Complex coastal systems

transdisciplinary learning on international case studies

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COMPLEX COASTAL SYSTEMS
TRANSDISCIPLINARY LEARNING ON INTERNATIONAL CASE STUDIES

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Preface

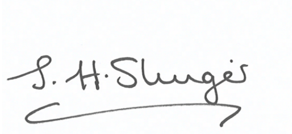
The project Co-designing Coasts using natural Channel-shoal dynamics (CoCoChannel), funded by the Dutch Research Council (NWO), commenced in 2015 with the broad aim of increasing knowledge on the nested scale behaviour of inlet and estuary coasts. The Texel inlet on the Dutch Wadden Sea coast was selected as the central case study of the project with two of the three sub-projects focussing here. The other sub-project, Multi-actor Systems - Co-Designing Nature-based interventions in Coastal Systems, under the leadership of Delft University of Technology, undertook the development of a co-design approach on Texel, but also initiated an international cross-comparative study to anchor the knowledge development within international experience.

Seven case studies located in South Africa, Sri Lanka, California, Suriname, Ireland and the Netherlands (2x) form the objects of inquiry. The case studies, focussing on tidal inlet or estuary mouth management issues, were selected to provide learning on the biophysical and the social systems. For this reason each of the authors invited to contribute a chapter and engage in a week-long workshop was deeply familiar with their specific case study. The workshop, convened in September 2017, was designed to facilitate transdisciplinary learning through consecutive divergent and convergent knowledge exchange phases. This book documents the learning from this international cross-comparative component of the CoCoChannel project.

This book is intended for:

- Transdisciplinary scholars who are interested in interdisciplinary learning and knowledge exchange,
- Policy analysts, environmental historians and coastal policy specialists who are interested in the role of science in the evolution of coastal policy and management,
- Coastal scientists and engineers interested in the dynamics of tidal inlets and estuary mouths,
- Coastal managers looking to learn about tidal inlet and mouth management practices,
- Educators focussed on interdisciplinary skills or interested in using the case studies in coastal, management and engineering classes or as the basis for problem structuring exercises by policy students, and
- Students interested in coastal systems management and wanting to broaden their interdisciplinary competence.

Enjoy learning from the reflective experience of the scientists involved in this transdisciplinary learning endeavour!



Authors

The authors contributing to this transdisciplinary endeavour draw upon a broad spectrum of scientific backgrounds ranging from engineering, through the biophysical sciences to the policy sciences. They include (in alphabetical order): Janine Adams (Nelson Mandela University), Dane Behrens (Environmental Science Associates), David Dann (University of California, Davis), Trang Minh Duong (IHE Delft, Netherlands), Filipe Galiforni Silva (University of Twente, Netherlands), Kate Hewett (University of California, Davis), Floortje d'Hont (Delft University of Technology, Netherlands), Piet Huizinga (formerly CSIR, South Africa), Michael Koohafkan (California Department of Water Resources), Stephen Lamberth (Department of Environment, Forestry and Fisheries, South Africa), John Largier (University of California Davis, USA), Suzanne Linnane (Dundalk Institute of Technology, Ireland), Declan MacGabhann (), Priscilla Miranda (Staatsbosbeheer, Suriname), Jan Mulder (University of Twente, Netherlands), Rosh Ranasinghe (IHE Delft, Netherlands), Matt Robart (University of California, Davis), Robin Roettger (University of California, Davis), Alec Rolston (Dundalk Institute of Technology, Ireland), Jill Slinger (Delft University of Technology, Netherlands), Susan Taljaard (CSIR, South Africa), Ad van der Spek (Deltares, Netherlands), Mick van der Wegen (IHE Delft, Netherlands), Lara van Niekerk (CSIR, South Africa), and Kathelijne Wijnberg (University of Twente, Netherlands).

Editors

Prof. dr. Jill Slinger

Prof. dr. Jill Slinger is an interdisciplinary specialist focussing on the contribution of inclusive, participatory processes in the environmental management of water and coastal systems. She holds a faculty appointment at Delft University of Technology and an honorary visiting professorship at Rhodes University, South Africa. She currently serves on the Netherlands Centre for River Studies Programme Committee and the Coastal Working Group of the Dutch Expertise Network for Flooding Safety. Dr. Slinger has published in water, coastal, environmental science, policy, planning and engineering journals. She has undertaken research on the southern African coast, the Incomati and Volta Deltas, the Scheldt Estuary, the Rhine River and the North Sea coasts of the Netherlands, England, Scotland, Belgium, Germany and Denmark. She has an ongoing involvement in pan-European and African research projects related to environmental systems management, participatory planning and design. Her current research activities focus on using ecosystem-based design knowledge to support the interactions between people, science and policy in river and coastal systems – an inclusive co-design approach to planning and management.

Dr. Susan Taljaard

Susan Taljaard is a marine, coastal and estuarine researcher in the Coastal Systems Research Group of the CSIR, South Africa. Her research has been shaped by the CSIR's mandate to foster science that contributes to the quality of life of South Africa's people, in collaboration with the private and public sectors. She has developed water quality guidelines and operational policies for marine disposal of land-derived wastewater, both nationally and internationally through the Benguela Current Large Marine Ecosystem Programme, as well as the Western Indian Ocean Land-based Activities Programme. In 2014, she was project leader of the CSIR's team that assisted the national environmental authority with the development of South Africa's first National Coastal Management Programme under the Integrated Coastal Management Act. Her research outputs focus on the implementation of environmental policies, protocols and guidelines for sustainable growth and development. Her recent appointment as Adjunct Professor at the Institute for Coastal and Marine Research, Nelson Mandela University, offers her the opportunity to share her transdisciplinary insights and learn from the next generation of marine and coastal scientists.

ir. Floortje d'Hont

Floortje d'Hont is in the final year of her PhD in the Policy Analysis section of the Faculty of Technology, Policy and Management of Delft University of Technology. She holds a MSc degree in Systems Engineering, Policy Analysis and Management from the same faculty. Her research is funded by the CoCoChannel (Co-designing Coasts using natural Channel-shoal dynamics) project and focuses on design-oriented collaborative activities that promote innovative solutions for coastal systems. She has drawn on her teaching and student supervision experience to design, analyze and report on stakeholder engagement and expert workshops. She is particularly interested in methods to support creative collaboration between citizens, experts, scientists, and governmental actors.

ir. Aashna Mittal

Aashna Mittal is a graduate from TU Delft with a Master's in Engineering and Policy Analysis cum laude. She has an academic background covering diverse domains such as engineering, liberal arts, and policy analysis. Her Master's research focussed on the potential of a community-based approach to groundwater management in peri-urban areas of India. Throughout her Master's, she was actively involved in teaching activities at the faculty of Technology, Policy, and Management at TU Delft. Continuing this interest forward, she is currently supporting the development of a new MOOC called Beyond Engineering: Building with Nature, and the editorial work of this book.

Reviewers

The reviewers attended the final public presentations (on 28th September 2017) that concluded the week-long workshop. They did not participate in the preceding discussions and transdisciplinary learning on which the final presentations were based. As such, they were familiar with the workshop outcomes, and through this and their individual experience in the fields of policy analysis and environmental management, were well positioned to review the methods, system understanding and learning that occurred through the transdisciplinary engagement of scientists with a range of international case studies.

Prof. (em.) dr. ir. Wil Thissen

Wil Thissen is emeritus Professor of Policy Analysis at the Faculty of Technology, Policy and Management of Delft University of Technology, where he pioneered the development and realisation of the teaching and research program in Systems Engineering, Policy Analysis and Management. He has served as an editorial board member of Technological Forecasting and Social Change, Impact Assessment and Project Appraisal, and The Environmental Impact Assessment Review, and has authored many scientific papers and books. His research interests are in developing and testing concepts and methods for supporting strategic policymaking in multi-actor environments, with particular emphasis on applications in the fields of infrastructure, energy policy and environmental and water management. Recently, he has developed a strong focus on working in the water and environmental management field in partnership with actors in developing countries, including Bangladesh, India, Indonesia, Rwanda, Suriname and Senegal.

Dr. ir. Heleen Vreugdenhil

Heleen Vreugdenhil is an innovation specialist in the Marine and Coastal Systems Department of Deltares in the Netherlands. Her PhD from the Delft University of Technology focussed on pilot projects in water management, analyzing their effects on changing management practices along the Rhine River in Switzerland, Germany and the Netherlands. After a post-doctoral period at the University of Maastricht in which she analyzed stakeholder engagement processes from national to local level, she joined Deltares where she currently employs her systems analysis skills to engage in the public domain and advance nature-based, participatory planning approaches. She complements these environmental management and innovation activities by a strong reflective focus through her part-time employment in the Policy Analysis Section of Delft University of Technology.

Acknowledgements

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Contents

1

Introduction

By Jill Slinger, Susan Taljaard and Floortje d'Hont

1.1. Background

This book captures the learning from a cross-comparison of seven international inlet or estuary mouth management situations. The conceptual framing is provided by a focus on systems knowledge and its development and use within coastal management. Systems and systems knowledge have been described as holistic, embodied ways of conceptualising reality, forming "both a way of inquiry and an object of inquiry" (Nelson, 2008). To date there has been little research focussing on the role of systems approaches in informing coastal management despite the early development of systems thinking (late 1950's onwards) (Ison et al., 1997), the general acceptance of the adaptive learning cycle of integrated coastal management (Group of Experts on the Scientific Aspects of Marine Environmental Protection [GESAMP], 1996; Olsen et al., 1999), and ongoing engineering infrastructural and urban development along our coasts. Recently, Reis et al. (2014) undertook a study on systems approaches for implementing integrated coastal management principles in Europe, concluding that there is evidence that systems approaches provide a significant step in advancing multidisciplinary sustainability science. Accordingly, this study adopted a systems approach (the way of inquiry) in seeking to learn across a diversity of case studies (the objects of inquiry), each exhibiting complex bio-geophysical and social dynamics on multiple, nested spatial scales and time horizons. In particular, an international cross-comparison was undertaken to garner knowledge on the role of system understanding in designing and managing nature-based interventions (Slinger, 2016; Waterman, 2010) in a range of inlet and estuary systems. Here, the interventions are regarded as the product of the involved network of scientists, engineers and other stakeholders within the case studies, and their social dynamics over time. In this sense the interventions are knowledgeable actions (Ison, 2008).

Specifically, a transdisciplinary systems approach is reported, in which the linkages between the social, economic and biophysical (ecological) aspects that are studied in the coastal environment form the focus of inquiry, as well as the use of a range of different knowledge types (see Max-Neef, 2005). By explicitly recognizing different types of knowledge, such as model-based knowledge, technical design knowledge, and local community knowledge, new and deeply relevant insights for coastal management in the Netherlands and internationally are obtained. The embedding of deep case-based knowledge within a broad international perspective, yet with a focus on the role of system knowledge, makes the learning useful for coastal decision making worldwide.

The case studies in the international cross-comparison satisfy the following criteria:

- There is an inlet management or estuary mouth management issue,
- The issue is understood to be nested within a broader ecological and social system context,
- Place-based knowledge is used,
- Scientists have been, and are, engaged with coastal management.

More generally, the coastal management situations in the case studies exhibit characteristics of 'wicked', 'messy' or 'unstructured' problems where complexity is inherent, outcomes are uncertain, and there are diverse viewpoints on what is known, and which outcomes are desired (Ackoff, 1980; Enserink et al., 2010; Rittel & Webber, 1973). Schön & Rein (1994) claim that such situations are fundamentally about competing values rather than gaps in scientific knowledge. So developing comprehensive and deeper scientific knowledge in individual disciplines will not necessarily help in solving the coastal management problems. However, like Head and Alford (2015), we argue that partial, provisional solutions can be pursued through scientific learning within and across such situations. The aim of the book, therefore, is to engender such learning across a diversity of case studies in estuary and inlet management.

The diversity of the case studies presents its own particular challenge to learning. Each of the case studies occurs within a different bio-geophysical coastal system and within a different socio-economic context. Which aspects can usefully be compared? In addressing this challenge, we examine a number of theoretical perspectives at the outset. Systems thinking (Ackoff, 1971; Checkland, 1981; Ison et al., 1997; Meadows, 2008) and policy analysis (Thissen & Walker, 2013; W. E. Walker, 2000) are fundamental to our approach, so they are introduced first. These theoretical perspectives provide the analytical lens and the methods (the ways of inquiry) through which we seek to learn about the case studies. Next, a number of integrated environmental management paradigms that have been established as underpinning integrated coastal management (Frantzeskaki et al., 2010; Taljaard et al., 2011) are described. These paradigms include environmental assessment, objectives-based management, adaptive management and ecosystem-based management. Social-ecological systems theory is then described (Berkes & Folke, 1998; Redman et al., 2004) and the move to include multi-disciplinary, place-based learning that rests upon system understanding in the management of the environment is highlighted. Each of the case studies (the objects of inquiry) is subsequently positioned against an integrated environmental management paradigm or social-ecological systems theory.

1.2. Theoretical framing

1.2.1. Systems thinking

Systems thinking tackles complex problems by treating the system - the set of interrelated and interdependent component elements (Ackoff, 1971) - as a whole (Checkland, 1981). In 1968, Von Bertalanffy (1968) stated that an entire system's behaviour cannot be understood by understanding the behaviour of each of the component parts in isolation. Instead systems and their behaviour are more than the sum of the parts and they have emergent properties that do not exist in the parts but are found in the whole (Weinberg, 1975). Many different types of systems have since been recognised in nature and society, ranging from ecosystems, through organisations and industrial systems to information systems and architectures (Costa et al., 2019; Ison et al., 1997). Common across these systems is the need to explore the implications of human interventions and decision making on the system properties and behaviour (Meadows, 2008). This has given rise to diverse fields of study such as cybernetics and simulation modelling (e.g., Forrester, 1961), and policy analysis (Thissen & Walker, 2013; W. E. Walker, 2000), all informed by systems thinking.

1.2.2. Policy analysis

Policy analysis employs a purposeful, systematic process to assist public policy decision makers in choosing which interventions to adopt in a system by (i) clarifying the problem, (ii) outlining the alternative intervention solutions and (iii) displaying the trade-offs amongst the outcomes (W. E. Walker, 2000). Policy analysis has a problem focus, conceptualising the problem as a system (see Enserink et al., 2010), rather than a method focus. A wide range of methods are adopted in organising and presenting information to those involved in policy making to help them in decision making. Indeed, the field of policy analysis recognises that in most complex problems there are many potential interventions, many factors over which the decision maker has no control, many interested stakeholders and many potential outcomes of interest. Frequently, there is more than one decision maker involved and preferences regarding the desirability of the outcomes are diverse. In short, an optimal choice for an action or intervention is seldom possible (see Thissen & Walker, 2013).

This contrasts with decision analysis, a rational, technical approach that assumes that a logically consistent choice can be made based on adequate information and the careful specification of desired targets, provided appropriate methods are applied. The argument is that a knowledge-based approach supports high quality decisions, reducing the risk of ill-informed or emotionally-based decisions. While policy analysis employs techniques from decision analysis in identifying decision criteria, listing out the various alternatives, and deliberating the present and future consequences of each alternative, it does not always ascribe weights to each criterion and rate each alternative on each criterion. Instead the focus lies on understanding the problem. Policy analysis recognises that an individual cannot have complete information, nor can they fully comprehend all alternatives and their consequences. In addition, an individual's preferences may fluctuate or alter over time. In reality, therefore, individuals do not exhibit fully rational decision making behaviour. Indeed, Simon (1955, 1957, 1991) defined an 'administrative' being rather than a purely 'economic' decision-maker, introducing the concepts of 'bounded rationality' and 'satisficing'. Situations in which individuals hold divergent interests and values on the one hand and divergent perceptions of reality on the other hand continue to present a challenge to decision making (Kørnøv & Thissen, 2000; March, 1991; Van de Riet, 2003).

Currently, the field of policy analysis accommodates a range of styles, drawing on a systems thinking base (Mayer et al., 2004). Where rational, technical views predominate the choice amongst alternatives can be supported by decision analysis. Where differences in values, and different perceptions of the problem predominate, the problem structuring (Enserink et al., 2010) and game structuring approaches (Cunningham et al., 2014; Slinger et al., 2014) of participatory policy analysis are most applicable. Three cornerstones for realizing participatory decision making in complex problem settings have been identified, namely: (i) valid policy- or decision-relevant scientific knowledge, (ii) process management whereby the involved stakeholders consent to a process designed to achieve appropriate and information-based decision outcomes, and (iii) stable stakeholders participation that acknowledges different roles and contributions (Agre & Leshner, 2010; Kørnøv & Thissen, 2000; Miser & Quade, 1985; Van de Riet, 2003).

In addition to these participatory engagement methods, there are numerous methods and techniques available to support policy analytic decision making. In particular, a graphical representation method, the system diagram (Figure 1.1), can be used to depict: (i) the demarcation of the problem under consideration (the boundary), (ii) the relationship between factors influencing the system behaviour, (iii) whether these influencing factors are external, internal or comprise the interventions of (managing and other) actors in the system, (iv) the outcomes from the system and how these relate to management objectives. In the system diagram (see Enserink et al., 2010), the policy makers, scientists and societal actors are not included explicitly, but are viewed as sources of knowledge, or as controlling the interventions.

1.2.3. Integrated Environmental Management

For a long time, the management of natural resources and the environment occurred via specific uses or sectors such as forestry, fisheries, agriculture, freshwater supply, wastewater discharge, and housing development (United Nations Environment Programme [UNEP], 2006). Where this approach has persisted, increasing demands on limited natural resources have resulted in conflicts between the different uses, aggravated by ineffective management. The concept of Integrated Environmental Management was introduced in the 1980's to address these issues by adopting a more holistic and interconnective approach (Margerum, 1999; Margerum & Born, 1995), and focussing on system goals through a strategic approach (Born & Sonzogni, 1995; Lang, 1986). This conceptual development

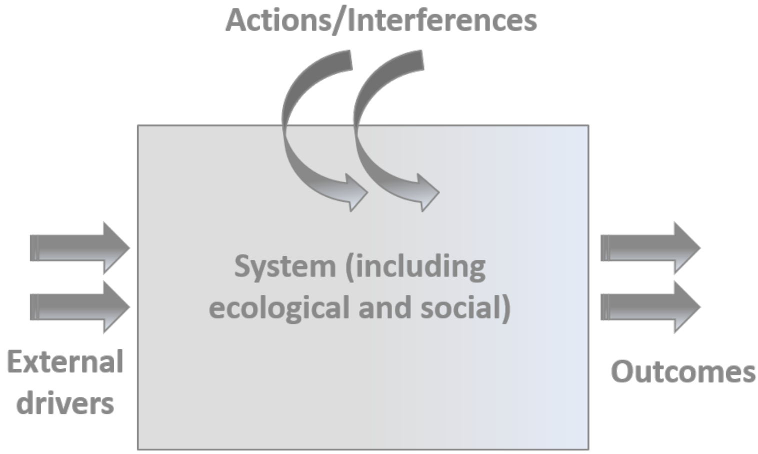

Figure 1.1. The system diagram

in environmental management was mirrored in the coastal environment. In the 1980s, the need became clear for an inter-sectoral approach to the coast taking account of all the activities affecting the coastal environment and its resources, and dealing with economic and social issues as well as environmental (ecological) concerns (Post & Lundin, 1996). Today, the Integrated Coastal Management approach aims to balance development and conservation, to ensure multi-sectoral planning, and to facilitate participation and conflict mediation (Christie, 2005).

Literature on the management of coastal environments emphasises the importance of country-specific knowledge and contextual factors in evaluating implementation of management policies (Cicin-Sain et al., 1998; Olsen et al., 1997; UNEP & Global Programme of Action for the Protection of the Marine Environment from Land-Based Activities [GPA], 2006). Indeed, Taljaard et al. (2011) note that there is no international, generic blueprint for integrated coastal (environmental) management that can be applied routinely to yield predictable and desirable outcomes. However, a number of paradigms have been established as underpinning integrated coastal (environmental) management (Frantzeskaki et al., 2010; Taljaard et al., 2011) and these provide a means of characterizing the predominant management approaches adopted in each of the research case studies.

Environmental assessment paradigm

Internationally, the National Environmental Policy Act of the United States in 1969, represents the first legal requirement for environmental assessment (Jay et al., 2007). Environmental assessments may be undertaken at two levels, namely the individual project level, referred to as Environmental Impact Assessment (EIA) and the plans, programme or policy level referred to as Strategic Environmental Assessment (SEA) (Fischer, 2003). Essentially, environmental impact assessment is a systematic process for determining the potential environmental consequences of a proposed project (or action) (Jay et al., 2007). The primary purpose of this anticipatory and participatory environmental management instrument is to inform decision makers of the likely environmental consequences of a project (or action) so as to support environmentally sound development decisions (Fischer, 2003; Jay et al., 2007). Strategic environmental assessment encompasses a range of analytical and participatory approaches that aim to integrate environmental considerations into policies, plans and programmes so as to clarify the inter-linkages with wider economic and social systems and so include environmental considerations into strategic decision making (Partidário, 1996, 2008; Wallington et al., 2007). Actor participation, appropriate process management, and sound scientific knowledge are viewed as essential to environmental assessment (Taljaard et al., 2011). In this, the environmental assessment paradigm agrees with characteristics of the participatory policy analysis paradigm.

Objectives-based management paradigm

The core concept of objectives-based management as outlined by Drucker (1954) is improving the performance of an organisation by clearly defining and agreeing objectives at all levels within an organisation. By aligning objectives across an organisation, managers and employees can avoid becoming so involved in day-to-day activities that their main purpose or objective is forgotten – the so-called 'activity trap'. Fundamental to the objectives-based management approach is the call for participatory involvement in the strategic planning process, so that implementation is expedited. In applying this concept to environmental management, the participatory involvement of actors at all levels naturally comes to the fore. Involved actors aid in determining environmental objectives. Management strategies (or environmental management programmes) are then developed with the aim of attaining the objectives, which are specified in terms of outcome indicators and associated target values. The implementation and assessment for compliance is undertaken primarily by civil servants at national, regional, and local levels (Edvardsson, 2004; Wibeck et al., 2006). A strength of the objectives-based management paradigm is the emphasis placed on setting objectives holistically for the environment (i.e., incorporating the biophysical environment, the social and the economic environment). In this aspect, the paradigm differs from the primarily biophysical/ecological (and sometimes local social) focus of the environmental assessment paradigm.

Adaptive management paradigm

According to Haber (1964) and Bornmann et al. (1999), the adaptive management concept originated in the early 1900s when ideas of scientific management were pioneered. Fundamental to the adaptive management paradigm is a healthy scepticism regarding predictive environmental assessments, typically undertaken prior to action. Instead, the limitations of model-based or predictive assessments in dynamically uncertain environmental systems are understood, and the value of experiential learning is appreciated. Adaptive management builds on learning from experience, by experimenting and monitoring the results of experiments and then adjusting practices based on the learning attained (Bornmann et al., 1999). Sound environmental monitoring and evaluation programmes to support learning and subsequent adaptation are central to this paradigm. By actively accommodating system changes and the unexpected (Noble, 2000), adaptive management introduces the use of iterative, incremental adjustments as a requirement in managing complex environmental systems.

Ecosystem-based management paradigm

The realisation that natural resources and the environment can be managed more effectively if the ecosystem is placed centrally (Costanza, 1998; Pretty & Ward, 2001) and management occurs through cooperative governance between different sectors led to the concept of ecosystem-based management (UNEP, 2006). Ecosystem-based management recognises that plants, animals, and human communities are interdependent and interact dynamically within a particular physical environment forming distinct spatial units or ecosystems (UNEP, 2006). Humans and development are viewed as an integral part of an ecosystem. There is a shift from centralised, top-down governance of the environment to a decentralised regional and local approach to resource management in which multiple stakeholder groups are involved. Ecosystem-based management further requires that the carrying capacity of the ecosystem is not exceeded, but that development occurs sustainably (Balchand et al., 2007; United Nations, 1987). The ecosystem-based management paradigm supports participatory actor involvement, requiring multi-sector, cooperative governance systems to be established (Taljaard et al., 2011). Management of the environment in its biophysical, social and economic aspects characterises the ecosystem-based management paradigm.

1.2.4. Social-ecological systems and transdisciplinarity

A social-ecological system is a coherent system of biophysical and social factors that regularly interact in a resilient, sustained manner, through coupled, non-linear interactions. Moreover, this coupled, complex system is dynamic, exhibiting continuous adaptation (Redman et al., 2004). So, the concept of social-ecological systems as linked systems of people and nature emphasises that humans are viewed as a part of, not apart from, nature (Berkes & Folke, 1998). Inherently a social-ecological system is a nested system with several spatial, temporal and organisational scales that may be hierarchically linked (Redman et al., 2004). The resilience of a social-ecological system is conceived as the capacity of a social-ecological system to absorb or withstand perturbations so as to maintain its structure and functions, and provides an indication of the degree to which the system is capable of self-organisation, learning and adaptation (Gunderson & Holling, 2002; Holling, 1973; B. Walker et al., 2004). A set of seven principles have been identified for building resilience and sustaining ecosystem services in social-ecological systems, namely: maintaining diversity and redundancy, managing connectivity, managing slow variables and feedbacks, fostering complex adaptive systems thinking, encouraging learning, broadening participation, and promoting polycentric governance systems (Biggs et al., 2012).

Social-ecological systems theory embodies a co-evolutionary view of the relationship between humans and nature. Humans and the whole social system are viewed as essentially part of the social-ecological system – an all-encompassing system present at multiple, nested scales. In Figure 1.2, a complex coastal system decomposed into a coupled ecological and social system is depicted. In this view, humans participate

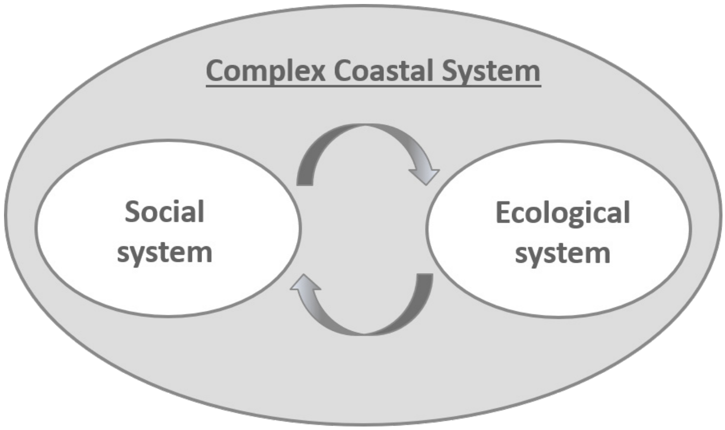

Figure 1.2. A complex coastal system conceptualised in terms of its social and ecological components

naturally in decision making on the environment, and participatory approaches aimed at environmental stewardship are embraced. Social-ecological theory therefore represents an extension of ecosystem-based management, and simultaneously incorporates the incremental learning of adaptive management. Where objectives-based management is applied without accounting for other potential environmental objectives or long-term sustainability, this would lie outside of social-ecological theory. Social-ecological theory recognises multiple sources of (disciplinary) knowledge for system understanding namely, environmental and social science, practice, local stakeholder knowledge and governance or decision making knowledge. Place-based contextual knowledge is also explicitly valued. This leads naturally to the adoption of a transdisciplinary approach in studying complex social-ecological systems.

Transdisciplinary is defined as a scientific approach that seeks to learn across disciplines (multi-disciplinary), using place-based knowledge, involving scientists and society, through convergent and divergent phases of learning and reflection. It seeks to usefully combine the reductionist thinking of scientific disciplines with the local knowledge of a place, and by reflecting on actions and effects now and in the past to make science and scientific practice relevant to society (Bergmann et al., 2012; Max-Neef, 2005).

1.3. The seven case studies

Seven case studies form the basis for the analysis. The case studies are located in the following countries: The Netherlands (2x), The United States of America, Ireland, Sri Lanka, Suriname and South Africa (Figure 1.3). The case studies focus on inlet or estuary mouth management, comprising four micro-tidal estuaries, two larger inlets and a wetland lake intermittently connected to the sea. Each of the case studies is nested within the context of scientific engagement in their respective countries with certain author(s) having a deep familiarity with the study site and its biophysical and/or social context. As such, the material presented here is only a selection of the full range of knowledge on each of the systems and is presented with its own particular slant. Whereas the overarching

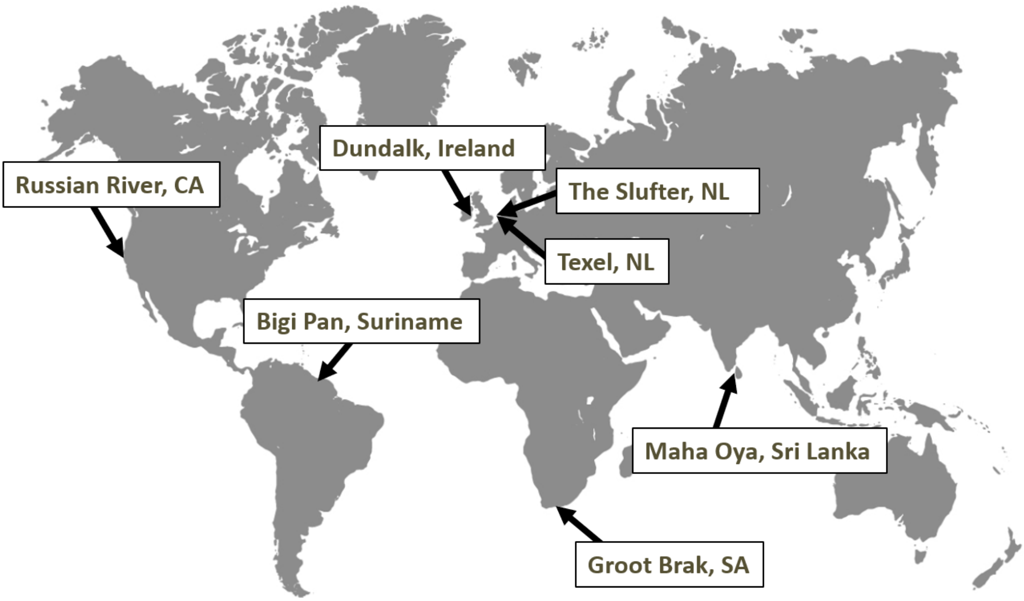

Figure 1.3. Locations of the seven case studies

research approach draws on systems thinking and policy analysis, each of the case studies differs in terms of its predominant theoretical paradigm, as listed in Table 1.1. and described briefly per case study thereafter.

Case study 1: Texel Inlet, the Netherlands

Texel Inlet represents a case study in Dutch coastal management. The imperative to protect the Dutch coast from flooding has been the central issue in coastal management for centuries. The damming of the Zuiderzee, a salt water inlet of the North Sea, formed a fresh water lake - the IJsselmeer, and initiated a process of coastal sedimentary re-adjustment of which the Texel Inlet forms part. However, since 1990 Dutch coastal policy is aimed at preventing structural erosion by maintaining the Dutch coastline at the 1990 position through sand nourishments. This objectives-based policy and associated sand nourishment strategy now ensures that south-west Texel receives a large portion of the sand nourishment budget as it is an erosion hotspot. However,

Table 1.1. Orientation of the case studies against the theoretical paradigms

Case Study	Country	Predominant theoretical paradigm
Texel Inlet	The Netherlands	Objectives-based Management
Dundalk Bay	Republic of Ireland	Environmental Assessment
Maha Oya	Sri Lanka	Environmental Assessment
Russian River	California, United States of America	Objectives-based Management
Groot Brak	Republic of South Africa	Adaptive Management
Bigi Pan	Suriname	Ecosystem-based Management
The Slufter	The Netherlands	Social-Ecological Systems

recent geomorphological insights on the dynamics of the ebb-tidal delta suggest that a large sandy shoal on the north-eastern margin of the ebb-tidal delta will in time attach to the south-western side of the island of Texel. This calls into question the wisdom of continuing to nourish this part of Texel. In essence, the Texel Inlet case study highlights how a single issue – flood risk management - can dominate in determining the objectives for coastal management. It draws attention to the role of scientific insights in improving management and highlights the need for collaborative, participatory approaches in designing alternative coastal management strategies that address multiple objectives.

Case study 2: Dundalk Bay, Republic of Ireland

Dundalk Bay is located on the northeast coast of Ireland and is of social and ecological importance, particularly as a fishing resource and regional harbour. The water quality issues associated with the rivers flowing into Dundalk Bay are the primary driver for the study. There is a need for catchment management to improve the quality of the inflowing water as well as a need for holistic and integrated management approaches. Here, scientists are actively involved in supporting community-based engagement with a view to enhancing integrated management of the water and coastal systems. The case study highlights the need to progress from environmental assessment to engaged co-management approaches in an effort to support learning within a social-ecological system.

Case study 3: Maha Oya, Sri Lanka

The case study of the Maha Oya Estuary in Sri Lanka focusses on the issue of climate change. Modelling research on the effects of climate change on small, wave-dominated estuaries led to the understanding that the frequency, period and duration of mouth closure of the estuary could change owing to both sea level rise and changing river flows. This new knowledge represents a pro-active environmental assessment and serves as a signal to Sri Lankan coastal managers that these external factors cannot be ignored. Coastal management will have to alter to accommodate these effects, particularly as the subsistence fishermen, sand miners and tourism-dependent occupations rely on estuary functioning for their incomes. This case study illustrates the role of scientific knowledge in alerting coastal managers of the need for change.

Case study 4: Russian River, California, United States of America

A 2010 Biological Opinion, a legal instrument, to ensure that the Russian River in California is managed for maintaining the habitat of the juvenile steelhead trout, represents a significant stage in the management of this estuary. Years of research by Californian scientists, particularly the Bodega Marine Laboratory (UC Davis), together with observation records of a citizen living near the mouth, are used to determine the relationship between the state of the mouth of this intermittently closed estuary and the habitat requirements for the endangered species. The biophysical system knowledge based on an extensive data set is shown to be crucial in managing for this single species objective.

Case study 5: Groot Brak, South Africa

Since the construction of the Wolwedans Dam upstream of the Groot Brak Estuary, South Africa, in 1990, this small, wave-dominated system has received both research and management attention. Initially the attention focussed on designing a water release and mouth management policy for the estuary to prevent flooding and a decline in estuary health, and to ensure that local socio-economic activities were not impacted adversely. The focus of the case study presented in this book is the incremental learning on mouth management practices over a thirty-year period, and the adaptation of the management of water releases and mouth breaching in response to this. The case study reveals ongoing learning regarding the character and functioning of the estuary and highlights how this growing scientific understanding then influenced management practice and policy.

Case study 6: Bigi Pan, Suriname

The Bigi Pan in Suriname is a wetland lake that is intermittently connected to the sea. The case study analyses the implementation problems of the Bigi Pan Multiple Use Management Area (MUMA). The MUMA was designed to accommodate people living within, using and drawing benefit from, an ecologically significant conservation area. It embodies the principles of ecosystem-based management, and institutionalises co-management. The case study draws upon an extensive round of stakeholder interviews regarding the functioning of the MUMA. It highlights the need for system understanding as the foundation for effective coastal management, and identifies a number of strategies to address this gap and improve management.

Case study 7: The Slufter, Texel, the Netherlands

New coastal modelling insights that the estuary mouth may not need to be straightened periodically as a means of mitigating the flood risk to the dike landward of the Slufter Estuary, led to a desire on the part of the Water Board to re-evaluate their mouth management strategy. A social-ecological systems lens was adopted by researchers from the outset. This means that the issue of mouth management was not interpreted only as a biophysical problem, nor only as a flood-risk management issue, but as a multi-facetted issue arising from an increasing awareness of the ecological and social value of the Slufter Estuary, and a desire to work with nature rather than against nature - the Building with Nature philosophy (Ecoshape, 2019; Slinger, 2016; Waterman, 2010). A process of stakeholder engagement was undertaken in which the divergent perspectives and values of local stakeholders in regard to mouth management were explored with the aid of system dynamics modelling (D'Hont, 2014). In this case study, the role of system understanding is shown to be fundamental to learning on coastal management within the social-ecological system.

1.4. Transdisciplinary approach

Diverse environmental concepts (C) and methods (M) - ways of inquiry - are employed by the scientists involved in the seven coastal case studies that form the objects of inquiry in this endeavour (Table 1.1, Figure 1.4). However, the fundamental strategy of inquiry in this book is informed by the systems concepts and methods of the policy analysis scientists. Together, the coastal environmental scientists and the policy analysts have sought to learn from each case study and across the case studies by sharing experiences and reflecting jointly on the theoretical concepts employed, the methods applied, and the particularities of the individual coastal systems (S). The new insights from this transdisciplinary approach were reported in the proceedings of the intensive week-long

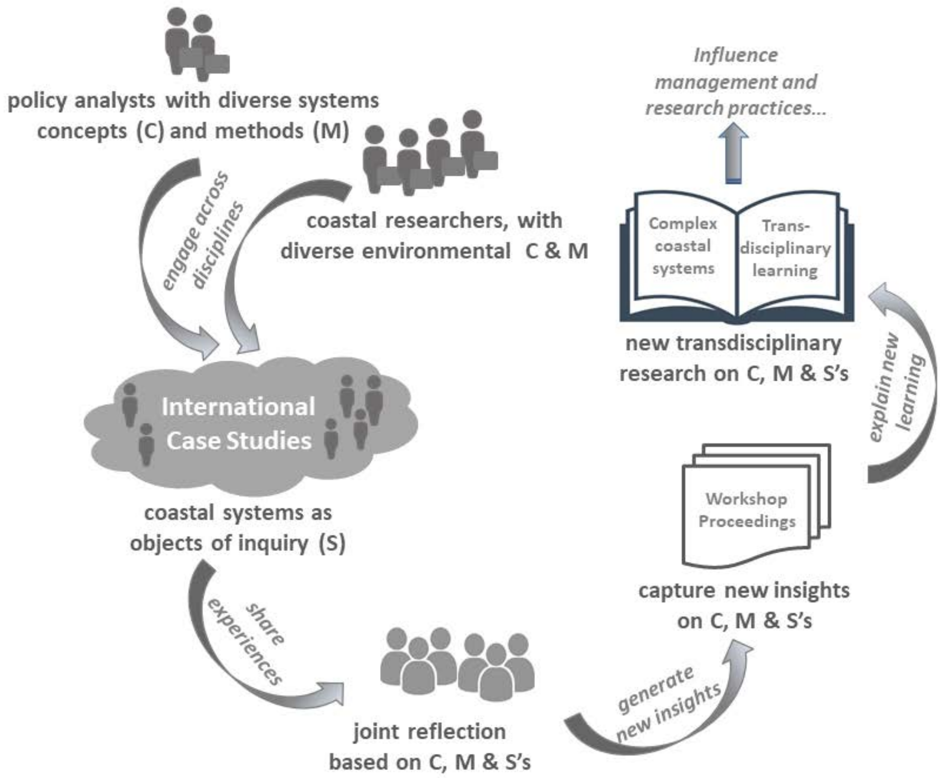

Figure 1.4. The transdisciplinary learning process applied in the cross-comparison of the international coastal case studies

workshop held at Delft University of Technology in September 2017 (D'Hont & Slinger, 2017). In this book, the learning from the full transdisciplinary research endeavour is synthesised by cross-comparing the coastal systems (S), the methods (M) applied and the concepts employed by the involved scientists (C). The cross-comparison is itself informed by concepts from systems thinking and policy analysis, with the aim of influencing coastal management and research practice internationally.

1.5. Reading and use guide

This introductory chapter has established the theoretical underpinning of the book in systems thinking and policy analysis, and has positioned the seven coastal case studies against the paradigms underlying integrated environmental management or social-ecological systems. Each of the case studies differs in terms of its predominant theoretical paradigm in combination with the insights offered and the type of biophysical and/or social system described. Readers primarily interested in big bay or inlet systems are advised to focus on the Texel Inlet and Dundalk Bay case studies. Readers interested in small, wave-dominated estuaries are invited to concentrate on the Maha Oya, Russian River, Groot Brak and Slufter estuaries. Those interested in the social aspects are directed towards the Bigi Pan and Slufter case studies, while those more interested in the biophysical aspects can focus on the other case studies. While each case study chapter can be read as a stand-alone unit, valuable insights are gained from cross-comparing and learning across the case studies as described in the concluding chapter.

1.6. References

Ackoff, R. L. (1971). Towards a System of Systems Concepts. *Management Science,* 17(11), 661–671. https://doi.org/10.1287/mnsc.17.11.661

Ackoff, R. L. (1980). The systems revolution. In M. Lockett & R. Spear (Eds.), Organisations as systems (pp. 26–33). The Open University Press.

Agre, P., & Leshner, A. I. (2010). Bridging science and society. *Science*, 327(5968), 921. https://doi.org/10.1126/science.1188231

Balchand, A., Mooleparambil, S. M., & Reghunathan, C. (2007). Management Strategies for Urban Coastal Zones: Integrating DPSIR Concepts with GIS Tools in People's Participatory Programs. *IHDP Update 2*.

Bergmann, M., Jahn, T., Knobloch, T., Krohn, W., Pohl, C., Schramm, E., … Faust, R. C. (2012). Methods for transdisciplinary research: a primer for practice. Campus Verlag.

Berkes, F., & Folke, C. (1998). Linking social and ecological systems for resilience and sustainability. *Linking Social and Ecological Systems: Management Practices and Social Mechanisms for Building Resilience*, 1–25.

Biggs, R., Schlüter, M., Biggs, D., Bohensky, E. L., BurnSilver, S., Cundill, G., … West, P. C. (2012). Toward Principles for Enhancing the Resilience of Ecosystem Services. *Annual Review of Environment and Resources*, 37(1), 421–448. https://doi.org/10.1146/annurev-environ-051211-123836

Born, S. M., & Sonzogni, W. C. (1995). Integrated environmental management: strengthening the conceptualization. *Environmental Management*, 19(2), 167–181. https://doi.org/10.1007/BF02471988

Bornmann, B. T., Martin, J. R., Wagner, F. H., Wood, G., Alegria, J., Cunningham, P. G., … Henshaw, J. (1999). Adaptive management. In N. C. Johnson, A. J. Malk, W. Sexton, & R. Szaro (Eds.), *Ecological Stewardship: A common reference for ecosystem management* (pp. 505–534). Amsterdam, Netherlands: Elsevier Science Ltd.

Checkland, P. (1981). *Systems thinking, systems practice*. Chichester, UK: J. Wiley.

Christie, P. (2005). Is integrated coastal management sustainable? *Ocean & Coastal Management,* 48(3), 208–232.

Cicin-Sain, B., Knecht, R. W., Jang, D., & Fisk, G. W. (1998). *Integrated coastal and ocean management: concepts and practices*. Island Press.

Costa, J., Diehl, J. C., & Snelders, D. (2019). A framework for a systems design approach to complex societal problems. *Design Science*, 5(e2), 1–32. https://doi.org/10.1017/dsj.2018.16

Costanza, R. (1998). The value of ecosystem services. *Ecological Economics*, 25(1), 1–2. https://doi.org/10.1016/s0921-8009(98)00007-x

Cunningham, S. W., Hermans, L. M., & Slinger, J. H. (2014). A review and participatory extension of game structuring methods. *EURO Journal on Decision Processes*, 2(3–4), 173–193.

D'Hont, F. M. (2014). *Does deepening understanding of make sense? A partial success story from the Slufter, Texel*. MSc Thesis. Delft University of Technology. Retrieved from http://repository.tudelft.nl/view/ir/uuid%3Af79420ff-9c9a-42b5-a7d4-65bb5db10ac3/

D'Hont, F. M., & Slinger, J. H. (2017). Co-designing estuary inlet management: Policies, people and practices. In *Proceedings International Workshop*. Delft, The Netherlands: Delft University of Technology.

Drucker, P. F. (Peter F. (1954). *The practice of management*. New York and Evanston: Harper & Row.

Ecoshape. (2019). Building with Nature. Retrieved December 4, 2019, from https://www.ecoshape.org/nl/

Edvardsson, K. (2004). Using goals in environmental management: The Swedish system of environmental objectives. *Environmental Management*, 34(2), 170–180. https://doi.org/10.1007/s00267-004-3073-3

Enserink, B., Hermans, L., Kwakkel, J., Thissen, W., Koppenjan, J., & Bots, P. (2010). *Policy Analysis of Multi-Actor Systems*. Lemma.

Fischer, T. B. (2003). Strategic environmental assessment in post-modern times. *Environmental Impact Assessment Review*, 23(2), 155–170. https://doi.org/10.1016/S0195-9255(02)00094-X

Forrester, J. (1961). *Industrial dynamics*. Cambridge, Massachusetts: M.I.T. Press.

Frantzeskaki, N., Slinger, J., Vreugdenhil, H., & van Daalen, E. (2010). Social-Ecological Systems Governance: From Paradigm to Management Approach. *Nature and Culture*, 5(1), 84–98. https://doi.org/10.3167/nc.2010.050106

Group of Experts on the Scientific Aspects of Marine Environmental Protection. (1996). *The contributions of science to integrated coastal management (GESAMP Report and Studies* No. 61). Rome, Italy: United Nations Food and Agriculture Organisation.

Gunderson, L. H., & Holling, C. S. (2002). *Panarchy: understanding transformations in systems of humans and nature*. Island, Washington.

Haber, S. (1964). *Efficiency and uplift; scientific management in the progressive era*, 1890-1920. Chicago: University of Chicago Press.

Head, B. W., & Alford, J. (2015). Wicked Problems: Implications for Public Policy and Management. *Administration and Society*, 47(6), 711–739. https://doi.org/10.1177/0095399713481601

Holling, C. S. (1973). Resilience and stability of ecological systems. *Annual Review of Ecology and Systematics*, 4, 1–23. https://doi.org/10.1146/annurev.es.04.110173.000245

Ison, R. L. (2008). Systems thinking and practice for action research. In P. W. Reason & H. Bradbury (Eds.), *The Sage Handbook of Action Research Participative Inquiry and Practice* (2nd ed., pp. 139–158). London, UK: Sage Publications.

Ison, R. L., Maiteny, P. T., & Carr, S. (1997). Systems methodologies for sustainable natural resources research and development. In Agricultural Systems (Vol. 55, pp. 257–272). https://doi.org/10.1016/S0308-521X(97)00010-3

Jay, S., Jones, C., Slinn, P., & Wood, C. (2007). Environmental impact assessment:

Retrospect and prospect. Environmental Impact Assessment Review, 27(4), 287–300. https://doi.org/10.1016/j.eiar.2006.12.001

Kørnøv, L., & Thissen, W. A. H. (2000). Rationality in decision- and policy-making: Implications for strategic environmental assessment. Impact Assessment and Project Appraisal, 18(3), 191–200. https://doi.org/10.3152/147154600781767402

Lang, R. (1986). Integrated approaches to resource planning and management. Calgary, Alberta, Canada: University of Calgary Press. Retrieved from file://catalog.hathitrust.org/Record/001534391

March, J. G. (1991). How Decisions Happen in Organizations. Human–Computer Interaction, 6(2), 95–117. https://doi.org/10.1207/s15327051hci0602_1

Margerum, R. D. (1999). Integrated environmental management: The foundations for successful practice. Environmental Management, 24(2), 151–166. https://doi.org/10.1007/s002679900223

Margerum, R. D., & Born, S. M. (1995). Integrated Environmental Management: Moving from Theory to Practice. Journal of Environmental Planning and Management, 38(3), 371–392. https://doi.org/10.1080/09640569512922

Max-Neef, M. A. (2005). Foundations of transdisciplinarity. Ecological Economics, 53(1), 5–16. https://doi.org/10.1016/j.ecolecon.2005.01.014

Mayer, I. S., Van Daalen, C. E., & Bots, P. W. G. (2004). Perspectives on policy analyses: A framework for understanding and design. International Journal of Technology, Policy and Management, 4(2), 169–191. https://doi.org/10.1504/IJTPM.2004.004819

Meadows, D. H. (2008). Thinking in systems : a primer. (D. Wright, Ed.). White River Junction, VT: Chelsea Green Publishing. Retrieved from http://catdir.loc.gov/catdir/toc/ecip0825/2008035211.html

Miser, H. J., & Quade, E. S. (Edward S. . (1985). Handbook of systems analysis: overview of uses, procedures, applications, and practice. Chichester, UK: John Wiley and Sons Ltd.

Nelson, H. G. (2008). Design communication: Systems, service, conspiracy, and leaderships. In Dialogue as a Collective Means of Design Conversation (pp. 39–49). New York, USA: Springer. https://doi.org/10.1007/978-0-387-75843-5_3

Noble, B. F. (2000). Strengthening EIA through adaptive management: A systems perspective. Environmental Impact Assessment Review, 20(1), 97–111. https://doi.org/10.1016/S0195-9255(99)00038-4

Olsen, S. B., Lowry, K., & Tobey, J. (1999). A manual for assessing progress in coastal management (Coastal Management Report 2211). Rhode Island, USA: Coastal Resources Centre, University of Rhode Island.

Olsen, S. B., Tobey, J., & Kerr, M. (1997). A common framework for learning from ICM experience. Ocean and Coastal Management, 37(2), 155–174. https://doi.org/10.1016/S0964-5691(97)90105-8

Partidário, M. R. (1996). Strategic environmental assessment: Key issues emerging from recent practice. Environmental Impact Assessment Review, 16(1), 31–55. https://doi.org/10.1016/0195-9255(95)00106-9

Partidário, M. R. (2008). Theory and Practice of Strategic Environmental Assessment.

Towards a more systematic approach - By Thomas B. Fischer. Natural Resources Forum, 32(1), 86–87. https://doi.org/10.1111/j.1477-8947.2008.173_4.x

Post, J. C., & Lundin, C. G. (1996). Guidelines for integrated coastal zone management. Environmentally Sustainable Development Studies and Monographs Series No. 9. Washington, DC: World Bank.

Pretty, J., & Ward, H. (2001). Social capital and the environment. World Development, 29(2), 209–227. https://doi.org/10.1016/S0305-750X(00)00098-X

Redman, C. L., Grove, J. M., & Kuby, L. H. (2004). Integrating social science into the long-term ecological research (LTER) network: social dimensions of ecological change and ecological dimensions of social change. Ecosystems, 7(2), 161–171.

Reis, J., Stojanovic, T., & Smith, H. (2014). Relevance of systems approaches for implementing integrated Coastal Zone management principles in Europe. Marine Policy, 43, 3–12. https://doi.org/10.1016/j.marpol.2013.03.013

Rittel, H. W. J., & Webber, M. M. (1973). Dilemmas in a general theory of planning. Policy Sciences, 4(2), 155–169. https://doi.org/10.1007/BF01405730

Schön, D. A., & Rein, M. (1994). Frame reflection : toward the resolution of intractable policy controversies. New York, USA: BasicBooks.

Simon, H. A. (1955). A Behavioral Model of Rational Choice. The Quarterly Journal of Economics, 69(1), 99–118. https://doi.org/10.2307/1884852

Simon, H. A. (1957). Models of man: social and rational mathematical essays on rational human behavior in society setting. New York, USA: John Wiley and Sons Inc.

Simon, H. A. (1991). Bounded Rationality and Organizational Learning. Organization Science, 2(1), 125–134. Retrieved from http://www.jstor.org/stable/2634943

Slinger, J. H. (2016). Engineering: Building with Nature 101x: series of 11 videos [dataset]. Delft, Netherlands: Delft University of Technology. https://doi.org/https://doi.org/10.4121/uuid:721edfdb-a984-470d-be4e-d66161c6c811

Slinger, J. H., Cunningham, S. W., Hermans, L. M., Linnane, S. M., & Palmer, C. G. (2014). A game-structuring approach applied to estuary management in South Africa. EURO Journal on Decision Processes, 2(3–4), 341–363. https://doi.org/10.1007/s40070-014-0036-7

Taljaard, S., Slinger, J. H., & Van Der Merwe, J. H. (2011). Criteria for evaluating the design of implementation models for integrated coastal management. Coastal Management, 39(6), 628–655. https://doi.org/10.1080/08920753.2011.616670

Thissen, W. A. H., & Walker, W. E. (Eds.). (2013). Public Policy Analysis: New Developments. Public Policy Analysis: New Developments. Springer US: Springer. https://doi.org/10.1007/978-1-4614-4602-6

United Nations. (1987). Our Common Future: Report of the World Commission on Environment and Development. General Assembly Resolution 42/187. New York, USA: UN General Assembly. https://doi.org/10.1080/07488008808408783

United Nations Environment Programme. (2006). Ecosystem-based management - Markers for assessing progress. Office. The Hague: UNEP/GPA. Retrieved from www.unep.orgwww.unep.org

United Nations Environmental Programme & Global Programme of Action for the Protection of the Marine Environment from Land-Based Activities. (2006). Protecting coastal and marine environments from land-based activities : a guide for national action. The Hague: UNEP/GPA.

Van de Riet, O. (2003). Policy analysis in multi-actor policy settings: Navigating between negotiated nonsense and superfluous knowledge (PhD dissertation). Delft University of Technology, Delft, Netherlands.

von Bertalanffy, L. (1968). General System Theory: Foundations, Development, Applications. George Braziller New York (Vol. 1). New York: George Braziller.

Walker, B., Holling, C. S., Carpenter, S. R., & Kinzig, A. (2004). Resilience, adaptability and transformability in social-ecological systems. Ecology and Society, 9(2), 5. https://doi.org/10.5751/ES-00650-090205

Walker, W. E. (2000). Policy analysis: a systematic approach to supporting policymaking in the public sector. Journal of Multi-Criteria Decision Analysis, 9(1–3), 11–27. https://doi.org/10.1002/1099-1360(200001/05)9:1/3<11::AID-MCDA264>3.0.CO;2-3

Wallington, T., Bina, O., & Thissen, W. (2007, October). Theorising strategic environmental assessment: Fresh perspectives and future challenges. Environmental Impact Assessment Review, 27(7), 569–584. https://doi.org/10.1016/j.eiar.2007.05.007

Waterman, R. E. (2010). Integrated coastal policy via Building with Nature (PhD dissertation). Delft University of Technology, Delft, the Netherlands.

Weinberg, G. M. (1975). An introduction to general systems thinking. New York: John Wiley & Sons.

Wibeck, V., Johansson, M., Larsson, A., & Öberg, G. (2006). Communicative aspects of environmental management by objectives: Examples from the Swedish context. Environmental Management, 37(4), 461–469. https://doi.org/10.1007/s00267-004-0386-1

2

Texel Inlet Dynamics and Shoreline Management

By Jan Mulder, Filipe Galiforni-Silva, Floortje d'Hont, Kathelijne Wijnberg, Ad van der Spek, Mick van der Wegen and Jill Slinger

2.1. Introduction

Texel Inlet represents a case study in Dutch coastal management. The imperative to protect the Dutch coast from flooding has been the central issue in coastal management for centuries. The damming in 1932 of the Zuiderzee, a major salt water branch of the Dutch Wadden Sea, formed a fresh water lake –the IJsselmeer – and initiated a process of coastal sedimentary readjustment of which the Texel Inlet and adjacent coasts are parts. However, since 1990 Dutch coastal policy is aimed at preventing structural erosion by maintaining the Dutch coastline at the 1990 position through sand nourishments. This objectives-based policy and associated sand nourishment strategy now ensures that south west Texel receives a large portion of the national sand nourishment budget as it is an erosion hotspot. In this case study, we focus on the evolution of integrated flood risk management at Texel Island, showing how scientific insights into coastal dynamics have influenced coastal policy in the past (section 2.4), and how recent advancements in knowledge on the natural dynamics of the system (section 2.3) and on the importance of stakeholder involvement in environmental management, may play a role in a potential adaptation of the policy (section 2.5). In essence, the Texel Inlet case study highlights how a single issue – flood risk management – can dominate in determining the objectives for coastal management, and highlights the role that new scientific insights can potentially play in influencing coastal management into the future.

2.2. Study area

Texel island, Texel Inlet and the adjacent North Sea and Wadden Sea represent a coherent system of high natural value, largely protected under the European environmental law Natura2000. The Texel Inlet is a mixed-energy inlet system connecting the Wadden sea tidal basin to the North Sea (Figure 2.1). It is located in the north-western part of the Netherlands and is the largest inlet system of the Dutch Wadden sea. To the south, it is bordered by the city of Den Helder where the coastline is fixed by the use of groins and dikes (Figure 2.2). To the north lies the island of Texel, characterised by an eroding sandy shore with a dynamic sand flat at its southern tip - De Hors – covering an area of roughly 3 km. Over the past 18 years a dune field has been establishing at De Hors. Just north of De Hors, the coast is protected by groins for 9 km (Figure 2.2).

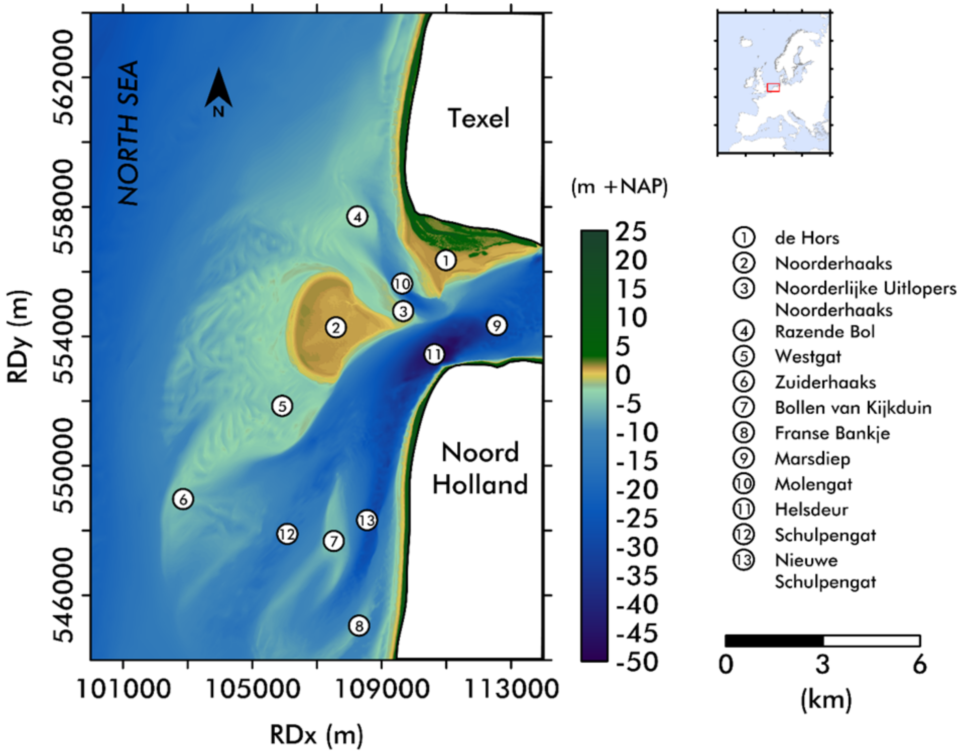

Figure 2.1. The Texel Inlet study area, showing the channel, ebb-tidal delta and shoals and the affected parts of the adjacent shorelines (after Elias et al., 2014)

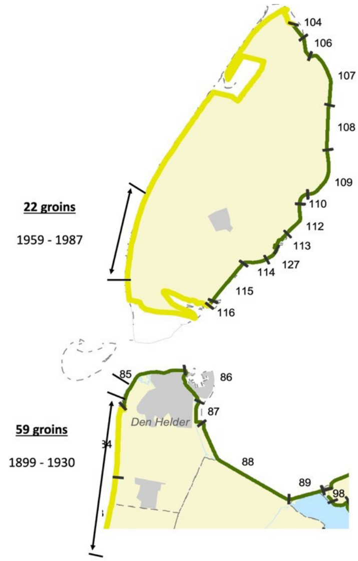

Figure 2.2. Characteristics of the shoreline along the Texel Inlet indicating dunes in yellow, dikes in green, and groins together with year of construction (after HHNK, 2008 and Verhagen and van Rossum, 1990)

2.3. Natural dynamics of the inlet

2.3.1. Hydrodynamics

According to Hayes (1979), the Texel Inlet can be classified as a mixed-energy, wave dominated inlet, although some tide-dominated characteristics such as a large ebb-tidal delta are evident. The tide-dominated features derive from the large tidal prism relative to wave energy (Elias and Van der Spek, 2017). The tide is semi-diurnal with a mean tidal range of 1.4 m, mean high tide level of 0.65 m NAP and a mean high spring tide level of 0.84 m NAP (Wijnberg et al., 2017). The average tidal prism is 990 x 10^9 m^3, with a seaward directed residual prism of 17 x 10^9 m^3 and peak ebb and flood velocities ranging from 1 to 2 $m.s^{-1}$ (Duran-Matute et. al., 2014, Buijsman & Ridderinkhof, 2007). The system is influenced by meteorological distortion of the water levels due to air pressure and wind set-up or set-down, which can reach values of up to 2 m during major storm events (Elias and Van der Spek, 2017). Daily maximum water levels show median values of 0.69 m (Figure 2.3). Data from 1997 up to 2015 show a maximum water level of 2.71 m with values above 2 m occurring less than 0.37 % of the time. The wave climate in the area is dominated by wind-generated waves coming from the North Sea. The mean significant wave height is 1.3 m, with a corresponding period of 5 seconds and a mean direction of west-southwest (Elias and van der Spek, 2006). The largest waves are associated with energetic events coming from the west and northwest owing to the longer wind fetch of the North Sea over these stretches (Sha, 1989; Van der Vegt & Hoekstra, 2012).

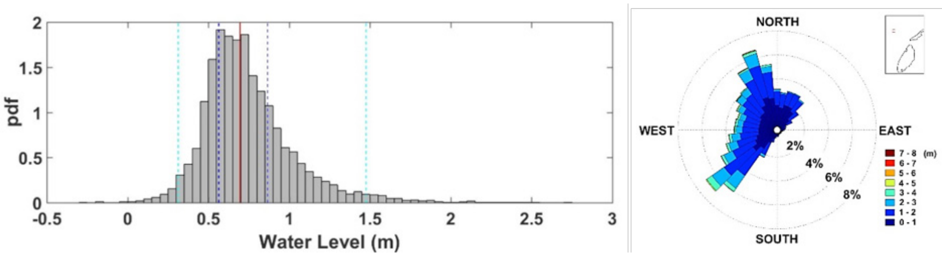

Figure 2.3. Left: Histogram of water level measured over the past 18 years. The median is located at 0.7 m to NAP. The 25 and 75 percentiles are at 0.56 m and 0.87 m, and the 2.5 and 97.5 percentiles are 0.31 m and 1.47 m respectively.
Right: Wave rose based on Eierlandse Gat buoy, near Texel.

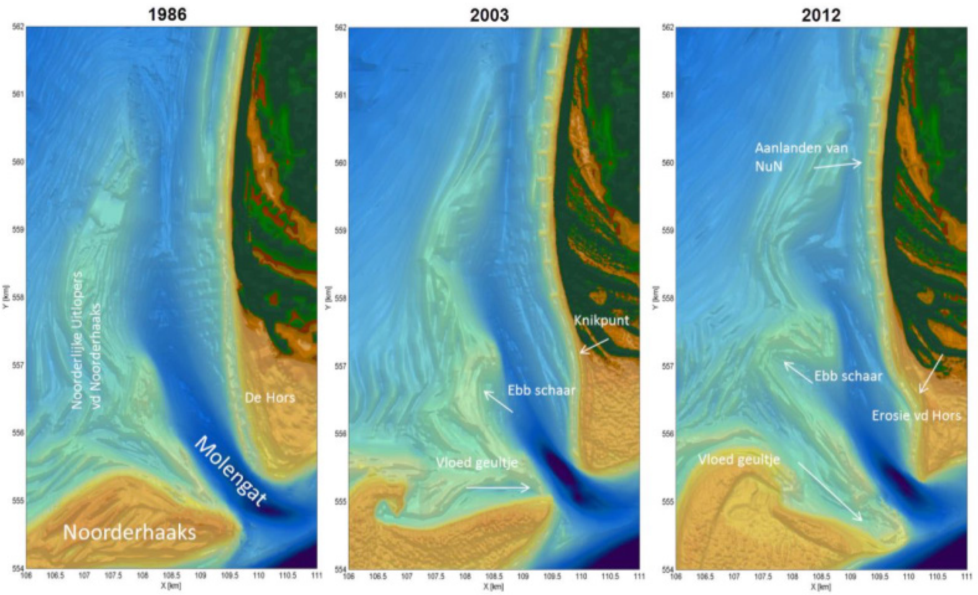

Figure 2.4. The northern parts of the sandy shoal Noorderhaaks in 1986, 2003 and 2012 exhibit a landward movement (from: Elias et al., 2014)

2.3.2. Morphodynamics

Regarding the ebb-tidal delta and the bed sedimentology, average surface grain size varies from 150 µm to 450 µm, depending on the location. Shoals present the smallest average grain size, ranging from 150 to 200 µm, whereas coarser sediments can be found in the Marsdiep area (Elias and van der Spek, 2017). The system presents an asymmetric ebb-tidal delta (Figure 2.1). The closure of the Zuiderzee in 1932 changed overall characteristics of the area by increasing the tidal range and consequently the tidal prism. This led to morphological responses in both the channel and the ebb-tidal delta (Elias and Hansen, 2013). The main channel of the ebb-tidal delta switched southward and developed into two southerly directed channels: Schulpengat and Nieuwe Schulpengat, whereas the delta extended towards the south and north (Elias and Van der Spek, 2006).

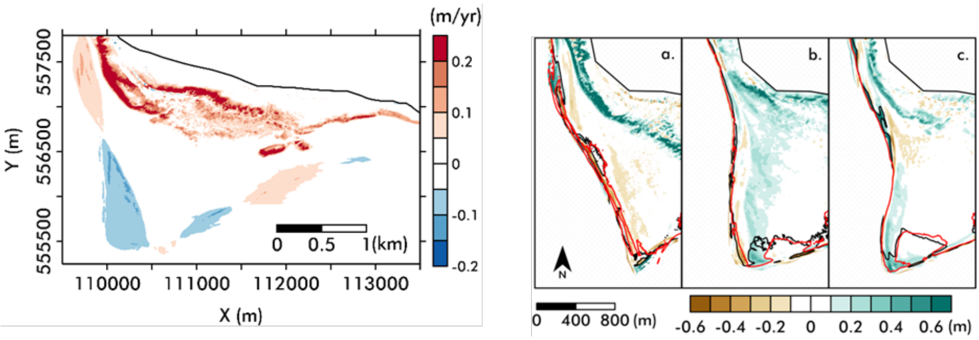

Figure 2.5. Average rate of change between 1997-2015, highlighting subtidal patterns of accretion and erosion, as well as dune growth on the northern end of the sand flat (left) with examples of yearly sand flat elevation changes observed on the west side of the sand flat (right). In each subplot, black and red contours represent the mean spring high tide level (MSHTL) of the first and second year, respectively. Subplot a (2014-2015) shows no deposition above MSHTL, whereas subplots b (2003-2004) and c (2009-2010) highlight deposition onto the flat in regions above MSHTL (Adapted from Wijnberg et.al., 2017)

The sandy shoal Noorderhaaks altered dramatically on the seaward side with the concave southerly directed spit changing to a more northward direction. In contrast, the landward side remained relatively stable due to the high flows and sediment redistribution related to the Molengat channel (Figure 2.4).

In previous centuries, parts of the Noorderhaaks periodically merged with the island extending the southern tip of Texel, De Hors. The closure of the Zuiderzee has affected this process, but recent interaction between Molengat and the adjacent coastline can be seen as an indication of a restoration of this bypassing mechanism. According to Elias and Van der Spek (2017), erosion of the southwest coastline of Texel can be attributed to a lack of sediment bypassing because of spit and channel migration.

The sand flat De Hors exhibits a steady dune growth, with a net accretion of 1.2 x 10^6 m^3 of sand over the last 18 years. The plain is stable in height, with more variation in shoreline movement in the west and the dune growth in the north. The average height of 0.89 (± 0.4) m between the waterline and the dune foot means that it is only flooded during energetic events. Since the east side of the plain is lower than the average height of the rest of the plain, thus being more prone to inundation. Morphologically, the dunes can be categorised in three zones: a western zone, with a high continuous foredune; a central zone, with a field of coppice-like dunes; and the eastern zone, with a continuous foredune that is lower than that in the west. The western zone accounts for around 60% of the total dune growth, followed by the central zone with around 30 percent and the eastern zone with only 10%. The observed spatial variability may be attributed to sediment supply limitations due to high groundwater levels and higher inundation frequency. Wijnberg et. al. (2017) hypothesise that two mechanisms are responsible for linking subtidal and subaerial sediment transfer and determining abundant dune growth regardless of beach plain stability. One mechanism is related to deposition of sand in the intertidal zone and consecutive transport by aeolian processes during lower tide levels. The second is related to deposition of sediment above spring high tide during flooding events. The sediment deposited during storm surge flooding, then is available for aeolian transport during non-energetic periods (Figure 2.5).

2.3.3. Biodiversity

The dynamics of the Wadden system are critical to the ecological functioning of Texel Island and its surroundings. In particular, the hydro- and morphodynamics, and the accompanying gradients in salinity and nutrients within the system, are responsible for mudflat, salt marsh- and dune development (Baptist et al., 2016; Van Puijenbroek et al., 2017) as well as the rich biology and vast diversity of flora and fauna (IJsseldijk et. al., 2015; Hoekendijk et al., 2015). Tidal effects have been observed on primary production, larval distribution and shellfish development (Cadée and Hegeman, 2002; Beukema and Vlas, 1989; Capelle et al., 2017). The area is an important breeding ground for juvenile fish and shellfish, an essential feeding ground for numerous migratory birds, and home to the largest population of seals in the Netherlands (Min. LNV, 2018). The dynamic dune development on SW Texel is rare in the Netherlands. This means that the sandy shoal of De Hors itself, the salt marshes along the Mokbaai just north of De Hors and the Texel Inlet including the dynamic shoal of Noorderhaaks are of high biological importance.

2.4. Evolution of coastal policy in response to new scientific insights

2.4.1. Early policies

For centuries, coastal protection in the Netherlands has been mainly a matter of 'trial and error' (Bijker, 1996). The first written notice in the country Holland dates from 1105 and refers to a Zanddijk (sand dike) near Egmond. The construction of primitive dikes using local materials like sand, clay sods and kelp reinforced by wooden piles, became common practice at locations where dunes were absent or very weak. The planting of marram grass from 1650 on, offered a first opportunity to stabilise and enhance growth of dunes and sand dykes. Up until the second half of the 18th century the use of stones for coastal protection was very rare. In 1776, the first stone groin was constructed at the coast

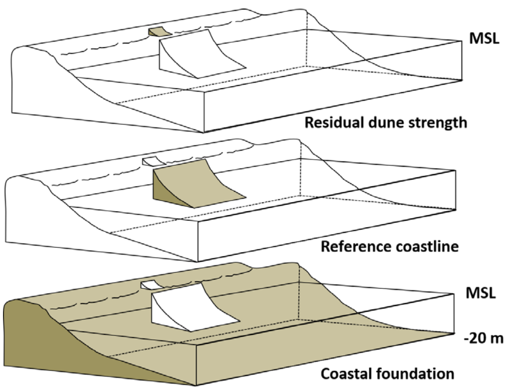

Figure 2.6. Schematic representation of the Dutch coast, indicating management objectives at three different scales (Mulder et al., 2011)

of South Holland, to counteract erosion. More were built until 1890 and more groins followed at the coast of Zeeland, North Holland, Texel (see Figure 2.2) and the island of Vlieland.

It is most likely that originally individual landowners were responsible for the construction and maintenance of dikes. Very soon however, the complicated character of water management led to the development of Water Boards where all inhabitants shared equal responsibility for flood safety. Management of the dunes, for centuries mainly hunting grounds, has been very limited for very long. This changed in the early 19th century when the national government (Rijkswaterstaat, founded in 1798) took increased responsibility. At Texel for instance, the dunes largely became state property. Rijkswaterstaat is responsible for maintenance of the most seaward dunes, and the State Forest Authority (SBB) is responsible for maintenance of the rest of the dune area. SBB plants marram grass and deciduous and pine forests, to prevent dune blow outs and to produce wood (RWS, 1950).

The knowledge and experience of the Water Boards has been documented. The 'Tractaat van Dyckagie' (Discussion on Dikes) by Andries Vierlingh (1507-1579) formed the state-of-the-art at the beginning of the 20th century (Bijker, 1996). The practice of flood protection gradually changed to accommodate learning from large engineering projects like the damming of the Zuiderzee (Afsluitdijk Project) and the inlets of the south-western Netherlands (Delta Project). This led to the establishment of safety standards of flood defence in the Delta Act (1958). However, the maintenance of the sandy coast and dunes as flood defence barriers still rested on experiential knowledge.

2.4.2. Start of a nourishment policy

Things gradually changed around 1965 with the start of a yearly monitoring programme of the Dutch coast, measuring coastal profiles at intervals of 250 m. Between 1985 and 1990,

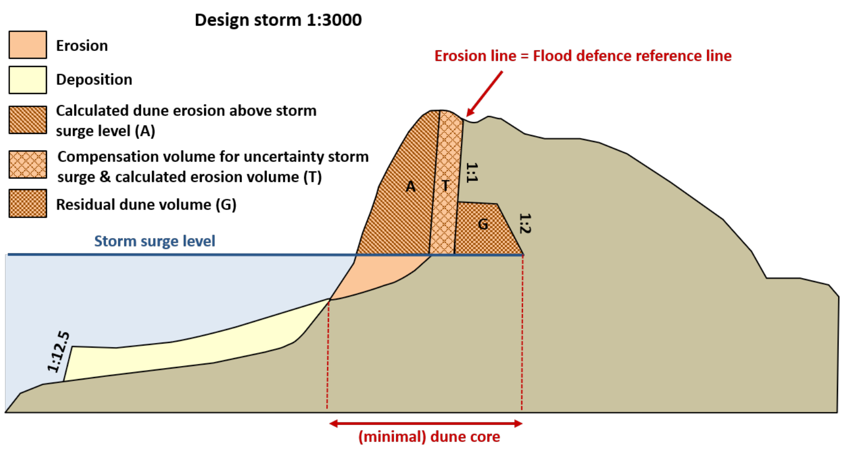

Figure 2.7. Definition sketch of dune strength calculation (after Technische Adviescommissie Waterkeringen [TAW], 2002)

scientific research undertaken as part of the Coastal Genesis Programme (Kustgenese) put forward significant new insights (e.g., Bijker, 1996, van Koningsveld et al., 2003) on flood protection based on a geologically-informed, large scale understanding of coastal processes. The Coastal Genesis Programme represented the first multidisciplinary coastal research programme of the Dutch government, involving engineers, geologists, physical- and historical geographers. For instance, at Texel, hard structures like groins proved to be ineffective in preventing erosion (Verhagen and van Rossum, 1990). The underlying cause of coastal erosion appears to be a structural sand deficit in the wider coastal system.

Following a severe coastal storm in 1990, the government adopted a new coastal policy called 'Dynamic Preservation' (Hermans et al., 2013). This policy identified three different scales to be considered in coastal protection, namely the small-scale residual dune strength ('reststerkte'), the medium-scale 1990 reference coastline ('Basiskustlijn') and the large-scale active coastal system concept ('kustfundament') (Mulder et al., 2011) (Figure 2.6).

The test procedure to determine the actual strength of a dune is based on a model calculation of dune erosion under design conditions, i.e., a storm with a probability of occurrence of 1 in 3000 years (Figure 2.7).

The model provides information to define the geographical characteristics of the dune water defence (the so called 'legger') which is then laid down in a legal document. The position of the flood defence reference line ('waterkering referentielijn') coincides with the erosion line under design conditions. The dune core ('kernzone') is the minimum volume required to meet the standard. In turn, the core comprises three components,

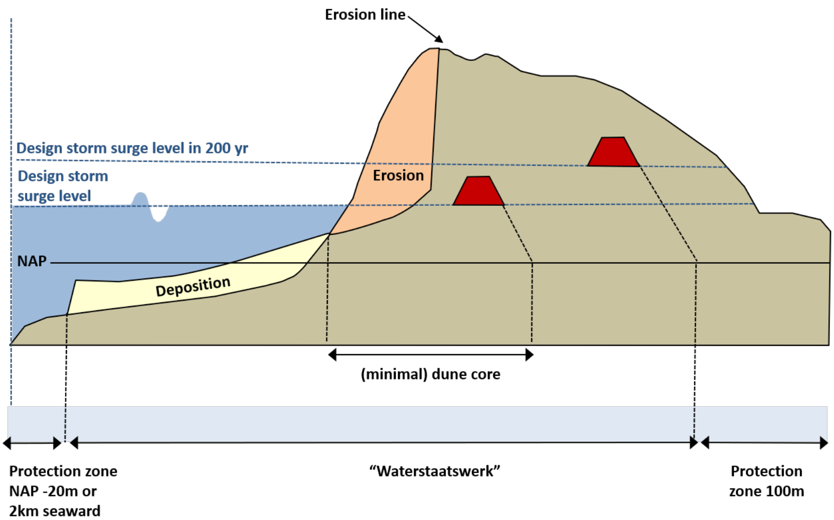

Figure 2.8. The different zones within a dune water defence (TAW, 2002).

namely the erosion zone (A), a small zone to compensate for uncertainties (T), and a zone comprising the required residual dune volume (G) (Figure 2.7). Next, the total spatial extent of the existing dune water defence ('waterstaatswerk') is defined by extending the core zone to include the deposition zone at its seaward side and at its landward side, a reservation zone to account for developments in hydrodynamic conditions over the next 200 years. Finally, summing up all zones that are subject to restrictions in use, two protection zones are defined: landward of the 'waterstaatswerk' a 100 m wide zone, and a seaward zone down to a depth of 20 m or extending 2 km offshore (Figure 2.8).

The dune strength is determined using this procedure every 5 years. The standard for safety against flooding of Texel has been redefined in 2014 as a probability of flooding of 1:3000 (Delta Programme, 2014). The standard applied in coastline management is the position of the coastline in 1990, referred to as the Base Coastline (Basiskustlijn [BKL]), based on principles as depicted in Figure 2.9. It relates the position of the momentary coastline to a volume around the Mean Low Water Level.

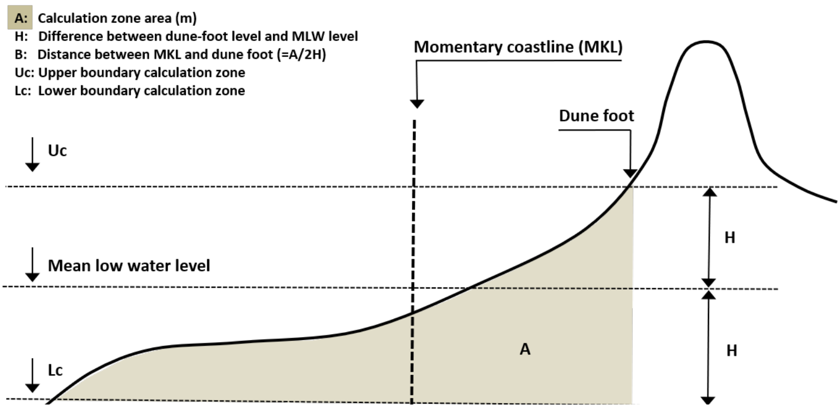

Figure 2.9. Definition sketch of momentary coastline (adapted from TAW, 2002)

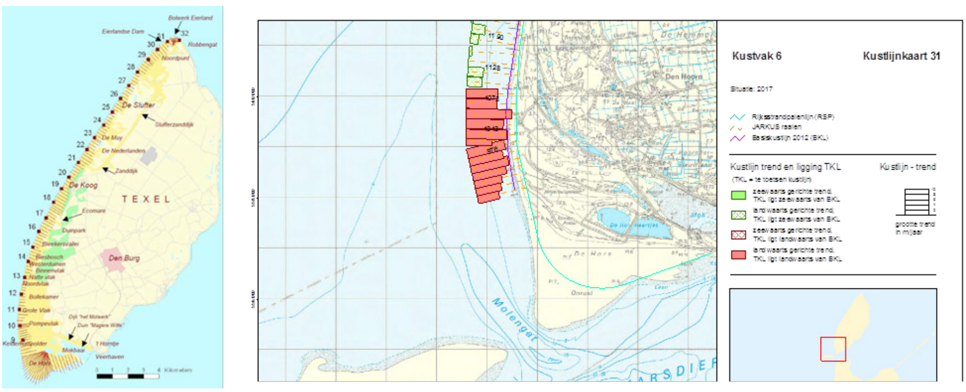

Figure 2.10. Position of all transects on Texel in the JARKUS monitoring programme (left; RWS, 2005). Right: results of coastline testing in 2017, indicating an erosive trend between km 9 and km 11 (right; RWS, 2016)

Based on the realisation that any momentary coastline only represents a snapshot and is not fully representative of the dynamic behaviour of the coastline, the standard 'basiskustijn' (BKL) was defined as the average momentary coastline position derived from its trend over the period 1981 – 1990. The coastline position is determined analytically on a yearly basis by comparing the BKL position with a predicted trend coastline position (TKL) as derived at that time from the momentary coastline trend over the preceding decade (Figure 2.10).

Coastline positions have been established and laid down in an appendix to the Water Law, at 250 m intervals along the entire sandy coast. An exception applies for SW Texel. To allow for natural dynamics at the southern tip of the island (De Hors) no BKL has been defined. The legal obligation to maintain the coastline only applies from Km9 northward (see Figure 2.10). This approach relies upon the yearly monitoring programme JARKUS that has been operational since 1965, and provides bathymetric transect data up to the 5 m contour of transects along the coast at 250 m intervals (Figure 2.10)

In 2000, the total sand volume of the active coastal zone between a depth of 20 m and the dune body massive has been defined as the 'kustfundament' or coastal foundation (Figure 2.6). This concept arose from research showing that maintenance of the BKL would be insufficient to meet the policy objective of sustainably maintaining coastal safety and other dune functions (e.g., Mulder et al., 2011). To comply with the objective of sustainability and maintain the sand volume of the coastal foundation, the yearly total nourishment volume of the Dutch coast was raised from 6 to 12 million m^3, from 2001 onwards.

2.4.3. Current legislative, organisational and social context

Currently, safety standards for all flood defences in the Netherlands, including the dunes, have been established by law (Delta Act, 1958; Flood Defence Act, 1996; Water Act, 2009). The Flood Defence Act and the Water Act define the need to preserve the coastline, in terms of the policy of 'Dynamic Preservation' (MIN V&W, 1990). In addition, natural values of the area, including the tidal inlet, adjacent dunes and sandy shoals such as De Hors, are protected under the European legislation Natura 2000 (Bird- and Habitat Directives apply here), and related national and regional legislation (e.g., Wet Natuurbescherming, 2017). Preservation of flood defences of the sandy coast of Texel involves three governance levels: a) the State or national level, Ministry of Infrastructure & Water management, Rijkswaterstaat (RWS); b) at regional level the province of North Holland and the water board Hoogheemraadschap Hollands Noorderkwartier (HHNK); and c) at local level, the municipality of Texel. In the case of Texel, the state (RWS) is responsible for the design and implementation of nourishments. At regional level, the province is responsible for co-ordination of spatial, economic and nature developments. The regional office of the State Forest Authority (SBB) is responsible for the protection of the natural environment (Figure 2.12). The Water Board (HHNK) is tasked with ensuring coastal safety against flooding. At local level, the municipality of Texel is responsible for maintenance of local infrastructure and economic development.

In implementing the coastal policy, repeated sand nourishments totalling 48 x 10^6 m^3 have been applied along the entire Texel coast between 1990 and 2015, with some 9 x 10^6 m^3

being applied between the transects 9 and 13 in the south-western part (Figure 2.11). The current coastal policy is successful in maintaining the coastline, but consideration of other aspects such as the natural environment and local economic development lags behind (Lubbers et al., 2007; Mulder et al., 2011). Indeed, where in other countries natural protection legislation generally constrains human interventions in the environment that affect protected habitats or species, Dutch legislation offers explicit exceptions for water safety objectives. The vital importance of flood protection in the Netherlands – where 59% of the country is prone to flooding (Pieterse et al., 2009) – means that the natural environment and local economic development often are secondary to the primary objective of flood protection.

Because Texel's dune system and intertidal area have significant ecological value, changes in nourishments and maintenance of the dunes affect the delivery of ecosystem services. The water board tries to align flood protection measures and water quality management actions with nature conservation. Indeed, many stakeholders have an interest in, responsibility for, or are affected by management decisions regarding the Texel coast (Figure 2.12). These include nature managers, tourists, environmental organisations, and people living on Texel. Tourism is a main contributor to the Texel local economy, and Texel's rich nature is what attracts tourist and recreants to the island. Also, visitors and owners of beach restaurants and beach houses have an interest in the width of the south-western Texel beach. A specific stakeholder is the Ministry of Defence, exploiting a training centre and small harbour at the north-eastern fringe of the sandy shoal De Hors. Additionally, the navy Harbour of Den Helder and local fisheries frequently uses the channel Molengat for navigation purposes.

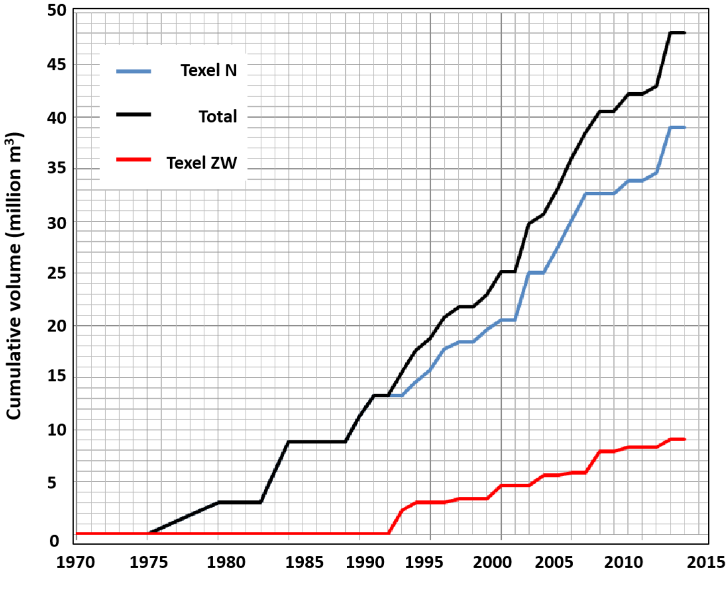

Figure 2.11. Total nourishment amounts (million m³) on Texel between 1990 and 2013, divided between the northern and south western part of the island (after Elias et al., 2014)

2.5. System understanding and insights gained

A structural sand deficit is determining the long-term evolution of the coastline of Texel. Recurring sand nourishments form an effective counter measure, as the coastline position has been maintained and safety against flooding from the sea has been guaranteed over a long period. However, the recurrence interval of nourishments and the associated costs are rather high, recreational beach widths are constantly under pressure and the lack of dynamics in the dune area has led to a deterioration of the natural environment. The latter is in strong contrast to the dynamic and highly valued area of De Hors where there is no legal obligation to maintain the coastline and no nourishments are applied.

New scientific insights shed more light on the relation between shoreline and dune development at De Hors (Section 2.3) and in general, on the link between inlet dynamics and ecological functioning. Present understanding is that the long-term, coastline movement at Texel may be regarded as a component of inlet dynamics (Elias and van der Spek, 2014, Van Heteren et al., 2006). In future, migration of the sandy shoal Noorderhaaks may have a significant effect on shoreline development (Cleveringa, 2001; Van Heteren et. al, 2006; Elias and Van der Spek, 2006). All in all, this challenges existing management approaches to take the (long-term) natural dynamics of the inlet into account. Questions that would need to be addressed include how channel-shoal dynamics in the inlet affect shoreline development, and whether manipulation of inlet dynamics is a feasible alternative to existing nourishment practices. Issues to be considered in addition to the sediment dynamics include flood safety, the status of Texel as part of a UNESCO world heritage area, biodiversity, tourism, navigation and fisheries.

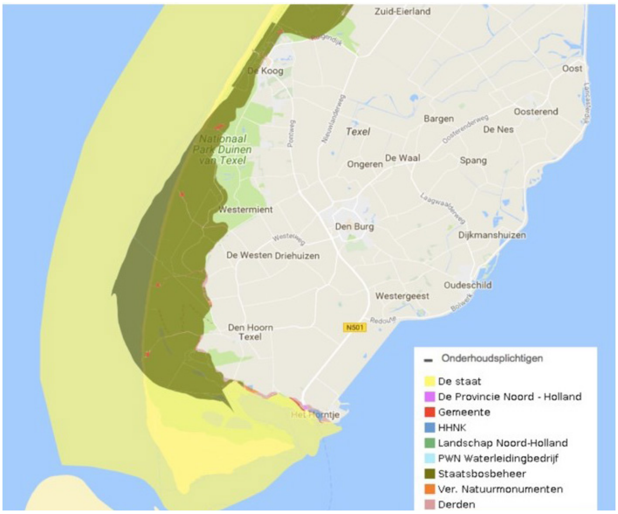

Figure 2.12. Indication of the main actors responsible for maintenance of different parts of SW Texel: state (yellow), municipality (red), state forest authority (dark green). Another main actor, not indicated in the map, is the Water Board, responsible for maintenance of the dune water defence (HHNK, n.d.)

A coastal system is perceived differently by different stakeholders (Costanza et al., 1997; Farber et al., 2002), and the interests of these stakeholders are affected by changing policies and management strategies (Hermans et al., 2013). Accordingly, active stakeholder engagement is considered necessary for effectively managing the environment (Ostrom, 2009). Since the single issue of flood protection has dominated the management of the Texel coast and inlet over time, different types and sources of knowledge (e.g., model-based, technical design knowledge and local community knowledge) will be needed in expanding to address the full range of objectives held by local inhabitants and other relevant stakeholders. The south-western part of Texel provides a wide range of ecosystem services, such as flood protection, biodiversity, recreational opportunities, navigation. As such, the interests of different stakeholders will be affected, highlighting the need for a collaborative exchange of perceptions, knowledge and understanding of the coastal system for the purpose of designing new strategies for managing this part of the Dutch coast.

2.6. Concluding remarks

The present coastal policy is effective in maintaining the coastline and ensuring safety from flooding. It requires frequent and costly sediment nourishments, particularly at south-western Texel. Scientific insights regarding large scale inlet dynamics indicate a future merging of the sandy shoal Noorderhaaks with the south-western tip of Texel. This holds implications for management approaches, implying that they should also take the (long-term) natural dynamics of the inlet into account. Perhaps, nourishments can be replaced by other methods aimed at steering large scale inlet dynamics. Scientific understanding of the sediment dynamics is essential in determining how alternative approaches could affect the natural dynamics of the coastal system. Similarly, governance knowledge is necessary to determine what is possible under existing regulations and what modifications to regulations might be necessary. Critical considerations involve balancing natural sediment dynamics, the long-term effects of climate change on the dynamic Wadden Sea system, and the envisaged human interventions. The south-western part of Texel provides a wide range of ecosystem services, such as flood protection, biodiversity, recreational opportunities, navigation, all of which dependent to some extent on the sediment cycle. The interests of different stakeholders will be affected by any changes in management approach. As such, the need for collaborative, participatory approaches in designing alternative multi-functional coastal management strategies into the future becomes apparent.

2.7. References

Baptist, M. J., de Groot, A. V., & van Duin, W. E. (2016). Contrasting biogeomorphic processes affecting salt-marsh development of the Mokbaai, Texel, The Netherlands. *Earth Surface Processes and Landforms*, 41(9), 1241–1249. https://doi.org/10.1002/esp.3949

Beukema, J., & de Vlas, J. (1989). Tidal-current transport of thread-drifting postlarval juveniles of the bivalve Macoma balthica from the Wadden Sea to the North Sea. *Marine Ecology Progress Series*, 52, 193–200. https://doi.org/10.3354/meps052193

Bijker, E. W. (1996). History and Heritage in Coastal Engineering in the Netherlands (pp. 390–412). American Society of Civil Engineers (ASCE). https://doi.org/10.1061/9780784401965.010

Buijsman, M. C., & Ridderinkhof, H. (2007). Water transport at subtidal frequencies in the Marsdiep inlet. *Journal of Sea Research*, 58(4), 255–268. https://doi.org/10.1016/j.seares.2007.04.002

Cadée, G. C., & Hegeman, J. (2002). Phytoplankton in the Marsdiep at the end of the 20th century; 30 years monitoring biomass, primary production, and Phaeocystis blooms. Journal of Sea Research, 48(2), 97–110. https://doi.org/10.1016/S1385-1101(02)00161-2

Capelle, J. J., van Stralen, M. R., Wijsman, J. W. M., Herman, P. M. J., & Smaal, A. C. (2017). Population dynamics of subtidal blue mussels Mytilus edulis and the impact of cultivation. *Aquaculture Environment Interactions*, 9(1), 155–168. https://doi.org/10.3354/aei00221

Cleveringa, J. (2001). *Zand voor zuidwest Texel: Technisch advies RIKZ over vier mogelijke ingrepen in het Zeegat van Texel (*Report RIKZ/OS/2001/031) [in Dutch]. The Hague: Ministry of Transport and Public Works.

Costanza, R., D'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., … Paruelo, J. (1997). The value of the world's ecosystem services and natural capital. *Nature*, 387(6630), 253–260.

Delta Programme Commissioner. (2014). *Delta Programme*. Retrieved from https://english.deltacommissaris.nl/delta-programme/delta-decisions/water-safety-delta-decision

Delta wet [Delta Act] (1958). The Hague: Ministerie van Infrastructuur en Milieu (Ministry of Infrastructure and the Environment).

Duran-Matute, M., Gerkema, T., de Boer, G. J., Nauw, J. J., & Gräwe, U. (2014). Residual circulation and freshwater transport in the Dutch Wadden Sea: a numerical modelling study. *Ocean Science*, 10(4), 611–632. https://doi.org/10.5194/os-10-611-2014

Elias, E. P. L., & Hansen, J. E. (2013). Understanding processes controlling sediment transports at the mouth of a highly energetic inlet system (San Francisco Bay, CA). *Marine Geology*, 345, 207–220. https://doi.org/10.1016/j.margeo.2012.07.003

Elias, E. P. L., & van der Spek, A. J. F. (2006). Long-term morphodynamic evolution of Texel Inlet and its ebb-tidal delta (The Netherlands). *Marine Geology*, 225(1–4), 5–21. https://doi.org/10.1016/j.margeo.2005.09.008

Elias, E. P. L., & van der Spek, A. J. F. (2017). Dynamic preservation of Texel Inlet, the Netherlands: Understanding the interaction of an ebb-tidal delta with its adjacent coast. *Geologie En Mijnbouw/Netherlands Journal of Geosciences*, 96(4), 293–317. https://doi.org/10.1017/njg.2017.34

Elias, E. P. L., van Oeveren, C., & Breuns, A. (2014). *Beheerbibliotheek Texel; Feiten en cijfers ter ondersteuning van de jaarlijkse toetsing van de kustlijn (Deltares report 1209381-007)* [in Dutch]. Deltares. Retrieved from http://publications.deltares.nl/1209381_007a.pdf

Farber, S. C., Costanza, R., & Wilson, M. A. (2002). Economic and ecological concepts for valuing ecosystem services. *Ecological Economics*, 41(3), 375–392. https://doi.

org/10.1016/S0921-8009(02)00088-5

Hayes, M. O. (1979). Barrier island morphology as a function of tidal and wave regime. In *S. P. Leatherman (Ed.), Barrier Islands: from the Gulf of St. Lawrence to the Gulf of Mexico.* (pp. 1–27). New York: Academic Press.

Hermans, L. M., Slinger, J. H., & Cunningham, S. W. (2013). The use of monitoring information in policy-oriented learning: Insights from two cases in coastal management. *Environmental Science and Policy*, 29, 24–36. https://doi.org/10.1016/j.envsci.2013.02.001

Hoekendijk, J. P. A., de Vries, J., van der Bolt, K., Greinert, J., Brasseur, S., Camphuysen, K. C. J., & Aarts, G. (2015). Estimating the spatial position of marine mammals based on digital camera recordings. *Ecology and Evolution*, 5(3), 578–589. https://doi.org/10.1002/ece3.1353

Hoogheemraadschap Hollands Noorderkwartier. (n.d.). *Legger zandige kust txl*. Retrieved December 8, 2019, from https://hhnk.webgispublisher.nl/Viewer.aspx?map=Legger-zandige-kust-txl

Hoogheemraadschap Hollands Noorderkwartier. (2008). *Legger van de primaire waterkeringen* [in Dutch]. Hoogheemraadschap Hollands Noorderkwartier, Edam. Retrieved from https://www.hhnk.nl/document.php?m=1&fileid=19324&f=e7505409d85da6bf9b8fdf172c6d6be1&attachment=0

IJsseldijk, L. L., Camphuysen, K. C. J., Nauw, J. J., & Aarts, G. (2015). Going with the flow: Tidal influence on the occurrence of the harbour porpoise (Phocoena phocoena) in the Marsdiep area, The Netherlands. *Journal of Sea Research*, 103, 129–137. https://doi.org/10.1016/j.seares.2015.07.010

Kustverdediging na 1990; Beleidskeuze voor de kustlijnzorg [First Coastal Policy Document] (1990). The Hague: Ministerie van Transport and Public Works (Ministry of Transport and Public Works).

Lubbers, B., De Heer, J., Groenendijk, J., van Bockel, M., Blekemolen, M., Lambeek, J., & Steijn, R. (2007). Evaluatie Derde Kustnota [in Dutch]. Amersfoort. Retrieved from http://publicaties.minienm.nl/documenten/evaluatie-derde-kustnota

Ministerie van Landbouw Natuur en Voedselkwaliteit. (2018). Beschermde natuur in Nederland; soorten en gebieden in wetgeving en beleid (Nature protection in the Netherlands; varieties and regions in law and policy) [in Dutch]. Retrieved from https://www.synbiosys.alterra.nl/natura2000/

Mulder, J. P. M., Hommes, S., & Horstman, E. M. (2011). Implementation of coastal erosion management in the Netherlands. *Ocean and Coastal Management*, 54(12), 888–897. https://doi.org/10.1016/j.ocecoaman.2011.06.009

Ostrom, E. (2009). A general framework for analyzing sustainability of social-ecological systems. *Science*, 325(5939), 419–422. https://doi.org/10.1126/science.1172133

Pieterse, N., Knoop, J., Nabielek, K., Pols, L., Tennekes, J., & Deltares. (2009). Overstromingsrisicozonering in Nederland. Hoe in de ruimtelijke ordening met overstromingsrisico's kan worden omgegaan [in Dutch]. Den Haag/Bilthoven. https://doi.org/978-90-78645-30-6

Rijkswaterstaat. (1950). *Beschrijving van de provincie Noord Holland behorende bij de* *waterstaatskaart* [in Dutch]. Staatsdrukkerij Den Haag.

Rijkswaterstaat. (2005). *Toetsing Waterkering Texel 2005* – Hoofdrapport [in Dutch]. Rijkswaterstaat Noord Holland.

Rijkswaterstaat. (2016). Kustlijnkaarten 2017 [in Dutch]. Rijkswaterstaat, Min. Infrastructuur & Milieu. Retrieved from publicaties.minienm.nl/download-bijlage/85791/kustlijnkaarten-2017.pdf

Sha, L. P. (1989). Sand transport patterns in the ebb-tidal delta off Texel Inlet, Wadden Sea, The Netherlands. *Marine Geology*, 86(2–3), 137–154. https://doi.org/10.1016/0025-3227(89)90046-7

Technische Adviescommissie Waterkeringen (Technical Advisory Committee on Water Defences). (2002). Leidraad Zandige Kust. Delft, December 2002, DWW-2003-046, ISBN90-369-5541-6, 223 p.

van der Vegt, M., & Hoekstra, P. (2012). Morphodynamics of a storm-dominated, shallow tidal inlet: The Slufter, the Netherlands. Geologie En Mijnbouw/Netherlands *Journal of Geosciences*, 91(3), 325–339. https://doi.org/10.1017/S0016774600000470

van Heteren, S., Oost, A. P., van der Spek, A. J. F., & Elias, E. P. L. (2006). Island-terminus evolution related to changing ebb-tidal-delta configuration: Texel, The Netherlands. *Marine Geology,* 235(1-4 SPEC. ISS.), 19–33. https://doi.org/10.1016/j.margeo.2006.10.002

Van Koningsveld, M., De Vriend, H. J., Ruessink, B. G., Stive, M. J. F., Mulder, J. P. M., & Dunsbergen, D. W. (2003). Usefulness and effectiveness of coastal research: A matter of perception? *Journal of Coastal Research*, 19(2), 441–461. https://doi.org/10.2307/4299184

van Puijenbroek, M. E. B., Teichmann, C., Meijdam, N., Oliveras, I., Berendse, F., & Limpens, J. (2017). Does salt stress constrain spatial distribution of dune building grasses Ammophila arenaria and Elytrichia juncea on the beach? *Ecology and Evolution*, 7(18), 7290–7303. https://doi.org/10.1002/ece3.3244

Verhagen, H. J., & van Rossum, H. (1989). Strandhoofden en paalrijen: evaluatie van hun werking (Discussienota "Kustverdediging na 1990") [in Dutch]. The Hague. Retrieved from https://library.wur.nl/WebQuery/hydrotheek/513953

Waterwet [Water Act] (2009). The Hague: Ministerie van Infrastructuur en Milieu (Ministry of Infrastructure and the Environment). Retrieved from https://wetten.overheid.nl/BWBR0025458/2018-07-01

Wet Natuurbescherming [Nature Conservation Act] (2017). The Hague: Ministrie van Landbouw, Natuur en Voedselkwaliteit (Minister of Agriculture, Nature and Food Quality). Retrieved from https://wetten.overheid.nl/BWBR0037552/2019-01-01

Wet op de Waterkering. Gewijzigd Voorstel van Wet [Flood Defence Act] (1996). The Hague: Ministry of Transport, Public Works and Water Management.

Wijnberg, K. M., van der Spek, A. J. F., Silva, F. G., Elias, E., Wegen, M. van der, & Slinger, J. H. (2017). Connecting subtidal and subaerial sand transport pathways in the Texel inlet system. *Coastal Dynamics Proceedings*, (235), 323–332.

3

Integrated Coastal and Catchment Management of Dundalk Bay

By Suzanne Linnane, Alec Rolston and Declan MacGabhann

3.1. Motivation for interest and approach

A shift in integrated management of the environment of Dundalk Bay in Ireland is being initiated through the development of a community-led process. Ireland's implementation of the second cycle of the Water Framework Directive (Directive 2000/60/EC, 2000) has placed increased emphasis on Integrated Catchment Management as a mechanism for the management of its water resources (Department of Housing, Planning, Local Community and Government [DHPLCG], 2017). As water flows across jurisdictional boundaries into the sea and, given the significance of the catchment as the geographical unit of water management, increased collaboration and integrated coordination will be required to ensure the delivery of actions. This includes not just communication between scientists, statutory agencies and policy makers, but the introduction of 'real' and sustained engagement with local communities and other interested stakeholders - in partnership and with a support network comprising scientists, relevant government agencies and others. Recent research has identified that although the majority of people wish to have a voice in local water management issues; most do not feel included in decisions regarding their local water environment. This is despite communities strongly valuing their local waters. Therefore an opportunity exists to facilitate stronger connections between local communities and their water environment, fostering bottom-up initiatives that empower and give ownership of local water management issues to these communities (Rolston et al., 2017). For this reason, a new community-led partnership, the Dundalk Bay Rivers Partnership is in the process of being established in the region, being based on the UK Rivers Trust model (The Rivers Trust, n.d.). The aim of the partnership is to:

- Increase awareness of catchment management actions undertaken at the local scale,
- Increase community involvement in such management actions,
- Increase communication between governing agencies and communities through increased transparency, increased stakeholder engagement and the provision of opportunity for feedback and interaction at the local and regional scale.

At the catchment scale, the importance of local rivers flowing into Dundalk Bay (with a catchment area of over 1,000 km^2) is being highlighted. Although in its infancy, the Partnership has engaged multiple stakeholders to develop a series of community-led visions for the rivers that flow in to Dundalk Bay. The aim is to assemble priority community-identified themes which can subsequently be used to develop funding bids, local delivery projects and to identify management actions which can be implemented to achieve the community visions.

The key management drivers acting within Dundalk Bay are principally its port, its fisheries, the WFD and its Natura 2000 status, although active management for the latter is limited due to resource restrictions. Unfortunately, the only WFD High Status River flowing in to Dundalk Bay has been lost since the previous WFD cycle, meaning that is more crucial than ever that locally-led schemes are launched to assist in addressing land management practices which are impacting on river and estuarine quality in the locality. Despite its frequent recreational use (with public walkways and beaches featuring strongly), the majority of inhabitants of Dundalk Bay and its surrounds could be unaware of the site's environmental and economic importance. Indeed, despite an increased focus on integrated catchment management processes in water management in Ireland, the initial development of the aforementioned Rivers Partnership failed to include Dundalk Bay and its estuaries, instead focussing solely on the freshwater regions of the catchment. This case study adopts a holistic, integrated environmental management view in positioning current efforts towards a new community-led partnership, the Dundalk Bay Rivers Partnership.

3.2. Study area

Dundalk Bay is a large bay located on the north-east coast of Ireland, spanning some 16km from the Cooley Peninsula in the north to Annagassan and Dunany Point in the south (Figure 3.1). While the Bay encompasses a number of habitat types, there are extensive saltmarshes and stretches of inter-tidal areas exposed at low water. The inner bay

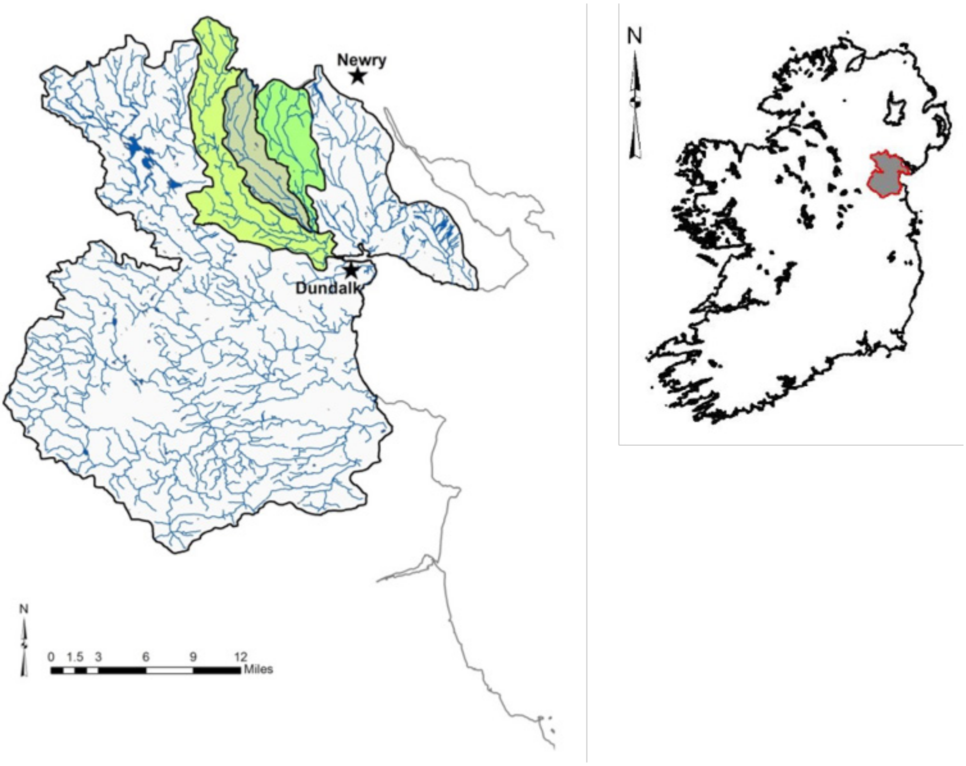

Figure 3.1. Location of Dundalk Bay and its wider catchment, located on the north-east coast of Ireland - shaded areas represent those areas where engagement work is already ongoing (adapted from Veerkamp & Rolston, 2017)

(33.35 km^2) is shallow, sandy and intertidal with up to 70% of its surface exposed during low tide. The hydraulics of the Bay are dominated by the sea but the Bay encompasses the mouths and estuaries of the Rivers Dee, Glyde, Fane, Castletown and Flurry. The Bay is designated under the EU Habitats Directive as a Special Area of Conservation (SAC) selected for the following habitats and/or species listed on Annex I/II of the EU Habitats Directive (Council Directive 92/43/EEC, 1992) such as perennial vegetation of stony banks, tidal mudflats and sandflats, Atlantic and Mediterranean salt meadows, Salicornia mud, and estuaries. These widespread mud and sandflats have a rich fauna providing an important food source to the tens of thousands of waterfowl inhabiting the Bay (National Parks and Wildlife Service [NPWS], 2014).

In addition, Dundalk Bay is designated a Special Protection Area (SPA) under the EU Birds Directive (Directive 2009/147/EC, 2010), and is listed as a Wetland of International Importance under the Ramsar Convention.

3.3. Natural dynamics of bay

3.3.1. Abiotic system characteristics

The tidal range at Dundalk is relatively large for Irish waters with a maximum spring range of 4.7 m and a mean neap range of 2.6 m (Marine Irish Digital Atlas [MIDA],

Figure 3.3. Dundalk Bay Navigation Channel into the Castletown Estuary (Source: Ordinance Survey Ireland)

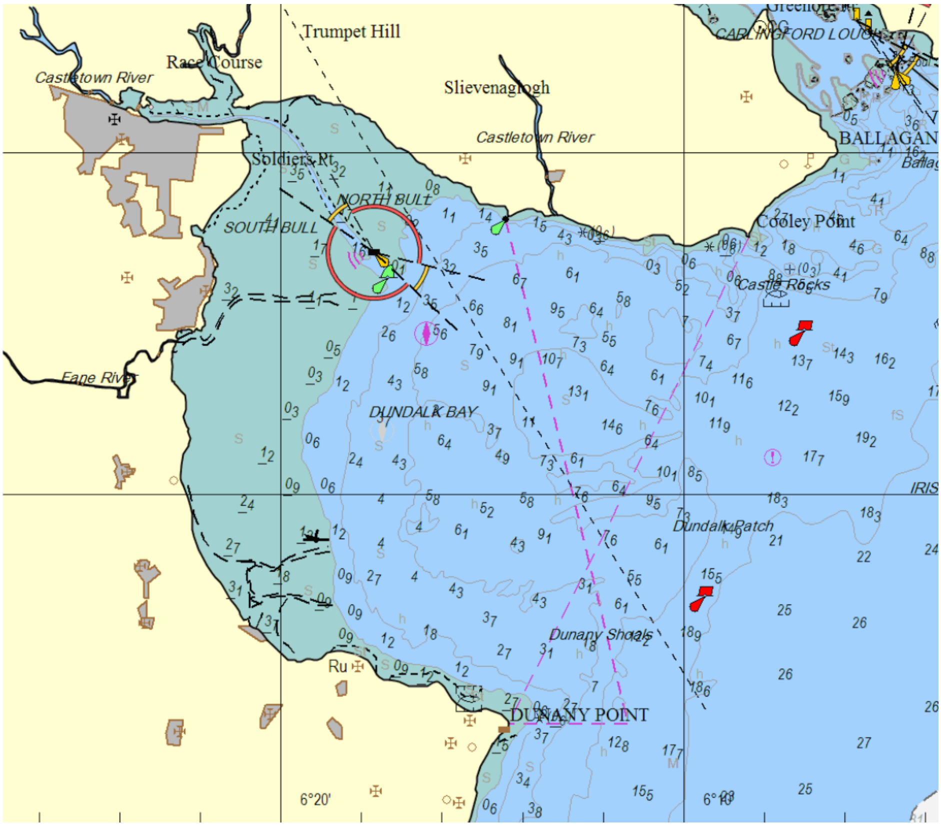

Figure 3.2. Dundalk Bay and approaches to Dundalk (Source: Ordinance Survey Ireland)

2017). The Bay is exposed to waves generated in the Irish Sea from an east north east to south east direction. The Castletown River cuts through the intertidal zone in the north-west corner of the bay and provides a channel which has been used by boats to access the Harbour for many years. Figures 3.2 and 3.3 show the location of this channel within Dundalk Bay.

The National Ports Policy recognises the importance of smaller ports around the country such as Dundalk Harbour and how they fulfil an important role in the local economy. In fact, Dundalk Harbour is listed in the National Ports Policy as a port of regional significance. The National Ports Policy states that "while commercial shipping in Ireland is centred on the five Ports of National Significance, 14 other ports [including Dundalk Harbour] handle commercial traffic and function as important facilitators of trade for their regional and local hinterland".

Dundalk Bay itself is of environmental, social and economic significance at both local and national scales. The cockle and razor fishing industry alone has been estimated as worth approximately €3 million per annum to the local economy and there is currently a proposal to extend the fishing area to 78km^2. At present, the fishing area does not extend more than approximately 20-25km^2.

The region is of national cultural importance given its multiple pre-historical archaeological sites (e.g., Proleek Dolmen) and has strong national folklore importance as the birthplace of mythological hero warrior Cú Chullain. The rivers around Dundalk Bay were the roadways of the Ancient East leaving behind an enviable heritage legacy from the Viking forts and the only castle in Ireland built by a woman - Castle Roche - to the land of legends and Irish chieftains.

The Bay's importance as a Natura 2000 site and a Ramsar site along with its historical significance presents tourism and recreational opportunities, particularly with regard to birdwatching. A national integrated marine mapping programme, Infomar (Geological Survey Ireland & the Marine Institute, n.d.), undertook a bathymetric survey of Dundalk Bay, including the navigation channel of the Castletown Estuary (Figure 3.4).

3.3.2. Biotic system characteristics

Saltmarsh vegetation occurs in four main areas with two types represented - Atlantic and Mediterranean salt meadows. Shingle beaches are particularly well represented and support a wide variety of flora including species such as Spear-leaved Orache (Atriplex prostrata), Sea Mayweed (Matricaria maritima), Sea Beet (Beta vulgaris subsp. maritima), Sea Rocket (Cakile maritima), Wild Carrot (Daucus carota), Sea-holly (Eryngium maritimum), Sea Sandwort (Honkenya peploides) and Sea Radish (Raphanus raphanistrum subsp. maritimus) (NPWS, 2011).

Vast sandflats and mudflats occur over 4,000ha, with ecological communities such as muddy fine sand communities and fine sand community complexes well represented. These habitats host a rich fauna of bivalves, molluscs, marine worms and crustaceans and are the main food resource of the tens of thousands of waterfowl (including waders and gulls) which feed in the intertidal area of Dundalk Bay. The saltmarshes are used as high-tide roosts by all of these species, while the grazing birds (particularly Brent Goose

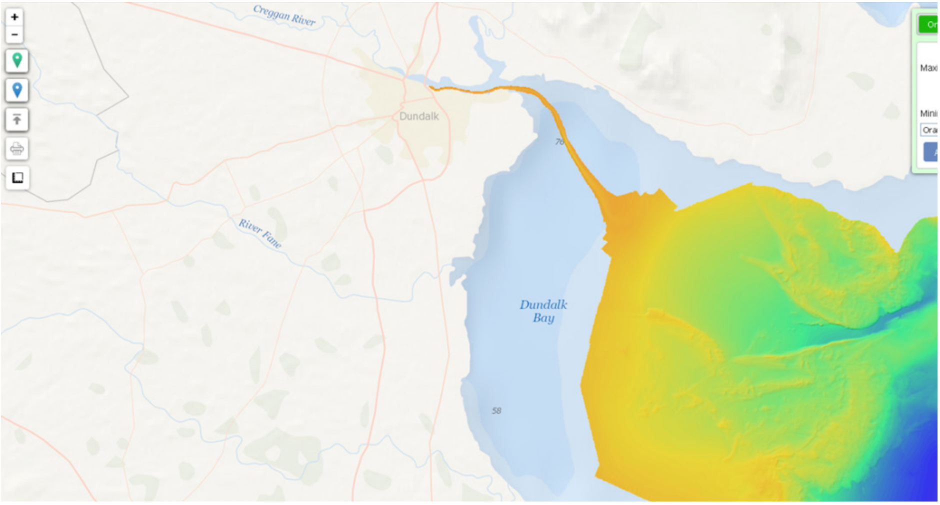

Figure 3.4. Bathymetric survey of Dundalk Bay undertaken by the Infomar Project (Geological Survey Ireland & the Marine Institute, n.d.)

and Wigeon) feed on the salt marsh grasses, sea grass and other grassland vegetation. The site is internationally important for waterfowl because it regularly holds over 20,000 birds (up to 57,000 have been recorded) and supports over 1% of the North-West European/East Atlantic Flyway populations of Brent Goose, Bartailed Godwit and Knot. Additionally, it is nationally important for Golden Plover, Great Crested Grebe, Greylag Goose, Shelduck, Mallard and many more (NPWS, 2014).

Dundalk Bay was surveyed in 2009 as part of the fish monitoring programme for the WFD (Kelly et al., 2010). A total of 16 fish species were recorded in Inner Dundalk Bay in September 2009. Sprat was the most abundant species captured, followed by cod, plaice and founder. Dundalk Bay has been included in Irish Salmon, Celtic Sea Trout and Eel surveys in recent years (Celtic Sea Trout Project [CSTP], 2016; Standing Scientific Committee on Salmon [SSCS], 2017; Standing Scientific Community for Eel [SSCE], 2015) and is part of Inland Fisheries Ireland's Marine Sportfish tagging programme (Clarke et al., 2016). In addition to this, Dundalk Bay was surveyed on a seasonal basis over a 5-year period (2007-2011) looking at the juvenile flatfish species which inhabit the intertidal areas (MacGabhann, unpublished). The Centre for Freshwater and Environmental Studies (CFES) at Dundalk Institute of Technology (DkIT) have also carried out fisheries surveys over the past three years (2014-2016) to supplement this data and have found flatfish to be abundant in shallow regions (various unpublished undergraduate theses). These shallow waters of the Bay are a vital nursery area for all the commercial fish species in the Irish Sea and the Bay is anecdotally considered to be the most important nursery area along the East coast of Ireland.

Furthermore, the Inner Bay receives the waters from the Castletown and Flurry Rivers. These rivers contain a number of important species which migrate through the Bay, including salmon, sea trout, eels and sea lamprey.

Figure 3.5. Examples of fishing fleet and surrounding landscape of Dundalk Bay (Pictures: Declan MacGabhann)

3.3.3. Human use

The major industrial uses of Dundalk Bay are the port and the fisheries (Figure 3.5). Leisure boats are unable to use the harbour for berthing as facilities do not exist at the existing piers for leisure craft to safely moor and for people to disembark easily.

A number of fishing boats, particularly small trawlers, moor at the quays in Dundalk on a permanent basis (n=16). These boats are involved in shellfish fishing in Dundalk Bay. The fishermen are organised and represented at both local and national levels in the form of Local, Regional and National Inshore Fisheries Forums where they can feed directly into policy making decisions affecting Dundalk Bay. Feedback suggests that there is an amicable relationship between the fishery interests and commercial interests using Dundalk Harbour with both industries progressing without hampering or interfering with the other's operations.

Dundalk Bay is a classified shellfish production area for both razor clams (Ensis siliqua) and cockles (Cerastoderma edule) with the razor fishing taking place in the deeper waters and the cockles closer to the shoreline (Marine Institute, 2016).

The entire bay is designated under the quality of shellfish waters regulations (Statutory Instrument Number 269 0f 2006 European Communities (Quality of Shellfish Waters) Regulations 2006 and its subsequent amendments). The current shellfish classification for razor clams allows the shellfish to be placed directly on the Asian, North American and European markets. The deeper waters outside of the Bay are very important in the Nephrops (Dublin Bay Prawn) fishery with up to 50 vessels operating on a seasonal basis. The cockle fishery has approximately 32 permitted vessels. Annual landings for the Razor fishery (approximately 81 vessels) is 300-700 Mt, but not all operate on a full time basis. The shrimp fishery comprises approximately 6-8 vessels with annual landings of 250-500+ Mt, and 6 - 10 vessels operate on a full-time basis for brown crab.

The main items imported into Dundalk Harbour are coal, timber and animal feed. Bord na Mona (an Irish semi state company) operate a large coal importing business at the Harbour and import loose coal which they then pack onto pallets for the Dublin market. Bord na Mona imports in the region of 35,000 - 40,000 tonnes per year of coal into Dundalk Harbour.

In addition to commercial fishery, the wider catchment of Dundalk Bay is host to a large number of angling clubs and recreational fishing is an important leisure activity undertaken within each of the estuaries located within Dundalk Bay. Walking, swimming and beach use are also popular pursuits.

The catchment around the bay is of mixed agriculture and urban land use.

3.4. Human interventions

Maintenance dredging of the Castletown estuary navigation channel is undertaken annually under the management of the Port Authority in order to remove sediment to ensure the safe navigation of vessels entering/leaving the port. The types of vessels which can access the Harbour are naturally restricted by the draught and in recent years, this has become even more restrictive as the navigation channel is becoming shallower despite dredging efforts.

Without these operations, the accumulation of sand particularly in the area of Soldier's Point (marked in red on Figure 3.2) causes difficulties for ships of certain draught entering and exiting the harbour. The navigation channel for Dundalk is currently maintained at a depth of -0.75 m CD. The berthing pockets at Dundalk Harbour are maintained at -0.1 m CD. Limited available historical surveys suggest that regular maintenance dredging will be required if the depths in the channel are to be maintained at a figure of -0.75 m CD. The amount and frequency of maintenance dredging required to keep the channel at -0.75 m CD depends upon prevailing weather conditions. According to a review of available historical data, bathymetric and sediment surveys, and modelling simulations undertaken prior to the latest foreshore application for maintenance dredging on behalf of the Dublin Port Company (RPS, 2011), a series of strong to gale south easterly winds could reduce the channel depth by 0.4 m in a relatively short period of time. The report also stated that in the past dredging has been poorly controlled and has resulted in humps and hollows in the sea bed rather than dredging to a consistent and required depth. The sediment analysis undertaken showed the material on the banks on either side of the channel to be mainly fine sand with a Dn50 size of 0.125 mm.

Dredging cannot be carried out during an annual specified 'closed period' between March and May to facilitate downstream migration of smolts of Atlantic salmon.

Generally, flood risk is not high in Dundalk Bay region and dredging is specifically carried out for navigational purposes rather than flood mitigation. As part of the recent national flood risk assessments, areas of Dundalk Bay have been identified as 'at risk' of flooding during either extreme, 200 or 100-year events (The Office of Public Works [OPW], 2018).

To assist in the integrated management of Ireland's coastal systems generally, the Irish Government has developed a web portal called FishingNET (Department of Agriculture, Food & the Marine, Ireland, n.d.) where it places all information pertaining to appropriate assessments for the Bay, its management plans, Natura 2000 determinations and vessel permits. From the food safety perspective, the Sea Fisheries Protection Authority (SFPA) has responsibility for shellfish production areas. The Marine Institute (MI) records harmful algal blooms in Dundalk Bay which may prevent the harvesting of shellfish and the Environmental Protection Agency (EPA) is responsible for Dumping at Sea permits which the port get for dumping the dredged spoil.

3.5. Social context

Whereas the primary responsibility with reporting on EU legislation requirements rests with the DHPCLG, several statutory agencies are involved in formulating and implementing strategies in this area including the Department of Agriculture, Food and the Marine which holds key management interests in the region.

National Agencies charged with implementing the management actions required for achieving relevant EU legislation and other international responsibilities (e.g., Ramsar) include the EPA and the NPWS. The SFPA Marine Institute (MI) and Bord Iascaigh Mhara (BIM) are responsible for the Natura 2000 fishing assessments. The SFPA are involved directly in the bi-annual classification of the shellfish beds through the national Molluscan Shellfish Safety Committee (MSSC). From a wider management perspective, the deterioration in water quality throughout the catchment from multiple stressors cannot be ignored.

From a stakeholder perspective, the newly formed Local Authorities Water and Community Office (LAWCO) are responsible nationally for water-related public engagement and have been driving the development of the Dundalk Bay Rivers Partnership (The Local Authority Waters Programme, n.d.). The Dundalk Bay Rivers Partnership is currently in the process of forming following initial public engagement activities which have developed a series of community-led visions for the rivers which flow into Dundalk Bay.

The fishermen in the catchment are well organised and as such there are two dedicated fishing association groups for Dundalk Bay - the North Irish Sea Razor Fishermen's' Association (NISRFA) and the Dundalk Cockle Fishermen's' Group (DCFG). The NISRFA feed directly into the National Inshore Fishermen Forum (NIFF) and also have representatives on the Regional Inshore Fishermen's Forums (RIFF) on all aspects relating to the sustainable exploitation of the razor clams in the bay. Coordination and collection of monthly shellfish sampling is undertaken by the association for the bi-annual classification of the shellfish beds.

The Cockle group, in conjunction with the MI, BIM and the SFPA, coordinate annual cockle surveys as part of the system set up for exploiting the cockle fishing under the appropriate assessment management plans. Sample coordination and collection is also undertaken by the group in partnership with the MI and SFPA. The survey is usually a week-long event and estimates the total biomass of the stock which then splits into three – one third left available for the bird stocks, a third to maintain the future stock and a third to be fished.

Voluntary agencies also play an important role as stakeholders in Dundalk Bay. These include various non-governmental organisations (NGOs) including Birdwatch Ireland and Coast Watch who carry out yearly surveys on behalf of the NPWS, as well as local interest groups including local angling groups.

The local agricultural community is a key stakeholder given the requirements of the WFD and the current status of the Dundalk Bay waters.

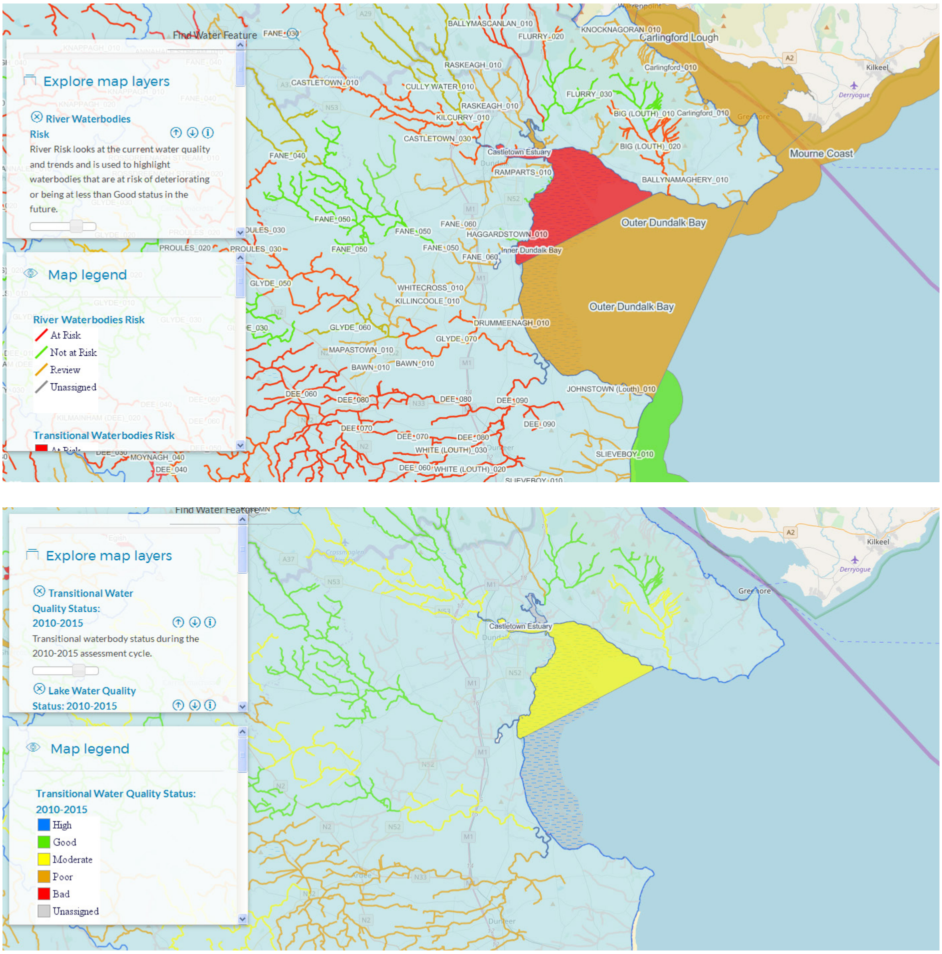

Figure 3.6. Above: current status of water bodies entering Dundalk Bay (note absence of high quality sites) (EPA Ireland, n.d.). Below: waterbodies in Dundalk Bay 'at risk' of not meeting their targets according to the WFD classification (EPA Ireland, n.d.)

The CFES at DkIT has an academic interest in the environmental, social and economic aspects of Dundalk Bay and has been involved in environmental assessment, integrated environmental management, and catchment-scale initiatives in the region for over a decade.

3.6. System understanding and insights gained

The integrated management of Dundalk Bay is complex given its ecological status under the EU Habitats Directive and Ramsar, in addition to being a commercial fishery and working port of significant regional economic importance. Despite sound environmental assessment activities over many years and multiple stakeholder involvement, there is limited integrated information available in relation to the dynamics of the system as a whole, particularly the management of the estuarine environment.

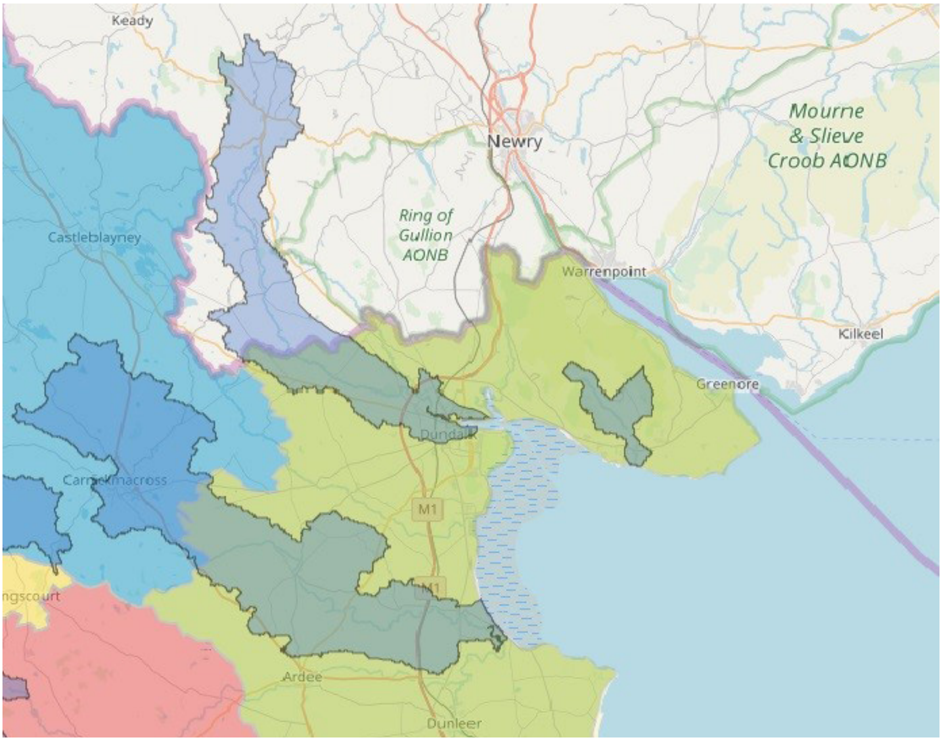

Figure 3.7. Priority areas identified in Dundalk Bay as part of Ireland's second River Basin Management Plan (areas shaded in blue-grey) (Environmental Protection Agency of Ireland, n.d.)

Although areas of the Bay have been identified as being at risk from flooding during extreme, 200 and 100-year events (OPW, 2018), dredging is carried out only on a maintenance basis as required for navigational purposes into the port. From an ecological and WFD status perspective, the loss of the only WFD High Status water body in the wider Dundalk Bay catchment is of concern. This, coupled with deteriorating water quality entering the Bay from its freshwater components, identifies catchment-scale management issues that cannot be ignored, and which are likely to drive much of the integrated management of water quality of the region, and the bay, into the future (Figure 3.6).

The abiotic and biotic characteristics of the system are intrinsically interlinked, but an understanding of the social aspects of the system, particularly how individual stakeholders perceive its value (economic, social and environmental), will be key to successful and sustainable long-term management. Following an initial environmental assessment and characterisation process undertaken as part of Ireland's second River Basin Management Plan, three catchments which flow in to Dundalk Bay have been prioritised for management actions to improve water quality (Figure 3.7).

Local stakeholder contributions (from landowners, businesses, wastewater treatment providers, etc.) to the management of Dundalk Bay will be key for long-term improvements in water quality. Through targeted stakeholder engagement as part of the ongoing formation of the Dundalk Bay Rivers Partnership, it is becoming increasingly clear that the formation of such entities can act as a catalyst for those who may have wished to get involved in local environmental management but did not know how they might do so. Successful implementation of such systems has the potential to result in multiple social, economic and environmental benefits, while also improving government-citizen relationships and building a strong platform for integrated and holistic environmental management.

The Dundalk Bay Rivers Partnership has, for the first time within this region, initiated a bottom-up, community-integrated course of action with the aim of addressing a range of environmental and social concerns with a specific catchment focus. This process has highlighted that quite often people are looking for ways to get involved in local water management initiatives, but frequently guidance and initiatives which encourage their participation and welcome their viewpoints are missing and subsequently people remain disengaged. In addition, by exploring individual and community values in relation to catchments and water resources it can be noted that there are generational differences in how water resources are viewed and valued. Much of the sentiment and heritage associated with water resources as valued by older generations is missing in younger (<40 years old) generations. However, often for older participants the chance to reminisce about how they used and 'valued' their water resources in the past can help to frame future management goals that work for everyone. Indeed, work undertaken in three sub-catchments of Dundalk Bay (Figure 3.1) has identified a 'missing generation' who have little or no interactions with their local water resources as a result of a water-related trauma and subsequent ingrained fear within the local community (Veerkamp & Rolston, 2017). Subsequently, community-led visions have been developed which have resulted in the delivery of targeted educational resources aimed at reconnecting the missing generation with their local waters. Engagement processes associated with this local situation have highlighted that a one-size-fits-all approach should not be applied. Rather an understanding of the local situation is required to inform a multi-faceted engagement strategy which addresses specific, locally-focussed criteria. This approach is undoubtedly more resource intensive but allows a tailoring of solutions to achieve multiple objectives in a sensitive manner.

That is not to say that collaboration and engagement are easy to undertake, and it would be naive to suggest so. Often, political pressures and legislative drivers focus works and measures. In addition, there can be multiple difficulties in engaging stakeholders and communities: there can be legacy issues regarding previous poorly delivered engagement initiatives; often it can be the same groups and individuals that express an interest in being involved in initiatives; and it can be very difficult to ensure engagement across multiple socio-economic backgrounds (Rolston et al., 2017).

3.7. Concluding remarks

Dundalk Bay is a habitat and resource of significant environmental, social and economic value. Multiple stakeholders are required to work together in order to achieve joined up, integrated management of the region to achieve multiple legislative and policy objectives and outcomes. Improving and maintaining the functioning of the Dundalk Bay ecosystem is vital not only to the future productivity of the shellfish industry but to the Bay's status as an Special Area of Conversation and Special Protection Area.

With a significant national policy shift in River Basin Management in Ireland leading to a greater focus on community-led initiatives, the development of the Dundalk Bay Rivers Partnership and the delivery of its agreed visions will clearly contribute to actions that are critical to achieving the targets and objectives set out in relevant legislation and policy. Including coastal communities into the wider catchment-scale engagement process ensures source to sea inclusion. This will facilitate the more holistic, integrated approach to environmental management being encouraged at the national water governance level. However, it must be recognised by all stakeholders that not everything can be done at once and that implementation (as well as tangible results) will take time. Therefore, the setting of expectations must be clearly and sensitively handled from the outset.

The involvement of local communities in water resources management presents an opportunity to improve the cost-effectiveness of management actions across the triple bottom line of social, economic and environmental processes. Yet, it is imperative that the national policy shift in River Basin Management towards this more bottom-up focussed approach is supported by sufficient and accessible funding opportunities.

It is envisioned that over time the Dundalk Bay Rivers Partnership will enable multiple key local stakeholders to employ a targeted management approach underpinned by inclusive social mobilisation and robust pre-planning. This has been proven successful through the UK Rivers Trust model, which is actively being implemented in Ireland under strong encouragement from national Government under Ireland's second River Basin Management Plan of the Water Framework Directive of the European Union.

3.8. References

Celtic Sea Trout Project (Milner, N., McGinnity, P. & Roche, W. E. (2016). Celtic Sea Trout Project – Technical Report to Ireland Wales Territorial Co-operation Programme 2007-2013 (INTERREG 4A). Inland Fisheries Ireland. Retrieved from http://celticseatrout.com/downloads/technical-report/

Clarke, M., Farrell, E., Roche, W., Murray, T., Foster, S., & Marnell, F. (2016). Ireland Red List No. 11: Cartilaginous fish [sharks, skates, rays and chimaeras]. Dublin, Ireland: National Parks and Wildlife Service, Department of Arts, Heritage, Regional, Rural and Gaeltacht Affairs. Retrieved from https://www.npws.ie/content/publications/red-list-no-11-cartilaginous-fish-sharks-skates-rays-and-chimaeras

Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. (1992). *Official Journal*, L 206, 7–50. Retrieved from https://eur-lex.europa.eu/legal-content/EN/TXT/HTML/?uri=CELEX:31992L0043

Department of Agriculture, Food & the Marine, Ireland. (n.d.). FishingNET - The Irish Government's Commercial Sea Fishing Network. Retrieved December 8, 2019, from http://fishingnet.ie/

Department of Housing, Planning, Local Community and Government. (2017). Public consultation on the river basin management plan for Ireland (2018-2021). Dublin, Ireland. Retrieved from http://www.housing.gov.ie/sites/default/files/public-consultation/files/draft_river_basin_management_plan_1.pdf

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. (2000). *Official Journal of the European Communities*, L 327, 1–71. Retrieved from https://www.dcu.ie/sites/default/files/library/apaguide2016.pdf

Environmental Protection Agency, Ireland. (n.d.). Catchments.ie - Water, from source to sea. Retrieved December 8, 2019, from https://www.catchments.ie/

Geological Survey Ireland & the Marine Institute. (n.d.). Infomar. Retrieved December 8, 2019, from https://www.infomar.ie/

Kelly, F., Harrison, A., Connor, L., Matson, R., Wightman, G., Morrissey, E., … Hayden, B. (2010). Sampling Fish for the Water Framework Directive: A Summary of the Central Fisheries Board's Surveillance Monitoring for Fish in Lakes, Rivers and Transitional Waters 2009. Dublin, Ireland: Central and Regional Fisheries Board. Retrieved from https://www.researchgate.net/publication/323342231

Marine Institute. (2016). Report supporting Appropriate Assessment of a Fisheries Natura Plan for cockle 2016-2020 on Dundalk Bay SAC and SPA. Renville, Oranmore, Galway: Marine Institute.

Marine Irish Digital Atlas. (2017). Marine Irish Digital Atlas. Retrieved from http://mida.ucc.ie/pages/information/phys/oceanography/tidesWavesCurrents/details.htm

National Parks and Wildlife Service. (2011). Dundalk Bay Special Protection Area (Site Code 4026). Version 1. Conservation Objectives Supporting Document. [Unpublished report]. National Parks and Wildlife Service.

National Parks and Wildlife Service. (2014). Dundalk Bay SAC – Site Synopsis. Retrieved from https://www.npws.ie/sites/default/files/protected-sites/synopsis/SY000455.pdf

Rolston, A., Jennings, E., & Linnane, S. (2017). Water matters: An assessment of opinion on water management and community engagement in the Republic of Ireland and the United Kingdom. *PLoS ONE,* 12(4), e0174957. https://doi.org/10.1371/journal.pone.0174957

RPS. (2011). Dundalk Harbour Navigation Channel Stability Study (Document no. IBE0623/AKB/Dundalk). A report prepared for the Dublin Port Company.

Standing Scientific Committee on Salmon. (2017). The Status of Irish Salmon Stocks in 2016 with Precautionary Catch Advice for 2017: Report of the Standing Scientific Committee on Salmon to Inland Fisheries Ireland. Dublin, Ireland. Retrieved from https://www.fisheriesireland.ie/documents/1412-the-status-of-irish-salmon-stocks-in-2016-with-precautionary-catch-advice-for-2017.html

Standing Scientific Community for Eel. (2015). Activity Report of the Standing

Scientific Committee for Eel: Report of the Standing Committee for Eel to Inland Fisheries Ireland and the Department of Communications, Energy and Natural Resources. Retrieved from https://www.fisheriesireland.ie/documents/513-ssce-report-2014-final-report-7-5-2015/file.html

The Local Authority Waters Programme. (n.d.). Local Authority Waters Programme - Working to protect and restore good water quality. Retrieved December 8, 2019, from http://watersandcommunities.ie/

The Office of Public Works. (2018). Flood risk management plan: Neagh Bann. Office of Public Works, Ireland. Retrieved from https://s3-eu-west-1.amazonaws.com/docs.floodinfo.opw/floodinfo_docs/Final_FRMPs_For_Publication/FRMP_Final2018_RiverBasin_06.pdf

The Rivers Trust. (n.d.). Homepage. Retrieved December 8, 2019, from https://www.theriverstrust.org/

Veerkamp, V., & Rolston, A. (2017). Shared Waters – Shared Landscapes Project Final Report. https://doi.org/10.13140/RG.2.2.36445.51684

4

Climate Proof Management of Maha Oya Inlet, Sri Lanka

By Trang Minh Duong and Roshanka Ranasinghe

4.1. Motivation for interest and approach

Maha Oya inlet is located on the south west coast of Sri Lanka, an island nation in the Indian Ocean, southeast of India. The inlet is connected to the 3rd largest river basin in the country. Maha Oya is an intermittently closed inlet, which closes whenever riverflow is low, regardless of the prevalent wave climate. It is either naturally opened when riverflow increases or artificially opened by the local community. The inlet and its surrounding are used for multiple purposes, such as sand mining, access to the ocean by fishing boasts, tourism including tourist hotels and recreation by both tourists and local people from the area. These activities bring significant economic and social benefits to local people and contribute to the national GDP. When the inlet closes, usually a few times per year, the main issues are flooding of low lying land and the problems for fishermen who can't travel through the inlet to the sea. Mouth closure therefore has direct negative impacts on the economy, and affects the lives of local people and tourists. Although intermittent mouth closure is a natural phenomenon and thus a component of the natural functioning of the ecosystem, climate change (CC) could cause these impacts to become more severe, particularly as there is low community resilience to coastal change impacts. Therefore, there is a demand for a better system understanding to support improved and sustainable inlet management. There are concerns about the potential impacts of CC on the system and uncertainty on how the system would behave in the future. This study about CC impacts on Maha Oya inlet was undertaken as a component of a comprehensive study about CC impacts on the stability of Small Tidal Inlets (STIs) (Duong, 2015). The broader study investigated potential impacts of different future CC scenarios on 3 main STI types based on their general morphodynamic behaviour. These include: Type 1 – Permanently open, locationally stable inlets, Type 2 – Permanently open, alongshore migrating inlets, and Type 3 – Seasonally/Intermittently open, locationally stable inlets. Maha Oya is a Type 3 inlet. Duong et al. (2016) defined STIs as systems with narrow (< 500 m wide) inlet channels, connected to shallow (average depth < 10 m) and less than 50 km^2 surface area estuaries/lagoons. These STIs are also called bar-built or barrier estuaries (Bruun and Gerritsen, 1960) and are commonly found in wave-dominated, microtidal mainland coasts (Slinger, 1997; Slinger, 2017). They occur on about 50% of the world's coastline (Ranasinghe et al., 2013) and are often present in tropical and sub-tropical regions (e.g., India, Sri Lanka, Vietnam, Florida (USA)), and South America (Brazil), South Africa, and SW/SE Australia) (Duong et al., 2016). These STIs and their adjacent coasts are dynamic systems, with complex feedbacks of system response to system forcing (Carter and Woodroffe, 1994; Slinger 1997). For decades there has been scientific interest in understanding the behaviour of these systems and the physical processes governing their behaviours (Bruun, 1978; Aubrey and Weishar, 1988; Prandle, 1992; Slinger et al., 1994; Slinger, 1997; Ranasinghe et al., 1999; 2013; Fitzgerald et al., 2008; Bertin et al., 2009; Dissanayake et al., 2012; Nahon et al., 2012; Zhou et al., 2014; Slinger et al., 2017). The CC driven variations in system physical processes and morphodynamic behaviours are of particular interest (Duong et al., 2016; Duong et al., 2017a; Duong et al., 2017b;

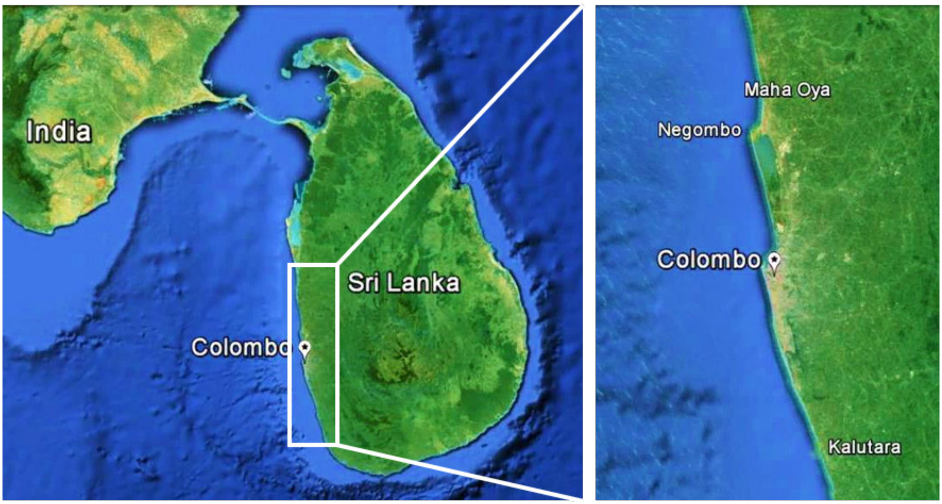

Figure 4.1. Location of Sri Lanka (left) and the case study site Maha Oya river (right) (from: Google Earth)

Slinger, 2017). CC is likely to influence system forcing including sea level rise (SLR), wave conditions and riverflows, which are in turn likely to influence system behaviour such as changing the stability of the inlet itself, erosion of inlet adjacent beaches and/or of estuary margin shorelines, permanent or more frequent inundation of low lying areas on estuary margins, eutrophication, and water quality. This study adopts an environmental science paradigm and aims to increase understanding of CC effects and so help to avoid potential socio-economic and environmental losses.

4.2. Study area

Maha Oya inlet is located in the southwest coast of Sri Lanka, 38 km North of the capital Colombo (Figure 4.1). Maha Oya river is the 3rd largest river basin in Sri Lanka. Sri Lanka is a wave-dominated, microtidal environment, with average offshore significant wave height of 1.12 m and mean tidal range of approximately 0.5 m (tides are predominantly semi-diurnal, 0.2 m neap and 0.8 m spring). It has a tropical monsoonal climate, with 2 monsoon seasons southwest (SW) (from May – September) and northeast (NE) (from November – February) and 2 inter monsoons in between. The southwest coast normally has the most energetic wave condition during SW monsoon (with significant offshore wave heights from 1 – 2 m, and mean wave direction from SW-W). The beaches around the country are sandy with grain size (D50) from 0.2 – 0.45 mm. One third of the total annual rainfall falls in October – December period (Zubair and Chandimala, 2006).

Figure 4.2. Maha Oya inlet, an intermittently open, locationally stable inlet, showing an open (left) and closed (right) state (from: Google Earth)

4.3. Natural dynamics of inlet system

Maha Oya river discharges (1571 million $m^3.yr^{-1}$) into the ocean via Maha Oya inlet. The behaviour of the inlet is governed by the balance of tide, wave and riverflow forcing. Maha Oya inlet is classified as an intermittently open and locationally stable inlet, which closes during low riverflows regardless of the wave climate. The inlet stays fixed in a location, not migrating along the coast (Figure 4.2). For example, via collected satellite images of the area, Maha Oya inlet was observed to open in August 2006 (SW monsoon); then it was closed in September (during SW monsoon) and also closed in October (inter monsoon) in the same year; and then it was opened again in November 2006 (Northeast monsoon). During the year 2006, the inlet stayed in the same location. Maha Oya inlet is considered to be a small tidal inlet (STI) with inlet width 100 m, inlet length 70 m, and inlet depth 3 m. It is connected to a lagoon with surface area of about 0.2 km^2 and average depth of 3 - 4 m. The net longshore littoral drift in the area is about 500,000 $m^3.yr^{-1}$ to the north (GTZ, 1994). The sediment size D50 at Maha Oya location is about 0.25 mm. Maha Oya catchment (approximately 1528 km^2) derives most of its riverflow during NE monsoon. The average river discharge is about 50 m^3/s, and the peak discharge is about 140 $m^3.s^{-1}$ in November (Duong et al., 2017b).

4.4. History of (mouth) management policies and practices

Maha Oya inlet is used for multiple purposes and activities including landing of small fishing boats, water front tourist hotels, dwellings, recreational boating, tourism, sand mining, and the exchange between fresh and sea water for water quality maintenance. It makes significant socio-economic contributions to the local community and to the country. Historically, there has been no official management plan from local authorities and also no thorough, detailed study about the system. A few times per year, there were problems when the inlet closed, such as the threat of inland flooding, water quality issues as well as obstruction of ocean access for fishing boats. The local community would then artificially open the entrance, sometimes with the help of the Coast Conservation Department (CCD). These activities represent ad hoc actions taken at moments of threat with no clear medium or long-term plan or system understanding. Over the years, the fast development of the area and the country have led to increased system usage. This, together with the anticipated CC impacts on the system, mean there is a need for research to support the management and planning processes of local authority and government.

4.5. Social context

Maha Oya inlet is used for multiple purposes and by different sectors. The threats on the system will directly or indirectly affect relevant stakeholders in negative ways. Local people are affected directly by the flooding of low lying land around the river as it threatens their life and properties. Fishermen are also affected directly when they cannot navigate through the inlet to the sea when it is closed. The deterioration of water quality when the inlet closes affects local people and tourists who live there and/or in the surrounding areas. In general, inlet closure which creates flooding, water quality, and ocean access issues, would have a negative impact on tourism revenue, and directly or indirectly affect tourist hotel owners, recreational boat operators, as well as the Department of Tourism.

Moreover, it could have significant effects on the environment too. Since these threats affect people's lives and jobs, there is a demand for improved management and policies on the part of the local authorities and government, particularly in view of the uncertain effects of CC.

4.6. Modelling climate change scenarios

The CC impacts on Maha Oya inlet were investigated using process based numerical modelling, particularly the state-of-the-art numerical model Delft3D (Lesser et al., 2004). Delft3D combines a short wave driver (SWAN), a 2DH flow module, a sediment transport model (Van Rijn, 1993), and a bed level update scheme. It is practically difficult to assess CC impacts on tidal inlets due to the limitations of numerical model ability and the long time modelling scales required for CC impact assessments (Ranasinghe and Stive, 2009; Dissanayake et al., 2012; Dodet et al., 2013; Duong et al., 2016). Ideally, for CC study, the coastal area morphodynamic model Delft3D needs to be run with 50-100 year time varying water levels, waves and riverflow forcing. However to date, even a coastal area morphodynamic simulation with concurrent tide, wave and riverflow forcing for more than a few years have not been successful (Lesser, 2009). Only morphodynamic simulations of wave-dominated inlets for a few months have shown successful results (Ranasinghe, et al., 1999; Bertin et al., 2009; Bruneau et al., 2011). Duong et al. (2016) developed a practical solution to this problem, namely the strategic process based 'snap-shot' modelling approach. Firstly, the contemporary morphodynamic model, forced with contemporary system forcing, is validated to reproduce system present condition. The model results are qualitatively validated with observed morphodynamic behaviour of the inlet using available aerial/satellite images of the area, and are compared against empirical relationships such as the A-P relationship (O'Brien, 1931; Jarrett, 1976), Escoffier curve (Escoffier, 1940), and the Bruun inlet stability criteria (Bruun, 1978). Subsequently, the validated model is used to simulate one year of system behaviour at the desired future time (such as 2050, 2100), with corresponding future CC modified forcing under different CC scenarios to assess potential CC impacts on the inlet. The CC forcing (wave and riverflow) was dynamically downscaled sequentially from Global Climate Model (GCM) projections, to Regional Climate Model (RCMs), then Regional wave/hydrodynamic/catchment models, and local wave models, following the modified version (Duong et al., 2016) of the ensemble modelling framework of Ranasinghe (2016). SLR was adopted from global mean sea level projection of The Fifth Assessment Report AR5 of the IPCC (2013). In the CC snap-shot simulations with SLR, the 'basin infilling' process (i.e., the raising of estuary/lagoon bed level due to SLR (or land subsidence)), was implemented by adjusting the initial bathymetry such that the shape of contemporary basin hypsometry curve was preserved (Ranasinghe et al., 2013; Duong et al., 2017b). All Delft 3D model parameter values were defined via a set of carefully designed sensitivity tests (for details see Duong et al., 2017b).

Inlet stability (i.e., open, close, migrating conditions) is the key criterion used in this study to investigate CC impacts on the inlet, since this directly drives the dynamic behaviour of the estuary/lagoon behind the inlet as well as the adjacent coast. This criterion (the Bruun inlet stability criterion r) is governed by the annual longshore sediment transport (M) and flow through the inlet (including tidal prism and riverflow) (P), so r=P/M (Bruun, 1978). Inlet stability refers to both inlet locational stability and inlet cross-sectional stability. Cross-sectionally stable inlets are the inlets where inlet dimensions will remain mostly constant over time. Locationally stable inlets are those inlets that stay fixed in their locations over time.

Furthermore, a reduced complexity model was developed (based on the Bruun inlet stability criterion r) to obtain rapid assessments of the temporal evolution of STI stability under CC forcing. This model is easy-to-use and gives very fast results, simulating 100 years in under 3 seconds on a standard PC. It is particularly useful for coastal zone managers and planners who need simple tools but can give reliable and fast results to aid the decision making process. (More details about methodology can be found in Duong, 2015; Duong et al., 2016; Duong et al., 2017a; Duong et al., 2017b).

4.7. System understanding and insights gained

This study provides new insights on processes governing the stability of intermittently open inlets. First, previous findings of Ranasinghe et al (1999) and Slinger et al (1994) that riverflow is a critical process in determining the stability of this type of inlets, are confirmed. Second, this study shows that the wave direction, which governs longshore sediment transport rates, could be a deciding factor where the stability of intermittently open inlets is concerned, particularly when riverflows are generally low. SLR, however, appears to be of minor importance for the stability of this type of inlet.

Detailed model results on CC impacts on the stability of Maha Oya inlet from the numerical modelling studies undertaken using the modelling approach described above can be found in Duong et al., 2017b. In summary, under CC scenario, Maha Oya inlet will not change inlet type by 2100, which means it will still be an intermittently open, locationally stable inlet. But the inlet stability level in 2100, which is measured by Bruun inlet stability criterion (r) as described above, will increase compared to the present, also with a longer open duration of the inlet than at present.

The results at 2100 of the reduced complexity model, developed specifically for this CC impacts study, also agree well with the modelling results that Maha Oya inlet won't change inlet type. However, since reduced complexity models can provide more information about the temporal evolution of the inlet stability from present till 2100, these results show that Maha Oya inlet can temporarily change inlet type, to a permanently open, locational stable inlet type for few years (as criterion r increases) in the 2070-2080 decade.

The system understanding gained from this study, including knowledge of the system characteristics and knowledge of physical processes governed the dynamics of the system, as well as new insights of the potential future inlet behaviour under CC impacts, may be of benefit and could play a key role in decision making processes and developing management policies geared for sustainable development. With the temporal variation of the inlet stability predicted from the present to a desired future time (e.g., 2050, 2100), local authorities and government can develop long-term management plans, which could incorporate different interventions corresponding to different levels of inlet stability over the future decades.

4.8. Concluding remarks

Due to the combination of rising sea levels, changing riverflow characteristics, wave conditions, increased utilisation of the waterway, lagoon and adjacent land, flooding of low lying land around Maha Oya has been increasing and more and more fishermen are affected by inlet closures and the lack of ocean access. Historically, whenever the inlet entrance was closed and created problems, either local people (such as: dwellers, fishermen) and/or the private sector (such as: sand mining companies, hotel owners, etc.), who were affected and suffering from the closure, would either individually or together open the mouth, sometimes with help from the Coast Conservation Department (CCD), without any scientific knowledge or awareness of the mechanisms and characteristics of the system. It is now recognised that this ad hoc inlet management is inadequate. Hence, the CCD has initiated the process of developing an efficient entrance management plan backed by sound science and designed with the involvement of local stakeholders with system knowledge. There is a significant concern about how climate change driven variations in mean sea levels (i.e., sea level rise), riverflow and wave conditions might affect entrance closure/open regimes (i.e., inlet stability).

This study on climate change impacts on the stability of Maha Oya inlet has shown, via a detailed numerical study, that the Maha Oya entrance will experience significant temporal variations in its stability over the next 100 years. Results show that the inlet may change from an intermittently closed/open tidal inlet to a permanently open inlet (during the 2070-2080 decade) and then revert back to an intermittently closed/open system, albeit with longer open durations than at present.

From a management perspective, these results seem to favour a 'do-nothing' approach for now, but highlight the importance of conducting detailed studies on the implications of the increased duration of the open state on inland flooding, water quality and associated local activities (inland fisheries, tourism), and ecosystem services. Knowledge from these studies may indicate that it is beneficial to artificially close the inlet from time to time during the 21st century and could lead to the development of new strategies for inlet management. Significantly, the results suggest that it is important not to develop new plans for a permanently open inlet as this condition is only likely to persist for about ten years from 2070 to 2080.

While this study provides indications on potential science-based management strategies appropriate to this particular system, in the modern era management strategies based on once-off studies may not gain local support. Therefore, it is imperative that the scientific results be considered together with local knowledge, local stakeholder views and preferences, in the light of national and local political considerations and socio-economic consequences. Thus, any future strategies should involve a comprehensive, participatory approach so as to ensure that the resulting interventions are environmentally, socially and economically appropriate to local conditions and needs, embracing the concept of co-design (Wijnberg et al, 2013).

The CCD is in the process of developing both short-term and long-term entrance management plans that take these new findings in to account to efficiently manage and use the Maha Oya system. For short-term management considerations, CCD has initiated a series of stakeholder meetings involving the different stakeholder groups to develop more effective inlet management strategies based on critical lagoon water levels and critical inlet closed periods to trigger artificial inlet breaching by the CCD. For long-term management considerations, CCD is in discussion with other relevant government agencies regarding commencing more detailed studies on: optimal inlet open/closed durations for fisheries and tourism, impact of longer inlet open times on ecosystem services, effects of sand mining on inlet stability and the adjacent coastline, and an updated Global Climate Model downscaling study for Sri Lanka.

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4.10. References

Aubrey, D. G., & Weishar, L. (1988). Hydrodynamics and sediment dynamics of tidal inlets. In Lecture notes on coastal and estuarine studies, 29 (p. 456). New York: Springer-Verlag.

Bertin, X., Fortunato, A. B., & Oliveira, A. (2009). A modeling-based analysis of processes driving wave-dominated inlets. Continental Shelf Research, 29(5–6), 819–834. https://doi.org/10.1016/j.csr.2008.12.019

Bruneau, N., Fortunato, A. B., Dodet, G., Freire, P., Oliveira, A., & Bertin, X. (2011). Future evolution of a tidal inlet due to changes in wave climate, Sea level and lagoon morphology (Óbidos lagoon, Portugal). Continental Shelf Research, 31(18), 1915–1930. https://doi.org/10.1016/j.csr.2011.09.001

Bruun, P. (1978). Stability of tidal inlets - theory and engineering (1st Editio). Amsterdam: Elsevier Scientific Pub. Co.

Bruun, P., & Gerritsen, F. (1960). Stability of coastal inlets. Amsterdam: North-Holland Publishing Company.

Carter, R. W. G., & Woodroffe, C. D. (1994). Coastal evolution : Late Quaternary shoreline morphodynamics, 517.

Dissanayake, D. M. P. K., Ranasinghe, R., & Roelvink, J. A. (2012). The morphological response of large tidal inlet/basin systems to relative sea level rise. Climatic Change, 113(2), 253–276. https://doi.org/10.1007/s10584-012-0402-z

Dodet, G., Bertin, X., Bruneau, N., Fortunato, A. B., Nahon, A., & Roland, A. (2013). Wave-current interactions in a wave-dominated tidal inlet. Journal of Geophysical Research: Oceans, 118(3), 1587–1605. https://doi.org/10.1002/jgrc.20146

Duong, T. M. (2015). Climate Change impacts on the stability of Small Tidal Inlets (Doctoral dissertation). UNESCO IHE/Delft University of Technology.

Duong, T. M., Ranasinghe, R., Luijendijk, A., Walstra, D., & Roelvink, D. (2017). Assessing climate change impacts on the stability of small tidal inlets: Part 1 - Data poor environments. Marine Geology, 390, 331–346. https://doi.org/10.1016/j.margeo.2017.05.008

Duong, T. M., Ranasinghe, R., Thatcher, M., Mahanama, S., Wang, Z. B., Dissanayake,

P. K., … Walstra, D. (2017). Assessing climate change impacts on the stability of small tidal inlets: Part 2 - Data rich environments. Marine Geology, 395, 65–81. https://doi.org/10.1016/j.margeo.2017.09.007

Duong, T. M., Ranasinghe, R., Walstra, D., & Roelvink, D. (2016). Assessing climate change impacts on the stability of small tidal inlet systems: Why and how? Earth-Science Reviews, 154, 369–380. https://doi.org/10.1016/j.earscirev.2015.12.001

Escoffier, F. F. (1940). The stability of tidal inlets. Shore and Beach, 8(4), 111–114.

FitzGerald, D. M., Fenster, M. S., Argow, B. A., & Buynevich, I. V. (2008). Coastal Impacts Due to Sea-Level Rise. Annual Review of Earth and Planetary Sciences, 36(1), 601–647. https://doi.org/10.1146/annurev.earth.35.031306.140139

GTZ. (1994). Longshore Sediment Transport Study for the South West Coast of Sri Lanka (Project report).

Intergovernmental Panel on Climate Change. (2013). Summary for policymakers. In T. F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, … P. M. Midgley (Eds.), Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

Jarrett, J. T. (1976). Tidal prism–inlet area relationships (Technical Report GITI No.3). Fort Belvoir, Virgina and Vicksburg, MS: U.S. Army Coastal Engineering Research Center, U.S. Army Engineer Waterways Experiment Station.

Lesser, G. R. (2009). An Approach to Medium-term Coastal Morphological Modeling (Doctoral dissertation). UNESCO IHE/Delft University of Technology.

Lesser, G. R., Roelvink, J. A., van Kester, J. A. T. M., & Stelling, G. S. (2004). Development and validation of a three-dimensional morphological model. Coastal Engineering, 51(8–9), 883–915. https://doi.org/10.1016/j.coastaleng.2004.07.014

Nahon, A., Bertin, X., Fortunato, A. B., & Oliveira, A. (2012). Process-based 2DH morphodynamic modeling of tidal inlets: A comparison with empirical classifications and theories. Marine Geology, 291–294, 1–11. https://doi.org/10.1016/j.margeo.2011.10.001

O'Brien, M. P. (1931). Estuary and tidal prisms related to entrance areas. Civil Engineering, 1(8), 738–739.

Prandle, D. (Ed.). (1992). Dynamics and Exchanges in Estuaries and the Coastal Zone (Volume 40, Vol. 40). Washington, D. C.: American Geophysical Union. https://doi.org/10.1029/CE040

Ranasinghe, R. (2016, September). Assessing climate change impacts on open sandy coasts: A review. Earth-Science Reviews. Elsevier B.V. https://doi.org/10.1016/j.earscirev.2016.07.011

Ranasinghe, R., Duong, T. M., Uhlenbrook, S., Roelvink, D., & Stive, M. (2013). Climate-change impact assessment for inlet-interrupted coastlines. Nature Climate Change, 3(1), 83–87. https://doi.org/10.1038/nclimate1664

Ranasinghe, R., Pattiaratchi, C., & Masselink, G. (1999). A morphodynamic model to simulate the seasonal closure of tidal inlets. Coastal Engineering, 37(1), 1–36. https://

doi.org/10.1016/S0378-3839(99)00008-3

Ranasinghe, R., & Stive, M. J. F. (2009). Rising seas and retreating coastlines. Climatic Change, 97(3), 465–468. https://doi.org/10.1007/s10584-009-9593-3

Slinger, J. H. (1996). Modelling the physical dynamics of estuaries for management purposes. University of Natal.

Slinger, J. H. (2017). Hydro-morphological modelling of small, wave-dominated estuaries. Estuarine, Coastal and Shelf Science, 198(B), 583–596. https://doi.org/10.1016/j.ecss.2016.10.038

Slinger, J. H., Taljaard, S., & Largier, J. L. (1994). Changes in estuarine water quality in response to a freshwater flow event. ECSA 22/ERF Symposium, Plymouth(UK), 13-18 Sep 1992, 1994, 51–56.

Slinger, J. H., Taljaard, S., & Largier, J. L. (2017, September). Modes of water renewal and flushing in a small intermittently closed estuary. Estuarine, Coastal and Shelf Science. Academic Press. https://doi.org/10.1016/j.ecss.2017.07.002

van Rijn, L. (1993). Principles of sediment transport in rivers, estuaries and coastal Seas (Part 1). Amsterdam, Netherlands: Aqua Publications.

Wijnberg, K. M., Hulscher, S. J. H. M., Roelvink, D., Slinger, J. H., Mulder, J. P. M., & de Groot, A. V. (2013). Co-designing Coasts using natural Channel-shoal dynamics (CoCoChannel). Grant application form: Building with Nature.

Zhou, Z., Coco, G., Jiménez, M., Olabarrieta, M., Van Der Wegen, M., & Townend, I. (2014). Morphodynamics of river-influenced back-barrier tidal basins: The role of landscape and hydrodynamic settings. Water Resources Research, 50(12), 9514–9535. https://doi.org/10.1002/2014WR015891

Zubair, L., & Chandimala, J. (2006). Epochal changes in ENSO-streamflow relationships in Sri Lanka. Journal of Hydrometeorology, 7(6), 1237–1246. https://doi.org/10.1175/JHM546.1

5

Managing the Mouth of Russian River Estuary, California

By John Largier, Dane Behrens, Kate Hewett, Michael Koohafkan, Matt Robart, David Dann and Robin Roettger

5.1. Introduction

For centuries the Russian River and its estuary has been used as a source of food, with impacts of human activity assumed to be minimal and transient. However, a more industrial era of resource extraction commenced in the mid-nineteenth century with the advent of forestry and extraction of old-growth redwood trees as well as more intense use of the salmon fishery. In spite of significant impacts, management interventions in this era were still transient and it is only a century ago that a more intentional, permanent management of the lower river and estuary started with the insertion of structures into the natural system. Although some structures were not fully implemented or did not stand the test of time, the intent was to tame the river permanently and allow for navigation, farming and water supply. This hubris is characteristic of an era of hydrological modification in many systems across the globe during a time when resources were seen as abundant and natural systems were seen as resilient (or not valued beyond their capacity to deliver water, food and transport opportunities).

In the last half century, management has focussed increasingly on resources and ecosystem functions and services. Whether resource-oriented or conservation-oriented, it became clear that the Russian River and comparable natural systems were in decline. However, management is now constrained by permanent hydrological modifications in the form of dams on the river, jetties at the estuary mouth, and levees protecting agricultural lands. In addition, somewhat unwittingly, houses and other structures have been placed in the floodplains of the estuary and lower river. The challenge of contemporary management is to balance multiple objectives (resilient ecosystem, flood protection, water quality, resource extraction, use of waters for recreation and transport) within a variable environment (sea level, river flow, waves, loading of inflows) while constrained by permanent structures (buildings, jetties, dams, levees).

The management of specific systems varies depending on the relative priority of these multiple objectives. While some systems are managed for system integrity and 'health', others are managed for a single species or objective – sometimes by design, and other times by the weight of stakeholder interests or the imperative of statutes, regulations and other legal tools. Permanent modifications may be removed through 'restoration', with the intent being to restore natural processes and ecosystem functions, but success is not assured and often not well tracked post-restoration. So, now we live in a management era that is again well intentioned, but with a focus more on ecosystems and resources – however, there are many examples of unintended consequences and a dearth of careful measurement and evaluation of the effects of management choices, whether soft (e.g., altering inflow rates and loading) or hard (e.g., insertion or removal of structures).

Management of the Russian River Estuary (RRE) is one example of this challenge to address multiple objectives, including issues related to flooding, water quality, fish habitat, and use of the waters for recreation. As elsewhere, this challenge is undertaken within a multi-agency landscape that includes many stakeholders, institutions and jurisdictions – each with their own priorities and approaches to the common aim of realizing the full potential of the RRE ecosystem and all the services it can provide.

5.2. Study area

The Russian River drains a 3850 km^2 watershed in northern California, with the mouth/inlet at 38.451°N and 123.127°W (Behrens et al., 2013). The 175 km long river exhibits a steep gradient, draining from an elevation above 1300 m. Precipitation is entirely via rainfall, which is heavy in winter months (specifically December to March) and typically absent in summer months (May to October). Rain and river flow is strongly pulsed with inflows exceeding 1000 m^3.s^{-1} in the days following major rainfall events – the strongest due to 'atmospheric rivers'. However, during dry periods in winter the flow can be much lower (10-100 m^3.s^{-1}). Lowest flow occurs in late summer and is managed by the Sonoma County Water Agency (SCWA) in accordance with a multi-agency agreement – typically 4 m^3.s^{-1}, but as low as 2 m^3.s^{-1} during dry years. In contrast to estuaries further south in California (or estuaries with smaller watersheds), the water budget is still positive during low-inflow periods and the water level rises in the estuary when the mouth is closed.

The estuary of the Russian River is a prototype of the bar-built, intermittently closing, drowned-river-canyon estuaries that are common in California (and along comparable

Figure 5.1. Aerial photo of Russian River Estuary showing the constricted estuary mouth that penetrates the wave-built sand barrier across the mouth of the fluvial valley (photo credit: John Largier)

steep-gradient, wave-exposed coastlines). The valley/canyon that the estuary occupies is blocked by a sand barrier at the mouth (Figure 5.1), constricting the flow of water between estuary and ocean. However, the river and estuary are larger than many comparable systems in California and it has received more attention than most, yielding a comprehensive set of monitoring data for science-based management. Recent classifications of west-coast estuaries have classed the RRE as a 'river mouth estuary' (Gleason et al., 2011; Sutula, 2011), but the propensity to close means the RRE has much in common also with 'lagoon estuaries' – together these two categories account for a few hundred small estuarine systems in California. An additional and often overlooked factor in California estuaries is the importance of ocean inflows that are biogenically important given the high natural loading of ocean waters in a coastal upwelling region (and the low organic loading of many unperturbed watersheds).

Management of intermittently closing estuaries in California has received considerable attention recently, after decades of ad hoc approaches. There is growing recognition of common drivers and responses across diverse systems – and an interest in learning from management in comparable regions, like South Africa and Australia. Recent activities include a state-wide assessment of nutrient criteria (Sutula, 2011), establishment of a Science Advisory Team for the multi-agency Wetlands Recover Project that addresses estuaries in southern California (Southern California Wetlands Recovery Project, 2018), a conservation assessment of estuaries along the US west coast (Gleason et al 2011), and a review of mouth breaching effects to advise permit decisions by the NOAA Fisheries (Largier et al., 2019). The most common drivers of inlet management are recognised as (i) water level and coastal flooding, (ii) water quality, specifically hypoxia, feacal bacteria and undesirable eutrophic smell/sight, and (iii) endangered species management. The impact of artificial breaching is felt in marsh and pelagic habitats that in turn impact broad ecological communities and specific endangered species.

Management of the mouth of the RRE is most active during the dry season, with SCWA obligated by a multi-agency agreement to open the mouth before the water level reaches 2.75 m above NAVD (9 foot). This is known as Agreement 1610 and the aim is to preclude flooding of low-lying structures and grazing lands (and also the coastal highway when water levels exceed 3.5 m). Typically a breach is planned and instituted when the water levels exceeds 2.1 m (7 foot). However, since 2009 SCWA has been operating under a Biological Opinion that directs them to manage the mouth in a way that promotes a closed/perched state during summer months to support the rearing of juvenile steelhead trout (an anadramous salmonid) in the estuary. The idea is to establish a perched-estuary state with an overflow mouth, as is typical of smaller estuaries in California and which has been shown to be conducive to high growth rates for juvenile steelhead (Bond, 2006; Hayes et al., 2008; Matsubu et al., 2019; Serghesio, 2011). This single-species imperative dominates management decisions, although management is also constrained by other interests, including recreational use of the estuary and lower river that depends on water quality and water level, regulations and perceptions relating to water quality in the estuary (including oxygen, bacteria, algal blooms and more), protection of a seal haul out on the beach at the mouth, and water delivery to SCWA clients.

The UC Davis Coastal Oceanography Group has been involved in studies of mouth state and controlling factors (Behrens et al., 2009, 2013), estuary circulation and stratification (Behrens et al., 2016; Largier & Behrens, 2010), dissolved oxygen (Hewett, 2014), and steelhead habitat (Largier & Koohafkan, 2016; Matsubu et al., 2019) – plus general advice to SCWA and collaborating agencies on the biophysical aspects of the estuary. To address hydrology and water quality during closures, we have deployed time series instruments and conducted hydrographic surveys (including BOD sampling) every year since 2009. Additional data is available prior to 2009. The RRE water column is markedly different between open-mouth and closed-mouth states (Figure 5.2): when open, a salt wedge intrudes tidally about 5 km from the mouth, occasionally delivering saline waters to the inner estuary; when closed, a 2-layer structure extends well into the inner estuary, with saline waters found in deeper regions and overlaid by a continuous and thickening layer of low-salinity water.

5.3. Biophysical dynamics

5.3.1. Estuary mouth closure

The mouth of the RRE closes multiple times annually, as illustrated in Figure 5.3.

There is seasonality in closures with the highest probability of a closed mouth in fall and early winter (October-December) and lowest probability in mid-summer (July-August). The mouth closes in winter during dry periods when big waves rebuild the sand barrier. There is a small increase in closure probability in spring. In addition to seasonality there is a marked interannual variability with the mouth closed for two thirds of the days in 1977 (a severe drought year) and zero days in 1982 (a high rainfall year). Other than in 1977, closures persist for days to weeks, seldom more than a month. Many closures end in natural breaching after a few days, but longer closures are typically ended by a mechanical breach (orange shading in Figure 5.3).

Freshwater inflow to the RRE is highly seasonal, following Mediterranean-climate, winter rainfall. This flow seasonality interacts with the seasonality of waves in determining the state of the mouth of the RRE (Behrens et al., 2013). Tidal forcing does not vary seasonally, but tides only affect the estuary mouth when it is open, driving flows between estuary and ocean that can maintain or enlarge the inlet channel: a spring-neap cycle in channel width is observed during summer (Behrens et al., 2013). Seasonality and interannual

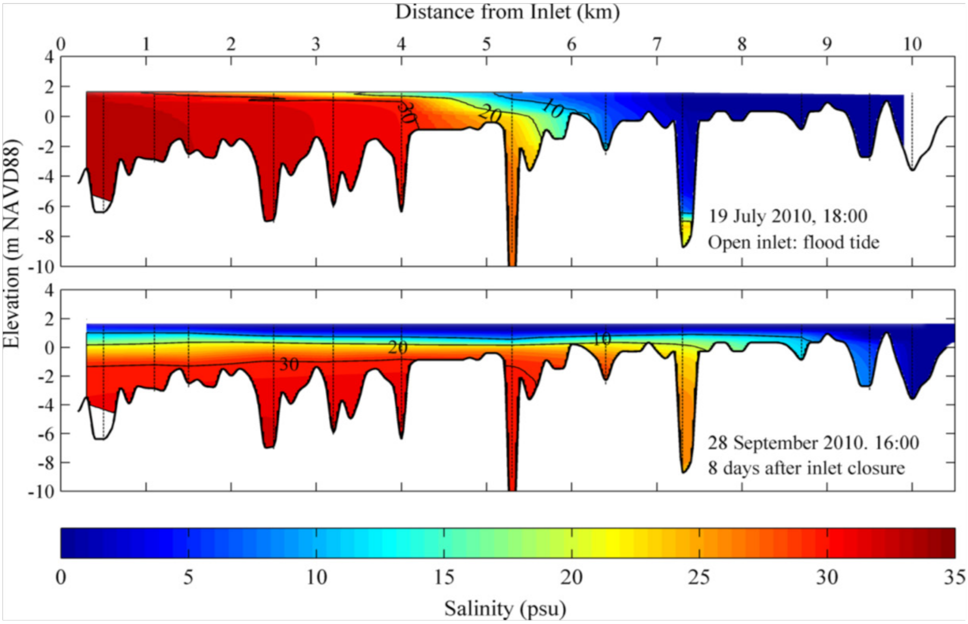


Figure 5.2. Longitudinal salinity structure in Russian River Estuary during representative open-mouth conditions (top panel: 19 July 2010) and closed-mouth conditions (bottom panel: 28 September 2010) (adapted from Behrens et al., 2016)

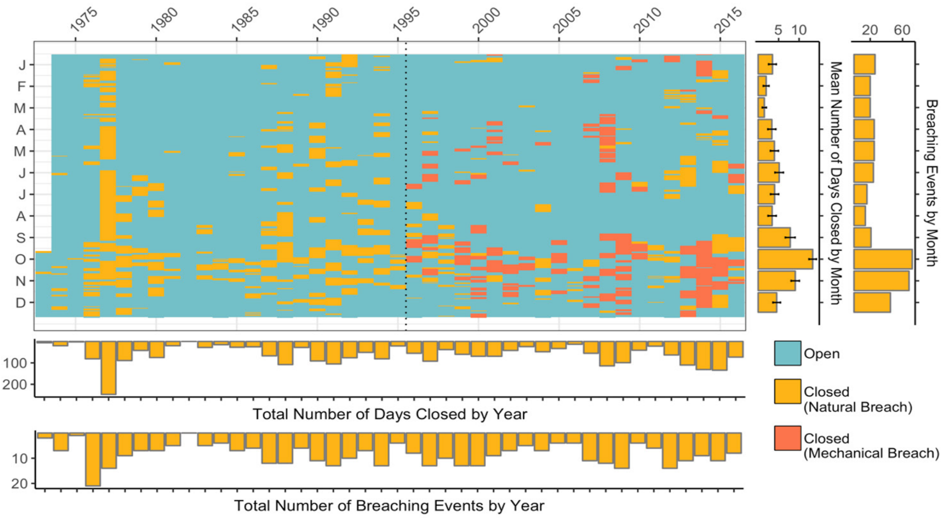


Figure 5.3. Figure 5.3: Daily record of RRE mouth state (1973 to 2016). Yellow/orange shading indicates closures; orange indicates closures terminated mechanically (since 1996)

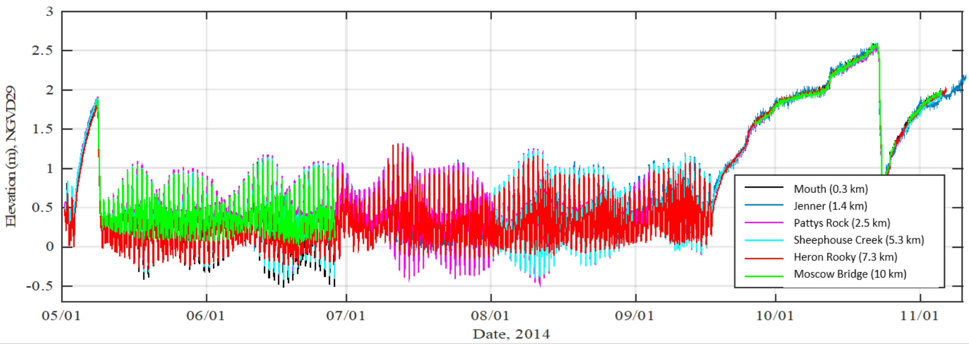


Figure 5.4. Water levels in Russian River Estuary during 2014 dry season, showing a brief closure in May, tidal variability May-September and a couple of prolonged closures in September-October and October-November.

variability is mostly related to the interaction of seasonal cycles in river flow and waves – the mouth tends to close during large waves and low river flow, but it will remain open if river flow is strong enough or if waves are small enough (in the absence of river flow, tidal flows can maintain an open mouth inlet). Closures happen most frequently in fall as wave power increases due to early winter storms in the Gulf of Alaska (October - November) while river flow remains very low until rainfall occurs in California some months later (November - December). In winter, both waves and river are strong, but closures may occur when river flow decreases during dry periods – however in both winter and early spring, closures are short-lived as inflow is high enough to fill the estuary basin and overflow the sand barrier within several days. In spring, closures occur if the seasonal hydrograph drops off earlier than the decrease in wave energy due to late storms in North Pacific. If a closure happens late enough in the season, river flow can be low enough to allow a prolonged closure before natural/artificial breaching occurs. However, if a closure does not occur in spring, the RRE mouth will remain open all summer. The interaction of river flow, tidal flows, and wave forcing in determining the probability of mouth closure has been further investigated by Behrens et al. (2013) and a conceptual model of mouth state has been outlined in a report on mouth state (ESA, 2016).

5.3.2. Estuary water level

In the RRE and most estuaries in northern California, there is a positive water balance so that the estuary basin fills during closures and the water level rises. The rate of water level rise is determined by river inflow, evaporation and seepage through the sand barrier (the exchange between surface waters and groundwater is unknown but assumed small) (Behrens et al., 2013; Slinger, 1996, 2017). Without intervention, the water level would rise to the height of the sand barrier, which is determined by the highest run-up event due to wave-tide concurrence since the mouth closed. A brief closure in spring 2014 that ended in a natural breach and a pair of prolonged closures in fall 2014 that were terminated by mechanical breaches are shown in Figure 5.4. Water level rises quickly during the spring closure owing to higher river flow rate, and the breach occurs naturally when water level is below 2 m as the sand barrier has not been built high yet. In fall closures the water level rises quickly up to 1.5 m or 2 m, but then slows down within a week in early October 2014 and again in early November 2014 when water level does not rise, indicating a steady water balance.

Recent efforts by the SCWA to develop an overflow channel and maintain a constant water level during closures (as required by the Biological Opinion) have been unsuccessful. While a perched state is common in smaller estuaries, the overflow rate in the RRE is too large and the critical shear stress for sand grains on the beach is exceeded, resulting in erosion and full breaching of the mouth. Higher through-barrier seepage occurs when estuary water levels are higher and it is possible that the residual overflow can be reduced sufficiently to allow non-eroding overflow (and over a berm that is above the height of any summer wave action that may close the mouth). However, high water levels impact a number of structures as well as some grazing land and also recreation facilities (e.g., submerging popular summer beaches along the lower river). Both the existing Agreement 1610 and political will preclude this more natural option being explored further. Typically the mouth is now breached when water levels reach 2.5 m, but reduced rates of water level rise are only seen when water levels exceed 2 m (e.g., Figure 5.4), allowing only a narrow window of operation for such an exploration. It is more likely that a perched state and steady estuarine water level can be sustained when water levels are allowed to rise to 3 m or above. In addition to the option of removing these buildings, there is also debate about removing a relict stone jetty from the beach that may be limiting seepage rates (thus also precluding the aim of sustaining a perched state in most summers).

5.3.3. Estuary stratification

In addition to rising water level during closure, intense stratification develops due to the trapping of a layer of seawater in the estuary basin (Figures 5.5 and 5.6). Bottom salinity at Paddy's Rock, about 2.4 km from the mouth, is persistently high (above 30) indicating the presence of seawater during both open and closed periods. However, tidally varying salinity during open-mouth periods gives way to persistent low salinity values near-surface during closures (below 10). A distinct low-salinity surface layer forms and thickens over time, resulting in a very stable water column that cannot be mixed by surface wind forcing, trapping the lower layer. Further, the salinity interface sharpens and turbulent mixing is suppressed in and immediately below this interface, precluding diffusive fluxes to/from the lower layer (Hewett, 2014; also see Slinger & Largier, 1990). In the week following mouth closure in mid-September, surface salinities drop from a tidal average of ~20 to below 5 while near-bottom oxygen drops from a tidal average of ~90% to zero (Figure 5.5). This rapid transition to deep-water anoxia was not observed in the brief closure in early May.

During closure periods a salt-stratified lake forms. This is evident in September 2010 (Figure 5.2) and May 2013 (Figure 5.6). High-salinity water is most obvious at depth in the outer estuary, but some saline water is also trapped at depth in deeper sections of the inner estuary. These deep waters are hypoxic – and even anoxic below the euphotic zone. Without replenishment of dissolved oxygen by photosynthesis or vertical mixing, even low levels of biological oxygen demand (BOD) may account for net respiration and remove all oxygen from water at depth (and over time BOD may be topped up by settling of organic material into this lower layer from freshwater overflow). In contrast, oxygen levels can exceed saturation levels at the interface depth where light can penetrate and account for net photosynthesis, and the absence of mixing allows oxygen to accumulate

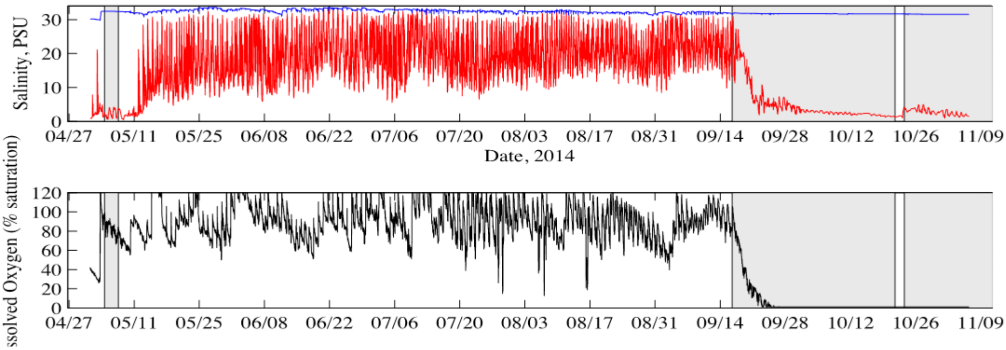


Figure 5.5. Hourly RRE salinity (near-surface and near-bottom) and dissolved oxygen (near-bottom) data from Paddy's Rock (2.4km from mouth) during 2014. Closure periods are demarcated by grey shading

(e.g., Figure 5.6c, at a depth of -1 m and within 3 km of the mouth). On this day, a strong algal bloom is evident with the top of the chlorophyll layer at this depth (Figure 5.6d). At the stations at ~5.3 km and ~7.3 km from the mouth, the algal bloom blocks the light (Figure 5.6e - 5.6f) and contributes to anoxia at depth (Figure 5.6c). Both water temperature and dissolved oxygen vary between night and day. In other surveys, a temperature maximum was observed at mid-depth associated with penetration of thermal radiation into the interface where the absence of turbulent mixing allows accumulation of heat during the day. Oxygen increases in the interface during daylight hours when photosynthesis dominates respiration, but decreases at night when respiration dominates (Hewett, 2014).

5.4. Estuary water quality and habitat volume

The management of the RRE and other estuaries in California focusses on the role of these environments as a preferred rearing habitat for juvenile steelhead, an iconic fish in the region that is important both in conservation and fishing. In recent work, the ranges of temperature, salinity and oxygen in which these fish can tolerate or thrive have been identified (Boughton et al., 2017). These tolerances define water quality from the perspective of juvenile steelhead and are summarised in 4 categories for each variable (see Largier & Koohafkan, 2016) – differentiating between freshwater-acclimated younger juveniles and saltwater-acclimated older juveniles. Optimal growth occurs at temperatures between 14 °C and 18 °C, with positive growth at temperatures below 14 °C or between 18 °C and 21 °C. No growth or negative growth occurs at temperatures above 21 °C, while temperatures above 25 °C are unsuitable. Similarly, minimal impairment occurs when dissolved oxygen concentrations exceed 6 $mg.L^{-1}$, there is some impairment in the range 4 to 6 $mg.L^{-1}$, severe impairment in the range 3 to 4 $mg.L^{-1}$, and conditions below 3 $mg.L^{-1}$ are unsuitable. When the salinity is below 10, or between 10 and 15, there is a low to negligible energy demand, whereas the brackish range from 15 to 28 poses a high energy demand for freshwater-acclimated younger juvenile salmonids and a low energy demand for saltwater-acclimated older juvenile salmonids. Likewise, a marine environment with salinity above 28 is unsuitable for the freshwater-acclimated younger juvenile salmonids and only imposes a moderate energy demand on saltwater-acclimated

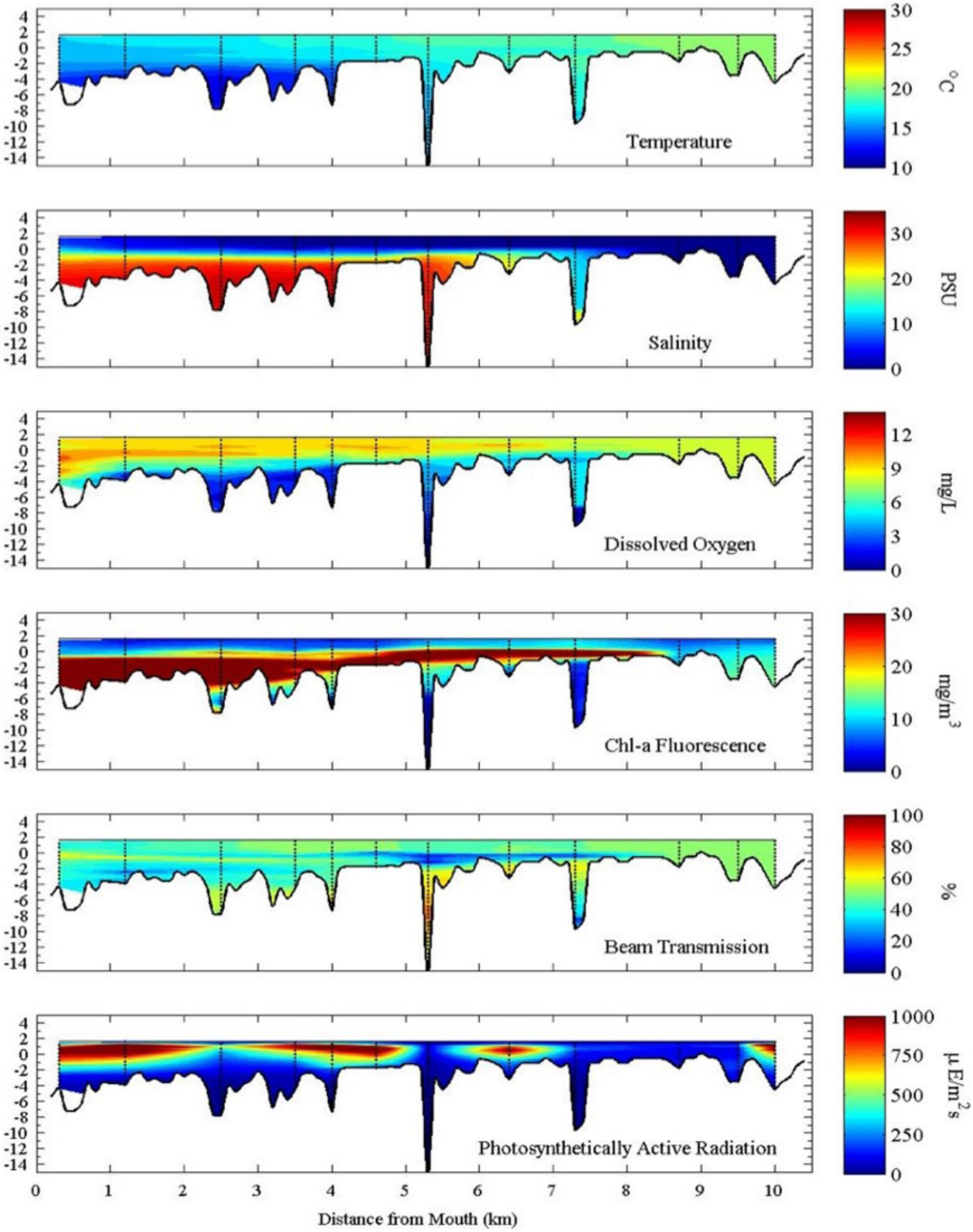


Figure 5.6. Longitudinal-vertical sections showing the spatial patterns of temperature, salinity, dissolved oxygen, chlorophyll fluorescence, light transmission and photosynthetically active radiation (PAR) along the thalweg of the RRE on 28 May 2013.

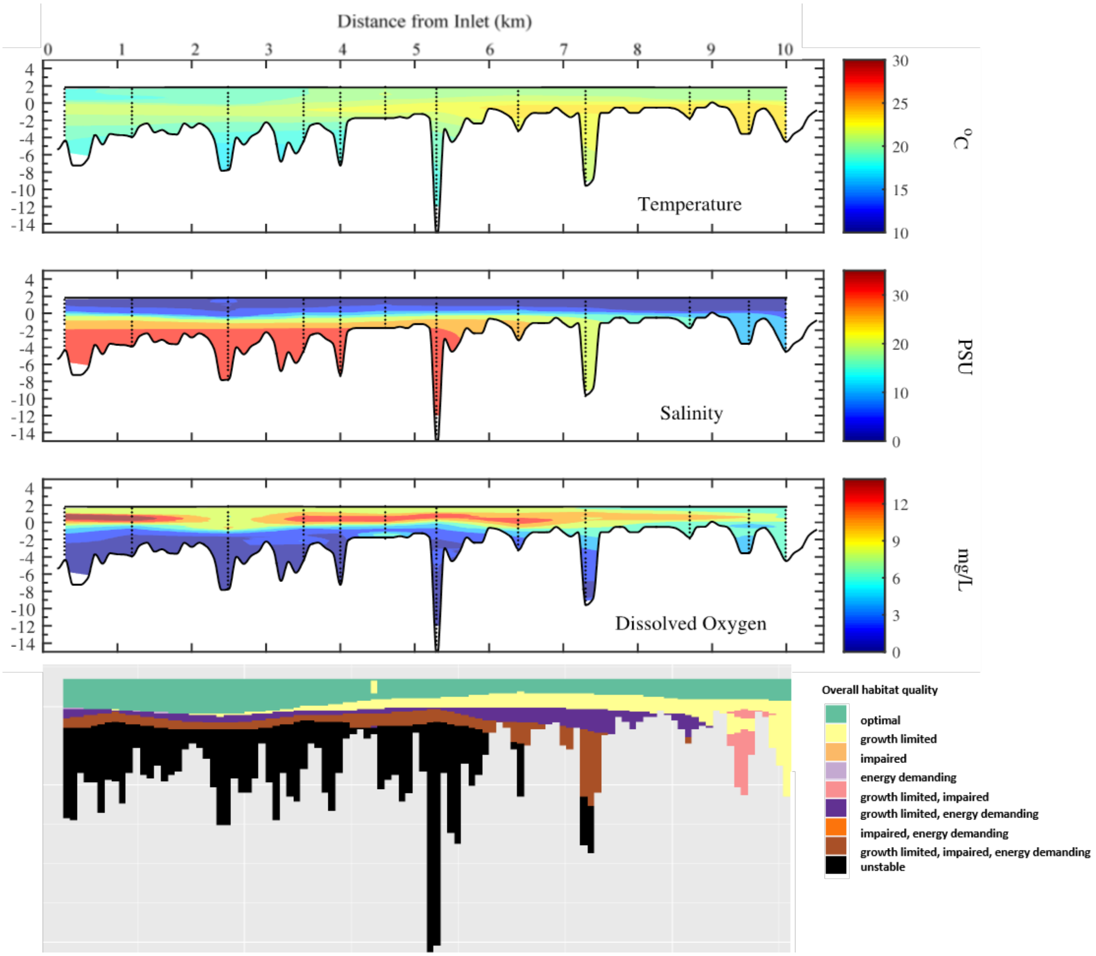


Figure 5.7. Longitudinal-vertical section showing temperature, salinity, dissolved oxygen, and habitat categories for freshwater-acclimated juvenile steelhead during survey on 30 September 2014.

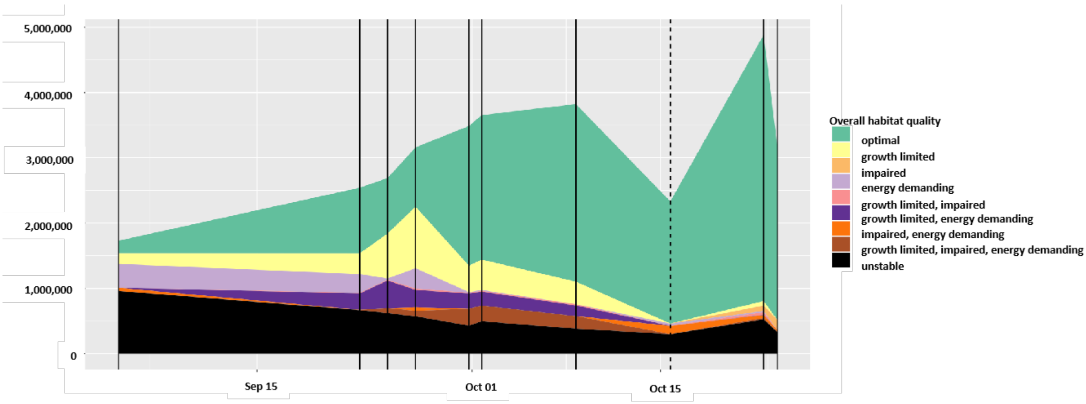


Figure 5.8. Time dependence of water-quality-defined habitat categories for freshwater-acclimated juvenile steelhead during closure from 4 September to 23 October 2014

older juvenile salmonids.

These physiological criteria allow quantification of water-quality habitat based on temperature, salinity and oxygen data at specific times in the estuary – and changes in the volume of available habitat can be tracked over time during a closure event (Figure 5.8). At depth, habitat is not available for juvenile steelhead, primarily due to the absence of oxygen (e.g., 'unsuitable', Figure 5.7), and above the hypoxic layer, habitat is still marginal due to high salinities ('energy demanding') and warm water ('growth limited'). Near-surface waters are 'optimal' for the freshwater-acclimated juveniles that like cool, low-salinity, oxygenated waters. However, sub-surface waters are warmer and growth limiting (yellow shading in bottom panel of Figure 5.8).

Using these water-quality defined habitats, it is then possible to track changes in volume during a closure event, as in September-October 2014 (Figure 5.8). On 4 September, at the time of closure, more than half of the available volume of the estuary is unsuitable ($1x10^6$ m^3), but this volume changes over time as oxygen conditions improve in deep water. At the same time the optimal, near-surface layer thickens as the water level rises and it also spreads laterally eventually accounting for 4×10^6 m^3 when the mouth breaches on 23 October. While sub-optimal habitats shrink over time, there is an interesting temporary increase in growth-limited habitat as mid-depth waters warm in late September (yellow shading in Figure 5.8).

In addition to slow day-to-day habitat changes, there are day-night changes in habitat volumes associated with diurnal variability in temperature and oxygen at mid-depth. However, high-frequency variability is much higher in open-tidal conditions, and ongoing research is directed at determining to what extent fish move with tidal flows or remain in one geographical location, experiencing tidally varying water-column habitat (Matsubu et al., 2019). Juvenile steelheads are also subject to predation and dependent on prey availability. Following Boughton et al. (2017), by defining depth/bottom-depth categories (littoral, surface limnetic, epibenthic, subsurface limnetic and profundal), one can identify additional habitat categories that account for major habitat differences in terms of predation pressure and prey availability.

These depth-defined habitat categories also vary over time (Figure 5.9), with little change in profundal and subsurface limnetic volumes, but major growth in the volume of the subsurface epibenthic habitat that represents a high-prey, low-predation environment. Although overbank flooding into vegetated areas is believed to provide excellent rearing habitat for juvenile steelhead, the shallow littoral zone does not expand significantly with rising water levels, but rather migrates landward and actually decreases in volume as the water line approaches steeper canyon banks.

5.5. Stakeholders and mouth management

The estuary of the Russian River and adjacent watershed and ocean are valued by many people, arranged into groups that have different hopes and expectations from the estuary. The system supplies water to nearby towns and farms, low-lying land in the valley is used for buildings and farms but these may be inundated during high water levels in the estuary, the estuary and lower river provide recreational opportunities that can be

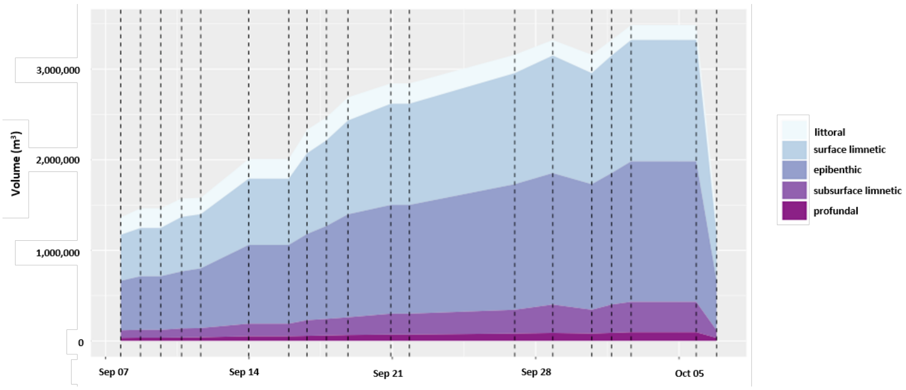


Figure 5.9. Time dependence of depth-defined habitat categories for juvenile steelhead during closure from 4 September to 23 October 2014

compromised by water levels or water quality, the estuary provides important fish habitat that can be diminished by low oxygen or high temperature, and the system provides important wildlife habitat that may be diminished by water quality or disturbance.

For each of these values, there are actors comprised of stakeholders including government agencies, non-governmental organisations, loose coalitions of local residents, and regional interest groups.

- *Water:* multiple stakeholders are primarily represented as clients of the Sonoma County Water Agency. There is limited additional extraction of water from the lower river and estuary. SCWA participates in community meetings.
- *Land:* inundation risk to buildings and farmlands is reduced through Agreement 1610 that was negotiated between multiple parties in 1992, which obligates SCWA as management agency to breach the mouth before water level reaches 2.75 m. Landowners are typically no longer present at community meetings.
- *Recreation:* local residents and regional visitors enjoy estuarine and river beaches that may be flooded as well as kayaking and other modes of experiencing the natural surroundings and wildlife, including hiking and bird watching. State Parks representatives represent this group at community meetings, although their interest is focussed on lands owned by State Parks. The SCWA also represents local residents who provide comment to them on water levels in the system. In addition to facilities, water quality is perceived as an issue and of concern to the Regional Water Quality Control Board and local groups like the Sonoma Environmental Alliance in addition to participation of individuals in community meetings.
- *Fish:* the fish community in general and specifically the endangered steelhead trout are sampled by SCWA, with strong interest and specific responsibilities exercised by federal and state agencies (NOAA National Marine Fisheries Service and California Department of Fish and Wildlife) and also restoration groups (e.g., California Sea Grant). The impact of water quality on fish habitat is a central issue,

of interest also to the Regional Water Quality Control Board in addition to fish and wildlife agencies.

- *Wildlife:* the estuary, beach and nearshore provides important habitat for birds and mammals. Specifically the beach at the mouth of the estuary is an important harbour seal haul-out, with late spring a critical period owing to pupping. This priority is represented by both local organisations like Stewards of the Redwoods and federal agency NOAA Marine Mammal Commission.
- *Ecosystem:* in addition to specific interests like steelhead population and recreational water quality, there are also stakeholders with interest in the ecosystem as a whole – this includes California State Parks and local environmental groups like Stewards of the Redwoods.

Although there are many stakeholders pulling in different directions, most stakeholders are reasonably represented in community meetings and technical workshops that link water quality, water levels, wildlife health, fish habitat and recreation uses with management of mouth state and river flow. The contemporary dialog is centered on the value of the estuary as critical rearing habitat for juvenile steelhead, driven by the 2009 Biological Opinion and constrained by the 1992 Agreement 1610 on management of water levels in the estuary. This focuses discussion and leads to action, but in this setting some issues have received less attention than others.

Fish habitat has not always been the focus of attention. Priorities and management choices have shifted over past decades. Several phases of human-environment interaction can be identified (phases are not distinct and timing of transitions are loosely determined):

- *Prior to 1840:* little is known about human impacts and use of resources, but they are assumed to be light and sustainable – within the capacity of the natural system to assimilate.
- *From 1840 to 1870:* this is a period of intense logging of the watershed and banks of the lower river and estuary and one would expect that there was increase sediment loading. Historical records indicate seasonal closure of the mouth. Although there is no record of active management of the mouth, one can speculate that it would have been opened to allow transport of logs to the sea.
- *From 1870 to 1940:* timber harvest decreased and interest in the use of the estuary for navigation increased, which culminated in attempts to build jetties at the mouth of the estuary to ensure an open and navigable inlet. These efforts continued from 1920s until completion of a single jetty in 1941 (Behrens et al., 2013), after which there were no further attempts to develop the estuary in this way. Also, development of agriculture and towns in the watershed led to an increase in the use of the river as a large-scale water source, with inter-basin transfer from the Eel River to the upper Russian River implemented in 1908 (and increased later), enhancing summer flows. The local salmon fishery was also important.
- *From 1940 to 1960:* increasing value of the river as a source of freshwater with construction of Coyote Dam in 1958. The first records of breaching are from the 1950s, implemented on an ad hoc basis by local residents.
- *From 1960 to 1980: t*ransition to explicit management by public agencies occurred

in the 1960s and breaching events were implemented by these agencies – and the inlet was also dredged to create a deeper channel. A daily record of mouth state and harbour seal presence was started in 1973 by local resident Elinor Twohy.

- *From 1980 to 2010:* ongoing increase in the use of the river for water supply led to construction of the Warm Springs Dam in 1982 and increasing interest in the river and estuary as habitat led to implementation of minimum flow requirements in 1986. The majority of breaching events were artificial (Figure 5.4), conducted by the SCWA, and the current breaching protocol was implemented in 1994 following the adoption of Agreement 1610. Also recreational use of the estuary and lower river increased with beach-going, kayaking and swimming becoming popular in summer months. During this period, many near-water residences were connected to centralised sewage systems, removing septic systems from impacting the lower river and estuary. This period also saw an increase in conservation interests and the role of conservation stakeholders in the system, specifically the formation of the Russian Riverkeeper organisation in 1995.
- *From 2010:* with declining salmon populations and recognition of the critical role of estuarine habitat for juvenile steelhead, the National Marine Fisheries Service wrote a Biological Opinion that obligates the SCWA to manage the estuary mouth in a way that promotes formation of a perched estuary with high water levels and retention of a thick freshwater/brackish water layer.

5.6. Towards systems understanding in management

In this case study we have described recent monitoring of the Russian River Estuary (RRE) and the capturing of this information in a management-oriented habitat-volume index. This decision-support tool explicitly states the goals of steelhead-oriented management and provides a metric that captures much of the biophysical space and time variability. Ongoing discussion and tool development is directed at potential use in forecasting the impact of policy choices, management decisions and changing environment (including climate change) on steelhead habitat and at potential use as an operational decision-support tool. And in doing this, we should guard against managing for an optimal solution when the habitat has always exhibited high variability that is a critical element of the habitat diversity that supports a genetically diverse fish population – a key attribute of population resilience in changing environments.

While the approach is to consider the entire biophysical system in which juvenile steelhead are reared, it remains a single-species approach and may overlook or undervalue many other important dimensions of a complex social-ecological system. The value of taking a systems approach is well appreciated, linking multiple issues and factors. But to date there has been no rigorous systems analysis of the RRE to integrate biophysical and socio-economic considerations. Without this analysis we are left with questions whether the species-specific approach driven by the 2004 Biological Opinion is diminishing the ecosystem or social value in other ways. And, if there are concurrent losses, there are questions whether this is a desirable (or at least acceptable) trade-off for individual stakeholder groups and for a combined stakeholder forum. Beyond conducting a systems analysis, there is the challenge of transferring this insight to management and policy institutions, with the aim being improved delivery of ecological services both on long-term and more immediate time scales.

Towards this broader aim, it will be useful to identify how scientific and stakeholder knowledge is actually used in management and policy decisions on the RRE – and how it could be used better. What changes are needed in the role of scientists, advisors, and managers to accommodate this? And how do we all work with nature to allow the natural variability to continue in a human-impacted system?

5.7. References

Behrens, D. K., Bombardelli, F. A., & Largier, J. L. (2016). Landward Propagation of Saline Waters Following Closure of a Bar-Built Estuary: Russian River (California, USA). *Estuaries and Coasts*, 39(3), 621–638. https://doi.org/10.1007/s12237-015-0030-8

Behrens, D. K., Bombardelli, F. A., Largier, J. L., & Twohy, E. (2009). Characterization of time and spatial scales of a migrating rivermouth. *Geophysical Research Letters*, 36(9), L09402. https://doi.org/10.1029/2008GL037025

Behrens, D. K., Bombardelli, F. A., Largier, J. L., & Twohy, E. (2013). Episodic closure of the tidal inlet at the mouth of the Russian River - A small bar-built estuary in California. *Geomorphology*, 189, 66–80. https://doi.org/10.1016/j.geomorph.2013.01.017

Bond, M. H. (2006). Importance of Estuarine Rearing to Central California Steelhead (Oncorhynchus mykiss) Growth and Marine Survival (Masters thesis). University of California, Santa Cruz.

Boughton, D., Fuller, J., Horton, G., Larson, E., Matsubu, W., & Simenstad, C. (2017). Spatial structure of water-quality impacts and foraging opportunities for steelhead in the Russian River Estuary: an energetics perspective (Report no. NOAA-TM-NMFS-SWFSC-569). https://doi.org/10.7289/V5/TM-SWFSC-569

ESA. (2016). Russian River Estuary Outlet Channel Adaptive Management Plan 2016: Prepared for Sonoma County Water Agency. Retrieved from https://evogov.s3.amazonaws.com/185/media/164926.pdf

Gleason, M. G., Newkirk, S., Merrifield, M. S., Howard, J., Cox, R., Webb, M., ... Carter, J. (2011). A Conservation Assessment of West Coast (USA) Estuaries. Arlington, VA.

Hayes, S. A., Bond, M. H., Hanson, C. V., Freund, E. V., Smith, J. J., Anderson, E. C., ... MacFarlane, R. B. (2008). Steelhead Growth in a Small Central California Watershed: Upstream and Estuarine Rearing Patterns. *Transactions of the American Fisheries Society*, 137(1), 114–128. https://doi.org/10.1577/t07-043.1

Hewett, K. M. (2014). Oxygen dynamics of the Russian River estuary during periods of inlet closure (Masters thesis). University of California Davis.

Largier, J. L., & Behrens, D. (2010). Hydrography of the Russian River Estuary, Summer-Fall 2009, with Special Attention on a Five-Week Closure Event: UC Davis Report to Sonoma County Water Agency. Bodega Marine Laboratory, University of California Davis. Retrieved from https://www.yumpu.com/en/document/read/26153998/hydrography-of-the-russian-river-estuary-sonoma-county-water-

Largier, J. L., O'Connor, K., & Clark, R. (2019). Considerations for Management of the Mouth State of California's Bar-Built Estuaries (Final Report). National Marine Fisheries Service - West Coast Region. Retrieved from https://www.fisheries.noaa.gov/resource/

document/considerations-management-mouth-state-californias-bar-built-estuaries-0

Largier, J. L., & Koohafkan, M. (2016). Calculation of Volume of Juvenile Steelhead Habitat in Russian River Estuary during Closure Events and Development of Habitat Browser: UC Davis Report to Sonoma County Water Agency. University of California Davis.

Matsubu, W., Simenstad, C., Horton, G., & Largier, J. (2019). Juvenile steelhead display an ontogenetic shift in environmental preferences in an intermittently closed estuary (in review). *Transactions of the American Fisheries Society.*

Southern California Wetlands Recovery Project. (2018). Wetlands on the Edge: The Future of Southern California's Wetlands: Regional Strategy 2018. Oakland, California: Prepared by California State Coastal Conservancy. Retrieved from https://scwrp.org/wp-content/uploads/2018/10/WRP-Regional-Strategy-2018-100518_lowRes.pdf

Serghesio, E. E. (2011). The Influence of an Intermittently Closed, Northern California Estuary on the Feeding Ecology of Juvenile Steelhead (Oncorhynchus mykiss) and Chinook Salmon (Oncorhynchus tshawytscha) (Masters thesis). University of Washington.

Slinger, J. H. (1996). Modelling the physical dynamics of estuaries for management purposes (Doctoral dissertation). University of KwaZulu-Natal, Pietermaritzburg, South Africa.

Slinger, J. H. (2017). Hydro-morphological modelling of small, wave-dominated estuaries. *Estuarine, Coastal and Shelf Science,* 198(B), 583–596. https://doi.org/10.1016/j.ecss.2016.10.038

Slinger, J. H., & Largier, J. L. (1990). The evolution of thermohaline structure in a closed estuary. *Southern African Journal of Aquatic Sciences*, 16(1–2), 60–77. https://doi.org/10.1080/10183469.1990.10557367

Sutula, M. (2011). Review of Indicators for Development of Nutrient Numeric Endpoints in California Estuaries (Technical Report no. 646). Costa Mesa, CA: Southern California Coastal Water Research Project.

6

Advancing Mouth Management Practices in the Groot Brak Estuary, South Africa

By Lara van Niekerk, Janine Adams, Susan Taljaard, Piet Huizinga and Stephen Lamberth

6.1. Motivation for management

The early 1990s in South Africa were characterised by a strongly hierarchical and technocratic regime where planning and development decisions regarding the environment were made at national government level with little or no public consultation (Slinger et al., 2005). One such a decision was the construction of the 70 m high and 270 m wide Wolwedans Dam (with a capacity of 23×10^6 m^3) only 3 km upstream of the Groot Brak Estuary (also known as the Great Brak Estuary) by the South African Department of Water Affairs and Forestry (DWAF) (Figure 6.1). However, then the local community of the Town of Groot Brak feared the effects of reduced water supply on the health of the estuary, as well as the risk of flooding during dam failure. Increasing public pressure, and consequent media coverage, culminated in the DWAF setting up a steering committee, the Groot Brak River Environmental Committee (GEC). This committee was tasked to investigate the effect of the dam on the estuary, and to establish a management

Figure 6.1. Wolwedans Dam, just upstream of Groot Brak Estuary (Source: DWS, South Africa)

plan for the optimal use of the reserved water (1 x 10^6 m^3) to maintain current ecological health. The Council for Scientific and Industrial Research (CSIR) were commissioned to undertake this assessment (CSIR, 1990; Slinger et al., 2005).

In 1994 South Africa became a deliberative democracy, with a strong emphasis on environmental protection and public participation in decision making. Stringent environmental legislation followed, for example the National Water Act (Act 36 of 1998) (Republic of South Africa, 1998) that gave the country's water resources a 'right' to water for the protection of the aquatic ecosystem functions. This Act led to the development of official methods to determine ecological water requirements (freshwater requirements) of these resources which included estuaries. The National Environmental Management: Integrated Coastal Management Act (ICM) (Act 24 of 2008) (Republic of South Africa, 2009) was another important piece of legislation that facilitated integrated coastal and estuarine management in South Africa. From this came the National Estuarine Management Protocol (NEMP) that sets norms and standards for estuarine management, for example the requirement to develop estuarine management plans for all estuaries in the country (including mouth management plans). It was within this strong pro-environmental legal framework that estuarine scientists in South Africa were able to improve ecosystems understanding and incrementally advance mouth management practices in the Groot Brak Estuary – the focus of this case study (CSIR, 1992, 1993, 1994, 1998, 2003, 2011). The refinement of mouth breaching practices in the Groot Brak Estuary, therefore, followed the adaptive management paradigm, acknowledging that incomplete knowledge on the system needs to be supplemented through long-term monitoring, and that management practices then need to be amended in accordance with the developing understanding to meet environmental objectives. Although not the focus of this study - and therefore not elaborated in this chapter - the adaptive management process was driven by a complex stakeholder coalition involving scientists, government and local citizens (Taljaard et al., 2012).

6.2. Study area

The Groot Brak Estuary (34°03'23"S; 22°14'18"E) is a temporarily open/closed estuary (TOCE) situated along the warm temperate south coast of South Africa (Figure 6.2). The estuary is about 6.2 km long covering an area of about 0.6 km^2 at high tide with a tidal prism about 0.3 x 10^6 m^3. The lower estuary is relatively shallow (0.5 m to 1.2 m deep) with a few deeper scour holes near the rocky cliffs and bridges. The middle and upper estuary is deeper (~2 m water depth), but also has scour holes (3-4 m water depth) (Slinger et al., 2017). The mouth of the Groot Brak Estuary generally closes when high waves coincide with periods of low river flow. The estuary mouth is bound by a low rocky headland on the east and a sand-spit to the west. Immediately inland of the mouth, the estuary widens into a basin containing a permanent island with dimensions of about 400 m x 250 m.

The biodiversity importance of the Groot Brak Estuary is not high on a national scale, and it does not enjoy any statutory protection status (Turpie et al., 2002). However, the system is one of the larger TOCEs in the country with a high diversity of estuarine habitats (including salt marsh) and is of national importance as a fish nursery (Van Niekerk et al., 2015). The Groot Brak is a popular coastal destination along the Garden Route, both for holiday makers and retirees. The permanent residents and the tourists rely on the estuary for recreation such as swimming, boating and fishing, with some limited subsistence use. Over the years low lying developments have been permitted by the municipality and, as a result, artificial breaching now also serves to prevent flooding of these properties.

6.3. Inlet dynamics

The Groot Brak Estuary is isolated from the sea by the formation of a sand berm across the mouth during periods of low, or no river inflow. The estuary stays closed until the basin fills up and the berm is breached, either as a result of high water level or flooding. The mouth of the Groot Brak Estuary generally closes when high waves coincide with periods of low river flow. High wave energy in conjunction with the generous supply of marine sand along this part of the coast suspends large amounts of marine sediment which is then transported into the estuary on the flood tides and deposited in the narrow mouth (Slinger et al., 2017). Under natural run-off condition breaching would have occurred when the water level in the estuary exceeded the berm height at levels estimated between +3.0 m and +3.5 m to MSL. The subsequent outflow velocities from the estuary would have been high, flushing large volumes of sediment out to sea. At such high berm levels back-flooding would have been extensive. Accordingly, artificial breaching has been practiced at Groot Brak for more than a century to reduce flooding in the floodplain.

When freshwater inflow to this system was dramatically reduced as a result of the construction of the Wolwedans Dam, the ability of the estuary to breach naturally was compromised. In an attempt to restore some of the natural functionality, the tactic of artificial breaching was introduced. The main difference between artificial and natural breaching is that in the case of artificial breaching, a channel is excavated at a pre-determined water level which allows outflow to begin earlier and usually at a lower water level than would be the case naturally. However, artificial breaching at lower levels compared with natural breaching, results in a reduced volume and duration of water flow out to sea, and reduced sediment scouring. Indeed, the sediment flushing potential

Figure 6.2. Groot Brak Estuary, indicating the estuarine functional zone (below +5 m MSL contour) (from: Google Earth)

is known to decrease exponentially with a decrease in outflow velocities after breaching (Beck & Basson, 2008). This means that the long-term sediment erosion/depositional cycles in the estuary are altered as wave action still deposits the same volume of marine sediment in the mouth, but less is scoured out by artificial breaching. In the long-term, the tactic of artificial breaching therefore results in increased sedimentation in the lower estuary. There is an additional complication. As even more low-lying development occurs there is a need to breach artificially at even lower levels. This reduces the volume of sediment flushed from the mouth even more, posing a major threat to the inlet dynamics and sedimentation processes in this estuary in the long-term. The challenge to estuarine scientists, therefore, is to design artificial breaching practices acknowledging the ever increasing human pressures on the estuary.

6.4. Building systems understanding for management

Following the construction of the Wolwedans Dam, a volume of 1 x 10^6 m^3 per annum was allocated as environmental flows to maintain connectivity and estuarine ecological function in the Groot Brak Estuary (Slinger et al., 2005). Initially the environmental flow allocation was not envisaged as an adaptive management process. However, through greater awareness of potential environmental consequences, it evolved into an adaptive management process in which new scientific learning was incrementally embedded in refining the mouth management practices. The scientific learning is reported in Table 6.1, with an overview of reports and published articles on the Groot Brak Estuary summarising major findings on hydrodynamic, water quality and ecological responses.

Table 6.1. Reports and published articles on the Groot Brak Estuary summarising major findings on hydrodynamic, water quality and ecological responses.

Knowledge gained	Reference
Estuary environmental study of the effects of the Wolwedans Dam on the Groot Brak Estuary. Reports on the creation of a water release management plan for the estuary along with future monitoring requirements. The plan also takes into account the socio-economic effects of the dam construction.	EMATEK (1990)
The hydrodynamics and water quality of the Groot Brak Estuary were studied under various seasonal inflow conditions.	Taljaard & Slinger (1993)
The effects of a planned freshwater release in summer 1990 on the water quality of the Groot Brak Estuary was monitored. Only the surface water was flushed by the release; efficacy of flushing was enhanced by tidal intrusion. A volume of freshwater comparable to the volume of the tidal prism should be released.	Slinger et al. (1994)
The integrated research approach of the Consortium for Estuarine Research and Management (CERM) is discussed along with the importance of an integrated modelling approach when considering the freshwater requirements of estuaries.	Slinger & Breen (1995)
A discrete simulation model for the dynamics of a submerged macrophyte, *Zostera capensis* is presented. This can be used to analyse the response to various freshwater inflow scenarios and mouth breaching scenarios.	Wortmann et al. (1997)
A cellular automation model was used to analyse the distribution and growth of estuarine macrophytes (*Zostera capensis*, *Ruppia cirrhosa* and *Phragmites australis*). It models vegetative spread and above ground biomass.	Wortmann et al. (1998)
Overview of the management programme, flow releases, mouth condition and transect monitoring for salt marsh and invertebrates.	Huizinga (2003)
The review of the original management programme highlighted importance of open mouth condition for mud prawn and marsh crabs. Under these conditions the salt marsh flourishes and brackish submerged macrophytes are replaced by seagrass (*Zostera capensis*).	CSIR (2004)
The Contingent Valuation Method (CVM) was applied to the Groot Brak Estuary under various inflow scenarios. This method uses the economic and social value of changes in estuarine services and a 'willingness to pay' for water quality changes.	Dimopoulos (2005)
Overview of the lessons learnt from the construction of the Wolwedans Dam, its management plan, monitoring and the revision of the environmental management plan to incorporate greater public participation.	Slinger et al. (2005)
Preliminary Environmental Flow Requirement study (ecological water requirement study. Detailed sampling in winter and summer showed that biotic stress occurred during the closed mouth winter periods as a result of hypoxia/anoxia and elevated ammonium levels. Monitoring protocol adjusted to reflect this stress.	Department of Water Affairs and Forestry (2008)
The first record of the invasive grass in South Africa (*Spartina alterniflora*) is documented in the Groot Brak Estuary. Sediment and plant characteristics were reported for the years 2009 to 2011.	Adams et al. (2012)
The role of monitoring information on policy-orientated decisions are reviewed. Results showed that monitoring information generally focusses on issues relating to core responsibility of decision makers and often overlooks other role players.	Hermans et al. (2013)

Table 6-1 (continued). Reports and published articles on the Groot Brak Estuary summarising major findings on hydrodynamic, water quality and ecological responses.

Knowledge gained	Reference
The decomposition characteristics of the dominant submerged macrophytes *Zostera capensis* and *Ruppia cirrhosa* and a macroalga *Cladophera glomerata* are documented. There was a high release of DIN within 28 days and this increased with temperature. The study showed that the health of the Groot Brak deteriorates during macroalgal blooms.	Lemley et al. (2014)
The effect of a prolonged period of mouth closure (2009 to 2011) was investigated. Nutrient input into the Groot Brak Estuary has increased due to urbanisation and industry development resulting in an increase of micro- and macroalgae. Reduced tidal exchange further exacerbates the problem. Macroalgal mats smothered salt marsh and open water where submerged macrophytes would normally establish.	Nunes & Adams (2014)
A multi-metric approach was used to classify estuaries with variable nutrient inputs. Indicators (dissolved oxygen, DIN, DIP, phytoplankton, epiphytes and microphytobenthos) were used to develop critical thresholds to classify estuaries into good, fair and poor conditions. The Groot brak Estuary has high daily inorganic nutrient input and low flushing time. The overall classification was given as fair to poor.	Lemley et al.(2015)
Benthic regeneration; the flux of inorganic nutrients, as well as total N and P were investigated across the sediment-water interface using microcosms. The estuary was found to act as a source of N and P in both summer and winter and any increased organic load would increase this flux. The authors recommended that increased flushing would reduce this.	Human, Snow, Adams, & Bate (2015)
A nutrient budget approach of the Groot Brak Estuary was used to determine the effect of submerged macrophytes and macroalgae on the storage of N and P in the estuary during 2011. Sediment contributed between 30 to 40 % of nutrients while the macrophytes between 20 to 38 %.	Human et al. (2015)
The adaptive potential of the invasive grass *Spartina alterniflora* is documented as well as the effectiveness of control measures on the plant.	Adams et al. (2016)
The effects of an artificial breach versus a natural breach are documented. Artificial breaching did not flush the estuary as well as the natural breach and as a result a macroalgal bloom still persisted. This was flushed out after the natural breach/flood event.	Human et al. (2016)
Synthesis of management/control and complete removal of the invasive halophytic grass *Spartina alterniflora* between 2013 to 2015. Regular and repeated control has greater success than previous attempts.	Riddin et al. (2016)
The hydro-morphological modelling of small, wave dominated estuaries is discussed. A simple, parametric, system dynamics model to simulate the opening and closure of the mouths of small, wave-dominated estuaries is reported on.	Slinger (2017)
Modes of flushing in the Groot Brak Estuary - salinity, temperature, dissolved oxygen and nutrient changes before, during and after a breach.	Slinger et al. (2017)

This informed mouth management practices in the estuary, as well as associated environmental flow allocations, as is illustrated in Table 6.2.

Table 6.2. Incremental progress in systems understanding and subsequent adaptation of inlet management practices in the Groot Brak Estuary – driven by various assessment and legislative processes

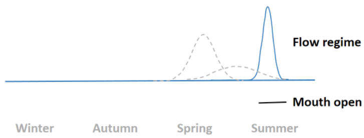
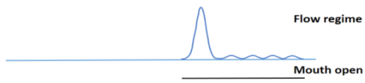
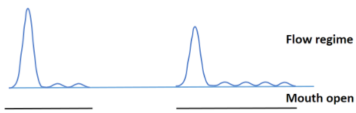
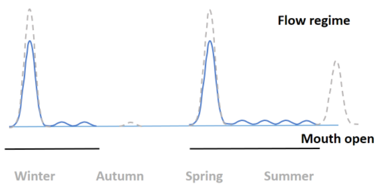
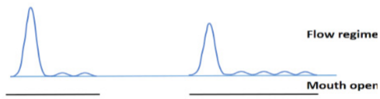
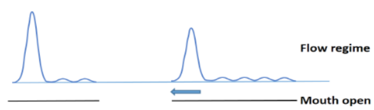
Progress in system understanding	Adaptation of mouth management practice
1988: Confirmation of Construction of Wolwedans Dam (1988) for oil refinery and urban development	
» Government allocate 1 x 10^6 m^3 per annum as environmental flows to maintain estuarine ecological function.	
1990: Environmental Impact Assessment (EIA)	
» Realised annual volume of 1 x 10^6 m^3 was insufficient to keep estuary mouth open for long periods » First link made between river flow and mouth dynamics, previous perception was that high waves open mouth, but monitoring showed differently » Invertebrate and fish recruitment require open mouth condition early summer (September/October) » Early summer breaching partially ties in with aesthetic requirements for holiday season providing tidal flushing to limit presence of excessive macroalgal blooms and objectionable smells.	 Optimum distribution of 1 x 10^6 m^3 to maintain mouth open in spring/summer.
2004: Review of Original Management Programme (as required in EIA study)	
» Long-term monitoring of mouth dynamics and water level showed importance of breaching at the highest water level possible to scour sediment from the estuary and keep mouth open for longer. » Small releases at neap tide enhance tidal flow that, subsequent to breaching, can further extend period of mouth opening » At onset of mouth closure a relatively small pulse release from the dam can be used to re-open the inlet, rather than a major release at a later stage when the berm has built up again. » At onset of mouth closure a relatively small pulse release from dam can be used to re-open the inlet, rather than conducting a bigger release to breach at a later stage when the berm has built up again. » Abundance of mud prawns (e.g., *Upogebia africana*) and marsh crabs (e.g., *Sesarma catenata*) fluctuate significantly depending on duration of mouth closure. These invertebrates have a marine larval development phase which ceases to happen if mouth closure extends across their summer recruitment period (Wooldridge, 1994). » Fluctuating water levels during open mouth periods were found to decrease submerged macrophyte biomass and extent in the estuary whereas high water level for more than 3 months caused die-back of the succulent salt marsh (Adams et al., 1999).	 Neap tide release increase open period.  Actual releases (based on surplus water being available during this period). Government provisionally allocated 3 - 4 x 10^6 m^3 (volume used).

Table 6-2 (continued). Incremental progress in systems understanding and subsequent adaptation of inlet management practices in the Groot Brak Estuary – driven by various assessment and legislative processes

Progress in system understanding	Adaptation of mouth management practice
2008: Preliminary Environmental Flow Requirement study (requirement under National Water Act)	
» Seasonal monitoring of abiotic and more extensive biological study (i.e., including fish and birds for the first time) showed biotic stress during closed periods as a result of hypoxia/anoxia and elevated ammonia levels. » Thus, earlier assumptions that only monitoring of the system in summer will pick-up worst-case condition after the summer holiday was incorrect, as typical winter closed periods (associated with hypoxia/ anoxia and elevated ammonia) can be even more detrimental to the ecology. » Proper assessment of flow requirements showed 11 x 10^6 m^3 was realistic to prevent further deterioration in estuary health. » Effects of drought on ecological health were evident during 2009-2010 when no flow releases were made to the estuary.	 Recommended ecological flow requirements (dotted line - 11 x 10^6 m^3) relative to provisional allocation (at the time).
2014/2015: Development of Estuary Management Plan (requirement under ICM Act)	
» Human et al. (2016), Human, Snow, Adams, & Bate (2015), Human, Snow, Adams, Bate, et al. (2015) and Slinger et al. (2017) showed inability of current mouth breaching practices to effectively flush estuary, thus not accounting for re-setting water quality in the entire system. » Identified need to improve stormwater runoff to estuary as a means of reducing additional nutrient and organic inputs, thus also addressing management of riparian activities, in addition to mouth management » Filamentous macroalgae showed to be an important storage compartment for nutrients, releasing it back into estuary through benthic remineralisation » Excessive macroalgal blooms cause dieback of salt marsh during closed periods (Nunes & Adams, 2014). » Without effective flushing estuary will continue to experience macroalgal ‘boom-and-bust’ cycles (Human et al., 2016; Human, Snow, Adams, & Bate, 2015; Human, Snow, Adams, Bate, et al., 2015).	 Recommended flow allocation was not approved, allocation remained at 3 - 4 x 10^6 $m^{3.}$
2017: Development of Estuary Mouth Management Plan (supplementary to EMP under ICM Act)	
» Identified the need to commence mouth breaching in early spring (1 September) to improve juvenile fish recruitment.	 Shifted breaching to early September to accommodate juvenile fish recruitment.

The original environmental impact assessment study commissioned by the government in 1990, set out to investigate the impact that the construction of the dam would have on the estuary, as well as to develop a management plan to mitigate, as best as possible, the envisaged impacts – the 1st Groot Brak Estuary Management Plan. This study included the evaluation of hydrodynamics, sediment processes and water quality (e.g., nutrients, dissolved oxygen, turbidity), but also two ecological components, namely estuarine macrophytes and invertebrates. It became one of the first environmental studies in South Africa that included ecological components in an assessment of this nature. Initially high waves were assumed to assist with keeping the inlet open, but monitoring showed the opposite – the mouth closed under high wave conditions. A socio-economic assessment was also included to identify and account for local community needs in the future management of the system; for example, having good water quality conditions during holidays and no nuisance algal blooms. A 1-dimensional hydrodynamic model was used to numerically simulate salinity distributions in the estuary for different freshwater flow regimes. Monthly freshwater flow scenarios for a 64-year period were simulated for natural, pre-Wolwedans Dam, and post-Wolwedans Dam hydrological flow distributions. Specifically, the effect of various flow scenarios on mouth state was also included in this assessment to inform a mouth breaching plan for later implementation by responsible managers. The primary objective of the initial Groot Brak Management Plan was to maintain connectivity and ecological health as best as possible. Specifically, the mouth had to be opened during early summer (October) to allow invertebrate and fish recruitment. Early summer breaching partly tied in with requirements from the socio-economic assessment, namely to have good water quality, limited presence of excessive macroalgal blooms and objectionable smells during the holiday season. This was achieved through tidal flushing during the open state. The study also included the development of a detailed long-term environmental monitoring programme (including mouth conditions, river flow, water levels, water quality, vegetation, invertebrates) to guide refinement of the initial findings in accordance with the adaptive management paradigm. A key conclusion, however, was that the 1 x 10^6 m^3 allocation environmental flow to the estuary was insufficient to maintain ecological health, especially during dry years.

The review of the 1st Groot Brak Estuary Management Plan in 2004 represented the next major refinement focussing on mouth management (CSIR, 2004). The review included an assessment of the ecological status of the system (in terms of hydrodynamics, water quality, estuarine macrophytes and invertebrates), but also of the public perceptions on the effectiveness of mouth management practices. Breaching procedures were upgraded to incorporate incremental learning from the monitoring over the preceding 10 years. One of the refinements was the requirement to increase pre-breaching water levels as high as possible to optimise sediment scouring for the lower estuary and mouth, and to extend the duration of the open state. Breaching the estuary a few days prior to spring tide to obtain optimum tidal flushing and facilitate the development of an outflow channel was another refinement. Breaching at neap tides was found to lead to premature closure. The best moment to breach is at high tide, or as near as possible after high tide, waves permitting. If it appears unlikely that waves will interfere, breaching can even be undertaken up to two hours earlier to allow for good channel formation. Small pulse releases over neap tides (~0.4 $m^3.s^{-1}$ for 4 to 6 days) were also to be found sufficient to

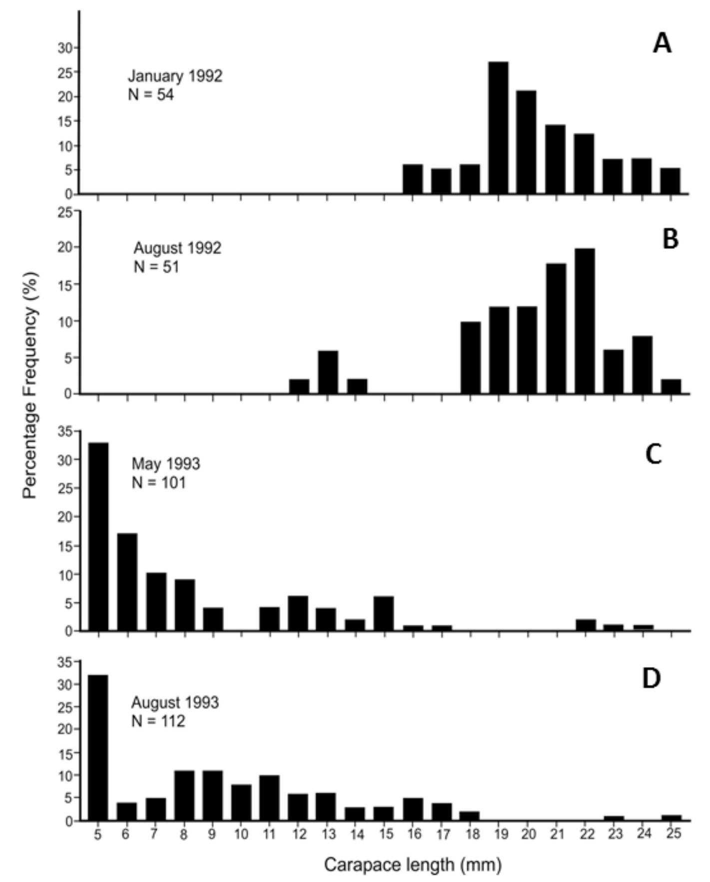


Figure 6.3. Size class distribution of Upogebia africana population in the Groot Brak Estuary on four sampling occasions (Jan 1992, Aug 1992, May 1993 and Aug 1993) (DWAF, 2008)

maintain open mouth conditions. Monitoring revealed that at the onset of mouth closure, a relatively small pulse release can re-open the inlet. Following the review process, the Groot Brak Management Plan was updated, including mouth management practices that incorporated new learning. The roles and responsibilities of all the relevant parties assisting with regular and emergency breaching were specified.

An open mouth is also important for the input of larvae into the estuary from the marine environment for recruitment and vice versa. Estuaries serve as nursery for estuarine and marine fish. Migration of juvenile fish into estuaries requires open mouth conditions in spring and summer. The abundance of mud prawns (e.g., *Upogebia africana*) and marsh crabs (e.g., *Sesarma catenata*) fluctuate significantly depending on the duration of mouth closure. These invertebrates have a marine larval development phase which ceases to happen if mouth closure extends across their summer recruitment period (Wooldridge, 1994). The impact of extended mouth closure over a 28-month period on *Upogebia africana* recruitment is shown in Figure 6.3. Recruitment ceased during this period resulting in discontinuous distribution of size classes. Recruitment only occurred when

Figure 6.4. Macroalgal blooms (left) and succulent salt marsh (right) in the Groot Brak Estuary (Photo credit: Janine Adams)

the mouth was open to the sea. The smallest cohorts (16-18 mm carapace length) again sampled in January 1992 were about 17 - 21 months old, indicating that recruitment last occurred in April 1990, shortly before the period of extended mouth closure. Some recruitment occurred in November 1991 when the mouth opened for 30 days; reflected in the three smallest size classes (12-14 mm carapace length) shown in Figure 6.3. Good recruitment occurred after the mouth again opened during most of the summer of 1993 (2C and D) (DWAF, 2008).

The Groot Brak Estuary has extensive salt marsh areas which germinate in spring and summer (Figure 6.4). Open mouth conditions during this period facilitate the development of intertidal habitat for salt marsh as well as reed and sedge growth. Whereas high water level during closed mouth conditions for greater than 3 months caused die-back of the succulent salt marsh (Adams et al., 1999). Open mouth conditions are linked to increased salinity values and opportunities for marine and brackish invertebrate species to increase in biomass and abundance if salinity increases from a low base (<10). Artificial breaching and lower water levels advantage waders and piscivorous birds that are associated with tidal and more saline conditions. While closed mouth conditions and associated high water levels and lower salinities tend to advantage waterfowl species and the spread of submerged macrophytes.

In 2007/08 an ecological water requirement (EWR) study was conducted on the Groot Brak Estuary to review and determine the ecological water requirements of the system (DWAF, 2008). As part of this study a summer (open mouth) and winter (closed mouth) assessment was done on nine parameters – hydrology, hydrodynamics, water quality, physical habitat, microalgae, macrophytes, invertebrates and for the first-time fish and birds. The study showed that while the system was functioning relatively well under the open conditions, it showed signs of severe stress under the closed state. This was especially the case for the higher trophic levels which were responding to poor water quality such as anoxia and high ammonium concentrations. This study also showed that earlier assumptions that only the summer conditions (after peak holiday season) need to be monitored to track ecosystem condition were erroneous and provided a false picture of the condition of the estuary.

Also emerging from the EWR study was the importance of addressing ecosystem functions, such as important fish nursery habitat. In addition to the general benefit estuaries provide to coastal and estuarine fish, some of South Africa's important and often critically overexploited fish species (<5% of historical levels) are dependent on estuaries in their first year (e.g., white steenbras and dusky kob). Although estuaries should not be managed for single species, nursery requirements of threatened species need to be addressed, especially in the larger systems, in an effort to rebuild these fish stocks.

In 2018, a comprehensive Estuarine Management Plan (a legal requirement under the Integrated Coastal Management Act) was developed for the estuary that incorporates all incremental leaning to date and more of the social-economic considerations (Anchor Environmental, 2018). For example, this plan highlights the importance of nutrient management and water quality monitoring and the need to reduce nutrient inputs, as well as proposing a zonation plan for the estuary.

Also in 2018, the provincial government (Western Cape Province) initiated a process to review all the estuary mouth breaching plans in the province, including the Groot Brak Mouth Management Plan. The Groot Brak Estuary served as a critical case study and formed the 'Blue Print' for most of the other breaching plans, but with relevant, site-specific adjustments. These plans are intended to assist with acquiring pre-approval for artificial breaching under South Africa's Environmental Impact Assessment legislation. The plans also clearly assign roles and responsibilities to ensure that there is no unnecessary confusion during emergencies, e.g., floods. The review process reinforced the need for long-term monitoring and regular feedback to residents and other concerned parties. Under current legislation all mouth management plans needs to be reviewed every 5 years. An important update in the Groot Brak Mouth Management Plan, based on scientific learning through monitoring, was that breaching should be done in early September to accommodate the peak fish recruitment window (September to November) (unpublished data, Department of Environment, Forestry and Fisheries).

Long-term monitoring of the water levels in the estuary has shown that flooding can occur through river flows from land, but also as a result a result of high waves and storm

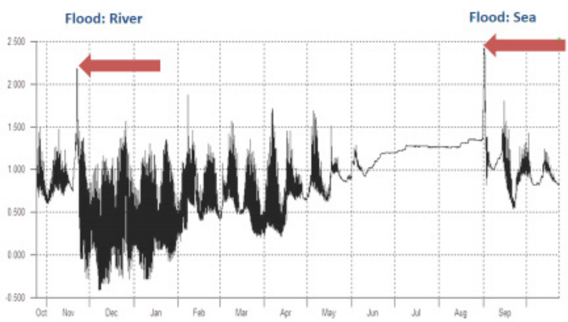


Figure 6.5. Water levels to mean sea level in the Groot Brak Estuary from November 2007 to November 2008 showing major flooding events from the land (river flow) and sea

surges from the sea, especially during winter. Water level data from 2007/2008 shows one of the five highest water levels ever recorded in the system was as a result of high waves that occurred in late winter (Figure 6.5). Therefore, keeping the inlet closed during winter also acts as a buffer to dampen flooding effects on the estuary from winter storms at sea.

Artificial breaching is not advocated as a solution to water quality problems (e.g., low oxygen levels) under the Mouth Management Plan. Instead water quality problems should be fixed at source as the reductions in water level associated with breaching can expose the remaining fauna to even poorer water quality conditions post-breaching. If the water depth of an estuary is lowered, the buffer between oxygen-depleting organic bottom material and the surface layer that can be oxygenated by wind is reduced. A similar response can occur in eutrophic estuaries where low oxygen events are linked to high loads of decaying plant material.

6.5. Concluding remarks

The Groot Brak Estuary has been negatively impacted by flow reduction (Wolwedans Dam), artificial breaching at low water levels, increased nutrient loading (e.g., agricultural return flow, storm water and septic tanks), ongoing sedimentation and high fishing pressure. Over 30 years of monitoring and research has resulted in more 20 scientific publications on the system to validate the science and disseminate the insights gained. This in-depth system understanding has incrementally advanced the mouth management practices over the decades. In addition, the South African legislative framework has provided the opportunity to integrate the learning on mouth breaching into formal approaches, with specified roles and responsibilities.

6.6. References

Adams, J. B., Bate, G. C., & O'Callaghan, M. (1999). Primary Producers. In B. R. Allanson & D. Baird (Eds.), Estuaries of South Africa (pp. 91–117). Cambridge, United Kingdom: Cambridge University Press.

Adams, J. B., Grobler, A., Rowe, C., Riddin, T., Bornman, T. G., & Ayres, D. R. (2012). Plant traits and spread of the invasive salt marsh grass, Spartina alterniflora Loisel., in the Great Brak Estuary, South Africa. *African Journal of Marine Science*, 34(3), 313–322. https://doi.org/10.2989/1814232X.2012.725279

Adams, J. B., van Wyk, E., & Riddin, T. (2016). First record of Spartina alterniflora in southern Africa indicates adaptive potential of this saline grass. *Biological Invasions*, 18(8), 2153–2158. https://doi.org/10.1007/s10530-015-0957-5

Anchor Environmental. (2018). Great Brak River Estuarine Management Plan (draft).

Beck, J. S., & Basson, G. R. (2008). Klein River Estuary (South Africa): 2D numerical modelling of estuary breaching. *Water SA*, 34(1), 33–38. https://doi.org/10.4314/wsa.v34i1.180759

Council for Scientific and Industrial Research. (1990). Great Brak River Estuary: Estuary environmental study with reference to a management plan for the Wolwedans Dam and the Groot Brak River mouth (CSIR Report EMAS-C 9036). Stellenbosch, South Africa.

Council for Scientific and Industrial Research. (1992). Great Brak management programme. Interim report: Report on the monitoring results for the period April 1990 to March 1992 (CSIR Report EMAS-C 93051). Stellenbosch, South Africa.

Council for Scientific and Industrial Research. (1993). Great Brak Estuary management programme. Interim report. Report on the monitoring results for the period April 1992 to March 1993 (CSIR Report EMAS-C 93051). Stellenbosch, South Africa.

Council for Scientific and Industrial Research. (1994). Great Brak management programme. Interim Report: Report on the monitoring results for the period April 1993 to March 1994 (CSIR Report EMAS-C 940013). Stellenbosch, South Africa.

Council for Scientific and Industrial Research. (1998). Great Brak management programme. Interim Report: Report on the monitoring results for the period April 1997 to March 1998 (CSIR Report ENV/S-C 98122). Stellenbosch, South Africa.

Council for Scientific and Industrial Research. (2003). Great Brak Estuary management programme: Review Report March 2003 (CSIR Report ENV-S-C-2003-092). Stellenbosch, South Africa.

Council for Scientific and Industrial Research. (2004). Great Brak Estuary management programme: Management Plan 2004 (CSIR Report ENV-S C 2004-111). Stellenbosch, South Africa.

Council for Scientific and Industrial Research. (2011). Great Brak Estuary management programme: Report on the monitoring results November 2011 (CSIR Report CSIR/NRE/ECOS/ER/2011/0098/B). Stellenbosch, South Africa.

Department of Water Affairs and Forestry. (2008). Reserve Determination studies for selected surface water, groundwater, estuaries and wetlands in the Outeniqua (Groot Brak and other water resources, excluding wetlands) catchment: Ecological Water Requirements Study – Estuarine RDM Report, Volume 1.1: As. Pretoria, South Africa.

Dimopoulos, G. (2005). Applying the contingent valuation method to value fresh water inflows into the Knysna, Great Brak and Little Brak estuaries (Masters thesis). Nelson Mandela Metropolitan University, Port Elizabeth.

EMATEK. (1990). Great Brak River environmental study with reference to a management plan for the Wolwedans Dam (CSIR Report EMA-C9036). Stellenbosch, South Africa.

Hermans, L. M., Slinger, J. H., & Cunningham, S. W. (2013). The use of monitoring information in policy-oriented learning: Insights from two cases in coastal management. *Environmental Science and Policy*, 29, 24–36. https://doi.org/10.1016/j.envsci.2013.02.001

Huizinga, P. (2003). The Great Brak Estuary management programme. Review Report (CSIR Report No. ENV-S-C-2003-092). Stellenbosch, South Africa: Council for Scientific and Industrial Research.

Human, L. R. D., Snow, G. C., & Adams, J. B. (2016). Responses in a temporarily open/closed estuary to natural and artificial mouth breaching. *South African Journal of Botany*, 107, 39–48. https://doi.org/10.1016/j.sajb.2015.12.002

Human, L. R. D., Snow, G. C., Adams, J. B., & Bate, G. C. (2015). The benthic regeneration of N and P in the Great Brak estuary, South Africa. *Water SA*, 41(5), 594–605. https://doi.org/10.4314/wsa.v41i5.2

Human, L. R. D., Snow, G. C., Adams, J. B., Bate, G. C., & Yang, S. C. (2015). The role of submerged macrophytes and macroalgae in nutrient cycling: A budget approach. *Estuarine, Coastal and Shelf Science*, 154, 169–178. https://doi.org/10.1016/j.ecss.2015.01.001

Lemley, D. A., Adams, J. B., Taljaard, S., & Strydom, N. A. (2015). Towards the classification of eutrophic condition in estuaries. *Estuarine, Coastal and Shelf Science*, 164, 221–232. https://doi.org/10.1016/j.ecss.2015.07.033

Lemley, D. A., Taljaard, S., Adams, J. B., & Strydom, N. A. (2014). Nutrient characterisation of river inflow into the estuaries of the Gouritz Water Management Area, South Africa. *Water SA*, 40(4), 687. https://doi.org/10.4314/wsa.v40i4.14

Nunes, M., & Adams, J. B. (2014). Responses of primary producers to mouth closure in the temporarily open/closed Great Brak Estuary in the warm-temperate region of South Africa. *African Journal of Aquatic Science*, 39(4), 387–394. https://doi.org/10.2989/16085914.2014.980773

Republic of South Africa. (1998, August 26). National Water Act, Act 36 of 1998. *Government Gazette of the Republic of South Africa*. Retrieved from http://www.energy.gov.za/files/policies/act_nationalwater36of1998.pdf

Republic of South Africa. (2009, February 11). National Environmental Management: Integrated Coastal Management Act, Act 24 of 2008. *Government Gazette of the Republic of South Africa*. Retrieved from https://www.environment.co.za/documents/legislation/NEMA-National-Environmental-Management-Act-Integrated-Coastal-Management-24-2008-G31884.pdf

Riddin, T., van Wyk, E., & Adams, J. (2016). The rise and fall of an invasive estuarine grass. *South African Journal of Botany*, 107, 74–79. https://doi.org/10.1016/j.sajb.2016.07.008

Slinger, J. H. (2017). Hydro-morphological modelling of small, wave-dominated estuaries. *Estuarine, Coastal and Shelf Science*, 198(B), 583–596. https://doi.org/10.1016/j.ecss.2016.10.038

Slinger, J. H., & Breen, C. M. (1995). Integrated research into estuarine management. *Water Science and Technology*, 32(5–6), 79–86. https://doi.org/10.1016/0273-1223(95)00649-4

Slinger, J. H., Huizinga, P., Taljaard, S., van Niekerk, L., & Enserink, B. (2005). From impact assessment to effective management plans: Learning from the Great Brak Estuary in South Africa. *Impact Assessment and Project Appraisal*, 23(3), 197–204. https://doi.org/10.3152/147154605781765562

Slinger, J. H., Taljaard, S., & Largier, J. L. (1994). Changes in estuarine water quality in response to a freshwater flow event. In K. R. Dyer & R. J. Orth (Eds.), *Changes in fluxes in estuaries: implications from science to management* (pp. 51–56). Fredensborg, Denmark: Olsen & Olse; International Symposium Series. https://doi.org/10.1016/0022-0981(95)90098-5

Slinger, J. H., Taljaard, S., & Largier, J. L. (2017, September 5). Modes of water renewal and flushing in a small intermittently closed estuary. *Estuarine, Coastal and Shelf*

Science. Academic Press. https://doi.org/10.1016/j.ecss.2017.07.002

Taljaard, S., & Slinger, J. H. (1993). Investigation into the flushing efficiency of 1. A freshwater release and 2. Seawater overwash in the Great Brak (CSIR Research Report 713). Stellenbosch, South Africa.

Taljaard, S., Van Niekerk, L., Huizinga, P., & Slinger, J. H. (2012). How scientists learnt about their role in governance: The case of Great Brak. In K. Belpaeme, O. McMeel, T. Vanagt, & J. Mees (Eds.), *Book of Abstracts. International Conference Littoral 2012: Coasts of Tomorrow* (VLIZ Speci, p. 111). Kursaal, Oostende, Belgium: Vlaams Instituut voor de Zee (VLIZ). https://doi.org/10.1007/s10334-015-0487-2

Turpie, J. K., Adams, J. B., Colloty, B. M., Joubert, A. J., Harrison, T. D., Maree, R. C., ... Mackay, H. (2002). Classification and prioritization of South African estuaries on the basis of health and conservation priority status for determination of the estuarine water reserve. Pretoria, South Africa.

Van Niekerk, L., Taljaard, S., Adams, J. B., Fundisi, D., Huizinga, P., Lamberth, S., Wooldridge, T. (2015). Desktop Provisional Ecoclassification of the Temperate Estuaries of South Africa: Report to the Water Research Commission (WRC Report No. 2187/1/15). Stellenbosch, South Africa: CSIR, Natural Resources and the Environment. Retrieved from http://www.wrc.org.za/

Wooldridge, T. H. (1994). The effect of periodic inlet closure on recruitment in the estuarine mudprawn, Upogebia africana (Ortmann). In K. R. Dyer & R. J. Orth (Eds.), *Changes in Fluxes in Estuaries: Implications from Science to Management* (pp. 329–333). Fredensborg, Denmark: Olsen & Olsen.

Wortmann, J., Hearne, J. W., & Adams, J. B. (1997). A mathematical model of an estuarine seagrass. *Ecological Modelling*, 98(2–3), 137–149. https://doi.org/10.1016/S0304-3800(96)01910-2

Wortmann, J., Hearne, J. W., & Adams, J. B. (1998). Evaluating the effects of freshwater inflow on the distribution of estuarine macrophytes. *Ecological Modelling*, 106(2–3), 213–232. https://doi.org/10.1016/S0304-3800(97)00197-X

7

The Practice of Managing the Bigi Pan Multiple-Use Management Area in Suriname

By Priscilla Miranda and Jill Slinger

7.1. Motivation for interest and approach

Over time, protected area management has moved from excluding people from ecologically valuable and vulnerable areas to multiple use management areas where humans are viewed as an integral part of the ecosystem (United Nations Environment Programme [UNEP], 2009). However, such ecosystem-based management is not without problems (Leech et al., 2009). Issues of conflicting resource uses, confusion in roles and responsibilities, issues of scale and local versus regional or national interest are commonplace (Walters & Ahrens, 2009). In recent years, sustainability has become an explicitly stated, even legislatively mandated goal of the institutions charged with ecosystem management (Christensen et al., 1996; The International Union for Conservation of Nature [IUCN], 2017). In practice, however, finding the balance between maximizing short-term gains on the one hand and long-term sustainability on the other seems harder.

Christensen et al. (1996) identify several factors that contribute to this gap between goals and practices, including:

- gross under sampling. Few monitoring activities are carried out in ecosystem environments, resulting in poor baseline data and information on biological diversity,
- widespread ignorance of the functioning and dynamics of ecosystems. Attention mostly goes to the output of the ecosystem,
- the openness and interconnectedness of ecosystems on spatial and temporal scales that exceed greatly the bounds of any management authority, and
- managing exploitation of presumably renewable resource ecosystems, is far more difficult than was assumed at the time that resource management institutes were established.

These factors mean that management strategies are derived from a simplified understanding of the structure and composition of the ecosystem, and tend to focus on immediate benefits instead of the sustained ecosystem functioning with the continuing and alarming loss of biodiversity as a result (Christensen et al., 1996; IUCN, 2017).

This is exemplified by the loss of thirty-five percent of the mangrove forests on a global scale (Valiela et al., 2001). In particular, the Guianan mangroves, stretching from the mouth of the Amazon River to the mouth of the Orinoco River are under threat. The 386 km coastline of Suriname, where eighty-five percent of the population resides, forms a component of this complex and extensive ecosystem (McGinley, 2014). In several areas along the coast of Suriname, e.g., the mouth of the Nickerie River and in Coronie, the estuarine ecosystems and their associated mangrove forests have been damaged by drainage and agricultural cultivation (Teunissen, 2008). In an attempt to redress the loss of biodiversity and accommodate the integral role of local communities in ecosystem management, the Surinamese government declared several areas along the entire coast Multiple-Use Management Areas (MUMAs) in 1987 (Suriname Coastal Protected Area Management Project [SCPAMP], 2013). The specific management goals for these MUMA's were formulated as:

- protection of the coast against erosion,
- protection of the nursery function of the estuarine area for the benefit of the coastal and sea fishery, and
- conservation of biodiversity.

In this chapter we explore the practice of managing a particular MUMA, the Bigi Pan MUMA in Suriname. We analyse the original management plan and the extent to which it has been implemented in practice over the two decades from 1990 to 2010 (Miranda, 2010). We follow Scharpf (1997) in distinguishing the system to be managed (the Bigi Pan MUMA and its local social and ecological system) from the managing system (the regional and national authorities concerned with the overarching management of the Bigi Pan MUMA). Drawing on the perspectives of the people involved in management up till 2010 or living in the area, we identify problems and clarify their underlying causes. We contribute to the body of knowledge on ecosystem-based management and indicate how this study informed the new Bigi Pan Management Plan 2013-2023 (SCPAMP, 2013).

Information on the envisaged ecosystem-based management of the Big Pan was collated from McCormick (1990). Primary data on existing and past management practices in the Bigi Pan MUMA were collected through face-to-face interviews. A total of 25 in-depth interviews were conducted in the period November 2009 to February 2010. The interviewees were selected from the five groups of actors (cf. Enserink et al., 2010) listed below, covering the time span from plan formulation to practice up till 2010 and the full range of involvement from inhabitants of the area to strategic and operational managers:

- actors involved in the design of the Bigi Pan MUMA management plan (1990 and 1996),
- actors involved in the actual management on the national level (past and present),
- actors involved in the actual management on the local level (past and present),
- actors using the area (past and present), and
- other interested actors (e.g., researchers, media).

The semi-structured interviews (Verschuren & Doorewaard, 2005) were conducted in the working or home environment of the interviewees. The interviews lasted approximately 1.5 hours and covered the topics of the formulation and implementation of the MUMA plan and the use and biophysical condition of the area. Insights were accumulated regarding the practice of managing the Bigi Pan MUMA with each subsequent interview. Refinement of the insights occurred through cross-checking with subsequent interviewees or existing literature, or through scheduling of extra interviews. The interviews were recorded and the digital voice files were transcribed into text files. These were returned to the interviewees for checking. Thematic analysis (Boyatzis, 1998; Silverman, 2010) was subsequently performed on the text and a composite picture of management practice and the existing problems in the Bigi Pan area emerged. These are represented by 124 issues, categorised into 22 sub-themes and four main themes.

The practice of managing the Bigi Pan MUMA from 1990 to 2010 is analysed in the light of the four main themes that emerged from the thematic analysis, namely: the biophysical condition of the ecosystem (Section 7.3.1), the original MUMA plan (Section 7.4.1), existing management of the area (Section 7.4.2) and existing use of the area (Section 7.5.1).

7.2. Study area

The Bigi Pan is located in north-western Suriname, in the districts of Nickerie and Coronie (Figure 7.1). The Bigi Pan is an estuarine ecosystem influenced by tides and experiencing variations in salinity. The Bigi Pan MUMA (Figure 7.1) has a terrestrial area of 68 300 hectares and extends to the 6 m-depth contour along the coast. The population living in the Nickerie district is approximately 37 000 people, whereas 3 000 people live in the Coronie district (Opdam et al., 2006). Several communities in the surrounding the area depend directly on the Bigi Pan MUMA for their livelihood (Parahoe & Wortel, 2009). Human activities in, and uses of, the area include rice and cattle farming, commercial fishery, apiculture, sand mining, hunting, tourism and research. In addition, the East-West access road was constructed in the area in 1964. Commercial fishing has been practiced in the Bigi Pan for over 40 years, employing as many as 185 full-time and part-time fishermen in 1990. Today, commercial fishing and tourism are the dominant activities.

Despite the existence of the Bigi Pan MUMA, problems with the management of the area have been reported in the media. These problems, including poaching, inaccessibility of the area and conflicts over dike breaching by fishermen, acted to trigger our interest in the practice of ecosystem-based management in the Bigi Pan, Suriname.

The mangrove forests that characterise the Bigi Pan estuarine ecosystem are important for shoreline stabilisation. They accelerate the accretion of the coastline when large mudflats are passing westward along the Surinamese coast (at an average speed of 1.5 $km.yr^{-1}$). The accreted sediments also retard shoreline erosion when the mudflats have moved further along the coast leaving the mangrove areas exposed once again (Winterwerp et al., 2005). Parwa (Black Mangrove; *Avincennia germinans*) is the dominant pioneer vegetation. White Mangrove (Akira; *Laguncularia racemosa*) and Red Mangrove (Mangro; *Rhizophora mangle*) are also recorded in the area, mainly along the tidal creeks and the Nickerie River (Parahoe & Wortel, 2009; SCPAMP, 2013).

The productivity of the mangrove forests is related to the periodicity and frequency of tidal flushing and to the quality of the inundating waters. Mangrove forests are highly dependent on fresh water to maintain an optimum salinity balance and for the import of inorganic nutrients, which are present in terrestrial runoff (McCormick, 1990; Rajkaran & Adams, 2012). The mangrove forests produce organic matter, which forms the basis of a complex food web. The mangrove-covered estuarine ecosystem forms an excellent breeding area and nursery of juvenile fish and crustaceans before they migrate to the open sea waters in sub-adult or adult stage (SCPAMP, 2013). At least two species of shrimps are found in the area, one of which is the *Penaeus subtilis*. Twenty-nine fresh and brackish water species of fish are known to occur in the Bigi Pan (based on commercial fishing statistics) and an additional three coastal species occur in the nearshore zone. The important commercial fish include the Tilapia (*Oreochomus* spp.), Snook (*Centropomus* spp.), Trapon (*Trapon altanticus*) and the Brasilian Mullet (*Mugil brasiliensis*).

A total of 122 bird species have been recorded in the Bigi Pan. Fifty are migrant species from either North America or other parts of South America. The remaining 72 bird species are year-round residents. The area is considered to be of international importance to at least 16 species and is home to three range-restricted species, namely: Guyanan Piculet, Blood-coloured Woodpecker, and Rufous Crabhawk (SCPAMP, 2013). The colony of Scarlet Ibis is the second largest in the world and accounts for 25% of the world population. The Bigi Pan currently has the status of Hemispheric Reserve within the Western Hemisphere Shorebird Reserve Network.

In addition, 38 mammal species occur in the MUMA and the area is known for its large population of white tailed deer, brown brocket and jaguar (SCPAMP, 2013).

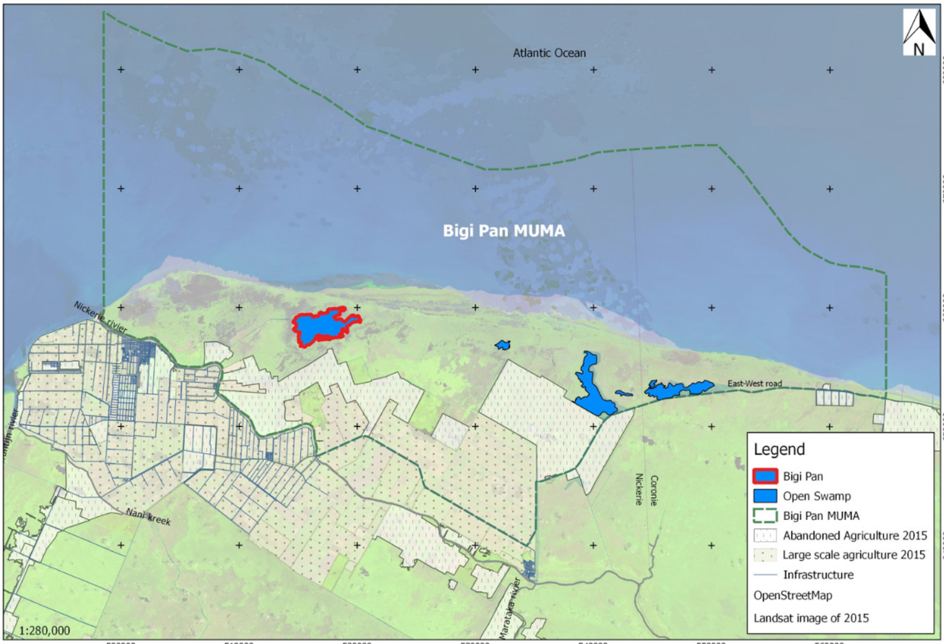


Figure 7.1. The Bigi Pan Multiple-Use Management Area

7.3. Natural dynamics of system

7.3.1. Biophysical condition

The freshwater-salt water balance in the Bigi Pan has altered over the period 1990 to 2010. Different reasons for this change were given by interviewees. First, the altered connection with the sea was mentioned as one of the major causes. Many creeks that connected the area with the sea are now silted up. A person familiar with the area over many years stated that about 40 years ago when the mud bank was positioned in front of Coronie, there were between 30 and 40 creeks open. Now, only a handful of creeks are still open. The altered position of the mud bank was identified as the underlying cause of this change. The result is that the inflow of saltwater has decreased and now it is only at spring tide that salt water enters the system. Second, the changed freshwater inflow was mentioned as a cause of alterations in the freshwater-salt water balance in the Bigi Pan. The construction of the East-West road in 1964 completely cut off the inflow of freshwater from the Coronie swamp. This resulted in high salinity levels within the estuary and associated changes in vegetation and the occurrence of freshwater fish in a particular area. One person gave an example of the change in vegetation, saying that the north side of the road used not to have Parwa-trees. This is confirmed by McCormick (1990). In 1993, culverts were introduced in an attempt to restore the freshwater inflow from the Coronie swamp (Naipal et al., 2008). This caused a shift in the freshwater-salt water balance, only this time towards freshwater dominance in some areas. The combination of creeks silting up, reducing the influence of the sea, and increased freshwater inflow resulted in an expansion of the freshwater areas. The colonisation of areas previously under salt-tolerant species by freshwater vegetation is indicative of this change, as is the clogging and obstruction of waterways by freshwater reeds (Naipal et al., 2008).

A further change in the biophysical conditions within the Bigi Pan lies with the fish population. Six interviewees from the fisheries sector indicated that fish numbers have changed over time. There are now less fish in the Bigi Pan area than in the past. These statements refer to the areas in which swamp fishing is undertaken. The fish population and the associated fish catches have declined. A fisherman gave an explicit example. In earlier years, he said that a fisherman could harvest 2 to 3 iceboxes of fish in about 2 hours with 2 net lengths. Now he says that he can barely harvest 1 icebox with 7 net lengths. Another indication of the declining fish population is found by comparing the effects of droughts, namely a drought that occurred 10 years ago (late 1990s) and the drought that occurred in 2009. Interviewees stated that during the drought of 10 years ago, massive fish deaths occurred, whereas only minor fish deaths occurred in 2009.

Besides the decline in the fish population, a shift in fish diversity has also been noticed. This shift applies to both swamp fishing areas and the coastal fishing areas. Interviewees stated that economically valuable fish are more difficult to catch these days. People had different, but non-conflicting opinions about the cause of the changes in the fish population. Both human and natural causes were identified. People identified the reduced connection with the sea as a natural cause of the decline in fish diversity. Overfishing was identified as the human cause of these changes. A variant of overfishing that several people identified as causing a change in fish diversity over the years is fishing down. Not all of them call it fishing down, but their description fits with the concept. Overfishing of the commercially valuable fish species which are bigger and slow-growing causes declining stocks, in response commercial fishermen choose to fish down the food chain and start to catch the fish and prawn species upon which the more commercially valuable fish prey (Pauly et al., 1998).

Related to the decline in the fish population (numbers and diversity) is the decline in the bird population. The first reason for this change identified by interviewees is the decline in their food source. Causes identified in addition to the decline in their food source are habitat disturbances. One of these disturbances is noise nuisance from the hunting sector. This is exacerbated by poaching. A further cause was stated to be the pollution by the pesticides used in the farming of rice in adjacent fields. This could not be confirmed by ancillary scientific sources.

7.4. History of (mouth) management policies and practices

7.4.1. The 1990 MUMA plan

According to interviewees, the 1990 MUMA plan is a paper plan because the strategies necessary to reach the management goals are not implemented for the most part. This is confirmed by Parahoe & Wortel (2009). The three main causes identified by the interviewees for the plan not being implemented are: (i) the plan had no 'leader', (ii) the implementation and management of the area was not a priority on the political agenda, and (iii) there was a poor transfer of the plan to the responsible actors.

7.4.2. Issues with the management of the area

Many of the authorities envisaged as being involved in the management of the Bigi Pan MUMA, as stated in the management plan of 1990, did not take up their responsibilities (Miranda, 2010). The tasks linked to these responsibilities are also not divided as the plan envisaged. For instance, the responsibility for the maintenance of the waterways was allocated to the Ministry of Public Works, but they have no permanent presence in the area. Instead, before the area was declared a MUMA and up until the early nineties, the Fisheries Service was active in the area and undertook this task. When a district commissioner with affinity for the area held office in the period of 1997, he took this task under his wing. Now that the Nature Conservation Division is active in the area, they have taken on this task in addition to their allocated tasks.

Further, the responsibility for water management has not been taken up explicitly. The quantity and quality of the water is fundamental to the continued health of the estuarine ecosystem, yet the responsibility for such an important aspect is not adequately addressed. In the original MUMA plan, a local authority responsible for the water management of the rice sector, the Multi-purpose Corantijn Project (MCP), was assigned this role. However, they have little knowledge or interest in nature conservation or estuarine systems and a long-term interest in using water for rice cultivation. They do have some practical experience with the management of the water of the Bigi Pan. However, by late 2009 and early 2010 there was no local authority with the willingness and capability to undertake the water management of the Bigi Pan.

Collaboration problems were named as one of the main issues concerning the implementation of management. These problems occur particularly on a regional level between the actors involved in management, but also between the national and regional actors. No structural cooperation exists on either level. A number of explanations for the lack of collaboration exist. First, there have been shifts in the active participation and presence in the area of the actor groups over time. These shifts have not always been agreed and have led to clashes. Second, differences in opinions about how the area should be managed have also led to clashes. For example, the Fishery Service is in favour of measures that promote the fishery sector, while Nature Conservation takes the precautionary approach and is against any interference in the natural processes. Third, conflicts of interest are present. Nature Conservation is supposed to facilitate all use functions in Bigi Pan. However, they are accused by interviewees of favouring the tourism sector above the other sectors (e.g., fishery), who state that staff of Nature Conservation are involved in, and profit from, tourism themselves. A final reason, which applies only within the region, is the strong identification with their role in managing Bigi Pan that local managers exhibit. This personal involvement and the associated emotions have led to clashes between people who in their private life are citizens of the same town, for instance.

Concerning the operational phase of the MUMA management, few surveillance activities are undertaken in the area. Reasons for the limited surveillance include the shortage of staff and equipment in relation to the size of the area - in short, there are capacity problems. Several people blamed the shortage of staff on the unattractiveness of the job. Game wardens have to be away from home for relatively long periods and stay overnight in rough terrain with all the dangers this entails. One person gave an example of the 'chicken and egg' situation that leads to a shortage of equipment. When staff members are creative in solving the equipment shortage and take actions like borrowing boats from private owners, the response of the government is hands-off. Supplying a boat for surveillance activities then receives a lower place on the priority list. This hampers operational surveillance and enforcement of compliance with the MUMA regulations. Fifty-six percent of interviewees stressed the importance of enforcement of regulations, stating that sanctions should be severe enough to have a discouraging effect.

Besides surveillance activities, Nature Conservation also focuses on education and building awareness. They aim to educate people to make sustainable choices. For instance, to stop hunting the Scarlet Ibis, because they understand how rare the bird species has become worldwide.

In late 2009, early 2010 people were of the opinion that management measures did not follow logically from the 1990 management plan. This was confirmed by the authority charged with general management of the area. Apart from surveillance activities, very few management activities are executed. Instead, political willingness and priorities are catalysts for the execution of measures. As one of the interviewees said, "Other countries cope with natural disasters, we have politics!" Over half of the interviewees mentioned the effects of political influences. They also said that with political support (political will) a lot more can be realised.

7.5. Social context

7.5.1. Use of the area

Due to the declining fish catch and the declining value of the catch, the swamp fishing activities have also reduced and become economically less feasible. Fishermen have moved to other sectors to earn their living (tourism, construction or banana farming). Fishing families however, always return to the fishing when possible. There is export potential for the Bigi Pan MUMA, but they cannot live up to export standards in quality and quantity. Export standards are a driving force to improve their fishing methods and make them more sustainable. However, this positive influence is disrupted by the existing illegal export to Guyana. To maintain the fishing activities, people indicated that the availability and diversity of fish, accessibility of the area and facilities like fishing camps need improvement.

The tourism sector is an upcoming sector. Activities are seasonal, mainly held in the wet seasons because the area is less accessible in dry seasons. This sector is not yet regulated or organised in the Bigi Pan MUMA. Anyone with a boat can access the area without registering. To maintain tourism activities, the beauty of nature, accessibility of the area, and facilities like tourist camps were identified as necessary.

Local communities do not depend on hunting for their livelihood, except when the rice and banana sectors are struggling economically. Then excessive poaching occurs. Otherwise recreational hunting occurs, mostly for people from Paramaribo. Hunting is heavily regulated. Hunters object to the restrictions, because they go against customary practices and increase the costs. This statement was based on an example that a hunter gave. When hunting for birds, it is almost impossible to find an isolated bird because birds live in flocks. So aiming at one bird with a shotgun is almost impossible, because the bullet scatters and will definitely kill more than one. The bag limit that applies will be exceeded. Hunting activities can have a noise nuisance for tourism. Tourism however, can have an awareness effect on people involved in the tourism sector who (used to) go hunting for recreation. By seeing the negative reaction of tourists towards hunting and their love of nature, the hunters become aware of the intrinsic value of the natural ecosystem.

Although a large part of Bigi Pan is dedicated to cattle and rice farming, these activities have declined enormously. This has to do with declining rice prices. It was indicated that when rice prices are up again, rice farming will be expanded. Buffer areas, which have not yet been cultivated, could be cultivated. These areas are located near to the nursery area of the Bigi Pan MUMA and when cultivated, the nursery function of the area could be severely affected with implications for the fishery sector. Also, large parts of state-owned rice farm (SML) were sold to Staatsolie (the national oil and gas company of Suriname), who will use the area for the production of bio-fuel (ethanol). The effects of this development on the Bigi Pan in the future require consideration.

7.6. System understanding and insights gained

Characteristics of the practice of managing the Bigi Pan MUMA in the period 1990 to 2010, include:

- The greater part of the Bigi Pan MUMA plan of 1990 was not implemented, because it had low priority on the political agenda, there was no leader for the implementation of the plan and there was an incomplete transfer of the plan to the actors involved,
- The existing management problems arose because the management authorities did not take up the responsibilities as envisaged by the plan, there are collaboration problems between managing authorities, surveillance activities are limited and management measures do not follow logically from the plan but from political willingness and priorities,
- Users of the Bigi Pan MUMA are not satisfied, they expect more from the area,
- There are detrimental changes in the biophysical conditions, namely: the altered freshwater-salt water balance in the Bigi Pan, a decline in fish numbers and diversity, and a decline in the bird population.

The identified issues are in partial agreement with the observations of Christensen et al. (1996). In particular, we confirm that few monitoring and surveillance activities are undertaken, that there is a fundamental lack of understanding of the ecosystem functioning and dynamics of the Bigi Pan, and that the management of the ecosystem is far more difficult than was imagined at the time that the MUMA institution was formed. We were not able to clearly distinguish the spatial and temporal scale effects mentioned by Christensen et al. (1996).

So the management plan formed an initial step in determining how to manage the area.

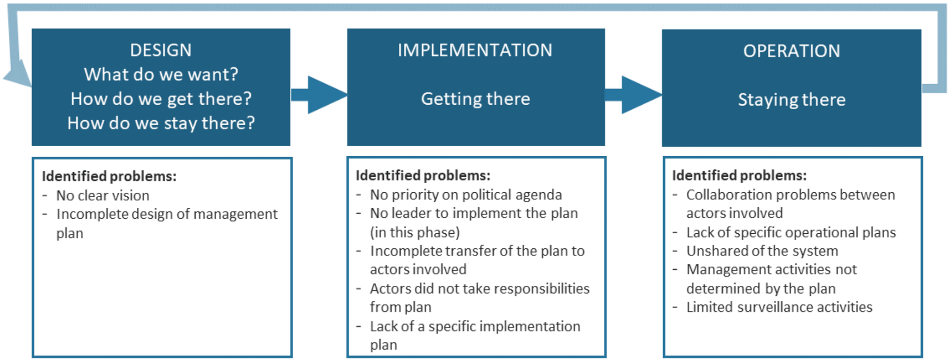


Figure 7.2. Analysis of the issues associated with the design, implementation and operational phases of managing the Bigi Pan MUMA from 1990 to 2010 (adapted from Miranda, 2010)

To assess the coherence between the plan and management activities up till 2010, we distinguish the planning phase, the implementation phase and the operational phase of the management of the Bigi Pan MUMA (Figure 7.2). As indicated at the top of the diagram, we view the design phase as answering the questions 'What do we want?', 'How do we get there?' and 'How do we stay there?' The implementation phase is all about 'Getting there' and the operational phase is about 'Staying there'. The blue arrows connecting these phases at the top of the diagram represent the ongoing learning cycle associated with Integrated Coastal Management (Olsen et al., 1997, 1999; Taljaard et al., 2013). The problems identified via the textual analysis with each of the phases are summarised in the lower boxes of Figure 7.2.

In Figure 7.3 we specify the components required to move from the design of a management plan through the implementation phase to the operational phase of successfully managing the Bigi Pan MUMA. First, we identify the primary goals of each of the phases and the requirements for achieving these phases, including the necessary information and resource inputs. The process-based activities undertaken in each of the phases are identified. Most importantly, the outputs from one phase that are necessary for the subsequent phase are indicated by the connecting arrows in the lower part of Figure 7.3. The issue identification per phase in Figure 7.2 together with the structured analysis in Figure 7.3, reaffirm that to increase the interconnectedness between the design of the MUMA plan, its implementation and operation, the following aspects need to improve in future:

- the understanding of the system,
- the social support.

7.7. Concluding remarks

This study has established that the promulgation of the Bigi Pan Multiple-Use Management Area (MUMA) and the management plan formed an initial step in determining how to manage the area. However, the next steps of ensuring that the plan was accepted and

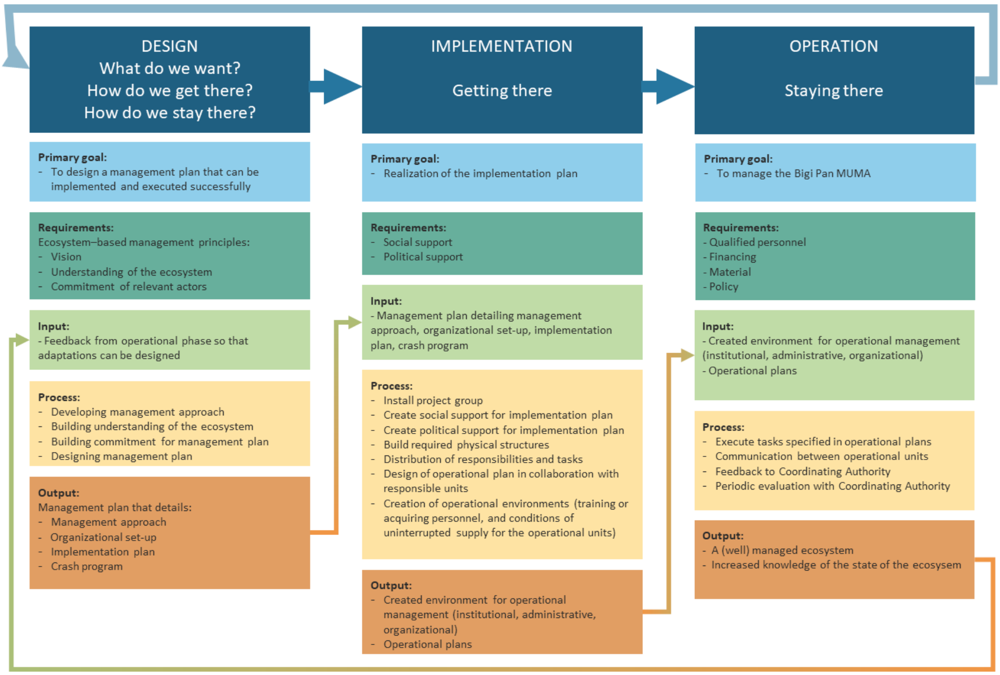


Figure 7.3. Required components for the design, implementation and operational phases of managing the Bigi Pan MUMA (adapted from Miranda, 2010)

followed through by the actors concerned, or adapted to make it workable were not taken consistently. Attention for the subsequent implementation phase, and the operational phase, of the Bigi Pan MUMA was lacking. Failure to recognise and consistently undertake these steps from planning to practical management in the period from 1990 to 2010 has meant that the Bigi Pan MUMA was managed in an ad hoc fashion for about twenty years.

The management plan for the Bigi Pan MUMA was adapted in 2013. The engagement with diverse stakeholders and the analysis undertaken in our study (Miranda, 2010, this chapter) spawned a reassessment of the efficacy of the management of the Bigi Pan, as exemplified by the following quote: “less than 10% of the original plan of 1990 has been implemented” (SCPAMP, 2013). Finally, a recommendation from this study on a composite intervention likely to exert a positive influence because it combines interventions focussed on improving the understanding of the biophysical system, and increasing the social support, is the establishment of a local committee. This committee can be constituted so as to improve structural cooperation between actors, and can be tasked with developing and implementing a joint vision for management of the area that is based on an understanding of the inherently dynamic biophysical conditions (Miranda 2010). This recommendation was incorporated into the new Bigi Pan Management Plan in the form of a “local-based foundation, governed by representatives from government authorities and agencies, as well as local user groups” (SCPAMP, 2013) tasked with implementation. It remains a challenge to ensure that the collaborative efforts required for locally supported management of the Bigi Pan Multiple Use Area are sustained and indeed lead to both improved understanding of the biophysical system and increased social support.

7.8. Acknowledgements

Dr. Els van Daalen and Dr. Marcel Marchand are gratefully acknowledged for their inputs regarding the research approach in 2009 and 2010.

7.9. References

Boyatzis, R. E. (1998). Transforming Qualitative Information Thematic Analysis and Code Development. Sage Publications.

Christensen, N. L., Bartuska, A. M., Brown, J. H., Carpenter, S., D'Antonio, C., Francis, R., ... Woodmansee, R. G. (1996). The report of the ecological society of america committee on the scientific basis for ecosystem management. *Ecological Applications. Ecological Society of America*, 6(3), 665–691. https://doi.org/10.2307/2269460

Enserink, B., Hermans, L., Kwakkel, J., Thissen, W., Koppenjan, J., & Bots, P. (2010). *Policy Analysis of Multi-Actor Systems*. Lemma.

Leech, S., Wiensczyk, A., & Turner, J. (2009). Ecosystem management: a practitioners' guide. *BC Journal of Ecosystems and Management*, 10(2), 1–12. Retrieved from http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.579.3470&rep=rep1&type=pdf

McCormick, K. J. (1990). *Bigi Pan Multiple Use Management Area Management Plan, Environment Canada and Ministry of Natural Resources*. Paramaribo, Suriname.

McGinley, M. (2014, June 30). Guianan mangroves. *The Encyclopedia of Earth*. Retrieved from https://editors.eol.org/eoearth/wiki/Guianan_mangroves

Miranda, P. A. (2010). *Balancing Human Needs and Nature Conservation: A study on the gap between design and management of the Bigi Pan Multiple-Use Management Area in Suriname,* SA (Masters thesis). Delft University of Technology. Retrieved from https://repository.tudelft.nl/islandora/object/uuid%3Aa43d4f50-37b8-4e3d-bf51-da5f190dd430

Naipal, S., Wesenhagen, G., Wortel, V., Parahoe, M., Soetosenojo, A., & Jadhav, Y. (2008). Final Report on Biodiversity and Economic Valuation of Bigi Pan Multiple Use Management Area Part 2 - Hydrology and Water Quality.

Olsen, S. B., Lowry, K., & Tobey, J. (1999). A Manual for Assessing Progress in Coastal Management (Coastal Management Report 2211). Rhode Island, USA: Coastal Resources Centre, University of Rhode Island.

Olsen, S. B., Tobey, J., & Kerr, M. (1997). A common framework for learning from ICM experience. *Ocean and Coastal Management,* 37(2), 155–174. https://doi.org/10.1016/S0964-5691(97)90105-8

Opdam, H. J., Monbaliu, J., Maren, M. van, Naipal, S., Prado, N. del, Mohadin, K., … Toorman, E. (2006). Towards Integrated Coastal Zone Management (ICZM) in Suriname. Final Report to the Inter-American Development Bank, Suriname Branch. Paramaribo, Suriname.

Parahoe, M., & Wortel, V. (2009). The Bigi Pan Multiple Use Management Area. Paramaribo, Suriname: World Wide Fund (WWF). The Center for Agricultural

Research in Suriname (CELOS). ESEM Production.

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., & Torres, F. (1998). Fishing down marine food webs. Science, 279(5352), 860–863. https://doi.org/10.1126/science.279.5352.860

Rajkaran, A., & Adams, J. (2012). The effects of environmental variables on mortality and growth of mangroves at Mngazana Estuary, Eastern Cape, South Africa. Wetlands *Ecology and Management*, 20(4), 297–312. https://doi.org/10.1007/s11273-012-9254-6

Scharpf, F. W. (1997). Games real actors play: Actor-centered institutionalism in policy research. Westview Press.

Silverman, D. (2010). Doing qualitative research: A practical handbook (Third edit). London: SAGE Publications, Inc.

Suriname Coastal Protected Area Management Project. (2013). Bigi Pan Management Plan 2013-2023. Suriname Coastal Protected Area Management Project.

Taljaard, S., Slinger, J. H., & van der Merwe, J. (2013). Dual adaptive cycles in implementing integrated coastal management. *Ocean and Coastal Management,* 84, 23–30. https://doi.org/10.1016/j.ocecoaman.2013.07.003

Teunissen, P. A. (2008). Nota van toelichting op instelling noordelijke verkavelingsgrens voor Groot-Paramaribo. Op verzoek van de Ministerie van Ruimtelijke Ordening, Grond- en Bosbeheer [in Dutch].

The International Union for Conservation of Nature. (2017). IUCN Global Protected Areas Programme. Retrieved November 21, 2019, from https://www.iucn.org/theme/protected-areas/about

United Nations Environment Programme. (2009). Ecosystem Management Programme - A new approach to sustainability. Nairobi.

Valiela, I., Bowen, J. L., & York, J. K. (2001). Mangrove Forests: One of the World's Threatened Major Tropical Environments. BioScience, 51(10), 807. https://doi.org/10.1641/0006-3568(2001)051[0807:mfootw]2.0.co;2

Verschuren, P., & Doorewaard, H. (2005). Designing a Research Project. Utrecht: Lemma.

Walters, C., & Ahrens, R. (2009). Oceans and estuaries: Managing the commons. In Principles of Ecosystem Stewardship: Resilience-Based Natural Resource Management in a Changing World (pp. 221–240). Springer New York. https://doi.org/10.1007/978-0-387-73033-2_10

Winterwerp, J. C., Borst, W. G., & de Vries, M. B. (2005). Pilot Study on the Erosion and Rehabilitation of a Mangrove Mud Coast. *Journal of Coastal Research,* 21(2), 223–230. https://doi.org/10.2112/03-832a.1

8

On the Role of System Understanding in the Slufter, Texel, the Netherlands

By Floortje d'Hont and Jill Slinger

8.1. Motivation for research approach

8.1.1. Location, type of coast

The Slufter is an estuary located within a nature reserve in the North Sea dunes of the island of Texel, the most westerly Wadden Sea island of the Netherlands (Figure 8.1). The Slufter comprises coastal dunes, an estuarine channel, a salt-marsh and an intertidal zone landwards of the coastal dunes. The entire Slufter area is about 1 km wide (from mouth to sand dike) and over 2 km long. The Slufter is a small system, with an intermittently closed mouth and seasonal freshwater inflow of unknown total volume. The dynamic intertidal zone is bounded by a sand dike and sandy dunes. Diversity in the substrates and a lack of disturbance mean the Slufter exhibits high species richness in its vegetation (Pedroli & Hoekstra, 1992). The Slufter area, including the sand dike, forms a component of the primary flood defence of Texel, and protects the hinterland from flooding from the North Sea.

Figure 8.1. Figure 8.1: The Slufter is situated along the North Sea coast of Texel, the Netherlands (Picture: Flying Focus)

8.1.2. Purpose of mouth management: Flood defence

According to Dutch law (Water Act, 2009), the water board Hollands Noorderkwartier (HHNK) is formally responsible for managing parts of the Dutch coast, including the Texel coast and ensuring that it adheres to the legally prescribed safety standards for flood defence (see Chapter 2). For the purpose of flood defence, HHNK's existing management policy for the Slufter is to excavate a straight mouth channel to the west of the existing mouth every four to six years. The aim is to maintain the integrity of the dune front to the northeast, and to reduce the penetration and potential erosive action of storm waves at the sand dike.

However, simulations from new storm wave models (van Rooijen & van Thiel de Vries, 2013) indicate that it may not be necessary to intervene in this way as the mouth dynamics have only a limited effect on flooding safety. Because of these indications, and because flooding safety is not the only issue at stake, HHNK is considering intervening less with the mouth of the estuary as part of their coastal policy and letting nature take its course in the Slufter in the future.

8.1.3. Additional societal and ecological value

Indeed, the Slufter is a tourist attraction, drawing nature lovers, particularly bird watchers, as well as hikers and cyclists to the island of Texel, and generating economic value for medium and small business enterprises. There are more slufters and slufter-like nature areas in the Netherlands, but the Slufter is the largest natural salt-marsh and the most stable one in the Netherlands (Pedroli & Hoekstra, 1992). As a protected nature area, the Slufter forms part of several ecological networks established and safeguarded by national and European legislation.

Other stakeholders include governmental authorities, environmental organisations, nature managers and the citizens of the island. However, the value of such an estuary is perceived differently by the different actors (Costanza et al., 1997; Farber et al., 2002), each of whom may have an interest in, some responsibility for, or be affected by decisions regarding the Slufter. The multi-actor environment and the formal and informal responsibilities of HHNK result in a playing field in which HHNK wants to enhance (collaborative) long-term decision making about the Slufter. For HHNK this means maintaining safety standards efficiently and effectively, while minimizing the negative effects on the ecosystem, and maintaining good relations with the stakeholders.

8.2. Research motivation and approach

As researchers at Delft University of Technology, this case study provided a unique opportunity to explore the role of system understanding in support of integrated management of a small estuary. The Slufter and similar small estuary systems are under-researched in the Netherlands. By gathering a wide range of knowledge from different sources and sharing new knowledge in a collaborative workshop setting, we aimed to deepen understanding of both the social and the ecological aspects of the small estuary system, with the end objective of including values besides flood defence in the policy making process. We investigated social-ecological knowledge via the design and application of an action research study that aimed to improve system understanding and influence policy in the long term. This study is extensively reported in D'Hont (2014) and D'Hont et al. (2014). Although there was no need for policy change purely from a flood defence perspective, there was an opportunity for more natural dynamics in the area and a resulting regime that is more in line with societal and ecological values. As such, the case study of the Slufter represents a small exemplar of the friction between ecological and safety values in Dutch coastal management, as well as signalling a larger scale trend of increasing integration and stakeholder consultation in coastal management. The undertaken approach is characterised by a combination of a stakeholder analysis and problem modelling, and requires the adoption of a dynamic, multi-actor, and social-ecological systems conceptual lens. In particular, the utility of combining information gathered through desk research, a simulation model study and stakeholder interviews for enhancing system understanding is explored. We designed and applied a knowledge intervention in the form of a workshop, where stakeholders were able to use this synthesised understanding of the dynamic system, as well as information from the stakeholder analysis, as starting points for discussions.

We view the Slufter as a social-ecological system (SES), where system knowledge among stakeholders is important. The structure of this book chapter follows the steps undertaken in the case study research. Following some theoretical background, a systems analysis was performed, so as to be able to assess the functioning of the Slufter with respect to ecological and social aspects. Values, interests, functions, system understanding and individual perspectives were elicited through stakeholder analysis and stakeholder interviews. In the interviews stakeholders were encouraged to explain their view of the system, revealing their own scale perspectives and preferences, and supplying information-rich insights and answers (Vreugdenhil et al., 2010). Desk research revealed a high degree of nestedness of the Slufter as hydro-morphological system (cf. Slinger 2017) and as multi-level governance system (Section 8.4). Accordingly, part of the research involved using a system dynamics modelling study to illustrate how the abiotic dynamic processes that occur within archetypical estuaries such as the Slufter, influence the biotic environment. The hydro-morphological (abiotic) processes are the main driver for the dynamic behaviour of the Slufter, particularly the Slufter mouth, as sediment disposition and erosion shape the landscape, enhancing freshwater-seawater

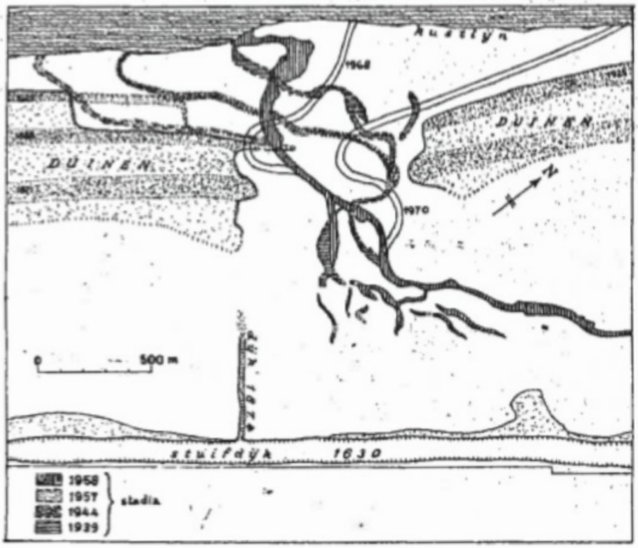


Figure 8.2. Dynamic behaviour of the Slufter channel and mouth from 1939 to 1958 (Rijkswaterstaat, n.d.)

gradients and contributing to the highly valued biodiversity (i.e., diversity in vegetation, invertebrates and birds). Then, we move on from describing the ecological system and its abiotic and biotic natural dynamics to consider the origination of the Slufter, and the role that humans interventions have played, and are playing, in the physical system and the social, governmental and institutional structures in Section 8.5.

Then the results from the first two analytical stages are synthesised and reported back to selected stakeholders, forming the knowledge intervention (Section 8.6). The knowledge intervention was designed with the aim of increasing the shared understanding and enhancing individual system understanding in a stakeholder setting. In D'Hont et al. (2014), we describe the design and application of the knowledge intervention within a potentially contentious situation, owing to the existing degree of discussion among stakeholders and the extent and variety of the values associated with the nature reserve, the Slufter. Clearly, the long-term influence of the knowledge intervention cannot be understood fully immediately after the workshop, nor can it be understood in isolation of other knowledge acquisition opportunities or events. Instead, this book chapter focuses on the shifting social values and perspectives regarding human interventions in the ecological and social aspects of the Slufter estuary.

8.3. Theoretical background

The choice for knowledge-sharing is grounded in scientific literature and policy practice. The field of integrated coastal management (ICM) has a substantive issue-based focus (e.g., coastal erosion issues, flood defence issues, and conservation issues). ICM is also characterised by extensive evaluation of issue-based pilot projects and their contribution to integrated management programmes (Olsen et al., 1997; Olsen, 2009), delivering insights at the national and regional level. Findings from these evaluations indicate that a participatory approach can have success in generating public acceptance for new national policies or programmes, e.g., the South African Coastal Policy (Department of Environmental Affairs, 2008) and the Dutch New Delta Programme (Delta Commission, 2016). We therefore research a knowledge-based, stakeholder-inclusive approach to coastal management aimed at deepening system understanding. For this, the case study of the Slufter is suitable, because the Netherlands has a strong tradition in collaborative governance. Activities to encourage inclusion and consultation of the public are increasingly favoured, although claims are made regarding success based primarily on the personal experience of the initiators or on the number of people informed (Newig & Fritsch, 2009; Reed, 2008). Accordingly, we aimed at an intervention in a collaborative and transdisciplinary setting, combining knowledge of local stakeholders and specialists of different backgrounds to think together on an area they all know, either personally or professionally, using a social-ecological perspective. We know that local actors perceive and value complex systems differently (Costanza et al., 1997; Farber et al., 2002; Mayer et al., 2004). Ostrom (Ostrom, 2009) famously argues that increased system understanding can lead to better long-term management supported by local stakeholders: "When users share common knowledge of relevant SES attributes, how their actions affect each other, and rules used in other SESs, they will perceive lower costs of organizing" (Ostrom, 2009). Acordingly, we choose to focus on increasing the system understanding of local stakeholders using a three stage analysis process, namely (i) system analysis, (ii) system dynamics simulation, and (iii) a knowledge intervention with stakeholders in the form of a workshop. The results of the system analysis and the knowledge intervention are described in Sections 8.4 and 8.5. respectively. The results of the system analysis were communicated to the stakeholders through an oral presentation and discussion during the workshop.

8.4. Natural dynamics of inlet

The Slufter has a highly dynamic character with a narrow and sinuous channel meandering through a dune valley. Pioneer plant species grow on the bare areas, succeeded by other species over time. The Slufter mouth is particularly dynamic, as sediment disposition and erosion shape the inlet and associated intertidal landscape, enhancing freshwater-seawater gradients and contributing to the highly valued biodiversity (i.e., diversity in vegetation, invertebrates and birds). Figure 8.2 provides a representation of the changing location of the Slufter mouth and the meandering behaviour of the Slufter channel in the early to mid 20th century (Rijkswaterstaat, n.d.).

8.4.1. Abiotic characteristics and dynamics

The Slufter is located in a coastal area with semidiurnal and spring-neap tidal variations, which are associated with high-low variations in water level on a 12 hour 40 minute and 28 day time scale respectively. The sill height increases when sediment is deposited in the mouth channel and decreases when erosion occurs in the mouth channel. The sediment is transported by the water flowing through the mouth on the ebb and flood flows. During the ebb, sediment is eroded from the mouth channel and transported out to sea. This erosion causes the sill height to decrease. During flood flows, the action of waves in the breaker zone means that the capacity of the seawater to transport sediment is enhanced. As the water floods into the estuary, it is no longer able to transport all the sediment that it is carrying in suspension. This excess sediment is deposited and causes the sill height to increase and the mouth cross-section to decrease. It is this mechanism which can cause the mouth to close and the tidal influence on the estuary to be cut off. This usually happens under high wave conditions, but not necessarily storm conditions. Communities and authorities can intervene in such a situation by choosing to breach the mouth.

Storms typically occur in the winter season between October and March. For the Slufter, the highest storm wave intensity near the sand dike (within the Slufter basin) would be caused by surges during spring tide with a North Westerly wind direction. High waves during storms can deposit sandy sediments deep within the Slufter area. For instance, the 'Sinterklaasstorm' of 5 December 2013 happened during the course of the case study research, elevating water levels to 2.54 m above NAP in the Slufter, and causing concern that the sand dike would burst (From: stakeholder interviews in D'Hont, 2014). This did not occur. Indeed, as mentioned before, the district water board HHNK maintains safety levels through intervening in the Slufter to reduce the storm wave intensity near the sand dike by maintaining the integrity of the dune front.

8.4.2. Biotic characteristics and dynamics

In the Slufter, the combination of wind- and water-driven sediment transport and the transition from fresh to salt water results in a high diversity of vegetation types in a relatively small area (Balke, 2013). There is valuable vegetation in the salt-fresh water transition areas near and in the Slufter channel, as well as in the brackish habitats (Balke, 2013, p. 13), for instance the Dutch-termed 'Fraaiduizendguldenkruid', 'Hertshoornweegbree', 'Zilte rus', 'Rood zwenkgras' en 'Engels gras' (Pranger, 1999). The Slufter area attracts a wide diversity of bird species. This is mainly due to the peace and quiet in the area and the wide diversity of biotopes (Durieux, 2004). The area is a birds habitat, as it is used

Figure 8.3. Texel (here spelled Teßel) and Eierland (here spelled Eyerland) on a northern fragment of a map of The Netherlands, published circa 1743. Sand embankments between Eierland and Texel later connected the two islands to form Texel as we know it today (Image: Wikipedia - Eierland, 2018)

for incubation by birds, such as eider ducks, the common shelduck, and the pied avocet. Other organisms that live in The Slufter channel are crabs, shrimps and flounder. Sheep and cattle graze the Slufter area (Nationaal Park Duinen van Texel, 2010).

The ecological situation in the Slufter was measured and mapped by Pranger and Tolman (Pranger & Tolman, 2011; Balke, 2013). The largest part of the salt-marsh is taken up by vegetation types from the middle salt-marsh. The pioneer zone is characterised by an abundance of species. The humid dune slacks and typical brackish salt-marsh vegetation indicate the presence of freshwater near the dune front. The vegetation and bird species are indicative of the presence or absence of abiotic factors such as wind, freshwater and seawater. The Slufter serves a foraging function; animals, such as geese, ducks, waders, gulls and other birds search for food resources and exploit them. These food resources include vegetation, insects and benthic organisms (Durieux, 2004, p. 28). Additionally the area serves a refuge function; water birds and waders seek refuge on the high dunes when the water rises high (Durieux, 2004, p. 28).

8.5. The role of human interventions on the Slufter

Aside from the ecological value of the Slufter area, the nature reserve is also socially valuable. First and foremost, the sand dike of the Slufter functions as a primary flood defence barrier and is an important link in Texel's flood defence infrastructure. Historically, the Slufter is arguably a remnant of a 'failed' land reclamation centuries ago. The area has a cultural-historical value, and is an area of pride to the islanders. Poems have been written about the area which is unique along the Dutch coast. Its dynamic (a) biotic richness attracts educational school excursions to teach children about different species and ecosystem dynamics. Lastly, associated with its rich vegetation and highly dynamic character, the Slufter attracted tourism and recreationists.

8.5.1. Origin of the Slufter

Present-day Texel and the Slufter have been shaped by consecutive geomorphological processes from the penultimate glacial period and by human interactions from the 17th century. Settlement on Texel has adapted to the elevation differences. The villages and older buildings are located on the higher areas, whereas agricultural lands are mostly located in the polders (Municipality of Texel, 2006).

Pleistocene

The Wadden Islands are the result of the effects of the tides, waves, wind and a rising sea level in the time period after the last glacial period, approximately 11 000 years ago. An increasingly warmer climate caused the North Sea to fill up with melted land ice. The irregular pleistocene landscape was flooded with sea water and waves and streams formed a beach ridge ('strandwallenreeks') along the Dutch coast from present day Belgium to the Elbe river mouth in Germany. New primary dunes were shaped from new, fresh sand on the beach areas and the beach ridge evolved into a island ridge with local openings. The area between the islands and the Dutch coast became an intertidal zone, comparable to the present day Wadden Sea (Rijkswaterstaat, 2013b).

Texel is the only Dutch Wadden Island that is positioned from southwest to northeast. This is mainly caused by the origin and geological structure of the island: Texel is the only

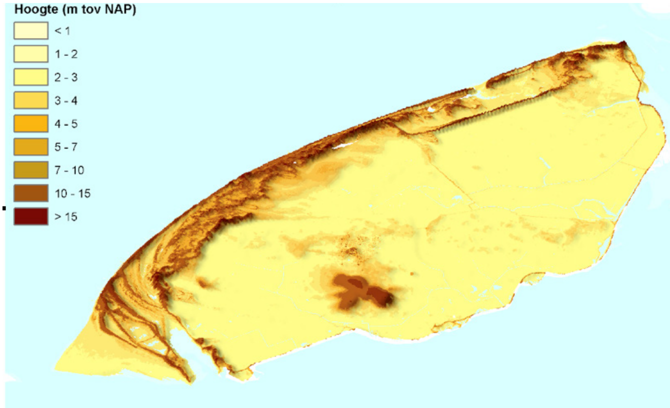


Figure 8.4. Elevation on Texel in metre above NAP (Hoogheemraadschap Hollands Noorderkwartier, 2013)

Wadden Island with boulder clay (keileem) close to the surface. Boulder clay bulges kept the West coast of Holland and Texel from moving towards the east, not following the movements of the other Wadden Islands. Another determining factor in the morphology of the present day Wadden Islands was the occurrence of storm surges in the 10th to 12th century A.C. that divided the beach ridge up into islands (Hoogheemraadschap Hollands Noorderkwartier, 2013).

Land reclamation attempts

Where present-day Texel is shaped through land reclamation and embankment constructions, the Slufter can be considered to be a remnant of a "failed" land reclamation. Centuries ago, the island consisted of two separate, smaller islands: Texel in the south and Eierland (or: Yerland) in the north, with washover systems in between the two. Around 1630, the Staten van Holland decided to connect Eierland to Texel by constructing a sand embankment (Figure 8.3. and Figure 8.4.), to prevent the development of a new inlet from the North Sea to the Zuiderzee (i.e., present-day Wadden Sea) (Rijkswaterstaat, n.d.). Through natural sediment accretion and human interventions (e.g., planting of marram grass), dunes on the North Sea side of the connecting sand dike were formed and, in line with the Dutch reclamation tradition of those times, attempts were made to empolder the area in 1855 (Water Act, 2009). However, the closing was unsuccessful and three tidal channels came into being: 'De Muy', the small Slufter and the large Slufter. The Dutch persisted in trying to close the North Sea dune ridge, aiming for a continuous sandy flood defence line. As such, the smaller 'De Muy' was closed relatively easily (Water Act, 2009). The large Slufter, located near the 'Krimduinen' on former Eierland, proved to be a bigger challenge, with failed enclosure attempts in 1886 and again 1888. Closing these tidal inlets affected the freshwater discharge in the others. After the large Slufter and the Muy were closed through dune and dam ridge construction, the small Slufter channel experienced rapid growth, because it had to process much more water than before. After the final attempt to close the small Slufter in 1925 failed, the State decided to leave the estuary open. The Slufter evolved into the nature area we know today – a highly dynamic nature reserve, by Dutch standards (Hoogheemraadschap Hollands Noorderkwartier, 2013).

The map in Figure 8.4 represents the range of elevation with respect to NAP; the vertical datum that approximately equals the Mean Sea Level along the Dutch coast (Ministry of Defence, 2013). The map shows the dunes and the sand dike forming the main coastal barrier on the island. The lower parts are most likely to flood in case of a dike break and high tide. The Slufter forms an inlet in the line of North Sea dunes. The sand dike behind the Slufter, which is the same sand dike that was built in the 17th century, protects the polders behind the Slufter. The elevation of the lands behind the coastal barrier vary from 2 m below NAP in the polders to 23 m above NAP at the top of the dunes (Municipality of Texel, 2006). The Slufter area comprises approximately 700 hectare of land. The Slufter area has silted up during the past 50 years, caused by natural processes and by the addition of sediment to the coastal system through interventions such as shore-face nourishments or beach nourishments (Rijkswaterstaat, 2013a). However, the exact effect of human interventions in the North Sea on the Slufter nature reserve is unknown.

8.5.2. Institutional context

The nature reserve the Slufter is part of the primary water barrier, i.e., the part of the coast that protects the island of Texel from high water levels in the North Sea. The Delta Act (1958), Flood Defence Act (1996), and Water Act (2009) establish the safety standards against flooding in the Netherlands. Mulder et al. (Mulder et al., 2011) describe the institutionalisation of coastal erosion management (see section 2.4). The water board HHNK is one of 25 water boards in the Netherlands. These regional governmental authorities are amongst the oldest forms of government, the earliest ones known existed in the 13th century, forming collectives of local people with an interest in water safety. Now the water boards lay down the conditions for achieving the strategic objectives of flood defence, they define concrete measures to achieve these objectives and they execute projects.

In addition to its function in flood defence, the Slufter's unique natural characteristic and location make it the object of environmental protection as well. The legislative context regarding the flood protection and environmental protection of the Slufter is complex, because it is not an isolated ecosystem, but embedded in a larger nature network. On the European level, the Slufter is part of the Natura 2000 area 'Duinen and Lage Land Texel', and is protected under both the Habitat Directive (92/43/EEC) and the Birds Directive (2009/147/EEC) (Rijkswaterstaat, 2013a). Natura 2000 forms the core of European Union nature and biodiversity policy. Further, the Ramsar Convention (an international convention on protection of wetlands) applies to the Slufter, and it is included in the broader objectives for the Wadden Sea and the North Sea as specified in the Fifth White Paper on Spatial Planning (Vijfde Nota Ruimtelijke Ordening [VIJNO], 2001). The Slufter forms part of a National Park that is in turn part of the Ecological Network (Ecologische Hoofdstructuur, EHS). Additionally, there are many institutionalised forms of cooperation between managers of nature reserves on the regional, national and international levels (e.g., Wetlands convention, EHS, National Park Dunes of Texel).

8.5.3. Current management practice

Within current management practices, the district water board HHNK is formally responsible for maintaining the sandy coast of Texel so that it adheres to the legally prescribed safety standards for flood defence. Most notably, HHNK intervenes in the physical system by straightening the channel near the opening every four to six years.

The interventions in the Slufter (periodical dredging of the mouth) to maintain the integrity of the flood defences have negative side-effects on the ecological system in the area. In line with the more dynamic approach to Dutch coastal management in the recent past, and new insights from flood risk models, the water board is considering a less drastic way of intervening in the system. This change is likely to have only limited effects on flooding safety and could potentially have substantial beneficial effects on the bio-geomorphological dynamics of the Slufter. The simulations from new storm wave models commissioned by HHNK (Rooijen & van Thiel de Vries, 2013) indicate that the sand dike is in principal strong enough to protect the hinterland from flooding even if the Slufter opening becomes larger. Their simulation study reveals that a wider Slufter channel at different locations will not create unsafe situations, and thus provides the district water board with a justification to review the current practice of channel straightening.

Although the current practice of HHNK includes stakeholder management in relation to flood management, HHNK is interested in discovering whether another type of stakeholder involvement could deliver deeper insights or enhanced engagement. Although environmental protection legislation is in place, water managers have the right and obligation to intervene if (water) safety is at stake, even when this affects a protected ecological system. However, HHNK aims to keep flood protection measures and water quality management aligned with nature preservation, and only wants to have to cross ecological boundaries when this is absolutely necessary. HHNK partners with other actors, such as governmental authorities, environmental organisations and nature managers on the island. These actors may hold different opinions. Besides water safety, broadly speaking, stakeholders of the Slufter are interested in other aspects, such as ecology, economy, tourism and recreation.

Owing to the natural dynamic nature of the Slufter, the vegetation growth in the Slufter can change significantly over the course of 6 years (Balke, 2013). Human interventions influence the natural dynamics too: the vegetation, and the proportions between bare sand, water, and pioneer vegetation differ from 2005 to 2011, because the Slufter channel was straightened in 2010.

8.6. The knowledge intervention – intervening in the social system

The synthesised understanding from the preceding system analysis was combined with simulation model outcomes and a stakeholder analysis and presented to a selection of participants in a workshop setting, forming the knowledge intervention. After some deliberation, we chose for a level of detail of supplied information that was thought to be appropriate for the participants with real-world understanding of the estuary, but limited specialised, disciplinary or abstracted conceptual knowledge. The participants group was a mixture of researchers familiar with modelling techniques and local actors from the island, all with individually different viewpoints and substantial, ready, real-world knowledge of the Slufter. An ex ante survey of what participants valued about the Slufter was conducted. Thereafter, we presented three archetypical estuary characteristics and estuary behaviour (Figure 8.5 and Figure 8.6), followed by discussion on estuary dynamics in relation to the Slufter. Participants were encouraged to consider the situation of normal weather conditions and ordinary tidal dynamics, as opposed to other meetings and workshops on the Slufter that commonly emphasised flood defence and consequently the situation of exceptional storm weather conditions. The aim in this regard was to increase dynamic system understanding of the participants by discussing known dynamic behaviour and system boundaries that related to the individual real-world experiences of the participants. As expected, the discussion quickly diverted from water safety, and participants were able to communicate regarding the potential consequences of dynamic estuary behaviour on vegetation and birds, based on the information supplied on the abiotic dynamics.

Next, information on stakeholder perceptions and values derived from the interviews were presented and discussed. Contrary to expectations that the discussion would focus on differences in the perceptions of stakeholders and what they could learn from each other, participants repeatedly came back to discussing the importance of wild nature versus human interference. They agreed that finding a balance between human interventions and wild nature remains difficult. Participants did communicate their individual values and exchanged some knowledge on the system, thereby creating some

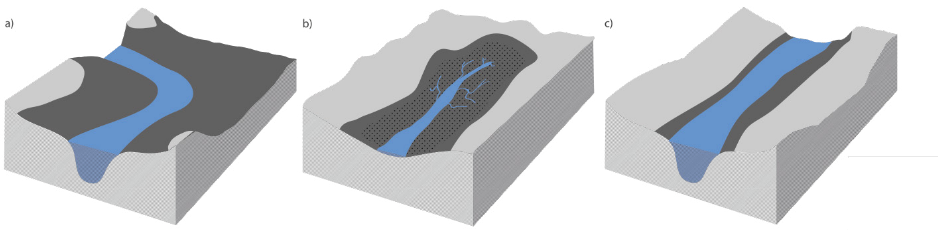


Figure 8.5. Archetypal estuary systems – a) small estuary with a shallower basin form and a higher sill height b) mudflat-like, perched, shallow formed basin c) long, deep estuary (from: D'Hont, 2014)

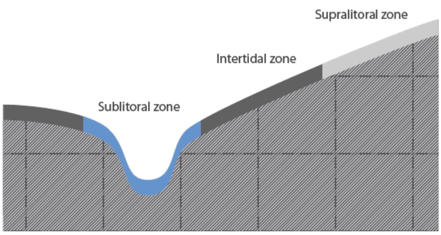


Figure 8.6. Archetypical estuary bathymetries were used in a stakeholder setting alongside summarised descriptions of estuary behaviour (from: D'Hont, 2014)

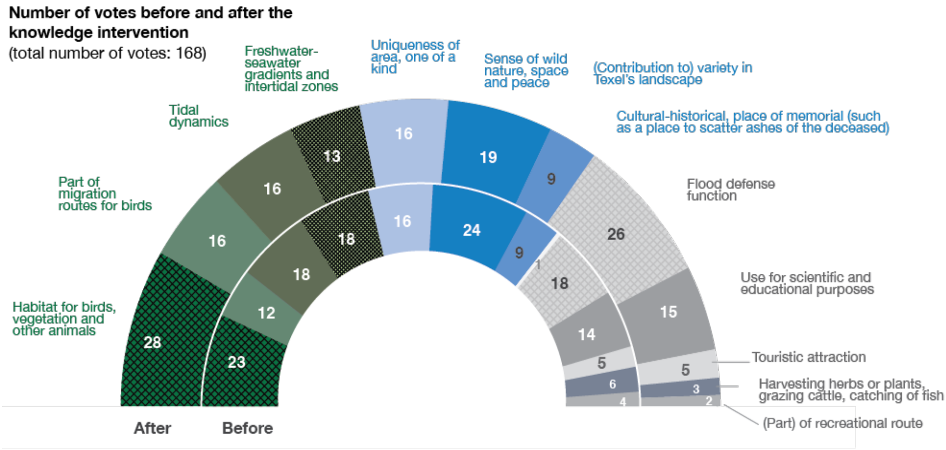


Figure 8.7. Participant voting before and after the knowledge intervention (Participants each had 12 votes to rank the qualities of the Slufter before and after the knowledge intervention) (from: D'Hont, 2014)

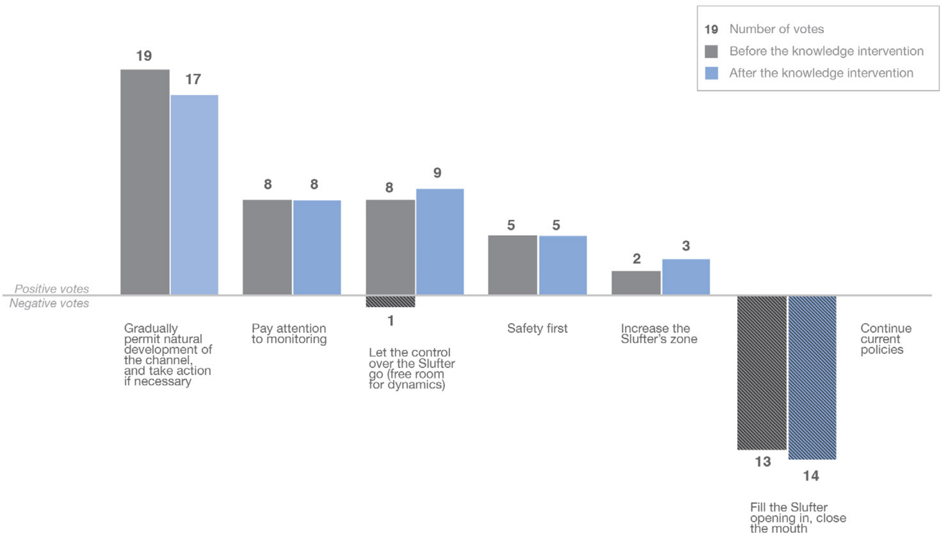


Figure 8.8. The ex-ante and post workshop voting distribution (Participants each had 3 positive votes and 1 negative vote to express how their preferences were to be translated to (hypothetical) policy options) (from: D'Hont, 2014)

common knowledge. For example the unknown volume and seasonal variability of the freshwater inflow to the Slufter estuary was discussed and whether the freshwater inflow should be considered significant was debated. An additional discussion was started regarding the values of a participant who emphasised the function of the Slufter as a bird habitat and a link in global migration routes. After a coffee break and a stroll outside during which the discussion and sharing continued, the participants voted again by sticking dots on the posters hanging in the room, which provided the same options as the ex-ante measure. As depicted in Figure 8.7, the greatest change in the participants' perceptions lay in the increased recognition of the nature reserve's function as a habitat and migration route for birds, vegetation and other animals, as well for the flood defence function of the Slufter. This change can be explained by the topics discussed during the session. In reacting to proposed policies, the participants agreed almost unanimously that the Slufter mouth should not be closed for the purpose of embankment (Figure 8.8). Figures 8.7 and 8.8 reveal that participants' opinions did not change radically, although the quality of the Slufter as a bird habitat or migration route was more valued than before the event. However, the knowledge intervention undertaken provides an indication that a shared understanding of the ecological and social functions of the Slufter estuary can be enhanced by an integration of a stakeholder approach and problem modelling.

8.7. Conclusions: system understanding and insights gained

This intervention brought new system knowledge on stakeholders' perceptions and estuary morphodynamics into a collaborative setting in which current practices in managing the inlet of the Slufter were under discussion. In this research, we used system knowledge in an experimental setting, with stakeholders within a potentially contentious situation. The system was understood in terms of its social-ecological system characteristics, the biophysical and social components, and their interactions. Participants were recognised as forming an integral part of this system. There was a high level of discussion ongoing amongst policy makers and stakeholders. The diversity and extent of the values that stakeholders associate with the island and the nature reserve the Slufter is such that it is an emotive issue. Despite this connection to the Slufter, there was a lack of urgency, which could also have affected the engagement of actors with the new knowledge (De Bruijn & Herder, 2009; Roeser, 2012). Additionally, the insights from the focussed knowledge intervention and its prior system analysis were later not fully adopted by the water board 'Hoogheemraadschap Hollands Noorderkwartier' (HHNK). HHNK initiated the research project, and also hosted, facilitated and controlled the intervention. Although HHNK gladly accepted this role, and strives for participation, they appear unaware that stakeholders would not necessarily perceive the activity as neutral. This could affect the efficacy of policy activities in the Netherlands (Deelstra et al., 2003; Kolb et al., 2008). The duality of the role of the water board (i.e., governance authority vs. stakeholder, task-oriented and stakeholder-engagement-oriented),and their habit of being in the driving seat are issues to consider when designing such activities. Different workshop participants (e.g., stakeholders with less connections to policy makers), small or one-on-one groups might be more effective conditions for knowledge interventions to improve system understanding and support ongoing coastal management (Andersen et al., 1997). Also, changing the way participants work with the supplied information, creating a more interactive approach or a comparison between different kinds of knowledge interventions could be implemented (Hommes et al., 2009; Reddel & Woolcock, 2004).

In conclusion, well-designed collaborative engagement interventions add local knowledge and stakeholder involvement in early design phases. This fits with the current practice of stakeholder management, especially for smaller projects that include interventions in coastal, ecological systems. The developed approach is useful to assess human values and use functions, in addition to biophysical qualities of these systems. The choice to conceptualise a dynamic coastal nature reserve as a social-ecological system allowed for the involvement of a wide range of stakeholders and accommodated dealing with the dynamic behaviour of the ecological system.

Today, the Integrated Coastal Management approach (ICM) aims to facilitate participation and conflict mediation, to ensure multi-sectoral planning and to balance conservation and development (Christie, 2005). The contextual nature of implementation means that site-specific knowledge is valued in the ICM field, and that articles describing the evolution of learning on, and about, ICM stress this notion (Cicin-Sain et al., 1998; Olsen et al., 1997). Engaging stakeholders to elicit knowledge, and other forms of participation, are becoming common practice (Reed, 2008) especially in water and coastal governance (Morinville & Harris, 2014; Taljaard et al., 2013).

As described in the previous sections, when discussing the flood defence in the Netherlands, we are mainly talking about interventions in the physical system. While safety levels are meeting basic levels of flood protection, there is room and recognition for societal values, such as ecosystem values, environmental protection, and other forms of (anthropocentric) human enjoyment of ecosystems. Consequently in more recent years, the focus on resolving flood defence issues has shifted from morphological solutions, to more inclusive interventions in the social system.

8.8. References

Andersen, D. F., Richardson, G. P., & Vennix, J. a M. (1997). Group Model Building: Adding More Science to the Craft. *System Dynamics Review,* 13(2), 187–201. https://doi.org/10.1002/(SICI)1099-1727(199722)13:2<187::AID-SDR124>3.0.CO;2-O

Balke, T. (2013). Opstellen monitoringsplan Slufter Texel: Plan van Aanpak [in Dutch]. Delft.

Christie, P. (2005). Is integrated coastal management sustainable? *Ocean & Coastal Management*, 48(3), 208–232.

Cicin-Sain, B., Knecht, R. W., Jang, D., & Fisk, G. W. (1998). *Integrated coastal and ocean management: concepts and practices.* Island Press.

Costanza, R., D'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., … Paruelo, J. (1997). The value of the world's ecosystem services and natural capital. *Nature*, 387(6630), 253–260.

D'Hont, F. M. (2014). Does deepening understanding of make sense? A partial success story from the Slufter, Texel. MSc Thesis. Delft University of Technology. Retrieved from http://repository.tudelft.nl/view/ir/uuid%3Af79420ff-9c9a-42b5-a7d4-65bb5db10ac3/

D'Hont, F. M., Slinger, J. H., & Goessen, P. (2014). A knowledge intervention to explore stakeholders' understanding of a dynamic coastal nature reserve. In 32nd International Conference of the System Dynamics Society. *System Dynamics Society.* Retrieved from http://www.systemdynamics.org/conferences/2014/proceed/index.html

De Bruijn, H., & Herder, P. M. (2009). System and actor perspectives on sociotechnical systems. Systems, *Man and Cybernetics, Part A: Systems and Humans*, IEEE Transactions On, 39(5), 981–992.

Deelstra, Y., Nooteboom, S. G., Kohlmann, H. R., van den Berg, J., & Innanen, S. (2003). Using knowledge for decision-making purposes in the context of large projects in The Netherlands. *Environmental Impact Assessment Review*, 23(5), 517–541. https://doi.org/10.1016/S0195-9255(03)00070-2

Delta Commission. (2016). Delta Programme. Retrieved from http://english.deltacommissaris.nl/delta-programme

Delta wet [Delta Act] (1958). The Hague: Ministerie van Infrastructuur en Milieu (Ministry of Infrastructure and the Environment).

Department of Environmental Affairs. (2008). Integrated Coastal Management Act. Cape Town, South Africa. Retrieved from https://www.environment.gov.za/sites/default/files/legislations/nem_integratedcoastal_management40_0.pdf

Durieux, M. (2004). De stabiliteit van de Slufter op Texel [in Dutch]. Delft University of Technology. Retrieved from http://repository.tudelft.nl/view/ir/uuid%3Aac0b9379-169e-42b3-90f8-b1f111ca27d4/

Farber, S. C., Costanza, R., & Wilson, M. A. (2002). Economic and ecological concepts for valuing ecosystem services. *Ecological Economics*, 41(3), 375–392. https://doi.org/10.1016/S0921-8009(02)00088-5

Hommes, S., Hulscher, S. J. M. H., Mulder, J. P. M., Otter, H. S., & Bressers, H. T. A. (2009). Role of perceptions and knowledge in the impact assessment for the extension of Mainport Rotterdam. *Marine Policy,* 33(1), 146–155.

Hoogheemraadschap Hollands Noorderkwartier. (2013). Concept beheerplan Natura 2000 Texel [in Dutch].

Kolb, J. A., Jin, S., & Song, J. H. (2008). A model of small group facilitator competencies. *Performance Improvement Quarterly,* 21(2), 119.

Mayer, I. S., Van Daalen, C. E., & Bots, P. W. G. (2004). Perspectives on policy analyses: A framework for understanding and design. *International Journal of Technology, Policy and Management*, 4(2), 169–191. https://doi.org/10.1504/IJTPM.2004.004819

Ministry of Defence. (2013). Coordinate systems. Retrieved from http://www.defensie.nl/english/navy/hydrographic_service/geodesy_and_tides/coordinate_systems/vertical

Morinville, C., & Harris, L. M. (2014). Participation, politics, and panaceas: exploring the possibilities and limits of participatory urban water governance in Accra, Ghana. *Ecology and Society* 2, 19(3), 36. https://doi.org/http://dx.doi.org/10.5751/ES-06623-190336

Mulder, J. P. M., Hommes, S., & Horstman, E. M. (2011). Implementation of coastal erosion management in the Netherlands. *Ocean & Coastal Management*, 54(12), 888–897.

Municipality of Texel. (2006). Hoogtekaart Texel. Retrieved from http://www.texelgevoel.nl/portalinfo.php?id=1258#.UjbvvGSWmt8

Nationaal Park Duinen van Texel. (2010). De Slufter en de Muy [in Dutch]. Retrieved from http://www.npduinenvantexel.nl/page/Natuur/De-Slufter-en-de-Muy

Newig, J., & Fritsch, O. (2009). Environmental governance: participatory, multi-level–and effective? *Environmental Policy and Governance,* 19(3), 197–214.

Olsen, J. P. (2009). Change and continuity: an institutional approach to institutions of democratic government. *European Political Science Review*, 1(01), 3–32.

Olsen, S., Tobey, J., & Kerr, M. (1997). A common framework for learning from ICM experience. *Ocean and Coastal Management*, 37(2), 155–174. https://doi.org/10.1016/S0964-5691(97)90105-8

Ostrom, E. (2009). A General Framework for Analyzing Sustainability of Social-Ecological Systems. *Science*, 325(419). https://doi.org/10.1126/science.1172133

Pedroli, A. I. J., & Hoekstra, G. B. M. (1992). Sluftervorming en natuurontwikkeling [in Dutch]. Delft. https://doi.org/TPG190123

Pranger, D. P. (1999). Vegetatiekartering Duinen - Noord Texel.

Pranger, D. P., & Tolman, M. E. (2011). Toelichting bij de vegetatiekartering Slufter en andere kwelders op Texel [in Dutch].

Reddel, T., & Woolcock, G. (2004). From consultation to participatory governance? A critical review of citizen engagement strategies in Queensland. *Australian Journal of Public Administration*, 63(3), 75–87.

Reed, M. S. (2008). Stakeholder participation for environmental management: A literature review. *Biological Conservation,* 141(10), 2417–2431. https://doi.org/10.1016/j.biocon.2008.07.014

Rijkswaterstaat. (n.d.). Texel - Toestand van de Duinen rondom de Slufter.

Rijkswaterstaat. (2013a). Duinen en Lage Land Texel - Natura 2000 beheerplannen. Retrieved from http://natura2000beheerplannen.nl/pages/duinen-en-lage-land-texel.aspx

Rijkswaterstaat. (2013b). Natura 2000-beheerplan Texel (Concept 1 July 2013).

Roeser, S. (2012). Risk communication, public engagement, and climate change: a role for emotions. *Risk Analysis*, 32(6), 1033–1040.

Slinger, J. H. (2017). Hydro-morphological modelling of small, wave-dominated estuaries. *Estuarine, Coastal and Shelf Science*, 198(B), 583–596. https://doi.org/10.1016/j.ecss.2016.10.038

Taljaard, S., Slinger, J. H., & van der Merwe, J. (2013). Dual adaptive cycles in implementing integrated coastal management. *Ocean and Coastal Management*, 84, 23–30. https://doi.org/10.1016/j.ocecoaman.2013.07.003

van Rooijen, A. A., & van Thiel de Vries, J. S. M. (2013). Stormgedreven morfodynamiek van De Slufter, Texel. Modelstudie naar het effect van de monding op de kustveiligheid en morfodynamiek [in Dutch]. Delft.

Vijfde Nota ruimtelijke ordening [Fifth Spatial Planning Policy Document] (2001). The Hague: Ministrie van Volkshuisvesting, Ruimtelijke Ordening en Milieubeheer (Ministry for Housing, Spatial Planning and the Environment). Retrieved from https://www.eerstekamer.nl/pkb/vijfde_nota_ruimtelijke_ordening

Vreugdenhil, H., Slinger, J., Kater, E., & Thissen, W. (2010). The Influence of Scale Preferences on the Design of a Water Innovation: A Case in Dutch River Management.

Environmental Management, 46(1), 29–43. https://doi.org/10.1007/s00267-010-9524-0

Waterwet [Water Act] (2009). The Hague: Ministerie van Infrastructuur en Milieu (Ministry of Infrastructure and the Environment). Retrieved from https://wetten.overheid.nl/BWBR0025458/2018-07-01

Wet op de Waterkering. Gewijzigd Voorstel van Wet [Flood Defence Act] (1996). The Hague: Ministry of Transport, Public Works and Water Management.

9

Transdisciplinary Learning Across Case Studies

By Jill Slinger and Susan Taljaard

Seven international case studies in coastal management are examined in this book. Each of the case studies dealt with an inlet or mouth management issue, but they were not selected based on similarity in their ecological (biophysical) or social systems. The case studies also differed in terms of the dominant environmental paradigm used by the scientists in their original analyses, as clarified in the introductory and other chapters. During the international cross-comparative workshop, aimed at transdisciplinary learning within and across the case studies, the marked differences in the research emphasis placed on the ecological and/or the social system (the object of inquiry), in the connections made to management practice and policy decision making (the theoretical lenses informing the way of inquiry), and in the knowledge sources used (the way of inquiry), became apparent. Accordingly, in the following sections the learning from the transdisciplinary research endeavour is synthesised by cross-comparing the coastal systems (S), the methods (M) applied and the concepts employed by the involved scientists (C). The cross-comparison is itself informed by concepts from systems thinking and policy analysis, with the aim of influencing coastal management and research practice internationally.

9.1. Insights from the case studies

In this section, the insights deriving directly from each of the case studies are summarised. The focus of each of the case studies, conceived as nested within a broad social-ecological system, is clarified and the relevance for coastal management practice is explained.

The two case studies of big bays or inlet systems, namely Texel Inlet and Dundalk Bay, reveal a move to involve stakeholders and communities in the integrated planning and management of these large systems. So, the focus shifts from the biophysical system alone to encompass an integrated social-ecological system. The case studies involving small, wave-dominated estuaries, namely the Maha Oya, Russian River, Groot Brak and Slufter estuaries, are more diverse in their emphases, varying from science pro-actively signalling a need for management, to intensive co-management of an estuary and its mouth. All case studies highlight the need for improved understanding of the estuaries as social-ecological systems, with due consideration for interlinkages between the social and ecological systems. As such, they also indicate a shift towards a broader contextualised understanding of the inlet and mouth management issues. Indeed, the Bigi Pan reveals that biophysical system understanding is fundamental to effective and sustainable management, and the Slufter case study reveals the value of social system understanding in moving to new integrated management strategies.

The Texel Inlet case study highlights how flood risk management has dominated other potential concerns in determining the objectives for coastal management. This single issue focus and the intensive and sustained collection of monitoring data to enable analysis of the efficacy of management interventions, has succeeded in the goal of protecting the inhabitants of south west Texel from flooding over a long period of time. Scientific insights on the geomorphological dynamics of the ebb-tidal delta at the inlet reveal that continued investment in ongoing sand nourishments may not be required to maintain coastal safety in the near future. Indeed, there is an indication that the sandy shoal Noorderhaaks may conjoin with the adjacent coastline delivering vast quantities of sand in a natural manner. Uncertainty remains as to the timing of this anticipated dynamic change and the precise mechanisms by which it could occur (Wijnberg et al., 2017). The societal costs and consequences are not addressed in this case study. Instead, the need to move to a collaborative, participatory approach in designing alternative coastal management strategies that take this new understanding into account is highlighted.

The Dundalk Bay case study rests on a thorough environmental assessment of the catchment and bay systems, but also highlights a move towards engaged co-management approaches at the community level. These efforts are directed at learning and supporting sustained social involvement with the integrated management of the catchment-bay system.

The Maha Oya case study illustrates the role of scientific knowledge in alerting coastal managers of the need to plan for future environmental change, because of the strong effects on the linked social system. The intermittent closure regime of the Maha Oya Estuary mouth is anticipated to change, affecting local fisherman, sand mining, tourism and many other livelihood associated functions in the future. The strong influence of mouth condition on the ecological health of the estuary is similar to the Russian River in California. Here, the biophysical system knowledge drawn from an extensive data set is used to determine habitat suitability for an endangered species, and the management of the mouth is then optimised for this single species objective. Science thus serves to give form to policy, and mouth management is focussed on one critical indicator.

In the Groot Brak Estuary the condition of the estuary impacts the social system associated with it. Activities and (dis)services deriving from the estuary such as tourism, aesthetic value, fishing and flooding are affected by the condition of the mouth. The case study reveals ongoing learning regarding the character and functioning of the estuary, and highlights how the growing scientific understanding, commencing in the late 1980's with the construction of an upstream reservoir, has influenced management practice and policy. The co-evolution of the Groot Brak Estuary and her people (see Slinger et al., 2012) highlights the strong interlinkages in a social-ecological system.

The Bigi Pan case study evaluates the management of the Multiple Use Management Area (a social-ecological system), drawing upon an extensive round of stakeholder interviews amongst people living in the area and the managing authorities. It highlights the need for (biophysical) system understanding as the foundation for effective coastal management, and identifies a number of strategies to address this gap and improve the existing ecosystem-based management of the Bigi Pan wetland in Suriname.

An extensive process of stakeholder engagement also characterises the Slufter case study. Here, the divergent perspectives and values of local stakeholders in regard to inlet management were explored with the aid of system dynamics modelling (D'Hont, 2014). The role of (social-ecological) system understanding is shown to be fundamental to learning in regard to coastal management.

9.2. Framing the case studies in terms of the system diagram of policy analysis

The seven international case studies were undertaken within diverse theoretical paradigms, leading to diversity in their focus - which object of study, or system (S) they see. As the overarching goal of this book is to learn within and across case studies, we adopt a broad social-ecological systems lens and draw upon systems thinking and policy analysis methods to locate each of the case studies on the system diagram. This enables us to clarify the emphasis of each of the studies relative to one another, and to understand which insights they contribute towards coastal management practice.

First, we locate Texel Inlet at the interface between actions/interferences and the system box (ellipse 1 in Figure 9.1), because this case study focuses on interventions to maintain coastal safety taking the physical dynamics of the environment into account. The objectives for coastal safety are even formulated in terms of the physical environment, although there is growing awareness of managing for multiple objectives. Indeed, the case study highlights the need to move towards more stakeholder-inclusive approaches in coastal management. The Dundalk Bay case study provides an overall assessment of the bay system, including the inflowing rivers, examining the impacts of actions and relating

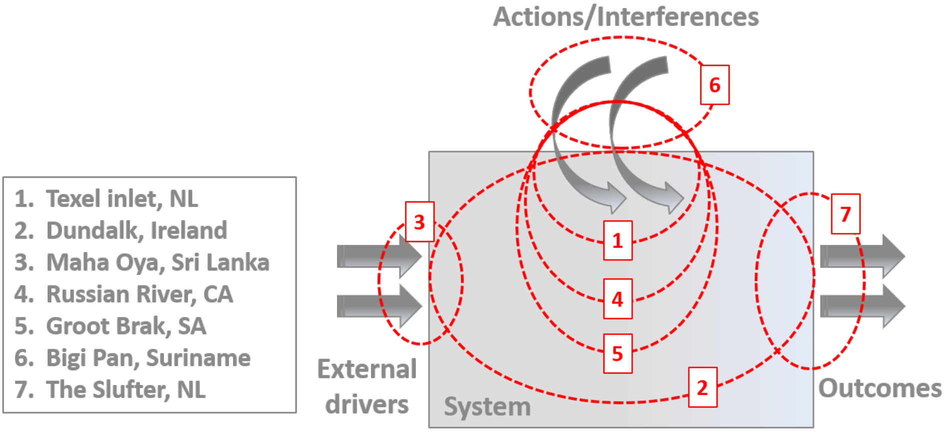


Figure 9.1. Framing the dominant focus of the case studies in terms of a system diagram

these outcomes, but focussing on the state of the environment of the bay system itself. It is therefore located entirely within the system box (ellipse 2 in Figure 9.1). In contrast, the Maha Oya case study provides a pro-active environmental assessment focussing on an external driver, namely climate change, and its effects primarily on the mouth of the estuary. It is therefore located at the interface between external drivers and the system box (ellipse 3 in Figure 9.1). The Russian River case study investigates the implementation of proposed actions in the system to achieve given objectives, elucidating the physical dynamics, water quality and biotic (single species) responses. Some associated social issues, e.g., fisheries, conservation are also addressed. Accordingly, this case study encompasses actions/interferences and protrudes into the systems box (ellipse 4 in Figure 9.1), to accommodate the depth of environmental systems knowledge that is used. The Groot Brak case study adaptively implements flow release and mouth management actions in the system focussing on physical, water quality and multiple biotic components (vegetation, invertebrates and fish) and associated social use, e.g., flood protection, tourism, and ecosystem health linked to aesthetics. It therefore encompasses actions/interferences and also protrudes into the systems box a little further than the Russian River to indicate the multiple species considerations (ellipse 5 in Figure 9.1). The Bigi Pan case study used an ecosystems-based approach to engage with stakeholders in seeking to identify improved actions/interventions than those implemented at the time of the study, to deal with the problems then experienced with the management of the coastal wetland. The understanding of the biophysical system held by those interviewed was not extensive, and the case study is therefore located at the interface between actions/interventions and the systems box (ellipse 6 in Figure 9.1). The case study highlighted the need for system understanding in informing actions. Finally, in the Slufter case study a social-ecological systems approach was used to engage with stakeholders on their expectations/values as potentially affected by management actions in the system, so as to negotiate desirable outcomes. Knowledge on biophysical systems functioning was also used in the engagement process. This means that the Slufter is located at the interface between the system box and the outcomes (ellipse 7 in Figure 9.1).

The location of the case studies on the system diagram enables us to understand that just as the underlying theoretical paradigms (C) were diverse, each of the case studies differs in their focus (their object of study - S), yielding a range of insights and using different types of knowledge in doing so (their strategy of inquiry - M). Some focus on the biophysical system, signalling to the social system, some focus on the social system emphasising the need for biophysical understanding, and others reveal co-evolutionary management of the social-ecological system. Next, the contribution of key knowledge sources to each of the case studies will be explored.

9.3. Framing the key knowledge sources of the case studies

In coastal management practice and research, three generic groups of stakeholders (termed actors) are involved, namely scientists, policy makers and coastal citizens. It follows that three actor-based knowledge sources can be distinguished. First, there is scientific knowledge, encompassing the many disciplines concerned with the coast. These include (i) the empirically focussed disciplines of geology, geomorphology, physics (hydrodynamics), (water and soil) chemistry and ecology, (ii) the pragmatically oriented disciplines of resource management, policy analysis, simulation modelling and agriculture, (iii) the normative disciplines of planning, engineering design, law, and education, as well as the purposive disciplines of ethics and philosophy (cf. Max-Neef, 2005). The actor-based knowledge of scientists includes knowledge of the research methods (M) appropriate to their disciplines, how to conduct field work, and methods for communicating their research findings, for instance.

Second, there is the knowledge of policy makers regarding the decision making processes operative in the context within which they work. This knowledge is founded on their underlying disciplinary expertise and training, but in the context of coastal management we take it as including the experiential knowledge of their own governance context and how to work effectively within this context (M). It therefore includes knowledge on the formal and informal rules at play, who has what influence, who controls which resources and the distribution of power relations (Ostrom, 2009). Furthermore, it includes knowledge of the actor-network of citizens, scientists and (local) coastal management practitioners in their area.

Third, there is the knowledge of citizens resident in the coastal area of interest. This knowledge is place-based and spread amongst different people. It is often integrated in nature, rather than reductionist, having to do with living in the particular coastal area and experiencing the effects of different coastal management interventions over time. This knowledge can reveal the unexpected side effects of interventions as opposed to the effects envisaged at the time of their introduction. As such, it represents valuable input to effective coastal management, but requires social science methods and policy analysis expertise (M) to access and involve appropriately.

A comparison across the case studies reveals that the key knowledge sources that were used differ substantially. This is reflected in Figure 9.2, which depicts the ex-post positioning of the case studies undertaken using policy analysis methods. The further a case study is from an actor-based knowledge node, the less this knowledge source has

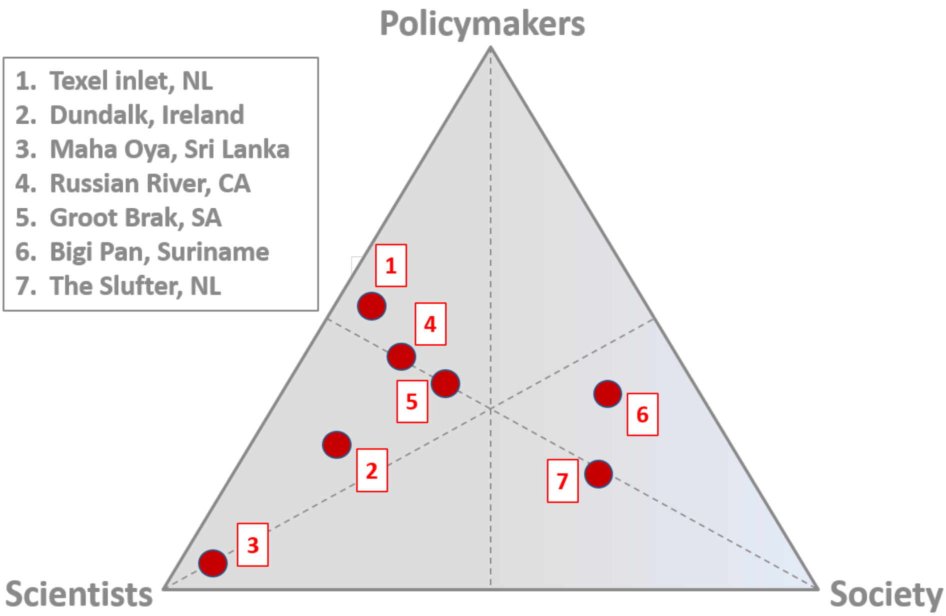


Figure 9.2. Relative contribution of three actor-based knowledge sources to the case studies.

been used in the case study. A case study located in the middle of the triangle would have an equal contribution from all knowledge sources. A case study located at the middle of a border between two knowledge sources would have a relative contribution of 50% from each of the adjacent knowledge sources and 0% from the opposing node. A case study located at a node would have a relative contribution of 100% from this knowledge source.

The case studies drawing most extensively on knowledge from society (local, place-based knowledge of citizens) are the Bigi Pan in Suriname and the Slufter in the Netherlands (Figure 9.2). Both case studies employed interviews with stakeholders as a research and knowledge acquisition strategy. The case study with the most knowledge sourced from policy is the Texel Inlet in the Netherlands. Although, it should be remembered that this analysis merely reflects the knowledge delivery as presented in these case descriptions. A wider issue focus than mouth management in the Groot Brak Estuary in South Africa would have revealed strong policy knowledge sources in the estuary management (see Slinger et al., 2005, 2012). In this instance, however, the Groot Brak Estuary case study is positioned as drawing very equally on knowledge sources from science and policy, as is the Russian River in California (Figure 9.2). In contrast, the Maha Oya Estuary in Sri Lanka is located close to the scientific knowledge node, because it is scientific knowledge that is being used to alert coastal managers and policy makers to the potential changes in mouth closure owing to climate change and the associated potential impacts on citizens (Figure 9.2). In effect, it is a wake-up call. The Dundalk Bay case study is similar in that scientists are collating information, assessing environmental status in order to begin to engage effectively in a dialogue about the integrated management of the bay system and its catchment (Figure 9.2).

The positioning of the case studies in terms of the relative contribution of actor-based knowledge sources is useful for clarifying (i) the means of learning employed within the case studies (M), and (ii) indicating potential additional sources of information and knowledge to supplement the existing efforts in coastal management (potential new M). At the same time, it sheds light on the extent of the object of study within the broad social-ecological coastal system (S). Where scientists have been involved for a longer period of time and the subject of involvement is well established, the knowledge sources accessed seem to be more distributed. Where new issues are raised or scientists are in the early stages of involvement, the knowledge sources are localised between one or two nodes. This is not to imply that studies will or should necessarily broaden their focus over time. The reason for scientists engaging in these case studies at the outset remains an inlet or mouth management issue. This study has highlighted that engaging with environmental management on the part of scientists, policy makers and local stakeholders is beneficial even when the state of knowledge of the ecological system is still a limiting factor. Such engagement appears to stimulate the development of different types of knowledge that then act to anchor the management approach within the social system, arguably making it more robust owing to the distributed learning of the involved stakeholders. So it is not necessarily the passage of time that engenders this effect, but the experiential learning of the people involved and its distribution amongst the actor-groups concerned. This serves to widen the object of study (S), that is it broadens the focus of the system that is seen.

9.4. Stepping towards the future

The goal of this research was to learn from a number of coastal case studies. The case studies in the international cross-comparison are each characterised by an inlet management or estuary mouth management issue that is understood by the involved scientists to be nested within a broader ecological and social context. We did not require the case studies to be located within similar biophysical or social systems. This means that specific, disciplinary insights were unlikely to emerge from cross-comparison. Instead, drawing upon systems thinking and policy analysis approaches (the way of inquiry of the overarching transdisciplinary endeavour), similarities and differences in the foci of the coastal case studies (S), and the types of knowledge employed (M) based on the diverse theoretical paradigms (C) adopted by the researchers, were distinguished.

In all of the case studies, deep place-based knowledge was used. Sometimes this came primarily from the scientists through measurements and modelling (e.g., Maha Oya, Texel Inlet) and sometimes this derived from local citizens and scientists (e.g., Russian River, Groot Brak). Some cases were predominantly focussed on the social aspects and some on the environmental system only. In summary, these analyses:

- reveal that no case study examined the social-ecological system as a whole,
- identify which aspects would need to be address should the focus broaden to address the whole system, and
- clarify the under-utilised actor-based sources of knowledge.

Drawing on this cross-comparative analysis with its social-ecological systems view and policy analytic strategy of inquiry, we can infer strengths and weaknesses of the individual case studies, and so clarify which aspects, deriving initially from the predominant paradigm underlying each case study, should still be maintained in their further study and management. In Table 9.1, we also identify which aspects need to be included in taking steps to broaden the case studies to address the full social-ecological system and strengthen coastal management into the future.

Finally, we conclude by reflecting that the transdisciplinary research synthesised in this book represents an endeavour to learn by action research and policy analysis methods (M) across a (nested) social-ecological research system comprising scientists with their case studies. We trust that this endeavour will inspire others to undertake transdisciplinary learning and contribute to wise coastal research and practice.

Table 9.1. Stepping into the future – which aspects need to be maintained and included as the predominant paradigm underlying each of the case studies is broadened to the full social-ecological system

Case Studies	Predominant theoretical paradigm per case study	Transdisciplinary learning implies the following actions per case study	
Texel Inlet	Objectives-based Management	Include	Multiple objectives Co-design with stakeholders
		Maintain	Environmental monitoring
Dundalk Bay	Environmental Assessment	Include	Community engagement Enhanced integration of catchment and bay
		Maintain	Environmental monitoring
Maha Oya	Environmental Assessment	Include	Engagement with authorities and community Environmental monitoring
		Maintain	Climate change predictive science
Russian River	Objectives-based Management	Include	Multiple objectives
		Maintain	Environmental monitoring Community engagement
Great Brak	Adaptive Management	Include	Co-design
		Maintain	Adaptive scientific and community learning Environmental monitoring
Bigi Pan	Environmental Assessment	Include	Biophysical system understanding Stakeholder-engagement at local level Ecosystem-based management implemen
		Maintain	Ecosystem-based management planning
The Slufter	Social-Ecological Systems	Include	Co-design Stakeholder perception monitoring
		Maintain	Environmental monitoring

9.5. References

D'Hont, F. M. (2014). *Does deepening understanding of make sense? A partial success story from the Slufter, Texel.* MSc Thesis. Delft University of Technology. Retrieved from http://repository.tudelft.nl/view/ir/uuid%3Af79420ff-9c9a-42b5-a7d4-65bb5db10ac3/

Max-Neef, M. A. (2005). Foundations of transdisciplinarity. *Ecological Economics*, 53(1), 5–16. https://doi.org/10.1016/j.ecolecon.2005.01.014

Ostrom, E. (2009). A general framework for analyzing sustainability of social-ecological systems. *Science,* 325(5939), 419–422. https://doi.org/10.1126/science.1172133

Slinger, J. H., Huizinga, P., Taljaard, S., van Niekerk, L., & Enserink, B. (2005). From impact assessment to effective management plans: Learning from the Great Brak Estuary in South Africa. *Impact Assessment and Project Appraisal,* 23(3), 197–204. https://doi.org/10.3152/147154605781765562

Slinger, J. H., Linnane, S., Taljaard, S., Palmer, C., Hermans, L., Cunningham, S., … Clifford-Holmes, J. (2012). From policy to practice: enhancing implementation of water policies for sustainable development. The Story of the Great Brak: Water and Society. Delft, The Netherlands: Delft University of Technology.

Wijnberg, K. M., van der Spek, A. J. F., Silva, F. G., Elias, E., Wegen, M. van der, & Slinger, J. H. (2017). Connecting subtidal and subaerial sand transport pathways in the Texel inlet system. *Coastal Dynamics Proceedings*, (235), 323–332.

Complex coastal systems

transdisciplinary learning on international case studies

Jill H. Slinger · Susan Taljaard · Floortje M. d'Hont *Editors*

The transdisciplinary research synthesised in this book represents an endeavour by a group of coastal researchers and policy analysts to learn from a cross-comparison of seven international case studies on tidal inlet or estuary mouth management situations, located in South Africa, Sri Lanka, California, Suriname, Ireland and the Netherlands. The conceptual framing is provided by a focus on systems knowledge and its development and use within coastal management.

This book is intended for:

- Transdisciplinary scholars who are interested in interdisciplinary learning and knowledge exchange,
- Policy analysts, environmental historians and coastal policy specialists who are interested in the role of science in the evolution of coastal policy and management,
- Coastal scientists and engineers interested in the dynamics of tidal inlets and estuary mouths,
- Coastal managers looking to learn about tidal inlet and mouth management practices
- Educators focussed on interdisciplinary skills or interested in using the case studies in coastal, management and engineering classes or as the basis for problem structuring exercises by policy students, and
- Students interested in coastal systems management and wanting to broaden their interdisciplinary competence.

We trust that this endeavour will inspire others to undertake transdisciplinary learning and contribute to wise coastal research and practice.

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