sub-units by the presence of mud drapes. Moreover, Fujiwara and Kamataki (2007) observed the presence of a vertical stack of many coarse-grained sub-layers separated by mud drapes interpreted as due to incremental deposition from multiple sediment flows separated by flow velocity stagnation stages and concluded that it is unlikely that the mud drapes were deposited by short-period storm waves.

More recently, different techniques have been explored to identify the sedimentary structures in tsunami deposits. An example of this is ground penetrating radar, used by Switzer et al. (2006) to survey the erosional contact between an event layer and the under and overlying units. Koster et al. (2011) also used ground penetrating radar in combination with electrical resistivity tomography measurements and sedimentology for tsunamiite recognition in Greece and Spain. According to these authors, ground penetrating radar data indicated unconformable thicknesses of tsunamigenic beddings, channel-like structures (backwash deposits) and to some extent basal erosion, as well as abrasion-scours in various places, and boulder accumulation inside the deposits (see Table 1.1).

1.5.2 Sedimentary Sources

Several authors have argued that tsunamis are frequently associated with the deposition of continuous and discontinuous sediment sheets across large areas of the coastal zone, provided that there is an adequate sediment supply (e.g., Dawson et al. 1996b; Hindson et al. 1996). The decrease of energy associated with the run-in of tsunami inundations is evidenced in stratigraphic and sedimentary architecture by the fining inland and thinning inland and ramping upwards of tsunamigenic deposits. This is probably the most common feature/criteria to recognize tsunami events in the stratigraphy of any given coastal area mainly due to the settlement of the particles through the water column, related to a decrease of the turbulence of the flow, generally forming fining-upward depositional sequences. Grain size characteristics of the tsunami deposits reflect both the origin of the displaced sediment and hydrodynamic conditions of sedimentation (Sugawara et al. 2008), with normally graded sand layers related to the decrease of the hydrodynamic energy during sedimentation (e.g. Dawson et al. 1988, 1991; Shi et al. 1995; Minoura et al. 2000). Although not a frequent situation, each fining-upward sequence can be attributed to individual tsunami waves as referred to by Ota et al. (1985) and Clague and Bobrowsky (1994). In contrast, coarsening upwards sequences have also been recognized and were ascribed to the long duration time of the tsunami (Higman and Jaffe 2005) or high-density flow, as cited above. The same authors stated that tsunamis with narrower source regions are more likely to deposit sediment that is normally graded than those with wider sources who produce more complex deposits. Although local topography plays a decisive role (e.g. Hori et al. 2007) the thickness and mean grain size of tsunami deposits generally decrease landwards (e.g. Shi et al. 1995; Hindson et al. 1996; Minoura et al. 1997; Gelfenbaum and Jaffe 2003; Goff et al. 2004; Paris et al. 2009). Landward coarsening deposits have also been exceptionally observed (e.g. Higman and Bourgeois 2008). In fact, the sediment texture of tsunami deposits is mostly related to material available for transport in the coastal zone tsunami deposits can therefore vary immensely from location to location. Differences in the tsunami records preserved tend to reflect the unique character of each tsunami, and may be attributed to source differences, coastal configuration, tide level, and sediment supply. For example, tsunami sediment source has been attributed to beaches and berms (e.g. Sato et al. 1995; Gelfenbaum and Jaffe 2003; Costa et al. 2012b), aeolian grains (e.g. Switzer et al. 2006; Costa et al. 2012b) or to the inner shelf (e.g. Switzer and Jones 2008).

The use of heavy minerals to establish provenance of tsunamigenic deposits has also been investigated by several authors (e.g. Switzer et al. 2005; Bahlburg and Weiss 2007; Szczucinski et al. 2006; Babu et al. 2007; Morton et al. 2007, 2008; Narayana et al. 2007; Higman and Bourgeois 2008; Switzer and Jones 2008; Jagodzinski et al. 2009, 2012; Nakamura et al. 2012). For example, Bahlburg and Weiss (2007) observed the presence of thin heavy-mineral concentrations at the base of individual sand layers inferred to have been laid down by different waves from the same event. Furthermore, Switzer and Jones (2008) identified a mixed heavy mineral assemblage characteristic of barrier sediments with a component of inner shelf material characterised by immature platy minerals in a tsunami deposit. Morton et al. (2008) observed that vertical textural trends showed an overall but non-systematic upward fining and upward thinning of depositional units with an upward increase in heavy mineral laminations at some locations. However, most of these studies were limited to one study area and also by local differences in source material. Jagodzinski et al. (2009) tried to compare tsunami deposits, beach sediments and pre-tsunami soils in Thailand. The difference between tsunami deposits and beach sediments and soils was reflected in differences in the respective proportions of mica and tourmaline. These differences were attributed to the mode of sediment transport and deposition with mica, due to its low density, being more abundant in the topmost part of the tsunami deposit.

Scanning Electron Microscopy (SEM) of mainly quartz grains has also been used to establish the source material of tsunami deposits (e.g. Bruzzi and Prone 2000;; Dahanayake and Kulasena 2008; Costa et al. 2009, 2012b; Mahaney and Dohm 2011). Bruzzi and Prone (2000) compared SEM microtextural signatures of quartz grains deposited by the AD 1755 tsunami (Boca do Rio, Portugal) and other quartz grains deposited by a storm in the Rhone delta (France) in November 1997. According to the authors, several features were associated with a specific event (e.g. tsunami) such as upturned plates, fractures and marks of considerable size. Dahanayake and Kulasena (2008) identified diagnostic criteria to distinguish tsunami sediments from storm-surge sediments in southern Sri Lanka noting that in tsunami sediments, reworked marine microfauna are abundant, quartz sand is not well rounded, and heavy minerals were rare, when compared with storm-surge sediments, although they do not explain the reasons underlying these differences.

Anisotropy of magnetic susceptibility has also been used to provenance studies of tsunamigenic deposits (e.g. Sugawara et al. 2008; Font et al. 2010; Wassmer et al. 2010) but the application of this technique is still in its early days and always require that the data produced is normalised in respect of the grain size distribution. One example of these studies was conducted by Font et al. (2010) in Boca do Rio (Portugal) where the magnetic data showed a dominance of paramagnetic minerals (quartz) mixed with lesser amount of ferromagnetic minerals, namely titanomagnetite and titanohematite both of a detrital origin and reworked from the underlying sedimentary units.

1.5.3 Palaeontological Signature

Macrofossils and microfossils have been used to identify and interpret sedimentary units as tsunamigenic. To date, the use of palaeontological characteristics to recognise tsunami deposits has focused on diatoms, foraminifera, ostracods, nannoplankton, pollen, molluscs and plant fragments. Typically the palaeontological signature is characterized by marked changes in the population indicating the increase in abundance of marine to brackish fossils and/or the high-energy of the event (e.g. presence of broken shells, etc.).

Diatoms have been widely used as a proxy to detect extreme marine inundations (e.g. Dawson et al. 1996a, b; Hemphill-Haley 1996; Chagué-Goff et al. 2002; Abrantes et al. 2005; Dawson 2007; Nichol and Kench 2008; Sawai et al. 2009). Generally, diatom assemblages in tsunami deposits are chaotic (mixture of freshwater and brackish–marine species), because tsunami crosses coastal and inland areas eroding, transporting and re-depositing freshwater taxa (Dawson et al. 1996b; Smith et al. 2004).

Dawson et al. (1996b) analysed the diatom assemblages contained within tsunami deposits in Scotland, related to the Second Storegga Slide and also associated with Grand Bank tsunami of 1929, and detected the presence of exceptionally large numbers of the species Paralia sulcata with most individuals exhibiting evidence of breakage. Normally, tsunami deposits are characterized by a high percentage of broken valves (e.g. more than 65 %: Dawson et al. 1996b; more than 75 %: Dawson 2007, 90 % of pinnate: Dawson and Smith 2000; more than 60 %: Sawai 2002). This is in contrast with Sawai et al. (2009) who analysed diatoms in tsunami deposits from the Indian Ocean tsunami of 2004 and concluded that the breakage of diatom valves was relatively low. Moreover, low breakage of diatoms in tsunami deposits has also been reported in the Pacific coast of Washington State and Puget Sound, USA (Hemphill-Haley 1996), and considered to have resulted from rapid sedimentation. At first sight it may appear that large percentages of broken diatoms may be indicative of former tsunamis. However, this issue is made problematic since owing to the varying robustness of lenticular and circular diatoms, tests of some species are more able than others to withstand fracturing.

In a study of tsunami sediments deposited by the Papua New Guinea tsunami of 1998, Dawson (2007) observed a contrast between the sedimentological/textural data suggesting that the beach shore-face, the berm as well as the sand spit were the source of the tsunami deposit. However, the examination of the diatom content of the sand suggested that the majority of diatoms were originated from the area immediately offshore. Benthic marine-brackish species (e.g. *Surirella sp*., *Cocconeis scutellum* and *Diploneis smithii*) were dominant within the tsunami sands along the length of transect including the sample furthest inland. Moreover, in one sample located 200 m inland, the presence of the fully marine *Triceriatum favus* attests to the allochthonous (transported) nature of the species within the deposit.

Sawai et al. (2009) analysed the diatom assemblages of Phra Thong's (Thailand) 2004 tsunami deposit and concluded that it contained surprisingly few freshwater specimens considered to have been a result of strong currents of the tsunami. Turbulent tsunami currents can cause rapid entrainment of a mixture of freshwater species, eroded soil and benthic marine species within a mass of coastal sand. However, if the currents are very strong, only benthic marine diatoms attached to heavier sandy substrate are able to settle out of the water column. When current velocity slows, the suspended freshwater specimens and soil fractions are able to settle out of the water column and deposit on top of the sandy, marine diatom-dominated portion of the deposit.

Foraminiferal content is also a common micropalaeontological proxy that has been used in tsunami sediment provenance studies (e.g. Hindson et al. 1996; Patterson and Fowler 1996; Shennan et al. 1996; Andrade et al. 1997; Dominey-Howes et al. 1998; Hindson and Andrade 1999; Clague et al. 1999; Hawkes et al. 2007; Kortekaas and Dawson 2007; Mamo et al. 2009). Bahlburg and Weiss (2007) observed in Kenya that the samples contained abundant tests of benthic foraminifera (*Quinqueloculina* and *Spiroloculina*) typically derived from shallow and protected shelf regions in water depth of less than 30 m. Also present were several species of *Amphistegina sp*. including *Amphistegina lessonii d'Orbigny* which may occur down to water depths of 80 m. The foraminifera content indicated that the tsunami very likely entrained most of the sediment in shallow depths of less than 30 m. Hawkes et al. (2007) analysed tsunami deposits in Malaysia and Thailand, and observed that the pre-tsunami assemblages were mainly composed of intertidal and inner shelf species (i.e. *Ammonia* spp., *Elphidium hispudula*) while the tsunami sediment also contained a minor but important addition of mangrove species, such as *Haplophragmoides. wilberti* and *Haplophragmoides manilaensis* and some radiolarian species. According to the authors, the mangrove and radiolarian species reflected the chaotic nature of deposition where swash up and backwash combine to create turbulence, mixing the assemblages together. Foraminiferal assemblages within tsunami sediments were also able to provide information about sediment provenance and wave characteristics. In one location (Sungai Burong), species assemblages in the tsunami sediment revealed at least two separate episodes of deposition that contained inrushed species from the inner-shelf, as well as backwash species from the mangrove environment. In the study by Dahanayake and Kulasena (2008), also on the 2004 Indian Ocean tsunami, more abundant planktonic species such as *Globigerinita glutinata*, *Hantkenina sp*. and also benthic *Quinqueloculina sp*. as well as *Amphistegina lessonii D'Orbigny* were detected. Kortekaas and Dawson (2007) analysing a tsunami deposit in Martinhal (Portugal) noted a clear abrupt change from the brackish foraminifera assemblage of the underlying layer to a fully marine assemblage consisting of *Elphidium macellum*, *Elphidium crispum*, *Quinqueloculina seminulum*, *Cibicides refulgens*, *Eponides repandus* and *Ammonia beccarii var. batavus*.

More recently, Mamo et al. (2009) summarized the many procedures, characteristics and limitations associated with foraminiferal assemblages and their use in the recognition of tsunami deposits. Characteristics such as changes in assemblage composition (Hindson et al. 1996; Hindson and Andrade 1999; Hawkes et al. 2007), for example, marine shelf species within a lagoon or brackish environment; changes in test size or in juvenile to adult ratios (Guilbault et al. 1996); a shift in population numbers (Cundy et al. 2000; Hawkes et al. 2007; Kortekaas and Dawson 2007); or a change in the taphonomic character of the tests (Hindson and Andrade 1999; Hawkes et al. 2007) can be used to recognise tsunami deposits. Given that the exact composition of an assemblage varies from location to location, it is impossible to expect to see a specific diagnostic specie(s) or assemblage in association with tsunami-deposited sediments. Some authors (Dominey-Howes et al. 1998; Nanayama and Shigeno 2006; Uchida et al. 2007) suggested that given ideal conditions a tsunami deposit might contain deeper water species that would not otherwise be expected from the shallow water.

Ostracods have also been used as tsunami indicators (e.g. Ruiz et al 2010; Mischke et al. 2010). For example, Mischke et al. (2010) analysed a tsunami deposit in Lake Hersek (Turkey) and suggested that the simultaneous occurrence of ostracods of different origin (lagoonal: *Cyprideis. torosa* and *Loxoconcha elliptica*; shallow marine: *Loxoconcha rhomboidea*, *Xestoleberis* sp., *Pontocythere* sp. and *Aurila* cf. *arborescens*; and inland waters: *Heterocypris salina* and *Eucyprinotus* cf. *rostratus*) within beds of brackish-marine mollusc shells and fragments indicates that the shell layers were deposited under high-energy environmental conditions (Ruiz et al. 2010). In the case of Lake Manyas (140 km west of Lake Hersek), ostracods of different origins were also interpreted as reflecting an event of large amplitude (seiche) (Leroy et al. 2002). In Hersek, the use of ostracods as tsunami indicators was argued on the basis of: (1) the large number of ostracod shells accumulated during the high-energy events, (2) the higher number of *taxa* which is not typical for an undisturbed lagoon setting, and (3) the mixture of ostracod valves with clear marine, lagoonal and non-marine origin.

Changes in Nannoplankton have also been discussed in association with tsunami deposits (e.g. Andrade et al. 2003; Paris et al. 2010a). Andrade et al. (2003) observed within the clay/sitl fraction changes in samples from the Tagus estuary (Portugal) subtle variations (i.e. increases) in calcareous nannoplankton that were correlated with magnetic susceptibility, foraminifera and geochemical changes. Paris et al. (2010a) observed that a characteristic of the Lhok Nga (Indonesia) tsunamigenic sediments is their nannolith coastal assemblages despite their relative impoverishment in clay content, which under normal marine hydrodynamic conditions would prevent nannoliths to settle. The abundance of nannoliths in the 2004 tsunami deposits tends to decrease landward and upward, despite variations due to successive phases of erosion/sedimentation by waves (see Table 1.1).

1.5.4 Geochemical Signature

Several studies were conducted in tsunami deposits with the aim of identifying a distinctive geochemical signature (e.g. Minoura and Nakaya 1991; Minoura et al. 1994; Andrade et al. 1998, 2003; Goff and Chagué-Goff 1999; Chagué-Goff et al. 2002; Goff et al. 2004; Srinivasalu et al. 2007; Chagué-Goff 2011). Usually geochemical features of tsunami deposits simply indicate the presence of saltwater inundation and thus do not provide information on the specific type of inundation. In fact, increases in the concentration of chemical elements of marine origin or elements indicative of coarser-sized sediments have been recognised in the past as a proxy in the study of extreme marine inundations. Minoura and Nakaya (1991) detected increases in Na, Ca, K, Mg and Cl. This was further supported by increases in Cl, Na, Ca, SO_4 and Mg observed by Minoura et al. (1994). Hindson and Andrade (1999) were able to detect increases in SiO_2 (indicating increase in sand material) CaO (indicating a larger presence of bioclasts), Cr, MgO, I and Cl (all indicating a marine water influx). On the other hand, increase in Fe and S and dilution of anthropogenic elements were observed by Goff and Chagué-Goff (1999) which suggested a sudden marine inundation. Van der Bergh et al. (2003) observed the presence of exotic sediments derived from outer coastal or continental shelf environments that were richer in heavy metals (Pb, Cu, Ni, Fe and Cr) when compared with in-situ sediments. A similar pattern was observed by Szczucinski et al. (2005) who studied sediments, deposited by the 2004 tsunami in Thailand, and noticed that they contained significantly elevated contents of salts (Na^+, K^+, Ca^{+2}, Mg^{+2}, Cl and SO_4) in water-soluble fraction, and of Cd, Cu, Zn, Pb and As.

Andrade et al. (2003) observed in the Tagus estuary (Portugal) in association with major compositional breaks increases in the ratios of SiO_2/Al_2O_3 and CaO/Al_2O_3 both accompanying increasing carbonate content in comparison with the under and overlying layers. The authors indicated that these elemental pairs have similar crystallochemical properties and changes of the ratios should primarily reflect variations in sediment source. The geochemical data coupled with palaeontological and magnetic susceptibility results allowed association with tsunami events that had affected that region.

Srinivasalu et al. (2007) reported, in India, a high content of dissolved salts in sediments (Na^+, K^+, Ca^{+2}, Mg^{+2}, Cl^-) indicating that Cu, Pb, Zn were more enriched in the tsunami deposit than the other neighbouring coastal regions. The geochemical signature is a valuable tool in the tsunami recognition but cannot be used *per si* as a diagnostic signature of tsunami deposits. In fact, the changes in geochemistry observable in a geochemical profile are mainly due to saltwater inundation, carbonate enrichment (caused by increase in shells) and changes in sediment source, all these features being also observed in storm deposits provided (Table 1.1).

1.6 Onshore Fine-Grained Deposits

Most studied tsunami imprints in coastal stratigraphy are coarser sand-sized layers within low energy finer materials accumulated in depositional basins and exhibiting a distinctive sedimentological signature within the stratigraphic column (e.g. Atwater 1987; Dawson et al. 1988; Atwater and Moore 1992; Clague et al. 1994; Shi et al. 1995; Hindson et al. 1996; Bondevik et al. 1997; Minoura et al. 1997; Goff et al. 2000; Nanayama et al. 2000, 2007; Chagué-Goff et al. 2002; Moore et al. 2007; Peters et al. 2007; Paris et al. 2009; Costa et al. 2012b). This type of depositional arrangement is indicative of extreme marine inundations because they provide a stratigraphic context and facilitate accurate spotting and dating of individual events. Fine tsunami or storm deposits are typically sand-sized with a clearly-defined clay/silt fraction (e.g. Ota et al. 1985; Srinivasalu et al. 2007). The differentiation of tsunami events in a clayish stratigraphy was successfully attempted by Andrade et al. (2003) in the Tagus estuary. The combine used of sedimentological, magnetic susceptibility, micropaleontological and chronological data allowed the identification of historical tsunamis that affected the studied areas.

The identification and differentiation of finer units deposited by tsunami or storms in coastal stratigraphy requires a multi-proxy approach that mainly focuses on the allochthonous sediment and/or palaeontological content in order to establish a marine or coastal provenance. Tsunami or storm- transported sediment is typically deposited during run-up, even though deposition also occurs during backwash and in the period of time between run-up and backwash, usually correspondent to a slack where currents fall to minimum intensity and the directional pattern is ill-defined. Furthermore, in some situations the erosional capacity of run-up or the backwash constrains the sedimentary recognition of events by removing sediments deposited by earlier tsunami waves.

1.7 Geomorphological Signature

Dawson (1994) discussed the importance of changes caused by tsunami in coastal landscapes not only by direct tsunami-driven flow orthogonal to the shoreline, but also by episodes of vigorous backwash and by water flow sub-parallel to the coastline. He suggested that the combined effect of these processes could produce coastal landforms dominated by the effects of high-magnitude erosion and deposition. Even prior to this statement, Andrade (1990, 1992) studied the Ria Formosa (Algarve) barrier chain and noticed severe damage of the barrier chain in field observations, supported by cartographic and written documents. Such damages included the drowning of the western barrier, truncation of the oriental extremity of the barrier chain and extensive overwash of two of the eastern barrier islands (Armona and Tavira) accompanied by fast progradation of the easternmost ribbon of the Algarve sandy coast in the middle eighteenth century, and attributed these changes to the AD 1755 earthquake. Andrade (1992) showed that most of the the backbarrier surface of Tavira and Armona islands revealed a unique geomorphological pattern, compatible with the exceptional overwash event and with the drainage network reorganization process that must have followed the AD 1755 tsunami.

Shi and Smith (2003) described evidence for coastal erosion and retreat that occurred along the northern coastal line of Flores Island (Indonesia) as a result of the 1992 tsunami. The authors correlated the scale of geomorphological changes with the observed tsunami run-up heights over a wide area. Similarly, Regnauld et al. (2004) and Oliveira et al. (2009) described dune erosion caused by multiple tsunami events in New Zealand and to the AD 1755 tsunami in Portugal. Meilianda et al. (2007) presented a quantitative budget of shoreline sediment fluxes before and immediately after a tsunami in Banda Aceh, Indonesia. Through the study of remote sensing images they determined a chaotic shoreline retreat just after the tsunami. In the following 6 months 60 % of the sediment loss had been compensated by shoreline accretion on the west coast of Banda Aceh city whereas further erosion (15 % of the sediment loss during the tsunami) occurred on the northwest coast. The fact that not all locations showed a beach recovery after the tsunami stresses the importance of inner shelf processes and longshore currents in redistributing the sediment eroded at the coastline. In the same location, Fagherazzi and Du (2008) revealed that most of the morphological change occurred in the shore-normal direction, with large volumes of sand removed by the tsunami at the coastline later returned to the beach in a short time interval. A series of parallel, tapered incisions widening toward the coastline are characteristics of large flooding events such as tsunamis. In fact, flood scour features are indicator of tsunami events, given their unique morphology with width and depth of the same order of magnitude and their sharp boundaries. Goff et al. (2008) recognised region-wide dune remobilisation caused by tsunami inundation in New Zealand.

In general terms, and based upon field evidence from 2004 Indian Ocean tsunami, inundation by large, region-wide events is likely to cause multiple breaching of dune systems (Higman and Jaffe 2005). In other words, multiple tsunami-scour fan assemblages can be formed during a single inundation. The assemblages could include remnant dune ridges, or pedestals, between each breach, and individual overwash fans that could coalesce to form landward sand sheets that may or may not be mobile depending upon aeolian and dune swale conditions (Goff et al. 2008). If landward sand sheets infill or overlay a wetland they can stabilise and weather in situ to form a low-profile hummocky topography. If they remain dry and are exposed to aeolian onshore processes they can form extensive, region-wide parabolic dune systems (Goff et al. 2008). Kench et al. (2009) observed that reef islands in the Maldives were geomorphically resilient to the impact of the 2004 Indian Ocean tsunami with the most immediate impacts being relatively minor with reductions in island area ranging from <1 to 9 %. Overwash deposits, in the form of sand sheets and sand lobes, as well as strandlines and individual clasts of coral rubble, were the most common accretionary forms. These deposits represented a net addition to island surfaces, although their preservation potential as tsunami signatures may be low.

1.8 Conclusions

In summary, it is important to note that the criteria to recognize tsunami deposits (see Table 1.1) are still, at the present state of knowledge, ambiguous. Although the conjunction of the identification of sedimentary structures, the establishment of source material, the micropalaeontological analysis, geochemical characteristics and the study of geomorphological imprints in coastal landscape facilitates the identification of tsunami deposits; even if erosion and preservation is constrained by several facts. In fact, the contrast/peculiarity of the tsunami layers, especially when compared with under and overlying layers, provides in many cases the conclusive evidence for the recognition of such deposits. Data is commonly obtained through the use of sedimentological techniques - some have been widely used (e.g. textural and geomorphological) while others have been scarcely applied (e.g. microtextural analysis and heavy mineral assemblages); both have been used in modern tsunami sediments as well as in palaeotsunamis. Based in the references summarized in this sub-chapter and in Table 1.1 it is possible, through the use of diverse sedimentological proxies, to obtain information about the presence or absence of tsunami indicators, to establish their likely source or to collect valuable information about tsunami run-up, backwash or wave penetration inland. Recent events in Sumatra and Japan have been used to further develop the application and definition of sedimentary criteria to be used in the identification of tsunami deposits. However, in the study of palaeotsunamis a group of other questions (e.g. understanding mechanisms of inundation and deposition for each specific location, preservation of sedimentary structures and palaeontological evidences) still needs to be addressed in future studies in order to contribute to the development of more detailed and rigorous criteria that can further contribute to the accuracy of hazard maps.

References

Abrantes F, Lebreiro S, Rodrigues T, Gil I, Bartels-Jónsdóttir H, Oliveira P, Kissel C, Grimalt JO (2005) Shallow-marine sediment cores record climate variability and earthquake activity off Lisbon (Portugal) for the last 2000 years. Quat Sci Rev 24(23–24):2477–2494

Abrantes F, Alt-Epping U, Lebreiro S, Voelker A, Schneider R (2008) Sedimentological record of tsunamis on shallow-shelf areas: the case of the 1969 AD and 1755 AD tsunamis on the Portuguese Shelf off Lisbon. Mar Geol 249(3–4):283–293

Albertão GA, Martins PP (1996) A possible tsunami deposit at the Cretaceous-Tertiary boundary in Pernambuco, Northeastern Brazil. Sediment Geol 104(1–4):189–201

Andrade C (1990) O Ambiente de barreira da Ria Formosa (Algarve, Portugal). In: Departamento de Geologia, PhD, Lisboa, pp 645

Andrade C (1992) Tsunami generated forms in the Algarve Barrier Islands (South Portugal). Sci Tsunami Haz 10(1):21–34

Andrade C, Andrade A, Kortekaas S, Dawson A (1997) Sedimentological traces of tsunamigenic overwash of the Martinhal lowland (Western Algarve – Portugal). In: Eurocoast-Portugal A (ed) Seminário sobre a Zona Costeira do Algarve. Faro, Portugal, pp 11–18

Andrade C, Munhá JM, Paulino J (1998) Geochemical signature of extreme marine flooding in the Boca do Rio lowland (Algarve, Portugal). Com V Congresso Nacional de Geologia, Lisboa, IGN, SGP 84(1):51–54

Andrade C, Freitas MC, Miranda JM, Baptista MA, Cachão M, Silva P, Munhá J (2003) Recognizing possible tsunami sediments in the ultradissipative environment of the Tagus estuary (Portugal). In: Davis RA, Sallenger A, Howd P (eds) Coastal sediments'03 – The 5th international symposium on coastal engineering and science of coastal sediment processes, Clearwater Beach, Florida (CD-ROM) (ISBN: 978-981-238-422-5 CD). p 14

Andrade C, Freitas MC, Moreno J, Craveiro SC (2004) Stratigraphical evidence of late Holocene barrier breaching and extreme storms in lagoonal sediments of Ria Formosa, Algarve, Portugal. Mar Geol 210(1–4):339–362

Atwater BF (1987) Evidence for great Holocene earthquakes along the outer coast of Washington State. Science 236:942–944

Atwater BF, Moore AL (1992) A tsunami about 1000 years ago in Puget Sound, Washington. Science 258:1614–1617

Babu N, Suresh Babu DS, Mohan Das PN (2007) Impact of tsunami on texture and mineralogy of a major placer deposit in southwest coast of India. Environ Geol 52:71–80

Bahlburg H, Spiske M (2011) Sedimentology of tsunami inflow and backflow deposits: key differences revealed in a modern example. Sedimentology. doi:10.1111/j.1365-3091.2011.01295.x

Bahlburg H, Weiss R (2007) Sedimentology of the December 26, 2004, Sumatra Tsunami deposits in eastern India (Tamil Nadu) and Kenya. Int J Earth Sci 96(6):1195–1209

Blair TC, McPherson JG (1999) Grain-size and textural classification of coarse sedimentary particles. J Sed Res 69:6–19

Bondevik S (2003) Storegga tsunami sand in peat below the Tapes beach ridge at Harøy, Western Norway, and its possible relation to an early Stone Age settlement. Boreas 32(476–483)

Bondevik S, Svendsen JI, Mangerud J (1997) Tsunami sedimentary facies deposited by the Storegga tsunami in shallow marine basins and coastal lakes, Western Norway. Sedimentology 44:1115–1131

Bondevik S, Mangerud J, Dawson S, Dawson A, Lohne O (2003) Record-breaking height for 8000-year-old tsunami in the North Atlantic. EOS Trans Am Geophys Union 84:289–293

Bourgeois J (2009) Geologic records and effects of tsunamis, vol 15, The sea: tsunamis. Harvard University Press, Cambridge, MA, pp 53–91

Bourgeois J, Hansen TA, Wiberg PL, Kauffman EG (1988) A tsunami deposit at the Cretaceous-Tertiary boundary in Texas. Science 241:567–570

Bourgeois J, Petroff C, Yeh H, Titov V, Synolakis C, Benson B, Kuroiwa J, Lander J, Norabuena E (1999) Geologic setting, field survey and modeling of the Chimbote, Northern Peru, tsunami of 21 February 1996. Pure Appl Geophys 154:513–540

Bourrouilh-Le Jan FG, Beck C, Gorsline DS (2007) Catastrophic events (hurricanes, tsunami and others) and their sedimentary records: introductory notes and new concepts for shallow water deposits. Sediment Geol 199(1–2):1–11

Bruzzi C, Prone A (2000) A method of sedimentological identification of storm and tsunami deposits: exoscopic analysis, preliminary results. Quaternaire 11(3–4):167–177

Bryant E, Nott J (2001) Geological indicators of large tsunami in Australia. Nat Hazards 24:231–249

Bryant E, Young R, Price D (1992) Evidence of tsunami sedimentation on the southeastern coast of Australia. J Geol 100:753–765

Bussert R, Aberhan M (2004) Storms and tsunamis: evidence of event sedimentation in the Late Jurassic Tendaguru Beds of southeastern Tanzania. J Afr Earth Sci 39(3–5):549–555

Cantalamessa G, Di Celma C (2005) Sedimentary features of tsunami backwash deposits in a shallow marine Miocene setting, Mejillones Peninsula, Northern Chile. Sediment Geol 178(3–4):259–273

Caron V (2011) Contrasted textural and taphonomic properties of high-energy wave deposits cemented in beachrocks (St. Bartholomew Island, French West Indies). Sediment Geol 237(3–4):189–208

Chagué-Goff C, Dawson S, Goff JR, Zachariasen J, Berryman KR, Garnett DL, Waldron HM, Mildenhall DC (2002) A tsunami (ca. 6300 years BP) and other Holocene environmental changes, Northern Hawke's Bay, New Zealand. Sediment Geol 150(1–2):89–102

Chagué-Goff C, Schneider J-L, Goff JR, Dominey-Howes D, Strotz L (2011) Expanding the proxy toolkit to help identify past events – lessons from the 2004 Indian Ocean Tsunami and the 2009 South Pacific Tsunami. Earth Sci Rev 107(1–2):107–122

Chandrasekar N (2005) Tsunami of 26th December 2004: observation on inundation, sedimentation and geomorphology of Kanyakumari coast, South India. In: Satake PA (ed) 22nd international tsunami symposium, Xania, Greece, pp. 49–56

Choowong M, Murakoshi N, Hisada K-I, Charoentitirat T, Charusiri P, Phantuwongraj S, Wongkok P, Choowong A, Subsayjun R, Chutakositkanon V, Jankaew K, Kanjanapayont P (2008) Flow conditions of the 2004 Indian Ocean tsunami in Thailand, inferred from capping bedforms and sedimentary structures. Terra Nova 20:141–149

Choowong M, Phantuwongraj S, Charoentitirat T, Chutakositkanon V, Yumuang S, Charusiri P (2009) Beach recovery after 2004 Indian Ocean tsunami from Phang-nga, Thailand. Geomorphology 104(3–4):134–142

Cita MB, Camerlenghi A, Rimoldi B (1996) Deep-sea tsunami deposits in the eastern Mediterranean: new evidence and depositional models. Sediment Geol 104(1–4):155–173

Clague JJ, Bobrowsky PT (1994) Tsunami deposits beneath tidal marshes on Vancouver Island, British Columbia. Geol Soc Am Bull 106(10):1293–1303

Clague JJ, Bobrowsky PT, Hamilton TS (1994) A sand sheet deposited by the 1964 Alaska tsunami at Port Alberni, British Columbia. Estuar Coast Shelf Sci 38(4):413–421

Clague JJ, Hutchinson I, Mathewes RW, Petterson RT (1999) Evidence for late Holocene tsunamis at Catala Lake, British Columbia. J Coast Res 15(1):45–60

Clague JJ, Bobrowsky PT, Hutchinson I (2000) A review of geological records of large tsunamis at Vancouver Island, British Columbia, and implications for hazard. Quat Sci Rev 19: 849–863

Costa PJM (2006) Geological recognition of abrupt marine invasions in two coastal areas of Portugal. In: Geography and earth sciences, MPhil. Brunel University, London, p 139

Costa PJM, Andrade C, Freitas MC, Oliveira MA, Jouanneau JM (2009) Preliminary results of exoscopic analysis of quartz grains deposited by a palaeotsunami in salgados lowland (Algarve, Portugal). J Coast Res SI 56:39–43

Costa PJM, Andrade C, Freitas MC, Oliveira MA, Silva CM, Omira R, Taborda R, Baptista MA, Dawson AG (2011) Boulder deposition during major tsunami events. Earth Surf Proc Land 36:2054–2068

Costa PJM, Andrade C, Dawson AG, Mahaney WC, Paris R, Freitas MC, Taborda R (2012a) Microtextural characteristics of quartz grains transported and deposited by tsunamis and storms. Sediment Geol 275–276:55–69
Costa PJM, Andrade C, Freitas MC, Oliveira MA, Lopes V, Dawson AG, Moreno J, Fatela F, Jouanneau J-M (2012b) A tsunami record in the sedimentary archive of the central Algarve coast, Portugal: characterizing sediment, reconstructing sources and inundation paths. Holocene 22(8):899–914
Cundy AB, Kortekaas S, Dewez T, Stewart IS, Collins PEF, Croudace IW, Maroukian H, Papanastassiou D, Gaki-Papanastassiou P, Pavlopoulos K, Dawson A (2000) Coastal wetlands as recorders of earthquake subsidence in the Aegean: a case study of the 1894 Gulf of Atalanti earthquakes, central Greece. Mar Geol 170(1–2):3–26
Dahanayake K, Kulasena N (2008) Recognition of diagnostic criteria for recent- and paleo-tsunami sediments from Sri Lanka. Mar Geol 254:180–186
Dawson AG (1994) Geomorphological effects of tsunami run-up and backwash. Geomorphology 10(1–4):83–94
Dawson AG (2004) Tsunami hazards in Europe. Environ Int 30:577–585
Dawson S (2007) Diatom biostratigraphy of tsunami deposits: examples from the 1998 Papua New Guinea tsunami. Sediment Geol. doi:10.1016/j.sedgeo.2007.01.011, 8
Dawson S, Smith DE (2000) The sedimentology of Middle Holocene tsunami facies in northern Sutherland, Scotland, UK. Mar Geol 170(1–2):69–79
Dawson AG, Stewart I (2007) Tsunami deposits in the geological record. Sediment Geol 200(3–4):166–183
Dawson AG, Stewart I (2008) Offshore tractive current deposition: the forgotten tsunami sedimentation process. In: Shiki T, Tsuji Y, Yamazaki T, Minoura K (eds) Tsunamiites – features and implications. Elsevier, Amsterdam, pp 153–161
Dawson AG, Long D, Smith DE (1988) The Storegga slides: evidence from Eastern Scotland for a possible tsunami. Mar Geol 82(3–4):271–276
Dawson AG, Foster IDL, Shi S, Smith D, Long D (1991) The identification of past tsunamis in coastal sediment sequences. Sci Tsunami Haz 9:73–82
Dawson AG, Hindson R, Andrade C, Freitas C, Parish R, Baterman R (1995) Tsunami sedimentation associated with the Lisbon earthquake of 1 November AD 1755: Boca do Rio, Algarve, Portugal. The Holocene 5(2):209–215
Dawson AG, Shi S, Dawson S, Shuto N, Takahashi T (1996a) Coastal sedimentation associated with the June 2nd and 3rd tsunami in Rajegwesi, Java. Quat Sci Rev 15:901–912
Dawson S, Smith DE, Ruffman A, Shi S (1996b) The diatom biostratigraphy of tsunami sediments: examples from recent and middle Holocene events. Phys Chem Earth 21(1–2):87–92
Dominey-Howes D, Dawson A, Smith D (1998) Late Holocene coastal tectonics at Falasarna, Western Crete: a sedimentary study. In: Stewart I, Vita-Finzi C (eds) Coastal tectonics. Geological Society of London, London, pp 343–352
Dominey-Howes D, Cundy A, Croudace I (2000) High energy marine flood deposits on Astypalaea Island, Greece: possible evidence for the AD 1956 southern Aegean tsunami. Mar Geol 163:303–315
Donato SV, Reinhardt EG, Boyce JI, Rothaus R, Vosmer T (2008) Identifying tsunami deposits using bivalve shell taphonomy. Geology 36(3):199–202
Einsele G, Chough SK, Shiki T (1996) Depositional events and their records – an introduction. Sediment Geol 104(1–4):1–9
Etienne S, Paris R (2010) Boulder accumulations related to storms on the south coast of the Reykjanes Peninsula (Iceland). Geomorphology 114(1–2):55–70
Fagherazzi S, Du X (2008) Tsunamigenic incisions produced by the December 2004 earthquake along the coasts of Thailand, Indonesia and Sri Lanka. Geomorphology 99(1–4):120–129
Fichaut B, Suanez S (2010) Quarrying, transport and deposition of cliff-top storm deposits during extreme events: Banneg Island, Brittany. Mar Geol 283(1–4):36–55
Finkl CW, Pelinovsky E, Cathcart RB (2012) A review of potential tsunami impacts to the Suez Canal. J Coast Res 28(4):745–759

Font E, Nascimento C, Omira R, Baptista MA, Silva PF (2010) Identification of tsunami-induced deposits using numerical modeling and rock magnetism techniques: a study case of the 1755 Lisbon tsunami in Algarve, Portugal. Phys Earth Planet In 182(3–4):187–198

Fujino S, Naruse H, Matsumoto D, Jarupongsakul T, Sphawajruksakul A, Sakakura N (2009) Stratigraphic evidence for pre-2004 tsunamis in Southwestern Thailand. Mar Geol 262(1–4):25–28

Fujiwara O, Kamataki T (2007) Identification of tsunami deposits considering the tsunami waveform: an example of subaqueous tsunami deposits in Holocene shallow bay on southern Boso Peninsula, Central Japan. Sediment Geol 200(3–4):295–313

Fujiwara O, Masuda F, Sakai T, Irizuki T, Fuse K (2000) Tsunami deposits in Holocene bay mud in southern Kanto region, Pacific coast of central Japan. Sed Geol 135(1–4):219–230

Gelfenbaum G, Jaffe B (2003) Erosion and sedimentation from the 17 July, 1998, Papua New Guinea tsunami. Pure Appl Geophys 160:1969–1999

Gelfenbaum G, Vatvani D, Jaffe B, Dekker F (2007) Tsunami inundation and sediment transport in the vicinity of coastal mangrove forest. In: Sixth international symposium on coastal engineering and science of coastal sediment processes. ASCE, New York. ISBN: 978-0-7844-0926-8, http://dx.doi.org/10.1061/40926(239)86

Goff JR, Chagué-Goff C (1999) A late Holocene record of environmental changes from coastal wetlands: Abel Tasman National Park, New Zealand. Quat Int 56:39–51

Goff JR, Crozier M, Sutherland V, Cochran U, Shane P (1998) Possible tsunami deposits from the 1855 earthquake, North Island, New Zealand. Geol Soc Lond Spec Publ 146:353–376

Goff JR, Rouse HL, Jones SL, Hayward BW, Cochran U, McLea W, Dickinson WW, Morley MS (2000) Evidence for an earthquake and tsunami about 3100–3400 yr ago, and other catastrophic saltwater inundations recorded in a coastal lagoon, New Zealand. Mar Geol 170(1–2):231–249

Goff J, Chagué-Goff C, Nichol S (2001) Palaeotsunami deposits: a New Zealand perspective. Sediment Geol 143(1–2):1–6

Goff J, McFadgen BG, Chagué-Goff C (2004) Sedimentary differences between the 2002 Easter storm and the 15th-century Okoropunga tsunami, southeastern North Island, New Zealand. Mar Geol 204(1–2):235–250

Goff J, Dudley WC, deMaintenon MJ, Cain G, Coney JP (2006) The largest local tsunami in 20th century Hawaii. Mar Geol 226:65–79

Goff J, McFadgen B, Wells A, Hicks M (2008) Seismic signals in coastal dune systems. Earth Sci Rev 89(1–2):73–77

Goff J, Pearce S, Nichol SL, Chagué-Goff C, Horrocks M, Strotz L (2010a) Multi-proxy records of regionally-sourced tsunamis, New Zealand. Geomorphology 118(3–4):369–382

Goff J, Weiss R, Courtney C, Dominey-Howes D (2010b) Testing the hypothesis for tsunami boulder deposition from suspension. Mar Geol 277(1–4):73–77

Goff J, Chagué-Goff C, Nichol S, Jaffe B, Dominey-Howes D (2012) Progress in palaeotsunami research. Sediment Geol 243–244:70–88

Goto K, Chavanich SA, Imamura F, Kunthasap P, Matsui T, Minoura K, Sugawara D, Yanagisawa H (2007) Distribution, origin and transport process of boulders deposited by the 2004 Indian Ocean tsunami at Pakarang Cape, Thailand. Sediment Geol 202:821–837

Goto K, Kawana T, Imamura F (2010a) Historical and geological evidence of boulders deposited by tsunamis, southern Ryukyu Islands, Japan. Earth Sci Rev 102(1–2):77–99

Goto K, Okada K, Imamura F (2010b) Numerical analysis of boulder transport by the 2004 Indian Ocean tsunami at Pakarang Cape, Thailand. Mar Geol 268(1–4):97–105

Goto K, Chague-Goff C, Fujino S, Goff J, Jaffe B, Nishimura Y, Richmond B, Sugawara D, Szczucinski W, Tappin DR, Witter RC, Yulianto E (2011a) New insights of tsunami hazard from the 2011 Tohoku-oki event. Mar Geol 290:46–50

Goto K, Miyagi K, Kawana T, Takahashi J, Imamura F (2011b) Emplacement and movement of boulders by known storm waves -field evidence from the Okinawa Islands, Japan. Mar Geol 283:66–78

Gracia E, Vizcaino A, Escutia C, Asioli A, Rodés A, Pallàs R, Garcia-Orellana J, Lebreiro S, Goldfinger C (2010) Holocene earthquake record offshore Portugal (SW Iberia): testing turbidite paleoseismology in a slow-convergence margin. Quat Sci Rev 29:1156–1172

Guilbault J-P, Clague JJ, Lapointe M (1996) Foraminiferal evidence for the amount of coseismic subsidence during a late holocene earthquake on Vancouver Island, West Coast of Canada. Quat Sci Rev 15(8–9):913–937
Hall AM, Hansom JD, Williams DM, Jarvis J (2006) Distribution, geomorphology and lithofacies of cliff-top storm deposits: examples from the high-energy coasts of Scotland and Ireland. Mar Geol 232:131–155
Hansom JD, Barltrop NDP, Hall AM (2008) Modelling the processes of cliff-top erosion and deposition under extreme storm waves. Mar Geol 253(1–2):36–50
Hawkes AD, Bird M, Cowie S, Grundy-Warr C, Horton BP, Shau Hwai AT, Law L, Macgregor C, Nott J, Ong JE, Rigg J, Robinson R, Tan-Mullins M, Sa TT, Yasin Z, Aik LW (2007) Sediments deposited by the 2004 Indian Ocean tsunami along the Malaysia-Thailand Peninsula. Mar Geol 242(1–3):169–190
Hemphill-Haley E (1996) Diatoms as an aid in identifying late-Holocene tsunami deposits. Holocene 6(4):439–448
Higman B, Bourgeois J (2008) Deposits of the 1992 Nicaragua tsunami. In: Shiki T, Tsuji Y, Yamazaki T, Minoura K (eds) Tsunamiites – features and implications. Elsevier, Amsterdam, pp 81–103
Higman B, Jaffe B (2005) A comparison of grading in deposits from five tsunamis: does tsunami wave duration affect grading patterns? EOS Trans Am Geophys Union 86(52):T11A–T0362A, Abstract
Hindson RA, Andrade C (1999) Sedimentation and hydrodynamic processes associated with the tsunami generated by the 1755 Lisbon earthquake. Quat Int 56(1):27–38
Hindson RA, Andrade C, Dawson AG (1996) Sedimentary processes associated with the tsunami generated by the 1755 Lisbon earthquake on the Algarve coast, Portugal. Phys Chem Earth 21(1–2):57–63
Hori K, Kuzumoto R, Hirouchi D, Umitsu M, Janjirawuttikul N, Patanakanog B (2007) Horizontal and vertical variation of 2004 Indian tsunami deposits: an example of two transects along the western coast of Thailand. Mar Geol 239:163–172
Huntington K, Bourgeois J, Gelfenbaum G, Lynett P, Jaffe B, Yeh H, Weiss R (2007) Sandy signs of a tsunami's onshore depth and speed. Eos (Trans Am Geophys Un) 88(52):577–578
Imamura F, Lee HJ, Takahashi T, Shuto N (1997) The highest run-up of the 1993 Hokkaido Nansei-Oki earthquake tsunami. In: IAMAS – IAPSO Joint Assemblies, Melbourne, Australia
Jagodzinski R, Sternal B, Szczucinski W, Lorenc S (2009) Heavy minerals in 2004 tsunami deposits on Kho Khao Island Thailand. Pol J Environ Stud 18:103–110
Jagodzinski R, Sternal B, Szczucinski W, Chague-Goff C, Sugawara D (2012) Heavy minerals in the 2011 Tohoku-oki tsunami deposits – insights into sediment sources and hydrodynamics. Sediment Geol 282:57–64
Kelletat D (2008) Comments to Dawson, A.G. and Stewart, I. (2007), tsunami deposits in the geological record. – sedimentary geology 200, 166–183. Sediment Geol 211(3–4):87–91
Kench PS, Brander RW, Parnell KE, O'Callaghan JM (2009) Seasonal variations in wave characteristics around a coral reef island, South Maalhosmadulu atoll, Maldives. Mar Geol 262(1–4):116–129
Kortekaas S, Dawson AG (2007) Distinguishing tsunami and storm deposits: an example from Martinhal, SW Portugal. Sediment Geol 200(3–4):208–221
Koster B, Reicherter K, Vött A, Grützner C (2011) The evidence of tsunamigenic deposits in the gulf of Corinth (Greece) with geophysical methods for spatial distribution. In: 2nd INQUA-IGCP-567 International Workshop on Active Tectonics, Earthquake Geology, Archaeology and Engineering, Corinth, pp 107–110
Krumbein WC, Sloss LL (1963) Stratigraphy and sedimentation. Freeman, San Francisco
Le Roux JP, Vargas G (2005) Hydraulic behavior of tsunami backflows: insights from their modern and ancient deposits. Environ Geol 49:65–75
Leroy S, KazancI N, Ileri Ö, Kibar M, Emre O, McGee E, Griffiths HI (2002) Abrupt environmental changes within a late Holocene lacustrine sequence south of the Marmara Sea (Lake Manyas, N-W Turkey): possible links with seismic events. Mar Geol 190(1–2):531–552
Long D, Smith DE, Dawson AG (1989) A Holocene tsunami deposit in Eastern Scotland. J Quat Sci 4:61–66

Lowe DJ, de Lange WP (2000) Volcano-meteorological tsunamis, the c. AD 200 Taupo eruption (New Zealand) and the possibility of a global tsunami. The Holocene 10(401–407)
Mahaney WC, Dohm JM (2011) The 2011 Japanese 9.0 magnitude earthquake: test of a kinetic energy wave model using coastal configuration and offshore gradient of Earth and beyond. Sediment Geol 239(1–2):80–86
Mamo B, Strotz L, Dominey-Howes D (2009) Tsunami sediments and their foraminiferal assemblages. Earth Sci Rev 96(4):263–278
Martin ME, Bourgeois J (2012) Vented sediments and tsunami deposits in the Puget Lowland, Washington – differentiating sedimentary processes. Sedimentology 59(2):419–444
Matsumoto D, Naruse H, Fujino S, Surphawajruksakul A, Jarupongsakul T, Sakakura N, Murayama M (2008) Truncated flame structures within a deposit of the Indian Ocean Tsunami: evidence of synsedimentary deformation. Sedimentology 55:1559–1570
Meilianda E, Dohmen-Janssen CM, Maathuis BHP, Hulscher SJMH, Mulder JPM (2007) Beach morphology at Banda Aceh, Indonesia in response to the tsunami on 26 December 2004. In: Coastal sediments, New Orleans, pp 14
Minoura K, Nakaya S (1991) Traces of tsunami preserved in inter-tidal lacustrine and marsh deposits: some examples from Northeast Japan. J Geol 99:265–287
Minoura K, Nakaya S, Uchida M (1994) Tsunami deposits in a lacustrine sequence of the Sanriku coast, Northeast Japan. Sediment Geol 89(1–2):25–31
Minoura K, Imamura F, Takahashi T, Shuto N (1997) Sequence of sedimentation processes caused by the 1992 Flores tsunami: evidences from Babi Island. Geology 25:523–526
Minoura K, Imamura F, Kuran U, Nakamura T, Papadopoulos GA, Takahashi T, Yalciner AC (2000) Discovery of Minoan tsunami deposits. Geology 28(1):59–62
Mischke S, Schudack U, Bertrand S, Leroy SAG (2010) Ostracods from a Marmara Sea lagoon (Turkey) as tsunami indicators. Quat Int. doi:10.1016/j.quaint.2010.11.013
Moore GW, Moore JG (1984) Deposits from a giant wave on the island of Lanai, Hawaii. Science 226:1312–1315
Moore JG, Moore GW (1986) Large-scale bedforms in boulder gravel produced by giant waves in Hawaii. Geol Soc Am Spec Pap 229:101–110
Moore JG, Normark WR, Holcomb RT (1994) Giant Hawaiian landslides. Annu Rev Earth Planet Sci 22:119–144
Moore AL, McAdoo BG, Ruffman A (2007) Landward fining from multiple sources in a sand sheet deposited by the 1929 Grand Banks tsunami, Newfoundland. Sediment Geol 200(3–4):336–346
Morton RA, Gelfenbaum G, Jaffe BE (2007) Physical criteria for distinguishing sandy tsunami and storm deposits using modern examples. Sediment Geol 200:184–207
Morton RA, Richmond BM, Jaffe BE, Gelfenbaum G (2008) Coarse-clast ridge complexes of the Caribbean: a preliminary basis for distinguishing tsunami and storm-wave origins. J Sediment Res 78:624–637
Morton RA, Gelfenbaum G, Buckley ML, Richmond BM (2011) Geological effects and implications of the 2010 tsunami along the central coast of Chile. Sediment Geol 242:34–51
Nakamura Y, Nishimura Y, Putra PS (2012) Local variation of inundation, sedimentary characteristics, and mineral assemblages of the 2011 Tohoku-oki tsunami on the Misawa coast, Aomori, Japan. Sediment Geol 282:216–227
Nanayama F, Shigeno K (2006) Inflow and outflow facies from the 1993 tsunami in southwest Hokkaido. Sediment Geol 187(3–4):139–158
Nanayama F, Shigeno K, Satake K, Shimokawa K, Koitabashi S, Miyasaka S, Ishii M (2000) Sedimentary differences between the 1993 Hokkaido-nansei-oki tsunami and the 1959 Miyakojima typhoon at Taisei, Southwestern Hokkaido, Northern Japan. Sediment Geol 135(1–4):255–264
Nanayama F, Furukawa R, Shigeno K, Makino A, Soeda Y, Igarashi Y (2007) Nine unusually large tsunami deposits from the past 4000 years at Kiritappu marsh along the southern Kuril Trench. Sediment Geol 200(3–4):275–294

Nandasena NAK, Paris R, Tanaka N (2011) Numerical assessment of boulder transport by the 2004 Indian ocean tsunami in Lhok Nga, West Banda Aceh (Sumatra, Indonesia). Comput Geosci 37(9):1391–1399

Narayana AC, Tatavarti R, Shinu N, Subeer A (2007) Tsunami of December 26, 2004 on the southwest coast of India: post-tsunami geomorphic and sediment characteristics. Mar Geol 242:155–168

Nichol SL, Kench PS (2008) Sedimentology and preservation potential of carbonate sand sheets deposited by the December 2004 Indian Ocean tsunami: South Baa Atoll, Maldives. Sedimentology 55:1173–1187

Noda A, Katayama H, Sagayama T, Suga K, Uchida Y, Satake K, Abe K, Okamura Y (2007) Evaluation of tsunami impacts on shallow marine sediments: an example from the tsunami caused by the 2003 Tokachi-oki earthquake, Northern Japan. Sediment Geol 200(3–4): 314–327

Noormets R, Felton EA, Crook KAW (2002) Sedimentology of rocky shorelines: 2: Shoreline megaclasts on the north shore of Oahu, Hawaii–origins and history. Sediment Geol 150(1–2):31–45

Nott J (1997) Extremely high-energy wave deposits inside the Great Barrier Reef, Australia: determining the cause–tsunami or tropical cyclone. Mar Geol 141(1–4):193–207

Oliveira MA, Andrade C, Freitas MC, Costa PJ (2009) Modeling volume transfer between beach-Foredune and the Backshore by the 1755 Lisbon tsunami at Boca Do Rio Lowland, Algarve (Portugal). J Coast Res, 1547–1551

Ota Y, Pirazzoli PA, Kawana T, Moriwaki H (1985) Late Holocene coastal morphology and sea-level records on three small islands, the South Ryukyus, Japan. Geogr Rev Jpn 58B(2):185–194

Paris R, Lavigne F, Wassmer P, Sartohadi J (2007) Coastal sedimentation associated with the December 26, 2004 tsunami in Lhok Nga, West Banda Aceh (Sumatra, Indonesia). Mar Geol 238(1–4):93–106

Paris R, Wassmer P, Sartohadi J, Lavigne F, Barthomeuf B, Desgages E, Grancher DB, Vautier F, Brunstein D, Gomez C (2009) Tsunamis as geomorphic crisis: lessons from the December 26, 2004 tsunami in Lhok Nga, West Banda Aceh (Sumatra, Indonesia). Geomorphology 104:59–72

Paris R, Cachao M, Fournier J, Voldoire O (2010a) Nannoliths abundance and distribution in tsunami deposits: example from the December 26, 2004 tsunami Lhok Nga (Northwest Sumatra, Indonesia). Géomorphologie: relief, processus, environnement 1

Paris R, Fournier J, Poizot E, Etienne S, Morin J, Lavigne F, Wassmer P (2010b) Boulder and fine sediment transport and deposition by the 2004 tsunami in Lhok Nga (Western Banda Aceh, Sumatra, Indonesia): a coupled offshore-onshore model. Mar Geol 268(1–4):43–54

Paris R, Giachetti T, Chevalier J, Guillou H, Frank N (2011a) Tsunami deposits in Santiago Island (Cape Verde archipelago) as possible evidence of a massive flank failure of Fogos volcano. Sediment Geol 239(3–4):129–145

Paris R, Naylor LA, Stephenson WJ (2011b) Boulders as a signature of storms on rock coasts. Mar Geol 283:1–11

Paskoff R (1991) Likely occurance of a mega-tsunami in the Middle Pleistocene, near Coquimbo, Chile. Rev Geol Chile 18(1):87–91

Patterson RT, Fowler AD (1996) Evidence of self organization in planktic foraminiferal evolution: implications for interconnectedness of paleoecosystems. Geology 24(3):215–218

Peters R, Jaffe B, Gelfenbaum G (2007) Distribution and sedimentary characteristics of tsunami deposits along the Cascadia margin of western North America. Sediment Geol. doi:10.1016/j.sedgeo.2007.01.015, 42

Pratt BR (2001) Oceanography, bathymetry and syndepositional tectonics of a Precambrian intracratonic basin: integrating sediments, storms, earthquakes and tsunamis in the Belt Supergroup (Helena Formation, ca. 1.45 Ga), Western North America. Sediment Geol 141–142:371–394

Pratt BR (2002) Tepees in peritidal carbonates: origin via earthquake-induced deformation, with example from the Middle Cambrian of western Canada. Sediment Geol 153(3–4):57–64

Regnauld H, Nichol SL, Fontugne M (2004) Maoris, middens and dune front accretion rate on the NE coast of New Zealand: resilience of a sedimentary system after a tsunami. Géomorphol: Relief Process Environ 1:45–54
Regnauld H, Planchon O, Goff J (2008) Relative roles of structure, climate, and of a tsunami event on coastal evolution of the Falkland Archipelago. Géomorphol: Relief Process Environ 1:15–26
Reinhart MA (1991) Sedimentological analysis of postulated tsunami-generated deposits from Cascadia great-subduction earthquakes along southern coastal Washington. MSc, University of Washington, Seattle, WA, p 77
Richmond BM, Watt S, Buckley M, Jaffe BE, Gelfenbaum G, Morton RA (2011) Recent storm and tsunami coarse-clast deposit characteristics, southeast Hawai'i. Mar Geol 283:79–89
Ruiz F, Abad M, Cáceres LM, Rodríguez Vidal J, Carretero MI, Pozo M, González-Regalado ML (2010) Ostracods as tsunami tracers in Holocene sequences. Quatern Res 73(1):130–135
Sato H, Shimamoto T, Tsutsumi A, Kawamoto E (1995) Onshore tsunami deposits caused by the 1993 Southwest Hokkaido and 1983 Japan Sea earthquakes. Pure Appl Geophys 144:693–717
Sawai Y (2002) Evidence for 17th century tsunamis generated on the Kuril-Kamchatka subduction zone, Lake Tokotan, Hokkaido, Japan. J Asian Earth Sci 20:903–911
Sawai Y, Jankaew K, Martin ME, Prendergast A, Choowong M, Charoentitirat T (2009) Diatom assemblages in tsunami deposits associated with the 2004 Indian Ocean tsunami at Phra Thong Island, Thailand. Mar Micropaleontol 73:70–79
Scheffers A (2008) Tsunami boulder deposits. In: Shiki T, Tsuji Y, Yamazaki T, Minoura K (eds) Tsunamiites – features and implications. Elsevier, Amsterdam, pp 299–317
Scheffers A, Kelletat D (2003) Sedimentologic and geomorphologic tsunami imprints worldwide– a review. Earth-Sci Rev 63(1–2):83–92
Scheffers A, Kelletat D (2005) Tsunami relics on the coastal landscape west of Lisbon, Portugal. Sci Tsunami Haz 23(1):3–16
Scheffers A, Scheffers S (2007) Tsunami deposits on the coastline of west Crete (Greece). Earth Planet Sci Lett. doi:10.1016/j.epsl.2007.05.041, 32
Scheffers A, Kelletat D, Vött A, May SM, Scheffers S (2008) Late Holocene tsunami traces on the western and southern coastlines of the Peloponnesus (Greece). Earth Planet Sci Lett 269(1–2): 271–279
Schnyder J, Baudin F, Deconinck J-F (2005) A possible tsunami deposit around the Jurassic-Cretaceous boundary in the Boulonnais area (northern France). Sediment Geol 177(3–4): 209–227
Shennan I, Rutherford MM, Innes JB, Walker KJ (1996) Late glacial sea level and ocean margin environmental changes interpreted from biostratigraphic and lithostratigraphic studies of isolation basins in Northwest Scotland. Geol Soc Lond 111:229–244
Shi S (1995) Observational and theoretical aspects of tsunami sedimentation. PhD, Coventry University, Coventry, UK, p 333
Shi S, Smith DE (2003) Coastal tsunami geomorphological impacts and sedimentation processes: case studies of modern and prehistorical events. In: International Conference on Estuaries and Coasts, Hangzhou
Shi S, Dawson AG, Smith DE (1995) Coastal sedimentation associated with the December 12th, 1992 tsunami in Flores, Indonesia. Pure Appl Geophys 144:525–536
Shiki T, Tachibana T, Fujiwara O, Goto K, Nanayama F, Yamazaki T (2008) Characteristic features of tsunamiites. In: Shiki T, Tsuji Y, Yamazaki T, Minoura K (eds) Tsunamiites – features and implications. Elsevier, Amsterdam, pp 319–340
Smit J, Montanari A, Swinburne NHM, Alvarez W, Hilderbrand R, Margolis SV, Claeys P, Lowrie W, Asaro F (1992) Tektite-bearing deep-water clastic unit at the Cretaceous-Tertiary boundary in North-Eastern Mexico. Geology 20:99–103
Smith DE, Shi S, Cullingford RA, Dawson AG, Dawson S, Firth CR, Foster IDL, Fretwell PT, Haggart BA, Holloway LK, Long D (2004) The Holocene Storegga Slide tsunami in the United Kingdom. Quat Sci Rev 23:2291–2321

Srinivasalu S, Thangadurai N, Switzer AD, Ram Mohan V, Ayyamperumal T (2007) Erosion and sedimentation in Kalpakkam (N Tamil Nadu, India) from the 26th December 2004 tsunami. Mar Geol 240:65–75

St-Onge G, Chapron E, Salas M, Viel M, Mulsow S, Debret M, Foucher A, Mulder T, Desmet M, Costa P, Ghaleb B, Jaouen A, Locat J (2012) Comparison of earthquake-triggered turbidites from the Saguenay (Eastern Canada) and Reloncavi (Chilean margin) Fjords: implications for paleoseismicity and sedimentology. Sediment Geol 243–244:89–107

Sugawara D, Minoura K, Imamura F (2008) Tsunamiites features and implications – chapter 3 – tsunamis and tsunami sedimentology. Elsevier, Armsterdam, pp 9–49

Switzer AD (2008) 20 years of palaeotsunami studies on coastal sandsheets: a review. In: 2nd International Tsunami Fiedl Symposium, Italy

Switzer AD, Burston JM (2010) Competing mechanisms for boulder deposition on the Southeast Australian coast. Geomorphology 114:42–54

Switzer AD, Jones BG (2008) Large-scale washover sedimentation in a freshwater lagoon from the Southeast Australian coast: tsunami or exceptionally large storm? The Holocene 18:787–803

Switzer AD, Pucillo K, Haredy RA, Jones BG, Bryant EA (2005) Sea-level, storms or tsunami; enigmatic sand sheet deposits in sheltered coastal embayment from southeastern New South Wales Australia. J Coast Res 21:655–663

Switzer AD, Bristow CS, Jones BG (2006) Investigation of large-scale washover of a small barrier system on the Southeast Australian coast using ground penetrating radar. Sediment Geol 183:145–156

Synolakis CE, Imamura F, Tsuji Y, Matsutomi H, Tinti S, Cook B, Chandra YP, Usman M (1995) Damage conditions of East Java tsunami of 1994 analyzed. Eos Trans AGU 76(76):257–261–262

Szczucinski W (2012) The post-depositional changes of the onshore 2004 tsunami deposits on the Andaman Sea coast of Thailand. Nat Hazards 60:115–133

Szczucinski W, Niedzielski P, Rachlewicz G, Sobczynski T, Zioła A, Kowalski A, Lorenc S, Siepak J (2005) Contamination of tsunami sediments in a coastal zone inundated by the 26 December 2004 tsunami in Thailand. Environ Geol 49:321–331

Szczucinski W, Chaimanee N, Niedzielski P, Rachlewicz G, Saisuttichai D, Tepsuwan T, Lorenc S, Siepak J (2006) Environmental and geological impacts of the 26 December 2004 isunami in Coastal Zone of Thailand – overview of short and long-term effects. Pol J Environ Stud 15(5):793–810

Szczucinski W, Niedzielski P, Kozak L, Frankowski M, Zioła A, Lorenc S (2007) Effects of rainy season on mobilization of contaminants from tsunami deposits left in a coastal zone of Thailand by the 26 December 2004 tsunami. Environ Geol. doi:10.1007/s00254-007-0639-4, 12

Tappin DR (2007) Sedimentary features of tsunami deposits – their origin, recognition and discrimination: an introduction. Sediment Geol 200(3–4):151–154

Terrinha P, Pinheiro LM, Henriet JP, Matias L, Ivanov MK, Monteiro JH, Akhmetzhanov A, Volkonskaya A, Cunha T, Shaskin P, Rovere M (2003) Tsunamigenic-seismogenic structures, neotectonics, sedimentary processes and slope instability on the Southwest Portuguese Margin. Mar Geol 195(1–4):55–73

Uchida J, Abe K, Hasegawa S, Fujiwara O (2007) Studies on the source of run-up tsunami deposits based on foraminiferal tests and their hydrodynamic verification. Quatern Res 46(6):533–540

van den Bergh GD, Boer W, de Haas H, van Weering TCE, van Wijhe R (2003) Shallow marine tsunami deposits in Teluk Banten (NW Java, Indonesia), generated by the 1883 Krakatau eruption. Mar Geol 197(1–4):13–34

Wassmer P, Schneider J-L, Fonfrage A-V, Lavigne F, Paris R, Gomez C (2010) Use of anisotropy of magnetic susceptibility (AMS) in the study of tsunami deposits: Application to the 2004 deposits on the eastern coast of Banda Aceh, North Sumatra, Indonesia. Mar Geol 275(1–4):255–272

Weiss R, Bahlburg H (2006) A note on the preservation of offshore tsunami deposits. J Sediment Res 76:1267–1273

Wentworth CK (1922) A scale of grade and class terms for clastic sediments. J Geol 30:377–392
Williams DM, Hall AM (2004) Cliff-top megaclast deposits of Ireland, a record of extreme waves in the North Atlantic–storms or tsunamis? Mar Geol 206(1–4):101–117
Yawsangratt S, Szczucinski W, Chaimanee N, Chatprasert S, Majewski W, Lorenc S (2012) Evidence of probable paleotsunami deposits on Kho Khao Island, Phang Nga Province, Thailand. Nat Hazards 63:151–163
Young R, Bryant E, Price D (1996) Catastrophic wave (tsunami?) transport of boulders in southern New South Wales, Australia. Geomorphology 40(2):191–207

Chapter 2
Advances in Beach Management in Latin America: Overview from Certification Schemes

Camilo-Mateo Botero, Allan T. Williams, and Juan Alfredo Cabrera

Abstract Beach management in Latin America is described, focused on analysis of beach certification schemes currently implemented in this continent. Initially, core concepts about beach management are discussed, in order to establish a common framework. Moreover, several initiatives to measure quality are analysed, looking for identifying those tools able to give a certification. Afterwards, 9 beach certification schemes applied in 12 Latin American countries are described (Argentina, Brasil Colombia, Costa Rica, Cuba, Dominican Republic, Ecuador, Mexico, Peru, Puerto Rico and Uruguay), according to their origin, administrative framework, structure and implementation. Finally, advance in beach management in Latin America is discussed, pointing out main conceptual, methodological and practical challenges to be achieved for scientific and decision makers of the continent.

2.1 Beach Management Principles and Techniques

Physical systems information flows from the bottom upwards, i.e. from beach management – BM where a theoretical ideal puts into practice Integrated Coastal Management – ICM aims and objectives, i.e. it is a sub set of the wider scoped ICM

C.-M. Botero (✉)
Grupo Joaquín Aaron Manjarres, Universidad Sergio Arboleda,
Calle 18 No. 14-18, Santa Marta, Colombia
e-mail: playascol@yahoo.com; camilo.botero@usa.edu.co

A.T. Williams
Built Environment, Swansea Metropolitan University, Swansea SA1 6ED, Wales, UK
e-mail: allan.williams@virgin.net

J.A. Cabrera
Grupo COSTATENAS, Universidad de Matanzas,
Autopista a Varadero, Km 3 ½, Matanzas, Cuba
e-mail: alfredo.cabrera@umcc.cu

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8,
DOI 10.1007/978-3-319-06305-8_2,

umbrella and a very effective management tool (Williams and Micallef 2009). However, it usually lacks the proactive element and is frequently geared and based on award/rating tools/schemes, especially in tourist areas, such as, the Mediterranean, Caribbean. For example, the National Healthy Beaches Campaign; the Blue Wave initiative, (USA), Good Beach Guide, (UK), Blue Flag (Denmark).

Many BM schemes have a very narrow sectoral focus (safety, engineering, etc.), which is invariably the beach itself and not the bathing area hinterland (Radic et al. 2006) and many such schemes fail to recognize the catholic variety of beach types that exist, with the probable exception of traditional resort/remote beach areas (Cagilaba and Rennie 2005). Consequently, current progressive thinking has concerned itself with a holistic integrated approach to BM, recognizing beach classifications (dissipative-reflective; heavy-light usage; resort, urban, village, rural, remote; urban (e.g. Paris plage) – water (sea, river, lake), etc. acknowledging that different beach types need different management schemes. Quality indicators have been introduced into the literature e.g. those of Cagilaba and Rennie (2005) and Espejel et al. (2007), arguing that the following are essential elements for any BM plan. These are: clean shallow water and sand beach, no dangerous animals/odours, a pleasant water temperature, security and finally sound infrastructure/services (toilets, access, lifeguards, shade and a small shopping precinct).

So what is BM? Beach management should aim to achieve an optimal physical usage/resource development that is in line with the physical whilst satisfying basic beach user social needs i.e. integration of the physical beach fundament onto which is superimposed the cultural environment (Williams et al. 2002). Two common definitions are:

> that process of managing a beach, whether by monitoring, simple intervention, recycling, recharge, the construction or maintenance of beach control structures or by some combination of these techniques, in a way that reflects an acceptable compromise in the light of available finance, between the various coastal defence, nature conservation, public amenity and industrial objectives. (Simm 1996:147)

> Beach management seeks to maintain or improve a beach as a recreational resource and a means of coast protection, while providing facilities that meet the needs and aspirations of those who use the beach. (Bird 1996:212)

In essence, it should provide a sustainable, optimum usage of natural, e.g. profiling and sediment characteristics, wave, tide and current data, flora and fauna, socio-economic e.g. beach user preferences and priorities, economics of beach resources, willingness to pay, etc. and cultural resources (history, etc.) whilst providing in certain localities an increase in recreation, in others conservation. To the above, must be added the utilisation of expert personnel and appropriate techniques with which to obtain baseline of information regarding the beach area especially in understanding the nature of coastal processes; follow a sound management philosophy – working with nature rather than against it; long, rather than short range planning; more public participation and properly educated management people; consider ecological rather than economic considerations; co-ordinate public agencies; provision of legislation and more importantly implementation of the enforcement mechanisms. BM should input into any planning/regulation/decision

Table 2.1 Benefits accruing from beach management strategy adopted by Breton (1998)

Establishment of native flora following decreased human disturbance
Better representation of natural plant distribution typical of this part of the coast
Reconstruction of dune systems as a by-product of increased vegetation cover
Contribution to a community based management approach through the involvement of the public in litter collection
Academic value from data recorded on the ability of 'sand-loving' plants to adapt to low water, high temperature and substrate mobility and poor soil conditions
Opportunities to utilize this area for educational purposes for both academic and general public
Data generation for use in future beach management and rehabilitation programmes
Development of a beach management strategy utilizing low human and financial resources

making processes, especially with respect to engineering structures needed. It reflects by its very nature multi-disciplinary management, with sound aims, objectives and methodologies.

An example of sound management philosophy, which enhanced a degraded dune system in Catalonia, Spain, considered both socio-economic as well as environmental interests related to beach use (Breton and Esteban 1995; Breton 1998). The local council's main objective was integration of public recreational beach usage with conservation. Access to prevent trample damage was controlled by pathways, litter was hand-picked and signage used for educational purposes (Table 2.1).

2.2 State of Art of Beach Management in Latin America

The beaches, in Latin America, like in the entire world, have a strategic importance. Nevertheless, only in the recent years it has begun to understand the necessity of application of a coastal integrated management in this ambit. Nowadays is recognized that problems are complex, and cannot be approached correctly from the traditional reductionist's approach and methods (analysis-synthesis). In consequence, they demand adoption of conceptual and methodological focuses framed in the paradigms of the complexity and systems.

In fact, the integrated management of beaches, like a derivation of the integrated management of coastal areas, shifts from the decade of the 1990, mainly after establishes of this focus during the Conference of the United Nations on Environment and Development – CNUMAD of Rio de Janeiro, in 1992. In this way, the integrated management of beaches conforms as a new professional field, in full formation, starting from adoption of diverse conceptual-methodological approach.

As an example, in a quick search on scientific articles that had the term beach, in the database of Latin American scientific publications SciELO, was obtained that of 105 publications in the lapse of time between 2002 and 2008, 24 % was referred to ecology of beaches, 14 % to aspects of the coast dynamics, 2 % to environmental and alone contamination 2 % was framed in administration of beaches (Botero 2013).

Much more recently, in January of the 2012, during the I Ibero-American Congress of Integrated Management of Coastal Areas, in Cadiz (Spain), 40.7 % of the total speech presented in the section “Management for the sustainability and adaptation to the climatic change” were about beaches. This result allows appreciating a gradual growth in this type of studies and their applications, although it was also evidenced that main attention follows setting in the descriptive analyses, in particular geomorphological and ecological issues. Moreover, majority of studies are still in stage of diagnostic, more than in the implementation and evaluation of a management truly *integrated*.

Great relevance for advance of integrated management of beaches in Latin America has been organization and operation of a virtual forum of discussion in the theme: the Ibero-American Beach Management and Certification Network – PROPLAYAS. This initiative starts from year 2006, integrating a wide number of professionals and work groups related with the topic, coming from Argentina, Brazil, Colombia, Cuba, Costa Rica, Mexico, Peru, Portugal, Venezuela, Uruguay, among other countries, which continues growing. This network has become the most active space in coordination of efforts and knowledge in management of beaches of the continent.

In this moment, in Latin America are develop diverse management models and certification schemes of beaches, mostly influenced by exogenous models. Many of these initiatives are framed in international cooperation projects, type north-south, although it is appreciated a slow growth of the focuses and initiatives of local scale and south-south approach. There is also evidenced the concepts and instruments dominating the field of the integrated management of beaches has strong influences of diverse scientific disciplines. Among those scientific areas, it is highlighted contributions from biology, geography and oceanography, very valuable, but always slanted by vision to a unique discipline and, for the same question, with a potential of partial application. The first forms and actions of administration of beaches in the region were directly related with solve erosion problems and physical degradation. As a result, practical and solutions were mainly from coastal Engineering, generating a very controversial and polemic topic, with defenders and detractors.

The beaches, against the predominant opinion during the decade of the 1960s, are much more than a fringe of sand, and therefore they should be interpreted, just as they are in fact, complex systems, in those modifications of anyone on their behalves supposes to general unbalance. The application of *hard solutions*, such as jetties, barriers breakwater, and other of this type, implicate negative factors for a natural dynamics and a convenient transport to natural system of the sand. Not mean that it is fair an absolutely negative valuation of this actuation type, because there are some examples of applications with positive results, but it is unquestionable that majority actions of this type has been negative consequences. This appreciation can corroborate in numerous critical analysis of the matter carried out in Colombia, Argentina, Brazil and other countries.

During years 1970s and 1980s starts a more integrated vision of the performances engineers, when being emphasized by several authors that they should be articulated with the scientific studies that allow a deeper knowledge of coastal processes and causes and magnitude of erosion. Additionally, regulations and laws applicable to coastal environment and marine areas start to be included. Based on these principles, *soft solutions* have been applied, highlighting regenerations of beaches by means of the artificial injection of sand (beach feeding). According to Tristá et al. (2012) this technique presents ecological and aesthetic advantages with regard to other options. These authors point out that it does not introduce radical changes to natural conditions of beaches and environmental and visual impacts are low when normal conditions are recovered. Also, morphological profile and dunes of beaches suffer less with these *soft solutions*.

The fact is that in last three decades application of artificial injection of sand, it has become the alternative for majority of administration of beaches at world level, and in the Latin America case, applications show notably successful in Cuba, Dominican Republic, Jamaica, Mexico and Brazil, among other countries. However, they are many cases in which these coastal performances respond to owners' interests or punctual tourist companies, but not to true programmes of integrated administration of beaches, scientifically based and with a holistic vision.

On the other hand, this beach feeding is an expensive technique that can bear problems for increasing of sedimentary flow in the marine ecosystem and that, usually, requires to repeat the operation to restore reservations (Medina et al. 2001). Several detractors of marine artificial feeding with sand coming from submarine banks argue that this method is ecologically degrading, as long as it affects negatively to the marine ecosystems in those extraction take place. Also, in some cases, these projects cause a progressive burial of meadows of seagrasses and corals for the haulage of the poured sand (Rodríguez-Perea et al. 2002).

All above-mentioned explains importance to distinguish between integrated management of beaches and coastal Engineering. Several solutions to face erosion and to regenerate or maintain physical conditions of the beach, constitute an important component, but they are not equivalent, in its entirety, to the Programmes of integrated coastal management.

Another distinction is the equivalence between integrated management of beaches and conservation of the coastal ecosystems. The latter constitutes an action approach that has reinforced in the last years, derived of the concern for degradation of biotic component and affectation of the marine-coastal environmental quality. It is so, in an extreme position, defends the option of not acting, and one pleads for *to conserve and to stop in the time* to the ecosystems, what constitutes, in our opinion, a deviation of true integrated management of beaches. Indeed, it is also important involve these efforts into the Programmes of integrated management of beaches, and there are very interesting applications of these focuses, even in the context of programmes of handling conservation areas. Despite of many conservationist initiatives verified in Costa Rica, Ecuador and other countries, it is clear that integrated administration of beaches should advance toward a complex vision. In consequence, a group of management actions, even with dissimilar and concrete instruments, should guide to preserve the basic mechanisms of self-regulation and dynamic-functional balance of coastal systems, with all its components.

This way, and amid these burning debates, integrated management of beaches in Latin America has gone evolving toward the design and application of models characterized by more holistic base conceptions. Also, an articulate group of diverse performances and "less aggressive" techniques allow a progressive change toward a management of beaches and dunes, every time "more integrated".

In a recent work, under the title The integrated management of beaches and dunes: experiences in Latin America and Europe (Rodríguez-Perea et al. 2012), more than 60 authors contribute with their vision to these scientific and technical debates. In this publication, several authors defend new conceptual-methodological approach and operative tools for a more integrated management of beach-dunes systems. As a result, topics such as restoration actions and protection of dunes, implications of geomorphological factors in implementation initiatives, use of studies of environmental perception, and performance based on observation systems, are expanding for Latin America.

In many cases, there are experiences of advanced applications, based in technical very particular, as non-retreat of remains of seagrasses, to counteract the erosive processes. Another example are installation of captor's tramps to retain the sands and to facilitate formation of embryonic dunes. More proposals are substitution of mechanized cleaning by manual cleaning of the beach, and elimination of facilities and equipment's beach that affects the sandy fringe. All of these initiatives are proliferating and begin a new tendency of integrated beach management.

In this context, Williams and Micallef (2009) have been emphatic in defining the central aspects that should be guaranteed in the integrated management of beaches. These authors consider important to be attentive of environmental impact that cause the structures and processes, to maintain the "amenity" of the beach, cleaning and security, observation to the quality of the water and appropriate benefit of the tourist services, among other. Meanwhile, Cabrera et al. (2009) propose a wide group of addresses of the integrated management, supporter in the experience of more than 20 years in the Varadero's beach (Cuba). These authors propose several actions, such as evaluation of erosion causes and tendencies of physical degradation of the beach, maintenance and improvement of the fringe of sand, restoration and protection of dunes, handling of residual liquids and of solid waste, environmental planning, prevention of natural risks and anthropogenic, inspection and systematic audits, education and actors' training, and a rigorous integral scientific monitoring of the beach.

It is necessary to highlight that beaches management has been frequently associate, sometimes in an absolute way, with administration of tourist certification. Nevertheless, many authors have insisted in that management actions in tourist beaches imply the articulation of environmental, sociocultural and institutional aspects. In any case, it has played a fundamental role application of the Norms ISO 9001 (Systems of Quality) and ISO 14001 (Systems of environmental management) to the beaches, that suppose an advance in the continuous improvement of facilities and in coastal environment.

In the case of the Colombian Caribbean, it has been proposed a management approach toward an environmental and tourist quality of beaches. Botero (2013) has design a model formed by a group of parameters organized in five categories: environmental, facilities, safety/security, information and education, and management; scope of this proposal should be translated in concrete actions to advance toward improve sanitary, ecosystem and recreational quality on beaches. Another initiatives in Latin America can be seen forward, when beach certification schemes will be described. As a conclusion, very positive fact is that in all Latin America, gradually, have gone entering in implementation diverse models of beaches management, based on national legal frameworks and certification schemes, at different scales. That is a signal of maturity of this field in the continent.

2.3 Beach Quality Assessment Tools on the World

There exist a rather large number of award schemes, all ostensibly providing quality indicators of quality. The following are but a few examples of these awards and the reader is directed to Williams and Micallef (2009) for a detailed examination of the various award schemes in existence.

2.3.1 Costa Rica

The oldest scheme is based upon the Marine and Terrestrial Act (Ley Maritimo Terrestre), Coast Rica, Chaverri (1989), identified beaches suitable for tourism development. Its basis was 113 *positive* and *negative* factors, divided into six groups. These were: Water (10 positive and 16 negative), Beach (9 and 7); Sand (6 and 10); Rock (11 and 11); General beach environment (11 and 12); Surrounding area (5 and 5). Each group had a score ranging from 0 (*bad*) to 4 (*good*). Summation of the resultant scores gave a rating division that was very subjective and of doubtful validity – a very subjective methodology indeed. Parameters, such as, vegetation quality, coastal area wealth, were not explained and no weighting was carried out.

2.3.2 The Blue Flag

The most well known of European award schemes, it is run by the non-profit Foundation for Environmental Education, based in Denmark (FEE 2013). A beach is eligible if it is nationally/internationally designated as a bathing area, with at least one sampling point for water quality analysis and also has the necessary facilities and standards needed to comply with the criteria needed. The aim is coastal sustainability at all levels and it is awarded for only one season. Two criteria need emphasizing. The FEE place great store on what are termed Imperative (I) and Guideline (G) standards for Total coliform, Faecal coliform and Faecal streptococci counts. For an Imperative standard, a beach must comply in order to receive an award; Guidelines standards are not mandatory.

2.3.3 UK. Quality Coast Awards

The Quality Coast award introduced in 2007 by Environmental Campaigns (ENCAMS; – www.encams.org), recognizes different well managed, quality but diverse parts of a coastline available to different users and recognizes four categories: (a) Fun in the Sea: essentially water sport activities; (b) Away from it all: need to get away from the city, appreciate wildlife, scenic beauty etc. Essentially they want rural beaches; (c) Bucket and Spade: day trippers/holiday makers, need car parks, build sand castles, often with grandparents. Beach entertainment for children e.g. cricket, football, kite flying are important. Safety and water cleanliness are important; and (d) Relaxed recreation: a nice day out at the seaside. Food outlets, car parks, toilets etc. are needed. Often people have dogs and like long walks. Certain elements need to be in place before (e.g. carrying capacity assessments, no dog fouling) and during inspection (e.g. signage is of a good standard). It strongly recommends that a beach management plan is in place.

2.3.4 The Good Beach Guide

This book is published annually by the Marine Conservation Society (MSC), a Non Governmental Organisation (NGO) based at Ross on the Wales/England border, UK. It is divided into two main sections. The first part relates to water quality criteria: (a) Recommended (minimum sewage contamination); (b) MSC Guideline pass (mscG; sewage affected in heavy rain/certain tides); (c) European Union, Guideline pass (G). Fail EU mandatory test 5 % of time; (d) European Union Mandatory Pass (P) Pass above but large pollution risk; and (e) European mandatory Fail (F) Contaminated. The second section provides a range of information such a, beach descriptions, safety, litter, facilities, wildlife, seaside activities, accessibility and parking, public and tourist information.

2.3.5 Ukraine

The Ministry of Resorts and Tourism of Crimea have categorised award beaches as: Yellow shell. In 2012, as an initiative, beaches were classified into five categories based on the services provided. The ratings were: one to two Shells – toilets, showers, changing rooms. For example, Kapsel beach on Cape Meganom; three to five Shells – as above, plus web-cameras, Wi-Fi, sport water activities and games, sport yards, cafes and bars. For example, Sudak, Novyi Svet; five Shells – given for the good ecological conditions. For example, Massandra Beach, Yalta.

In addition, Blue Flag beaches occur. Currently there are seven beach areas that fly this flag. This evaluation technique does not cover any remote beach evaluations: Yalta – Massandra beach, beach of the hotel 'Yalta-Intourist'; Evpatoria – the

'Northern Medical Centre' beach together with the beach associated with the 'Interregional Centre for Labour, Health and Social Rehabilitation of Disabled People'; Sudak – 'Kapsel' beach on Cape Meganom and the 'Novyi Svet' Municipal beach; Illichivsk (Odessa province) – the city Municipal beach.

2.3.6 Portugal: Gold Award Quality Beaches

'The environmental organisation Quercus (National Association of Conservation of Nature), awards a 'Gold quality award' to bathing areas which over the past 5 years, have obtained a 'good' grade in water quality standards. The rating scheme is limited in scope, as water quality is the only criterion and based on annual data provided by the Portuguese Institute of Water.

Myriad rating schemes exist that assess beach quality and give an award. But why do people go to a particular beach? Cutter et al. (1979) found that they went to the nearest beach. Work done by a variety of authors – see Williams and Micallef (2009), have confirmed this finding. However, this might be the first priority but for preference, >4,000 beach users who were questioned in Europe and the USA expressed a desire for five beach elements, in no particular order: safety, good water quality, no litter, facilities and scenery. In Latin America, Botero et al. (2013) findings added a *good warm beach atmosphere* to this list. But how many schemes have interviewed beach users? Most existing rating schemes have not taken cognisance of beach user views. In addition, how many schemes have weighted their questions? Inevitably all questions have been given the same weight, but some are more important than others. Do beach users know about rating schemes, do they care? Nelson et al. (2000) and Nelson and Botterill (2002) found that many people associated a flag with *danger*.

Do we need awards? If a beach is underperforming and an award will improve it, the answer must be *Yes*. However, the essence of good BM is that the manager knows his/her bathing area and can take steps to improve it irrespective of awards. McKenna et al. (2011) paper gives a salutary warning as to the significance of ratings/awards. Nevertheless, a wide perspective of schemes related with beach certification, classification and evaluation is shows at Table 2.2.

2.4 Beach Scheme Certifications in Latin America

Within this context of beach management and a *jungle* of awards, a detailed review of Latin American situation was done by Botero (2013). Main objective of this research was to propose a common framework for existing awards in this continent, but only those defined as a Beach Certification Schemes – BCS in the sense of "*set of administrative and operative elements, which through a systematic assessment, endorse continuous improvement of holistic conditions of the beach and recognize to the public an effective management*" (Botero 2013:87).

Table 2.2 Certification, classification and evaluation tools on the world

Name	Kind of tool	Application area	References
Blue Flag (www.blueflag.org)	Certification (ecolabel)	Europe, Africa, Oceania and America	Cagilaba and Rennie (2005), Vaz (2008), Williams and Micallef (2009), ECOSAMBITO (2007), Botero et al. (2012), FEE (2013),
Bandera Azul Ecológica (www.aya.go.cr)	Certification (ecolabel)	Costa Rica	ECOSAMBITO (2007), Botero et al. (2012), MINAET (2011)
Bathing Area Registration and Evaluation (www.bare-beach.com)	Evaluation and classification	Mediterranean	Cagilaba and Rennie (2005), Vaz (2008), Williams and Micallef (2009);
Beach Quality Index	Evaluation	Spain	Williams and Micallef (2009); Ariza (2007)
Beach Safety in Australia (www.surflifesaving.com.au)	Evaluation	Australia	Cagilaba and Rennie (2005); Williams and Micallef (2009)
Blue Wave and Clean Beach (www.cleanbeaches.org)	Certification (ecolabel)	United States	Cagilaba and Rennie (2005), Williams and Micallef (2009), ECOSAMBITO (2007),
Certificación Turística de Playas (www.turismo.gob.ec)	Certification (ecolabel)	Equator	INEN (2012)
Clean Beaches (www.kabq.org.au)	Certification (award)	Australia	Cagilaba and Rennie (2005)
Environmental Campaigns (www.encams.org)	Certification (ecolabel)	United Kingdom	Cagilaba and Rennie (2005), Williams and Micallef (2009)
Escala Delfín	Evaluation	Rumania	Williams and Micallef (2009)
Good Beach Guide (www.goodbeachguide.co.uk)	Classification	United Kingdom	Cagilaba and Rennie (2005), Williams and Micallef (2009)
Green Sea Initiative (www.dwrcymru.com)	Certification (ecolabel)	United Kingdom	Williams and Micallef (2009)
Guida Blu (www.legambiente.it)	Evaluation	Italy	Williams and Micallef (2009)
Índice CEDEX (www.cedex.es)	Evaluation	Spain	Williams and Micallef (2009)
National Healthy Beaches Campaign (www.healthybeaches.org)	Evaluation	United States	Cagilaba and Rennie (2005), Williams and Micallef (2009)
Norma Q (www.aenor.es)	Certification (Sistema gestión)	Spain	Williams and Micallef (2009)

Playa Ambiental	Certification (ecolabel)	Cuba	Botero et al. (2012), JCPV (2008)
Playa Natural (www.ceadu.org.uy)	Certification (ecolabel)	Uruguay	ECOSAMBITO (2007), Botero et al. (2012), MINTURD (2008)
Playas and Balnearios de Calidad (www.ambiente.gov.ar)	Certification (ecolabel)	Argentina	ECOSAMBITO (2007), Botero et al. (2012), Dadón (2005)
Praia Dourada	Evaluation	Portugal	Williams and Micallef (2009), Vaz (2008)
Award Ecoplayas (ecoplayas.rcp.net.pe)	Certification (award)	Peru	Botero et al. (2012), ECOPLAYAS (2007)
Award Ecoplayas (www.ategrus.org)	Certification (award)	Spain	ATEGRUS (2012)
Quality Coast Award (www.keepbritaintidy.org)	Certification (award)	United Kingdom	Vaz (2008), Williams and Micallef (2009)
Sostenibilidad para Destinos Turísticos de Playa (www.mincomercio.gov.co)	Certification (ecolabel)	Colombia	ECOSAMBITO (2007), Botero et al. (2012), ICONTEC (2011)
Sustentabilidad de Calidad de Playas (www.semarnat.gob.mx)	Certification (ecolabel)	Mexico	ECOSAMBITO (2007), Botero et al. (2012), SEMARNAT (2006)

Information of nine BCS was obtained after 4 years of searching in many different sources: (a) Official documents provided or published by organizations in charge of each BCS (i.e. Foundation for Environmental Education for Blue Flag); (b) Scientific documents such as papers, books and thesis related with beach management in Latin America; (c) Interviews with several officials of governments and organizations related with tourist beaches; (d) Interviews and discussion with many researchers of Latin America, mainly members of PROPLAYAS Network; (e) Visits to beaches in Argentina, Brazil, Colombia, Cuba, Ecuador and Uruguay.

All information acquired was organized in a common structure for the nine BCS. This procedure allows analyse each scheme in the same pathway, in order to describe its origin, management framework, structure and implementation. First part describes the establishment of the BCS in its own country, analysing the legal, political and technical framework existing in these times. Second part, management framework, is focused in those organizations with any role or function related with promotion, implementation or assessment of the BCS, and also describes procedure to obtain the award. Third part is the most technical, due to it describes all criteria required to endorse quality within particular sense of 'beach quality' gave for each BCS; this part is focused in those 'categories' in which criteria is grouping. Finally, a general description of implementation of BCS during years and geographical coverage reached is commented in the fourth part. In order to synthesize all this information, originally in Spanish, each BCS in Latin America will be shown.

2.4.1 Argentina: IRAM 42100

The Argentinian BCS, IRAM 42100, starts at 2003, when is created a commission for writing the Guidelines for Beach and Bathing Areas Quality Management. This commission was steered for the University of Buenos Aires and they tested these guidelines in three coastal cities: Mar del Plata, Necochea and Villa Gesell (Dadon 2005). At the same time, at 2004 was approved the Environmental Policy for Growing and Equity and afterwards, at 2005, the Strategic Federal Plan for Sustainable Tourism. Within this context the BCS was created as a Technical Guideline for the Argentinian Institute of Standardization – IRAM, therefore its name and number (IRAM 42100).

As a result of its origin, the IRAM 42100 is managed by the Argentinian Institute of Standardization. This organization do all tasks related with promotion, auditing and certification of bathing areas interested in be awarded. Punctually, main difference of this BCS over others is its focus, which is not a beach, unless a small portion of coastal zone called *bathing area*. This model is very common in Argentina and Italy, but not in other countries (see Fig. 2.1). This scheme has the advantage that defines clearly the responsible of management, but at the same time the disadvantage is fragmentation of beach in several units; in a beach can be more than 20 *bathing areas* independent among them.

Fig. 2.1 Typical use of the beach in Argentina

The BCS is awarded for 3 years, although each year is an auditing procedure done by IRAM's consultants. All visits for auditing are paid for owner of the bathing area, and also guidelines to start implementation of scheme have a cost. Majority of improvements of the beach are done for the private sector (bathing areas), with a slight cooperation with public institutions. Nevertheless, guidelines ask for users' perception surveys, allowing continuous improvement of quality.

About its structure, IRAM 42100 does not depend of type of beach, due to scheme is focused in subunits (bathing areas) of the beach. It has six categories of requirements: (1) Environment, coastal scenery and resources; (2) Infrastructure and facilities; (3) Safety and security; (4) Staff; (5) Environmental information and education; and (6) Management system (Fig. 2.2). This BCS is very close to ISO standards, but it can be considered as an ecolabel because all requirements are predefined.

Finally, implementation of IRAM 42100 has been less than expected. After 8 years, only three *bathing areas* have been awarded and only one in coastal zones; other two were obtained in rivers. Nevertheless, the owner of this *bathing area* highlighted advantages of BCS, specially related with improvements of infrastructure and demand for next tourist season. In the other side, no one of the three 'pilot' beaches had any bathing area awarded, as was expected in 2005. In conclusion, success of this BCS is very limited, perhaps due to high costs of implementation, strictness of guidelines and weak interest of public institutions at local scale.

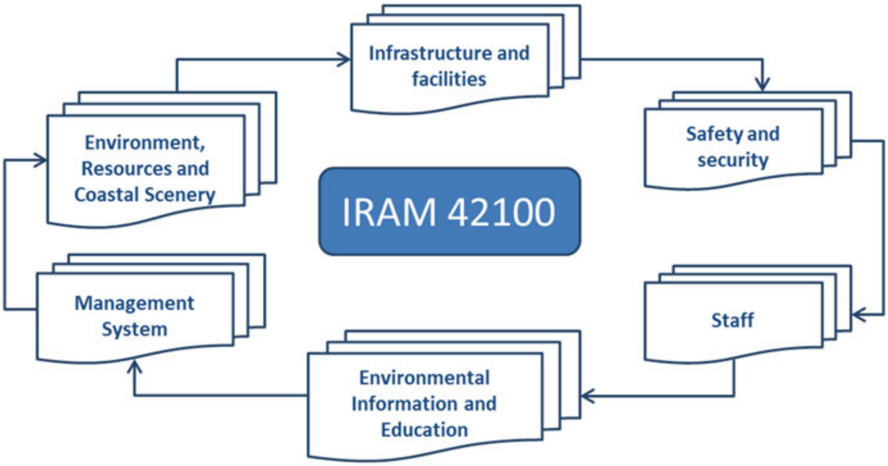


Fig. 2.2 Structure of categories of IRAM 42100

2.4.2 Colombia: NTSTS 001-2

The Colombian beach scheme certification was an initiative of Ministry of Trade, Industry and Tourism, following the National Tourism Law (Law 300 of 1996). This institution created the Sustainable Tourism Sectorial Unit, which was in charge of formulate six Technical Standards for tourism industry, being one of them about beaches. As a result, in 2007 the National Standards for Sustainable Tourism in Beach Destinations was approved with code NTSTS-001-2. Despite of public support, after 4 years without implementation, in 2011 the certification scheme was evaluated and approved a new version.

About its management framework, this certification is property of Ministry of Trade, Industry and Tourism; therefore its implementation should follow national tourism guidelines. Nevertheless, assessment of a particular beach, in order to give certification, can be done for any company approved for auditing technical standards. When a beach passes the evaluation procedure, the Ministry give the certification NTSTS-001-2 for 3 years, although each year has an auditing visit.

One of the most interesting aspects of Colombian BCS is its scope. Normally beach certifications cover strictly the ridge of sand and the bathing area of beaches, but NTSTS-001-2 is focused in classical tourism concept of “destination”. This particular scope, wider than normal certification schemes, allows managing not only the beach, but also its surrounding areas. At the same time, its scope has as a disadvantage the difficulty to establish an exact edge of beach management area, because it is according to the destination borders.

Structure of the Colombian beach certification scheme is composed by three general categories and four specific ones (Fig. 2.3). The formers are related with legal issues, a sustainable management system and territorial destination area. About specific categories, they are related with environmental, security/safety,

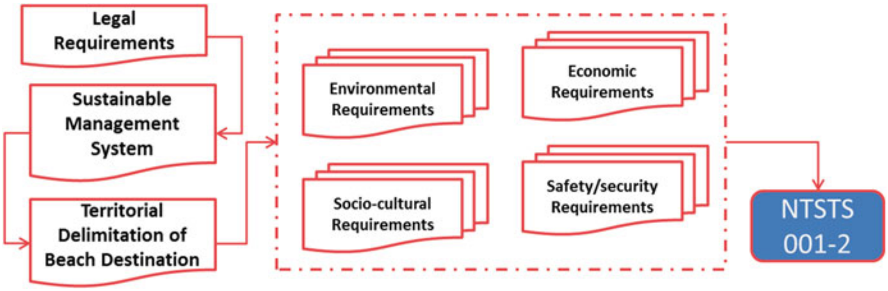


Fig. 2.3 Structure of Colombian beach certification NTSTS-001-2

socio-cultural and economic issues, each one with several requirements to fulfil. A beach with 100 % of these aspects covered will obtain the beach certification, but normally it is so difficult for any initiative of a single beach. Moreover, when all requirements are compulsory, continuous improvement is not necessary, because the beach must fulfil all aspects before obtain the certification.

As it was commented above, after 4 years this certifications was evaluated and a new version approved. However, it happens because no one beach was awarded with the NTSTS-001-2 scheme, despite of efforts of Ministry of Trade, Industry and Tourism for its implementation. During 4 years this Ministry spent almost 100,000 US dollars promoting a pilot case in the north coast of Colombia, in a highly conserved beach within a national park. Nevertheless, this initiative was leaded from Bogota, 1,000 km away of Caribbean coast, and for consultants with limited experience in beach management; as a result, the beach was not certificated. There are other examples of initiatives supported by the Ministry, such as the beach *El Morro* in Tumaco Islands, at the pacific coasts, but nowadays no one has the award NTSTS-001-2.

2.4.3 Costa Rica: Bandera Azul Ecológica (Ecological Blue Flag)

Beach certification scheme of Costa Rica was the first created in Latin America. It was established in 1996, by the Water and Sewage Institute, with many other institutions such as National Tourism Institute, Ministry of Environment, Energy and Communications and Ministry of Public Health. This BCS is the only one approved and updated by presidential decrees, during its more than 15 years of existence. Due to success of Ecological Blue Flag – EBF, since 2001 this award is also given to local communities inland. As a result, in 2012 this certification scheme had nine categories, being beaches only one of them.

Administration of EBF is leaded by a national commission conformed for ten different institutions, among public and private ones. Main responsibilities of this commission are: (a) To prepare an annual working plan; (b) To assess, audit and

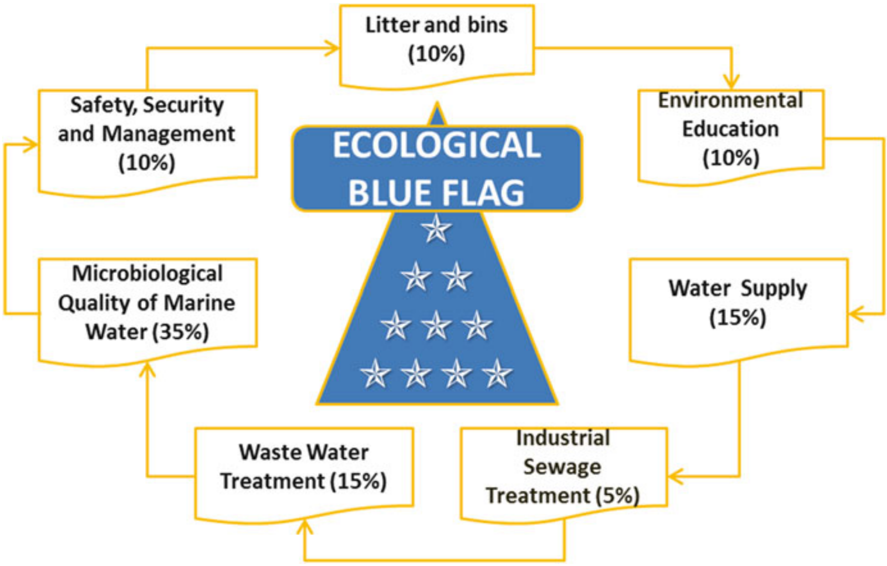


Fig. 2.4 Structure of beach certification scheme of Costa Rica

monitor local committees winners of previous certification; (c) To adjust any requirement of the certification scheme for future implementation. In brief, almost whole administration of BCS is in charge of this national commission.

At local level, communities linked to beaches must create a *Pro-EBF Committee*, which will lead application for the award. Procedure to obtain the award starts in March, when each committee must send documentation to the national commission. In April local committees must send a working plan and in December they must submit an annual inform. A particularity of this scheme is that water quality sampling is done by AYA, meanwhile in others BCS is done by each beach's responsible. The Pro-EBF Committees have to be conformed at least for five members and they must receive two or three auditing visits per year.

Structure of certification has seven categories, each one is weighted accordance with importance for beach management; in Fig. 2.4 can be seen proportion of each category. Majority of requirements are focused in environmental and sanitary aspects, as a result of its origin in an institute of water and sewage. EBF has a big advantage due to its multilevel assessment, which is represented in five starts as a maximum recognition; first stars are related with environmental aspects and latter with improvements in services and facilities.

Ecological Blue Flag is the most implemented BCS in Latin America. Its success stem from the huge public support received during more than 15 years, and very clever strategies in its administration, such as committees with local community and the multilevel structure mentioned before. Moreover, assessment of beaches awarded is rigorous, as it is demonstrated for many flags lost each year in different localities.

Despite of that, during year 2012 the national commission awarded hundred beaches in this small Centro American country, an evidence of its success. Perhaps only weakness is absence of beach typologies and measure of carrying capacity, zoning and public perception.

2.4.4 Cuba: Playa Ambiental (Environmental Beach)

Beach Certification Scheme applied in the Republic of Cuba is called Environmental Beach. This scheme is the result of implementation of the Environmental Management Programme of Varadero Beach, run by the Office of Varadero's Coastal Management (JCPV 2008). The scheme is created in May 2009 by the Ministry of Science, Technology and Environment – CITMA, framed by Decree Law No. 212 of 2000. Previous, between 2002 and 2005, several meetings between authorities of tourism and environment were conducted to analyse the application of any of the existing schemes, such as the Blue Flag Programme and the ISO 14001 standard. The result was the creation of a scheme adapted to the Cuban socio-political reality, in a context of high state control, which is hardly compatible with the schemes supported by NGOs.

The certification *Environmental Beach* is a scheme of provincial character, so far it was designed to hotels and entities located in the first line of Varadero beach. Certification must be requested by a group, committee or body in charge of beach management and conformation of it is one of the general requirements. The brand is owned by the Provincial Delegation of the Ministry of Science, Technology and Environment, and in particular the Department of Environment, that's who can grant or to remove the award according to compliance and maintenance requirements.

Regarding evaluation of requirements, it begins once the beach committee or body formally request to the Territorial Delegation of CITMA, submitting the required documentation. After reviewing the documentation, an Evaluation Committee must visit the beach and check that it meets the general requirements as well as the 13 categories of criteria established by its rules (JCPV 2008). Once this committee reviews that the beach meets the minimum qualifications, the awards is delivery for 3 years. From this point can be displayed Environmental Beach emblems, although it must pass an annual review and periodic audits performed by CITMA, which may involve proceedings for annulment and withdrawal of certification if failures are evident.

The Cuban BCS is quite solid, being composed of two distinct blocks. Initially, there are four general requirements, which are prior to the improvement of the beach conditions. These requirements begin with the creation of the group, committee or manager in charge to certify the beach and after it should establish a public environmental commitment. Accompanied by these two requirements, it is compulsory identify current legislation, corroborating its full implementation. Finally, it is verify easy and safe access throughout the length of the beach, protecting its property as a public area.

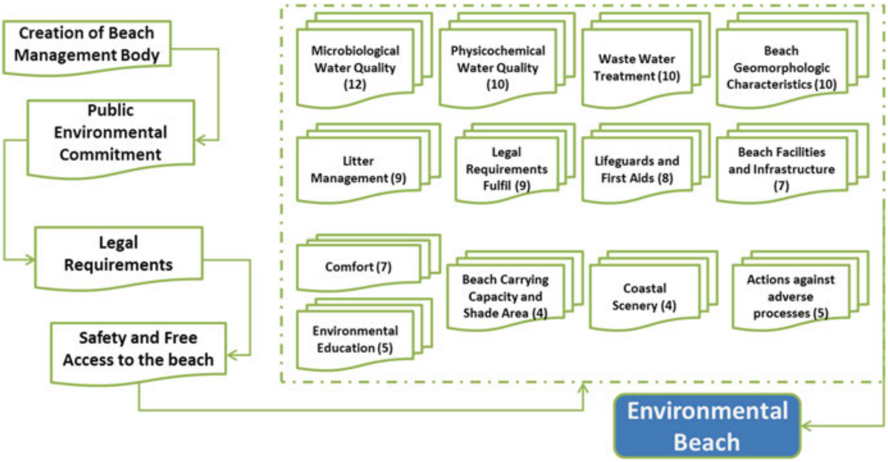


Fig. 2.5 Structure of Cuban beach certification scheme

Once the general requirements area achieved, there are 13 categories that must meet the beach, as shown in Fig. 2.5. Each category has a percentage value of points, which were established by a panel of experts, following the Delphi Method (JCPV 2008). This scheme considers itself as an "Environmental Management System", thus the categories with higher values are related to water quality and wastewater treatment. Main weaknesses are the absence levels of assessment, differentiation of types of beaches and measurement of user perception.

About its implementation, between 2010 and 2012 had been delivered three certified beaches in Varadero and were still under evaluation five applications, not only in Varadero but also on the beaches of the city of Matanzas. Likewise, in the second half of 2012 was considering extending the scheme to the five provinces of the GEF's Project Sabana-Camagüey. If this implementation is materialized, and considering the state of Cuban State hierarchy, we can predict that within 5 years, this BCS would be one of the most implemented in Latin America.

2.4.5 Ecuador: INEN 2631

Beach certification scheme in the Republic of Ecuador comes from sustained and funded effort in integrated coastal management for over 20 years. In 1984 Ecuador began a coastal zone management programme, funded by resources of the cooperation agency of the United States of America – USAID, for two million dollars. Subsequently, the Inter-American Development Bank – IDB granted two loans to implement the Programme of Coastal Resources Management – PMRC, funded with 12 million dollars between 1987 and 1996, and 14 million dollars between 2001 and 2004; most PMRC actions focused on support aquaculture, but also were invested resources in tourism.

In 2006 the consulting company perform various studies and consultation processes in the Ecuadorian beaches. As a result, they write the Handbook of Technical Standards for Beaches Tourist Certifications (PMRC 2007), as well as a consultancy report with worthy information and excellent comparative analyses (ECOSAMBITO 2007). Finally, in 2011, between September and December, the Technical Sub-Committee on Sustainable Tourism of Sun, Sea and Sand was created in the Ministry of Tourism; in March 2012 they officially approved the beach scheme certification, called INEN 2631:2012.

The INEN 2631 standard is managed by the Technical Steering Committee and Monitoring – CTSM, which is led by the Ministry of Tourism and composed of representatives from 12 institutions. To start the certification process on a beach, a Local Beach Management Committee – CLGP should be formed, consisting of at least five members in urban and rural beaches and six in uninhabited ones; members stay in their roles for 1 year minimum. The main responsibilities of CGLP are: (a) Represent the beach enrolled in the certification process; (b) Manage compliance with the quality requirements of the beach to obtain, maintain and renew the certification of tourist beach.

One of the most interesting aspects of the administration of the Ecuadorian certification is the ongoing collection of information on the beach. The Local Beach Management Committee must make permanent monitoring all weekends and holidays of the year, recording it in a form provided in the Standards for this function. Monthly, the CLGP sends the information collected from monitoring to the Technical Steering Committee and Monitoring for global processing (INEN 2012). Similarly, the CLGP must submit an annual activity report to the CTSM with respect to compliance with the requirements of the standard. For certification and renewal of the award, an assessment is made during the peak months, without notice.

The Ecuadorian scheme, as a result of specific studies for its creation and review of other certification schemes, has a structure more consistent and clear that most schemes previously created. The three main aspects that define its structure are: (a) Assessment requirements in three distinct types of beach; (b) Certification levels; and (c) Homogeneous categories of requirements. In fact, about requirements to assess and their organization, the scheme has a first category of *General Requirements*, which include criteria such as gathering of information and the creation of the Local Beach Management Committee. Then, they must meet circa 70 aspects, grouped in 7 categories or axes (Fig. 2.6). Although not all requirements must be applied in the first level of certification, it is considered that in any case their amount is excessive to start a management process in a Latin American beach.

Finally, in the second half of 2012 was barely starting the implementation of the ECP, given its recent creation. However, in interviews conducted in Guayaquil in July 2012, as well as the fluid contact with those responsible for certification in Ecuador, it is known that the Ministry of Tourism is promoting the certification of four beaches in the Ecuadorian coast, with assistance of a working group led by an expert in tourism and community organization. The goal of this organization was to begin certifying beaches in 2013, although there is no information of this result.

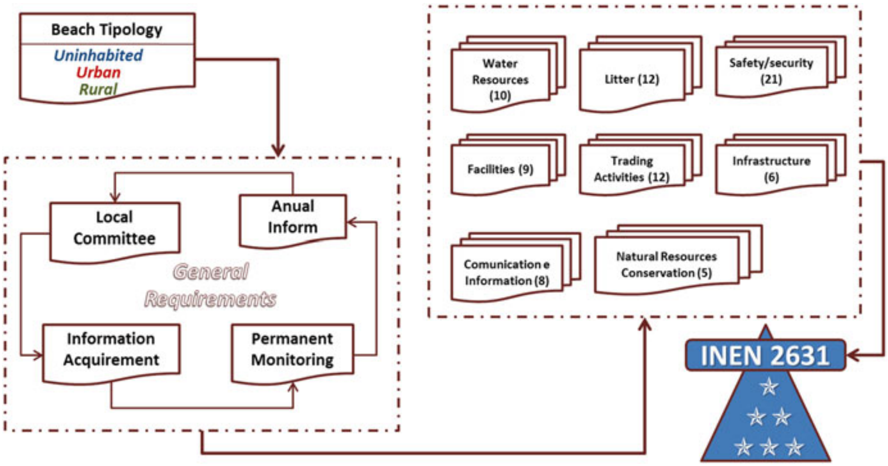


Fig. 2.6 Structure of Ecuadorian certification INEN 2631

2.4.6 Mexico: NMX-AA120-SCFI-2006

The Mexican BCS has its beginnings in the Clean Beaches Programme, which focuses on water quality at beaches. This was a joint initiative of the Departments of Navy, Environment, Health and Tourism. Subsequently, the Clean Beaches Committees, who are responsible for managing the beach, arise as a derivation of the Watershed Councils. In May 2003, in a meeting in Nuevo Vallarta, the institutions most closely related to the management of tourist beaches agree to create a Mexican beach certification. After 3 years of work, coordinated by the Secretariat of Environment and Natural Resources – SEMARNAT, in 2006 is published in the Official Gazette the Mexican Standard NMX-AA-120-SCFI-2006.

The coordination of the BCS is done by the environmental authority (SEMARNAT), in contrast with other countries where certification is dependent on the tourism authority. About managing the beach to be certified, the Standard NMX -AA -120 states that may be requested by municipalities, Clean Beaches Committees and/or individuals and companies interested in assessing the quality of the beach. To begin the certification process, they must initially submit an application to the SEMARNAT that is downloaded from the website of this institution, attaching the questionnaire included right there. Upon receiving the request, SEMARNAT directs it to the Mexican Institute of Standardization and Certification – IMNC, who sends a new request form certifying the quality of beaches to the applicant, which must be completed and handed back to IMNC. This standardization institute is in charge of doing the audit, which has a significant cost. If the result is approving, the applicant receives a flag that should be placed on the beach. This certification is valid for 2 years, but must undergo an assessment to audit a year of fulfilled granted, which follows a similar procedure to obtain.

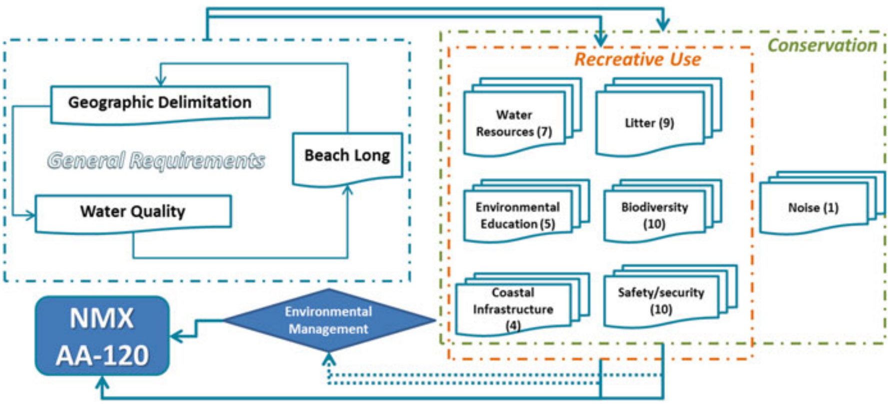


Fig. 2.7 Structure of categories of Mexican beach certification scheme

The NMX-AA-120 Standard considers two types of beaches for certification: (a) Recreational, defined as those where leisure activities are performed; and (b) Beaches under Priority Conservation, being those located within the territorial limits of various types of protected areas. NMX-AA-120 standard has a first category of *General Requirements*, which include mapping and bathing water quality aspects. Once these are fulfilled, each type of beach has specific criteria, but categories are almost identical: (a) Water Resources, (b) Litter, (c) Coastal Infrastructure, (d) Biodiversity, (e) Safety and Security, and (f) Environmental Education. In beaches with priority for conservation, further requirement is included relating to noise pollution (Fig. 2.7).

Since its creation in 2006, the NMX-AA-120 standard has been increasing year by year coverage on the wide Mexican coastal zone. For the second half of 2012, 14 beaches were certified between the Pacific coast and the Caribbean, being the former where the majority (11) are located. Certified beaches are available on the website of SEMARNAT, which is updated monthly with information provided by the IMNC. Although the amount of certified beaches is higher than that achieved in Argentina or Colombia, countries with similar certification schemes and coastal lengths, value is less than what would be expected for a BCS with 6 years of implementation.

2.4.7 Peru: Premio Ecoplayas (Ecoplayas Award)

The Beach Certification Scheme in Peru is not a permanent management system, unless an award that is given annually. It was originated in 2006 as a private initiative of the Peruvian Beaches Ecological Organization – ECOPLAYAS, which has administered it from that date to 2012. Due to it is a private initiative, which also only gives three awards per year, one per category as will be explained further, its development has been minimal during its 6 years of existence.

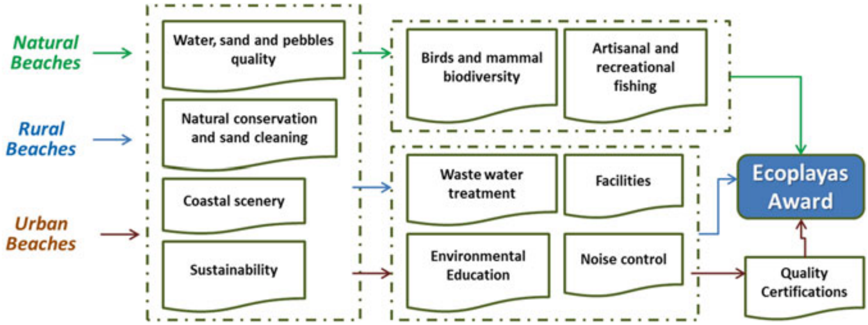


Fig. 2.8 Structure of categories of Ecoplayas award

The administering organization exercises all functions of the award, from promotion to evaluation. The participatory process has five steps, which are followed consecutively. Initially, between mid-December and mid-January, the institutions interested in participating should register online at the website of the Organization ECOPLAYAS. Once all applications are received, the managing organization team visits each of the beaches and evaluates requirements according to the category in which was inscribed the beach; this visit is prearranged with the requesting institution. After gathering information and verified on-site conditions, a Jury reviews the evaluation reports and decides in closed session to reward three beaches, one per category. Finally, the last days of March, the awards ceremony is done which public authorities and press participation.

The prize is awarded annually, consisting of a trophy-winning permanent possession and delivery the *Ecoplayas Flag* for 1 year, which shall be hoisted on the award-winning beach. As a particular case, if a beach wins for three times the award, it will be granted in perpetuity possession of Ecoplayas Flag. When a beach earns the award, to be presented again to the prize, it must wait at least 1 year.

The ECOPLAYAS award has three categories of types of beaches: natural, urban and rural. *Natural Beaches* are considered those with minimal human intervention and which can make sports and ecotourism. The *Urban Beaches* are coastal areas that are near or in cities with large populations. These beaches must have roads and public transport, a significant flow of visitors in the tourist season and good urban infrastructure and public services. Finally, the *Rural Beaches* are those coastal areas located in the vicinity of cities with small populations and rural geographical environment, but with seasonal influx of visitors; they must have minimum infrastructure services to the public.

Assessment requirements are very similar among the three categories, difference being some specific criteria for each type and strength of aspects, such as services to the public. All categories have four common requirements, which are: (a) Natural conservation and cleaning; (b) Water, sand and/or pebbles quality; (c) Coastal scenery; and (d) Sustainability (Fig. 2.8). Specific requirements for Natural Beaches are: (a) Presence of typical biodiversity of birds and/or mammals, and (b) Freedom to practice artisanal and recreational fishing. On the other side, urban and rural beaches share most of their specific requirements: (a) Wastewater treatment; (b) Facilities (baths, showers, bins/cans, umbrellas, sports facilities, etc.); (c) Noise Control; and d. Environmental education; as an exclusive requirement for urban beaches is included existence of other quality certifications, which is quite striking, since it allows (and promote) multiple certification.

ECOPLAYAS award has been given since 2006 without interruption, which means 18 beaches awarded until 2012, although some have repeated the award. Due to this award is granted only for three beaches per year, its application has been restricted to this number, without have been considered extension to more winning beaches until the end of 2012. As a weakness of this BCS, historical information about winning beaches is not available, neither about structural changes during its implementation.

2.4.8 Puerto Rico, Dominican Republic and Brasil: Blue Flag

As already mentioned above, Blue Flag is the most recognized and ancient beaches certification scheme in the world, being as well the most studied in the literature (Botero 2008, 2009; Cagilaba and Rennie 2005; ECOSAMBITO 2007; Vaz 2008; Williams and Micallef 2009; McKenna et al. 2011). However, their success has roots in December 1975, when the European Commission adopted the European Bathing Water Directive 76/160/EEC, marking a change in the assessment of the environmental quality of bathing areas and maritime beaches.

Due to an innovative blend of environmental conservation with tourism promotion, the programme impact quickly the European countries and after abroad, being present in 46 countries in 2012. However, their creation in Latin America dates back to 2004, when it was the first Blue Flag raised in the continent. It is noted that Blue Flag is a registered mark of an NGO, which only depends on itself to fit each country, avoiding the lengthy administrative processes of public bodies and standards organizations that have had to follow other Latin American BCS for its creation. In this sense, if a country wishes to participate in the Blue Flag programme, the first step should be to register as a non-profit organization called *National Operator*, for which it must follow a series of steps: (a) Organization of a Blue Flag seminar/workshop; (b) Establishment of a National Blue Flag Committee; (c) Assessing the feasibility of permanence Blue Flag; and (d) Implementation of a pilot phase of the Blue Flag programme. Once this procedure is completed, authorization to use the brand is given to the National Operator.

The management of the Blue Flag Programme is focused on this National Operator, although its application guidelines are not defined for each country, but they are centralized at the Head Office of FEE, in Denmark. The role of the National Operator focuses on two aspects: promoting and evaluating the beaches in their jurisdiction. In regard to the promotion of the programme, one of the great successes of this BCS is it public and fully mediated delivery of the certifications to beaches that have passed the evaluation process. Regarding the assessment of the beaches, a Blue Flag beach can only be requested by the municipal authority (mayor), which is very relevant in countries with strong local governments, such as the case of Europe, but it is a weakness in countries like those of Latin America, where the political responsibility for environmental issues is still scarce on the local scale.

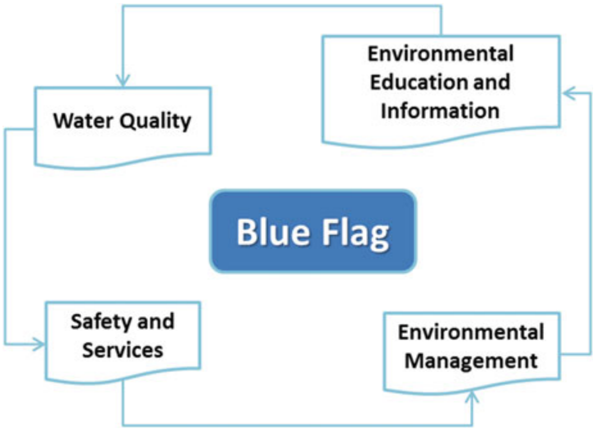
The Blue Flag award is given annually, receiving nominations for a period established by each country (e.g. Puerto Rico: May to June). Applications are selected for a National Jury that which main authorities with beach management responsibilities are represented. Once applications are approved at the national level, they are forwarded to the International Jury, which is composed of representatives of the FEE and the international partners of the Blue Flag Programme. The jury ultimately decides which flags are to be delivered for the following year, in each of the 46 countries that make up the programme. The certification comes through referral documentation, not an audit in situ, marking a profound difference with those BCS based on standards. Therefore, fulfilment of criteria is ensured by various means, such as monitoring of local authorities, results of official analysis of bathing water and inspections carried out by the National Operators and the International Blue Flag Coordination Programme. Again there is a weakness of the Blue Flag programme in Latin America, because after a decade of the century, it is not still common systematic and rigorous monitoring of bathing water quality on beaches in the continent.

However, as noted by the Blue Flag Programme itself, the best observers for meeting the requirements are the users of the awarded beaches. In fact, if the National Operator receives repeated complaints from users, it can remove the flag, or at least make a visit verification conditions that can also result in the cancellation of certification. A very special situation allows for this BCS, because if some beach not met sometime certain requirements, such as the quality of the bathing water, it is allowed to lower the flag until it reach again the level of quality and so it can be raised again.

The Blue Flag Programme requirements are grouped into four categories: (a) Water Quality; (b) Environmental information and education; (c) Environmental management; and (d) Safety and services (Fig. 2.9). It is noted that the BCS has a strong bias towards environmental requirements, being the centre of three of the four categories. The Blue Flag Programme is applicable in any type of beach, but due to the minimum required infrastructure and equipment, has been widely criticized for its use on not urban beaches (Nelson and Botteril 2002; Nelson et al. 2000). However, going against several of the most critical scientific papers to Blue Flag (Rees 1997; McKenna et al. 2011), the BCS does not have much more demanding requirements than other certifications such as Argentinian or Ecuadorian ones. Perhaps the weakness of the programme is not so much the application of equipment, such as the inability to adapt to the types of beach according to the natural variable.

As was commented before, Blue Flag Programme began in Latin America in 2004, when the first four beaches in Puerto Rico were certified. Subsequently, it was implemented in Dominican Republic (2004) and Brazil as the last country that has joined the ECP in the continent (2009). In Blue Flag website appears also an

Fig. 2.9 Structure of Blue Flag requirements



institution in Mexico, however in 2012 this country had not yet received any flag. Regarding the number of certified beaches in the continent, for 2012 were given 21 Blue Flags, divided in 7 to Puerto Rico and 14 to Dominican Republic and no one in Brazil. In proportion to other BCS less recognized, as the Ecological Blue Flag Programme or NMX -AA -120 Standard, the Blue Flag Programme coverage is still low.

2.4.9 Uruguay: Playa Natural (Natural Beach)

The Natural Beach Standard was developed in 2003 by a group of experts from the NGO Centre for Studies, Documentation and Analysis of Uruguay, from a request of the Ministry of Tourism and Sports – MINTURD. Officially, the Standard was approved by the Executive Decree 406 of 2003, in order to use it as promoting environmental certification on beaches. For processing, other schemes were consulted, such as Blue Flag and Blue Wave, although the approach finally proposed was much closer to the ISO 14001 environmental management standard, which profoundly defined their ultimate design.

The Natural Beach management is focused entirely on the Ministry of Tourism and Sports, who is also the owner of the trademark. This issue should be emphasized, since it demonstrates a unique consistency of *national brand* (Uruguay Natural) with the beach certification scheme. The relevance is supported because a certification is primarily *recognition*, then for making it more convenient for a BCS national in scope, it should have same image as the country. Moreover, management the certification on the beach is via a Responsible Party Beach – GRP, which may be a public or private organization with demonstrable environmental management authority over the beach, such as the Mayor Municipal Commission of Development, a local NGO, a business chamber or to a neighbourhood association.

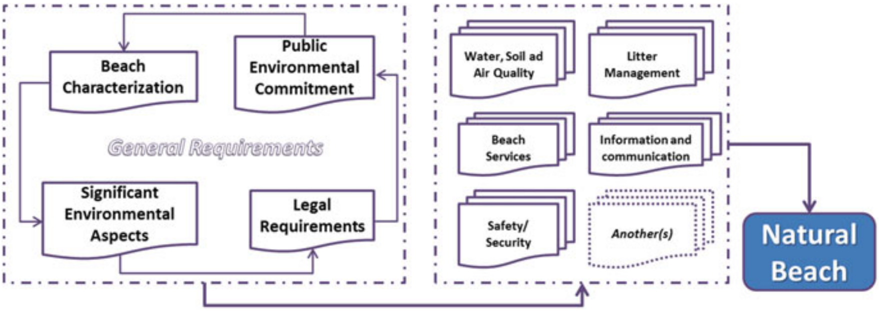


Fig. 2.10 Structure of categories of Uruguayan beach certification scheme

The assessment process begins with requirements to MINTURD and a certification body accredited to this standard. Subsequently, the external audit is performed by the certifying body, which sends the results to MINTURD who, if satisfied that all requirements are met, gives the award to the beach. As is common in the certification process, the GRP should bear all the expenses arising from the external audit, as well as improving the condition of the beach. The duration of certification is up to 12 months, as explicitly stated in the rule is valid until 31 March following delivery, being consistent with the summer in the southern hemisphere. To renew, it should hire a certification body, which performs the external audit and send the results to MINTURD again.

Natural Beach scheme is more consistent with the certification of management systems, both environmental and quality. Although other BCS, like Colombian or Argentinian, promote the implementation of management systems, Uruguayan scheme really works as such. Importance of this structure is that instead of placing specific requirements to meet, the scope of certification is the quality of health, ecosystem and recreational conditions of the beach. This aspect is quite confusing, but part of understanding is the ability to differentiate a management system of an eco-label, such a Blue Flag Programme (WTO 2002).

Uruguayan BCS has a category of general requirements, framing four aspects: (a) Beach characterization; (b) Public environmental commitment; (c) Significant environmental issues; and (d) Identification of legal requirements. Additionally, it must establish five environmental management programmes, depending on the following areas: (a) Water, air and soil quality: (b) Litter Management; (c) Beach Services; (d) Information and communication; and (e) Safety and security. In addition to these five programmes, the Responsible Party Beach may add it deems necessary for the proper environmental management of the beach (Fig. 2.10).

Baseline information of this standard is curiously low, since BCS is very focused on brand recognition. However, in 2008, during the 5 years after its adoption, a workshop organized by the PROPLAYAS Network review its implementation and make adjustments in the text for their improvement (PROPLAYAS 2008). In this activity it was concluded that after 5 years of implementation, it had accumulated enough experience to review the BCS and create a new version; however, a part of it, not much more information is known.

2.5 Discussion of Beach Management in Latin America

Integrated beach management in Latin America has become a new professional field, in full development. Every time there are more researchers, government's authorities, non-government organizations, managerial technicians, educators, and other sectors of the society that study, promote and apply this kind of management. As a consequence, it demands to systematize the lessons from the theoretical-methodological approach adopted and the own practical experiences.

It is still frequent that management's actions do not focus on ecosystem-based and adaptive approaches, but rather it is opted to maintain rigid and static procedures to natural and dynamic systems. The coastal management still implies planning, organization, address and control of coastal resources towards certain objectives of uses, particularly tourist, conceiving beaches like a service, and not, like it should be, as a natural system. As consequence of this incorrect interpretation, many administration efforts and tourist certification of beaches have been configured more as marketing mechanisms that management tools.

In front of this conception, the vision of a true integrated management of beaches should be prefixed, it means a holistic and dialectical character, in that multiple actions are developed, very articulate to each other, and always looking for maintaining functional base, and multiple and sustainable use of the coastal system. After that, feasible and coherent efforts of certification can be constituted in an exceptional value added for tourist-recreational activity.

Although reductionist focus, dominant until our days, has caused a vision and a fragmented practice of the coastal management in general, integrated beach management will advance toward an increased character of *interdisciplinary*. In consequence, it is necessary to develop a varied and sufficiently solid arsenal of tools, among which is necessary to highlight the approaches of functional zoning and carrying capacity, risks' studies and public perception, the best variants in the coastal engineering, monitoring systems focused on making decisions, and other many, but always based on the particularities of the system beach, so that they can be applied with true effectiveness.

A notable influence has been exercised in several countries of Latin America, among those that is necessary to highlight to Cuba, Uruguay, Mexico and Colombia, the conception of the cycles of coastal integrated management, suggested from 1996, for the Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection – GESAMP. This method translates continuous development to coastal management, through diverse phases that go from characterization and diagnosis, until conformation, implementation and evaluation of programmes, improving in a systematic way.

However, in many cases, the practice demonstrates that these processes have been characterized by expensive and long periods of time, and erratic results in its implementation. In this sense, it is necessary to criticize persistence of focus and partial actions in some initiatives of integrated beach management, as a result of the very frequent fragmentation of the reality, and exhaustive processes of characterization and diagnosis that are carried out starting from multidisciplinary studies. In those, specialists of different areas, very expert in their topic, describe elements and problems from their perspective, that constituted later in the bases for elaboration and implementation of integrated beach management programmes, that end up in excessive detailed working guides, what impregnates whole process with a classic top-down focus.

This gets more complicated in the face of the reality, pointed out by Botero (2013), due to authorities usually have too short periods of time for support these programmes, making impossible than they complete a whole cycle of coastal management. Therefore, local authorities usually concentrate only their interest on those matters that come closer to their government objectives, and within this reality, it is practically impossible that integrated beach management process can be successful, because it requires continuity in the time.

Stem from this reality, with too much frequency coastal management is developed over conceptions, forms and instruments decided by few ones, for a short term and directed by people that hardly ever have the holistic formation, neither time that is required. As a consequence, evaluation stage is reached very few times, and the most common solution is that new studies of diagnostic are restarted, what believes a scenario of continuous frustrations, in which credibility of efforts is lost.

The learning of this analysis, it is that a less mechanical and absolutist integrated beach management should be adopted in its concepts and methods, and that it should advance toward a wider vision to each particular coastal system, in all its dynamic complexity, and in permanently evolution. It is very important that programmes of integrated beach management continue moving from current emphasis in descriptions and diagnoses, toward an operative design focused on what and how to manage these singular and complex ecosystems integrally, and taking in consideration particularities of socio-political, normative and technician-scientific contexts in that they are developed.

Another learned lesson, and outstanding for immediate future, it is that being beaches in majority of Latin American countries a public goods, in which converge a wide range of stakeholders with diverse interests and intervention forms, it is necessary coordination of actions that is only possible through conformation and operation of participative bodies for integrated management of beaches, that act as valves of the interrelations among all stakeholders. This way, in Latin America are indispensable conformation of Beach Management Organs at local scale, and they should be constituted as the basic decision making structure, with a public-private character, non-profit nature, participative, pluralistic and contributive regime, and relatively autonomous as for their organization, conformation and decisions (Botero et al. 2008).

Moreover, in the broader context of Beach Certification Schemes that have been created in Latin America, we should note the lack of communication between the organizations managing the nine schemes. As a result, it has not taken advantage of the synergy that can generate joint work in a region that shares many similarities and the same interest of management. Additionally, the BCS still are applied paltry amount compared to the number of tourist beaches of the continent, being an indicator of reduced success in terms of implementation of this tool. However, the progress of research in integrated beaches is evident on the continent, starting with the continuity of the Ibero American Network of Beach Management and Certification – PROPLAYAS and increased scientific publications on the subject, like this one which has presented in this book.

References

Ariza E (2007) A system of integral quality indicators as a tool for beach management. Doctorate thesis, Univesidad Politécnica de Cataluña. Blanes, Spain.

ATEGRUS (2012) Bases premio bandera ecoplayas. Asociación Técnica para la Gestión de Residuos y Medio Ambiente. Bilbao, Spain

Bird ECF (1996) Beach management. John Wiley & Sons, Chichester

Botero C (2008) Proposal of management framework for tourist beaches based on ICM. Master thesis in water and coastal management, Universidade do Algarve, Faro, Portugal

Botero C (2009) Utilidad de los esquemas de certificación de playas para el manejo integrado costero: Evaluación de ocho certificaciones en Iberoamérica. Rev Cienc en su PC 4:27–41

Botero C (2013) Evaluación de los esquemas de certificación de playas en América Latina y propuesta de un mecanismo para su homologación. PhD thesis, University of Cadiz, Spain

Botero C, Diaz LH, Hurtado Y et al. (2008) Determinación de un sistema de calificación y certificación de playas turísticas – Informe final contrato 012. Instituto de Investigaciones Tropicales, University of Magdalena, Santa Marta, Colombia

Botero C, Zielinski S, Noguera LA (2012) Esquemas de Certificación de playas en América Latina: Diagnóstico de una herramienta de Manejo Integrado Costero. Rev Costas-UNESC 1:49–63

Botero C, Anfuso G, Zielinski S et al (2013) Reasons for beach choice: European and Caribbean perspectives. J Coast Res SI65:850–885

Breton F (1998) Metodologia d'ordenació i gestió dels espais lliures en les zones litorals, Support to social science emerging groups, UAB CIRIT 1995–1997. Barcelona

Breton F, Esteban P (1995) The management and recuperation of beaches in Catalonia. In: Healy MG, Doody P (eds) Directions in European coastal management. Samara Publishing, Cardigan

Cabrera JA, Orellanes O, López L et al (2009) Evaluación del programa de manejo integrado de la playa de Varadero (Cuba): 7 años de experiencias y retos. Rev Medio Ambiente Turismo y Sustent 2:67–79

Cagilaba V, Rennie HG (2005) Literature review of beach awards and rating systems. Environment Waikato Technical Report 2005/24. Waikato, New Zealand

Cutter SL, Nordstrom KF, Kuema GA (1979) Social and environmental factors influencing beach site selection. In: West N (ed) Proceedings of the 5th annual conference on resource allocation issues in the coastal environment. Newport, RI, USA, pp 183–194

Dadon J (2005) Playas y balnearios de calidad: gestión turística y ambiental. Directrices y guía de autoevaluación. Secretaria de Ambiente y Desarrollo Sustentable, Buenos Aires

ECOPLAYAS (2007) Bases del premio Ecoplayas -Descripción y guía evaluativa. Organización Ecológica Playas Peruanas. Lima, Peru

ECOSAMBITO (2007) Establecimiento de un programa de certificación de playas turísticas en la faja costera. Informe final contrato PMRC-022-2006, Programa de Manejo de Recursos Costeros, Presidencia de la República del Ecuador. Guayaquil, Ecuador

Espejel A, Espinoza-Tenorio O, Cervantes I et al (2007) Proposal for an integrated risk index for the planning of recreational beaches: use at seven Mexican arid sites. J Coast Res SI 50:47–51

FEE (2013) Official web page for the Foundation for environmental education. www.blueflag.org/Criteria/Beaches. Accessed 20 Oct 2013

ICONTEC (2011) Norma Técnica Sectorial Colombiana NTS-TS-001-2 que establece los requisitos de sostenibilidad para destinos turísticos de playa – Second Edition. Instituto Colombiano de Normas Técnicas y Certificación. Bogotá D.C., Colombia

INEN (2012) Requisitos de certificacion turistica de playas – Norma INEN 2631:2012. Instituto Ecuatoriano de Normalización, Quito

JCPV (2008) Fundamentación y Reglamento del Sistema de Certificación Playa Ambiental. Junta Coordinadora Playa de Varadero. Varadero, Cuba

McKenna J, Williams AT, Cooper JAG (2011) Blue Flag: red herring: do beach awards encourage the public to visit beaches? J Tour Manag 32(3):576–588

Medina JR, Tintoré J, Duarte CM (2001) Las praderas de Posidonia oceánica y la regeneración de playas. In: Pericás J. La recuperación de playas, objeto de polémica. www.elmundo-eldia.com. Accessed 15 March 2002

MINAET (2011) Decreto Reglamentario del Programa Bandera Azul Ecológica. Ministerio de Ambiente, Energía y Telecomunicaciones, San José

MINTURD (2008) Especificaciones de desempeno ambiental y requisitos para playas. Ministerio de Turismo y Deportes, Montevideo

Nelson C, Botteril D (2002) Evaluating the contribution of beach quality awards to the local tourism industry in Wales – the green coast award. Ocean Coast Manag 45:157–170

Nelson C, Morgan R, Williams AT et al (2000) Beach awards and management in Wales, UK. Ocean Coast Manag 43(1):87–98

PROPLAYAS (2008) Informe de la reunión de expertos en gestión ambiental de playas – Revisión Norma Playa Natural. Red Iberoamericana de Gestión y Certificación de Playas. Montevideo, Uruguay

Radic M, Micallef A, Williams AT et al (2006) Application of the Bathing Area Registration and Evaluation (BARE) system – experience from Istria County on the northern Croatian coastline. In: Proceedings of the Second International Conference on the management of coastal recreational resources – beaches, yacht marinas and ecotourism. ICoD, University of Malta, Malta

Rees G (ed) (1997) Lies, damned lies and beach awards. Current issues, vol 1. Robens Institute, University of Surrey, Guildford, pp 1–2

Rodríguez-Perea A, Servera N, Prieto M (2002) Alternatives a la Dependència de les Platges de les Baleares de la regeneració artificial continuada: Informe Metadona. In: Pericás J. La recuperación de playas, objeto de polémica. www.elmundo-eldia.com. Accessed 15 March 2002

Rodríguez-Perea A, Pons GX, Roig-Munar FX, et al (2012) La gestión integrada de playas y dunas: experiencias en Latinoamérica y Europa. Monografíes de la Societat d'Historia Natural de les Balears, Mallorca, Spain

SEMARNAT (2006) Norma Mexicana NMX-AA120-SCFI-2006 que establece los requisitos y especificaciones de sustentabilidad de calidad de playas. Secretaria de Medio Ambiente y Recursos Naturales. Mexico D.F

Simm JD (1996) Beach management manual. CIRIA report 153. CIRIA, London

Tristá E, Juanes JL, Caballero V, et al (2012) Causas y alternativas para el control de los procesos de erosión en las playas. Experiencias de las aplicaciones en las playas del Caribe. In: Rodríguez-Perea A, Pons GX, Roig-Munar FX, et al. La gestión integrada de playas y dunas: experiencias en Latinoamérica y Europa. Monografíes de la Societat d'Historia Natural de les Balears, Mallorca, Spain

Vaz B (2008) Contributos para a avaliação e gestão de praias: a importância da percepção dos seus utilizadores. Master thesis in Ecologia Humana y Problemas Sociales Contemporáneos, Universidade Nova de Lisboa, Lisboa
Williams AT, Micallef A (2009) Beach management: principles and practices. Earthscan, London
Williams AT, Micallef A, Ergin A (2002) A theoretical framework for beach management guidelines. In: Ozhan E (ed) Beach management in the Mediterranean and the Black Sea – MEDCOAST. Middle East Technical University, Ankara
WTO (2002) Voluntary initiatives for sustainable tourism. World Tourism Organization, Madrid

Chapter 3
New Methods to Assess Fecal Contamination in Beach Water Quality

Sarva Mangala Praveena, Kwan Soo Chen, and Sharifah Norkhadijah Syed Ismail

Abstract The emerging paradigm in assessment of recreational water quality needs a basic information on microbial contamination. Recreational water is susceptible to fecal contamination which may increase in health risk associated with swimming in polluted water. Recreational water quality is monitored for fecal indicator bacteria to help prevent swimming-associated illnesses. This study aims to determine the concentrations of total coliforms and Escherichia coli (E. coli) in recreational water, Teluk Kemang beach, Port Dickson (Malaysia). This study was also aimed to determine relationship between total coliforms, E. coli and physico-chemical parameters of marine recreational water. Exposure behaviors and perceived health symptoms among beach visitors were also assessed in this study. A total of eight water sampling points were selected randomly taken at 100 m from the wash zone and 20 cm below the water surface along Teluk Kemang beach. Total coliforms and E. coli concentrations were analyzed using membrane filtration technique. Total coliforms concentrations were found between 20 and 1,940 cfu/100 ml. E. coli concentrations were between 0 and 90 cfu/100 ml. Total coliforms and E. coli were the highest at sampling point A and exceeded USEPA (Report of Task Force on guide standard and protocol for testing microbiological water purifiers. USEPA, Washington, DC, pp 1–29, 1986) guideline as it located near sewage outfall. Significant correlations were found between total coliforms and E. coli with pH, temperature and oxidation reduction potential. Microbiological water quality in Teluk Kemang public beach was generally safe for recreational activities among the public except sampling location near with sewage outfall. Output of this study will act as a crucial model for protection of marine ecosystem health. Moreover, output

S.M. Praveena (✉) • K.S. Chen • S.N.S. Ismail
Department of Environmental and Occupational Health, Faculty of Medicine and Health Sciences, Universiti Putra Malaysia (UPM), Serdang, Selangor Darul Ehsan 43400, Malaysia
e-mail: smpraveena@gmail.com; ogenki2kwan@yahoo.com; norkhadijah@medic.upm.edu.my

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8,
DOI 10.1007/978-3-319-06305-8_3,

of this study can be used as a monitoring design tool for other public beaches in Malaysia, as such studies are lacking and have not been reported so far.

3.1 Introduction

Swimming at recreational water sites such as beach is one of the most popular activities during holidays. However, according to Alm et al. (2003) recreational waters such as beach water are susceptible to fecal contamination which may increase in health risk associated with swimming in polluted beach water. Exposure to these pathogens can occur during swimming or other recreational activities through ingestion, inhalation, or direct skin contact with polluted beach water. However, there are limited estimation available on the magnitude of recreational water exposures in numerous epidemiological studies (Nevers and Whitman 2008, 2011). High public health risks posed by polluted bathing beaches are growing year by year (Sampson et al. 2006). Pruss (1998) found an increase in health risk associated with swimming in waters containing elevated levels of fecal bacteria indicators. In recent years, efforts have been made to improve recreational health risks in public beach.

According to Shuval (2003), there are more than 120 million cases of health risk cases caused by swimming and bathing in wastewater-polluted beach waters associated with exposure to microbial pathogens, which externally introduce as biological contaminants (Abdelzaher et al. 2010; Wade et al. 2006). Biological contaminants which have received huge attention are fecal indicator bacterias (FIB) such as total coliform and Escherichia coli (E. coli). Hamzah et al. (2011) have showed that total coliform and E. coli are appropriate indicators to be used as pollution indicators especially for public places like beaches. Moreover, according to Byappanahalli et al. (2008), total coliform and E. coli have been suggested as reliable indicators of sewage contamination in environmental waters. Therefore, total coliform and E. coli can be significant threats to human health through their presence in coastal waters used for recreation. By quantifying FIB presence, risk to human health are crucial requirements for an understanding its interactions, growth as well as the pathways transported from sources to regions of potential human exposure (Dyble et al. 2008).

Port Dickson is the only coastal district in the state of Negeri Sembilan, one of the longest beaches in Malaysia with white sandy beaches facing calm sea of Straits of Malacca with a coastline stretching up to 18 km. Sandy white beaches of Port Dickson have been also documented by top international tourist magazines as one of the best and most beautiful in the country (Lee and Mohamed 2009; Schwartz 2005; Thanapalasingam 2005). Teluk Kemang beach attracts the most visitors each year, supported by numerous hotels and resorts that offer a wide range of accommodation, as well as with delicious seafood restaurants. During the last five decades, Straits of Malacca received a large of waste loadings from municipal, industrial, agricultural and shipping discharges (Chua et al. 1997). Similarly, Port Dickson sea received heavy metal pollution due to growth in tourism, shipping, small industries and urbanization (Schwartz 2005; Thanapalasingam 2005). According to Kadaruddin (1997), there are 82 wastewater pipe lines discharge wastewater including sewage from hotels and houses directly into the sea in northern part of Port Dickson which lead to degradation of the marine water quality causing significant negative impacts on marine ecosystem. Additionally, Department of Environment found that majority of the Port Dickson coastal water was polluted by E. coli (Department of Environment 2006). According to Law et al. (1990), insufficient sewage water treatment is one the most pressing environmental problems that have caused deterioration of water quality in Port Dickson. Previous studies in Port Dickson beach water have focused in nutrients (nitrate, phosphate, ammonia) to evaluate its spatial behavior (Law and Chu 1990; Law et al. 1991, 2000, 2001, 2002; Japar Sidik et al. 1995; Abu Hena et al. 2000). Latest study done by Praveena and Zaharin (2013) concluded that increase level of nutrients concentration compared to previous studies showed that the water quality of Malacca Strait is deteriorating. In terms of microbiological pollution, there is only one study done by Hamzah et al. (2011) on fecal indicator bacteria concentration related with Port Dickson beach water quality. However, the study only focused on main four beaches with has limited samples and lacking in terms of exposure behavior and perceived health symptoms. Review done by Praveena at al. (2011) concluded that microbiological pollution datasets on continuous and systematic on fecal indicator bacteria have not been reported so far although insufficient sewage water treatment and wastewater pipe lines discharge directly to the sea are main serious problems facing by beach water of Port Dickson. Dada et al. (2012) stated that there is no single study has aimed at the occurrence of fecal indicator bacteria among recreational beaches and recognized health risks associated with the subject raises in Malaysia.

Looking into the context, this study aims to determine and detect spatial variation of fecal indicator bacterias (total coliform and Escherichia coli) in Teluk Kemang public beach, Port Dickson. This study was also intended to study the relationship between concentrations of fecal indicator bacterias and physicochemical parameters of beach water. This study was also designed to determine exposure behaviors and commonly reported perceived health symptoms among beach visitors resulted from exposure to beach water. His study is considered as the first study to gather information on fecal indicator bacterias, exposure behaviors and perceived health risks symptoms among beach visitors in Teluk Kemang, Port Dickson. Output of this study will act as a crucial public health model for protection of human for accurate risk assessment and remediation of beach water bodies with impaired water quality in Teluk Kemang. Moreover, output of this study can be used as a monitoring tool for other public beaches in Malaysia, as such studies are lacking and have not been reported so far.

3.2 Materials and Methods

3.2.1 Study Area

Port Dickson is located in Negeri Sembilan (Lat. 2°30.08′N; Long. 101°29.94′E) on the western coast of Peninsular Malaysia (Fig. 3.1). Beaches in Port Dickson extend up to 18 km from north Tanjung Gemuk to south Tanjung Tuan facing the Strait of Malacca. The Strait of Malacca is one of the world's busiest sea-lanes (Law et al. 2002). Teluk Kemang is the largest and most popular beach of Port Dickson, located between 7th and 8th mile from Port Dickson town. This beach stretch has the most number of hotels, resorts and apartments providing family accommodation. On weekends and public holidays, this beach gets crowded with visitors, while the sea becomes active with water activities and motorized water sports. Port Dickson enjoys a tropical climate and receives an average annual rainfall of 2,381 mm. The annual temperature ranges from 21 to 32 °C and the humidity varies between 80 and 90 %. Generally, the coastal waters of Port Dickson are shallow at about 20 m and are well mixed (Law et al. 2002).

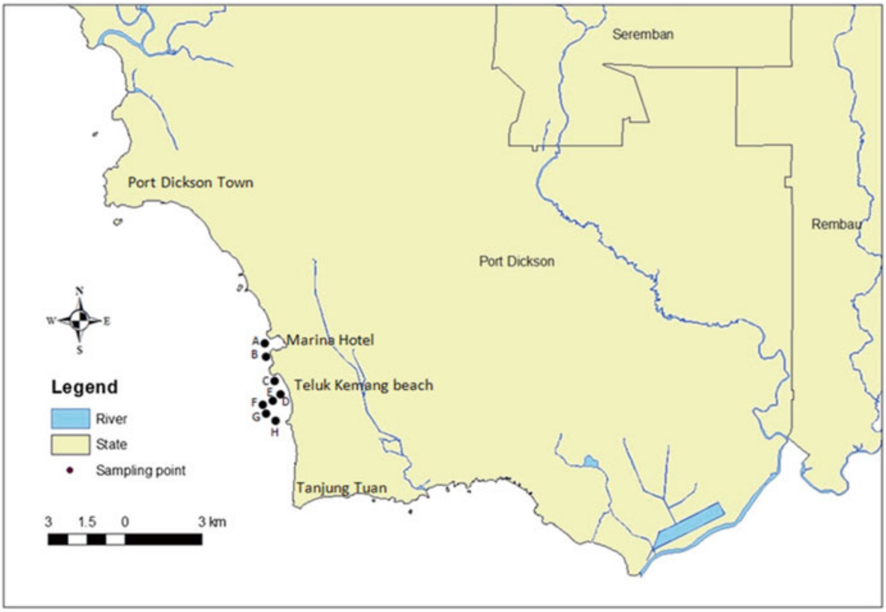


Fig. 3.1 Sampling points in this study

3.2.2 Sample Collection and Analysis

This study covers Teluk Kemang beach, an area of 1.8 km facing Strait of Malacca. Water sampling was conducted in March 2013 during school holidays to represent the busiest condition of Teluk Kemang beach. A total of eight sampling points were selected randomly with 200–300 m distance between each station where bathers have their recreational water activities (Fig. 3.1). A total of 100 ml water samples were taken triplicates using Schott bottles at 100 m from wash zone of the beach (zone that is in contact with the outer fringes of the water between low and high tide), and 20 cm below the water surface (Hamzah et al. 2011). This was to obtain surface water representative of that in contact with the bathers. After the sample collection, the water samples were placed immediately into the insulated ice box with ice pack and maintained under the temperature of 1–4 °C during the transit to the laboratory (APHA 1995). The samples were brought back to the laboratory on that day itself and analyzed within the 12 h of collection. In-situ parameters, namely pH, temperature, turbidity, salinity, dissolved oxygen (DO) and oxidation-reduction potential (ORP) were measured using YSI model 32 and Salinity Refractometer MASTER-S/MillM, respectively. The readings were taken three times for each parameter at each sampling point.

Total coliforms and E. coli were analyzed by membrane filtration method using a simultaneous detection technique with m-ColiBlue24® broth as described by Grant (1997). A total of 100 ml of water sample was filtered through 0.45-µm pore size cellulose nitrate membrane filter that retained the bacteria present in the water sample. The filter was then placed on an absorbent pad saturated with 2 ml of m-ColiBlue24® broth in Petri dish. The plate was incubated at 35 ± 0.5 °C in an incubator for up to 24 h. The bacterial colonies that grow on the plate were inspected for the presence of red and blue color. The red and blue colonies indicated the total coliforms; while the blue colonies were formed specifically from the breakdown of 5-bromo-4-chloro-3-indolyl-b-D-glucoronide (BCIG) by the E. coli enzyme β-glucoronidase.

Quality control and assurance were applied on each sample to provide greater data confidence regarding bias and variability. All laboratory equipments were pre-sterilized by autoclave at 121 °C for 15 min before use to minimize possibility of contamination, all equipments. Equipment blank which consists of distilled water was used to test for bias from possible contamination. This is done to verify that decontamination procedures and laboratory protocols are adequate. Triplicate samples were analyzed to estimate the variability resulting from sampling and analytical procedures (Koterba et al. 1995). The mean and standard deviation values were used to judge the accuracy and precision of each parameter measured. In addition, analytical errors of beach water analyses are shown in standard deviation values.

3.2.3 Questionnaire Survey

Beach visitors to Teluk Kemang beach were approached to participate questionnaire survey to determine exposure behaviors and perceived health symptoms of beach visitors. A total respondent of 117 was calculated based on formula by Daniel (1999). These respondents were interviewed individually to fulfill the questionnaire. Questionnaire was pretested before the actual administration of the questionnaire on the respondents to ensure comprehensibility and precision of the questionnaire. Cronbach's alpha value of 0.71 was obtained to ensure consistency of the questionnaire.

3.2.4 Statistical Analysis

The statistical analysis was performed using IBM SPSS (Statistical Package for Social Science) software Version 21. Descriptive statistics were used to detect spatial variation of fecal indicator bacterias (total coliform and Escherichia coli) in Teluk Kemang public beach, Port Dickson. Correlation analysis was undertaken to study relationship between concentrations of fecal indicator bacterias and physicochemical parameters of beach water.

3.3 Results and Discussion

3.3.1 Total Coliform and E. coli Concentration

Figure 3.2 shows mean concentrations of total coliforms and E. coli at eight sampling locations in Teluk Kemang public beach, Port Dickson. Total coliforms concentrations were found highest at sampling point A (1,940 ± 191 cfu/100 ml) and lowest at sampling point H (20 ± 20 cfu/100 ml). Sampling point F has the second highest number of colony followed by sampling point D. E. coli concentrations were found highest also at sampling point A (90 ± 30 cfu/100 ml) and lowest at sampling point G and H where E. coli was not detected. The second highest number of E. coli colony was found at sampling point D and E.

The highest concentrations of total coliforms and E. coli were found at sampling point A as it is located at the sewage outlet near Marina Hotel. Study by Law and Othman (1990) showed that the sampling point that received wastewater discharged from town has the significantly highest fecal coliforms counts. Thus, bacteriological water quality in Teluk Kemang public beach is still affected by sewage discharge from the surrounding area. Sampling point B was situated at the coastal inlet of Teluk Kemang where waterway connection between the sea and a bay, a lagoon, or a river entrance through which tidal and other currents flow (Hughes and Kraus 2006). Total

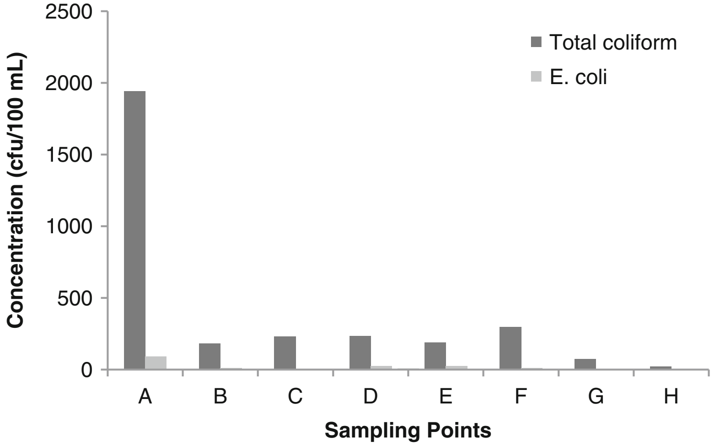


Fig. 3.2 Total coliforms and *E. coli* at eight sampling locations

coliforms and E. coli concentrations at sampling point B reduced dramatically. This may be due to the dilution of the coliforms by tide or water flowing into the sea (Medema et al. 2003). Sampling points C to H were situated at the bathing zones of the Teluk Kemang public beach. There was a decreasing trend of total coliforms from sampling point C to H (except for point F) as the distance from point A increased. This is probably due to dilution of bacterial density as the sewage outfall distance increase. Point F showed an exceptionally high number of total coliforms. This might due to high numbers of bathers in the study area which contributed to the high total coliforms. E. coli concentrations were equally less in numbers at all sampling points of the bathing zones in Teluk Kemang with no more than 10 cfu/100 ml. However, there was a double fold of E. coli at sampling point D and E. This phenomenon can be explained by the large number of boating activities in the vicinity of the two sampling points. The presence of E. coli may not be caused by recent fecal contamination of water as recreational boating activities might have resuspended bottom sediments with bound E. coli, contributing to the pollution of the water (An et al. 2002). Stephenson and Rychert (1982) found that E. coli concentrations of bottom sediments were many times greater than the overlying water. Besides, suspended matter due to increase of turbidity protects the bacteria from a bactericidal sunlight effect by light scattering, bringing about lower rate of bacterial die-off (Alkan et al. 1995; Pommepuy et al. 1992), thereby increase the total coliforms and E. coli concentrations.

Currently, Malaysia does not have total coliforms and fecal coliforms standard guidelines for bathing water. The limit for total coliforms recommended by USEPA (1986) at recreational marine water is 1,000/100 ml. In this study, total coliforms concentrations at sampling point A near the sewage outfall area had exceeded the USEPA limit by almost double while other sampling points were safely under the limit. For E. coli concentration, Malaysian Marine Water Quality Criteria and Standards (DOE 2011) set the limit at 100 MPN/100 ml while USEPA (1986) recommended 200 cfu/100 ml for recreational purposes. Based on Malaysian Marine Water Quality Criteria and Standards, none of these sampling points have exceeded the standard value. Therefore, all the sampling points were safe for public recreational activities such as swimming except for point A.

3.3.2 Physicochemical Parameters Measurements

Table 3.1 shows the measurement values of physicochemical parameters for each sampling points. pH measurement values were in the range of pH 8.10–8.14. There was an increasing trend from sampling point A to H. The maximum value was recorded at sampling point G and H while the minimum value was recorded at sampling point A. pH values from sampling point B to F were constant at pH 8.13. For temperature, the values were recorded in the range of 30.2–30.4 °C. There was a decreasing trend from sampling point A to H. The highest temperature was recorded in sampling point A while the lowest was recorded at sampling point H. DO measurements were recorded in the range of 5.20–5.57 mg/l. There were fluctuations on the DO values across the sampling points. The maximum value was recorded at sampling point F while the minimum values were recorded at sampling point A. Salinity was measured in the range of 30–31.67 ppt. The highest salinity was recorded at sampling point A and B while the lowest was recorded at sampling point E and F. There were also slight fluctuations of salinity across the sampling points. ORP was recorded in the range of 280.6–342.8 mV. The highest value was recorded at sampling point D while the lowest at sampling point H. Turbidity was recorded in the range of 2–2.3 m. The highest turbidity was recorded at sampling point A while the lowest were at sampling point D.

3.3.3 Relationship Between Concentrations of Fecal Indicator Bacterias and Physicochemical Parameters

Shapiro-wilk test of normality showed total coliforms and E. coli concentrations were not normally distributed ($p < 0.05$). Therefore, Spearman's rho correlation coefficient test was performed to assess relationships between the concentrations of total coliforms, E. coli, and physicochemical parameters (Table 3.2).

Physicochemical parameters of the water at each sampling points can affect the survival or growth of total coliforms and E. coli in the marine water. Both concentrations of total coliforms and E. coli were negatively correlated with pH. Study by Pearson et al. (1987) found that fecal coliform numbers were lowest when pH was high. According to Carlucci and Pramer (1959), the death of E. coli was more rapid at alkaline than at acid or neutral reactions. However, pH in seawater does not

Table 3.1 Physicochemical parameters measurements at eight sampling points

	Mean ± standard deviation							
Parameters	A	B	C	D	E	F	G	H
pH	8.10 ± 0.02	8.13 ± 0.01	8.13 ± 0.00	8.13 ± 0.00	8.13 ± 0.00	8.13 ± 0.00	8.14 ± 0.00	8.14 ± 0.00
Temperature (°C)	30.40 ± 0.00	30.40 ± 0.00	30.30 ± 0.00	30.30 ± 0.00	30.27 ± 0.06	30.27 ± 0.06	30.27 ± 0.06	30.20 ± 0.00
DO (mg/l)	5.20 ± 0.08	5.47 ± 0.01	5.55 ± 0.05	5.46 ± 0.03	5.42 ± 0.01	5.57 ± 0.10	5.50 ± 0.05	5.54 ± 0.08
Salinity (ppt)	31.67 ± 0.58	31.67 ± 1.15	31.30 ± 0.58	31.33 ± 0.58	30.00 ± 0.00	30.00 ± 0.00	30.33 ± 0.58	30.33 ± 0.58
ORP (mV)	304.03 ± 27.25	313.57 ± 17.25	319.57 ± 32.78	342.83 ± 43.69	323.23 ± 42.54	320.33 ± 24.32	282.77 ± 0.75	280.60 ± 1.87
Turbidity (m)	2.33 ± 0.29	2.10 ± 0.10	2.13 ± 0.06	2.00 ± 0.00	2.17 ± 0.06	2.23 ± 0.06	2.23 ± 0.06	2.30 ± 0.00

DO dissolved oxygen, *ORP* oxidation-reduction potential

Table 3.2 Spearman's rho correlation coefficient between total coliforms, *E. coli* and physicochemical parameters

Variables	pH	Temp.	DO	Salinity	ORP	Turbidity	TC	EC
pH	1.00	–0.729**	0.525**	–0.530**	–0.548**	0.262	**–0.813****	**–0.795****
Temp.		1.00	–0.363	0.634**	0.358	–0.233	**0.524****	**0.450***
DO			1.00	–0.242	–0.060	0.079	–0.273	**–0.610****
Salinity				1.00	0.227	–0.270	0.376	0.358
ORP					1.00	–0.607*	**0.504***	**0.536****
Turbidity						1.00	–0.229	–0.208
TC							1.00	**0.624****
EC								1.00

DO dissolved oxygen, *ORP* oxidation-reduction potential, *TC* total coliforms, *EC E. coli*
*Significant at $p<0.05$; **Significant at $p<0.01$
Coefficients significant at the 0.05 and 0.01 level are indicated by bold text

contribute much to the survival of fecal coliform as they are almost constant in seawater (Solic and Krstulovic 1992).

Temperature was significantly positively correlated with the number of total coliforms and E. coli. The number of coliforms found increased with temperature. This is contrary to Solic and Krstulovic's (1992) report that survival rate of bacteria decreases with increase in temperature. However, for bacterial growth, the optimal temperature is 37 °C equivalent to that of body temperature (Nguyen 2006). Gothier et al. (2001) reported for E. coli growth, the optimal temperature was between 40.2 and 41.2 °C. Thus, this might be the reason of the positive correlation in the specific range of temperature of tropical waters.

Besides, total coliforms and E. coli concentrations were positively correlated with oxidation reduction potential. This result was contradictory as it had been shown that high oxidation reduction potential is associated with inactivating bacteria such as E. coli (Kim et al. 2000; Liao et al. 2007). However, as the sampling points were situated near beach area, E. coli might have been protected by the resuspended bottom sediments from the effects of oxidation reduction potential after discharge from the sewage outfall (WHO 2004). Increased water turbulence caused by the boating activities might have both increased the oxygen level and also resuspended the bottom sediments in the seawater. This might have caused the positive correlation between both variables. In addition, oxidation reduction potential and bactericidal activity of water decrease with increasing pH (James et al. 2004; Park et al. 2005).

E. coli was found negatively correlated with dissolved oxygen might cause by consumption of dissolved oxygen resulting from microbial decomposition of organic compounds such as that discharged from wastewater treatment facility (Pope 1995). Dissolve oxygen deficiencies were commonly found as the result of discharge of oxygen demanding material from municipal wastewater treatment facility in excess of the assimilation capacity of water bodies. Bacteria use oxygen as energy to break down organic molecules into simpler, more stable end-products such as carbon dioxide (Todar 2009). High concentrations of oxygen can cause oxidative stress on E. coli cells which resulted in reduced or inhibited growth and cause the inverse relationship (Baez and Shiloach 2013). High concentration of dissolved oxygen in seawater could also enhance microbial inactivation by solar ultraviolet radiation (Hughes 2003), thereby reducing the number of bacteria.

3.3.4 Exposure Behaviors and Perceived Health Symptoms Among Beach Visitors

Table 3.3 shows the result for exposure behaviors among the respondents in Teluk Kemang public beach namely frequency of visit, water activities, ways of exposure, duration of exposure, and accidental ingestion of seawater.

Table 3.3 Exposure behaviors of respondents

Exposure behaviors	N = 108	Percentage (%)
Frequency of visit		
≥1 every week	33	30.6
≥1 every month	19	17.6
≥1 every 6 months	34	31.5
≥1 every year	22	20.4
Water activities		
Sea bathing/swimming	62	57.4
Nautical activities	24	22.2
Walking at the seawater edge	22	20.4
Way of exposure		
Whole body exposure including head	68	63
Whole body exposure not including head	7	6.5
Exposure to waist level only	25	23.1
Exposure to leg level only	8	7.4
Duration of exposure		
30 min	30	27.8
1 h	25	23.1
1 h 30 min	4	3.7
2 h	13	12.0
2 h 30 min and above	36	33.3
Accidental ingestion of seawater		
Yes	65	60.2
No	43	39.8

Most respondents were found to visit the beach either at least once every 6 months or every week. Those who reported visiting every 6 months were largely come from other surrounding nearby states such as Seremban and Kuala Lumpur, while the weekly visitors were constituted of locals as it is convenient for them, residing near to the beach. As expected, most beach visitors' activities involved swimming or sea bathing as it was a public beach and required no charges. Small number of respondents was also involved in the boating activities of which many boat renting business could be found on the beach. Some of the respondents reported only walking at the seawater edge. For the extent of exposure, consistent with the most popular activity i.e. sea bathing, most respondents reported having whole body exposure including head during activities on the water. About 6.5 % of the respondents reported having whole body exposure not including head during water activities. It was noted that the whole body exposure may not merely include the activity of sea bathing. It might also include other activities such as jet-ski and banana boat where the water splashes on the body of the visitors. From the logic point of view, sea bathing activity was prone to the accidental ingestion of sea water as it involves the whole immersion of head into the water. Other activities such as banana boat might also cause accidental ingestion as the boat capsizes suddenly during the course of activity. Besides, for the duration of exposure, most respondents spent from 30 min to 2 h and 30 min during water activities. In this

Table 3.4 Perceived health symptoms among respondents

	Yes		No	
Health symptoms	N	Percentage (%)	N	Percentage (%)
Gastroenteritis	2	1.9	106	98.1
Respiratory symptoms	15	13.9	93	86.1
Skin symptoms	22	20.4	86	79.6
Eye symptoms	21	19.4	87	80.6
Ear symptoms	4	3.7	104	96.3
Open wound infection	6	5.6	102	94.4
Presence of symptoms	55	47	62	53

case, there might be confusion in the question that causes the inconsistent answers. The duration of exposure was one of the factors determining the risk of health symptoms (Mugglestone et al. 2000). Naturally, the longer the duration of exposure, the higher the risk of getting water related infections. Significant correlation was found between gastrointestinal symptoms and ingestion of water during sea bathing (Pike 1994; Prieto et al. 2001).

Beach visitors were asked of any health symptoms from the exposure to the bathing water in Teluk Kemang public beach in the questionnaire interview based on their own perceptions and past experiences. Table 3.4 shows results of perceived health symptoms among the respondents. Results showed that there were not many respondents complaining of having symptoms related to exposure of bathing water at the Teluk Kemang public beach. Over half of the respondents reported having no symptoms at all.

Skin symptoms and eyes symptoms, although in small number, were highest in report. Study by Fleisher et al. (2010) showed that there was correlation between skin symptoms and increased exposure from non-point source microbial contaminants. However, for eyes symptoms, WHO (2003) reported that evidence has been shown that swimming reduces eyes' immune defense regardless of water microbial quality. Skin symptoms in recreational water might also be caused by over exposure to the ultraviolet radiation from sunlight (WHO 2003) especially in tropical country like Malaysia. Both skin and eyes symptoms might be caused by the high pH value found in the seawater of Teluk Kemang (WHO 2003). Other symptoms such as ear symptoms and open wound infections fared an insignificant number of reports. Gastroenteritis which is the most prominent symptoms in relation to indicator bacteria count in fecally contaminated recreational water (Pruss 1998) was reported at the lowest number in this study. This is corresponding to the low number of total coliforms and E. coli found in the water. According to USEPA (1986), the recommended limit of E. coli set at 200 CFU/100 ml would cause approximately 19 illnesses per 1,000 swimmers at recreational marine beaches. In addition, the sample size might be too small in this study to demonstrate the correlation between the concentration of indicator bacteria and health symptoms. Another reason may be the lack of awareness of the relationship between health symptoms and exposure to polluted bathing water among the public.

3.3.5 *Study Limitation*

There are few limitations in this study. Tidal condition of the beach was not taken into account during water sampling. Questionnaire interview was based on the previous experiences of the respondents on the perceived health symptoms resulted from exposure to the public beach bathing water. Thus, respondents might present with recall bias in recalling the time, exposure and symptoms experienced by them. Direct cause-effect relationship could not be established between the concentration of indicator bacteria and the perceived health symptoms because the health symptoms enquired was not caused by exposure to the water at the same time as water sampling.

3.4 Conclusion

To our knowledge, this is the first study to assess and report fecal indicator bacterias microbial associated with exposure behaviors and commonly reported perceived health symptoms among beach visitors in public beach, Malaysia. All the sampling points at the Teluk Kemang public beach were within the bacteriological standard for recreational water except for total coliforms at sampling point A which exceed the 1,000/100 ml limit by USEPA (1986). Concentrations of total coliforms and E. coli were highest at sampling point A compared to other points as it was near the sewage outfall. Concentrations of total coliforms and E. coli in the beach water were significantly influenced by various environmental factors such as pH, temperature, dissolved oxygen, and oxidation reduction potential. For the exposure behaviors, most beach visitors were involved in sea bathing or swimming activities which is prone to accidental seawater ingestion. There was no significant number of complaints on perceived health symptoms among the respondents from exposure to the bathing water at Teluk Kemang public beach. The microbiological water quality in Teluk Kemang public beach was generally safe for recreational activities for public. Further monitoring program on public beach quality in Teluk Kemang is necessary for better understanding of microbiological fluctuations and environmental conditions which will affect its concentrations. Moreover, further research need to focus on the actual health symptoms from exposure to the bathing beach by increasing the sample size and taking into account of the various confounders such as food intake. Besides, to establish direct cause effect relationship, bathers can be asked of health symptoms after about 1 week after sampling or bathers' exposures. Output of this study will act as a crucial model for protection of human health in public beach water in Malaysia. Importantly, output of this study can be used as a monitoring tool for other public beaches in Malaysia, as such studies are lacking and have not been reported so far.

Acknowledgments The authors would like to thanks Miss Nurul Syazani Yuswir and Miss Raihanah Chokeli for their help during sampling. Sincere thanks to Science Officer (Madam Nur Hayati Bakar) and lab assistants of Microbiology Laboratory, Department of Medical Microbiology and Parasitology, Faculty of Medicine and Health Sciences for their help during sample analysis. Special thanks to Centre of Marine Science (COMAS), Universiti Putra Malaysia for their boat service and sampling assistances.

References

Abdelzaher AM, Wright ME, Ortega C, Solo-Gabriele HM, Miller G, Elmir S, Fleming LE (2010) Presence of pathogens and indicator microbes at a nonpoint source subtropical recreational marine beach. Appl Environ Microbiol 76:724–732

Abu Hena MK, Kusnan M, Sidik BJ, Omar H, Hashim H (2000) A study of leaf growth and productivity of Tropical Ellgrass Enhaslus Acroides Royle in Port Dickson, Negeri Sembilan, Malaysia. In: Shariff M, Yusoff FM, Gopinath N, Ibrahim HM, Mustapha RAN (eds) Towards sustainable management of the Straits of Malacca. Malacca Straits Research and Development Center (MASDEC), Serdang

Alkan U, Elliot DJ, Evison LM (1995) Survival of enteric bacteria in relation to simulated solar radiation and other environmental factors in marine waters. Water Resour 29:2071–2081

An YJ, Kampbell DH, Breidenbach GP (2002) Escherichia coli and total coliforms in water and sediments at lake marinas. Environ Pollut 120:771–778

APHA (1995) Standard methods for the examination of water and wastewater, 19th edn. American Water Works Association, Water Environment Federation, Washington, DC

Baez A, Shiloach J (2013) Escherichia coli avoids high dissolved oxygen stress by activation of SoxRS and manganese-superoxide dismutase. Microb Cell Fact 12:23–32

Byappanahalli MN, Przybyla-Kelly K, Shively DA, Whitman RL (2008) Environmental occurrence of the enterococcal surface protein (esp) gene is an unreliable indicator of human fecal contamination. Environ Sci Technol 42:8014–8020

Carlucci AF, Pramer D (1959) An evaluation of factors affecting the survival of Escherichia coli in sea water. Appl Microbiol 8:247–205

Chua TE, Ross SA, Yu H (1997) Aquatic resource and environmental studies of the Straits of Malacca: managing the straits through science and technology. In: Malacca straits environmental profile. Malacca Straits Research and Development Center (MASDEC), Serdang

Dada AC, Asmat A, Gires U, Heng LY, Deborah BO (2012) Bacteriological monitoring and sustainable management of beach water quality in Malaysia: problems and prospects. Glob J Health Sci 4:126–138

Daniel WW (1999) Biostatics: a foundation for analysis in health sciences, vol 7. Wiley, New York

Department of Environment (2006) Environmental quality report. Department of Environment, Kuala Lumpur

DOE (2011) Environmental quality report. Department of Environment, Kuala Lumpur

Dyble J, Bienfang P, Dusek E, Hitchcock G, Holalnd F, Laws E, Lerczak J, McGillicuddy DJ, Minnett P, Moore SK, O'Kelly C, Solo-Gabriele H, Wang JD (2008) Environmental controls, oceanography and population dynamics of pathogens and harmful algal blooms: connecting sources to human exposure. Environ Health 7(Suppl 2):S5. doi:10.1186/1476-069X-7-S2-S5

Fleisher JM, Fleming LE, Solo-Gabriele HM, Kish JK, Sinigalliano CD, Plano L et al (2010) The beaches study: health effects and exposures from non-point source microbial contaminants in subtropical recreational waters contaminated with domestic sewage. Int J Epidemiol 39:1291–1298

Gothier A, Guerin-Faublee V, Tilly B, Delignette-Muller ML (2001) Optimal growth temperature of O157 and non-O157 Escherichia coli strands. Lett Appl Microbiol 33:352–356

Grant MA (1997) A new membrane filtration medium for simultaneous detection and enumeration of Escherichia coli and total coliforms. Appl Environ Microbiol 63:3526–3530

Hamzah A, Kipli SH, Ismail SR, Una R, Sarmani S (2011) Microbiological study in coastal water of Port Dickson, Malaysia. Sains Malays 40:93–99

Hughes KA (2003) Influence of seasonal environmental variables on the distribution of presumptive fecal coliforms around an Antarctic research station. Appl Environ Microbiol 68:4884–4891

Hughes SA, Kraus NC (2006) Frequently asked questions about coastal inlets and U.S Army Corps of Engineers' Coastal Inlets Research Program (CIRP). US Army Corps of Engineers, Vicksburg

James CN, Copeland RC, Lytle DA (2004) Relationships between oxidation-reduction potential, oxidant, and pH in drinking water. AWWA water quality technology conference, San Antonio, 14–18 November 2004

Japar Sidik B, Arshad A, Hishamuddin O, Sahmsul Bahar A (1995) Halophila decipiens Ostenf: a new record of seagrass for Malaysia. Aquat Bot 52:151–154

Kadaruddin A (1997) Penggunaan dan pengurusan zon pinggir pantai Negeri Sembilan. Akademika 50:3–23

Kim C, Hung YC, Brackett RE (2000) Roles of oxidation reduction potential in electrolyzed oxidizing and chemically modified water for the inactivation of food-related pathogens. J Food Prot 63:19–24

Koterba MT, Wilde FD, Lapham WW (1995) Ground-water data-collection protocols and procedures for the National Water-Quality Assessment Program – collection and documentation of water-quality samples and related data. U.S. Geological Survey Open-File Report, Virginia

Law AT, Chu CH (1990) Advances in fisheries and marine science. Pertanian University Publication, Serdang

Law AT, Othman A (1990) Sewage pollution in the coastal waters of Port Dickson, Straits of Malacca. Partanika 13:375–380

Law AT, Abdul-Rashid MK, Tan CH, Yuen YS (2000) Malacca straits expedition cruise no. 1: nitrogen and phosphorus distribution from Langkawi to Johore. In: Proceedings of international conference on the straits of Malacca, Malacca, Malaysia, 19–22 April, p 13

Law AT, Kuan CY, Jan JK, Hong S (2002) Hydrocarbon and sewage pollution in the coastal waters of Port Dickson, Straits of Malacca. In: Yosuff FM, Ibrahim KM, Tan SG, Tai SY, Shariff M (eds) Tropical marine environment: charting strategies for the millennium. Malacca Straits Research and Development Center (MASDEC), Serdang

Law AT, Shazili NA, Yaacob R, Chark LH, Saadon MS, Shamsudin L (1991) Coastal oceanography studies in the Straits of Malacca and South China Sea. Pulau Pinang

Liao LB, Chen WM, Xiao XM (2007) The generation and inactivation mechanism of oxidation–reduction potential of electrolyzed oxidizing water. J Food Eng 78:1326–1332

Lee JN, Mohamed CAR (2009) Trace metal contents in the Porites corals of Peninsular Malaysia. Int J Environ Res 3:85–94

Medema GJ, Shaw S, Waite M, Snozzi M, Morreau A, Grabow W (2003) Catchment characterization and source water quality. In: Assessing microbial safety of drinking water: improving approaches and methods. World Health Organization, London

Mugglestone MA, Stutt ED, Rushton L (2000) A review of the health effects of bathing water. MRC Institute for Environment and Health, Leicester

Nevers MB, Whitman RL (2008) Coastal strategies to predict Escherichia coli concentrations for beaches along a 35 km strecth of southern Lake Michigan. Environ Sci Technol 42:4454–4460

Nevers MB, Whitman RL (2011) Efficacy of monitoring and empirical predictive modeling ar improving public health protection at Chicago beaches. Water Res 45:1659–1668

Nguyen MT (2006) The effect of temperature on the growth of the bacteria Escherichia coli DH5α. St Martin Univ Biol J 1:87–94

Park CM, Hung YC, Lin CS, Brackett RE (2005) Efficacy of electrolyzed water in inactivating Salmonella enteritidis and Listeria monocytogenes on shell eggs. J Food Prot 68:986–990

Pearson HW, Mara DD, Mills SW, Smallman DJ (1987) Physicochemical parameters influencing faecal bacteria survival in waste stabilization ponds. Water Sci Technol 19:145–152

Pike EB (1994) Health effects of sea bathing (phase III). Final report to the Department of the Environment. Water Research Centre Plc, Medmenham

Pommepuy M, Guillad JF, Dupray E, Derrien A, Le Guyader F, Cormier M (1992) Enteric bacteria survival factors. Wat Sci Technol 25:93–103

Pope LM (1995) Surface-water-quality assessment of the lower Kansas River basin, Kansas and Nebraska: dissolved oxygen and Escherichia coli in streams during low flow, July 1988 through July 1989. Geological survey water-resources investigations report 94-4077, USGS, Washington, DC

Praveena SM, Zaharin AZ (2013) A baseline study of tropical coastal water quality in Port Dickson, Strait of Malacca, Malaysia. Mar Pollut Bull 67:196–199

Praveena SM, Siraj SS, Suleiman AK, Aris AZ (2011) A brush up on water quality studies of Port Dickson. Malays Res J Environ Sci 5:841–849

Prieto MD, Lopez J, Revilla JA, Llorca J, Delgado-Rodriguez M (2001) Recreations in coastal waters: health risk associated with bathing in seawater. J Epidemiol Community Health 55:442–447

Pruss A (1998) Review of epidemiological studies on health effects from exposure to recreational water. Int J Epidemiol 27:1–9

Sampson RW, Swiatnicki SA, Mcdermott CM, Kleinheinz GT (2006) The effects of rainfall on Escherichia coli and total coliform levels at 15 lake superior recreational beaches. Water Resour Manag 20:151–159

Schwartz ML (2005) Encyclopedia of coastal science. Springer, Netherlands

Shuval H (2003) Estimating the global burden of thalassogenic diseases: human infectious diseases caused by wastewater pollution of the marine environment. J Water Health 1:53–64

Solic M, Krstulovic N (1992) Separate and combined effects of solar radiation, temperature, salinity, and pH on the survival of faecal coliforms in seawater. Mar Pollut Bull 24:411–416

Stephenson GR, Rychert RC (1982) Bottom sediment: a reservoir of Escherichia coli in Rangeland Streams. J Range Manag 35:119–123

Thanapalasingam V (2005) Pollution status of the Sungai Skudai River system through heavy metals. Master's thesis, Universiti Teknologi Malaysia, Johor, Malaysia

Todar K (2009) Todar's online textbook of bacteriology. University of Wisconsin–Madison, Madison

United States Environmental Protection Agency USEPA (1986) Report of Task Force on guide standard and protocol for testing microbiological water purifiers. USEPA, Washington, DC, pp 1–29

Wade TJ, Calderon RL, Sams E, Beach M, Brenner KP, Williams AH, Dufour AP (2006) Rapidly measured indicators of recreational water quality are predictive of swimming associated gastrointestinal illness. Environ Health Perspect 114:24–28

World Health Organization (WHO) (2003) Chapter 4: Faecal pollution and water quality. In: Guidelines for safe recreational water environment, vol 1, Coastal and fresh water. WHO, Geneva, pp 51–96

World Health Organization (WHO) (2004) Water treatment and pathogen control: process efficiency in achieving safe drinking water. IWA Publishing, London

Part II
Ocean Governance, Fisheries and Aquaculture: Advances in the Production of Marine Resources

Chapter 4
New Approaches in Coastal and Marine Management: Developing Frameworks of Ocean Services in Governance

Luz Paramio, Fátima Lopes Alves, and José António Cabral Vieira

Abstract The Ocean's role as a global system is increasingly being recognized. "Modern" and "post-modern" views of ocean uses, as a source of resources and space for economic development is now supplemented by the functions that the ocean provides as human life and well-being support. "Ocean services" are a current focus of discussion for policymakers aiming to address sustainability principles and perspectives in a more effective way. International initiatives acknowledge the need to value the benefits provided by nature and include them in new economic models. This chapter summarizes and reviews these new approaches to ocean services, highlighting the main challenges related to its incorporation in ocean governance frameworks.

4.1 Introduction

Current economic and social transition trends include recognition of the direct consequences of how humans use the environment and its resources. Societies acknowledge the need for better management models addressing sustainability principles and benefits derived from goods and services provided by nature. Environmental, economic and social values measures are the best way to retrieve the basic value of "Nature" as life support and welfare. The recognition of the degree to which human activities can affect and depend on the health and preservation of marine and coastal ecosystems and their goods and services is today an indisputable fact (Halpern et al. 2012).

L. Paramio (✉) • J.A.C. Vieira
CEEAplA – Centre of Applied Economics Studies of the Atlantic, University of Azores, RuaMãe de Deus, 9500-855 Ponta Delgada, Portugal
e-mail: lparamio@uac.pt

F.L. Alves
CESAM – Centre for Environmental and Marine Studies, Department of Environment and Planning, University of Aveiro, 3810-193 Aveiro, Portugal

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8,
DOI 10.1007/978-3-319-06305-8_4,

Failed policies and standards (Adger and Jordan 2009) have led to design new ways to cover complexity associated to natural systems, as the case of the ocean. The complexity, variety and dynamism of the ocean favor the concept of governance as an ideal approach, addressing its analysis and the formulation of proposals for its management (Suarez de Vivero et al. 2008). Development involves considering new forms of governance (Crowder et al. 2006) adapted to the new dynamics. These implies a new kind of relationship among political processes (politics), institutional structures (polity) and policy content (policy)[1] (Lange et al. 2013) and innovative ways of steering, by the state, market, and civil society (Juda and Hennessey 2001; Juda 2003; VanLeeuwen and Van Tatenhove 2010).

This chapter examines current ocean governance perceptions and status. Governance and sustainability principles need to work together as a condition to overcome the current environmental crisis that affects global economics as well as ecological and social systems. This chapter focuses particularly on the newest range of approaches, propelled by a paradigm shift in which the ocean is considered as a planetary system. The relation between ocean economy and sustainability need to be analyzed. New concepts, such as a Green and Blue Economy, aim to promote an effective ocean services governance framework.

4.2 Ocean Services: The New Chance for Ocean Governance

The term "ocean" includes coastal and marine domains, and is chosen over other terms such as sea, oceans, or marine environment. This designation implies a global system including the multitude of issues, scales and spheres that it covers. Obviously, the ocean plays a fundamental role in socioeconomic development. Utilization of the ocean has increased exponentially, on one hand due to the intensification of "traditional uses", such as fisheries, transport, and on the other hand through the diversification of "new uses", such as prospecting, offshore energy etc. (Vallega 2001). Consideration of the main categories of the use of the ocean, from coastal areas to the depths, is vital to develop national and trans-national economies. Nowadays, there is a new factor in the equation, supported by the approach of ecosystem services, the functions of the ocean system plays. Ocean services definition covers uses, resources and functions, including its related goods and services, in other words, all benefits that the oceans provides to humans and earth life. The main benefits provided by the ocean are (CBD 2012):

- Seafood; habitats; fuel wood; energy sources (oil and gas, wind, wave, tidal, thermal); wetland protection; and genetic resources.
- Regulation of weather and climate; protects coasts; detoxifies and traps sediments. Wetlands mitigate storm damage.

[1] VanLeeuwen and Van Tatenhove (2010) explore more about the dimensions of the new governance, Policy, Politics and Polity and its applicability to the marine environment.

- Cultural services, including recreational; educational; esthetic; and spiritual.
- Economic activity, including fisheries, seafood, maritime transportation, trade, fuel, and energy.
- Nutrient cycling and primary production

Mechanisms are needed to ensure the preservation of these functions. The richness of benefits provides by the ocean, regard the sustainability dimensions. The survival and nurturing of marine and coastal ecosystems and biodiversity is essential to the nutritional, spiritual, societal and economic well-being of many millions of people, even those who may not think that they have any strong reliance on the ocean (CBD 2012). Addressing the opportunities and problems associated with coastal and marine systems is strategically important for sustainable development and should be supported by integrated and coordinated management (Partidário 2010). Lacking governance able to synchronize the efforts of coastal and marine management and development, taking account the ecosystem services, the ocean system degradation will intensify.

The value of benefits that the ocean provides (Noone et al. 2012) to human society and to the Earth is moving from "immeasurable" to "measurable." Measuring methodologies has been a controversial discourse, lead new approaches to management the ocean into the natural capital context.

4.3 Ocean Services and Management

As discussed above, existing literature illustrates several ways of addressing ocean management affairs, from marine environment to maritime activities, to marine ecological functions. There is no one method to approach it. There is also clear evidence of a fundamental rethinking about ocean management (Juda 2003).

Table 4.1 lists the variety of ocean management approaches. These need to be placed into a context among other internal and external processes.

In this circumstance, it is appropriate to review and analyze the status of ocean services management, encouraged by the rise of new governance claims to address the ocean services. The benchmarking practices and capacities to approach the ocean can bring greater effectiveness and efficiency to its management and use; also they allow building a new dialogue (Commonwealth 2012). Moreover, complex systems, as the case of the ocean, addressed from governance involve explicit appreciations of complexity and uncertainty, likelihood of surprise and need for flexibility and adaptive capacity (Kemp et al. 2005). It is obvious that attention to management process and structures is necessary. However an emphasis on trust building, institutional development, and social learning takes the management into the realm of governance (Armitage et al. 2008).

At global, regional and national levels, the discussion is launched (CBD 2012; EC 2012; UNEP et al. 2012; UN 2012). How to address the set of values of goods

Table 4.1 Literary diversity of marine and coastal and management approaches

Designations	Approach	Definitions/scope
Marine resources management	Sector approach	Fisheries, oil energy, blue carbon
Integrate coastal areas Management (ICAM)/ Integrate Coastal Zone Management (ICZM)[a]	Spatial approach	A continuum pro-active and adaptive process of resource management toward a sustainable development of coastal areas (Humphrey et al. 2000; Beatley et al. 2002)
Marine/maritime spatial Planning (MSP)[b]	Spatial approach	Analyzing and allocating parts of three dimensional marine spaces to specific uses or non-use, to achieve ecological, economic and social objectives that are usually specified through a political process (Ehler and Douvere 2007)
Coastal and marine spatial Planning	Spatial approach	Encompasses of Coastal and marine management process and its integration
Ocean planning	Holistic approach	Adopted by US and Canada at regional level, is an opportunity for sectors to put aside their conflicts and work together to find solutions to create sustainable economies, vibrant communities and healthy oceans
Marine protected areas	Conservation approach	Any area of the intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment. Include, conservation zone, natural reserve area, non-take area among others (Gubbay 1995; Leslie 2005; Dudley 2008)
Marine ecosystem based management	Ecological and place based approach	Ecosystem-based management is an integrated approach to management that considers the entire ecosystem, including humans. The goal of ecosystem-based management is to maintain an ecosystem in a healthy, productive and resilient condition so that it can provide the services humans want and need. Ecosystem-based management differs from current approaches that usually focus on a single species, sector, activity or concern; it considers the cumulative impacts of different sectors". (McLeod et al. 2005; Arkema et al. 2006; Crowder and Norse 2008; Curtin and Prellezo 2010)
Ocean zoning	Spatial and cross-sector approach	Comprehensive management in order to regulate activities in the Ocean (Halpern et al. 2008)

[a]Similar management processes are called different in United States and Europe. This is the case for ICAM and ICZM, respectively

[b]Some approaches prefer the term maritime over marine. However an integrate vision is given by marine, that is more accepted by the scientific literature

and services provided by the ocean? And how integrate them into new governance arrangements? Identification, quantification and engagement of services provided by the ocean are a fundamental way toward an effective governance system.

4.4 Management and Governance Approaching the Ocean

Understanding the ocean system implies a multidisciplinary vision that involves synergies between environmental and human sciences. Moreover, science integration has been a fundamental stage in management process (McFadden 2007). Tools coming from the environmental research, sustainability science, ecosystem services and the socio-ecological component are some of the elements that contribute to better understanding our relations within the ocean.

Ocean governance has been improved through the methodologies and instruments development, allowing the passage from last discussed theory to practice, a more efficient and effective governance. As above mentioned, several management concepts emerge from the terrestrial planning, with a great potential for cross-learning between marine and terrestrial types of planning[2] (Tyldesley 2004; Ehler and Douvere 2007; Kidd and Ellis 2012). The main differences are related to the biophysical and administrative boundaries, the spatial and temporal scales, the dimensionality of the systems (2D compared to 3D), ecosystem structure, dynamics and inter-connectivity, uses, schemes and uncertainties. Nevertheless, in both systems dimensions of governance are more similar than they may appear (Portman 2008).

4.4.1 Reflections and Perceptions About Governance

In order to embrace ocean services governance, some reflections and central points are collected. Governance has assumed a prominent place in the political, scientific and technical discourse (de Alcántara 1998; Vallega 2001). The concept was originally derived from the economics concept of "governance", emerging as "the exercise of authority, control, administration, power of a government in a given sphere" (World Bank 1992). It has since been transported to the sphere of politics to reinforce the role of management in governance "as the way in which power is exercised in the management of social and economic resources (…) for the development, also implying the ability of governments to plan, formulate and implement policies and accomplish tasks" (Gonçalves 2005). The introduction of the objective of development and its implementation through management go beyond the realm of policy (Bihim 2004). It is also a wider use of the term of governance, where inclusive and

[2] The application of concepts processes and tools of territorial planning and management to the marine environment have resulted in effective mechanisms for the management the sea, as example "The Irish Sea Pilot Project".

co-responsible management contrasts with the traditional models, which were more hierarchical and imposing (Dasi 2008). New modes of governance are more flexible, less hierarchical and directed by specific situations, within which incorporation of a variety of actors occurs from different spheres: public administration, civil society, media, science and business (Martinelli 2003). Thus new rules should be set, in order to influence both the distribution of power and the mechanisms of regulation and control (Suarez de Vivero et al. 2008). This chapter chooses to focus on a contemporary vision (Kooiman 2003):

> Governance as a system composed of the structure, dynamics and standards, where the government, governance (as interactive) and governability[3] are complementary elements of this system.

In practice, each governance system has a set of rules and formal and informal procedures that make up an institutional framework in which the various actors should be involved. In this system, there should be a flow of information that connects its various components. The system will have so much governability as more predictable, transparent and legitimate for this board (IGCC 2002).

This chapter adheres to this conception of governance regarding the approach to the ocean. In the case that the scale of analysis is applied to the realm of the environment (Krahmann 2003), the governance interpretations discussed above are also complemented in a comprehensive framework by the implicit features addressed by three visions: "global governance" (Rosenau 1999, 2002), a "governance for sustainability" (Kimball 2001; Kemp and Parto 2005; Bosselmann et al. 2008) and a more recent "Governance of the Earth System" (Biermann 2007).

4.4.2 Demystifying Ocean Governance Complexity

In order to demystify ocean governance, it is appropriate to redefine the difference between governance and management. In many cases, there is a replacement of both terms, appearing to be justified only by the need to use 'governance', which emphasizes the importance of human strategies and actions regarding the ocean and the role of integrated political approaches (Vallega 2001). The main differences are as follows (Juda 1999; Juda and Hennessey 2001; Olsen et al. 2006):

- Governance is a more inclusive term that includes all the other mechanisms and institutions that contribute to changing human behavior. It considers longer-term trends and requirements with regard to natural resources, based on an assessment of institutions and a discussion of the values to be attained. It shows that the fundamental objectives and institutional processes are the basis for planning and decision making.
- Management is related to the actions, irrespective of the organizational system as a whole, while governance focuses on the organization of the system as a whole.

[3] Inside the issue it is also important difference two complementary terms, governance and governability. Kooiman (2003) describes governability concerns the conditions of governance, understood as a status.

Fig. 4.1 Governance dimensions applied to the ocean namely, the normative, institutional arrangements and policies (Paramio 2012)

Management is the process by which humans and resources are utilized to achieve a distinct objective within a known institutional structure. Finally, governance sets the stage within which management occurs.

The current visualization of the term "governance", especially regarding the ocean, considers a reassessment of social objectives and a review of structures and their behaviors:

> The governance of the ocean is seen as an integrated approach to values, functions, policies, laws and institutions of the ocean system allowing multi-perspective planning and resolution of various issues. (Olsen 1993)

The "government" consists of multiple entities and "governance" involves multiple scales (international, regional, national and local), indicating transversal spheres (economic activities, conservation issues and social implications) and focusing on the connection between different levels (Krahmann 2003).

New perspectives in the fields of ocean governance and sustainable development seek to meet current challenges resulting from the articulation between scales and sectors, and also from the management integration of uses, function and values (Fig. 4.1). A central challenge is to reconnect governance efforts to the changing preconditions for societal development as active stewards of the Earth System (Folke et al. 2005), including the ocean.

4.4.3 Address Sustainability: The Implicit Mission of Ocean Governance

The implementation of standardized policies and current uses of the ocean has compelled different countries to rethink their governance systems (Young et al. 2007). Ocean governance and sustainable development is intimately connected, seeking to achieve similar goals: the integration of existing uses and new economic uses; the recognition and inclusion of ocean services, protection against threats from human activities, and the introduction of the ocean into national development policies.

In 1998,[4] the Independent World Commission on the Oceans, ICWO, (Costanza et al. 1998; ICWO 1998) defined a set of principles to approach ocean governance. Despite the evolution of the concept, some points need to be integrated into the current framework:

- The recognition of the dependence on ecological, economic and social well-being from the sustainability of the oceans;
- An integrated ecological and economic governance of the oceans, which recognizes the value of natural capital and ecosystem services, as well as the large uncertainties inherent in ocean science and governance.
- The importance of the problem of mismatch of scale between ecosystems and institutions of human governance, and the limitations of existing property rights regimes in relation to issues of ocean governance.
- The Lisbon principles: Responsibility, scale-matching; precautionary principle; adaptive management; full allocation of costs; participation.

Pardo (1967) proposes an integrated and interdisciplinary approach in the development and management of the ocean encompassing resources, services and human activities. Pardo approaches implicit dimensions of ocean governance, namely:

- Ethical dimension, where the ocean should be regulated and managed on behalf of humanity as a whole, with special attention to the needs of the poor (equity);
- Economic dimension, where the ocean's resources need to be developed and sustainably managed, including the high seas (common heritage);
- Environmental dimension, where the ocean's resources should be preserved to be shared with future generations (sustainability);
- Peace and security dimension, where the ocean should be reserved for peaceful uses, in order to benefit humanity as a whole (human security).

This vision enhances the ecosystem and places the human in context regarding management and behavior. In this way, Pardo outlines the basis of the current Ecosystem Based Management (EBM) applied to the ocean. Mann-Borgese's approach (1998) connects the ocean governance issues to sustainability, indicating

[4] In 1998 in the context of the Oceans International Year and the EXPO-98 – "Oceans: a Patrimony for the Future, in Lisbon, was create an Independent World Commission on the Oceans (IWCO). The IWCO meet different experts in the area of the ocean and policy.

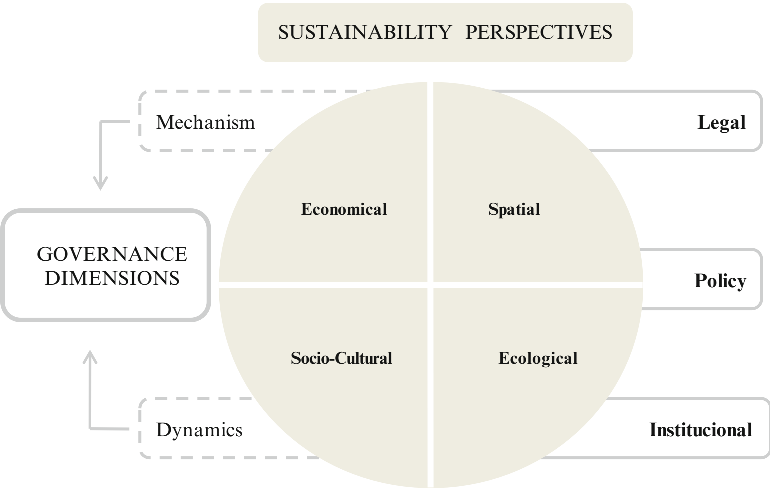


Fig. 4.2 Linkage approach to governance dimensions and sustainability perspectives regarding the ocean (Paramio 2012)

the magnitude of the ocean as a system through main five perspectives: physical, cultural, economic, legal and institutional. In the way to address the current challenges the link between governance and sustainability. This chapter proposes to approach sustainable ocean governance by taking into account four perspectives:

- *Social and cultural perspective*, understood as equity principles, values and a cultural component related to heritage and knowledge (Steinberg 2001);
- *Economic perspective*, in the sense of efficiency of the system that provides goods and services (Mann-Borgese 2000).
- *Ecological perspective*, which considers carrying capacity and resilience, and
- *Spatial perspective*, which underlines the importance of the geography related to specificities and explores the tools to establish priorities and allocate services.

In an attempt to understand the relations between ocean governance and sustainability, Fig. 4.2 shows the main components of an ocean governance system. As previously discussed, the ocean governance dimensions cut across legal, institutional, policy and external processes, and are related to mechanisms of implementation and dynamics. All these perspectives and principles should be taken into account in order to support new design policy frameworks.

It is clear that the economic perspective, more than ever, is the drive to development and action; however, the ocean framework also needs to consider sustainability principles and a balance between perspectives. This chapter names some tools that have been demonstrated as benchmarking in management, namely:

- Ecosystem-Based Management, as a governance tool that allow increasing proximity between ecologic perspectives, management and decision processes.

- Integrated Coastal Zone Management (ICZM), as a governance tool allowing the integration of the interface, ocean and land, regarding system particularities. The development in governance provided by the ICZM is given at the planning and participatory level, fuelled by the application of a multidisciplinary and holistic vision.
- Marine-Protected Areas, as a governance tool capable to extrapolate the ocean complexity into manageable dimensions, allowing a better perception of the challenges and opportunities.
- Marine Spatial Planning, as a governance tool related to the economic, spatial and ecological perspective. MSP allows for activities allocation and integration, simplifying the decision process and priorities establishment.

Maritime Cluster, understood as a platform to develop the economic and social perspective of the ocean. The engaging of the stakeholders into the governance process is the main goal of this tool.

Kullenberg (2010) suggests that the sustainability of the ocean can be organized through the implementation of an oriented-services economy. The inclusion of these services into the governance framework aims to answer the global challenge denoted by Folke et al. (2011) to adapt governance efforts to the changing preconditions for society as active stewardship of the earth.

4.4.4 Ocean Governance Framework Status

The development and evolution of ocean governance is sustained in a complex and comprised system of international and regional agreements. The “international regime” is a tapestry of conventions and instruments related to ocean affairs. The international framework is mainly supported by two independent but related frameworks (Frank 2007). One is supported by international law principles, in which the United Nations Convention of Law of the Sea (UNCLOS) is considered as the constitution of the sea and the sustainability framework, supported in the United Nation Conference Sustainable Development Agenda (1992) recently updated in 2012 at the conference RIO+20. The second is composed of complementary international agreements addressing specific activities, regions, habitats, etc.

The Law of the Sea (LOS), launched 30 years ago, and its subsequent marine environment agreements are insufficient to confront the new challenges and expectations regarding ocean development. Notwithstanding, LOS has the implicit idea of sustainability, on a planetary scale, and emphasizes the balance of interests not only among nations but also between earth systems. Even though the sustainability principles are present in this theory, concern about equity in access to the ocean and its resources arises. Other contemporary principles of governance are included in the LOS, such as ecosystem-based management, stakeholder engagement and resilience. In the governance framework, complementary international agreements address specific activities or regions. This international sphere of these agreements has an integrative and negotiation capacity, often being the catalyst for policy development initiatives, especially at regional levels.[5]

It is now recognized that the ocean of today is greatly changed from the one from three decades ago. The accompanying framework is comprehensive but remains poorly implemented. The governance frameworks need also to be reinforced in response to the current threats. The current system of governance and management is in need of urgent and significant reform (Friedheim 1999). More practical, cooperative and integrated arrangements are needed to enforce and implement new systems of governance and management at better spatial scales. Contemporary problems call for an improvement in this framework (Scovazzy 2012) and the need to understand better the relationships between the components of the system (rules, processes, institutions, actors). This process can only be fully achieved if it is driven in a strategic way from the hearts of governments across all sectors (Commonwealth 2012).

In recent years, in the international scenario, there have arisen some reactive responses regarding the ocean. The economic crisis and the continental shelf extension process are central drivers for increased investment in the knowledge of the ocean. The implications resulted also in increased responsibilities for the signatory countries, but at the same time provided the rights of exploration and use of the sea. Leading governments also assume legal and institutional responsibilities regarding constitutional matters that concern the management of maritime territory, deciding the rules of exploitation of natural resources, the protection of species and habitats, economic frameworks for royalties, etc. Development of a framework to protect and to regulate the services provided by the ocean should be a priority at international and national levels.

4.5 Economic Perspective of the Ocean: Opening Doors to Ecosystem Services

An economic paradigm arises about the ocean as a "common mankind heritage"[6] (Pardo 1967) and "open access resource" (Hardin 1968; Ostrom 1990) and generates international discussion (Bromley 2008; Block 2011). These features are particularly susceptible, when it is in mind the ocean as a global and dynamic system and witch in healthy conditions provides: resources, space and wellbeing.(Commonwealth 2012). As it was remarked, the potential of the ocean is vast, yet the basis for future growth of benefits from ocean goods and services (Costanza 1999) is being eroded or lost. Barriers to realizing greater opportunities and benefits need to be addressed (Lamb 2013). There is a need for urgent scientific analysis that supports the incorporation of valuation into governance frameworks (Daily et al. 2009) regarding the ocean (Remoundou et al. 2009).

[5] Some regional agreement, as example the OSPAR Conventions, among others, plays a fundamental role in policy development for the Atlantic.

[6] Later E. Mann-Borgese applied to the Ocean the Common heritage of humankind as governance philosophy.

Assigning economic values to the impacts of this decline can be very useful in overcoming economic-based objections to sustainable policies at all levels of government. There are very good approaches to economic values services provided by marine and coastal ecosystems, marketed and non-marketed, that should be used to justify their sustainable use and management (Mee et al. 2008; Douvere and Ehler 2009). Ecosystem-based management is seen as a solution to achieve the sustainability of the ocean. It allows for managing a diversity of human activities in a specific space. Some countries have started to introduce an ecosystem services framework, including the accounting stage, into their policies. The aim is to support the planning and management process through the identification and uses of different ecosystems that provide distinct services. The management instruments and documents, discussed above, need to embrace a detailed identification, analysis and assessment of existing ecosystems services (Partidário 2010) provide by the ocean. Harmonizing traditional economic activity and ecosystem-dependent economic values is a challenge that the governance must address (CBD 2012).

4.5.1 Ecosystem Services Provided by the Ocean

One of the main approaches to ecosystem services is provided by the Millennium Ecosystem Assessment, MEA (2003).It scope is understand and assess the interrelation between ecosystems and human well-being in the global scale. According to MEA the ecosystem goods and services from which people benefit can be grouped into four categories: provisioning, regulating, cultural and supporting services. MEA defines ecosystem services as “the benefits people obtain from ecosystems” (MEA 2005). Fisher and Turner (2008) expand on this definition to propose that “ecosystem services are the aspects of ecosystems utilized (actively or passively) to produce human well-being” (Bateman et al. 2011). MEA has fundamentally changed the approach that has characterized natural and social scientists, and has mapped the complex relationships between biodiversity, ecosystem functioning, and human welfare (Nunes and Ghermandi 2013). The ‘measurement’ of monetary values reflect the social importance of ecosystem services, and is seen as a prerequisite for better management decisions (Laurans et al. 2013). The spread of the ecosystem services framework has been accompanied by the promotion of market-based policy instruments for environmental governance (Muradian and Rival 2012).

In this context, the urgent need to mainstream the economic value of marine and coastal ecosystems and biodiversity seem to be increasingly firmly established on the agenda of high-level policy and decision-makers (Nunes and Ghermandi 2013). Despite controversy about “putting price to the nature”,[7] the approach must refine the simplistic idea that a lack of price-setting is equivalent to a lack of values (Nunes et al. 2009). The economic valuation of ecosystem goods and services is a challeng-

[7] Term used by Richard Conniff “What’s wrong with putting a price on nature?”http://www.theguardian.com/environment/2012/oct/18/what-wrong-price-on-nature.

ing task, but one where even approximate estimates of the values of ecosystems used in decision-making can lead to better resource management and policy, especially where the alternative assumption is that nature has zero (or infinite) value (Commonwealth 2012). The desire to create favorable economic valuations for things that we know to be ecologically important is understandable, but bypassing accepted scientific and economic principles in order to do so sets a dangerous precedent and risks many unintended consequences (Lamb 2013).

The sustained supply of goods and services from the ocean is central to human prosperity. This supply depends not just on the presence, but also the quantity and quality of marine biodiversity (UNEP et al. 2012). Biodiversity is defined as richness and composition among species and functions; that is, groups of species can carry out the same ecological functions (Beaumont et al. 2005). Marine biodiversity plays a fundamental role in supporting a wide range of goods and services essential for the maintenance of the social and economic well-being of our society. Notwithstanding the term, biodiversity has many different definitions (Sheppard 2006) but in the particular case of ecosystem services valuation; it is used to refer to richness and composition at species and functional levels. Goods and services are defined as indirect or direct benefits to human society which arise from the natural environment. Current environmental management tends to focus on market-linked goods and services, such as tourism and fisheries. It is this narrow approach that has contributed to the over exploitation and degradation of marine biodiversity. Only by understanding all the goods and services provided by marine biodiversity (Randall 1988) is it possible to appreciate the true value of this resource. This understanding is the key to developing sustainable management plans to maximize the benefits received from marine biodiversity (Beaumont et al. 2005). Marine biodiversity is a key element to support ocean services.

The economic valuations of biodiversity take a place on international policy agendas in order to understand the benefits of natural capital. The "blue capital," understood as the natural capital of the ocean, needs to be recognized accordingly.

4.6 Ocean Services: New Approaches, Different Governance Scales

In order to fill the existing gaps relative to governance and its implementation mechanism for the ocean, there is an emerging number of studies regarding the ocean economy. It is possible to highlight the approaches relative to maritime activities, zonal or conservation scopes. On the one hand, there are the socio-economic studies that approach ocean activities or sectors (fishing, maritime transport, etc.) and their importance to national economies in terms of GDP[8] (Pugh 2008; Kildow and McIlgorm 2010). In order to trace conditions created by legal frameworks, such as

[8] Gross Domestic Product.

UNCLOS, some studies analyze the economic value of the zonal approach, such as EEZ, regional approach, the Pacific Ocean (Seidel and Lal 2010) or the Baltic Sea (Sweden Agency 2008) among others. Conservation interests also have the need to justify their policies beyond environmental and social progress, resorting to economic benefits provide by the marine-protected areas (Alban et al. 2006) or valuation about impacts, species, habitats or environmental concerns, such as ocean acidification (Armstrong et al. 2012).

Despite the strong specificities, these approaches represent major advances in a global governance vision of the ocean. However, this chapter only examines ocean services from a governance and decision making perspective.

4.6.1 Global Scale: Rio+20 and the Green Economy

In 2012, the United Nations Conference of Sustainable Development (UNSD)-Rio+20 included more than 193 member states, 20 years after the first conference in 1992. Several ocean governance issues were discussed, such as new jurisdictions, the high seas, etc. However, special attention was given to the "green economy," that in the case of the ocean has evolved into the so-called "blue economy". Some changes in the "color concept" (Basorie 2012) are echoed in the conference context, from green at Rio+20 to blue at Yeosu, EXPO 2020. The theme of Rio+20 was "a green economy in the context of sustainable development and poverty alleviation".

A green economy is an economic development model based on sustainable development and knowledge of ecological economics.

The direct valuation of natural capital and ecosystem services, are actives that should be introduce in the global economy. Currently, the green economy initiatives focus the research of process to develop an accounting framework to convert the nature in an active. The challenge is look for ways to ocean economy develops a Green Economy policy, to strengthen its social, economic and environmental linkages and to reform current governance arrangements. These goals will require some fundamental changes in the way the ocean is managed at national, regional and global scales to create a more harmonized and integrated approach (Commonwealth 2012).

4.6.2 Toward the Blue Economy

The United Nations' project, "Nature's 100 Best," aimed to find sustainable solutions for economy and society that are inspired by the principals of nature. In this spawned the Blue Economy concept. This definition has origins in Gunter Pauli's book, *The Blue Economy: 10 years – 100 Innovations – 100 Million Jobs* (Pauli 2010). The book analyses the best nature-inspired technologies that could affect the economies of the world, while sustainably providing for basic human needs.[9] The

[9] See http://www.blueeconomy.eu/ and the summary of blue economy principles.

Table 4.2 Blue print report approach to blue-green economy

Blue-green economy key dimensions (IOC/UNESCO et al. 2011)
Protection and restoration of ocean ecosystems and biodiversity, including beyond national jurisdictions;
Development of blue carbon markets;
Active sea-floor management (including oil and gas, mining, and cables) both within and outside national jurisdictions;
Change in fisheries and aquaculture management regimes at regional and national levels toward, equitable, non-subsidized, and sustainable practices;
Adaptation to sea level rise and climate change;
Integrated coastal management;
Increasing sustainable use of bioresources, including biotechnology;
Recognition and adoption of ocean/coastal carbon sinks and the creation of a market for trading ('blue carbon')
Dramatically enhanced recycling of major ocean pollutants such as nutrients through market mechanisms
Greater adoption of renewable energy from the ocean (a move away from land based focus)

projects were evaluated by a team of business strategists, financial analysts and a public policy maker. The Blue Economy is a new economic model understood as a further development of the Green Economy. With appropriate policies to support research and development, and promotional strategies that accomplish their delivery through market mechanisms, such materials and methods offer abundant opportunities for addressing pressing global issues.

The Blue Economy is a term that also embraces maritime activities as well as other services, such as ecosystem services, provided by the ocean. In this sense, it adheres to the previously presented philosophies of a green economy and blue economy. Many global initiatives[10] have developed frameworks to address the ocean economy. Some of them aim to mitigate the continued degradation of the ocean, and to restore and sustain ecosystem services.

The UNESCO Blue print Report (IOC 2011) a review of ocean issues prepared for the RIO+20 Conference, includes an approach to the blue-green economy key dimensions, as seen in Table 4.2. The blue economy was the principal issue at World Expo 2012. The Expo 2012 topic "The Living Ocean and Coast: Diversity of Resources and Sustainable activities" defines a "blue economy" as the global sharing of green growth based on new science and technology applied to the ocean.

The ocean and its governance system need to be able to answer the new economic and social requirements. An integrate approach of the ocean through an ecosystem based management needs to embrace the oceans services. This represents an opportunity to create strategic frameworks that allow the establishment of priorities at national and regional level to achieve effective global ocean governance. The main changes in governance suggested in the report by the International

[10] See the initiatives: Center for American Progress, http://www.americanprogress.org/issues/green/report/2012/06/27/11794/the-foundations

Table 4.3 Governance objectives proposed by the blue print report

Blue print report governance objectives
Objective 1. Actions to reduce stressors and maintain or restore the structure and function of marine ecosystems for equitable and sustainable use of marine resources and ecosystems
Objective 2. Actions that support the Green Economy concept leading to alleviation of poverty and promotion of sustainable ocean sectors and livelihoods including actions to improve implementation at local levels through participatory processes
Objective 3. Actions resulting in Policy, Legal and Institutional Reforms for effective Ocean Governance, including in the High Seas, and strengthening the institutional framework, mandate and coordination of UN bodies with marine competencies
Objective 4. Actions supporting marine research, monitoring and evaluation, technology and capacity transfer as a mean for improving knowledge, addressing emerging issues, developing capacities in support of sustainable use of the ocean

Oceanographic Commission[11] are related to physical, behavioral and institutional objectives, as seen in Table 4.3.

Despite these proposals at the RIO+20 Conference, the results were a disappointment, in that more action at the decision making level was expected.

4.6.3 Regional Scale: European Blue Growth

Regional approaches reveal how governance mechanisms assume, often in the absence of legal instruments, a central role. The experience at the regional scale shows the importance of the components of governance of the ocean, namely, participation in the precautionary principles, adaptive management and ecosystem management. Regional Ocean Governance is seen (Rosenberg 2005) therefore as a new way of proactive governance due to its trans-jurisdictional nature of the uses, resources and problems.

The European Union is a prime example, through the development of marine policies and spatial planning. The EU aimed at a holistic view in the policy process, from formulation to implementation. What with the UNCLOS and its global and regional agreements, the European Union has, explicitly or implicitly, transferred part of its powers to the member states.

The European Ocean framework has a base in the Integrate Maritime Policy (IMP),[12] a pack of instruments that aims at a comprehensive policy and a more coherent and structured governance. IMP is based on a clear recognition that all issues related to oceans and seas are interlinked and that the policies should be

[11] For more information, see the objectives and proposal descriptions at the report of A Blueprint for Ocean and Coastal Sustainability, IOC/UNESCO et al. 2011.

[12] EC (2008), COM (2008) 395, Guidelines for an integrated approach to Maritime Policy: Towards best practice in integrated maritime governance and consultation.

Table 4.4 European strategic areas regarding the governance of the ocean

European IMP strategic areas
Integration of maritime governance, based in intersectoral collaboration and stakeholder consultation
Development of cross-cutting policy tools, as maritime spatial planning, marine knowledge and comprehensive data, and integrated maritime surveillance
Defining the limits of sustainability, within the Framework Directive "Marine Strategy", and trends to a green economy
Strategies for developing regional sea basin level allow the adjustment of priorities and political tools for the only geographic, economic and political development of each marine region
Development of the international dimension of the IMP, to strengthen the EU position in multilateral and bilateral relations
Renewed approach on sustainable economic growth

drawn in an articulated way. The marine affairs addressed by the IMP framework aim to enhance the responsiveness of Europe and its member states to the challenges of globalization and competitiveness, climate change, degradation of the marine environment, maritime safety and maritime security, and energy security and sustainability. The progress of IMP[13] follows developments in six strategic areas, Table 4.4.

In 2010, the European Commission launched "Strategy 2020"[14] a policy drive to cope with economic difficulties and renew a European vision. This strategy is characterized by the following growth priorities: Smart growth, aimed at developing a knowledge-based economy and innovation; a sustainable economy that promotes more efficient use of resources, greener and more competition and inclusion; and promoting high employment to ensure social and territorial cohesion. In 2011, the EU launched the Atlantic Strategy, pointing to the sea economy as a key component. The strategy includes five themes: implementation of the ecosystem approach; reduction of carbon footprints; seabed exploration; responding to threats and emergencies and social and inclusive growth. The implementation of this strategy presents a methodology for integrating initiatives and actions supported by the member states and stakeholders, the promotion of international cooperation and the adoption of a strategic plan, supported by a "smart governance" from the existing structures.

In October 2012, the "Limassol Declaration" (EC 2012) a Marine and Maritime Agenda for growth and jobs, echoed the strong maritime pillar supported by the Europe 2020 Strategy. Moreover, there was the proposal of the European Union Directive[15] to establish a framework for maritime spatial planning and integrated coastal management. It required (as a binding tool) members to establish maritime spatial plans and integrated coastal management strategies by 2020.

[13] EC (2009), COM (2009) 540 final, The state of development of the action plan.

[14] EC (2010), COM (2010) 2020 EUROPE 2020: A strategy for smart, sustainable and inclusive growth.

[15] EU(2013). COM (2013) 133 final, of 12.3.2013.

In 2013, the Galway Statement reinforced the transatlantic cooperation between Europe, Canada and the United States invoking the ocean governance principles applied to the Atlantic. The Declaration includes:

- A global vision of ocean systems that takes into account the relations between the Atlantic and Arctic, especially related to the governance developments.
- The recognition of ocean services for human health and well-being.
- The notion of sustainability, taking into account all perspectives with particular attention to a legacy for future generations.

The European Union's blue growth approach is based on the historical weight of certain economic activities in the European Union. Europe published blue growth reports that from an analytical approach try to identify the economic areas related to the ocean. Ocean economy functions include: Maritime transport and shipbuilding, food and nutrition, health and ecosystem services, energy and raw materials, leisure and recreation, coastal protection, and maritime monitoring and surveillance. Only recently, the European Union assumed the broad term of "Blue Economy" in the sense:

> Europe's Blue Economy can mean growth, employment opportunities and competitiveness. Through Blue Growth, Europe is unlocking the wealth that exists in the coastal and marine environment.[16]

In this way, the legal and financial instruments aim to support two fundamental tools: the Regional Strategies Basins and the local maritime clusters as local synergistic concentrations of higher education, research and industry.

4.6.4 Common Wealth and the "Natural Blue Capital"

The Commonwealth Secretariat prepared an important report titled "A new approach to the ocean governance: the practical ways to fast track the green economy" (Commonwealth 2012). The report sets out a practical approach to building governance and institutional developments to make the Green Economy a reality for the ocean. It is a guideline to support decision-making of Commonwealth Environment Ministers. This initiative is motivated by the wide extension of overseas territories of Commonwealth countries, in particular its small island states that are strongly dependent on ocean resources for their economic and social well-being. The Commonwealth aims to position itself as a global leader in ocean governance.

The interpretation of the concept of the Green Economy by the Commonwealth report is inextricably linked with environmental and social considerations. The report remarks that the linkages among economic sectors, human impacts, and all aspects of environmental health are stronger and more challenging to manage.

[16] Maria Damanaki, European Commissioner for Maritime Affairs and Fisheries, personal communication, European Maritime Malta 22 May 2013.

Table 4.5 Commonwealth framework to address a green economy for the ocean

Commonwealth practical actions toward green economy transition
Recognizing 'natural blue capital' and its potential for growth: Transforming our approach to biodiversity in the ocean from one seen to be based around habitats and species to one based around 'blue capital', where biodiversity has multiple values and can act as natural solutions to inform and answer wider challenges;
Applying marine spatial planning: reinvesting in a wider spatial perspective to management and governance of the ocean area by linking the ocean to coastal and inland issues in such a way that previous conflicts, barriers and costs are reduced or removed, and new multiple benefits and new growth are promoted
Transforming patterns of economic investment: altering the economic investment portfolio to remove perverse incentives that act as barriers to progress, recognize the financial value of natural blue capital, and reinvest the money released in new areas that achieve greater long-term growth and deliver multiple benefits within social, economic and environmental priorities
Creating new approaches to ocean governance: enhancing cooperation and communications to create more political will to remove and reduce barriers to implementation, making growing the Green Economy easier to achieve with ocean resources and more politically desirable to do
Growing the institutional and human capacity to act: devising new ways of working that lever greater capacity from current systems to make change happen through for example increased regional cooperation, sharing of costs and knowledge, public/private partnerships and creating 'fast-track' approaches to translating aid into action

The framework sets out five practical actions, listed in Table 4.5, will lead to a transformation in the governance of the ocean.

The framework draws on the work of UNEP, the World Bank and many others, but sets a new context of ocean governance reform. This framework values the goods and services offered by the ocean, set actions toward a more spatially beneficial context, and re-deploys incentives to create investment in the productive capacity of oceans.

4.7 Walking to Accounting Ocean Services

Natural capital includes all resources (forest, mineral, fisheries, etc.) and ecosystems that produce services that are often 'invisible'. These values are not readily captured in markets, so we don't really know how much they contribute to economy and prosperity. Humans often take these services for granted and do not know what it would cost if they were lost.

Despite inherent problems related to the measurement of natural capital (Boyd and Banzhaf 2007) the promotion and use of 'green markets' have expanded recently, as a policy response to the ecological crisis (Muradian and Rival 2012). Without going too deeply into the nature of assessment methodologies, this chapter aims to provide a discussion from a governance perspective: Is it possible to link economic maritime activities and marine ecosystem services through the ocean services definition? Which kind of accounting is applicable to ocean services? Are these the panacea to securing ocean sustainability?

The importance and urgency to mainstream the economic value of marine and coastal ecosystems and biodiversity are today firmly established on the agendas of high-level policy and decision-makers (Nunes and Ghermandi 2013). One of the most dynamic initiatives was commenced by The Economics of Ecosystems and Biodiversity (TEEB). The TEEB's studies interpret the natural capital, such as the range of species, genes and ecosystems and its critical components, and aims to identify the scale and benefits derived. The TEEB focused attention on the economic benefits of biodiversity and ecosystems as well as the costs of biodiversity loss and ecosystem degradation.

In particular, the TEEB's Oceans and Coasts initiative (2012) seeks to draw attention to the economic benefits of ocean biodiversity and healthy ecosystems and emphasizes the unrealized benefits of preserved and enhanced whole ecosystem structures, functions and processes to humans and nature. The initiative aims to extend the TEEB studies around the world.

Through the WAVES Project (Wealth Accounting and the Valuation of Ecosystem Services) the World Bank is leading the efforts to incorporate natural capital – including that of coastal and marine resources – into the accounts of its implementing partner nations. The Marine Ecosystem Services Partnership hosts and updates an extensive library of valuations of coastal and marine ecosystem services. Strong attention is paid to the economic aspects of marine and coastal development and conservation in the work of the United Nations World Ocean Assessment. All these are effective tools to support governance and policies regarding sustainable management of marine natural resources (Nunes and Ghermandi 2013).

4.8 Opportunities and Barriers for Oceans Services Governance

These numerous initiatives show the need to incorporate the opportunities afforded by ocean services into an ocean governance framework. Since planning to management and assessment process the recognition of oceans services is appointed as a key to achieve the sustainability. The current international ocean governance framework has new modes and new configurations in which international and regional agreements are strongly conditioned by economic drivers. The dimension of goods and services that the ocean provides are only partially known and not fully recognized.

Ocean services appear as a catalyst for a new dynamic in global ocean governance. The appreciation of ocean services as a tool for governance arises following the international discussion about the earth's capacity and boundaries. How to address global sustainability models in our developed world is a key issue. A green economy is now recognized at the global level as a solution, but what is the role of the ocean? How do we address the governance challenges in the ocean system? Are ocean services able to integrate human activities and marine ecosystem functions within the new economic transition?

The need to study the ocean system further is a prerequisite for effective management. This implies a great investment in technology and logistic resources as well as in knowledge transfer and communications networks. In the last decade, global efforts have allowed us to extend our knowledge of the sea accompanied by strong economic investment by countries. Now, economic and social crises drive a strong commitment to research on sustainability mechanisms that allows for investment in the uses of the ocean and its ecosystems.

They are many points to consider regarding a new economic model for the ocean. Different perspectives about oceanic economic development and policy support constitute a "wicked" problem (Jentoft and Chuenpagdee 2009). The chapter trusts in governance as being able to engage in these challenges. However, ethical paradigms need to be discussed in the realm of contemporary ocean governance needs.

Governments must begin a transition towards a Green Economy, creating a viable socio-economic framework that generates jobs, assists in poverty alleviation, adapts to and mitigates climate change and other existing and emerging challenges, and embraces integrated environmental management (UNEP et al. 2012). Economic valuations play an informative role for general influence and awareness-raising about more and better conservation. Later reaffirmed by the release of The Economics of Ecosystems and Biodiversity (TEEB) report (TEEB 2010), the Tenth Conference of the Parties (COP) to the Convention on Biological Diversity in Nagoya in 2010 concluded that economic valuation is expected to serve as a governance resource that could change our individual and collective choices (Laurans and Rankovic et al. 2013).

At national, regional and global levels, a new dialogue about the value of oceans goods and services has been launched, along with visions for new governance arrangements, practices and capacities that can bring greater effectiveness and efficiency to their management and use. Other important issues are linked to global governance dynamics, such as the Earth Condominium Project by the UN that represents an international initiative aimed at "building the intangible natural heritage mankind principle" (Barreira 2012). This initiative is supported by the Common Heritage of Mankind principles, where the ocean should have a prominent place.

Acknowledgments The authors gratefully acknowledge the final revision from Teresa Tiago and Maria da Luz Fernandes. The financial support was provided by the Azores Science Fund, FRC, to the project M3.1.7/F/016/2011, *SEAValue Project: the value of services provides by the Sea of Azores. Contributions to Regional Development.*

References

Adger WN, Jordan A (2009) Governing sustainability. Cambridge University Press, Cambridge

Alban F, Appere G, Boncoeur J (2006) Economic analysis of marine protected areas: a literature review. EMPAFISH Project, p 51

Arkema KK, Abramson SC, Dewsbury BM (2006) Marine ecosystem-based management: from characterization to implementation. Front Ecol Environ 4:525–532

Armitage D, Marschke M et al (2008) Adaptive co-management and the paradox of learning. Glob Environ Chang 18(1):86–98
Armstrong CW, Holen S, Navrud S, Seifert I (2012) The economics of ocean acidification – a scoping study. University of Tromsø, Tromsø, 57 pp
Barreira A (2012) Builidng a natural intangible heritage of mankind. In: Second earth condominium conference, Gaia, Portugal, 12–13 May. Accessed at http://www.condominiodaterra.org/media/cms_page_media/39/ABarreira_Earthcondomium.pdf
Basorie WD (2012) Green economy and Blue economy Jakarta opinion paper. Published June 05 2012. http://www.thejakartapostcom/news/2012/06/05/green-economy-and-blue-economyhtml. Accessed December 2012
Bateman I, Mace G et al (2011) Economic analysis for ecosystem service assessments. Environ Resour Econ 48(2):177–218
Beatley T, Brower DJ, Schwab AK (2002) An introduction to coastal zone management, 2nd edn. Island Press, Washington, DC, p 331. ISBN 1-55963-915-6
Beaumont N, Townsend M, Mangi S, Austen M C (2005) Marine biodiversity an economic valuation building the evidence base for the Marine Bill. DEFRA, p 73
Biermann F (2007) Earth system governance as a crosscutting theme of global change re-search. Glob Environ Chang 17:326–337
Bihim JÁ (2004) Governação nas Autarquias Locais Colecção, Inovação e Governação nas Autarquias http://www2.spipt/inovaut/docs/Manual_Ipdf (In Portuguese)
Block WE (2011) Review of Ostrom's governing the commons. Libertarian papers 3, 21, libertarianpapers.org. Accessed November 2012
Bosselmann K, Engel R, Prue T (2008) Governance for sustainability: issues, challenges and successes. IUCN environmental policy and law paper no 70. Gland: IUCN, p 278. ISBN 978-2-8317-1105-8
Boyd J, Banzhaf S (2007) What are ecosystem services? The need for standardized environmental accounting units. Ecol Econ 63(2–3):616–626. doi:10.1016/jecolecon200701002
Bromley DW (2008) The crisis in ocean governance: conceptual confusion, spurious economics, political indifference. MAST 6(2):7–22
CBD (2012) Marine biodiversity – one ocean, many worlds of life. Montreal: Secretariat of the Convention on Biological Diversity, p 77
Commonwealth Secretariat (2012) A new approach to ocean governance: practical ways to fast track the Green Economy Nairobi, Kenya, February 2012 January CCGE (12)1, 19 pp, http://www.secretariatthecommonwealthorg/doclist/190676/36161/243527/ccge2012/. Accessed January 2013
Costanza R (1999) The ecological, economic, and social importance of the oceans. Ecol Econ 31(2):199–213
Costanza R, Andrade F, Antunes P, van den Belt M, Boersma D, Boesch DF, Catarino F, Hanna S, Limburg K, Low B, Molitor M, Pereira JG, Rayner S, Santos R, Wilson J, Young M (1998) Principles for sustainable governance of the oceans. Science 281:198–199
Crowder L, Norse E (2008) Essential ecological insights for marine ecosystem-based management and marine spatial planning. Mar Policy 32(5):772–778
Crowder L, Osherenko G, Young OR, Airamé S, Norse EA, Baron N et al (2006) Resolving mismatches in US ocean governance. Science 313:617–618
Curtin R, Prellezo R (2010) Understanding marine ecosystem based management: a literature review. Mar Policy 34:821–830
Daily GC, Polasky S, Goldstein J, Kareiva PM, Mooney HA, Pejchar L, Ricketts TH, Salzman R, Shallenberger R (2009) Ecosystem services in decision making: time to deliver. Front Ecol Environ 7(1):21–28
Damanaki M (2013) The future economics of the sea, SPEECH/11/178, Meeting of the Employers' Group of the European Economic and Social Committee Brussels, 14 March 2011
Dasi JF (2008) Governança territorial para el desarrollo sostenible: estado de la cuestión e agenda. Boletin AGE Nº46-2008, pp 11–32 (In Spanish)

De Alcantara CH (1998) Uses and abuses of the concept of governance. Int Soc Sci J 155:105–113
Douvere F, Ehler C (2009) Ecosystem-based marine spatial management: an evolving paradigm for the management of coastal and marine places. Ocean Yearbook 23:1–26
Dudley N (ed) (2008) Guidelines for applying protected area management categories. IUCN, Gland
Ehler C, Douvere F (2007) Visions for a sea change report of the first international workshop on marine spatial planning. In: Intergovernmental Oceanographic Commission and Man and the Biosphere Programme. IOC manual and guides, vol 46. ICAM Dossier, 3. UNESCO, Paris, 83 pp
European Commission (2012).Blue growth scenarios and drivers for sustainable growth from the oceans, seas and coasts. MARE/2010/01. Ecorys. EU (2012) http://ec.europa.eu/maritimeaffairs/policy/documents/limassol_en.pdf
Fisher B, Turner RK (2008) Ecosystem services: classification for valuation. Biol Conserv 141:1167–1169
Folke C, Hahn T, Olsson P, Norberg J (2005) Adaptive governance of social-ecological systems. Annu Rev Environ Resour 30:81–833
Folke C, Jansson A, Rockström J, Olsson P, Carpenter SR, Chapin FS, Crépin AS, Daily G, Danell K, Ebbesson J, Elmqvist T, Galaz V, Moberg F, Nilsson M, Österblom H, Ostrom E, Åsa Persson A, Peterson G, Polasky S, Steffen W, Walker B, Westley F (2011) Reconnecting to the biosphere. Ambio 40(7):719–738
Frank V (2007) The European community and marine environmental protection in the International law of the sea: implementing global obligations at the regional level. Martinus Nijhoff, Leiden, p 505 ISBN: 978 90 04 15695 1
Friedheim RL (1999) Ocean governance at the millennium: where we have been – where we should go. Ocean Coast Manag 42:747–765
Gonçalves A (2005) O conceito de governança. In: Conselho Nacional de Pesquisa e Pós Graduação em Direito – CONPEDI, Anais, 16 p Manaus. (In Portuguese)
Gubbay S (1995) Marine protected areas – past, present and future marine protected areas. Conserv Biol 5:1–14
Halpern BS, McLeod KL, Rosenberg AA, Crowder LB (2008) Managing for cumulative impacts in ecosystem-based management through ocean zoning. Ocean Coast Manag 51:203–211
Halpern BS, Longo C, Hardy D, McLeod KL, Samhouri JF, Katona SK, Kleisner K, Lester SE, O'Leary J, Ranelletti M, Rosenberg AA, Scarborough C, Selig ER, Best BD, Brumbaugh DR, Chapin FS, Crowder LB, Daly KL, Doney SC, Elfes C, Fogarty MJ, Gaines SD, Jacobsen KI, Karrer LB, Leslie HM, Neeley E, Pauly D, Polasky S, Ris B, Martin KS, Stone GS, Sumaila UR, Zeller D (2012) An index to assess the health and benefits of the global ocean. Nature 488:615–620
Hardin G (1968) The tragedy of the commons. Science 162(3859):1243–1248. doi:10.1126/science16238591243
Humphrey S, Burbridge P, Blatch C (2000) US lessons for coastal management in the European Union. Mar Policy 24:275–286
ICWO (1998) The ocean our future. Cambridge University Press, Cambridge, p 248. ISBN 9780521642866
IGCC (2002) La Gobernanza para el desarrollo sostenible en Cataluña: conceptos,requerimientos institucionales y elementos de análisis. Documents de Recerca 5, Institut Internacional de Governabilitat de Catalunya, Generalitat de Catalunya, Barcelona (In Spanish)
IOC/UNESCO, IMO, FAO, UNDP (2011) A blueprint for ocean and coastal sustainability Paris: IOC/UNESCO
Jentoft S, Chuenpagdee R (2009) Fisheries and coastal governance as a wicked problem. Mar Policy 33:553–560
Juda L (1999) Consideration in developing a functional approach to the governance of large marine ecosystems. Ocean Dev Int Law 30(1):43–69

Juda L (2003) Changing national approaches to ocean governance: the United States, Canada and Australia. Ocean Dev Int Law 34:161–187

Juda L, Hennessey T (2001) Governance profiles and the management of the uses of large marine ecosystems. Ocean Dev Int Law 32:43–69

Kemp R, Parto S (2005) Governance for sustainable development: moving from theory to practice International. J Sustain Dev 8:12–30

Kidd S, Ellis G (2012) From the land to sea and back again? Using terrestrial planning to understand the process of marine spatial planning. J Environ Policy Plan 14(1):49–66

Kildow JT, McIlgorm A (2010) The importance of estimating the contribution of the oceans to national economies. Mar Policy 34(3):367–374

Kimball LA (2001) International ocean governance: using international law and organizations to manage marine resources sustainably. The World Conservation Union, Gland, p 123 pp

Kooiman J (2003) Governing as governance. Sage, London, p 256. ISBN 0761940367

Krahmann E (2003) National, regional, and global governance: one phenomenon or many? Glob Gov 9:323–346

Kullenberg G (2010) Human empowerment: opportunities from ocean governance. Ocean Coast Manag 53:405–420

Lamb W (2013) Commentary on economic valuations of biodiversity. Ecol Econ 89:170–17

Lange P, Driessen PPJ, Sauer A, Bornemann B, Paul Burger P (2013) Governing towards sustainability – conceptualizing modes of governance. J Environ Policy Plan 15(3):403–425

Laurans Y, Rankovic A et al (2013) Use of ecosystem services economic valuation for decision making: questioning a literature blindspot. J Environ Manage 119:208–219

Leslie HM (2005) A synthesis of marine conservation planning approaches. Conserv Biol 19(6):1701–1713

Mann-Borgese, E (1998) The oceanic circle: governing the seas as a global resource. United Nations University Press, Tokyo/New York. ISBN-10: 92-808-1013-8

Mann-Borgese E (2000) The economics of the common heritage. Ocean Coast Manag 43:763–779

Martinelli A (2003) Markets, governments, communities and global governance. Int Sociol 18:291–323

McFadden L (2007) Governing coastal spaces: the case of disappearing science in integrated coastal zone management. Coast Manag 35:429–443

McLeod KL, Lubchenco J, Palumbi SR, Rosenberg AA (2005) Communication partnership for science and the sea scientific consensus statement on marine ecosystem-based management. www.compassonlineorg/pdf_files/EBM_Consensus_Statement_v12pdf. Accessed 2012

Mee LD, Jefferson RL, Laffoley D, d'A Elliott M (2008) How good is good? Human values and Europe's proposed marine strategy directive. Mar Pollut Bull 56:187–204

Millennium Ecosystem Assessment (MA) (2003) Ecosystems and human well-being: a framework for assessment. Island Press, Washington, DC

Millennium Ecosystem Assessment (MA) (2005) Ecosystems and human well-being: synthesis report. Millennium Ecosystem Assessment. Island Press, Washington, DC, p 155

Muradian R, Rival L (2012) Between markets and hierarchies: the challenge of governing ecosystem services. Ecosyst Serv 1:93–100

Murawski SA (2007) Ten myths concerning ecosystem approaches to marine resource management. Mar Policy 31:681–690

Noone K, Sumaila R, Díaz RJ (2012) Valuing the ocean draft executive summary. Stockholm Environment Institute. http://www.sei-international.org/publications?pid=2064. Accessed January 2013

Nunes PLD, Ghermandi A (2013) The economics of marine ecosystems: reconciling use and conservation of coastal and marine systems and the underlying natural capital. Environ Resour Econ 56(4):459–465

Nunes PALD, Ding H, Markandya A (2009) The economic valuation of marine ecosystems. In: Briand F (ed) Economic valuation of natural coastal and marine ecosystems, CIESM workshop monograph, 37, Monaco, pp 23–34

Olsen S (1993) Will integrated coastal management programs be sustainable; the constituency problem. Ocean Coast Manag 21:201–226
Olsen SB, Sutinen JG, Juda L, Hennessey TM, Grigalunas (2006) TA Handbook on governance and socioeconomics of large marine ecosystems RI: Coastal Resources Center. University of Rhode Island, Kingston, p 94
Ostrom E (1990) Governing the commons: the evolution of institutions for collective action. Cambridge University Press, Cambridge. ISBN 0 521 40599 8
Paramio L (2012) GOVERNANÇA OCEÂNICA: Bases Estratégicas de Desenvolvimento para o Mar dos Açores. Tese de Doutoramento Universidade dos Açores.Ocean governance: Strategic Basis for the Development the Sea of Azores. PhD thesis, University of Azores, Ponta Delgada, Portugal, p 346 (In Portuguese)
Pardo A (1967) Address to the United Nations General Assembly, Twenty-Second Session, First Committee, 1516th Meeting United Nations, New York. http://www.un.org/Depts/los/convention_agreements/texts/pardo_ga1967pdf
Pardo A (1993) Perspectives on ocean governance. In: Van Dyke JM, Zaelke D, Hewison G. (eds) Freedom for the seas in the 21st century: ocean governance and environmental harmony, pp 38–40. Washington, DC: Island Press.
Partidário MR (2010) TEEB case: SEA for including ecosystem services in coastal management, Portugal. www.TEEBweb.org. Accessed June 2013
Pauli G (2010) The blue economy – 10 years, 100 innovations, 100 million jobs: report to the Club of Rome. Paradigm Publications, Taos, New Mexico, USA
Portman ME (2008) From land to sea: the role of land trusts in marine protection. Soc Nat Resour 22:12–26
Pugh, D (2008) Socio-economic indicators of marine-related activities in the UK economy. The Crown Estate, London, p 68. ISBN: 978-1-906410-01-8
Randall A (1988) What mainstream economists have to say about the value of biodiversity. In: Wilson EO (ed) Biodiversity. National Academy Press, Washington, DC, pp 217–223
Remoundou K, Koundouri P, Kontogianni A, Nunes PLD, Skourtos M (2009) Valuation of natural marine ecosystems: an economic perspective. Environ Sci Pol 12(7):1040–1051, ISSN 1462-9011, http://dx.doi.org/10.1016/j.envsci.2009.06.006
Rosenau JN (1999) Toward an ontology for global governance. In: Hewson M, Sinclair TJ (eds) Approaches to global governance theory. State University of New York Press, Albany
Rosenau JN (2002) Governance in a new global order. In: Held D, McGrew A (eds) Governing globalization: power, authority, and global governance. Polity Press, Cambridge
Rosenberg AA (2005) Regional governance and ecosystem-based management of ocean and coastal resources: can we get there from here. Duke Envtl L Pol'y F 16:179
Scovazzy T (2012) Marine protected areas in waters beyond national jurisdiction. In: 30 years of signature of the United Nations convention on the law of the sea: the protection of environment and the future of the law of the sea. Abstract books, 15–17 November 2012, Porto (Portugal)
Seidel H, Lal PN (2010) Economic value of the pacific ocean to the pacific islands and territories. IUCN, Gland, 74 pp
Sheppard C (2006) The muddle of 'biodiversity'. Mar Pollut 52(2):123–124
Steinberg PE (2001) The social construction of the ocean. Cambridge University Press, Cambridge
Suarez de Vivero JL, Mateos JCR, Corral DF (2008) The paradox of public participation in fisheries governance: the rising number of actors and the devolution process. Mar Policy 32(2008):319–325
Swedish Environmental Protection Agency (2008) The economic value of ecosystem services provided by the Baltic Sea and Skagerrak. Report 5874, Stockholm, Sweden. Available at:http://www.naturvardsverket.se/Documents/publikationer/978-91-620-5874-6.pdf. Downloaded at 1 Oct 2013
TEEB (2010) The economics of ecosystems and biodiversity: mainstreaming the economics of nature: a synthesis of the approach, conclusions and recommendations of TEEB, p 39, http://www.teebweb.org
TEEB (2012) Why value the oceans – a discussion paper (Yannick B, Linwood P, eds). http://www.teebweb.org. Accessed May 2013

Tyldesley D (2004) Coastal and marine spatial planning framework for the Irish sea pilot project report to the joint nature conservation committee, p 35. http://jncc.defra.gov.uk/pdf/Tyldesley%20Marine%20spatial%20planningpdf
UN (2012) The future we want. United Nations resolution 66/228 adopted by the General Assembly. Available online at http://sustainabledevelopment.un.org/futurewewant.html
UNCLOS (1982) United Nations Convention on the Law of the Sea, 10 Dec 1982, 1833 UNTS
UNEP (2006) Marine and coastal ecosystems and human wellbeing: a synthesis report based on the findings of the Millennium Ecosystem Assessment United Nations Environmental Program, Nairobi, Kenya
UNEP (2011) Towards a Green Economy: pathways to sustainable development and poverty eradication. United Nations Environment Programme. www.unep.org/greeneconomy. ISBN: 978-92-807-3143-9
UNEP, FAO, IMO, UNDP, IUCN, World Fish Center, GRID Arendal (2011) Green economy in a blue world www.unep.org/greeneconomy; www.unep.org/regionalseas24pp
UNEP, FAO, IMO, UNDP, IUCN, World Fish Center, GRID Arendal (2012) Green economy in a blue world. http://www.unep.org/pdf/Green_Economy_Blue_Full.pdf. Accessed May 2013
Vallega A (2001) Ocean governance in post-modern society – a geographical perspective. Mar Policy 25(6):399–414
VanLeeuwen J, Van Tatenhove J (2010) The triangle of marine governance in the environmental governance of Dutch offshore platforms. Mar Policy 34(3):590–597
World Bank (1992) World Bank's report, governance and development. World Bank, Washington, DC
Young OR, Osherenko G, Ekstrom J, Crowder LB, Ogden J, Wilson JA, Day JC, Douvere F, Ehler CN, McLeod KL et al (2007) Solving the crisis in ocean governance: place-based management of marine ecosystems. Environ Sci Policy Sustain Dev 49(4):20–32

Chapter 5
Interaction of Fisheries and Aquaculture in the Production of Marine Resources: Advances and Perspectives in Mexico

Roberto Pérez-Castañeda, Jesús Genaro Sánchez-Martínez, Gabriel Aguirre-Guzmán, Jaime Luis Rábago-Castro, and María de la Luz Vázquez-Sauceda

Abstract Marine capture fisheries in Mexico are dominated by sardine, shrimp, and tuna, representing as a whole 60 % of the total catch. However, shrimp and tuna are the most important fishery resources in terms of economic value. Capture shrimp fishery in Mexico has exhibited stagnating catches (around 65 thousand tonnes) since the last two decades, and shrimp stocks have been clearly depleted in some parts of the country. Conversely, shrimp (*Litopenaeus vannamei*) aquaculture has shown an exponential growth in the number of farms and shrimp production since mid-1980s. As a result, currently, shrimp aquaculture production has almost doubled the total production of the capture shrimp fishery. On the other hand, total catch of tuna by Mexican tuna fleet has fluctuated around 130 thousand tonnes during the last 5 years (2007–2011). Contribution of farmed tuna (*Thunnus orientalis*) has been negligible in terms of volume; however, the price of 1 tonne of farmed tuna is about 7–13 times that of tuna caught by the fishing fleet, making it an attractive alternative as source of employment and income. The case studies presented here are indicative of the potential value of aquaculture as a complementary productive activity to meet the growing human demand for food from the sea. This is especially relevant in terms of global fisheries production because the maximum fisheries catch potential from the oceans around the world has apparently been reached. However, there are still concerns associated with aquaculture impacts on the environment that must be addressed.

R. Pérez-Castañeda (✉) • J.G. Sánchez-Martínez • G. Aguirre-Guzmán
J.L. Rábago-Castro • M. de la Luz Vázquez-Sauceda
Facultad de Medicina Veterinaria y Zootecnia, Universidad Autónoma de Tamaulipas, Carretera Victoria-Mante km 5, AP 263, Cd. Victoria 87000, Tamaulipas, Mexico
e-mail: roperez@uat.edu.mx

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8,
DOI 10.1007/978-3-319-06305-8_5,

5.1 Introduction

The fundamental purpose of aquaculture and fishing is providing seafood products for human consumption. Wild capture fishing is a much older activity than aquaculture, which is referred as the farming of aquatic animals and plants (FAO 2012). Fishing has been practiced by humanity since the earliest stages of human evolution as a source of food supply (Gartside and Kirkegaard 2009). This practice allowed many ancient cultures around the world to settle in the coastal areas, obtaining seafood products from the intertidal zone and from shallow waters along the seashore (Squires 2009).

Apparently, fishing in the ancient world did not only play a marginal role in the economy, since there is archeological evidence of fish processing on a commercial scale along the Mediterranean, Atlantic coast and in the Black Sea (Bekker-Nielsen 2005). However, during the last century the prominent intensification of fishing effort, and the fisheries industrialization in general, made possible a pronounced increase of total landings as well as of the diversity of catches, including the capture of new species that had not been exploited before, such as pelagic and deep sea organisms. However, while global fishing production and the total number of fishing species captured underwent a meaningful increase during the last decades, on the other hand, the intensification of the fishing effort caused the depletion of fishing stocks due to overfishing, overstressing the marine systems, which is considered to be largely responsible for the current global fisheries crisis (Pauly et al. 2002).

Total landings of worldwide marine fisheries have fluctuated from 16.8 million tonnes in 1950 up to 86.4 million tonnes in 1996, stabilizing at about 80 million tonnes during the following years; with a slight decrease in 2010, when the global marine capture fisheries was of 77.4 million tonnes. Stabilization of global fisheries production over the past recent years might suggest that the maximum fisheries catch potential from the oceans around the world has probably been reached. Actually, according to recent data from FAO (2012) 14.1 % of the world marine fishery stocks are underexploited or moderately exploited, 57.3 % are fully exploited, 13.7 % are overexploited and 7.6 % are depleted or in recovering. Additionally to the impacts of fishing per se, it has been pointed out that capture fisheries are also affected by climate-related threats, including climate change (Brander 2007).

Under this scenario there is apparently little chance to increase the production of marine resources based solely on captured fisheries production, and thus seafood supply as a source of animal protein for human consumption could be seriously compromised in the following years. Moreover, the world human population has increased exponentially, from 2.53 billion in 1950 to 7.16 billion in 2013 (United Nations 2013) and, as a result, the human population demand for seafood is continuously growing year after year. In this regard, during the last years, aquaculture has been playing an increasingly important role in contributing to the production of marine resources, and, although fisheries production remains stable, aquaculture production is still expanding globally and diversifying (FAO 2012). Aquaculture is the fastest growing sector of the world food economy, and its production is projected

Fig. 5.1 Geographic location of Mexico showing both coastlines along the Pacific Ocean and Gulf of Mexico

to overtake that of other sectors of animal food production. Indeed, since 2011 world farmed fish production topped beef production (Larsen and Roney 2013).

Aquaculture was probably first practiced in China more than 4,000 years ago, and particularly in Asia, this activity has been traditionally important as source of food (Rabanal 1988). In fact, Asian countries have historically been the major producers of cultured aquatic organisms in the world (FAO 2012). The contributions of capture fisheries and the aquaculture industry to the world's production of marine resources for human consumption have evolved to reach total production levels of 148 million tonnes of fish in 2010 (FAO 2012). The combined contribution of fisheries and aquaculture might be critical to guarantee the supply of nutritious food and animal protein to the escalating demands of a growing world population. However, the future trends of this interaction might not be easily predicted, in part because fisheries catch is primarily dependent on the natural boundaries of the environment, whereas aquaculture productivity is mostly limited by water quality and biotechnological aspects of the aquatic species of interest. However, it is clear that contribution of aquaculture to seafood production is increasingly significant in many countries around the world.

Mexico is a country that has 8,475 km of coastline along the Pacific Ocean and 3,118 km along the Gulf of Mexico and the Caribbean Sea (Cifuentes-Lemus et al. 1995) (Fig. 5.1). Seafood production in Mexico was mostly based on capture fisheries, but since the mid-1980s aquaculture has become a growing contributor for the production of marine resources for human consumption. Even in certain regions of the country the relative contribution of fisheries and aquaculture production has been reversed in the last years, generating higher production of seafood by aquaculture than by fishing; such is the case of shrimp production along the Mexican Pacific coast.

In this chapter, we analyze the historical interactions of capture fisheries and aquaculture as complementary sectors in the production of marine resources for human consumption in Mexico. Advances and perspectives regarding the interaction between fishing and aquaculture activities of the most important fishery resources in the country (shrimp and tuna) are analyzed considering their production levels, inherent characteristics and concerns.

5.2 Inherent Characteristics and Global Situation of Marine Fisheries and Aquaculture

World marine fisheries are based on the capture and removal of marine organisms from the natural environment. This activity can be view as a form of hunting of wild animals in the marine ecosystem. The productivity and sustainability of fisheries are mostly dependent on natural constraints such as the population size and fish stocks regeneration, environmental variability and ecosystems health, but also of the fishery management practices. Most fisheries in the world are characterized by an open access regime and are impacted by additional stressors such as global climate, leading to fishing stocks to be overharvested and in some cases overexploited to the point of collapse (Pauly et al. 2005). Apparently, more than 3,000 years of scientific understanding of the phenomena, theirs causes, and suitable mitigation measures, are not sufficient to prevent the destruction of natural resources, as stated by Ludwig et al. (1993).

The global marine catch in 2009 was mostly comprised of pelagic species (41 %) such as herrings, sardines, anchovies (small pelagics) as well as tunas, bonitos, and billfishes (large pelagics) (Ye and Cochrane 2011). It has been pointed out that as part of the global marine fisheries crisis, a decrease in the mean trophic level of species groups in landings from 1950 to 1994 has been experienced, a phenomenon known as "fishing down marine food webs". This situation not only denotes the gradual transition in catch composition (from long-lived, high trophic level piscivorous bottom fish toward short-lived, low trophic level invertebrates and planktivorous pelagic fish) but also implies that the exploitation rates of fishing marine stocks are unsustainable (Pauly et al. 1998). Moreover, from 1950 to 1989 global marine catches increased continuously at an average rate of 1.6 million tonnes per year, however, a shift in this trend has been observed from 1989 to 2011 exhibiting a phase of stagnating catches stabilizing at about 80 million tonnes (Fig. 5.2), suggesting that its maximum productive potential has probably been reached. Irrespective of the global fisheries crisis, human demand for fish and seafood will inevitably continue to grow; therefore, additional production of seafood is necessary to meet the global demand for food from the continuously growing world human population. Under this scenario, aquaculture production might be an important complementary source of seafood for human consumption.

Aquaculture is the farming of aquatic organisms (e.g. fish, mollusks, crustaceans, aquatic plants) in a controlled environment, which implies some form of intervention such as rearing, feeding and protecting from predators (FAO 2011).

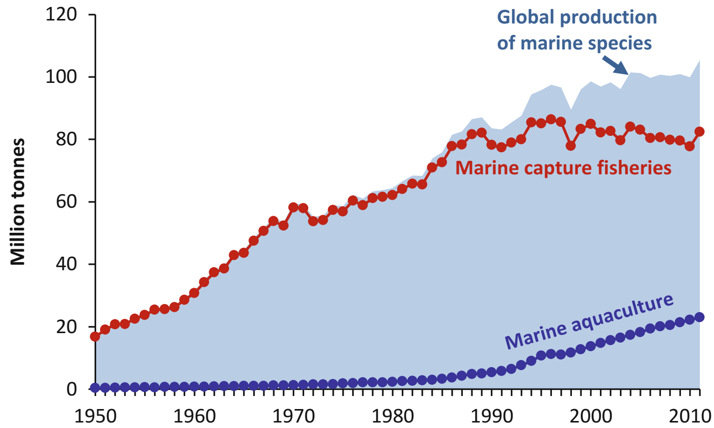


Fig. 5.2 World production of marine capture fisheries and aquaculture of marine species (1950–2011) (Data from FAO statistics)

Aquaculture of marine species usually occurs in ponds or tanks (land based aquaculture) as well as in existing water bodies utilizing some type of artificial enclosure, like cages or pens, for containing the aquatic organisms (water based aquaculture).

Composition of farmed marine species varies among countries and is dependent of the culture environment (brackish or marine water). According to FAO data from 2011 (http://www.fao.org/fishery/statistics/en), freshwater aquaculture is dominated by the farming of carps and cichlids. However, aquaculture production of brackish and marine species is mainly characterized by mollusks (clams, oysters, mussels) representing about 61 % of the production, followed by crustaceans (shrimps, prawns, crabs) representing 15 %, and marine fishes (miscellaneous coastal and pelagic fishes) representing 8 % of the total. Aquaculture by volume is dominated by Asian countries; which all together account for about 80 % of world aquaculture production in brackish and marine environments.

In contrast with the marine fisheries crisis, aquaculture of marine and brackish species has experienced a continued growth, increasing its production from 340 thousand tonnes in 1950 to 23 million tonnes in 2011 (Fig. 5.2). In fact, when comparing marine fisheries and aquaculture production from 1993 to 2011, a significant negative trend is observed for capture fisheries whereas a significant positive pattern is evident for aquaculture production of brackish and marine species (Fig. 5.3). It is clear that fisheries alone cannot fill the global demand for fish and seafood; however, aquaculture production combined with wild fisheries production might be a plausible alternative.

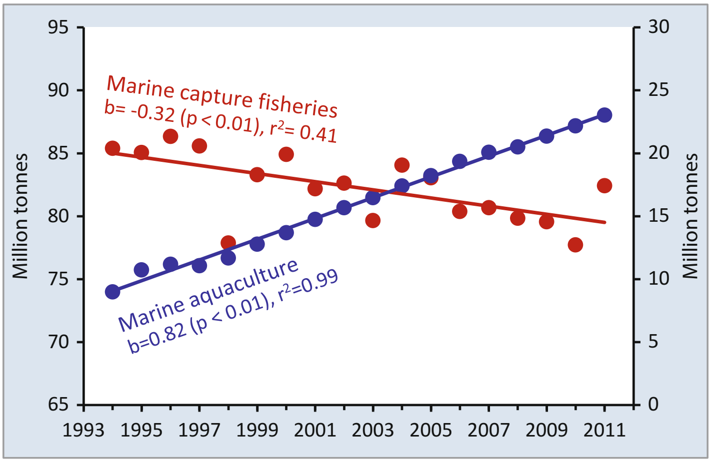


Fig. 5.3 Trends in global production of marine capture fisheries and marine aquaculture for the period 1994–2011. In both cases a significant linear relationship was fitted. The slope coefficient (*b*), its significance, and determination coefficient are reported (Data from FAO statistics)

5.3 Contribution of Fisheries and Aquaculture to Seafood Production in Mexico

Commercial landings from marine capture fisheries in Mexico are dominated by sardines, shrimp, and tuna, which as a whole represent about 60 % of the total catch in the country. However, the most economically important species or species groups are shrimp and tuna accounting for 43 % and 7 %, respectively, of the total value of fishery production in the country (CONAPESCA 2011). According to fisheries and aquaculture statistics in Mexico, capture fisheries increased from 118,043 tonnes in 1954 to a maximum of 1.5 million tonnes in 1981; nevertheless, during the following years catches have fluctuated around 1.2–1.4 million tonnes (Fig. 5.4). Temporal trends of fishery landings in Mexico could be the result of different extractive phases of the fishery stocks over time, as described for Latin American benthic shell fisheries (Castilla and Defeo 2001). In this regard, an initial exploitation phase occurred during 1954 to early 1970s; afterward an expansive extraction phase began from mid-1970s extending to early 1980s. Finally, a trend toward stabilization of catches is observed from mid-1980s to the present.

With regard to aquaculture production in Mexico, its first record in official statistics was 109,061 tonnes in 1983, reaching 262,855 tonnes in 2011 (Fig. 5.4). To date, the contribution of aquaculture to the national fisheries production is about 16 % by volume. Nonetheless, aquaculture production accounts for 40 % in terms of economic value of the national fisheries production, which could be explained by the growing industry of shrimp farming, the most economically important species in Mexico (CONAPESCA 2011).

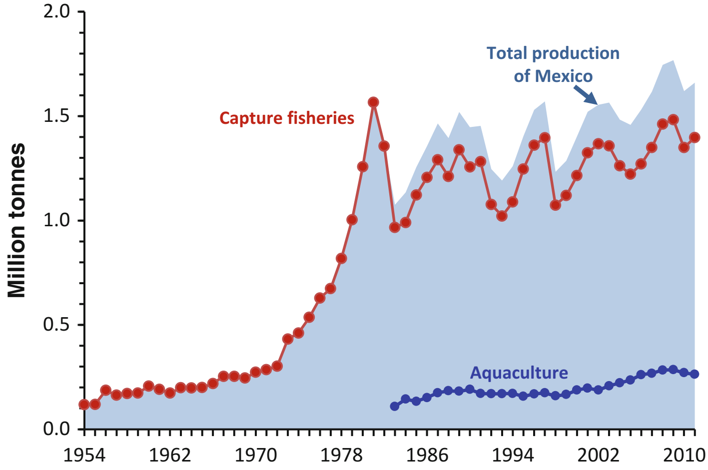


Fig. 5.4 Total production (freshwater and marine) of capture fisheries and aquaculture in Mexico (Data from CONAPESCA (Mexico))

5.4 Trends and Advances on the Interaction of Fishing and Aquaculture for Shrimp Production

5.4.1 Situation of the Shrimp Fishery in Mexico

Shrimp is the most important marine resource in terms of economic value in Mexico, representing about 550 million dollars in 2011; its fishery occurs along the Gulf of Mexico and Caribbean coasts as well as the Pacific coast. Shrimps are captured as juveniles by an artisanal fleet in coastal lagoons, whereas the industrial fleet off-shore captured adult shrimps, resulting in a sequential fishery. In the Gulf of Mexico and the Caribbean Sea the main species captured by the shrimp fishery are *Farfantepenaeus aztecus*, *F. duorarum*, *Litopenaeus setiferus* and *F. brasiliensis*. On the other hand, catches of Mexican Pacific shrimp fishery are mainly conformed by *F. californiensis*, *Litopenaeus stylirostris* and *L. vannamei* (INAPESCA 2006).

Shrimp catches in the Gulf of Mexico have shown an overall decreasing trend from 27 thousand tonnes in 1980 to 20 thousand tonnes in 2011, exhibiting remarkable differences in historical landings according to the region (Fig. 5.5). For instance, shrimp captures in the Campeche Sound (Southern Mexico) declined sharply from 14 thousand tonnes in 1979 to 2.6 thousand tonnes in 2006, with a slightly increase to 7.5 thousand tonnes in 2011 (Fig. 5.5). The shrimp stocks in this region of the Gulf of Mexico are depleted, particularly the pink shrimp (*F. duorarum*). During the 1960s this region accounted for about 90 % of total shrimp catches in the Gulf of Mexico and the Caribbean Sea; however over recent years its contribution has been usually lower than 20 % (Fig. 5.5). The decreasing trend in shrimp catches has been related to recruitment failures; however, the future scenario of the shrimp fishery in

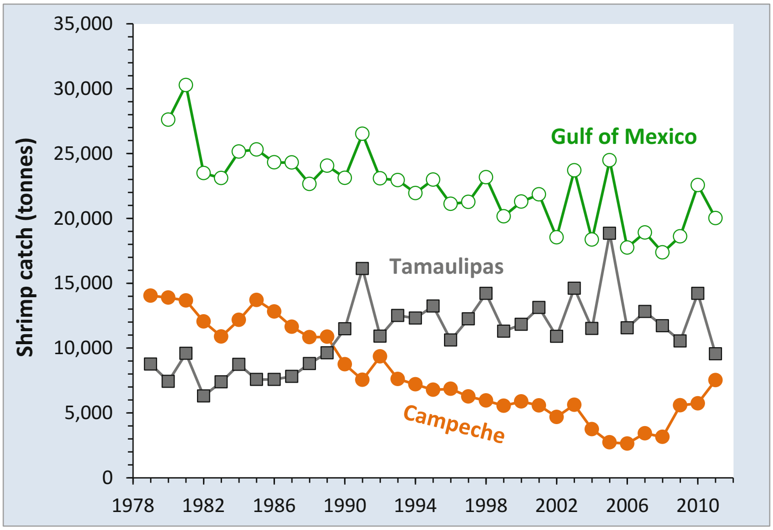


Fig. 5.5 Total shrimp catches in Mexican coasts of the Gulf of Mexico and in the two most productive shrimp fishery zones along the coasts of Tamaulipas and Campeche (Data from CONAPESCA (Mexico))

the Campeche Sound is not promissory because if the current levels of fishing intensity are maintained, then the stock density will continue to decrease (Ramírez-Rodríguez et al. 2000) aggravating the present situation of fishery depletion.

Conversely, the trend in shrimp catches in the Tamaulipas coast (Northeastern Mexico) has shown an increasing trend during the years 1950–1990, reaching an apparent stabilization of catches at around 12–13 thousand tonnes since 1992. It is worth noting that since 1990 the shrimp catches in Tamaulipas have been higher than catches from the Campeche Sound (Fig. 5.5). In fact, during the last years the shrimp fishery of Tamaulipas has provided annually more than 50 % of the total shrimp catch from the Gulf of Mexico, while Campeche usually contributes with no more than 20 %, except for the period of 2009–2011 where it was from 25 to 38 % (Fig. 5.5), which might be mostly due to the increase in captures of other shrimp species such as the Atlantic seabob (*Xiphopenaeus kroyeri*) and rock shrimp (*Sicyonia brevirostris*) from the Caribbean waters.

On the other hand, captures in the Mexican Pacific shrimp fishery have fluctuated around 45 thousand tonnes during the last years, representing the most important shrimp fishery in Mexico in terms of volume, contributing in average about 70 % of the national shrimp catch, whereas the Gulf of Mexico shrimp fishery accounts for the remaining 30 %. Historically, higher shrimp catches have been obtained on the west side coast of Mexico (Fig. 5.6). Therefore, national trends in shrimp catch follow the same pattern of the Mexican Pacific shrimp fishery, being the states of Sonora and Sinaloa the most productive for this fishery. Contrary to the declining trend of shrimp catch in the Gulf of Mexico, the Mexican Pacific shrimp fishery has

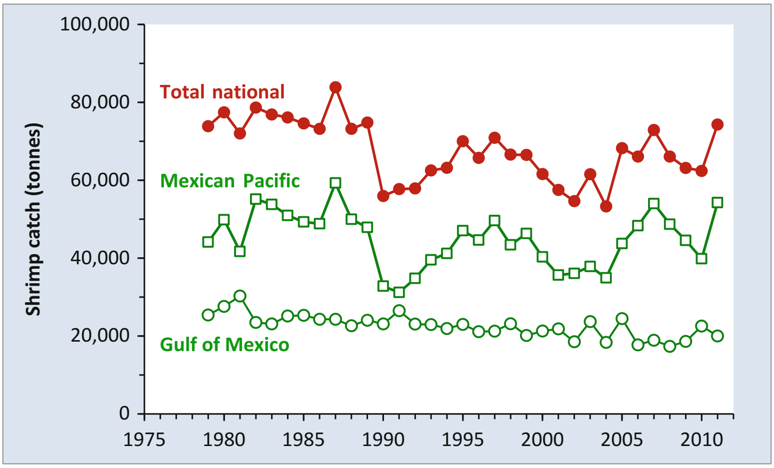


Fig 5.6 Fluctuations of shrimp catch in Mexico (1979–2011) from the Pacific Ocean and Gulf of Mexico (including Caribbean Sea). Total national catch is also plotted (Data from CONAPESCA (Mexico))

exhibited an apparent stabilization since the early 1990s; this pattern is also mirrored in the national trend (Fig. 5.6). According to the above, it is clear that the shrimp fisheries in Mexico are at their maximum catch potential and indeed in some particular regions the shrimp stocks have declined steadily; and the most severe decline has been experienced by the pink shrimp (*F. duorarum*) in the Campeche Sound, Gulf of Mexico (Ramírez-Rodríguez et al. 2000).

Production of farmed shrimp in Mexico has been recorded in official statistics since 1985 with 35 tonnes (Secretaría de Pesca 1993). However, shrimp farming has grown exponentially since then, reaching an annual production of 133 thousand tonnes in 2009 and about 110 thousand tonnes in 2011 (Fig. 5.7). Moreover, since 2003 shrimp farming production exceeded the total national production of the capture shrimp fishery, and nowadays aquaculture production of shrimp nearly doubles the capture shrimp fishery production in Mexico. Farmed shrimp has made it possible to reach a national production of 170 to almost 200 thousand tonnes of shrimp during recent years (Fig. 5.7).

Along with the substantial increase in aquaculture production of shrimp, there has been an increase in the number of shrimp farms, particularly along the Mexican Pacific; while in contrast, a continuous reduction in the number of shrimp vessels has been observed since 2004. Afterward, in 2011 the number of shrimp vessels and shrimp farms was almost the same (Fig. 5.8). Although there has been a fast increment of farmed shrimp production in the Mexican Pacific during the last two decades, shrimp aquaculture has remained incipient in states located along the Mexican coasts of the Gulf of Mexico and consequently most shrimp production,

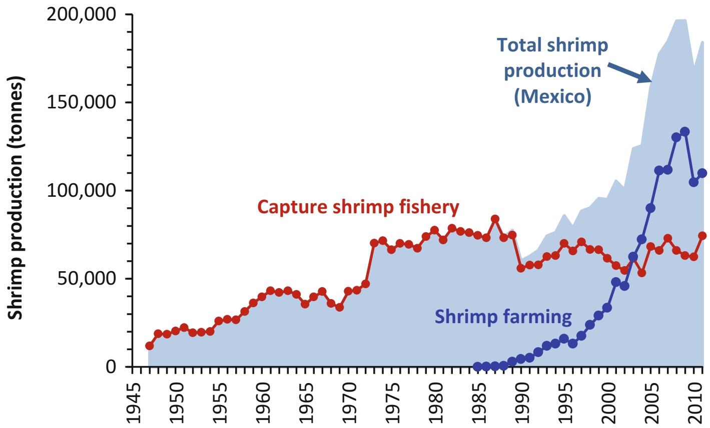


Fig. 5.7 Total production of shrimp in Mexico from capture shrimp fishery and aquaculture (shrimp farming) (Data from CONAPESCA (Mexico))

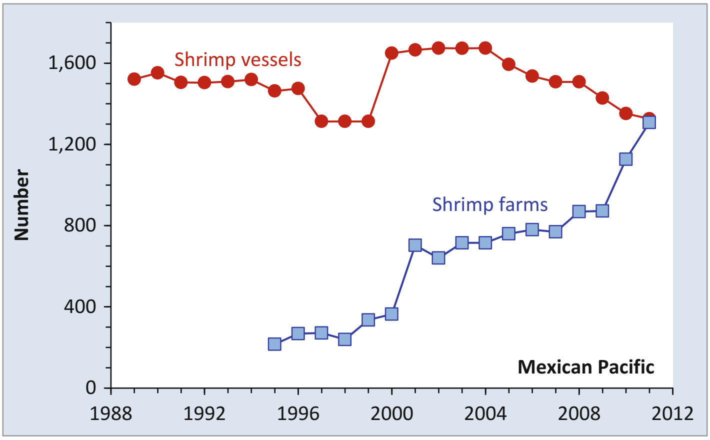


Fig. 5.8 Trends in the number of shrimp vessels and shrimp farms in the Mexican Pacific coasts during several years (Data from CONAPESCA (Mexico))

from the fishery and aquaculture sectors, comes from the Mexican Pacific (Fig. 5.9). In this regard, according to official statistics of fisheries and aquaculture production in Mexico provided by CONAPESCA (2011), the Mexican Pacific accounts for 73 % of the national capture shrimp fishery production and 97 % of shrimp farming production in the country (Fig. 5.10).

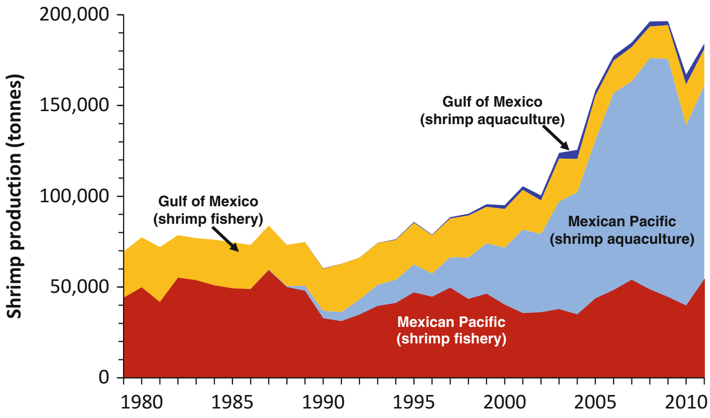
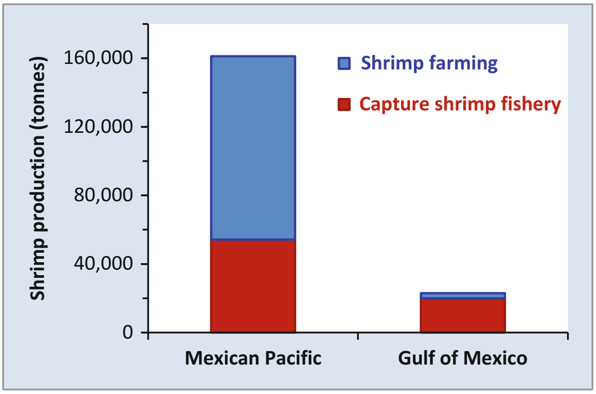


Fig. 5.9 Shrimp production in Mexico by fishing and aquaculture from the Gulf of Mexico and Pacific Ocean (1979–2011) (Data from CONAPESCA (Mexico))

Fig. 5.10 Contribution of the Mexican Pacific and Gulf of Mexico to the national production of shrimp in 2011. Production from fishing and aquaculture activities is presented separately (Data from CONAPESCA (Mexico))



5.4.2 *Aquaculture of Shrimp in Mexico: Technological and Biological Aspects*

Global production of farmed marine shrimp was around 3.5 million tonnes in 2011, being dominated by the culture of *L. vannamei* (white leg shrimp) and *Penaeus monodon* (giant tiger prawn) (Fig. 5.11). In the case of Mexico, the main shrimp species utilized in aquaculture industry is the Western white shrimp or white leg shrimp (*L. vannamei*). The culture of *L. vannamei* in Mexico began in the early 1980s because of the viral disease problems with *L. stylirostris* that until then used to be the most widely exploited in shrimp farming along the Mexican Pacific (SAGARPA 2012).

Production of the white shrimp (*L. vannamei*) represents about 75 % of the world shrimp production and it is the main species used in shrimp farms of America

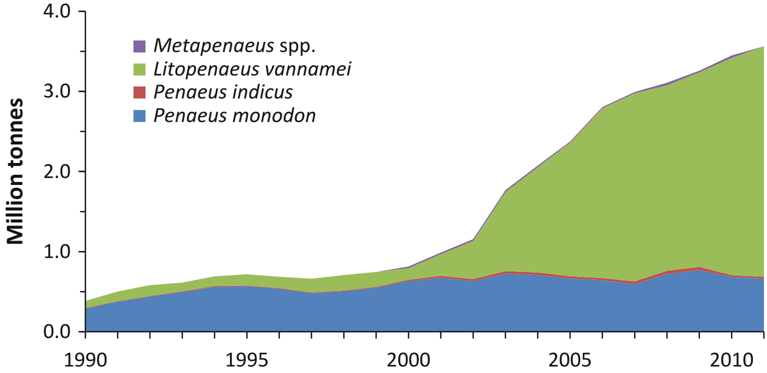


Fig. 5.11 World aquaculture production of principal shrimp species (FAO 2012)

(including Mexico), Asia, and Africa (FAO 2012); being the main introduced species for aquaculture purposes in the last two mentioned continents.

This shrimp species reaches a maximum total length of a 23 cm, and shows a uniform growth rate. The white shrimp tolerates salinities from 15 to 40, and some experimental evidence suggests the possibility for farming shrimp in freshwater areas with special acclimation (Araneda et al. 2008, 2013; Miranda et al. 2010; Ortega-Salas and Rendón 2013). Even, some commercial farms in Mexico, particularly in the Northeastern region, operate under very low salinity to freshwater conditions.

Before the 1980s, a high number of shrimp farms used to obtain post larvae directly from the wild environment or brood stock collected from natural habitats for reproduction purposes in production laboratories (maturation and hatcheries). During the period of mid-1980s to the early 1990s, the traditional method to obtain the post larvae and brood stock from wild habitats changed to obtain organisms from shrimp farms, which support the use of quality control measures against shrimp diseases (Martínez-Córdova 1999; Walker and Winton 2010). In this period, all Mexican farms employed specific pathogen free (SPF) or specific pathogen resistant (SPR) post larvae. At same time, the collection of wild post larvae was prohibited in order to prevent dispersion of shrimp diseases and as an environmental protection measure for wild shrimp populations that support shrimp fisheries (Páez-Osuna 2001; Lightner 2005; Walker and Winton 2010). Nowadays, the brood stock individuals are collected from special shrimp ponds where SPF of SPR organisms are maintained under low density, suitable feeding, and a high control of water quality (Lightner 2005). Collected organisms are analyzed in certificated aquatic animal health laboratories (Ortega 2012) before being transported to maturation tanks.

Maturation and reproduction of brood stock shrimp is carried out in tanks under controlled conditions in a maturation laboratory (Treece 2000). Shrimp females with attached spermatophore are transferred to the spawning area for spawning. Afterward, shrimp females are returned to their specific maturation tank. The spawned eggs are maintained in the spawning area until hatching. The nauplii are

Table 5.1 General characteristics of the Mexican shrimp culture systems (CONAPESCA 2011)

	Shrimp culture systems			
	Extensive	Semi-intensive	Intensive	High intensive
Pond area (ha)	1.5–280	5–15	0.3–5	0.03–1
Post larvae/m^2	1.7–15.4	0.4–40	21–58	450–500
Feeding	Natural	Moderate natural + commercial	Low natural + commercial	Commercial feed
Technology	Low level	Moderate level	High level	Higher level
Production (cycle/year)	1	1–2	2–3	3–4
Production (kg/ha)	13–500	500–1,000	1,000–5,000	>5,000
Shrimp and water quality management	No quality control	Low to moderate in nursery, moderate to high in grow-out phase	High during all production cycle	Higher during all production cycle
Number of farms	2	1,367	13	–
Production area	3	71,057	384	–

collected, washed, and evaluated for quality (Treece 2000). Later, they are cultured under standard conditions to protozoea (I–III) and mysis stages (I–III) before assuming the body plan of the adult as a post larva (Treece 2000).

In 2012, Mexico had 58 official companies of post larvae production (commonly named as production laboratories) with 10,000–15,000 million of post larvae production of *L. vannamei* and/or *L. stylirostris* distributed in the Mexican Pacific (states of Baja California, Baja California Sur, Nayarit, Sinaloa, and Sonora) and the Gulf of Mexico (Tamaulipas and Yucatan). The majority of these laboratories (75 %) are located in the states of Sinaloa and Sonora; in fact, these states from the Mexican Pacific are the largest producers of farmed shrimp in the country and have the largest number of shrimp farms.

Extensive shrimp farming was the first shrimp production system installed in Mexico, before private investment in the aquaculture industry was allowed by the government (Martínez-Córdova 1999; DeWalt et al. 2002). As in other countries, extensive production of shrimp was initially based on the capture of wild post larvae; however, presently post larvae are supplied from controlled farms with SPF or SPR organisms (Lightner 2005; Walker and Winton 2010). Table 5.1 displays the general characteristics of the main shrimp farming systems in Mexico; the number of shrimp farms and the extension in hectares used for shrimp aquaculture in 2011 is also reported. Extensive shrimp culture systems are disappearing compared to intensive and semi-intensive systems. The historical transition from extensive to intensive systems is evident; in 1995 there were 71 extensive shrimp farms (2,884 ha) whereas there were 147 semi-intensive (10,872 ha), and 13 intensive (546 ha) production systems (SEMARNAP 1996). Afterward, by 2011 the number of extensive shrimp farms decreased to only two farms (3 ha); however, there was a notable increase of semi-intensive shrimp farms reaching the amount of 1,367 farms

Fig. 5.12 Shrimp culture pond of extensive (**a**) and intensive (**b**) systems. (**c**) Nursery greenhouse of *L. vannamei* post larvae with freshwater in Reynosa, Tamaulipas, Mexico. (**d**) Shrimp harvest under HACCP conditions (Photographs provided by Zeferino Blanco-Martínez (**b**, **c**) and Gabriel Aguirre-Guzmán (**a**, **d**))

(71,057 ha), while the number of intensive shrimp production units did not change, remaining at 13 farms (383 ha) (CONAPESCA 2011).

In general, a few farms have continued using the extensive, which require relatively low levels of technology and investment. The production of shrimp depends on the natural productivity of the enclosed body water which is enhanced by fertilization (organic and inorganic); additionally, shrimp are fed a commercial feed close to harvest time, in order to increase shrimp size (Treece 2001; Martínez-Córdova et al. 2004). In this type of aquaculture system, shrimp farms take advantage of natural tides for water exchange in the shrimp pond which at the same time helps to maintain proper water quality in the ponds. This system can use pumps to facilitate this process during harvest time or during extreme low water quality conditions (Páez-Osuna et al. 2003).

Semi-intensive shrimp farming have been utilized since 1980s, being the main shrimp farming systems used in Mexico today (Table 5.1, Fig. 5.12a, b). The increase of semi-intensive and intensive production systems was possible because the Mexican government allowed private investment in the shrimp aquaculture industry, although the exploitation of shrimp species had been exclusively reserved to organizations of fishermen named ‘cooperatives’ (INAPESCA 2006). This situation generated an exponential growth in shrimp farming enabling it to reach the actual levels of productivity, where farmed shrimp production has practically doubled the capture shrimp fishery production (Fig. 5.7).

Mexican farms that use these systems recognize two culture phases. The first phase or nursery (from post larva to juvenile shrimp ≤0.5 g) use natural food obtained through previous fertilization (inorganic) with triple superphosphate or TSP [$Ca(H_2PO_4)_2$] + Urea [$CO(NH_2)_2$] on shrimp ponds (Treece 2001; Martínez-Córdova et al. 2004). The post larvae acclimatization is very relevant to reduce post larval stress caused by changes in temperature, salinity and pH when they are transferred from hatcheries to shrimp farms.

During this phase, shrimp ponds are filled with less than 100 % of their water capacity, which is increased as the shrimp grows. Pond fertilization is strongly monitored during the shrimp culture focusing on obtaining a proper productivity of algae, mollusks, polychaetes, organic matter, small crustaceans, and diverse benthic organisms that provide a natural feed source for shrimp (Treece 2001; Martínez-Córdova et al. 2004). However, as shrimp grow natural feeding is replaced by artificial feed (pellets). Several farms with intensive systems have specialized areas such as greenhouse or nursery areas with a high control of water quality, where shrimp are maintained until reaching a suitable body size before they are transported to the grow-up ponds.

In the second phase or growth phase (shrimp ≥0.5 g to harvest) there is a constant control of the different parameters of shrimp production, such as disease prevention and/or control, feeding and growing rates, feed conversion ratio, feed intake, shrimp density, among other parameters. Commercial feed is supplied one or two times per day. Dissolved oxygen levels, pH, and water temperature in the shrimp pond is assessed daily, whereas ammonium, nitrites, nitrates and primary productivity parameters are monitored weekly. Semi-intensive and intensive shrimp farms utilize water exchange and aeration supply to maintain water quality. Additionally, shrimp survival, their size and weight, and intestinal fullness are monitored biweekly in order to calculate the feed necessary for the next weeks (Arredondo-Figueroa 2002; Rojas et al. 2005).

Recently, the use of liners or geo-membrane of synthetic polymers (high-density polyethylene or HDPE, and ethylene-propylene-diene-monomer or EPDM) is used in shrimp farms with intensive systems (Horowitz et al. 2001; Samocha 2011a). Shrimp farms use these polymeric geosynthetic barriers principally as pond cover to prevent water filtration (Horowitz et al. 2001; Samocha 2011a); this type of material also facilitates the collection, treatment and/or removal of organic material and sediment produced during shrimp farming. Initially, these geo-membranes were used in maturation and larval rearing tanks, later they were also utilized in nursery ponds and finally in grow-out ponds. However, the use of EPDM has decreased due to its apparent toxicity to shrimp (Samocha 2011b). In most shrimp farms one to two crops a year are obtained; in tropical climates, even three crops a year are possible.

Some farms in Mexico grow shrimp in freshwater conditions. To this end, shrimp post larvae are previously adapted to freshwater in tanks before they are transferred to the grow-out ponds (Araneda et al. 2008, 2013; Miranda et al. 2010; Ortega-Salas and Rendón 2013). Some of these types of farms that culture shrimp in freshwater conditions, are located in the Mexican states of Baja California, Colima, Sinaloa, Sonora, and Tamaulipas which operate at commercial scale, using freshwater from artesian wells (2–10 ‰) (CONAPESCA 2011; Ortega-Salas and Rendón 2013). Early post larvae of *L. vannamei* (PL 2–7) are acclimated from seawater to freshwater at a rate of 1 ‰ per day until obtaining a salinity water level similar to that in grow-up ponds (Fig. 5.12c). The post larvae are acclimated in greenhouses at low densities for a better control of the environmental conditions, and they are feed with *Artemia* spp. nauplii and microparticle feed (Treece 2001; Araneda et al. 2008; Miranda et al. 2010).

When organisms show an adequate body size, they are transferred to the grow-out ponds (land and/or liners). The freshwater shrimp farm close to Reynosa, Tamaulipas (Mexico) located at 65–70 km from the coast is a good example of this type of production system (Gutiérrez-Salazar et al. 2011).

5.4.3 Future Considerations and Concerns

The further development of the shrimp aquaculture industry may generate a high environmental pressure (mangrove deforestation, increase of organic material and sediments in estuaries and coastal lagoons) that at the same time might diminish or stop the continuous growth of this industry. In this regard, the implementation of measures in order to develop aquaculture methods environmentally sustainable is critical (Páez-Osuna 2001; Ferreira et al. 2007).

The rapid expansion of the shrimp farming industry brought with it frequent outbreaks of diseases affecting the shrimp growth and survival (Walker and Winton 2010). Pathogens represent a critical problem to the shrimp industry, causing important declines in shrimp production with the concurrent economic losses. Viruses are known as the most important pathogens faced by the shrimp farming industry. In Mexico, the Infectious hypodermal and hematopoietic necrosis virus (IHHNV) was introduced during early 1980s into shrimp farms, producing the collapse of young farmed *L. stylirostris* which encouraged shrimp farmers to change the cultured shrimp species to *L. vannamei* (Aguirre-Guzmán and Ascencio-Valle 2000). Later, the Taura Syndrome Virus (TSV) affected the shrimp cultures of *L. vannamei* during the 1990s causing the return of *L. stylirostris* to the shrimp cultures (Lightner 1999; Aguirre-Guzmán and Ascencio-Valle 2000; Lightner 2005). Currently, *L. vannamei* is the most important shrimp cultured in Mexico; however, in early 2000s the White Spot Syndrome Virus (WSSV) generated important shrimp losses in shrimp farms particularly on *L. vannamei* (Lightner 1999, 2005; Aguirre-Guzmán and Ascencio-Valle 2000; Sánchez-Martínez et al. 2007). Recently, the Mexican government stopped all shrimp imports from Asia due to a new pathogen (*Vibrio parahaemolyticus*) detected in those shrimps; which has produced high shrimp mortality in the Mexican Pacific region where the vast majority of shrimp farms are located. These bacteria are transmitted orally, colonizing the shrimp gastrointestinal tract, and producing toxins that cause tissue destruction and dysfunction of the hepatopancreas. This bacterial disease is called early mortality syndrome (EMS) or acute hepatopancreatic necrosis syndrome (AHPNS) (FAO 2013).

The future of shrimp farming in Mexico is relevant for shrimp production in the country. For that reason and according to previous experience where significant losses of shrimp production were due to several pathogens, some strategies have been established with the aim to obtain a healthy shrimp aquaculture industry. The strategies implement are based on animal health and technological elements, such as the use of SPF (specific pathogen free) and SPR (specific pathogen resistant) post larvae (Walker and Winton 2010). Mexico implemented this measure of diseases control in the mid-1990s after TSV outbreaks in shrimps from Hawaii, USA, France and Tahiti. To date, this biosecurity strategy has been complemented with a reduced water exchange, hazard analysis and critical control points (HACCP) systems, and the careful monitoring and management of shrimp culture (Fig. 5.12d). The Mexican government decreed a law project (NOM-062-ZOO-1999 and Project NOM-000-ZOO-2009) on aquatic animal health showing the relevance of using SPF organisms in shrimp production (Lightner 2005), with special attention to quarantine laws to prevent the import of exotic pathogens.

Shrimp farmers associations (SFA) in Mexico are highly communicated and organized (intra and inter-associations) allowing them to achieve common objectives for the shrimp aquaculture industry. They also have a close relationship with research groups to work in common areas of interest; this is especially true in the Mexican Pacific, which accounts for 97 % of total farmed shrimp (Fig. 5.10). This organizational arrangement has also enabled them to get financial support for technology to improve shrimp production, as well as to get better national and/or international prices. Currently, the SFA have been working on the issue of animal health in shrimp farms in real time with different laboratories focused on the diagnosis of shrimp diseases in Mexico, USA, and France.

5.5 Trends and Advances on the Interaction of Fishing and Aquaculture for Tuna Production

5.5.1 Situation of the Tuna Fishery and Wild Tuna Stocks in Mexico

Tuna represents the third most important marine resource in Mexico in terms of volume, after sardine and shrimp; but it is the second most economically important fishery resource, just after shrimp, reaching about US$86 million in 2011. The Mexican tuna fishery occurs along the Pacific Ocean and Gulf of Mexico; however, the Mexican Pacific is the most important region for tuna fishing in the country. In fact, the catch of tuna in the Mexican Pacific in 2011 was 106 thousand tonnes (107 tuna vessels) whereas in the Mexican Atlantic (Gulf of Mexico and the Caribbean

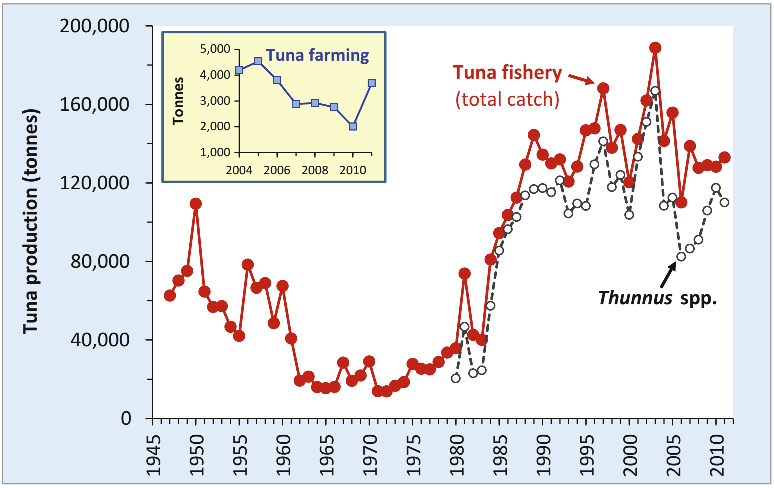


Fig. 5.13 Historical production of tuna in Mexico by the tuna fishery and aquaculture. Total catch of tuna includes skipjack tuna (*Katsuwonus pelamis*) and bonito (*Sarda* spp.). The catch of *Thunnus* spp. is also displayed separately (1980–2011) (Data from CONAPESCA (Mexico))

Sea) was 1.5 thousand tonnes (31 tuna vessels) (CONAPESCA 2011). This fishery on the Pacific side of Mexico is supported primarily by the catch of the yellowfin tuna (*Thunnus albacares*) accounting for about 80 % of total tuna catch, followed by skipjack tuna (*Katsuwonus pelamis*), bluefin tuna (*T. orientalis*) and bonito (*Sarda* spp.) among others (INAPESCA 2006), which are mainly caught with purse seine gears. In the Gulf of Mexico, the fishery is based on yellowfin tuna primarily caught with long-line fishing gears.

Tuna catch by Mexican vessels started in the 1950s in the Pacific Ocean reaching around 109 thousand tonnes; before this year the fishery was artisanal (INAPESCA 2006). However, after the following two decades tuna catches decreased to 13,825 tonnes in 1972 reaching its maximum in 2003 with 188,821 tonnes; but during the last 5 years (2007–2011) tuna catch has fluctuated around 130 thousand tonnes. Additionally, tuna farming has also contributed to tuna production in Mexico; however, its contribution in terms of volume has been much lower than from the tuna fishery. Tuna aquaculture production in Mexico was recorded in official statistics of CONAPESCA since the mid-2000s, and its production has fluctuated from 4,193 tonnes in 2004 to 3,689 tonnes in 2011 (Fig. 5.13).

There are three bluefin tuna species: Southern bluefin tuna (*T. maccoyii*), Atlantic and Mediterranean bluefin tuna (*T. thynnus*), and Northern Pacific bluefin tuna (*T. orientalis*); of these three species, the Northern Pacific bluefin tuna is currently farmed in Mexico. *Thunnus orientalis* is a highly migratory species, and its commercial value exceeds that of most other species of tunas (Kitagawa et al. 2007). This species also has the largest home range of any tuna of the genus *Thunnus* (Withlock et al. 2012).

Thunnus orientalis spawns in the North Pacific in the sea between Japan and the Philippines from April to June, south of Honshu Island in July, and in the Japan Sea in August (del Moral-Simanek and Vaca-Rodríguez 2009a); however, their reproductive biology is not well understood (Chen et al. 2006). Juvenile bluefin tuna remain in the Western Pacific during their first year of life (Kitagawa et al. 2007), and then, when the carrying capacity of the sea begins to saturate, part of the 2-year-old bluefin tuna population migrate to California and Baja California as a survival strategy, using the cold water of the Subarctic Frontal Zone (Kitagawa et al. 2009). These organisms constitute the tuna fishery found in the East Pacific from May to October, where they stay until they are 6 years old; no tuna larvae have been reported from the East Pacific, and it is surmised that the tuna return to Japan to spawn. Several studies have reported on the seasonal movements of pacific bluefin tuna in the Eastern North Pacific using archival tags, and they have revealed that in the summer, tuna are located primarily over the continental shelf of Baja California and migrate north to Central California in the fall, to return to Southern California by mid-winter (Kitagawa et al. 2007). In another study, electronic tagged bluefin tuna showed repeatable seasonal movements along the west coast of North America, and were found farthest south in the spring off southern Baja California, Mexico and farthest north in the fall when fish were found predominately off central and northern California (Boustany et al. 2010), concentrating in areas of high productivity, with dispersed fish in areas of low productivity.

There are commercial fisheries for Pacific bluefin tuna throughout their distribution range, from the Western to the Eastern Pacific Ocean (Withlock et al. 2012). Of the total population of Northern Pacific bluefin tuna, about 15 % is captured in the Eastern Pacific (California and Baja California), where due to its high value in the Japanese market, it is ranched and fattened for export to Japan (del Moral-Simanek et al. 2010). The Mexican fleet usually captures tuna from May to October (del Moral-Simanek and Vaca-Rodríguez 2009a).

There are other valuable tuna species that are valuable alternatives to bluefin tuna; these species are bigeye tuna (*T. obesus*) and yellowfin tuna (*T. albacares*), which have been farmed in the Cedros Island farm, located on the Mexican Pacific (Belle et al. 2003).

5.5.2 Tuna Culture in Mexico

Global demand for tuna has increased at a rapid rate, resulting in a downward trend in supplies of tuna, which in turn has resulted in increased catch restrictions by the Tuna Regional Fisheries Management Organization (RFMO); at the same time there has been an increase in tuna stock originating from farms (Ariji 2010).

The tuna industry in Mexico consisted principally in the capture of wild tuna for canning. However, after the 1990 US tuna embargo on Mexico, tuna captures stopped being one of the main export fisheries products, and revenue from tuna capture fell (del Moral-Simanek and Vaca-Rodríguez 2009a). Because of this, Baja

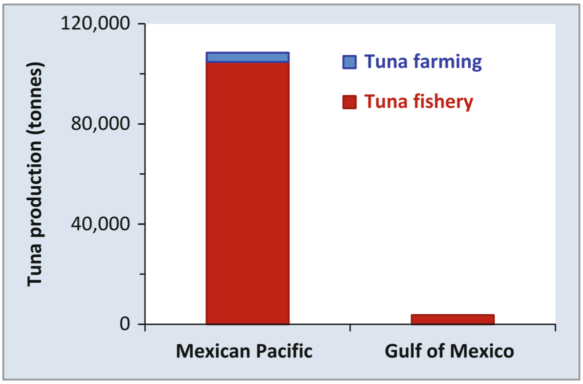


Fig. 5.14 Contribution of the Mexican Pacific and Gulf of Mexico to the national production of tuna in 2011. Production from fishing and aquaculture activities is presented separately (Data from CONAPESCA (Mexico))

California businessmen saw the opportunity to establish North Pacific bluefin tuna (*T. orientalis*) farms, with the purpose of capturing, farming and exporting tuna to the Japanese market (del Moral-Simanek et al. 2010).

Tuna farming operations in Mexico constitute about 3 % of the world total (Belle et al. 2003). Most tuna farming operations in Mexico are located on the Baja Peninsula, on the Pacific side. Tuna farming in Mexico started in 1996 (Belle et al. 2003) because of its temperate water, low labor costs, and abundant supply of local feed, especially sardines (*Sardinops sagax*) (del Moral-Simanek et al. 2010). The first bluefin tuna farm was established in Isla de Cedros, Baja California in 1997, with a total production of 64 tonnes of live tuna in 3 years (del Moral-Simanek et al. 2010) and a second farm, Maricultura del Norte, S de RL de CV in 1998. By 2003 there were already five tuna farms in operation, and by 2007 there were ten farms (del Moral-Simanek and Vaca-Rodríguez 2009a). From 1999 to 2003, the tuna production from farming increased from less than 50 to about 600 tonnes (Belle et al. 2003), and up to 4,535 tonnes in 2005 (CONAPESCA 2005) by the ten companies in operation (del Moral-Simanek and Vaca-Rodríguez 2009b). However, due to the world financial crisis, by 2009, there were only two farms in production, Baja Acuafarms and Maricultura del Norte (Anonymous 2009), which limited the number of captures. An economic reactivation in Baja California, Mexico, began in August 2010, when three more farms were opened and newly capture tuna were transported and caged (Anonymous 2010). By 2013, 79 % of farmed tuna in Tokyo was imported from Mexico, which exported from January to April 2013 about 7,000 tuna (Anonymous 2013). Tuna aquaculture in Mexico is only practiced on the Pacific side of the country; in fact, the Mexican Pacific accounts for almost 99 % of the total national production of tuna with 108 thousand tonnes (capture fishery and aquaculture) whereas catches of tuna from the Gulf of Mexico fleet accounts for 1.5 thousand tonnes (CONAPESCA 2011) (Fig. 5.14).

Tuna farming in Mexico is done by fattening wild caught tuna (del Moral-Simanek and Vaca-Rodríguez 2009a). The production cycle lasts from 3 to 8 months, depending on the fish size. When the fish are small they are usually held for longer periods of time. Water temperatures range from 18 to 22 °C and the cage systems

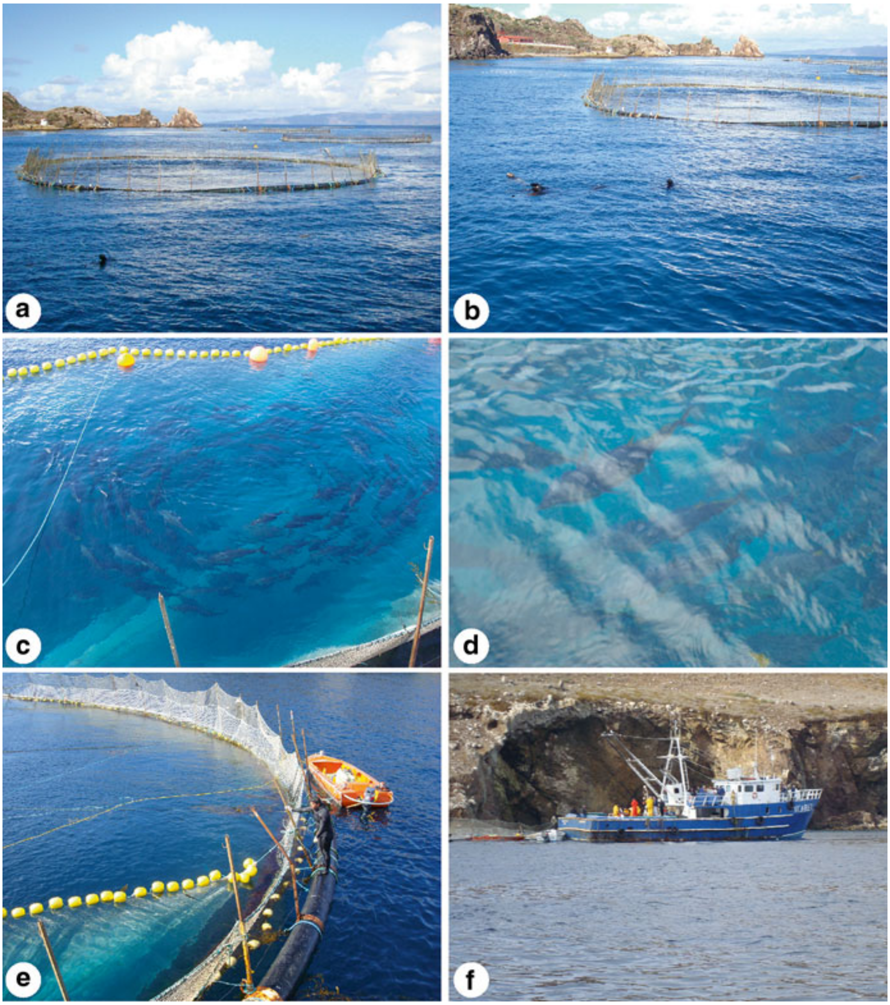


Fig. 5.15 Tuna farming on the Pacific Ocean in Baja California (Mexico): (**a**) circular cages for capture-based aquaculture of tuna located along the coast; (**b**) sea lions surrounding the cages attracted by the uneaten food; (**c**, **d**) bluefin tunas (*Thunnus orientalis*) swimming inside the cages which are fattened for several months; (**e**) tuna farmer overseeing the feeding of fish; (**f**) harvesting of tuna from the cages to be iced and transported to sashimi markets of Japan and USA (Photographs provided by Luz Estela Rodríguez-Ibarra)

are 40–50 m in diameter, 15–20 m deep with holding volumes of 18,000–20,000 m^3, with fish densities ranging from 2 to 5 kg/m^3. Fish are fed both fresh and frozen sardines, mackerel and squid. Weight gain ranges from 30–90 % of initial weight. Fish are harvested from December to April/May, using the Australian method (Belle et al. 2003) (see Fig. 5.15).

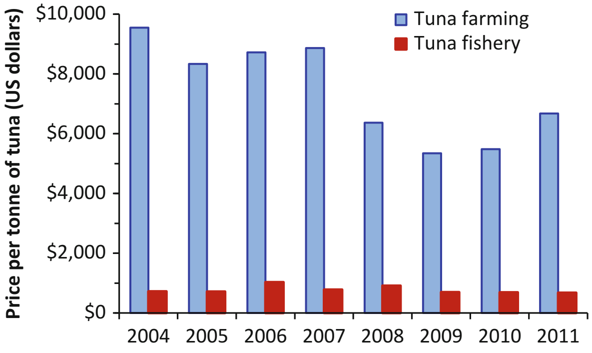


Fig. 5.16 Economic value of tuna production (price per tonne) from farming and fishing activities of tuna in Mexico. Farming is based exclusively on Pacific bluefin tuna (*Thunnus orientalis*) whereas the capture fishery targets mostly yellowfin tuna (*T. albacares*) (Data from CONAPESCA (Mexico))

A particular problem of tuna farming in Mexico is the presence of predators, in the form of sea lions, since many areas along the coastline contain large colonies, which are attracted to the tuna cages by the excess feed that falls through the cages and that is discarded. Therefore, stress and poor performance of fish has been a problem and is common in most farms. Tuna farmed in Mexico is generally smaller in size (Belle et al. 2003) (Fig. 5.15).

The primary market for tuna farmed in Mexico is Japan. However, the rapidly expanding US market for sashimi products has increased the number of exports from Mexico to the US. It is predicted that a larger percentage of farmed tuna from Mexico will be marketed in the US. Bluefin tuna farmed in Mexico is highly prized for its meat within the sashimi market, and consequently the price per tonne of the farmed tuna is about tenfold the price per tonne of tuna caught in the fishery. For instance, during 2004–2011 the price of 1 tonne of farmed tuna in Baja California (Mexico) was 7–13 times the price per tonne of tuna caught by the Mexican fleet (Fig. 5.16).

The increase in tuna farming in Mexico has also resulted in an increased demand of live feed to fatten the tuna in the form of sardines, which has resulted in an increased price per tonne, benefiting the sardine local fishery with prices of up to US$100–120 per tonne (del Moral-Simanek et al. 2010).

Capture-based aquaculture of bluefin tuna in Mexico is similar to that occurring in other parts of the world, and has been expanding rapidly but little is known about its environmental impact. Studies on Mediterranean tuna farms, have reported high nutrient concentrations at the cage station, however, monitoring of physico-chemical parameters, nutrients, and chlorophyll in the water column together with organic carbon in sediment did not show detectable impact of fattening of Atlantic bluefin tuna, probably caused by strong currents present in the area, water depth, controlled feeding, and periodic presence of tuna farming activity in the study area (Aksu et al. 2010). However, fish farms cause wild fish to aggregate nearby, but the spatial and temporal extent of the attraction effect around farms is still poorly understood (Bacher et al. 2012).

5.5.3 Future Considerations and Concerns

Tuna from the genus *Thunnus* are high commercial value species that until very recently could only be acquired by wild captures (Munday et al. 2003) which has resulted in the decline of tuna populations worldwide. Japan continues to be the most important market for tuna products, however the fishery productivity has decreased, especially for bluefin tuna (*T. orientalis*, *T. thynnus*), because of the strict capture limits implemented, which are needed to conserve the resources (Ariji 2010). Tuna production from farms or ranches has become increasingly significant, but, since these farms rely on natural young tuna as a raw material, they are also affected by the resource management efforts (Ariji 2010).

Complete tuna domestication may be a solution to stop the tuna population decline due to overexploitation, fortunately, a full-cycle farming technology for *T. orientalis* has recently been developed in Japan (Sawada et al. 2005; Ariji 2010); however, in Japan, there is a negative image of farmed products; therefore it is necessary to change the consumer perception.

Particularly, the reproductive biology of Pacific bluefin tuna (*T. orientalis*) appear to be poorly known when compared with other tunas (Juan-Jordá et al. 2013); while larval production technology is underdeveloped (Sawada et al. 2005). There are studies that report the reproductive stock parameters associated with this species; and it seems that condition factor decreases from late May to early June, and the sex ratio might be 1:1 for spawners, with a relatively high gonadosomatic index, which is markedly increased from late May to early June. Histological examination of oocytes revealed that all specimens were sexually mature at the start of the spawning activity which starts in May and peaks in late May to early June and fecundity increased with fork length, and preliminary estimates of spawning frequency between batches range 2–4.5 days based on analysis of postovulatory follicles (Chen et al. 2006).

Completion of the life cycle of Pacific bluefin tuna may open the possibility of genetic improvement by selecting useful biological traits such as growth, feed conversion rate, meat quality, disease resistance, resistance to environmental stress, etc. (Sawada et al. 2005). However, difficulties in maturation, spawning and seedling production of Pacific bluefin tuna still exist, and the key factors initiating spawning have not been clearly identified. Furthermore, there are high levels of cannibalism in tuna larvae, which needs to be suppressed to ensure high survival rates for practical larval and juvenile production of Pacific bluefin tuna (Sawada et al. 2005).

A concern in tuna farming is mercury accumulation. Studies on southern bluefin tuna (SBT) edible tissues showed that there is a reduction in the mercury concentration over a typical farming period of 136 days, due to growth dilution from 0.51 mg/kg down to 0.33 mg/kg. Culture beyond 136 days resulted in an increase in mercury concentration due to the combined effects of mercury accumulation and seasonal lipid depletion (Balshaw et al. 2008).

Intensive aquaculture is affected by disease epizootics and the introduction of parasites as a consequence of the transport of live fish (Guo and Woo 2009). Marine fish parasites are usually considered benign in the wild, except when they may impact its host's fecundity or induce their death (Jones 2005). However, in aquaculture settings, parasites are associated with high mortalities and reduced productivity, especially when these parasites are direct cycle (Hayward et al. 2007, 2008) and they propagate from fish to fish, reaching high infection rates due to the high density at which fish are kept.

Tuna are energetic pelagic fish with a remarkable migratory activity, with a unique physiology reflected in high metabolic rates. However, knowledge of microbial and environmental diseases of tuna is still limited (Di and Mladineo 2008). Many pathogens can be potentially dangerous for tuna of the genus *Thunnus*, both in natural populations, as those kept in captivity for aquaculture (Munday et al. 2003), although only a few have caused noticeable economic losses.

Among the different pathogens affecting tuna are viruses. There are two major viral diseases affecting Pacific bluefin tuna; juvenile production has often failed in Japan because of the occurrence of viral nervous necrosis (VNN) caused by betanoda viruses (Sugaya et al. 2009). The mortalities mostly occur at larval stages, and in some, but not all mortality cases, the diseased fish was characterized by vacuolation in the central nervous systems and retina. VNN is a major cause of larval mortality of Pacific bluefin tuna (Nishioka et al. 2010). Another virus that affects under 1 year old juveniles is the Red Sea Bream Iridovirus (RSIV) (Masuma et al. 2011), which causes high mortality at the grow-out phase (Munday et al. 2003).

Bacteria are another source of disease for tuna. Most bacterial diseases described for tuna, mainly *Aeromonas* sp. have been reported as secondary infections following skin damage caused by ectoparasites, such as *Caligus elongatus* (Munday et al. 2003). Infections with *Photobacterium damselae* subsp. *piscicida* have been associated with high mortalities of blue fin tuna in the Adriatic Sea (Mladineo et al. 2006). Tuna are also susceptible to *Mycobacterium* sp., which causes piscine tuberculosis (Munday et al. 2003).

Protozoan infections include Coccidia (*Goussia auxidis*) and Scuticociliates (*Uronema nigricans*), which cause significant mortalities in Pacific bluefin tuna larvae (Munday et al. 2003). However, most parasitic infections in tuna are due to metazoan infections, although just a few species are of economic importance. *Kudoa* sp. is a myxozoon that is capable of liquefying the fish muscle in *T. maccoyii* and *T. thynnus*, which results in high economic losses. There are also many monogenean species that infect tuna, affecting gills and skin. *Hexostoma* thynni is one of the most common monogenean species found in tuna, causing branchial hyperplasia, lamellar fusion and hemorrhages (Mele et al. 2010). Copepod crustacea are also another group of problematic parasites, with three potentially pathogenic species for tuna: *Caligus elongatus*, *Eyryphorus brachypterus* and *Penella filosa*, which cause external lesions, including branchial epithelia hyperplasia, lamellar fusion or hemorrhages (Mele et al. 2010). *Cardicola forsteri*, a digenean sanguinicolid blood fluke has been identified as a moderate risk for farmed southern bluefin tuna (Nowak 2004), although the fish are able to control de blook fluke infection on its own (Aiken et al. 2006) growth may be compromised. Furthermore, studies demonstrate the development of acquired resistance in fish against a parasite in an aquaculture environment under natural infection conditions (Aiken et al. 2008).

5.6 Conclusions

From the viewpoint of food security, the particular case of shrimp aquaculture in Mexico has been a very significant alternative to increase the shrimp production for human consumption, and perhaps it represents one of the few examples worldwide where aquaculture production of a native shrimp species (i.e. *Litopenaeus vannamei*) has practically doubled the national shrimp fishery production. According to official statistics of fishery landings in Mexico, the maximum level of shrimp caught by the fishing fleet was reached in a period of 40 years (1947–1987) with 83.8 thousand tonnes. However, it only took about 20 years for shrimp aquaculture industry to reach a similar production which was even almost doubled in the following 5 years up to 133 thousand tonnes of farmed shrimp.

Contrarily to the exponential development of shrimp aquaculture industry in the Mexican Pacific, the number of shrimp farms along the Gulf of Mexico is still incipient accounting only for 2.6 % of the total farmed shrimp production. The opportunity to increase the development of shrimp farming in the Gulf of Mexico is evident. However, any future expansion of shrimp aquaculture along the Gulf of Mexico coast should be made after a complete analysis of the potential environmental risks associated with this activity, which is based on the culture of *L. vannamei*, an exotic species from the Pacific Ocean. Recently, seven individuals of this exotic species were reported in a coastal lagoon on the Southern Gulf of Mexico; but, the establishment of a local population of this species has not been confirmed (Wakida-Kusunoki et al. 2011). Irrespective of this, the introduction of exotic species to the marine ecosystem of the Gulf of Mexico could have negative and unforeseen impacts to the biota. However, the lack of culture technologies for native shrimp species is also a limiting factor to avoid the use of *L. vannamei* in shrimp farms along the Gulf of Mexico.

Shrimp pond effluents may contribute to nutrient over-enrichment and eutrophication of coastal ecosystems (Páez-Osuna et al. 2003) and the residual waters from shrimp farms may also contain chemical products such as fertilizers, pesticides and antibiotics (Páez-Osuna 2001). In this regard, the rapid increase in the number of shrimp farms along the Pacific Ocean also increase the pollution risks to the coastal environment which have not been sufficiently documented excepting some particular cases.

Because of the capture shrimp fishery in Mexico is at its maximum level of exploitation exhibiting stagnating catches, coupled with the fact that in some regions this fishery resource is clearly depleted (Campeche Sound, Gulf of Mexico), shrimp aquaculture might be a potential alternative to maintain or increase the current shrimp production levels. However, aquaculture development should be supported by a most participative environmental research in order to avoid possible negative effects to coastal ecosystems and their resources.

On the other hand, the interaction between fishing and farming of tuna is very different to the case of the interactive contribution of the fishery and aquaculture of shrimp mentioned above. Tuna fishing is the main contributor to the total tuna production in Mexico with approximately 130 thousand tonnes per year, with a very little contribution of bluefin tuna (*Thunnus orientalis*) farming with around 3–4 thousand tonnes per year. Contrary to the advanced development regarding shrimp farming, capture-based aquaculture of tuna continues to prevail. This means that tuna farming depends entirely on wild captures of bluefin tuna to be transported and fattened in circular cages. Furthermore, this relatively new industry requires catching small pelagic, primarily sardines, which are utilized as feed for tunas. These two facts concerning tuna farming make it difficult to expect that tuna aquaculture may have an important contribution to national fish production, because it is directly dependent on the capture of wild tunas to be fattened and of small pelagics to be utilized as feed. Indeed, amount of bluefin tuna farmed accounts for only 3.4 % of the total tuna production in Mexico, a completely opposite situation to farmed shrimp production. However, the price of 1 tonne of farmed tuna is about 7–13 times that of tuna caught by the Mexican fleet. Therefore, capture-based aquaculture of bluefin tuna might not be a plausible option to increase the availability of fish for human consumption; but it is a good alternative as source of employment and incomes.

The occurrence of outbreaks of infectious diseases in aquatic organisms is one of the major threats to aquaculture production. Disease outbreaks have resulted in important losses in several countries around the world. In this regard, the use of SPF and SPR post larvae has been an important tool in shrimp aquaculture in Mexico as biosecurity strategy. However, knowledge of certain microbial diseases in farmed tuna is still limited. Thus, future research on this critical topic would be required.

In the case of Mexico, the combined contribution of fisheries and aquaculture to food security has allowed to reach seafood production and income levels that could not have been obtained from the fishing or aquaculture activities individually. Therefore, aquaculture should not be considered as a replacement of capture fisheries, but as a complementary productive activity to meet the growing human demand for food from the sea.

References

Aguirre-Guzmán G, Ascencio-Valle F (2000) Infectious disease in shrimp species with aquaculture potential. In: Pandalai SG (ed) Resent research developments in microbiology, vol 4. Research Signpost, Trivandrum, pp 333–348

Aiken HM, Hayward CJ, Nowak BF (2006) An epizootic and its decline of a blood fluke, Cardicola forsteri, in farmed southern bluefin tuna, *Thunnus maccoyii*. Aquaculture 254:40–45

Aiken HM, Hayward CJ, Crosbie P, Watts M, Nowak BF (2008) Serological evidence of an antibody response in farmed southern bluefin tuna naturally infected with the blood fluke *Cardicola forsteri*. Fish Shellfish Immunol 25:66–75

Aksu M, Kaymakci-Basaran A, Egemen O (2010) Long-term monitoring of the impact of a capture-based bluefin tuna aquaculture on water column nutrient levels in the Eastern Aegean Sea, Turkey. Environ Monit Assess 171:681–688

Anonymous (2009) Disminuye producción de atún en Ensenada. http://www.iaes.gob.mx/index.php?pag=m_blog&gad=detalle_entrada&entry=19

Anonymous (2010) Comienza Siembra de atún aleta azul en Baja California, México. Panorama Acuícola Noticias del día. http://www.panoramaacuicola.com/noticias/2010/08/23/comienza_siembra_de_atun_aleta_azul_en_baja_california_mexico.html

Anonymous (2013) México copó el 70% del atún de criadero llegado a Tokio. Dinero en Imagen Economia. http://www.dineroenimagen.com/2013-05-13/20107

Araneda M, Pérez EP, Gasca-Leyva E (2008) White shrimp *Penaeus vannamei* culture in freshwater at three densities: condition state based on length and weight. Aquaculture 283:13–18

Araneda ME, Hernández JM, Gasca-Leyva E, Vela MA (2013) Growth modelling including size heterogeneity: application to the intensive culture of white shrimp (*P. vannamei*) in freshwater. Aquacult Eng 56:1–12

Ariji M (2010) Conjoint analysis of consumer preference for bluefin tuna. Fish Sci 76:1023–1028

Arredondo-Figueroa JL (2002) El cultivo de camarón en México, actualidades y perspectivas. ContactoS 43:41–54

Bacher K, Gordoa A, Sague O (2012) Spatial and temporal extension of wild fish aggregations at *Sparus aurata* and *Thunnus thynnus* farms in the north-western Mediterranean. Aquacult Environ Interact 2:239–252

Balshaw S, Edwards JW, Ross KE, Ellis D, Padula DJ, Daughtry BJ (2008) Empirical models to identify mechanisms driving reductions in tissue mercury concentration during culture of farmed southern bluefin tuna *Thunnnus maccoyii*. Mar Pollut Bull 56:2009–2017

Bekker-Nielsen T (2005) Ancient fishing and fish processing in the Black Sea region. Aarhus University Press, Aarhus

Belle S, Smart A, Sylvia P (2003) Current status and future prospective of bluefin tuna (*Thunnus thynnus orientalis*) farming in Mexico and the West Coast of the United States. In: Bridges CR, García A, Gordin H (eds) Domestication of the bluefin tuna *Thunnus thynnus thynnus*. CIHEAM, Zaragoza

Boustany AM, Matteson R, Castleton M, Farwell C, Block BA (2010) Movements of pacific bluefin tuna (*Thunnus orientalis*) in the Eastern North Pacific revealed with archival tags. Prog Oceanogr 86:94–104

Brander KM (2007) Global fish production and climate change. Proc Natl Acad Sci U S A 104:19709–19714

Castilla JC, Defeo O (2001) Latin American benthic shellfisheries: emphasis on co-management and experimental practices. Rev Fish Biol Fish 11:1–30

Chen K-S, Crone P, Hsu C-C (2006) Reproductive biology of female Pacific bluefin tuna *Thunnus orientalis* from south-western North Pacific Ocean. Fish Sci 72:985–994

Cifuentes-Lemus JL, Torres-García MP, Frías-Mondragón M (1995) El océano y sus recursos, IX. La pesca, 2nd edn. Fondo de Cultura Económica, Mexico City

CONAPESCA (2005) Anuario estadístico de acuacultura y pesca 2005. SAGARPA, CONAPESCA, Mazatlán

CONAPESCA (2011) Anuario estadístico de acuacultura y pesca 2011. SAGARPA. CONAPESCA, Mazatlán

del Moral-Simanek RJ, Vaca-Rodríguez JG (2009a) Administración de la pesquería del atún aleta azul en Baja California: Una visión global. Front Norte 21:151–175

del Moral-Simanek RJ, Vaca-Rodríguez JG (2009b) Captura de atún aleta azul en Baja California, México: ¿pesquería regional o maquiladora marina? Región y Sociedad XXI:159–190

del Moral-Simanek RJ, Vaca-Rodríguez JG, Alcalá Alvarez MC (2010) Análisis socioeconómico e interrelación de las pesquerías de sardina y atún aleta azul en la región noroeste de México. Región y Sociedad XXII:9–29

DeWalt BR, Zavala JRR, Noriega L, González RE (2002) Shrimp aquaculture, the people and the environment in coastal Mexico. Report prepared under the World Bank, NACA, WWF and

FAO Consortium program on shrimp farming and the environment. The World Bank, NACA, WWF, FAO, Washington
Di MA, Mladineo I (2008) Ultrastructure of *Didymocystis semiglobularis* (Didymozoidae, Digenea) cysts in the gills of Pacific bluefin tuna (*Thunnus orientalis*). Parasitol Res 103:641–647
FAO (2011) Aquaculture development. 6. Use of wild fishery resources for capture based aquaculture. FAO technical guidelines for responsible fisheries. No. 5, Suppl. 6. FAO, Rome
FAO (2012) The state of world fisheries and aquaculture 2012. FAO, Rome
FAO (2013) Report of the FAO/MARD technical workshop on early mortality syndrome (EMS) or acute hepatopancreatic necrosis syndrome (AHPNS) of cultured shrimp (under TCP/VIE/3304). Hanoi, Vietnam, on 25–27 June 2013. FAO Fisheries and Aquaculture Report No. 1053. FAO, Rome
Ferreira JG, Hawkins AJS, Bricker SB (2007) Management of productivity, environmental effects and profitability of shellfish aquaculture—the farm aquaculture resource management (FARM) model. Aquaculture 264:160–174
Gartside DF, Kirkegaard IR (2009) A history of fishing. In: Equires VR (ed) The role of food, agriculture, forestry, and fisheries in human nutrition, vol 2. EOLSS Publishers, Paris, pp 105–139
Guo FC, Woo PTK (2009) Selected parasitosis in cultured and wild fish. Vet Parasitol 163:207–216
Gutiérrez-Salazar GJ, Molina-Garza ZJ, Hernández-Acosta M, García-Salas JA, Mercado-Hernández R, Galaviz-Silva L (2011) Pathogens in Pacific white shrimp (*Litopenaeus vannamei* Boone, 1931) and their relationship with physicochemical parameters in three different culture systems in Tamaulipas, Mexico. Aquaculture 321:34–40
Hayward CJ, Aiken HM, Nowak BF (2007) Metazoan parasites on gills of Southern Bluefin Tuna (*Thunnus maccoyii*) do not rapidly proliferate after transfer to sea cages. Aquaculture 262:10–16
Hayward CJ, Aiken HM, Nowak BF (2008) Epizootics of metazoan gill parasites did not threaten feasibility of farming southern bluefin tuna (*Thunnus maccoyii*) in a trial extending over summer months. Vet Parasitol 154:122–128
Horowitz A, Samocha TM, Gandy RL, Horowitz S (2001) Toxicity tests to assess the effect of a synthetic tank liner on shrimp survival and nitrification in a recirculating superintensive production system. Aquacult Eng 24:91–105
INAPESCA (2006) Sustentabilidad y pesca responsable en México. Evaluación y manejo. SAGARPA. INAPESCA, México
Jones B (2005) Mass mortalities in the oceans. In: Rohde K (ed) Marine parasitology. CSIRO, Canberra, pp 371–374
Juan-Jordá MJ, Mosqueira I, Freire J, Dulvy NK (2013) The conservation and management of tunas and their relatives: setting life history research priorities. PLoS ONE 8:e70405
Kitagawa T, Boustany AM, Farwell CJ, Williams TD, Castleton MR, Block BA (2007) Horizontal and vertical movements of juvenile bluefin tuna (*Thunnus orientalis*) in relation to seasons and oceanographic conditions in the eastern Pacific Ocean. Fish Oceanogr 16:409–421
Kitagawa T, Kimura S, Nakata H, Yamada H, Nitta A, Sasai Y, Sasaki H (2009) Immature Pacific bluefin tuna, *Thunnus orientalis*, utilizes cold waters in the Subarctic Frontal Zone for trans-Pacific migration. Environ Biol Fish 84:193–196
Larsen L, Roney JM (2013) Farmed fish production overtakes beef. Plan B updates. June 12, 2013. Earth Policy Institute. http://www.earth-policy.org/plan_b_updates/2013/update114. Accessed 25 Nov 2013
Lightner DV (1999) The penaeid shrimp viral pandemics due to IHHNV, WSSV, TSV and YHV: current status in the Americas, available diagnostic methods, and management strategies. J Appl Aquac 9:27–52
Lightner DV (2005) Biosecurity in shrimp farming: pathogen exclusion through use of SPF stock and routine surveillance. J World Aquacult Soc 36:229–248

Ludwig D, Hilborn R, Walters C (1993) Uncertainty, resource exploitation, and conservation: lessons from history. Science 260:17–36

Martínez-Córdova LR (1999) Cultivo de camarones peneidos: principios y prácticas. AGT Editor, México City

Martínez-Córdova LR, Campaña-Torres A, Martínez-Porchas M (2004) Manejo de la productividad natural en el cultivo del camarón. In: Cruz-Suárez LE, Ricque-Marie D, Nieto-López MG, Villarreal D, Scholz U, González M (eds) Avances en nutrición acuícola VII. Memorias del VII simposium internacional de nutrición acuícola, Hermosillo, 2004. Universidad Autónoma de Nuevo León. Monterrey

Masuma S, Takebe T, Sakakura Y (2011) A review of the broodstock management and larviculture of the Pacific northern bluefin tuna in Japan. Aquaculture 315:2–8

Mele S, Merella P, Macias D, Gomez MJ, Garippa G, Alemany F (2010) Metazoan gill parasites of wild albacore *Thunnus alalunga* (Bonaterre, 1788) from the Balearic Sea (western Mediterranean) and their use as biological tags. Fish Res 102:305–310

Miranda I, Valles JL, Sánchez R, Álvarez Z (2010) Culture of marine shrimp *Litopenaeus vannamei* (BOONE, 1931) in freshwater. Rev Cien FCV-LUZ XX:339–346

Mladineo I, Miletic I, Bocina I (2006) *Photobacterium damselae* subsp. *piscicida* outbreak in cage-reared Atlantic bluefin tuna *Thunnus thynnus*. J Aquat Anim Health 18:51–54

Munday BL, Sawada Y, Cribb T, Hayward CJ (2003) Diseases of tunas, *Thunnus* spp. J Fish Dis 26:187–206

Nishioka T, Mori K, Sugaya T, Tezuka N, Takebe T, Imaizumi H, Kumon K, Masuma S, Nakai T (2010) Involvement of viral nervous necrosis in larval mortality of hatchery-reared pacific bluefin tuna *Thunnus orientalis*. Fish Pathol 45:69–72

Nowak BF (2004) Assessment of health risks to southern bluefin tuna under current culture conditions. Bull Eur Assoc Fish Pathol 24:45–51

Ortega SC (2012) Veterinary medical education and veterinary involvement in aquatic-animal health and aquaculture in Mexico. J Vet Med Educ 39:195–199

Ortega-Salas AA, Rendón MLA (2013) Hyper-intensive farming of white shrimp *Litopenaeus vannamei* in a freshwater tank under semi-controlled conditions (Decapoda: Penaeidae). Cuad Invest UNED 5:63–68

Páez-Osuna F (2001) The environmental impact of shrimp aquaculture: causes, effects, and mitigating alternatives. Environ Manag 28:131–140

Páez-Osuna F, Gracia A, Flores-Verdugo F, Lyle-Fritch LP, Alonso-Rodriguez R, Roque A, Ruiz-Fernández AC (2003) Shrimp aquaculture development and the environment in the Gulf of California ecoregion. Mar Pollut Bull 46:806–815

Pauly D, Christensen V, Dalsgaard J, Froese R, Torres F Jr (1998) Fishing down marine food webs. Science 279:860–863

Pauly D, Christensen V, Guénette S, Pitcher TJ, Sumaila UR, Walters CJ, Watson R, Zeller D (2002) Towards sustainability in world fisheries. Nature 418:689–695

Pauly D, Watson R, Alder J (2005) Global trends in world fisheries: impacts on marine ecosystems and food security. Philos Trans R Soc B 360:5–12

Rabanal HR (1988) History of aquaculture. ASEAN/UNDP/FAO Regional small-scale coastal fisheries development project. ASEAN/SF/88/Tech. 7. ASEAN/UNDP/FAO, Manila

Ramírez-Rodríguez M, Chávez EA, Arreguín-Sánchez F (2000) Perspective of the pink shrimp (*Farfantepenaeus duorarum* Burkenroad) fishery of Campeche Bank, Mexico. Cienc Mar 26:97–112

Rojas AA, Haws MC, Cabanillas JA (eds) (2005) Buenas prácticas de manejo para el cultivo de camarón. Prácticas de desarrollo sostenible en ambientes costeros de prioridad de los ecosistemas del Golfo de California camaronicultura. The David and Lucile Packard Foundation. United States Agency for International Development (Cooperative Agreement No. PCE-A-00-95- 0030-05)

SAGARPA (2012) Actualización de la carta nacional acuícola. Secretaría de Agricultura, Ganadería, Desarrollo Rural, Pesca y Alimentación. Diario Oficial. pp 33–112, 6 June 2012

Samocha TM (2011a) The effect of different synthetic polymer liners in tank-systems stocked with Litopenaeus vannamei. Panor Acuic Mag 17:11–15

Samocha TM (2011b) The effect of different synthetic polymer liners on growth, survival and selected water quality indicators in tank-system stocked with *Litopenaeus vannamei* under limited water discharge and in the presence of natural productivity. Panor Acuic Mag 17:16–21

Sánchez-Martínez JG, Aguirre-Guzmán G, Mejía-Ruíz H (2007) White spot syndrome virus (WSSV) in cultured shrimp: a review. Aquacult Res 38:1339–1354

Sawada Y, Okada T, Miyashita S, Murata O, Kumai H (2005) Completion of the Pacific bluefin tuna *Thunnus orientalis* (Temminck et Schlegel) life cycle. Aquac Res 36:413–421

Secretaría de Pesca (1993) Anuario estadístico de pesca 1990. Secretaría de Pesca, México City

SEMARNAP (1996) Anuario estadístico de pesca 1995. SEMARNAP, México City

Squires VR (2009) The role of food, agriculture, forestry, and fisheries in human nutrition. In: Equires VR (ed) The role of food, agriculture, forestry, and fisheries in human nutrition, vol 1. EOLSS Publishers, Paris, pp 1–28

Sugaya T, Mori K, Nishioka T, Masuma S, Oka M, Mushiake K, Okinaka Y, Nakai T (2009) Genetic heterogeneity of betanodaviruses in juvenile production trials of Pacific bluefin tuna, *Thunnus orientalis* (Temminck & Schlegel). J Fish Dis 32:815–823

Treece GD (2000) Shrimp maturation and spawning. In: Tamaru CCT, Tamaru CS, McVey JP, Ikuta K (eds) Spawning and maturation of aquaculture species. US-Japan Natural Resources Technical Report No. 28. University of Hawaii Sea Grant College Program, Hawaii, pp 121–133

Treece GD (2001) Fertilizacion. In: Boyd CE, Haws MC (eds) Métodos para mejorar la camaronicultura en Centroamérica. UCA, Managua, pp 93–106

United Nations (2013) World population prospects. The 2012 revision. Highlights and advance tables. Department of Economic and Social Affairs, Population Division. Working Paper No. ESA/P/WP. United Nations, New York

Wakida-Kusunoki AT, Amador-del Angel LE, Carrillo P, Quiroga C (2011) Presence of Pacific white shrimp *Litopenaeus vannamei* (Boone, 1931) in the Southern Gulf of Mexico. Aquat Invasions 6:139–142

Walker PJ, Winton JR (2010) Emerging viral diseases of fish and shrimp. Vet Res 41:51–75

Withlock RE, McAllister MK, Block BA (2012) Estimating fishing and natural mortality rates for Pacific bluefin tuna (*Thunnus orientalis*) using electronic tagging data. Fish Res 119–120:115–127

Ye Y, Cochrane K (2011) Global overview of marine fishery resources. In: FAO (ed) Review of the state of world marine fishery resources. FAO fisheries and aquaculture technical paper, no. 569. FAO, Rome, pp 3–18

Part III
Exploration and Management of Coastal Karst

Chapter 6
Advances in the Exploration and Management of Coastal Karst in the Caribbean

Michael J. Lace

Abstract Coastal karst is a dominant landform within the Caribbean that has played a significant role in shaping human migration and settlement patterns, modern economic development in the region and its evolving biodiversity. Recent exploration of coastal karst, associated models of littoral cave development and comparative coastal resource management strategies are examined in three distinct island settings in the Caribbean: The Republic of Haiti, Grenada and Isla de Mona (Puerto Rico). These regional island examples display distinct and complex coastal karst morphologies ranging from intensely karstic carbonate platforms to complex non-carbonate island cores overlain by extensive and thickly-bedded coastal and interior carbonate cover or predominantly volcanic landscapes with limited fringing carbonates. Previously unreported field research in these selected island settings support an emerging view of the complex karst development in the region and indicate that significant karst areas remain to be explored while illustrating the associated landform vulnerabilities, anthropogenic effects and range of coastal resource management and preservation initiatives applied to date.

6.1 Introduction

The Caribbean Sea defines an area of approximately 2.6 million km^2 with over 55,000 km of coastline and more than half of its total associated landmass is predominantly carbonate (Day 2010; Stanley 1995). Similar to the assessment of marine and terrestrial biodiversity in the region (Lutz and Ginsburg 2007; Miloslavich et al. 2010), thorough exploration and study of coastal karst in the Caribbean remains markedly incomplete even though recent research efforts have spurred advances in developing modern models of coastal speleogenesis

M.J. Lace (✉)
Coastal Cave Survey, 313 ½ West Main St., West Branch, IA 52358, USA
e-mail: Michael-lace@uiowa.edu

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8,
DOI 10.1007/978-3-319-06305-8_6,

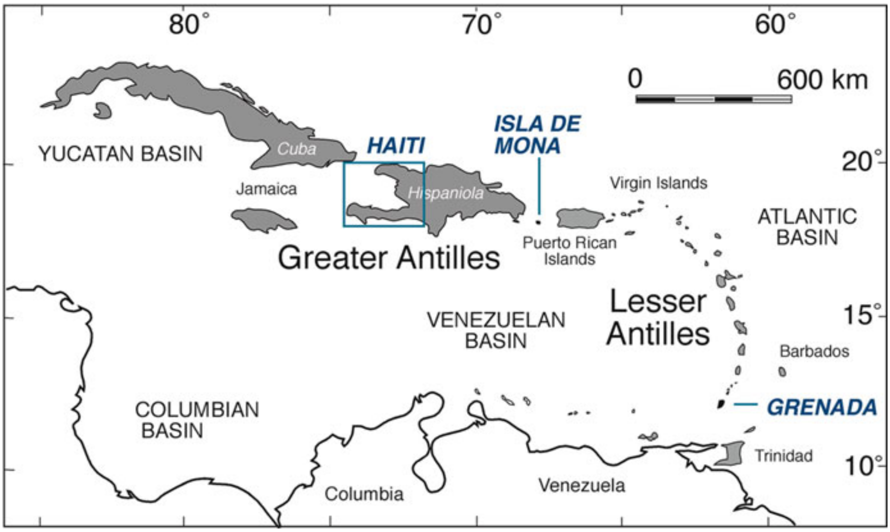


Fig. 6.1 Regional study area

(Mylroie and Mylroie 2013). Perceptions of the inherent values of coastal karst vary widely and encompass a diverse range of attributes specific to these complex landforms, including geological, biological, cultural and economic resources. Three contrasting case studies serve to illustrate a range of coastal karst management approaches that hinge upon recent exploration, inventory and study of cave and karst development on these geographically and geomorphologically diverse island landscapes (Fig. 6.1). The potential role coastal caves play in shaping sustainable management of coastal zones at risk is also examined in each setting.

Field research applied to all three settings in this communication included the generation of detailed site maps (Lace 2008), associated resource inventories and specific management assessments consistent with karst resource monitoring approaches (Palmer 2007; Toomey 2009). No sampling of geological, biological or cultural materials at any cave sites was conducted as the preliminary site assessments (many reported for the first time) were designed to lay the initial groundwork, within the context of a broader coastal cave and karst database, to support more detailed studies by future researchers in multiple fields.

6.2 Coastal Karst of the Republic of Haiti: Exploration, Research and Resource Management

The Republic of Haiti (19 00N, 72 25W) is one of the least understood karst areas in the Caribbean. It is one of the most mountainous islands in the region with elevations reaching 2,700 m (asl), encompassing an area of 27,500 km^2 (Fig. 6.2). Some of its rugged interior still remains inaccessible other than by foot or pack animal.

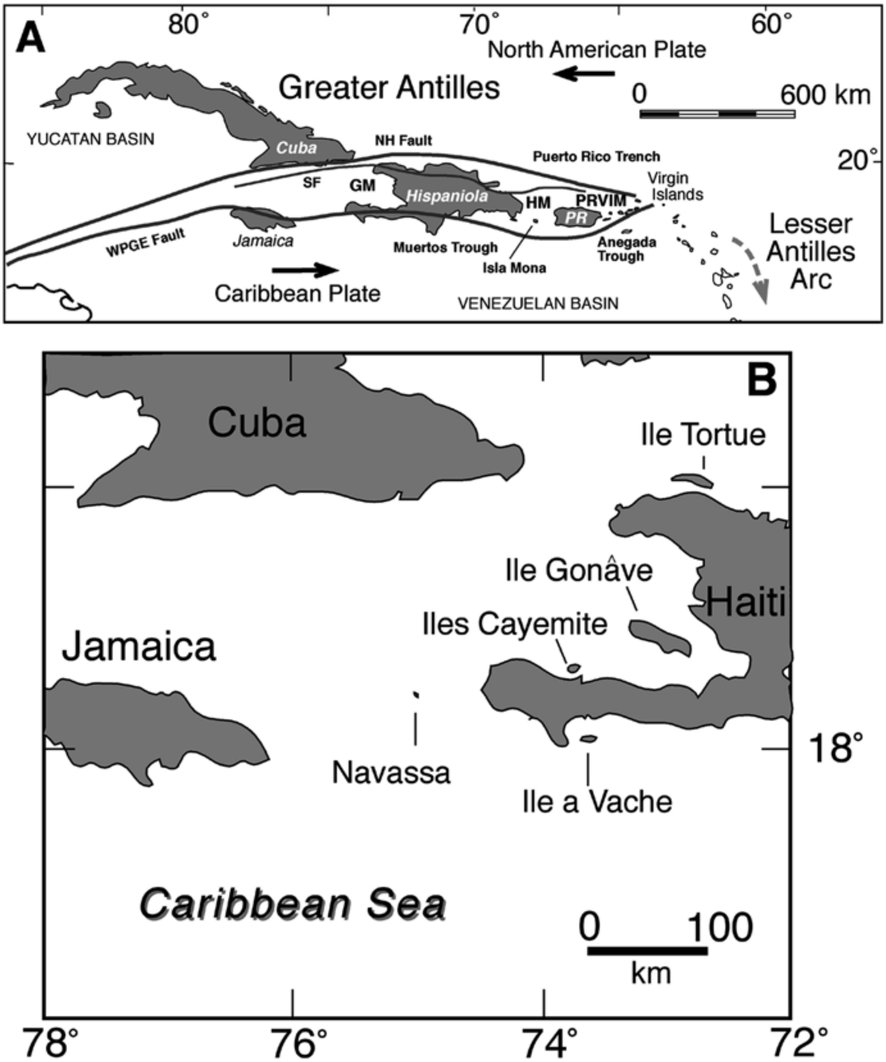


Fig. 6.2 (**a**) Tectonic setting of Hispaniola and the Puerto Rican Islands, where the Gonave (*GM*), Hispaniola (*HM*) and Puerto Rico-Virgin Islands (*PRVIM*) microplates are bounded by the *WPGE* Walton-Plantain Garden-Enriquillo, *NH* North Hispaniola and *SF* Septentrional faults (Adapted from Cotilla 2007) (**b**) Map of Haiti and associated offshore islands

However, Haiti is one of the most densely populated landmasses in the western hemisphere with approximately 9.5 million inhabitants. Such a population density on a landscape which is estimated to be more than 70 % karst carries stark implications for groundwater quality and clear complications to the dynamic balance between sustainable long-term development and preservation of one of the most fragile and biologically diverse terrains in the region.

Fig. 6.3 Coastal geomorphologies in Haiti. Embayments and cave development on the (**a**) north coast and (**b**) south coast of the southern peninsula. (**c**) Karstic Pleistocene-aged coastline in the Gulf of Gonave. (**d**) Cave and arch development (Jeremie area) and (**e**) beach development on the coastal plain, southern Peninsula

6.2.1 Geographic and Geologic Settings

The Republic of Haiti occupies the western third of the Island of Hispaniola and contains 1,800 km of coastline exhibiting a variety of morphologies, including extensive karst areas (Fig. 6.3). Haiti also has several significant offshore islands, most of which exhibit degrees of karst development (Figs. 6.2b and 6.4). Inland karst areas are

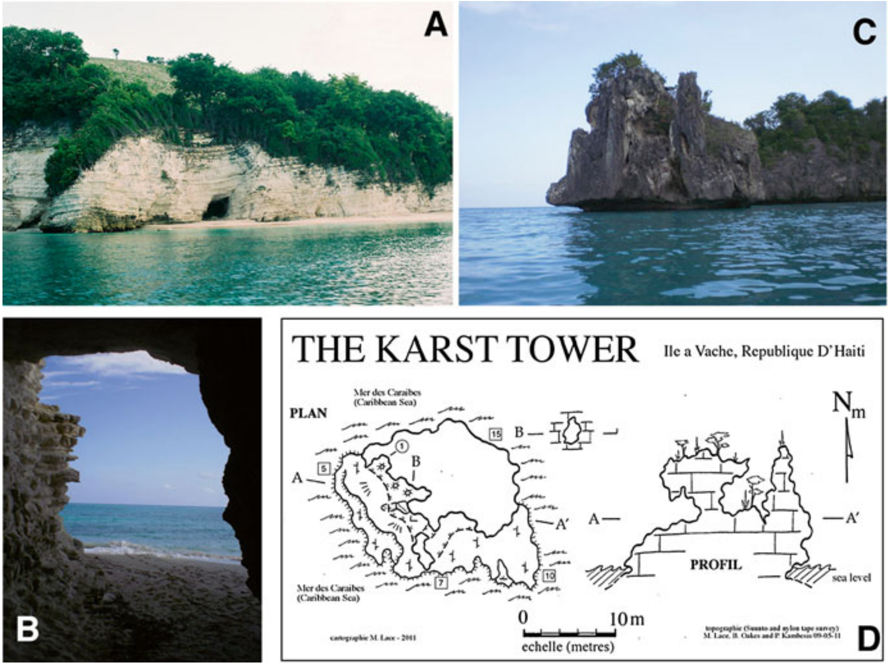


Fig. 6.4 Coastal karst examples from Ile a Vache, Haiti. (**a**, **b**) Sea cave in Abaka Bay. (**c**, **d**) offshore karst pinnacle exhibiting remnant of flank margin cave development

similarly extensive and complex encompassing a range of lithologies associated with mature epikarst and numerous examples of classical continental cave and karst development. Though periodic geologic studies have been conducted across the island, coastal landforms in Haiti have remained the least studied in the Caribbean (Florentin and Maurasse 1982; Hadden and Minson 2010). Coastal geomorphologies include well-developed embayments, estuaries, extensive coastal plains and cliffs as well as some of the best-preserved uplifted Pleistocene-aged terraces in the region (Dodge et al. 1983) (Fig. 6.5). Recent systematic survey efforts, in collaboration with the Haitian Speleological Survey, designed to document the coastal karst resources have revealed a complex inventory of cave types shaped and repeatedly modified by a range of coastal and continental karst processes coupled with changing sea levels within one of the most dynamic tectonic regimes in the Caribbean (Cotilla et al. 2007; Mann et al. 1991) (Fig. 6.2a).

6.2.2 *Cave and Karst Development*

Beginning in 2007, detailed field exploration and study of Haiti's caves and karst areas has been conducted by Haitian and American participants as part of an ongoing collaborative effort between private landowners, university and community

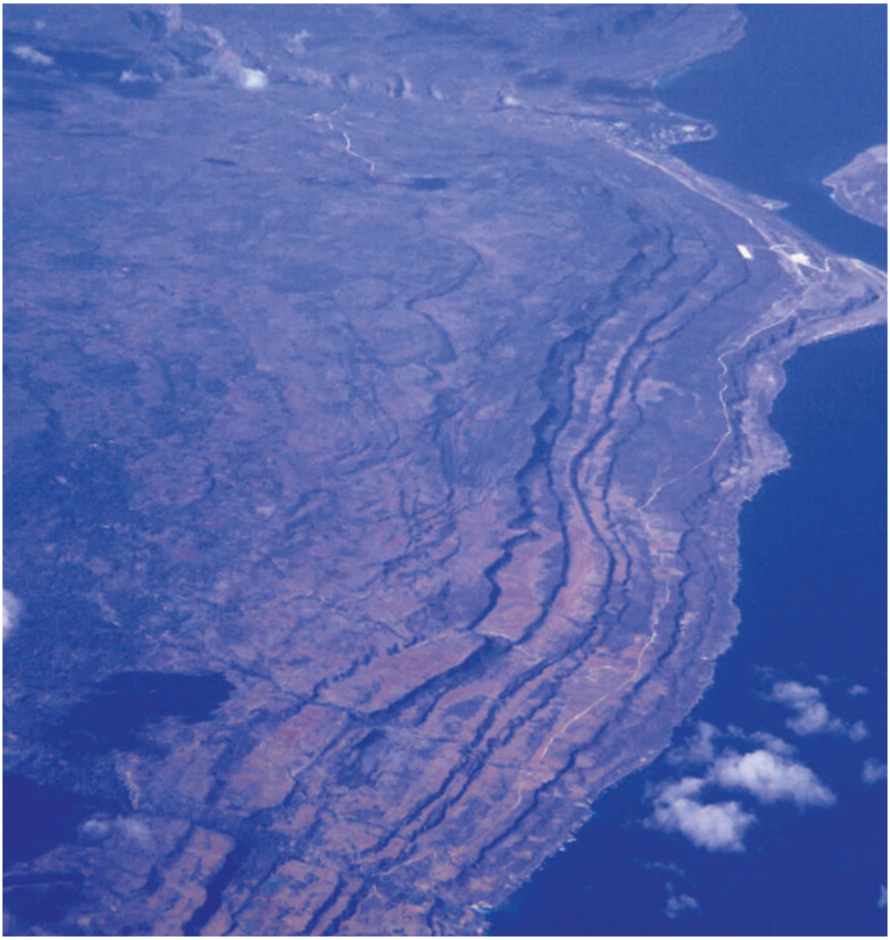


Fig. 6.5 Complex network of uplifted Pleistocene reef terraces, Northern Peninsula, Haiti

leaders, and government ministries on a countrywide scale. Over 250 karst and pseudokarst caves have been recorded to date. Karst cave examples include fluvial caves, flank margin caves (formed by mixing zone dissolution), Pseudokarst cave examples in Haiti include tufa caves (i.e., constructional caves formed by rapid carbonate deposition), sea caves (erosional structures formed by wave energy) fissures, talus caves (cavities within rock collapse), man-made caves (from limestone quarrying) and tafoni (cavities formed by salt air weathering). Grotte Marie-Jeanne (Port-a-Piment) is currently the largest surveyed cave in Haiti, defined by 5 km of passage reaching a maximum depth of 102 m within a coastal ridge (Figs. 6.7 and 6.9), yet the potential for the discovery of more extensive caves remains.

Significant exploration and documentation is required to define the clearly complex patterns of littoral and continental karst expression. While a comprehensive model of coastal cave and karst development in Haiti is still under construction, preliminary data demonstrates that field research in these complex and stunning littoral landscapes has enormous potential to contribute to a modern collective understanding of karst mechanisms in the Caribbean with direct implications to emerging long-term coastal preservation strategies.

6.2.3 Coastal Resource Management: Opportunities and Potential Pitfalls

Long-term anthropogenic impacts to interior and coastal ecosystems in Haiti have been significant and continue to present formidable challenges to resource management. Though significant coastal environments still remain intact, in many coastal zones population densities coupled with limited infrastructure development place-significant pressures on coastal resources, the effects of which are manifested in a variety of forms (E.I. 2012). Coastal land uses include: agriculture on varying scales, logging, fishing, salt production, mining (primarily limestone for construction) and infrastructure and housing development, with a limited tourism component to date (Fig. 6.8). However, not all projected large-scale tourism development appears to be consistent with models of sustainable management of natural coastal settings and preservation of both natural and cultural resources – a pattern that persists in many insular and continental coastlines worldwide.

6.2.3.1 Landscape Vulnerabilities and Coastal Watershed Management

Coastal areas in Haiti feature complex geomorphologies and equally complex management issues. While integrated coastal resource management approaches have seen limited success elsewhere in the Caribbean, they have yet to be widely applied to the karstic shorelines of Haiti. Thus, both longstanding and emerging land use issues continue to negatively impact landform stabilities, water availability/quality, economic development and resource preservation on a large scale.

Water quality and availability are persistent issues in coastal communities with karst areas prone to aquifer contamination (Lace and Mylroie 2013; Wampler and Sisson 2011). Watershed resources vary widely across the current as a function of its complex hydrogeology (Troester and Turvey 2004; USACE 1999). Significant karstic estuaries are used a primary water sources, for example in the southern peninsula community of Labaye (Fig. 6.6). Storm event vulnerabilities are also apparent with flooding effects to infrastructure, habitation, agriculture and more importantly recurrent loss of lives in coastal communities during hurricane events. Tectonic hazards (earthquakes, tsunami, terrestrial landslides) also pose significant collapse risks in coastal karst zones (Pikelj and Juracic 2013).

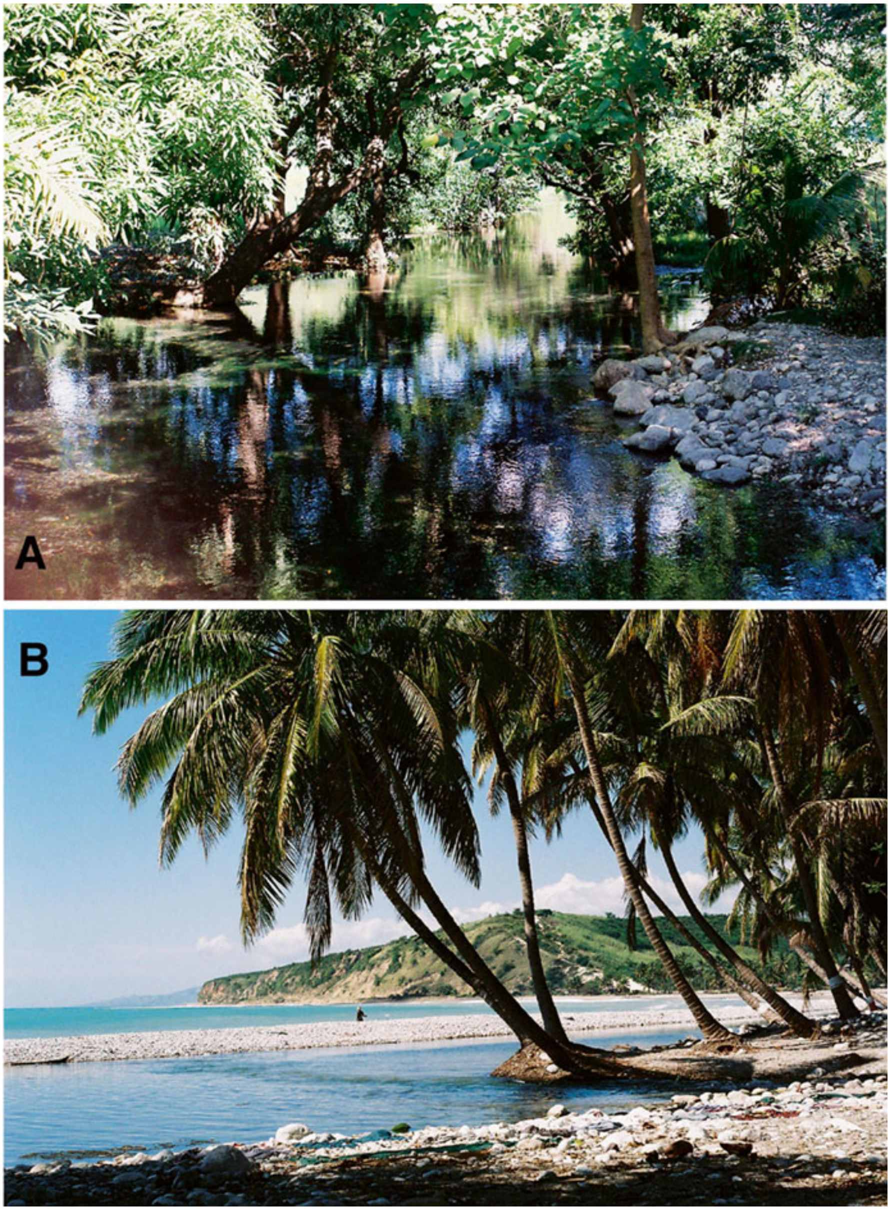


Fig. 6.6 Karstic coastal barrier estuary (Labaye, southern Peninsula)

Such vulnerabilities of karst landscapes in this area of the Caribbean are not unique to Haiti, for example, nearby Jamaica (two thirds of which is karstic terrain) is subject to the same risks of expanding anthropogenic influences (Day 2007). The accompanying case studies similarly illustrate the range of landform vulnerabilities in the region and contrasting sustainable management models. Recent coastal management initiatives also include the designation of the first Marine Protected Area in Haiti in 2013, encompassing a southwestern coastal zone associated with the mainland shore areas of Port Salut/Aquin as well as the offshore island of Ile a Vache (Fig. 6.2).

6.2.3.2 Cultural Resource Preservation and Haitian Karst

Archeology in the Caribbean is a complex and dynamic field of research, linking multiple cultural periods with diverse physical landscapes (Wilson 2007). Hispaniola occupies a pivotal position in regional models of cultural evolution but sporadic fieldwork has only begun to document the extent of cultural resources in Haiti (Rouse and Moore 1985). Preservation of cultural heritage in the Caribbean has seen renewed interest in recent years (Siegel and Righter 2011), yet the Republic of Haiti has been conspicuously absent in such discussions even though its rich cultural resources derived from its archaic, historical and modern periods, stand at significant immediate and long-term risk.

Geotourism has emerged as an expanding development pattern on a global scale, carrying with it controversial incumbent risks to resource preservation frequently at odds with the benefits of economic growth (Dowling 2011). As in any karst setting, a delicate balance must be struck between access to the resource and the incumbent impacts of varying degrees of development and visitation. In Haiti's southern peninsula, for example, a sustainable regional ecotourism plan can be further developed using the initial community-based development of Grotte Marie-Jeanne as a focal point for a selection of area destinations, as demonstrated in other settings (Pavlovich 2002; Timcak et al. 2010).

Grotte Marie-Jeanne (Port-a-Piment) presents an intriguing test case for sustainable eco-tourism development in the southern peninsula that is applicable to other karst settings across Haiti. Consistent with community-based approaches applied elsewhere (Hobbs 2004), integration of a broad a cross-section of the local populace both in development and long term management phases of the Grotte Marie-Jeanne project potentially enhances local economic growth while increasing community investment in the long term preservation of the resource. Consistent with initial assessments of the potential for the project (Kambesis et al. 2010), recent developments include improved access to the cave, construction of an interpretive center, development of a low-impact trail within the accessible entry segment of the cave system (Fig. 6.7) and a management plan supporting limited tours by trained local guides and appropriate safety equipment. The project involves support from the community of Port-a-Piment, the UNEP and the Haitian Ministry of Tourism. As a direct result of detailed resource assessments and recently implemented improvements, the land above the cave has now been designated a protected area, which will ideally promote long-term reforestation plans to enhance area biodiversity and promote slope stabilization of the ridge harboring Grotte Marie-Jeanne and associated caves. Grotte Marie-Jeanne forms an integral part of a long range stewardship plan consistent with current karst resource management approaches (Cigna 1993; Jones et al. 2003; Kambesis et al. 2013) and previously proposed resource management initiatives in Haiti (Woods and Harris 1986).

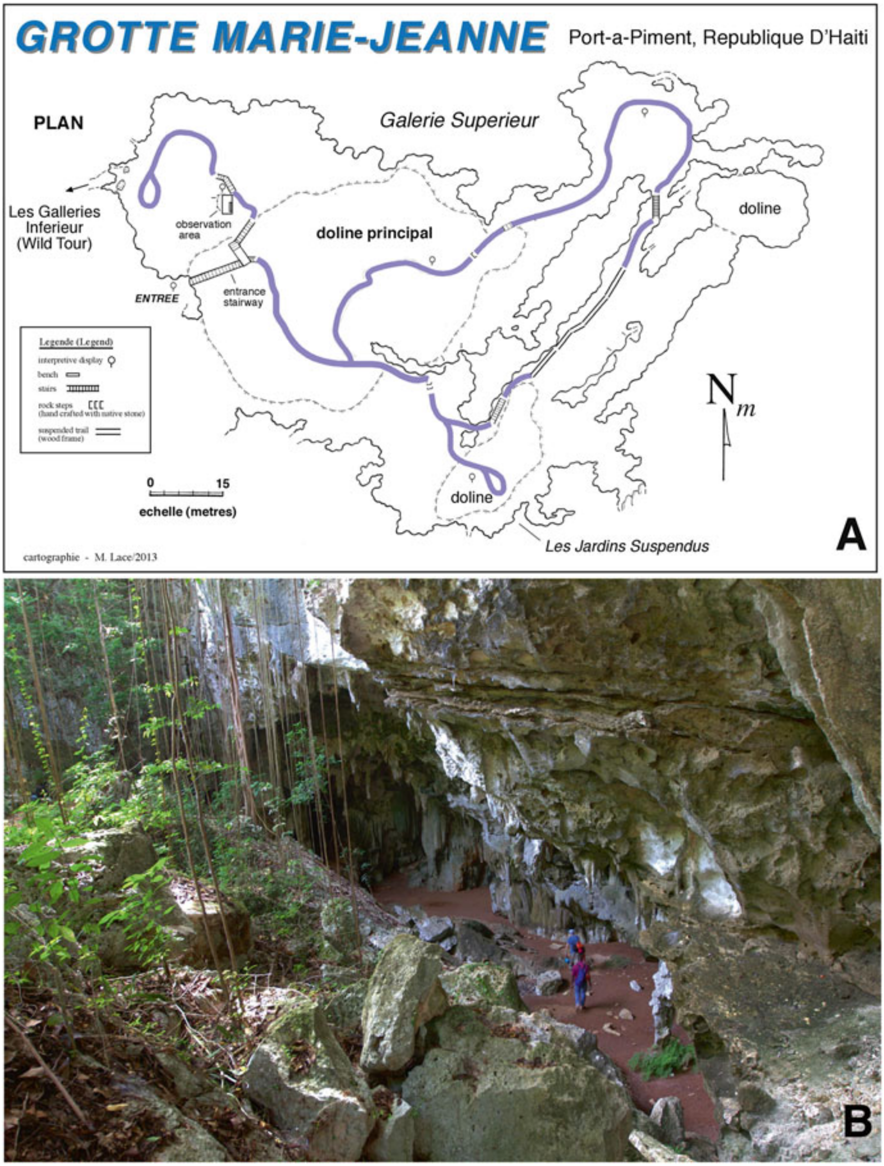


Fig. 6.7 Grotte Marie-Jeanne, Port-a'-Piment, Haiti (**a**) proposed trail map. (**b**) Principal doline entrance and tour route (photo by Brian D. Oakes)

At present, Grotte Marie-Jeanne remains unique in its sustainable site development strategy compared to the limited number of publicly accessible caves in Haiti, some of which have seen significant negative impacts from poorly structured visitation and vandalism. Clearly, not every cave site (coastal or inland) is suitable for ecotourism and detailed site assessments prior to any development planning must be carefully considered.

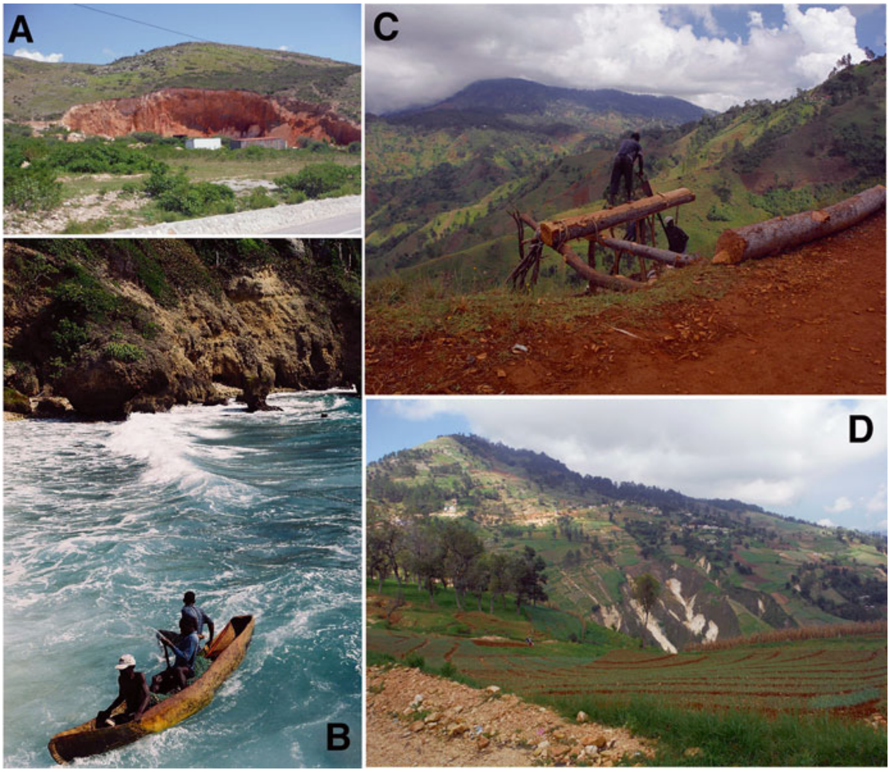


Fig. 6.8 Coastal land use panel (**a**) quarrying (**b**) fishing (**c**) deforestation and (**d**) slope instability and associated agricultural land use

Fig. 6.9 Lac Sourel – lower level of Grotte Marie-Jeanne

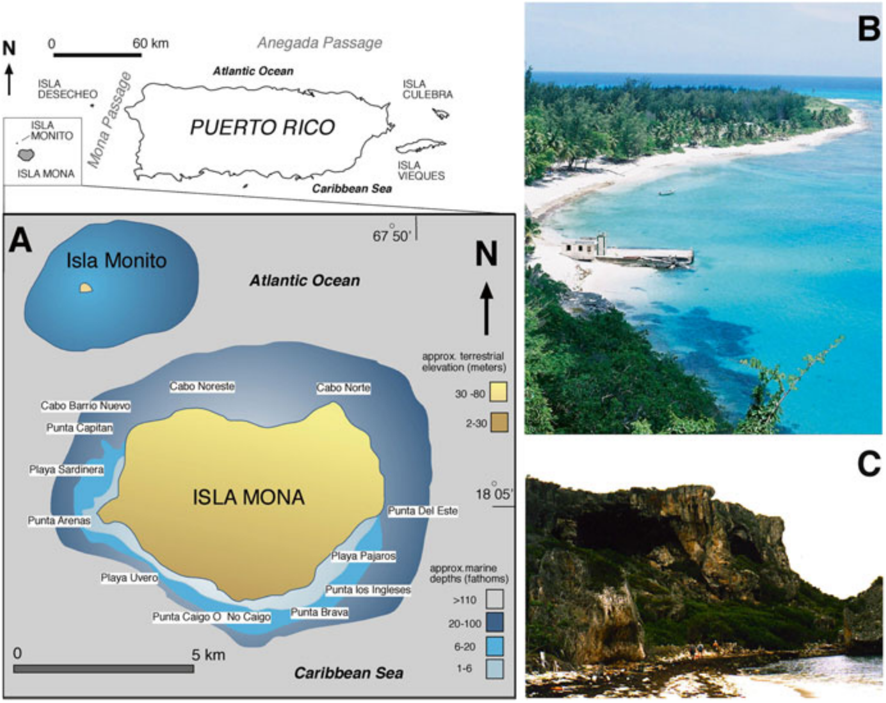


Fig. 6.10 (**a**) Map of Isla de Mona with regional inset (Benthic profile adapted from Kaye 1957). (**b**) Playa Sardinera, west coast of Isla de Mona. (**c**) Karstic cliffline, east coast of Isla Mona

6.3 Modeling Cave Development Patterns on Isla de Mona, Puerto Rico

6.3.1 Geologic and Geographic Setting

Located 68 km west of Puerto Rico (18 05N, 67 53W), Isla de Mona (58 km^2) and adjacent Isla Monito (0.17 km^2) are emergent surfaces of an uplifted carbonate block within the tectonically active Mona Passage (Fig. 6.1). The island platform rests on the Hispaniola-Puerto Rico Microplate within the complex continental margin defined by the North American Plate to the north and the Caribbean Plate to the south (Mann et al. 1991) (Fig. 6.2a). Isla de Mona is bounded by deep rift structures within a dynamic submerged landscape marked by periodic structural modification (i.e., linear extension) driven by strike-slip plate motion. The Mona coastline (32 km in length) is overwhelmingly karstic yet evidence of karst development associated with the platform is not confined to the emergent surfaces as a limited selection of submerged cave and karst features have also been mapped (Fig. 6.10).

Isla de Mona is Mio-Pliocene aged platform composed of two primary carbonate units. The volumetrically predominant platform base is composed of Mona Dolomite while the Lirio limestone forms a surface layer of limited vertical extent. The karst and pseudokarst landscapes of Isla de Mona provide an enduring record of episodic void development as a function of repeated sea level fluctuations and long term tectonic uplift. The predominant expressions of cave development are dissolutional voids formed within the *flank* of the landmass by processes associated with the *margins* of freshwater lens (i.e. *flank margin caves*), as defined by Mylroie (2013) and discussed in the next section.

6.3.2 Cave and Karst Development

Over 200 coastal caves have been documented on the island to date (Lace 2013), the majority of this inventory is composed of flank margin caves, including the largest known flank margin cave in the world – Sistema Faro (Lace et al. 2014), but also includes sea caves, pit caves and talus caves. Detailed cartography and analysis of the structural morphologies of these karst and pseudokarst expressions support a complex model of carbonate island cave development as a function of tectonic uplift, sea level changes, karst hydrogeology and cliff retreat (Fig. 6.11).

6.3.3 Karst Exploration and Research

Cave surveys and inventories formed the critical baseline database to which multiple analyses were applied, as previously reported (Lace 2008). Cave distribution analyses provide an overview of cave development trends in terms of lithology of the host rock and specific cave type (as an end proxy for tracking primary speleogenetic processes) (Fig. 6.13a, b). Yet more detailed approaches were necessary to quantitatively define the karst morphologies within the inventory and model their associated mechanisms of development.

6.3.3.1 Denudation Mechanisms

The coastal environment of Isla Mona is a geologically dynamic zone, with continual modification by cliff retreat, driven by energetic wave energy and progressive structural failure of its karstic escarpments. Though the estimated extent of cliff retreat is limited compared to other island settings, it has profoundly altered the majority of cave expressions on the island which are associated with the platform periphery. Figure 6.12 illustrates that coastal denudation mechanisms act in a similar manner on contrasting coastal cave structures. Flank margin caves are preexisting voids that are subsequently breached, exposed and denuded by littoral processes (Fig. 6.12a–d). In contrast, sea caves are initially formed by the same littoral processes that subsequently modify the erosional void (Fig. 6.12e–h), potentially resulting in a common endpoint structure (Fig. 6.12h).

Fig. 6.11 Map – Cueva Capitan, example of complex cave development within the Isla de Mona shoreline

6.3.3.2 Fractal Analysis of Coastal Cave Morphologies

Fractal geometry has long been used to mathematically define complex natural structures with particular attention to varied coastal landscapes and modeling the effects of a range of coastal processes (Tanner et al. 2006; Schwimmer 2008). Fractal analysis has previously been applied to a range of karst structures, from fluvial origins (i.e. stream cave networks) to paleokarst and fracture networks in a

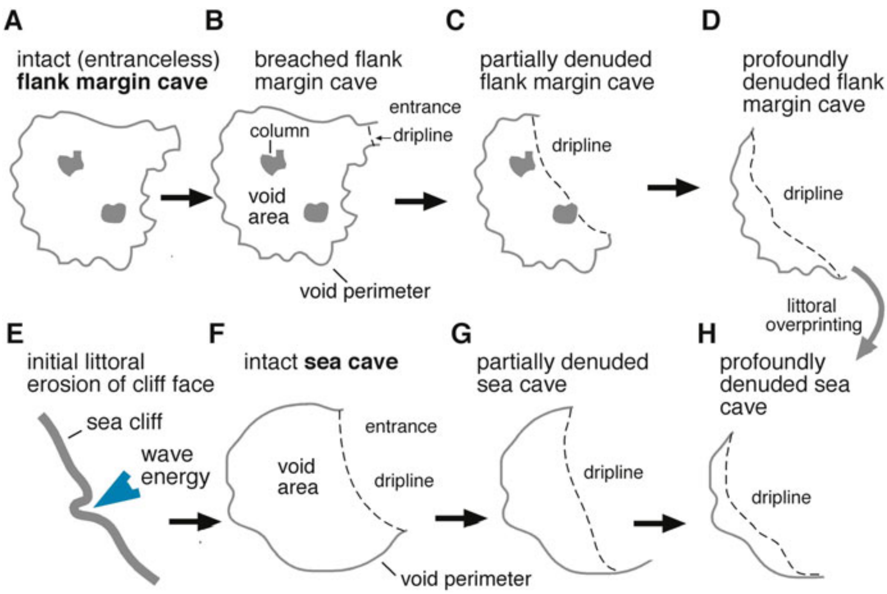


Fig. 6.12 Coastal denudation model. Progressive structural evolution of a flank margin cave (**a**–**d**) and sea cave (**e**–**h**) in a littoral environment

variety of landscapes (Kusumayudha et al. 2000; Skoglund and Lauritzen 2010; Robledo et al. 2002; Ferer et al. 2013). In contrast to the limitations associated with traditional methods of cave measurement, i.e. applying Euclidian algorithms to "fractal" objects (in this case – cave structures), fractal analyses can offer a more sophisticated approach to defining the distinctive, complex morphologies observed in karst (Curl 1986; Laverty 1987).

In terms of analyzing coastal karst expressions, this potential is underscored as measurements of total cave length, long applied to cave morphologies in traditional settings (Ford and Ewers 1978), have proven of limited value in coastal settings where caves are often complex, non-linear structures (Mylroie 2007). Thus, coastal caves which are morphologically complex represent prime candidates for such analyses. Similar to other karst features in carbonate coastal settings, the caves of Isla de Mona are clearly "fractal" objects (Fig. 6.13c). Fractal objects, whether theoretical constructs or shapes occurring on nature, exhibit a range of fractal dimensions, typically expressed within a range of 1.0 and 2.0. The fractal dimension (D) of a given object can be calculated using a variety of methods, where N is the number of spatial increments (i.e. boxes) required to define the shape and C is the scale (i.e., box size) applied. The fractal box count (D) is then determined from the negative slope of the regression line derived from the plot of the log of the box count versus the log of the box size (Fig. 6.13d).

The flank margin caves of Isla de Mona and many other coastal settings display limited vertical passage development – consistent with their model of genesis and in stark comparison to other continental karst expressions (Palmer 1991; Mylroie

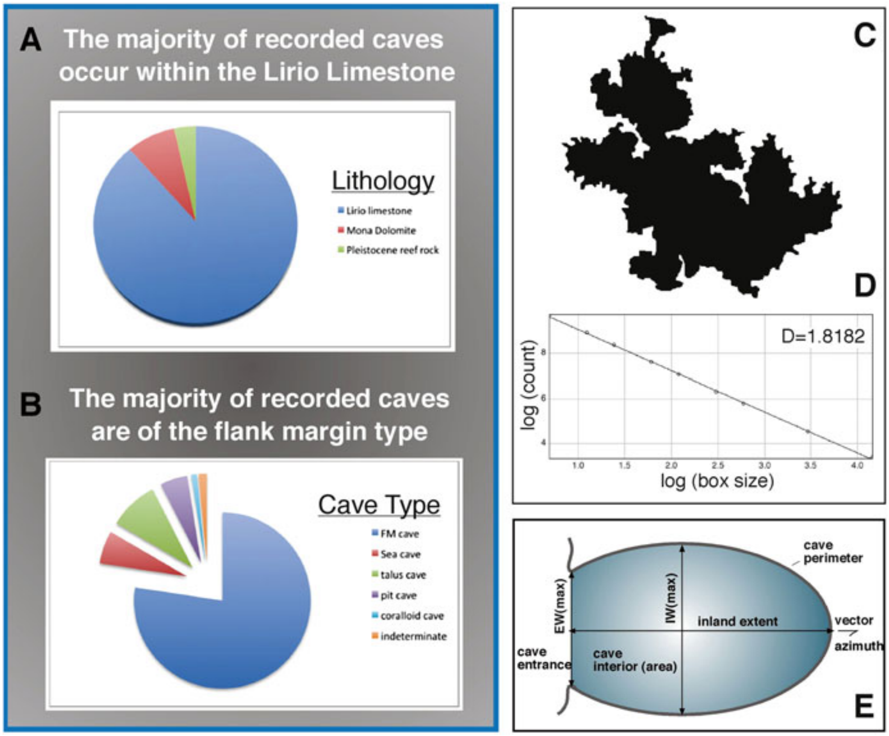


Fig. 6.13 Cave distributions by (**a**) lithology and (**b**) cave type. (**c**, **d**) Modeling cave morphologies on Isla de Mona, where *D* fractal dimension, *EW* cave entrance width and *IW* internal passage width

2013). In this island setting, they are comparatively planar structures which generally feature limited vertical extent compared to lateral passage development, as defined by total cave perimeter and total aerial footprint (or total "cave area"). Thus, two-dimensional data, as depicted in detailed cave maps (Fig. 6.11), can serve as an adequate proxy for morphological complexity in this case, where tectonic uplift in conjunction with sea level fluctuations have not conspired to generate multiple elevations of elaborate passage development within a single cave.

Figure 6.14 graphically compares each Mona cave to multiple spatial (morphometric) parameters: (i) aerial footprint (i.e "total cave area"), (ii) relative degree of structural denudation, based on maximum entrance width to maximum internal passage width ratio (Fig. 6.13e), and finally (iii) fractal dimension (as determined by a standard fractal box count described above). The average fractal dimension [$D = 1.84 \pm .05$] for the cave data set from Isla de Mona was measured from sample of over 140 documented cave sites, exhibiting a narrow dimensional range of 1.63–1.94 and encompassing examples of all recorded karst and pseudokarst cave types, cave areas (11–28,072 m^2) and measured effects of denudation (0.13 to a theoretical maximum of 1.0, as per Stafford et al. 2004). The fractal dimension range of the cave data set is persistent throughout, serving as a continuous morphological thread,

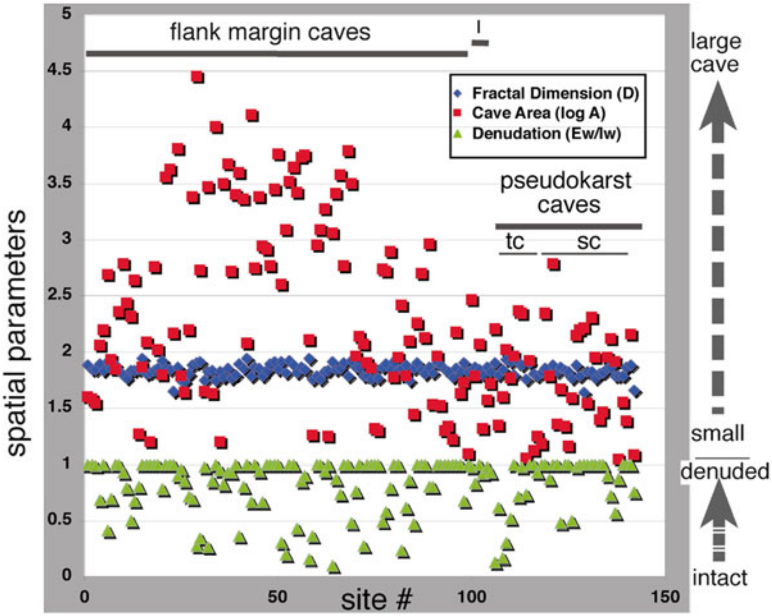


Fig. 6.14 Spatial analysis of Mona caves, using Image J software analysis of cave maps (National Institutes of Health, Bethesda, MD, www.NIH.gov); where *tc* talus cave, *sc* sea cave and *I* indeterminate

as it were. Interestingly, though the cave sites examined have been exposed to a range of coastal modification and denudation effects due to repeated cycles of coastal modification, cave formation and cliff retreat, the fractal profile indicates that the overall complex morphologies of these structures have been retained through a span of profound episodic patterns of coastal landscape evolution.

Though the sample size is sufficiently large, the range of cave types within the cave inventory is limited in that flank margin caves represent the predominant form of coastal cave expression on the island, with a minority segment composed of pseudokarst cave types. Thus, the overall morphology would be expected to be similar (as Fig. 6.14 indicates) and consistent with karst development within the theoretical framework of a simple carbonate island setting within the Carbonate Island Karst Model (CIKM) (Mylroie and Mylroie 2007). Figure 6.14 further demonstrates that flank margin caves compose the majority of the recorded cave area (or degree of void development) and subsequently can be identified as the principal potential instability component (i.e. recorded void type) in an overall coastal karst landform stability profile of Isla de Mona.

Expanding the morphological comparison of the Isla de Mona cave data to other island karst settings that display greater diversity in observed cave types and island platform complexity may prove useful in delineating regional trends in coastal speleogenesis, biodiversity and landform stabilities. Nevertheless, the comparison of multiple spatial criteria that specify cave and karst morphologies, as shown here, provides a comparative morphometric profile of cave development on Isla de Mona that spans the measured extent and diversity of these resources – an approach that can readily be applied to other settings.

Previous analyses using morphometric parameters to better define cave patterns in coastal settings have shown promise but greater quantitative definition is still needed to differentiate cave and karst structures based on their distinctive morphologies (Basso et al. 2012; Owen 2013; Lace 2008; Waterstrat et al. 2010). Inclusion of additional fractal indices, such as lacunarity or tortuosity (i.e. generating a multi-fractal profile) may prove useful in future expanded studies by further discriminating between structures based on speleogenic origins in this or other coastal settings. Three-dimensional modeling of cave structures has shown considerable promise but such analyses have yet to be applied to a sufficiently broad sampling of both coastal and continental karst landscapes to permit discrimination of type-specific cave morphologies (Boggus and Crawfis 2007; Kincaid 1999; Pardo-Iguzquiza et al. 2011; Labourdette et al. 2007). As demonstrated in this chapter, simultaneous analysis of multiple spatial parameters specific to coastal karst may reveal previously undefined patterns, potentially contributing to a better understanding of coastal cave development, karst landform stability profiles and the distribution of critical cave habitats in coastal environments.

6.3.4 Cave and Karst Resource Management

The coastal karst of Isla de Mona is considered world class in terms of its density, scale and complexity. As a natural reserve (designated by the U.S. Dept, of the Interior in 1975) and part of the largest marine protected area in the Puerto Rican islands, the island and its resources have remained largely untouched by expanding commercial and residential development that has at times inflicted profound impact on coastal landforms on the Puerto Rican mainland. As a U.S. territory, Isla de Mona is considered a U.S. border, thus a coast guard and terrestrial border security presence is a key component of its routine management operations. Geographically, the island continues to occupy a pivotal position in the western Caribbean as a U.S. border control point refugees, many in small vessels of questionable seaworthiness, periodically make landfall on the Mona coast.

In addition to the unique geology and biologic diversity, the caves also reflect a complex human history associated with the island and its resources, spanning archaic occupation (4,300 ybp) through Euro-colonial expansion to present day karst preservation efforts within the context of a natural reserve. Recent field research has revealed that the caves of Isla de Mona offer a unique glimpse into archaic and historical periods associated with the complex role Isla de Mona has played in shaping cultural progression in the Puerto Rican islands and in the broader region (Dávila 2003; Lace 2012; Rodriguez Ramos 2010) (Fig. 6.15). This unique geoarcheological profile carries distinct cultural resource management requirements.

Isla de Mona continues to offer a unique carbonate island setting in which field research in coastal speleogenesis, tropical cave climate modeling, island karst

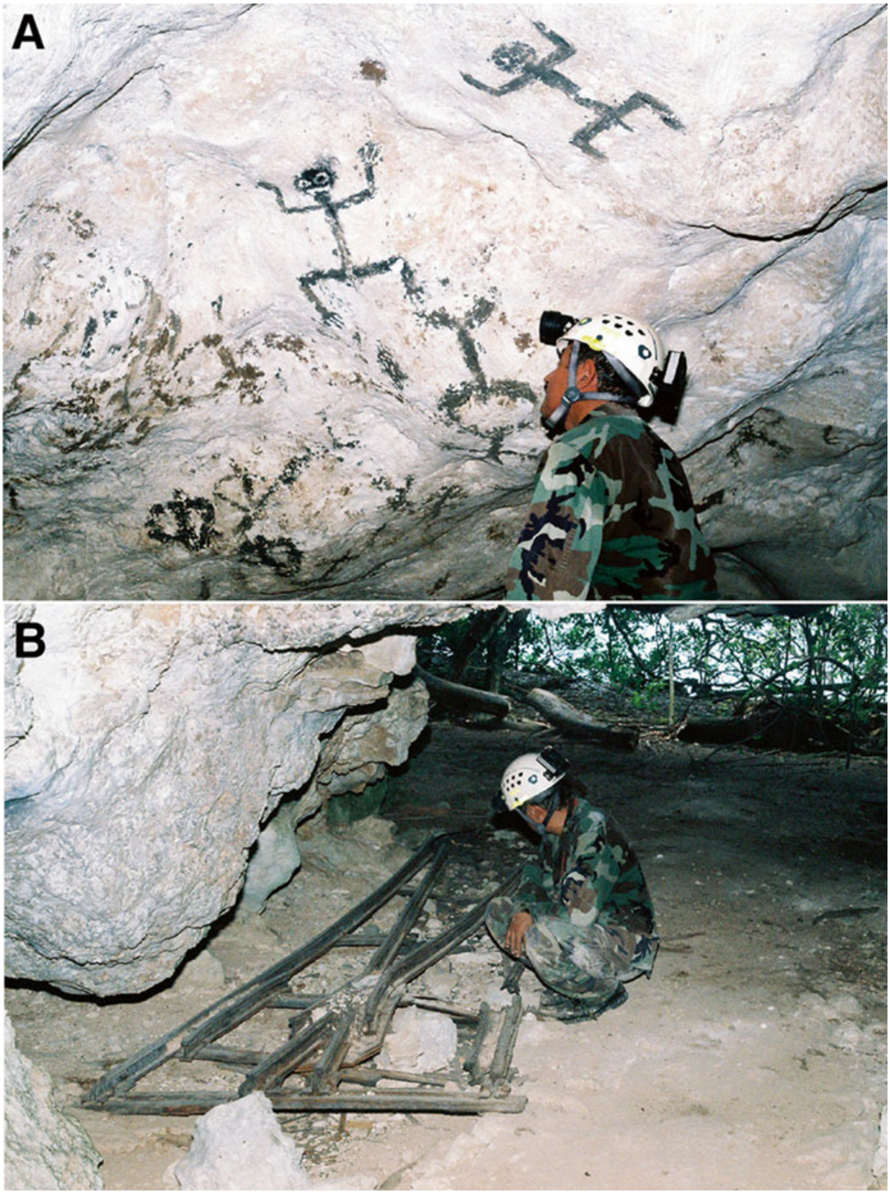


Fig. 6.15 (**a**) Cave rock art on Isla Mona. (**b**) Remnants of nineteenth century guano mining

biodiversity and geoarcheology of caves in the Caribbean region can be conducted. All research on Isla de Mona is performed via a permit system administered by the Puerto Rico Department of Natural Resources (Departamento Recursos Naturales y Ambientales de Puerto Rico). Though we have collectively learned a great deal about the geological, biological and cultural significance of the many caves and karst features on Isla de Mona to date, significant research in many disciplines remains to be accomplished. Field research continues on Isla de Mona with the goal of completing the cave and karst inventory as a critical baseline tool for future studies and for supporting long term preservation and management goals.

6.4 Coastal Caves of Grenada

6.4.1 Geologic Setting

The island of Grenada occupies the southernmost position in the Lesser Antilles island chain (12 07N, 61 40W), where the pre-Miocene and post-Miocene arcs of volcanic islands converge, >150 km north of Trinidad, Tobago and the South American coast (Fig. 6.1). The island rests on the eastern edge of the Caribbean Plate in a subduction zone with the underlying South American Plate. It is bounded by the Grenada Basin the west and the Tobago Basin to the east. The emergent island platform has a total area of 344 km^2 with 121 km of coastline. Geologically the island is composed of complex pyroclastic flows, igneous rock (including andesite domes) and significant areas of reworked volcanic detritus with only isolated fringing carbonate deposits in the form of uplifted reef relicts of Pleistocene age (Arculus 1976). The oldest rock is composed of sedimentary deposits called the Tufton Hall Formation, placing the age of the island within the Eocene to lower Miocene periods. No contemporary volcanic activity has been recorded on the island, with the exception of minor geothermal features in the form of hot springs, there is one active submarine volcano off the northwest coast (Kick'em Jenny) whose last eruption was reported in 1974. Inland morphologies include relict volcanic craters, deep ravines, numerous rivers and volcanic peaks reaching a maximum elevation of 840 m, with comparable elevations supporting a perennial orographic effect and accompanying diverse landcover development.

Coastal morphologies are diverse, featuring arid to heavily vegetated slopes, rocky headlands, cliffs, sea stacks, arches, wave-cut benches, deep embayments, beaches and numerous smaller offshore islands composed of varying lithologies (Fig. 6.16). Caves are also prominent features of the Grenada coast. From a geoarcheological, ecological and hydrogeological perspectives, the inland and coastal landforms on this and other small islands are intimately interconnected (Benz 2010).

6.4.2 Cave Development

Though carbonate coastal landforms are dominant in the Greater Antilles, non-carbonate coastlines are prevalent in the Lesser Antilles Arc and must also be considered when comparing regional coastal zone management approaches as a

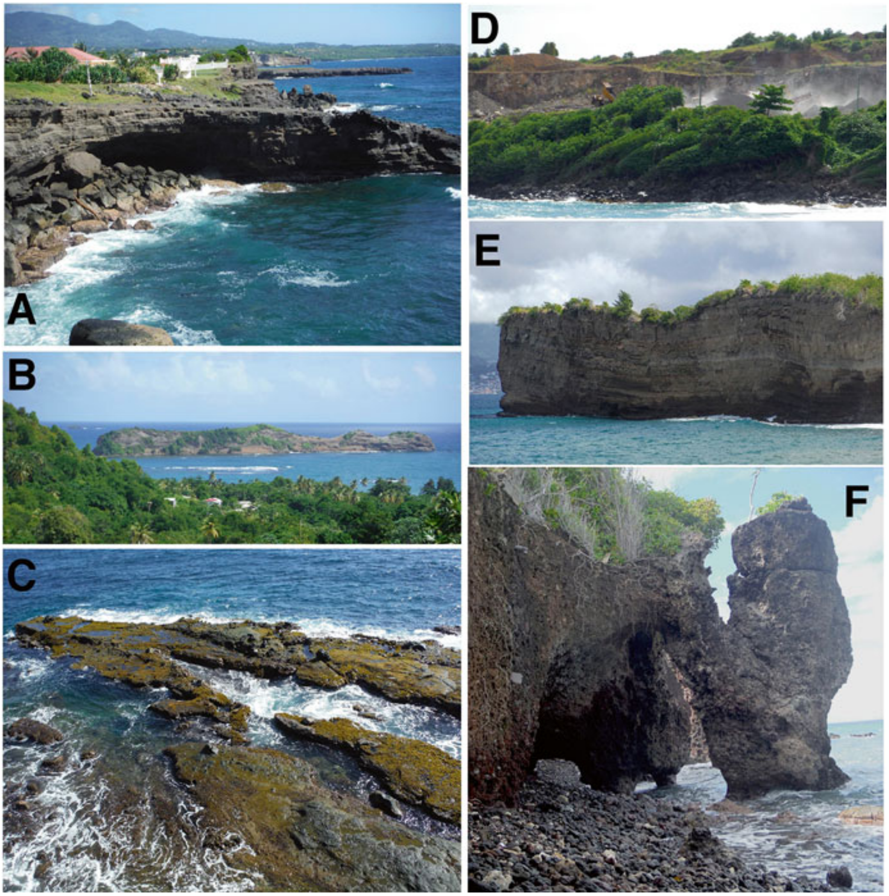


Fig. 6.16 Coastal landscapes of Grenada. (**a**) Coastal outcrop composed of reworked volcanics. (**b**) One of several offshore islands (Ile Marquis). (**c**) Shoreline terrace. (**d**)Active coastal quarry. (**e**)Sea cliff (Canoe Bay, south coast). (**f**) Sea arch development

function of the cultural resources and overall biodiversity unique to such settings. Grenada features minimal carbonate exposures therefore limited potential for true karst development in its coastal settings. Yet, coastal zone management strategies are not limited solely to carbonate landforms. Pseudokarst examples are confined to coastal areas and found in varying locations across the island in developed, undeveloped zones and MPAs (Fig. 6.17). No submerged caves have been recorded of the coast of Grenada with the exception of anecdotal reports from divers of submerged caves off the coast of nearby Carriacou. Though caves in Grenada play important roles in the cultural history and biodiversity, they had previously been poorly defined. This report offers the first formal report of pseudokarst development on Grenada.

Fig. 6.17 Pseudokarst examples: (**a**) Ft. Jeudy cave (**b**) St. David's bat cave (**c**) Lover's Bay cave. (**d**) Sea cave formed within Pleistocene-aged limestone, north coast

A total of 41 caves have been documented on the island to date though exploration of all coastal areas and submerged areas in the nearby Grenadine islands remains incomplete. All of the caves defined on Grenada are pseudokarst, or not formed by karstic dissolution processes, encompassing varied expressions of sea cave development (Moore 1954) primarily within contemporary shorelines composed of reworked volcanics with a minority located within uplifted coastal basalt and Pleistocene-aged limestone. The following examples serve to illustrate this pattern:

Black Sand Punchbowl (Fig. 6.18). Three separate entrances breach the coastal outcrop of reworked volcanics, leading to an amphitheater-like, open-air chamber laterally connected to the sea. The structure resembles a littoral sink (commonly referred to as a "punchbowl") similar to numerous examples recorded on the western U.S. coast (Bunnell and Kovarik 2013) (Fig. 6.18a, b). Continual wave energy has been focused on structural defects in the coastal exposure, resulting in progressive cavity development and eventual collapse of limited overburden to generate the structure observed today (Fig. 6.19a). A second site to the east of the punchbowl likely represents a remnant structure formed by similar processes,

Fig. 6.18 Littoral sinks. (**a**) Aerial image of pseudokarst features within the Grenada coastline (Courtesy of GoogleEarth). Sites *I* and *II* indicate similar features at different stages of development. (**b**) Black Sand Punchbowl (Site *I*) and (**c**) Natural Bridge (site *II*)

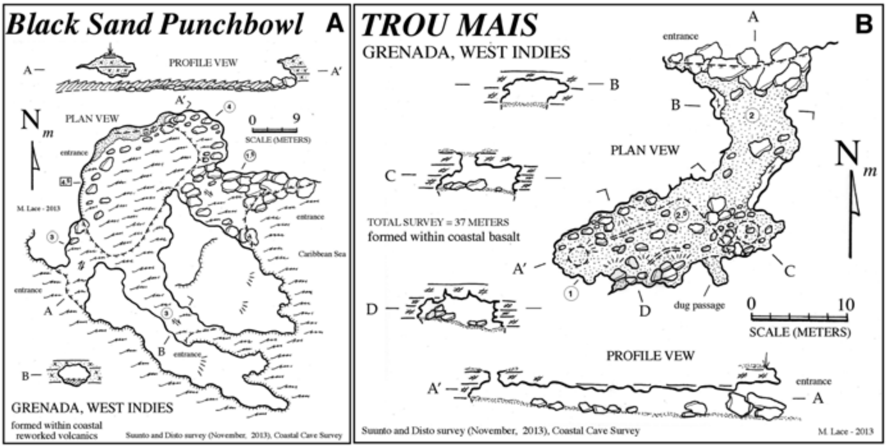


Fig. 6.19 (**a**) Map of Black Sand Punchbowl. (**b**) Map of Trou Mais, Grenada

with only a partial perimeter segment remaining as a natural bridge (Fig. 6.18c). If it were in a carbonate coastal setting, however, a karstic model of structural genesis would invoke a different interpretation as a breached flank margin cave, that is, a preexisting cavity initially formed by dissolution processes associated with a freshwater lens, as discussed in the section on Isla de Mona and documented in many other carbonate island settings (Mylroie 2013). While limestone exposures are restricted to the northern end of the island – no flank margin caves have been identified. Such a dissolutional model is not tenable in the majority of the coastal settings on the island given the porosity and structural integrity associated with reworked coastal volcanics and basalts.

Trou Mais (Fig. 6.19b) is the longest recorded cave on Grenada with 37 m of surveyed passage formed within a Pliocene-aged basalt flow (Fig. 6.20). The cave has been reported as entirely man-made but close inspection revealed that this is not the case. Located at ~30 m asl and proximal to the current shoreline, the structure is most likely a sea cave formed by previous sea level stillstand. The structure was subsequently uplifted to its current position, which is comparable to maximum elevations associated with uplifted Pleistocene-aged limestones on the northern end of the island (Arculus 1976). The cave is an important pre-contact cultural site, exhibiting anthropogenic passage modification consistent with its reported pre-Columbian use as agricultural storage (i.e. corn) by the Caribs (hence the cave name). The cave is also a significant bat habitat, hosting multiple species, as well as a range of opportunistic species including anole and frogs. Modern anthropogenic site uses include minimal tourist visitation, including occasional local school fieldtrips, and sporadic, non -commercial harvesting of guano as a garden fertilizer.

St. David's Bat Cave. Two entrances lead into a large sea cave chamber that passes 15 m through a narrow coastal ridge (Fig. 6.17b). It is the most significant cave bat roost identified on the island to date with more than several hundred occupants and clear signs of long term bat use. As such, the site warrants long-term colony monitoring and immediate proactive preservation efforts.

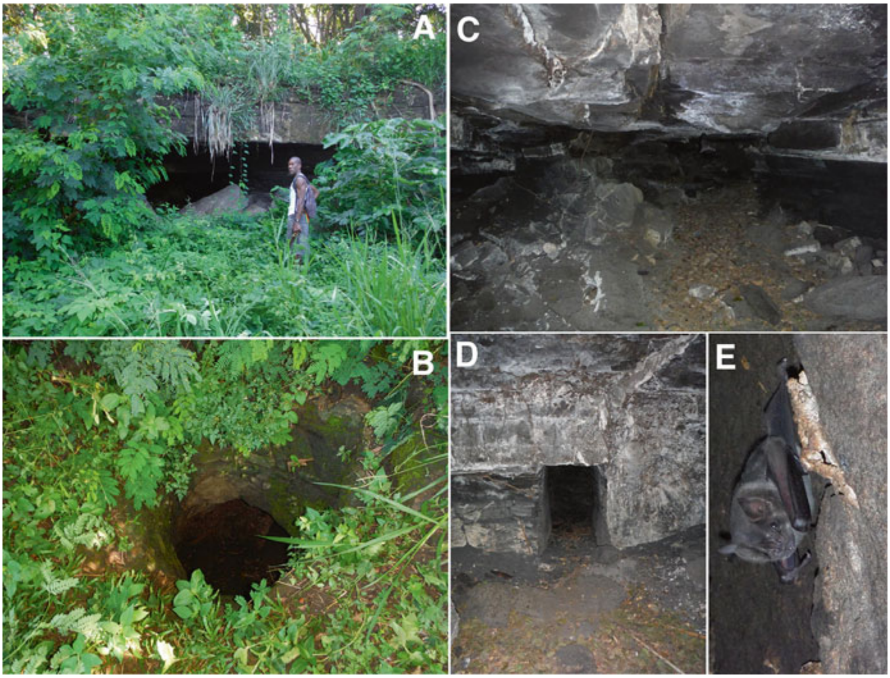


Fig. 6.20 Trou Mais. (**a**) Main entrance. (**b**) Internal passage cross section. (**c**) Anterior opening. (**d**) Excavated side passage and (**e**) roosting bat

The remainder of the cave inventory features variations of sea cave expressions as well as man-made cavities. Not surprisingly, the majority of sea cave development is associated with embayments exposed to the prevailing currents which flow westward through the Grenada Passage, supplying a significant amount of the inflow into the Caribbean Sea. Embayments and associated peninsulas of varying scales focus the wave energy associated with these currents onto structural defects within littoral exposures of volcanics, resulting in erosional void development.

6.4.3 Coastal Geoarcheology

The pre-Columbian cultural sequence of Grenada spans pre-Arawak, Arawak and Carib occupation, partly based on surface excavations across the island (Bullen 1965). At least five rock art sites also attest to past cultural use of both coastal and inland landscapes – all associated with non-carbonate boulders locally known as "Carib Stones", for example, at Duquesne Bay (Fig. 6.21a, b). At least four other coastal and inland rock art sites have been reported on the island (Fig. 6.21c, d) (Cody 1990; Marquet 2002). Rock art examples on Grenada range from simplistic to elaborate anthropomorphic, zoomorphic and geometric forms and are thought to date to the late pre-contact period (900-1400 B.C.), though no confirming dating

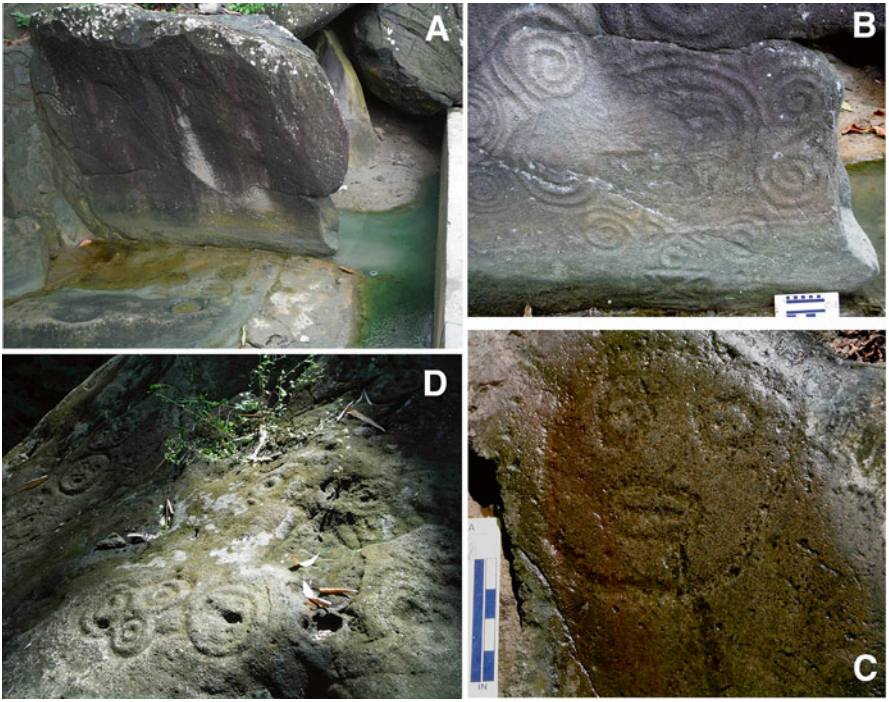


Fig. 6.21 Rock art of Grenada. (**a**–**c**) Coastal sites (**d**) interior river site (note 10 cm scale)

technique has been applied to date. Prior efforts to apply UNESCO heritage status to one of these sites have so far been unsuccessful. All documented rock art sites currently remain at risk, displaying a range of rock surface instabilities and land use pressures (Allen and Groom 2013). The only cave exhibiting pre-contact use (consistent with site ethnography) is Trou Mais (Fig. 6.19d), as previously discussed. Excavations at a coastal site adjacent to the cave yielded ceramics dating to A.D. 1-300. Such coastal cultural sites must be an integral component of a long-term preservation plan that has yet to be fully implemented – a recurrent trend in the region and a common denominator present in all three settings described in this report.

6.4.4 Coastal Cave and Karst Resource Management

Grenada has a population of over 100,000 with an economy based primarily on tourism, agriculture and local industries as reflected in land use patterns across the island (Ternan et al. 1989). Coastal areas figure prominently in this economic model. Management strategies have been enacted to both support local economic viability and resource preservation with the context of sustainable models (Thomas 2000;

UNEP 2012; Watson 2007), utilizing integrated GIS methods to define the complex coastal resources at risk (DeGraff and Baldwin 2013; Helmar et al. 2008). The caves of Grenada and adjacent insular environments also feature substantial biodiversity associated with coastal and interior landscapes (Pederson et al. 2013). However, cave development is limited compared to other islands in the Lesser Antilles and are not prominent components of existing tourism destination models or coastal zone management plans. Though environmentalism is well-developed on the island, spanning community and governmental levels, commercial land use pressures still pose a significant risk to coastal landform integrity (Fig. 6.16d).

6.5 Summary

Coastal karst and pseudokarst form one of the dominant emergent landscapes in the Caribbean. As the preceding examples demonstrate, exploration and study of these diverse landscapes is still far from complete. Haiti, Isla de Mona and Grenada feature contrasting coastal geomorphologies which have directly influenced the diversity and degree of coastal cave development. This has further influenced endemic biodiversity (Willig,et al. 2010) and patterns of human settlement and cultural uses of coastal landscapes from Archaic through modern periods (Fitzpatrick and Keegan 2007). Yet, all three settings share the same imperative and general strategies to preserve and effectively manage these natural and cultural landscapes. A more comprehensive understanding of cave and karst development patterns in these and other coastal settings has multi-disciplinary implications, supporting management of at-risk karst resources, with the potential to influence the quality of life of significant coastal populations throughout the region. In depth field exploration remains a critical component of this process. Integrated analysis of the geological, biological and geoarcheological significance of coastal caves and karst offers critical perspectives to emerging models of both physical and cultural landscape evolution not only on a regional scale but with broader applications as well.

Acknowledgments The author wishes to thank Brian D. Oakes, Patricia N. Kambesis and expedition participants of the Isla de Mona Project for their assistance. This work was supported by the Coastal Cave Survey and in part by a grant from the International Exploration Section of the National Speleological Society in collaboration with the Haitian Speleological Survey (Recensement Spéléologique D'Haiti – RHS) with logistical support from the Puerto Rico Department of Natural Resources (Departamento Recursos Naturales y Ambientales – DRNA).

References

Allen CD, Groom KM (2013) Evaluation of Grenada's "Carib Stones" via the rock art stability index. Appl Geogr 42:165–175

Arculus RJ (1976) Geology and geochemistry of the Alkali Basalt – Andesite Association of Grenada and the Lesser Antilles Island Arc. Geol Soc Am Bull 87(4):612–624

Basso A, Bruno E, Parise M, Pepe M (2012) Morphometric analysis of sinkholes in a Karst coastal area. Geophysical research abstracts, vol 14. EGU General Assembly
Benz EJ (2010) A Paleoenvironmental reconstruction from the Island of Grenada, Caribbean environments through the time of human occupation. MA thesis, Department of Anthropology, Washington State University, 94p
Boggus M, Crawfis R (2009) Procedural creation of 3D solutional cave model. Proceedings of the 20th IASTED International Conference on Modeling Simulation. Banff, Alberta, pp 180–186
Bullen RP (1965) An archaeological chronology for Grenada. Am Antiq 31(2):237–241
Bunnell DA, Kovarik JL (2013) Caves of the Western U.S. coast. In: Lace MJ, Mylroie JE (eds) Coastal karst landforms, vol 5, Coastal research library series. Springer, Dordrecht, pp 299–316
Cigna AA (1993) The environmental management of tourist caves: the examples of Grotta di Castellano and Grotte Granda del Vento, Italy. Environ Geol 21:173–180
Cody A (1990) Faces and figures on Grenada: their historical and cultural relations. In: Rock art papers, vol. 7, San Diego Museum of Man, San Diego
Cotilla MO, Cordoba D, Calzadilla M (2007) Morphotectonic study of Hispaniola. Geotectonics 41(5):368–391
Curl RL (1986) Fractal dimensions and geometries of caves. Mar Geol 18(8):765–783
Dávila O (2003) Arqueología de la Isla de la Mona. Editorial del Instituto de Cultura Puertorriqueña, San Juan
Day M (2007) Natural and anthropogenic hazards in the Karst of Jamaica. Geol Soc Lond Spec Publ 279:173–184
Day M (2010) Human interaction with Caribbean karst landscapes: past, present and future. Acta Carsologica 39(1):137–146
DeGraff AK, Baldwin K (2013) Participatory mapping of heritage sites in the Grenadines. CEMRES technical report no. 65. Centre for Resource Management and Environmental Studies, Barbados
Dodge RE, Fairbanks RG, Benninger LK, Maurasse F (1983) Pleistocene sea levels from raised coral reefs of Haiti. Science 219(4591):1423–1425
Dowling RK (2011) Geotourism's global growth. Glob Herit 3(1):1–13
EI (2012) Integrated baseline study: Port-a-Piment watershed, Sud Department, Haiti. Report compiled, in cooperation with the Cote Sud Initiative, by the Earth Institute, Columbia University, New York, 26p
Ferer MV, Smith DH, Lace MJ (2013) Fracture networks from a deterministic physical model as "forerunners" of maze caves (Abstract). In: Proceedings of the AGU Fall meeting, San Francisco
Fitzpatrick SM, Keegan WF (2007) Human impacts and adaptations in the Caribbean islands: an historical ecology approach. Earth Environ Sci Trans Royal Soc Edinb 98:29–45
Florentin JM, Maurasse R (1982) Survey of the geology of Haiti: a guide to field excursions in Haiti. Miami Geological Society, Miami, 103p
Ford DC, Ewers RO (1978) The development of limestone cave systems in the dimensions of length and depth. Can J Earth Sci 15:783–1798
Hadden RL, Minson SG (2010) The geology of Haiti: an annotated bibliography of Haiti's geology, geography and earth science. USACE Army Geospatial Center, Alexandria, VA, 254p
Helmar EH, Kennaway TA, Pedreros DH, Clark ML, Marcano-Vega H, Tieszen LL, Ruzycki TR, Schill SR, Sean Carrington CM (2008) Land cover and forest formations distribution on St. Kitt's, Nevis, St Eustatius, Grenada and Barbados from decision tree classification of cloud-cleared satellite imagery. Caribb J Sci 44(2):175–198
Hobbs JJ (2004) Guidelines for the participation of local and indigenous peoples in Karst conservation and management. In: Batelaan O, Dusar M, Masschelein J, Thanh Tam Vu, Tran Tan Van, Nguyen Xuan Khien (eds) Proceedings of the interdisciplinary conference on development and conservation of Karst Regions, Hanoi, Vietnam. Trans-KARST 2004, pp 86–87
Jones WK, Hobbs HH, Wicks CW, Currie RR, Hose LD, Kerbo RC, Goodbar JR, Trout J (2003) Recommendations and guidelines for managing caves on protected lands. Prepared for the National Cave and Karst Research Institute, National Park Service. Karst Waters Institute Special Publication, no. 8, 81p

Kambesis PN, Lace MJ, Despain J, Goodbar J (2010) Assessment of Grotte Marie-Jeanne, Port-a-Piment, Republique D'Haiti. Final report submitted to the Haitian Ministry of Tourism. Available www.grottemariejeanne.com, 40 p

Kambesis PN, Oakes B, Lace MJ (2013) Challenges of cave management in a developing country: a case study of Grotte Marie-Jeanne, Departmente Sud, Haiti. Proceedings of the 20th National Cave and Karst Management symposium, Carlsbad, NM, pp 143–146

Kincaid TR (1999) Volumetric fractal dimension as a quantitative descriptor of saturated cave morphology. In: Palmer AN, Palmer MV, Sasowski ID (eds) Karst modeling. Karst Waters Institute, Charlottesville, p 186 (Abstract)

Kusumayudha SB, Zen MT, Notosiswoyo S, Gautama RS (2000) Fractal analysis of the Oyo River, cave systems, and topography of the Gunungsewu Karst area. Hydrogeol J 8(3):271–278

Labourdette R, Lascu I, Mylroie JE, Roth M (2007) Process-like modeling of flank margin caves: from genesis to burial evolution. J Sediment Res 77:965–979

Lace MJ (2008) Coastal cave development in Puerto Rico. J Coast Res 24(2):508–518

Lace MJ (2012) Anthropogenic use, modification and preservation of coastal caves in Puerto Rico. J Island Coast Archaeol 7:378–403

Lace MJ (2013) Coastal caves and karst of the Puerto Rican islands. In: Lace MJ, Mylroie JE (eds) Coastal karst landforms, vol 5, Coastal research library series. Springer, Dordrecht, pp 207–226

Lace MJ, Mylroie JE (2013) Coastal cave and karst resource management. In: Lace MJ, Mylroie JE (eds) Coastal karst landforms, vol 5, Coastal research library series. Springer, Dordrecht, pp 127–143

Lace MJ, Kambesis PN, Mylroie JE (2014) Sistema Faro: evolution of the world's longest flank margin cave. In: Robledo PA, Duran JJ (eds) Large cave systems of the world. Boletin Geologico y Minero. Journal of Earth Sciences, Instituto Geologico y Minero de Espana. (in press)

Laverty M (1987) Fractals in karst. Earth Surf Proc Land 12:475–480

Lutz SJ, Ginsberg RN (2007) The state of deep coral ecosystems of the United States Caribbean region: Puerto Rico and the U.S. virgin islands. In: Bumsden SE, Hourigan TF, Bruckner AW, Dorr G (eds) The state of the deep coral ecosystems of the United States. NOAA Technical Memorandum CRCP-3, Silver Springs

Mann P, Draper G, Lewis JF (1991) An overview of the geologic and tectonic development of Hispaniola. Geol Soc Am Spec Pap 262:1–28

Marquet SJ (2002) Les Pétroglyphes des Petites Antilles Méridionales. Contextes Physique et Culturel. Paris monographs in American archaeology. BAR international series 1051, Archaeopress, Oxford. 371p

Miloslavich P, Diaz JM, Klein E, Avarado JJ, Diaz C (2010) Marine biodiversity in the Caribbean: regional estimates and distribution patterns. PLoS ONE 5(8):e11916

Moore DG (1954) Origin and development of sea caves. Nat Speleol Soc Bull 16:71–76

Mylroie JE (2007) Cave surveys, cave size and Flank margin caves. Compass Tape 17(4):8–16

Mylroie JE (2013) Coastal karst development in carbonate rocks. In: Lace MJ, Mylroie JE (eds) Coastal karst landforms, vol 5, Coastal research library series. Springer, Dordrecht, pp 77–109

Mylroie JE, Mylroie JR (2007) Development of the carbonate island karst model. J Cave Karst Stud 69:59–75

Mylroie JE, Mylroie JR (2013) Flank margin caves in carbonate islands and the effects of sea levels. In: Schroder J, Frumkin A (eds) Treatise on geomorphology. Academic, San Diego, pp 351–362, 6

Owen A (2013) Tafoni development in the Bahamas. In: Lace MJ, Mylroie JE (eds) Coastal Karst landforms, Coastal Research Library Series, No. 5. Springer, Dordrecht

Palmer AN (1991) Origin and morphology of limestone caves. Geol Soc Am Bull 103:1–25

Palmer AN (2007) Cave geology. Cave Books, Dayton, 464p

Pardo-Iguzquiza E, Duran-Valsero A, Rodriguez-Galiano V (2011) Morphometric analysis of 3-dimensional networks of karst conduits. Geomorphology 132:17–28

Pavlovich K (2002) The evolution and transformation of a tourism destination network: Waitomo Caves, New Zealand. Tour Manage 24:203–216

Pederson SC, Genoways HH, Kwiecinski GG, Larsen PA, Larsen, RJ (2013) Biodiversity, biogeography and conservation of bats in the Lesser Antilles. In: Vernier J-L, Burac M (eds)

Biodiversite Insulaire: La Flore, La Faune, et L'Homme dans les Petites Antilles. Schoelchers, Martinique. Direction de L'Environnement, de L' Amenagament et du Logement de Martinique et Universite des Antilles et de la Guyane, Paris, France, pp 62–73, 330

Pikelj K, Juracic M (2013) Eastern Adriatic Coast (EAC): geomorphology and coastal vulnerability of a karstic coast. J Coast Res 29(4):944–957

Robledo P, Duran JJ, Gumiel P (2002) Fractal analysis applied to the study of the karst collapse paleostructures in the upper Miocene of the Eastern Coast of Mallorca, Spain. Geogaceta 31:27–29

Rodriguez Ramos R (2010) Rethinking Puerto Rican Precolonial history. University of Alabama Press, Tuscaloosa, 267p

Rouse I, Moore C (1985) Cultural sequence in Southwest Haiti. In: Proceedings of the 10th International congress for the study of pre-Columbian cultures of the Lesser Antilles. Fort de France, Martinique, pp 1–21

Schwimmer RA (2008) A temporal geometric analysis of eroding Marsh shorelines: can fractal dimensions be related to process? J Coast Res 241:152–158

Siegel PE, Righter E (eds) (2011) Protecting heritage in the Caribbean, vol 3, Caribbean Archaeology and Ethnohistory Series. University of Alabama Press, Tuscaloosa, 216p

Skoglund RO, Lauritzen S (2010) Morphology and speleogenesis of Okshola, Fauska, Northern Norway: an example of a multi-stage network cave in a Glacial landscape. Nor J Geol 90:123–139

Stafford KW, Mylroie JE, Jenson JW (2004) Karst geology of Aguijan and Tinian, CNMI: cave inventory and structural analysis of development. Technical report no. 106, Water and Environmental Research Institute of the Western Pacific, University of Guam

Stanley S (ed) (1995) A global representative system for marine protected areas, Marine Region 2: The Wider Caribbean, vol 2. Great Barrier Reef Marine ark Authority, The World Bank, The World Conservation Union (IUCN), pp 15–42

Tanner BR, Perfect E, Kelley JT (2006) Fractal analysis of Maine's Glaciated shoreline test established coastal classification schemes. J Coast Res 225:1300–1304

Ternan JL, Williams G, Francis C (1989) Land capability classification in Grenada, West Indies. Mt Res Dev 9(1)

Thomas A (2000) Grenada, Carriacou and Petite Martinique, National report on the implementation of the U.N. convention to combat desertification and/or drought. Forestry Dept. MALFF, Grenada

Timcak GM, Jablonska J, Jurkasova Z (2010) Destination calibration process for tourism management of defined regions. Acta Geoturistica 2:1–12

Toomey RS III (2009) Geological monitoring of caves and associated landscapes. In: Young R, Norby L (eds) Geological monitoring. Geological Society of America, Boulder, pp 27–46

Troester JW, Turvey MD (2004) Water-resources reconnaissance of Ile de la Gonave, Haiti. Hydrogeol J 12:224–236

UNEP (2012) Road map on building a green economy for sustainable development in Carriacou and Petit-Martinique, Grenada. Report prepared by the Division of Sustainable Development of the United Nations Department of Economic and Social Affairs, 131p

US Army Core of Engineers (1999) Water Resources Assessment of Haiti. 35p

Wampler PJ, Sisson AJ (2011) Spring flow, bacterial contamination, and water resources in Rural Haiti. Environ Earth Sci 62:1619–1628

Waterstrat WJ, Mylroie JE, Owen AM, Mylroie JR (2010) Coastal caves in Bahamian eolian calcarenites: differentiating between sea caves and flank margin caves using quantitative morphology. J Cave Karst Stud 72:61–74

Watson J (ed) (1997) Guidelines for cave and karst protection. IUCN, Cambridge

Willig MR, Presley SJ, Bloch CP, Genoways HH (2010) Macroecology of Caribbean bats: effects of area, elevation, latitude and hurricane-induced disturbance. In: Fleming TH, Racey PA (eds) Island bats: evolution, ecology and conservation. University of Chicago Press, Chicago, pp 216–264

Wilson SM (2007) Archaeology of the Caribbean. Cambridge University Press, Cambridge, p 56

Woods CA, Harris L (1986) Stewardship plan for the National Parks of Haiti. Report prepared for USAID. 272p

Part IV
Coastal Marine Environmental Conflicts: Advances in Conflict Resolution

Chapter 7
Mud Crab Culture as an Adaptive Measure for the Climatically Stressed Coastal Fisher-Folks of Bangladesh

Khandaker Anisul Huq, S.M. Bazlur Rahaman, and A.F.M. Hasanuzzaman

Abstract The coastal belt of Bangladesh endowed with diverse resources bears a significant scope in ensuring the security of food and livelihood of the coastal poor people. But the coastal belt of Bangladesh is highly affected by the natural calamities and salinity intrusion which are occurred due to climate change. Considering the above situation the present study was developed to conduct on-farm adaptive research on crab fattening/culture in Barguna and Patuakhali districts as a livelihood option for the fisher folks including female from September 2010 to August 2012. In case of crab fattening the main target is gonad development of female crab and shell hardening of male crab rather than weight gain. To conduct the experiment 24 study sites were selected where six cages and six pens were set in each district. The size of each cage was 2 m × 1 m × 0.3 m (length × width × height) comprising of 50 cells. And the length, width and height of each cell were 0.2, 0.2 and 0.3 m respectively. One lean crab (>180 g) was stocked in each small cell of the cage. The area of the pen was 100 m^2 (10 × 10 m) where crab (50–150 g) stocked at three different densities (3, 5 and 7/m^2). Low valued small waste fishes and formulated pellet feed were fed. Raw feed was given as wet weight basis at the rate of 8–10 % of the total body weight twice a day. Suitable salinity (3–15 ppt) for crab fattening/culture was found during November to June survival and gonad maturation were found higher during the time. Gonad maturation of mud crab occurred over the year but higher was found from December to May. Economic analysis revealed that it is possible to earn net profit 1,313.83–1,4761.30 BDT per cycle (cage 10–20 days, pen 20–30 days) from crab fattening/culture in one cage (50 celled) and pen (100 m^2) respectively. Recommendation can be made to conduct comprehensive crab culture extension program for building capacity and improving economic condition of the climatically stressed coastal community.

K.A. Huq (✉) • S.M.B. Rahaman • A.F.M. Hasanuzzaman
Fisheries and Marine Resource Technology Discipline, Khulna University, Khulna 9208, Bangladesh
e-mail: huqka@yahoo.com; riti_rahaman@yahoo.com; mhzaman.bd@gmail.com

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8,
DOI 10.1007/978-3-319-06305-8_7,

7.1 Introduction

Mud crab is a very potential commodity for the earning of foreign exchange of Bangladesh. Worldwide *Scylla serrata*, mud crab is popular as a palatable and valuable food item. Besides 15–25 % protein, mud crab contains 1 % fat and 2–3 % minerals. Mud crabs are collected from the coastal region of Bangladesh, especially from Khulna, Satkhira, Bagerhat, Barguna, Potuakhali, Noakhali, Vola, Hatia, Cox's Bazar etc. Mud crabs can survive 2 ppt to more saline marine environment. They are distributed in various channels of coastal regions. During 2010–2011 fiscal year, total production of mud crab in Bangladesh was about 1,150.70 tons that contributed to a revenue income of 4,323,740,740.30 taka (FD 2011), in the international market its value was more than 540 million Taka (DoF 2012) however, it is assumed that a significant portion of total catch has not been recorded in Forest Department year after year. In the country crab is sold up to 400–1,500 BDT/kg for export to the international market. Usually the demand and export value of 180–500 g weighing crab is higher in foreign countries.

Crab of Bangladesh is exported to various countries such as China, Taiwan, Thailand, Singapore, Malaysia, Japan, Hong Kong, America and different countries of Europe. Although demand of crab has recently been increased in the international market but in Bangladesh its production or culture system has not been developed. Usually the collectors collect all sizes of crabs from the natural sources. The market value of small sized, soft shelled and gonad immature crab is lower, and it is sold in the local market at a very low price. However, the price will be very higher after shell hardening and gonad maturation by stocking the crabs in cage or pen for 10–20 days with proper feeding and other management. This process called crab fattening. On the other hand, if a certain area of open water is enclosed with bamboo fence or nylon net for crab fattening or culture, then it is called pen culture.

The coastal belt of Bangladesh is endowed with diverse resources that indicate a significant scope of ensuring food security and livelihood of the coastal poor people. But this region is highly prone to the effect of climate change. The effect of climate change is deteriorating food security and livelihood of coastal resource-poor people. Human life and crops in the coastal areas of Bangladesh are damaged by excess tidal fluctuation, salinity intrusion, cyclone, flood and tidal surge due to climate change. Though man cannot control this adverse impact of climate change but it is possible to coup these natural disasters to certain level by introducing various crop variety and culture systems which are adaptive in changing climatic condition. Fortunately, the perpetual sea-level rise and salinity intrusion is making the coastal waters economically and environmentally feasible for aquaculture. Among the potential species, the Mud crab, *Scylla serrata*, is a lucrative candidate with its high market price. Mud crab fattening in cage and pen will be a good alternative crop by which the poor coastal people can maintain their livelihood.

The mud crabs inhabits abundantly in the estuaries, tidal rivers of the Sundarbans mangrove swamps, and vast coastal ghers or shrimp polders along the coastal region of Bangladesh (Khan and Alam 1991). Export of mud crab started in 1977–1978, and became a stable business in 1982 and ranked third among frozen foods exported from Bangladesh (SEAFDEC 1998; Ali et al. 2004). The export of live mud crabs from Bangladesh has increased manifolds in the last decade (BEPB 2004). Generally Hong Kong, Malaysia, Taiwan, Singapore, and European Union import mud crab from Bangladesh. Zafar and Ahsan (2006) reported that a substantial part of wild crabs has no market value simply because of soft shell or inadequate female gonad. Such lean crabs can easily reach marketable size when extra value is added through fattening technology. The crab fattening technology feasible in the coastal belt of Bangladesh has been developed by the Bangladesh Fisheries Research Institute, Brackishwater station, Khulna (Saha and Ahammad 1999).

Begum et al. (2006) worked on fattening crab in bamboo cage (10″ L × 10″ W × 12″ H) and in fish ghers (0.5–0.1 ha) in south-western coastal region of Bangladesh. In cage, one individual/cell and 80 individual/dec in gher, crab of >175 g was stocked, and feeding at the rate of 8–5 % of the total biomass was given. Zafar and Hossain (2008) conducted an experiment on pen culture of mud crab in the mangrove swamp (20 × 20 m) of Cox's Bazar. They stocked male crabs of 180–250 g, and soft shells female crabs of 130–170 g at the rate of 1/m^2 and 0.5 m^2. The crabs were fed with eel, goby and trash fish at the rate of 3 % body weight.

Though crab fattening is practiced by the local poor fisher-folks along the coasts of Khulna region, the crab culture technology in terms of cage and pen aquaculture is not hitherto widespread in the coastal belts of Bangladesh, site-specifically Patuakhali district. But both these culture technology of crab farming are very likely feasible in this coastal areas. The pen or cage culture in tidal rivers, swamps, and shrimp ghers are also deemed environment friendly aquaculture technology. These culture techniques can be disseminated into the coastal fish farmers through hands-on training and workshop. Once the coastal communities get trained with these technologies, they may use tidal rivers, their fish ghers, and mangrove swamps for cultivating crab and thus will improve their socio-economic condition. This study was conducted with the aim of building adaptive capacity of climatically vulnerable poor marginal coastal fisher-folks with special focus on improvement of livelihood.

7.2 Methodology

7.2.1 Selection of Experimental Sites and Target Farmers

After several field surveys appropriate sites (Fig. 7.1) were selected for the implementation of the project. Kathaltoli, Charduani and Pathorghata Sadar union of Pathorghata upazila, Barguna district, and Tiakhali, Khaprabhanga and Mithagonj union of Kalapara upazila, Patuakhali district were finally selected as the study points (Table 7.1). During the survey, the project team consulted and discussed with

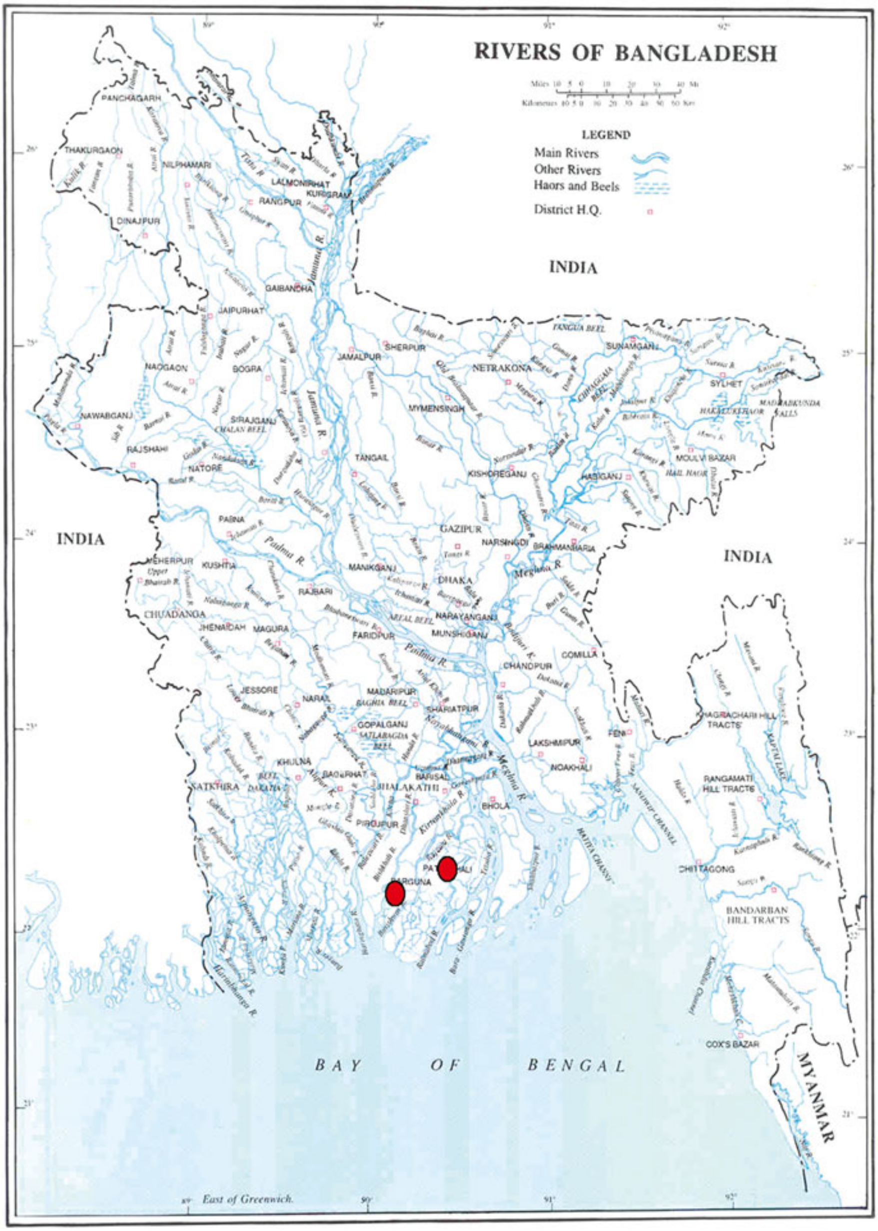


Fig. 7.1 Map of Bangladesh indicating two experimental districts

ASPS II personnel, CBOs formed by RFLDC Barishal, FFS, UFO of Department of Fisheries, and other stakeholders to refresh and improve the project implementation plan, as well as to select farmers from CBOs for adaptive research.

Table 7.1 Name and location of the experimental sites

District	Upazila	Union	Latitude	Longitude
Barguna	Pathorghata	Pathorghata Sadar	22°01′–22°04′	89°56′–89°59′
		Kathaltoli	22°05′–22°08′	89°56′–90°00′
		Charduani	22°03′–22°07′	89°54′–89°58′
Patuakhali	Kalapara	Nilgoanj	21°54′–21°59′	90°07′–90°15′
		Mithagonj	21°51′–21°58′	90°12′–90°16′
		Khaprabhanga	21°52′–21°54′	90°06′–90°10′

7.2.2 Training of the Farmers for Adaptive Research Trial

Training workshops were organized at Pathorghata and Kalapara upazila for CBOs, FFS, LFs and other stakeholders to carry out the research works successfully. Selected farmers were trained up through different sessions of the workshop program about site preparation, cage and pen setting, stocking and feeding, water quality management, health and gonad maturation checking, harvesting, grading etc. Emphasis was given to group discussion where farmers shared their farming experience and knowledge. Feedbacks from the participants were noted down and later it contributed to further refreshment of the project implementation plan.

7.2.3 Adaptive Research Trial

The research trial was conducted with the involvement of CBOs/FFS of ASPS II component. This trail was comprised of mud crab culture in pen, and crab fattening in cage in previously mentioned experimental locations. Adaptive research was conducted in three unions of each upazila where trained CBO/FFS farmers were available. A total of 24 study sites were selected at six unions of Kalapara and Patharghata upazila. However, very few study points had to be relocated after conducting the experiment for a certain period due to unwillingness of the particular farmers and improper management of the farms.

7.2.4 Crab Fattening in Cage

Tidal channel, estuaries, rivers, ghers, canals and low lands etc. water bodies where salinity remain more than 5 ppt at least 6–7 months in the year are suitable for crab fattening. Crab cultivable salinity has been found from November to June in the coastal rivers of the study area. Growth and gonad development of mud crab were not desirable when the water salinity was almost zero/freshwater in the rainy season. In rainy season during heavy rainfall crab reduced its food intake; therefore their growth and gonad development were not satisfactory and in this condition crab showed high mortality. In the winter during ‘cold wind blow’ if water temperature fall down abruptly to 15 °C or lower, then food intake and growth reduced and at last they might die. For this reason it is better to set the cages where light and air is enough.

7.2.5 Cage Preparation

Cages were made with matured bamboo, thread and wire which are locally available in Bangladesh. The size of each cage was 2 m × 1 m × 0.3 m (length × width × height) comprising of 50 cells. And the length, width and height of each cell were 0.2, 0.2 and 0.3 m respectively. The size of the cage might vary with requirement. During cage preparation the bamboo slots are arranged in such way, so that the appendages will not be broken by entangling between the open spaces. The cover of the cage was fitted in such way, so that it can open easily to give feed and also be used for other management. The cage was placed in such way, so that it will not touch the bottom during the low tide. In many experimental points plastic drums were tightened at four upper corner of the cage so that at least 5 cm top level of the cage remains above the water surface. This placement of the cages helped the stocked crabs to take oxygen from the air.

7.2.6 Stocking of Crabs in Cages

It is already mentioned that the experiment was conducted in six unions of Kalapara and Pathorghata Upazila. Crab fattening was done in two locations of each union i.e. in total there were 12 cages in 2 upazilas. Fifty lean crabs (weight >180 g) were stocked in each floating bamboo cage comprising of 50 cells. One individual lean crab was stocked in each cell i.e. 50 individual crabs were stocked in each cage.

7.2.7 Feeding Management in Cages

For feeding the crab, low valued small tapashi, vola, kukur jib (which are locally called ‘bairagi’), Chewa, kuchia, tilapia, small shrimp, mollusk meat and formulated pellet feed were used. Raw feed is given at the rate of 8–10 % of the crab’s body weight, and the feeding frequency was twice a day (morning and evening). The feeding rate was adjusted according to the feeding tendency of the crabs. The crab is carnivorous base omnivore animal. Supplied feed should contained 30–40 % protein. Adult crabs were found hardly to eat commercial pelleted feed. It is assumed that crabs can be habituated to supplementary feed if it is started at their larval stages. Crab fattening experiment was conducted during both summer and winter to see the effects of temperature and salinity variations on growth and survival of crabs. Gonad maturation was found best (80–100 %) during January to May. This time might be the peak spawning season of mud crab. The salinity was found suitable (3–15 ppt) during the dry season (November to June) for mud crab culture/fattening.

7.2.8 Post Stocking Management in Cages

The cages were cleaned properly after few days' interval so that the open spaces are not closed by waste materials and aquatic weeds or algae. After 8–10 days of stocking gonad of the crabs was checked to observe the development stage. If light does not pass through the abdomen near the base of the chelate legs then it becomes gonadally matured. Gonad maturation was preferably tested under sunlight or using electric bulb. Gonad development can also be checked by a torch light. If the gonad is found empty or partially fulfilled with egg then it is kept in the chamber again and feed regularly.

7.2.9 Crab Culture/Fattening in Pen

When a part of an open water body is enclosed by bamboo fence/nylon net for crab fattening/culture then that part of water body is called pen. Rivers, estuaries, ghers, ponds and other low lying water bodies of coastal areas of Bangladesh where salinity remain more than 5 ppt in the most of the time of the year are suitable for pen culture. Loamy or sandy loamy soil is suitable for crab culture. Three suitable sites have been selected from each of two upazilas and in each location two CBO farmers were involved in pen culture or fattening activities. Pen culture of crabs was also conducted in summer and winter to evaluate the influence of temperature and salinity fluctuations on growth and gonad maturation.

7.2.10 Pen Preparation and Setting

Similar to cage the pens were also made with ripe bamboo, thread and wire which are available in Bangladesh. Selected areas were enclosed by bamboo fence. The size of the pen was 100 m^2 and the length, width and height of the pen were 10, 10 and 1.5 m respectively. The size of the pen may vary depending on the total stocked crabs. During pen preparation, the bamboo slots were arranged properly so that the appendages are not broken by entangling between the open spaces. Management for crab culture/fattening in pen would be easier if the area of the pen ranges between 100 and 1,000 m^2. In case of long time culture the area of the pen might be larger. To prevent the crabs from escaping the pen should be kept 0.5 m above the surface water. During unusual high tide or any other situation the pen can be encircled with nylon net at the top of the pen. The pen should be set at least 0.25 m under the bottom mud to prevent escape through burrowing.

7.2.11 Stocking of Crabs in Pens

Mud crab has high resistance capacity against environmental stress and disease; therefore it can be fattened/cultured in pens with comparatively high stocking density. Small crabs weighing 50–100 g were stocked in pens for 4–6 months at a stocking density of 3, 5 and 7/m^2. However, after 45–60 days of stocking many of these, comparatively small sized crabs were found to become gonadally developed with an average size of 125 g. When gonadally matured small crabs continued for further few days then the crabs faced mortality and the rate of mortality increased with the extension of the stocking period. After 3 months about 40 % mortality was occurred. The market price (200–250 Tk/Kg) of small sized gonad developed crab is very low comparing large one (>400 Tk/Kg). Therefore, after then comparatively large (weighing 150 g or above) gonadally undeveloped female or soft shelled male crabs were stocked in pens for fattening, so that the crabs can become gonadally matured or hard shelled within short time (20–30 days) and to get higher market price. It is better to stock female crab, because culture of female mud crab is more profitable. During stocking, the size of the crabs should be more or less similar; otherwise larger crabs may eat or damage the newly molted small crabs physically or even dominate over the small sized crabs during feeding.

7.2.12 Feeding Management in Pens

In case of pen culture, bairagi (low valued small tapashi, vola, kukur jib) Chewa, kuchia, tilapia, small shrimp, mollusk meat and formulated pellet feed were also used. Raw feed was applied at the rate of 8–10 % of the total body weight of crabs. Feeding was done twice a day (morning and evening). Using feeding trays the rate was readjusted following the feeding affinity of the crabs. Crabs showed very little tendency to eat commercial pellet feed. Feeding tendency of the crabs was observed by supplying required amount of feeds in trays as it is very important to estimate the feeding rate depending on feed consumption.

7.2.13 Post Stocking Management in Pens

Water quality was maintained very carefully for better growth, gonad maturation and to keep crabs free from diseases or other contamination. During new and full moon efforts were taken for water exchange. Sometimes lime (100 Kg/ha) was applied to maintain water quality in pen. Growth performance, gonad maturation and health of the crab were checked at a regular interval.

7.3 Results and Discussion

7.3.1 Water Quality Monitoring

The water quality of the two crab aquaculture systems, namely pen and cage culture being practiced in Kalapara, Potuakhali district and Patharghata, Barguna districts have been being recorded since March 2011–March 2012 (Tables 7.2 and 7.3). Water quality of cage and pen of the research site was recorder fortnightly. The recorded water parameters were salinity, temperature, pH, dissolve oxygen, hardness, transparency, alkalinity, free CO_2 and acidity. At Patharghata the average highest salinity of six pens and six cages was found 12.58 and 12.92 respectively with a maximum of 14 ppt (May 2011), whereas at Kalapara the average highest salinity in pen and cage was found 13.16 and 13.67 respectively with a maximum of 15 ppt in May, 2011. During rainy season from July to October, 2011 the salinity was found almost zero due to heavy rainfall and fresh water flow from the upstream throughout the experimental sites of both Upazilla. After rainy season salinity started rising from November and continue to May until rain start in monsoon.

The highest pH was found to be 7.83 in pen at May, 2011 and 7.89 in cage at March, 2011 at Patharghata Upazilla of Barguna District. And the lowest pH was found to be 7.32 and 7.27 in pen and cage respectively at December, 2011. However, the highest pH at Kalapara Upazilla was found to be 7.83 and 7.75 in pen and cage at March and August, 2011. On the other hand the lowest value was found to be 7.37 and 7.44 at December and October, 2011 respectively.

The highest level of dissolve oxygen was found to be 6.90 mg/l in pen and cage at November, 2011 at Patharghata Upazilla. And the lowest level was found to be 5.80 mg/l in pen at March–April, 2011 and 5.93 mg/l in cage at March–July, 2011. However, the highest level of dissolve oxygen at Kalapara was found to be 6.22 and 6.30 mg/l in pen and cage respectively at January, 2012. And the lowest level of dissolve oxygen was found to be 5.63 and 5.60 mg/l in pen and cage respectively at April, 2011 and October, 2011 & February, 2012.

The highest average water temperature was found to be 29.33 and 29 °C in pen and cage respectively. And the average lowest temperature was found to be 20.43 and 19.84 °C in pen and cage respectively in January, 2012. On the other hand, the

Table 7.2 Month wise water quality parameters in pen and cage at Patharghata Upazilla from March 2011 to March 2012

	Pen					Cage				
Month	Salinity	pH	DO	Transp.	Temp.	Salinity	pH	DO	Trans.	Temp.
March	8.33	7.73	5.80	27.75	25.67	8.58	7.89	5.93	27.83	26.33
April	10.75	7.78	5.80	27.92	27.33	10.92	7.74	5.93	27.75	26.92
May	12.58	7.83	5.80	27.92	28.33	12.92	7.78	5.93	27.58	28.50
June	4.00	7.78	5.80	27.92	29.25	3.83	7.83	5.93	27.75	29.00
July	0.00	7.79	5.80	27.92	28.33	0.00	7.75	5.93	27.75	27.50
August	0.00	7.74	5.80	27.92	28.75	0.00	7.76	5.95	27.50	28.50
September	0.00	7.78	6.64	29.25	27.50	0.00	7.71	6.65	28.33	27.50
October	0.00	7.55	6.61	28.50	28.50	0.00	7.55	6.60	27.42	28.50
November	2.67	7.55	6.90	29.50	26.00	2.83	7.55	6.90	29.50	26.00
December	3.50	7.32	6.45	25.42	21.99	3.42	7.27	6.52	26.50	21.17
January	5.17	7.35	6.41	25.83	*20.43*	5.25	7.36	6.25	25.83	19.84
February	7.50	7.42	6.31	26.33	22.77	7.83	7.46	6.33	26.17	22.94
March	10.00	7.35	6.13	27.83	26.10	9.50	7.40	6.12	26.50	25.83

Table 7.3 Month wise water quality parameters in pen and cage at Kalapara Upazilla from March 2011 to March 2012

	Pen					Cage				
Month	Salinity	pH	DO	Transp.	Temp.	Salinity	pH	DO	Trans.	Temp.
March	11.50	7.83	5.75	26.50	26.25	11.50	7.69	5.73	28.00	26.92
April	12.25	7.68	5.63	27.33	27.75	12.50	7.59	5.83	28.17	27.42
May	13.16	7.73	5.68	27.50	28.17	13.67	7.72	5.81	26.58	28.00
June	4.58	7.65	5.80	27.33	28.42	4.42	7.70	5.77	27.58	28.08
July	1.25	7.73	5.64	27.17	29.42	1.25	7.70	5.78	28.00	29.25
August	0.00	7.77	5.85	27.83	27.75	0.00	7.75	5.78	27.67	28.25
September	0.00	7.44	5.85	28.58	28.75	0.00	7.50	5.69	27.83	28.50
October	0.00	7.36	5.78	30.08	27.00	0.00	7.44	5.60	29.08	27.17
November	3.25	7.42	5.99	27.50	25.67	2.75	7.48	5.63	26.33	25.17
December	5.17	7.37	6.10	25.08	20.17	5.25	7.68	5.78	24.42	20.25
January	7.31	7.65	6.22	25.83	21.00	6.83	7.70	6.30	26.67	20.83
February	9.89	7.56	5.65	27.00	22.78	9.33	7.55	5.60	26.17	23.00
March	11.17	7.60	5.90	28.17	24.33	11.33	7.60	5.90	28.50	25.00

highest average level of temperature at Kalapara Upazilla was found to be 29.42 and 29.25 °C in pen and cage respectively with a maximum of 32 °C in July, 2011. However, the lowest average level of temperature was found to be 20.17 and 20.25 °C in pen and cage respectively in December, 2011. Sometimes unusual weather condition like abrupt fall of temperature and salinity might cause lower food intake which is ultimately responsible for higher mortality of crab. At the end of December, 2011 due to cold wind blowing for 3 days water temperature was fallen down to 15 °C at patharghata, therefore, unusual higher mortality (about 40 %) was occurred in pen which was under a shadow of big tree.

The highest transparency level was found to be 29.50 cm in both pen and cage at Patharghata upazilla at November, 2011 (Tables 7.2 and 7.3). And the lowest level of transparency was found to be 25.42 and 25.83 cm in pen and cage respectively at December, 2011 and January, 2012. However the highest level of transparency was found to be 30.08 and 29.08 cm at Kalapara upazilla at October, 2011. On the other hand, the lowest level of transparency at Kalapara was found to be 25.08 and 24.42 cm in pen and cage respectively at December, 2012. Except salinity other water parameters in both cage and pen of the research area were found within the cultivable range over the year.

7.3.2 Biological Performance of Mud Crab

7.3.2.1 Biological Performance in Pen at Patharghata

In case of pen, comparatively small sized crablings (50–100 g) were initially stocked with the objective of rearing them for long time (4–6 months). But about 45 days after of stocking when crabs attained 100–120 g weight then gonad maturation was found in the crabs and after a certain period crabs were found to face slow growth and higher mortality. As a result for the next time the farmers stocked comparatively large size crabs (>100 g) in all the pens for fattening where gonad development occurred within 20–30 days except some unusual situation. The average stocking density in pen PP-1 was $3/m^2$ in seven consecutive fattening/culture cycle. The average individual weight of all stocked crabs in PP-1 was 116.74 g. In all pens the trash fish, Chewa, kuchia, tilapia, small shrimp, mollusk meat used as wet weight basis and rate was 10 % of the total body weight of crabs. Average mortality of the pond PP-1 was 6.33 %, whereas the highest mortality of the pond was 8.67 % and lowest 4.00 %. Total average harvest weight 9.62 kg with maximum and minimum harvesting weight of 12.8 kg and 6.65 kg respectively. The mean stocking density in pen PP-2 was $3/m^2$ in seven consecutive fattening/culture cycle. The average individual weight of all stocked crabs was 136.99 g. Average mortality of the pond PP-2 was 7.55 %, whereas the highest mortality of the pond was 15 % and lowest 3 %. Total average harvest weight 12.61 kg with maximum and minimum harvesting weight of 12.74 kg and 9.64 kg respectively. The average stocking density in pen PP-3 was $5/m^2$ in eight consecutive fattening/culture cycle. The average individual weight of all stocked crabs was 127.13 g. Average mortality of the pond PP-3 was 8.06 %, whereas the highest mortality of the pond was 13.0 % and lowest 5.0 %. Total average harvest weight 15.78 kg with maximum and minimum harvesting weight of 18.3 kg and 12.5 kg respectively (Table 7.4).

The average stocking density in pen PP-4 was $5/m^2$ in five consecutive fattening/culture cycle. The average individual weight of all stocked crabs was 102.90 g. Average mortality of the pond PP-4 was 11 %, whereas the highest mortality of the pond was 20 % and lowest 5 %. Total average harvest weight 16.22 kg with maximum and minimum harvesting weight of 19.62 kg and 14.35 kg respectively.

Table 7.4 Biological performance of crab in pen

Pathorghata				Kalapara		
Pen no	Average stocking weight (g)	Average mortality	Average harvest weight/pen (kg)	Average stocking weight (g)	Average mortality	Average harvest weight/pen (kg)
1	116.74	6.33	9.62	150.28	8.5	11.99
2	136.99	7.55	12.61	150.47	7.74	11.85
3	127.13	8.06	15.78	150.71	9.53	19.26
4	102.9	11	16.22	145	7.78	18.72
5	131.32	8.97	22.21	151.73	7.15	28.05
6	135.84	19.75	20.32	151.43	11.94	26.57

The average stocking density in pen PP-5 was 7/m^2 in eight consecutive fattening/culture cycle. The average individual weight of the crabs during stocking was 131.32 g. Average mortality of PP-5 was 8.97 % with the maximum and minimum value of 15.5 % and 3 % respectively. Total mean harvest weight was 22.21 kg with maximum and minimum harvesting weight of 27.04 kg and 17.45 kg respectively. The average stocking density in pen PP-6 was 7/m^2 in six consecutive fattening/culture cycle. The average individual weight of all stocked crabs was 135.84 g. Average mortality of the pond PP-6 was 19.75 %, whereas the highest mortality of the pond was 40 % at the end of December 2011 when cold wind flowed for some days and lowest 9.5 %. Total mean harvest weight 20.315 kg with maximum and minimum harvesting weight of 11.07 and 25.2 kg respectively.

7.3.2.2 Biological Performance in Pen at Kalapara Upazilla

The stocking density in pen KP-1 was 3/m^2 in six consecutive fattening/culture cycle. The average individual weight of all stocked crabs was 150.28 g with the maximum and minimum individual weight of 66.67 and 175 g respectively. The average mortality of the pond KP-1 was 8.5 %. The mean total harvest weight was 11.99 kg. The stocking density in pen KP-2 was 3/m^2 in seven consecutive fattening/culture cycle. The average individual weight of all stocked crabs was 150.47 g with the maximum and minimum individual weight of 93.33 and 170 g respectively (Table 7.4). The average mortality of the pond KP-2 was 7.74 %. The mean total harvest weight was 11.85 kg. The stocking density in pen KP-3 was 3/m^2 in seven consecutive fattening/culture cycle. The average individual weight of all stocked crabs was 150.71 g with the maximum and minimum individual weight of 100 and 170 g respectively. The average mortality of the pond KP-3 was 9.53 %. The mean total harvest weight was 19.26 kg. The stocking density in pen KP-4 was 5/m^2 in seven consecutive fattening/culture cycle. The average individual weight of all stocked crabs was 145 g with the maximum and minimum individual weight of 80 and 165 g respectively. The average mortality of the pond KP-4 was 7.78 %. The mean total harvest weight was 18.72 kg.

Table 7.5 Biological performance of crab in cage

	Pathorghata			Kalapara		
Cage no	Ava. stocking weight (g)	Average mortality	Average harvest weight/cage (kg)	Average stocking weight (g)	Average mortality	Average harvest weight/cage (kg)
1	184.75	4.14	9.02	184.75	4.87	9.23
2	186.68	4.29	9.12	186.68	4.7	9.29
3	179.7	6	8.37	180.7	4.22	9
4	183.72	5.5	8.66	183.72	3.38	9.27
5	182	4.63	8.65	182	4.15	9.1
6	182.82	7.38	8.57	182.84	4.16	8.97

The stocking density in pen KP-5 was 7/m^2 in seven consecutive fattening/culture cycle. The average individual weight of all stocked crabs was 151.73 g with the maximum and minimum individual weight of 97.14 and 175 g respectively. The average mortality of the pond KP-5 was 7.15 %. The mean total harvest weight was 28.05 kg. The stocking density in pen KP-6 was 7/m^2 in seven consecutive fattening/culture cycle. The average individual weight of all stocked crabs was 151.43 g with the maximum and minimum individual weight of 90 and 170 g respectively. The average mortality of the pond KP-6 was 11.94 %. The mean total harvest weight was 26.565 kg.

7.3.2.3 Biological Performance in Cage at Pathorghata

The average individual weight of all stocked crab in cage PC-1 was 184.75 g with the minimum and maximum individual weight of 160 and 205 g. In all cages the trash fish locally called bairagi (low valued small tapashi, vola, kukur jib) Chewa, kuchia, tilapia, small shrimp, mollusk meat used as wet weight basis and rate was 10 % of the total body weight of crabs. The fattening culture period in all cages ranged 10–20 days except some unusual situation. Average mortality of the cage PC-1 was 4.14 %, whereas the highest mortality was found 6 % and lowest 2 %. Total average harvest weight was 9.02 kg with minimum and maximum harvesting weight of 8.3 and 10.08 kg respectively (Table 7.5). The average individual weight of all stocked crab in cage PC-2 was 186.68 g with the minimum and maximum individual weight of 170 and 215 g. Average mortality of the cage PC-2 was 4.29 %, whereas the highest mortality of crab was 10 % and lowest 2 %. Total average harvest weight was 9.12 kg with minimum and maximum harvesting weight of 8.3 and 9.9 kg respectively. The average individual weight of crab in cage PC-3 was 179.70 g with the minimum and maximum individual weight of 176 and 182 g. Average mortality of the cage PC-3 was 6.0 %, whereas the highest mortality of crab was 10 % and lowest 4 %. Total average harvest weight was 8.37 kg with maximum and minimum harvesting weight of 8.9 kg and 7.03 kg respectively. The average individual weight of all stocked crab in cage PC-4 was 183.72 g with the minimum and maximum individual weight of 175 and 202 g. Average mortality of the cage PC-4 was 5.5 %, whereas the highest mortality of crab was 8 % and lowest 0 %. Total average harvest weight was 8.66 kg with minimum and maximum harvesting weight of 7.6 and 10.25 kg respectively. The average individual weight of all stocked crab in cage PC-5 was 182.00 g with the minimum and maximum individual weight of 173.10 and 192 g. Average mortality of the cage PC-5 was 4.63 %, whereas the highest mortality of crab was 8 % and lowest 0 %. Total average harvest weight was 8.65 kg with minimum and maximum harvesting weight of 7.62 and 9.75 kg respectively. The average individual weight of all stocked crab in cage PC-6 was 182.82 g with the minimum and maximum individual weight of 174.6 and 202 g. Average mortality of the cage PC-6 was 7.38 %, whereas the highest mortality of crab was 15 % and lowest 2 %. Total average harvest weight was 8.57 kg with minimum and maximum harvesting weight of 6.84 and 9.5 kg respectively.

7.3.2.4 Biological Performance in Cage at Kalapara Upazilla

The average individual weight of all stocked crabs in cage KC-1 was 184.75 g with the minimum and maximum individual weight of 180 and 205 g. Average mortality of the cage KC-1 was 4.87 %, whereas the highest mortality of crab was 12 % and lowest 0 %. Total average harvest weight was 9.23 kg with minimum and maximum harvesting weight of 8.8 and 10.3 kg respectively (Table 7.5). The average individual weight of all stocked crabs in cage KC-2 was 186.68 g with the minimum and maximum individual weight of 170 and 215 g. Average mortality of the cage KC-2 was 4.70 %, whereas the highest mortality of crab was 10 % and lowest 0 %. Total average harvest weight was 9.29 kg with minimum and maximum harvesting weight of 7.9 and 10.8 kg respectively. The average individual weight of all stocked crabs in cage KC-3 was 180.70 g with the minimum and maximum individual weight of 176 and 205 g. Average mortality of the cage KC-3 was 4.22 %, whereas the highest mortality of crab was 8 % and lowest 0 %. Total average harvest weight was 9.00 kg with minimum and maximum harvesting weight of 8.5 and 9.34 kg respectively. The average individual weight of all stocked crabs in cage KC-4 was 183.72 g with the minimum and maximum individual weight of 182.5 and 210 g. Average mortality of the cage KC-4 was 3.38 %, whereas the highest mortality of crab was 7 % and lowest 0 %. Total average harvest weight was 9.27 kg with minimum and maximum harvesting weight of 8.25 and 10.46 kg respectively. The average individual weight of all stocked crabs in cage KC-5 was 182.00 g with the minimum and maximum individual weight of 173.10 and 192 g. Average mortality of the cage KC-5 was 4.15 %, whereas the highest mortality of crab was 6 % and lowest 0 %. Total average harvest weight was 9.1 kg with minimum and maximum harvesting weight of 8.51 and 9.57 kg respectively. The average individual weight of all stocked crabs in cage KC-6 was 182.84 g with the minimum and maximum individual weight of 174.6 and 202 g. Average mortality of the cage KC-6 was 4.16 %, whereas the highest mortality of crab was 6 % and lowest 2 %. Total average harvest weight was 8.97 kg with minimum and maximum harvesting weight of 8.61 kg and 9.4 kg respectively.

Table 7.6 Month wise mortality of crab in cage and pen of Ptahghata and Kalapara Upazilla from March, 2011 to March, 2012

		Patharghata		Kalapara	
Month		Pen	Cage	Pen	Cage
2011	M	6.10	3.50	6.51	2.00
	A	7.60	4.17	6.51	2.70
	M	5.40	4.40	6.97	2.23
	J	6.58	5.33	7.97	4.67
	J	8.30	5.67	8.74	5.61
	A	8.00	6.00	8.27	5.78
	S	10.00	8.00	10.56	6.35
	O	12.86	8.70	9.89	5.84
	N	12.96	3.33	6.04	3.34
	D	13.04	7.03	12.15	5.67
2012	J	12.73	7.50	11.41	6.67
	F	7.50	4.00	5.93	3.20
	M	5.96	3.33	4.60	2.35

7.3.3 Crab Mortality in Cage and Pen

Mortality is a very important factor in aquaculture and fisheries management. During the study period per cent mortality in cage and pen was recorded at both the areas Patharghata and Kalapara in each month/cycle (Table 7.6). Crab showed mortality over the year reflecting little bit higher in winter (December–January) and also in rainy season (July–October) in both research areas. Crab showed comparatively higher mortality in pen (5.40–13.04 %) than the cage (2.00–8.70 %). There was no remarkable difference in mortality between two culture areas.

7.3.4 Maturation Status of Crab

Gonad maturation performance of crab was checked regularly during the culture period. At the same time month wise gonad maturation (%) was also evaluated throughout the culture period (Fig. 7.2). From this study gonad maturation performance was found higher during December to May (80–100 %). From the study it was also found that salinity range was higher during this period (5–15 ppt). Gonad development (%) was found slow and less during rainy season (July–October) when freshwater was found in the study area. So, it could be assumed that salinity might have impact on the gonad maturation performance of *Scylla serrata*. Gonad development of female crab and shell hardening of male crab were started to checked after 8–10 days of stocking with 3–5 days interval to observe the development stage until marketable condition (>80 % gonad maturation). If proper management and feeding were done then gonad maturation and shell of hardening were occurred within 10–20 days in cage and 20–30 days in pen (Fig. 7.2).

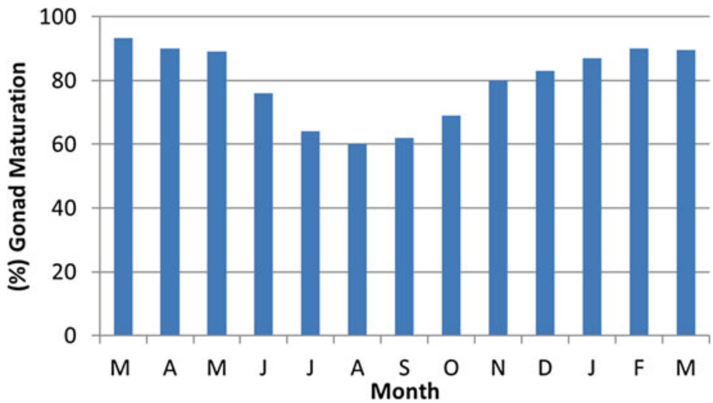


Fig. 7.2 Month wise gonad maturation status of mud crab using trash fish from March, 2011–March, 2012

7.3.5 Advantages of Fattening Crabs in Cages

- Preparation of cage is less costly and fattening is comparatively easy and economically viable in open water.
- Any family member can participate in the activities of crab fattening since feeding, checking, cleaning, harvesting etc. is very easy.
- As crab is stocked in separate chamber they cannot eat or damage each other and do not compete for feed.
- Checking of gonad maturation and shell hardening can be done easily and at last harvesting and marketing are less laborious.
- During sudden excessive high tide or tidal surge crabs cannot escape from the cage.
- Gonad maturation and shell hardening in cage is quicker (10–20 days) than pond or pen culture.

7.3.6 Effects of Feed on Gonad Maturation

The effect of different feeds (Kuchia, trash fish, chewa, pelleted supplementary feed) on gonad maturation of crab was checked. It was found that, crab did not like to eat the supplementary feed. Therefore, gonad maturation was found higher where crab was fed with Kuchia, trash fish, chewa rather than pelleted supplementary feed. At Patharghata crab showed highest gonad maturation where kuchia was used as feed, whereas at Kalapara highest gonad maturation was found when chewa was used as feed. The wild collected crab was not habituated to eat the pelleted supplementary feed. This situation can be try to solved to make the crab habituated with supplementary feed from its larval stage or to domesticate the animal producing in hatchery (Fig. 7.3).

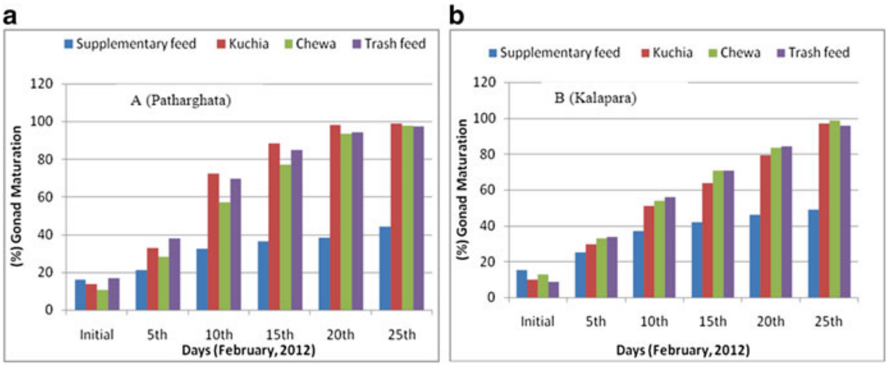


Fig. 7.3 Effect of different feeds on the gonad maturation of mud crab at Patharghata (**a**) and Kalapara (**b**)

Table 7.7 Proximate composition of different feeds and crabs

Feed item	Moisture	Protein	Lipid	Fiber
Male crab	74.2	19.5	2.8	
Female crab	72.3	18.82	3.1	
Formulated feed (CP) (dry weight basis)	12	38	5	4
Trash fish	74.93	19.08	3.38	
Chewa	74.22	24.8	4.69	
Kuchia	70.01	17.09	2.88	

7.3.7 *Proximate Composition of Cultured Crab and Crab Feeds*

During crab fattening/culture mainly low cost small trash fish (locally called bairagi), chewa and kuchia were used as feed. Formulated shrimp feed (CP brand) also used as crab feed to know the effect of formulated pillate feed. Proximate composition of feeds and crabs were analyzed in the 'Fish Post Harvest and Quality Control Laboratory' of Fisheries and Marine Resource Technology Discipline, Khulna University to know the nutritional value of feeds and products (Table 7.7). Among three raw feed (trash fish, chewa and kuchia) highest value of both protein and lipid was found in chewa. Incase dry weight basis protein and lipid was higher in formulated CP feed. Incase cultured crab protein level almost same.

7.3.8 *Crab Harvesting from Pen and Cage and Marketing*

After proper management and feeding the gonad of female crabs were fulfilled with eggs and the shell of male became hardened within in 10–20 days in cage, then the crabs were harvested by a clamp or lifting the cage. In case pen culture/

Table 7.8 Grading system of gonad matured/hard shell mud crab in the study area

Item	**Female**				
Name of grade	F1	KS-1[a]	F2	F3	F4
Weight (g)	>180	>180	150–<180	120–<150	<120
Price range (BDT)	250–1,000	100–350	200–500	150–400	100–300
Item	**Male**				
Name of grade	XXL	XL	L	M	SM
Weight (g)	>500	>400	>300	>200	>100
Price range (BDT)	250–800	200–650	150–450	100–350	50–200

[a]Immature female crab

fattening large sized (>100 g) gonadally undeveloped female and soft shelled male crabs became gonadally matured and hard shelled respectively between 20 and 30 days. After 10 days of stocking crabs were checked to know the status of gonad maturation and shell hardening with 3–5 days interval. The marketable sized crabs were harvested and sorted out for sell. Crabs were harvested from the pens using different types of gear such as bamboo traps, cast net, net trap, clamps, hooks and sometimes with the hands only. Harvesting can also be done by dewatering the cultured pond. After harvesting the chelate legs were tied with plastic tape and kept in bamboo/plastic basket for marketing. During harvesting and handling sufficient care was taken to avoid any physical damage or loss of any appendages of the crabs. Sell value of the crabs will significantly be dropped if it is physically damaged.

Usually live crabs are exported from Bangladesh to foreign countries. By spraying saline water it might be possible to keep the crabs alive up to 7 days. The marketing channel of crab is started from collectors to fatteners, farmers, different middlemen and at last exporters. Exporters export the crabs to the foreign countries.

Grading of crab is very important before packing and marketing. Grading has been done on the basis of sex (male and female), crab size, condition of gonad, shell hardness, color of the product etc. Large sized soft shell and gonadally immature crab has less demand in the local market and are not exported to the abroad. Comparatively large and gonadally developed crab has high demand and higher price in market (Table 7.8).

7.3.9 Economic Performance of Mud Crab Fattening

The economic analysis shows that there are substantial differences in net profit with respect to places where the culture has been being practiced. In Patharghata the highest and lowest net profit in cage (50 celled) were 1,313.83–706.91 BDT respectively for each fattening/culture cycle (10–20 days, Table 7.9). Whereas the highest and lowest profit in cage at Kalapara were 1,197.94 and 695.65 BDT respectively per culture cycle (Table 7.10). If the cage is made of bamboo with good quality, it will last more than a year. Crab fattening in cage can be continued for 18 cycles in a year.

Table 7.9 Economic evaluation of mud crab culture in cage at Patharghata (March, 2011–March, 2011)

	Cage ID					
Items/parameter	PC-1	PC-2	PC-3	PC-4	PC-5	PC-6
Stocking/cage	50	50	50	50	50	50
Individual wt range of crablings (g)	160–205	170–215	176–182	175–202	173.10–192	174.6–202
Average individual wt (g)	184.75	186.68	179.70	183.72	182.00	182.82
Total weight (kg)	9.24	9.33	8.98	9.18	9.10	9.14
Lean crab price/kg	248.33	220	228	236.88	243.75	250
Crab purchasing cost (tk.)	2,428.67	2,059.75	2,049.80	2,189.33	2,225.13	2,300.91
Feeding cost (tk.)	450	301.5	890.00	335.38	374.63	318.13
Total cost (tk.)	2,924.17	2,101.29	2,337.40	2,605.96	2,724.75	2,694.03
Average individual harvest wt (g)	190.63	192.17	185.10	188.11	186.44	187.25
% of mortality	5.33	5	9.60	8	7.13	8.50
Total harvest weight (kg)	9.02	9.12	8.37	8.66	8.65	8.57
Selling price/kg	429.17	370.83	400	394.38	356.25	415
Total selling price (tk.)	3,933.08	3,415.13	3,360.64	3,436.45	3,114.21	3,572.56
Net profit (Tk.)	1,008.92	1,313.83	1,023.24	790.74	706.91	878.53

Table 7.10 Economic Evaluation of mud crab culture in cage at Kalapara (March, 2011–March, 2012)

	Cage ID					
Items/parameter	KC-1	KC-2	KC-3	KC-4	KC-5	KC-6
Stocking/cage	50	50	50	50	50	50
Individual wt range of crablings (g)	180–205	170–215	176–205	182.5–210	173.10–192	174.6–202
Average individual wt (g)	184.75	186.68	180.70	183.72	182.00	182.84
Total weight (kg)	9.24	9.33	9.03	9.18	8.08	9.14
Lean crab price/kg	230.5	250	275.5	286.88	263.5	250
Crab purchasing cost (tk.)	2,137.93	2,343.38	2,487.75	2,636.98	2,126.16	2,296.86
Feeding cost (tk.)	450	176.67	172.78	168.75	182.50	188.89
Total cost (tk.)	2,650.43	2,576	2,726.42	2,839.48	2,319.91	2,542.42
Average individual harvest wt (g)	189.53	191.25	186.20	189.51	185.55	186.25
% of mortality	6	5.37	7.56	7.25	7.5	7.56
Total harvest weight (kg)	9.23	10.27	9.00	9.23	9.1	8.97
Selling price/kg	361.25	380.50	410	405.25	386.25	415
Total selling price (tk.)	3,346.06	3,545.12	3,689.05	3,784.13	3,517.86	3,724.52
Net profit (Tk.)	695.65	968.96	962.63	944.64	1,197.94	1,182

Table 7.11 Economic evaluation of mud crab culture in pen at Patharghata (March, 2011–March, 2012)

	Pen ID					
Items/parameter	PP-1	PP-2	PP-3	PP-4	PP-5	PP-6
Stocking/m^2	3	3	5	5	7	7
Individual wt of crablings (g)	61.2–157.33	82.22–180	94–160	37.50–164	100–160	60–180
Average individual wt (g)	116.74	136.99	127.13	102.9	131.22	135.84
Total weight (kg)	35	40.56	63.52	55.24	91.84	96.064
Crabling price/kg (Tk)	118.99	115.43	183.75	116	183.13	181.67
Crab purchasing cost (Tk.)	4,164.65	4,681.841	11,671.8	6,407.84	16,818.66	17,451.95
Feeding cost (Tk.)	1,375.44	1,245.72	1,788.52	4,043.2	3,866.52	4,085.32
Total cost (tk.)	6,000	6,719.24	15,399.96	12,734.4	23,756.36	25,044.68
Average individual harvest wt (g)	148.9	155.91	139.43	145.37	141.74	144.3
% of mortality	6.33	8	9.25	12	9.36	19
Total harvest weight (kg)	38.48	42.44	63.12	64.88	88.84	81.26
Selling price/kg	244.67	278.57	370	262	374	343.33
Total selling price (Tk.)	9,414.902	11,822.51	23,354.4	16,998.56	33,226.16	27,899
Net profit/loss (Tk.)	3,414.902	5,103.271	7,954.44	4,264.16	9,469.8	2,854.316

In case of pen (100 m^2) at Patharghata the highest and lowest net profit were 9,469.8 and 2,854.35 BDT respectively in each culture cycle (20–30 days, Table 7.11). Whereas the highest and lowest profit in pen at Kalapara were 14,761.30 and 3,689.88 BDT respectively per culture cycle (Table 7.12). In a pen, culture may be continued for eight cycles in a year. Mud crab has high resistance capacity against environmental stress and disease; therefore it can be fattened/cultured with comparatively high stocking density (3–7 crab/m$^{2)}$. There was no remarkable difference in growth, mortality and gonad development among the three different stocking densities 3, 5 and 7 crabs/m^2. Highest benefit was occurred where stocking density was 7 crab/m^2 in pen KP-6. Investment cost was higher in pen than the cage. The variation of profit depends on crab growth, amount of gonad maturation, survival rate, up-down of market price etc. Sometimes the growth, gonad development and survival rate varied with the intensity of care and proper management exercised by the farmers. Both pen and cage culture of mud crab in Patharghata in Barguna district fetched profits and were found economically viable. The selection of cage or pen culture could be depend on the suitability of the culture site and investment ability of the farmers.

Table 7.12 Economic evaluation of mud crab culture in pen at Kalapara (March, 2011–March, 2012)

	Pen ID					
Items/parameter	KP-1	KP-2	KP-3	KP-4	KP-5	KP-6
Stocking/m^2	3	3	5	5	7	7
Individual wt of crablings (g)	66.67–175	93.33–170	100–170	80–165	97.14–175	90–170
Average individual wt (g)	150.28	150.47	150.71	145	151.73	151.43
Total weight (kg)	45.08	45.16	75.36	70.04	104.96	104.76
Crabling price/kg (Tk)	93.33	114.28	150	133.33	142.857	146.429
Crab purchasing cost (Tk.)	4,207.316	5,160.885	11,304	9,338.433	14,994.27	15,339.9
Feeding cost (Tk.)	266.68	682.84	414.28	460	531.44	1,391.44
Total cost (tk.)	4,899.16	6,027.12	12,075	10,030.16	15,948.52	17,648
Average individual harvest wt (g)	164.25	162.28	163.57	161.667	165.857	166.78
% of mortality	5	3.55	5.95	4.12	2.31	8.23
Total harvest weight (kg)	47.96	47.4	77.04	74.88	112.2	106.26
Selling price/kg	205	205	246.43	211.7	255.7	305
Total selling price (Tk.)	9,831.8	9,717	18,984.97	15,852.1	28,689.54	32,409.3
Net profit/loss (Tk.)	4,932.64	3,689.88	6,909.967	5,821.936	12,741.02	14,761.3

7.3.10 Training, Demonstration and Extension

For training and demonstration part of the project, several workshops and on-farm training programs were arranged in Pathorghata and Kalapara upazila of Barguna and Patuakhali districts where selected farmers, local facilitators, monitoring officer, other stakeholders of selected six unions were participated. It is already mentioned that a total of 12 CBO/FFS farmers were selected from the set six study locations through group discussion along with consultation and collaboration of ASPS II RFLDC-Barisal technical support units. At the beginning of adaptive research trial training was given to the farmers on mud crab culture (crab selection, stocking, feeding, gonad check, harvesting and marketing). Then on-farm training program was organized for the farmers where they were trained up for culture and fattening of mud crabs in their study locations and further suggestions/advice were given if there is any lacking/mistake on activities. Considering the importance of cage and pen preparation, handicrafts makers were invited in the study areas and they gave hands-on training to the farmers for making cage and pen using bamboo and other necessary materials. Training manual, booklet, leaflet, presentation and abstract in national and international seminar, webpage circulation was made and manuscript is preparing to publish in reputed journal. Two theses have been made for the fulfillment of Master of Science degree from Fisheries and Marine Resource Technology Discipline of Khulna University which will be kept in library and students, academician and research would use this to earn knowledge about crab fattening in cage and pen.

7.3.11 Involvement of CBOs in Linking the Farmers to the Regional Market

In taking into account of the marketing of the produced crab from the demonstrated project; CBOs were highly encouraged to be involved. A product promotion group was formed by selecting two CBO members from each of six unions of two upazillas. This group was principally involved as intermediary in promoting produced product, and analyzing marketing approaches. Marketing the product at depots, hotels, and markets; communicating with traders; collecting market-relevant information, and dispensing them to the farmers were the activities of the group.

During farmers filed days, training program and workshops; different crab marketing issues such as product supply and demand, price, grading system, transportation, marketing channel, export requirements etc. were discussed and suitable marketing strategies were determined.

7.3.12 Developing Marketing Group

For the development of marketing system in the proposed areas a marketing group was established with the involvement of CBO/FFS members. They assessed the market condition of crabs in different areas of the country in particular south west coastal region and tried to provide information to the farmers. They had also been trained to make a social network in marketing their products and to make proper profit. Training workshops entitled “Training on Marketing of Mud Crab for the Fisher-Folks of Patharghata and Kalapara Upazilla” were arranged in Mongla, Paikgacha and Shyamnagar upazila of Bagerhat, Khulna and Satkhira district during project implementation period. During workshops, group discussions were made on different topics such as crab’s biology, their culture system including pond/gher preparation, stocking, feeding, water quality and other post stocking management, disease control, gonad and growth checking, harvesting, transportation and various marketing issues like, grading, pricing, marketing channel, export market, buyers preference, peak and off peak season in international market, etc. Crab market stakeholders exchanged their views about the present status of linkage among the local crab markets, scope and necessity of linkage, existing problems of grading, pricing and transportation system, and many other issues. Several field visits were also arranged for the members of the marketing group in crab markets of the respective areas to make them aware about marketing system and to build up linkage between the farmers and buyers of the local market.

7.3.13 Marketing Status

- Marketing channel is not well established and convenient to farmers particularly.
- Market prices fluctuate rapidly, and it is controlled by local depot owners.
- The main export market is very far from the main collection area and the tracks loaded with crab have to be cross a big river. But river crossing facility is not enough and most often it is too late.
- Illegal money demanded by the police man during their duty on the transport of crab.
- Usually it is not possible to reach the track on due time in the airport and catch the flight. As a result a substantial quantity of the products is at the risk of being destroyed.

7.4 Achievements of the Research

- Target farmers have motivated and accepted the crab culture technology as their additional income generation option and some of them have already started practicing such culture in large scale with their own initiatives.
- The neighbors of the target farmers have been encouraged to do crab culture by realizing the profitability of this simple technology, and some have already started such crab culture or fattening practice.
- The Target farmers have been trained to make cage and pen by themselves using bamboo, rope and other necessary materials; they could use these cage and pen either for fattening and culturing crabs, or selling in the market among farmers for income generation.
- Women were found very much encouraged in crab culture and thus they took part to keep their contribution in the economy of their family.

7.5 Recommendations

- The large scale crab culture along the coastal belts of Bangladesh needs further meticulous study on artificial feed development and breeding program to reduce pressure on natural food and wild crab seed leading to achieve biodiversity and conservation goals.

- A comprehensive crab culture extension program for building capacity and improving economic condition of the climatically stressed coastal community need to regular training, and motivation approaches by Department of Fisheries, Department of Forests, and other concerned departments.
- To reduce the exploitation pressure on natural source, culture of mud crab is very much urgent to start. Therefore, GOs, NGOs and concerned departments should take initiative on large scale mud crab culture along the coastal regions of Bangladesh.

Acknowledgement A research team of Fisheries and Marine Resource Technology Discipline of Khulna University carried out the research project in two coastal districts namely Patuakhali and Barguna with the financial support of DANIDA, RFLDC-Barisal. The on-farm adaptive research project was executed by the joint efforts of the project team, participating CBO and FFS members of ASPS II, and resource personnel of RFLDC-Barisal. The research team is very much grateful to the organization and personnel who are involved to conduct this study smoothly.

References

Ali MY, Kamal D, Hossain SMM, Azam MA, Sabbir W, Mushida A, Ahmed B, Azam K (2004) Biological studies of the mud crab, *Scylla serrata* (Forskål) of the Sundarbans mangrove ecosystem in Khulna region of Bangladesh. Pak J Biol Sci 7(11):1981–1987

Begum M, Shah MR, Alam MG (2006) *Ghere O Khachai Jugopoth Kakra Fattening Kolakowshol* (Concurrent crab fattening technology in gher and cage). Bangladesh Fisheries Research Institute, Brackish water Station, Paikgacha

BEPB (2004) Bangladesh export promotion bureau. Government of the People's Republic of Bangladesh, Dhaka

DoF (2012) Annual report (Fish fortnightly) published by the Department of Fisheries, Ministry of Fisheries and Livestock, Government of the People's Republic of Bangladesh

Forest Department (2011) Annual report published by the Forest Department, Ministry of Forest and Environment, Government of the People's Republic of Bangladesh

Khan MG, Alam MF (1991) The mud crab (Scylla serrata) fishery and its bio-economics in Bangladesh. BOBP/REP 51:29–40

Saha MR, Ahammad SU (1999) Fattening technique of mud crab. Bangladesh Fisheries Research Institute, Mymensingh, p 8

SEAFDEC (1998) Newsletter, vol 21(1), January–March

Zafar M, Ahsan MN (2006) Marketing and value chain analysis of mud crab (*Scylla sp.*) in the coastal communities of Bangladesh. Bangladesh Fisheries Research Forum (BFRF), Dhaka

Zafar M, Hossain MI (2008) Community-based pen culture of mud crab, *Scylla serrata* in the mangrove swamp of Cox's Bazar. In: Bangladesh Fisheries Research Forum (BFRF) Abstracts book. 3rd Fisheries Conference and Research Fair 2008. Bangladesh Agricultural Research Council, Dhaka, p 170

Chapter 8
The Guadalquivir Estuary: A Hot Spot for Environmental and Human Conflicts

Javier Ruiz, Mª José Polo, Manuel Díez-Minguito, Gabriel Navarro, Edward P. Morris, Emma Huertas, Isabel Caballero, Eva Contreras, and Miguel A. Losada

Abstract The Guadalquivir estuary has an important place in history as mainland Europe's most southern large river-estuary-delta system. Intensification of human pressure combined with a limited understanding of its functioning have resulted in increasing socio-economic and environmental conflicts over the estuary. Within this context, the existing scientific framework could not answer the concerns raised about the consequences of dredging to substantially increase its depth, allowing large ships to serve the port of Seville.

A holistic approach to understanding the estuarine ecosystem, including it's physical and biogeochemical dynamics and how these act to control biodiversity, was identified as the first step towards making knowledge-based decisions about its sustainable use. Intensive use of satellites and remotely operated sensors was critical to this endeavour, allowing a detailed diagnosis to be produced quickly (within 30 months). A real-time monitoring network allowed the critical spatio-temporal scales (intratidal to inter-annual) necessary to provide a baseline understanding of the major processes to be resolved. Discrete sampling of variables, such as nutrients and dissolved inorganic carbon, helped to fill in the gaps and synoptic satellite images allowed the estuaries influence on the inner shelf to be evaluated.

The overall diagnosis depicts an estuary in a poor condition. Numerous human interventions have drastically altered the geomorphology, tidal dynamics and

J. Ruiz (✉) • G. Navarro • E.P. Morris • E. Huertas • I. Caballero
Department of Ecology and Coastal Management, Instituto de Ciencias Marinas de Andalucía ICMAN-CSIC, 11510 Puerto Real (Cádiz), Spain
e-mail: Javier.Ruiz@icman.csic.es

Mª J. Polo • E. Contreras
Fluvial Dynamics and Hydrology Research Group, Interuniversity Research Institute of Earth System in Andalusia, University of Córdoba, Campus de Rabanales, Edif. Leonardo da Vinci, Córdoba 14071, Spain

M. Díez-Minguito • M.A. Losada
Environmental Fluid Dynamics Group, Andalusian Institute for Earth System Research, University of Granada, CEAMA, Avda. del Mediterráneo, s/n, 18006 Granada, Spain

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8,
DOI 10.1007/978-3-319-06305-8_8,

freshwater inputs of the system. Physical conditions favour high residence times that interact with the perturbation of upstream sediment supply by dams and local organic matter loading to create poor water quality. The present day conditions are a system where low dissolved O_2, high CO_2, high suspended solids and a lack of stable benthic-intertidal habitats limits biodiversity, more than is typical of other estuaries.

Recovery of the ecosystem-services supplied by the Guadalquivir is a challenge, but entirely possible, given careful balancing of the cultural, social, economic and environmental demands of all stakeholders within a knowledge-based framework. This study demonstrates that the application of robust and cost-efficient technology to estuarine monitoring can quickly generate the scientific foundations necessary to meet these societal and legal demands, and can provide a suitable tool by which the cost-effectiveness of remedial solutions can quickly be evaluated.

8.1 Introduction

The Guadalquivir estuary is located on the southwestern coast of the Iberian Peninsula (Fig. 8.1) and comprises the last 110 km of the Guadalquivir river, the first 85 km of which are navigable. The source of the river is in the Cazorla mountains, approximately 1,400 m above sea level, and it flows into the Gulf of Cádiz; it has a total length of 680 km and drainage basin of 63,822 km^2 (Granado-Lorencio 1991).

The morphology of the estuary has significantly evolved throughout history, initially due to natural processes and later because of human modifications. The first historical records seem to associate the mouth of the Guadalquivir with the mythical city of Tartessos (Rodríguez Ramírez et al. 1996). These records also stress the intense dynamism of the estuary. During Roman times, the estuary was known as the *Lacus Ligustinus*, indicating it was characterized by open, shallow waters (García et al. 2004). Some towns, now well inland (e.g., Mesas de Asta or Lebrija) were port cities at that time. Natural sedimentation and infilling eventually formed the Guadalquivir marshes. This transition occurred in a very short period (geological speaking) and generated the present-day physical environment of Doñana National Park, a freshwater wetland considered a UNESCO-MAB Biosphere Reserve and World Heritage site.

Because the land surrounding the estuary is essentially very flat, the morphology of the main river-estuary, before human modification, included many meanders and several ramifications. This was not very convenient for navigation to the port of Seville, which was to be the major hub for commercial exchanges between Spain and the newly discovered American continent. Hence, in the eighteenth century the first large-scale human modifications of the estuary began with short-cutting of meanders, these continued until the present day, reduced the length of the route between the mouth and the port by 50 km (at present the distance is 84 km).

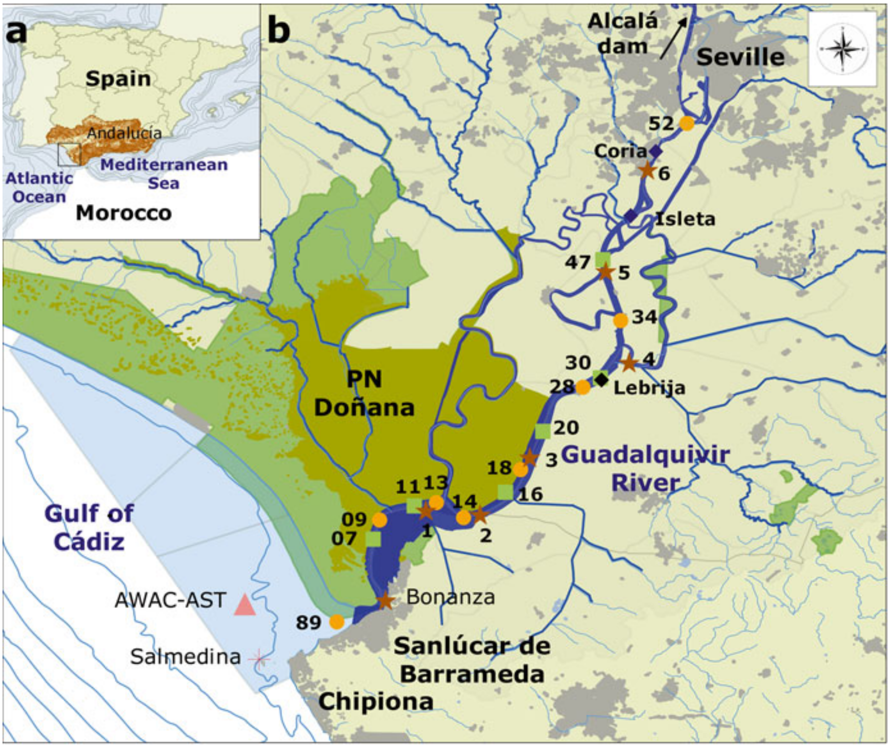


Fig. 8.1 Map of region, showing (**a**) Andalucía in Spain and (**b**) the Guadalquivir estuary with the indications of the instruments deployed and measurements made within the Guadalquivir monitoring network: *Green squares* correspond to water dynamics stations, *orange dots* to water quality stations, *brown stars* to tide gauges, *pink triangle* to AWAC-AST, *pink star* to Salmedina meteorological station, and *diamonds* to temporary quality stations

In the twentieth century, these modifications were joined by pressure for agricultural land, leading to widespread drainage of the marshes and massive modifications in the fresh water regime. More than 80 % of the original marsh surface area was transformed, representing one of the largest losses of marshes in Europe. In addition, the burgeoning agriculture demands led to a strong modification of fresh water inputs and the building of numerous reservoirs throughout the drainage basin. The furthest downstream, Alcala del Río dam, situated only 110 km from the estuary mouth, was so close it resulted in distortion of the tidal dynamics in the estuary. Fresh water inputs to the estuary have decreased by an average of 60 % (from approximately 5,000 hm^3/year in 1931–1981 to 2,000 hm^3/year in 1981–2000) with a greater reduction in dry-years.

More recently, as a result of the public outcry generated by the Aznalcollar mining spill, the remaining marshes of the Doñana area were completely isolated from the estuary using dykes and floodgates (Contreras and Polo 2010). Salt production and aquaculture have also contributed to the isolation of the estuary from its surroundings. As a result of all these modifications, the estuary is now composed of a main channel with a few tidal creeks without any significant intertidal zones. This channel is periodically dredged from its mouth to where it accesses the Port of Seville in order to guarantee a minimum navigation depth of 6.5 m.

These continuous perturbations have been intense enough to not only alter the dynamics of the estuary, but also the morphology of the adjacent coastal zone (López-Ruiz et al. 2012). They are indicative of the acute anthropogenic pressures and conflicts of interests that accrue around the estuary. The estuary and its immediate surroundings are presently home to 1.7 million people clustered in 90 population settlements. The process of decision making is extremely difficult in this heavily over exploited environment where numerous, conflicting stake holders interact. It includes several administrative bodies with different competences (environment, transport, agriculture, etc.) belonging to the regional and national government, all of them operating under the umbrella of legal instruments of the European Union. Within this difficult socio-economic context, the recent decision by the Port Authority of Seville to extend the programme of dredging so as to significantly increase the depth of the navigational section of the estuary, triggered social alarm, and created the need for a comprehensive analysis of the estuary and the potential consequences of human actions on its physical and ecological dynamics.

To provide this comprehensive, high quality baseline understanding, necessary to meet the demands of the stakeholders and leave no room for ambiguity, an exhaustive research programme was initiated in the summer of 2007. This included detailed field studies combined with hierarchical monitoring and modelling of many aspects of the Guadalquivir estuary ecosystem; including its' morphology, sedimentology, hydrology, hydrodynamics, biogeochemistry and biodiversity. Here we focus on the physical environment, how it affects biogeochemical processes and discuss the consequences for biodiversity in the estuary and adjacent self. To provide the high resolution measurements needed to resolve the critical scales of variability in this dynamic estuarine system, an ad-hoc real-time remote monitoring (RTRM) network was deployed along the length of the estuary. This also had the advantage of making the maintenance of instrumentation more efficient. The RTRM was complemented by several moorings, monthly longitudinal transect sampling of water quality parameters, the acquisition of remote sensing images and the compilation of other data sources. This chapter gives an overview of the Guadalquivir estuary monitoring network, summarises the main physical and biogeochemical processes, their potential impacts on biodiversity and suggests options for rational management.

8.2 Guadalquivir Estuary Monitoring Network

The core of the monitoring network comprised of a real-time remote monitoring (RTRM) system that provided online, high temporal resolution meteorological, hydrographic and water quality information. RTRM technologies have recently become an economically-viable means of assessing key hydrological parameters (Gutiérrez et al. 2009). They offer a number of important advantages, such as increasing the efficiency of system maintenance and data collection, providing an early warning system for decision-makers, and the possibility for near-real-time, transparent presentation of environmental information to stakeholders (Glasgow et al. 2004).

This core-system was supplemented by several moorings and monthly longitudinal transect sampling of water quality parameters, which provided an independent data set for validation of in-situ instruments. Remote sensing was used to provide synoptic snapshots of parameters such as total suspended solids (TSS) and chlorophyll *a* (chl*a*), particularly at the mouth of the estuary and within the surrounding coastal zone.

The positioning of instruments and sampling sites (Fig. 8.1) was strategically designed based on a priori knowledge about the estuaries dynamics, whilst taking advantage of the availability of existing infrastructures, such as navigation buoys. For example, shorter distances were selected between sites close to the mouth, where previous existing information seemed to indicate that environmental gradients were more acute.

A single system for real-time assessment of meteorological parameters was situated in the coastal zone in front of the estuary (Meteorological node). Vertical profiles of water velocities (1 m resolution) and surface waves were recorded using an upward-looking acoustic Doppler profiler (AWAC-AST) also situated in front of the estuary. Tidal propagation was assessed by deploying seven tide gauges along the banks of the estuary (Tidal gauges). Real-time water column velocity data was provided by six acoustic Doppler profilers (Hydrodynamics nodes) and real-time water quality parameters were assessed at eight sites along the length of the estuary (Water quality sites). Surface water samples and vertical profiles of physiochemical parameters were also collected at these eight water quality sites, as well other positions, during longitudinal transect sampling campaigns. A number of these sites coincided with the sampling positions of the monitoring programme carried out by the regional management agency (Consejería de Medio Ambiente, Junta de Andalucía).

Monthly transect sampling of water quality parameters started in June 2007 and was maintained until August 2009 (Fig. 8.2). The RTRM network began functioning in Feb. 2008 and moorings were installed in March 2008. The RTRM network and moorings were in place 24 months and were routinely maintained every 1 and 3 months (depending on the season).

Meteorological measurements were sampled every second, and processed in-situ to give time-ensemble mean, maximum, and minimum values for 10 min. intervals (Table 8.1). Tidal gauges collected samples every 10 min. The AWAC-AST ADP situated in front of the estuary collected a current profile for 60 s every 10 min and wave data every hour, whilst the rest of the ADP measurements at the hydrodynamics sites, situated along the length of the estuary, were collected for 180 s every 15 min. Finally, real-time water quality measurements were collected every 30 min.

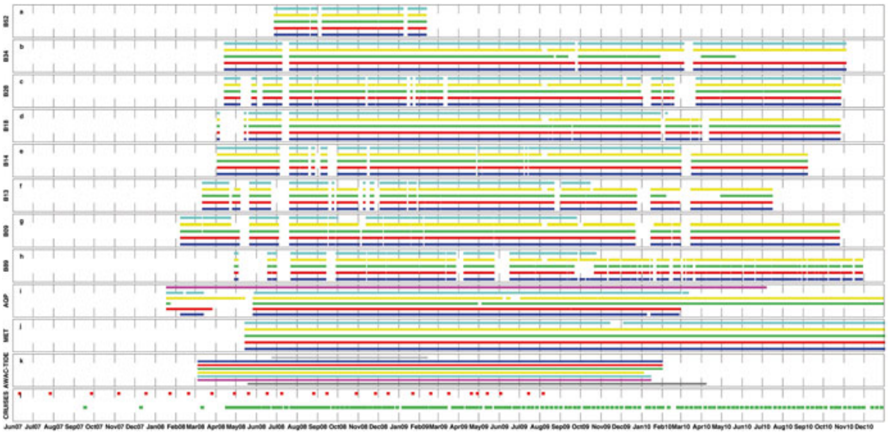


Fig. 8.2 Bar chart of data availability in Guadalquivir estuary during the 2007–2010 monitoring programme. Panels **a**–**k** show the daily data availability (more than 95 %). Panels **a**–**h** represent real-time water quality nodes. Color indicates parameter type: *Blue* is temperature, *red* is conductivity, *green* is dissolved oxygen, *yellow* is turbidity, and *cyan* is chlorophyll fluorescence. Panel **i** represents hydrodynamic date. Color indicates each node: *blue, red, green, yellow, cyan*, and *magenta* are buoys 7, 11, 16, 20, 30, and 47, respectively. Panel **j** is meteorological data. Color indicates parameter; *blue, red, green, yellow*, and *cyan* are wind, temperature, radiation, barometric pressure, and relative humidity, respectively. Panel **k** is AWAC-AST ADP (*dark grey*) and tidal gauges data (M1 in *magenta,* M2 in *cyan*, M3 in *yellow*, M4 in *green*, M5 in *red*, M6 in *blue* and M7 in *grey*). Panel **l** represent the longitudinal transect water sampling carried out by ICMAN-CSIC (*red symbols*) and the regional management agency, Consejería de Medio Ambiente, Junta de Andalucía (*green symbols*)

8.2.1 Real-Time Remote Monitoring (RTRM) Network

The general system architecture of each node within the RTRM network included; (1) a power source, (2) a combined telemetry unit and data-logger, (3) combined sensor-interface and data-logger, and (4) an array of sensors. For the water quality nodes, an extra pumping module was added that allowed water from different depths to be sampled by a single sensor array. Considering the accessible nature of the deployments and the large risk of instrument vandalism and opportunistic thievery, great lengths were taken to camouflage the modules within and below the navigational buoy structures.

Power sources consisted of a rechargeable gel battery, a solar charge regulator and a number of solar panels. The exact combination of these units was selected depending upon the nodes characteristic operating demands, with the aim of allowing high-temporal resolution and extended autonomy (see Gutiérrez et al. 2009 for details).

The 'brain' of each node was an ad-hoc module developed at the Instituto de Ciencias Marinas de Andalucía, ICMAN-CSIC (recently licensed to South-Tek Sensing Technologies) that combined a data logger with a Global System for Mobile Communications/General Packet Radio Service (GSM/GPRS) modem and high-gain antenna. A detailed description of the design can be found in Gutiérrez et al. (2009).

Table 8.1 Details of the instruments deployed and measurements made within the Guadalquivir monitoring network

Node	Parameter	Sampling interval	Characteristics	Model, manufacturers
Meteorological (node 01)	Wind speed and direction	1 s, time-ensemble mean, maximum and minimum derived for 10 min. intervals	Range: <100 m/s, accuracy: ±1 m/s	Wind Monitor, R.M. Young
	Air temperature/humidity		T range: −30 to 70 °C, accuracy: ±0.1 °C	STH-5031, Geonica
	Atmospheric pressure		H range: 0–100 %, accuracy: ±3 %	61,202L, Young
	Solar radiation		Range: 600–1,100 hPa, accuracy: ±0.5 hPa	Pyranometer Li200, Licor
			Range: 400–1,100 nm, error: <5 %	
Hydrodynamical (nodes 07, 11, 16, 20, 30, 47)	Current profiler (ADP)	180 s measurement every 15 min	Down-looking	Aquadopp Profiler (1 Mhz), Nortek
			Maximum profiling range: 20 m	
			Cell size: 0.3–4 m	
			Minimum blanking: 0.20 m	
			Velocity range: ±10 m/s	
			Accuracy: 1 % of measured value	
Water quality (nodes 89, 09, 13, 14, 18, 28, 34, 52)	Temperature	30 min	Range: −5 to 35 °C, accuracy: 0.0001 °C	SBE16plus, Seabird Electronics Inc
	Conductivity		Range: 0–9 S/m, accuracy: 0.0001 S/m	SBE16plus, Seabird Electronics Inc
	Dissolved oxygen		Range: 120 % saturation, accuracy: 2 % sat	SBE43, Seabird Electronics Inc
	Fluorescence		Range: 0–500 µM/L, accuracy: 0.03 µM/L	Fluorometer Cyclops-7, Turner Designs
	Turbidity		Range: 0–3,000 NTU, accuracy: 0.04 NTU	Turbidimeter Cyclops-7, Turner Designs
Tidal gauges	Pressure?	1 measurement every 10 min.	DETAILS	Aqualogger 520PT
				NKE-SP2T logger

(continued)

Node	Parameter	Sampling interval	Characteristics	Model, manufacturers
AWAC-AST ADP	Current profiler (ADP)	X s measurement every 10 min	Upward-looking	AWAC-AST profiler, Nortek
Longitudinal transects	Temperature, conductivity, dissolved oxygen, optical backscattering, and in-vivo chla fluorescence	Approximately every month		Seabird-SBE-19plus-V2 SeaCAT Profiler, Seabird Electronics Inc
	Dissolve inorganic nutrients, dissolved organic nitrogen and carbon, particulate organic carbon, total alkalinity, pH, chla and TSS			Water samples collected with Niskin bottle

This data logger can handle up to 24 analogue channels and when combined with the GSM/GPRS unit allowed bi-directional communication between each node and a central server located in the ICMAN-CSIC. The data loggers were programmed to handle node-type specific data logging and telemetry tasks (Gutiérrez et al. 2009, 2011). For the water quality nodes they also ran the pumping cycle that brought water from different depths to the sensor array. To facilitate data communication, the hydrodynamics nodes were configured as transmission control protocol/internet protocol clients, improving the handling of dynamic internet protocol-address assignation from the network provider.

The sensor interface and presence of an instrument specific data-logger for each node depended on the instrument-sensor combination. For example, the specialised nature of the data collected by the ADPs meant much more information than could be reasonably transmitted to the central server was generated. Thus, only a part of the data was selected for real-time transmission; battery level, head pressure and temperature, compass pitch and roll, ADP error/status and the three velocity components of the upper six water column cells. The rest of the information (velocity of the other 15 cells, beam amplitudes, etc.) was stored in internal memory and recovered during routine maintenance operations. For the water quality and meteorological nodes, data was stored on the data-logger (only as a back up) and all information was transmitted to the central server, including sensor data as well as control parameters related to the pumping performance and battery level.

The water quality node pumping module comprised of a polyester enclosure with four suction pumps and a single flow meter (1–15 L min^{-1}) and a 50-μm filter batch to avoid early degradation of the pumps and sensors, and some silicon piping.

Meteorological measurements included air temperature, relative humidity, downwelling solar radiation, barometric pressure, wind speed and wind direction, derived using an array of instruments (Table 8.1, see Navarro et al. 2012 for technical details). Hydrodynamic measurements were collected with down-looking ADPs operating at a frequency of 1 MHz with a measurement period of 180 s. The ADPs provided water velocity components (u, v, and w in the x, y, and z directions, respectively), temperature, head pressure, pitch and roll, allowing the main flow velocity and direction to be derived at a vertical resolution of 1 m within the water column. With the pumping and filtration module, water quality parameters could potentially be measured at depths of 1, 2, 3 and 4 m below the water surface.

8.2.2 Monthly Sampling of Water Quality Parameters

Longitudinal transect sampling of surface water samples was carried out from a small boat using Niskin bottles. Parameters measured included the concentration of dissolve inorganic nutrients (nitrate, nitrite, silicate and orthophosphate), dissolved organic nitrogen and carbon, dissolved oxygen, particulate organic carbon, total alkalinity, pH, chl*a* and TSS. In-situ profiles of temperature, conductivity, dissolved oxygen, optical backscattering, and in-vivo chl*a* fluorescence were derived using a Seabird-SBE-19plus-V2 SeaCAT Profiler (SBE19) equipped with an array of external sensors (Table 8.1).

Nutrients were analysed with a TRAAC800 auto-analyser following the techniques of (Grasshoff and Kremling 1983). Two sub-samples of filtered water (12 mL, Whatman GF/F filters) were taken at each site and stored at−20 °C until analysis in the laboratory. The standard deviation of each replicate was <0.06 and 0.01 μM for nitrate and phosphate, respectively. Chlorophyll analysis was conducted by filtering samples through Whatman GF/F glass fibre filters (0.7 mm pore size), extracting in 90 % acetone, and measuring chl*a* by standard fluorometric methods using a Turner Designs Model-10 following JGOFS protocols (IOC/UNESCO 1994). Concentrations of TSS were measured gravimetrically on pre-weighed Whatman GF/F filters according to JGOFS protocols (IOC/UNESCO 1994).

Total alkalinity was measured with a Metrohm 794 Titroprocessor using the method described by Mintrop et al. (2000). Water samples were collected and stored in 500 mL borosilicate bottles poisoned with 100 μL of $HgCl_2$ saturated aqueous solution until analysis in the laboratory. The accuracy was determined from regular measurements of batches of certified reference material (CRM, supplied by Prof. Andrew Dickson, Scripps Institution of Oceanography, La Jolla, CA, USA). Water pH measurements in NBS (National Bureau of Standards) scale were carried out using a Metrohm 780 pH meter equipped with a glass electrode combination. The speciation of inorganic carbon in water was calculated using the thermodynamic equations and the constants for C and sulphate of Cai and Wang (1998) and Dickson (1990) respectively. Dissolved oxygen was determined following the Winkler method through potentiometric titrations using a Metrohm 794 Titroprocessor. Flasks containing the water samples were sealed and stored in darkness, for at least 48 h until analysis. Concentrations of dissolved organic nitrogen and carbon were derived by catalytic oxidation at high temperature (720 °C) and chemiluminescence, respectively using a Shimadzu TOC-VCPH analyser.

8.2.3 Calibration and Maintenance

All instruments came with basic manufacturers calibrations pre-installed, however a number of extra calibration steps were also carried out. The compass of the ADP was individually calibrated for each instrument according to the manufacturers instructions (Nortek Aquadopp User Guide 2005). The in-situ turbidimeters and fluorometers were individually calibrated in the laboratory (Navarro et al. 2011). Because of the highly turbid nature of the estuary, a modified method, where turbidity and fluorescence were simultaneously measured (Turner Designs Cyclops User's Manual 2012), was used to calibrate the in-situ fluorometer for the quantification of chl*a*. The laboratory fluorometer used for the quantification of chl*a* in water samples was calibrated using pure chl*a* from the cyanobacterium *Anacystis nidulans* (Sigma Chemical Co.)

with the concentration being determined spectrophotometrically. Routine maintenance of the water quality stations was carried out regularly and involved replacing the pumping module as well as cleaning the CTD and sensor arrays. When necessary, the anti-fouling cylinders (SBE A24173) inside the CTDs were replaced.

8.2.4 Complementary Data Sources

DEIMOS-1 (DEIMOS Imaging, Spain) multi-spectral images with a spatial resolution of 22 m from between Aug. 2010 and Sept. 2011 were acquired and processed to produce synoptic maps TSS concentrations in the surface waters of the estuary following the methodology of Caballero et al. (2013). In addition, RGB MODIS images were acquired from the National Aeronautics and Space Administration (NASA) server (http://lance-modis.eosdis.nasa.gov).

Daily water volumetric flow rates from the Alcala del Río dam, meteorological data (precipitation, air temperature, wind velocity and humidity) from a site in Lebrija, as well as hydrological parameters (conductivity, dissolved oxygen and turbidity) and water quality parameters at a number sites situated along the length of the estuary were obtained from the regional management agency (Consejería de Medio Ambiente, Junta de Andalucía, http://www.juntadeandalucia.es/medioambiente).

8.2.5 Validation and Quality Assessment

Validation of the meteorological data collected through the RTRM network was done by contrasting with the nearby site operated by the regional management agency; very good correlation was observed between both data sets (see Navarro et al. 2011 for details). Current velocities measured at the hydrodynamics nodes were validated using a different ADP (ADP-1,000 kHz SonTek, USA) that integrates a GPS and bottom tracking during a tidal cycle (Navarro et al. 2012). Data from the water quality nodes were compared to the data collected during the monthly transects and the data independently collected by the regional management agency, and showed good correspondence with both data sets (Navarro et al. 2011, 2012).

One criterion for assessing the quality of time series data sets is the availability of data. Using this criterion the quality of the data generated by the RTRM network can be considered excellent since only about 5 % of data collected did not meet quality standards (Fig. 8.2). This high level of data quality, even under severe conditions, including large storms and huge, extremely turbid river discharge events, suggest that the network was a reliable and robust monitoring solution. The few gaps in the time-series were caused by CTDs being transported to the laboratory for specific maintenance, problems with the stainless steel cases of the Cyclops fluorometers and issues with the data reception pin of the ADP. An example of the time series for water quality at node 28 is shown in Fig. 8.3.

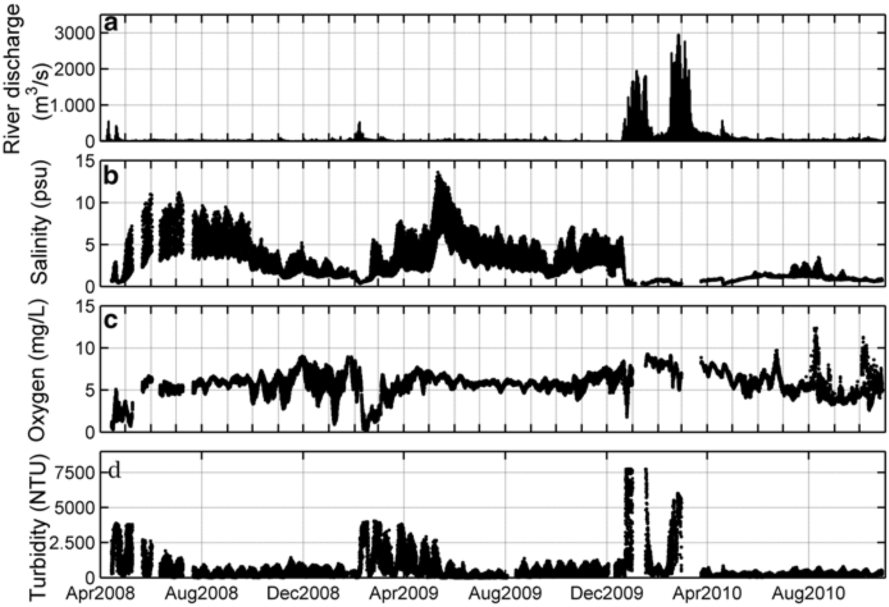


Fig. 8.3 Time series of volumetric flow rates from Alcalá del Rio dam and water quality recorded by the RTRM at node 28

8.3 Physical and Biogeochemical Processes

The physical environment of the waters within the estuary is strongly controlled by tides and freshwater inputs as well as, to a lesser extent, wind (Álvarez et al. 1999). The tides are mesotidal, about 3.5 m during spring tides with maximum tidal currents of 1 m/s and penetrate from the estuaries only mouth near Sanlúcar de Barrameda to the Alcalá del Río dam (Fig. 8.1).

According to the dominant processes controlling tidal propagation, the estuary can be divided into three regions that each roughly comprise one third of the estuary (Díez-Minguito et al. 2012). In the region nearest the mouth, there is a significant reduction of amplitude in tidal elevation. Here the dominant process is tidal diffusion generated by the presence of tidal flats, and by secondary circulation, related to the propagation of the tidal wave in the sinuous lower region of the estuary. At the head of the estuary, tidal motion is partially standing due to reflection on the Alcalá del Río dam. This causes notable increases in tidal elevation, and harmonisation of the tidal phases, leading to a close-to-zero lag phase between high water and high-water slack. In the intermediate region the balance between channel convergence and bottom friction appears to be the dominant process.

Wind also influences the exchange between the estuary and inner shelf (Scully et al. 2005). The region has a distinctive climate dominated by strong winds from the east (*Levantes*) (Prieto et al. 2009). Winds from the east favour baroclinic circulation, augmenting the injection of saltier water, replacing superficial, low-salinity water flushed out of the estuary. This contributes to an increase in the saline intrusion front (Fig. 8.11) and the longitudinal dispersion coefficient of solutes, resulting in lower estuarine retention times and an increase in nutrient exchange between the estuary and surrounding shelf (Reyes-Merlo et al. 2013).

Precipitation within the Guadalquivir catchment basin exerts an influence on volumetric flow rates from Alcalá del Rio dam (Q_d), at the head of the estuary. This dam represents 80 % of the estuary freshwater inputs and discharges are strongly regulated. Q_d is generally very low (<40 m^3/s), resulting in a tidally-energetic system throughout most of the year. However, sudden, rapid increase in discharge rates are not infrequent during autumn and winter when high precipitation upstream rapidly overwhelms the very low storage capacity reservoir. In spring and summer, water is usually released to irrigate land downstream (Contreras and Polo 2011). Periods with high discharge rates caused by increases in upstream flow, correlate with inter-annual variations in precipitation, and are more frequent during years when the North Atlantic Oscillation (NAO) is in a negative phase (Prieto et al. 2009).

Based on the interaction between tide and Q_d, Díez-Minguito et al. (2012) identified three different characteristic hydrodynamic regimes for the Guadalquivir estuary; (1) when Q_d is <40 m^3/s the system is tidally dominated and well-mixed, (2) when Q_d is >400 m^3/s the system is fluvially dominated and stratified on the inner shelf, and (3) an intermediate regime.

8.3.1 Low-Flow Regime

By far the most typical hydrodynamic regime for the Guadalquivir estuary is low-flow, which is present 75 % of the year. Under this regime, the mesotidal range and shallow depth of the estuary, indicate that the water column is well-mixed, except adjacent to the estuary mouth. This is confirmed by calculations of dimensionless numbers that express the ratio between mixing and buoyancy, such as the Estuary number (N) and estuarine Richardson number (N_R) (Díez-Minguito et al. 2013a). N represents the ratio of river flow in a semi-diurnal cycle to the tidal prism. Highly stratified estuaries exhibit values of N greater than 1, and well-mixed estuaries typically exhibit flow ratios less than 0.1. Whereas, N_R gives an indication of the relative importance of freshwater-derived buoyancy versus mixing by tidal currents; values >0.8 and <0.08 indicate stratified and well-mixed conditions, respectively (Hansen and Rattray 1966; Fischer et al. 1979). Values of N and N_R in the Guadalquivir estuary during the low-flow regime range from 0.025 to 0.05 and 0.01 and 0.08, respectively, clearly indicating well-mixed conditions.

The consequence of tidally-dominated mixing during periods with low Q_d is the absence of vertical structure in the water column. Vertical profiles of a conservative tracer like salinity are relatively homogeneous (Fig. 8.4). Although low, Q_d are sufficient to compensate evaporation losses, thus salinity monotonically decreases

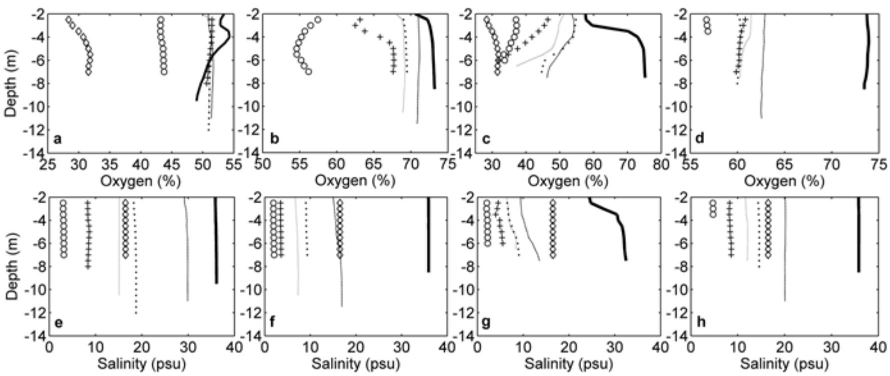


Fig. 8.4 Vertical profiles of oxygen and salinity at nodes 89 (*thick line*), 09 (*thin line*), 13 (*dots*), 14 (*grey line*), 18 (*crosses*), 28 (*circles*) and 34 (*rombs*). Panel **a**–**d** and **e**–**h** correspond respectively to dates 09/15/2008, 11/27/2008, 03/17/2009 and 06/03/2009

from the estuary mouth to the head. Tidally-averaged effective dispersion coefficients vary depending on the estuary stretch under consideration. Typical values in the lower stretches of the estuary are around 500 m^2/s and larger values, ca. 1,400 m^2/s, are observed upstream (Díez-Minguito et al. 2013a). However, these values are not large enough to prevent the existence of sharp gradients in salinity and O_2 along the estuary (Fig. 8.4).

TSS are very high during the low-flow regime (Fig. 8.5). However, external inputs of TSS are minimal given the low input of freshwater and its relatively low TSS concentration (below 200 mg/L; Contreras and Polo 2010), hence the majority of the suspended material appears to be a result of resuspension by tidal currents (Díez-Minguito et al. 2013b). Remote-sensing images provided high resolution (22 m) synoptic views of TSS in the lower third of estuary (Fig. 8.6, Caballero et al. 2013) and suggested that the highest concentrations tend to be located between 20 and 30 km from the estuary mouth. These observations correlate well with the tidal harmonic analysis of Díez-Minguito et al. (2012), where maximum longitudinal current velocities of the most energetic, semi-diurnal constituents (M2, S2 and N2) are predicted at approximately 20 km from the mouth. In addition, this stretch of the estuary is also characterized by two adjacent low radius curvature bends that further enhance secondary flow, contributing to the maintenance of high turbidity levels.

High concentrations of TSS acutely attenuate light in the water column, severely reducing the availability of light for photosynthesis by phytoplankton and thus, photoautotrophic growth. This means photosynthetic organisms are unable to make use of the abundant nutrients within the estuary, concentrations of nitrogen and phosphorous are orders of magnitude above those considered limiting for phytoplankton growth (Elser et al. 2007; Fig. 8.5), resulting in very low levels of chla-containing photosynthetic biomass (Figs. 8.5 and 8.7).

This very-low photosynthetic biomass means very low rates of photoautotrophic oxygen production. Hence, despite the well mixed conditions within the estuary,

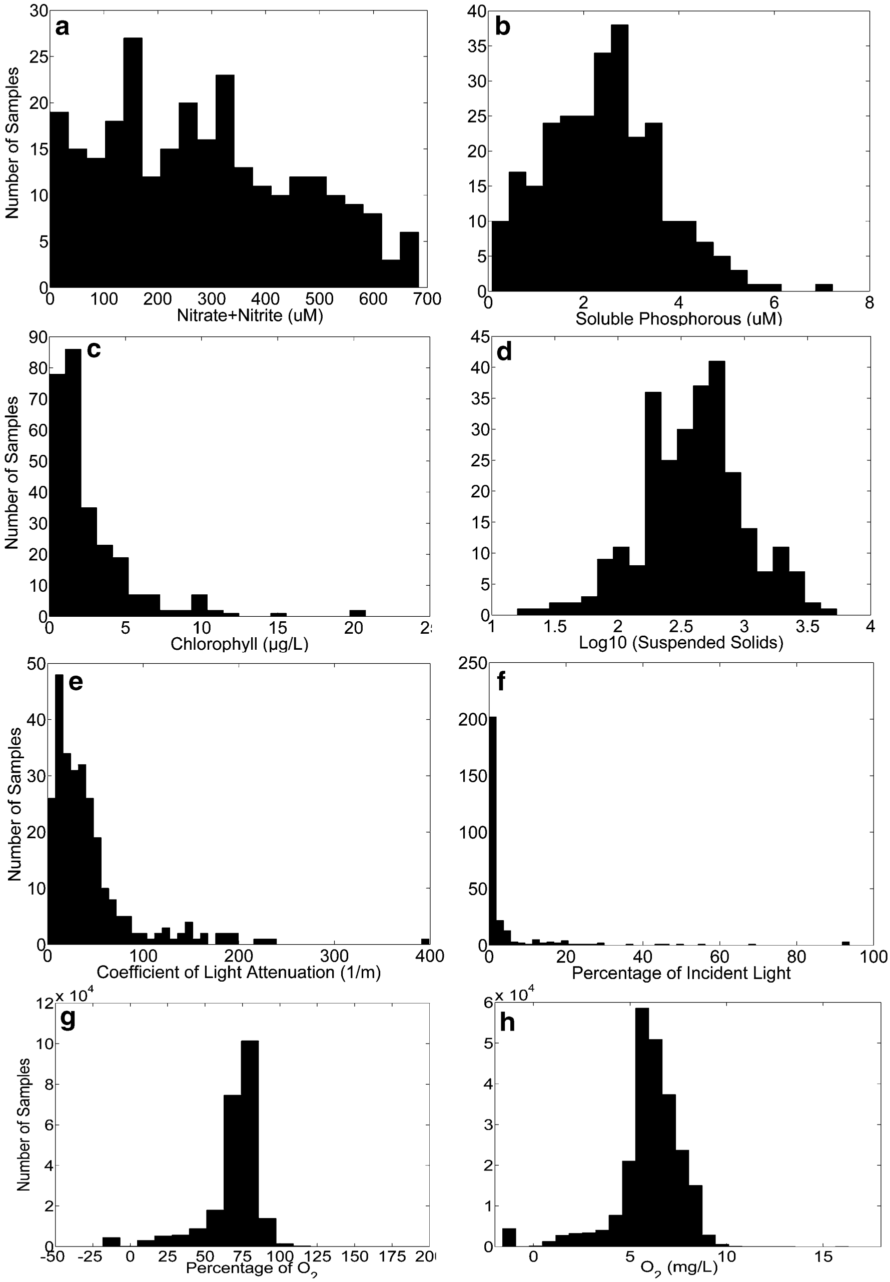


Fig. 8.5 Histograms of chemical and physical variables recorded at the estuary

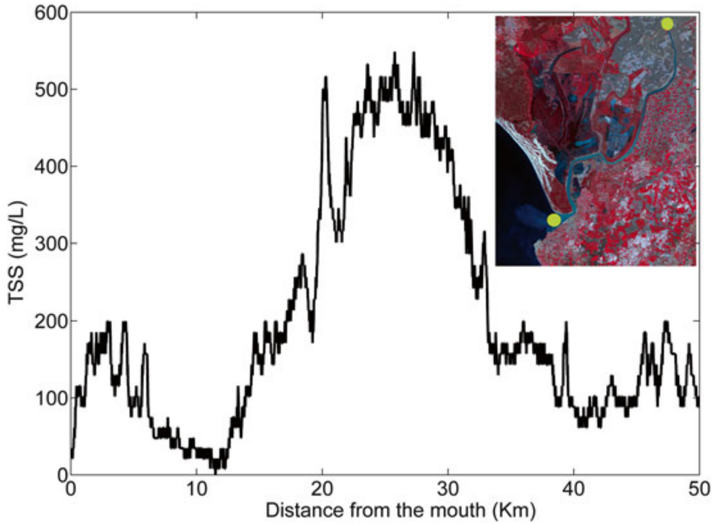


Fig. 8.6 TSS along a longitudinal transect from DEIMOS-1 sensor for April 16, 2011

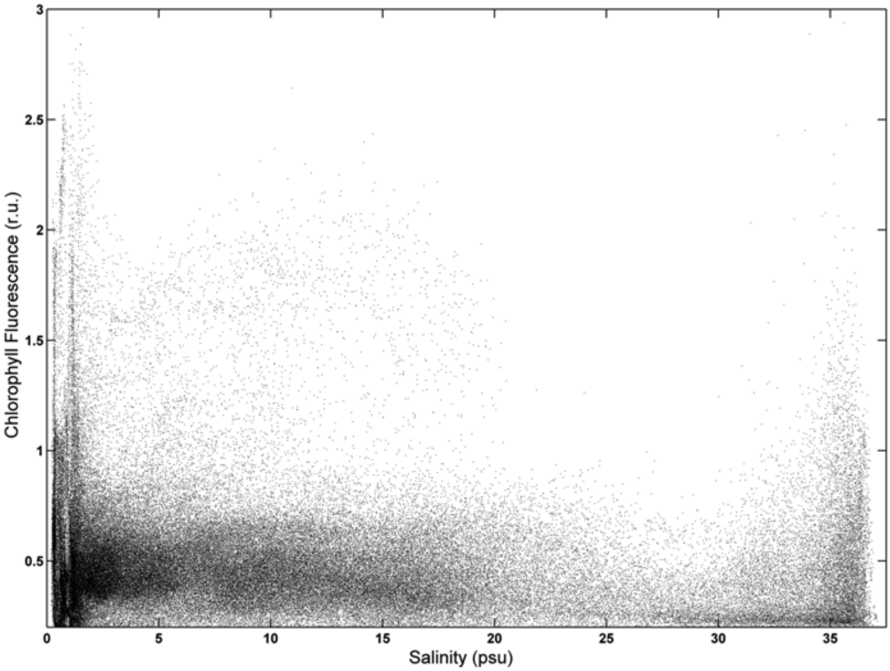


Fig. 8.7 Chlorophyll fluorescence vs salinity

under saturation (compared to atmospheric values) of dissolved O_2 is regularly observed (Fig. 8.5). This indicates that re-aeration (i.e., air-water O_2 supply) occurs at lower rates than O_2 consumption, suggesting intense heterotrophic processes that, because of the lack of light, are not compensated by oxygen production. This high rate

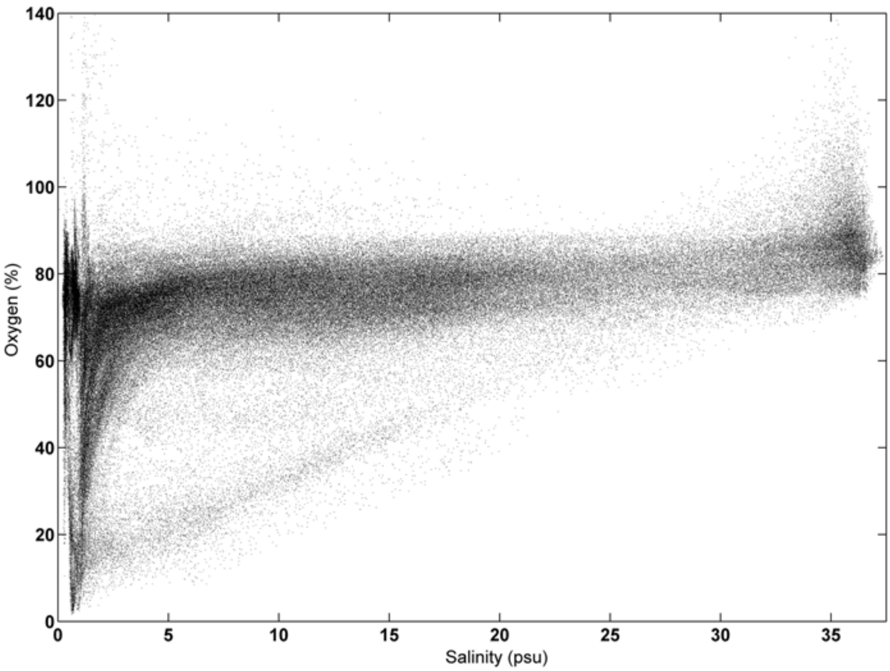


Fig. 8.8 %O_2 vs salinity

of oxygen consumption is the result of the large volumes of organic material entering the system, either from up-stream of the dam (Contreras and Polo 2010) and/or from the nearby municipalities and agricultural activities (Mendiguchía et al. 2007).

As for other degraded estuaries (e.g., Damme et al. 2005), in terms of oxygen concentrations, two regions can be identified within the Guadalquivir; the upper section with lower salinity, where fluctuations are dominated by biogeochemical processes, and the lower section, where the marine influence keeps values close to atmospheric equilibrium (Benson and Krause 1984). Thus, very low %O_2 events tend to be more frequently associated with mesohaline waters (<5), whereas in euhaline waters (>30) %O_2 is almost always >60 % (Fig. 8.8). Periods when dissolved oxygen concentrations are so low that the water column can be termed anoxic (<0.5 mg/L) are sometimes observed in the upper region at salinities of <2. Indeed, the isohaline with a value of 2 is usually taken to define the tidally-averaged, estuarine saline intrusion position (X_2) relative to the mouth of the estuary (Díez-Minguito et al. 2013a). In the low flow regime X_2 ranges between 35 and 75 km from the mouth (Fig. 8.9), suggesting up to 70 % of the estuary maybe affected by hypoxia.

Further evidence that biologically mediated processes, such as organic matter degradation, determine the biogeochemical properties within the estuary is given by the close coupling observed between dissolved CO_2 partial pressure (pCO_2) and %O_2 saturation (Fig. 8.10c). Indeed, the dominance of heterotrophic metabolism results in values of pCO_2 permanently higher than atmospheric values (~382 ppm during the monitoring period) along almost the whole length of the estuary, suggesting the system is a net exporter of gaseous CO_2 to the atmosphere (Fig. 8.10a).

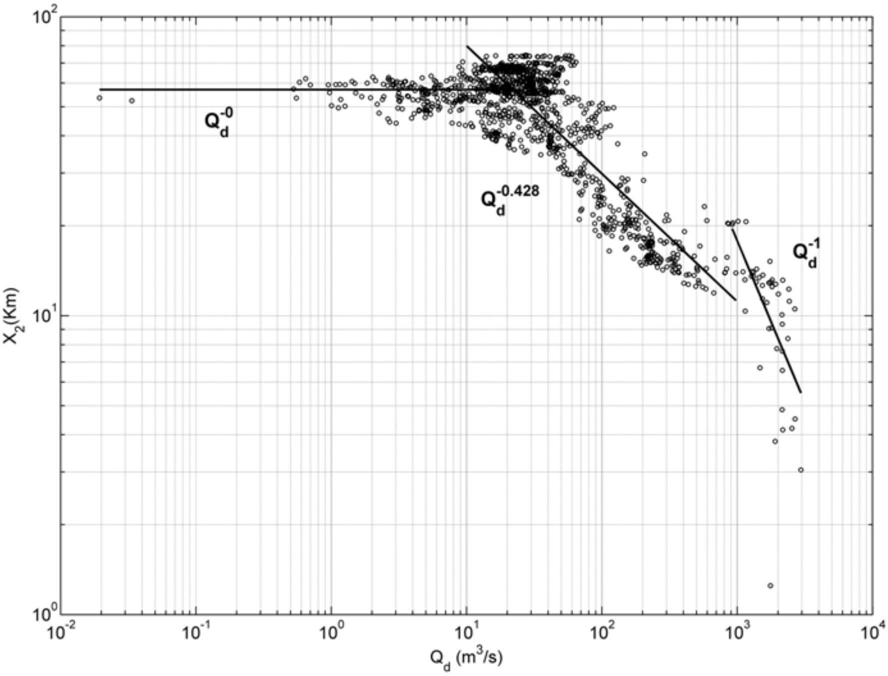


Fig. 8.9 Length of the saline intrusion X_2 compared to the freshwater discharge Q_d

Only very rarely are the extreme light-limiting conditions in the water column relaxed, allowing phytoplankton to proliferate. These events are usually observed at both ends of the salinity gradient, as evident from the signal of chlorophyll fluorescence (Fig. 8.7), and lead to increases in $\%O_2$ (Fig. 8.8). The fact that these events do not occur at intermediate salinities indicates that these blooms result from a relaxation of light limitation at upper (fresher) and lower (saltier) extremes of the estuary.

8.3.2 High-Flow Regime

When discharges are greater than 400 m^3/s, the estuary becomes fluvially dominated resulting in low salinities at the mouth. In this hydrodynamic regime, the salt intrusion position practically reaches the mouth of the estuary (X_2 ranges between 20 and 5 km), increasing stratification of the water column and a salt-wedge forms on the continental shelf (Figs. 8.9 and 8.11).

High sediment loads in the outflow from the Alcalá del Río dam combined with sediments entrained from the bed by high turbulence result in extreme turbidity

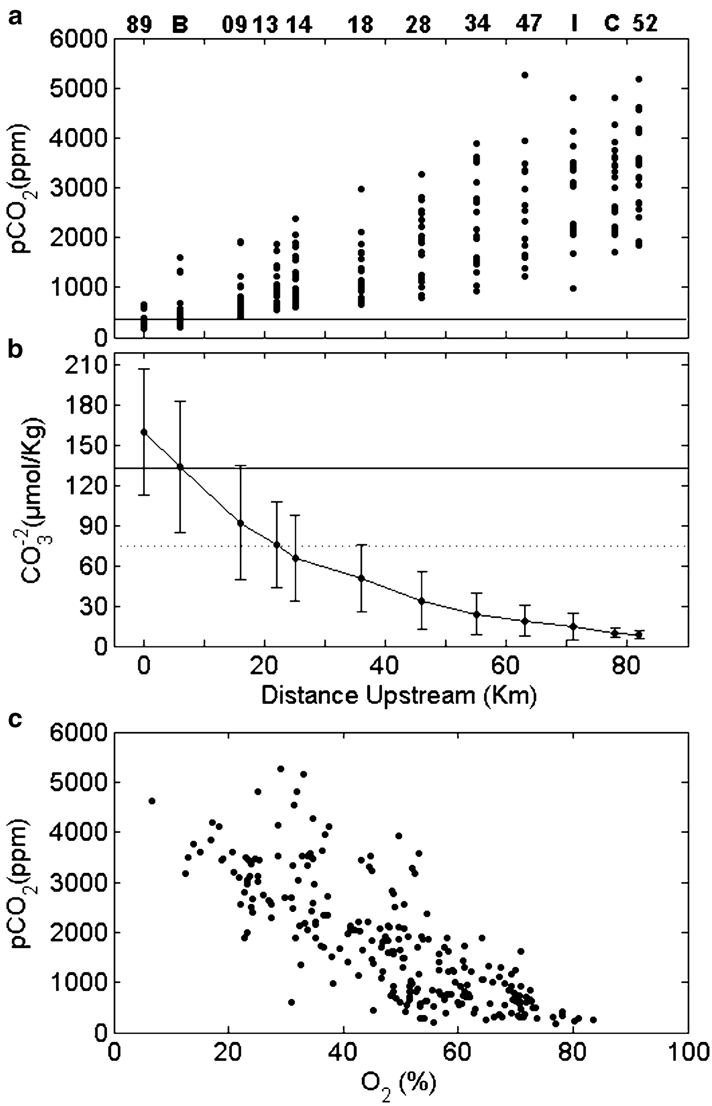


Fig. 8.10 (**a**) pCO_2 and (**b**) concentration of calcium carbonate vs distance upstream. (**c**) pCO_2 vs % O_2

events. Prolongation of these events long after peak discharges, can occur (Fig. 8.3) due to non linear interactions (see Sect. 8.3.4 below).

Longitudinal gradients in oxygen are non-existent during these events. Concentrations of chl*a* increase probably because of the advection of freshwater phytoplankton from upstream of Alcalá del Rio dam.

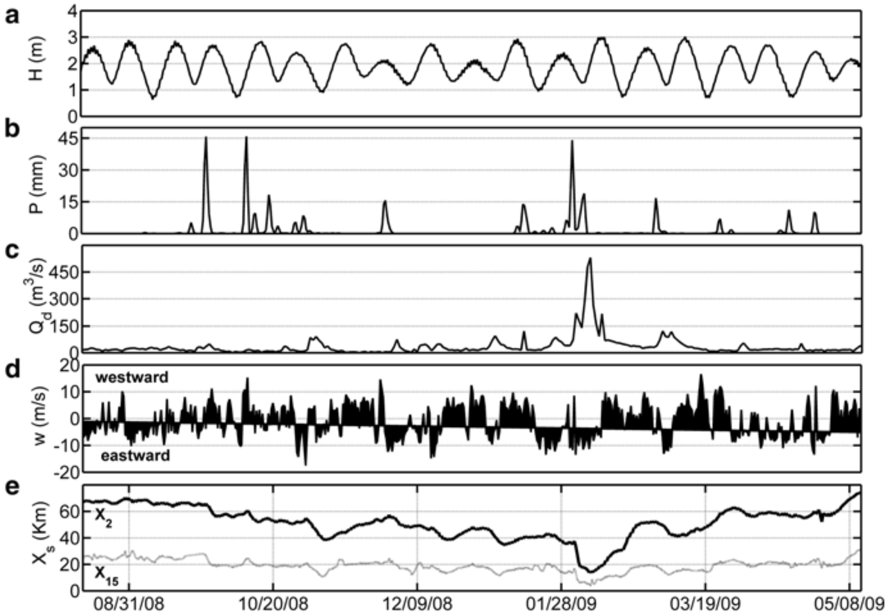


Fig. 8.11 Time series of tidal range (**a**), precipitation (**b**), volumetric flow rate at Alcala del Rio dam (**c**), wind velocity at the meteorological station (**d**) and extension of saline intrusion (**e**)

These high-flow events have important consequences for the neighbouring coastal zone. Large quantities of sediments are discharged from the estuary and transported along the coast (Figs. 8.12 and 8.13a). Indeed, fluvial sediments from Guadalquivir and Guadiana rivers are the main source of Holocene sediments on this shelf (Lobo et al. 2004). These events also pump large quantities of nutrients into the coastal zone, stimulating growth of phytoplankton that tend to be nutrient limited (Navarro and Ruiz 2006), creating large phytoplankton blooms (Fig. 8.13b) that sustain higher trophic levels (Prieto et al. 2009). Under extreme conditions, oxygen depletion may potentially extend to the neighbouring coastal waters creating 'dead zones' as is observed in New Jersey (Glenn et al. 2007) and in the Narragansett Bay (Saarman et al. 2008).

8.3.3 Intermediate Regime

The intermediate regime mainly corresponds to controlled discharges from Alcalá del Río dam that provide irrigation water for crops downstream (Díez-Minguito et al. 2013a). Within this regime, Q_d varies between ca. 40 and 400 m^3/s depending on precipitation and the capacity of the reservoir network upstream to satisfy the

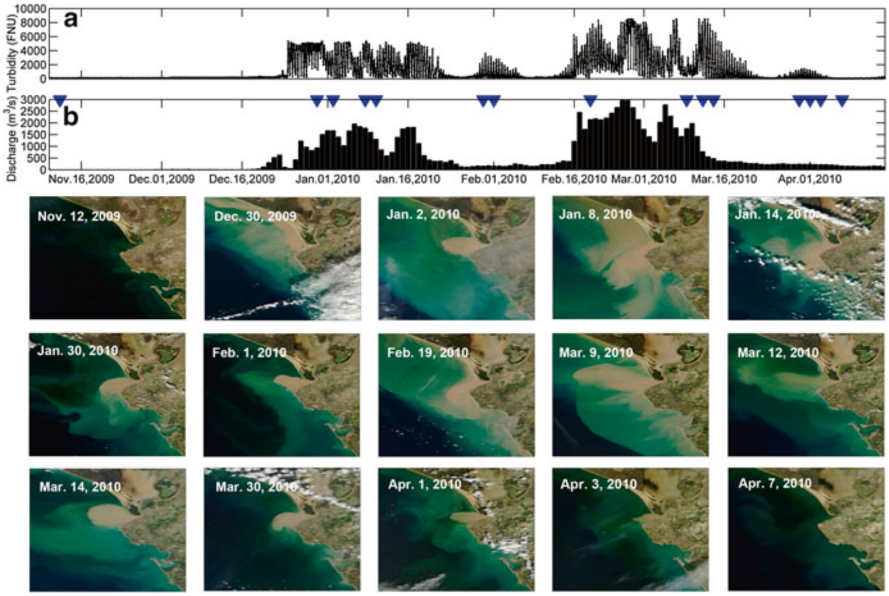


Fig. 8.12 Turbidity recorded at node 89, Q_d, and RGB MODIS images

freshwater demand. With the present conditions of the estuary, these moderate to high river discharges do not translate into an efficient increase in estuarine circulation since vertical mixing is still strong. In this condition the residence time of the estuary is higher than could be achieved if the estuarine circulation were more developed.

8.3.4 Recovery From High-Flow Regime

Of particular relevance to the estuarine dynamics is the transient state that persists after a period of high flow. Figure 8.11 shows the evolution of X_2 and X_{15} in February 2009 during a high discharge (531.2 m^3/s) event. Six days after Q_d had returned to low levels, the salinity at the estuary mouth returned to its previous records (not shown) but the salt intrusion front took another 6 days to enter the first section of the estuary (17 km), after which the rate of salinity propagation upstream increased to 10 km/day (Díez-Minguito et al. 2013a). Overall, previous salinity levels were reached throughout the estuary in16.5 days, a little more than a spring-neap cycle. This represents a net celerity of 4 cm/s (3.5 km/day). However, recovery of dissolved oxygen concentrations to pre-high-flow conditions lagged behind salinity (Fig. 8.14), with pre-event levels not reached 1 month after the high flow event.

This is intriguing, as it would seem rational to expect that transport of saline waters upstream would also replenish the dissolved oxygen within the estuary. This

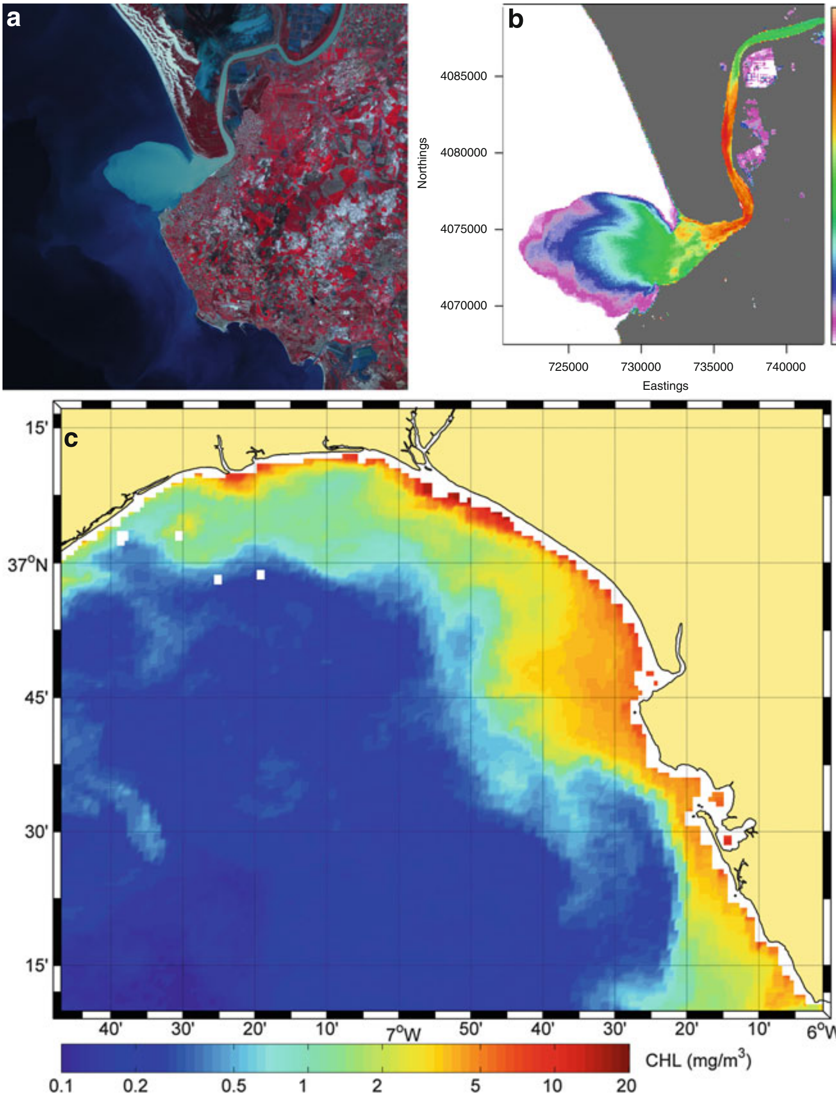


Fig. 8.13 DEIMOS-1 images in 03/03/2011, (**a**) RGB and (**b**) TSS. (**c**) MODIS chla in/11/23/2009

suggests that high-flow events modify the rates of biogeochemical processes within the estuary and that these effects have different relaxation times compared to the post-river flood recovery of salinity. A strong indication as to the mechanism that is probably responsible for maintaining enhanced net rates of oxygen consumption long after recovery of the estuaries physical structure can be seen in the persistence of very high turbidity levels long after salinity has recovered (Figs. 8.14 and 8.3). As a tracer of suspended solids, very high turbidity, on the one hand indicates negation

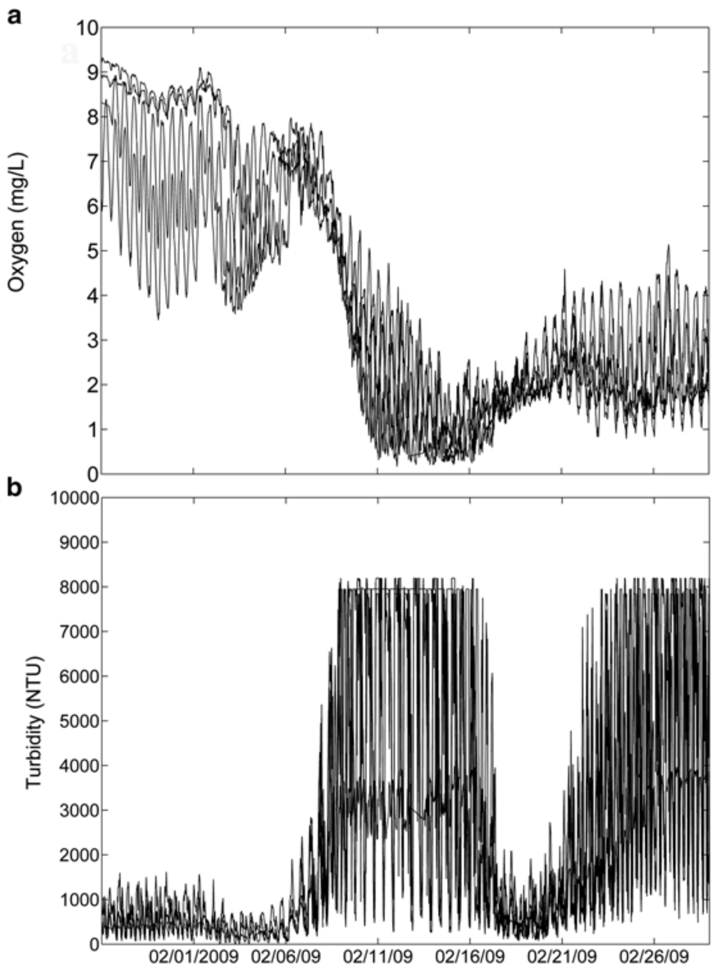


Fig. 8.14 Time series of O_2 (**a**) and turbidity (**b**) in nodes 14, 18, 28 and 89 after a high flow event

of photoautotrophic oxygen production and on the other hand, an abundance of suspended organic matter, with a subsequent stimulation of heterotrophic metabolism. Very high turbidity levels can persist after high-flow events for up to 5 months and are associated with enhanced tidal elevations and currents (Fig. 8.3). This is due to a reduction in the bottom friction resulting from a loss in sediment cohesion caused by the high-flow event (Wang and Fernández Bermejo 2010; Reyes-Merlo et al. 2013) as observed in numerous turbid estuaries (Kineke et al. 1996; King and Wolanski 1996; Rockwell Geyer et al. 1996; Álvarez et al. 2003). Once triggered, the positive feedback of enhanced near-bed turbulence resulting from the elevated current velocities reduces flocculation and deposition of suspended material (Winterwerp and Van Kesteren 2004) leading to a transient state with a rather long recovery time. The estuary eventually recovers its previous turbidity levels after a long but continuous process of sedimentation. In addition, westward winds seem to significantly contribute to quicker recovery times, injecting saltier, low-TSS loaded water into the estuary.

In terms of affects on oxygen consumption this mechanism potentially increases the absolute quantity of organic matter in contact with the water column, i.e., via resuspension and increasing sediment porosity, releases previously trapped pools of nutrients (Almroth-Rosell et al. 2012) and may substantially increase benthic fluxes via enhancement of transfer terms, i.e., turbulence and water velocities in the benthic boundary layer (Boynton et al. 1981; Janssen et al. 2005).

8.4 Potential Impacts on Biodiversity

Whilst the dynamic conditions of estuaries represent a particular set of problems for organisms, many of which require special adaptations that limit the diversity of biological communities, certain aspects of the physiochemical conditions in the Guadalquivir estuary are extreme, even in terms of estuaries.

Background concentrations of suspended solids are very high in the estuary. The mean concentration recorded in this study was 600 mg/L, which is high compared with other estuaries in the world (Table 8.2) and values may reach as high as tens of grams per litre during extreme turbidity events (Contreras and Polo 2010).

Table 8.2 Background concentrations of total suspended solids (TSS) in other estuaries in the world

River	Mean concentration (mg/L)
Amazonas	200
Colorado	0.5
Columbia	39
Congo	34′
Danube	325
Ganges	1,092
Indo	247
Mackenzie	137
Mekong	340
Mississippi	362
Niger	208
Orinoco	136
St. Lawrence	9
Guadalquivir	600

Table 8.3 Percentage of the time that oxygen concentrations in the different nodes was below thresholds identified by the EC Directive on Fresh water for Fish (EC), the Canadian Council of Environmental Ministries (CCME) the river channel (GES and CCME), Vaquer-Sunyer and Duarte (biodiversity) as well the Environmental protection Agency for hypoxia and anoxia

Node	EC (<9 mg/L)	CCME (<5.5 mg/L)	Biodiversity (<4.59 mg/L)	EPA-hypoxia (<2.9 mg/L)	EPA-anoxia (<0.5 mg/L)
89	99.88	10.49	0.39	0.02	0.00
09	99.83	24.57	7.13	1.64	0.00
13	99.58	22.35	7.46	3.73	0.03
14	99.45	18.90	7.86	3.96	0.03
18	98.21	15.57	4.23	2.48	0.14
28	99.44	30.28	11.82	5.95	0.31
34	98.89	60.49	41.10	23.39	13.60

This results in very high light attenuation, essentially excluding the vast majority of pelagic and submerged benthic photosynthetic organisms. Only emergent photosynthetic organisms, such as benthic microalgae, macroalgae and halophytes, that can take advantage of light available during tidal exposure on tidal flats may find a niche in this type of system. However, because of the successive anthropogenic modification of the morphology of the estuary, deepening and enclosing the main channel in rigid structures (Gallego Fernández and García Novo 2007, relatively little "natural" intertidal areas remain (Contreras and Polo 2010; Fernández-Delgado et al. 2000).

Prolonged exposure to high concentrations of TSS has a number of direct effects on aquatic fauna (Bilotta et al. 2012; Newcombe and MacDonald 1991). These include clogging of the filter feeding apparatus and digestive organs of invertebrates, reduction in fecundity and brood size (Kirk 1992), retarded development of eggs and larvae and increased water pumping rates (Appleby and Scarratt 1989). At very high concentrations, sediment-clogged gills can lead to death for many fish species (Ritchie 1972) and deleterious effects are observed at much lower concentrations (in the order of 100 mg/L) (Boehlert and Morgan 1985; Buck 1956; McLeay et al. 1984; Redding et al. 1987). These effects become more acute for younger life stages (Alexander and Hansen 1986; Appleby and Scarratt 1989; Newcombe 1994), which is particularly relevant considering the estuary is an important nursery area for economically important fish species (Drake et al. 2007).

Oxygen concentrations are also likely to be one of the key elements determining the quantity and quality of the biota within the estuary (Rabalais et al. 2009). Mean values of dissolved oxygen concentrations decreased upstream from the estuary mouth (Table 8.3). In the region near Seville, where the main sewage outfalls are located, oxygen concentrations below the anoxic limit of 0.5 mg/L established by the Environmental Protection Agency (USEPA 2000) were observed 13.6 % of the time. This means that this portion of the estuary is subject to anoxic conditions for more than 1½ months every year, with expected devastating effects on its biota (Riedel et al. 2012). This section of the estuary is below the hypoxic limit of 2.9 mg/L established by EPA about 3 months a year. Hypoxia can cause death, increase vulnerability to diseases (Dauer et al. 1992), alter food-web connections such as prey–predator interactions (Pihl et al. 1992) and lead to general decreases in ecosystem biodiversity (Vaquer-Sunyer and Duarte 2008). As we move downstream, hypoxic conditions decrease, nevertheless days to weeks of hypoxic conditions are observed even near the mouth of the estuary (Table 8.3).

Oxygen concentrations below the mean lethal concentration (LC50) of 4.59 mg/L found by Vaquer-Sunyer and Duarte (2008) for marine biota, i.e., the concentration that may have noticeable effects on biological communities, and below the values recommended by the Canadian Council of Environmental Ministries (CCME) (5.5 mg/L) to guarantee life in rivers (Chambers et al. 2006) are very common in the whole estuary and are the standard condition in its upper reaches (Table 8.3). Concentrations of over 9 mg/l or 90 % oxygen saturation, indicative of good quality waters in terms of oxygen, suitable for salmon and trout (EC Directive on Fresh water for Fish, 78/659/EEC) are very rarely observed (Table 8.3).

In the Guadalquivir, low oxygen events are also combined with high concentrations of dissolved CO_2. High aquatic dissolved CO_2 concentrations (hypercapnia) produces acidosis within the tissue of organisms that maybe compensated by physiological mechanisms, but nevertheless can represents a substantial stress (Burnett 1997).

Hypercapnia also results in reduced aquatic carbonate ion concentrations that can potentially impact calcium carbonate synthesiser organisms. Calcification rates of mussels and oysters decline linearly with increases in dissolved pCO_2 (Gazeau et al. 2007). Hence, 25 % reductions in calcification rates are predicted at ambient pCO_2 values of 750 ppm ($CO_3^{-2} \approx 133$ μM), which corresponds to the atmospheric CO_2 concentration expected by the end of this century under the IS92a IPCC scenario (IPCC 2007). The shells of mussels completely dissolve when pCO_2 levels exceeds 1,800 ppm ($CO_3^{-2} \approx 75$ μM) (Gazeau et al. 2007), a pCO_2 level observable in the estuary.

Inhibition of calcification is also important in the early phases of molluscan larvae, many of which have a high capacity to resist hypoxia (Vaquer-Sunyer and Duarte 2008), but are highly sensitivity to hypercapnia, such as clams, scallops and oyster (Miller et al. 2009; Talmage and Gobler 2009).

Indeed, carbonate concentrations in the estuary are not high enough to completely support calcifying organisms life-cycles (Fig. 8.10b). This is consistent with recent analysis of the benthic macrofauna community in the estuary that indicates low abundances of calcifying organisms (Arias et al. 2010). Initially this might be attributed to the salinity gradient, however, if that was the case, a higher presence of shell-forming species should be found in the vicinity of the estuary mouth, where the oxygen saturation levels were also favourable, and only one mollusc species was present. Except for summer months, Ω, the saturation state of a system with respect to aragonite and calcite, was <1.0 indicating a deficit of aragonite in the estuaries waters during most of the annual cycle (Fig. 8.15). Even in summer, the saturation state remained below the level that is considered optimum for calcification in coastal surface waters (Langdon and Atkinson 2005). Since bivalve larvae exclusively use aragonite to form their shells, and this form is more

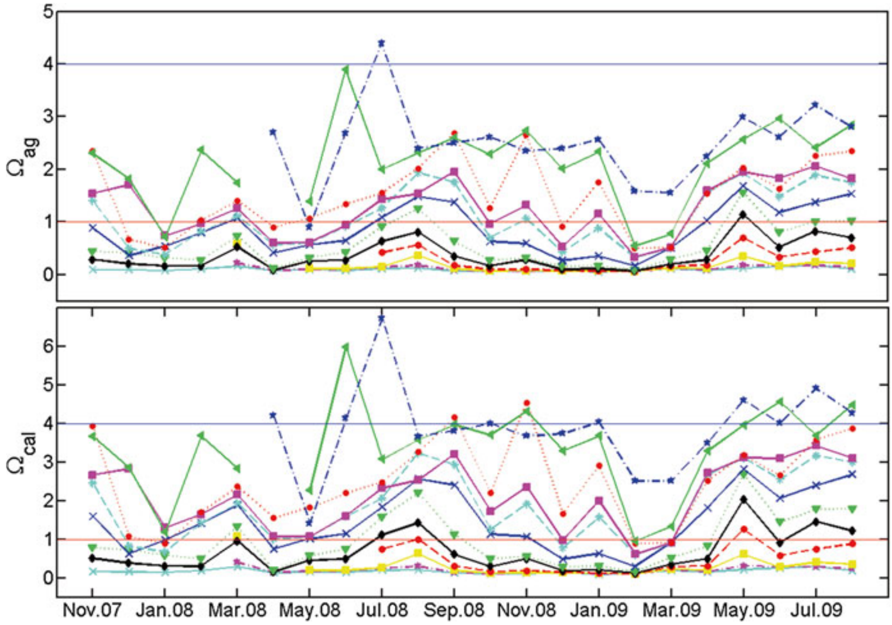


Fig. 8.15 Aragonite (calcium carbonate) saturation levels vs months

soluble than calcite, the low values of Ω present in the estuary seem to indicate a critical inhibition of bivalve shell formation in early life-stages. The adverse scenario for calcifying organisms maybe further enhanced by the local acidification resulting from the degradation of organic matter in the upper sediment layers (Jahnke and Jahnke 2000).

These combined direct effects on organisms are the most likely explanation of the very low pelagic-benthic biodiversity in the Guadalquivir. This results in biological communities with a broad distribution typical of degraded environments (Keister et al. 2000). Toxicity from chemicals is also another issue that interacts with the harsh physiochemical environment to limit biodiversity. Furthermore, a number of potential indirect impacts and feedbacks may also be relevant, including toxic algal toxins that are persistent in some sections of the estuary (E. Costas Pers. Comm.) and shifts in food webs (Rabalais 2002), but are hard to assess considering the present degraded environment.

8.5 Summary

The Guadalquivir estuary has an important place in western European history. It is thought to be the centre of the mythical Tartessos civilization and cardinal to the medieval Al-Andalus culture. It's waters greeted the Nao Victoria after the first circumnavigation of the globe, and were the springboard for the first exchanges between

Europe and the New World, ushering in Spain's Golden Age. Nevertheless, this long historical role has taken its toll on the estuary functioning. The intensification of human pressure since the eighteenth century combined with a limited understanding of the systems dynamics has resulted in ever increasing socio-economic and environmental conflicts related to Andalucia's largest river-estuary-delta system.

Within this context, the existing scientific framework could not answer the multiple concerns raised about the consequences of dredging the estuary to substantially increase its depth, so as to allow larger ships to serve the port of Seville. The cultural and environmental significance of the estuary, magnitude of the planned dredging, and numerous conflicting economic interests, generated great expectations from all levels of society that demanded a rigorous and unquestionable diagnosis of the present state of the system. This was particularly the case considering Spain is a member of the European regulatory framework, which increasingly demands knowledge-based decision-making, and where ecosystem-based management is a fundamental core of environmental legislation. Since the Fifth and Sixth Environmental Action programmes,[1,2] EU policies and directives have encourage the integrated management ecosystems. This approach has inspired a number of ecosystem-based legal instruments that directly affect the estuary, such as the Water,[3] Marine Strategy[4] and Habitat[5] directives, as well as the emerging EU Biodiversity Strategy for 2020.

A basic holistic approach to understanding the estuarine ecosystem including its physical and biogeochemical dynamics and how these act to control biodiversity was identified as the first step towards making knowledge-based decisions about its sustainable use.

The hierarchical monitoring programme described here demonstrates the feasibility of this approach. Intensive use of satellites and remotely operated sensors was critical to this endeavour, allowing detailed information to be collected in a very short time period (30 months). The real-time monitoring network allowed us to resolve the critical spatio-temporal scales necessary to provide a baseline understanding of the major physical and biogeochemical processes within the estuary. Discrete sampling of variables, such as nutrients and dissolved inorganic carbon, helped to fill in the gaps and synoptic satellite images allowed the estuary influence on the inner shelf to be evaluated.

[1] Decision No 2179/98/EC of the European Parliament and of the Council of 24 September 1998 on the review of the European Community Programme of policy and action in relation to the environment and sustainable development "Towards sustainability".

[2] Decision No. 1600/2002/EC of the European Parliament and of the Council of 22 July 2002 laying down the Sixth Community Environment Action Programme.

[3] Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy.

[4] Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive).

[5] Directive 92/43/EEC of the Council of 21 May 1992 on the Conservation of natural habitats and of wild fauna and flora.

The overall diagnosis depicts an estuary in a poor condition. Agriculture and other land use demands have eliminated most (85 %) of the existing marshes and tidal flats during the twentieth century (González Arteaga 1993; Losada et al. 2010). Together with canalisation for navigation, these modifications have meant a drastic change in geomorphology; transforming the estuary from a multi-channel, meandering system intricately connected to tidal flats and the surrounding delta, into a single, almost straight channel closed off from the surrounding natural areas. The construction of the dam at Alcalá del Río in 1930 has modified tidal propagation and competing interests for water use, have in recent decades, reduced freshwater inputs by 60 % (Contreras and Polo 2010). Furthermore, agricultural land use and large municipalities introduce large loads of organic matter and inorganic compounds, particularly into the upper reaches of the estuary (Contreras-Arribas 2013; Mendiguchía et al. 2007).

The net cumulative consequences of this intense anthropogenic pressure, are the acute alteration of numerous processes important for maintaining the good environmental status of the estuary. The multiple damming of the river basin (55 dams >15 m high) has resulted in mainly fine-grained sediments reaching the estuary. Combined with the changes in geomorphology that have created enhanced velocities which favours the continuous resuspension sediments and erosion of the shore (Díez-Minguito et al. 2013b; Caballero et al. 2013). The generally very high residence time of the estuary means this fine grained material is only rarely flushed out into the surrounding coastal zone. The problem is compounded by the isolation of the estuary from tidal flats and marshes that would usually provide the function of high accretion areas (Hatton et al. 1983), effectively removing large volumes of suspended material from the system. Indeed, recent paleo-geological dating of sediments within Doñana National Park suggests changes in accretion rates related to anthropogenic modifications of the Guadalquivir and Guadiamar (Gascó et al. 2006). Worryingly the most recent decreases appear to correspond with the complete isolation of the park from the estuary following the Aznalcollar mine accident.

Persistent high levels of suspended solids means very little light is available for photosynthesis and results in a water column where consumption dominates over oxygen production. This issue is enhanced by the high organic matter and nutrient loading (particularly nitrate) from agricultural and urban land uses, which stimulates respiratory decomposition-catabolism and denitrification. The negative net metabolic balance of the estuary also results in very high CO_2 concentrations. As for suspended sediments, the generally low, fresh water flow rates resulting in little or no stratification, combined with the geomorphological modifications, have resulted in estuarine waters with generally high residence times. These strongly contribute to water quality problems, particularly in the upper section with the highest matter loadings, both for chronic problems, but also for rare accidental events such as the Aznalcollar mining spill. Finally, the loss of tidal flats and marshes represents a substantial reduction in the photosynthetic organisms that may potentially interact in nitrogen and carbon cycling and a direct reduction in the surface area of the estuary, and thus potential for air-water transfer of O_2, CO_2 and N_2.

The present defining conditions within the estuary are that of a system where low dissolved O_2, high CO_2, high TSS and a lack of stable benthic-intertidal habitats limits biodiversity, more than is typical of other estuaries. Although a lack of data impedes historical comparisons. Examples of other ecosystem services that have been lost or damaged by anthropogenic deterioration of the estuary exist. For example, the previously better water quality and the existence of tidal marshes is implicated in the enhanced suitability of the estuary as a nursery for juveniles fish species and higher yields in nearby fisheries (Cattrijsse et al. 1997).

Recovery of the ecosystem-services supplied by the Guadalquivir is a challenge, but entirely possible (Howden et al. 2010); given careful balancing of the cultural, social, economic and environmental demands of all stakeholders within a knowledge-based framework. Compromises will have to be made by all, but considering that most of the issues that affect the estuary will only become more acute in the future, it is in the interests of all to work towards sustainable solutions. The high resolution monitoring data described in this paper can act as a valuable resource for models aimed at risk-scenario analysis, such as extensive dredging or accidents, where uncertainties in the system response are high. The ecosystem response of other remedial actions that may be positive for the environmental status of the estuary, such as an increment in the flow of freshwater, reductions or repositioning of point source loading or an increment in the tidal surface, can also be investigated at a low cost with the RTRM. The potential of the RTRM to follow up the ecosystem response to any human or natural forcing in real time and for all the scales involved facilitates the implementation of adaptive policy designs. These policies should incorporate the knowledge and ecosystem based process of decision-making demanded by the society and European legislation. Overall the application of robust and cost-efficient technology to estuarine monitoring can quickly generate knowledge on key physical and biogeochemical processes and their potential impacts on biodiversity, providing the necessary scientific foundations to meet societal and legal demands.

Acknowledgments This research was funded by the Andalusian Regional Government as well as by projects CTM2009-10520/MAR, P09-TEP-4630 and P09-RNM-4735, P09-RNM-4853, and DMI-COR-LTR753-S. E.P.M. is supported by a Junta para la Ampliación de Estudios (JAE-Doc-2010) contract, part-funded by the European Union (European Social Fund, ESF2007- 2013) and the Spanish Ministry for Economy and Competitiveness.

References

Alexander GR, Hansen EA (1986) Sand bed load in a brook trout stream. North Am J Fisher Manag 6(1):9–23

Almroth-Rosell E, Tengberg A, Andersson S, Apler A, Hall PO (2012) Effects of simulated natural and massive resuspension on benthic oxygen, nutrient and dissolved inorganic carbon fluxes in Loch Creran, Scotland. J Sea Res 72:38–48

Alvarez O, Izquierdo A, Tejedor B, Mañanes R, Tejedor L, Kagan B (1999) The influence of sediment load on tidal dynamics, a case study: Cádiz bay. Estuar Coast Shelf Sci 48:439–450

Appleby J, Scarratt DJ (1989) Physical effects of suspended solids on marine and estuarine fish and shellfish, with special reference to ocean dumping: a literature review. Canadian technical report of fisheries and aquatic sciences No 1681

Arias AH, Marcovecchio JE, Freije RH, Ponce-Velez G, Vázquez Botello A (2010) Análisis de fuentes y toxicidad equivalente de sedimentos contaminados con PAHs en el estuario de Bahía Blanca, Argentina. Hidrobiológica 20(1):41–56

Benson BB, Krause D Jr (1984) The concentration and isotopic fractionation of oxygen dissolved in freshwater and seawater in equilibrium with the atmosphere. Limnol Oceanogr 29(3): 620–632

Bilotta G, Burnside N, Cheek L, Dunbar M, Grove M, Harrison C, Joyce C, Peacock C, Davy-Bowker J (2012) Developing environment-specific water quality guidelines for suspended particulate matter. Water Res 46(7):2324–2332

Boehlert GW, Morgan JB (1985) Turbidity enhances feeding abilities of larval Pacific herring, Clupea harengus pallasi. Hydrobiologia 123(2):161–170

Boynton WR, Kemp WM, Osborne CG, Kaumeyer KR, Jenkins MC (1981) Influence of water circulation rate on in situ measurements of benthic community respiration. Mar Biol 65(2): 185–190

Buck DH (1956) Effects of turbidity on fish and fishing. Trans 1956 N Am Wildl Conf Oklahoma Fish Res Lab 21:249–261

Burnett LE (1997) The challenges of living in hypoxic and hypercapnic aquatic environments. Am Zool 37(6):633–640. doi:10.1093/icb/37.6.633

Caballero I, Morris EP, Ruiz J, Navarro G (2013) Assessment of suspended solids in the Guadalquivir estuary using new DEIMOS-1 medium spatial resolution imagery. Remote Sens Environ 146:148–158, doi:http://dx.doi.org/10.1016/j.rse.2013.08.047

Cai WJ, Wang Y (1998) The chemistry, fluxes, and sources of carbon dioxide in the estuarine waters of the Satilla and Altamaha Rivers, Georgia. Limnol Oceanogr 43(4):657–668

Cattrijsse A, Dankwa HR, Mees J (1997) Nursery function of an estuarine tidal marsh for the brown shrimp *Crangon crangon*. J Sea Res 38(1):109–121

Chambers PA, Culp JM, Glozier NE, Cash KJ, Wrona FJ, Noton L (2006) Northern rivers ecosystem initiative: nutrients and dissolved oxygen–issues and impacts. Environ Monit Assess 113(1–3):117–141

Contreras Arribas EP (2013) Influencia de los aportes fluviales en la calidad del agua del estuario del Guadalquivir. Ph. D. dissertation, Department of Agronomy, University of Córdoba

Contreras EP, Polo MJ (2010) Capítulo 2: Aportes desde las cuencas vertientes. Propuesta metodológica para diagnosticar y pronosticar las consecuencias de las actuaciones humanas en el estuario del Guadalquivir. Group of Fluvial Dynamic and Hydrology, University of Córdoba, Córdoba

Contreras EP, Polo MJ (2011) Influence of reservoirs and land use changes on the river contribution: case study of the Guadalquivir estuary. Geophys Res Abstr 13:12719

Dauer DM, Rodi AJ, Ranasinghe JA (1992) Effects of low dissolved oxygen events on the macrobenthos of the lower Chesapeake Bay. Estuaries 15(3):384–391

Dickson AG (1990) The oceanic carbon dioxide system: planning for quality data. US JGOFS News 2(2)

Díez-Minguito M, Baquerizo A, Ortega-Sánchez M, Navarro G, Losada M (2012) Tide transformation in the Guadalquivir estuary (SW Spain) and process-based zonation. J Geophys Res Oceans 117(C3):1978–2012

Díez-Minguito M, Contreras E, Polo M, Losada M (2013a) Spatio-temporal distribution, along-channel transport, and post-riverflood recovery of salinity in the Guadalquivir estuary (SW Spain). J Geophys Res Oceans 118(5):2267–2278

Díez-Minguito M, Baquerizo A, de Swart HE, Losada M (2013b) Structure of the turbidity field of the turbidity field in the Guadalquivir estuary: Analysis of field data and a box model approach. Submitted to J Geophys Res

Drake P, Borlán A, González-Ortegón E, Baldó F, Vilas C, Fernández-Delgado C (2007) Spatio-temporal distribution of early life stages of the European anchovy Engraulis encrasicolus L.

within a European temperate estuary with regulated freshwater inflow: effects of environmental variables. J Fish Biol 70(6):1689–1709
Elser JJ, Bracken ME, Cleland EE, Gruner DS, Harpole WS, Hillebrand H, Ngai JT, Seabloom EW, Shurin JB, Smith JE (2007) Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. Ecol Lett 10(12):1135–1142
Fernández-Delgado C, Drake P, Arias A, García-González D (2000) Peces de Doñana y su entorno. Ministerio de Medio Ambiente, Madrid
Fischer HB, List EJ, Koh RC, Imberger J, Brooks NH (1979) Mixing in inland and coastal waters. Academic, New York, 483 p
Gallego Fernández JB, García Novo F (2007) High-intensity versus low-intensity restoration alternatives of a tidal marsh in Guadalquivir estuary, SW Spain. Ecol Eng 30(2):112–121
García E, Alonso C, Jiménez M, Maclino I (2004) Perspectivas de investigación sobre puertos y fondeaderos en el sur de Hispania. In: Le strutture dei porti e degli approdi antichi. ANSER, Seminar 2, Rome,16–17 Apr 2004
Gascó C, Antón MP, Pozuelo M, Clemente L, Rodríguez A, Yanez C, González A, Meral J (2006) Distribution and inventories of fallout radionuclides (239+ 240Pu, 137Cs) and 210Pb to study the filling velocity of salt marshes in Doñana National Park (Spain). J Environ Radioact 89(2):159–171
Gazeau F, Quiblier C, Jansen JM, Gattuso JP, Middelburg JJ, Heip CH (2007) Impact of elevated CO2 on shellfish calcification. Geophys Res Lett 34(7):1–15
Glasgow H, Glasgow J, Burkholder R, Reed A, Lewitus J, Kleinman M (2004) Real-time remote monitoring of water quality: a review of current applications, and advancements in sensor, telemetry, and computing technologies. J Exp Mar Biol Ecol 300(1):409–448
Glenn S, Schofield O, Chant R, Kohut J, Roarty H, Bosh J, Bowers L, Gong D, Kerfoot J (2007) Wind-driven response of the Hudson River plume and its effect on dissolved oxygen concentrations. Environ Res Eng Manag 1(39):14–18
González Arteaga J (1993) Valencianos y arroz en las marismas del Guadalquivir. Revista de estudios Andaluces 19:67–95
Granado-Lorencio C (1991) The effect of man on the fish fauna of the River Guadalquivir, Spain. Fish Res 12(2):91–100. doi:http://dx.doi.org/10.1016/0165-7836(91)90034-D
Grasshoff KEM, Kremling K (1983) Methods of seawater analysis, 2nd edn. Weinheim, Verlag Chemie
Gutierrez FJ, Roque D, Navarro G (2011) Ocean data manager handles large, diverse data sets. Sea Technol 52:25–30
Gutiérrez F, Navarro G, Ruiz JH (2009) Real-time telemetry of hydrodynamic and hydrological variables. Sea Technol 50(8):47–51
Hansen DV, Rattray M Jr (1966) New dimensions in estuary classification. Limnol Oceanogr 11:319–326
Hatton RS, Delaune RD, Patrick WH Jr (1983) Sedimentation, accretion, and subsidence in marshes of Barataria Basin, Louisiana. Limnol Oceanogr 28(3):494–502
Howden NJK, Burt TP, Worrall F, Whelan MJ, Bieroza M (2010) Nitrate concentrations and fluxes in the River Thames over 140 years (1868–2008): are increases irreversible? Hydrol Processes 24(18):2657–2662. doi:10.1002/hyp.7835
IPCC AR4 SYR (2007) Climate change 2007: synthesis report, contribution of working groups I, II and III to the fourth assessment report of the Intergovernmental Panel on Climate Change, IPCC, ISBN 92-9169-122-4
Jahnke RA, Jahnke DB (2000) Rates of C, N, P and Si recycling and denitrification at the US Mid-Atlantic continental slope depocenter. Deep Sea Res Part I Oceanogr Res Pap 47(8): 1405–1428
Janssen F, Huettel M, Witte U (2005) Pore-water advection and solute fluxes in permeable marine sediments(II): Benthic respiration at three sandy sites with different permeabilities (German Bight, North Sea). Limnol Oceanogr 50(3):779–792
Keister JE, Houde ED, Breitburg DL (2000) Effects of bottom-layer hypoxia on abundances and depth distributions of organisms in Patuxent River, Chesapeake Bay. Mar Ecol Prog Ser 205:43–59

Kineke G, Sternberg R, Trowbridge J, Geyer W (1996) Fluid-mud processes on the Amazon continental shelf. Cont Shelf Res 16(5):667–696
King B, Wolanski E (1996) Bottom friction reduction in turbid estuaries. Coast Estuar Stud 50:325–337
Kirk KL (1992) Effects of suspended clay on Daphnia body growth and fitness. Freshw Biol 28(1):103–109
Langdon C, Atkinson M (2005) Effect of elevated pCO2 on photosynthesis and calcification of corals and interactions with seasonal change in temperature/irradiance and nutrient enrichment. J Geophys Res Oceans 110(C9):1978–2012
Lobo FJ, Sánchez R, González R, Dias JMA, Hernández-Molina FJ, Fernández-Salas LM, Díaz del Rio V, Mendes I (2004) Contrasting styles of the Holocene highstand sedimentation and sediment dispersal systems in the northern shelf of the Gulf of Cádiz. Cont Shelf Res 24:461–482
López-Ruiz A, Ortega-Sánchez M, Baquerizo A, Navidad D, Losada MÁ (2012) Nonuniform alongshore sediment transport induced by coastline curvature. Coastal engineering proceedings 1 (33):sediment. 29
Losada M, Baquerizo A, Bramato S, Díez-Minguito M, Ortega M (2010) Capítulo 3: Dinámica mareal. Propuesta metodológica para diagnosticar y pronosticar las consecuencias de las actuaciones humanas en el estuario del Guadalquivir. Grupo de Dinámica de Flujos Ambientales, University of Granada, Granada
McLeay DJ, Ennis GL, Birtwell IK, Hartman GF (1984) Effects on Arctic grayling (Thymallus arcticus) of prolonged exposure to Yukon placer mining sediment: a laboratory study. Can Tech Rep Fish Aquat Sci 1241:96
Mendiguchía C, Moreno C, García-Vargas M (2007) Evaluation of natural and anthropogenic influences on the Guadalquivir River (Spain) by dissolved heavy metals and nutrients. Chemosphere 69(10):1509–1517
Miller AW, Reynolds AC, Sobrino C, Riedel GF (2009) Shellfish face uncertain future in high CO2 world: influence of acidification on oyster larvae calcification and growth in estuaries. PLoS One 4(5):e5661
Mintrop L, Pérez FF, González-Dávila M, Santana-Casiano JM, Körtzinger A (2000) Alkalinity determination by potentiometry: intercalibration using three different methods. Cienc Mar 26(1):23–27
Navarro G, Ruiz J (2006) Spatial and temporal variability of phytoplankton in the Gulf of Cádiz through remote sensing images. Deep Sea Res II: Top Stud Oceanogr 53(11):1241–1260
Navarro G, Gutiérrez F, Díez Minguito M, Losada M, Ruiz J (2011) Temporal and spatial variability in the Guadalquivir estuary: a challenge for real-time telemetry. Ocean Dyn 61(6):753–765
Navarro G, Huertas IE, Costas E, Flecha S, Díez-Minguito M, Caballero I, López-Rodas V, Prieto L, Ruiz J (2012) Use of a Real-Time Remote Monitoring Network (RTRM) to characterize the Guadalquivir estuary (Spain). Sensors 12(2):1398–1421
Newcombe CP (1994) Suspended sediment in aquatic ecosystems: Ill Effects as a function of concentration and duration of exposure. Province of British Columbia, Habitat Protection Branch, Ministry of the Environment, Land and Parks, Victoria
Newcombe CP, MacDonald DD (1991) Effects of suspended sediments on aquatic ecosystems. N Am J Fish Manag 11(1):72–82
Nortek AS (2005) Aquadopp user guide. Vangkroken, Norway
Pihl L, Baden SP, Diaz RJ, Schaffner LC (1992) Hypoxia-induced structural changes in the diet of bottom-feeding fish and Crustacea. Mar Biol 112(3):349–361
Prieto L, Navarro G, Rodríguez Gálvez S, Huertas IE, Naranjo JM, Ruiz J (2009) Oceanographic and meteorological forcing of the pelagic ecosystem on the Gulf of Cádiz shelf (SW Iberian Peninsula). Cont Shelf Res 29(17):2122–2137
Rabalais NN (2002) Nitrogen in aquatic ecosystems. AMBIO: J Human Environ 31(2):102–112
Rabalais NN, Turner RE, Díaz RJ, Justić D (2009) Global change and eutrophication of coastal waters. ICES J Mar Sc: J du Conseil 66(7):1528–1537

Redding JM, Schreck CB, Everest FH (1987) Physiological effects on coho salmon and steelhead of exposure to suspended solids. Trans Am Fish Soc 116(5):737–744

Reyes-Merlo M, Díez-Minguito M, Ortega-Sánchez M, Baquerizo A, Losada M (2013) On the relative influence of climate forcing agents on the saline intrusion in a well-mixed estuary: medium-term Monte Carlo predictions. J Coast Res 65:1200–1205

Riedel B, Zuschin M, Stachowitsch M (2012) Tolerance of benthic macrofauna to hypoxia and anoxia in shallow coastal seas: a realistic scenario. Mar Ecol Prog Ser 458:39–52

Ritchie JC (1972) Sediment, fish, and fish habitat. J Soil Water Conserv 27(3):124–125

Rockwell Geyer W, Beardsley RC, Lentz SJ, Candela J, Limeburner R, Johns WE, Castro BM, Dias Soares I (1996) Physical oceanography of the Amazon shelf. Cont Shelf Res 16(5):575–616

Rodríguez Ramírez A, Rodríguez Ramírez J, Rodríguez Vidal L, Cáceres L, Clemente G, Belluomini L, Manfra S, Improta J, de Andrés J (1996) Recent coastal evolution of the Doñana National Park (SW Spain). Quat Sci Rev 15(8):803–809

Saarman E, Prell WL, Murray DW, Deacutis CF (2008) Summer bottom water dissolved oxygen in upper Narragansett Bay. In: Science for ecosystem-based management. Springer, pp 325–347

Scully ME, Friedrichs C, Brubaker J (2005) Control of estuarine stratification and mixing by wind-induced straining of the estuarine density field. Estuaries 28(3):321–326

Talmage SC, Gobler CJ (2009) The effects of elevated carbon dioxide concentrations on the metamorphosis, size, and survival of larval hard clams (Mercenaria mercenaria), bay scallops (Argopecten irradians), and Eastern oysters (Crassostrea virginica). Limnol Oceanogr 54(6):2072

Turner Designs (2012) Application note S-0035 Cyclops user's manual. Turner Designs, Sunnyvale

UNESCO (1994) Protocols for the Joint Global Ocean Flux Study (JGOFS) core measurements, IOC manuals and guides 29. UNESCO, Paris

USEPA (2000) Ambient aquatic life water quality criteria for dissolved oxygen (Saltwater): cape cod to cape hatteras. Environmental Protection Agency, Office of Water, Washington, DC PA-822-R-00-012

Van Damme S, Struyf E, Maris T, Ysebaert T, Dehairs F, Tackx M, Heip C, Meire P (2005) Spatial and temporal patterns of water quality along the estuarine salinity gradient of the Scheldt estuary (Belgium and The Netherlands): results of an integrated monitoring approach. Hydrobiologia 540(1–3):29–45

Vaquer-Sunyer R, Duarte CM (2008) Thresholds of hypoxia for marine biodiversity. Proc Natl Acad Sci 105(40):15452–15457

Wang ZB, Fernández Bermejo M (2010) Impact of fluvial sediment input to tidal amplification in an estuary. In: EGU general assembly conference abstracts, 2010, p 4868

Winterwerp JC, Van Kesteren WG (2004) Introduction to the physics of cohesive sediment in the marine environment, Development in sedimentology. Elsevier, Amsterdam

Chapter 9
Shrimp Farming as a Coastal Zone Challenge in Sergipe State, Brazil: Balancing Goals of Conservation and Social Justice

Juliana Schober Gonçalves Lima and Conner Bailey

Abstract The coastal zone of Sergipe State in northeastern Brazil has undergone abrupt changes in recent decades due to multiple use of its resources, including farming of the exotic shrimp *Penaeus vannamei*. The rearing of marine shrimp became an important local economic activity in the late 1990s, occupying increasingly large areas on the coast. Shrimp farming in Sergipe State is practiced mainly through extensive family-based production systems in mangrove areas that have been declared Permanent Preservation Areas by Brazilian law. As a result, these family shrimp farms are considered illegal but the farms themselves long predate promulgation of the law and represent an important source of livelihood for hundreds of families. The rearing of marine shrimp has emerged as an important challenge faced by the Brazilian government in the management of national coastal resources. More intensive shrimp farms have been established just outside the mangrove and, while legal, have a more serious environmental impact on coastal ecosystems while generating limited social benefit compared to low-input family shrimp farms. This chapter discusses marine shrimp farming in the state of Sergipe from the perspective of both social justice and environmental conservation.

J.S.G. Lima (✉)
Department of Fisheries and Aquaculture Engineering, Federal University of Sergipe, Avenida Marechal Rondon s/n, Jardim Rosa Elze, São Cristovão, Sergipe CEP 49100-000, Brazil
e-mail: jsglima@googlemail.com

C. Bailey
Department of Agricultural Economics & Rural Sociology, Auburn University, Auburn, AL, USA
e-mail: cbailey@ag.auburn.edu

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8,
DOI 10.1007/978-3-319-06305-8_9,

9.1 Introduction

Approximately half of the world's population lives within 200 km of the coast (Creel 2003), and population density in the coastal zone is three times higher than global averages (IPCC 2007). People are drawn to the coast for a variety of reasons, including the wealth of natural resources and diversity of both ecological and economic niches that support coastal populations (Bailey and Pomeroy 1996).

The concentration of human populations and the diversity of economic activities found in the coastal zone create conditions where resource use conflicts are all too common. Government policies to protect the coastal zone have been established in many countries, including Brazil, the focus of this chapter. Brazil's coastline extends over 8,698 km, has a population density (105 people per km^2) five times higher than the national average, and is the zone where resource use conflicts are most extreme due to the competing interests of urbanization, industrialization, recreation, and other forms of development including offshore oil and gas exploration and coastal aquaculture (Seraval and Alvest 2011).

Our particular interest is in the extensive mangrove ecosystems found in the coastal zone of Sergipe State in northeast Brazil. Mangrove forests are highly complex and productive ecosystems, serving as important shelter for many species of birds, fish, crustaceans and shellfish (Saint-Paul and Schneider 2010). In Brazil and elsewhere in the world, mangrove represent sources of many valuable resources including wood for construction, charcoal production and cooking, as well as the harvest of shrimp, crabs, fish, honey, and tannins (Panitz 1997; Rebelo-Mochel 1997). Historically, the only significant activity that involved clearing of mangrove in Sergipe State was for the purpose of producing salt in shallow flats through evapotranspiration. Before refrigeration, salt was an important commodity for food preservation, and a small area of mangrove in Sergipe was cleared for salt production over 100 years ago. During the last two decades, these salt flats have been converted first to fish ponds and then, beginning in 1998 to shrimp ponds because the income potential, even for low intensity shrimp production, was higher than from the production of fish or salt.

The Brazilian government has declared mangrove forests throughout Brazil to be Permanent Protected Areas (PPA). In Sergipe State, the government is considering what actions might be taken – including the possibility of prohibiting the production of shrimp by local small-scale farmers. The issue takes on a social justice dimension because small-scale shrimp farmers have established rights based on historical use of the resource. In this chapter we will refer to such rights as "traditional resource use rights," which have been the focus of a number of authors (e.g., Christy 1982; Posey and Dutfield 1996; Redford and Padoch 1992). This concept is similar to what is sometimes referred to as traditional ecological knowledge (e.g., Inglis 1993) and is further grounded in the broader literature on common property management (e.g., Agrawal 2001; Berkes 1989). The semi-extensive production process used by these farmers relies on limited inputs in the form of stocking materials and commercial feeds. As a result, the ecological impact of these small-scale farmers tends to be relatively benign.

Also in this chapter we describe a second set of shrimp producers who have government permits to operate because their ponds are outside the PPA. The production system used by these farmers generally is referred to as semi-intensive, meaning that they employ relatively high stocking densities and relatively large amounts of commercial feed. Even though these farms are not physically located within the mangrove they have a direct impact on the PPA because they withdraw water from the mangrove and, in particular, because they discharge large volumes of nutrient-rich effluent into the mangrove.

In addition to describing the two production systems and their environmental impacts, we examine the impact of both systems on local food security, employment generation, and income distribution of the two systems. We develop an argument in favor of balancing resource management and social justice to allow existing small-scale producers to continue using current practices in areas already in production.

Clear and enforceable policies to protect mangrove ecosystems and the coastal zone in Brazil are necessary. Within the framework of a broad conservation policy, we argue that there are clear social and environmental advantages of providing limited opportunity for existing small-scale shrimp farmers to continue operations. At the same time, we argue that the Ministry of the Environment should not allow the adoption of more intensive shrimp farming practices within the PPA because of the potential for adverse environmental consequences. Before comparing in detail the two production systems, we briefly describe the administrative and legal framework of coastal zone management in Brazil and the rationale for establishing PPAs in mangrove ecosystems. We then discuss experience elsewhere in the world where conservation policies have benefitted from incorporating local populations into effective management strategies. After our discussion of the two production systems used in Sergipe State, we conclude with a discussion of policy formulation that takes into account both conservation and social justice.

9.2 Coastal Zone Management in Brazil

Production of penaeid shrimp in brackish water ponds has become a major industry in many tropical nations in Asia and Latin America since the 1980s (Bailey 1988). Shrimp farming came relatively recently to Brazil, with development occurring only in the 1990s and concentrated primarily in the northeast where it has become a central coastal zone management concern (Seraval 2010). The Federal Constitution of 1988, in § 4 of Article 225, defines the coastal zone as "national patrimony" of unique value and charges the Ministry of the Environment with responsibility for environmental preservation and management of natural resources. The National Coastal Zone Management Plan (PNGC – Plano Nacional de Gerenciamento Costeiro) was established in 1988 charging the Ministry of the Environment with implementation (Muñoz 2001; Seraval 2010). The PNGC is designed to be integrative and participatory in nature, with an overall goal of establishing general standards for management of the country's coastal zone. These standards in turn are to be used as the basis of coastal zone policies at federal, state, and local levels.

Mangrove forests are defined by law as PPAs. Most recently, in 2012, the Forest Code (Law No. 12.651/12) reaffirmed that PPAs include every protected area, covered or not covered by native vegetation, with the environmental function of preserving water resources, landscape, geological stability and biodiversity, facilitating the genetic flow of fauna and flora, soil protection and ensuring the well-being of human populations. Because mangroves are considered PPAs, current law does not allow for issuance of licenses to operate shrimp farms in any PPA. Inclusion of human well-being as a criterion for managing natural resources provides justification to modify conservation policies in the interests of social justice. In this chapter, we recommend that current laws governing PPAs be modified to allow very limited use of mangrove areas for sustainable harvests by small-scale producers who have traditional use rights over the land.

The current national Forest Code allows the practice of certain productive activities in PPAs only if such activities are of public utility or social interest and have a low environmental impact. Sustainable agroforestry activities are included in this provision as long as these activities are conducted by rural small farm families and they do not cause a distortion of the vegetal cover (mangroves in case of shrimp farms) and do not affect environmental functioning of the area. The definition of who constitutes a small rural family farm according to the law 4771/65 is that the land be operated primarily with labor of the family. Moreover, 80 % of the family's gross income is to be derived from the agroforestry activity (which we believe covers some types of shrimp farming, depending on its management, in an ecosystem dominated by mangrove). This legal provision is important because it explicitly allows some small-scale producers to operate within a PPA as long as they do not disrupt the ecosystem. Wealthier producers who are unlikely to rely primarily on their own labor or whose other incomes would not exceed 20 % of their total income, on the other hand, would be excluded from operating within a PPA.

Managing mangrove ecosystems is the responsibility of the Ministry of the Environment, which has to work closely with other ministries, including the Ministry of Fisheries and Aquaculture, which is responsible for promoting aquacultural development. Not surprisingly, these two ministries have different perspectives, with the Ministry of the Environment concerned with conservation issues and the Ministry of Fisheries and Aquaculture primarily concerned with technical approaches to expand production. In practice, the Ministry of Fisheries and Aquaculture has promoted more intensive, large-scale producers. Problems of coordination between different government agencies with different mandates and constituencies are not unique to Brazil.

Aquaculture is not the only source of pressure on PPAs and the coastal zone generally. Urban sprawl, industrial development, and the growth of coastal tourism all impinge on the coastal zone. In addition, certain illegal activities occur in PPAs because these areas are isolated and activities, such as smuggling drugs, can be carried out with little likelihood of discovery. Brazil has a growing drug problem (Forero 2013) and drug smuggling through the mangrove PPAs in Sergipe State has contributed to endemic violence in the coastal zone.

9.3 Managing Nature Reserves and PPAs

Over the past 30 years, natural resource managers have come to realize that involving local populations in conservation of biodiversity makes sense. Prior to this change, a whole generation of resource managers were influenced by Garrett Hardin (1968), who argued that people behave in individually selfish ways in the face of a declining resource base and the only way to protect natural resources is for strong government action. This view has been widely challenged by economists, sociologists, political scientists, geographers and biologists, who have documented the existence of local management systems that are effective in meeting both conservation and social equity goals (e.g., McCay and Acheson 1987; Bromley 1992). Among the leaders of this work are Nobel Prize winner in Economics, Dr. Elinor Ostrom (1990; see also Ostrom et al. 2002, 2004).

This understanding – that local people have a combination of local knowledge and motivation to protect the resource for the long-term benefit of local residents – has carried over into management of national parks and other areas of special conservation interest in different parts of the world. In the Philippines, local fishers set aside 20 % of a reef as a sanctuary to allow for fish stocks to rebuild (White 1984). More recently, a comparative study of marine protected areas in the Philippines and Indonesia showed "that a strong linkage exists between social and biological success, with social considerations determining long-term biological success where both biological and social criteria were used" (Christie 2004:155). Adams et al. (2004) describe the links between poverty alleviation and biodiversity conservation. Bini and Diniz-Filho (2005) make the case that nature preserves in the Brazilian Cerrado region are positively correlated with incomes of local residents.

At the start of the twentieth century, the United States was among the first to establish national parks and wildlife preserves and did so largely for the benefit of wealthy recreationalists (Gottlieb 2005). In the 1960s, early conservation efforts in Africa focused on big game species which attracted hunters from around the world. Opportunities to simultaneously maintain healthy game populations and earn foreign exchange earnings led governments of east Africa to introduce policies to remove people from areas favored by hunters. Doing so, however, led to an increase in poaching activities by outsiders which game wardens were ill equipped to stop. Governments rethought their policies and created opportunities for local residents to have a stake in the hunting industry, creating incentives for residents to help game wardens eliminate poachers (Anon. 2010) and improve rural livelihood. Equally importantly, involvement of local residents has created incentives to protect habitat that wildlife depend upon.

Conservation of biodiversity and the protection of rural livelihoods are compatible and often mutually supportive public policy objectives. One reason is that people who are economically desperate are more likely to engage in ecologically destructive actions, while those who are able to earn a living from the ecosystem are more likely to protect this resource. A considerable body of research exists to support this thesis, with cases on all continents and in both terrestrial and maritime settings (e.g., Brechin et al. 2002; Cernea and Schmidt-Soltau 2006; Hayes 2006; West and Brechin 1991). The logic behind this approach is that local residents can find economic advantage from protecting endangered species or ecosystems, and that this approach provides multiple benefits. The first of these is that people living in the area to be dedicated as a nature reserve or protected area do not have to be displaced (Wilkie et al. 2006). The second is that local residents become active participants in protecting habitat and species of interest because doing so is in their interest. Finally, because local people are involved, enforcement costs are low and effectiveness is high. Rather than displacing local populations, increasingly governments are utilizing local knowledge and populations to manage biodiversity (Ghimire 2008).

9.4 Shrimp Farming in Sergipe State

Carvalho and Fontes (2007) provide some basic descriptive statistics, and therefore a useful starting point for considering policy issues related to Sergipe State's shrimp farming industry. Shrimp farming in Sergipe began in 1998 and currently represents only about 3 % of national production (IBAMA 2007). Most shrimp farms are small in scale and family-based operations that use limited inputs and are found in the river basins of the São Francisco, Sergipe, Vaza-Barris, Piauí, and the Real rivers. According to 2004 data from a state agency for industry and mineral resources, CODISE (Companhia de Desenvolvimento Industrial e de Recursos Minerais de Sergipe), there were about 640 ha of active shrimp farms at Sergipe State. The Sergipe River basin had the highest concentration with 36 % of all shrimp farms. The Vaza-Barris River Basin, according to the same statistic source, has about 13 % of all shrimp farms in the state (cited in Carvalho and Fontes 2007).

In 2006 there were reported to be about 987 ha of shrimp farms in Sergipe State. There is no more recent published data on the current extent of shrimp farms, but direct observations by the lead author and review of satellite images make clear that there has been some loss of mangrove through conversion to shrimp ponds. However, the main causes of mangrove loss have been urban sprawl, the expansion of industrial development, and conversion of mangrove to agricultural purposes. Similar patterns are found in other Brazilian states (Panizza and Fournier 2008), reflecting growing anthropogenic pressure on coastal, estuarine, and mangrove resources in Brazil.

9.4.1 Comparing the Two Production Systems

We report here on on-going research on two distinct shrimp farming systems found in Sergipe State. One system is characterized by the use of relatively large numbers of post-larval (PL) shrimp to stock their ponds along with a large volume of

Table 9.1 Key characteristics of the two types of shrimp farms in Sergipe State, 2010

Characteristic	Outside PPA	Inside PPA
Average total area of ponds by owner (ha)	6 a 45	≤10
Stocking density (shrimp/square meter)	20–35	5–10
Feed (kg per ha per season)	2,700	532
Feed conversion ratio	1.25	0.84
Productivity (kg per ha per season)	2,150	631
Survival (%)	70.0	71.1
Practice of polyculture	Not observed	26.67 %
Rearing cycles per year	2.5	3.0
Physico-chemical analysis of water quality	Carried by all the properties observed	6.7 %
Use of fertilizers	Used by all farms observed	6.7 %
Use of antibiotics	Used by all farms observed	6.7 %
Presence of sedimentation basin	Used by all the properties observed	Not observed
Social contribution	Generation of direct and indirect jobs	Food security for the local population and jobs, often indirect, among others

commercial processed feed necessary for the shrimp to grow. Farms of this type generally are described in the Brazilian context as “semi-intensive,” with 20–35 PL shrimp/m^2 and 2.7 mt of feed per hectare each season (Table 9.1). Operation of these ponds requires pumps to move water and aerators to maintain oxygen levels, necessary because heavy organic loading in the ponds creates such a high level of oxygen demand that the entire crop could be lost. The owners of these semi-intensive shrimp farms are relatively well educated, generally live in urban centers, hire people to manage the farms, have access to technical expertise necessary to operate their farms, and of course must have access to financial resources necessary to support this complex operation. The number of such farms in Sergipe State is relatively small. Santos (2009) was able to identify only nine semi-intensive farms that have been licensed in that state.

The second group of producers is made up of families with limited financial resources who we will refer to as “small-scale” producers. These families generally have limited formal education, live near their farms and provide most or all of the necessary management and labor. They buy limited numbers of PL shrimp to stock their ponds and use only small amounts of commercial feed to supplement the natural productivity of their ponds (Table 9.1). Such farms would be described in the aquaculture literature as “semi-extensive” because they do not depend exclusively on natural inputs for stocking and feeding, but rather supplement the natural system to a limited degree. These farms rely on tidal movements rather than pumps to move water into their ponds. Low stocking densities mean that they have relatively few problems with disease and organic loading from effluent is relatively low. Similarly, low stocking densities mean that these producers do not rely on aerators to oxygenate the water. Many but not all of the farms of this group were carved out of the mangrove generations ago for salt production.

Data related to farm practices were collected from 19 farms, 7 of which used semi-intensive production systems. The owners of these 7 farms all belong to the Association of Shrimp Farms of Sergipe State. These farms are located in Estancia, Itaporanga Da'juda, Santa Luzia do Itanhy and Santo Amaro das Brotas municipalities. The remaining 12 farms for which data are available fall under the "small-scale" category. There are 32 such farms in the municipality of São Cristóvão alone with the remainder found in other coastal municipalities. All of the small-scale farmers who participated in this study belong to the Association of Aquaculture Farms of São Cristovão.

For both groups of farms, the selection of study sites was based on farmer willingness to collaborate with the study and the ability of the lead author to access the farm during all months of the year. Data on production (inputs, mortality, harvests) were collected from June 2008 to June 2010. Information on the production systems were collected through focus groups, semi-structured interviews, and direct observations of life in and around these farms. The second author spent 2 weeks in Sergipe State in 2012 and has worked on coastal aquaculture in other parts of the world. In the remainder of this section we describe key differences which distinguish these two systems based on their ecological and social impacts as well as their legal status. Comparative data on production practices of shrimp farms using semi-intensive and semi-extensive systems are shown in Table 9.1.

9.4.1.1 Production Practices

Farms using semi-intensive systems vary in size from 6 to 45 ha, generally are larger than – and are very much more intensively managed than – the farms of small-scale producers. These more intensively managed farms are located just outside the mangrove zone and so are outside the PPA. That said, these farms are intimately related to the mangrove and estuarine ecosystems; water is pumped from and pond effluents are released first into sedimentation ponds and then into the adjacent mangrove. Effluents from shrimp farms include organic wastes from the production process (uneaten feed, fecal material, dead shrimp) as well as any pathogens which may have become established in the ponds. Shrimp producers around the world are engaged in a constant struggle to control bacterial and viral diseases and frequently use an array of drugs and chemicals to keep pathogens under control. In recent years a variety of probiotics have been used by aquaculture producers (Cruz et al. 2012). All of the semi-intensive shrimp farms used probiotics compared to only 7 % of all small-scale producers. Shrimp diseases are most common where semi-intensive production practices are used due to the added physiological stress caused by crowding and by impaired water quality due to heavy feeding. The higher stocking and feeding rates of semi-intensive farms result in higher yields at harvest, which average 2,150 kg/ha compared to 631 kg/ha for small-scale producers.

Feed conversion ratios show that small-scale producers are more efficient in their use of feed than more intensively managed farms. This is so because feed used by small-scale producers is used as a supplement to the natural productivity found in pond waters linked to the mangrove. In contrast, shrimp in more intensively managed farms rely exclusively on processed feed. In addition to processed feeds, all semi-intensive farms used fertilizer to increase the productivity of pond water, a practice used only by 7 % of all small-scale farms, who are able to rely on the natural productivity of water flowing into the ponds during high tides whenever farmers open gates.

Owners of semi-intensive farms are able to hire technical experts to help them manage their ponds. The relatively high stocking densities and use of processed feeds creates water quality problems that must be managed carefully to prevent loss of the crop due to disease or oxygen depletion. Small-scale producers do not have access to formal technical assistance but these farmers have long experience living in the area and intimate knowledge of the mangrove ecosystem. This form of traditional knowledge regarding how the ecosystem functions is directly relevant to their management of shrimp ponds and knowledge is freely shared between this group of producers.

9.4.1.2 Connectedness to Local Needs

The two groups of shrimp farms are socially distinct. All small-scale low intensity farmers work on their farms, often supplemented with household labor. Occasionally a neighbor will be hired should there be a need for additional labor. In contrast, semi-intensive farms depend on hired labor. In some cases, these owners may provide managerial oversight, but in other cases day-to-day operations are managed by a full-time farm manager.

The land and ponds of semi-intensive shrimp farms are clearly marked as private property and the ponds are often surrounded by barbed wire fences and guarded at night. Local residents are not permitted to pass through this private property. In contrast, the ponds of small-scale producers are generally located in immediate proximity to residential areas and mangrove. The origins of this physical proximity stem from the nature of land ownership in Brazil, which is highly unequal (Wolford 2001). Those who are landless in urban settings have established *favelas* (loosely translated as a shanty town), establishing permanent residences in marginal land over which no clear title is held. The town of São Cristóvão is located on a hill overlooking mangrove, ponds, and homes of the owners of small-scale shrimp farms (Fig. 9.1). Thus most of the low-intensity ponds are located in close proximity not only to the homes of the owners but also to the homes of many other people in the larger community. It is common for local residents to walk across the bunds that separate ponds in order to reach mangrove areas where they will harvest a variety of daily needs including wood, crabs, fish, or other resources. No efforts are made to

Fig. 9.1 Small-scale shrimp farms showing proximity to community of São Cristovão, Sergipe (Photo by C. Bailey)

impede such public access. Small-scale shrimp farmers also grow more than shrimp. Because they frequently open gates to their ponds during high tides, to replenish water which has evaporated and to replenish the ponds with natural nutrients from the mangrove, it is not uncommon for indigenous species of fish to enter the ponds and grow alongside the shrimp in a natural polyculture. These fish may have limited market value but are a valuable source of food for local residents. The lead author observed numerous cases where local residents – often but not always kin of the pond owner – were invited into the ponds after the shrimp had been harvested to collect the remaining shrimp and fish for their own consumption.

The ponds of small-scale producers are located in close proximity to daily markets where shrimp are sold to local consumers (Fig. 9.2). Shrimp can be sold in any quantity, including very small amounts that a poor family may use in a daily meal. Some of the shrimp harvested by small-scale producers are sold to middlemen who take the shrimp to more distant urban markets, but a significant portion of the local production goes to feed local residents and is available at a relatively low price so that anyone can afford to purchase shrimp. In contrast, all the shrimp grown in semi-intensive farms are sold middlemen who distribute to urban supermarkets in the state capital of Aracaju or beyond, wherever the best prices may be found. Because the owners generally are not locally resident and the shrimp are not available to

Fig. 9.2 Local marketing of small shrimp in São Cristóvão, Sergipe (Photo by C. Bailey)

local consumers, the local benefit of these semi-intensive farms in the communities where such farms are located is limited.

9.4.1.3 Environmental Interactions

The environmental impact of shrimp farming is a matter of global concern. In Brazil, establishment of PPAs reflects national concern about the health of coastal ecosystems, including mangrove forests. Muhlert et al. (2013) have identified indicators of resource use efficiency and environmental performance appropriate to shrimp farming in Sergipe State. Our comparative data show that semi-intensive shrimp farms use higher stocking densities, larger quantities of feed, and produce more shrimp per hectare (Table 9.1). Feed is the single largest expense for semi-intensive shrimp producers, often accounting for half or more of total production costs. In environmental terms, the impact of feed is to increase nutrient loading which can cause a number of problems, including eutrophication and oxygen depletion as well as promote conditions where bacterial and viral diseases can flourish. Waste released into adjacent ecosystems of cultivated marine shrimp ponds are associated with negative environmental impacts (Boyd 2003; Lucas 2003; Freitas et al. 2008; Machado et al. 2011) and commercial feed is the most important source

Table 9.2 Measures of environmental impact for eight shrimp farms

		Semi-intensive farms				Semi-extensive farms			
		A1	A2	A3	A4	B1	B2	B3	B4
Dry matter content	Summer	3.49	4.32	3.96	4.54	2.77	4.00	4.00	2.95
	Winter	5.26	5.54	5.29	4.43	1.55	2.45	1.84	5.00
Conversion rate of protein	Summer	0.34	0.42	0.38	0.44	0.27	0.39	0.39	0.29
	Winter	0.51	0.54	0.51	0.43	0.15	0.24	0.18	0.49

of waste in shrimp farming (Jackson et al. 2003; Martin et al. 1998; Páez-Osuna et al. 1997).

One measure for determining how efficiently feed is utilized is the feed conversion ratio (Boyd et al. 2007). If 1 kg of feed results in growth of 1 kg of shrimp, the ratio would be 1.0. Small-scale shrimp farmers had a feed conversion ratio of 0.85 (Table 9.1), meaning that processed feed was used as a supplement to the natural productivity of pond water and most probably does not result in excessive net nutrient loading into the ecosystem. Ponds with semi-intensive systems were less efficient, with a feed conversion ratio of 1.25. Shrimp farming systems that depend on commercial feeds rather than using such feeds in a supplementary manner to complement natural productivity impose a burden on the surrounding environment (Martinez-Cordova et al. 2002; Souza et al. 2009).

A second measure of environmental impact is to use the weight of dry matter in both feed and shrimp. Boyd and Tucker (1998) estimate that commercial feed is 90 % dry matter and that shrimp are 25 % dry matter. Measuring dry matter provides a more accurate depiction of feed conversion efficiency and therefore ecological impact. The comparison is between the dry matter content of feed and the dry matter content of the shrimp expressed as a ratio and multiplied by the feed conversion ratio.

Comparisons of dry matter content ratios shown in Table 9.2 are for eight ponds, four of which (A1, A2, A3 and A4) were on semi-intensive farms, with the remaining ponds (B1, B2, B3, and B4) operated by small-scale producers) over the summer and winter seasons from 2010 to 2011. Semi-intensive farms had the highest surplus of dry matter added to ponds, ranging from 3.49 to 4.54 kg/ha in the summer, with somewhat higher loadings (lower efficiency) in the winter season (4.43–5.54 kg/ha). Low input farms operated by small-scale producers had lower loadings of dry matter, ranging from 2.77 to 4.0 kg/ha in the summer and 1.55–2.45 kg/ha in the winter.

The crude protein content used in artificial diets is a key point in determining the sustainability of shrimp farming due to the limited supply of fish meal and fish oils on international markets and the global impact of fisheries that target anchoveta, caplin, menhaden, and other species that, while not consumed directly by humans, are a key part of the oceanic food web (Naylor et al. 2000). Commercial feed in Sergipe State is approximately 35 % crude protein. The conversion rate of protein provides a measure of how many kg of crude protein is required to produce 1 kg of shrimp and is calculated by multiplying the crude protein content by 100 and multiplying again by the feed conversion ratio. By this measure, small-scale farms are far more efficient that the semi-intensive farms (Table 9.2).

In sum, data from the eight ponds show that small-scale farms have a far smaller ecological impact than the semi-intensive farms. Small-scale producers depend much more heavily on the natural productivity of waters that flow into their ponds from adjacent mangrove and estuaries. Data on feed conversion ratios (simple and dried matter) as well as protein conversion efficiency show that small-scale producers are more efficient in their use of natural resources and on this basis are more sustainable in ecological terms both locally (mangrove and estuarine health) and globally (in terms of marine resources). These findings are consistent with Lima (2011) and Lima et al. (2012), who showed that small-scale limited input shrimp farmers interact positively with the adjacent ecosystem because they convert excess organic matter and nutrients from the ecosystem into shrimp biomass which are sold to local consumers.

9.5 Discussion

The category of small-scale producers is a complex and dynamic one. The overall characterization of these producers as using limited inputs on ponds which have been in existence for a long time is, in general, accurate. But it is important to acknowledge that there have been new ponds carved into the mangrove by small-scale producers who do not have long-standing cultural and historical connections to the mangrove and estuary that characterize those who have traditional resource use rights and detailed local ecological knowledge. In the absence of a workable management plan, we believe it likely that additional ponds will be created by small-scale producers who do not have this background. Further, some of these small-scale producers will be interested in increasing the intensity of their production which would have more serious negative environmental impacts than shrimp farmers using semi-extensive systems. Indeed, anecdotal evidence collected during field research indicates that this transition is occurring and is in some cases is limited only by the financial and technical resources of the new shrimp farmers. Thus what we have is not a simple dichotomy between semi-extensive and semi-intensive but a more dynamic blend of systems that is expanding, shifting in composition, and evolving in the direction of semi-intensive production systems with consequent negative environmental impacts.

In this evolving situation, the Ministry of the Environment needs to take action to maintain the ecological integrity of the mangrove and estuarine ecosystem while also protecting the legitimate interests of long-established small-scale producers. Doing so will require managing the ecosystem as a whole, including effluent from shrimp farms outside the PPA and stopping construction of new ponds within the PPA. But this will not be enough. There may need to be a local policy that limits the sale of existing ponds within the PPA to members (or immediate family of members) of producer groups such as the Association of Aquaculture Farms of São

Cristovão representing the interests of small-scale producers. Not all communities where small-scale producers operate have such organizations, but creation of such organizations and limits to the transfer of ownership can be a condition of permits to operate within the PPA being issued.

Policies restricting the sale of shrimp ponds should be established in any event, but such policies will become particularly important if no new ponds are allowed within the PPA. In this situation, the economic value of existing ponds is likely to increase, creating an incentive for owners to sell to the highest bidder. This development would undermine any effort to maintain a small-scale and semi-extensive production system that provides modest livelihoods to local residents and supplies high quality protein to local consumers, not only because the new buyers are likely to be outsiders but there is some likelihood that the new buyers would adopt semi-intensive production technologies to recoup their investment costs. A shift to semi-intensive production systems within the PPA would have a detrimental ecological impact due to effluent discharge among other factors. The new owners would be likely to follow the pattern of other semi-intensive shrimp farmers of selling their harvest in bulk to middlemen supplying major urban and international markets. The new owners may not allow local residents to glean fish and shrimp from ponds after the harvest is completed or to cross their lands to gain access to the mangrove given the fear of theft of shrimp from the more densely stocked ponds.

Well intended policies can sometimes have unintended consequences. The intent behind establishing PPAs in mangrove ecosystems is to protect these ecosystems from destruction. However, a blanket prohibition of any human activity in the mangrove of Sergipe State, or elsewhere in Brazil, would impose undue burdens on coastal residents, particularly among the poor, who rely on mangrove for important sources of food and fuel. Clearing of mangrove for urban or industrial development is different from allowing local residents to harvest crabs, fish and wood as has been done for centuries. Similarly, in the case of Sergipe State, the presence of small-scale low input shrimp farmers who sell their product in local markets and whose environmental impact can be described as benign do not represent a threat to the integrity of mangrove ecosystems. Far more problematic in this regard are the small number of semi-intensive farms who use higher stocking densities and larger volumes of commercial feed, and who discharge effluent into the mangrove. Even though these more intensively managed farms are located outside the PPA, their ecological impact is significant. Machado et al. (2011) did a general estimation of the nutrient and organic matter balance in a typical semi-intensive shrimp farm and showed that this production model has a negative impact on the adjacent environment because a high proportion of the commercial feed used was not effectively converted into shrimp biomass. If the area under such relatively intensive production of shrimp expands significantly, the impact on the coastal zone could be serious.

Experience from other countries suggests that removal of local populations from areas designated as nature reserves and protected areas not only is detrimental to the livelihoods of the affected people but also undermines the effectiveness of the intended conservation efforts. The common lesson learned is that involving local people in the management of protected areas provides a mutually beneficial incentive. The potential for non-extractive nature tourism is considerable. São Cristóvão is of historic interest as one of the oldest towns in Brazil and is located approximately 1 h from the state capital (and airport) of Aracaju. Bird populations and other wildlife in the mangrove would attract domestic and international tourism, and the ecological knowledge of local residents would be vital to developing this form of nature tourism. Local residents with a stake in maintaining the ecological integrity of the PPA would also serve as the eyes and ears of the Ministry of the Environment, reducing the costs of enforcement.

We realize that there are difficulties involved in creating such a partnership between a federal Ministry, or even state and local governments, with poor rural people who have limited education. This is true everywhere in the world. We also realize that owners of shrimp farms using semi-intensive systems will seek to influence policy and ensure their interests are considered, protected, or advanced. Resource management is a highly political arena and coastal management in Brazil is no exception. Based purely on conservation values, all shrimp farming that has any impact on the mangrove should be eliminated where a PPA is established. A balanced approach that protects both a vulnerable ecosystem and also the livelihood of limited resource shrimp farmers is, however, an achievable goal. We make several recommendations in the following section that would allow for such a balanced approach. The recommendations are tailored for conditions found in Sergipe State but may be more broadly applicable elsewhere in Brazil and quite possibly elsewhere in the world.

9.6 Recommendations

We believe that coastal zone management policies must incorporate and find an optimal balance between conservation and social justice values. Many small-scale shrimp farmers of Sergipe State today are the descendants of people who have worked in the mangrove for many generations. They have developed an indigenous knowledge of the ecosystem which allows them to work with nature in gaining a livelihood from the coastal zone and it is fair to say that the ecological health of the PPA in Sergipe State is a product of this long-standing human involvement. This said, it is clear that there are forces at work which require some form of intervention to control or prohibit the expansion of shrimp farming or other human activities in the PPA. The question is how to do this in a way that finds the appropriate balance between conservation and social justice.

In general, we recommend that small-scale farming using semi-extensive methods be a permitted activity in the PPA, and that licenses should be given to producers who fit this profile. Specifically, we recommend:

- The permits should specify that stocking densities are limited to 10 PL shrimp/m^2. This stocking level would require little in the way of supplemental feed and would not generate an excessive net ecological burden on the ecosystem in the form of nutrient and organic matter discharge. Multitrophic systems which incorporate fish or shellfish into a synergistic production process can be encouraged to diversify harvests, increase incomes, and reduce further environmental impacts in keeping with the standards of the National Environmental Council (CONAMA) for marine shrimp farms in Brazil (Resoluçao CONAMA n° 20/1986).

- Seventy percent of licenses to operate shrimp ponds in the PPA would be limited to local residents who have a family history of being involved in utilization estuarine resources, including the management of ponds for fish and/or shrimp, provided that they meet the legal criteria of small family producers who depend primarily on household labor and who earn at least 80 % of their income from their licensed shrimp farms.
- All licenses would be limited to a maximum area of 10 ha, with the expectation that many farms would be smaller than that.
- To be eligible for a license, a farmer would have to belong to an association such as the Association of Aquaculture Farms of São Cristovão. Membership to these associations would be limited to local residents who have a family history of being involved in utilization of estuarine resources, including the management of ponds for fish and/or shrimp. These associations would be self-governing and develop ecologically sustainable production practices based on local knowledge and external technical assistance.
- Producers holding a license allowing them to produce shrimp would be able to transfer ownership to their children or sell their ponds to local residents who have a family history of being involved in utilization of estuarine resources, including the management of ponds for fish and/or shrimp, and who would qualify for a license based on their reliance on household labor and earning at least 80 % of household income from their ponds.
- Producers holding a license to farm shrimp should allow (as is currently the practice) the transit of local residents from communities into the mangrove to gain access to the natural resources which are important to food security and other needs of local people.
- Members of these associations should work closely with local offices of the Ministry of the Environment and other government agencies to report illegal activities or other problems that they observe in the course of their activities in the PPA.
- Members of these associations should develop alternative activities in the PPA that enhance economic stability of local communities, in particular non-extractive activities such as ecotourism.
- Effluent from semi-intensive shrimp farms located immediately outside the PPA needs to be monitored to determine if it represents a threat to water quality or a source of disease organisms that would affect other shrimp producers. Effluent discharge permits should be issued to all semi-intensive shrimp farms and these permits should be withdrawn in the event that problems are identified.
- The number and intensity of production of shrimp ponds per estuary either inside or outside the PPA cannot exceed the estuary's carrying capacity. It will be necessary to establish estimated carrying capacities for each estuary. Such data does not currently exist but is of fundamental importance to establish the number of ponds and the level of intensity of production permitted per estuary.

9.7 Conclusion

In Brazil, as in many other parts of the world, coastal resources are under intense pressure and the federal government of Brazil has established a legal framework to protect vulnerable ecosystems. Legal frameworks established at the national level do not always or adequately take into account local conditions or histories of resource use. Such is the case in Sergipe State, where a blanket enforcement of a PPA would displace small-scale shrimp farmers who have established traditional resource use rights, who earn a modest livelihood, whose production supports food security in the larger community, and whose production practices have minimal and by some measures beneficial impacts on the natural environment.

The Forest Code of 2012 provides for inclusion of human well-being as a criterion for establishing and implementing policy. We believe the small-scale shrimp farmers of Sergipe State represent a case where use of this criterion is legitimate based on the data presented here. We have presented a number of recommendations tailored to conditions found in Sergipe State. These recommendations may be applicable elsewhere in Brazil and beyond, though further research would be needed before the specific conditions and needs could be determined. The recommendations are designed to balance the needs for conservation and social justice in a setting where an established population of small-scale shrimp farmers are operating without licenses but have traditional resource use rights based on generations of occupation and use of the land.

References

Adams WM, Aveling R, Brockington D, Dickson B, Elliott J, Hutton J, Roe D, Vira B, Wolmer W (2004) Biodiversity conservation and the eradication of poverty. Science 306:1146–1149

Agrawal A (2001) Common property institutions and sustainable governance of resources. World Dev 29(10):1649–1672

Anon. (2010) Horns, claws and the bottom line. The Economist, 2 Sept 2010. Available online at http://www.economist.com/node/16941705. Accessed 31 Dec 2011

Bailey C (1988) Social consequences of tropical shrimp mariculture development. Ocean Shoreline Manag 11:31–44

Bailey C, Pomeroy C (1996) Resource dependency and development options in coastal Southeast Asia. Soc Nat Res 9(2):191–199

Berkes F (1989) Common property resources: ecology and community-based sustainable development. Belhaven Press, London

Bini LM, Diniz-Filho JAF (2005) Conservation does not compromise social development in Brazilian Cerrado. Science. Published online 24 May 2005. Available at http://www.sciencemag.org/content/306/5699/1146/reply#sci_el_1353?sid=107a27e2-9534-4dbc-acdf-20fdbbdf5455. Accessed 31 May 2014

Boyd CE (2003) Fertilizantes químicos na aquicultura de viveiros. Rev ABCC 5:79–81

Boyd CE, Tucker CS (1998) Pond aquaculture water quality management. Kluwer Academic Publishers, Boston

Boyd CE, Tucker C, Mcnevin A, Bostick K, Clay J (2007) Indicators of resource use efficiency and environmental performance in fish and crustacean aquaculture. Rev Fish Sci 15:327–360

Brechin SR, Fortwangler CL, West PC (2002) Reinventing a square wheel: critique of a resurgent "protection paradigm" in international biodiversity conservation. Soc Nat Res 15(1):17–40
Bromley DW (1992) Making the commons work; theory, practice, and policy. Institute for Contemporary Studies, San Francisco
Carvalho MES, Fontes AL (2007) A carcinicultura no espaço litorâneo sergipano. Revista da Fapese 3(1):87–112
Cernea MM, Schmidt-Soltau K (2006) Poverty risks and national parks: policy issues in conservation and resettlement. World Dev 34(10):1808–1830
Christie P (2004) Marine protected areas as biological successes and social failures in Southeast Asia. Am Fish Soc Symp 42:155–164. Available at https://depts.washington.edu/smea/sites/default/files/patrickc/MPAs.Christie.AFS%20book.2004.pdf
Christy FT (1982) Territorial use rights in marine fisheries: definitions and conditions. FAO Fisheries technical paper 227. Food and Agriculture Organization of the United Nations, Rome
Creel L (2003) Ripple effects: population and coastal regions. Population Reference Bureau, Washington, DC. Available at http://www.prb.org/pdf/RippleEffects_Eng.pdf. Accessed 22 Dec 2013
Cruz PM, Ibáñex AL, Hermosillo OAM, Saad HCR (2012) Use of probiotics in aquaculture. ISRN Microbiol 2012: Article ID 916845. doi:10.5402/2012/916845
de Souza FMMC, Messias GA, Fialho DH, Soares RB, de Correia ES (2009) Crescimento do camarão marinho *Farfantepenaeus subtilis* (Pérez- -Farfante, 1967) cultivado em tanques com diferentes protocolos de fertilização orgânica. Acta Sci Biol Sci 31:221–226
Freitas U, Niencheski LFH, Zarzur S, Manzolli RP, Vieira JPP, Rosa LC (2008) Influência de um cultivo de camarão sobre o metabolismo bêntico e a quali- dade da água. Rev Bras Eng Agríc Amb 12:293–301
Forero J (2013) Brazil's fight against drug trafficers stretched thin along its porous borders. Guardian Weekly, 29 Jan 2013. Available at http://www.theguardian.com/world/2013/jan/29/brazil-drugs-trade-south-america. Accessed 11 Jan 2014
Ghimire KB (2008) Parks and people: livelihood issues in National Parks Management in Thailand and Madagascar. Dev Chang 25(1):195–229
Gottlieb R (2005) Forcing the spring; the transformation of the American environmental movement. Island Press, Washington, DC
Hayes TM (2006) Parks, people, and forest protection: an institutional assessment of the effectiveness of protected areas. World Dev 34(12):2064–2075
Hardin G (1968) The tragedy of the commons. Science 162:1243–1248
IBAMA (2007) Estatística da Pesca 2007. Ministério do Meio Ambiente. Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis. Diretoria de Fauna e Recursos Pesqueiros. Coordenação Geral de Gestão de Recursos Pesqueiros. 147 p. 2011
Inglis J (1993) Traditional ecological knowledge: concepts and cases. International Development Research Centre, Ottawa
IPCC (Intergovernmental Panel on Climate Change) (2007) Climate change 2007: working group II: impacts, adaptation and vulnerability. 6.2.2 Increasing human utilization of the coastal zone. Available at https://www.ipcc.ch/publications_and_data/ar4/wg2/en/ch6s6-2-2.html. Accessed 22 Dec 2013
Jackson C, Preston NP, Thompson P, Burford M (2003) Nitrogen budget and effluent nitrogen components at an intensive shrimp farm. Aquaculture 218:397–411
Lima JSG (2011) Licenciamento da Carcinicultura Marinha Praticada em Áreas de Preservação Permanente no Estado de Sergipe: Diretrizes para a Sustentabilidade Socioambiental. São Cristóvão, Sergipe: Grupo de Estudos sobre Aquiculturae Sustentabilidade, Universidade Federal Sergipe. 96 p
Lima JSG, Rivera EC, Focken U (2012) Energy evaluation of organic and conventional marine shrimp farms in Guaraíra Lagoon, Brazil. J Clean Prod 35:194–202
Lucas JS (2003) The future. In: Lucas JS, Southgate PC (eds) Aquaculture: farming aquatic animals and plants. Blackwell Publishing, London, pp 476–513

Machado LS, Santos IS, Lima JGL (2011) Flow of nitrogen and phosphorus in *Penaeus vannamei* ponds in Sergipe State, Brazil: implications for sustainable production. World aquaculture, Natal, Brasil. Abstracts World Aquaculture Society, p 676

Martin JM, Vernan Y, Guerlorguet O, Pham D (1998) Shrimp rearing: stocking density, growth, impact on sediment, waste output and their relationships studied thought the nitrogen budget in rearing ponds. Aquaculture 164:135–149

Martínez-Córdova LR, CampañaTorres A, Porchas-Cornejo MA (2002) Promotion and contribution of biota in low water exchange ponds farming blue shrimp, *Litopenaeus stylirotris* (Stimpson). Aquacult Res 33:27–32

McCay BM, Acheson JM (eds) (1987) The question of the commons; the culture and ecology of communal resources. University of Arizona Press, Tuscon

Muhlert ACS, Lima JSG, Machado L, Evangelista RA (2013) Indicadores Numéricos como Ferramenta para Avaliaçao da Sustentabilidade Ecológica da Carcinicultura Marinha em Sergipe, Brasil. Interciência 38(8):615–620

Muñoz JMB (2001) The Brazilian national plan for coastal management (PNGC). Coast Manag 29:137–156

Naylor RL, Goldburg RJ, Primavera JH, Kautsky N, Beveridge MCM, Clay J, Folke C, Lubchenco J, Mooney H, Troell M (2000) Effect of aquaculture on world fish supplies. Nature 405:1017–1024

Ostrom E (1990) Governing the commons: the evolution of institutions for collective action. Cambridge University Press, Cambridge

Ostrom E, Dietz T, Dolsak N, Stern PC, Stonich S, Weber EU (eds) (2002) The drama of the commons. Committee on the Human Dimensions of Global Change. National Academy Press, Washington, DC

Ostrom E, Gardner R, Walker J (2004) Rules, games, and common-pool resources. University of Michigan Press, Ann Arbor

Páez-Osuna F, Guerrero-Galvan SR, Ruiz-Fernandes AC, Espi- noza-Angulo RE (1997) Fluxes and mass balances of nutrients in semi-intensive shrimp farm in North-Western Mexico. Mar Pollut Bull 34:290–297

Panitz CMN (1997) Ecological description of the Itacorubi mangroves, Ilha de Santa Catarina, Brazil. In: Kjerive B, de Lacerda LD, Diop EHSD (eds) Mangrove ecosystem studies in Latin America and Africa. 170, SC-97/WS/5. United Nations Educational, Scientific and Cultural Organization, Paris, pp 204–223

Panizza AC, Fournier J (2008) O litoral do Rio Grande do Norte: dinâmica e modelo espacial. Confins, Paris, n. 3, julho-outubro 2008. Available at http://confins.revues.org/document3473.html

Posey DA, Dutfield G (1996) Beyond intellectual property: toward traditional resource rights for indigenous peoples and local communities. International Development Research Centre, Ottawa

Rebelo-Mochel F (1997) Mangroves in São Luis Island, Maranhão, Brazil. In: Kjerive B, de Lacerda LD, Diop EHS (eds) Mangrove ecosystem studies in Latin America and Africa. SC-97/WS/5. United Nations Educational, Scientific and Cultural Organization, Paris, pp 145–170

Redford KH, Padoch C (1992) Conservation of neotropical forests: working from traditional resource use. Columbia University Press, New York

Saint-Paul U, Schneider H (2010) Mangrove dynamics and management in North Brazil. Springer, New York

Santos IS (2009) Caracterização do manejo das carciniculturas de terras altas licenciadas no estado de Sergipe. Universidade Federal de Sergipe, Monografia, 53 p

Seraval TA (2010) International Trends in Ocean and Coastal Management in Brazil. MS thesis, Departamento de Ambiente e Ordenamento, Universidade de Aveiro, Portugal. Available at http://ria.ua.pt/bitstream/10773/3492/1/236978.pdf. Accessed 9 Nov 2013

Seraval TA, Alvest FL (2011) International trends in ocean and coastal management in Brazil. J Coast Res Special Issue 64:1258–1262

West PC, Brechin S (eds) (1991) Resident populations and protected natural areas in developing nations: interdisciplinary perspectives and policy implications. University of Arizona Press, Tucson

White A (1984) Effects of protective management on coral reefs in the Philippines. ICLARM Newsl 7(4):9–11

Wilkie DS, Morelli GA, Demmer J, Starkey M, Telfer P, Steil M (2006) Parks and people: assessing the human welfare effects of establishing protected areas for biodiversity conservation. Conserv Biol 20(1):247–249

Wolford W (2001) Grassroots-initiated land reform in Brazil: the rural landless workers' movement. In: de Janvry A, Gordillo G, Platteau JP, Sadoulet E (eds) Access to land, rural poverty, and public action. Oxford University Press, New York

Chapter 10
Regional Environmental Assessment of Marine Aggregate Dredging Effects: The UK Approach

Dafydd Lloyd Jones, Joni Backstrom, and Ian Reach

Abstract In regions where multiple marine dredging activities take place in a spatially restricted area, cumulative effects may impact sensitive receptors, both locally and regionally. In the UK, marine dredging of sand and gravel generally occurs in spatially limited regional blocks, with approximately 20 million tonnes of resource dredged each year from English and Welsh waters.

The UK marine aggregate industry has approached the assessment of cumulative marine dredging activities through a voluntary Marine Aggregate Regional Environmental Assessment (MAREA) approach. Each MAREA assesses the cumulative impacts of marine dredging activities using regional-scale hydrodynamic and sediment transport models linked to regional-scale mapping of sensitive receptors. Each MAREA characterises the physical, biological, and human context of a region, and eases the regulatory burden for site-specific Environmental Impact Assessment. This chapter will describe the MAREA methodology, and proposes that similar regional assessment exercises could contextualise the effects and impacts of multiple marine dredging activities in other parts of the world.

10.1 Introduction

Over the last 50 years marine aggregate dredging in the United Kingdom (UK) has been an important contributor to the supply of essential minerals, with approximately 20 million tonnes of sand and gravel being mined by dredging each year

D. Lloyd Jones (✉) • I. Reach
MarineSpace Ltd, Ocean Village Innovation Centre,
Ocean Way, Southampton SO14 3JZ, UK
e-mail: dafydd.lloydjones@marinespace.co.uk

J. Backstrom
Fugro EMU Ltd, 1 Mill Court, The Sawmills, Durley, Southampton SO32 2EJ, UK

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8,
DOI 10.1007/978-3-319-06305-8_10,

from English and Welsh territorial waters. These minerals are widely used in concrete for construction and also “as dredged” for beach replenishment and for the protection of coasts from erosion and flooding. UK Government policy recognises the fact that marine dredging plays a key role in both servicing the needs of society and in supplying materials for coastal maintenance and flood defence (HM Government 2011). Maintaining continuity of marine mineral supply is, therefore, vital, especially given the stress (in the UK) on constrained and declining land-based mineral resources.

In the UK, marine dredging is an activity which occurs within individually licensed areas, contained within regional blocks, off the coastlines of the Humber Estuary, East Anglia, the outer Thames Estuary, the east English Channel, the South Coast and in the Bristol Channel. For uses in concrete products the ideal aggregate blend consists of approximately 55 % gravel and 45 % sand. Naturally occurring resources rarely occur in exactly this ratio; therefore blending of cargoes from different licence areas is usually required to provide the necessary aggregate quality. Marine aggregates are usually delivered to wharves that are close to construction markets and are commonly integrated with existing ready-mix concrete and concrete block plants.

Dredging in England and Wales requires a marine licence (environmental consent) to be issued by Government regulators, and a commercial agreement with The Crown Estate before dredging is allowed to begin. The Crown Estate (governed by a UK Act of Parliament) acts as the property manager for Her Majesty the Queen and works supportively with Government and at a local level regarding leasing the UK Continental Shelf (UKCS) to allow business development.

The statutory regulations responsible for the control of marine aggregate dredging are administered by the Marine Management Organisation (MMO) on behalf of the Secretary of State in the English inshore region (within 12 nautical miles of the coast), and by Natural Resources Wales/Cyfoeth Naturiol Cymru (NRW) in the Welsh inshore region (within 12 nautical miles). All English, Welsh and Northern Ireland offshore regions (greater than 12 nautical miles offshore) are administered by the MMO. The MMO is an executive non-departmental public body responsible for most activities licensed within the English marine environment while NRW is a Welsh Government sponsored body, charged with the management of the natural resources of Wales, and has regulatory powers for licensing marine activities in Welsh territorial waters.

A licence application (for permission to extract marine minerals) will trigger the requirements of the European Council’s Environmental Impact Assessment (EIA) Directive (85/337/EEC as codified 2011/92/EU). This is transposed into domestic legislation in England and Wales through the Marine Works (Environmental Impact Assessment) Regulations (as amended 2011). Where interaction with European marine sites (Marine Protected Areas (MPAs)) designated under the EU Birds and Habitats Directives (2009/147/EC and 92/43/EEC) may occur, The Conservation of Habitats and Species Regulations 2010 (or The Offshore Marine Conservation (Natural Habitats, &c.) (Amendment) Regulations 2012 for waters >12 nm offshore) may also be considered applicable. Therefore, given the regulatory structure within the UK, the environmental acceptability of all UK marine dredging operations is rigorously tested by means of an EIA and also Appropriate Assessment where required. An Appropriate Assessment is a recognised step-by-step process which helps determine the likely significant effects and assesses potential adverse impacts on the integrity of a European marine site, including an examination of alternative solutions.

Since the 1980s a good understanding has been developed of the local, site-specific, impacts of marine aggregate dredging. It has also been recognised that where multiple activities occur within a spatially limited block, there is the potential for effects to occur both cumulatively (i.e. effects arising from multiple dredging activities within the block), and in-combination (effects that occur as a result of all industrial sectors which operate within the block). Until recently, awareness and consideration of these regional-scale effects has not been as robust as those regarding site-specific considerations; and Regulators have increasingly required developers to assess the significance of their activities on a regional scale.

In order to address this, and to assist in the licensing of new dredging areas, and the re-licensing of areas due to expire, assessments of the regional cumulative and in-combination effects of marine aggregate dredging have been carried out in a series of Marine Aggregate Regional Environmental Assessments (MAREAs) for the Humber (ERM Ltd 2012); East Anglia (EMU Ltd 2012b); the Thames Estuary (ERM Ltd 2010); and the South Coast (EMU Ltd 2012a) (Fig. 10.1). These built on a regionally-based assessment approach adopted during licensing of dredging in the east English Channel (Posford Haskoning 2002). No MAREA has been conducted for the Bristol Channel or the Irish Sea, as the licence areas are more dispersed than in the other regional blocks.

There was no regulatory requirement to undertake the MAREAs, and the UK marine aggregate dredging industry led the initiative as a voluntary commitment to better inform regulators and stakeholders and to streamline the regulatory process.

10.2 The MAREA Process

10.2.1 Defining the MAREA Approach

Whilst the MAREAs were undertaken as a voluntary approach by the UK aggregate industry, the assessment methodology was not developed by the industry in isolation, but was based on collaborative process between the industry and a Regulatory Advisory Group (RAG). The RAG was made up of representatives of statutory and technical advisory bodies to the MMO; English Heritage, Natural England, the Joint Nature Conservation Committee and the Centre for Environment, Fisheries and Aquaculture Science, and preliminary funding was provided by The Crown Estate and the marine aggregate industry.

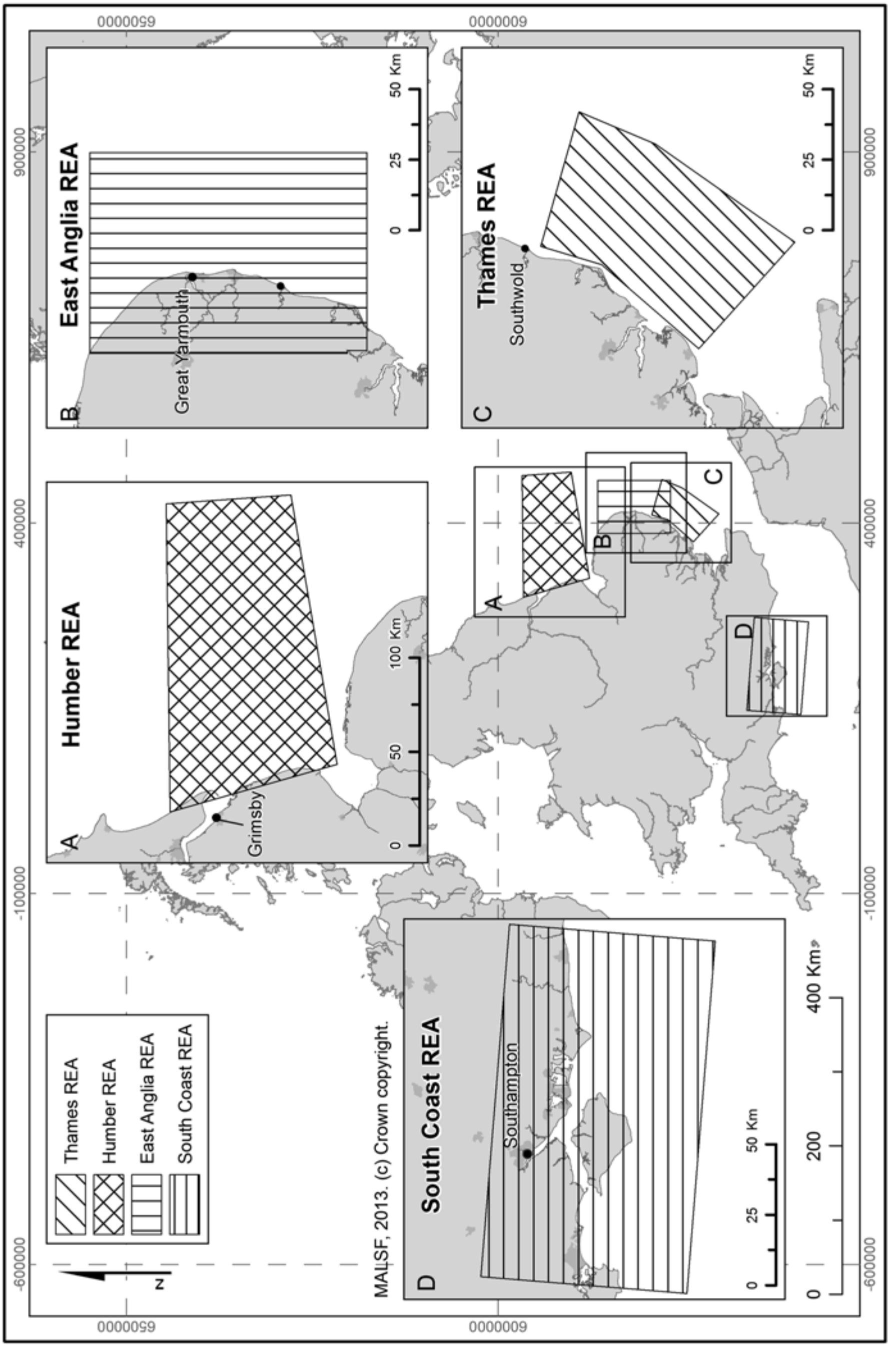


Fig. 10.1 Location of the marine aggregate regional environmental assessment regions around the UK coastline (Lloyd Jones et al. 2013)

The challenge for Regulators is to ensure that marine activity at a regional scale does not unacceptably affect the natural environment. The RAG, therefore, guided the development of the MAREA rationale to focus on regional-scale issues relating to the marine historic environment, nature conservation, fish and fisheries, ornithology, benthic ecology, and physical processes and sediment transport (RAG 2008). The RAG guidance also formulated the fundamental questions to be answered by the MAREAs – are current levels of marine dredging environmentally acceptable; if so, could they be increased without causing significant impacts; and should existing marine dredging continue and new areas for dredging be licensed?

EMU Ltd (2012a) summarised the five broad aims of the MAREA process as:

- To provide objective, evidence-based assessments of the distribution and importance of regional resources (living and non-living) and the potential impacts from the proposed activities on these resources at a regional level;
- To provide a context for site-specific EIAs within the relevant MAREA region and to identify site-specific issues that individual EIAs may need to focus more specifically;
- To provide a robust assessment of cumulative and in-combination impacts at the regional level using consistent definitions and interpretations of such impacts, and thus contribute towards assessments of magnitude and scale of such impacts in individual EIAs;
- To provide an assessment of future development planned by the aggregate extraction industry, based on using projections, and taking account of other human activities and natural variability; and
- To make recommendations for a regional monitoring programme to be addressed at the MAREA level.

The objective of the process was, therefore, to develop a suite of data and assessments that could be utilised to establish environmental baselines, and enable development and licensing of future activities within a region. The MAREA approach can also, therefore, be applied to any region where marine activities occur close to one another, resulting in possible cumulative and in-combination impacts.

10.2.2 The MAREA Methodology

The MAREA methodology was designed to identify and assess potential regional cumulative and in-combination impacts of future marine dredging activities within four regions (see Fig. 10.1). The MAREA definitions of cumulative and in-combination impacts are as follows:

- Cumulative: Impacts that arise from multiple marine aggregate dredging activities within a region and/or sub-region; and
- In-combination: The total impacts of all industrial sectors operating within the same region in the context of natural variability or trends.

There are currently no recognised standards for conducting such assessments, despite the legislative requirements to promote cumulative and in-combination assessments. A number of studies were reviewed as part of the process (e.g. Lane and Wallace 1988; US Council on Environmental Quality 1997; Hegmann et al. 1999; Hyder 1999; Cizek 2007). A key finding of the review, and a potential weakness of many previous assessments, was their strong reliance on professional judgement. The MAREAs, therefore, were designed to adopt a systemic and auditable evidence-based approach as far as possible, although professional judgements are still required where little quantitative data exist.

10.2.2.1 Scoping, Baseline Data Collection and Source-Effect-Receptor Conceptualisation

Each MAREA began with a scoping process, which was designed to identify existing regional information. Where data gaps existed, additional baseline surveys and specialist desk-based studies were defined and commissioned. Additional studies carried out included:

- Additional geophysical surveying across currently licensed and proposed future marine dredging areas. These surveys included sub-bottom seismic, sidescan sonar and multibeam bathymetry and were designed to assist in the characterisation of shallow geology, seabed sediments and palaeo-landscapes; and provided further information on regional seabed sediment transport;
- A detailed coastal characterisation, including data on coastal defence, geology, morphology and physical processes; nearshore bathymetry and sediment distribution; present day hydrodynamics and sediment transport;
- A desk-based navigation study, assessing shipping densities and activity within the region; and
- Archaeological studies, characterising the known and potential wreck resource within the region, and assessing the palaeo-landscape and prehistoric archaeological potential.

Following the additional data gathering, a suite of integrated data sets was therefore compiled for each MAREA region, with the included topic areas summarised in Table 10.1.

In order to understand the relationship (and pathways) between potentially sensitive receptors and the effects generated by the marine dredging process, all relevant physical effects, and the receptors considered at risk from predicted future extraction activities were identified. For the purposes of the MAREAs, the physical effects of dredging that cause changes to the environment were identified as: seabed removal; vessel displacement; noise and vibration; suspended sediments; fine sand dispersion; bathymetry; waves; tides; and sediment flux (a proxy for sediment erosion and accretion). The potentially sensitive receptor groups identified for assessment were the coastline and inshore banks; benthic ecology; fish and shellfish ecology; marine mammals and turtles; ornithology; nature conservation; commercial

Table 10.1 Data sets compiled for each MAREA region following scoping and baseline data collection

Physical environment	Biological environment	Human environment
Coastal characterisation	Benthic habitats	Commercial fisheries
Coastal defences	Benthic communities	Recreational fisheries
Shallow geology	Protected habitats	Marine infrastructure
Bathymetry/topography	Protected species	Marine recreation
Seabed sediment	Commercial species	Shipping
Sediment transport	Fish ecology	Navigation
Temperature/salinity	Ornithology	Archaeology
Water quality	Chelonia (turtles)	Cultural heritage
Sediment contamination	Pinnipeds (seals)	Designated heritage sites
Underwater noise	Cetaceans (porpoises, dolphins and whales)	
	Designated sites	

and recreational fisheries; navigation; infrastructure and other marine users (e.g. offshore renewables; and oil and gas); and archaeology (ship and aircraft wrecks; and prehistoric landscapes).

10.2.2.2 Modelling

An important stage in each MAREA was the undertaking of a detailed numerical modelling exercise for each MAREA region. The aim of the modelling was to calculate the scale of cumulative effects of proposed future dredging activities; to differentiate these from naturally-induced changes and to assess the significance of the effects over time.

Modelling for the South Coast, Thames Estuary and East Anglian regions was undertaken by HR Wallingford (2010a, b, 2011a), while ABPmer (2012) carried out the modelling for the Humber region. The tidal modelling for the South Coast, Thames and Anglian regions was carried out using TELEMAC, with waves modelled using SWAN (HR Wallingford 2010a, b, 2011a). MIKE21 was used to model tidal flows in the Humber region, with MIKE SW used for waves (ABPmer 2012). Each model used a flexible mesh spacing, meaning that the model resolution could be increased (smallest elements) over the existing and proposed marine dredging areas, with lower resolution (largest elements) further offshore (Fig. 10.2). The flexible model mesh allowed the seabed in and around the marine dredging licence areas to be well reproduced, while reducing the amount of bathymetric detail required further offshore.

As the modelling studies were designed to investigate how past dredging, as well as proposed future dredging, might alter the hydrodynamics and associated sediment transport in a region, representations of seabed levels in each region were produced which differed from each other only in those areas where dredging had

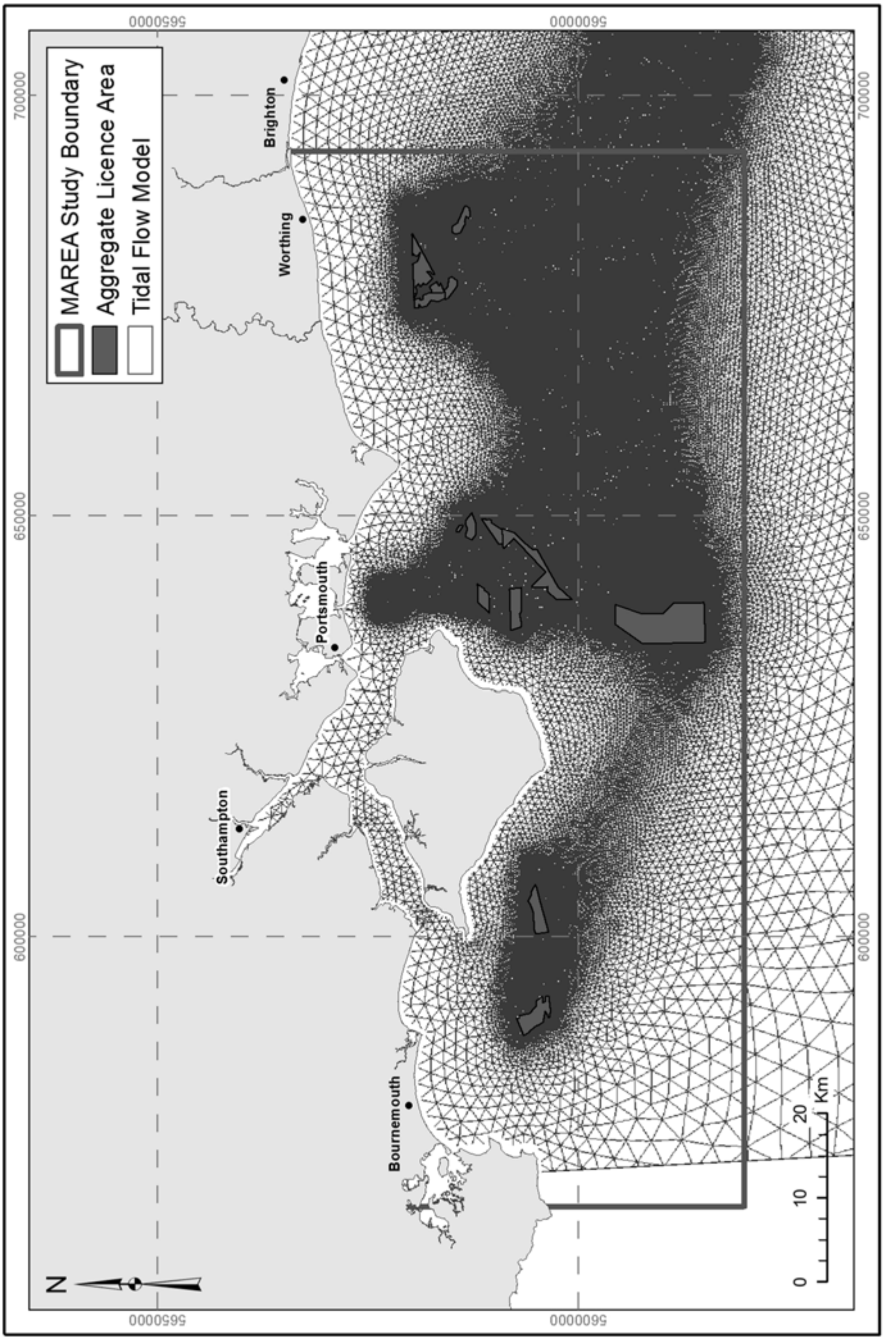


Fig. 10.2 Example of a flexible mesh used in tidal modelling for the South Coast MAREA region, UK (After HR Wallingford 2010a)

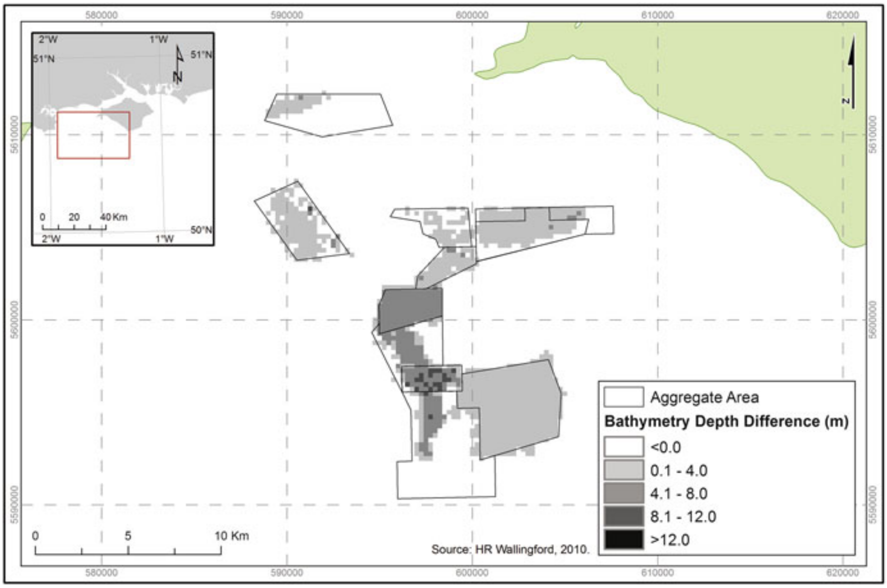


Fig. 10.3 Depth differences between the present day seabed and post-dredging seabed, for the maximum extraction scenario, west of the Isle of Wight, South Coast, UK (After HR Wallingford 2010a)

occurred in the past, or was proposed to take place in the future. Three different bathymetries were, therefore, produced in each modelling exercise:

- A baseline or 'pre-dredged' bathymetry, in which the seabed levels in each existing or relinquished dredging area were restored to the levels of the seabed surrounding each area, in order to represent the situation before any dredging had occurred;
- A 'present day' bathymetry in which seabed levels in each existing or relinquished dredging area were established using the most recent survey data available from those areas. The differences in bed levels between this and the 'pre-dredged' bathymetry were the result of past dredging in existing or relinquished licence areas; and
- A 'post-dredging' bathymetry in which future seabed levels in each existing and proposed new dredging area were defined by combining the present-day bed levels and the present plans of dredging companies for extraction to the year 2030 (approximately). The post-dredging bathymetries therefore represented the worst-case bathymetric change predicted to occur over the lifetime of the proposed marine dredging activity.

Figure 10.3 shows an example of the depth changes between 'present day' and 'post dredging' for marine dredging licence areas west of the Isle of Wight, South Coast, UK. The models took a precautionary approach to scales of effects, and hence dealt with regional maximum extraction scenarios. This maximum extraction scenario assumed that all licence renewals, applications and prospecting licences within each MAREA region would be permitted and dredged concurrently, and therefore the modelled changes in hydrodynamics and sediment transport represented a worst case situation. In reality, since extraction plans include an element of 'double-counting' arising from competition for market shares between dredging companies, to realise all the bathymetric changes shown in Fig. 10.3 by 2030 would actually require a much greater intensity of aggregate dredging than is expected to occur.

Any changes in hydrodynamics resulting from future dredging could potentially alter the local and regional sediment transport rates and pathways. The outputs from the hydrodynamic models described above were, therefore, coupled to sediment transport models in order to assess the scale and intensity of potential sediment transport changes. Sediment transport in the Thames, South Coast and East Anglian regions was simulated using the HR Wallingford SANDFLOW model, which is a non-equilibrium finite element sediment transport model that simulates the total load (suspended and bedload), with input flows from TELEMAC (HR Wallingford 2010a, b, 2011a). A different approach was employed for the Humber, where a desk-based empirical approach was used. This involved calculating the bed shear stress under currents and waves at key locations within the study area and presenting this alongside calculated theoretical sediment mobility thresholds for the different size fractions within the grain size distribution at each site. This enabled mobilisation events to be identified and demonstrated the extent to which changes in waves or tidal currents could affect sediment mobility and hence the potential for sediment transport. Both techniques provided spatial 'envelopes' of changes to sediment flux within the model boundaries.

The model outputs were validated and calibrated against tidal current and wave data collected from within the modelled regions, and also (for the Thames, Anglian and Humber MAREAs) data from the Southern North Sea Sediment Transport Study 2 (HR Wallingford et al. 2002). The outputs of the modelling exercise provided, therefore, a series of 'envelopes' of maximum extraction dredging effects, which could be taken forward to the MAREA assessment process (see Sect. 10.2.2.3).

The use of worst case scenarios extended throughout the MAREA reports, enabling the Regulator and its advisors (the RAG) to use the results and determinations within an appropriate precautionary envelope. This proved particularly useful where conclusions were considered in the context of potential interactions with designated heritage or nature conservation features, or other sensitive receptors such as coastlines.

10.2.2.3 MAREA Assessment Methodology

Each MAREA assesses the cumulative and in-combination effects of dredging using a multi-stage process that is common across all four regions:

1. Identification of potentially sensitive receptors – for each region, the outputs of the specialist desk studies and the new baseline datasets are combined and analysed in order to characterise and identify the physical, human and biological

receptors potentially sensitive to the future effects of dredging (e.g. benthic and fish ecology, birds, marine mammals, fisheries, archaeology etc.). This is done on an individual MAREA basis, as potentially sensitive receptors are not necessarily common across all regions.

2. Conceptualisation of effects – often the terms 'effect' and 'impact' are used interchangeably, however for the MAREAs they are separately defined. An effect is defined as a physical change in the environment that occurs as a consequence of the dredging process. The effects that have the potential to impact the sensitive receptors present in a region are identified and include, but are not limited to, alterations to the local hydrodynamics and the associated changes to sediment transport, deposition and plume effects; underwater noise; vessel presence and loss of access to areas; changes to benthic communities; changes to distribution of fish; changes to seabed features. Impacts are defined as the possible changes in potentially sensitive receptors as a result of an effect, and may be positive or negative (ERM Ltd 2010; EMU Ltd 2012a).
3. Prediction of effect magnitude – the magnitude of each predicted dredging effect is assessed based upon three variables, namely the extent, duration, and frequency of the effect; and by comparing the predicted changes with the baseline conditions. This is a critical step and allows the extent of the effect to be quantified according to the spatial area affected, and allows inferences to be made on potential changes to receptors. Definitions for each of variables were agreed by the Industry and the RAG, before the commencement of the MAREAs.

The extent describes the spatial scale of the effect, and is important in distinguishing between near-field and far-field effects. The extent can be defined as:

- Site-specific – this is an effect that occurs within the licence area where dredging is occurring or predicted to occur in future. This classification includes effects associated with the movement of the draghead across the seabed;
- Local – an effect that extends beyond the immediate footprint of dredging but does not affect receptors at the regional scale. Examples of effects acting at the local scale typically include alteration to tidal currents, noise propagation etc.; and
- Regional – an effect that covers much, or all, of a MAREA region, but does not extend outside it. Examples may include changes to populations of mobile marine species, or to regional sediment transport pathways.

The duration of an effect is a measure of its temporal persistence, and the levels defined are:

- Temporary – an effect that only occurs during dredging, or for a few hours or days after dredging stops;
- Short-term – an effect that is no longer visible after up to 1 year following cessation of dredging;
- Medium-term – an effect that lasts between 1 and 10 years following cessation of dredging; and
- Long term – an effect that lasts for more than 10 years after dredging.

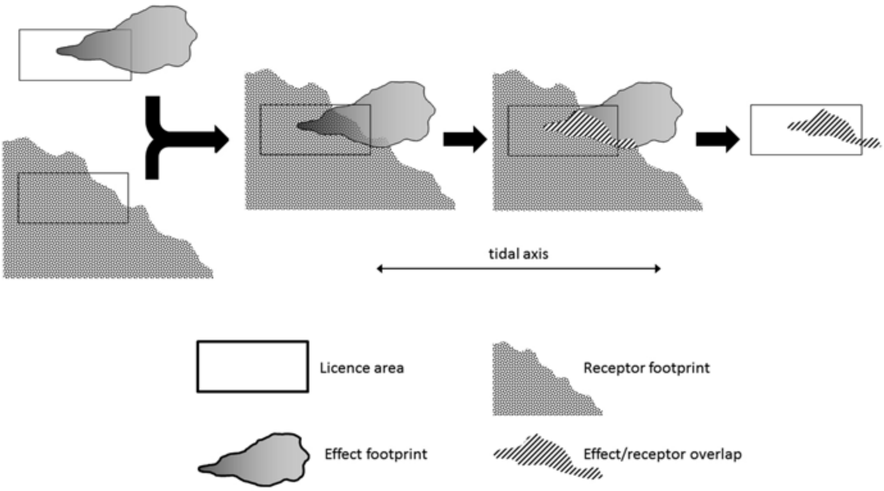


Fig. 10.4 Mapping overlap between effect and receptor to determine likely extent of potential impacts (After EMU Ltd 2012a)

The frequency of an effect is a measure of how often it occurs, and the agreed frequency definitions are:

- Routine – occurs during all normal dredging activities (95–100 %);
- Intermittent – occurs often, but not during all dredging activity (25–95 %);
- Occasional – only occurs during a small proportion of operations (<25 %); and
- Rarely – effect only occurs as an unplanned event e.g. oil spills.

The elevation above baseline is an estimation of how far an effect changes from its pre-existing condition, as a result of the dredging. The MAREA process recognises that assigning categories to these variables can be subjective, however since magnitudes are largely based on quantifiable data outputs from the modelling (described in Sect. 10.2.2.2) uncertainties can be reduced in this part of the methodology (EMU Ltd 2012b).

4. Identifying the degree of interaction between effect and receptor – the purpose of this step is to identify whether a potentially sensitive receptor occurs within the footprint of an effect, i.e. to answer the question, ‘Is there an exposure pathway present?’ It is a GIS-based mapping approach; using the outputs of the regional-scale numerical models (the maximum scenario effect envelopes) overlaid with the locations of potentially sensitive receptor groups. An impact is only predicted to occur where there is an Effect-Receptor overlap (pathway) (Fig. 10.4). If there is no Effect-Receptor overlap then that receptor can be screened out of further impact assessment.

ERM Ltd (2012) points out that a certain degree of expert judgement is also required when evaluating effect receptor overlaps, since the magnitude of effects may vary within the effect footprint, with higher intensities experienced towards the source of the effect (e.g. deposition and plume effects). In addition, while some receptor footprints may represent fixed locations (e.g. wreck sites, dredge disposal areas), others may represent mobile receptors (e.g. fishing activity, foraging seals).

5. Determination of receptor value and sensitivity – following the prediction of effect magnitude, an assessment of the value of the receptor is carried out. This considers whether a receptor is rare, protected, threatened or (in the case of biological receptors) whether it provides an important ecosystem service (e.g. whether it is a keystone species, or a habitat architect species) (ERM Ltd 2012). The sensitivity of a receptor is assessed based upon its tolerance, adaptability and recoverability to a particular effect, with all definitions again being agreed in advance of the MAREAs by the industry and the RAG. Assigning sensitivity categories is based on a combination of quantifiable data and published indices (if available), stakeholder consultation, and expert judgement.

The tolerance of a receptor is defined as the extent to which a receptor (at a regional scale) is adversely affected by a particular dredging effect; i.e. the receptor's ability to be either unaffected or affected, temporarily and/or permanently. The agreed tolerance categories are:

- Low – the receptor is unable to tolerate the effect, leading to a permanent change in abundance or quality;
- Medium – the receptor has some ability to tolerate the effect, but there will be a detectable change at the regional scale e.g. a change in distribution; and
- High – the receptor is unaffected, or positively affected, by the effect.

The adaptability of a receptor is a measure of its ability to avoid or adapt to/accommodate an effect. The adaptability definitions are:

- Low – the receptor is unable to avoid the effect;
- Medium – the receptor has some ability to tolerate the impacts, or can partially adapt; and
- High – the receptor can completely avoid or adapt, with no changes detectable at the regional scale.

The recoverability of a receptor is a measure of its ability to return to a state similar its state before any changes occurred. The agreed recoverability categories are:

- Low – the receptor is unable to recover, resulting in permanent or long term changes (>10 years);
- Medium – the receptor recovers to an acceptable status over the short to medium term (1–10 years); and
- High – Receptor recovers fully within 1 year.

EMU Ltd (2012a, b) describes Sensitivity as the benchmark against which changes and levels of exposure can be compared to evaluate significance. Sensitivity can be expressed quantitatively where data are available. When sensitivity is less well understood the assessment is based on scientific literature and professional judgement (EMU Ltd 2012a, b).

Degree of Change (Sensitivity)	Degree of Change (Magnitude): Very Low	Low	Medium	High
High	Minor Significance	Moderate Significance	High Significance	High Significance
Medium	Not Significant	Minor Significance	Moderate Significance	High Significance
Low	Not Significant	Not Significant	Minor Significance	Moderate Significance

Fig. 10.5 Impact assessment matrix, combining magnitude of effect and sensitivity of receptor, used in the South Coast and East Anglia MAREAs (EMU Ltd 2012a, b)

6. Evaluating impact significance – the significance of an impact depends on there being an overlap between an effect and a receptor, the magnitude of the effect, and the value and sensitivity of the receptor (Fig. 10.4), however assigning cumulative impact significance is the stage of the process that involves the greatest subjectivity. The assessment methodologies for all MAREAs standardise the assessment of impact significance by using defined significance criteria, applied across all effects and receptors in a systematic, consistent and auditable way, and these metrics were agreed in advance between the RAG and industry. Figure 10.5 shows an example of the relationship between the magnitude of effect and the sensitivity of the receptor used to assess impact significance in the South Coast and East Anglian MAREAs (EMU Ltd 2012a, b).

   The overall impact significance criteria for the MAREA process are defined as:

- Not significant – impacts that after assessment are not significant in the context of the MAREA objectives;
- Minor significance – impacts that warrant the attention of particular stakeholders but no action is required if impacts can be controlled by adopting normal good working practices;
- Moderate significance – regional impacts that should be recognised and addressed in consultation with particular stakeholders; and
- Major significance – regional impacts that are not environmentally sustainable and compromise the continuation of extraction activity in the region.

7. Cumulative and in-combination assessment – the final stages of the MAREA process determine the cumulative and in-combination impacts of the regional dredging where cumulative impacts are impacts arising from multiple dredging

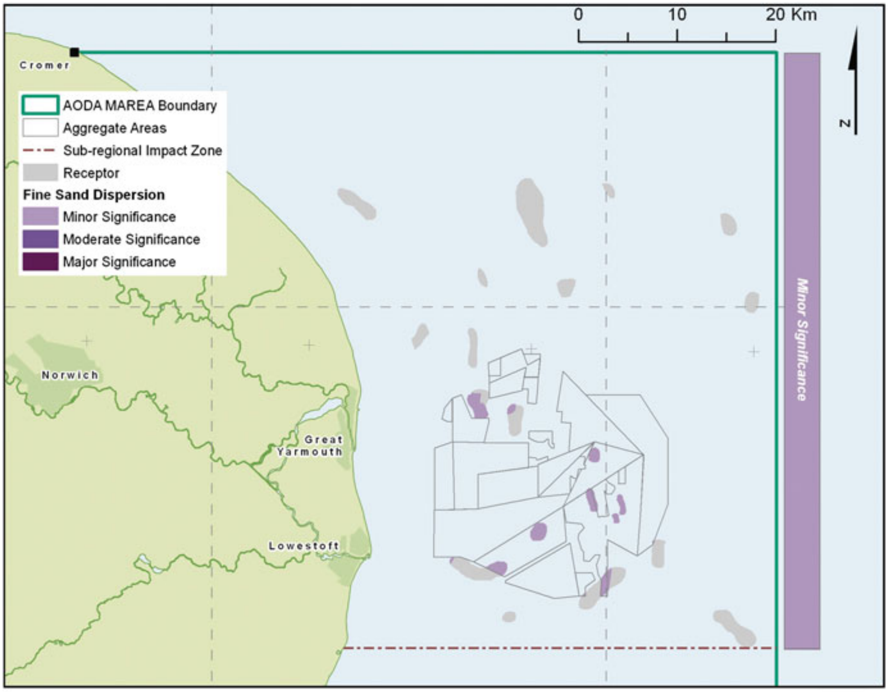


Fig. 10.6 Mapped impact significance of suspended sediment effects on sensitive benthic receptors in the northern component of the East Anglian MAREA region, UK (EMU Ltd 2012b)

operations, and in-combination impacts are from all industrial sectors operating within the same region. Figure 10.6 shows an example from the Anglian Offshore MAREA (EMU 2012b), illustrating the assessed cumulative impact significance of suspended sediment on sensitive benthic receptors off the East Anglian coast.

Multiple impact footprints of all potential impacts acting on a single receptor are also considered within the MAREAs. ERM Ltd (2012) indicates that this is not simply a case of summing impact layers, as some impacts may act together to increase the level of impact while others may act antagonistically to reduce the overall level of impact. Therefore, while impact layers can be collectively assessed by overlaying multiple effects in a GIS to produce a 'heat map' an interpretation of what this means for the receptor must be descriptive rather than quantitative (ERM Ltd 2012). An example heat map for the East Anglian region is shown in Fig. 10.7, and is a graphical representation of data mapped as a colour gradient, where the greatest overlap of variables is represented by a more intense colour.

The outputs from the cumulative assessments are taken forward to an in-combination assessment where interaction tables summarise which dredging effects are also produced by other industries and activities in the region. The in-combination assessment takes a spatial approach and only considers activities that are currently

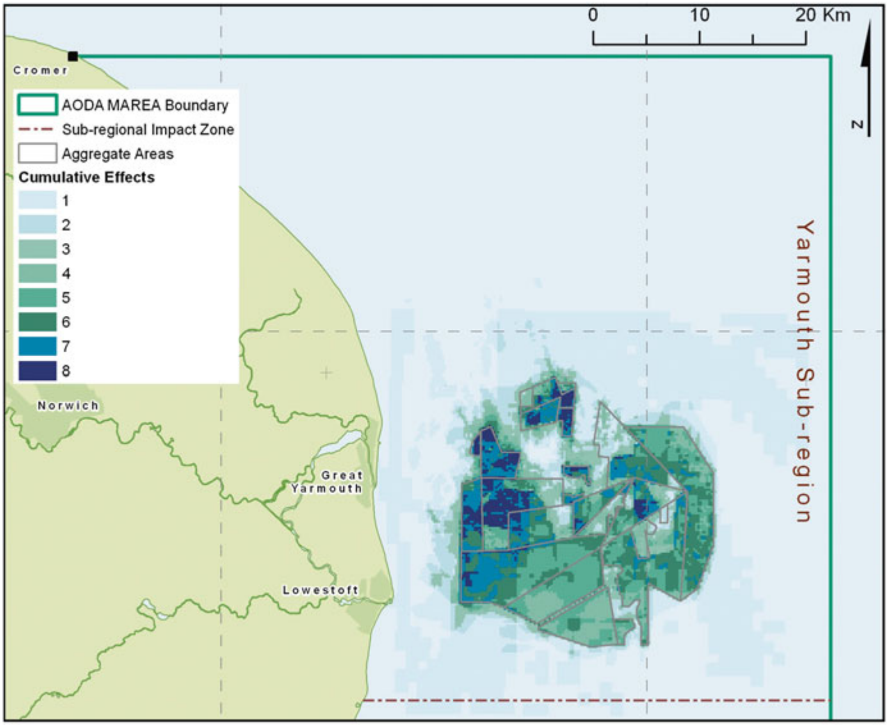


Fig. 10.7 Multiple effects of dredging in the northern component of the East Anglian MAREA region, UK. Note: the majority of overlapping effects are concentrated within the aggregate areas (EMU Ltd 2012b)

operating or which are reasonably foreseeable (i.e. projects in the planning process). Those dredging areas where in-combination effects are most likely to occur are highlighted by the MAREAs so that greater attention can be paid to these issues within the site-specific EIA.

10.3 Easing Consenting Using the MAREAs

10.3.1 Identification of Strategic Issues Using the MAREAs

Ahead of a large-scale marine aggregate licence renewal programme in English territorial waters, the consistency of application advice provided to marine aggregate operators by the Regulator and its advisors has been assessed by reviewing responses to the four recently conducted MAREAs (Humber, Anglian, Outer Thames and South Coast) (MarineSpace Ltd 2013a, b, c, d).

The reviews, which were commissioned by the marine aggregates industry, have compared statutory advice provided on licence applications already in the application process, validated against the assessments of the relevant MAREA. The reviews demonstrate that the MAREAs can be utilised as a quality control mechanism, bench-marking EIA scoping exercises and assisting streamlined statutory advice in practice. Regional-scale cumulative and/or in-combination effects are being referenced within individual EIAs and effort is being efficiently expended on assessing the licence-specific issues within an EIA.

An additional benefit from the delivery of a UK national MAREA programme is the identification of consistent regional-scale issues between MAREAs. These have been collated and used to establish UK national-level strategic topics, and to determine if opportunities exist for collaborative, strategic solutions to address these at various scales within the regulatory process. A programme of national strategic assessment protocols and methodologies to address over-arching themes (beyond the remit of any single MAREA) has been implemented which addresses themes such as consistent EIA and AA of disturbance on sensitive designated populations of seabird species e.g. Red-throated Diver *Gavia stellata* (Reach et al. 2013); and the mapping and assessment of Atlantic Herring *Clupea harengus* and sandeel potential spawning habitat (MarineSpace Ltd et al. 2013a, b).

Alongside a characterisation of the regional-scale cumulative and in-combination impact issues, the MAREAs also highlight, where possible, any licence-specific effects that are likely to require specific attention within an individual EIA. These effects are local-scale in nature, and are typically associated with discrete features that have a limited spatial or temporal distribution within the wider region. It is therefore appropriate to address these at a licence-specific scale.

Additionally the MAREAs allow quality control at a licence-specific scale, facilitating consistency for any particular marine aggregate company's portfolio of licence applications within any one region. An Applicant can check that all of their applications are receiving consistent statutory advice and consideration and flag any inconsistencies with the Regulator.

10.3.2 Reduction in Site-Specific EIA Burden

To further illustrate how the MAREAs can reduce the burden of site-specific EIAs the example of Coastal Impact Studies (CISs) in the UK can be examined. Under the current system it has previously been a requirement for Applicants to conduct site-specific CISs to demonstrate that dredging at the Application Area will not significantly impact coastal processes. CISs have typically comprised a coastal characterisation; wave modelling evaluating how the proposed lowering of the seabed will alter waves at the coastline; tidal flow modelling simulating and identifying the effects of aggregate dredging on tidal currents; and a sediment transport assessment evaluating the influence of dredging on local sediment transport, and the potential consequences for sediment transport pathways in the region.

In regions containing multiple dredging areas, the result of this requirement has been repeated site-specific modelling exercises, despite the dredging effects having a regional scale. In addition site-specific modelling gives, arguably, incomplete results as it does not take into account the effects on hydrodynamics (and hence sediment transport), of other dredging-related bathymetric change in the region. The MAREAs have therefore undertaken, through a single regional study process, all of the components previously assumed as part of a site-specific CIS. Each regional coastline has been characterised to highlight local receptor sensitivities, and to understand the processes occurring at each part of the coast. Furthermore, historical reviews incorporating data from the ongoing regional coastal monitoring programmes have been undertaken for some regions, to provide a context to the coastline changes that are currently observed.

A tiered approach has been accepted by the Regulators and the RAG for the use of the MAREA outputs in lieu of a site-specific CIS. In the majority of cases, where future dredging will result in seabed lowering that lies within the thresholds assessed through the MAREA maximum offtake modelled bathymetry, then the MAREA findings may be used as the CIS and no further site-specific CIS will be required for the individual licence application; removing the requirement to produce such a study within the EIA.

In order to ensure that the dredging effects remain within the limits assessed by the MAREA, data from each bathymetric monitoring survey will be compared with the modelled 'post-dredging bathymetry' to show that the seabed remains within the modelled thresholds. There may be instances where, because of newly acquired data or amended resource assessments, dredging may be proposed beyond the thresholds assessed in the MAREA. In these cases further hydrodynamic and sediment transport modelling may be required to assess the additional impact of this seabed lowering.

10.4 Conclusions

The MAREAs establish, for the first time, the context of each study region in terms of its physical, biological, and human (heritage and socio-economic) value; and the interaction with future aggregate extraction activities. They deliver a range of regional datasets, modelling outputs and regional cumulative and in-combination assessments which will aid the progression of site-specific licence renewals and applications, reduce the site-specific application costs and Regulatory burden.

The MAREA methodology links the outputs of regional hydrodynamic and sediment transport models with the identification of potentially sensitive receptors from specialist desk-based studies and baseline data collection and analysis. The model outputs and receptor locations are compared through a GIS-based 'effect-receptor overlap' mapping approach. Where there is no overlap of effect with receptor then the receptor can be scoped out of the impact assessment. The impact assessment integrates a prediction of effect magnitude (based upon extent, duration, frequency, and elevation above baseline) with a determination of receptor value and sensitivity (based upon its tolerance, adaptability and recoverability to a particular effect) to evaluate the impact significance. While the MAREA methodology was developed for the UK offshore aggregate dredging industry this Chapter shows that the 'effect-receptor overlap' methodology can be successfully applied to datasets from world-wide regions where multiple dredging operations occur and a similar approach could be adopted by other countries or developers where regional cumulative and in-combination impacts of those developments may occur.

As well as streamlining the Environmental Impact Assessment process for dredging applications, the MAREAs also provide a range of integrated datasets which can assist wider coastal zone and marine management, at a regional seas scale. The integrated datasets provide details on the effects and impacts of multiple dredging operations in the regions; as well as their interaction with other users of the marine environment.

References

ABPmer (2012) Humber MAREA – Physical processes study: assessment of dredging effects

Cizek P (2007) Meta-cumulative effects: mapping cumulative impacts of resource extraction and development throughout the NWT. Canadian Arctic Resources Committee

The Conservation of Habitats and Species Regulations (2010) Statutory instrument 2010/490. Her Majesty's Stationary Office

Department for Communities and Local Government (DCLG) (2006) Mineral policy statement 1. TSO

East Channel Association (2011) The first regional monitoring review for the East Channel Region

ECA, Emu Ltd (2005) Regional monitoring blueprint related to marine aggregate extraction operations in the Eastern English Channel

EMU Ltd (2012a) South coast marine aggregate regional environmental assessment. Report for the South Coast Dredging Association

EMU Ltd (2012b) Anglian Marine aggregate regional environmental assessment. Report for the Anglian Offshore Dredging Association

ERM Ltd (2010) Marine aggregate regional environmental assessment of the outer Thames Estuary

ERM Ltd (2012) Marine aggregate regional environmental assessment of the Humber and Outer Wash Region

European Council Directive 2009/147/EC The Birds Directive

European Council Directive 92/43/EEC Council Directive on the conservation of natural habitats and of wild fauna and flora

European Council Directive 2011/92/EU The assessment of the effects of certain public and private projects on the environment. Codification of EU Directive 85/337/EEC

Hegmann G, Cocklin C, Creasey R, Dupuis S, Kennedy A, Kingsley L, Ross W, Spaling H, Stalker D (1999) Cumulative effects assessment practitioners guide. Prepared by AXYS Environmental Consulting Ltd. and CEA Working Group for the Canadian Environmental Assessment Agency, Hull, Quebec

HR Wallingford (2010a) South Coast Dredging Association MAREA tidal flows and sediment transport study

HR Wallingford (2010b) Thames Estuary Dredging Association MAREA tidal flows and sediment transport study

HR Wallingford (2011a) Anglian Offshore Dredging association marine aggregate regional environmental assessment tidal flows and sediment transport study

HR Wallingford (2011b) Area 473 East: Dredger and plume monitoring study

HR Wallingford, Cefas/UEA, Posford Haskoning, D'Olier B (2002) Southern North Sea sediment transport study Phase 2, sediment transport report, report produced for Great Yarmouth Borough Council, HR Wallingford Report EX4526, August 2002
Hyder (1999) Consulting guidelines for the assessment of indirect and cumulative impacts as well as impact interactions. EC DGX1 Environment, Nuclear Safety and Civil Protection, Brussels
Lane P, Wallace R (1988) Reference guide, feasibility study and overview of institutions interested in cumulative effects assessment: vol. 1. Reference guide to cumulative effects assessment in Canada. Canadian Environmental Research Council, Hull
Lloyd Jones D, Backstrom J, Reach I (2013) The marine aggregate regional environmental assessments – new, integrated, coastal datasets to assist regional management. Paper presented at the 10th ICE coasts, marine structures and breakwaters conference, Edinburgh, 18–20 Sept 2013
MarineSpace Ltd (2013a) A review of national, regional and site-specific issues related to applications for marine licences to extract marine aggregate from the Humber Region
MarineSpace Ltd (2013b) A review of national, regional and site-specific issues related to applications for marine licences to extract marine aggregate from the Anglian Region
MarineSpace Ltd (2013c) A review of national, regional and site-specific issues related to applications for marine licences to extract marine aggregate from the outer Thames Region
MarineSpace Ltd (2013d) A review of national, regional and site-specific issues related to applications for marine licences to extract marine aggregate from the South Coast Region
MarineSpace Ltd, ABPmer Ltd, ERM Ltd, Fugro EMU Ltd, Marine Ecological Surveys Ltd (2013a) Environmental effect pathways between marine aggregate application areas and Atlantic Herring potential spawning habitat: regional cumulative impact assessments. Version 1.0. A report for the British Marine Aggregates Producers Association
MarineSpace Ltd, ABPmer Ltd, ERM Ltd, Fugro EMU Ltd and Marine Ecological Surveys Ltd (2013b) Environmental effect pathways between marine aggregate application areas and Sandeel Habitat: regional cumulative impact assessments. A report for BMAPA
The Offshore Marine Conservation (Natural Habitats, &c.) (Amendment) Regulations 2012. Statutory Instrument 2012/1928. Her Majesty's Stationary Office
Posford Haskoning (2002) Regional environmental assessment
Reach IS, Henson K, Golding TJ, Murphy KJ, Langman RJ, Coates AS, Warner IC, Hatton L, Wright S, Leake S (2013) Marine aggregate licence, renewal and application areas: Anglian and Outer Thames Estuary Region: a report to inform an appropriate assessment (in combination), with regard to disturbance, and any displacement effects, on the classified Red-throated Diver *Gavia stellata* population of the Outer Thames Estuary special protection area – version 1.0. A report produced for British Marine Aggregate Producers Association to inform the Marine Management Organisation
Regulatory Advisory Group (2008) Regional environmental assessment: a framework for the Marine Minerals Sector
US Council on Environmental Quality (1997) Considering cumulative effects under the National Environmental Policy Act. US Executive Office of the President

Part V
Examples of Advances in Environmental Management: Analyses and Applications

Chapter 11
Advances in Large-Scale Mudflat Surveying: The Roebuck Bay and Eighty Mile Beach, Western Australia Examples

Robert J. Hickey, Grant B. Pearson, and Theunis Piersma

Abstract The shores of Roebuck Bay and Eighty Mile Beach in northwestern Australia are amongst the richest known intertidal mudflats worldwide. They are both listed as Wetlands of International Importance under the Ramsar Convention, primarily because of the high numbers of shorebirds that migrate to and from these sites every year. There are only a dozen or so areas in the world with extensive intertidal flats rich in shorebirds.

Shorebird studies by a collaboration between The Department of Environment and Conservation, The University of Western Australia, The Royal Netherlands Institute for Sea Research, Central Washington University, Broome Bird Observatory, and local community volunteers in northwestern Australia have focused on understanding the geological and biological processes of coastal tidal mudflats. Studies have established that invertebrates are abundant and they are used for feeding by resident and migratory shorebirds.

In addition to requiring equipment, software, and considerable organization, these labor intensive studies were only possible with the assistance of large numbers of community volunteers, professionals, and donated equipment.

R.J. Hickey (✉)
Department of Geography, Central Washington University,
400 East University Way, Ellensburg, WA 98926, USA
e-mail: rhickey@cwu.edu

G.B. Pearson
Bennelongia Environmental Consultants, 5 Bishop Street, Jolimont, WA 6014, Australia

T. Piersma
Department of Marine Ecology, NIOZ Royal Netherlands Institute for Sea Research,
P.O. Box 59, 1790 AB Den Burg, Texel, The Netherlands

Animal Ecology Group, Centre for Ecological and Evolutionary Studies, University of Groningen, P.O. Box 11103, 9700 CC Groningen, The Netherlands

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8,
DOI 10.1007/978-3-319-06305-8_11,

11.1 Introduction

Roebuck Bay and Eighty Mile Beach (Fig. 11.1) are world-renowned, non-breeding sites for migratory shorebirds (Rogers et al. 2003). These small to medium-sized birds – sandpipers, plovers, curlews and the like – nest in the far northern hemisphere in habitats ranging from Mongolian steppes to high arctic tundra. In the non-breeding season, they inhabit a very different world – depending on intertidal mudflats where they feed on benthic animals (van de Kam et al. 2004; van de Kam et al. 2010). The rich and diverse benthos of these mudflats supports a very large and diverse shorebird population. In the East-Asian – Australasian Flyway, Eighty Mile Beach is the only site that supports a larger number of shorebirds than Roebuck Bay, while the diversity of benthic or bird species occurring in internationally significant numbers in Roebuck Bay is unparalleled (Piersma et al. 2006). About 150,000 roosting shorebirds use the place. Indeed, there are few places on earth where soft bottom intertidal mudflats support larger numbers of migratory shorebirds. Roebuck Bay is one of less than only 20 comparable coastal areas scattered around the globe. The features that characterize this Bay and make it so outstanding are varied, complex, and have been the subject of considerable scientific and community investigation over the past 10 years (Pepping et al 1999; Piersma et al. 2005), especially during large expeditions to Roebuck Bay (1997, 2002, and 2006) and Eighty Mile Beach (1999).

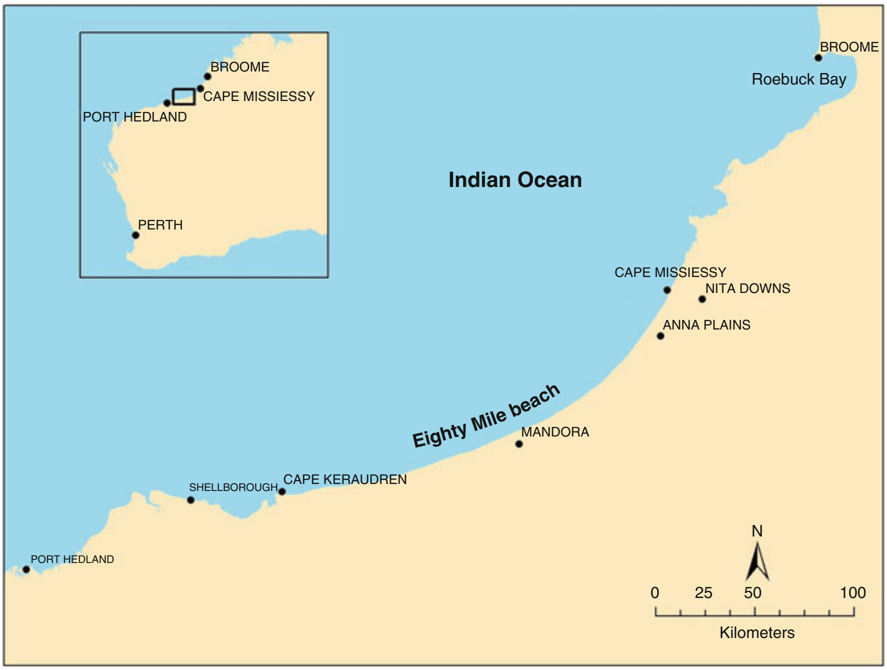


Fig. 11.1 Location of Eighty Mile Beach and Roebuck Bay, Western Australia

This information is essential if we are to conserve the immense and internationally shared natural values of these important shorebird sites and to find informed compromises between the increasing use of the foreshore by the ever increasing human population in the Kimberley Region and their use by the beasts and birds. A considerable proportion of the world's Great Knots (*Calidris tenuirostris*) depends on (very specific portions of) Roebuck Bay for molt, survival, and fuelling for migration. This is also true for perhaps all the Red Knots (*Calidris canutus piersmai*) and Bar-tailed Godwits (*Limosa lapponica menzbieri*) of specific, reproductively isolated and morphologically and behaviorally distinct subspecies. The intertidal macrobenthic community of places like Roebuck Bay contains a unique assemblage of species (De Goeij et al. 2003; Honkoop et al. 2006). Some of these species were new to science.

11.2 Project Description

Recent studies in North-western Australia have focussed on understanding the geological and biological processes of these coastal tidal mudflats. Preliminary research reports for each expedition can be downloaded from http://www.onlinegeographer.com/birds.html.

- Monroeb (*Mon*itoring *Roe*buck *B*ay 1996) was established in 1996 through the Broome Bird Observatory in collaboration with The Royal Netherlands Institute for Sea Research (NIOZ) and Department of Conservation and Land Management (CALM) to begin a long term study of seasonal dynamics of intertidal benthos of a tropical mudflat at Roebuck Bay. The project has sampled four sites at two locations on a monthly basis for 18 years, and is ongoing.
- Roebim-97 (*Roe*buck B*ay* *B*enth*i*c *I*nvertebrate *M*apping Project 1997) was the first large scale mapping of intertidal benthos at Roebuck Bay (and probably worldwide).
- Annabim99 (*Anna* Plains *B*enthic *I*nvertebrate *M*onitoring project 1999) was a follow up on Roebim97 to examine the nature and distribution of the benthic invertebrates at Eighty Mile Beach, south of Broome. This site compares with Roebuck Bay in terms of importance to shorebirds as feeding grounds and high tide refuge.
- Tracking2000 (Radio Telemetry and other studies of Great Knots and Red Knots in Roebuck Bay February to April 2000) was a large scale multi-disciplinary study of the ecology of Red Knots and Great Knots at Roebuck Bay.
- SRoebim02 (*S*outh *Roe*buck Bay *B*enthic *I*nvertebrate *M*apping Project 20*02*) was an extension of Roebim97 that undertook to map the benthos of the whole of Roebuck Bay. This required large numbers of volunteers.
- Roebim06 (*Roe*buck Bay *B*enthic *I*nvertebrate *M*apping Project 20*06*) was an extension of Roebim97 that undertook to repeat map the benthos of the northern shores of Roebuck Bay.

All of these projects incorporated local community volunteers for the assessments of the benthic organism content of the mudflats of Roebuck Bay and Eighty Mile Beach and studies of migratory waders. Local community involvement in ongoing research on the mudflats and migratory waders is now an ever-present and increasingly important aspect of the conservation effort at Roebuck Bay and Eighty Mile Beach.

11.3 Site Description

Northwest Australia is truly unique for a particular biogeographic reason. It is the *only* area in the entire Indo-Pacific faunal region where intertidally foraging molluscivore shorebirds such as Red and Great Knots comprise a dominant part of the avifauna (Rogers et al. 2003). Everywhere else in the Indo-Pacific, the shorebird community consists predominantly of crab-eating species such as Whimbrels (*Numenius phaeopus*), Grey Plovers (*Pluvialis squatarola*), Greater and Lesser Sand Plovers (*Charadrius leschenaultii* and *C. mongolus*), Terek Sandpipers (*Xenus cinereus*), and Grey-tailed Tattler (*Heteroscelus brevipes*) (Rogers et al. 2000, 2001, 2003; Rogers and Taylor 2001; Battley et al. 2012).

A shallow sloping coast, soft sediments, and large tides are all prerequisites for the formation of extensive exposed mudflats such as those of Roebuck Bay and Eighty Mile Beach. For these tidal mudflats also to be rich in shorebirds, one additionally needs a food supply in the form of the abundant invertebrate wildlife living on and in the mud. Not surprisingly, such conditions are seldom fulfilled. Many coastlines where the tides are large are steep-sloped and rocky. Further, many shallow sloping soft sediment shorelines do not have sufficient tidal range (i.e. the embayments in southeast Australia, for example) or face such climatic extremes some times of the year (Fig. 11.2).

Most areas of this intertidal hall of fame are found in temperate climates and/or are situated in river deltas. Only two areas, the extensive Amazon deposits off the Guyanas in South America and Roebuck Bay/Eighty Mile Beach in northwest Australia, are truly tropical. In addition, the northwest Australian intertidal flats are examples of areas without connections to large river outlets. To make them even more special, Roebuck Bay and Eighty Mile Beach find themselves in the 'Indopacific biogeographic region' (which covers the tropical Indian Ocean and Pacific basins). Here, biodiversity of marine organisms is the highest in the world. Furthermore, the tundras and taigas of the Russian Far East harbor the most diverse shorebird fauna of the whole circumpolar region. Roebuck Bay is at one end of the most species-rich migration flyway in the world. Be it within the context of Australia, Australasia or the world, Roebuck Bay has no match (Pepping et al. 1999; Piersma et al. 2005).

In 1971, the International Convention on Wetlands (called the Ramsar Convention) was signed in Ramsar, Iran. It was formally adopted on 21 December 1975. Designed to promote the conservation and wise use of wetlands of international

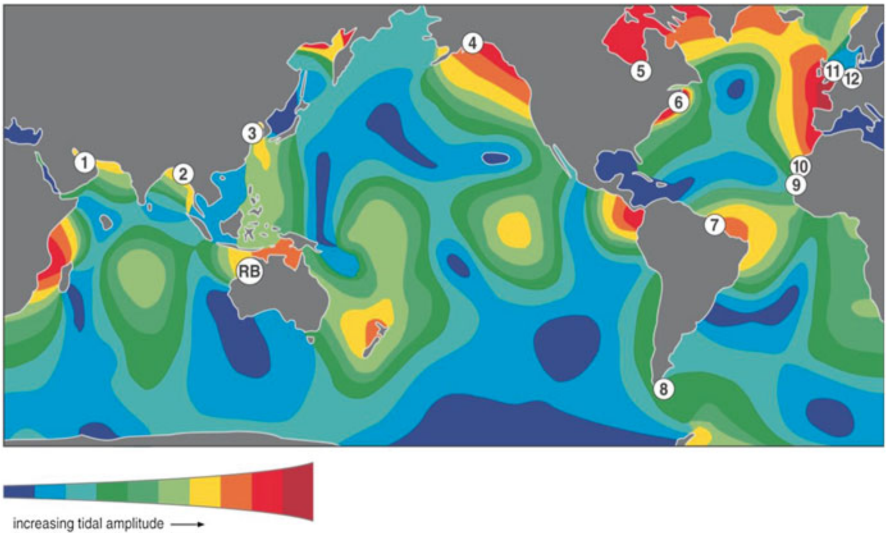


Fig. 11.2 Tidal ranges of the oceans of the world, and the world's most important large mudflat systems for shorebirds. Roebuck Bay and the mudflats off Eighty Mile Beach (RB) share distinction with (*1*) the shores around the Persian Gulf, (*2*) the Ganges and Brahmaputra Delta in Bangladesh, (*3*) the mudflats of the northern Yellow Sea, (*4*) the Copper River Delta in Alaska, (*5*) James Bay and (*6*) the Bay of Fundy in Canada, (*7*) the broad soft-sediment foreshores of the Guyanas, (*8*) the great bays of Tierra del Fuego, the West-African (*9*) Archipelago dos Bijagos in Guinea-Bissau and (*10*) Banc d'Arguin in Mauritania, (*11*) the large estuaries of Great Britain, and (*12*) the Wadden Sea (Rogers et al. 2003)

significance within each signatory's territory, the Convention placed an obligation for contracting parties to list and protect qualifying wetlands as Ramsar Sites (Ramsar 2013). Australia was one of the first countries to become a contracting party to the Ramsar Convention.

There are six objectives that each contracting party to the agreement has committed to:

1. Designate wetlands as Ramsar Sites for inclusion on the list of Wetlands of International Importance according to set nomination criteria.
2. Ensure the ecological characters of listed Ramsar Sites are protected;
3. When developing national planning mechanisms ensure they include wetland conservation
4. Ensure there is a mechanism to establish nature reserves on wetlands even if not included in the Ramsar List
5. Ensure there is adequate training in the fields of wetland research, management and conservation.
6. Promote and participate in international efforts to protect wetlands

Eighty Mile Beach and Roebuck Bay are two of the more significant inclusions in the Ramsar listings for Australia and, consequently, have seen increased interest in recent years for proper management through improved public awareness, progress towards a management plan, and legislative process.

We are constantly informed of new threats to more of the Earth's biological diversity. Much has been written about this increasing loss of not just species, genes, and memes, but of whole ecosystems and the functional processes necessary to support healthy, living communities (Piersma et al. 2005).

The proper conservation of the wonders of places like Eighty Mile Beach or Roebuck Bay is now more than ever dependent upon the supportive communal goodwill. The human communities, their willingness for social commitment and the political economies within them will shape the level of conservation of the ecological and the biophysical features of these and many other heritage jewels. There is a clear moral and pragmatic need to include indigenous owners and custodians in the process of joint management of lands and seas, especially in reference to areas such as Eighty Mile Beach and Roebuck Bay. The success and extent of the conservation effort may be judged by our ability to embrace these concepts and to convey the importance of these areas to the wider community.

11.4 Volunteers

Both Roebuck Bay and Eighty Mile Beach are large places, so any project must start with a logistical analysis – how much can we do? As we were also dependent upon volunteers, the projects had to be scaleable; the amount of work we could complete would depend upon the often last-minute success of recruiting efforts. The total number of involved individuals at each early expedition was: ROEBIM97, 30; ANNABIM, 80; SROEBIM02, 140; and ROEBIM06, 38.

In 1996, the Broome Bird Observatory was successful in acquiring a large grant from the Western Australian Lotteries Commission that enabled the construction of a dedicated laboratory in preparation of the first planned benthic mapping expedition in Roebuck Bay in 1997. The laboratory provided a modern, ergonomic and air-conditioned work space for volunteers and proved essential for the successful operation of benthic mapping projects between 1997 and 2013 and the many shorebird expeditions that have ensued since 1996.

Included in the package was a sum for the purchase a secondhand two person hovercraft, essential for traversing the deep muddy parts of the Bay. The hovercraft provided many researchers of benthos and shorebirds with unparalleled access to remote parts of the bay for several years until it was finally replaced with a larger four person hovercraft donated to BBO as surplus-to-requirements by The Western Australian Department of Health.

The primary key to our volunteer success was working with CALM (now the Department of Parks and Wildlife [DPaW]). DPaW maintains a strong policy of volunteer involvement in a range of programs that has extended through the period of benthic monitoring at Roebuck Bay and Eighty Mile Beach. In 2010–2011 more than 725,000 volunteer hours were contributed to the Department through the

Community Involvement Unit, The Bush Rangers and the Keep Australia Beautiful programs. Since the program began in the early 1990s, the number of volunteers registered with DPaW and the number of projects in which volunteers are involved has steadily increased. In 2010–2011, there were 12,759 volunteers registered with the department, 3,602 of whom actively contributed 436,216 h of support to various conservation and community projects across WA. This volunteer contribution was valued at around $15.12 million in 2010–2011. For every DPaW employee, around 2,000, there are more than six DPaW registered volunteers contributing their time to caring for WA's natural areas. Since 1999, DEC volunteers have contributed over 4 million hours of work (Department of Environment and Conservation DEC 2011).

The DPaW Landscope Expeditions program was involved in our mudflats work on four occasions: 1997, 1999, 2002, and 2006. Expeditioners not only volunteered their time, but they contributed funds towards the operating costs of the research projects. One hundred percent of all contributions were directed towards the research and field cost of the projects. Our expeditions would have been difficult without the hard cash provided through this program.

In addition to DPaW and individual volunteers, we partnered with a number of local organizations. These include Environs Kimberley Inc. (EK) (the 1997, 1999, 2002, and 2006 expeditions), who was responsible for local publicity, recruitment from the local community, account and expenditure management, and liaison. The successful incorporation of the local community into the field program was essential, and the continued promotion of the benthic work by EK in their regular newsletter helped maintain local enthusiasm for benthic research. The Broome Bird Observatory (BBO) (1996 to present) has traditionally provided an important link between the local community and research activities and continued to assist with the provision of the BBO mudflat monitoring equipment. BBO staff also participated in the field preparations and subsequent collection management and data analysis. In addition to the mapping efforts, as a reach-out to the Broome community, the project incorporated the 'Celebrate the Bay Forum' on 17 June 2006 on the CALM grounds in Broome. This 1-day event was widely publicized in local media, visited by about 150 people, and generated enthusiasm for the ecology of the bay and concerns about its future well-being.

Throughout the 17 year program of mapping the benthos of the two intertidal wetlands, industry and Government organizations have contributed essential funds and equipment. Every donation of money, equipment, or time, however small, was important and highly valued. Repeated and timely access across the vast intertidal flats at low tide was an integral part of the programs' success. This was achieved by foot where sample sites were close to access points, by boat where access to distant parts of the sites involved transporting small teams at high tide thus allowing them to follow the tide out to the waiting vessel. Hovercraft were used on each survey and provided rapid access across deep mud or flooded sites. A large hovercraft was donated and, piloted by Jamie Wallis of Wallis Drilling for ROEBIM97, ANNABIM99, and SROEBIM02 expeditions.

The survey at Eighty Mile Beach took place in the month of October, at the cusp of the wet season and a time that could be subjected to very high day time temperatures. The camp site at Anna Plains Station provided the teams with some respite from the heat. The Station also provided power and water that supplied a converted caravan, loaned to the expedition by Western Australian Main Roads, with electricity for microscopes and air conditioning. These facilities enabled teams to work long into the night to complete sample identification.

Finally, the university faculty often had undergraduate, honors, and graduate students working on individual research projects which tied into our overall goals (Crean 2000; Carew and Hickey 2000; Wade and Hickey 2008; Wade 2004; Hickey et al. 2000; Compton et al. 2007a, b; Rogers et al. 2000, 2001).

11.5 Survey and Data Management Methods

This section will describe the methods used during the 2006 expedition. They were the culmination of a multi-expedition learning process, and were shown to be extremely successful. In 2006, our team comprised 38 participants, 2 Landscope Expeditioners, 11 local volunteers, 13 logistical support, 13 Science support, and 10 scientific coordinators. A core group representing each collaborating institution established science objectives and time scales. Members of this group were responsible for the recruitment of suitable support personnel for their respective roles.

Initially, we developed a sampling grid which covered the entire bay (Hickey et al. 2003). This required writing a small program which generated a regular grid of points and a unique ID (available at www.onlinegeographer.com). The grid was on a 200 m spacing in the northern part of the bay (the area closest to human habitation and impacts). This grid file was then imported into our geographical information system (GIS) software and edited such that only possible sampling locations were shown. This represented the beginning of our database, to which every bit of data gathered was tied. In 2006, we used the identical grid created in 2002; prior expeditions also used a 400 m grid for the eastern and southern portions of the bay.

The study took place at Roebuck Bay between Crab Creek in the northeast and Town Beach in the northwest. With a neap tide on 23 June, sampling during the first week took place with tidal ranges that did not expose the full extent of the intertidal flats. For most of the project, the range (or distance from the shore) of our sampling was constrained by these neap tides. We tried to cover as much as possible of the areas sampled in June 1997, March-April 2000 (during the *Tracking-2000* expedition) and June 2002. Every sampling station received a unique station number composed of a row number (from south to north), a column number (from west to east) and an indicator of north (n) or south (s), and example being “r14c56n.” Navigating to the stations by GPS, teams of two to four people visited each of the stations based upon the geographical co-ordinates that were pre-assigned to them. Most samples were taken by teams on foot, but the whole area east of the BBO that included the deep muddy areas around Crab Creek, was sampled by the two hovercraft teams. Every field team was provided custom GIS maps and an associated spreadsheet of grid coordinates of the area they were to sample (Fig. 11.3).

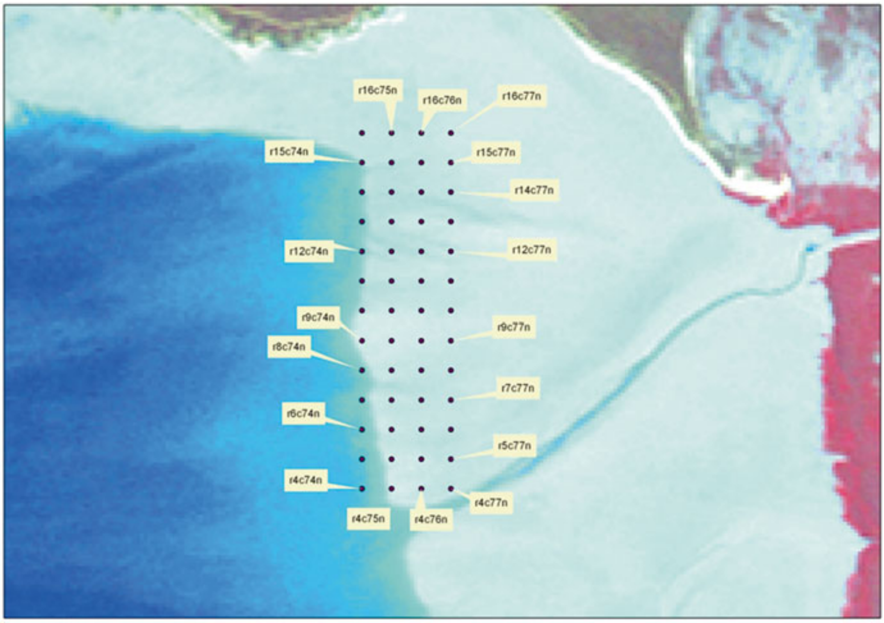


Fig. 11.3 Example of the field map with 'hopeful' sampling stations for the hovercraft team on 13 June 2006, such as they were routinely prepared. Naming in rows (*r*) and columns (*c*)

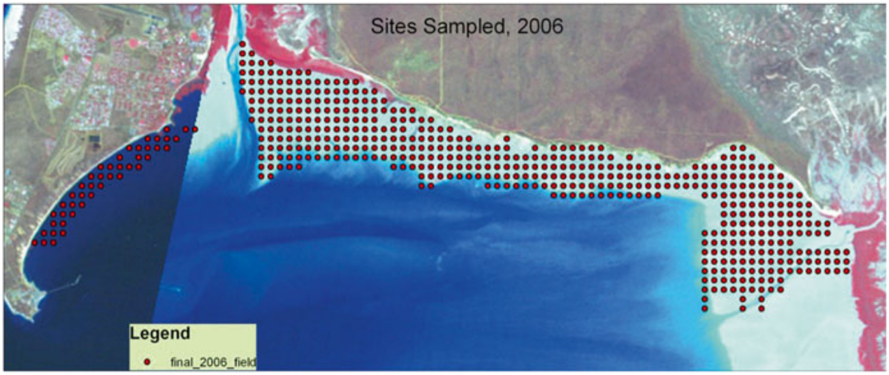


Fig. 11.4 Sites sampled in 2006

In the period 11–20 June 2006, we visited 532 sample stations and mapped both the invertebrate macrobenthic animals (those retained by a 1 mm sieve) over the whole of the northern intertidal area of Roebuck Bay (Fig. 11.4) and the shorebirds that depend on this food resource. More than 12,000 individual invertebrates were identified and measured. These animals represented 185 taxa at taxonomic levels ranging from species (bivalves, gastropods, brachiopods, some of the echinoderms,

Fig. 11.5 Sieving a core sample (Photo by Theunis Piersma)

and sipunculids), families (polychaete worms, crustaceans, and sea anemones) to orders and phyla (Phoronida, Echiura, Nemertini, and Tunicata).

Prior to heading into the field, training sessions were held with both the equipment and the sampling protocols. These were especially important, as they sped up the sampling times considerably. Of particular note was using GPS receivers – and explaining the built-in inaccuracies and that getting close to the sample point was more important than attempting to find the exact coordinate. This was much more relevant in the pre-2,000 expeditions, prior to selective availability being turned off.

Sample points were located in the field using handheld GPS receivers of multiple different models. They were invaluable for finding sample sites on the otherwise nearly featureless mudflats. For those that were keen, sample points were entered as waypoints into GPS receivers – thereby making the finding of those points even simpler. Daily progress maps showing sites sampled to date were generated daily and used during evening briefings.

At each station, three corers made of PVC-pipe were pushed down to a depth of 20 cm (less if the corer hit a hard shell layer below which we expect no benthic animals to live), and the core samples, each covering 1/120 m^2, removed (Fig. 11.5). The samples with a total surface area of 1/40 m^2 were sieved over a 1 mm mesh and the remains retained on the sieve placed into a plastic bag, to which a waterproof label indicating the station was added. At the same time, a sediment sample was taken with a depth of 3–5 cm and a diameter of 4.4 cm (surface area = 1/650 m^2), stored in a labeled plastic bag, and kept at outside temperature for transport to the laboratory. These sediment samples were analyzed in a laboratory either in Perth or at NIOZ, Texel.

Fig. 11.6 The sorting process, in makeshift conditions, just outside the BBO mudlab in full swing (Photo by Jan Drent)

In the field, records were made of the penetrability (depth of footsteps made by an average person, in cm – a surrogate for grain size) and the presence of visible larger, and therefore more uncommon, animals on the mud surface, the sort of animals (sentinel crabs, anemones, Ingrid-eating snails *Nassarius* sp.) that may be missed by our sampling technique. The sheets also allowed us to record which of the predetermined stations were actually visited, the names of the observers, and the times of sampling.

The biological samples were taken back to the Broome Bird Observatory and immediately sorted in low plastic trays in the sorting area just outside the mudlab Laboratory (Fig. 11.6) or stored in a fridge at 4 °C for a maximum of 1.5 days and then sorted. All living animals were then kept in seawater, again at 4 °C for a maximum of 1 day, upon which they were examined under a microscope by specialists seated indoors in the mudlab (Fig. 11.7). All invertebrates were assigned to a single taxonomic category. At the same time, the maximum length (in case of molluscs and worm-like organisms), or the width of the core body (in brittle stars), was measured in mm. The latter information will be of use in making predictions of the benthic biomass values using existing predictive equations.

We also upgraded the historical reference collection for more detailed study of the species at a later stage. Some of the polychaetes collected were preserved for later detailed examination. Most bivalves were dissected and the flesh dried and incinerated for determination of biomass values. We added to the ethanol-collection of bivalve tissues to be used for genetic screening of species differences (Compton et al. 2007a, b.

Fig. 11.7 Identification of sorted samples in the BBO mudlab (Photo by Theunis Piersma)

11.6 Mapping

On every expedition, maps were the foundation upon which a benthic sampling expedition was based. Fortunately, the databases from all prior expeditions were available. The primary base maps were 1994 (low tide) and 1995 (high tide) Landsat images. The grid spacings for the expedition point data were 200 m for the northern shore and 400 m for the eastern and southern shores. Everything used the AMG zone 51 coordinate system on the AUSGEOID 66 datum.

Once the field sampling was complete, all field, bird, and species data were entered into the GIS database – often requiring considerable gyrations to get everything in the proper format. Because all the surveys used consistent sampling locations (except the first, which used latitude/longitude and had to be converted to the nearest AMG sample location), we could run comparisons of change over time at Roebuck Bay (1997–2006). The extent of the surveys along the northern shores during these two previous surveys in comparison with the mapping efforts in 2006 are shown in Fig. 11.8.

11.7 Other Essentials

On each expedition, we had a dedicated cook. He/she was responsible for making sure food was available for breakfast, sandwiches (or similar) for lunch, and a nice dinner. Everyone kicked in on dishwashing duties. This individual was invaluable,

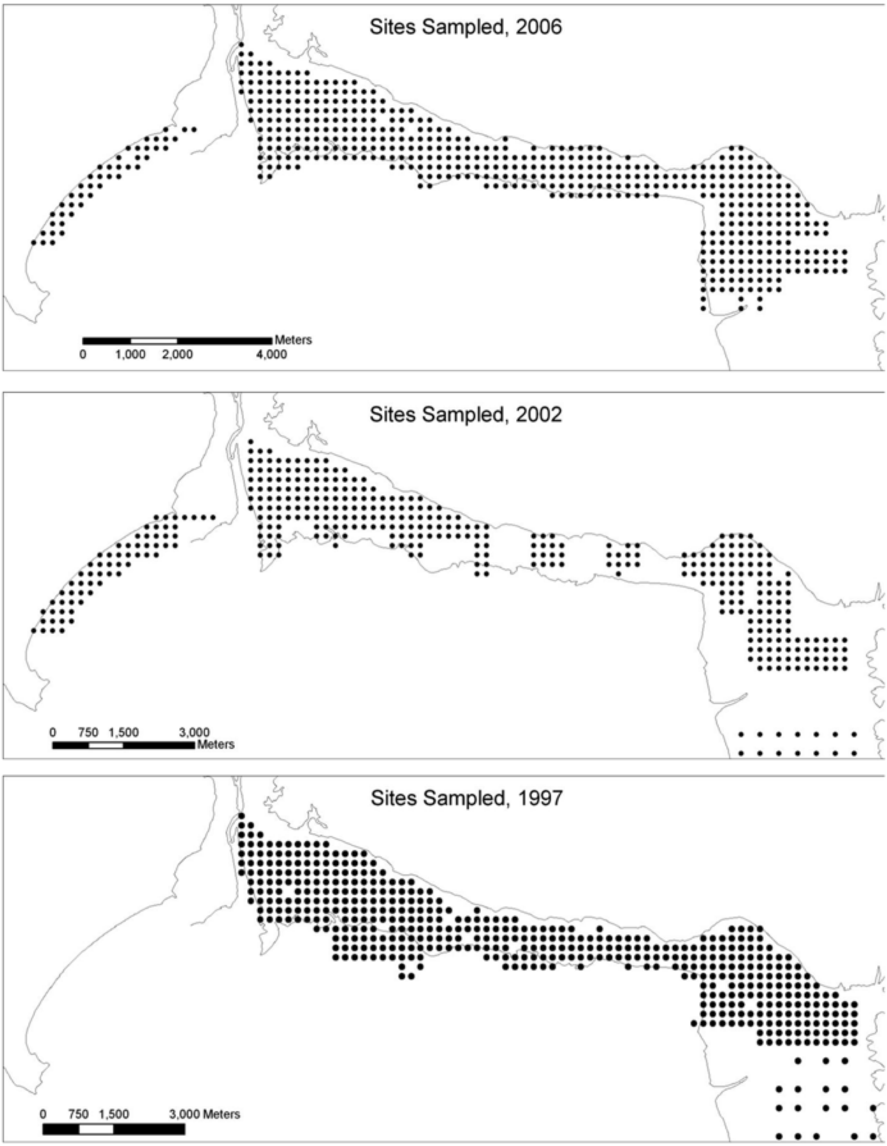


Fig. 11.8 The extent of the grids along the northern shore sampled in 1997, 2002 and 2006. In 1997

for not only did we receive consistently good food, but everyone could focus on their duties without worrying about (and spending time on) cooking.

Mechanics were also important to have on-site, as there were always issues with the boats, hovercraft, and other equipment. As our time to sample was limited, keeping the equipment up and running was absolutely necessary.

First Aid was available from trained DEC personnel in addition to the presence on the logistic support crew of a trained nurse in 1999 and in 2002 by a GP experienced in remote area medicine.

Effective communication between sample teams and community support organizations was essential. Where cell phones didn't work, we had radio communications between the different sampling teams.

11.8 Conclusions

Roebuck Bay and Eighty Mile Beach are nationally (ANCA 1996) and internationally recognized (Ramsar 2013) as important locations for shorebirds providing a key migratory stopover in spring and autumn. These areas also provide a home for up to 150,000 birds in the non-breeding season. They are two of less than 20 locations worldwide characterized by soft bottom intertidal mudflats supporting large numbers of migratory shorebirds, and are the foremost internationally important sites for shorebirds in the Asia-Pacific flyway system (Pepping et al. 1999). The ability of Roebuck Bay and Eighty Mile Beach to support large numbers of shorebirds and to facilitate their annual migration to and from the arctic relates to the particularly abundant and diverse food source, comprising the benthic macro/meiofauna resident in the extensive tidal mudflats.

Much of our recent knowledge and understanding of the biodiversity of the mudflats and their usage by migratory shorebirds has arisen through campaign surveys of the areas, undertaken with military precision and organization, and adhering to standardized and repeatable methods. These extensive surveys have only been possible through the tireless contribution by many community volunteers. If fully costed, these projects would never have been funded. It is anticipated that future research on the intertidal mudflats will continue to rely heavily on community involvement. Local community involvement in research on the mudflats and migratory waders is now an ever-present and increasingly important aspect of the ongoing conservation effort at Roebuck Bay and Eighty Mile Beach. It is absolutely true that the success of these expeditions hinged upon the ability to recruit volunteers and receive donations of time, personnel, and equipment from both the Government and private sources.

Acknowledgements A great many people have contributed to the success of these community based research projects that have established Roebuck Bay and Eighty Mile Beach as special wetlands with unique properties. To all these people we express our gratitude and our hope that we can continue to work together on projects that strengthen the arguments for conservation of these wetlands.

References

Australian Nature Conservation Agency (ANCA) (1996) A directory of important Wetlands in Australia, 2nd edn. ANCA, Canberra

Battley PF, Warnock N, Tibbitts TL et al (2012) Contrasting extreme long-distance migration patterns in bar-tailed godwits *Limosa lapponica*. J Avian Biol 43:21–32

Carew R, Hickey R (2000) Determination of a generalized tidal inundation model for Roebuck Bay (Western Australia) using an integrated GIS and remote sensing approach. Trans GIS 4(2):99–111

Compton TJ, Drent J, Kentie R et al (2007a) Overlap in the feeding morphology of bivalves from species-rich and species-poor intertidal flats using gill: palp ratios for comparative analyses of mollusc assemblages. Mar Ecol Prog Ser 348:213–220

Compton TJ, Rijkenberg MJA, Drent J et al (2007b) Thermal tolerance and climate variability: a comparison between bivalves from differing climates. J Exp Mar Biol Ecol 352:200–211

Crean M (2000) GIS and remote sensing to tidal mudflat mapping Eighty-Mile Beach, Western Australia. Honours thesis, Curtin University of Technology, Perth, Australia, p 77

De Goeij P, Lavaleye M, Pearson G et al (2003) Seasonal changes in the macro-zoobenthos of a tropical mudflat. NIOZ-Report 2003–4. Netherlands Institute for Sea Research (NIOZ), Texel, p 49

Department of Environment and Conservation (DEC) (2011) Community Involvement Annual Report 2010–2011. Available via http://www.dec.wa.gov.au/community-and-education/volunteer-programs.html. Accessed 13 Nov 2013

Hickey, R, Carew R, Watkins R et al (2000) Tidal inundation modelling, Roebuck Bay, Western Australia. GIS Asia Pacific, February/March, pp 47–50

Hickey R, Pearson G, Piersma T (2003) Maps, mud, and birds. Geospat Solutions 13(7):40–43

Honkoop P, Pearson G, Lavaleye M et al (2006) Spatial variation of the intertidal sediments and macrozoo-benthic assemblages along Eighty-mile Beach, North-western Australia. J Sea Res 55:278–291

Pepping M, Piersma T, Pearson G et al (eds) (1999) Intertidal sediments and benthic animals of Roebuck Bay, Western Australia. Netherlands Institute for Sea Research (NIOZ), Texel

Piersma T, Pearson G, Hickey R et al (eds) (2005) The long mud: benthos and shorebirds of the foreshore of Eighty-Mile Beach, Western Australia. NIOZ Report 2005-2. Netherlands Institute for Sea Research (NIOZ), Texel, 218 p

Piersma T, Pearson G, Hickey R et al (2006) Roebuck Bay invertebrate and bird mapping 2006. Research report. Available from http://www.onlinegeographer.com/roebim06/ROEBIM06_prog_rep140806.pdf. Accessed 13 Nov 2013

Ramsar (2013) The Ramsar convention manual, 6th edn. Available via http://www.ramsar.org/pdf/lib/manual6-2013-e.pdf. Accessed 13 Nov 2013

Rogers, DI, Taylor I (2001) Conservation and ecology of migratory shorebirds in Roebuck Bay, North-western Australia. Final Report for the Wetlands Unit. Environment Australia. Albury: Charles Sturt University

Rogers DI, Battley P, Russell M et al (2000) A high count of Asian Dowitchers in Roebuck Bay, North-western Australia. Stilt 37:11–13

Rogers DI, Boyle A, Hassell C (2001) Wader counts on Kidneybean Claypan and adjacent Roebuck Plains, Northwestern Australia. Stilt 38:57–63

Rogers DI, Piersma T, Lavaleye M et al (2003) Life along land's edge. Wildlife on the shores of Roebuck Bay, Broome. Department of Conservation and Land Management, Perth

van de Kam J, Ens BJ, Piersma T et al (2004) Shorebirds. An illustrated behavioural ecology. KNNV Publishers, Utrecht

van de Kam J, Battley PF, McCaffery BJ et al (2010) Invisible connections. Why migrating shorebirds need the Yellow Sea. CSIRO Publishing, Melbourne

Wade S (2004) Using spatial analysis to aid in managing Eighty-Mile Beach, Western Australia. Unpublished MS thesis, Central Washington University, Ellensburg, WA, USA, 102p

Wade S, Hickey R (2008) Mapping migratory wading bird feeding habitats using satellite imagery and field data, Eighty-Mile Beach, Western Australia. J Coast Res 24(3):759–770

Chapter 12
Sea-Level Indicators

Niki Evelpidou and Paolo A. Pirazzoli

Abstract Because changes in sea level may have a great impact on the distribution of mineral resources, the exploration and exploiting of these resources should not ignore the changes in sea level that may have occurred in the past in the area considered. The study of relative sea-level changes is an essential element of ocean observation and technological advances are often necessary to improve this study that includes the determination of levels (elevation or depth), chronological estimations, and the identification of appropriate sea-level indicators.

Indicators of fossil or present-day sea-level positions are the most important elements for a sea-level reconstruction, because they provide information not only on the former level but also on the accuracy of the reconstruction.

A classification is proposed of the main criteria that can be used to deduce appropriate sea-level indicators from geomorphological, stratigraphical, biological or archeological coastal data. Two cases studies are used as examples of sea-level reconstructions that may be useful to clarify the geology in certain areas, or to coastal engineering and coastal protection: (1) on the impact of the recent sea-level rise in the interpretation of sea-level indicators; and (2) on the foreseeable impacts of the predicted near-future sea-level rise on the coasts of NE Italy.

N. Evelpidou (✉)
CNRS – Laboratoire de Géographie Physique, Meudon, France

Faculty of Geology and Geoenvironment, National and Kapodistrian University of Athens, Athens, Greece
e-mail: evelpidou@geol.uoa.gr

P.A. Pirazzoli
CNRS – Laboratoire de Géographie Physique, Meudon, France
e-mail: paolop@noos.fr

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8,
DOI 10.1007/978-3-319-06305-8_12,

12.1 Introduction

Because changes in sea level may have a great impact on the distribution of mineral resources, the exploration and exploiting of these resources should not ignore the changes in sea level that may have occurred in the past in the area considered. On the other hand, due to the fact that recent and ongoing sea-level changes have a great impact on the biologic resources in the near shore environment, the possibility of such changes should be considered by mariculture farms and coastal engineers. The study of relative sea-level changes is therefore an essential element of ocean observation and technological advances are often necessary to improve this study that includes the determination of levels (elevation or depth), chronological estimations, and the identification of appropriate sea-level indicators.

Levels may be determined in many manners (with satellites, oceanographic vessels, geophysical equipments, leveling techniques, tide-gauge devices, or even direct measurement by an observer). Chronological estimations may result from radiometric analysis of samples, comparison with stratigraphic sequences, archeological or historical data, assumptions on erosion or deposition processes, or even from glacio-isostatic or climate modeling.

Indicators of fossil or present-day sea-level positions are nevertheless the most important elements for a sea-level reconstruction, because they provide information not only on the former level but also on the accuracy of the reconstruction.

We propose in this chapter a classification of the main criteria that can be used to deduce appropriate sea-level indicators from geomorphological, stratigraphical, biological or archeological coastal data, followed by a couple of case-studies of sea-level reconstructions that may be useful to clarify the geology in certain areas, or to coastal engineering and coastal protection: (1) on the impact of the recent sea-level rise in the interpretation of sea-level indicators; and (2) on the foreseeable impacts of the predicted near-future sea-level rise on the coasts of NE Italy.

12.2 How Can Fossil Paleoshorelines Be Identified?

Fossil paleoshorelines can be identified and traced from geomorphological, biological, sedimentological, stratigraphical or archeological sea-level indicators.

The coastal geomorphological features that are used as sea-level indicators are the result of either erosional or depositional processes. Erosional features can only be preserved on hard, solid rocks and in some cases they constitute indicators of sea-level change. Such indicators are marine notches, potholes, abrasion platforms, etc. Among the depositional formations, marine terraces and beachrocks stand out as the most important sea-level indicators.

Fig. 12.1 Uplifted wave cut notch at western part of Corfu island. Although it indicates that an uplift has taken place it is not a precise sea level indicator that could provide the exact magnitude of the uplift

12.2.1 Erosional Geomorphological Sea-Level Indicators

Erosional processes along coastlines include the direct effects by waves, abrasion, salt weathering, bioerosion and chemical attack (Rampino 2005).

Marine notches on coastal cliffs near sea level may be produced by various processes. The imprecise term ‘wave-cut’, often found in the literature, generally refers to erosion by wave activity slightly above sea level. The effect of waves transporting sand or gravel, on the rock is abrasive, forming wave-cut platforms and notches, which are easily recognizable by their polished surfaces. The accuracy of abrasion features as sea-level indicators is often weak, depending mainly on exposure (Fig. 12.1).

Salt weathering takes place when salt crystals grow within pore spaces in rocks and is most effective in areas exposed to salt spray and subject to alternative wetting and drying. Salt weathering may produce a honeycomb pattern or a cavernous weathering called tafoni which should not be used however as sea-level indicator, because tafoni have also been observed in arid tropical areas remote from the sea (Fig. 12.2).

Biological erosion may be especially important in tropical or subtropical areas on carbonate coasts. Algae are probably the most important bioerosional agent on

Fig. 12.2 Tafoni formations may not be used as sea-level indicator since they have been observed also in areas remote from sea. Tafoni of this figure have been formed in a granodiorite hill of Naxos island (Cyclades) and are not related to the sea level changes that have taken place in this area and are generally of subsidence (Evelpidou et al. 2013a)

rocky coasts. The substrate under algal mats is exposed to the products of metabolism and organic waste, which can directly etch the rocks. Algae also support grazing organisms such as gastropods and echinoids, which abrade rock surfaces. Boring and browsing organisms erode rocks most effectively in the intertidal zone. The rate of maximum undercutting (near MSL) varies with the rock type and the local climate and has been roughly estimated to be of the order of 1 mm/a (Laborel et al. 1999). However, this is only a first order value, as lower rates are generally observed in hard limestones, especially in non tropical areas. More detailed estimations show a range varying from 0.2 to 5 mm/a, depending on lithology, location, and probably duration of bioerosion (for references, see Pirazzoli 1986, Table 1; Laborel et al. 1999, Table 1; Evelpidou et al. 2011a, Table 1).

In the midlittoral zone, several types of notches can be useful as sea-level indicators, though with variable accuracy: tidal notches, surf notches and solution notches.

Tidal notches are well known as precise sea level indicators that usually undercut limestone cliffs in the mid-littoral zone (e.g. Pirazzoli 1986), and constitute the most important erosional geomorphological sea-level indicators. In microtidal areas sheltered from wave action, elevated or submerged notches are used to indicate former sea-level positions, with up to a decimetre confidence.

Bioerosion by endolithic organisms and surface feeders grazing upon epi- and endolithic algae are generally acknowledged to play an important role in tidal-notch development. The erosion rate is generally highest near mean sea level (MSL) and

Fig 12.3 A well developed continuous tidal notch at Cephalonia island, which is emerged nowadays. During its formation the vertex was located near MSL, its base near the lowest tide and its top near the highest tide level

decreases gradually towards the upper and lower limits of the intertidal range. Accordingly, in places sheltered from continuous wave action, if MSL remains stable, tidal-notch profiles will be typically reclined U-shaped or V-shaped, with their vertex located near MSL, their base near the lowest tide and their top near the highest tide level (Fig. 12.3). In a moderately exposed site, continuous wave action may splash sea water onto the roof, thus shifting the top of the notch upwards, above the highest-tide level. When a tidal notch is uplifted or submerged, its profile may provide valuable information on whether the movement was co-seismic or gradual and eventually on the type of paleoseismic history, e.g. whether one or more events took place. Figure 12.4 depicts the three most common profiles for emerged and submerged tidal notches. Profile a and a′ corresponds to a rapid uplift and subsidence accordingly, greater than the tidal range, resulted to a preserved former notch. In Fig. 12.5a an example of a-type profile is shown from Qwambu, Huon Peninsula (Papua New Guinea) while in Fig. 12.5b an example of a′ type profile from Karpathos island (Greece) is shown. Profile b and b′ corresponds to two uplifted and submerged accordingly former notches, which were preserved after two rapid uplift and subsidence accordingly events, greater than the tidal range. In Fig. 12.6a repeated b-type profiles are shown from Tewai, Huon Peninsula (Papua New Guinea) while in Fig. 12.6b an example of b′-type profile from Kaminakia, Antikythira Island (Greece) is shown.

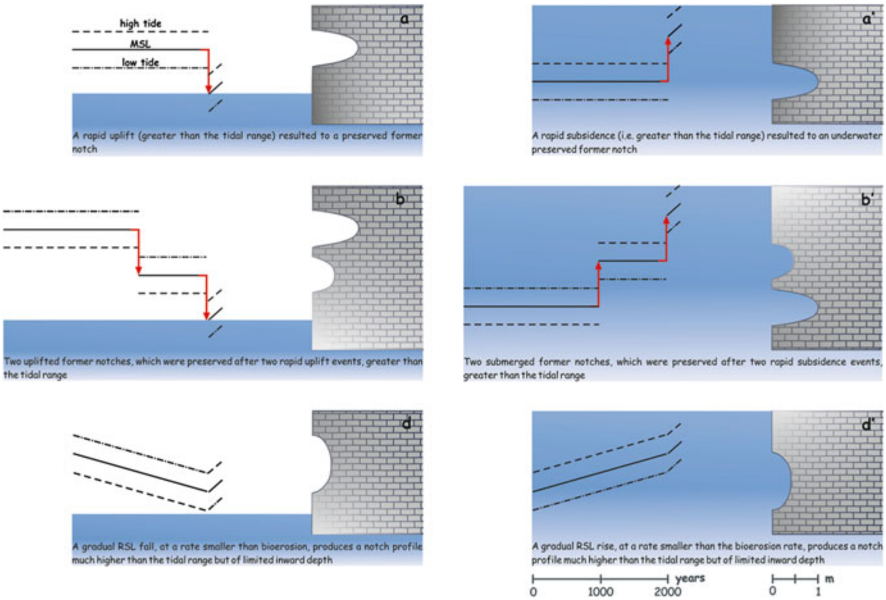


Fig. 12.4 Various profile types of tidal notches providing information on the type of the event that uplifted or submerged this sea level indicator

Finally, profile d and d′ depict a gradual RSL fall and rise, at a rate smaller than bioerosion, which produces a notch profile much higher than the tidal range but of limited inward depth. In Fig. 12.7a an example of d type profile is shown from Cheng-Kung, eastern coast of Taiwan, while in Fig. 12.7b an example of d′-type profile from north Corinth gulf (Greece) is shown.

Carbonate coasts exposed in stronger wave action commonly have a distinctive morphology. In fact, as water turbulence increases, so does the height of the notch and organic incrustations may develop on the floor of the notch. These organic incrustations often comprise calcareous algae (*Lithophyllum*, *Lithothamnium*, *Porolithon*, *Neogoniolithon*) and vermetids (*Denropoma*, *Petaloconchus*, *Spiroglyphus*) (Kempf and Laborel 1968; Focke 1978a; Laborel 1979) (Fig. 12.5). These organisms protect the substrate rock and thus locally inhibit erosion. While erosion proceeds above the accretion level, a surf bench may begin to form protruding seawards, which may extend as high as 2 m above high-tide level (Focke 1978b). Surf notches may therefore develop on more exposed sites above high tide level. Fossil surf notches can be easily identified and distinguished from fossil tidal notches when a bioconstructed accretion exists near the notch floor. In this case, indications on the former sea-level may be provided by the organic accretion rather than by the notch developed above it.

Solution notches are frequent in carbonate rocks near sea level, in the case of fresh-water arrivals. Higgins (1980) even believes that solution notches in calcareous rocks occur *only* in proximity to coastal springs, where surface seawaters are locally diluted by freshwater.

Fig. 12.5 (**a**) Holocene raised tidal notch of a-type, at Qwambu, Huon Peninsula (Papua New Guinea). Repeated Holocene coseismic uplifts occurred in this area, with recurrence of 1,000–1,300 years and average uplift of about 3 m (Chappell et al. 1996). C. Jouannic gives scale (Photo P.A.P. A898, 5 Aug. 1988). (**b**) Example of a' type profile from Karpathos island (Greece)

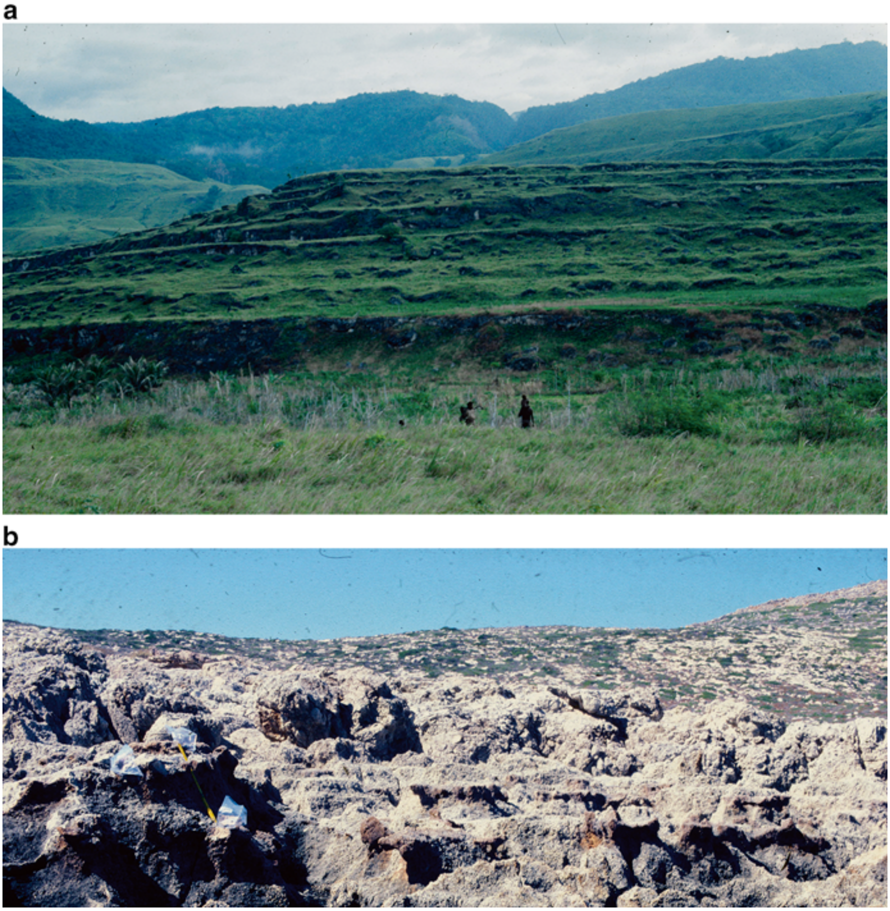


Fig. 12.6 (**a**) Repeated b-type profiles in Tewai, Huon Peninsula (Papoua New Guinea). The repeated Holocene co-seismic uplifts with a recurrence of 1,000–1,300 years and average uplift of about 3 m caused the development of sequences of Holocene regressive terraces (Chappell et al. 1996). Persons in the foreground give scale (Photo P.A.P. A954, Aug. 1988). (**b**) b′-type tidal notch at Kaminakia, Antikythira Island (Greece). Tidal notches initially formed at sea level and submerged repetitively, have been subsequently raised all together co-seismically in AD 365. Former shorelines may be distinguished at about +2.20 m (its submergence has been dated 1,490 ± 70 BP), at +2.00 m (dated 1880 ± 70 BP and at +1.7 m (dated 2180 ± 70) (Pirazzoli et al. 1981) (Photo P.A.P. 5655, Sept. 1979)

12.2.2 Depositional Sea Level Indicators

Beachrocks are hard, coastal sedimentary formations (Fig. 12.9), consisting of beach sediments that are rapidly cemented by the precipitation of carbonates (High-Magnesium Calcite or Aragonite) (Bricker 1971). The formation of beachrocks takes places in the intertidal zone, and constitutes a diachronic process. Since

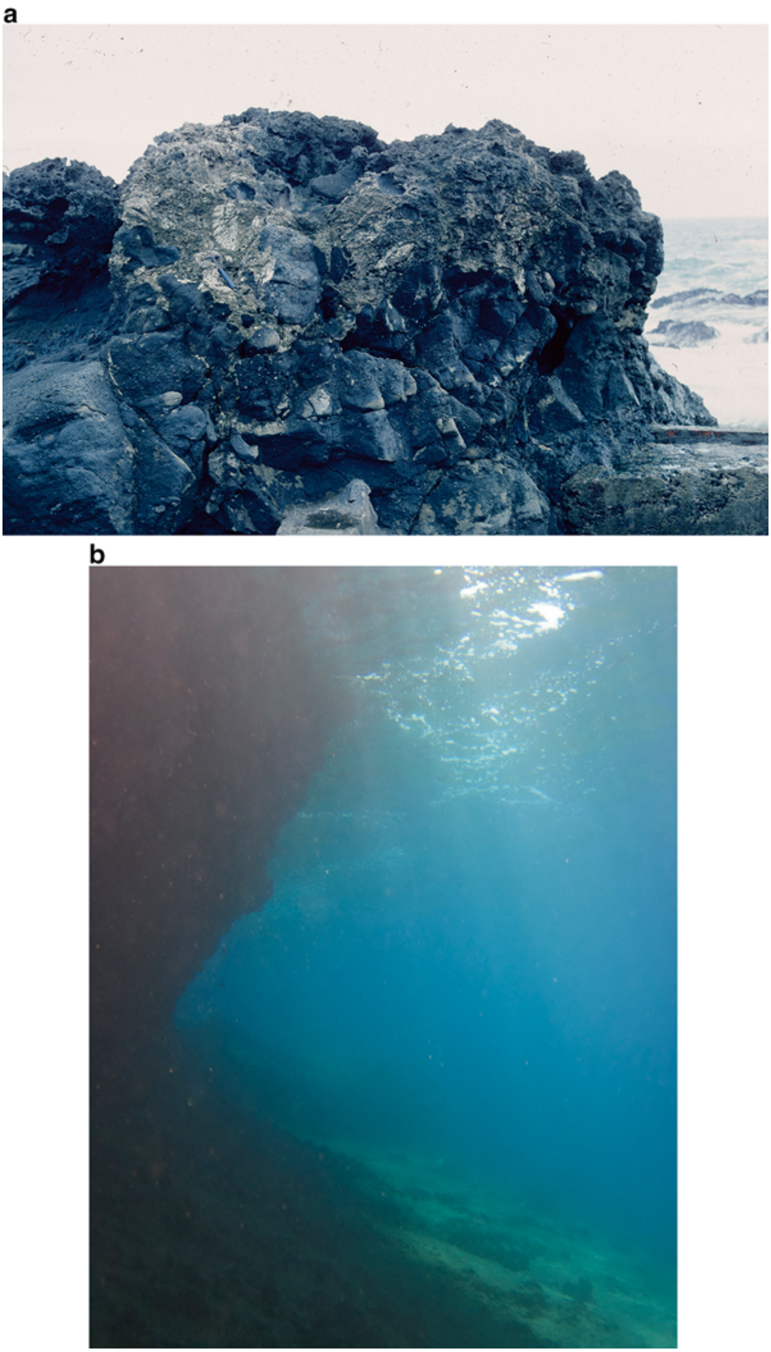


Fig. 12.7 (**a**) d-type tidal notch of height greater than the local tidal range bordering a raised coral reef reaching +3.5 m above MSL N of Cheng-Kung, eastern coast of Taiwan. Average tidal range = 1.1 m, Spring tidal range = 1.45 m. This area is characterized by irregular average uplift rates, varying between 2.5 and over 8.0 mm/a (Pirazzoli et al. 1993). At least part of the uplift of the coral reef probably occurred gradually (Photo P.A.P B737, 18 Jan. 1990). (**b**) d′-type tidal notch in north Corinth Gulf (Greece)

lithification takes place on the coast, beachrocks (also in correlation with other sea level indicators) have often been used as indicators of sea level changes and neotectonic movements (e.g. Tatumi et al. 2003; Kelletat 2006; Desruelles et al. 2009; Mourtzas 2012; Stattegger et al. 2013).

There is still some debate concerning the reliability of beachrocks as sea level indicators. According to Hopley (1986), beachrocks are more reliable in areas with small tidal range as opposed to low-latitude areas and furthermore only the upper part of beachrocks constitutes a reliable indicator of the paleo-tidal level. Laboratory analysis of beachrock samples is necessary in order to determine the mineralogical composition of the cement and not confuse them with other lithified materials of the coast.

A marine terrace is any relatively flat surface of marine origin, bordered by a steeper ascending slope on one side and by a steeper descending slope on the opposite side (Pirazzoli 2007) (Fig. 12.10). The development of marine terraces is linked to Quaternary sea level changes associated with global climatic changes and/or to active tectonic processes. A steadily and rapidly rising coastline is the best mark for measuring major long term sea-level fluctuations and tectonic uplift, since marine terraces are the geological records of former sea levels. A series of uplifted marine terraces could be even correlated with different interglacial periods, when the uplift rate is rapid enough. Marine terraces may be very useful at extracting rates of long-term tectonic deformation.

Speleothem formations of karstic origin may also provide information for past sea level fluctuations, when they are covered by marine biogenic overgrowths. Although speleothems are typical continental features, developed only in subaerial conditions, when they are submerged by a rising sea level, they are covered by marine biogenic overgrowths. By dating this biogenic overgrowth, it is possible to determine the timing of cave flooding due to rising sea level and the establishment of marine conditions. Speleothems have often been used for sea level reconstructions (e.g. Lundberg and Ford 1994; Antonioli and Oliverio 1996; Alessio et al. 1996; Bard et al. 2002; Antonioli et al. 2004).

12.2.3 Sedimentological Sea Level Indicators

In the study of coastal changes, environments such as coastal marshes and wetlands may provide powerful data as they are sensitive to environmental changes. In particular, sea level fluctuations affect low-lying coastal areas, and any change in sea level will be recorded by vegetation succession, changes in lithology and the microfossil assemblages within deposits. The most common method, for locating material that will provide information regarding sea-level change, is the use of a vibrating sampler. In such environments, the study of lithostratigraphy (sedimentary characteristics) and biostratigraphy (associated flora and fauna) allow to distinguish various sub-environments. Micropaleontological analysis allows distinguishing transitions between sub-environments within the sedimentary sequence.

In order to develop the chronology of the sedimentary sequence, the age of biomarkers and organic, peaty deposits can be established by radiocarbon dating. Biomarkers found in sedimentary sequences, through core analysis, should be related to sea level in a consistent and quantifiable way.

The reconstruction of sea level changes, through the use of drillings, stems from the combined interpretation of lithostratigraphy, biostratigraphy and chronostratigraphy. Such sea level reconstructions should take into consideration some limitations; compaction should be taken into account for the positional uncertainty of the dated samples and samples should be disturbed as little as possible.

12.2.4 Biologic Sea Level Indicators

As the research of sea level changes developed and started taking into account multidisciplinary criteria, the use of fossilized biological remains has developed greatly recently. On a rocky coast, several zones may be distinguished, where littoral flora and fauna are developed in parallel horizontal layers: supralittoral, mid-littoral and infralittoral zones.

Species that live in very narrow depth ranges are considered as the most reliable sea level indicators (Laborel and Laborel-Deguen 2005); such are for instance, the corallines *Lithophyllum lynoides* and *Lithophyllum onkodes*, vermetid gastropods of genus *Dendropoma* and *Spiroglyphus* (Laborel 1986) and annelids such as *Idanthyrsus* and *Galeolaria* (Baker and Haworth 1999). In order to accurately determine a past sea level position, one needs to distinguish and measure the altimetric difference between the fossilized remains and their present day counterpart (Fig. 12.8).

Apart from fossilized biological indicators, it has recently been shown that bioerosional textures in limestone coastlines (Kázmér and Taboroši 2012; Taboroši and Kázmér 2013) may help to determine the chronological order of vertical displacements. For example, traces of sea urchins on a -today- uplifted tidal notch in Cephalonia allowed to determine a complex history of subsidence followed by uplift.

Apart from biological indicators found in rocky coasts, biological markers from core samples are also used to determine past sea level positions, provided that compaction is minimum and the sample is disturbed as little as possible.

12.2.5 Archeological Sea Level Indicators

Geoarcheological analysis in coastal areas provides with valuable evidence for past sea levels, when it is possible to correlate the coastal structures with sea level.

The continuous presence of man during the last thousand years near the coasts has left numerous evidences, such as production structures, town structures and ports. The vast majority of archeological remains cannot provide evidence for the

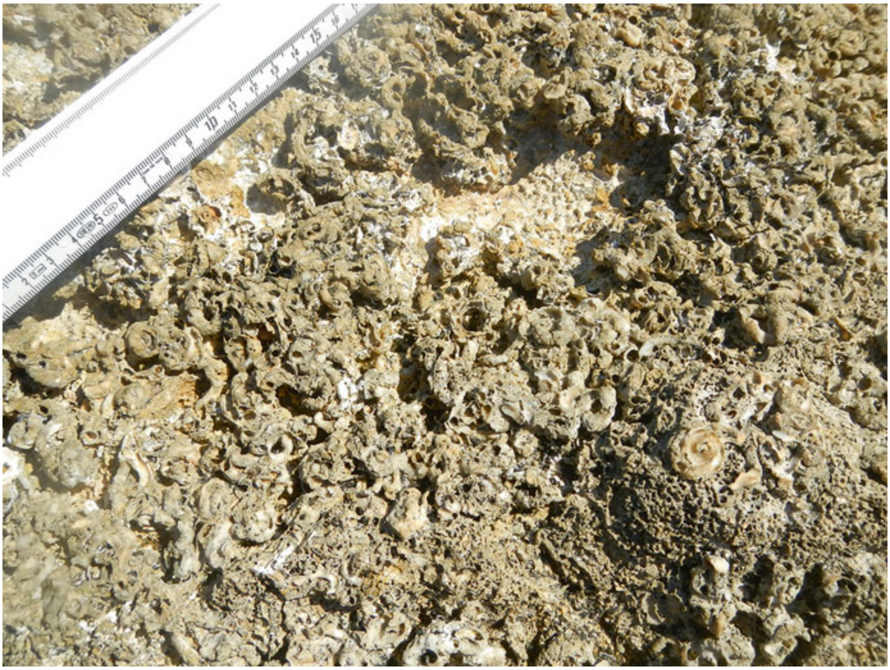


Fig. 12.8 Continuous incrustations of Vermetids up to +60 cm at Karavomylos area (Cephalonia island)

location of sea level and can only be used in a few cases to obtain information on past sea level fluctuations. This is owed to the fact that a site on land, which has been uplifted, is difficult to be distinguished from a site that was initially built in a higher altitude. In addition, although submerged sites suggest that a relative sea level rise has occurred, it is difficult to quantify such a rise. Nevertheless, some structures were directly "connected" with sea level and during their function were partly submerged, depending on tide and marine conditions (harbor structures, breakwaters, quays, docks, fish tanks, etc.) (Flemming 1979-1980); these may be considered as reliable sea level indicators (Fig. 12.11). A review of various archeological sea level indicators and their accuracy has been provided by Auriemma and Solinas (2009).

In order to reliably reconstruct ancient sea level based on archeological remains, it is necessary to have a good understanding of the structure's usage and functionality and the local hydrographic and climatic conditions.

12.3 Dating Relative Sea Level Changes

Dating sea level changes may be direct or indirect, depending on the type of sea level indicator available, whether it is found emerged or submerged and, therefore, whether datable material is available.

Fig. 12.9 Uplifted beachrock formation near Ancient Diolkos at Corinth Gulf. Scale is given by Niki Evelpidou

Fig. 12.10 Marine terrace at Arkitsa area (North Euboean Gulf, Greece)

The easiest and most trustworthy method is radiocarbon dating of biomarkers. This is usually the case for uplifted fossil shorelines, or samples deriving from coastal cores; in the latter the accuracy of the relationship of the biological marker to mean sea level should be taken into account.

In the case of submerged sea level indicators, indirect ways may be used to date a past sea level; correlation with other dated sea level indicators (e.g. archeological or from coastal cores) found at about the same depth is commonly used.

Archeological remains used as sea level indicators are not always easily and accurately dated. For instance, buildings may be dated with more precision by their intrinsic and distinctive features in comparison to quarries or breakwaters, whose typology remained unchanged for centuries (Auriemma and Solinas 2009). A special exception are Roman fish tanks, whose use was very frequent along the

Fig. 12.11 Punta della Vipera fish tank located near the town of Civitavecchia (Tyrrhenian coast of Italy). Features such as foot walks (crepidines), sluice gates, tops of channels, and moles may provide information regarding relative sea level at the time of construction

Tyrrhenian coasts of central Italy between the 1st c. BC and the 1st c. AD (Plinius, Naturalis Historia, IX; Columella, De Re Rustica, XVII; Varro, De Re Rustica, III).

A special mention should also be made to beachrocks, whose dating presents some difficulties. Beachrocks can be dated either by biogenic materials with radiocarbon, or by dating the cement. In the case of biogenic materials, it is possible to acquire an age much older than that of the beachrock lithification, because the time interval between the death of the organism and the cementation of the beachrock may have lasted very little time to even thousands of years. In the case of dating the cement, difficulties arise when trying to extract enough material and furthermore, the acquired age may be much younger due to subsequent alteration of the cement.

12.4 Case-Studies

12.4.1 The Impact of the Recent Sea-Level Rise on the Interpretation of Sea-Level Indicators

Several recent studies (e.g. Jevrejeva et al. 2008; Kemp et al. 2011) have shown from tide-gauge records that the global sea level has risen during the last two centuries at an average rate of the order of 2 mm/a, i.e. about 20 cm.

One may wonder what the impact of this sea-level rise could be on the interpretation of several sea-level indicators available today in the coastal zone. For depositional geomorphological sea-level indicators, the impacts would be quite variable, depending on the features considered. The recent sea-level rise at the rate of 2 mm/a exceeds the possibilities of bioerosion in the intertidal zone. This means that, especially on carbonate rocks, the deepening of tidal notches is interrupted. As a consequence, in microtidal areas, no new tidal notches could have formed during the last two centuries, leading to the disappearance of this type of feature, as shown by Evelpidou et al. (2012). In contrast, tidal notches developed before the nineteenth century will be submerged and preserved in fossil form (they are called 'modern' tidal notches) at a depth of the order of about 20 cm. The reality of such a recent disappearance has been verified in most areas of the Mediterranean, where submerged 'modern' tidal notches testify of the local MSL position preceding the recent period of sea-level rise, e.g. in Greece, in the Cyclades and the Sporades Islands (Evelpidou et al. 2013a, b), or in several Ionian Island (in preparation).

On the other hand, the response of other types of notches will be completely different. In surf notches containing a biologic accretion the sea-level rise may be absorbed by an increase in the development of the organic accretion, which will remain the only relatively valuable sea-level indicator (if its position in the vertical biological zonation can be taken into account), while the surf notch developed above it and the notch roof will remain close or above the present sea level, with no value as sea-level indicator. Such an example is the notch observed undercutting the limestone cliff in the Orosei Gulf (Sardinia). A photo of this notch near the present sea level has been published by Antonioli et al. (2007, Fig. 12.6) with the caption "A well-developed tidal notch at Orosei Gulf, Sardinia, Italy". However, if it was a tidal notch, its position at the present sea level, after the recent sea-level rise, would have been abnormal. It seems that it was not taken into account that the almost horizontal base is capped, by an organic platform up to 10–20 cm thick; it is made, according to Carobene (1972), mainly by Serpulids and Lithotamnia, with the presence also of *Patella caerulea*, *Chama gryphoides* (L.), *Barbatia barbatia*, *Chthamalus stellatus* and *Balanus* sp. This notch is therefore not a tidal one, but a surf notch, in which the recent sea-level rise has been absorbed by a thickening of the organic accretion.

The situation is also different for solution notches, where a freshwater spring undercuts a limestone cliff at sea level. Accompanying a rise in sea level, dissolution by freshwater will tend to displace the roof of the notch that protrudes above the waterline continuously upwards, while the base of the notch, dissolved, will tend to be missing. For this isolated roof of a solution notch, protruding above the waterline (Fig. 12.12), the term "visor" has been proposed by Evelpidou et al. (2011b).

For marine terraces, the case of a small sea-level rise can probably be neglected, but for beachrocks the problem is completely different, especially in microtidal areas, because the upper level of beachrock cementation, which is the most significant sea-level indicator, will have been displaced upwards.

For biological sea-level indicators, a gradual upwards displacement of the biological vertical zonation should be taken into account.

In addition, for sea-level indicators deduced from archeological remains related to constructions in the sea (like harbors or fish tanks), where the biological marine

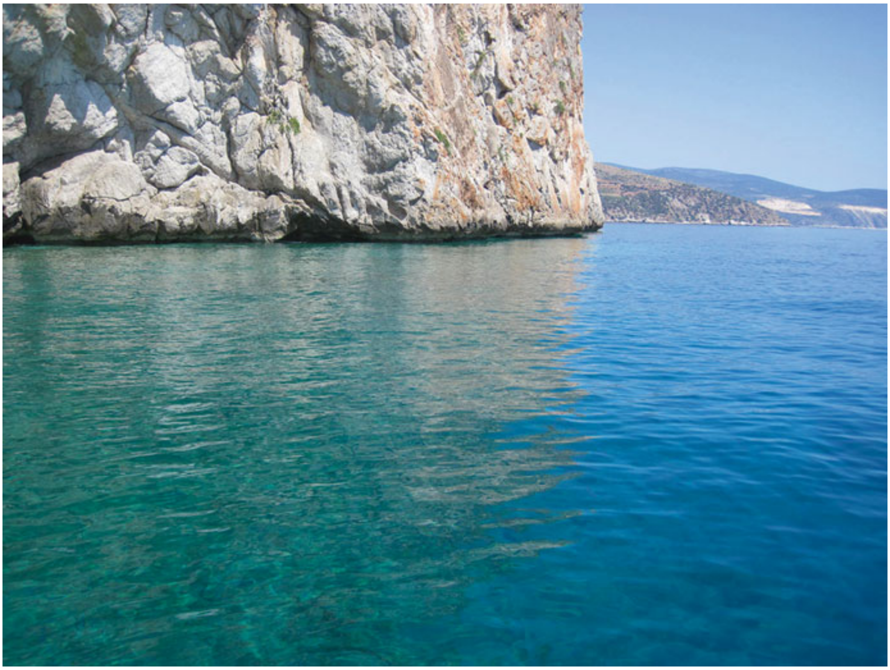


Fig. 12.12 A visor developed by the dissolution of freshwater which tends to displace the roof of the notch, at North Corinth Gulf

zonation may leave fossils related to former sea levels, the impacts of a small sea-level rise should be taken into account, while this rise can most often be neglected in archeological remains unrelated to coastal activities.

12.4.2 Foreseeable Impacts of the Predicted Near-Future Sea-Level Rise on the Coasts of NE Italy

The coasts of northeastern Italy in Roman times have been described by Pliny as a continuous, almost impassable sequence of lagoons, marshes and estuaries. The Po delta had not yet developed at that time and the openings of the various branches of this river to the sea were called the Septem Maria (seven seas) area. Most of the marshes have subsequently been filled by sediments or drained by man. Among the lagoon basins remaining today, the largest ones are the Lagoons of Marano and Caorle (the Grado Lagoon developed in late Roman times), the Lagoon of Venice, the Comacchio fisheries ("Valli") and the salt marshes ("Saline") of Cervia.

Due to the Sirocco, a persistent wet wind blowing from SSE (140–180°) along the longitudinal axis of the Adriatic, especially in the autumn and winter seasons, sea water tends to be channeled between the Apennines and the Dinaric Alps and

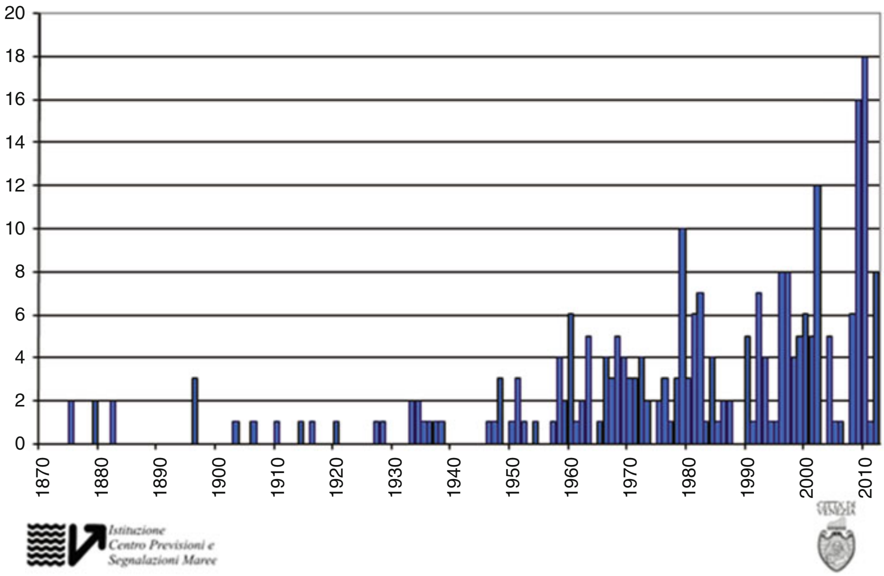


Fig. 12.13 Annual frequency (number of flooding events = tides >1.10 m a. the local datum) in Venice, from 1872 to 2012). At the level of 1.10 m, 14 % of the city if flooded

pushed northwards, producing sea-level surges in all the Adriatic, with maximum values in its northern part, from where there is no possible way out (Pirazzoli and Tomasin 2008).

Flooding events, often reported in coastal areas, may result from sea-level surges or from river overflow. Great sea-level surges are generally related to deep atmospheric depressions and are often accompanied by heavy rain and in certain cases by river flood; therefore, during the most devastating coastal floods, it is usually difficult to distinguish the boundary between water coming from the sea from that of the river, even in inland areas.

Coastal areas of the NE Adriatic Sea, covering a surface of almost 2,400 km^2 along over 300 km of coast between Monfalcone and Cattolica, are depressed below sea level (Bondesan et al. 1995). Man-induced or natural subsidence has affected these areas, especially near the Po delta area, where an altitude of over 2.5 m was lost in some places during the past century. They are therefore exposed to the risk of flooding by sea-level surges and rivers.

According to the summary for Policymakers of the IPCC AR5, near-future relative sea-level changes between 30 cm and 1 m (depending on the emission of greenhouse gases scenario assumed) are expected to occur during this century. Among the areas at risk, the lagoon of Venice represents specific problems for coastal engineering.

In the historical city of Venice, flooding at street level is a frequent phenomenon (Fig. 12.13). During the great flooding of 1966 the tide reached the level of 1.94 m above the local datum. A special Italian law, in 1984, suggested constructing adjustable barriers at the lagoon inlets. The barriers should have been "experimental, gradual and reversible". The study of the barriers has been committed, in monopoly, to a group of private companies (*Consorzio Venezia Nuova*) who proposed to construct a project called Experimental Electromechanical Module (MoSE). The project consists of 79 mobile gates at the three inlets of the lagoon (Fig. 12.13).

Each gate 20 m long would lie on the seabed during normal time but would be raised by injecting compressed air. The mobile gates are expected to be closed when the height of the tide threatens to reach the level of 110 cm above the local datum. Each gate will oscillate independently with waves. Thus narrow passages for water will have to remain open at all times and the barrier will not be watertight.

Each gate would be connected with a hinge to an enormous submerged concrete caisson. Below the caisson, thousands of foundation stakes, several dozen meters long, would be capped by a continuous concrete slab across the inlet. Indeed, such a huge construction could not be "gradual and reversible", as requested by the law of 1984.

A commission of the Italian Ministry of the Universities and Research (MURST) estimated in 1999 the following scenarios of relative sea-level rise for the year 2100 in Venice: most probable: 16.4 cm; prudent (recommended for the MoSE project): 22 cm; pessimistic: 31.4 cm. These underestimations, which were ignoring the work of international sea-level experts (e.g. IPCC) constitute a wrong basic assumption for the MoSE project. They were immediately denounced (Pirazzoli 2002), but all criticism was ignored.

The foundation stone for the MoSE project was set in 2003 by the Berlusconi government, in spite of the opposition of the Municipality of Venice and the strong negative impacts feared for the environment.

The MoSE gates cannot face a sea-level rise because they do not form a watertight barrier. Their oscillations with waves will enlarge the spaces between the gates and will permit sea water to raise the lagoon level even when the gates are closed. Rainfall and river discharges will also raise the lagoon level. This has been demonstrated by simulating the occurrence of certain storms of the past with the flood gates assumed to be fully operational (Pirazzoli 2002; Pirazzoli and Umgiesser 2006). It was shown that problems would start for a sea-level rise of about 20–30 cm, or even today with a repetition of the 1966 event.

In 2006 a hydrodynamic study commissioned by the Municipality of Venice to the French Company PRINCIPIA R.D., has shown that for certain steep wave conditions (Hs = 3.2 m; TP = 8 s), not rare in the area, an unstable behavior is obtained for the MoSE gates (https://www2.comune.venezia.it/mose-doc-prg/->4.2). In these conditions of resonance, the flow of sea water into the lagoon through the gates would increase to a level that cannot be specified by modeling and the gates could even overturn. The MoSE experts ignored the results of this study and did not try to explain how the resonance phenomena could be avoided.

In short, the MoSE Project, for which the heart of the construction is just starting, is inadequate to the safeguard of Venice. If it is completed (at best in 2016), it will be necessary to demolish it shortly after its construction. It is therefore urgent that the Italian Government considers realistically the limits of this project and studies the possibilities, when flooding would become unavoidable, of conversion of what has already been constructed into the rapid building of a permanent water-tight dyke separating the lagoon from the sea.

References

Alessio M, Allegri L, Antonioli F, Belluomini G, Improta S, Manfra L, Preite M (1996) La curva di risalita del Mare Tirrenonegliultimi 43 karicavata da datazionisuspeleotemisommersi e datiarcheologici. Memorie Descrittive del Servizio Geologico Nazionale 52:235–256

Antonioli F, Oliverio M (1996) Holocene sea-level rise recorded by a radiocarbon-dated mussel in a submerged speleothem beneath the Mediterranean sea. Quat Res 45:241–244

Antonioli F, Bard E, Potter EK, Silenzi S, Improta S (2004) 215-ka history of sea-level oscillations from marine and continental layers in Argentarola Cave speleothems (Italy). Glob Planet Chang 43(1–2):57–78

Antonioli F, Anzidei M, Lambeck K, Auriemma R, Gaddi D, Furlani S, Orrù P, Solinas E, Gaspari A, Karinja S, Kovačić V, Surace L (2007) Sea-level change during the Holocene in Sardinia and in the Northeastern Adriatic (central Mediterranean Sea) from archaeological and geomorphological data. Quat Sci Rev 26:2463–2486

Auriemma R, Solinas E (2009) Archaeological remains as sea level change markers: a review. Quat Int 206:134–146

Baker RGV, Haworth RJ (1999) Evidence for the nature of late Holocene sea level fall on the New South Wales coast from fixed biological indicators: was the fall smooth or fluctuating. In: Kesby JA, Stanely JM, McLean RF, Olive LJ (eds) Geodiversity readings in Australian geography at the close of the 20th century, Special publications 6. Institute of Australian Geographers, Canberra, pp 351–360

Bard E, Antonioli F, Silenzi S (2002) Sea-level during the penultimate interglacial period based on a submerged stalagmite from Argentarola cave (Italy). Earth Planet Sci Lett 196(3–4): 135–146

Bondesan M, Castiglioni GB, Elmi C, Gabbianelli G, Marocco R, Pirazzoli PA, Tomasin A (1995) Coastal areas at risk from storm surges and sea-level rise in Northeastern Italy. J Coast Res 11(4):1354–1379

Bricker OP (1971) Introduction: beachrock and intertidal cements. In: Bricker OP (ed) Carbonate cements, vol 19, Studies in geology. Johns Hopkins Press, Baltimore, pp 1–3

Carobene L (1972) Osservazioni sui solchi di battenteattualiedantichinel Golfo di Orosei in Sardegna. Boll Soc Geol It 91:583–601

Chappell J, Ota Y, Berryman K (1996) Late quaternary coseismic uplift history of Huon Peninsula, Papua New Guinea. Quat Sci Rev 15:7–22

Columella, De Re Rustica, XVII, 1–11 (Libb. VIII-IX, ed. Ǻ. Josephson (Coll. Script. vet. Upsaliensis), Upsaliae 1955)

Desruelles S, Fouache E, Ciner A, Dalongeville R, Pavlopoulos K, Kosun E, Coquinot Y, Potdevin J-L (2009) Beachrocks and sea level changes since Middle Holocene: comparison between the insular group of Mykonos–Delos–Rhenia (Cyclades, Greece) and the southern coast of Turkey. Glob Planet Chang 66(1–2):19–33

Evelpidou N, Pirazzoli PA, Vassilopoulos A, Tomasin A (2011a) Holocene submerged shorelines on Theologos area (Greece). Z Geomorphol 55:31–44

Evelpidou N, Pirazzoli PA, Saliège J-F, Vassilopoulos A (2011b) Submerged notches and doline sediments as evidence for Holocene subsidence. Cont Shelf Res 31:1273–1281

Evelpidou N, Kampolis I, Pirazzoli PA, Vassilopoulos A (2012) Global sea-level rise and the disappearance of tidal notches. Glob Planet Chang 92–93:248–256

Evelpidou N, Melini D, Pirazzoli PA, Vassilopoulos A (2013a) Evidence of repeated late Holocene rapid subsidence in the SE Cyclades deduced from submerged notches. Int J Earth Sci. doi:10.1007/s00531-013-0942-0

Evelpidou N, Koutsomichou I, Pirazzoli PA (2013b) Evidence of late Holocene subsidence events in Sporades Islands: Skopelos and Alonnisos. Cont Shelf Res 69:31–37

Flemming NC (1979–1980) Archaeological indicators of sea level. In: Les Indicateurs de Niveaux Marins. Oceanis 5, Institut Océnographique, Paris, Fascicule Hors-Série: 149–166

Focke JW (1978a) Limestone cliff morphology and organism distribution on Curaçao (Netherlands Antilles). Leidse Geol Meded 51:131–150

Focke JW (1978b) Limestone cliff morphology on Curaçao (Netherlands Antilles), with special attention to the origin of notches and vermetid coralline algal surf benches. Z Geomorphol 22:329–349

Higgins CH (1980) Nips, notches, and solution of coastal limestone: an overview of the problem with examples from Greece. Estuar Coast Mar Sci 10:15–30

Hopley D (1986) Beachrock as a sea-level indicator. In: van de Plassche O (ed) Sea-level research. Geo Books, Norwich, pp 157–173

Jevrejeva S, Moore JC, Grinsted A, Woodworth PL (2008) Recent global sea level acceleration started over 200 years ago? Geophys Res Lett 35:L08715, http://dx.doi.org/10./1029/2008GL033611

Kázmér M, Taboroši D (2012) Bioerosion on the small scale – examples from the tropical and subtropical littoral. Hantkeniana 7:37–94

Kelletat D (2006) Beachrock as sea-level indicator? Remarks from a geomorphological point of view. J Coast Res 22(6):1555–1564

Kemp AC, Horton BP, Donnelly JP, Mann ME, Vermeer M, Rahmstorf S (2011) Climate related sea-level variations over the past two millennia. Proc Natl Acad Sci 108:11017–11022

Kempf M, Laborel J (1968) Formations de Vermets et d'Algues calcaires des côtes du Brésil. Recueil des Travaux de la Station Marine d' Endoume 43:9–23

Laborel J (1979–80) Les Gasteropodes Vermetides: leurutilisation comme marqueurs biologiques fossiles. Oceanis 5 Fasc Hors-Sér: 221–239

Laborel J (1986) Vermetids. In: van de Plassche O (ed) Sea-level research: a manual for the collection and evaluation of data. Geo Books, Norwich, pp 281–310

Laborel J, Laborel-Deguen F (2005) Sea-level indicators, biologic. In: Schwartz ML (ed) Encyclopedia of coastal science. Springer, Dordrecht, pp 833–834

Laborel J, Morhange C, Collina-Girard J, Laborel-Deguen F (1999) Littoral bioerosion, a tool for the study of sea-level variations during the Holocene. Bull Geol Soc Den 45:164–168

Lundberg J, Ford DC (1994) Late Pleistocene sea level change in the Bahamas from mass spectrometric U-series dating of submerged speleothems. Quat Sci Rev 13(1):1–14

Mourtzas ND (2012) A palaeogeographic reconstruction of the seafront of the ancient city of Delos in relation to Upper Holocene sea level changes in the central Cyclades. Quat Int 250:3–18

Pirazzoli PA (1986) Marine notches. In: Van de Plassche O (ed) Sea-level research: a manual for the collection and evaluation of data. Geo Books, Norwich, pp 361–400

Pirazzoli PA (2002) Did the Italian Government approve an obsolete project to save Venice? EOS Trans Am Geophys Union 83(20):217–223

Pirazzoli PA (2007) Sea level studies; geomorphological indicators. In: Elias SA (ed) Encyclopedia of quaternary science. Elsevier, Oxford, pp 2974–2983

Pirazzoli PA, Tomasin A (2008) Sea level and surges in the Adriatic Sea: recent trends and possible near-future scenarios. Atti Ist Ven Sci Lett Arti CLXVI:61–83

Pirazzoli PA, Umgiesser G (2006) The projected «MoSE» barriers against flooding in Venice (Italy) and the expected global sea-level rise. J Mar Environ Eng 8:247–261

Pirazzoli PA, Thommeret J, Thommeret Y, Laborel J, Montaggioni LF (1981) Les rivages emerges d'Antikythira (Cerigotto): corrélations avec la Crète Occidentale et implications cinématiques et géodynamiques. Actes du Colloque «Niveaux marins et tectonique quaternaire dans l'aire Méditerranéenne», CNRS & Universite de Paris I, pp 49–65

Pirazzoli PA, Arnold M, Giresse P, Hsieh ML, Liew PM (1993) Marine deposits of late glacial times exposed by tectonic uplift on the east coast of Taiwan. Mar Geol 110:1–6

Plinius, Naturalis Historia, IX, 168 (Libb.VII-XV, ed. L.Jan-C. Mayhoff (Teubner), Lipsiae 1909)

Rampino MR (2005) Erosion processes. In: Schwartz ML (ed) Encyclopedia of coastal science. Springer, Dordrecht, pp 432–433

Stattegger K, Tjallingii R, Saito Y, Michelli M, TrungThanh N, Wetzel A (2013) Mid to late Holocene sea-level reconstruction of Southeast Vietnam using beachrock and beach-ridge deposits. Glob Planet Chang 110:214–222, http://dx.doi.org/10.1016/j.gloplacha.2013.08.014

Taboroši D, Kázmér M (2013) Erosional and depositional textures and structures in coastal karst landscapes. In: Lace MJ, Mylroie J (eds) Coastal karst landforms. Coastal research library 5. Springer, Berlin

Tatumi SH, Kowata EA, Gozzi G, Kassab LRP, Suguio K, Barreto AMF, Bezerra FHR (2003) Optical dating results of beachrock, eolic dunes and sediments applied to sea-level changes study. J Lumin 102–103:562–565

Varro, De Re Rustica III, 3, 2 (Ed. Keibel-Goetz (Teubner), Lipsiae 1929)

Chapter 13
Advancement of Technology for Detecting Shoreline Changes in East Coast of India and Comparison with Prototype Behaviour

Ramasamy Manivanan

Abstract Developments in the coastal area have significant impact on the adjacent shorelines. Mathematical modelling provides a useful tool for predicting such changes in shorelines in advance. Processing and analysis of satellite imageries of coastal area enables us to estimate and monitor the shoreline changes, which is otherwise extremely difficult, time consuming and costly by field surveying. In this paper the shoreline changes obtained by mathematical modelling and by image processing technique are compared by applying these techniques to shoreline adjacent to Ennore region. The study indicated that the cross-shore and longshore impact predicted by mathematical model and satellite information match satisfactorily. Thus the satellite information is useful for calibrating the mathematical model which can be further used for predictive purposes.

13.1 Introduction

The coastline is an interface between the land and sea where winds, waves, tides and currents attack the land. The land responds to this attack by dissipating the energy of the sea and changing the shape and alignment of the coastline continuously. Waves play a prominent role in the nearshore processes. The action of waves is the principal cause for changes in the shorelines.

Human interventions into the natural processes taking place in the coastal zone have an impact on the shorelines. These are construction of ports and coastal

R. Manivanan (✉)
Mathematical Modeling for Coastal Engineering (MMCE), Central Water and Power Research Station, Khadakwasla, Pune 411 024, India
e-mail: vananrmani@rediffmail.com

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8,
DOI 10.1007/978-3-319-06305-8_13,

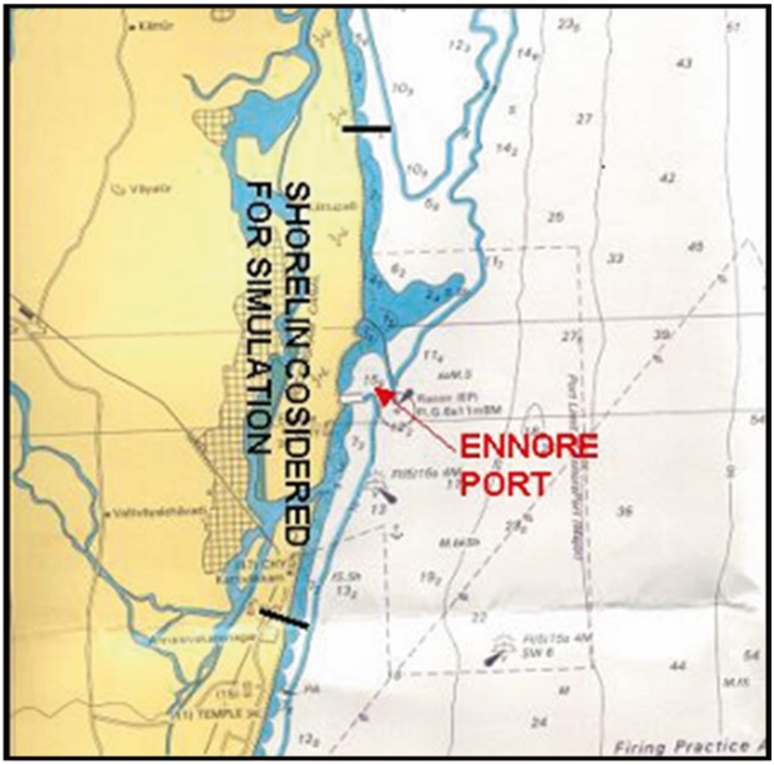


Fig. 13.1 Location of Ennore region

structures, dredging, mining etc. In order to anticipate the impact and suggest remedial measures to minimize the adverse effects, it is desirable to predict the changes in the shorelines before undertaking the developments. Mathematical modelling provides a useful tool for predicting such changes in the shorelines.

Satellite remote sensing technique is a very useful tool for studies regarding coastline changes. It can provide synoptic view in the form of satellite imageries of coastal area at different time instants. Processing and analysis of these imageries enables us to estimate, monitor and understand the shoreline changes, which is otherwise extremely difficult, time consuming and costly by field surveying.

This study is carried out to demonstrate the usefulness of the satellite derived information for mathematical modelling of the shoreline changes. The coastal site around Ennore region is considered for this study (Fig. 13.1). Shoreline changes due to the development of the port are simulated by mathematical modelling techniques and compared with the shoreline changes derived from remote sensing/ image processing techniques. The methodology and findings of this study is described below.

13.2 Development of Port at Ennore

It is located on the east coast of India about 20 km north of Chennai port. It was constructed in 1999 to meet the increasing demand of cargo. It is in operation since 2001 with coal as major cargo. The port is sheltered by northern breakwaters of 3.2 km length and southern breakwaters of 1.1 km length with entrance from south-east direction. Since the region is located in highly sensitive zone of littoral drift, the development of the port was expected to cause significant impact on the coastline. After the construction of the port, the southern shoreline has witnessed accretion due to the southern breakwater and also due to presence of Ennore creek while the northern coastline has experienced severe erosion.

13.3 Site Conditions

For simulation of littoral drift and shoreline changes, data on bathymetry, tidal levels, currents, waves and sediments at the site are required. The site at Ennore is near Chennai where the tidal levels MHWS, MHWN, MSL, MLWN and MLWS are 1.1 m, 0.8 m, 0.6 m, 0.4 m and 0.1 m respectively. Tidal currents are unidirectional, northward during SW and non-monsoon period and southward during NE monsoon period with average magnitude of the order of 0.1–0.2 m/s. The major data required for simulation of shoreline changes is the nearshore wave climate at the site.

Instrumentally observed wave data at the site over a period of several years, if available, are best suited for design purposes. As such data were not available, the ship observed deep water wave data reported by India Meteorological Department (IMD) covering a period of about 30 years for quadrant between latitude 10–150° N and longitude 80–85° E were analysed to obtain seasonal and annual wave climates in the offshore region of Ennore. It is seen from the annual wave climate (Fig. 13.2a) that the predominant wave direction is from SW and NW quadrants with maximum wave height of the order of 4.5 m. This wave climate in the deep sea was transformed to get the nearshore wave climate at Ennore using a mathematical model which is described subsequently. The orientation of the coastline in this region of Ennore is approximately in North-South direction.

Littoral drift rates near the site at Ennore were computed by various researchers. Chandramohan et al. (1990) estimated northward drift of 1.027×10^6 m^3 and southward drift of 0.683×10^6 m^3 with net drift towards north to be 0.344×10^6 m^3 along Madras coast. Natesan and Subramanian (1994) indicated that along the coastal stretches of Grid I (Lat. 9–12 N, Long. 80–83 E) and Grid II (12–15 N, 81–84 E), a net transport of 0.35–0.4 million m^3 of sand towards north occurs. Indomer (2005) estimated northward drift of 0.977313×10^6 m^3 and southward drift of 0.512757×10^6 m^3 with net drift towards north to be 0.464556×10^6 m^3 along Minjur coast. Shoreline studies were carried out by ICMAM (2007) in which the net transport towards north at Ennore coast was assumed to be 0.35×10^6 m^3.

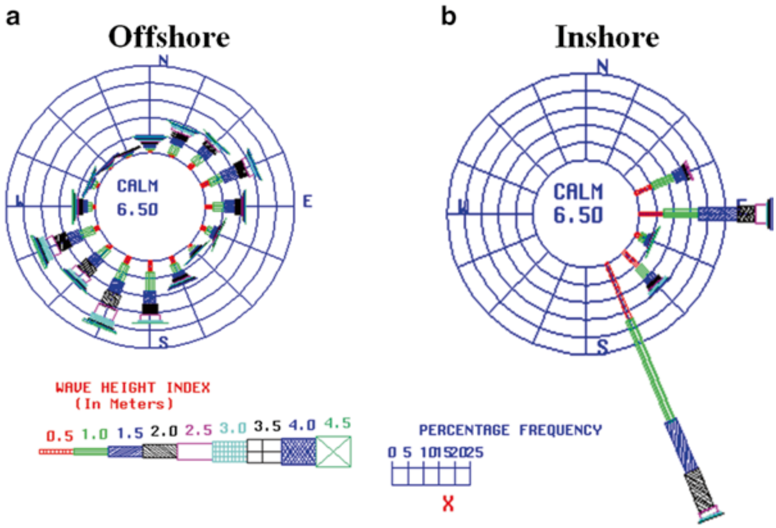


Fig. 13.2 Rose diagram for wave heights for annual period

From the above discussions it is seen that there is considerable variation in the estimation of littoral drift in the region due to the uncertainties involved. However, in general, the annual rate of northward and southward transport at the Chennai coastline are of the order of 0.6 and 0.1 million m^3 respectively. Hence the average northward and southward transports for Ennore coastline are considered to be 0.60 and 0.1 million m^3 so that net and gross transports are 0.5 and 0.7 million m^3.

13.4 Transformation of Waves

The wave climate prevailing at the site is one of the important parameters for estimating littoral drift and shoreline changes which is derived by wave transformation from deep to shallow waters using OUTRAY (1989) model. This model takes the deep water wave height, period and direction as input and computes the wave height, period and direction at the inshore point of interest. The wave height and period are input to the model by specifying the spectral distribution of wave energy at the offshore boundary. The offshore wave climate described earlier was transformed to the inshore location near Ennore in 12 m contour to obtain the seasonal and annual inshore wave climates. It is seen that during SW monsoon (June to Sept.), the waves approach predominantly from SSE direction with 90 % occurrence. During NE monsoon (Oct. to Jan) the predominant wave directions are East, and SSE at the nearshore location with percentages 48 % and 29 % respectively. In non-monsoon period (Feb. to May), the predominant wave directions are East, ESE, SE and SSE with percentages 23 %, 17 %, 15 % and 40 % respectively. The maximum wave height of 3.5 m is seen to occur. The annual frequency distribution of waves at the inshore location near Ennore is shown as rose diagram in Fig. 13.2b which indicates that the waves approach predominantly from East and SSE directions with percentages 25 % and 48 % respectively and waves up to 4.0 m are seen.

13.5 Simulation of Littoral Drift and Shoreline Changes

For these studies, LITPACK (2000) software was used to simulate the longshore transport and impact of the construction of Ennore port. Computation of longshore sediment transport is based on the local wave, current and sediment characteristics. The sediment transport model takes into account time-varying distribution of both suspended load and bed load in combined wave and current motion including the effect of wave breaking, when relevant. The model gives a deterministic description of cross-shore distribution of longshore sediment transport for an arbitrary, non-uniform bathymetry and sediment profile. The seasonal/annual sediment budget is found by the contribution of transport from each of the wave incidents occurring during the season/year. For simulation of shoreline evolution, the following continuity equation for sediment volumes is numerically solved.

$$\frac{\partial y_c}{\partial t} = -\frac{1}{h_{act}(x)}\frac{\partial Q(x)}{\partial x} + \frac{Q_{sou}(x)}{h_{act}(x)\Delta x} \tag{13.1}$$

where, y_c = distance from the base line to the coastline, h_{act} = height of the active cross-shore profile, Q = longshore transport of sediment expressed in volume, x = longshore position, Δx = longshore discretisation step, Q_{sou} = source/sink term expressed in volume. Thus the term Q_{sou} can be conveniently used to include sand bypassing arrangement.

13.5.1 Littoral Drift Distribution

The LITPACK (2000) model assumes the depth contours parallel to the coast and longshore transport is computed along a representative cross-shore profile. The longshore transport was computed for 2.5 km long bed profiles normal to shore with depth varying from +2.9 to −15 m. The profiles were divided into 250 grid points with grid size of 10 m. For these studies, besides the inshore wave climate and bathymetry along the cross-shore profile, grain size distribution over the profiles is also required. At the site grain size was observed to be of the order of 0.2 mm Hence grain size D50 over the profiles was assumed to be 0.2 mm.

Since the general orientation of the coastline at Ennore is North-South, depending on the wave direction with respect to the shoreline, the littoral drift will be

Table 13.1 Seasonal and annual transport rates (million cu m)

Profile	Season	Northward	Southward	Net	Gross
South profile	SW monsoon	–0.34	0.00	–0.34	0.34
	NE monsoon	–0.11	0.07	–0.04	0.18
	Non-monsoon	–0.16	0.02	–0.14	0.18
	Annual	–0.61	0.09	–0.52	0.70
North profile	SW monsoon	–0.29	0.00	–0.29	0.29
	NE monsoon	–0.13	0.08	–0.05	0.21
	Non-monsoon	–0.15	0.02	–0.13	0.17
	Annual	–0.57	0.10	–0.47	0.67

Note: +ve sign indicates southward transport, –ve sign indicates northward transport

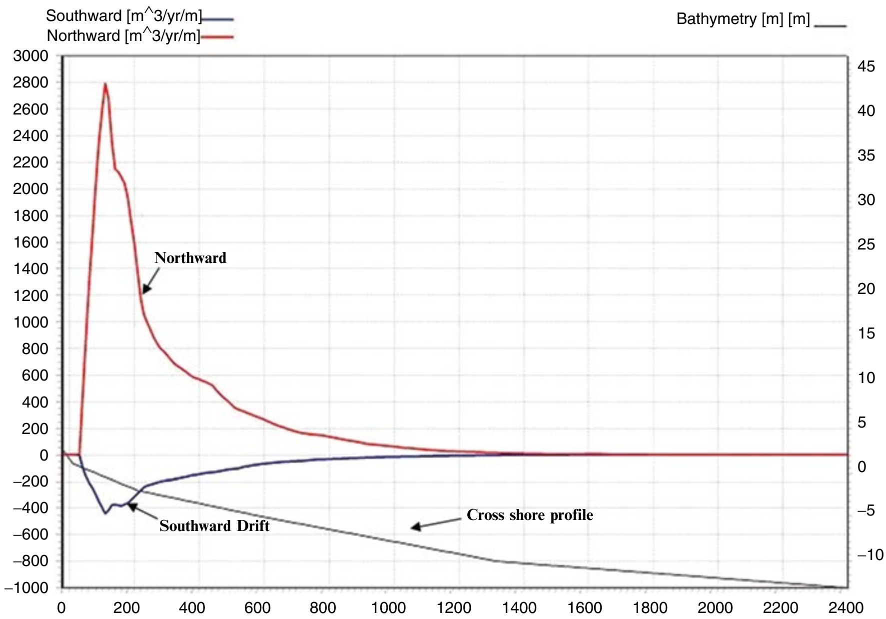


Fig. 13.3 Distribution of littoral drift during annual period

directed towards north or south along the shoreline. The model was run for the two cross-shore profiles for seasonal and annual inshore wave climates. The model was calibrated for the net and gross annual transport of 0.70 and 0.50 million cum and the seasonal/annual northward, southward, net and gross transport quantities computed are given in Table 13.1.

The distributions of seasonal and annual transports over the profile are shown in Fig. 13.3 which indicate that the transport is mainly confined within 400 m from the shore. The maximum transport occurs at about 50–70 m from the shore.

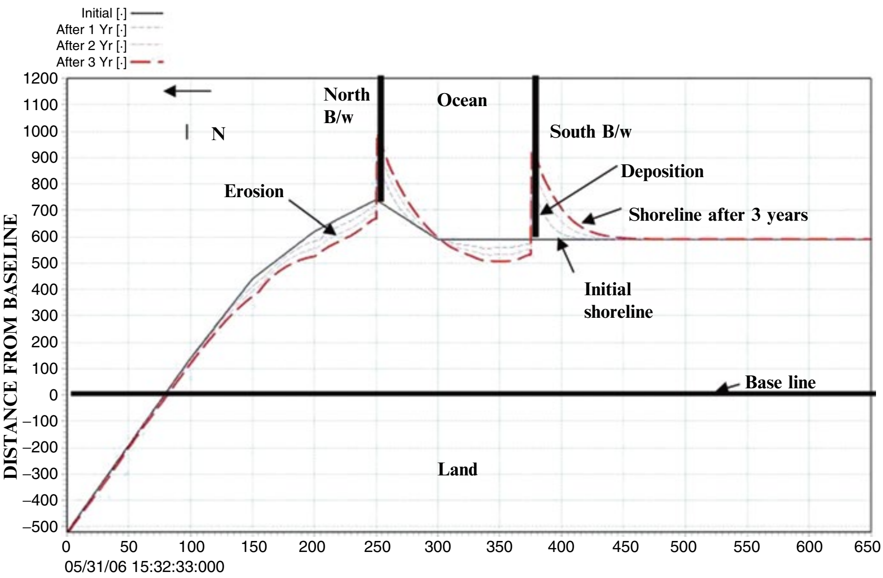


Fig. 13.4 Shoreline changes after 1, 2, 3 years

13.5.2 Shoreline Changes

For the shoreline evolution model, shoreline of 14.4 km length extending 6 km towards north and 6 km on the south of Ennore region was considered. This was divided into 721 grid points with grid size of 20 m. The port with 3.2 km long north breakwater and 1.1 km long south breakwater was considered in the model. The effective blocking lengths normal to shoreline of the north and south breakwaters are considered to be 1,500 and 1,000 m. With these conditions the shoreline evolution model is run with the schematic layout of the breakwaters for 3 years period. The shorelines evolved after each year is shown in Fig. 13.4.

As the net transport is towards north, the coastline on the south side of the south breakwater has advanced and the coastline on the north side of the north breakwater has receded. The maximum cross-shore advancement on the south side is about 300 m after 3 years and its longshore accretion effect on south side of south breakwater is felt for about 1.5–2 km. The maximum cross-shore recession on the north side is about 100 m after 3 years and its longshore erosion effect on north side of north breakwater is felt for about 2 km.

13.6 Shoreline Changes by Image Processing

Remote sensing satellite records the features of earth's environment in the form of spectral response without physically coming in contact with the earth's surface. The electro-magnetic energy in the discrete bands of different wavelengths of electro-magnetic spectrum reflected or radiated by different objects enables identification of different objects, their locations, spatial distribution and helps in obtaining their properties. The advantage of remote sensing satellites is their capability to map regions including coastal, oceanic and land features like wetlands, intertidal zones etc., at regular interval of time. The temporal data obtained from remote sensing satellite can effectively be used to assess the dynamic changes in the shorelines by delineation of water and land. The infrared band in the spectral range of 0.77–0.86 μm is found to be suitable for demarcation of shoreline as the contrast between land and water is very sharp. Multi-date digital satellite images can be compared to detect any changes in the shoreline using digital satellite image processing software. While comparing two imageries of different dates, it is important to ensure that both the imageries are at similar tidal condition.

In order to evaluate the shoreline, satellite data sets of two dates covering a period of 3 years were selected. The details of the imageries selected for this study are listed in the following table.

Sl. no.	Satellite	Sensor	Date	Path	Row	Spatial resolution (m)
1	IRS-1D	LISS-III	12.02.2001	102	064	23.8
2	IRS-1D	LISS-III	17.04.2004	102	064	23.8

The imageries are in digital form having visible and infrared spectral bands. Subset imageries bounded by latitude 13.2795 and longitude 80.3574 were used for the analysis of shoreline changes. The images of shorelines of 2001 and 2004 near Ennore region as viewed by LISS-III sensor of IRS-1D satellite are compared.

13.7 Comparison of Shoreline Changes Obtained by Mathematical Model and Image Processing

The shoreline changes obtained by mathematical modelling and image processing are shown in Fig. 13.5 for comparison. It can be seen that the cross-shore and long-shore shoreline changes obtained by the model and image processing match satisfactorily. With further refinement in the model results by tuning model parameters and analysis of additional imageries, the matching of the two results can be improved.

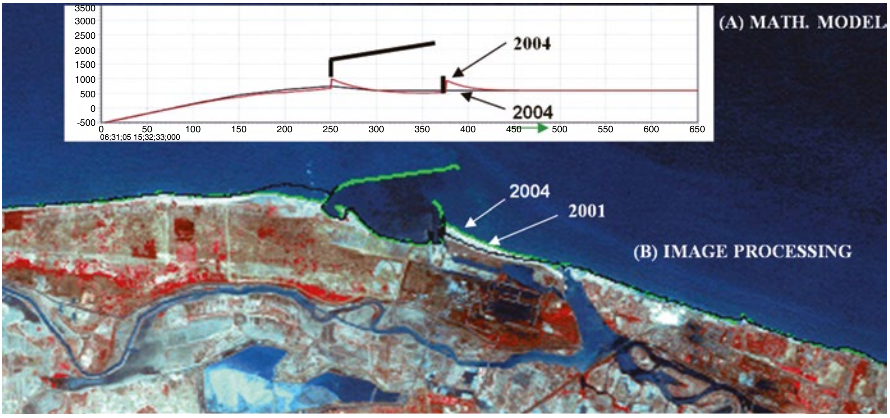


Fig. 13.5 Comparison of shoreline changes obtained by mathematical model

13.8 Conclusion

Human interventions into the natural processes taking place in the coastal zone such as construction of ports has significant impact on the shorelines. In order to assess the effect of such developments and suggest remedial measures in anticipation, it is desirable to predict the shoreline changes in advance. Mathematical modelling provides a useful tool for predicting such changes in advance. Processing and analysis of satellite imageries of coastal area enables us to estimate and monitor the shoreline changes, which is otherwise extremely difficult, time consuming and costly by field surveying. Satellite remote sensing is a very useful technique for studies regarding coastline changes. Processing and analysis of satellite imageries of coastal area enables us to estimate and monitor the shoreline changes, which is otherwise extremely difficult, time consuming and costly by field surveying.

Shoreline changes due to the development of Ennore port are simulated by mathematical modelling technique and compared with the shoreline changes derived from remote sensing/image processing technique. The study indicated that the cross-shore and longshore impacts predicted by mathematical model and satellite information match satisfactorily. Thus the satellite information is useful for calibrating the mathematical model which can be further used for predictive purposes. With further refinement in the model results by tuning model parameters and analysis of additional imageries, the model could be improved.

Acknowledgement Authors are thankful to Shri. S. Govindan Director, Central Water and Power Research Station, Pune for kind consent for publishing this paper.

References

Chandramohan P, Nayak BU, Raju VS (1990) Longshore sediment model for south Indian and Shri Lankan coasts. J Waterw Port Coast Ocean Eng Proc ASCE 116:408–424
ICMAM (2007) Shoreline management plan for Ennore coast, Tamil Nadu
Indomer Coastal Hydraulics (P) Ltd (2005) Rapid marine EIA for CWDL desalination plant at Minjur
LITPACK (2000) An integrated modelling system for littoral processes and coastal kinetics, Danish Hydraulic Institute
Natesan U, Subramanian SP (1994) Study of transport processes along Tamil Nadu coast. Proc Indian Natl Conf Harbour Ocean Eng (INCHOE-94) 1:D-103–D-108
OUTRAY (1989) Refraction model-training and user manual, hydraulics research, Wallingford

Chapter 14
Coastal Dunes: Changes of Their Perception and Environmental Management

Tomasz A. Łabuz

> *"There is no real ending. It's just the place where you stop the story."*
> *"There is no escape—we pay for the violence of our ancestor."*
>
> Frank Herbert, author of famous Dune saga SF stories

Abstract Human capacity to adapt to various living conditions has always been a challenge to every civilization. Since the dawn of time people have been transforming environment to make their living easier. In the coastal zone these changes occurred the earliest, stretching as far as sandy barriers covered by coastal dunes. On the low sandy shores, in the vicinity of river mouths first settlements, towns and seeds of countries were established. We inherited from our ancestors these areas in some sense already developed or used. Nowadays, in many places coastal areas are transformed to such extent that they no longer resemble the natural wild landscape, for which we yearn and which is necessary for humans to function healthily in their natural environment.

Learning more and more about the mutual relations among factors influencing dune habitats, we are constantly changing the coastal dunes practical use. We know that while implementing the changes we must have in mind their availability for future generations. Yet, in some parts of the world this has not been acknowledged so far. In other countries, with higher ecological awareness and with high degree of landscape transformation, have been making numerous attempts to restore the natural environment.

How do we currently use this environment? What do we know about it? What is its value for humans? Do we care about it enough? Or, do we need to do more to preserve coastal dune habitats for future generations?

T.A. Łabuz (✉)
Institute of Marine and Coastal Sciences, University of Szczecin,
Mickiewicz 18, 70-383 Szczecin, Poland
e-mail: labuztom@univ.szczecin.pl

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8,
DOI 10.1007/978-3-319-06305-8_14,

The chapter will briefly outline the coastal dune types and the conditions for their development. The second part is devoted to functions and practical use of the coastal dunes. It presents human economical activities based on coastal dunes values. In the third part today's aims and methods of current research are shown. Also, the changing attitudes to the environmental management of coastal dunes is discussed. In the last part various approaches to the use and the perception of coastal dunes, resulting from cultural otherness and stages of society development are examined.

14.1 Introduction

This chapter presents past and current methods of the use, research and environmental management of coastal dunes. It contains the overview of scientific publications that have appeared in the last 20 years, with a special focus on the period starting in 2005. Since that time a number of new studies have appeared that are taking into account the sea level rise and climate change that affect our environment, including coastal dunes. Furthermore, since that time a new measurement technologies and presentations of GIS data have been used in this subject. Over the past 10 years many conferences have been organized around the world that have been devoted exclusively to coastal dune issues. Some of them were of just national character, but they were among the first organized in their geographical regions. The international ones attracted scientists from around the world, who presented their latest findings and solutions in the field of coastal dunes management. Three of those conferences should be mentioned here because of a huge input of scientific papers: Dunes and Estuaries 2005 – International Conference on Nature Restoration Practices in European Coastal Habitats organized in Koksijde, Belgium in 2005; International Conference on Management and Restoration of coastal Dunes (ICCD) organized in Santander, Spain in 2007; and International Symposium on Aeolian Processes from 2011 organized in Eilat, Israel.

Some of them, such as organized every 2 years: International Coastal Symposium organized by Coastal Education and Research Foundation (CERF) and the Journal of Coastal Research (JCR) or Littoral, the major international conference of Coastal & Marine Union – EUCC, International Conference on Geomorphology (IAG) or International Conference on Aeolian Research (ICAR), have separate panels where processes and phenomena shaping coastal dunes, methods of environmental management and legal solutions for management are discussed. Those are the main subjects of present scientific concern about the coastal dune environment. After each thematic conference or panel there are dozens of valuable publications. The last few years is the time when several thematic publications have appeared on the market that are devoted solely to the management of coastal dune environments in general and are accompanied by case studies (Martínez and Psuty 2004; Doody 2013; Martínez et al. 2013) and academic books presenting the latest scientific achievements in this field (Bird 2003, new edition; Nordstrom 2000, 2008; Woodroffe 2002; Pye and Tsoar 2009; Hesp 2011).

14.2 Types of Coastal Sand Forms: The Review of Current State of Knowledge

Coastal dunes are developed on sandy barriers – i.e. higher origin forms – that accumulated in the past or have been formed nowadays and shaped by the sea. Such accumulative features as barriers and spits are formed by currents and waves. Dunes are developed as aeolian forms shaped by winds and influenced by a vegetation pattern (Figs. 14.1, 14.2, and 14.3). These amazing land structures may develop in conditions of permanent sand load from the sea in accumulative parts of the coast. The sand comes from erosion of other sections of the coast or is transported from the seabed or supplied by rivers to the sea. That is why dunes are typical coastal forms near river deltas or estuaries where there is enough sand.

The shape of the coastal dunes is a non-linear function of coast exposure to the predominant winds and waves, resulting in a coastal waters circulation exerting alongshore sediment transport causing accumulative or erosive tendencies in time

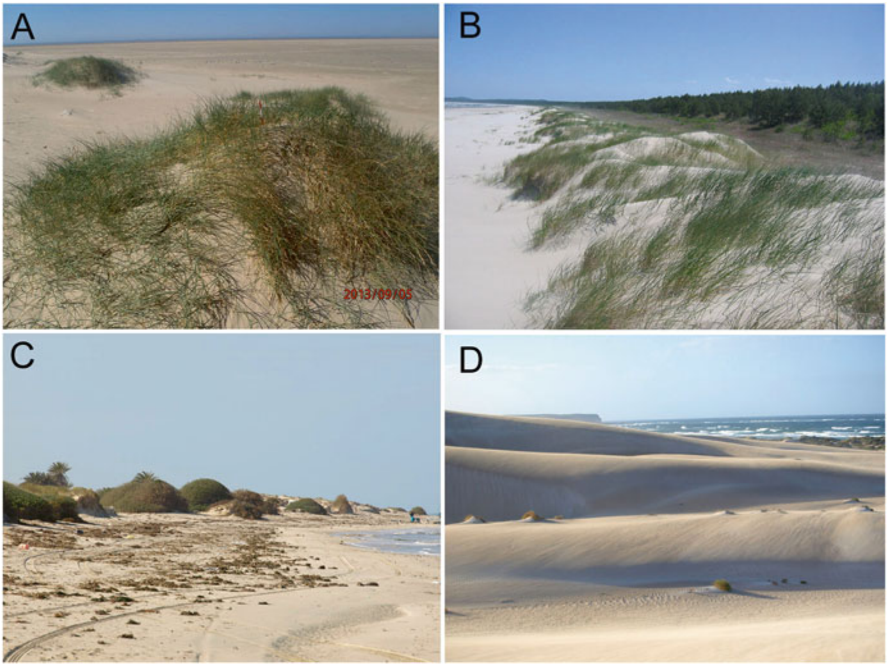


Fig. 14.1 Coastal dunes. (**a**) Embryo dunes (1 m high) on beach covered by *Elymus farctus*, Calais France. (**b**) Foredune ridge (5 m high) covered by *Ammophila arenaria*, behind runnel covered by heat and next ridge by pine forest (typical plants zonation on foredunes) Swina Gate Sandbar, Poland. (**c**) Coppice dunes, Djerba Island, Tunisia. (**d**) Shifting, transverse dunes (1.5 km wide), visible shifting sand during wind action and sparse vegetation, Lake Newland Conservation Park, Eyre Peninsula, South Australia

Fig. 14.2 Coastal sand forms. (**a**) Narrow sandspit (50 m wide and 0.5 km long), Otuwhero Inlet, Abel Tasman National Park, New Zealand (with environment information board). (**b**) Pocket beach, Devon, United Kingdom. (**c**) Barchan dune entering dune slack with small pine trees, Slowinski National Park, Poland. (**d**) Dunes encroaching rocky hill slopes, Kekerengu, New Zealand

(Fig. 14.3). The coastal dunes can be divided into two groups: typical coastal ridges of height to 30 m and inland sand bodies built by shifting dunes that may create coastal areas elevated up to 200 m. Their height and width is influenced by sand availability and coast development trends (types of material that build the dunes will be described in Sect. 14.3.1.).

14.2.1 Typical Coastal Dunes: Foredunes

The simplest classification of the coastal dunes includes morphological and dynamic characteristics of the dunes highlighted by the type of vegetation growing on them, which determines the stage of development of dunes (Hesp 2000, 2002, 2011; Woodroffe 2002; Bird 2003; Pye and Tsoar 2009).

A typical dune profile shows a dune ridge or ridges of a general height of 5–15 m amsl (above the mean sea level). To the back of such sandy belts there generally are lagoons, lakes or wetland lowland up to 3 m amsl. Under some favourable

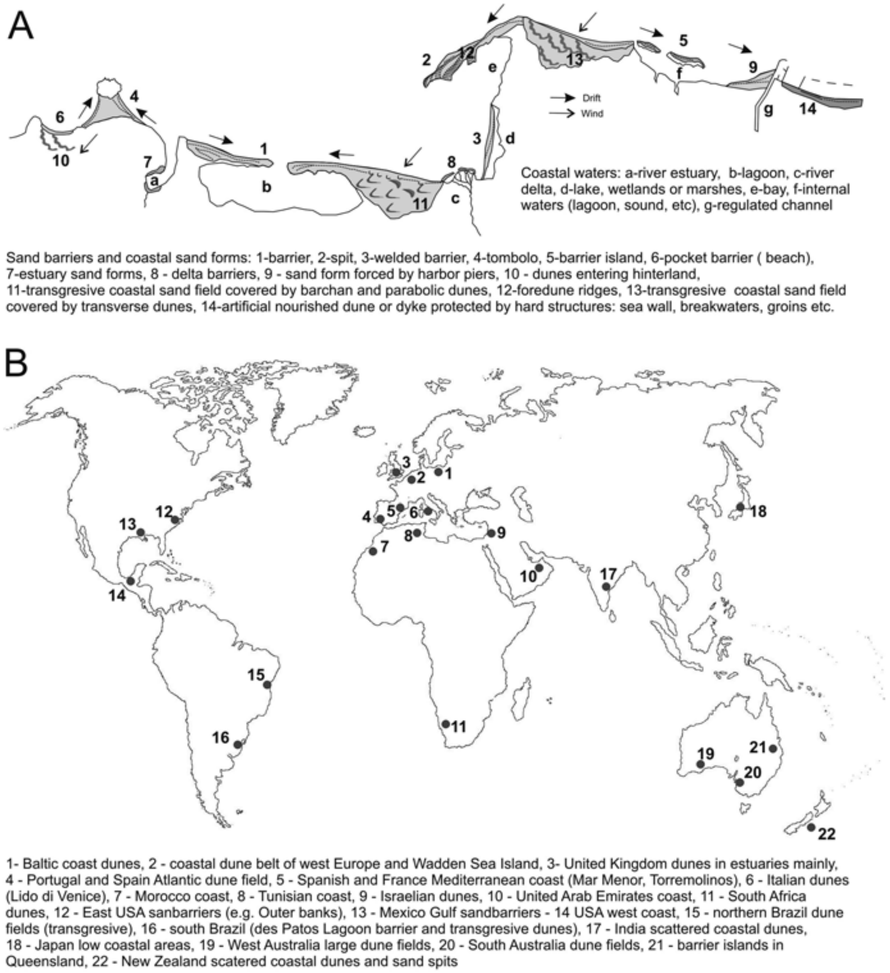


Fig. 14.3 Coastal sand forms. (**a**) Theoretical example of coastal sand forms distribution, (**b**) coastal dunes areas described in chapter

accumulation conditions the lowland behind the coastal dune belt is covered by wandering or stabilised inland dunes, mainly parabolic or barchans (e.g. Curonian Sandbar). At the front of the foredune is a beach, of the height up to 3 m, neighbouring a shallow shore face with one or several sandy shoals. The beach may be divided into a lower beach – mainly shaped by water, with smaller forms, such as water ridges and beach lagoons or micro-cliffs – and an upper beach shaped mainly by the wind, where embryo dunes appear, or by the sea during heavy storm surges.

The foredune is an aeolian structure developed at the rear of the beach as a parallel ridge along the coast (Hesp 2002, 2011), which may protect the lowland. An incipient foredune is an unstable dyke covered by pioneer grasses which grows until a new one appears in the form of embryo hillocks on the upper beach. The established foredunes are older ridges that are stabilised by plants of the next succession stages – these ridges are mainly stable forms, and due to erosion tendencies they may be eroded by sea and slightly reshaped by wind.

We can distinguish foredunes that at different development stages are called white, grey or brown depending on the development of soil processes associated with the age and type of vegetation cover. Foredunes are parallel to coastline, convex forms formed on the backshore and neighboring a sandy beach (Hesp 2002). This beach sand is an factor vital to their development. On the coast with more or less stable coastline just one foredune ridge may grow. The coast with positive sand balance on the beach may be built up by several foredeunes (Fig. 14.1b). In places, where the beach progradation takes place new foredunes are developing on the upper part of the beach. There, vegetation, initially rare but getting denser, stops the transported sand. On such a beach the first aeolian forms that may withstand being blown off and washed off are called embryo dunes (Fig. 14.1a). These incipient foredunes may grow above a high tide line or a storm range until they are destroyed by a severe storm overflowing the beach. When the upper part of the beach is higher than the typical storm surge range, the dunes on the beach embryo grow and may be transformed into a more or less asymmetric foredune that is shaped occasionally by water and mainly by winds (Fig. 14.1b, c).

On the coast with accumulative tendencies there are several foredune ridges in different development phase from white, not stabilized with pioneer plants and most frequently covered by grasses, through biodiverse dune heat to the ones naturally (or nowadays often artificially) covered by forest. There are different species of grasses that cover incipient foredunes around the world but the most cosmopolitan is Marram grass (*Ammophila arenaria*). This one, used as dune stabilization plant in Europe, was introduced in the nineteenth century in both Americas, Australia or in New Zealand.

On the coast with dominating erosion tendencies the foredune is slowly abraded and may retreat with the beach into the neighboring hinterland (Woodroffe 2002; Bird 2003; Hesp 2011). Low ridges of foredunes are often broken by storm surges. Such storm surge gates lead to the washover fan development in the foredune slacks or to a typical runnel dividing two ridges (Fig. 14.1b). This phenomenon is beneficial for wild stretches of the coast where bio and geodiversity increases in the coastal landscape. It should be mentioned here that in some European countries, where dunes become very stable and overgrown by the species of the next phases of succession, artificial breach-gates are created (Belgium, Netherlands, United Kingdom), (Fig. 14.23).

Foredunes with scarped slopes may be influenced by mass movements responsible for landslides. Moderate beach-dune erosion may result in removal of the whole vegetation and consequently lead to renewal of the aeolian processes on foredunes. So the thesis that storm surges help in the foredune development is true on coasts with rapid embryo dunes development that stops sediment, thus discontinuing its supply to a dune (Łabuz 2013). A foredune that is still developing is called an incipient dune while the one which is fully stabilized by plants with minor shifting parts is called an established dune. The rate of the foredune growth depends both on sand load and the aeolian accumulation that is predominant over abrasion and deflation. It may take 3–4 years for a new foredune to develop from the beach covered by embryo dunes fixed by sparse vegetation (called also nebkhas) as it was observed in many countries (Hesp 2002, 2011; Łabuz 2003).

In some conditions instead foredune ridges on the coast, may be form coppice or nebhka dunes. The morphology of these forms are shaped by clumps of plants. Such forms are mainly irregular and separated sand mounds of several meter height. Nebkhas or so called coppice dunes can be found throughout arid and semiarid regions (Fig. 14.1c). The present plant communities were evaluated on the coppice dunes and dune interspaces. For such forms the term hummock dunes are also used (Pye and Tsoar 2009).

14.2.2 Sandspits, Sand Barriers and Other Sandy Coastal Forms

The widest sand barriers including dunes are developed in a humid climate where river regime is able to discharge material from the land. On lowlands, plains or coastal mouths of river valleys dunes cover the largest areas. In such circumstances typical sand barriers develop as coastal sand bodies built on marine or land deposits due to permanent long accumulation in favouring conditions (Fig. 14.3).

There are different sand barrier types covered by similar sandy dunes but in a different development phase and geographical shape or pattern. Sand barriers are developed on low coasts of fluvial or limnic and organic origin with shallow water in bays or behind rocky promontories. They are structures fully joint to the land, just as a sandy coast between France and Belgium with typical foredunes and inland dunes that are mainly rebuilt by man. Other ones are developed on shallow seashores separating bays, coastal lagoons, sounds and lakes from the open ocean/sea (Fig. 14.2a). Some are wider with a complex of coastal and inland shifting dunes (Figs. 14.1d and 14.2c) as the Curonian Sandbar in Lithuania and Russia, others are very narrow as the Mar Menor sandbar in Spain (23 km long) or the Lido di Venice sandbar in Italy (ca. 50 km long), both heavily damaged by tourism and erosion. The exact opposite may be the El-Bardaweel Lake sandbar – far from civilization, deserted and narrow 80 km long structure on the Sinai Peninsula in Egypt. One of the longest barriers covered by foredune ridges is the Younghusband Peninsula, a 100 km long, narrow barrier in South Australia. The longest complex of sand barriers covered by foredunes and a transgressive dune field has been developed in South Brazil in the country of Rio Grande do Sul (Hesp et al. 2005), where three barriers, each over 150 km long, are separating lagoons or salt swamps from the open ocean with one of the largest world lagoons – 280 km long des Patos, separated by the 8 km wide barrier. The complex of different sandy forms in this area is not easy to define (Hesp et al. 2005).

Some of them are made of one sandspit – a peninsula developing due to longshore material transport from one direction and its aggradation along the coast. It is a very frequent form on the coast that is varying in shape, where longshore sand transport is entering area with rapidly changing coastline relief, e.g. from a straight to a bay. One of the furthest out to sea from the shoreline is Farewell Spit on the north end of the South Island of New Zealand – it stretches for 30 km. However, there are longer spits as the one on the smaller Baltic Sea coast in the east part of Polish coast – the Hel Peninsula – stretching along 36 km (Furmańczyk 1995; Łabuz 2013), very narrow on its base and densely developed in that part. One of the spits that are interesting in shape is Cape Cod in the state of Massachusetts in the USA, where two T shaped spits are developing in opposite directions. On both of them shifting inland dunes and foredune ridges are forming. In other places a sandy coast is built by two sandspits that create one sandbar with a water channel that connects the sea and a land reservoir. One of the places with plenty of such forms is the north coast of the Black Sea in Russia and Ukraine (Vykhovanets 1999). The Arabat Spit, a 112 km long, narrow sand belt with low dune ridges or the Tenderov Spit, 100 km long form built by two spits subject to progradation in opposite directions (as the above mentioned Cape Cod in the USA) are the most characteristic: long and narrow sandy forms. A lot of embryonic barriers and spits are developed on the African Mediterranean coast where in dry conditions narrow sandy belts separate shallow or dried lagoons filled with salt sediment from the sea: the Farwah Spit or Flamingo Island (spit) on Djerba Island. Those are simple spits with low dunes and the length of 7–12 km.

Sandspits are important relief forms of river mouths estuaries and deltas. Some are separate sandy belts and some covered by the complex of foredunes marking delta's progradation. A very good example of such structure is the simple Ebre River delta with two spits (Bastos et al. 2012) and a developed dune belt or a sand barrier at the front of the Nile River Delta.

The third possibility is a sand barrier built as separated islands developed in off the main coast – the barrier islands. The best known ones are the Frisian Islands or the Wadden Islands in north-west Europe which belong to Denmark, Germany and Netherlands – they form a 400 km long chain of sandy islands separating the Wadden Sea from the North Sea. There is also an 800 km long complex of sandy barriers and island chains separated from each other and stretching south from Mississippi delta in Texas to Mexico. Those are mainly simple sandy barriers, narrow with one low dune ridge often washed off by hurricanes. This type of a sand barrier may exist as a separate island developed in the areas of favoring sea currents accumulating sand as e.g. Hiddensse Island in Germany on the Baltic Sea coast. Barrier islands are coastal forms that are very dynamic in a long and short term, that are also strongly affected by washing off storm surges (Ladage and Kunz 2002; Fritz et al. 2007; Houser et al. 2008).

The fourth possibility is a sandbarrier developed as a secondary structure joining the primary land with a lonely island. Such a form is called a tombolo – a sand sheet often covered by dunes that develops due to waves refraction in a shallow and narrow channel between the land and the island. The name of this kind of spit comes from Italian, since in Italy many rock islands are connected to the main land by such forms. One of the most spectacular ones is Monte Argentario in Tuscany, where on the Tyrrhenian Sea two sandy spits separated by a lagoon connect a rocky island with the land. Both of them have been built by opposite longshore currents meeting together in the area where the tombolo is now (the town of Orbetello). The shape, length and width of each form result from coastal factors responsible for the past and current coast development. A very interesting structure can be found between two French islands located near the Canadian east coast. On the south coast of Newfoundland Island there are two small islands: Saint Pierre and Miquelon, both are connected by a gravel and sandy strip which is 11 km long. It is similar to a bridge-tombolo form and is covered by dunes. In its middle part the strip is narrower than 0.1 km. The islands' beach strip is used as a road. There are also similar gravel and sandy strips connecting primary chalk structures of Rugen Island in Germany. The first stage of tombolo developments may be also cuspate spits built in current convergence areas on shallow shores where permanent sand accumulation leads to dune development on the low sandy coast. There is also possibility to find low sandy areas with foredune ridges at the front of the cliff coast, developed in bays, often with river outlets, surrounded by high cliffs which are known as pocket beaches – due to its small size (Fig. 14.2b). When this type of coast is built not only by gravel (shingle beach) but also by sand, it is narrow, concave and sometimes devoid of dunes but still with shifting sand on the beach.

The widest sandbars with parallel dune ridges pointing to progradation tendencies of the land can be found on the low coast of organic origin. They can be as long as the barriers of the Usumacinta River delta between the towns of Frontera and La Veleta in the south of Mexico on the Yucatan Peninsula with the coast over 100 km long and 15 km wide or as extended towards the sea as the Darss Peninsula promontory on the German Baltic Sea coast. One of the longest barriers, narrow and built mainly by one or two dune ridges, can be found along the North American coast of the Atlantic Ocean (called the Outer Banks) – a chain of long barrier islands stretching for over 320 km from North Carolina to Virginia or the Lakes Entrance barrier with the Ninety Mile Beach which is a 145 km long dune belt in Victoria state in Australia.

14.2.3 Inland Dunes on the Coast and Transgressive Dunefields

In a climate with relatively high winds blowing onshore, sparse vegetation on dunes and permanent supply of sand, blowouts and larger parabolic or barchan dunes may develop (Fig. 14.4). Coastline variation and exposition to winds, species composition and sand supply patterns lead to the development of these forms at the hinterland of the foredune belt. Blowouts are mainly deflation forms with a deflation basin and erosional walls that develop in sand ridges. A parabolic, crescent-shaped dune with long arms fixed by vegetation often develops in areas where strong winds blow

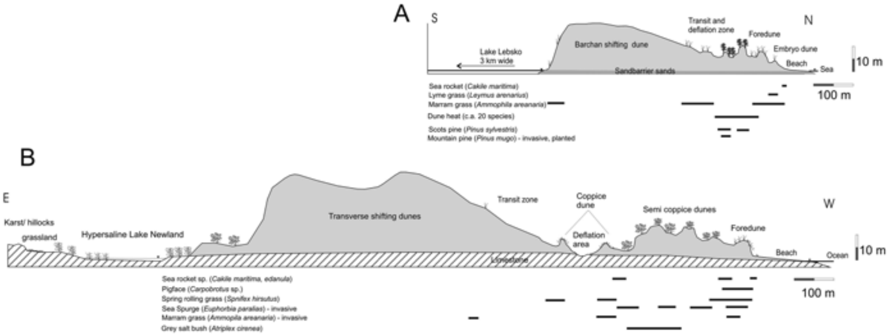


Fig. 14.4 The comparison of form size and selected plant distribution on two shifting sand fields. (**a**) Lebsko Lake Sandbar in Slowinski National Park, Poland. (**b**) Transverse dune system in Lake Newland Conservation Park, South Australia

sand in their dominant direction all the year long. The arms of a barchan type dune are short and directed in the direction to the prevailing wind. Such landscape is similar to a small desert where sand movement and shifting dunes, separated by deflation pans or blowouts, are the main landscape features often wandering on a substrate of the origin different than Aeolian (Fig. 14.2c). The excellent example of primary origin transformation into a new relief of these forms can be found on a long sandbarrier in Southern Brazil in the region of Rio Grande do Sul. There, foredunes, which are also called beach ridges, transform into shifting sands with numerous blowouts, barchans or parabolic forms, thus shaping the so called transverse dunefield (Hesp et al. 2005). It is a large complex of numerous dune forms shifting due to the prevailing wind direction and forming mainly one long belt of sand. The farther away from the source of sediment (mainly from the coast), the slower the dune movement and the faster the stabilization which occurs through plant colonization (accelerated by a shallow groundwater level). In some places the transgressive dunes fields enter lagoons or coastal lakes existing in the conjunction with sandy barriers on which dunes develop, just as on the low dynamic Baltic Sea coast in Northern Europe: in Poland on the Lebsko Lake Sandbar or in Lithuania on the Curonian Sandbar or in Anxious Bay in South Australia, where in Lake Newland Conservation Park high transgresive form is entering hypersaline lake (Fig. 14.1d). Such large shifting dune fields stretching for hundreds of kilometers off the coast are, to a larger or smaller extent, made of by easy to recognize dune features or asymmetric sand bodies. One of the most famous and the biggest sand bodies on the coast is Namib Desert stretching for 2,000 km, with high barchans and star dunes developed on the coast due to permanently arid air, cooled by the cold Benguela current flowing around this part of Africa's coast. On the other hand, the development of shifting inland dunes may be affected by a nearby river estuary and material load, as e.g. the Great Dune of Pyla in the entrance to the Arcachon Bay formed close to the estuary of the Garon River in France. It is a small sand body: just 2.5 km long and 0.5 km wide but with a 110 m high dune that is shifting and thus increasing its area.

Such inland dunes developing on the coast can even enter a higher land. Due to sand supply and eastern winds they are widely developed on the Brazilian coast (e.g. the 50 km long and 15 km deep in-land shifting sand structure of Lençóis Maranhenses National Park near São Luis). They have also developed on the south coast of Australia, stretching from West Australia (The Walpole National Park) through the Nullarbor Plain cliff coast (near Border Village) to the Eyre Peninsula in south Australia (the Coffin National Park). On a smaller scale they have also developed in the coastal areas of New Zealand (the coast of South Island near Kekerengu), (Fig. 14.2d). These climbing dunes may be also older coastal forms which were originally developed on the low coast that has been slowly getting uplifted by neotectonic movements, as mentioned in New Zealand or in Morocco.

14.3 Coastal Dunes Environment Rich Values Versus Human Use

Dynamic coastal sandy system is a complex of sensitive factors and presents a nonlinear morphological response to change. Coastal sand dunes are worldwide habitats that are vulnerable to human and natural factors influence (Martinez et al. 2004). For a long period of time people thought that this environment was stable and worth to use. The reason for that was simple: the dune coast was favorable to human settlement due to its low coast height in relation to high erosive cliffs or swampy coasts. People decided to settle on the dune coasts and exploit its values for the sake of the so called civilization development.

14.3.1 Natural Values of Coastal Dunes

Coastal dunes are very sensitive and unique transitional ecosystems (Martínez and Psuty 2004). Coastal dunes as an environment shaped by winds from the sea sand and stabilized by pioneer vegetation can develop in almost all climates. Therefore, the composition of plant and animal species that have found their habitat on coastal dunes must differ. The mineral origin of sand varies as well. The value of dunes lies in morphologically diverse landscape shaped by storm surges and wind with the participation of pioneering vegetation.

They provide nesting, nursery and foraging for many species. Coastal dunes have also been inhabited since the beginning of human settlement in the coastal areas. They have been used as a sand source, the area for grazing, gathering food, timber, minerals hidden in the sand dune and, finally, as a place for recreation.

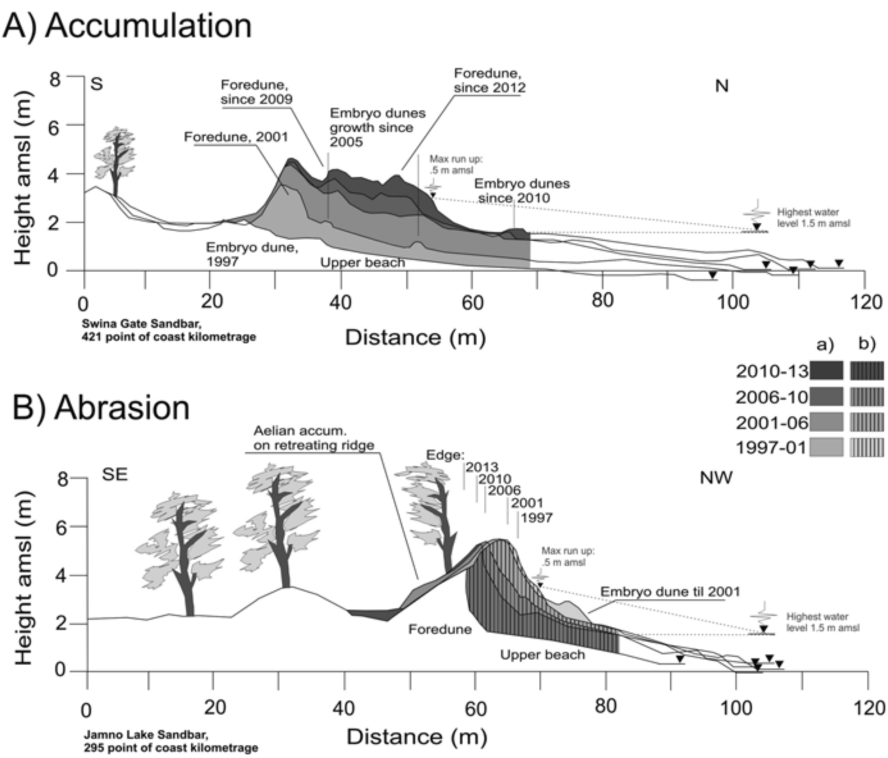


Fig. 14.5 The comparison of two foredune system dynamics in period of 15 years, examples from Poland (based on field surveys). (**a**) Accumulative section of the coast, Swina Gate Sandbar. (**b**) Erosive section of the coast, Jamno Lake Sandbar

14.3.1.1 Landscape and Nature of Instability

Coastal dunes vary due to their morphology and age of development. Different dune types are described in Chap. 1. Natural richness of this habitat depends on a large diversity in the area, along and across the coast as well as on the time of its development. As a result of the sea and the winds impact, in a short term, these forms can be rebuilt, and fixed. The previously stabilized dunes may be shifted again and its development may be renewed. This is the value of this dynamic landscape.

They are known to exist in each climate zone, from cold areas through moderate climate zones, tropical areas with high precipitation to arid areas stretching along the equator (Hesp 2004; Tsoar 2005). They develop in varying conditions of each climatic zone or a particular coastal area. The main factor responsible for their development is ongoing sand load from the sea or from the neighboring river outlets. Recent coastal dunes and sand fields of inland dune forms are seen in the areas that indicate accumulative coast conditions (Fig. 14.5a). Nowadays, due to fast rising sea levels, many of coastal dunes are in a retreat condition (Fig. 14.5b). Shrinking beaches, attacking waves and slow growth of the sea level in the oceans lead to their rapid decrease. It is an important issue for human existence in the areas where settlement or industry have been developed since the civilization sprawl. This coastal habitat is one of the most dynamic ones, where older sand dunes that have already been fixed by vegetation start to shift once more and where new sand bodies are overgrowing on them due to slow coast retreat. The dunes that are not stabilized may enlarge its area due to shifting sands blown by winds. Such landscape is very similar to a typical sandy desert. The rate of sand bodies movements depend on their size and the annual wind strength and direction.

Sand is a unique substance which builds up dunes. In this sense sand is understood as material composed of fine or very fine grain of different origin and the average size from 0.15 to 0.35 mm diameter. Sand dunes can be constructed from quartz, volcanic or even with fine limestone sand derived from crushed shells or coral reef. In some places differences in the sand mineral composition indicate its barrier and dune origin. One of the most interesting are both spits of Monte Argentario tombolo in Italy (described in Sect. 14.2.2). The southern spit and the wider beach with 6 m high foredunes is built of bright quartz sand, while the northern one covered by low dunes with narrow beaches is composed of dark volcanic minerals. In some places local sand is excavated due to its richness in precious minerals.

Progradation of the coast accompanied by beach widening, vegetation succession and the positive balance of sediment on the beach leads to the development of new foredune ridge. The rate of new dune development may be shortened to 1–2 years, when on the beach some new embryo forms appear of up to 1 m high. Such forms may be transformed into a 3–4 m high foredune ridge in the next 3–8 years (Hesp 2002, 2011; Łabuz 2009).

On the shores with tendencies to erosion, coastal dunes are destroyed but they can also withdraw onto the adjacent land. Such dunes are referred to as transgressive dunes associated with the transgression of the seas. When the sea cuts far into the coastal areas, the dunes enter the adjacent land of different origin, mainly coastal wetlands, lakes or even highlands built by older deposits. After each storm surge we can observe how dunes retreat up to 10 m. It is known that storms developed by hurricanes may completely erode dunes and even break the whole sand spit (Fritz et al. 2007; Merrell et al. 2011).

The diversification of the coastal dune landscape is a result of varying dune morphology and distribution of vegetation. Foredunes in moderate wind climate have 6–15 m in height. On the coasts with strong winds and abundant sand we can observe foredunes as high as 30–40 m. These forms are separated by low interdune depressions: blowouts, runnels or niches. The coast built by the dunes shifting so far inland is more diversified in height. Parts of the coast built by convex forms of shifting dunes up to 200 m in height are separated by low-lying deflation areas that can even reach 2 m over the sea level. Many other forms of the deflationary or marine origin may appear. Due to different age and origin of each relief part varying composition of vegetation type may be observed. These values lead to the creation of the unique landscape where everything is composed by the same forms. Such landscape is also a great tourist attraction. Sand beaches and shallow waters offer a great opportunity for leisure and sport activities. The dunes themselves may be used for business purposes. Many natural reserves or coastal parks are established on coastal dunes and the surrounding area (more in Sect. 14.5.8).

14.3.1.2 Biodiversity

In all climate zones, dunes, being at different stages of development, are usually covered by vegetation habitats in subsequent phases of succession. These habitats are usually arranged in belts covering forms of different age or being at a certain stage of development: incipient, stabilized, deflated or withdrawing due to the coast erosion. On this basis we can specify the stage of the dune dynamics and development (more in Sect. 14.2.1). This pioneer vegetation grows only when it is buried in sand. Growing plants contribute to the development of coastal dunes – firstly the incipient, embryo dunes, then the foredunes and eventually their established forms. The succession of the dune habitat vegetation stabilizes its surface and contributes to strengthening of the established dunes. In all the accumulative dune habitats, the first vegetation type are pioneer species: the salt-loving halophytes and the psammophytes that thrive when buried in sand. These plants appear on the upper beach where, due to sand transportation, the first embryo dunes develop. Among them there is the Sea rocket (*Cakile maritima*), occurring everywhere in different varieties, or other salt plants of northern latitudes as the Sea sandworth (*Honcenya peploides*) as well as such succulent plants as the Pigface (*Carpobrotus* sp.) from the lower and southern latitudes (also from South America and Australia). These species are the first sand catchers and dune creators, boosting the development of shadow or embryo dunes.

The Sea sandworth is responsible for creation of 0.5 m high dunes covered by a green plant carpet. The Pigface trailing on the sand with its fast grooving rhizomes fixes the unstable surface on larger areas (Global Invasive Species Database 2009). The psammophilous grasses vary depending on climatic zones and continents. These grasses are found almost on each incipient dune, embryo dune and foredune or each shifting coastal sand body. Their extensive systems of creeping underground roots and rhizomes allow them to grow on shifting sand. These rhizomes lead to germinate and spread on larger and larger areas, stabilizing sand surface. One of the pioneer grasses that live on areas influenced by salt from the sea are sand couch-grass species (*Elymus* sp.), that can be found in Europe and Asia. The most cosmopolitan is the Marram grass (*Ammophila arenaria*), native to the North Atlantic coasts and secondarily planted in the settled areas of South America, Africa, Australia or New Zealand (Hertling and Lubke 2000; Van der Putten et al. 2005). There is also its cousin grass *Ammophila breviliquata*, native only to North America (Cheplick 2005). The Lyme grass (*Leymus arenarius*) is a rarer, neighbor plant growing natively in the northern hemisphere. The psammophilous grasses native to the antipodes are *Spinifex* sp. and *Pingao* (Herbert and Oliphant 1991; Bergin and Kimberley 1999; Gadgil 2002; Bergin 1999), (Fig. 14.6a). There are other grasses

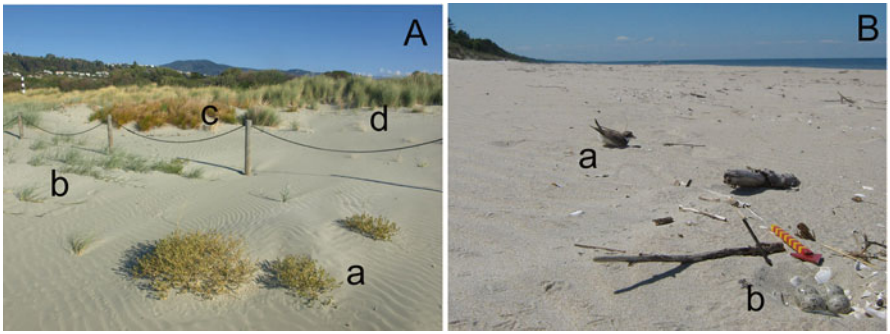


Fig. 14.6 Coastal dunes biodiversity. (**a**) Dunes in Nelson, New Zealand: *a* halophilous Sea rocket, *b* Spinifex, *c* Pingao, *d* Marram grass. (**b**) Beach in Poland with nesting *Charadrius* sp.: *a* bird, *b* nest

from the *Festuce* family or even the *Ammophila* hybrids that cover unstable dune forms. In a warmer climate other plants can be found that may grow on the shifting dunes.

On more stabilized dunes, that appear due to the decrease of sand load, dune heat or dune meadow vegetation grow with different herbaceous plants and other psammophytes. The last stage of plant succession are deforested meadow dunes rich in many plants or forested by Pinue trees whose varieties differ depending on the climate.

The coastal dunes are a habitat for many animals from invertebrates through amphibians, reptiles to mammals. Some bird species have nests and feeding areas on dunes, e.g. the wide spread Plovers family (*Charadrius*) that live on wide sandy beaches as well as on the embryo dunes (Fig. 14.6b). In each climatic zone, many of these animals are related and adapted to the harsh conditions on dunes. In each climatic zone many birds and animals managed to adapt to the harsh living conditions on dunes. Some of them can live on stabilized dunes and some on the shifting only.

14.3.2 Coastal Dune Environment as a Habitat for Human Civilization: Hard Beginnings

Coastal areas have always attracted human development. There is no indication that the trend toward increasing this development will be reversed (Nordstrom 2000). This fact poses the problem of human security in coastal areas which continue to be affected by urban sprawl and tourist industry investments. Some of the countries have become aware of the risks emerging in connection with the destruction of the dunes that protect the low coast. Other countries are located entirely on the low sandy coast and they still develop these areas for settlement, industry or tourism purposes.

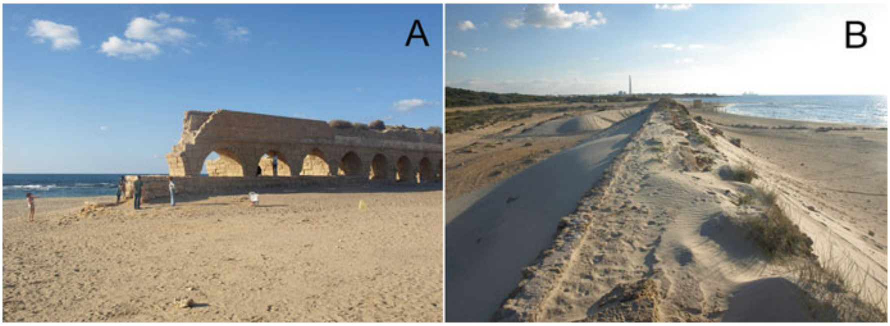


Fig. 14.7 The Roman aqueduct in Caesarea, Israel, (**a**) destroyed by erosion, (**b**) buried in sand

The beginning of human development on coastal dunes, so important for our civilization, dates from the Greek or Roman times. On the today's Israelian coast one of the oldest coastal towns of Caesarea is located, whose origins are dated to 90 BC. It was rebuilt by Herod the Great in 25 BC to become a rich harbor town thriving on trade with its neighbors. Through centuries new generations and new civilizations kept developing the city. The 12 m high cliff made of pottery shells discarded for centuries by local residents is a silent witness of the times. Nowadays, the ancients remains of the palaces, houses and a Roman aqueduct have been destroyed by coast erosion (Fig. 14.7). Sparse dunes in the retreat phase have been slowly burying them with sand. On the other hand, the first coastal harbor in Italy – Magna Graecia dated to 540 BC, also known as Veila, developed to become a rich seaside trading town maintaining its position for hundreds of years. Then, due to the slow land uplifting process that had been continuing for 600 years, the sea retreated by 750 m. Consequently, the town lost its trade and military importance. At the front of it, during the first 200 years a sand barrier developed, the port became dry and the coastal dune was covered with houses. In this specific coastal ecosystem nothing is stable. Studying this system without wider acknowledgement of the relations among human and natural development is worth nothing but a handful of sand, which in some places buries port channels and in others is missing, which means that people will either fight desperately with nature or be forced to abandon their homes.

14.4 Renewed Aims and New Techniques of Coastal Dunes Research

Research of coastal dune environment may be conducted in field or in laboratory. There are many tools and techniques that can be used to collect spatial data for topographic mapping (Table 14.1). These methods include conventional ground surveys as well as different types of remote sensing techniques including photogrammetry and laser scanning.

Table 14.1 The utilities comparison of main surveying methods in coastal dunes

Group, name	Beginning to use (estimated time based on literature)	Applications	Limitations
Remote sensing methods and spatial analyses	–	Varied accuracy of environment components, mainly development, vegetation and beach width or coastline changes	Quality of the data and detailed analyzes Restrictions on short-term surveys or extreme events
Airborne imagery	Since 1970s'	Lower accuracy data of land cover, land use and coast dynamics, shifting dunes dynamics	Varied accuracy of conventional images, resolution, need to be transformed into digital data, weather
Satellite imagery	Since 1980s'	Digital comprehensive data of land cover, land use and coast dynamics, shifting dunes dynamics	Access to high resolution images, weather
LIDAR	Since 1990s' and widely since 2000s'	Mainly for DTM and sand volume changes of bigger areas	Access to data, also financial
Satellite radar interferometry (InSAR)	Widely since 2000s'	Large size relief area changes: mass movement, erosion, coastline changes, earthquakes land use, etc.	Restrictions on short-term surveys or extreme events, financial
Equipment for various morphological field measures	–	Varied accuracy of environment parts as buildings, vegetation habitats, dune relief and beach changes	Bigger areas surveys, e.g. whole sandbars
Traditional leveling	Since 1950s'	Small area relief measurements and vegetation distribution	Simple information, profiling
DGPS and RTK GPS technique	Since 1995 and widely since 2002	Mainly relief changes, volume and height, vegetation location, coastline changes	Area size up to 0.03 sq. km per day, long time of survey, availability of satellites or AGN ground bases, forest, weather
Total Station (TS)	After 2000	Mainly relief changes, each object location	Area size up to 0.02 sq. km per day, long time, weather, relief complexity

(continued)

Table 14.1 (continued)

Group, name	Beginning to use (estimated time based on literature)	Applications	Limitations
Terrestrial Laser Scanning (TLS)	End of 2000, widely since 2005	Very high accuracy, digital data of dunes sand volume dynamics, relief changes, object location, just 3D image of environment	Vegetation (grasses to trees), weather (strong wind, rain). Range of 50–800 m from the laser position
Mobile DGPS	Since 2005	Mainly for beach relief, volume and width	Relief of terrain, not adequate for dunes
Mobile (TLS)	After 2010	Mainly for beach relief, volume and width, also other object will be registered	Relief of terrain, not adequate for dunes, weather, vegetation
Smartphone GPS	New in 2012	Forms position, plants positions, measurements, cooperation with other devices. Not clear	Probably size of surveyed area and data sets. Not clear
Fixed video cameras and imagery	Since 2000s'	Mainly permanent beach changes	Observed small area, difficult analysis and image processing

The conventional field measurements methods are best if we are collecting data about changes happening on short time scale. The traditional way of beach and dune monitoring through profiles perpendicular to shoreline has low accuracy because there are areas between profiles that are impossible to record (Andrews et al. 2002). In the past it was the only method to collect data on the coast shape and dynamics. In the mid 1990s these methods became diversified since new techniques and electronic equipment had appeared which made data collecting faster. Of course that required the use of stronger computers and software in order to analyze more data. Dune investigation moved forward from simple relief leveling to three dimensional relief or elevation models (3D DEMs). Nowadays coastal dune issues may be resolved by means of geospatial analysis using GIS software and database (Mitasova et al. 2004, 2005).

In era when remote methods were being developed, it was possible to cover larger areas with research to understand the processes and phenomena in neighboring areas (Campbell 2002). At the beginning it was aerial photography – a method quickly developing in the mid 1970s. The availability of satellites and measuring devices diversified the sources of data acquisition. The multi-faceted examination of terrain relief, vegetation dynamics and aeolian processes became possible.

Nowadays the large sets of data are analyzed by computer programs and techniques. Processed data are stored in the GIS database and are used for coast change prediction (Mitasova et al. 2010a, b; Teodoro et al. 2011). These methods are costly and cannot capture short-term changes in the dunes, if they are not repeated several times a year. Other studies lead to better understanding of indicators influences on dunes dynamics and of statistical analyzes of their relationships. There are also several issues concerning the dunes development that are studied with modeling methods and laboratory testing, such as wind tunnels and water channels. However, field observations are still needed for short term investigations of rapid dune changes (Hesp 2013).

There are permanent geological studies of sedimentation conditions and age dating of dune sand bodies around the world. Traditional geological survey is strengthened by non invasive methods using similar equipment as in the case of terrestrial or remote investigation.

There are wide surveys on vegetation dynamics that are conducted in order to determine the proper environmental management of coastal dunes. Moreover, in the last 10 years, a number of studies have appeared that discuss management plans for areas of the coast and coastal dunes including the legal and public participation. These issues are being developed at in different parts of the world as a result of growing environmental awareness and social demands. More and more problems in the functioning of coastal dune areas are recognized. These issues and conducted research are presented in this part of the work.

14.4.1 Remote Sensing Methods and Spatial Analyses of Sandy Coasts

For years historic maps have been used as a tool to survey remote barrier islands and sandbars development in the USA (Houser et al. 2008; Hapke et al. 2013) or Europe (Zawadzka-Kahlau 1999; Ladage and Kunz 2002; Borges et al. 2002). Currently, for many years the coastal areas are being presented by remote imaging methods of aerial and satellite photography. Since the middle of the twentieth century the aerial photography has been as a remote sensing method for surveying coastal processes. The satellite remote sensing methods developed quickly as images from satellites orbiting above the Earth had become available. The quantitative nature of the wide range remote sensing data allows for clear indication of change occurring within beach-dune ecosystem (Furmańczyk 1995; Schwarzer et al. 2003; Levin and Ben-Dor 2004; Saye et al. 2005; Houser et al. 2008; Gonçalves et al. 2010; Teodoro et al. 2011; Hapke et al. 2013).

In recent years we have been observing rapid development in acquisition techniques of high-quality data on the Earth surface topography (Bishop 2013). Among these new sources of data there is airborne laser scanning (ALS), known as LIDAR (Light Detection and Ranging), which provides high resolution information about the configuration of land surface (Baltsavias 1999; Saye et al. 2005; Mitasova et al.

2010a, b; Bishop 2013), and a space-borne interferometric synthetic aperture radar (InSAR), that can measure the ground deformation over a large spatial scale (Ruiz et al. 2012). The remote sensing methods such as photogrammetry, LIDAR and SAR provide large amount information about dune topography changes, land use and vegetation cover. Each method has its own restrictions on the use and, therefore, are used for different purposes in environmental studies of coastal dunes.

14.4.1.1 Airborne and Satellite Imagery

The aerial photography of coastal processes have long history. Photogrammetry is based on processing of images, with such main products as: DTMs, DSMs, ortho-images, 2D and 3D reconstruction and classification of objects for mapping or thematic applications, as well as visualization maps, 3D views, animation and simulation (Baltsavias 1999).

When converting old maps and aerial images into a digital format it can be possible to investigate long-term dune changes (Anthonsen et al. 1996). Scientists used maps since 1870 (and until 1992 – aerial photos) to quantify vegetation cover and aeolian processes dynamics across Denmark dunes. This is a popular method used for dynamic recognition of coastal areas. The quantitative information obtained from older images is limited due to the quality of data (Borges et al. 2002; Houser et al. 2008). The elevation errors of a few decimeters in digital surface models (DSMs) are still acceptable for the detection of essential relief changes in coastal areas (Borges et al. 2002). The photogrammetrist's ability to accurately plot elevation is restricted by shadows produced by buildings and trees. Ortho-rectification is not possible without good ground survey control. The digital photogrammetry has been developed from the analogue method used for decades. This method operates on images of a scene captured from different locations by means of a standard digital camera, creating one or more "stereo pairs", used for 3D optical studies of the surface. The traditional aerial imagery as well as the digital one is widely used for monitoring beach dynamics and dunes (Smith and Zarillo 1990; Brown and Arbogast 1999; Łabuz 2002; Levin et al. 2004). It may be used for dune dynamic determination basing on vegetation interpretation (Łabuz 2003). Nowadays, the application of a wide range of remote sensing technologies with GIS requirement and programs helps to reveal dune development, which gives us the opportunity to learn more and to manage coastal zones better (Mitasova et al. 2005, 2009).

In remote sensing (RM) methods we can distinguish several types of satellite imagery resolution: temporal, spatial, spectral, and radiometric (Campbell 2002). Since it is a vast subject it is not going to be discussed here. Instead, the author briefly presents the potential use of the remote sensing in the studies on dunes. The satellite imagery make steady progress due to the availability of geospatial datasets which have been standardized. There are several satellites allowing the ground data acquisition, such as Landsat, MODIS or ASTER.

The last decade has brought improvements in spectral and spatial resolution of data obtained from remote sensing imagery method that can be used for the beach morphodynamics (Borges et al. 2002; Houser et al. 2008; Gonçalves et al. 2010; Yang et al. 2010) or the morphodynamics and vegetation dynamics of dunes (Levin and Ben-Dor 2004; Levin et al. 2004; Kutiel et al. 2004; Mitasova et al. 2005; Teodoro et al. 2011). This method may be applied in the search for mineral resources in rich dune sands (Jagannadha et al. 2008). On the basis of remote imagery thematic maps and coast monitoring can be made.

14.4.1.2 LIDAR

The altimetric airborne laser ranging systems (LIDAR) typically use solid-state lasers with short pulses. Laser scanners operate by firing pulses of laser light in a known direction and waiting for the reflection. The scanner can determine the location of demanded objects by measuring the direction the laser was fired in and the time it takes for the light to return (Brock et al. 2002; Vosselman and Maas 2010). The large number of pulses in a regular pattern (typically several thousand per second) allows to create a Digital Terrain Model (DTM). Some systems allow the recording of multiple echoes from one laser pulse (Baltsavias 1999). Many systems provide additional on-board standard video or digital cameras. The laser footprint is approximately circular and varies with the scan angle and the topography. The points spacing along and across track differ and the latter varies also along the scan line with the scan angle often along the track spacing too, depending on the scan pattern. Thus, it is impossible to image the whole area, homogeneously and without gaps and overlaps; plus, the further visualization, processing, etc. of the image requires the interpolation of a regular grid (Baltsavias 1999; Saye et al. 2005). The LIDAR can provide fully automatically raw X, Y, Z data. This data still needs manual editing for error correction and fill-in of gaps but LIDAR method is used for many purposes in geomorphological mapping (Brock et al. 2002; Mitasova et al. 2005; Höfle and Rutzinger 2011; Bishop 2013). The advantages of LIDAR include its ability to penetrate through forest canopies and to detect minor topographic features. The second advantage to using airborne LIDAR data is that a spatially dense data set is generated over short periods of time, which can be used to provide comprehensive and accurate spatial representation of coastal dunes (Woolard and Colby 2002).

LIDAR data are used for the determination of dune heights, beach dynamics and globally to determine coast position and resistance to storms and hurricanes (Woolard and Colby 2002; Sallenger et al. 2003; Robertson et al. 2004; Mitasova et al. 2009, 2010; Saye et al. 2005; Houser et al. 2008; Hapke et al. 2013). From this data coast profiles and sand volume changes may be extracted and compared to other remote or conventional data bases. Of course, high-density collection techniques such as airborne or developing terrestrial LIDAR methods are clearly ideal to acquire accurate 3-D beach and dune morphology. A 2D–3D Digital Terrain Model (DTM) is a main output that is constructed basing on the collected data. Several DTMs (or DEMs – Digital

Elevation Models) from the same area may be a source of foredunes and beach dynamics surveys (Saye et al. 2005). However, the airborne LiDAR technology is expensive (Lee et al. 2013). The method is costly and more productive on large scale areas (Pardo-Pascual et al. 2005). The hindrance to accurate data using LIDAR in local sale is low system accuracy (Woolard and Colby 2002).

14.4.1.3 Satellite Radar Interferometry

This method evolved after the location of SAR radars in the Earth orbit. Nowadays the ERS-1 and ERS-2 satellites are generally used for data acquisition (Ruiz et al. 2012). Synthetic Aperture Radar (SAR) is a side active radar-ranging system that uses microwaves of electromagnetic spectrum (Carnec et al. 1996; Hanssen 2001). InSAR requires two SAR images acquired over the same scene. Data are mainly used for measuring land mass movement and subsidence, and are also used in coastal areas (Ruiz et al. 2012).

14.4.2 Equipment for Various Morphological Field Measures

There are several geodesic devices which can directly measure dunes for the GIS (Geographical Information System). They can be divided into two groups:

- traditional optical leveling, amplified by links to the GPS network,
- modern electronic measurements with tied real global coordinates by the DGPS network.

In the first group of equipment optical theodolites and laser levels are still used in various types of leveling instruments. This is a relatively conventional method of relief measurements. Due to the availability of the GPS (Global Positioning System) network these methods are more accurate in terms of the GIS database creation.

Currently, the majority of GPS receivers are based on the American GPS network or the Russian GLONASS (Globalnaja Nawigacionnaja Sputnikowaja Sistiema). In the near future the access to the European (Galileo), the Chinese and other networks is expected to be released, which will significantly expand the area of scientific research in the world.

In the second group there is technologically advanced equipment directly or indirectly working with the global coordinate system. The devices from the second group have been widely used for about 15 years and are built upon the new digital technologies and network of GPS receivers. This allows the measurement of the actual coordinates and the relative height tied to the GPS satellite navigation system. Among the devices for measuring the height of dunes the most popular are:

- Differential Global Positioning System (DGPS),
- Real-Time Kinematic Global Positioning System (RTK GPS),
- Total Station (TS).

New techniques are more accurate as far as surface measurements and 3D relief models creation are concerned. Conventional and DGPS methods are often integrated in coastal field work (Harley et al. 2011).

14.4.2.1 Traditional Measurements Reinforced by the GPS Network

This group of equipment allows for measurement of relative heights in relation to the “reader” – the device that reads relative height differences between the points of interest. Knowing the actual height of a point where the device is located, the reader can determine the real height and the actual height differences between points at which the measurement was performed, e.g. using poles for optical or laser reading. These devices can read relative altitude, which may be sufficient for temporary or short-term measurements of aeolian processes and relief changes.

Surface measures performed by means of conventional methods still provide information about topographic and volumetric changes of dune areas that can be then used for 2D and 3D mapping (Andrews et al. 2002). Technological development made it possible to connect GPS receivers with other traditional devices – such as the theodolite – enabling traditional GIS measurements of: surface, distance, height and its differences. Moreover, the development of GIS tools permits more and more accurate and complete data collection. These devices, along with the appropriate GIS software, are used for monitoring such geographical phenomena on coastal dunes as the changes in terrain height, vegetation dynamics and distribution, the effects of erosion, the rate of dunes development, the changes in the beach height and width, and many others. Profiling collected over several decades in Australia shows long and short-term changes in dune and beach relief caused by aeolian processes and storm surges (Harley et al. 2011; Hesp 2013). Over 15 years of measurements in the same places led to determining the rate of new dunes development in Poland (Łabuz 2009). These methods give immediate information about registered relief changes and can be used for computations of sand volume changes. These devices still are and will be used in areas where electronic reference network is not available. Such conditions exist in many remote areas of coastal dunes that located away from human settlements.

The location measurement with the use of popular GPS receivers marks the beginning of modern topographical surveying. These devices measure the time of arrival of a radio signal from satellites to the receiver. The GPS signal information about the layout of satellites in the sky and their theoretical paths, as well as their deviations from those paths (the so-called ephemera). In the first measurement phase the GPS receiver updates this information. Then it uses the updated data to determine the distance between itself and individual satellites in the sky that are currently available. Using trilateration (explained below), the receiver can calculate the geographical position: longitude, latitude and ellipsoidal altitude, and then send it to the selected reference system.

14.4.2.2 DGPS Technique and RTK GPS

Differential Global Positioning System (DGPS) involves the cooperation of ground-based DGPS reference stations with satellite network. It can also be a part of the so called Active Geodetic Network (AGN) consisting of several fixed ground bases. Many countries have created their independent AGN networks, which has reduced the time spent on field base station location relative to the satellites. These networks are part of the emerging Global Navigation Satellite System (GNSS). Such stations broadcast calculated and corrected X, Y, Z coordinates converted from coming from the available satellites, between the base and the receiver on the ground. The communication between a measuring device in the field and an AGN transmitter can be realized through the Internet or a cellular network. Therefore, there are a portable GPS receivers called rovers that can communicate over a cellular telephone network with the nearest base transmitting corrected coordinates. In areas where there is no AGN network, the GPS rover measurements without ground-base are not possible. After the first experiments with the DGPS RTK – a mobile base and a rover for obtaining coordinates of points located on partly forested and secluded coastal dunes – a local coordinate network was established which was based on a local triangular network (Łabuz et al. 2012). Each point was set to measure coordinates for 6 h. Such procedure can make the research longer, but it guarantees accurate results of the future surveys conducted in the same area.

RTK-DGPS is a technique used world-wide for beach and dune surveying (Bastos et al. 2012; Lee et al. 2013). The most popular devices based on DGPS for measuring the height of the dunes include:

– Differential Global Positioning System (DGPS) stations, two receivers – one stationary/stabilized ground-base reference station to broadcast the difference between the positions indicated by the satellite systems and the known fixed positions and the other one – a rover used to rove around, making position distance measurements from the base.
– Real-Time Kinematic (RTK) DGPS, a mobile version of DGPS with a mobile ground-base used to enhance the precision of position data derived from satellite-based positioning systems, and one or more rovers operating with connection to the base, computing a local position linked to a periodically stabilized base. Both are connected by e.g. wireless or telephone network,
– Real-Time Kinematic (RTK) GPS, a single RTK rover, that is independent from the field ground base but communicates with AGN network.

With these devices we can measure coastal dunes and beaches relief or total sand volume changes (Pardo-Pascual et al. 2005; Harley et al. 2011; Bastos et al. 2012; Łabuz et al. 2012; Lee et al. 2013), (Fig. 14.8a). The DGPS RTK field measures taken after storm surges helped to collect data characteristics of its impact on coastal areas caused by Katrina hurricane in the USA (Fritz et al. 2007). It is a quite fast method to register dune and beach relief changes after storm surges (Łabuz 2013). It was also used for surveying the shifting dunes (Navarro et al. 2011) or the beach and dune balance relations (Bastos et al. 2012). To improve the quality of the work

Fig. 14.8 The equipment used for dunes morphology surveying, (**a**) Real-Time Kinematic (RTK) GPS, (**b**) Total Station (TS), (**c**) Terrestrial Laser Scanner (TLS)

when using the RTK-GPS system the proposed optimum reading interval between points is 20 cm (Kizil and Tisor 2011). The higher the reading resolution is more unproductive it gets (Pardo-Pascual et al. 2005; Lee et al. 2013). A large proportion of errors may either be made by an operator or can be the result of an unexpected movement of the equipment during operating (Lee et al. 2013). Moreover, it should be underlined that this accuracy depends on the relief diversity, mainly of small embryo dunes, that need to be measured in more detail. Until recently, the margin of error when reading the exact height by means of GPS was almost 20 cm. Currently we can do it with the precision of a few millimeters, depending on the user's skills.

Another surveying technique where the RTK-GPS system is mounted on an all-terrain vehicle (ATV) is widely adopted for efficient surveys having a wider spatial coverage compared to the on-foot GPS surveys (Pardo-Pascual et al. 2005; Baptista et al. 2008; Lee et al. 2013). The high-accuracy RTK-GPS technology mounted to an ATV enables the entire three dimensional subaerial beach to be measured at dense spatial scales without incurring the significant costs associated with LIDAR surveys (Harley et al. 2011).

14.4.2.3 Total Station vs RTK GPS

Total Station (TS) is an electronic theodolite integrated with an electronic distance meter (EDM) that allows to measure the angle and distances coordinate (X, Y, and Z) from the total station to points under survey (Fig. 14.8b). These devices are used to read distances and heights of particular measured points relative to the instrument. These surveys should be done in the real world coordinate system so that a comparison can be made to other data, such as aerial imagery, vector datasets, political boundaries (Khalil 2013). The proposition is to have two data sources from at least two points (Khalil 2013). In geodesy there is a popular method for determining geodetic coordinates which is called trilateration. It is based on the measurement of three points in a triangular system tied to Global Navigation Satellite System (GNSS), being usable in conjunction with GPS, GLONASS and Galileo satellite network. This method allows to obtain correct coordinates of the points. Total station is a widely used tool to survey land form evolution and to monitor land use (Khalil 2013) also in beach dune ecosystem (Delgado and Lloyd 2004; Baptista et al. 2011; Harley et al. 2011).

RTK-GPS system helps acquire data of higher quality than Total Station (Kizil and Tisor 2011). The signal blockage by trees or other obstacles can make the use of RTK system limited or impossible (Kizil and Tisor 2011). In such cases Total Station is used. The elevation errors between the both devices was defined as 12 cm. Measurements taken by means of Total Station take much more time than by RTK-GPS (Lee et al. 2013). On the other hand, terrain is better mapped by RTK-GPS than by Total station. The RTK-GPS is more accurate than Total Station, because it can be mounted on vehicles (Pardo-Pascual et al. 2005; Harley et al. 2011; Bastos et al. 2012; Lee et al. 2013), which makes the measurements faster, especially on the beaches.

14.4.2.4 Ground Laser Scanning

There are different ground scanners operating in varying conditions, such as Terrestrial Laser Scanner (TLS) and a Mobile Laser Scanner (MLS). The terrestrial fixed scanner is able to collect the point data at the 360° angle around the positioned equipment (Fig. 14.8c). Mobile terrestrial laser scanning is an emerging technology that combines the use of a laser scanner, a global navigation satellite system, and an inertial measurement unit on a mobile platform. Ground-based mobile laser systems are typically installed on a minivan or on boat. The scanner allows measurements without interfering with sand surface that takes place when surveying is performed using other equipment, such as GPS RTK or Total Station. The scanner can be used to measure seasonal relief changes of embryo dunes or dune front (Feagin et al. 2012; Montreuil et al. 2013a, b).

The TLS or MLS scanner device emits a coherent light beam that is reflected from the surface of the object and returns to the device (Vosselman and Maas 2010). Hereby information is achieved about the object position relative to the device location, with a horizontal and vertical angles, thanks to which local coordinates of a each point can be calculated. In the device its memory accumulates the point cloud with the known coordinates in space. On this basis it is possible to construct a real image of the measured object presented by this cloud. Complementary to the research, photographic images of the object are made that show its true colors.

The range of a scanner work depends on its technical parameters and assignment (Michoud et al. 2013). For example, Optech scanner can measure surfaces at a distance of 800 m. Smaller Faro scanner is suitable for the measurement of small surfaces located in close proximity ranging up to 80 m from the instrument. The equipment supplied by Topcon enables measurement within around 150 m from the scanner and Leica ScanStation – up to 100 m. Measurement error depends on the accuracy of the reading device and on its type and it is usually less than 3 mm. Scan accuracy depends on the scanned distance from the area and the predetermined number of points per unit area. The TLS point clouds are collected with the resolution of up to 3 cm while the MLSs are able to scan quickly a long shoreline with a resolution not less than 10 cm. The sampling rate is 2,000–3,000 points per second. It is unbelievable that such large amount of data can be obtained in such a short time. The point cloud data can be imported into various software packages including GIS or others.

However, these devices have several limitations when used on the coastal dunes. During strong wind action it is not possible to calibrate device for measurements. In sandy environment shifting sand grains may interfere image. The vegetation cover is a main obstacle for beam of laser scanners. This limits the accuracy of measurements of the foredunes covered with grasses or shrubs, as some researchers have also found out (Montreuil et al. 2013a), and completely eliminates this method as a tool for measuring the dunes covered with forests. The method works best for as a way to measure micro- and macro-relief of the beach, beach embryo dunes and the exposed slopes of eroded dunes and deflation forms.

The laser beam stops on exposed slopes, so the slopes that are located in the shadow must be scanned from another place. Depending on the object geometry it is necessary to run a scan survey on different equipment positions. In order to capture the long stretch of the relief, a scanner can be set along the form in two or more positions. In order to reflect the terrain changes across interdune depressions, dune ridges and beach relief the scanner must operate from three to four positions across forms. The distance to each position depends on the scanning range and the assumed accuracy. On foredunes and upper beaches scanning resolution may be set at 3–5 cm (1 point for 3–5 cm area). Such scanning on the area that is 100 m wide takes 1–1.5 h. The scanning resolution of a lower beach may be 5 and 10 cm, which takes between 30 and 40 min. The scanned area in the field should be outlined by reference points, marked with white objects or disks. It helps to easily distinguish reference points for production of tied point clouds when scanner is located in several positions.

The output of the TLS scans is a set of reflection point clouds defined by X, Y, Z coordinates, which can be used to construct 3D digital models of the scanned surface (Montreuil et al. 2013a, b). Post-processing of TLS data includes the four following steps: (1) scan registration, (2) model geo-referencing, (3) vegetation correction and (4) generation of a digital elevation model (DEM). The obtained results demonstrate the potential of the TLS to generate very precise three-dimensional models of dune topography.

14.4.2.5 Video Cameras and Imagery

Another, separate method is the registration of beach and fore dunes using one camera or a system of several cameras. This is a fully automated procedure to derive mainly beach width and elevation changes on video images or movies collected on a daily basis (Holland et al. 1997; Uunk et al. 2010). Such cameras are being installed around the world beaches for acquiring data on beach dynamics related to sea level fluctuations. In order to obtain quantitative information from these images several devices must be used. This method requires the use of images rectification procedure for measuring the observed relief changes. In combination with manual mapping by a conventional survey or DGPS, it may be a promising method for data collection for future computations (Turner et al. 2006; Uunk et al. 2010). However, single cameras are used for collecting information and predicting dune-beach changes related to extreme storm surges or to management for touristic purposes.

14.4.2.6 The Combined Techniques in Coastal Dune Mapping

The comparison of topographic profiles from different time periods remains a key tool for monitoring coastal change. For this purpose a variety of techniques is used including total station surveys, DGPS surveys and LIDAR data analysis (Harley et al. 2011; Montreuil et al. 2013b). For long-term studies it is therefore important to understand the relative accuracy of each of the different techniques. Previous research has reported a good relationship between the DGPS topographic profiles and the airborne LIDAR used for large spatial scale research (Saye et al. 2005). The latter is now being used in some locations for repeat annual surveys (Montreuil and Bullard 2012). The use of TLS for this kind of study suggests that it could be a valuable tool in coastal management. However, some expertise is required to use the technology effectively and its spatial coverage is more limited than that of LIDAR. Consequently, TLS is probably more suited to regular surveys of small coastal areas – for example to monitor changes associated with soft engineering structures – rather than regional scale monitoring.

14.4.3 Field Investigation on Aeolian Processes and Laboratory Modeling

The surveys of aeolian processes related to sand movement and transportation have long history. This section will summarize latest work on this subject. There is a wide range of literature on the subject of shifting sand and may be studied from mentioned below investigations. Throughout the twentieth century plenty of articles appeared describing relations of sand movement to wind velocity, topographic relief and vegetation presence (Bauer et al. 1990; Sherman and Hotta 1990; Arens 1996; Gilles et al. 2002; Lynch et al. 2009; Petersen et al. 2011; Houser and Sojan 2011; Davidson-Arnott et al. 2012; Bastos et al. 2012; Anthony 2013; and others). Some of them were related to such sand grain parameters as size, surface humidity and surface roughness (Arens 1996; Davidson-Arnott et al. 2008; Bauer et al. 2009; and others). These investigations are being in use in field and laboratory conditions to achieve information on particle movements in different environmental conditions and to create appropriate models (Sherman et al. 1998).

14.4.3.1 Field Surveys

The sand catching and wind velocity measures are main surveys. The aeolian transport is very important in coastal environment, therefore all over the world many studies are being undertaken in field for determining the above mentioned dependences (Juanaidi 2009). In field, these studies are basing on the usage of different type sand traps and anemometers (Arens 1996; Walker 2005; Bauer et al. 2009; Hesp et al. 2009). Also, new ultrasonic anemometers are used for resolving problems in sediment transportation (Van Boxel et al. 2004; Walker 2005; Delgado-Fernandez et al. 2011). In experimental, field work, the rate of sand transport is measured by sand traps, the narrowest as possible to avoid transport disruption. Such studies on aeolian processes are also carried out to determine the rate of stabilization of dunes by recognizing sand content in ground, for example its carbonate composition (Meerkerk et al. 2007). These all methods and field solutions vary in terms of the employed equipment and the obtained results and are still discussed by the researchers, due to used techniques and later post-processing data acquisition. However many of the obtained results present similarities of sand behavior in relation to plant distribution or measured wind velocity (Arens 1996; Kocurek and Ewing 2005). Similar investigations on sand transport are carried out in wind tunnels in artificially created windy conditions (Walker and Nickling 2003). It may be caused when hard meteorological conditions do not for proper field study (Hesp et al. 2009). The equipment may be damaged by strong wind action, which I have experienced once during a heavy storm.

14.4.3.2 Laboratory Experiments and Modeling

We must not forget to mention in this chapter the laboratory experiments and modeling of sand movements on different surfaces. Laboratory and computer simulations, supported by field and experimental data, indicate that a given wind regime produces a simple dune-field pattern – a dune environment, that is a major component of this study. In recent years there several publications have appeared on sand blow that has been surveyed in the course of experimental or laboratory research run in varying conditions of wind tunnels (Gilles et al. 2002; Walker and Nickling 2003; Dong et al. 2004; Parsons et al. 2004; Udo and Takewaka 2007). There are several variables that are involved in wind tunnel simulation work (Bauer and Davidson-Arnott 2003; Pye and Tsoar 2009). The purpose of all these surveys is to try to understand the physical conditions of the aeolian processes.

14.4.4 Sedimentology Recognition: OSL and GPR Methods

Conventional studies on the dunes geological conditions consist of excavations or drilling accompanied by a wide range of paleo-environmental methods that help to date the dunes origin and to study their stratigraphy and sedimentology

(Clemmensen et al. 2001). Among them there are radiocarbon or luminescence sand dating techniques (Hoffmann et al. 2005; Pye and Tsoar 2009).

14.4.4.1 Optically Stimulated Luminescence (OSL)

Luminescence dating is an important method for timing the Holocene sand movement (Aagaard et al. 2007; Clarke and Rendell 2009; Kunz et al. 2010). By means of this method we can find out when mineral grains were last exposed to sunlight, and therefore OSL dating can be used to determine the depositional age of dune sediments (Jacobs 2008; Reimann et al. 2010a, b). The Optically Stimulated Luminescence (OSL) dating is a tool for the reconstruction of coastal spits evolution and foredune accretion (Molodkov and Bitinas 2006; Reimann et al. 2010a, b).

14.4.4.2 Ground Penetrating Radar (GPR)

With the development of new techniques, a radar for subsurface surveys was constructed. A ground penetrating radar (GPR) is a geophysical device using electromagnetic wave reflection being the result of electric properties of material in the ground (Schenk et al. 1993; Harari 1996; Bristow et al. 2000; Galgaro et al. 2000; Clemmensen and Nielsen 2010). The exploration of the internal structure of dunes carried out with the ground-penetrating radar and supported by data from topographic DGPS reveals geological conditions and development of sand dune bodies (Bennett et al. 2009; González-Villanueva et al. 2011). These analyses are of great importance to ascertain the state of the internal structure of dunes as an important element in their stability and, therefore, in their evolution. The internal structure shows the accretion and progradation sequences of dunes over beach deposits which depend on dune morphology (height, crest orientation) and location, as well as the processes affecting them (Bennett et al. 2009; Santalla et al. 2009; Clemmensen and Nielsen 2010).

14.4.5 Studies on Vegetation Dynamics and Landscape Changes

Vegetation in coastal dune systems is a factor that is important for determining the mobility of the whole system. Studies on the vegetation dynamics are conducted by means of different methods – such as remote sensing and spatial analyses as well as terrain studies in situ. This section highlights the influence of new methods on the diagnosis of the dune vegetation and the related problems of proper management. There are several factors responsible for vegetation occurrence on dunes. The mechanical stress from moving sand, nutrient availability, salt spray and water supply are key factors determining the species presence and their diversity (Acosta et al. 2006a, b; Grunewald and Schubert 2007; Łabuz and Grunewald 2007; Gracia Prieto et al. 2009; Provoost et al. 2009).

The described above methods of aerial photographs, satellite images, GPS research and GIS techniques are used for dune vegetation surveys (Shanmugam and Barnsley 2002; Acosta et al. 2005; Kollmann et al. 2009; Provoost et al. 2009; Peyrat et al. 2009; Faggi and Dadon 2011; Bitton and Hesp 2013). The species location, their diversity and abundance are the subject of base surveys carried out on coastal dunes. The typical field method for plant investigation are relevés, performed on cross-shore transects or in different size plots, from 1 m^2 to 25 m^2 (Moreno-Casasola 1988; Acosta et al. 2006a; Łabuz 2007; Łabuz and Grunewald 2007; Faggi and Dadon 2011; Santoro et al. 2012). There are several indices based on vegetation relevés that are used to measure plant species diversity of coastal dunes, e.g. Shannon and Evenness indices or Rarity index (García Mora et al. 2000; Acosta et al. 2009; Carboni et al. 2009). Furthermore, soil samples for nutrients investigations are collected (Berendse et al. 1998; Kutiel et al. 1999; Isermann 2005; Álvarez-Rogel et al. 2007). The pH and nitrogen indicate ecological conditions for vegetation dynamics and occurrence (Piotrowska and Gos 1995; Isermann 2005; Remke et al. 2009). The change in the physiochemical conditions produces an environment favorable to more plant species and therefore more species can become established (Isermann et al. 2007). These studies are carried out to analyze the biological and ecological mechanisms that influence vegetation dynamics or invasive plants occurrence (Van der Meulen and Salman 1995; Van Boxel et al. 1997; Feagin et al. 2005; García-Mora et al. 1999; Campos et al. 2004; Acosta et al. 2006b, 2007; Kollmann et al. 2009; Roig et al. 2009; Provoost et al. 2009).

14.4.6 The Methods and Indicators for Long-Term Management Plans

Coastal dune vulnerability and monitoring are mainly studied basing on the checklist of different indicators (García Mora et al. 2000, 2001; Shanmugam and Barnsley 2002; Martínez et al. 2006; Muńoz Vallés et al. 2011; Sousa et al. 2011; Doody 2013).

The problems in coastal dune management are related to coast erosion and sea level rise (Nordstrom 2000; Robertson et al. 2004; Dillenburg et al. 2004; Phillips and Jones 2006; Feagin et al. 2010; Bastos et al. 2012; Berry et al. 2013), sand movement (Van Boxel et al. 1997; Terlouw and Slings 2005; Meerkerk et al. 2007; Navarro et al. 2011), tourism impact and settlement development (Isermann and Krisch 1995; Nordstrom et al. 2002; Levin and Ben-Dor 2004; Hesp et al. 2005; Dahm et al. 2005; Grunewald 2006; Phillips and Jones 2006; Costa and Melo e Souza 2009; Klein and Osleeb 2010; Muńoz Vallés et al. 2011) and alien or native species growth and removal (Gómez-Pina et al. 2002; Leten et al. 2005; Tsoar et al. 2005; Wootton et al. 2005; Van der Putten et al. 2005; Meerkerk et al. 2007; Richards and Burningham 2011).

There are numerous works strictly devoted to management of dune landscape, from dealing with the general environment restoration (Van der Meulen and Salman 1995; Arun et al. 1999; Doody 2002, 2013; Provoost et al. 2002; De Lillis et al. 2004; Łabuz 2004; Martínez and Psuty 2004; Lemoine and Faucon 2005; Defeo et al. 2009; Kollmann et al. 2009; Nordstrom et al. 2009; Antunes do Carmo et al. 2010; Froede 2010; Marchante et al. 2010; Santoro et al. 2012; Del Vecchio et al. 2013; Weisner and Schernewski 2013), through vegetation management to animal impact as grazing (Kutiel et al. 2004; Provoost et al. 2004) to native dune animal protection (Kutiel et al. 2000b; Herrier et al. 2002; Baeyens and Martínez 2004).

In recent years numerous papers have been devoted to educational campaigns and public participation in the proper environmental management of coastal dunes and other associated habitats (Dahm et al. 2005; Houston 2005; Pontee and Morris 2011; Berry et al. 2013; Nicholls et al. 2013). The recently discussed management plans address many indicators, public awareness and long-term coast changes (Nicholls et al. 2013).

14.5 Coastal Dunes Values Use and Reuse by Human

Traditional economies thought that coastal dunes have little value, according to Nordstrom (2000). At the beginning, the human impact was small and local but the acceleration of cultural development led to the unprecedented change in landscape and the dynamics of species exchange. The most important for human development in the coastal areas was access to the sea from the low sandy coast. Small villages, big cities and important harbors are located on low sandy coasts in places where rivers have their outlets to seas and oceans. Dunes built by sand grains of similar diameter of 0.15–0.35 mm everywhere and with high salinity did not represent any economic value. Initially they were not fit even for the housing purposes.

Are the coastal dunes are really valuable for humans? The perception of the environment as a valuable area depends essentially on many social, economic, and even political factors (Everard et al. 2010). To find the objective answer to this question we need to present its utilitarian aspects, that are often difficult to quantify in the financial terms, and relate them to current social expectations and legal acts regulating the possibilities of their use. The approach to the environmental development of the space which we live in is the sum of the generations' experience, each civilizational change and evolution, the state of knowledge and the needs of a particular community. The values of this environment can be economic and natural. The economic values result from the opportunities to exploit what this environment offers to us. The natural values are the result of diverse and dynamic landscape full of unique land forms, inhabited by the unique plant and animal species. This environment will always have greater value and be of greater use for those who live in it, who are a part of the nature, who work and live there struggling with adversity, threats and disasters, including those of natural origin, such as floods. Visitors to the coastal dunes should be educated about these values and threats of natural or human origin before this exceptional environment will be lost. They are only seasonal users, occasional guests who need to acknowledge and respect this unstable and unique environment.

The first indisputable fact is that a man chose to settle on a low dune coast, which provided an easy access to the sea and its resources. In the beginnings of the Western European civilization, forests growing on coastal dunes were felled in order to acquire wood to build cities and ships. This of course resulted in dune sand starting to shift (Riksen et al. 2006; Danielsen 2008; Koster 2009). Pastures, fields and homes built close to the dunes started to be endangered by shifting sand. That situation required to take steps in order to re-stabilize the ground by means of re-planting the forest. Unintentionally, people began to plant non native plants that, as we know now, were responsible for rapid colonization of the sand. In this way humans began to destroy the natural flora of the dunes, which also had consequences for the animal world. On the dunes, treated as agricultural wasteland because of poor soil, rough grazing developed. This method is still in use in the developing countries. It has even re-appeared in Europe where it is used for nature landscape preservation. Its goal is to preserve as much as possible of the natural landscape of dunes and to protect biodiversity. The coastal dune landscape, which resembles the mystical desert dunes areas, is in itself a magnet for the tourist business.

In conclusion, the values of coastal dunes may be put into several groups:

- ecological, conservation related – which save natural values of the biodiverse land, where many natural reserves or national parks are established,
- recreational – with their unique natural and dynamic landscape, offering perfect conditions for sports and leisure,
- land for residential, industry or agriculture development – land close to the sea is the most attractive area, with its rich deposits of amber and valuable minerals, its shallow waters with the abundance of fish,
- natural belt for coast protection against erosion and sea flood – with high, natural dune dykes, sandy coast are the best protection for developed lowlands and coastal waters,
- intrinsic values (first time described by Nordstrom 1990), referring to the importance of their diverse environment as an essential part of the surrounding nature.

Despite their high conservation value, coastal dunes have been subjected to considerable human impacts changing in time: including the acquisition of vegetation, grazing livestock, or sand stabilization by means of non-native, invasive plants for industrial and intensive agricultural purposes (Table 14.2). These issues are discussed in the following sections.

14.5.1 Settlement Purposes

Settlement on the dunes is known in all the European countries (Fig. 14a). The 65 km long Belgium coast is in two thirds totally covered by settlement. The North American east coast and the Mexico Gulf coast are known for their sandy coasts,

Table 14.2 The dune management main problems related to world development and culture

Development issues	Highly developed and urbanized with complex management	Average development, nowadays creating management plans	Last inhabited areas but quickly developing with management plans	Developing countries with urbanization, agriculture and industry pressure, seeing some problems	Developing or not developed in some areas but with settlement pressure without plans for management	Not developed, without plans for management
World areas	USA, some areas of Canada, Europe (west and south part)	Rest of Europe, some South America countries	Australia, New Zealand	Middle East, North Africa, India, RPA, China, Brazil, Mexico	Africa, some remote areas of Brazil, Mexico	Africa, Asia
No.	1	2	3	4	5	6
Coast protection						
Littoral drift interruption	H	M–H	M–H	M	L	L
Waterfront constructions	H	H	M	M–H	L	L–M
Biodiversity decrease	M	M	M	M–H	L	L
Non native plants	H	M	H	M	L–M	M
Artificial dunes creation	H	M	M	L	L	L
Urbanization (roads, houses)						
Society	H	M–H	L–M	M–H	M	L
Private	H	L–M	M–H	L–M	L–M	M
Habitat subdivision	H	M–H	M–H	M–H	L–M	L
Tourism						
Massive tourism	H	M–H	L–M	M	L	L
Dense infrastructure	H	M	L	L	L	L
Recreational pressure	H	H	L	L–M	L	L
Golf courses	H	L–M	L	L	L	L
Pavements, boulevards	H	L–M	L–M	L	L	L
Roads, parking lots	H	M	M	L	L	L
Off road sports	M–H	M	M	H	L–M	L

Environment changes						
Erosion	H	H	H–M	M	M–L	L
Shifting dunes	M	L	L	M	H	L
Grass encroachment	H	H	H–M	M–L	L	L
Dune stabilization	H	H–M	H–M	M	L	L
Non native species	H	H–M	H–M	M	M	M
Soil, water pollution	M	H	M	M	M	L
Animal decrease	H	H–M	H–M	H–M	M	M
Industry						
Dune mining	L	L	M–L	M–L	M	M
Harbor infrastructure	H	M	M	M–L	M–L	H–L
Manufacturing	H–M	M	M	M–L	M–L	H–L
Agriculture						
Afforestation	M–H	M	M	H–M	M	M–L
Deforestation	H	H–M	H–M	M	M	H–L
Intensive farming	M–L	M–L	H–M	M–L	M	M
Pasture	L	L	M–L	M	H	H
Crops	M	M	M–L	M	M–H	M–H
Private use						
Protection works	H	L	M	L	L	L
Crops	L	L	L–M	M	M	M
Garbage dumping	L–M	L–M	L	M	H	H
Water extraction	M–H	M–L	M–L	M	L	L
Military use (camps, training grounds, state borders)	H	M	M	M–L	L	L

Impact: *H* heavy, *M* medium, *L* low

that are occupied by private houses that are built almost on the beach. Their protection costs million of dollars in places where coast erosion occurs, as in the Avalon municipality on New Jersey in the USA (Nordstrom et al. 2002). Now there is a question how to restore local environment without withdrawing private property from the coast. This issue seems to be important on the narrow barrier island in the USA, where millions of dollars come from private property and coastal tourism, and where many coastal protection measures have been taken (Nordstrom 2000; Feagin et al. 2010) Along the sandy coast of Brazil there are thousands of small villages that are built straight on low dunes. There dunes are used as pasture zones, as the waste disposal and the land where people build their households. These small coastal villages have problems with ecology, as Sergipe on Praia do Jatoba, the south coast of Brazil (Costa and Melo e Souza 2009). Near Rio Grande do Sul (RS) or on Praia do Cassino (RS) litter and waste from households is dumped on dunes (Rosa and Cordazzo 2007; Portz et al. 2010). Moreover, human settlement is developing in the neighborhood of shifting transgressive dunefields – Toress and Cidreira are towns in the region of Rio Grande do Sul, Brazil, where quick settlement sprawl is observed (Portz et al. 2010).

In New Zealand many coastal dunes were leveled or bulldozed for the purpose of new settlements, especially after the WW II (Dahm et al. 2005). As a result, completely removed native plants were replaced by a green carpet of grasses. In the twentieth century only several new towns were built on coastal dunes – mainly in Israel (Levin and Ben-Dor 2004). In the middle of 1950s the town of Ashkelon and Ashdod were built from scratch on the sand dunes of the Mediterranean coast of Israel, together with their industrial zones. They are surrounded by semi shifting dunes (Fig. 14.9d).

On the other hand, for some countries coastal areas are the only available land for settlement or industry development. The present Singapore development is possible only on low coastal areas. The Arabian Emirates future income is based on the creation of artificial sandy islands because new developments are not possible in the desert.

14.5.2 Agricultural Use, Pasture, Obtaining Food, Herbs and Timber

The longest period of when dunes were used by human civilization happened in Europe. Dunes covered by forest were used for acquiring timber and for pasture (Koster 2009). Dune heaths, were used for grazing (Fig. 14.10), cutting heather and excavating peat from dune slacks (WallisDeVries et al. 1998).

For a long time the mobile foredunes were regarded as a worthless ecosystem. After the arrival of Europeans on the American continent, sandy peninsulas and islands of the east coast were a convenient place where the horses that had been set free could breed and live.

Fig. 14.9 Settlement on dunes. (**a**) Developed Belgian coast, Oostende, (**b**) illegal private housing in Tunisia, (**c**) typical private housing in Australia, Melbourne Bay, (**d**) among the sands Ashdod town, Israel

In some cultures, e.g. in Asia, many plants growing on dunes were used as medicinal and edible herbs. Nowadays, coastal dunes in India are utilized for agriculture, industry development and sand mining purposes (Sridhar and Bhagya 2007).

Aboriginal people in Australia used dune plants as the source of food. Fruits and leaves of the pig plant – or Ice plant (*Carpobrotus glaucescens*) – were eaten by Aboriginal people. For many of them coastal sand dune plants were a part of life and culture (Elliot and Jones 1982). Early European explorers used the Pig plant as an anti-scurvy treatment and its roasted leaves were used as a salt substitute. The juice extracted from its leaves brings relief to pain from insect bites. The Pingao grass plant (*Demoshoenus spiralis*) indigenous only to New Zealand was, and still is used by Maori for weaving (Herbert and Oliphant 1991). After years of degradation of dunes covered by Pingao grass, now started programs to its restoration in New Zealand (Hilton et al. 2000; Hesp and Hilton 2013).

Forest covering the dunes were and still are used for timber production. Europe is a good example of forest exploitation form dunes, that caused land drying and launch of shifting sands. Dune forest resource were used for timber and coal production in developing Europe (Riksen et al. 2006; Koster 2009). For fast timber production and sand stabilization were used many non-native species. In Mediterranean region (Marchante et al. 2010) or in Morocco (Negre 1952) exotic

Fig. 14.10 Sheep of Palestinian farmers on dunes near Nitzanim, Israel

Acacia and Eucalyptus were used. Similar situation is known from South America or Australia and New Zealand (Holmes and Cowling 1997; Midgley and Turnbull 2003; Kutiel et al. 2004; Marchante et al. 2010).

In addition, on dunes are located fishing harbors or just places for boats. In developing countries, the boats are drawn directly on the dunes, in the rest countries, there also buildings that are used for fish processing (Fig. 14.11).

14.5.3 Excavation of Dunes: The Mining Industry

The industrial value of the coastal dune ecosystem is the possibility of sand excavation. In dunes where there is a large concentration of valuable minerals they are mined on an industrial scale. Even small scale sand mining may pose danger to the environment stability, as it happened in Northern Ireland (Carter et al. 1992) or on the Azores (Borges et al. 2002). According to the documentary records, on the Santa Barbara beach in São Miguel (the Azores) sand mining between the mid 1960s and mid 1970s led to reducing the foredune berm width to a half of its former size (Borges et al. 2002). That, of course, triggered dynamic changes and ecological imbalance. Some dune areas in Spain where sand was excavated (the La Bota beach or the Lincres Spit) have suffered from a continuous degradation process (Gómez-Pina et al. 2002).

Fig. 14.11 Fishing boats on the sand dunes in Campania, Italy

Due to different origin of sand building dunes, valuable minerals concentration can be found there. It is mainly possible in volcanic sands building dunes near former or present active volcanic zones. Along the Orissa-Bengal coast in India heavy mineral-rich sand dunes have been found which are now subject to mine industry development (Jagannadha et al. 2008). In the developing countries mining is one of the fastest growing industries, because it has a high income. On the border between South Africa and Namibia coastal sand hills are a mine area for diamonds while on Madagascar heavy minerals are mined (Lubke 2013). In the far south of the South Island in New Zealand, where Southern Alps connect to lowland near the town of Orepuki, the Gemstone beach is situated. This beach and the neighboring dunes contain such semi-precious gemstones as: garnets, orbicular jaspers, rodinguite quartzes, semi-nephrites or elusive sapphires (Fig. 14.12). These precious stones were traditionally mined by Maori people. After gold had been found in this volcanic sand, a gold rush broke out with its temporary settlement and transport investments.

Dunes can also be excavated in search of amber, that is deposited together with the sand building sandbars. The best known area for amber excavation is in the south east Baltic Sea coast in Poland and Russia. In Poland there are several small mining land plots rented by the companies. In Russia amber is excavated from coastal dunes and offshore sands on a large industrial scale. Some studies on offshore sand mining and its effects on the coastal systems show how dangerous it is for dunes and their ecological values (Hilton and Hesp 1996).

Fig. 14.12 Gemstone Beach and dunes, the place of sand excavation by early Maori people, New Zealand

14.5.4 Industrial Zones on Coastal Dunes

Near human settlements set up in coastal sandy areas many industrial zones have been located. The choice of venue usually serves economical purposes – the businesses are close to workforce resources, water supplies for cooling systems or

Fig. 14.13 Industrial zones on dune coast. (**a**) The Hadera power plant, Israel, (**b**) the Zeebruge harbor in Belgium and developed sand dune spit that is under conservation

production, transport lines and to harbors. However, with urban sprawl many power plants or new factories have been built directly on dunes in the coastal zones. Such zones can be found in most developed countries, from Europe through Americas to Australia and India or China as well. Those areas are totally devoid of natural landscape and ecosystem values. Some of these structures are close to urban and recreational zones, as e.g. a power plant near the town of Torremolinos in Spain or the Hadera power plant near historic Caesarea site in Israel. Often the coastal location of industry is due to the lack of land for development in the hinterland as in Hadera or Hajfa in Israel, Zeebruge LNG gas harbor in Belgium, Rotterdam harbor and many more (Fig. 14.13). There are several nuclear power plants built on the low coast in the United Kingdom, Israel or India. The largest world harbors are located in low sandy coastal areas. Nowadays, they are also built on artificial offshore islands, due to the lack of land. Amongst them the biggest are: the Rotterdam, Le Havre, Calais or Singapore harbors. The large jetties disturb sand transport along the coast, usually causing coast progradation and dunes development on the one hand and erosion on the other. However, such structures may lead to new supply of land which is valuable and needs to be protected as in Zeebruge in Belgium (Herrier et al. 2002). There are also questions whether it is appropriate now to build ports on the valuable natural stretches of coast? Economic and political interests led to new industrial developments in the valuable natural dune areas. This happened in Poland, where in 2012 a new LNG harbor was built on a natural part of the sandy dune coast of high ecological values near the town of Swinoujscie (Łabuz 2013).

14.5.5 *Turistic and Recreational Industry*

In Europe wide beaches neighboring dune coasts have been seen as a recreational area since the eighteenth century. Low sandy coast, beautiful beaches, adjacent lakes and river mouths (or deltas) full of animals are ideal for the development of

Fig. 14.14 Tourism impact examples. (**a**) Hotels built on dunes and mass tourism causes a variety of degradation of the natural landscape, Torremolinos touristic resort, Spain. (**b**) Leisure on dune destroys vegetation and increases deflation, Tallinn, Eastland. (**c**) The beach harrowing and raking for cleaning purposes destroys organic matter and pioneer beach habitat, Tel- Aviv Israel. (**d**) Typical American and Australian leisure method – off road vehicles on the beach, but there are also destroyed dunes by irresponsible drivers, South Australia

different types of tourism. They are a perfect place not only for sunbathing but also for ecological and educational tourism and recreation.

Nowadays mass coastal tourism is perceived as a main impact factor affecting the coastal natural environment (Fig. 14.14). There are well known examples of this impact such as the North America sandpit barriers (Buerger et al. 2000), or the European coasts in Spain, France, Belgium, Germany, Poland, Bulgaria, Netherlands or the United Kingdom (Petrova and Apostolova 1995; Piotrowska and Gos 1995; Buerger et al. 2000; Gómez-Pina et al. 2002; Łabuz 2004; Arens et al. 2013; Hapke et al. 2013 and others). Since the late 1980s' many Spanish coastal resorts have suffered considerable economic decline caused by the devastation of local natural environment (Pollard and Dominguez Rodriguez 1993). These serious tourism-related threats are clearly evident in Malaga on the Mare Manor barrier, where dozens of high rise hotels have been built in tourist resorts. Torremolinos is a typical coastal town where since 1960s the devastation of natural values has transformed the coast into a flourishing tourist business. Ironically, because of the destruction of its natural environment, the town has been losing its tourist attractiveness. In the La Bota beach (Huelva) or the Liencres Spit near Santander in Spain coastal dunes were degraded by off-road vehicles and human trampling (Gómez-Pina et al. 2002). The problem of illegal off road vehicle rides is notorious on many world dunes, which has been described at length in Brazilian publications (Costa and Melo e Souza 2009; Portz et al. 2010). On the other hand there is famous racetrack build on dunes in Zandvoort in Netherlands. Coastal dunes are widely used as golf courses, around the world. In areas of their occurrence dune relief and vegetation is strongly transformed. Firstly they were introduced on United Kingdom dunes, and now are part of each larger touristic resort in the world.

14.5.6 Coastal Dunes as Country Borders, Military Camps and Protection Zones Against Enemies

Coastal dunes may form natural, strongly protected borders between conflicted countries, the countries facing illegal immigration or smuggling. Such a strongly protected and fenced dune coast separates Mexico from the United States to the south of San Diego. The fence is several meters high, but built as an openwork designed for water flow and sand movement. A very similar construction divides the two countries on the sandy dune coast of Mexico Gulf where Rio Grande riverbed is a borderline. Sometimes the river is not connected to the bay, which makes it difficult to protect the border and makes it easier to cross.

Another, but more problematic coastal border is the one between Israel and Jordan on the coast of the Red Sea, where barbed wire and minefield separate natural habitats, beaches and low forms of coastal dunes (Fig. 14.15a). A similar situation is on the Israeli border with Palestinian Gaza where military patrols drive in heavy vehicles along their routes through the dunes. Nearby there are the Israeli military bases and outposts sentry. What is more, in Israel the coastal dunes situated to the north of the border are both the nature reserve (the Nizanim Nature Reserve) and the safety belt for the nearby cities of Ashdod and Ashkelon, which are constantly suffering from rockets fired from Gaza. Fragments of these missiles can be found on the dunes which are only periodically available for tourists. Also, at the Polish border with Russia on the Vistula Spit, the 200 m wide belt of the Baltic Sea coast is fenced which makes it unavailable to people and animals.

Coastal dunes as lands remote from urban zones are used for military training camps or are covered by such special military structures as airports, rocket launch sites or bunkers (Fig. 14.15b). A good example of such a former military training camp can be found in Poland where on the shifting dunes of the today's Slowinski National Park German troops were preparing for the invasion in Africa during WWII. Nowadays probably each country has such a camp within its borders. These areas are under military jurisdiction in which special security measures are employed to prevent unauthorized entry because of heavy equipment use and strategic purposes. These camps can be found by means of Google Earth maps, but I will not deal with the description of their purpose in this chapter. My knowledge and observations allow to describe the overall effect of such use on the dunes environment.

Fig. 14.15 Military use of the dunes. (**a**) State border between Israel and Jordan on the coast of the Red Sea. (**b**) Military training ground with destroyed tanks, Tunisia. (**c**) Bunkers of Atlantic Wall in Belgium. (**d**) Bunker on Bribie Island, Queensland Australia

Dunes are trampled, destroyed by explosives and leveled by heavy machinery such as tanks. During WWII or the Cold War kilometers of bunkers traps and trenches were dug on coastal dunes. Dune relief was good for building underground bunkers and shelters. The sandy coasts in France, Belgium to Netherlands and Denmark are strewn by bunkers of the so called Atlantic Wall (Fig. 14.15c). Some of the beaches and dunes in France, as Omaha Beach are witnessed some of the bloodiest battles during WWII in June 1944 during the Allied landing in Normandy. On Omaha Beach coast were killed hundreds of soldiers.

Some of the former bunkers are used as museums, storages or shelters for grazing animals (France, Belgium, Netherlands, Germany, Poland). Due to permanent erosion many of them were eluted from the retreating dunes. In Denmark they are treated as obstacles increasing water erosion and are going to be removed. Such structures are now a tourist and historic attraction, therefore they are protected by special measures against erosion. The example of bunkers protected by means of sand bags can be found in the Bribie Island National Park in Eastern Australia (Queensland) near Brisbane (Fig. 14.15d). These unique structures may be used as indicators of coast progradation or erosion. In the areas where there are no other landmarks of coastal processes, after 70 years they can be found on beaches where erosion prevails while the others are buried in sand, tens of meters from the present coastline where coast progradation has been observed (e.g. Calais in France).

14.5.7 Coastal Dunes Used for Protection Against Land Erosion and Flooding

The sea-level rise, coupled with storm surges and high tides, will ultimately pose severe problems for beach managers, coastal engineers and tourism business (Phillips and Jones 2006). In the past, coast protection meant heavy structures and the approach to stop erosion at all costs – hold the line. The basic method of shore protection is hard structures: concrete bands, rock boulders or artificial ridges – shielding the destroyed waterfront (Nordstrom 2000). These procedures restrict the natural development of the coast and deepen the erosion process. The longer the section of coast artificially protected by hard structures, the smaller the material load from the so far eroded coast. The construction of hard structures leads to slow degradation of natural processes and coastal habitats. To tackle the problem, some countries discourage investments in the areas threatened by erosion by changes in law, limiting building permits, reducing the investment insurance against erosion. In many countries dune belts are treated as safety zones against erosion or flooding that cannot be built-up.

Erosion is a natural process on some coasts which may provide sediment to rebuild sandy coast in other places. However this process is undesirable in urban areas and other places of human activity. Many examples of the developed countries show that because of private household or tourist infrastructure located on coastal dunes certain measures for the sake of coast protection must be undertaken (Fig. 14.16a).

Heavy storm surges developed by hurricanes are the main problem for the coast safety on the USA south and east coast. It has been observed that during extreme storm surges, dunes and beaches of narrow barriers may be washed out (Fritz et al. 2007; Houser et al. 2008). Foredune and beach height, relative to storm surge run up, is a primary factor responsible for coast protection.

Coastal dune ridges are natural dykes protecting coast against erosion. Reinforced by wide beaches, developing by aeolian accumulation processes, they constitute a buffer zone between the sea and the land. This zone is constantly being remodeled and rebuilt after storm erosion, resulting in the accumulation of sediment – the reconstruction of beaches and dunes. This process is known as cut and fill. Of course, the reconstruction requires material from rivers, the sea and, in particular, from the destruction of adjacent sections of the coast. The hard regulatory work on rivers cause a decrease in the amount of material delivered to the construction of the beaches and dunes. In present times the availability of sand is rapidly decreasing due to river regulation schemes and to shrinking of the eroded

Fig. 14.16 Coastal dune erosion issues. (**a**) Coastal erosion, dune destroyed by the sea, Salerno Italy. (**b**) Covering eroded dunes by rock blocks, near Kolobrzeg, Poland. (**c**) Instead dunes concrete dyke with promenade, the removal of naturally accumulated sand disturbing walkers, Koksijde, Belgium. (**d**) Artificial sand dyke resembling dune, Lido on the Venice Spit, Italy

coast sections (Aminti et al. 2002). Hard engineering structures have traditionally been used to manage storms and tides for the protection of developments within the coastal zone. Since these structures are expensive and tend to promote erosion, alternative soft engineering techniques that work in conjunction with natural coastal processes are increasingly being used (Nordstrom 2000). But still dunes are reinforced by hard structures or are occasionally and spontaneously rebuilt as the dykes to protect against storms (Fig. 14.16).

Nourishment of beach seems to be the best protection technique, less affective for nature (Nordstrom 2000). Nowadays dunes are rebuilt artificially, created from anew in front of tall hotels and promenades (Feagin 2005). In a long term, human actions can stop coast erosion, as it was presented in a study of the USA north-east coastline where since the nineteenth century the length of eroded coast has decreased by 8 % (Hapke et al. 2013). On the narrow island of Avalon in New Jersey the first program of coast protection by nourishment was done in 1987 and a project of dune restoration by fences was started in the early 1990s (Nordstrom et al. 2002). Artificially created dunes are the main goal for low coast protection measures in the Netherlands. Nowadays dunes are also

Fig. 14.17 There are two basic types of forms of protection of the dunes: (**a**) National Parks, (**b**) Nature Reserves

restored or rebuilt by means of natural vegetation (Nordstrom et al. 2002). The plantations of non-native species do provide excellent stabilization for this valuable landscape (Feagin et al. 2010).

14.5.8 Coastal Dunes as Environment Protection Areas

After years of coastal dunes transformation for human purposes, it is time we began our fight for the restoration of the natural character of coastal dunes. This is mainly observed in countries and areas where natural landscape has been considerably transformed, such as Belgium, the Netherlands or the United Kingdom. The nature reserves or national parks are a solution for providing protection of the last natural sections of coasts. There are a lot of such parks around the world. Worth mentioning are the Zuid-Kennemerland *National Park* in the Netherlands, Ainsdale Sand Dunes National Nature Reserve in UK, Réserve Naturelle Dune Marchand on the border between Belgium and France, the Nizanim Nature Reserve near Ashdod in Israel, the Lençóis Maranhenses National Park near São Luis in Brasil, the Slowinski National Park in Poland or the Walpole National Park and the Coffin National Park in Australia and many more (Fig. 14.17). These are ones of the largest famous parks opened for tourism and protecting the natural character of dune coasts. The purpose of these parks has been to preserve different coastal landscapes from coastal waters, swamps, to shifting dunes to dune heat. The Bribie Island National Park in Eastern Australia near Brisbane is also connected to a large recreational park. Both of them form a large area of undeveloped land, being the natural heritage and a good place for recreational and educational purposes. The Netherlands coastal reserves are wide open for non traffic use, mainly for hikers or cyclists, and a place where nature of dunes can be admired. They have also utilized for one more function – the Amsterdam Waterworks have been located on grey dunes, providing supplies of drinking water for the local community.

14.5.9 Coastal Dunes Perceived as a Worthless or Invalid for Society

In areas where fundamental problems of existence are of the most vital importance, coastal dunes are still treated as wastelands or herding grounds (Arun et al. 1999; Rosa and Cordazzo 2007; Defeo et al. 2009; Everard et al. 2010; Portz et al. 2010), the latter not being blameworthy. In the areas where ecological awareness of the need to preserve the natural environment loses to the need for survival, coastal dunes are not protected. In these parts of the world dunes are treated as a waste disposal site. The Sergipe, Praia do Jatoba on the south coast of Brazil is a small village exemplifying ecological problems with coast management (Costa and Melo e Souza 2009). Coastal dunes are inhabited, but the local community has been disposing of waste there. Another example are tourist resorts on the coast of Tunisia were waste is dumped directly onto the dunes and the beach. It is a result of poor awareness about environmental protection needs. Waste disposals, garbage and litter from the sea is an increasing problem for the ecology of coastal areas. One of the first studies on the qualitative and quantitative composition of litter disposed of on beaches presents a wide range of its types (Tudor et al. 2002). Tudor's study on the United Kingdom beaches show that plastic containers and their parts are its main components. They can be found in the so called washover fans or accumulation fans on the beaches or behind foredunes busted by storm surges. But this kind of surveys are still rare, and need to be undertaken.

This is the result of our civilization's cultural development, the impact of which varies in different parts of the world. Is this problem typical of developing regions? Not entirely. The problem of littering and dune devastation affects all countries. Everywhere beaches and dunes are littered by garbage deposited by the sea, mainly on accumulative coasts where water carries a considerable amount of packaging and artifacts of anthropogenic origin. The problem is mainly plastic packaging getting into the seas. After storm surges, it is collected in storm surge washover fans and on the beaches.

In many areas dunes are illegally used for off-road sports. Coastal habitats are destroyed by people recklessly driving their sports utility vehicles (Kutiel et al. 2000a). Local vegetation is devastated and the animals scared off. Even in the developed countries, illegal pathways or parking lots as well as vehicle raking are very popular (Bonanno et al. 1998; Peach 2006) Plants destruction by trampling and by off road cars is a well known problem (Phillips and Jones 2006; Santoro et al. 2012). An experimental study has been conducted which confirms that coastal vegetation is disturbed as a result of the increased off-road vehicle traffic, which is manifested by macrofauna densities lowered by 27–52 % (Walker and Schlacher 2011).

In conclusion, there are two main sources of waste pollution on the discussed area (Fig. 14.18). The first reason is throwing garbage into the sea. Generally, garbage is carried by the sea onto the beach together with typical natural wood and parts of plants (as seeds, roots and shells, that are creating on beach valuable habitat for settling pioneer dune creating plants). How they find their way to the sea is a completely different problem. The second source is garbage thrown by people

Fig. 14.18 There are two main sources of waste pollution on coastal dunes: (**a**) pushed into the sea, Tunisia. (**b**) Directly left on dunes, Israel

directly onto the dunes. In some countries, dunes that are worthless even for agriculture are used as landfills. Large amounts of waste is produced in the developing countries where people exploit nature to the maximum for the benefit of economic advancement. These garbage is thrown into the rivers or directly to de sea.

14.5.10 Diversified Utilization of Coastal Dunes

Coastal dunes have been or were used for many purposes, including both the sanctified and infamous ones. To the south to Madras in India there is Shore Temple (in Mahabalipuram) which is one of the oldest such structures. The Temple, dated on 700 AD, is permanently affected by storm surges and tsunami waves (e.g. on December 2004). It was built on a narrow promontory protruding from a sandy coast. In the 1990s in Casablanca in Morocco Hassan II Mosque, the third largest mosque of the world, was erected. The structure is located on an artificial bank constructed on a formerly low sandy coast covering shallow rock deposits. Ten years later, in 2004, concrete waterbreakers were put up to protect its walls against waves. Now they are to be demolished, because they do not become the structure splendor. Nearby, a modern shopping center – the Casablanca Mall being the second largest in Africa with its 200,000 m^2 of floor space – was built in 2011, only 50 m from Atlantic Ocean where sandy coast is slowly retreating. Now it is protected by a heavy longshore waterbreaker but in the future, due to retreating surrounding areas (mainly on the south coast) it will need better protection. In times when many areas have to deal with the problems with the protection of structures built on soft coasts and when people are trying to make their best to preserve natural coastal processes, it seems that some communities do not consider the problem of retreating coasts a vital issue. The Arab Emirates are an example of ignorance in coastal processes and future sea level rise.

Sandy dunes of relatively little value are sometimes used as cemeteries. In Poland there are three cemeteries on the coastal dunes. One of them is located on the first erosive ridge and the graves are 50 m from the beach. Similar utilization of coastal dunes can be seen in Morocco, which follows from the fact that the beaches are not used for full known leisure by the locals. Also in Australia first, old cemeteries can be found on sandy dunes. Usually it is practical rather than religious approach that determines such decisions.

As it has been mentioned, dunes serve not only as the location of military camps, but can also be a useful venue for other military purposes, such as testing the V1–V2 rocket weapons that were used by the Germans during WWII. The tests took place in the area which today is a part of Poland – on the Lebsko Lake Sandbar covered by wandering dunes. When sand moves it is possible to find the remains of the German military buildings, communication network or missile fragments. What is more, the Polish dunes were used for even more evil purpose – the German Nazis' concentration camp was located on the Vistula Sandbar. About 400 m from the shore, between dunes, during WWII the Germans established a concentration camp, together with the whole mass extermination infrastructure: the crematoria. Forest and dunes adjacent to the shore witnessed the atrocities of genocide.

14.6 Management of Coastal Dunes

Coastal dune management is defined as all measures aimed at the preservation and restoration (Table 14.3) of the natural values of a coastal sand dune area (Gómez-Pina et al. 2002). It is essential that beach and foreshore are seen as parts of a whole dune system. Management must take into account all processes within this system (Van der Meulen and Salman 1995).

For the management purposes several steps are usually undertaken:

– planning – aimed at proposed actions to be taken in order to achieve the desired state of the environment,
– survey – an action undertaken to measure and record changes in the environment, to determine the direction of change and the characteristics of the environment variables,
– carrying out techniques – activities aimed at the environment rebuild, restoration, conservation or preservation,
– monitoring – planned observations, surveys to determine ongoing changes caused by the previously undertaken actions.

The number of publications describing ongoing works on dune management is increasing rapidly. Between 1990 and 2004 the number of publications increased 12 times (Bonte and Hoffmann 2005). The majority of the studies (59 %) were conducted in the European coastal dunes (incl. Israel), followed by North America (22 %) and South Africa (9 %). In the analysis, both the above mentioned authors quoted a small number of studies in on coastal dunes from Australia, South America

Table 14.3 The management activities, goals and undertaken actions in costal dune environment

Coverage to the main management activities	Coast protection (Hard structures protection – Reconstruction)	Restoration (Reconstruction – Re-mobilisation)	Conservation (Stabilisation – Conservation)		Preservation (Conservation – Protection)	
Management in general	Hard structures protection	Reconstruction	Stabilisation	Re-mobilisation	Conservation	Protection
Goal	Land protection against retreat (erosion)	Coast protection, dunes creation, human property protection	Landscape management, dunes creation, human property protection	Landscape conservation with bio and geodiveristy increase, rejuvenation	Environment conservation in general (bio and geoconservation), habitats change eg. rejuvenation	Protection of whole environment: habitats, bio and geodiversity
Dune system element under action	Undertaken actions					
Sand surface	1. Filling gaps by other, no dune sediment 2. Artificial dykes 3. Hard structures covering slopes: bands, seawalls	1. Stabilisation (wooden fences, plastic nets, fashine fences, brushwood) 2. Sand nourishment on beach and dune 3. Dune ridges rebuild 4. Hard protection	1. Wooden fences, plastic nets, fashine fences, brushwood	1. Dunes destabilisation by blowouts creation, foredune breach	1. Cleaning with garbage and litter	1. Comprehensive protection by not interfering: Reserves, National Parks or no intervention

(continued)

Table 14.3 (continued)

Coverage to the main management activities	Coast protection	Coast protection / Restoration	Restoration / Conservation	Restoration / Conservation	Conservation / Preservation	Preservation
Vegetation	1. Different origin plants to stabilize slopes and ridges	1. Planting sand binding grasses (typical *Ammophila* sp.) on slopes, ridges, in blowouts 2. Native plants planting	1. Planting sand binding grasses (native more often and typical *Ammophila* sp.) 2. Plants removal from embryo dunes on beach 3. Afforestation and reaforestation	1. Grass removal (by hand, fires, machinery, chemicals, mowing, grazing) 2. Sod-cutting 3. Shrub removal (machinery) 4. Deforestation (felling, fires)	1. Grass tillage (removal by hand, planting, mowing, grazing) 2. Shrub removal (machinery) 3. Deforestation (felling, fires) 4. Invasive plants removal (machinery, mowing, chemical spray) 5. Native plants planting	1. Comprehensive protection by not interfering: Reserves, National Parks or no intervention
Animals	1. Not important	1. Not important	1. Limits for herbivores (indigenous and previously introduced))	1. Introduction of herbivores for different species grazing	1. Introduction of herbivores for different species grazing 2. Introduction of indigenous species 3. Getting rid of pests	1. Comprehensive protection by not interfering (as above)

Nutrients, soil	1. Not important	1. Nutrient input supporting soil development	1. Nutrient input supporting soil development	1. Soil removal	1. Soil removal	1. Comprehensive protection by not interfering (as above)
Surface and ground water	1. Drainage system for construction stabilization	1. Channels creation: foredune breaching 2. Ponds creation	None	None	1. Limiting recreation activities 2. Fences 3. Reserves est. 4. Outside pollution reduction	1. Comprehensive protection by not interfering (as above)
Human presence	1. Still present 2. Dykes used as bicycle roads or pavements 3. Bands used as recreation place or even parking place	1. Trampling and recreation restriction 2. Limiting activities, settlement and use 3. Public information	1. Trampling and recreation restriction 2. Limiting activities and use 3. Education and awareness increase	1. Limited permission for trampling 2. Recreational activities limits 3. Education and awareness increase	1. Trampling and recreation restriction (by fences and information boards) 2. Paths creation (fenced on sand, footbridges above sand) 3. Education and awareness increase	1. Comprehensive protection by not interfering (as above) 2. Rare paths for educational purposes 3. Highly specialized education 4. No enter

and Asia (Bonte and Hoffmann 2005). Since that elaboration more publications have appeared in South America, Australia and more in Europe and North America. These studies are mainly focused on the future sea level rise in relation to climate change and, of course, on the development of best management practice and the techniques to assess the value of coastal dune environment. With this new approach researchers and managers are expected to take steps to create natural coastal dune environment. There is still no research reports concerning North Africa, South-East Asia and the Arab countries. Please find below a brief outline of the world research on coastal dune issues that has been conducted over the past 20 years.

14.6.1 Present Perception of Coastal Dunes

The problem of shifting sands burying settlements was dealt with by planting vegetation. initially, this method was used widely on the old continent, and later it spread to the colonies overseas. Gradually, fixation of 'troublesome' dunes was introduced in other countries. The stabilized coastal dunes immediately were colonized by human settlements and business. Over time, it was understood that the coastal dunes were of a great intrinsic value, their diverse landscape and natural dyke protecting people from the destructive storm surges. Due to growing ecological awareness countries began to take care of the dunes and their habitats, their bio- and geodiversity that was rich in unique plants and animals. People realized that the dunes are the source of sediments which could be used for building up the natural coasts. Now the trend is reverse. Naturalization has become a worldwide goal for coastal dune management. After years of dunes stabilization and afforestation, steps have been taken to rebuild dunes for developed coast protection. Consequently, the new approach has resulted in the restoration of the natural character of the dune environment and its variety. The next step is the dunes re-mobilisation and habitats restoration to re-mobilize the dunes and to revive their habitats so much disturbed by high fertilizer load and grass and shrubs encroachment. In many areas where due to civilizational progress people fail to protect this ecosystem, dunes still are not regarded as utilitarian value.

14.6.2 Management Issues

In terms of scientific studies the management of coastal dunes is understood as (1) nature restoration or conservation carried out by establishing nature reserves, (2) stopping the moving dunes by stabilizing vegetation or putting up fences of different origin that catch the shifting sand, (3) re-mobilization of stabilized relief (4) regulation of the vegetation richness (5) removing non-native species (plants or animals), and (6) re-introducing endangered species of animals. Other goals of coastal management are the reconstruction of coastal dunes degraded naturally or artificially and their protection against erosion by means of hard structures or beach and dune

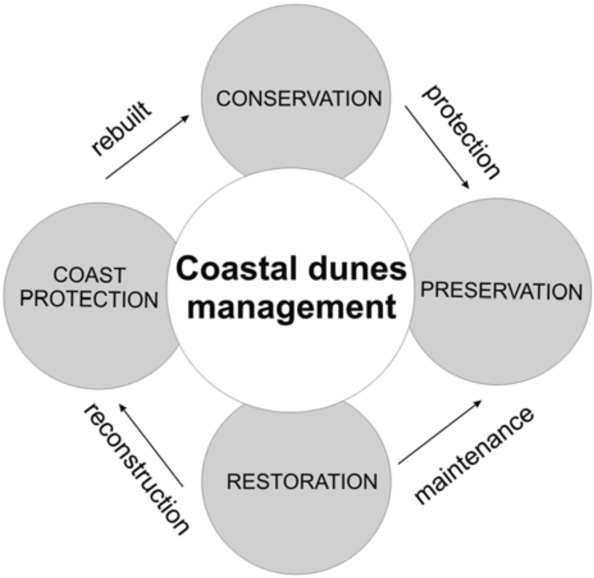


Fig. 14.19 The relation of management methods for coastal dunes

nourishment also for land and human property protection. To put it briefly, the purpose of management is to preserve the natural values of dunes, restore their natural or semi-natural values or to protect their environment and local communities against the effects of its changes, even the natural ones. No action in management may be deliberated still as action or the reason of lack of interest in community planning. This is evident in the developing countries where there are other problems that are more serious than the protection of dune environment. For instance, coastal law in India is ineffective and fails to manage properly its long coastline and coastal resources that are being degraded due to increasing population (Puthucherril George 2011).

Coastal dune environment management can be understood as (Fig. 14.19):

– dunes protection with hard structures built to defend the coast against storm erosion, retreat, loss of natural values and human property,
– dunes restoration– reviving the natural and landscape values of degraded dunes in order to protect local communities against landscape changes and erosion consequences,
– dunes conservation – understood as defending natural values from natural and man-made degradation, protection of biodiversity and geodiversity of forms,
– abandoning intervention or actions towards the natural environment development in places where preservation areas have been established.

14.6.2.1 Dune Protection with Hard Structures

There are different methods for dune coast protection. Some countries implement hard coast protection, other prefer soft or limited intervention. Also, some areas can be found where the settlement retreat from the dune ecosystem has been proposed.

These hard structures are still being in use in the areas where coast retreat is endangering local community and local properties. The town of Galveston located on the low coast of Mexico Gulf is an example of human fight with flooding. A seawall, constructed there at the beginning of the twentieth century after several hurricane floods, protects well the rebuilt town. unfortunately, the local beach has vanished. As years are passing by and the next destructive hurricanes come, new areas are being eroded. This is why an urgent need of new coast protection has arisen (Merrell et al. 2011).

Hard engineering methods include: (1) sea walls, (2) submerged waterbreakers and jetties, (3) artificial dykes, (4) groins, and (5) other artificially established structures. Soft methods, including the biological ones, are focused on dunes or beach rebuilding. They include: (1) nourishment, (2) natural vegetation plantation, which favors accumulation and sand stabilization, (3) wooden or plastic sand fences or even covering deflation surfaces by shrubs or branches, (4) plants removal by humans or animals.

On the other hand engineered protection structures: dikes, seawalls, waterbreakers, piers, and even the plantations of nonnative species do provide excellent stabilization for this valuable landscape (Feagin et al. 2010). The question is if hard protection measures are the best method? There are still areas where hard protection is seen as the optimum way to fight coast erosion. Breakwater construction in the Douro estuary has successfully protected local natural environment against sea erosion (Bastos et al. 2012). If houses cannot be removed from the coast, the only method is to take more hard protection measures. This modus operandi is still preferred in countries where coastal managers or coastal services do not realize that environment is undergoing permanent change. Of course, it is beyond dispute that in the areas where human property and life is truly at risk, the concrete bands and waterbreakers will continue to be established.

In some countries hard methods are a necessity because sand for nourishment is a scarce commodity, because, due to its nature, the low lying coast cannot be protected in any other way, and because of political solutions and financing. Submerged waterbreakers or sea walls are popular in Italy. Italian coast is rocky with several low dune sections that are poor in the sandy material. It is not possible to rebuild dunes, i.e. to find such amount of sand that will protect human settlements. As much as 34 km of Rimini coast are protected by waterbreakers made of boulders. Another good example is Lido di Venice where sandspits are protected by means of numerous hard engineering structures. Nowadays, huge storm gates are being constructed to protect the Venetian Lagoon and the city of Venice against flooding. Artificial dykes made of concrete and covered with an artificial dune is a popular method of coast defense in low lying areas in Netherlands. In Poland, since 2009 over 15 km of the coast has been covered with concrete or boulder sea walls and bands – everything financed from the European Union funds.

In certain circumstances hard structures are not enough to protect the coast. In 2011 the earthquake produced the disastrous tsunami that flooded the Japanese coast. The tsunami reached heights of 19 m above the mean sea level. The approximately 9.5 m high wave was able to break artificial dikes that were 6.2-m amsl high and flooded the town of Arahama (Takaaki Uda et al. 2012) where scouring holes were formed in the sandy material that was building the coast. Behind the dike deep trench was formed. After the 3.11 earthquake, as it is referred to in Japan, it has become obvious that we must develop new earthquake forecast tools. Worth mentioning is the 2004 tsunami in Indonesia, and 2013 in the Philippines, where low coast without the dunes have been completely submerged and developed areas destroyed. New Zealand, which is often hit by tsunamis, has been developing a wide public awareness network, starting from warning signs to tools preventing flooding. Increasing sea level or storminess will require the use of hard protection measures. In North America, where sandy spits and dune systems are suffering heavily from hurricanes (Feagin 2005; Feagin et al. 2005), protection measures are taken with the view to land owners' needs, to protection of private properties, of agricultural and settled areas and also to those who wish to preserve the natural environment (Doody 2002). Change is an important component of coastal systems and sand dunes that are well adapted to accommodate environmental perturbations. The perception of low and soft dune coast protection is changing. Nowadays more natural, 'soft' techniques are used because dunes are natural structures that protect coast against flooding. In many places dunes are artificially rebuilt, nourished and covered by native, sand binding grasses. The idea of coastal dunes being the natural coast protection is popular in Netherlands, where high foredunes have been achieved as a result of beach nourishment, grazing and vegetation plantations (Kooijman and van der Meulen 1996; Terlouw and Slings 2005).

14.6.2.2 Dune Restoration

The term restoration implies that attempts have been made to recover the form, function, and species inventory existing prior to human modification. Dune restoration is primarily required when natural dunes have been significantly modified or damaged by human activities. Natural environment restoration may have biological consequences, as it was observed in Denmark (Andersen 2002). Restoration efforts must be based on ecological, geomorphological and social criteria to maximize the goods and services dunes can provide (Provoost et al. 2004; Dahm et al. 2005; Roig et al. 2009; Feagin et al. 2010; Santoro et al. 2012; Nordstrom and Jackson 2013; Hesp and Hilton 2013). The so called soft techniques in dune restoration are now amongst best coast protection techniques (Gómez-Pina et al. 2002; Favennec 2002; Matias et al. 2005; Dahm et al. 2005; Roig et al. 2009; De Lillis et al. 2004; Arens et al. 2005; Bezzi et al. 2009; Nordstrom and Jackson 2013; Santoro et al. 2012). Dune restoration can be achieved by means of the following methods:

1. Creation of new dunes.
2. Favoring the dunes growth.
3. Social participation.
4. Limitation of human pressures.
5. Limitation of human access.

14.6.2.3 Dune Conservation

Coastal dune conservation is combined with the well known dunes restoration and preservation techniques. However, it is based more on active protection of the existing valuable dune habitats (Heslefeld et al. 2004; Houston 2005). Such techniques as sand stabilization or plant removal and grazing are typical of sand dune conservation (Plassmann et al. 2010; Millett 2013). In this approach dunes are seen as a more or less natural habitat that should be protected without any significant human impact. The dune conservation method uses techniques similar to restoration but it focuses more on preserving the natural character of the environment. The main goals for conservation are:

- to protect and maintain environment biodiversity,
- to protect and maintain landscape heterogeneity (geodiversity).

Conservation should preserve the existing habitats and species. Of course, the conservation techniques may be under discussion, but its main aim is to maintain balance between human use and natural protection. The conservation methods are devoid of direct action in sand surface and plans changes. They rely on restricted human influence on sand movement, using sand fences for sand stopping, as well as on native species diversification in order to conserve the typical dune landscape and environment. Nature conservation may be opportunity, realized due to new urbanization realization. In Belgium, Zeebruge harbor neighbors a few protection areas, such as a small dune that has developed following the harbor waterbreakers construction (Fig. 14.23b). That small section with a newly developed natural dune-beach system is now a protected habitat area for plants and birds (Herrier et al. 2002). In several other dune areas, biodiversity is expected to thrive after local management plans have been accomplished. It is worth mentioning that conservation is varying from undertaken activities. The conservation methods are focused on the health of the existing habitats rather than on intervention. They rely on the impact on the environment elements.

14.6.2.4 Dune Preservation

Preservation is a method of doing ‘just nothing’ to the environment. This method is employed in national parks or reserves that have been established for preservation purposes. The main idea is to do nothing in the dune environment. However, protection measures in parks differ from country to country as a result of varying national legal regulations.

Most of the other, remote dune areas are under protection. In such areas people refrain from any kind of action, or they cannot rich remote dune areas. However, in countries where dunes are largely transformed, even interfering in the process of restoration is seen as the restoration itself.

14.6.2.5 Dunes Failure to Interference

When natural dune areas are far from human settlements, they do not need protection. Are there still such areas on the Earth? It is true that there is not much natural dune environment left free from human interference. Such areas can only be found in remote and sparsely populated areas of the coasts of Canada, Mexico, Australia and New Zealand. Yet, even in these areas management plans aiming at environmental protection are implemented. The most effective protection measures are reserves and national parks. In the remote Australian land you can still find wild and untrodden dune areas. Although these areas seem to be completely desolate, they are subject to some kind of management. The entrance to parks is always paid, so that the costs of security guards, paths, board signs, brochures and the construction of recreation spots for tourists can be covered. But still are opened for research, where no one caused changes in natural environment and phenomenas' shaping landscape.

There are undertaken actions for less intervention or even managed realignment even in the densely populated and intensively used for recreation stretches of coastline dune. On the German Baltic coast in the Nature Reserve of Hütelmoor Heiliger See suppressed from the use of coast protection of heavily eroded narrow dune sandbarrier (Weisner and Schernewski 2013). Its authorities announced that coast may be broken by sea and dunes may slowly retreat entering wetlands in the reserve. Such example is rare and courage when public demand is to protect land, because from other hand in neighbor Poland almost 40 mln of euro is being spend for 8 km long, hard measure protecting uninhabited narrow strip of sand barrier with shallow lake on Kopan Lake Sandbarrier.

14.6.3 Management Purposes

Management objectives are implemented through a number of different management or just maintenance techniques (Fig. 14.20). Those will be described below. It is worth mentioning that these techniques are used in combination to achieve the correct target in dunes management.

14.6.3.1 Dune and Beach Nourishment and Rebuild

The beach quality has a major impact on the value of the coastal zone to both residents and visitors. The beach nourishment is a technique used to restore an eroding or lost beach. It involves the placement of sand fill along the shoreline to widen the beach and is the only beach management tool that serves the dual purpose of protecting coastal lands and providing a place for recreation (Phillips and Jones 2006; Nordstrom 2008; Bezzi et al. 2009; Klein and Osleeb 2010). A nourished beach may provide more space for natural sub-environment development (Nordstrom

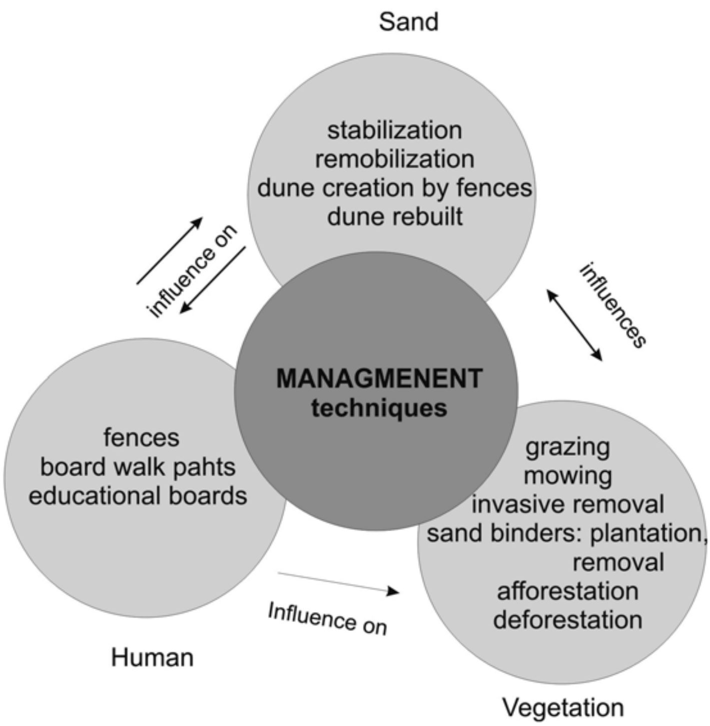


Fig. 14.20 The main management techniques undertaken in coastal dunes and relation of environment components

2008). On the widened beach new embryo dunes may form, due to sufficient sand supply and enough protection against storm surges washover.

However, the beach nourishment always influences neighboring dune environment, its relief and vegetation cover (Vestergaard and Hansen 1992; Van der Wal 2004; Nordstrom et al. 2009). During strong onshore wind actions, the additional sand load from the beach is blown onto the dunes. This is an effect intended by engineers, which may lead to natural restoration of the eroded dunes. Of course, not without significance are the physical and chemical parameters of the sediment used for this artificial nourishment.

Moreover, dunes can be constructed directly by means of bulldozers, sand fences, or vegetation plantings (Nordstrom 2000; De Lillis et al. 2004; Van der Wal 2004; Matias et al. 2005; Phillips and Jones 2006; Bezzi et al. 2009; Hesp and Hilton 2013). This method involves dune (1) ridge total reconstruction as a more or less natural ridge; (2) ridge reshaping: the formation of gentle seaward slopes or filling erosion niches, gates and blowout bowls; (3) partial rebuilding of ridges by their heightening and extension (Fig. 14.21). Extending the seaward side of the dune is more common and occurs where (Nordstrom 2008): (1) landward part of a dune system or ridge is restricted by land use, (2) a higher dune is undesirable because

Fig. 14.21 Dunes and beach nourishment phases. (**a**) Beach nourishment and dune rebuild, Mrzezyno, Poland. (**b**) Artificial dune ridge built for flood protection, scattered reeds is to support the soil development, Spiekeroog Island, Germany. (**c**) Completely rebuilt foredune, Hohe Düne near Warnemünde, Germany. (**d**) Built already mature dyke, pretending to be foredune, Zandvoort, Netherlands

views of the sea would be restricted, or (3) the surface cover of the existing foredune is considered valuable. Nourishment of the landward side of the foredune is rare. Such artificial dunes are often built to unnatural shapes, resembling earthen dikes but actions can be taken to make these dunes appear and function more naturally (Feagin 2005; Nordstrom et al. 2007a; Nordstrom 2008). In Poland, a natural extension of the foredunes in areas where there are wide beaches are obtained by mowing the grass growing on embryo dunes on the upper beach. The aim of this method is to stop stabilization of sediment accumulated by plants and to encourage its move onto a proper foredune ridge (Łabuz 2013). Unfortunately, the collateral effect of this method are destroyed pioneer dune vegetation habitats.

14.6.3.2 Dune Stabilization

The stabilization of coastal dunes relies on several techniques, the aim of which is to stop the movement of sand that is blown by the wind (Nordstrom et al. 2009). The main reason for dune stabilization is to protect human property from the dune

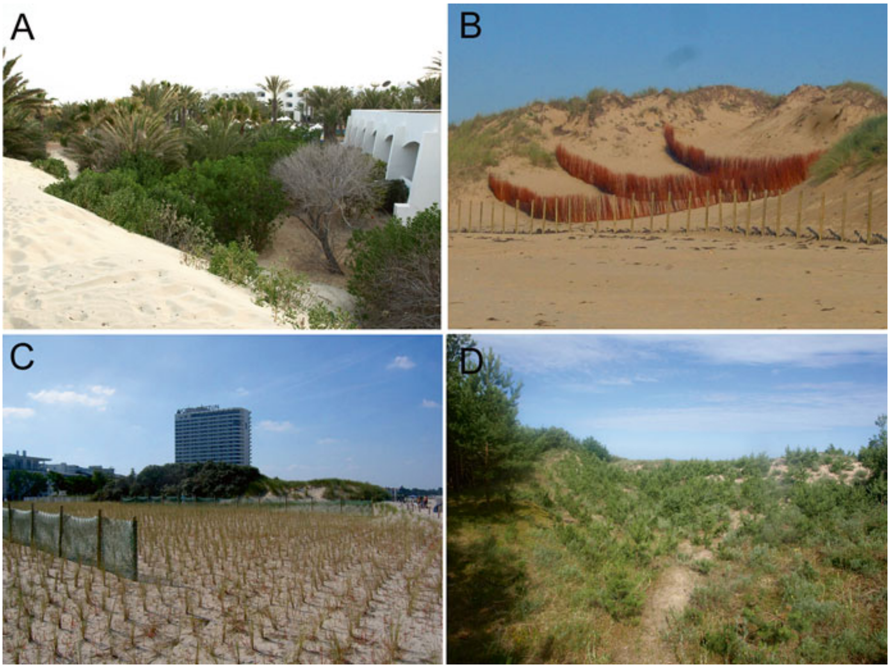


Fig. 14.22 Methods of dunes stabilization. (**a**) Not intended stopping of shifting dunes by vegetation surrounding hotel resort, Djerba Island, Tunisia. (**b**) Fashine sand fences stabilizing huge blowout, near Santander, Spain. (**c**) Marram grass plantation stabilizing reduced fast growing dune, that obscures the sea view of the hotels, Warnemunde, Germany. (**d**) Young pine plantation stabilizing dune heatland, but destroys the protected grey dune habitat, Poland

movement (Fig. 14.22). In order to stabilize the moving sand or to stop a shifting dune several action can be undertaken: (1) planting sand binding species, (2) afforestation of the formely shifting sand, (3) construction of stopping fences, (4) covering blow areas by brushwood.

The Marram grass is a main plant that may stabilize shifting dunes. This species is used around the world. In recent years, in order to stabilize the dunes and landscape protection using different species of native dune in specified areas (described in Sect. 14.6.4.3). The purpose of various fascine and wooden fences (discussed in Sect. 14.6.4.8) is to stop the sand. This technique is used not only to stabilize the moving sand but also to encourage the reconstruction and development of the dunes. Each rebuilt or reconstructed dune ridge is covered by plants to stabilize surface (Nordstrom 2008). The surfaces devoid of vegetation are covered with brushwood in order to increase the roughness of the surface and prevent further dismantling (Ovesen 2001). The history of dunes stabilization by afforestation is well known in Europe (Riksen et al. 2006; Danielsen 2008; Koster 2009).

In the area of Valdevaqueros (Cádiz, Spain) there is a 40 m high shifting transverse form that permanently buries an important road (Navarro et al. 2011). The

Fig. 14.23 Methods of dunes remobilization, by artificial breaks that may renewal aeolian processes. (**a**) Dune breach in Ijmuiden, Netherlands. (**b**) Dune breach and remobilization on France and Belgium border

efforts undertaken to stabilize this dune by fences or vegetation came to nothing. Nowadays, the plan is to build a tunnel over the road, over which sand will be shifting. In southern Brasil on the Torres to Tramandaí barrier the fast developing human settlement is endangered by shifting sand (Hesp et al. 2005). For that reason several steps have been undertaken to stabilize the sand. Dunes are stabilized also in areas where tourist infrastructure continues to sprawl. On the other hand shifting dunes may increase landscape management problems.

14.6.3.3 Dune Re-mobilization

Due to stabilizing activities local coast management policy and the climate change a coastal dune in Netherland have been stabilized (Arens et al. 2004). Actions have been undertaken by coast managers to re-mobilize some of the dune sections. There are few examples of dune processes remobilization around the world. In Netherlands this method is used to diversify the landscape of stabilized coastal forms (Van Boxel et al. 1997).

Another aim of this method is to rejuvenate the system. A study on Kennemerland parabolic coastal dune reports how for the period of 4 years the activated aeolian processes were shaping local environment and causing vegetations changes (Arens et al. 2004). Another place of dune re-mobilization is an artificial gate – a breach made in a foredune near Schoorl (Meerkerk et al. 2007). Similar actions were undertaken on the border between France and Belgium and near Calais in France (Fig. 14.23). Dunes destabilization can be generated through the blowouts remobilization (Van Boxel et al. 1997). In all cases the artificial interruption of the foredune was to increase the dynamics of dune relief by the supply of sediment from the beach and by allowing the inflow of water during storm surges in lower parts. These activities were intended to increase biodiversity and landscape diversity of quickly stabilizing dunes. The question is: will we lose the last active inland drift sands of Western Europe? (Riksen et al. 2006).

14.6.3.4 Dune Vegetation Maintenance

The dunes vegetation maintenance is a part of environment restoration and conservation processes which is focused on the growth of natural values. Vegetation maintenance consists in native species introduction, alien and native species removal, mowing and grazing. These methods have proven to be appropriate tools in dune grassland restoration (Provoost et al. 2002). The Netherlands was the first country that initiated programs of ecological values management (Provoost et al. 2004).

14.6.4 Techniques Used in Dune Management

There are several techniques of coastal dune management. Usually dune management combines more many techniques. Landscape may be preserved by invasive plant removal accompanied by native plants introduction. On the other hand plant removal encourages sand movement, that may be goal for geodiversity increase. These techniques – mainly field work activities – are aimed at environment rebuilding, restoration, conservation or even preservation.

14.6.4.1 Afforestation and Reforestation

These methods are based on forest establishment on the land that has never been forested (afforestation) or was formerly forested (reforestation). Both processes began in Europe. Some areas around the world are still being afforested in dune stabilization processes or as the protection against wind induced dune erosion. Afforestation and reforestation are mutually exclusive but they are used for:

- dune sand stabilization and protection,
- increase the agricultural productivity in dune areas (forestry or grazing animals).

The European history of dune stabilization is well known (Riksen et al. 2006). One of the examples is western part of Denmark where until 1867 almost 5 % of the total arable land had been destroyed by drifting sand formations, some of them reaching more than 10 km inland and forcing the relocation of entire villages (Stadsgaard 2004 after Wilkie 2002). Similar problems occurred in France, Belgium, the Netherlands and in Germany (Koster 2009). References to the damage caused to arable land and buildings by drifting sand in Denmark started to appear in the fifteenth and, in the Netherlands, in the sixteenth century. The first attempt to reverse the trend was a Danish Royal Decree of 1539 which prohibited removal of vegetation and grazing of animals in coastal areas susceptible to sand drifts (Stadsgaard 2004 after: Wilkie 2002). Denmark was one of the first countries to stabilize sand dunes by planting trees. By the mid-1800s a number of forest plantations were being established. in the Netherlands the State Forestry Department was established

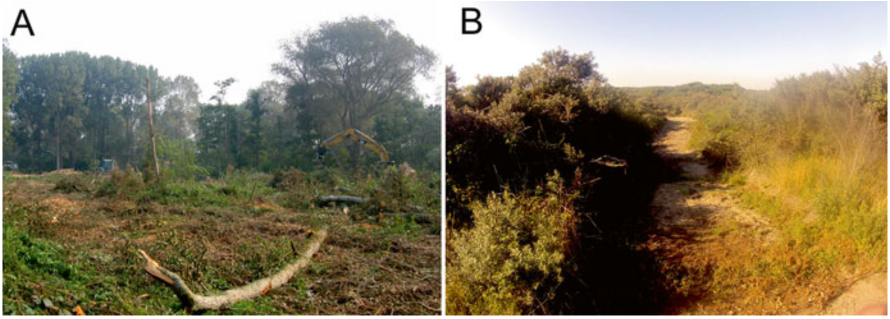


Fig. 14.24 Techniques for plants removal. (**a**) Forest cutting on the dunes, Belgium. (**b**) Trenching Sea buckthorn thicket on the dunes, Calais, Belgium

in 1899 to deal with the afforestation of shifting dunes (Riksen et al. 2006). In the eighteenth century afforestation was started in the southern part of Europe (Danielsen 2008), while in the nineteenth century coastal dunes along the Baltic coast were afforested (Piotrowska and Gos 1995; Koster 2009). For the purpose of afforestation various species of pines were used, often being alien species to a particular country. In the Mediterranean region exotic *Acacia* and *Eucalyptus* species were used too, and now they pose threat to native species (Kutiel et al. 2004; Marchante et al. 2010; Gutierres et al. 2011; Del Vecchio et al. 2013). This problem is also present on the Bulgarian coastal dunes, where they jeopardize valuable plant populations (Petrova and Apostolova 1995). The afforestation is killing heat land plants, even being under protection as *Eryngium maritimum* in Poland (Łabuz 2007), (Fig. 14.22d). Forest plantations are often considered to be limited in biological diversity – yet these dune plantations demonstrate that, with time and careful management, planted forests can become species-rich ecosystems (Stadsgaard 2004).

14.6.4.2 Deforestation

It is the removal of trees or large shrubs from dune sands (Fig. 14.24). After several centuries of dune afforestation in Europe there is scarce of bare and grassland habitats. This method is used to restore thenatural environment and to remobilize the dunes. Some of the coastal monocultural forests have been converted to a mixture of different trees or habitats. In 1992–99,850 ha of dune landscape was restored after pine plantations removal (Ovesen 2001). This method is hardly acceptable to local communities and therefore successful persuasion is vital before any deforestation decisions are made.

Nowadays in Denmark some of the *Pinus mugo* plantations on the fringes of the dune habitats will be converted back to their original heathland condition (Stadsgaard 2004; Jørgensen 2010). Defforestation in the Mediterranean region mainly involves the removal of non native tree species (Kutiel et al. 2004; Marchante et al. 2010; Del Vecchio et al. 2013). The voices against that particular deforestation make a case for

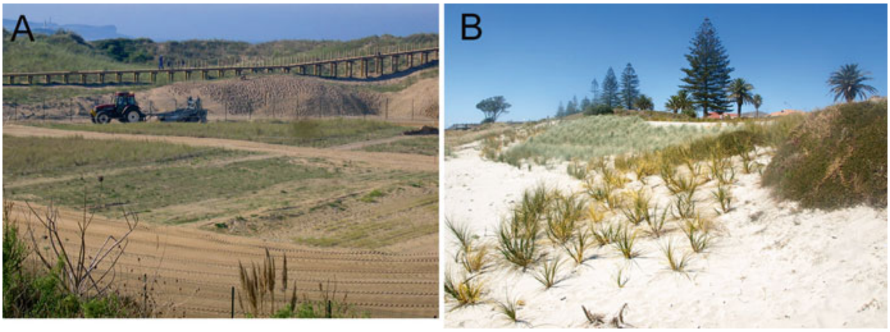


Fig. 14.25 Maintenance of natural vegetation. (**a**) Centre for introduction of natural dune plants, Cantabria, Spain. (**b**) Restored dunes by rebuild, and planting native plants: the first is Pingao sp. and second Spinifex sp., Tauranaga, New Zealand

a large number of species of plants and animals that have adapted to life in these plantations. The variety of planted tree species and habitat niches has encouraged the in-migration of numerous animals, there are also few rare plants, also with the protection status (Stadsgaard 2004 after Wilkie 2002).

14.6.4.3 Planting Natural Vegetation

Native vegetation refers to any naturally occurring local plant species which are indigenous.

The native dune plants are important because they provide essential habitat for native animal species and help to protect land against wind driven erosion. The loss of dune vegetation is a major trigger for deflation and unexpected sand and dune movement. The most important for land stabilization is a foredune ridge and its vegetation. That is why the majority efforts are undertaken to protect or to create a typical foredune with its native sand binding plants. Planting of nature vegetation may be also a goal for species enrichment in other degraded dune ecosystem as dune slacks or heathland. In conclusion, these methods depend on planting natural:

– sand binding grasses for dunes creation and landscape conservation,
– vegetation for landscape conservation and biodiversity increase.

Native plants, mainly sand binging species, are planted in deflation areas or on the sea slopes of foredunes. Some of these plants may be grown in special nurseries as in Cantabria, Spain (Fig. 14.25a). These plants are also used to stabilize reconstructed dunes. Most popular of them are: *Ammophila arenaria* (Europe), *Ammophila littoralis* and *Eltrygia junicea* (Southern Europe), *Ammophila brevigulata* (North America), *Panicum amarum* (southern latitudes of America), *Pingao* (New Zealand), *Spinifex* sp. (New Zealand, Australia), *Tamarix* sp. (Middle East). In several, mainly developing, countries non-native *Ammophila arenaria* is still used.

This method is also commonly used to restore native habitats, mainly in Australia and New Zealand (Bergin 1999; Bergin and Kimberley 1999; Hesp and Hilton 2013). People learns hard how important natural components of the environment are. Over the years, New Zealand's native *Pingao* or *Spinifex* dune grasses, so important for indigenous peoples (Herbert and Oliphant 1991; Bergin 1999) were destroyed as a result of replacing them with faster growing Marram Grass planted in order to protect the shore and stabilize sand expansion (Fig. 14.25b). On the other hand, some species are desirable for proper dune management in specific regions. Currently for several years underway the remedial programs, replacing Marram grass by native *Pingao* or *Spinifex*, both in New Zealand and Australia (Bergin 1999; Bergin and Kimberley 1999; Gadgil 2002; Hilton et al. 2006; Hesp and Hilton 2013).

14.6.4.4 Native Plant Removal

The main goal for native plant removal is to re-juvenile a dune ecosystem which quickly becomes mature. Due to fertilizer load, such as phosphorus or nitrogen, sand dunes become more fertile (Kooijman 2004; Remke et al. 2009), which leads to rapid development of vegetation. This affects the stabilization of conventional movement of dunes and results in their stabilization by overgrowing (Arens et al. 2005; Arens and Geelen 2006). Even sand binding grasses encroachment is well known, that is related to sand dune fertilization (Kooijman and van der Meulen 1996; Arens et al. 2013). These stabilizing sand plants modify the nature of the vegetation and fauna, decrease landscape heterogeneity and biodiversity, change the geomorphologic characteristics of sand dunes, and even massively reduce the area of diversified dune landscapes (Nordstrom 2000; Kutiel et al. 2000b; Tsoar et al. 2005; Cheplick 2005; Howe et al. 2012). In areas that are sensitive to changes in the dune environment it is very easy to destroy the ecological balance. In the Netherlands, the rabbit decline caused grass encroachment and the considerable decrease of the small-scale dynamics in grey dunes (van Til and Kooijman 2007).

In some circumstances eradication of even native dune plants is fully justified. The lack of natural dynamic processes may be compensated for by controlled mosaic burning, grazing or mechanical plant removal (Hewett 1985; Hester et al. 1994; Kooijman and van der Meulen 1996; Stadsgaard 2004; Tsoar et al. 2005; Plassmann et al. 2010; Millett 2013).

14.6.4.5 Invasive Plant Removal

There are numerous non-native plants that grow on dunes around the world. However, most of these invasive species were introduced after a new habitat had become harmful. Nowadays they have become alien and fast growing invaders that are changing all types of ecosystems and landscapes, not only the dunes. From grasses through shrub to the above mentioned trees, all these invaders 'feel the pinch' of dune management services in developed countries, where people have

Fig. 14.26 Invasive plants carpet on dunes. (**a**) Pig plant (*Carpobrotus* sp.) on Tunisian dunes. (**b**) Japanese Rose (*Rosa rugosa*) on Polish dunes

realized what harm those plants do to the ecosystem (Campos et al. 2004; Acosta et al. 2006b; Carranza et al. 2010; Hill et al. 2010).

Non native plants have been used for many purposes – mainly for stabilization of shifting dunes but also as the species enriching pasture for animals. Non native species in arid environments, especially on poor soils or infertile lands, are beneficial to human agriculture (e.g. D'Antonio 1993). However, coastal ecosystems are among the most severely threatened by plant invasions (Campos et al. 2004; Acosta et al. 2006a). Knowledge of the extent and persistence of invasive seed banks helps explain invasion processes and enables management planning (Marchante et al. 2010).

There are several methods used for removing invasive plants. You can do it (1) by hand, (2) using machinery, (3) chemicals, (4) grazing, or (5) mowing and sod-cutting.

Traditionally, non-native sand binders, particularly the fast-growing grasses known from Europe were spread around the world. However, stability that helped first colonizers in managing new lands came at the expense of local native species. Nowadays these plants, mainly Marram grass (*Ammophila arenaria*), are being removed from sand dunes in South Africa, Australia, New Zealand and USA (Bergin and Kimberley 1999; Hertling and Lubke 2000; Cheplick 2005; Hilton et al. 2006; Wootton et al. 2005; Van der Putten et al. 2005; Petersen et al. 2011; Hesp and Hilton 2013). In other parts of world as well as in Europe Marram grass is still used for sand stabilization.

Among first pioneer sand binding plans is the invasive *Carpobrotus* sp. – Pig (or Ice) plant, natively widespread on the African dunes. *Carpobrotus* sp. and other similar genera are trailing, mat-forming succulents that have now invaded diverse coastal areas in South Europe, North Africa and both Americas (Global Invasive Species Database 2009). This South African succulent *Carpobrotus edulis* has been invading native and non-native plant assemblages in California since its introduction in the early 1900s (D'Antonio 1993). Ice plant has been widely observed on dunes in Spain, Tunisia, Egypt and Morocco (Suehs et al. 2009; Carranza et al. 2010). At the beginning it was well-known for its ornamental purposes. Nowadays this plant is quickly colonizing dunes and displaces native dune species (Fig. 14.26a).

In the European, American and Canadian dune heathland the most difficult to get rid of are the shrubs of *Rosa rugosa* and *Hippophaë rhamnoides* (Weidema 2006; Isermann 2008a, b; Jørgensen and Kollmann 2009; Kollmann et al. 2009; Hill et al. 2010; Richards and Burningham 2011). The Japanese Rose (*Rosa rugosa*), an invasive neophyte in Europe, builds up large dominant shrublands especially in coastal areas (Isermann 2008a), (Fig. 14.26b). Dunes with *Hippophaë rhamnoides* shrubs have a large nutrient load and influence local biodiversity (Isermann et al. 2007). They can be found on most European dune coasts (Provoost et al. 2004). where they are mainly removed by bulldozers. There are several studies focused on their impact on the dune environment, some even point to their positive aspects for animal biodiversity. However, the invasion of *Rosa rugosa* on sand dunes was facilitated in the European coastal areas by planting aimed at sand stabilization and coastal protection (Isermann 2008a; Hill et al. 2010). It was also used as an ornamental plant, which may be the cause of its introduction in the proximity of health resorts.

In the Mediterranean region a plant posing an ecological problem is *Acacia* sp., formerly used for sand stabilization. Various species of Acacia are aggressively invading dunes in Portugal, Spain, Italy, Turkey, Cyprus and South Africa (Holmes and Cowling 1997; Holmes 2002; Hadjikyriakou and Hadjisterkotis 2002; Kutiel et al. 2004; Richardson and Kluge 2008; Marchante et al. 2010; Gutierres et al. 2011; Del Vecchio et al. 2013). Furthermore, there are often conflicts of interest over the desire to control acacias whilst continuing to commercially exploit them (Impson et al. 2009). *Acacia* and *Eucalyptus* species were planted in Morocco between 1926 and 1950 to provide wood for timber production (Negre 1952). It is the common reason for our today's problems with eradicating these rapidly expanding species. Why are they problematic? Eucalyptus trees, for instance, exhaust soil nutrients and water. Eucalyptus grows more quickly than native species, that is why it was introduced around the world from South America, North America, Africa to South Europe.

As aforementioned, the eradication of invasive plants includes the use of pesticides that are lethal to plants lethal spray for plants (Wootton et al. 2005). Is it safe for other, non-target species? Today the previously introduced sand binding grasses are removed manually, sometimes they are removed to restore dunes by native species. How to thread shrubs and trees that are fast growing in this gentle environment?

14.6.4.6 Grazing

Coastal dunes were grazed for agricultural purposes until the twentieth century (WallisDeVries et al. 1998). In poor, developing countries this method is still in use. Shepherds with their sheep can be found on dunes in Palestine, Egypt and Tunisia. However, even in such well developed countries as New Zealand sand dunes are used for grazing, or even intensive livestock farming. In the last 30 years this method is mainly designated for dunes management (Hewett 1985; Hester et al. 1994; Kooijman

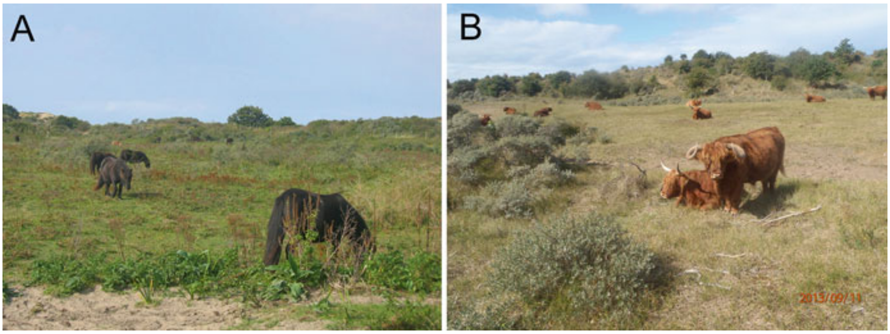


Fig. 14.27 Herbivores grazing on dunes. (**a**) Ponies in De Westhoek Nature Reserve, Belgium. (**b**) Cattle in Zuid-Kennemerland National Park, Netherlands

and van der Meulen 1996; Seliskar 2003; Provoost et al. 2004; Tsoar et al. 2005; Plassmann et al. 2010; Millett 2013). Good examples of such dune management come from the Netherlands, the United Kingdom or the United States (Hester et al. 1994; Kooijman and van der Meulen 1996; Provoost et al. 2004; Millett 2013). Grazing domestic livestock, such as horses, sheep or cattle as well as native herbivores, such as rabbits, have had the major impact on dune vegetation throughout Europe and has been instrumental in forming the present dune landscape (Hewett 1985; Kooijman and van der Meulen 1996; WallisDeVries et al. 1998; Provoost et al. 2004; Damgaard et al. 2013; Doody 2013; Millett 2013). Grazing of the introduced herbivores may be a solution in places where wild mammals are in decline, as it happened in the Netherlands or Belgium (Fig. 14.27) In the 1950s and 1960s, rabbit disease (myxomatosis) caused their population decline (van Til and Kooijman 2007).

Unfortunately, the herbivores may significantly alter dune plant community structure by differentially consuming plant species. Those animals have their favorite species of plants that they eat, causing damaging effects on local biodiversity or even ground stabilization of dunes. Excessive grazing causes the decrease in the number of both useful and unique plants and results in the formation of degraded surface which is vulnerable to erosion. Also, large numbers of birds' nests and eggs are destroyed. On the other hand in low, grazed grasses birds can build nests, which enriches the habitat. The absence of grazers is seen as a threat to the environment because it encourages tall grass encroachment (Hewett 1985; Kooijman and van der Meulen 1996). How to find the ecological balance using this management method?

In a variety of dune habitat vegetation, dune slacks, wetlands or dry dune ridges, covered by varied habitats are introduced various animals. According to this method the dune area is divided into smaller parts, where recessed animals graze for a specified period of time. The main objective of this management practice is to maximize the benefits from time control of each growing species. The grazing time depends on the goals and status of "degradation" the rate of degradation, of on the biodiversity or intensity of development of the invasive species. Long-term grazing management can play an important role for the conservation of dune communities and associated species (Damgaard et al. 2013; Millett 2013).

14.6.4.7 Mowing and Sod-Cutting

This methods facilitates the natural colonization of sand-dwelling species with a subsequent increase in biodiversity (Grootjans et al. 2001; Kooijman 2004; van Til and Kooijman 2007).

Mowing is a technique which limits grass encroachment, mainly on stabilized dune heathland. The traditional mowing meadows are used for dune management purposes. This measure promotes the rate of growth and the diversity of species as well as the composition of vegetation (Kooijman 2004). This method is an alternative to grazing but reduces the whole specimen from the threaded area. Of course the problem with this method is the dune relief shape, since mechanized moving is possible on flat areas and dune slopes must be mowed by hand.

Another method of man-made plants eradication is the reduction of vegetation by sod-cutting. This technique is mainly dedicated to dune slacks or heathland management. This measure is something in between mowing and top soil removal: it might be more sustainable than mowing and is less expensive than top soil removal (van Til and Kooijman 2007). Sod-cutting was first attempted in 1994 and 2002 on Wadden Sea Islands in the Netherlands. It brought positive results as some desired plant species and species richness reappeared (Grootjans et al. 2001; van Til and Kooijman 2007). The target species were mainly the fast growing *Calamagrostis epigejos* and Juncacea sp. of dune slacks. After sod-cutting experiments, conducted on Texel Island on the Wadden Sea the richness of species in the monitored plots exceeded that of the uncut reference plots (Grootjans et al. 2001). Sod-cutting naturally leads to decrease in soil organic matter and nutrient stocks as well as to the increased soil pH (Kooijman 2004).

14.6.4.8 Fences for Sand Restrain

Sand fences are a very old method used to stop wandering sands. Primitive tribes living in sand deserts use palm leaves to build fences lining the villages against sand.

There are different materials that can be used to construct sand fences (Fig. 14.28). From brush wood fences or surfaces covered by brush wood to wooden piles or plastic nets, that are placed in different configurations on upper beaches, dune seaslopes or in deflation gates. The amount of sand trapped in this way depends on the fence height, its porosity, shape and arrangement (Nordstrom 2008; Pye and Tsoar 2009). Of course the bulit-up sand gradually buries a fence and it needs to be built again. Sand-trapping fences can speed the process of foredune development. The fences are used against blowouts creation, to stop shifting sand or to separate valuable habitat from human impact, trampling or exploration (Nordstrom 2000; Peach 2006).

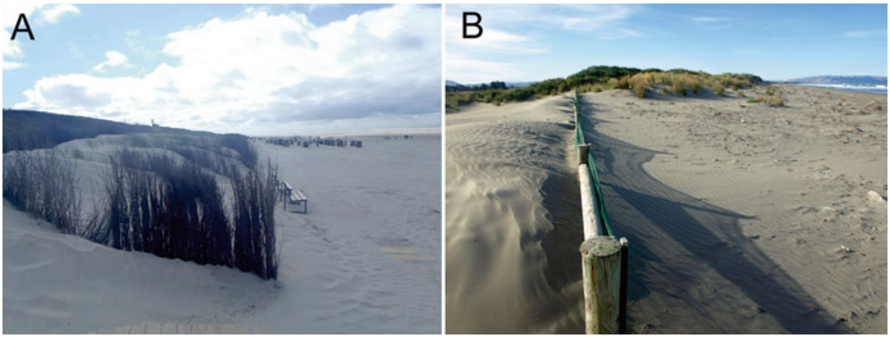


Fig. 14.28 The types of sand fences and their usefulness. (**a**) Natural fashine for foredune creation Spiekeroog Island, Germany. (**b**) Plastic net stopping shifting sand, Picton, New Zealand

14.6.4.9 Limitation or No Access for Human

Human trampling on dunes is the consequence of coastal recreation activities. It endangers the vegetation cover, soil structure and fragile fauna. Trampling always affects the richness and diversity of local species. Trodden paths of development are areas of deflation, which especially applies to the ridges and slopes of the dunes. Angling the pathways to be perpendicular to prevailing winds would help to ensure that the public pathways did not become erosion channels (Nordstrom et al. 2007a). We cannot prohibit people going to the beach through the dunes, but the tourists' limited access would be welcome. However, it would be highly beneficial to the dune habitat protection, if there was no human traffic at all. no access for human to the dune environment is a good help for habitat protection.

Dune trampling is prevented by fencing and establishing of foot bridges above ground (Fig. 14.29a, b). Dune trespassing is often limited or prohibited in view of dune conservation (Santoro et al. 2012). For example, in Denmark in summer local land management authorities outline additional paths by means of wooden chips to prevent overcrowding on the regular ones (Ovesen 2001). In Italy trampling limitation with dunes restoration was undertaken. The results of the project show that trampling limitation plays an important role in reshaping the plant diversity and may be a vital factor for bird nesting protection (Santoro et al. 2012). Very interesting approach to manage the dunes has been introduced in the Netherlands, where dunes are one of the most important environments. Dune areas between Dutch coastal towns are divided with fences in order to limit unwanted exploration and permit proper protection of habitats. Some areas are completely inaccessible to people, in others animals are grazing. The remaining parts are accessible but there are tracks designed for different purposes such as city bike, or MTB bike riding, horse riding or just for walking and hiking. Each zone is fenced off and dedicated to other forms of use. It is well channeled tourism and a deeply thought-out management plan.

Fig. 14.29 The examples of access restrictions to the dunes and educational information about environment. (**a**) Wooden boardwalk through dunes near Somo, Cantabria Spain. (**b**) Wire fence separating dune from beach, wooden boardwalk and environment information, Frankston, Melbourne Bay, Australia. (**c**) Information about the reasons for dunes fencing and restoration aims, Nelson, New Zealand. (**d**) Simple, natural fence between dune and beach with information on Natura 2000 habitats protection, Punta Sabbioni, Venice Spit, Italy

In recent years limitation has been put on car and other vehicle usage. Even in the countries where people traditionally can drive cars on the beach, i.e. in Australia, New Zealand, the USA, and the United Kingdom, some restrictions have been introduced recently. Mechanical destruction of dunes by sports cars, motorcycles and ATVs has a disastrous effect on the coastal fauna, flora and the dune relief, as, for example, the Israeli studies have shown (Kutiel et al. 2000a).

The American policy of beach fencing, board pathways and public education is a good example of how devastated dunes should be managed (Peach 2006). Such solutions are more widely used in developed areas (Nordstrom et al. 2007a).

14.6.5 Legal Aspects and Users' Inclusion

The value of coastal dunes and the importance of their mobility are being increasingly better recognized worldwide. Consequently, many efforts have been undertaken to initiate programs of 'dynamic conservation and restoration management'

(Nordstrom and Lotstein 1989). Due to widespread expansion of human population, dunes face destruction through land development and recreational usages, as well as through measures taken to prevent the encroachment of sand onto inhabited areas. Some countries have developed long-term programs of dune environment protection (Fig. 14.29c).

The restoration capacity of dunes appears to be considerable due to their dynamic nature. Removal of forests and other alien stabilizing vegetation as well as the re-establishment of dynamic, shifting dune formations are amongst the restoration measures being undertaken (Martínez and Psuty 2004; Hilton et al. 2006; Nordstrom 2008; Brooks et al. 2009; Brown et al. 2011; Doody 2013; Martínez et al. 2013). Policy strategies in North America, through Europe to Australia are: holding the line, managed realignment, limited intervention with nourishment or doing nothing (Nordstrom 2000; Lombardo et al. 2002; Robertson et al. 2002; Nordstrom et al. 2007a; Nicholls et al. 2013; Weisner and Schernewski 2013). Proposed adaptation strategies must take into account the sea-level rise (Berry et al. 2013). In Europe coastal dunes are also under protection as Natura 2000 habitats, where separate dune environments are under active conservation (Fig. 14.29d).

Long-term plans have been launched in the United Kingdom to counteract coastal erosion (Brown et al. 2011; Nicholls et al. 2013). Basing on the British experience the new plans adopt several strategies: the advance of the existing defense line, managed realignment or no active intervention. Plans are realized with wide coastal monitoring and stakeholders' participation. The future dunes management must be based on stakeholder engagement (Tompkins et al. 2008). There has been a positive shift in recent years as more and more local citizens appear to be more engaged in coastal conservation and in stewardship activities that will benefit their local community.

The education of local communities is one of the elements of legislation and environmental policies. Long-term management plans are established to ensure the success of the work what work. The ongoing actions are a sum of society awareness and needs, legal solutions and the availability of resources, including the financial ones. It should be emphasized that the cost of environmental protection and ecological restoration-values are very high and only a few countries can afford them (Van der Meulen et al. 2004; Tsoar et al. 2005).

14.7 Conclusions

The coastal dune landscape, being a part of coastal system, was used for centuries for various purposes. Over the last century there have been many changes in land use, which has greatly affected the dune systems.

This diverse and highly dynamic environment attracts tourists. It is rich in plant and animal species that are under protection. In countries with the high development and greatly transformed landscape, dune management methods are complex. Management techniques of dune stabilization, re-mobilization or simply restoration are still based to a high degree on interference. In the developing countries there are still no programs to protect coastal dunes. In areas untouched by human hands, you can still enjoy the natural, rich and enchanting environment of coastal dunes. It is important to note that regardless of latitude, the nature of dunes is similar, adapted to the adverse conditions of moving sand.

New methods and expanding knowledge helps to preserve the dune habitats for future generations. Hopefully, equipped with this knowledge we are making the right choices in order to protect this precious environment.

Acknowledgements This chapter is a based on 15 years of experience in my research into coastal dunes as well as on my observations undertaken during my scientific expeditions and private travels to coastal dunes in whole Europe, Australia, New Zealand, north and south Africa, Arabian and Israeli Middle East and India. I have walked along and examined more than 2,000 km of the world coastal dunes. The facts described in chapter are based on rich scientific literature that looks at many problems of coastal dunes development and management. I could write this chapter thanks to all the colleagues with whom I could discuss the issues relating to the dunes over the years at numerous conferences, and thanks to other scientists involved in dune environment surveys, those who wrote the numerous articles I am quoting above. The data used in this chapter are also collected under the project titled Anthropogenic-Natural Dunes Dynamics (ANDDY). The financial support for preparing this chapter came from the funds provided by the LIDER program by the Polish National Centre for Research and Development (NCBiR), which gave me support to realize the project titled Foredune Morphodynamics and Biodiversity (FoMoBi).

References

Aagaard T, Orford J, Murray AS (2007) Environmental controls on the coastal dune formation; Skallingen Spit, Denmark. Geomorphology 83:29–47
Acosta A, Carranza ML, Izzi CF (2005) Combining land cover mapping of coastal dunes with vegetation analysis. Appl Veg Sci 8:133–138
Acosta A, Izzi CF, Stanisci A (2006a) Comparison of native and alien plant traits in Mediterranean coastal dunes. Commun Ecol 7(1):35–41
Acosta A, Ercole S, Stanisci A, Blasi C (2006b) Sandy coastal ecosystems and effects of disturbance in Central Italy. J Coast Res 39:985–989
Acosta A, Ercole S, Stanisci A, De Patta Pillar V, Blasi C (2007) Coastal vegetation zonation and dune morphology in some Mediterranean ecosystems. J Coast Res 23(6):1518–1524
Acosta A, Carranza ML, Izzi CF (2009) Are there habitats that contribute best to plant species diversity in coastal dunes? Biodivers Conserv 18:1087–1098
Álvarez-Rogel J, Carrasco L, Marín CM, Martínez-Sánchez JJ (2007) Soils of a dune coastal salt marsh system in relation to groundwater level, micro-topography and vegetation under a semi-arid Mediterranean climate in SE Spain. Catena 69:111–121
Aminti P, Cipriani LE, Iannotta P, Pranzini E (2002) Beach erosion control along the Golfo di Follonica (Southern Tuscany). Actual hard protection vs. potential soft solutions. In: Conference proceedings littoral 2002, the changing coast. EUROCOAST/EUCC, Porto, p 9
Andersen UR (2002) Restoration of coastal habitats. To the benefit of birds, plants or toads? In: Conference proceedings littoral 2002, the changing coast. EUROCOAST/EUCC, Porto, p 9
Andrews B, Gares PA, Colby JD (2002) Techniques for GIS modeling of coastal dunes. Geomorphology 48(1–3):289–308
Anthonsen KL, Clemmensen LB, Jensen JH (1996) Evolution of a dune from crescentic to parabolic form in response to short-term climatic changes: Råbjerg Mile, Skagen Odde, Denmark. Geomorphology 17(1–3):63–77

Anthony EJ (2013) Storms, shoreface morphodynamics, sand supply, and the accretion and erosion of coastal dune barriers in the southern North Sea. Geomorphology 199:8–21

Antunes do Carmo J, Schreck Reis C, Freitas H (2010) Working with nature by protecting sand dunes: lessons learned. J Coast Res 26(6):1068–1078

Arens SM (1996) Patterns of sand transport on vegetated foredunes. Geomorphology 17:339–350

Arens SM, Geelen L (2006) Dune landscape rejuvenation by intended destabilization in the Amsterdam water supply dunes. J Coast Res 22:1094–1107

Arens SM, de Slings Q, Vries CN (2004) Mobility of a remobilised parabolic dune in Kennemerland the Netherlands. Geomorphology 59:175–188

Arens B, Geelen L, Slings R, Wondergem H (2005) Restoration of dune mobility in the Netherlands. In: Herrier J-L, Mees J, Salman A, Seys J, Van Nieuwenhuyse H, Dobbelaere I (eds) Proceedings 'Dunes and Estuaries 2005' – international conference on nature restoration practices in European coastal habitats. Koksijde, 19–23 Sept 2005. VLIZ Special Publication, vol 19, pp 129–138

Arens SM, Slings QL, Geelen LHWT (2013) Restoration of dune mobility in The Netherlands. In: Martínez ML, Gallego-Fernandez JB, Hesp PA (eds) Restoration of coastal dunes. Springer series on environmental management. Springer, Heidelberg/New York/Dordrecht/London, pp 107–124

Arun AB, Beena KR, Raviraja NS, Sridhar KR (1999) Coastal sand dunes – a neglected ecosystem. Curr Sci 77(1):19–21

Baeyens G, Martínez ML (2004) Animal life on coastal dunes: from exploitation and prosecution to protection and monitoring. In: Martínez ML, Psuty NP (eds) Coastal dunes, ecology and conservation, vol 171, Ecological studies. Springer, Berlin/Heidelberg, pp 279–296

Baltsavias E (1999) A comparison between photogrammetry and laser scanning. ISPRS J Photogramm Remote Sens 54:83–94

Baptista P, Bastos L, Bernardes C, Cunha T, Dias J (2008) Monitoring sandy shores morphologies by DGPS – a practical tool to generate digital elevation models. J Coast Res 24:1516–1528

Baptista P, Cunha TR, Matias A, Gama C, Bernardes C, Ferreria O (2011) New land-based method for surveying sandy shores and extracting DEMs: the INSHORE system. Environ Monit Assess 182(1–4):243–257

Bastos L, Bioa A, Pinho JLS, Granja H, da Silva J (2012) Dynamics of the Douro estuary sand spit before and after breakwater construction. Estuar Coast Shelf Sci 109:53–69

Bauer BO, Davidson-Arnott RGD (2003) A general framework for modeling sediment supply to coastal dunes including wind angle, beach geometry, and fetch effects. Geomorphology 49:89–108

Bauer BO, Sherman DJ, Nordstrom KF, Gares PA (1990) Aeolian transport measurement and prediction across a beach and dune at Castroville, California. In: Nordstrom KF, Psuty NP, Carter RWG (eds) Coastal dunes. Form and process. Wiley, Chichester, pp 39–55

Bauer BO, Davidson-Arnott RGD, Hesp PA, Namikas SL, Ollehead J, Walker IJ (2009) Aeolian sediment transport conditions on a beach: surface moisture, wind fetch and mean transport rates. Geomorphology 105(1–2):106–116

Bennett MR, Cassidy NJ, Pile J (2009) Internal structure of a barrier beach as revealed by ground penetrating radar (GPR): Chesil beach, UK. Geomorphology 104(3–4):218–229

Berendse F, Lammerts EJ, Olff H (1998) Soil organic matter accumulation and its implications for nitrogen mineralization and plant species composition during succession in coastal dune slacks. Plant Ecol 137:71–78

Bergin DO (1999) Spinifex on coastal sand dunes. Guidelines for seed collection, propagation and establishment. Coastal dune vegetation network technical bulletin no. 2. New Zealand Forest Research Institute, p 28

Bergin DO, Kimberley MO (1999) Rehabilitation of coastal foredunes in New Zealand using indigenous sandbinding species. Department of Conservation Science for Conservation series no. 122, p 55

Berry A, Fahey S, Meyers N (2013) Changing of the guard: adaptation options that maintain ecologically resilient sandy beach ecosystems. J Coast Res 29(4):899–908

Bezzi A, Fontolan G, Nordstrom KF, Carrer D, Jackson NL (2009) Beach nourishment and foredune restoration: practices and constraints along the Venetian shoreline, Italy. J Coast Res SI 56:287–291

Bird E (2003) Coastal geomorphology. An introduction. Wiley, Chichester, p 322

Bishop MP (2013) Remote sensing and GIS science in geomorphology: introduction and overview. In: Shroder J, Bishop MP (eds) Treatise on geomorphology, vol 3. Remote sensing and GIS science in geomorphology. Academic, San Diego, pp 1–24

Bitton MC, Hesp PA (2013) Vegetation dynamics on eroding to accreting beach-foredune systems, Florida panhandle. Earth Surf Process Land 2013:10, wileyonlineliblary.com

Bonanno SE, Leopold DJ, Hilaire LRS (1998) Vegetation of a freshwater dune barrier under high and low recreational use. J Torrey Bot Soc 125(1):40–50

Bonte D, Hoffmann M (2005) Are coastal dune management actions for biodiversity restoration and conservation underpinned by internationally published scientific research? In: Herrier J-L, Mees J, Salman A, Seys J, Van Nieuwenhuyse H, Dobbelaere I (eds) Proceedings 'dunes and estuaries 2005': international conference on nature restoration practices in European coastal habitats, Koksijde, 19–23 Sept 2005. VLIZ Special Publication, vol 19, pp 165–178

Borges P, Andrade C, Freitas MC (2002) Dune, bluff and beach erosion due to exhaustive sand mining – the case of Santa Barbara beach, *São Miguel (Azores, Portugal)*. J Coast Res Spec Issue 36:89–95

Bristow CS, Chroston PN, Bailey SD (2000) The structure and development of foredunes on a locally prograding coast: insights from ground penetrating radar surveys, Norfolk, UK. Sedimentology 47:923–944

Brock JC, Wright W, Jackson JA, Sallanger AH, Krabil W, Swift RN (2002) Basis and methods of NASA Airborne Topographic Mapper Lidar surveys for coastal studies. J Coast Res 18(1):1–13

Brooks N, Grist N, Brown K (2009) Development futures in the context of climate change: challenging the present and learning from the past. Dev Policy Rev 27(6):741–765

Brown DG, Arbogast AF (1999) Digital photogrammetric change analysis as applied to active coastal dunes in Michigan. Photogramm Eng Remote Sens 65:467–474

Brown S, Barton M, Nicholls R (2011) Coastal retreat and/or advance adjacent to defences in England and Wales. J Coast Conserv 15(4):659–670

Buerger R, Hill J, Herstine J, Taggart J (2000) The impact of recreation on barrier islands: a case study on Masonboro Island. Coast Manag 28(3):249–259

Campbell JB (2002) Introduction to remote sensing. The Guilford Press, New York/London

Campos JA, Herrera M, Biurrun I, Loidi J (2004) The role of alien plants in the natural coastal vegetation in central-northern Spain. Biodivers Conserv 13:2275–2293

Carboni M, Carranza ML, Acosta A (2009) Assessing conservation status on coastal dunes: a multiscale approach. Landsc Urban Plan 91:17–25

Carnec C, Massonnet D, King C (1996) Two examples of the use of SAR interferometry on displacement fields of small spatial extent. Geophys Res Lett 23(24):3579–3582

Carranza ML, Carboni M, Feola S, Acosta A (2010) Landscape-scale patterns of alien plant species on coastal dunes: the case of iceplant in central Italy. Appl Veg Sci 13:135–145

Carter RWG, Easwood DA, Bradshaw P (1992) Small scale sediment removal from beaches in Northern Ireland: environment impact, community perception and conservation management. Aqua Conserv Mar Freshw Ecosyst 2(1):95–113

Cheplick GP (2005) Patterns in the distribution of American Beachgrass (*Ammophila breviligulata*) and the Density and reproduction of annual plants on a coastal beach. Plant Ecol 180(1):57–67

Clarke ML, Rendell HM (2009) The impact of North Atlantic storminess on western European coasts: a review. Quat Int 195(1–2):31–41

Clemmensen LB, Nielsen L (2010) Internal architecture of a raised beach ridge system (Anholt, Denmark) resolved by ground-penetrating radar investigations. Sediment Geol 223:281–290

Clemmensen LB, Pye K, Murray A, Heinemeier J (2001) Sedimentology, stratigraphy, and landscape evolution of a Holocene coastal dune system, Lodbjerg, NW Jutland, Denmark. Sedimentology 48:3–27

Costa J, Melo e Souza R (2009) Paisagem costeira e derivações antropogênicas em sistemas dunares. Sci Plena 5(10):18
Dahm J, Jenks G, Bergin D (2005) Community-based dune management for the mitigation of coastal hazards and climate change effects: a guide for local authorities. Climate Change Office, New Zealand, 36
Damgaard C, Paarup Thomsen M, Borchsenius F, Nielsen KE, Strandberg M (2013) The effect of grazing on biodiversity in coastal dune heathlands. J Coast Conserv 17(3):663–670
Danielsen R (2008) Paleoecological development of the Quiaios-Mira dunes, northern-central littoral Portugal. Rev Palaeobot Palynol 152(1–2):74–99
D'Antonio CM (1993) Mechanisms controlling invasion of coastal plant communities by the alien succulent Carpobrotus edulis. Ecology 74:83–95
Davidson-Arnott RGD, Yang Y, Ollerhead J, Hesp PA, Walker IJ (2008) The effects of surface moisture on Aeolian sediment transport threshold and mass flux on b beach. Earth Surf Process Landf 33(2):55–74
Davidson-Arnott RGD, Bauer BO, Walker IJ, Hesp PA, Ollerhead J, Chapman C (2012) High-frequency sediment transport responses on vegetated foredune. Earth Surf Process Landf 37(11):1227–1241
Defeo O, McLachlan A, Schoeman DS, Schlacher TA, Dugan J, Jones A, Lastra M, Scapini F (2009) Threats to sandy beach ecosystems: a review. Estuar Coast Shelf Sci 81:1–12
Delgado I, Lloyd G (2004) A simple low cost method for one person beach profiling. J Coast Res 20(4):1246–1254
Delgado-Fernandez I, Jackson DWT, Cooper JAG, Baas ACW, Lynch K, Beyers JHM (2011) Reattachment zone characterization under offshore winds blowing over complex foredune topography. J Coast Res SI 64(Proceedings of the 11th International Coastal Symposium):273–277
De Lillis M, Costanzo L, Bianco PM, Tinelli A (2004) Sustainability of sand dune restoration along the coast of the Tyrrhenian Sea. J Coast Conserv 10:93–100
Del Vecchio S, Acosta A, Stanisci A (2013) The impact of *Acacia saligna* invasion on Italian coastal dune EC habitats. C R Biol 336(7):364–369
Dillenburg SR, Esteves LS, Tomazelli LJ (2004) A critical evaluation of coastal erosion in Rio Grande do Sul, Southern Brazil. An Acad Bras Cienc 76(3):611–623
Dong Z, Wang H, Liu X, Wang X (2004) The blown sand flux over a sandy surface: a wind tunnel investigation on the fetch effect. Geomorphology 57:117–127
Doody JP (2002) Protecting the coast–myth & magic. In: Conference proceedings littoral 2002, the changing coast. EUROCOAST / EUCC, Porto, p 6
Doody JP (2013) Sand dune conservation management and restoration. Coast Res Libr 4:303, Springer
Elliot WR, Jones DL (1982) Encyclopedia of Australian plants suitable for cultivation, vol 2. Lothian Publishing Company, Melbourne, p 466
Everard M, Jones L, Watts B (2010) Have we neglected the societal importance of sand dunes? An ecosystem services perspective. Aquat Conserv Mar Freshwat Ecosyst 20(4):476–487
Faggi A, Dadon J (2011) Temporal and spatial changes in plant dune diversity in urban resorts. J Coast Conserv 15:585–594
Favennec J (2002) The foredune/backshore system, an indicator of coastal dynamics. The example of the "Aquitaine Coast Observatory". In: Conference proceedings littoral 2002, the changing coast. EUROCOAST/EUCC, Porto, p 10
Feagin RA (2005) Project report: artificial dunes created to protect property on Galveston Island, Texas: the lessons learned. Ecol Restor 23:89–94
Feagin RA, Sherman DJ, Grant WE (2005) Coastal erosion, global sea-level rise, and the loss of sand dune plant habitats. Front Ecol Environ 3(7):359–364
Feagin RA, Smith WK, Psuty NP, Young DR, Martínez ML, Carter GA, Lucas KL, Gibeaut JC, Gemma JN, Koske RE (2010) Barrier islands: coupling anthropogenic stability with ecological sustainability. J Coast Res 26:987–992

Fritz HM, Blount C, Sokoloski R, Singleton J, Fuggle A, McAdoo BG, Moore A, Grass C, Tate B (2007) Hurricane Katrina storm surge distribution and field observations on the Mississippi Barrier Islands. Estuar Coast Shelf Sci 74(1–2):12–20

Froede CR Jr (2010) Constructed sand dunes on the developed barrier-spit portion of Dauphin Island, Alabama (U.S.A). J Coast Res 26(4):699–703

Furmańczyk K (1995) Coast changes of the Hel Spit over the last 40 years. J Coast Res Spec Issue 22:193–196

Gadgil RL (2002) Marram grass (*Ammophila arenaria*) and coastal sand stability in New Zealand. N Z J For Sci 32(2):165–180

Galgaro A, Finzi E, Tosi L (2000) An experiment on a sand-dune environment in Southern Venetian coast based on GPR, VES and documentary evidence. Ann Geofis 43(2):289–295

García-Mora MR, Gallego-Fernández JB, García-Novo F (1999) Plant functional types in coastal foredunes in relation to environmental stress and disturbance. J Veg Sci 10:27–34

García Mora MR, Gallego Fernández JB, García Novo F (2000) Plant diversity as a suitable tool for coastal dune vulnerability assessment. J Coast Res 16:990–995

García Mora MR, Gallego Fernández JB, Williams AT, García Novo F (2001) A coastal dune vulnerability classification. A case of study of the SW Iberian Peninsula. J Coast Res 17:802–811

Gilles JA, Nickling WG, King J (2002) Drag coefficient and plant form response to wind speed in three species: burning bush (Euonymus alatus), Colorado blue spruce (Picea pungens glauca) and fountain grass (Pennisetum setaceum). J Coast Res 107(24):47–60

GISD – Global Invasive Species Database (2009) *Carpobrotus edulis*. http://www.issg.org/database/species/ecology.asp. Accessed 9 Nov 2013

Gómez-Pina G, Muñoz-Pérez JJ, Ramírez JL, Carlos L (2002) Sand dune management problems and techniques, Spain. J Coast Res Spec Issue 36:325–332

Gonçalves JA, Bastos L, Perez B, Magalhães A (2010) Monitoring of beaches and sand dunes using digital aerial photography with direct georeferencing. In: Wagner W, Székely B (eds) ISPRS TC VII symposium–100 years ISPRS, Vienna, 5–7 Jul 2010. IAPRS, vol XXXVIII, Part 7B. pp 228–232

González-Villanueva R, Costas S, Duarte H, Pérez-Arlucea M, Alejo I (2011) Blowout evolution in a coastal dune: using GPR, aerial imagery and core records. J Coast Res SI 64(Proceedings of the 11th International Coastal Symposium):278–282

Gracia Prieto FJ, Sanjaume E, Hernández L, Hernández AI, Flor G, Gómez-Serrano MÁ (2009) Dunas marítimas y continentales. En: VV.AA., Bases ecológicas preliminares para la conservación de los tipos de hábitat de interés comunitario en España. Ministerio de Medio Ambiente, y Medio Rural y Marino, Madrid, p 106

Grootjans AP, Everts H, Bruin K, Fresco L (2001) Restoration of wet dune slacks on the Dutch Wadden Sea Islands: recolonization after large-scale sod cutting. Restor Ecol 9(2): 137–146

Grunewald R (2006) Assement of damages from recreational activities on coastal dunes

Grunewald R, Schubert H (2007) The definition of a new plant diversity index “H′dune” for assessing human damage on coastal dunes–derived from the Shannon index of entropy H′. Ecol Indic 7:1–21

Gutierres F, Gil A, Reis E, Lobo A, Neto C, Calado H, Costa JC (2011) Acacia saligna (Labill.) H.Wendl in the Sesimbra Council: invaded habitats and potential distribution modeling. J Coast Res SI 64(Proceedings of the 11th International Coastal Symposium):403–407

Hadjikyriakou G, Hadjisterkotis E (2002) The adventive plants of Cyprus with new records of invasive species. Z Jagdwiss 48(Supplement):59–71

Hanssen RF (2001) Radar interferometry–data interpretation and error analysis. Kluwer Academic Publishers, The Netherlands, p 328

Hapke CJ, Kratzmann MG, Himmelstoss EA (2013) Geomorphic and human influence on large-scale coastal change. Coastal geomorphology and restoration, 44th Binghamton geomorphology symposium. Geomorphology 199:160–170

Harari Z (1996) Ground-penetrating radar (GPR) for imaging stratigraphic features and groundwater in sand dunes. J Appl Geophys 36:43–52
Harley MD, Turner IL, Short AD, Ranasinghe R (2011) Assessment and integration of conventional, RTK-GPS and image-derived beach survey methods for daily to decadal coastal monitoring. Coast Eng 58(2):194–205
Herbert A, Oliphant J (1991) Pingao: the golden sand sedge. Nga Puna Waihanga, New Zealand Maori Artists and Writers, Rotorua, p 32
Herrier J-L, Killemaes I, Deboeuf Ch (2002) Nature conservation versus industrial development at Zeebrugee Harbour. In: Conference proceedings littoral 2002, the changing coast. EUROCOAST/EUCC, Porto, p 6
Hertling UM, Lubke RA (2000) Assessing the potential for biological invasion – the case of *Ammophila arenaria* in South Africa. S Afr J Sci 96:520–527
Heslefeld P, Jungerius PD, Klijn JA (2004) European coastal dunes: ecological values, threats, opportunities and policy development. In: Martínez ML, Psuty NP (eds) Coastal dunes ecology and conservation. Springer, Berlin/Heideberg, pp 335–351
Hesp PA (2000) Coastal sand dunes: form and function. Coastal dune vegetation network technical bulletin No. 4. Forest Research, Rotorua, p 28
Hesp PA (2002) Foredunes and blowouts: initiation, geomorphology and dynamics. Geomorphology 48:245–268
Hesp PA (2004) Coastal dunes in the tropics and temperate regions: location, formation, morphology and vegetation processes. In: Martínez ML, Psuty NP (eds) Coastal dunes ecology and conservation. Springer, Berlin/Heideberg, pp 29–49
Hesp P (2011) Dune coasts. In: Wolanski E, McLusky DS (eds) Treatise on estuarine and coastal science, vol 3. Academic Press, Waltham, pp 193–221
Hesp PA (2013) A 34 year record of foredune morphodynamics at Dark Point, NSW, Australia. In: Conley DC, Masselink G, Russell PE, O'Hare TJ (eds) Proceedings 12th international coastal symposium (Plymouth, England). J Coast Res Spec Issue 65:1295–1300
Hesp PA, Hilton MJ (2013) Restoration of foredunes and transgressive dunefields: case studies from New Zealand. In: Martínez ML, Gallego-Fernández JB, Hesp PA (eds) Restoration of coastal dunes. Springer series on environmental management. Springer, Berlin/Heidelberg, pp 64–92
Hesp PA, Dillenburg SR, Barboza EG, Tomazelli LJ, Ayup-Zouain RN, Esteves LS, Gruber NLS, Toldo-Jr EE, de Tabajara LLC, Clerot LCP (2005) Beach ridges, foredunes or transgressive dunefield? Definitions and an examination of the Torres to Tramandaí barrier system, Southern Brazil. An Acad Bras Cienc 3:493–508
Hesp PA, Walker IJ, Namikas SL, Davidson-Arnott R, Bauer BO, Ollerhead J (2009) Storm wind flow over a foredune, Prince Edward Island, Canada. J Coast Res SI 56:312–316
Hester MW, Wilsey BJ, Mendelssohn IA (1994) Grazing of *Panicum amarum* in an a Louisiana barrier island dune plant community: management implications for dune restoration projects. Ocean Coast Manag 23(3):213–224
Hewett DG (1985) Grazing and mowing as management tools on dunes. Vegetatio 62(1–3): 441–449
Hill N, Beveridge L, Flynn A, Garbary DJ (2010) *Rosa rugosa* as an invader of coastal sand dunes of Cape Breton Island and mainland of Nova Scotia. Can Field-Nat 124:151–158
Hilton MJ, Hesp PA (1996) Determining the limits of beach-nearshore sand system and the impact of offshore coastal sand mining. J Coast Res 12(2):496–519
Hilton M, Macauley U, Henderson R (2000) Inventory of New Zealand's active dunelands, vol 157, Science for conservation. Department of Conservation, Wellington, p 30
Hilton M, Harvey N, Hart A, James K, Arbuckle C (2006) The impact of exotic dune grass species on foredune development in Australia and New Zealand: a case study of *Ammophila arenaria* and *Thinopyrum junceiforme*. Aust Geogr 37(3):313–334
Hoffmann G, Lampe R, Barnasch J (2005) Postglacial evolution of coastal barriers along the West Pomeranian coast, NE Germany. Quat Int 133–134:47–60

Höfle B, Rutzinger M (2011) Topographic airborne LiDAR in geomorphology: a technological perspective. Z Geomorphol 55(Supplement 2):1–29

Holland KT, Holman RA, Lippmann TC, Stanley J, Plant NG (1997) Practical use of video imagery in nearshore oceanographic field studies. IEEE J Ocean Eng 22(1):81–92

Holmes PM (2002) Depth distribution and composition of seed-banks in alien-invaded and uninvaded fynbos vegetation. Aust Ecol 27(1):110–120

Holmes PM, Cowling RM (1997) The effect of invasion by Acacia saligna on the guild structure and regeneration capabilities of South African fynbus shrublands. J Appl Ecol 34:317–332

Houser C, Sojan M (2011) Alongshore variation in foredune height in response to transport potential and sediment supply: South Padre Island, Texas. Geomorphology 125:62–72

Houser C, Hapke C, Hamilton S (2008) Controls on coastal dune morphology, shoreline erosion and barrier island response to extreme storms. Geomorphology 100(3–4):223–240

Houston J (2005) The conservation of sand dunes in the Atlantic Biogeographical Region: the contribution of the LIFE programme. In: Herrier J-L, Mees J, Salman A, Seys J, Van Nieuwenhuyse H, Dobbelaere I (eds) Proceedings 'dunes and estuaries 2005': international conference on nature restoration practices in European coastal habitats, Koksijde, 19–23 Sept 2005. VLIZ Special Publication, vol 19. pp 29–44

Howe M, Litt E, Pye K (2012) Rejuvenating Welsh dunes. Br Wildl 24:85–94

Impson FAC, Moran VC, Kleinjan C, Hoffmann JH, Moore JA (2009) Multiple-species introductions of biological control agents against weeds: look before you leap. In: Julien MH, Sforza R, Bon C, Evans HC, Hatcher PE, Hinz HL, Rector BG (eds) Proceedings from the XII international symposium on biological control of weeds. pp 26–29

Isermann M (2005) Soil pH and species diversity in coastal dunes. Plant Ecol 178:111–120

Isermann M (2008a) Expansion of *Rosa rugosa* and *Hippophaë rhamnoides* in coastal grey dunes: effects at different spatial scales. Flora 203:273–280

Isermann M (2008b) Effects of Rosa rugosa invasion in different coastal dune vegetation types. In: Brock JH, Brundu G, Child L, Daehler CC, Pysek P (eds) Plant invasions: human perception, ecological impacts and management. Backhuys Publishers, Leiden, pp 289–306

Isermann M, Krisch H (1995) Dunes in contradiction with different interests. An example: the camping-ground Prerow (Darß/ Baltic Sea). In: Salman AHPM, Behrends H, Bonazountas M (eds) Coastal management and habitat conservation. EUCC, Leiden, pp 439–449

Isermann M, Diekmann M, Heemann S (2007) Effects of the expansion by *Hippophaë rhamnoides* on plant species richness in coastal dunes. Appl Veg Sci 10(1):33–42

Jacobs Z (2008) Luminescence chronologies for coastal and marine sediments. Boreas 37:508–535

Jagannadha RK, Subramanyam AV, Abhinav K, Sunil TC, Chaturvedi AK (2008) Discovery of heavy mineral-rich sand dunes along the Orissa-Bengal coast of India using remote sensing techniques. Curr Sci 94(8):983–985

Jørgensen H (2010) NOBANIS – Invasive alien species fact sheet–*Pinus mugo*–from: online database of the European network on invasive alien species – NOBANIS. www.nobanis.org. Accessed 30 Oct 2013

Jørgensen RH, Kollmann J (2009) Invasion of coastal dunes by the alien shrub *Rosa rugosa* is associated with roads, tracks and houses. Flora 204:289–297

Juanaidi AS (2009) Wind-blown sand and topographic changes of the coastal dune at the eroded beach. J Coast Res SI 56:322–326

Khalil R (2013) The accuracy of GIS tools for trans forming assumed total station surveys to Real Word coordinates. J Geogr Inf Syst 5:486–491

Kizil U, Tisor L (2011) Evaluation of RTK-GPS and total station for applications in land surveying. J Earth Syst Sci 120(2):215–221

Klein YL, Osleeb J (2010) Determinants of coastal tourism: a case study of Florida beach counties. J Coast Res 26(6):1149–1156

Kocurek G, Ewing RC (2005) Aeolian dune field self-organization–implications for the formation of simple versus complex dune-field patterns. Geomorphology 72(1–4):94–105

Kollmann J, Jørgensen RH, Roelsgaard J, Skov-Petersen H (2009) Establishment and clonal spread of the alien shrub Rosa rugosa in coastal dunes—a method for reconstructing and predicting invasion patterns. Landsc Urban Plan 93(3–4):194–200

Kooijman AM (2004) Environmental problems and restoration measures in coastal dunes in Netherlands. In: Martínez ML, Psuty NP (eds) Coastal dunes. Ecology and conservation, vol 171, Ecological studies. Springer, Berlin/Heidelberg, pp 243–258

Kooijman AM, van der Meulen F (1996) Grazing as a control against grass encroachment in dry dune grasslands in the Netherlands. Landsc Urban Plan 34:323–333

Koster EA (2009) The "European aeolian sand belt": geoconservation of drift sand landscapes. Geoheritage 1(2–4):93–110

Kunz A, Frechen M, Ramesh R, Urban B (2010) Luminescence dating of late Holocene dunes showing remnants of early settlement in Cuddalore and evidence of monsoon activity in south east India. Quat Int 222:194–208

Kutiel P, Zhevelev H, Harrison R (1999) The effect of recreational impacts on soil and vegetation of stabilized coastal dunes in the Sharon Park, Israel. Ocean Coast Manag 42:1041–1060

Kutiel P, Eden E, Zhevelev Y (2000a) Effect of experimental trampling and off-road motorcycle traffic on soil and vegetation of stabilized coastal dunes, Israel. Environ Conserv 1:14–23

Kutiel P, Peled Y, Geffen E (2000b) The effect of removing shrub cover on annual plants and small mammals in a coastal sand dune ecosystem. Biol Conserv 94:235–242

Kutiel P, Cohen O, Shoshany M (2004) Invasion rate of the alien species Acacia saligna within coastal sand dune habitats in Israel. Isr J Plant Sci 52(2):115–124

Łabuz TA (2002) The beach vegetation cover as a indicator of dynamics on the Pommeranian Bay coast (in Polish). Czas Geogr 73(4):245–255

Łabuz TA (2003) Initial foredune field as a factor of accumulative character of coastal dunes of the Swina Gate Barrier (West Polish coast). Oceanol Hydrobiol Stud 32(1):39–58

Łabuz TA (2004) Coastal dune development under natural and human influence on Swina Gate Barrier (Polish coast of Pomeranian Bay). In: Schernewski G, Löser N (eds) Managing the Baltic Sea. Coastline Reports, vol 2, pp 129–138

Łabuz TA (2007) Evaluation of past and present sea holly (Eryngium maritimum) habitats on Polish coastal dunes. Acta Univ Latv Biol 723:99–114

Łabuz TA (2009) The increase of the coastal dune area of the Swina Sandbar, West Polish coast. Z Dt Ges Geowiss 160(2):123–135

Łabuz TA (2013) Polish coastal dunes–affecting factors and morphology. Landf Anal 22:33–59

Łabuz TA, Grunewald R (2007) Studies on vegetation cover of the youngest dunes of the Świna Gate Barrier (West Polish coast). J Coast Res 23(1):160–172

Łabuz TA, Sławińska J, Osóch P (2012) Issues of coastal dunes mapping in Poland. In: Book of abstracts: IAG/AIG international workshop on objective geomorphological representation models: breaking through a new geomorphological mapping frontier. University of Salerno, Cilento & Vallo di Diano Geopark, 15–19 Oct 2012, pp 85–87

Ladage F, Kunz H (2002) Long-term morphological development of barrier islands: case study "Juist"–German North Sea coast. In: Conference proceedings littoral 2002, the changing coast. EUROCOAST/EUCC, Porto, p 8

Lee J-M, Park J-Y, Choi J-Y (2013) Evaluation of sub-aerial topographic surveying techniques using total station and RTK-GPS for applications in macro-tidal sand beach environment. Proceedings 12th international coastal symposium (Plymouth, England). J Coast Res Spec Issue 65:535–540

Lemoine G, Faucon L (2005) Managing the Flemish dunes: from eco-gardening to mechanical disturbances created by bulldozers. In: Herrier J-L, Mees J, Salman A, Seys J, Van Nieuwenhuyse H, Dobbelaere I (eds) Proceedings 'dunes and estuaries 2005': international conference on nature restoration practices in European coastal habitats, Koksijde, 19–23 Sept 2005. VLIZ Special Publication, 19. pp 239–246

Leten M, Van Nieuwenhuyse H, Herrier J-L (2005) Invasive scrub and trees in the coastal dunes of Flanders (Belgium): an overview of management goals, actions and results. In: Herrier J-L, Mees J, Salman A, Seys J, Van Nieuwenhuyse H, Dobbelaere I (eds) Proceedings 'dunes and estuaries 2005': international conference on nature restoration practices in European coastal habitats, Koksijde, 19–23 Sept 2005. VLIZ Special Publication, vol 19, pp 111–128

Levin N, Ben-Dor E (2004) Monitoring sand dune stabilization along the coastal dunes of Ashdod-Nizanim, Israel, 1945 – 1999. J Arid Environ 58:335–355

Levin N, Ben-Dor E, Karnieli A (2004) Topographic information of sand dunes extracted from shading effects using Landsat images. Remote Sens Environ 90:190–209

Lombardo S, Niesing H, Salman A, Lucius I (2002) Capitalizing knowledge for managing coastal erosion: the EUROSION initiative. In: Conference proceedings littoral 2002, the changing coast. EUROCOAST/EUCC, Porto, p 8

Lubke RA (2013) Restoration of dune ecosystem following mining in Madagascar and Namibia: contrasting restoration approaches adopted in region of high and low human population density. In: Martínez ML, Gallego-Fernández JB, Hesp PA (eds) Restoration of coastal dunes. Springer series on environmental management. Springer, Berlin/Heidelberg, pp 199–215

Lynch K, Jackson DWT, Cooper JAG (2009) Foredune accretion under offshore winds. Geomorphology 105:139–146

Marchante H, Freitas H, Hoffmann JH (2010) Seed ecology of an invasive alien species, *Acacia longifolia* (Fabaceae), in Portuguese dune ecosystems. Am J Bot 97(11):1780–1790

Martínez ML, Psuty NP (eds) (2004) Coastal dunes. Ecology and conservation. Ecological studies, 171. Springer-Verlag, Berlin/Heidelberg, p 386

Martínez ML, Maun MA, Psuty NP (2004) The fragility and conservation of the world's coastal dunes: geomorphological, ecological, and socioeconomic perspectives. In: Martínez ML, Psuty NP (eds) Coastal dunes ecology and conservation. Springer, Berlin/Heidelberg, pp 355–369

Martínez ML, Gallego Fernández JB, García Franco JG, Moctezuma C, Jiménez CD (2006) Coastal dune vulnerability along the Gulf of México. Environ Conserv 33:109–117

Martínez ML, Gallego-Fernandez JB, Hesp PA (2013) Restoration of coastal dunes. Springer series on environmental management. Springer, Heidelberg/New York/Dordrecht/London, p 347

Matias A, Ferreira I, Mendes I, Dias JA, Vila-Concejo A (2005) Artificial construction of dunes in the south of Portugal. J Coast Res 21(3):472–481

Meerkerk AL, Arens SM, van Lammeren RJA, Stuiver HJ (2007) Sand transport after a foredune breach: a case study from Schoorl. The Netherlands. Geomorphology 86:52–60

Merrell WJ, Reynolds LG, Cardenas A, Gunn JR, Hufton AJ (2011) The Ike dike: a coastal barrier protecting the Houston/Galveston Region from hurricane storm surge. In: Badescu V, Cathart RB (eds) Macro-engineering seawater in unique environments. Environmental science and engineering. Springer, Berlin/Heidelbeg, pp 691–716

Michoud C, Carrea D, Augereau E, Cancouët R, Costa S, Davidson R, Delacourt Ch, Derron M-H, Jaboyedoff M, Letortu P, Maquaire O (2013) Mobile laser scanning along Dieppe coastal cliffs: reliability of the acquired point clouds applied to rockfall assessment, EGU general assembly 7–12 Apr 2013, Vienna

Midgley SJ, Turnbull JW (2003) Domestication and use of Australian acacias: case studies of five important species. Aust Syst Bot 16(1):89–102

Millett J (2013) The impact of 36 years of grazing management on vegetation dynamics in dune slacks. J Appl Ecol 50(6):1367–1376

Mitasova H, Drake TG, Bernstein D, Harmon RS (2004) Quantifying rapid changes in coastal topography using modern mapping techniques and Geographic Information System. Environ Eng Geosci 10(1):1–11

Mitasova H, Overton M, Harmon RS (2005) Geospatial analysis of a coastal sand dune field evolution: Jockey's Ridge, North Carolina. Geomorphology 72:204–221

Mitasova H, Overton M, Recalde JJ, Bernstein D, Freeman C (2009) Raster-based analysis of coastal terrain dynamics from multitemporal lidar data. J Coast Res 25(2):507–514

Mitasova H, Hardin E, Kurum MO, Overton MF (2010a) Geospatial analysis of vulnerable beach-foredune systems from decadal time series of lidar data. J Coast Conserv Manag Plan 14(3):161–172

Mitasova H, Hardin E, Overton MF, Kurum MO (2010b) Geospatial analysis of vulnerable beach-foredune systems from decadal time series of lidar data. J Coast Conserv 14(3):161–172
Molodkov A, Bitinas A (2006) Sedimentary record and luminescence chronology of the Lateglacial and Holocene aeolian sediments in Lithuania. Boreas 35:244–254
Montreuil AL, Bullard JA (2012) A 150-year record of coastline dynamics within a sediment cell: eastern England. Geomorphology 179(15):168–185
Montreuil A-L, Bullard JE, Chandler JH (2013a) Detecting seasonal variations in embryo dune morphology using a terrestrial laser scanner. In: Conley DC, Masselink G, Russell PE, O'Hare TJ (eds) Proceedings 12th international coastal symposium (Plymouth, England). J Coast Res, Spec Issue 65:1313–1318
Montreuil A-L, Bullard JE, Chandler JH, Millett J (2013b) Decadal and seasonal development of embryo dunes an accreting macrotidal beach: North Lincolnshire, UK. Earth Surf Process Landf 38(15):1851–1868
Moreno-Casasola P (1988) Patterns of plant species distribution on coastal dunes along the Gulf of Mexico. J Biogeogr 15:787–806
Muńoz Vallés S, Gallego Fernandez JB, Dellafiore CM (2011) Dune vulnerability in relation to tourism pressure in Central Gulf of Cadiz (SW Spain), a case study. J Coast Res 27:243–251
Navarro M, Muñoz-Pérez JJ, Román-Sierra J, Tsoar H, Rodríguez I, Gómez-Pina G (2011) Assessment of highly active dune mobility in the medium, short and very short term. Geomorphology 129:14–28
Negre R (1952) Arid Zone Programme, introduction of exotic plants into Moroccan arid zone. The Moroccan Science Institute, Rabat, UNESCO NS/AZ/104, Paris, p 10
Nicholls RJ, Townend IH, Bradbury AP, Ramsbottom P, Day SA (2013) Planning for long-term coastal change: experiences from England and Wales. Ocean Eng 71(1):3–16
Nordstrom KF (1990) The concept of intrinsic value and depositional coastal landforms. Geogr Rev 80:68–81
Nordstrom KF (2000) Beaches and dunes of developed coasts. Cambridge University Press, Cambridge, p 338
Nordstrom KF (2008) Beach and dune restoration. Cambridge University Press, Cambridge, 201
Nordstrom KF, Jackson NJ (2013) Foredune restoration in urban settings. In: Martínez ML, Gallego-Fernández JB, Hesp PA (eds) Restoration of coastal dunes. Springer series on environmental management. Springer, Berlin/Heidelberg, pp 17–32
Nordstrom KF, Lotstein EL (1989) Perspectives on resource use of dynamic coastal dunes. Geogr Rev 79:1–12
Nordstrom KF, Jackson NL, Bruno MS, de Butts HA (2002) Municipal initiatives for managing dunes in coastal residential areas: a case study of Avalon, New Jersey, USA. Geomorphology 47:137–152
Nordstrom KF, Hartman JM, Freestone AL, Wong M, Jackson NL (2007a) Changes in topography and vegetation near gaps in a protective foredune. Ocean Coast Manag 50:945–959
Nordstrom KF, Lampe R, Jackson NL (2007b) Increasing the dynamism of coastal landforms by modifying shore protection methods: examples from the eastern German Baltic Sea Coast. Environ Conserv 34:205–214
Nordstrom KF, Gamper U, Fontolan G, Bezzi A, Jackson NL (2009) Characteristics of coastal dune topography and vegetation in environments recently modified using beach fill and vegetation plantings, Veneto, Italy. Environ Manag 44(6):1121–1135
Ovesen CH (2001) Management of Danish dunes today: theory and practice. In: Houston JA, Rooney PA, Edmondson SE (eds) Coastal dune management: shared experience of European conservation practice. Liverpool University Press, Liverpool, pp 302–308
Pardo-Pascual JE, Garcia-Asenjo L, Palomar-Vazquez J, Garrigues-Talens P (2005) New methods and tools to analyze the beach-dune system evolution using Real Time Kinematic Global Positioning System and Geographic Information Systems. J Coast Res SI 49:34–39
Parsons DR, Walker IJ, Wiggs GFS (2004) Numerical modelling of flow structures over idealized transverse aeolian dunes of varying geometry. Geomorphology 59(1–4):149–164

Peach G (2006) Management of Lake Huron's beach and dune ecosystems: building up from the grassroots. Great Lakes Geogr 13(1):39–49

Petersen PS, Hilton MJ, Wakes SJ (2011) Evidence of Aeolian sediment transport across *Ammophila arenaria*–dominated foredune, Mason Bay, Steward Island. N Z Geogr 67:174–189

Petrova A, Apostolova I (1995) Rare plant species on coastal sand dunes and sand beaches in Bulgaria–status and conservation problems. In: Salman AHPM, Berends H, Bonazountas M (eds) Coastal management and habitat conservation. EUCC, Leiden, pp 451–457

Peyrat J, Braun M, Dolnik C, Isermann M, Roweck H (2009) Vegetation dynamics on the Łeba Bar/Poland: a comparison of the vegetation in 1932 and 2006 with special regard to endangered habitats. J Coast Conserv 13(4):235–246

Phillips MR, Jones AL (2006) Erosion and tourism infrastructure in the coastal zone: problems, consequences and management. Tour Manag 27:517–524

Piotrowska H, Gos K (1995) Coastal dune vegetation in Poland. Diversity and development. In: van Dijk HWJ (ed) Management and preservation of coastal habitats. Proceedings of multidisciplinary workshop in Jastrzębia Góra. EUCC, Leiden, pp 71–82

Plassmann K, Jones LM, Edwards-Jones G (2010) Effects of long-term grazing management on sand dune vegetation of high conservation interest. Appl Veg Sci 13(1):100–112

Pollard J, Dominguez Rodriguez R (1993) Tourism and Torremolinos: recession or reaction to environment? Tour Manag 14(4):247–250

Pontee NI, Morris K (2011) CoastRanger MS: a tool for improving public engagement. J Coast Res 27(1):18–25

Portz LC, Manzolli Portantiollo R, Gruber Sambaqui NL, Correa Stalliviere IC (2010) Tourism and degradation in the coastline of Rio Grande do Sul: conflicts and management. Desenv Meio Ambien 22:153–166

Provoost S, Ampe C, Bonte D, Cosyns E, Hoffmann M (2002) Ecology, management and monitoring of dune grassland in Flanders, Belgium. In: Conference proceedings littoral 2002, the changing coast. EUROCOAST/EUCC, Porto, p 10

Provoost S, Ampe C, Bonte D, Cosyns E, Hoffmann M (2004) Ecology, management and monitoring of grey dunes in Flanders. J Coast Conserv 10:33–42

Provoost S, Laurence M, Jones M, Edmondson SE (2009) Changes in landscape and vegetation of coastal dunes in northwest Europe: a review. J Coast Conserv 15:207–226

Puthucherril George T (2011) Operationalising integrated coastal zone management and adapting to sea level rise through Coastal Law: where does India stand? Int J Mar Coast Law 26(4):569–612

Pye K, Tsoar H (2009) Aeolian sand and sand dunes. Springer, Berlin/Heidelberg, p 458

Reimann T, Tsukamoto S, Harff J, Osadczuk K, Frechen M (2010a) Reconstruction of Holocene coastal foredune progradation using luminescence dating an example from the Swina barrier (southern Baltic Sea, NW Poland). Geomorphology 132:1–16

Reimann T, Naumann M, Tsukamoto S, Frechen M (2010b) Luminescence dating of coastal sediments from the Baltic Sea coastal barrier-spit Darss-Zingst, NE Germany. Geomorphology 122:264–273

Remke E, Brouwer E, Kooijman A, Blindow I, Roelofs JGM (2009) Low atmospheric nitrogen loads lead to grass encroachment in coastal dunes, but only on acid soils. Ecosystems 12:1173–1188

Richards EG, Burningham H (2011) *Hippophae rhamnoides* on a coastal dune system: a thorny issue? J Coast Conserv 15(1):73–85

Richardson DM, Kluge RL (2008) Seed banks of invasive Australian Acacia species in South Africa: role in invasiveness and options for management. Perspect Plant Ecol Evol Syst 10(3):161–177

Riksen M, Ketner-Oostra R, Van Turnhout C, Nijssen M, Goossens D, Jungerius PD, Spaan W (2006) Will we lose the last active inland drift sands of Western Europe? The origin and development of the inland drift-sand ecotype in The Netherlands. Landsc Ecol 21: 431–447

Robertson A, McGrath J, Tomlinson R (2002) Formulation of master plan for the Palm Beach protection strategies. In: Conference proceedings littoral 2002, the changing coast. EUROCOAST/EUCC, Porto, p 8
Robertson W, Whitman D, Zhang K, Leatherman SP (2004) Mapping shoreline position using airborne laser altimetry. J Coast Res 20(3):884–892
Roig FX, Rodriguez-Perea A, Martin-Prieto JA, Pons GX (2009) Soft management of beach-dune systems as tool for their sustainability. J Coast Res Spec Issue 56:1284–1288
Rosa LS, Cordazzo CV (2007) Perturbações antrópicas na vegetação das dunas da Praia do Cassino (RS). Cadernos Ecol Aquát 2:1–14
Ruiz AM, Caro M, Sousa JJ (2012) Land subsidence monitoring in the Southern Spanish Coast using satellite radar interferometry. Eur Space Agency Publ ESA SP 697:1–54
Sallenger A, Krabill WB, Swift RN, Brock JC, List J, Hansen M, Holman RA, Manizade S, Sontag J, Meredith A, Morgan K, Yunkel JK, Fredrick EB, Stockdon H (2003) Evaluation of airborne topographic LIDAR for quantifying beach changes. J Coast Res 19:125–133
Santalla IR, García MJS, Montes IM, Ortiz DG, Crespo TM, Raventos JS (2009) Internal structure of the aeolian sand dunes of El Fangar spit, Ebro Delta (Tarragona, Spain). Geomorphology 104(3–4):238–252
Santoro R, Jucker T, Prisco I, Carboni M, Battisti C, Acosta ATR (2012) Effects of trampling limitation on coastal dune plant communities. Environ Manag 49:534–542
Saye SE, Van der Wal D, Pye K, Blott SJ (2005) Beach-dune morphological relationships and erosion/accumulation: an investigation at five sites in England and Wales using Lidar data. Geomorphology 72:128–155
Schenk CJ, Gautier DL, Olhoeft GR, Lucius JE (1993) Internal structure of an aeolian dune using ground penetrating radar. Spec Publ Int Assoc Sedimentol 15:61–69
Schwarzer K, Diesing M, Larson M, Niedermeyer RO, Schumacher W, Furmanczyk K (2003) Coastline evolution at different time scales: examples from the Pomeranian Bight, Southern Baltic Sea. Mar Geol 194(1–2):79–101
Seliskar DM (2003) The response of *Ammophila breviligulata* and *Spartina patens* (*Poaceae*) to grazing by feral horses on a dynamic mid-Atlantic barrier island. Am J Bot 90(7):1038–1044
Shanmugam S, Barnsley M (2002) Quantifying landscape-ecological succession in coastal dune systems using sequential aerial photography and GIS. J Coast Conserv 8:61–68
Sherman DJ, Hotta S (1990) Aeolian sediment transport: theory and measurement. In: Nordstrom KF, Psuty NP, Carter RWG (eds) Coastal dunes. Form and process. Wiley, Chichester, pp 17–37
Sherman DJ, Jackson DWT, Namikas SL, Wang J (1998) Wind-blown sand on beaches: an evaluation of models. Geomorphology 22:113–133
Smith GL, Zarillo GA (1990) Calculating long-term shoreline recession rates using aerial photographic and beach profiling techniques. J Coast Res 6:112–120
Sousa PHGO, Siegle E, Tessler MG (2011) Environmental and anthropogenic indicators for coastal risk assessment at Massaguaçú Beach (SP) Brazil, SI 64. In: Proceedings of the 11th international coastal symposium. pp 319–323
Sridhar KR, Bhagya B (2007) Coastal sand dune vegetation: a potential source of food, fodder and pharmaceuticals. Livest Res Rural Dev 84(19), Retrieved October 31, 2013, from http://www.lrrd.org/lrrd19/6/srid19084.htm
Stadsgaard JH (2004) Restoration of dune habitats along the Danish West Coast. International Forest Fire News (IFFN) No. 30(14–15), pp 1–3. http://www.fire.uni-freiburg.de/programmes/natcon/Denmark
Suehs CM, Médail F, Affre L (2009) Ecological and genetic features of the invasion by the alien Carpobrotus plants in Mediterranean island habitats. In: Brundu G, Brock J, Camarda I, Child L, Wade M (eds) Plant invasions: species ecology and ecosystem management. Backhuys Publishers, Leiden, pp 145–157
Takaaki Uda T, Sakai K, San-nami T, Shimizu T (2012) Two distinguished effects of the 2011 Great Tsunami to the coast along Sendai Bay: damage to Arahama and trench formation behind the dike on Yamamoto coast. Trans Jpn Geomorphol Union 33(4):365–384

Teodoro AC, Pais-Barbosa J, Gonçalves H, Veloso-Gomes F, Taveira-Pinto F (2011) Extraction of Cabedelo sand spit area (Douro estuary) from satellite images through image processing techniques. J Coast Res SI 64(Proceedings of the 11th International Coastal Symposium): 1740–1744, Szczecin, Poland

Terlouw L, Slings R (2005) Dynamic dune management in practice – remobilization of coastal dunes in the National Park Zuid-Kennemerland in the Netherlands. In: Herrier J-L, Mees J, Salman A, Seys J, Van Nieuwenhuyse H, Dobbelaere I (eds) Proceedings 'dunes and estuaries 2005': international conference on nature restoration practices in European coastal habitats, Koksijde, 19–23 Sept 2005. VLIZ Special Publication, vol 19, pp 211–217

Tompkins EL, Few R, Brown K (2008) Scenario-based stakeholder engagement: incorporating stakeholders preferences into coastal planning for climate change. J Environ Manag 88(4):1580–1592

Tsoar H (2005) Sand dunes mobility and stability in relation to climate. Physica A Stat Mech Appl 357(1):50–56

Tsoar H, Shachak M, Blumberg DG (2005) Ecological aspects of vegetation removal from the coastal sand dunes of Israel. In: Herrier J-L, Mees J, Salman A, Seys J, Van Nieuwenhuyse H, Dobbelaere I (eds) Proceedings 'dunes and estuaries 2005': international conference on nature restoration practices in European coastal habitats, Koksijde, 19–23 Sept 2005. VLIZ Special Publication, vol 19. pp 487–493

Tudor DT, Williams AT, Phillips MR, Thomas MC (2002) Qualitative and quantitative comparisons of some indices suitable for litter analysis. In: Conference proceedings littoral 2002, the changing coast. EUROCOAST/EUCC, Porto, p 7

Turner IL, Aarninkhof SGJ, Holman RA (2006) Coastal imaging applications and research in Australia. J Coast Res 22(1):37

Udo K, Takewaka S (2007) Experimental study on blown sand in vegetated area. J Coast Res 23(5):1175–1182

Uunk L, Wijnber KM, Morelissen RM (2010) Automated mapping of the intertidal beach bathymetry from video images. Coast Eng 57:461–469

Van Boxel JH, Jungerius PD, Kieffer N, Hampele N (1997) Ecological effects of reactivation of artificially stabilized blowouts in coastal dunes. J Coast Conserv 3(1):57–62

Van Boxel JH, Sterk G, Arens SM (2004) Sonic anemometers in aeolian sediment transport research. Geomorphology 59:131–147

Van der Meulen F, Salman AHPM (1995) Management of Mediterranean coastal dunes. In: Salman AHPM, Berends H, Bonazountas M (eds) Coastal management and habitat conservation. EUCC, Leiden, pp 261–278

Van der Meulen F, Bakker TWM, Houston JA (2004) The costs of our coast: examples of dynamics dune management from Western Europe. In: Martínez ML, Psuty NP (eds) Coastal dunes ecology and conservation. Springer, Berlin/Heidelberg, pp 259–277

Van der Putten WH, Yeates GW, Duyts H, Schreck Reis C, Karssen G (2005) Invasive plants and their escape from root herbivory: a worldwide comparison of the root-feeding nematode communities of the dune grass *Ammophila arenaria* in natural and introduced ranges. Biol Invasions 7:733–746

Van der Wal D (2004) Beach-dune interaction in nourishment areas along the Dutch coast. J Coast Res 20:317–325

van Til M, Kooijman A (2007) Rapid improvement of grey dunes after shallow sod cutting. In: Isermann M, Kiehl K (eds) Restoration of coastal ecosystems, vol 7, Coastline reports. EUCC, Leiden, pp 53–60

Vestergaard P, Hansen K (1992) Changes in morphology and vegetation of a man-made beach-dune system by natural processes. In: Carter RWG, Curtis TGF, Sheehy-Skeffington MJ (eds) Coastal dunes: geomorphology, ecology and management for conservation. A.A. Balkema, Rotterdam, pp 165–176

Vosselman G, Maas H-G (eds) (2010) Airborne and terrestrial laser scanning. CRC Press, p 320

Vykhovanets GV (1999) Aeolian processes and forms development on coast of Ukraine. Geogr Fis Dinam Quat 22:99–104

Walker IJ (2005) Physical and logistical considerations of using ultrasonic anemometers in Aeolian sediment transport research. Geomorphology 68:57–76

Walker IJ, Nickling WG (2003) Simulation and measurement of surface shear stress over isolated and closely spaced transverse dunes in a wind tunnel. Earth Surf Process Landf 28(10):1111–1124

Walker SJ, Schlacher TA (2011) Impact of a pulse human disturbance experiment on macrofaunal assemblages on an Australian Sandy Beach. J Coast Res 27(6A):184–192

WallisDeVries MF, Bakker JP, Van Wieren SE (eds) (1998) Grazing and conservation management, vol 11, Conservation biology series. Kluwer Academic Publishers, Dordrecht/Boston, p 374

Weidema I (2006) NOBANIS–invasive alien species fact sheet – Rosa rugosa – From: Online database of the European network on invasive alien species – NOBANIS. www.nobanis.org. Accessed 30 Oct 2013

Weisner E, Schernewski G (2013) Adaptation to climate change: a combined coastal protection and re-alignment measure in a southern Baltic tourism region. In: Conley DC, Masselink G, Russell PE, O'Hare TJ (eds) Proceedings 12th international coastal symposium (Plymouth, England). J Coast Res Spec Issue 65:1963–1968

Wilhjelm Committee (2001) Danish nature–status, trends and recommendations for future biodiversity policies. Copenhagen, Denmark, Danish Forest and Nature Agency. Available on the Internet: www.sns.dk/wilhjelm/endelig

Wilkie ML (2002) From dune to forest: biological diversity in plantations established to control drifting sand. Unasylva 209(53):64–69

Woodroffe CD (2002) Coasts, form process and evolution. Cambridge University Press, Cambridge, p 638

Woolard JW, Colby JD (2002) Spatial characterization, resolution, and volumetric change of coastal dunes using airborne LIDAR: Cape Hatteras, North Carolina. Geomorphology 48:269–287

Wootton LS, Halsey SD, Bevaart K, McGough A, Ondreicka J, Patel P (2005) When invasive species have benefits as well as costs: managing *Carex kobomugi* (Asiatic sand sedge) in New Jersey's coastal dunes. Biol Invasions 7(6):1017–1027

Yang JY, Choi CU, Jeon SW, Hong SY, Kim YS (2010) An analysis of coastal topography and land cover changes at Haeundae Beach, South Korea. Acta Astronaut 67(9–10):1280–1288

Zawadzka-Kahlau E (1999) The trends of Polish Baltic coast development, in Polish. IBW PAN, Gdańsk, p 147

Chapter 15
Advances in Brine Disposal and Dispersion in the Coastal Ecosystem from Desalination Plants

Ramasamy Manivanan

Abstract The unpredictable water pollution incident of the coastal environment is an important part of the emergency work. Based on the Navier stokes equation set, a 2-D pollutant dispersion model is built for the selected 2D domain in the coastal ecosystem. It calibrates key parameters with the various characteristics, rapidly and effectively achieves the forecasting of pollutant dispersion using implicit finite difference scheme in this paper. The model can maximize the information obtained from the available data and adapt to the fast evolution of the coastal bed as well as other external boundary changes. The experimental results show that the model provides a strong support for relevant decision makers with high usability and reliability. Reverse Osmosis (RO) Plant draws saline water from sea inlet and converts it into potable water and discharges brine water at outfall as effluent. The paper discusses mathematical model studies for dispersion of brine water with salinity of 63 ppt from a Reverse Osmosis (RO) plant into coastal waters at north Chennai. Studies were carried out to observe the brine water plume behavior in the vicinity of coastal area with different outfall locations. Initially the outfall was located at 1,000 m from shore and studied, subsequently studies were repeated for 750 and 500 m locations. The outfall plume dispersed in and around the outfall. The development of plume at each outfall location for different ambient conditions was studied. The possibility of effluent reaching the shore was also studied. The studies were carried out considering different orientation of port, and the dilution aspects of multiport and single port diffusers. The design discharge criteria and suitable outfall locations were determined from the studies. The study indicated that higher velocity and larger port diameter helps in enhancing dispersion rate and hence adverse effects on marine ecosystem can be minimized. Finally the outfall was located at 750 m from shoreline and the same was commissioned in the year 2007. The field

R. Manivanan (✉)
Mathematical Modeling for Coastal Engineering (MMCE), Central Water and Power Research Station, Khadakwasla, Pune 411 024, India
e-mail: vananrmani@rediffmail.com

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8,
DOI 10.1007/978-3-319-06305-8_15,

measurements were carried out for salinity concentration at different distances from the outfall (50–500 m in the interval of 50 m). The simulated salinity plume compared with field measurement. It could be seen that the model results with field measurement were considered to be satisfactory.

15.1 Introduction

Water, a limited finite resource, vital for the very existence of life on earth and a necessity for economic and social development and for environmental sustainability, is becoming a scarce commodity. This is caused by the population growth, the change of lifestyle, water pollution caused by human intervention, inefficient use of water and climatic changes with more frequent extreme events such as droughts and floods. Where, the availability of water cannot be increased by using conventional resources or by recycling or cannot be made available by demand management methods, the desalination of sea or brackish water offers an alternative solution.

The coastline separates the marine part of the world from the continents. It separates the salt-water communities from fresh water communities; it separates land life from water borne life. Some of the marine organisms move towards estuary for reproduction with less water salinity. 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 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 also attracts a lot of human activities. The coastal zone is an indispensable resource for mankind. It provides excellent opportunities for fisheries, housing, recreation, transport, defense purpose, industrial activities and many others. The influx of people with their social and economic activities, their needs for safe housing, food, drinking water, and with the tremendous quantities of waste they are producing pose serious threat to the unique features of the coastal zone. It is therefore necessary to study this valuable region so as to prevent those competing activities which gradually cause depletion of the resources of the coastal zone.

15.2 Desalination Plants

A large number of small desalination plants, based on the reverse osmosis (RO) process, are already functioning in India. Water managers and experts point out Chennai's satisfactory experience with the five units in the city, producing 500,000 l a day. Metrowater set up three plants at Nochikuppam, Kasimedu, and Velachery, in the area around Chennai, in 1977–1978, and two at Kasimedu and Ayodhyakuppam in 2001.

The seawater-based desalination plant at Narippaiyur in Tamil Nadu, a major plant in south Asia, is successfully supplying drinking water to 264 villages in the Ramanathapuram district, according to the Tamil Nadu Water Supply and Drainage board (TWAD). The plant, installed recently, has a capacity of 3.8 million liters per day and covers a population of 235,000, providing an average of 10 l of drinking water per capita.

The Tamil Nadu government has approved the installation of 45 desalination plants in Ramanathapuram district and the installation work has been entrusted to Bharat Heavy Electricals Limited (BHEL), a Government of India undertaking. Out of 45 plants, 2 major ones are at Narippaiyur and Rameshwaram (300,000 l per day). The remaining 43 smaller plants (20,000–300,000 l per day capacity) have been designed for treating brackish water from bore-well sources. Tamil Nadu is providing a leading example of how to deal with potable water shortages, not only just for India, but also for countries throughout the world. If combined with a major commitment to nuclear power as well, the potential for solving this life-threatening problem is clearly in sight. At least some in that state have come to realize that Tamil Nadu, with its long coastline, is not really water-short.

There are indications that Tamil Nadu may get what it needs. BARC, who played a key role in developing the desalination plant coupled to a nuclear electricity station in Kalpakkam, has already supplied 15 desalination plants to different Indian states, and constructed a large desalination plant adjacent to the Madras Atomic Power Station (MAPS) at Kalpakkam.

BARC has already commissioned a 1,800-cubic-meters-per-day Nuclear Desalination Demonstration Project (NDDP) at Kalpakkam, using reverse osmosis technology. The remaining 4,500-cubic-meters-per-day plant, which is under construction at Kalpakkam on MSF water purification technology, will be commissioned soon. Manivanan et al. (2006) described about the brine water dilution and dispersion in the coastal region at Ennore using CORMIX model. The study indicated that higher velocities of currents were influenced best dilution and dispersion of saline water.

15.3 Numerical Solution

Numerical solution of complete Navier-Stokes (N-S) equations for laminar flow and corresponding Reynolds averaged equations for turbulent flow have received a great deal of attention as they describe flow with any level of complexity, the only uncertainty being in the closure of turbulence. Analytical solution of N-S equations are available in case of varieties of parallel linear and simplified flow which are not at all true in case of real world fluid flow problems having series of nonlinear physical phenomena and irregular shaped physical boundaries inside and on the closed boundary of fluid domain. The vertically integrated 2D shallow water equations represent coastal hydrodynamics, which are derived from N-S equations with the assumption of hydrostatic pressure. The pioneering and leading contribution in numerical simulation of shallow water equation are Leendertse (1967) and Grotkop (1973) which are based on finite difference (ADI) method and finite element method respectively. Recently, the depth integrated shallow water equations are used to simulate flows with fractional step finite element method by Jiang et al. (2009). Caceres et al. (2008) simulated coastal hydrodynamics of Barcelona harbor located on the Spanish Mediterranean coast by using 2DH circulation model named Limicir using specific sponge type open boundary condition so as to reduce spurious boundary effects in ADI method. In a classical and philosophical paper by Maghsoudi, it was argued how a few simple laws (conservation of mass and momentum) can explain many natural phenomena meaning thereby numerical solution of N-S equations representing real world problems becomes a mixture of art and science.

The dispersion process under flowing currents is two-dimensional in nature and particularly complex in the near-field region. Although many numerical studies on the dispersion of line plumes have been conducted previously, most of them deal with a 1-D simplified configuration of an infinite diffuser under perpendicular currents. Also, because of the various length and time scales associated with the transport of effluent plumes, both the near-field modeling and the far-field modeling are performed separately. The problem with employing separate near and far-field models is disability of treatment of interface between two fields. In this study, the far-field transport of analysis of the plumes that were contained in wastewater effluents was investigated using the two-dimensional plume tracking model with normalized characteristic equations.

Chennai is situated all along the coast of Bay of Bengal. It has potential for commercial and industrial activities and has witnessed rapid growth of industries during past few decades. In the present case a study was carried out to assess the impact of brine effluent to be discharged into coastal area from a reverse osmosis plant containing high salinity of 63 ppt. Gupta et al. (2002) reported that the tidal currents which advect, disperse and dilute the effluent, play a dominant role in governing the pollutant levels in the coastal environment. Vaze et al. (2002) presented extensive studies on selection of location of intake and outfall for a power plant at Ennore. In their selection, the intake was located at Ennore port basin and outfall near the creek mouth. It restored the imbalance in the tidal influx and efflux. Balakrishna et al. (1997, 2003); Balakrishna and Kumar (2005) demonstrated the need for detailed studies before discharging the effluents from industries, power plants and desalination plants into natural waters like rivers, creeks and coastal waters. They carried out studies for thermal discharges using CORMIX model in Sikka creek, salt flow in Kharao creek, and thermal discharges using CORMIX model in Amba river. Yan et al. (1992) reported various similarity criteria developed from the momentum equation of the variable density flow and experimentally examined regarding the effect of model designs. The present study involves measuring of plume behavior in the vicinity of the coastal area with different outfall locations. The proposal includes a seawater intake and discharge system in the vicinity of North Chennai, Tamil Nadu, India (Fig. 15.1).

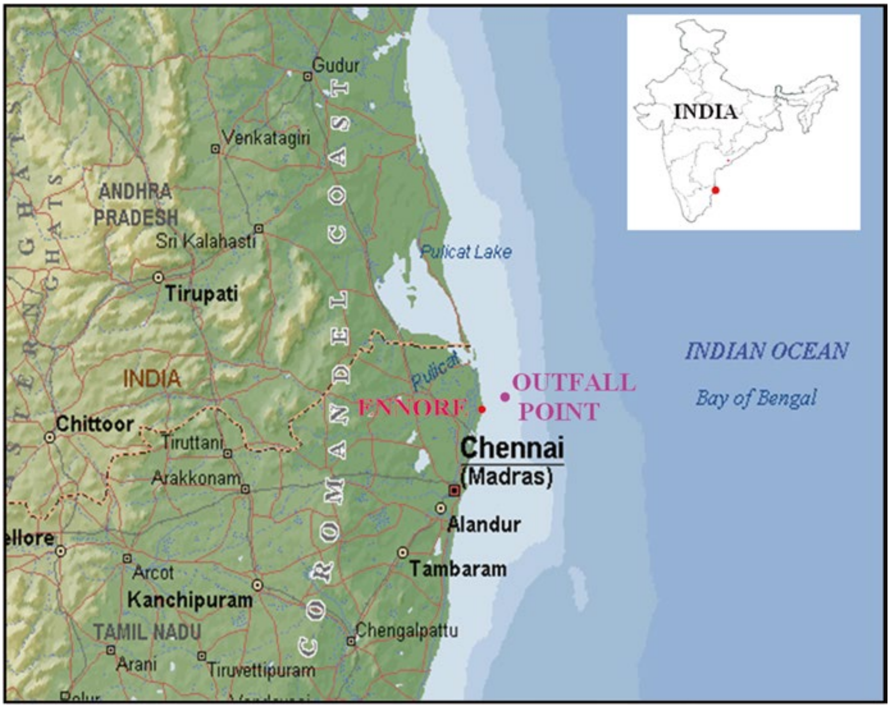


Fig. 15.1 Location plan of the desalination plant with outfall point

The brine water from the Reverse Osmosis plant is to be discharged at a suitable location in the sea such that the dilution efficiency maximum with higher velocity with the critical situations. The alternative outfall locations such as I, II and III were considered at 1,000 m, 750 m and 500 m distances respectively from the shoreline.

Finally, the outfall was located from the model studies is about 750 m from the shoreline. The desalination plant with intake and outfall was constructed during the year 2006 and running very smoothly. The field measurements were carried out for salinity concentration at different locations from the outfall (50–500 m in the interval of 50 m) for the monsoon and non-monsoon seasons. Tide, current and salinity measurement were carried out using tide gauge, current meter and salinity probe respectively. The measured data were compared with simulated data. It could be seen that the model results of salinity concentrations with field measured salinity concentrations were satisfactory. With this it could be drawn a conclusion that the model developed for the hydrodynamics and dispersion of salinity would be best option for the future salinity dispersion studies of east and west coastal ecosystem of India. The environmental impact of the salinity concentration also examined with the distribution of concentration in the coastal area.

15.4 Modeling Techniques

15.4.1 Governing Equations of Coastal Hydrodynamics

Mathematical model for hydrodynamics are basically based on three governing equations which have been simulated. However, the extended and modified form for large estuary has been developed by Singh and Ghosh (2001).

Continuity Equation

$$\frac{\partial d}{\partial t}+\frac{\partial (ud)}{\partial x}+\frac{\partial (vd)}{\partial y}=0 \tag{1}$$

Momentum Equations

$$\frac{\partial u}{\partial t}+u\frac{\partial u}{\partial x}+v\frac{\partial u}{\partial y}+g\frac{\partial h}{\partial x}+\frac{fu}{d}\left(u^2+v^2\right)^{1/2}+D\nabla^2 u-\Omega v+\frac{\tau_{sx}}{\rho d}=0 \tag{2}$$

$$\frac{\partial v}{\partial t}+u\frac{\partial v}{\partial x}+v\frac{\partial v}{\partial y}+g\frac{\partial h}{\partial y}+\frac{fv}{d}\left(u^2+v^2\right)^{1/2}+D\nabla^2 v+\Omega u+\frac{\tau_{sy}}{\rho d}=0 \tag{3}$$

where z is the elevation above datum (m); u and v are the depth-averaged components of velocity (m/s) in x and y direction, d is the total depth (=z+h); h is the depth below datum (m); f is the dimensionless friction coefficient; D is the horizontal eddy viscosity coefficient (m^2/s); Ω is the Coriolis parameter (radian/s); f is expressed by the following equation:

$$f=\frac{1}{32}\left[\log_{10}\left\{\frac{14.8d}{K_s}\right\}\right] \tag{4}$$

For refinement of curved boundary, a special technique proposed in Falconer (1990) has been adopted in which water depths at grid points outside the physical boundary curve are given appropriate values such that cross sectional area of such computational grid space is equal to that defined by actual prototype surface. At a grid point L located outside the actual boundary, the depth h is adjusted to give (Fig. 15.2)

$$h_i=h\left[\frac{2L_i}{\Delta x}-1\right] \tag{5}$$

where h is average depth at adjacent interval grid points and L_i is average distance between adjacent grid points and actual curved boundary. In the shallow water body system, this formula is adopted while defining the depth at tidal flat boundary at

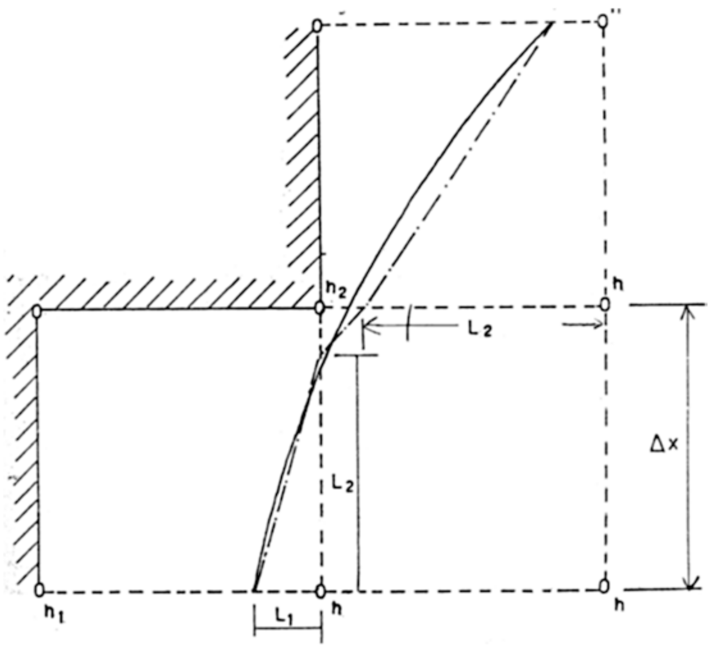


Fig. 15.2 Irregular boundary representation in finite difference method

both sides (North and South). Although for all intent and purpose, this technique conserves mass and momentum in general, special technique of weighted Taylor series approximation, as described by Falconer (1990), for curved boundary is taken particularly where the velocity field is very much governed by the momentum equations. Russell (1989) analyzed FDM and FEM and concluded that the two methods are actually quite similar and indeed FEM is a technique of deriving and justifying finite difference scheme and FEM can illuminate FDM properties that are difficult to perceive with customary FD framework.

15.5 Numerical Simulations

Taking into consideration all the above parameters/ constants, bathymetries and appropriate boundary conditions, the numerical integration has been started from the initial state of rest (i.e. z, u and v = 0) and the model run until it reaches a steadily repeating tidal state i.e. situation where the values of elevation and current components are periodic within 0.001 m and 0.001 m/s respectively, from one tidal cycle to next. The qualitative idea of steady state has been reported by Stephens (1986). Accordingly, model took five complete tidal cycles to bring into the state of steadily repeating tidal state.

The flow field in computational domain was simulated using a numerical model based on ADI finite difference technique. The theoretical formulations of the existing ADI numerical models as reported in Leendertse (1967) are not adequate to deal with such a high nonlinearity involved inside the computational domain.

Numerical solution of governing equations is always judged by numerical properties, viz. consistency, convergency, stability and accuracy. The more important in using variety of hydraulic problems is the stability and accuracy in case of high nonlinearity involved in computational domain. The traditional CFL criterion for the staggered grid (Stelling et al. 1986):

$$C_r = \Delta t \left\{ gh \left[\frac{1}{\left(\frac{\Delta x}{2}\right)^2} + \frac{1}{\left(\frac{\Delta y}{2}\right)^2} \right] \right\}^{\frac{1}{2}} \tag{6}$$

is to be replaced with T V criterion (Tengasanchole and Vongvisessomagi 1987; Singh et al. 2003), which is based on experimental criterion deciding stability).

$$\theta^{\frac{1}{3}} \frac{\Delta s}{\Delta t} \left(\frac{1}{\sqrt{gh} + \sqrt{U^2 + V^2}} \right) \geq \xi, \ \ for \ \ 0.8 \leq \theta \leq 1.0, \ \xi = 0.044 \tag{7}$$

On theoretical investigation, Leendertse (1967) found the ADI method to be unconditionally stable. But a source of inaccuracy known as ADI effect, reported by Weare (1979), Stelling et al. (1986) and Wilders et al. (1988), does arise where ADI models are used with large time step (High CFL number) in the flow domain with complex bathymetry. The ADI effect is reduced with a severe restriction on the step that limits the efficacy of the model. Such type of ADI effect has also been reported in Kim et al. (2003) who have applied ADI method with CFL number in the order of 2–3 in the case of a simple nonlinear domain of flume study. The important sources of error in computational hydraulics such as falsification of transportive and conservative properties of continuous equations during discretization and also other errors like aliasing, folding, nonlinear stability dispersion and diffusion reported by Maghsoudi and Simons (1992) in a finite difference numerical methods is a further topic for close agreement between field and numerical observation.

Looking into the hydraulic aspects of North Chennai coastal ecosystem, the schematization of the model area consists of computational lattice of (85 × 50) grid point, each grid having size of 50 × 50 m. Depth data (bathymetric information) are incorporated at all grid points with the help of measurement available. The water levels at three open boundaries were generated for a tidal range of 1.2 m with period of 15 days covering spring and neap phase of the tidal period.

With many trials, the model was found to be stable after extending it towards sea sides as well as reducing the CFL number to 1 as in case of explicit model, although it proved expensive in relation to CPU time. The model was calibrated and validated in respect of tidal variables (currents & levels) for neap as well as spring tide characteristics observed at the area.

15.6 Sensitivity Analysis

Sensitivity analysis in respect of various hydraulic parameters was also carried out. The sensitivity analysis shows that the numerical model for the proposed location is very sensitive to these parameters, such as bottom friction and eddy viscosity. Being computational domain of highly nonlinear in nature, the convective terms are significantly important and nonlinear terms in hyperbolic wave equation may induce instability by trapping short waves as time progresses (Singh and Prasad 2005; Singh and Bagwan 2004). The calibration procedure involved the assessment of sensitivity to variations in bottom friction and internal friction (eddy Viscosity) and to a certain extent a judgment and experience in running the model. Accordingly, a momentum diffusion coefficient of 10 m^2/s along with other smoothing technique at the cost of reducing CFL number in implicit model was found necessary to counter the nonlinear instability and inherent negative diffusion. Validation of such method including its accuracy aspects is reported by Singh and Prasad (2003).

15.7 Model Conditions

The diffuser system is intended to provide initial dilution of the effluent by mixing with surrounding seawater. After coming out from the circular ports, there would be transfer of momentum between the jet and the surrounding water due to lateral shear. The surrounding seawater would be entrained in the jet progressively with distance from the exit point. While the spread of jet would increase with distance, the centerline velocity as well as the density difference would reduce progressively. As the inertial effects diminish, buoyancy effects would dominate. If at this stage, the effluent is buoyant, the behavior would be like rising plume, however if it is negatively buoyant, as in the present case, the effluent would no more rise above this altitude and the effluent would start sinking to the seabed. In case the effluent discharge is released horizontally, it would reach the bottom quickly, however if the effluent is discharged through vertical port opening, it will take still relatively larger time and distance, dilution during this travel will also increase proportionately. As the total process is quite complex in nature, the mathematical model provides a handy tool for obtaining dilution under various tidal and input conditions for different alternatives.

A jet model is capable of taking into account ambient flow parameters like water depth, flow velocity, water density, salinity, diffuser characteristics like diffuser length, port diameter and discharge characteristics like flow rate, density and salinity. The model takes the above parameters as input and as output the model gives centerline position, dilution with respect to distance from the outfall. Gaussian

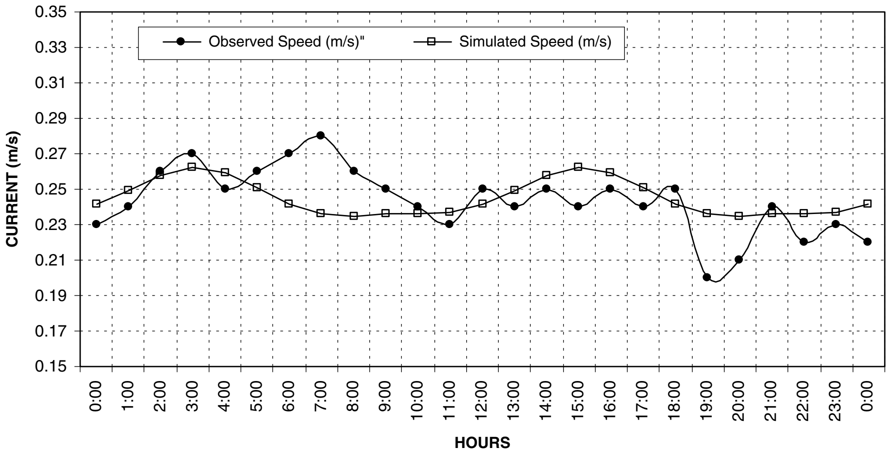


Fig. 15.3 Observed and simulated current at site

distribution is assumed for lateral spreading. As such half width of the jet at any distance from the outfall can be found out. In tidal situation, quasi-steady state condition is assumed for different phases of tide, and the model was run and the results were obtained.

As could be seen from the computational grid plan, there are three open boundary conditions that are required to be provided for simulating flow field in the project site. These boundary conditions have been derived from field observation data. It could be seen from the velocity field that the flow is unidirectional in north – south direction. Several trial runs of hydrodynamic model were made by varying Manning's bed roughness, wind speed/direction etc. to get the observed flow field in the region. With the Manning's bed roughness of 0.031, wind speed of 10 m/s and direction 90° N, the flow field corresponding to observed northward current of the order of 26 cm/s was simulated in the model. The observed and computed velocities are shown in Fig. 15.3. Considering the low velocities prevailing at the site, calibration of the currents is considered to be satisfactory (Manivanan 2008; Manivanan et al. 2006).

The model studies were conducted for the following conditions. Single port with diameter of 0.15, 0.20, 0.25 and 0.3 m and discharging horizontally and vertically with an angle of 90°. All the diffuser ports would be located 2 m above the seabed and in water depth of 9.0 m. The different velocities were considered such as 0.05, 0.10, 0.25, 0.40 and 0.6 m/s. Excess salinity of 31 ppt above ambient (i.e. total 63 ppt) is considered at diffuser discharge. The brine discharge would be 12 MGD (0.625 cumec).

15.8 Results

The field studies were carried out for the existing outfall locations considering following environmental conditions during monsoon (November 2010) and non monsoon (May 2010) seasons.

Table 15.1 Salinity gradient from the outfall location

Sl. no.	Distance (m) from outfall	Velocity-northward current (m/s)	Salinity (ppt) (monsoon season)	Velocity-southward current (m/s)	Salinity (ppt) (non-monsoon season)
1	50	0.25	35.5	0.15	35.75
2	100	0.28	35.48	0.15	35.70
3	150	0.30	35.40	0.17	35.68
4	200	0.31	35.3	0.15	35.55
5	250	0.32	35.29	0.16	35.51
6	300	0.29	35.49	0.18	35.48
7	350	0.26	35.48	0.14	35.84
8	400	0.28	35.48	0.15	35.72
9	450	0.31	35.3	0.18	35.47
10	500	0.30	35.2	0.2	35.4

15.8.1 Tides and Tidal Currents

The tides observed at the site are semi-diurnal with low amplitude (NIOT 2002). The maximum spring tidal range is 0.6 m while neap tidal range is 0.2 m. The minimum current speed ranged between 0.06 and 0.13 m/s, the maximum current speed range between 0.53 and 0.63 m/s. The current direction changes from south to north during February and from February to September the coastal currents are northerly.

15.8.2 Wind

In general, during SW monsoon, wind blows predominantly from SW with intensity of about 13–17 m/s. Wind with equal strength blows predominantly from NE direction during NE monsoon.

15.8.3 Salinity

The seawater salinity of the study area is 35 ppt. The salinity of effluent is of the order of 63 ppt, which is nearly twice the ambient salinity. The Table 15.1 and Fig. 15.4 show the salinity gradient from the outfall location. Photographs of dispersion of brine waste in the coastal ecosystem obtained on 21 May 2010, showing the surface brine water plume extending North/South from the outfall (Fig. 15.5).

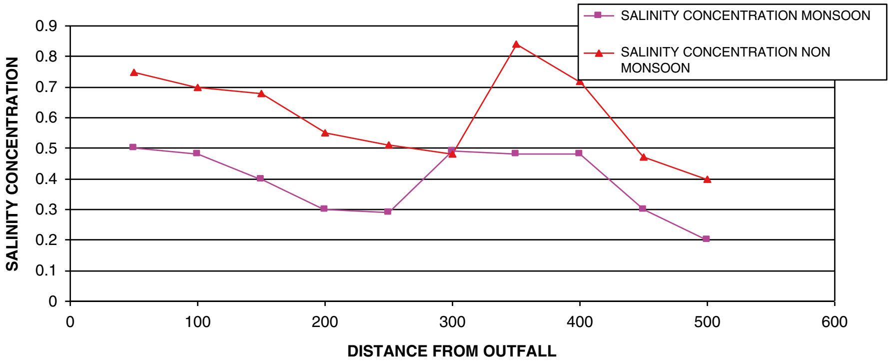


Fig. 15.4 Salinity concentration above ambient with distance from the outfall

Fig. 15.5 Photographs of dispersion of brine waste in the coastal ecosystem

15.8.4 Temperature

The surface temperature distribution in the area exhibits the presence of warm water (temp 30.0 °C) at the coast and relatively cold in the offshore region (temp 28.90 °C). The near bottom temperature varies from 29.3 °C at the coast to 28.5 °C offshore. The maximum seawater temperature variation was found to be 1.1 °C in the near shore zone. This water can be attributed to the release of warm water from the nearby thermal power station.

15.9 Model Prediction

From these studies it was possible to predict the development of salinity plume at the outfall location. The results provide, for the given ambient condition, discharge condition, the salinity and plume spreading in x, y, and z-axes.

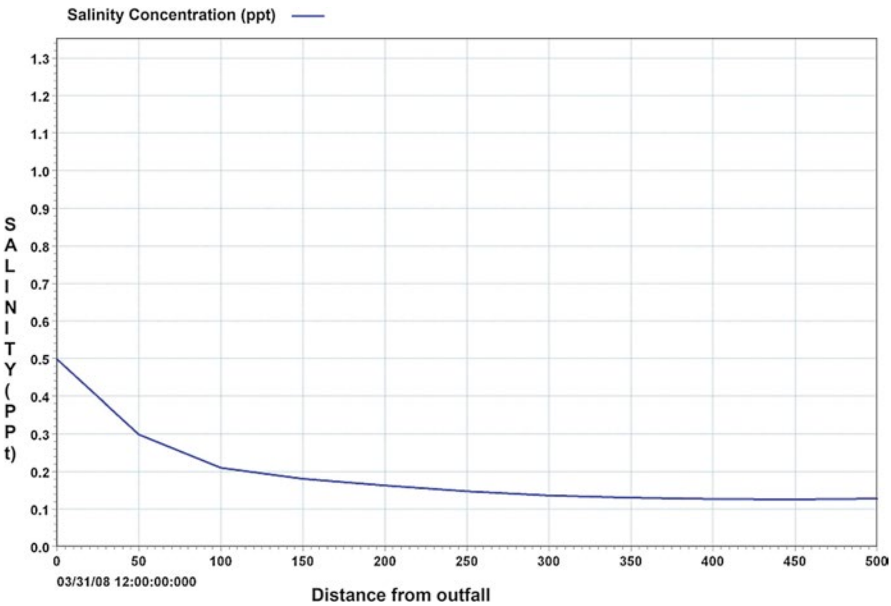


Fig. 15.6 Model prediction salinity concentration over the distance from the outfall

Further, hydrodynamic dilution with reference to concentration, top hat thickness measured vertically, top hat half thickness measured horizontally in y-direction, upper and lower plume boundaries in Z- coordinates can be obtained and presented. The region of interest in the downstream distance is considered up to the point where salinity of 0.1 ppt above ambient condition is reached (Fig. 15.6).

15.9.1 Northward Current

The plan view of the plume in the near field shows the concentration along downstream and along centerline for 0.05 m/s ambient velocity with port diameter of 0.15 m. It is seen that plume extends about 566 m from the outfall location. Similarly the plumes for 0.20, 0.25 and 0.30 m port diameters extend about 603, 739 and 908 m respectively. It may be noted that the plume boundary represents where the dilution reaches 0.1 ppt above ambient condition. For 0.1 m/s ambient velocity and 0.15, 0.20, 0.25 and 0.30 m port diameters salinity plume extends about 395, 597, 672 and 762 m respectively from the outfall location. For 0.26 m/s velocity and 0.15, 0.20, 0.25 and 0.30 m port diameters the salinity plume extends about 188 m 166, 438 and 603 m respectively from the outfall location. Similarly studies were conducted for 0.1, 0.25, 0.4 and 0.6 m/s ambient velocities with different port diameters and all above conditions. The summary of all cases is shown in Fig. 15.7. It is seen from Fig. 15.7 that as the current velocity increases, the length of the plume decreases. Also, the length of the plume increases as the port diameters increases (Table 15.2).

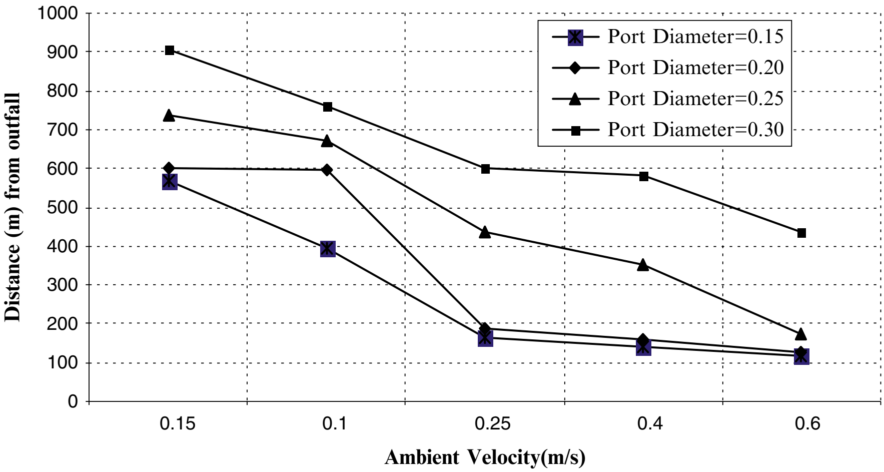


Fig. 15.7 Dispersion distance from outfall up to 0.1 ppt concentration of salinity

Table 15.2 Distance from outfall location with salinity concentration

			Distance (m) from the outfall	
Sl. no.	Velocities (m/s)	Port diameter (m)	0.1 ppt concentration	0.01 ppt concentration
1.	0.05	0.20	603	4,320
2.	0.10	0.20	597	3,818
3.	0.26	0.20	166	4,218
4.	0.40	0.20	160	3,751
5.	0.60	0.20	126	5,000

15.10 Discussion

After reviewing measured and simulation results, the scenario considering the prevailing south wind and wave without changing the current location of the outlet causes the greatest excess salinity in the inlet, and either the saline exceed more than an allowed amount. The brine plume would be felt up to distance of about 566 m in the near field dilution and about 1,500 m in the far field dilution and dispersion to obtain nearly ambient condition by self-purification, dilution and dispersion.

15.10.1 Brine Pollution Environmental Assessment in Coastal Development Plan

According to the Indian Department of the Environment (DOE) standards the brine water discharge into a water resource must not make an increase in the water ambient salinity more than 0.1 ppt in a distance equal to 200 m far from the discharging location. The measured and model result show that the brine water in Ennore coastal area exceeds than the permissible limit and in order to decrease its harmful impacts, some considerations, mentioned below, should be applied. In order to simulate each phenomenon by software, it is necessary to collect various data for a long period of time to use them as an input for the model. Availability of a complete time series data is too important to accomplish a simulation properly and to use the results of these basic models for later predictions and purposes. The final goal of data collection is to calibrate and verify a simulation. Therefore, the data -mentioned below- is necessary to be collected for Ennore region from meteorological stations in land and sea, and buoys: Wind direction and velocity, precipitation, evaporation, water temperature, water flow direction and velocity, wave conditions such as height, period, and direction and so on (Stephens 1986).

There are various methods to reduce and control the salinity pollution problem. Among which are: using the diluters because of existence of high amount of dissolved matters in the sea water that can be deposit in the pipes-, dilution through the outlet way by adding sea water that it does help to reduce the salinity concentration, discharging the brine water with the help of diffusers with vertical direction with high velocity currents for some time to get dilution, and finally, dividing the brine water amount and discharging each portion of it in different locations to reduce its harmful effects.

The above observations of the plume show that the flow in the surface layer responds within a few hours to changes in wind forcing. The shape and direction of the observed plume from the coastal outfall also depends on the antecedent current speed and direction during the previous days. The surface transport vectors were calculated from the Ekman transport data for the grid cell centerd on 14.70° N and 80.21° W, obtained from the survey group, the surface transport in shallow water was calculated on the assumption that it occurs at ~45° to the right of the wind direction. The wind forcing, causing upwelling, was more persistent during the summer of 2009 than that of 2010, which explains the larger decrease in sea surface temperature observed. The observations of 27 May 2010 and 8 November 2010, in which the brine wastewater discharge was seen as a narrow, long, southward moving plume, show that tidal effects can be important near the inlet channel. Nearshore, as water depth decreases, the interaction of swell waves with the shoaling bottom generates a turbulent water mass that is convergent on the shore. This onshore transport sustains a pressure gradient that blocks further onshore surface flow, as observed by the movement of the wastewater plume on 21 May 2010 and 1 November 2010. Consequently, at the interface between this mixed nearshore water mass and the waters further offshore, a distinct interface is marked by a foam line and an increase in turbidity nearshore. During this investigation, covering the spring and summer situations, offshore transport of the wastewater plume was not observed at the surface as might have been expected by a simple upwelling shelf circulation. Commonly the transport was to the south, in the direction of the inlet channel, or with an onshore component. Impacts of the pollutants released are, therefore, to be anticipated in the nearshore waters and potentially within the outfall location. Under the terms of the Water Act 1981, has designated its entire east coast as a ‘less sensitive area' but the Ennore has been designated as a 'sensitive area' in terms of eutrophication. The observations described above suggest that the eutrophication of the Ennore is unlikely to change as a result of the operation of the submarine outfall, which had as its objective to reduce the nutrient loading to the coastal region. Furthermore, this study demonstrates how simple observations of brine wastewater discharge from a submarine outfall can improve understanding of nearshore circulation.

15.11 Conclusion

The study resulted that as the effluent discharge with higher salinity is negatively buoyant (i.e. heavier than ambient water), the port with the nozzle opening should be kept vertically upwards and atleast 2 m above seabed. This would result in higher dispersion of effluent as compared to the horizontal nozzle. In all the combination runs, the effluent after dilution sinks to the bottom and then moves along the seabed.

As a result, the dilution process slows down. The field observations studies and the model studies indicated that the tidal influence due to the outfall is felt with in a distance of about 200–250 m. It could be concluded that the outfall location at a distance of 750 m from shoreline at a depth of 6.0 m with port opening of 0.15 m diameter is more suitable in achieving maximum dilution within a distance of 236 m from the point of discharge. Thus the distance of 500 m between the intake and outfall is found suitable. The study showed that higher velocity and larger port diameter helps in enhancing dispersion rate but considering the discharge of 12 MGD which is comparatively small, 0.15 m port diameter is found suitable. The influence of the plume would be felt up to distance of about 566 m.

The study indicated that higher velocity and larger port diameter helps in enhancing dispersion rate and hence adverse effects on marine ecosystem can be minimized. Finally 750 m was commissioned in the year 2007. The field measurements were carried out for salinity concentration at different distances from the outfall (50–500 m in the interval of 50 m). The simulated salinity plume compared with field measurement. It could be seen that the model results with field measurement were considered to be satisfactory.

Acknowledgement The author is thankful to Shri. S. Govindan, Director, CWPRS for giving permission to publish this paper.

References

Balakrishna S, Dilip Kumar R (2005) Model simulations for thermal outfall systems in rivers. Mumbai, HYDRO, India, pp 492–499

Balakrishna S, Dilip Kumar R, Ghosh LK (1997) Analysis of thermal discharges in coastal waters using numerical models. INCHOE, pp 595–604

Balakrishna S, Sharma VB, Singh AK (2003) Studies for disposal of effluents from power plants in to Thane creek water, Mumbai, HYDRO, India

Caceres I, Grifoll M, Gonzalez-Marco D, Sanchez-Arcilla A, Espino M (2008) Relevance of different driving terms in the Barcelona harbour winter circulation: field study and numerical model performance. ASCE J Waterw Port Coast Ocean Eng 134(5):275–285
Falconer RA (1990) Lecture series on mathematical modelling of flow and pollutant transport in river, lakes estuaries and coastal water, at Hydraulic Research Wallingford, Oxford, pp 6–32
Grotkop G (1973) Finite element analysis of long period waves. Comp Methods Appl Mech Eng 2:147–157
Gupta I, Shivani D, Chandorkar AA, Anjali S (2002) Water quality modeling of waste water discharges in Thane creek, HYDRO 2002, pp 490–494
Jiang C, Yang C, Liang D (2009) Computation of shallow wakes with the fractional step finite element method. J Hydraul Res 47(1):127–136
Kim CW, Yooh TH, Cho YS, Kim ST (2003) A two-dimensional conservative finite difference model in non-orthogonal coordinate system. J Hydraul Res IAHR 41(4):395–403
Leendertse JJ (1967) Aspects of computational model for long period water wave equation, Report RM-5294-PR. The Rand Corporation, Santa Monica, 165 pp
Maghsoudi N, Simons DB (1992) Important source of errors in computational hydraulics. In: Proceedings of the hydraulic engineering, ASCE, pp 392–397
Manivanan R (2008) Water quality modelling lakes, reservoirs and estuaries. NIPA, New Delhi
Manivanan R, Renganath LR, Kanetkar CN (2006) Mixing zone simulation of brine water and solid dispersion performance in the coastal environment using mathematical models. APD IAHR international conference held at IIT, Chennai
NIOT (2002) EIA & Risk Assessment Report for CPCL, Chennai NIOT, Chennai
Russell TF (1989) Finite elements and finite differences: are they really different, and does it matter? FEM analysis in fluids, edited by Chung TJ, pp 538–543
Singh CB, Bagwan A (2004) ANN model development for bank-line migration of river Brahmaputra using remote sensing data, ISH, Jl. Hydraul Eng 10(1):56–64
Singh C B, Ghosh LK (2001) Discussion on “tidal regime in Gulf of Kutch, West coast of India”. J Waterway Port Coastal Ocean Eng November:353–356
Singh CB, Prasad N (2003) Discussion on “three-dimensional numerical simulation for an open channel flow with a constriction”. J Hydraul Res IAHR 41(4):439–441
Singh CB, Prasad N (2005) Discussion on “a two-dimensional conservative finite difference model in nonorthogonal coordinate system”. J Hydraul Res IAHR 43(6):728–730
Singh CB, Atkekar ND, Sharma VB (2003) Mathematical model studies to determine length of approach jetty for fishing harbour. In: Proceedings of the conference on hydraulics and water resources, held at CWPRS, Pune, December, pp 240–243
Stelling GS, Weersma AK, Willemse JBTM (1986) Practical aspects of accurate tidal computations. J Hydraul Eng ASCE 112:802–817
Stephens CV (1986) A three dimensional model for tides and salinity in the Bristao channel. Cont Shelf Res 6(4):533–560
Tengasanchole T, Vongvisessomagi S (1987) Stability analysis of two-dimensional depth-averaged model. J Hydraul Eng ASCE 115(9):1204–1222
Vaze VV, Kale AG, Manjunatha SG (2002) Model studies for selection of location for intake and outfall for a power plant, HYDRO 2002, pp 502–507
Weare TJ (1979) Errors arising from irregular boundaries in ADI solutions of the shallow water equations. Int J Numer Methods Eng 14:921–931
Wilders P, van Stijn TL, Stelling GS, Fokkema GA (1988) A fully implicit splitting method for accurate tidal computations. Int J Numer Methods Eng 26:2707–2721
Yan Z, Jiang C, Zhou C, Liu M (1992) Study on hydraulic modeling of ocean outfalls. International symposium on hydraulic research in nature and laboratory, China, at Beijing

Chapter 16
Estuaries Ecosystems Health Status – Profiling the Advancements in Metal Analysis

Ahmad Zaharin Aris and Ley Juen Looi

Abstract Estuaries are highly productive areas, which play an important role in both ecological and socioeconomic aspects. It is an excellence sanctuary for a wide variety of aquatic organisms and wildlife that breed and inhabit in these areas. Despite the importance of estuaries, they are being threatened and damaged at an alarming rate. Metal pollution is among the most serious environmental crises in estuaries ecosystems. The anthropogenic activities had resulted in the increase of metal elements released or leached into the environment. As such, deterioration of estuaries ecosystem health status by these substances promoted an urgency to monitor the concentration of metals in the environment. Although metals pollution studies have been carried out extensively all over the world but unfortunately there are still no measurements that can be fully rely to predict the effect of metal pollution on estuarine ecosystems. Therefore, current chapter is indispensable to demonstrate the advances of analytical methods and detection techniques available for metals analyses. In addition, environmental forensic approaches and application of various metal pollution indicators, indices, modeling and statistical analysis in assessing estuarine ecosystem health status was also has been highlighted in this chapter. This chapter also pointed out the gap of knowledge which should be addressed for future risk assessment of metals pollution in estuaries ecosystem. Current chapter could be served as a reference for future metals comparative studies and monitoring works to be carried out effectively.

A.Z. Aris (✉) • L.J. Looi
Environmental Forensics Research Centre, Faculty of Environmental Studies,
Universiti Putra Malaysia, 43400 UPM Serdang, Selangor, Malaysia
e-mail: zaharin@upm.edu.my

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8,
DOI 10.1007/978-3-319-06305-8_16,

16.1 Introduction

Nowadays, estuarine ecosystem faces great challenges with regard to metal pollution caused by rapid industrialization and urbanization. Intense human activities in smelting, mining, petrochemical industry, printing, aquaculture, agriculture, electronic industry, sewage and municipal produces domestic wastes, industrial effluents, agricultural runoff, oil spills from vehicles and sewage discharges into estuaries systems (Milenkovic et al. 2005; Bainbridge et al. 2009; Sany et al. 2013; Tabinda et al. 2013; Wang et al. 2013). As final recipient of contaminants, estuaries systems are readily receiving air-based, land-based and sea vehicles-based pollutants from anthropogenic activities (Sany et al. 2013; Yarsan and Yipel 2013). Recent scientific studies found that many coastal areas including estuaries have been intensely contaminated by metals (Wan et al. 2008; Ding et al. 2009; Fang et al. 2009; Deng et al. 2010; Wang et al. 2013; Yarsan and Yipel 2013).

Generally, metals are present everywhere in the environment (Granero and Domingo 2002; Dorne et al. 2011; Yang and Liu 2012). They are highly persistent, toxic, non-degradable and readily dissolved in water, bioaccumulated and biomagnified in food chains (Mendil et al. 2005; Radojević and Bashkin 2006; Costa et al. 2009; Wang et al. 2013; Tabinda et al. 2013). Naturally, trace amount of metals existed in various matrices: water, soil or sediment, organism, and food (Yang and Liu 2012). Some metals such as manganese (Mn), copper (Cu), zinc (Zn) plays important roles as essential micronutrients for human body as structural and co-factor components of enzyme (Mushtakova et al. 2005). On the other hand, metals such as mercury (Hg), lead (Pb), cadmium (Cd) and arsenic (As) possess serious threat on the ecosystems and human health (Radojević and Bashkin 2006; Yang and Liu 2012; Zhang and Wang 2012). These poisonous substances are capable to disrupt endocrine functionality and affect central nervous system of organisms (Dyer 2007; Zheng et al. 2010; Yang and Liu 2012). However, either kind of metals in excessive amount than certain threshold concentration would impose negative effect to human health and ecosystems (Granero and Domingo 2002; Mushtakova et al. 2005; Liu et al. 2012; Sherman et al. 2013; Yang and Liu 2012).

Until earlier of twenty-first century, there are only a small amount of chemicals, belongs to the most crucial environmental contamination group (Yarsan and Yipel 2013). In fact, majority of these chemicals were originated from natural constituent such as plants, organisms and mineral background. However, Yarsan and Yipel (2013) recently reported that anthropogenically chemicals production was increased drastically from 7 million tons in 1950s to about 400 million tons today. Elevated amount of metals in environment can provoke many environmental crises such as loss of habitats and biodiversity, increase in mortality rate and extinction of aquatic organisms, metals pollutions in estuaries and coastal systems, and bioaccumulation in food chain (Essl et al. 2013; Sany et al. 2013).

Thus, in recent years, estuaries ecosystems health status deserved worldwide attention stem from its socioeconomic and ecological importance (Bayen 2012). Along with the increasing of world population and tremendous ability of marine environment in socioeconomic and ecological aspects, scientific studies has advance in recent years in order to fulfil the demands and benefit more from the seas (Yarsan and Yipel 2013). An overview of advancement on estuarine researches in term of matrices, detection limit, scope, analytical approaches, and data approaches is depict in Fig. 16.1. Much scientific studies have focussed on the potential threat of metals in aquatic environment owing to their persistence, toxicity and bioaccumulation ability (Pan and Wang 2012b; Sany et al. 2013). To date, many researchers have focused on ascertain metals concentration and its distribution, in various environmental matrices (the estuarine water, sediment and aquatic organisms) of the estuaries environment, in order to evaluate and assess the estuarine health status (Morillo et al. 2008; Sany et al. 2013; Yarsan and Yipel 2013). Among all the elements, arsenic, cadmium, lead, mercury, (Sun et al. 2012; Luque-Garcia et al. 2013) and copper (Sun et al. 2012) had received substantial focus on estuaries pollution studies.

16.2 Advances in Estuarine Water Research

Metals studies in estuaries water are limited due to its difficulty in the determination of metals in high salinity seawater samples (Wang et al. 2013). Besides, low concentration of dissolved metals in water matric, providing a representative data for estuaries water on metal pollution has become a great challenge. Until lately, chemical analysis has been used as the conventional methods for water pollution monitoring. Still, some metals have low stability in water matric making them difficult to be detected using conventional analysis (Ünlü et al. 2008). A chemical analysis alone is regarded as non-representative for the judgment of the pollution impacts for estuaries ecosystem since it does not delineate the detrimental effects and fate of pollutants through biotransformation within the body of living organisms (Yarsan and Yipel 2013). Yet, these methods are costly, tedious and insufficient to obtain representative data for dissolved metals in estuaries and coastal water (Sany et al. 2013).

Since water quality monitoring alone is insufficient to represent the health status of an estuaries, aquatic organisms are used to determine the water quality in an estuary system (Ahmed and Bibi 2010; Tabinda et al. 2013; Yarsan and Yipel 2013). Following this, many international programs have employed biological approach to complement with typical chemical approach in determining the effects of estuarine and coastal pollution (Yarsan and Yipel 2013). In this case, biomarkers have been used to accompany with the classic chemical analysis with a view to illustrate persistence biological responses of environmental pollution (Yarsan and Yipel 2013). However, for ecosystem health status purposes, metals levels in various matrices such as water, sediment and biota are equally important to elucidate the level of metal pollution in estuarine and marine ecosystems (Looi et al. 2013; Yarsan and Yipel 2013). Therefore, it is suggested that all environmental matrixes namely water, sediment, and biota should be incorporated in estuaries monitoring programs.

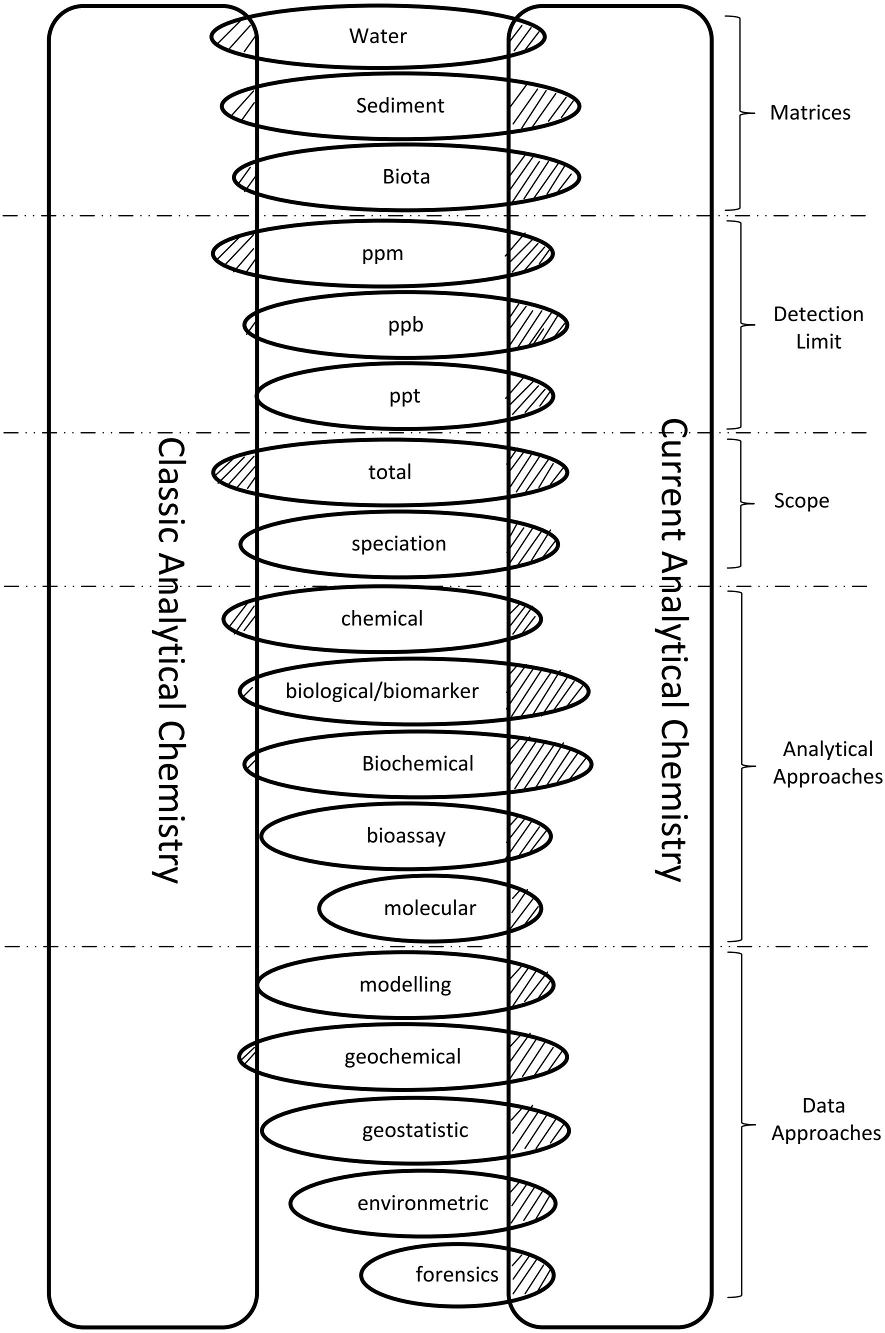


Fig. 16.1 Conceptual and technical evolution of analytical chemistry for metal analysis in estuaries system

16.3 Advances in Estuarine Biota Research

With the emergence of estuary pollution as one of the most important element of ecotoxicology, biomonitoring programs are indispensable to check the ecosystem health status (Shokrollahi et al. 2008; Cevik et al. 2008; Tabinda et al. 2013; Yarsan and Yipel 2013). In aquatic ecotoxicology studies, biomarker has been conventionally exposure to pollutants in *in vitro* test systems (Yarsan and Yipel 2013). These ecological studies have been assisting the development of guidelines for legislative means, which intended at reducing the negative impacts of man-made activities on aquatic environment. Now, biomarkers have become essential elements in environmental monitoring works as it could signal early warning of toxicological effects (Schettino et al. 2012; Sany et al. 2013).

Aquatic organisms (fish, crustaceans, algae, protozoa, macrophytes, bacteria and plankton) are extensively used as biomarker in determining the pollution status of aquatic systems for environmental pollutants (Alinnor and Obiji 2010; Rodrigo et al. 2013; Tabinda et al. 2013; Wang et al. 2013). Biomarker is a new method and has been recognized as sensitive, valuable and powerful tools to reflect the impact of contaminants in the period of biota lifespan due to their ability to accumulate both essential and non-essential elements (Sany et al. 2013; Wang et al. 2013; Yarsan and Yipel 2013). They inhabit and feed in polluted aquatic environment and are capable to response specifically to stressor or environmental changes by changing life functions through metals accumulation in their bodies (Yarsan and Yipel 2013). Bioaccumulation ability of biomarkers is largely depending on species of organisms and types of metals. For example: Pan and Wang (2012a) reviewed that mollusks have relatively high capacity to accumulate Cu, Zn, Cd, Pb and As. On the other hand, shellfishes are capable to accumulate Ni from sediments and water (Ip et al. 2005; Wang et al. 2013) while shrimps can accumulate high levels of Cr in their shell tissues (Wu and Yang 2011; Wang et al. 2013). Thus, these organisms are able to provide comprehensive and biologically relevant information of the potential impact of pollutants on the health status of ecosystem (Tabinda et al. 2013; Yarsan and Yipel 2013). There are two types of biomarker: biomarkers of effect and biomarkers of exposure (Galloway 2006; Schettino et al. 2012; Sany et al. 2013).

1. **Biomarker of effects**: organisms that can detect early changes in their health status
2. **Biomarker of exposure**: organisms that can detect micro-pollutant in their earlier exposure

To date, mollusc's species, such as clams, cockles and mussels have been widely used in estuarine monitoring program. Molluscs are sedentary organisms and thus conceivably been employed to reflect estuaries heath status. They are good evidence on the presence of one or more chemicals in the environment (Depledge et al. 1993; Yarsan and Yipel 2013). Subsequent study done by Rodrigo et al. (2013) proposed that cuttlefish, *Sepia officinalis* to be a novel potential biomarker for risk assessment in polluted estuaries. The cuttlefish inhabit in brackish water at least during part of his life, is substantially important for pollution monitoring of estuaries ecosystems

(Rodrigo et al. 2013). However, further investigation is vital to decipher the adaptation mechanisms of this organism to its habitat (Rodrigo et al. 2013).

In order to have better understanding on the primary pathway of exposure to environmental stressor, calculation on bioconcentration, bioaccumulation and biomagnification has been done to determine the ability of aquatic organisms to uptake and accumulate metals in tissues and organs (Yarsan and Yipel 2013).

1. Bioconcentration Factor (BCF)
   Bioconcentration is regarded as bioaccumulation of metals uptake by organism from water via respiratory surface and/or dermal absorption (Yarsan and Yipel 2013). Bioconcentration factor (BCF) can be calculated using Eq. 16.1:

$$\mathrm{BCF} = \frac{C_B}{C_{WT}} \text{ or } \frac{C_B}{C_{WD}} \tag{16.1}$$

   C_B = metal concentration in organism (mass of metal/kg of organism)
   C_{WT} = total metal concentration in water (mass of total metal/L)
   C_{WD} = dissolved metal concentration in water (mass of dissolved metal/L)

2. Bioaccumulation Factor (BAF)
   Bioaccumulated contaminants that are magnified in the food chain have damaging effect on all organisms including predatory animals and human being (Lawrence and Mason 2001; Wang et al. 2013; Yarsan and Yipel 2013). Therefore, it is also an important aspect in learning the potential effects of metals on organisms and human health (Yarsan and Yipel 2013). Bioaccumulation is a process whereby living organisms accumulating toxic and harmful substances in their body (Sun et al. 2012). Bioaccumulation is defined as the total uptake process of water and food. In addition to the metal concentration in water, metals uptakes by dietary route are take into consideration [(Bioaccumulation = bioconcentration + biomagnification – (elimination + growth dilution)]. Bioaccumulation factor (BAF) has the similar formula to that of BCF (refer to Eq. 16.2) (Mackay and Fraser 2000; Yarsan and Yipel 2013).

$$\mathrm{BAF} = \frac{C_B}{C_{WT}} \text{ or } \frac{C_B}{C_{WD}} \tag{16.2}$$

3. Biomagnification Factor (BMF)

   Biomagnification is the increase of metals concentration in a food web through dietary absorption (Macdonald and Ingersoll 2002; Yarsan and Yipel 2013). Biomagnification factor (BMF) is calculated as in Eq. 16.3:

$$\mathrm{BMF} = \frac{C_B}{C_A} \tag{16.3}$$

   C_B = metal concentration in organism (mass of metal/kg of organism)
   C_A = metal concentration in organism's diet (mass of metal/kg of food)

Recently, metals are found to affect various organs and systems of living organisms. Significant difference of metals concentration has been found between various body parts of biota (Yarsan and Yipel 2013). Still, metals concentration alone cannot be used to assess the toxicity and magnitude of pollution in estuaries ecosystem. In fact, organisms normally have various food sources with different concentration in the aquatic environment. Besides, the toxic effects toward organisms are largely depending on the characteristic of each pollutant and accumulated tissue. It is important to note that metals that bioaccumulate in organism's body do not necessarily biomagnify. These facts have led to some difficulties in the determination of BMFs (Yarsan and Yipel 2013). Therefore, parameters such as biomarker's species, the age of bioindication, food sources, chemical characteristics of metals, metals concentration and exposure time must take into account during future biomonitoring process (Yarsan and Yipel 2013).

Now, many researches have focussed on integration approaches (chemical analysis and biological survey) to estimate and predict the detrimental effects of chemical and biological communities toward metals contamination (Sany et al. 2013). Chemical analysis and biological surveys are complement to each other. Both of these methods are equally important because chemical analysis was unable to predict biological effect while biological survey only can perform toxicity bioassay in environmental matrices (Sany et al. 2013).

Another notable improvement of scientific methods is the development of new molecular techniques such as proteomics, genomic, and metabolomics to increase potential biomarkers for metal pollution monitoring purposes (Sany et al. 2013). Basically, biomarkers will response to contaminant exposure by changing their gene transcription, protein syntheses or metabolites secretion (Sany et al. 2013). During the exposure, pollutant's mechanisms of action can changed the specific set of protein synthesis level by disrupt the expression of genes which ultimately lead to alteration of metabolites secretion (Sany et al. 2013). These specific responses from biomarkers toward particular pollutants exposure are important tools that could aids environmental forensics studies in tracing specific pollutants. Thus, the advances in molecular techniques can be employed to control and conserve habitat policies and remediation actions.

16.4 Advances in Estuarine Sediment Research

Sediment has rapidly emerged as the most popular matrix to ascertain the health status of estuaries ecosystem (Bryan and Langston 1992; Sun et al. 2012). Estuaries sediments served as a great sanctuary for benthic organisms. They are capable to absorb metals to concentration many times higher than that in water (Vermeulen and Wepener 1999; Casper et al. 2004; Tabinda et al. 2013). Moreover, they are excellent sinks of persistence pollutants in the aquatic environment (Grant and Middleton 1993; Yarsan and Yipel 2013). They also served as an important medium for various chemical and biological processes (Turner and Millward 2002; Sun et al. 2012). Pollutants binding sediments can be transported to water column, resuspend when sediments are being disturbed, or be absorbed by benthic organisms as they feed

(Sun et al. 2012). In this manner, contaminated estuaries sediments can definitely affect the health status of living organisms and ecosystems.

Since 1990, increasing interest in looking for direct interaction between the concentration of sediment pollutants and detrimental biological effects has urged many scientists to concentrate on developing methods to predict metals distribution, potential negative effects and metals accumulation in estuarine and marine sediments (Nascimento 2007; Sany et al. 2013). Recently, sediment quality has been assessed by conducting toxicity tests on biota, profiling the level of sediment contamination, biomarker assessment and evaluating benthic communities (Nascimento 2007; Sany et al. 2013).

Bioassays have been frequently employed in toxicology study to investigate the toxic effects of on organisms' survival in exposed to various metals concentrations (Sany et al. 2013). In bioassay experiment, selective organisms are exposed to various metals concentrations in order to group the potential toxicity in marine environment (Sany et al. 2013). As a result, sediment quality standards are set up with different chemical benchmarks in order to protect the health status of marine environment (Sany et al. 2013). These standards were then revised to sediment quality guidelines (SQGs), such as, in Hong Kong, Australia/New Zealand, Canada, Netherland, Washington, Norway, Florida and New York (Hübner et al. 2009; Sany et al. 2013). Due to the difficulties in analyzing complex interference on ecosystem health status in regional and temporal extents, SQGs were commonly employed to predict the threat posed by metals contaminants in aquatic environment (Sany et al. 2013). However, by considering the fact that toxicity tests are expensive, tedious and time consuming, many researches have continue to focus only on determining the metals concentration and predicting metals pollution magnitude in aquatic environment (Sany et al. 2013).

Indices have been widely used as special indicators to evaluate the degree of contamination of metals in estuarine and coastal sediments (Sheela et al. 2012; Sany et al. 2013). Few indices that have been continuously employed to assess the metal pollution status in estuarine sediment are: geo-accumulation index, enrichment factor (EF), contamination factor (C_f), contamination degree (C_d), and pollution load index (PLI) (Sany et al. 2013). Among these methods, enrichment factor and contamination factor have become the most popular and simple way to characterize the pollution sources (natural and anthropogenic sources). It is measured based on total resistant elements (Al, Li, Fe, Rb, Sc, Cs). Over the last 10 years, Al and Fe were extensively used as normalizing element to measure the EF (Rubio et al. 2000; Sany et al. 2013).

16.5 Rapid Scientific Advances

Bibliometric analysis done by Sun et al. (2012) revealed that scientific research on estuary pollution has grown since last decade. In the 1950s, developed countries, such as UK and Germany were experienced severe water contamination due to vast population growth and industrialization (Sun et al. 2012). In year 1980s, similar situation was faced by developing countries, such as China and India as a result from industrialization (Wong et al. 2007; Wang et al. 2013), urbanization and economic development (Bowonder 1986; Wu et al. 1999; Sun et al. 2012). Therefore, in 1990s, rules and regulations were implementing worldwide to combat estuaries pollution (Bryan and Langston 1992; Sun et al. 2012). Since then, estuary pollution has received notable worldwide attention in environmental research field. Worldwide scientific paper publications regarding estuary pollution was gradually increased in this decade (Sun et al. 2012). In the report, Sun et al. (2012) revealed that scientific publication on biomarkers marked increment of 1.16 % from 0.24 % (1991–1995) to 1.4 % (2006–2010), suggesting increase of research interest to developed biomarkers for estuaries ecosystem monitoring purposes.

As scientific knowledge continues to advance, implementation of stringent quality assurance (QA) and quality control (QC) has improved the quality of sampling, preservation, storage and analytical procedures. In order to avoid cross contamination in various kind of metal analysis, it is now compulsory to have decontamination steps (acid wash) to make sure all apparatus are free from contamination. Besides, the effectiveness of decontamination steps can be verified by the preparation of equipment blank using ultra-pure (Milli-Q) water. For quality assurance process, it is recommended that minimum triplicate samples must be collected in order to measure the variability come from sampling and analytical procedures. In addition, standard reference materials (SRM) or certified reference materials (CRM) have been widely used to validate and calculate the recovery of analytical procedures. The acceptable range of recovery is (100 % ± 10 %). Furthermore, the sensitivity of method is essentially important in method development process. Calculation of limit of detection (LOD) and limit of quantification (LOQ) are needed to quantify acceptable level of method precision. According to Sanagi et al. (2009), LOD and LOQ are the concentration of the analyte that would yield signal-to-noise ratios of 3 and 10 respectively. Finally, in order to get representative data, the relative standard deviation (RSD) of analytical methods for each run must below 5 %.

16.6 Advances in Analytical and Detection Methods

The determination of metal elements in environmental matrices, such as water, sediment, and biota, is of considerable important. Rapid scientific advances are facilitating the improvement and development of analytical method for metals determination at ultra-trace levels in environmental samples (water, sediment, and biota) (Weiner 2013). Table 16.1 listed extraction and detection methods employed by recent scientific works for metals analyses. The growing impact of scientific knowledge on metal studies has contributed to high sensitivity, precision and recovery of metals analyses (Weiner 2013). Substantial methods have been applied to determine the total metals concentration in environmental matrices, such as flame atomic absorption spectrometry (FAAS), graphite furnace atomic absorption spectrometry (GFAAS), atomic fluorescence spectrometry (AFS), inductively coupled plasma atomic emission spectrometry (ICP-AES), and inductively coupled plasma mass spectrometry (ICP-MS).

Table 16.1 Sample extraction and detection methods applied to estuary water, estuary sediment, and estuary organisms for various metals analysis

Matrices	Elements	Extraction methods	Reagents used	Instrument/ detection	Detection limit	LOD	LOQ	RSD	Reference
Water	Cd, Cu	n.a	n.a	AAS	0.0005 $\mu g\ L^{-1}$	n.a	n.a	n.a	El-Serehy et al. (2012)
	Fe	n.a	n.a	AAS	0.002 $\mu g\ L^{-1}$	n.a	n.a	n.a	El-Serehy et al. (2012)
	Hg	n.a	HNO_3	CV-AFS	1 ng L^{-1}	n.a	n.a	n.a	Cardoso et al. (2013a, b)
	Mn, Pb	n.a	n.a	AAS	0.0005 $\mu g\ L^{-1}$	n.a	n.a	n.a	El-Serehy et al. (2012)
	Zn	n.a	n.a	AAS	0.001 $\mu g\ L^{-1}$	n.a	n.a	n.a	El-Serehy et al. (2012)
Sediment	Ag, Cd, Cr, Cu, Ni, Pb, Sn, Zn	Aqua regia	HCl, HNO_3	ICP-AES	n.a	n.a	n.a	1 %	Aly et al. (2013)
	As, Hg	Aqua regia	HCl, HNO_3	Hydride generation ICP-OES	n.a	n.a	n.a	1 %	Aly et al. (2013)
	Cd, Cu, Mn, Pb	Standard method 3110	H_2O_2, HCl, HNO_3	AAS	0.0005 $\mu g\ L^{-1}$	n.a	n.a	n.a	El-Serehy et al. (2012)
	Cd	n.a	HF-$HClO_4$-HNO_3	Air-acetylene flame AAS	0.01 $\mu g\ g^{-1}$	n.a	n.a	n.a	Abdallah (2013)
	Cd, Cr	n.a	HCl, HNO_3	GFAAS	0.02 mg kg^{-1}	n.a	n.a	n.a	Belabed et al. (2013)
	Co	n.a	HF-$HClO_4$-HNO_6	Air-acetylene flame AAS	0.1 $\mu g\ g^{-1}$	n.a	n.a	n.a	Abdallah (2013)
	Cr	n.a	HF-$HClO_4$-HNO_4	Air-acetylene flame AAS	0.2 $\mu g\ g^{-1}$	n.a	n.a	n.a	Abdallah (2013)

Cu	Aqua regia	HCl, HNO_3	Air-acetylene flame AAS	0.03 mg kg^{-1}	n.a	n.a	n.a	Belabed et al. (2013)
Fe	n.a	HNO_3, $HClO_4$	Air-acetylene flame AAS	0.007 mg L^{-1}	n.a	n.a	n.a	Naji and Ismail (2011)
Fe	n.a	H_2O_2, HCl, HNO_3	AAS	0.002 μg L^{-1}	n.a	n.a	n.a	El-Serehy et al. (2012)
Fe, Mn	Aqua regia	HCl, HNO_3	Air-acetylene flame AAS	0.04 mg kg^{-1}	n.a	n.a	n.a	Belabed et al. (2013)
Hg	n.a	n.a	AAS with gold amalgamation LECO AMA-254 (Advanced Mercury Analyzer)	0.01 ng	n.a	n.a	n.a	Cardoso et al. (2013a)
Hg	n.a	n.a	Direct mercury analyzer (MA 1S)	0.4 ng g^{-1}	n.a	n.a	n.a	Naji and Ismail (2011)
Hg	n.a	HNO_3	CVAAS flow injection mercury/ hydride analyzer	n.a	1.1 ng g^{-1}	3 ng g^{-1}	n.a	Hajeb and Jinap (2011)
Hg	EPA 7473	n.a	n.a	n.a	n.a	0.5 ng g^{-1}	n.a	Maggi et al. (2009)
Hg	Aqua regia	HCl, HNO_3	FIMS 400	0.01 mg kg^{-1}	n.a	n.a	n.a	Belabed et al. (2013)
THg	n.a	n.a	AMA-254	0.007 μg g^{-1}	n.a	n.a	n.a	Abi-Ghanem et al. (2011)
Dissolved CH_3Hg^+	n.a	$NaBH_4$	AFS	20 ng L^{-1}	n.a	n.a	n.a	Abi-Ghanem et al. (2011)

(continued)

Table 16.1 (continued)

Matrices		Elements	Extraction methods	Reagents used	Instrument/ detection	Detection limit	LOD	LOQ	RSD	Reference
		Particulate CH_3Hg^+	n.a	HNO_3, sodium tetrapropyl-borate	GC-ICPMS	0.02 ng g^{-1}	n.a	n.a	n.a	Abi-Ghanem et al. (2011)
		CH_3Hg^+	n.a	HCl, HBr, Toulene, L-Cysteine	Direct mercury analyzer (DMA-80)	n.a	0.4 ng $MeHg^+$	0.6 ng $MeHg^+$	n.a	Maggi et al. (2009)
		Ni	n.a	$HF-HClO_4-HNO_3$	Air-acetylene flame AAS	0.06 µg g^{-1}	n.a	n.a	n.a	Abdallah (2013)
		Ni, Pb	Aqua regia	HCl, HNO_3	Air-acetylene flame AAS	0.06 mg kg^{-1}	n.a	n.a	n.a	Belabed et al. (2013)
		Pb	n.a	$HF-HClO_4-HNO_3$	Air-acetylene flame AAS	0.02 µg g^{-1}	n.a	n.a	n.a	Abdallah (2013)
		Zn	Standard method 3110	H_2O_2, HCl, HNO_3	AAS	0.001 µg L^{-1}	n.a	n.a	n.a	El-Serehy et al. (2012)
		Zn	Aqua regia	HCl, HNO_3	Air-acetylene flame AAS	0.05 mg kg^{-1}	n.a	n.a	n.a	Belabed et al. (2013)
Biota	Bivalves and sponges	Ag, Cd, Cr, Cu, Ni, Sn, Zn, Pb	Aqua regia	HCl, HNO_3	ICP-AES	n.a	n.a	n.a	1 %	Aly et al. (2013)
	Bivalves and sponges	As, Hg	Aqua regia	HCl, HNO_3	Hydride generation ICP-OES	n.a	n.a	n.a	1 %	Aly et al. (2013)
	Bivalves (*D. trunculus*)	Cd, Cu, Mn, Pb	Standard method 3110	HNO_3	AAS	0.0005 µg L^{-1}	n.a	n.a	n.a	El-Serehy et al. (2012)

Mollusca species	Cd, Co, Ni	CEM digestion Application Manuel	HNO_3	Air-acetylene flame AAS	0.02 μg g^{-1}	n.a	n.a	n.a	Abdallah (2013)
Mollusca species	Cr	CEM digestion Application Manuel	HNO_3	Air-acetylene flame AAS	0.015 μg g^{-1}	n.a	n.a	n.a	Abdallah (2013)
Bivalves (*D. trunculus*)	Fe	Standard method 3110	HNO_3	AAS	0.002 μg L^{-1}	n.a	n.a	n.a	El-Serehy et al. (2012)
Suprabenthic peracarida	Hg	n.a	n.a	AAS with gold amalgamation LECO AMA-254 (Advanced Mercury Analyzer)	0.01 ng	n.a	n.a	n.a	Cardoso et al. (2013a)
Fish muscle sample	Hg	n.a	H_2SO_4, H_2O_2	GFAAS	n.a	0.014 mg kg^{-1}	0.047 mg kg^{-1}	n.a	Moraes et al. (2013)
Sand flathead (*Platycephalus bassensis*)	Hg	n.a	V_2O_5, H_2SO_4, HNO_2, H_2O_2	CVAAS	0.1 mg kg^{-1}	n.a	n.a	n.a	Jones et al. (2013)
Sand flathead (*Platycephalus bassensis*)	Hg	n.a	HNO_3, H_2SO_4, $KMnO_4$, $K_2S_2O_8$, $NH_2OH{\cdot}HCl$	CVAFS	n.a	0.02 mg kg^{-1}	n.a	n.a	Jones et al. (2013)
Fishes	Hg	EPA 7473		Direct mercury analyzer, AAS	n.a	n.a	n.a	4 %	Bonsignore et al. (2013)

(continued)

Table 16.1 (continued)

Matrices	Elements	Extraction methods	Reagents used	Instrument/ detection	Detection limit	LOD	LOQ	RSD	Reference
Bay scallop (*Argopecten irradias*)	CH_3Hg^+	Ultrasonic	Cupric chloride, zinc acetate, sodium chloride, hydrochloric acid, $HClO_4$, toulene	GC-μECD micro-electron capture detector	n.a	1.1 ng g^{-1}	n.a	13.70 %	Zhang and Yang (2013)
Bay scallop (*Argopecten irradias*)	$C_2H_5Hg^+$	Ultrasonic	Cupric chloride, zinc acetate, sodium chloride, hydrochloric acid, $HClO_4$, toulene	GC-μECD micro-electron capture detector	n.a	0.65 ng g^{-1}	n.a	14.00 %	Zhang and Yang (2013)
Bay scallop (*Argopecten irradias*)	C_6H_5Hg	Ultrasonic	Cupric chloride, zinc acetate, sodium chloride, HCl, HClO4, toulene	GC-μECD micro-electron capture detector	n.a	0.80 ng g^{-1}	n.a	11.20 %	Zhang and Yang (2013)
Mollusca species	Pb	CEM digestion Application Manuel	HNO_3	Air-acetylene flame AAS	0.05 μg g^{-1}	n.a	n.a	n.a	Abdallah (2013)
Bivalves (*D. trunculus*)	Zn	Standard method 3110	HNO_3	AAS	0.001 μg L^{-1}	n.a	n.a	n.a	El-Serehy et al. (2012)

n.a no data available

Among all this techniques, ICP-MS has been widely used in total metal analysis because of their high sensitivities, strong ability to anti-interference, simultaneously multiple elements detection, and short analysis time (He et al. 2013).

However, the toxicity of metal elements is largely depends on its chemical species (Ramalhosa and Río-Segade 2008). It is now universally accepted that, total metals concentration, either in water, sediment or biota, are insufficient to illustrate their fate in the environment and toxicological risks toward living organisms (Worms et al. 2006; Bayen 2012). As a substitute, it is indispensable that scientist should take chemical speciation into consideration in a view to predict bioavailability and probable toxic effects (Bayen 2012).

To date, biological and chemical testing had been widely used to assess speciation or bioavailability of metals in the environment (Bayen 2012). Bioaccessibility tests, biomimetic devices, and chemical extractability tools are among the chemical testing that been used in environmental analysis (Smith et al. 2010; Bayen 2012). Instead of total metals analysis, the speciation analysis for estuaries and water samples have been carried out by using dissolved or particulate metals (Bayen 2012). For sediment, sequential extraction techniques are the most commonly used chemicals testing to determine metals speciation or bioavailability (Liu et al. 2008; Zhou et al. 2010; Marchand et al. 2011; Yap et al. 2011; Zhou et al. 2011; Bayen 2012). The utilization of other useful tools, such as passive samplers and sensors, has also been applied to predict the bioavailability and speciation of metals in estuaries ecosystems (Bayen 2012).

In addition, methods for speciation analysis have been continuously developed to extract metal species in various environmental matrixes. For instance, the extensive researches on mercury speciation have led to methodology advances in speciation analysis. Various samples treatments have been established to quantitatively release toxic mercury compound, such as methylmercury (MeHg), from solid samples matrix, particularly acid leaching, distillation, alkaline digestion and supercritical fluid extraction (SFE) (Ramalhosa and Río-Segade 2008). Normally, microwave or ultrasound systems were used to facilitate the extraction process, because both irradiation methods can overcome disadvantages (in term of time, extraction efficiency and solvent utilization) of conventional extraction methods (Ramalhosa and Río-Segade 2008). Ramalhosa and Río-Segade (2008) reported that liquid liquid extraction (LLE) and solid phase extraction (SPE) have emerged as the most popular extraction techniques for metal speciation. In addition, several scientific studies also employed solid phase microextraction (SPME) to extract and preconcentrate analytes.

However, all of these extraction methods have their own disadvantages, for example poor extraction efficiency, incomplete derivation, and interference during detection (Ramalhosa and Río-Segade 2008). These drawbacks must strictly control and methodologies are urgently needed for emerging pollutants, to document their distinct complexation and interactions with other metals (Bayen 2012).

Next, the separation and detection techniques for metals speciation have also gradually advanced. Gas chromatography (GC) and high performance liquid chromatography (HPLC) has become the most widely used separation methods for speciation studies (Ramalhosa and Río-Segade 2008). The later one has several advantages over gas chromatography (GC) techniques. HPLC can separate compounds at ambient

Table 16.2 Interface systems for metal speciation analysis

Gas chromatography (GC)	High performance liquid chromatography (HPLC)
Atomic absorption spectrometry (AAS)	Ultraviolet – visible spectrophotometry
Atomic fluorescence spectrometry (AFS)	Amperometry/coulometry
Atomic emission spectrometry (AES)	Cold vapor-atomic absorption spectrometry (CV-AAS)
Microwave induced plasma-atomic emission spectrometry (MIP-AES)	Cold vapor-atomic fluorescence spectrometry (CV-AFS)
Glow discharge – atomic emission spectrometry (GD-AES)	Atomic emission spectrometry (AES)
Plasma emission detection (PED)	Cold vapor-microwave induced plasma emission spectrometry (CV-MIP-AES)
Electron capture detection (ECD)	Mass spectrometry (MS)
Mass spectrometry (MS)	Inductively coupled plasma–mass spectrometry (ICP-MS)
Inductively coupled plasma–mass spectrometry (ICP-MS)	Cold vapor – inductively coupled plasma–mass spectrometry (CV-ICP-MS)

Modified from Ramalhosa and Río-Segade (2008)

temperature without derivation. This in turn shorten the time used for compounds separation and also minimized the potential analytes losses during separation process. Moreover, HPLC has high flexibility: a great variety of stationary and mobile phase can be used to attain excellent separation. It is important to note that before choosing which method to be used, detection limit and linear range of the proposed method must be considered in order to obtain high quality of data.

After separation process, metals compounds can be quantified using variety of available detection techniques (Table 16.2). As instrumental technologies continue to advance in recent decade, machinery allows the quantification of metal species at ultra-trace (ppb and ppt) levels. A novel hyphenated technique, a microfluidic chip-based capillary electrophoresis (μchip-CE) hydride generation (HG) system was coupled with a microwave induced plasma optical emission spectrometer (MIP-OES), was developed to determine arsenic species (Lam et al. 2012). Lately, sensitive colorimetric method utilizing molybdenum blue complex has also been used to determine As(V) (Matusiewicz and Ślachciński 2011). However, according to Ramalhosa and Río-Segade (2008), atomic emission spectrometry (AES) and mass spectrometry (MS) have unique analytical ability to carry out speciation studies, due to their high sensitivity, selectivity, and multi-element detection. ICP-MS instrument is rapidly growing in popularity due to its highly sensitivity and capabilities to detect multi-elements at ultra-trace levels. However, their high running cost makes their application for routine monitoring work becomes difficult.

Due to these disadvantages, new biological techniques have been developed by many researchers to predict and generate better insights on the potential environmental hazards posed by toxic contaminants (Khim and Hong 2013; Luque-Garcia et al. 2013). Bioassay is now being widely used to complement instrumental analysis with the aim to provide more complete characterization of environmental samples (Giesy et al. 2002; Khim and Hong 2013). It can be used to measure the ability of environmental matrices in response to biochemical and physiological changes (Khim and Hong 2013). Integrated chemical analysis and biological methods, followed by instrument analysis, on combine water, sediment and biological samples, could generate better understanding on the behavior and fate of contaminants in the environment (Khim and Hong 2013).

Apart from that, scientific comprehension on biogeochemical processes, fate and transportation, accumulation and negative effects of metals in estuaries systems has continued to advance in this decade (Weiner 2013). Stable isotopes of metals (Fe, Cr, Mo, Hg, Cd, Zn, Pb, Cu) has been applied recently in isotopic signature and tracer studies to identify metals contamination sources, develop isotopic biosignatures, and decipher the fate and transformation process of metals in estuaries and coastal systems (Diefenbacher 2009; Bullen 2011; Weiner 2013).

16.7 Advances in Modeling Approaches

When data on metal toxicity are incomplete, predictive models are beneficial to predict the metals behaviors in estuarine ecosystem. For last few years, high-resolution modeling of contaminant dispersion has been improved. Ramšak et al. (2013) had improved the existing modeling tools (PCFLOW3D and Nafta3D) for pollution transport and dispersion. They refined the numerical grid; replaced spatial and temporal averaged input data with real–time forecasts; used grid nesting to correct the open boundary condition (Ramšak et al. 2013). Besides, Tong et al. (2012) also successfully developed an equivalence-based mass balance model to predict the methylmercury concentration and their thermodynamic behavior in aquatic system. This model is also capable to quantified methylmercury bioaccumulation in the food chain. From the modeling output, they hypothesis that metabolism and growth dilution could be the main mechanism contribute to the reduction of methylmercury concentration in aquatic organisms. They also found that prey is the main sources of methymercury in comparison to water. Therefore, food could be the dominant contribution of methylmercury magnification in food chain (Tong et al. 2012).

16.8 Advances in Statistical Approaches

In addition to modeling approaches, application of statistical methods in accessing estuaries health status continues to advance in this century (Praveena et al. 2011; Lim et al. 2012; Aris et al. 2013). Environmetrics is a new field of study used to analysis, model, interpret and predict environment occurrence (El-Shaarawi and Piegorsch 2013). It is employed to condense large amount of data, to classify probable pollution sources, to identify controlling factors that govern the pollution status and to interpret interrelationship between investigated variables. The outcome from environmetric analyses is especially important in spatial and temporal analyses, remote sensing and modeling process (Sany et al. 2013). It is useful as "end products" or evidence for metals pollution (Sany et al. 2013). Common environmetrics activities include principal component analysis (PCA), cluster analysis (CA), and discriminant analysis (DA). The advantages and disadvantages of environmetric methods are described in Table 16.3.

Table 16.3 Environmetric analysis employed to quantitative assessment for metal pollution in estuaries and coastal systems

Methods	Advantage	Disadvantage	Reference
Descriptive statistic	Provide a summary of central tendency (mean, median, mode) and variability It can be applied to any group of data which describe the quality of estuaries and coastal environment	Summary do not conclude about health status	Hare and Mantua (2000)
Frequency distribution (FD)	FD illustrates the profile of physico-chemical parameters It is a basic scale to portray estuaries health status	Normality pre-condition for probabilistic application is rarely achieved	Giovanardi and Tromellini (1992)
Outliers	Outlying extreme cases in the environment	Non-normally distributed parameters cause limitation of methodology	Karydis (1994)
Analysis of variance (ANOVA)	It compares means and identifies statistically significant difference between variables	It is only suitable for parametric data It assumes all data are normally distributed	Kitsiou and Karydis (2011)
Pearson product-moment correlation analysis	It interprets the relationships and interactions among variables	It required data transformation so that the data are normally distributed	Mustapha et al. (2013)
Hierarchical cluster analysis (CA)	It is used to classify sampling points based upon degree of association between variables It can be applied to non-parametric data	Limited access to significant test. There is different outcome due to various similarity clustering algorithms	Primpas et al. (2008), Aris et al. (2013)
Principal component analysis (PCA)	It extracts multivariate data and reduced to dominant factors that influence the health status It is user friendly and easy to understand the output	It is the preliminary method to evaluate marine environmental quality	Lundberg et al. (2009), Aris et al. (2013)
Discriminant analysis (DA)	It is used to classify variables into categorical dependent values. It is used to evaluate the outputs obtained by cluster analysis	It fails to determine the local geometrical structure of complex data	Cai et al. (2007), Juahir et al. (2010), Aris et al. (2013)

Modified from Sany et al. (2013)

1. Principal Component Analysis (PCA)

PCA is employed to ascertain the extent to which variables are group together (Jonathan et al. 2004; Aris et al. 2013). PCA extracts dominant factors (sources of variation) from original data according to the extent of variability in data set. PCA has turned out to be a favorable tool in determining the probable pollution sources in estuaries and coastal systems.

2. Cluster Analysis (CA)

This analysis enables us to classify various variables (metals concentrations) into groups based on the degree of association between the variables under studied (Aris et al. 2013). The output (dendrogram) is useful to manipulate, interpret, and represent processes that govern estuaries health status (Praveena et al. 2011). Variables that have high degree of similarity were grouped into same cluster while variables between clusters (from different clusters) if of high dissimilarity (Aris et al. 2012, 2013).

3. Discriminant Analysis (DA)

There is a growing interest in the application of DA in environmental analysis. It is usually employ to validate the results obtained by cluster analysis (Aris et al. 2013). It identifies metals that contributed significant impact in a cluster (Al-Odaini et al. 2011). DA identifies variables that differentiate between two or more naturally occurring clusters (Al-Odaini et al. 2011). According to Aris et al. (2013), DA can be applied on raw data based on stepwise mode to construct best discriminant function (DF) for each group in order to assess the spatial variations of metals in the environment.

16.9 Metal Forensics Approaches

Application of forensics investigation in relation to metal pollution crimes is essential to maintain the sustainability of estuaries ecosystem. However, inspection of estuaries metals pollution status in an environmental forensic perspective has had limited effort. To date, only few studies have been integrated metal forensics approaches by combining isotopic fingerprint (Lim et al. 2013), environmentric methods (Aris et al. 2013; Lim et al. 2013; Praveena et al. 2013), geostatistical, geochemical and indices (Lim et al. 2013) to trace and interpret diagnostic data concerning metals pollution status in estuaries health status. Such environmental forensics researches provide better perspectives and robust justifications of the complex underlying factors that governing the fate of metals in estuaries ecosystem.

16.10 Future Perspectives

Good quality of aquatic environment is among the fundamental factors that governing the health and disease of human and organism. Scientific researches in relation to estuary pollution will continue to grow rapidly in future. Holistic metal forensics investigation is of first concern in pollution monitoring programs. Pollution prevention is the main task to be accomplished in order to maintain the health status of estuaries ecosystems. Continuously minimization of anthropogenic metals emissions remains highest and urgent priority for lessens organism exposure to metals contamination. Since metals contamination covered local, regional and international dimensions, considerable advance has to be made by several industrialized countries in minimizing the point sources anthropogenic pollution at all scales. At global dimensions, abatement of trans-boundary metal pollution will require the collaboration at international levels.

16.11 Conclusion

The constant releases of contaminants require comprehensive understanding of metals cycling in estuarine ecosystems to apprise management policy and to assess the effectiveness of remedial actions on metals pollutions. Many studies on chemical, biological and ecological dimensions have been focused on sediment matrixes, considering it as final sink of metals, but not much integrated studies have been carried out to date. However, regardless of all the advances, there are still no measurements that can fully reliable to predict metals concentrations and their effect on estuarine ecosystems, leaving the scientist to encounter too much of uncertainty. Therefore, chemically and biologically integrated monitoring approach that comprise of water, sediment and biological matrixes is critically needed to monitor the effect of anthropogenically derived chemicals to estuaries ecosystems.

References

Abdallah MAM (2013) Bioaccumulation of heavy metals in Mollusca species and assessment of potential risks to human health. Bull Environ Contam Toxicol. doi:10.1007/s00128-013-0959-x

Abi-Ghanem C, Nakhlé K, Khalaf G, Cossa D (2011) Mercury distribution and methylmercury mobility in the sediments of three sites on the Lebanese Coast, Eastern Mediterranean. Arch Environ Contam Toxicol 60:394–405

Ahmed MS, Bibi S (2010) Uptake and bioaccumulation of water-borne lead (Pb) in the fingerlings of a freshwater cyprinid, Catla catla. J Anim Plant Sci 20(3):201–207

Alinnor IJ, Obiji IA (2010) Assessment of trace metal composition in fish samples from Nworie River. Pak J Nutr 9(1):81–85

Al-Odaini NA, Zakaria MP, Zali MA, Juahir H, Yaziz MI, Surif S (2011) Application of chemometrics in understanding the spatial distribution of human pharmaceuticals in surface water. Environ Monit Assess 184(11):6735–674

Aly W, Williams ID, Hudson MD (2013) Metal contamination in water, sediment and biota from a semi-enclosed coastal area. Environ Monit Assess 185(5):3879–3895

Aris AZ, Praveena SM, Abdullah MH, Radojevic M (2012) Statistical approaches and hydrochemical modelling of groundwater system in small tropical island. J Hydroinf 14(1):206–220

Aris AZ, Praveena SM, Isa NM, Lim WY, Juahir H, Yusoff MK, Mustapha A (2013) Application of environmetric methods to surface water quality assessment of Langkawi Geopark (Malaysia). Environ Forensic 14(3):230–239

Bainbridge ZT, Brodie JE, Faithful JW, Sydes DA, Lewis SE (2009) Identifying the land-based sources of suspended sediments, nutrients and pesticides discharged to the Great Barrier Reef from the Tully, ÄìMurray Basin, Queensland. Aust Mar Freshw Res 60(11):1081–1090

Bayen S (2012) Occurrence, bioavailability and toxic effects of trace metals and organic contaminants in mangrove ecosystems: a review. Environ Int 48:84–101

Belabed B-E, Laffray X, Dhib A, Fertouna-Belakhal M, Turki S, Aleya L (2013) Factor contributing to heavy metal accumulation in sediments and in the intertidal mussel *Perna perna* in the Gulf of Annaba (Algeria). Mar Pollut Bull. doi:10.1016/j.marpolbul.2013.06.004

Bonsignore M, Manta DS, Oliveri E, Sprovieri M, Basilone G, Bonanno A, Falco F, Traina A, Mazzola S (2013) Mercury in fishes from Augusta Bay (Southern Italy): risk assessment and health implication. Food Chem Toxicol. doi:10.1016/j.fct.2013.02.025

Bowonder B (1986) Environmental management problems in India. Environ Manag 10(5):599–609

Bryan GW, Langston WJ (1992) Bioavailability, accumulation and effects of heavy-metals in sediments with special reference to United-Kingdom estuaries-a review. Environ Pollut 76(2):89–131

Bullen TD (2011) Stable isotope of transition and post-transition metals as tracers in environmental studies. In: Baskarn M (ed) Handbook of environmental isotope geochemistry; advances in isotope geochemistry. Springer, Berlin, pp 177–203

Cai D, He X, Zhou K, Han JW, Bao HJ (2007) Locality sensitive discriminant analysis. In: Proc 20th IJCAI, pp 708–713

Cardoso PG, D'Ambrosio MD, Marques SC, Azeiteiro UM, Coelho JP, Pereira E (2013a) The effects of mercury on the dynamics of the peracarida community in a temperate coastal lagoon. Mar Pollut Bull. doi:10.1016/j.marpolbul.2013.04.004

Cardoso PG, Sousa E, Matos P, Henriques B, Pereira E, Duarte AC, Pardal MA (2013b) Impact of mercury contamination on the population dynamics of Peringia ulvae (Gastropoda): implication on metal transfer through the trophic web. Estuar Coast Shelf Sci. doi:10.1016/j.ecss.2013.06.002

Casper ST, Mehra A, Farago ME, Gill RA (2004) Contamination of surface soils, river water and sediments by trace metals from copper processing industry in the Churnet River Valley, Staffordshire, UK. Environ Geochem Health 26:59–67

Cevik U, Damla N, Koz B, Kaya S (2008) Radiological characterization around the Afsin-Elbistan coal-fired power plant in Turkey. Energy Fuel 22(1):428–432

Costa MF, Barbosa SCT, Barletta M et al (2009) Seasonal differences in mercury accumulation in *Trichiurus lepturus* (Cutlassfish) in relation to length and weight in a Northeast Brazilian estuary. Environ Sci Pollut Res 16(4):423–430

Deng B, Zhang J, Zhang G, Zhou J (2010) Enhanced anthropogenic heavy metal dispersal from tidal disturbance in the Jiaozhou Bay, North China. Environ Monit Assess 161:349–358

Depledge M, Ameral-Mendes JJ, Daniel B, Halbrook RS, Loepper-Sams P et al (1993) The conceptual basis of the biomarker approach. Biomarker: research and application in the assessment of environmental health. NATO ASI Series H Cell Biol 68:15–29, Springer Verlag, Berlin

Diefenbacher P (2009) Stable isotope signatures of heavy metals as tracers for anthropogenic pollution in the environment. http://www.ibp.ethz.ch/research/aquaticchemistry/teaching/archive_past_lectures/Term_Paper_HS2009/Diefenbacher_ termpaper_HS09.pdf. Accessed 8 Nov 2013

Ding ZH, Liu JL, Li LQ, Lin HN, Wu H, Hu ZZ (2009) Distribution and speciation of mercury in surficial sediments from main mangrove wetlands in China. Mar Pollut Bull 58:1219–1325

Dorne JL, Kass GE, Bordajandi LR, Amzal B, Bertelsen U et al (2011) Human risk assessment of heavy metals: principles and applications. Met Ions Life Sci 8:27–60

Dyer CA (2007) Heavy metals as endocrine disrupting chemicals. Endocrine-disrupting chemicals: from basic research to clinical practice. Humana Press, Totowa, pp 111–133

El-Serehy HA, Aboulela H, Al-Misned F, Kaiser M, Al-Rasheid K, El-Din HE (2012) Heavy metals contamination of a Mediterranean coastal ecosystem, Eastern Nile Delta, Egypt. Tuk J Fish Aquat Sci 12:751–760

El-Shaarawi AH, Piegorsch WW (2013) Encyclopedia of environmetrics. Environ Eng Manag J 12(5):1109–1112

Essl F, Moser D, Dirnbvock T, Dullinger S, Milasowszky N, Winter M, Rabitsch W (2013) Native, alien, endemic, threatened, and extinct species diversity in European countries. Biol Conserv 164:90–97

Fang TH, Li JY, Feng HM, Chen HY (2009) Distribution and contamination of trace metals in surface sediments of the East China Sea. Mar Environ Res 68:178–187

Galloway TS (2006) Biomarkers in environmental and human health risk assessment. Mar Pollut Bull 53(10):606–613

Giesy JP, Hilscherova K, Jones PD, Kannan K, Manchala M (2002) Cell bioassays for detection of aryl hydrocarbon (AhR) and estrogen receptor (ER) mediated activity in environmental samples. Mar Pollut Bull 45:3–16

Giovanardi F, Tromellini E (1992) Statistical assessment of trophic conditions. Application of the OECD methodology to the marine environment. In: Vollenweider RA, Marchetti R, Viciani R (eds) Marine coastal eutrophication. Elsevier, London, pp 211–233

Granero S, Domingo JL (2002) Levels of metals in soils of Alcala' de Henares, Spain: human health risks. Environ Int 28:159–164

Grant A, Middleton M (1993) Trace metals in sediments from the Humber estuary: a statistical analysis of spatial uniformity. Mar Estuarine Grad 27:111–120

Hajeb P, Jinap S (2011) Mercury exposure through fish and seafood consumption in the rural and urban coastal communities of Peninsular Malaysia. World J Fish Mar Sci 3(3):217–226

Hare SR, Mantua NJ (2000) Empirical evidence for North Pacific regime shifts in 1977 and 1989. Prog Oceanogr 47(2–4):103–145

He B, Yub ZJ, Shi JB, Jiang GB (2013) Research progress of heavy metal pollution in China: sources, analytical methods, status, and toxicity. Chin Sci Bull 58:134–140

Hübner R, Astin KB, Herbert RJH (2009) Comparison of sediment quality guidelines (SQGs) for the assessment of metal contamination in marine and estuarine environments. J Environ Monit 11(4):713–722

Ip CC, Li XD, Zhang G, Wong CSC, Zhang WL (2005) Heavy metal and Pb isotopic compositions of aquatic organisms in the Pearl River Estuary, South China. Environ Pollut 138:311–323

Jonathan MP, Ram-Mohan V, Srinivasalu S (2004) Geochemical variations of major and trace elements in recent sediments, off the Gulf of Mannar, the Southeast Coast of India. Environ Geol 45:466–480

Jones HJ, Swadling KM, Tracey SR, Macleod CK (2013) Long term trends of Hg uptake in resident fish from a polluted estuary. Mar Pollut Bull. doi:10.1016/j.marpolbul.2013.04.032

Juahir H, Zain SM, Aris AZ, Yusoff MK, Mokhtar MB (2010) Spatial assessment of Langat River water quality using chemometrics. J Environ Monit 12:287–295

Karydis M (1994) Environmental quality assessment based on the analysis of extreme values: a practical approach for evaluating eutrophication. J Environ Sci Health A 29(4):775–791

Khim JS, Hong S (2013) Assessment of trace pollutants in Korean coastal sediments using the trait approach: a review. Sci Total Environ. doi:10.1016/j.scitotenv.2013.07.052

Kitsiou D, Karydis M (2011) Coastal marine eutrophication assessment: a review on data analysis. Environ Int 37(4):778–801

Lam Y, Sze CW, Tong Y, Ng TB, Pang CS, Zhang Y (2012) A review of the potential issues of pollution caused by the mineral elements, mercury, lead and arsenic, its possible impacts on the human beings and the suggested solutions. Chin Med 3:167–178

Lawrence AL, Mason RP (2001) Factors controlling the bioaccumulation of mercury and methylmercury by the estuarine amphipod *Leptocheirus plumulosus*. Environ Pollut 111(2):217–231

Lim WY, Aris AZ, Praveena SM (2012) Application of the chemometric approach to evaluate the spatial variation of water chemistry and the identification of the sources of pollution in Langat River, Malaysia. Arab J Geosci 6(12):4891–4901

Lim WY, Aris AZ, Tengku Ismail TH (2013) Spatial geochemical distribution and sources of heavy metals in the sediment of Langat River, western Peninsular Malaysia. Environ Forensic 14(2):133–145

Liu JC, Yan CL, Zhang RF, Lu HL, Qin GQ (2008) Speciation changes of cadmium in mangrove (*Kandelia candel* (L))n rhizosphere sediments. Bull Environ Contam Toxicol 80:231–236

Liu JL, Feng XB, Qiu GL, Anderson CWN, Yao H (2012) Prediction of methylmercury uptake by rice plants (*Oryza sativa L*) using diffusive gradient in thin films techniques. Environ Sci Technol 46:11013–11020

Looi LJ, Aris AZ, Wan Johari WL, Yusoff FM, Hashim Z (2013) Baseline metals pollution profile of tropical estuaries and coastal waters of the Straits of Malacca. Mar Pollut Bull 74(1):471–476

Lundberg C, Jakobsson BM, Bonsdorff E (2009) The spreading of eutrophication in the eastern coast of the Gulf of Bothnia, northern Baltic Sea-an analysis in time and space. Estuar Coast Shelf Sci 82(1):152–160

Luque-Garcia JL, Cabezas-Sanchez P, Anunciação DS, Camara C (2013) Analytical and bioanalytical approaches to unravel the selenium-mercury antagonism: a review. Anal Chim Acta. doi:10.1016/j.aca.2013.08.043

Macdonald DD, Ingersoll CG (2002) A guidance manual to support the assessment of contaminated sediments in freshwater ecosystems. Volume I-an ecosystem-based framework for assessing and managing contaminated sediments. USEPA Great Lakes National Program Office, Chicago, EPA-905-B02-001-A

Mackay D, Fraser A (2000) Kenneth Mellanby review award. Bioaccumulation of persistent organic chemicals: mechanisms and models. Environ Pollut 110:375–391

Maggi C, Berducci MT, Bianchi J, Giani M, Campanella L (2009) Methylmercury determination in marine sediment and organisms by direct mercury analyser. Anal Chim Acta 614:32–36

Marchand C, Allenbach M, Lailier-Verges E (2011) Relationships between heavy metals distribution and organic matter cycling in mangrove sediments (conception Bay, New Caledonia). Geoderma 160:444–456

Matusiewicz H, Ślachciński M (2011) Interfacing a microchip-based capillary electrophoresis system with a microwave induced plasma spectrometry for copper speciation. Cent Eur J Chem 9:896–903

Mendil D, Uluözlü ÖD, Hasdemir E, Tüzen M, Sari H, Suiçmez M (2005) Determination of trace metals levels in seven fish species in lakes in Tokat, Turkey. Food Chem 90(1–2):175–179

Milenkovic N, Damjanovic M, Ristic M (2005) Study of heavy metal pollution in sediments from the iron gate (Danube River), Serbia and Montenegro. Pol J Environ Stud 14(6):781–787

Moraes PM, Santos FA, Cavecci B, Padilha CCF, Viera JCS, Roldan PS, Padilha PM (2013) GFAAS determination of mercury in muscle samples of fish from Amazonas, Brazil. Food Chem. doi:10.1016/j.foodchem.2013.05.008

Morillo J, Usero J, Rojas R (2008) Fractionation of metals and as in sediments from a biosphere reserve (Odiel salt marhes) affected by acidic mine drainage. Environ Monit Assess 139(1):329–337

Mushtakova VM, Fomina VA, Rogovin VV (2005) Toxic effect of heavy metals on human blood neutrophils. Biol Bull 32:336–338

Mustapha A, Aris AZ, Juahir H, Ramli MF, Kura NU (2013) River water quality assessment using environmetric techniques: case study of Jakara River Basin. Environ Sci Pollut Res Int 20(8):5630–5655

Naji A, Ismail A (2011) Risk assessment of mercury contamination in surface sediment of the Klang River, Malaysia. Aus J Basic Appl Sci 5(7):215–221

Nascimento IA (2007) Key issues on sediment quality assessment: a review of South America research with emphasis on Brazil. Aquat Ecosyst Health Manag 10:9–22

Pan K, Wang WX (2012a) Reconstructing the biokinetic processes of oysters to counteract the metal challenge: physiological acclimation. Environ Sci Technol 46:10765–10771

Pan K, Wang WX (2012b) Trace metal contamination in estuaries and coastal environments in China. Sci Total Environ 421–422:3–16

Praveena SM, Kwan OW, Aris AZ (2011) Effect of data pretreatment procedures on principal component analysis: a case study for mangrove surface sediment datasets. Environ Monit Assess 184(11):6855–6868

Praveena SM, Siraj SS, Aris AZ, Mustafa Al-Bakri N, Suleiman AK, Zainal AA (2013) Assessment of tidal and anthropogenic impacts on coastal waters by exploratory data analysis: an example from Port Dickson, Straits of Malacca, Malaysia. Environ Forensic 14(2):146–154

Primpas I, Karydis M, Tsirtsis G (2008) Assessment of clustering algorithms in discriminating eutrophic levels in coastal waters. Glob Nest J 10:359–365

Radojević M, Bashkin VN (2006) Practical environmental analysis, 2nd edn. The Royal Society of Chemistry, Cambridge, p 339

Ramalhosa ECD, Río-Segade S (2008) Analytical methods for mercury speciation in several matrixes: a review. Trends Chromatogr 4:43–77

Ramšak V, Malačič V, Ličer M, Kotnik J, Horvat M, Žagar D (2013) High-resolution pollutant dispersion modelling in contaminated coastal sites. Environ Res. doi:10.1016/j.envres.2012.12.013

Rodrigo AP, Costa PM, Costa MH, Caeiro S (2013) Integration of sediment contamination with multi-biomarker responses in a novel potential bioindicator (Sepia officinalis) for risk assessment in impacted estuaries. Ecotoxicology. doi:10.1007/s10646-013-1140-3

Rubio B, Nombela M, Vilas F (2000) Geochemistry of major and trace elements in sediments of the Ria de Vigo (NW Spain): an assessment of metal pollution. Mar Pollut Bull 40(11):968–980

Sanagi MM, Ling SL, Nasir Z, Hermawan D, Ibrahim WA, Abu Naim A (2009) Comparison of signal-to-noise, blank determination, and linear regression methods for the estimation of detection and quantification limits for volatile organic compounds by gas chromatography. J AOAC Int 92(6):1833–1838

Sany SBT, Hashim R, Rezayi M, Salleh A, Safari O (2013) A review of strategies to monitor water and sediment quality for a sustainability assessment of marine environment. Environ Sci Pollut Res. doi:10.1007/s11356-013-2217-5

Schettino T, Caricato R, Calisi A, Giordano M, Lionetto M (2012) Biomarker approach in marine monitoring and assessment: new insights and perspectives. Open Environ Sci 6:20–27

Sheela A, Letha J, Joseph S, Chacko M, Thomas J (2012) Water quality assessment of a tropical coastal lake system using multivariate cluster, principal component and factor analysis. Lakes Reserv Res Manag 17(2):143–159

Sherman LS, Blum JD, Franzblau A, Basu N (2013) New insight into biomarkers of human mercury exposure using naturally occurring mercury stable isotopes. Environ Sci Technol 47:3403–3409

Shokrollahi A, Ghaedi M, Niband MS, Rajabi HR (2008) Selective and sensitive spectrophotometric method for determination of sub-micro-molar amounts of aluminium ion. J Hazard Mat 151(2–3):642–648

Sun J, Wang MH, Ho YS (2012) A historical review and bibliometric analysis of research on estuary pollution. Mar Pollut Bull 64:13–21

Smith BA, Greenberg B, Stephanson GL (2010) Comparison of biological and chemical measures of metal bioavailability in field soils: test of a novel simulated earthworm gut extraction. Chemosphere 81:755–766

Tabinda AB, Bashir S, Yasar A, Hussain M (2013) Metals concentrations in the riverine water, sediments and fishes from river Ravi at Bolloki headworks. J Anim Plant Sci 21(1):76–84

Tong Y, Zhang W, Hu X, Qu L, Hu D, Yang T, Wei W, Wang X (2012) Model description of trophodynamic behaviour of methylmercury in a marine aquatic system. Environ Pollut 166:89–97

Turner A, Millward GE (2002) Suspended particles: their role in estuarine biogeochemical cycles. Estuar Coast Shelf Sci 55(6):857–883

Ünlü S, Topçuoğlu S, Alpar B, Kırbaşoğlu Ç, Yılmaz YZ (2008) Heavy metal pollution in surface sediment and mussel samples in the Gulf of Gemlik. Environ Monit Assess 144:169–178

Vermeulen LA, Wepener V (1999) Spatial and temporal variations of metals in Richards Bay Habour (RHB), South Africa. Mar Poll Bull 39(1):304–307

Wan L, Wang NB, Li QB, Sun B, Zhou ZC, Xue K, Ma ZQ, Tian J, Song L (2008) Distribution of dissolved metals in seawater of Jinzhou Bay, China. Environ Toxicol Chem 27:43–48
Wang SL, Xu XR, Sun YX, Liu JL, Li HB (2013) Heavy metal pollution in coastal areas of South China: a review. Mar Pollut Bull 76:7–15
Weiner JG (2013) Mercury exposure: advances in environmental analysis and ecotoxicology of a highly toxic metal. Environ Toxicol Chem 32:2175–2178
Wong CSC, Wu SC, Duzgoren-Aydin NS, Aydin A, Wong MH (2007) Trace metal contamination of sediments in an e-waste processing village in China. Environ Pollut 145:434–442
Worms I, Simon D, Hassler C, Wilkinson K (2006) Bioavailability of trace metals to aquatic microorganisms: importance of chemical, biological and physical processes on biouptake. Biochimie 88:1721–1731
Wu XY, Yang YF (2011) Heavy metal (Pb, Co, Cd, Cr, Cu, Fe, Mn and Zn) concentrations in harvest-size white shrimp *Litopenaeus vannamei* tissues from aquaculture and wild source. J Food Compos Anal 24:62–65
Wu C, Maurer C, Wang Y et al (1999) Water pollution and human health in China. Environ Health Perspect 107(4):251–256
Yang T, Liu J (2012) Health risk assessment and spatial distribution characteristic on heavy metals pollution of Haihe River basin. Environ Anal Toxicol 2:152. doi:10.4172/2161-0525.1000152
Yap CK, Azmizan AR, Hanif MS (2011) Biomonitoring of trace metals (Fe, Cu, and Ni) in the mangrove area of Peninsular Malaysia using different soft tissues of flat tree oyster *Isognomon alatus*. Water Air Soil Pollut 218:19–36
Yarsan E, Yipel M (2013) The important terms of marine pollution "biomarkers and biomonitoring, bioaccumulation, bioconcentration, biomagnification". J Mol Biomark Diagn S1:003. doi:10.4172/2155-9929.S1-003
Zhang W, Wang WX (2012) Large-scale spatial and interspecies differences in trace elements and stable isotopes in marine wild fish from Chinese waters. J Hazard Mater 215:65–74
Zhang QH, Yang GP (2013) Anatomical mercury speciation in bay scallops by thio-bearing chelating resin concentration and GC-electron capture detector determination. Food Addict Contam A. doi:10.1080/19440049.2013.856037
Zheng N, Liu JS, Wang QC, Liang ZZ (2010) Health risk assessment of metal exposure to street dust in the zinc smelting district, Northeast of China. Sci Total Environ 408:726–733
Zhou Y, Peng Y, Li X, Chen G (2011) Accumulation and partitioning of heavy metals in mangrove rhizophere sediments. Environ Earth Sci 64:799–807
Zhou YW, Zhao B, Peng YS, Chen GZ (2010) Influence of mangrove restoration on heavy metal accumulation and speciation in intertidal sediments. Mar Pollut Bull 60:1319–1324

Chapter 17
Floating Offshore Wind Farms and Their Application in Galicia (NW Spain)

Laura Castro-Santos and Vicente Diaz-Casas

Abstract The main objective of this chapter is to explain a methodology for the calculation of the life-cycle costs of developing a floating offshore wind farm. In this context, two steps will be considered: defining the life-cycle phases of a floating offshore wind farm (conception and definition, design and development, manufacturing, installation, exploitation, and dismantling) and estimating costs for each phase, whose value will be established as constant or dependent on the particular point of the geography. The result of the tool will be a geo-referenced map of the total cost implied in the development of a floating offshore wind farm. The methodology will be applied to a particular location, Galicia (NW of Spain), where offshore depth and wind resource satisfy the necessary conditions for this type of offshore energy installation. The study will be developed for a semisubmersible floating offshore wind platform and a general offshore wind turbine of 5 MW. The farm will be composed of 21 offshore wind turbines, with a total power of 107 MW. Results show the cost maps of each defined phase and the total cost, which is from 366 to 946 M€ depending on the location (deep, distance from shore, among others), of the life-cycle of a floating offshore wind farm.

17.1 Introduction

17.1.1 Objectives

This chapter aims to explain a methodology for the calculation of the life-cycle costs of developing a floating offshore wind farm. In this context, two steps will be considered: defining the life-cycle phases of a floating offshore wind farm

L. Castro-Santos (✉) • V. Diaz-Casas
Department of Naval and Oceanic Engineering, Integrated Group for Engineering Research (GII), University of A Coruña, C/Mendizábal, 15403 Ferrol, A Coruña, Spain
e-mail: laura.castro.santos@udc.es

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8,
DOI 10.1007/978-3-319-06305-8_17,

(conception and definition, design and development, manufacturing, installation, exploitation, and dismantling) and estimating costs for each phase, whose value will be established as constant or dependent on the particular point of the geography. The result of the tool will be a geo-referenced map of the total cost implied in the development of a floating offshore wind farm. Furthermore, the method will be applied to a particular location, Galicia (NW of Spain), where offshore depth and wind resource satisfy the necessary conditions for this type of offshore energy installation. The study will be developed for a semisubmersible floating offshore wind platform and a general offshore wind turbine of 5 MW. The farm will be composed of 21 offshore wind turbines, with a total power of 107 MW.

17.1.2 Renewable Energies

The electric market in Spain is composed of the 29.7 % of renewable energies, of which 15 % is onshore wind energy and 11 % hydraulic energy (Instituto para la Diversificación y el Ahorro de la Energía (IDAE) 2012). In addition, Galicia (North-West of Spain) has 27 % of renewable energies, of which 50.8 % is onshore wind energy and 24 % hydraulic energy (Instituto Enerxético de Galicia (INEGA) 2010).

However, the European Union (UE) has established one objective for the electric production: the 40 % should be obtained from renewable energies (Official Journal of the European Union 2009). This law has been transposed in Spain using the PER 2011–2020 (Renewable Energy Plan) (Ministerio de Industria Turismo y Comercio 2009).

Nowadays, Spain produces 30 % of electricity from renewable energy re-sources, but this value should rise by 40 %. Thus, the country will increase its electricity production from renewable energy resources.

Figure 17.1 shows that most of the onshore renewable energies have their offshore analogous.

Spain stands in fourth position in the world with 9.1 % of the installed power resulting from onshore wind energy, only overcome by China, United States and Ger-many (Global Wind Energy Council (GWEC) 2012). Therefore, this country has wide experience in wind energy. This is the main reason why it is an optimum area to develop complementary energy resources, as offshore wind power. It is due to the fact that the best wind areas in terms of resource located onshore are occupied by the oldest onshore wind farms (Maciel 2010). On the other hand, of all the offshore renewable energies, which can be seen in Fig. 17.1, the most developed is the offshore wind. Finally, Galicia is one of the best offshore wind areas in terms of wind resource. In fact, there are some areas where wind speed exceeds 10 m/s (Instituto para la Diversificación y el Ahorro de la Energía (IDAE) 2012).

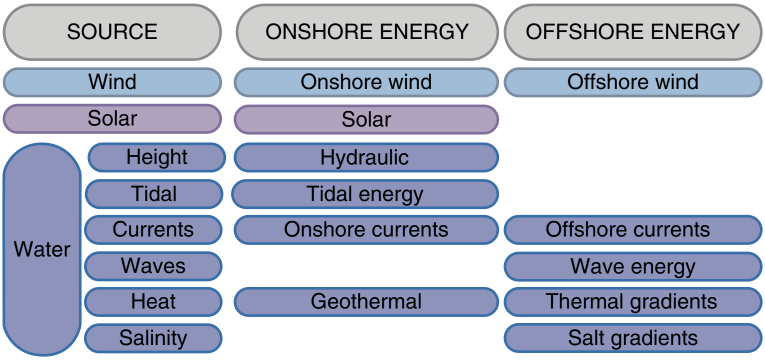


Fig. 17.1 Types of onshore and offshore renewable energies

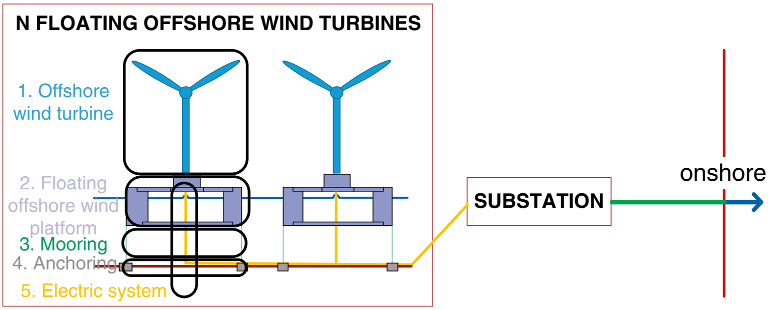


Fig. 17.2 Main components of a typical floating offshore wind farm

17.2 Main Components of a Floating Offshore Wind Farm

A typical floating offshore wind farm is composed of five main components (Fig. 17.2): offshore wind turbine, floating offshore wind platform, mooring, anchoring and electric system.

Firstly, there are two types of offshore wind turbines, depending on the position of their axis: vertical and horizontal (Vita et al. 2010). However, the most common is the horizontal axis wind turbine.

Regarding the floating offshore wind platform, there are many types of devices, but the three most important ones are the following (Jonkman et al. 2009), Fig. 17.3: semisubmersible, Tensioned Leg Platform (TLP) and spar. The semisubmersible platform is composed of three steel columns which helps the structure to keep its stability (ECN et al. 2002). The TLP is a platform ballasted and moored by sever-al vertical tendons in tension. Each tendon attaches to an horizontal spoke from the bottom of the substructure (Jonkman and Matha 2009). The spar platform is a slender spar buoy with several mooring lines. Nowadays, two floating prototypes have already been installed (European Wind Energy Association (EWEA) 2012): the

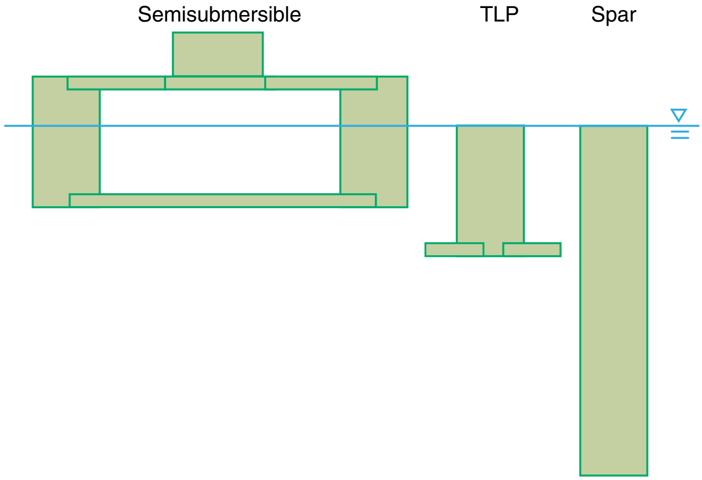


Fig. 17.3 Types of floating offshore wind platforms: semisubmersible, TLP and spar

Hywind spar substructure, which has been installed closed to Bergen (Nor-way), and the WindFloat semisubmersible platform, located in Aguçadoura (Portugal).

Referring to the mooring, three types of lines can be used (Tong 1998): chain, cable and synthetic fiber. On the other hand, four types of anchors can be used (American Petroleum Institute (API) 1996; Chakrabarti 2005): drag embedment anchor, suction pile, gravity anchor and plate anchor.

Finally, the electric system is composed of two main components (Kaiser and Snyder 2010): the substation, which can be installed onshore or offshore, and the electric cables, whose main characteristics differs depending on the location of the electric cable: onshore or offshore.

17.3 Floating Offshore Wind Energy Costs

Nowadays, one of the most important issues is the fact that there are no floating offshore wind farms in operation in the world. This is the main reason why its costs are yet unknown. However, if the floating offshore wind industry wants to start its development, it will be absolutely essential that entrepreneurs become aware of the main costs of this technology, in order to reduce them in the future. In this sense, this chapter will analyse the main costs of a floating offshore wind farm taking into account its life-cycle (Castro-Santos et al. 2013a).

In this context, six phases can be defined in the life-cycle of a floating offshore wind farm: conception and definition, design and development, manufacturing, installation, exploitation and dismantling. Each of them will be associated with a particular cost of the life-cycle as Figs. 17.4 and 17.5 shows.

In addition, each of these phases are made up of several sub-phases, composing the cost breakdown structure of a floating offshore wind farm. For instance, the

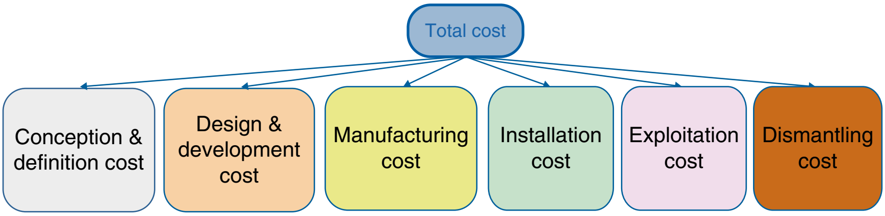


Fig. 17.4 Phases of the life-cycle of a typical floating offshore wind farm: conception and definition, design and development, manufacturing, installation, exploitation and dismantling

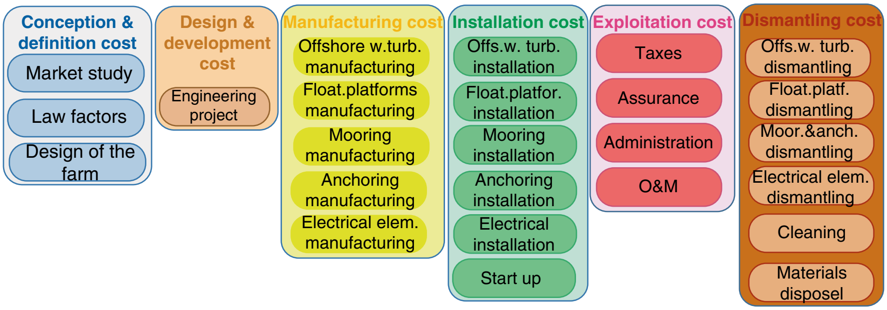


Fig. 17.5 Sub-phases of the life-cycle of a typical floating offshore wind farm

conception and definition phase will be based on the market study, legal factors, and farm design, while the design and development phase concerns project engineering. Otherwise, there are other phases whose components are focused on the five main components of the floating offshore wind farm cited previously (wind turbine, floating platform, mooring, anchoring and electric components). There-fore, the manufacturing, installation, exploitation, and dismantling phases will be disaggregated in order to consider each of these devices. Furthermore, the dismantling phase will also include clearing the geographical area where the floating off-shore farm is located and the decommissioning of all the elements that compose the farm.

The total cost will depend on, among others (Castro-Santos and Diaz-Casas 2013): the number of wind turbines (*NWT*), the power of each offshore wind turbine (*PWT*), the mass of the platform (m_p) (ECN et al. 2002), the diameter of the wind turbine (D_{wt}), the cost per MW of each wind turbine (C_{MW}), the probability of failure of each component of the floating wind farm ($P_{failure}$), the period (T_w) and height (H_w) of the waves (Cerda Salzmann 2004; Alari and Raudsepp 2012), the shape (k_w) and scale (c_w) parameter of the wind (Hunt 2009), the depth (D), the distance to shore (d), the distance from the farm to the storage area ($d_{storage}$) and the distance from the farm to the platform construction area ($d_{construction}$) (The Crown Estate 2009). In addition, the results of a previous technical study on electric cables, anchoring, and mooring dimensioning will support this cost estimation of a floating offshore wind farm. It will help to determine the failure probability of the mooring and anchoring systems, the weight of mooring and anchoring, and the length and section of the electrical cables (Castro-Santos et al. 2013b) (Fig. 17.6).

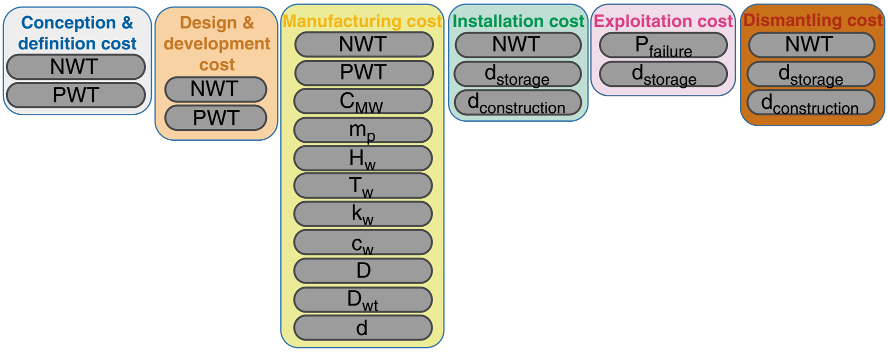


Fig. 17.6 Main dependences referring to the life-cycle costs of a floating offshore wind farm

In this context, the total Life-cycle Cost System (LCS) of a Floating Offshore Wind Farm (FOWF) will be formulated taking into account a geo-referenced map for each k point of the geography. Equation 17.1 shows that the conception and definition cost ($C_c\&_d$) and the design of development cost ($C_d\&_d$) are independent of the geography considered, as results will show for the particular case of Galicia. However, the other costs will depend on the location point considered (k): manufacturing cost ($C_m(k)$), installation cost ($C_i(k)$), exploitation cost ($C_e(k)$) and dismantling cost ($C_d(k)$). Obviously, they will directly be dependent on the distance to shore, regarding the fleet transport of the specialized vessels needed for installing, exploiting and dismantling the floating offshore wind farm, and the wind and wave resources.

$$LCS_{FOWF}(k) = C_{c\&d} + C_{d\&d} + C_m(k) + C_i(k) + C_e(k) + C_d(k) \tag{17.1}$$

All these costs are calculated for each point of the geography, obtaining a geo-referenced map of the total cost of developing a floating offshore wind farm. This method was carried out using MatlabTM software, which introduces the input variable matrices, the values of which depend on the study region's settings (wave period, height of waves, shape and scale wind parameters, depth, distance to shore, distance from farm to the storage area and distance from farm to the platform construction area) and transforms them into vectors (Fig. 17.7). Consequently, the software calculates the cost of each phase of the wind farm's life-cycle for each of the points involved. Thus, the total cost of the life-cycle of a floating offshore wind farm can be calculated for the selected floating platform.

17.4 Case of Study: Galicia (NW Spain)

The area selected to develop a floating offshore wind farm should be located in deep waters, where the floating offshore wind platforms can be installed. In this sense, the region chosen has been Galicia, located in the North-West of Spain, and whose

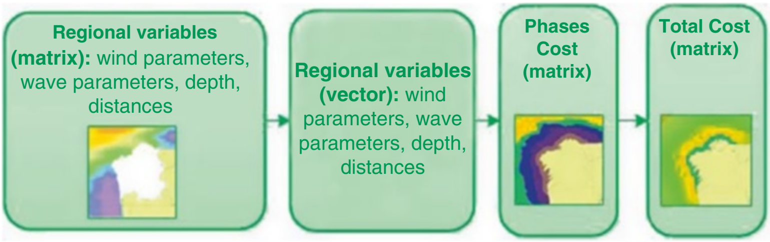


Fig. 17.7 The main components of the method proposed are: regional variables and phases cost matrix. They will calculate the total cost matrix for each point of a particular geography

Fig. 17.8 Location of Galicia in the North-West of the Iberian Peninsula (Google 2013)

offshore bathymetric characteristics are appropriate. In addition, this area has a great offshore wind resource whose exploitation can generate a great deal of electricity; a developed naval sector, which can build the floating offshore wind platforms; and several ports with appropriate characteristics for the specific installation and maintenance vessels (Fig. 17.8).

On the other hand, the selection of one of the three most common floating wind platforms previously defined will be considered. The present paper has taken into account a semisubmersible platform, because its installation process is well-known since the Portuguese WindFloat experience occurred in 2011 (Roddier and Cermelli 2009).

The method has been developed for a floating offshore wind farm composed of 21 offshore wind turbines (Repower 5 M) of 5 MW of power, it has a diameter of 126 m and 90 m of hub height. Therefore, the total power of the floating offshore wind farm is 107 MW. Otherwise, there will be 21 floating offshore semisubmersible platforms with three columns of 8 m of diameter, six mooring lines, 76 m of

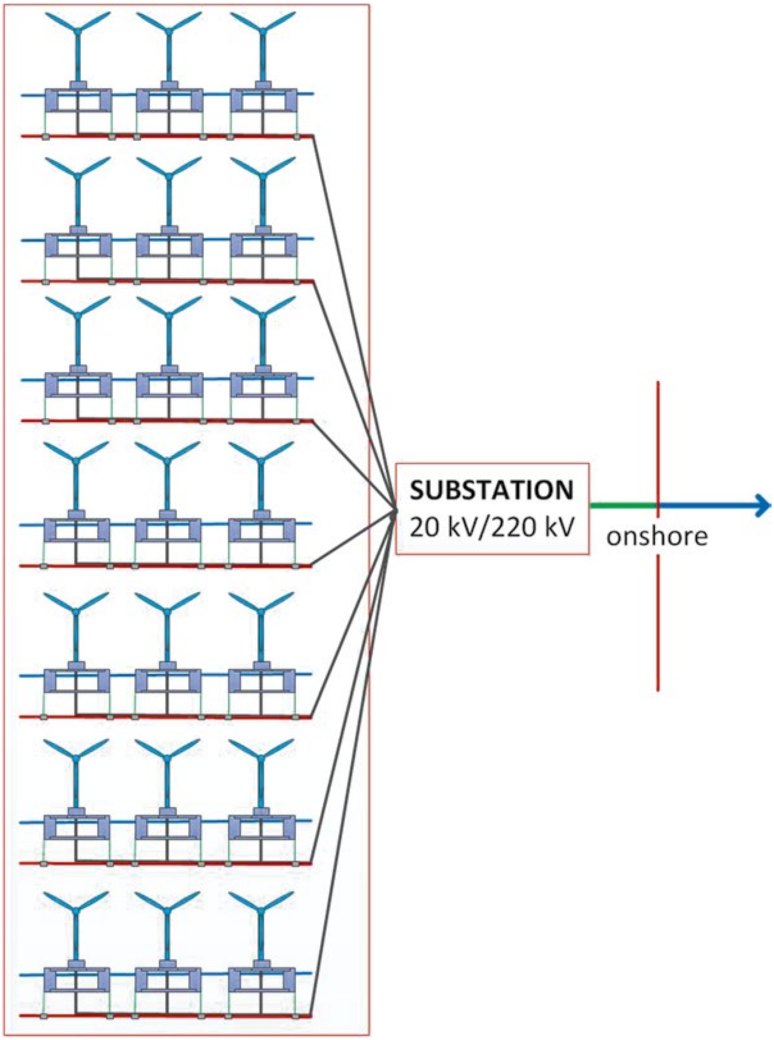


Fig. 17.9 General schema of a floating offshore wind farm composed of 21 wind turbines

length and 12 m of draft, following the Dutch Tri-Floater device (ECN et al. 2002). In addition, the substation will have a relation 20 kV/220 kV. The general schema of the planned floating offshore wind farm is shown in Fig. 17.9.

17.5 Results

The costs of the phases of conception and definition and the design and development are independent of the geographic point considered, their values being 6.79 M€ and 0.24 M€ respectively. However, the costs associated to the other phases of the

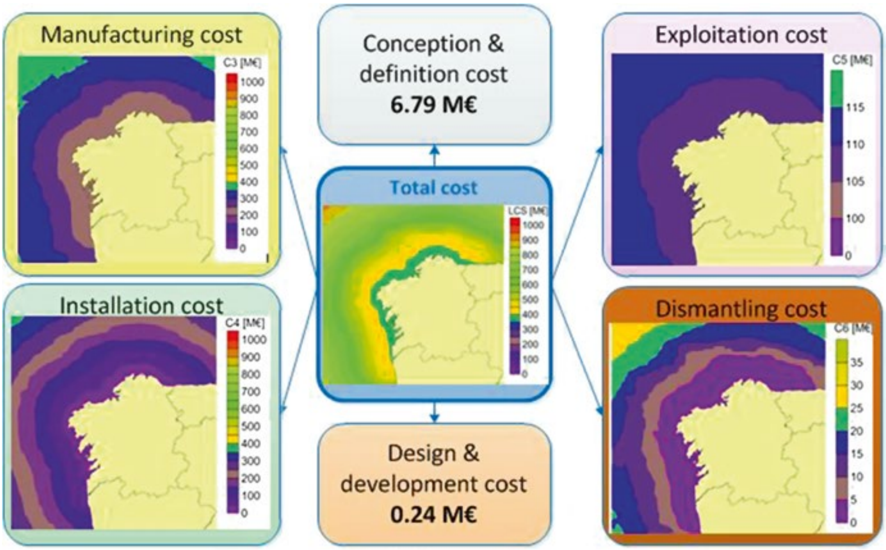


Fig. 17.10 Total life-cycle costs of a floating offshore wind farm depending on the location and for the particular case of the Galician region (NW of Spain)

life-cycle of a floating offshore wind farm: manufacturing, installation, exploitation and dismantling, depend on the depth of the location and the distance to the shore. This is the main reason why there is a particular map for the cost of each phase. In this sense, the cost of manufacturing all the components of a floating offshore wind farm (offshore wind turbine, floating offshore wind plat-form, mooring, anchoring and electric system) ranges from 215 to 405 M€ de-pending on the distance to the Galician shore. In addition, the installation cost ranges from 19 to 392 M€, exploitation cost ranges from 108 to 114 M€ and dismantling cost varies from 0.0058 to 30.87 M€. The value of exploitation is basically composed of the costs of operation and maintenance, and as expected for an earlier stage, it does not change considerably with the number of trips made by maintenance vessels. Furthermore, the total life-cycle cost, which includes the sum of the cost of each phase, ranges from 366 to 946 M€. Figure shows how the cost of installation increases in a different way from that of the manufacturing: the deeper the sea, the more expensive the installation (Fig. 17.10).

17.6 Conclusions

This chapter has explained the main components of the offshore wind energy sector. In addition, the differences between fixed and floating offshore wind farm have been established in terms of depth and platforms.

On the other hand, this study has proposed a method to calculate the cost of a floating offshore wind farm, which is one of the most important issues for investors. Therefore, these costs have been calculated for each point of the geography, which helps to develop economic maps for a particular geographical region. Furthermore, this methodology has been developed for the particular area of Galicia (North-West of Spain), where depth and offshore wind resource offer the best conditions to install a floating offshore wind farm.

References

Alari V, Raudsepp U (2012) Simulation of wave damping near coast due to offshore wind farms. J Coast Res 28(1):143–148. doi:10.2112/JCOASTRES-D-10-00054.1

American Petroleum Institute (API) (1996) API recommended practice 2SK. Recommended practice for design and analysis of stationkeeping systems for floating structures

Castro-Santos L, Diaz-Casas V (2013) Cost comparison of three floating offshore wind platforms. J Coast Res, pp 1–14

Castro-Santos L, Prado G, Costa P et al (2013a) Methodology to design an economic and strategic offshore wind energy roadmap in Portugal. In: SINTEF (ed) 10th deep sea wind R&D conference. Trondheim, Norway, pp 168–176

Castro-Santos L, Prado G, Costa P (2013b) Methodology to study the life cycle cost of floating offshore wind farms. In: 10th deep sea wind R&D conference. Trondheim, Norway, pp 179–186

Cerda Salzmann DJ (2004) Ampelmann, development of the access system for offshore wind turbines. TU Delft University. ISBN: 9789088911941

Chakrabarti SK (2005) Handbook of offshore engineering. Elsevier Ocean Engineering. ISBN: 978-0080443812

ECN, MARIN, Lagerwey the Windmaster et al (2002) Study to feasibility of boundary conditions for floating offshore wind turbines

European Wind Energy Association (EWEA) (2012) The European offshore wind industry key 2011 trends and statistics

Global Wind Energy Council (GWEC) (2012) Global wind statistics 2010

Google (2013) Google maps. Retrieved 11 Jun 2013, from https://maps.google.com

Hunt GL (2009) Maine offshore wind energy. Wind resources, technologies and energy production. Maine

Instituto Enerxético de Galicia (INEGA) (2010) Balance enerxético de Galicia 2010

Instituto para la Diversificación y el Ahorro de la Energía (IDAE) (2012) Web Atlas Eólico de España. Ministerio de Industria Turismo y Comercio. Retrieved 10 Dec 2012, from http://atlaseolico.idae.es/

Jonkman J, Matha D (2009) A quantitative comparison of the responses of three floating platforms. In: European offshore wind 2009 conference and exhibition. Stockholm, pp 1–21

Jonkman J, Butterfield S, Musial W et al (2009) Definition of a 5-MW reference wind turbine for offshore system development

Kaiser MJ, Snyder B (2010) Offshore wind energy installation and decommissioning cost estimation in the U. S. outer continental shelf. Louisiana

Maciel JG (2010) The WindFloat Project

Ministerio de Industria Turismo y Comercio (2009) Resumen del Plan de Energías Renovables

Official Journal of the European Union (2009) Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC

Roddier D, Cermelli C (2009) Windfloat: a floating foundation for offshore wind turbines. Part I: design basis and qualification process. In: ASME 28th international conference on ocean, offshore and arctic engineering (OMAE2009). Honolulu, pp 1–9
The Crown Estate (2009) A guide to an offshore wind farm
Tong KC (1998) Technical and economic aspects of a floating offshore wind farm. J Wind Eng Ind Aerodyn 76:1–12
Vita L, Paulsen US, Pedersen TF (2010) A novel floating offshore wind turbine concept. New developments, pp 1–10

Index

C.W. Finkl and C. Makowski (eds.), *Environmental Management and Governance: Advances in Coastal and Marine Resources*, Coastal Research Library 8,
DOI 10.1007/978-3-319-06305-8, © Springer International Publishing Switzerland 2015