Estuaries of the World

Salif Diop
Peter Scheren
John Machiwa *Editors*

Estuaries: A Lifeline of Ecosystem Services in the Western Indian Ocean

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Fishing boats resting on Pemba Island, Tanzania (photo by Peter Scheren, February 2010)

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"*. . ..Doubling the share of African University graduates in science and technology fields within a decade, by 2025 is the key to transform Africa into a knowledge-driven continent within a generation. . ...*"

From ***Makhtar Diop***, World Bank's Vice President for Africa. *Extracted from an article written 24th May, 2014 for an Ethiopian Journal* "*The Reporter*" *by Moctar Diop on* "*Powering Science, technology for Africa*'*s economic transformation. . .*

"***Put your thoughts*** *in order by reflection, pen in hand***.** *Appropriate you the power of the pen. Read, read,* ***read*** *every day, pen in hand.*"

From Nelson **Rolihlahla** *Mandela* "*Madiba*" (1918–2013).

Foreword

It is in estuaries and deltas where the richness of the land meets the abundance of the sea, creating an environment of high diversity, dynamism and productivity. Nevertheless, the important contributions estuaries make to local livelihoods and national economies, as well as to their complex role in the functioning of the land-ocean interface, are often overlooked.

The Western Indian Ocean is dotted with important estuaries and deltas, including the Tana and Sabaki in Kenya, Pangani, Rufiji and Ruvuma in Tanzania, Zambezi, Incomati, Maputo, Pungwe and Limpopo in Mozambique, Thukela in South Africa and Betsiboka in Madagascar. With numerous plans and investments in place for a massive acceleration in infrastructure development and energy and food production, human activities will increasingly impact these important estuarine and coastal ecosystems and the life-supporting services they provide. In this regard, the unique, diverse and productive estuaries and deltas of the Western Indian Ocean stand at a crucial crossroad.

The Western Indian Ocean is internationally recognized as a hot spot of biodiversity, hosting one-third of the 38 globally, recognized marine and coastal habitats, an abundance of fish species and marine mammals, all five marine turtle species, over 40 species of seabirds and the longest fringing reef in the world. The region is also home to the charismatic coelacanth, nicknamed the living fossil, and the critically, endangered sawfish and seahorse. Furthermore, the region's coastal and marine waters are important fishing grounds, supporting the livelihoods of the local population. Its marine parks and other protected areas are also the basis for an active tourism industry.

The unfortunate reality, however, is that human activities in these river catchments are having increasingly serious impacts on these sensitive downstream estuarine and coastal ecosystems. The damming of rivers over the past 50 years, combined with reduced rainfall, expansion of irrigated agriculture and other increasing water abstraction and land uses within various catchments, are among the underlying causes of those changes. Furthermore, pollution from municipal and industrial effluents is exacerbating the serious degradation of waters and sediment quality that is being observed in these rivers, estuaries and coastal waters, resulting in a loss of biodiversity, increasing eutrophication and reduced fish catches in many locations in the Western Indian Ocean.

To set the stage for addressing the continuing degradation of these important land-sea interfacing water systems, this publication was made possible due to the leadership of four scientists: Prof. Salif Diop from the University of Dakar; Dr. Peter Scheren from WWF; Prof. John Machiwa from the University of Dar es Salaam and Prof. Jean-Paul Ducrotoy from the Institute of Estuarine and Coastal Studies, The University of Hull, UK.

The focus of this book is on estuaries, but its scope and implications extend well beyond this particular coastal feature. Indeed, estuaries can only be considered as part of the life cycle of the entire river basins draining into them and the downstream marine areas that receive these riverine inputs. These interlinked systems and the life-supporting ecosystem services they provide are particularly sensitive to human and natural pressures; hence the title of this book "**Estuaries: a Lifeline of Ecosystem Services in the Western Indian Ocean**". It is our belief that this book will be a valuable source of information and guidance for the numerous scientists, researchers, managers and decision makers concerned with the integrated management of estuaries, deltas, lagoons, and the coastal and marine areas of the Western Indian Ocean and will help facilitating their sustainable use.

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Preface

This volume "Estuaries: a lifeline of ecosystem services in the Western Indian Ocean" published in the book series "Estuaries of the World" (EOTW) by Springer is the second of its nature focusing on Africa. The case studies presented in this book provide clear evidence of the fact that the estuarine ecosystems of this region are extremely valuable in providing cultural (recreational, spiritual, etc.), provisioning (food, timber, etc.) and regulatory (flood protection, climate regulation, etc.) services that are not only at the core of the coastal ecosystem functioning, but also an important basis of livelihoods of over 60 million inhabitants living in the region; the coastal ecosystems of the region, and in particular estuaries, represent important socio-economic values based on irreplaceable ecosystem functions. However, these valuable ecosystems are subject to a range of human pressures that may compromise the health of living human residents. These disturbances are multiple and include pollutants, excess nutrients (causing eutrophication), loss and transformation of habitats and disturbance of hydrological regimes causing flooding and unpredictable flow patterns. The effects of these impacts, often acting in cumulative and synergistic manners, affect the overall stability of the system and threaten its strength and resilience.

Unfortunately, due to inadequacies in the management and governance of these ecosystems, local management is often unable to control the basic causes of these attacks on ecosystem integrity, instead passively responding to their consequences without treating the cause. In addition, the exogenous pressures imposed by global climate change amplify the scale of stress on ecosystems. Its consequences (e.g. the increase in temperature, sea level rise, increased risks of flooding, etc.) may intensify the risk of seeing abrupt and nonlinear changes in natural systems. This will have an impact on flora and fauna, their structure (species richness and biological diversity), their functioning and their biological productivity.

At risk of compromising future development, policy makers are confronted with economic and legal constraints which often are antagonistic. The complexity of understanding human-marine coastal environment interactions as evidenced in this book, explain why the ecosystem-based approach constitutes one of the most valuable frameworks for promoting the sustainable development of marine and coastal ecosystems in the Western Indian Ocean as elsewhere in the world. Indeed, the book shows that adequate knowledge, scientific information and capacity, awareness and governance on ecological processes and the important role and value of ecosystems goods and services they provide are the key to allowing coastal communities and policy makers to define adequate responses to the threats at hand.

The potential of coastal or estuarine systems to provide crucial ecosystem functions and services depends on the stability of prevailing abiotic conditions, organised along gradual gradients and ecotones; a good understanding of the diversity of ecological processes in estuaries is therefore critical to apprehend the complexity of the system. More so, it is important to understand the linkages between these ecosystems and the economic realm, including aspects of market, land and other property rights regimes, as well as their related government structures and social networks. Unfortunately, more often than not, economic forces have considered ecosystems and the resources they provide as "free", not taking account of their economic externalities in terms of the adverse effects of their use.

Based on the above underlying philosophy, the present book chapters describe the various ecosystem functions and values of the region's estuarine ecosystems and their respective habitats, including the land/ocean interactions that define and impact ecosystem services. The Western Indian Ocean region covered by this volume consists of the continental coastal states of Kenya, Mozambique, South Africa and Tanzania and the island states of Madagascar, Seychelles, Comoros and Mauritius, all being signatories to the Nairobi Convention for the protection, management and development of the marine environment of the region. One of the main goals of the Nairobi Convention is "to promote a mechanism for regional cooperation, coordination and collaborative actions in the Eastern and Southern African region that enable the contracting parties to harness resources and expertise from a wide range of stakeholders and interest groups towards solving interlinked problems of coastal and marine environment, including critical and transboundary issues".[1]

The authors of the book do hope that their scientific contribution will help support decision making and promote robust capacity building and management programmes tailored to the region's needs. In this regard, this book aims to provide a good scientific assessment of ecological conditions prevailing in the region which undoubtedly will benefit any future commitment to sustainable development of coastal areas of the Western Indian Ocean.

Dakar-Fann, Senegal — Salif Diop
Dar es Salaam, Tanzania — Peter Scheren
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[1] *extract from the* "*Nairobi Convention*".

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The Western Indian Ocean: A Wealth of Life-Supporting Ecosystem Goods and Services

Peter Scheren, Salif Diop, John Machiwa, and Jean-Paul Ducrotoy

Abstract

The coastal and marine environment of the Western Indian Ocean (WIO) region is one of the least ecologically disturbed in the world. The region is a hot spot of biodiversity hosting over 2,200 species of fish, five species of marine turtles, more than thirty-five marine mammal species, including humpback whales, dolphins, whale sharks and the highly endangered dugong, and the enigmatic coelacanth; a prehistoric fish once thought to be extinct. The region furthermore boasts the longest unfragmented fringing reef in the world, with over 350 species of corals, and a diverse assemblage of coastal forests, mangrove forests and sea grass beds. It is estimated that about 22 per cent of the species found in the WIO region are unique to this region. The ecosystem services provided by this rich marine environment are estimated at over 25 billion US$ per year. At the crossroad between land and sea, the estuarine ecosystems of the region are the gathering point of a number of large river basins, many of which transboundary of nature, and bringing in influence from far away inland. This chapter provides an overview of the key ecological and physical characteristics of the region, setting the background for the deeper analyses presented in the following chapters.

Keywords

WIO Region • Mangrove forest • Coral reef • Seagrass bed • Fisheries • Coelacanth • Environmental change • Marine life • Ecosystem services

Geographical Setting

The Western Indian Ocean (WIO) region extends from approximately latitude 12° N to 34° S and longitude 30° E to 80° E, a total area of about 30 million km^2, equivalent to 8.1% of the global ocean surface (FAO 2007). The region has a combined coastline exceeding 15,000 km (including those of the island states) and a total continental shelf area of about 450,000 km^2 (GEO 2003).

Disclaimer: Parts of this chapter are based upon the recently concluded Transboundary Diagnostic Analysis of Land-based Sources and Activities in the Western Indian Ocean (UNEP/Nairobi Convention Secretariat and WIOMSA 2009). In this regard, no use of the Chapter may be made for any commercial purposes without the prior approval of the UNEP/Nairobi Convention Secretariat and WIOMSA.

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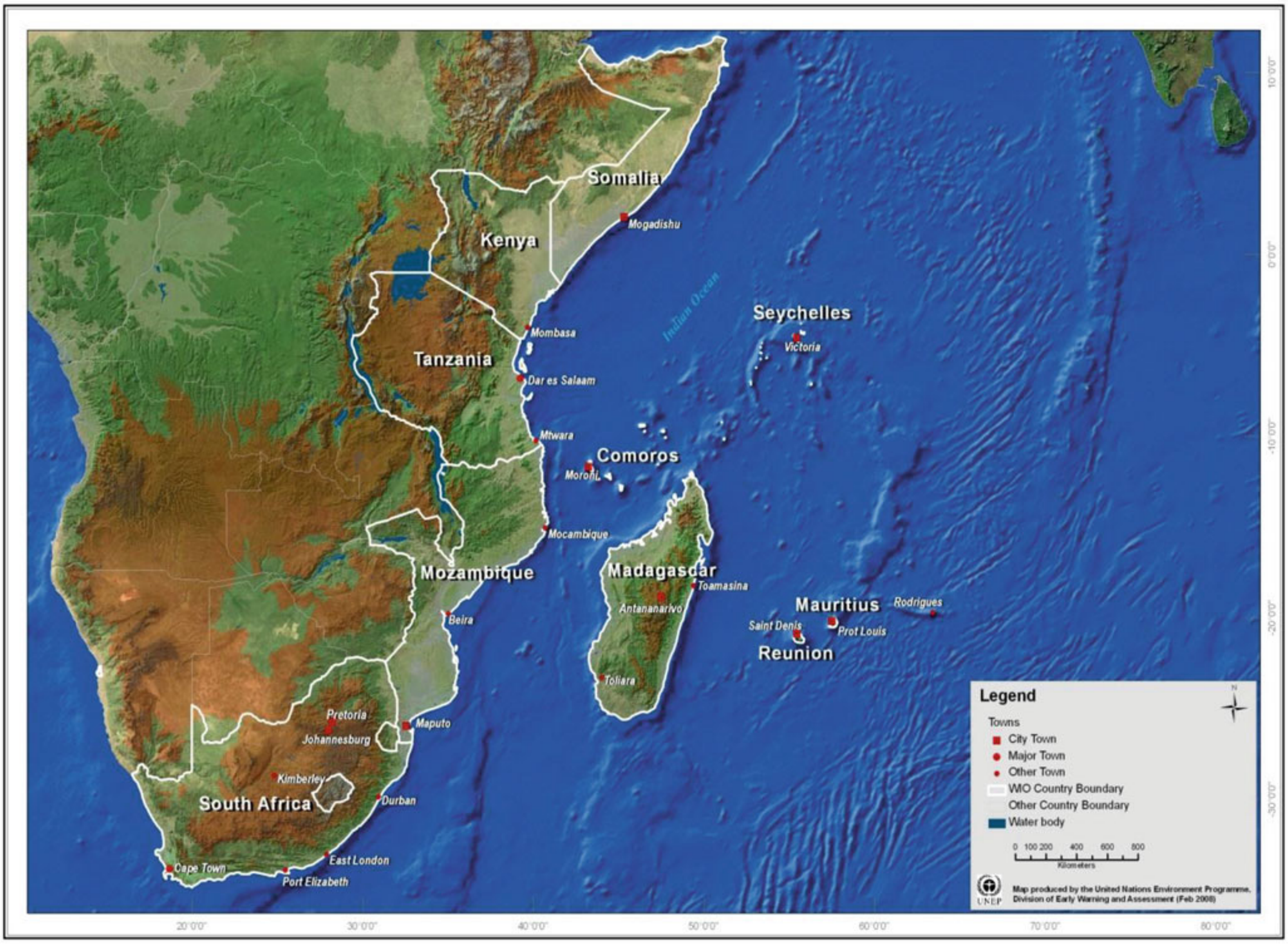

Fig. 1 Physical setting of the Western Indian Ocean region

The WIO encompasses a large array of marine and coastal settings, ranging from small volcanic and coral islands to large continental countries with extensive coastlines and tropical and subtropical climates. The African mainland states are Somalia, Kenya, Tanzania, Mozambique and South Africa, while the island states are Mauritius, Comoros, Seychelles, Madagascar and Réunion-France (Fig. 1).

Climatic Conditions

The climate in the WIO region ranges from sub-tropical to tropical (FAO 2005a). The mean daily temperatures in the northern parts range from 25 °C–29 °C, while the hottest summers reach 35 °C, usually during the months of December through February (Carbone and Accordi 2000). Further south, in Tanzania and northern Mozambique, mean annual temperature ranges between 18 °C and 36 °C (Hughes et al. 1992), while the average annual temperature on the east coast of South Africa is 21 °C, although dropping to 8–10 °C during the months of June to August in the southern extreme of the region (AFRISCO 1994; FAO 2005a).

The prevailing wind regimes in the WIO region can be divided into two distinct systems: the monsoon regime that dominates the Somali Current Large Marine Ecosystem (SCLME), and the subtropical high-pressure system that dominates the southern region (the Agulhas Current LME, or ACLME) (Beckley 1998; Okemwa 1998). The Northeast Monsoon affects the climate of the northwest Indian Ocean from November to March, and is characterized by northeasterly winds over the tropics and northern subtropics. The Northeast Monsoon exhibits winds of moderate strength, with dry terrestrially-derived air blowing from Arabia to Madagascar (Weller et al. 1998). In contrast, during the Southwest Monsoon (June to October), the wind direction reverses and the winds then tend to be much stronger, with an intense wind stream developing along the high Eastern African highlands (Ethiopian highlands; Kenya highlands; highlands of northern and southern Tanzania; etc.) (Slingo et al. 2005).

The rainfall pattern in the region decreases northwards from Mozambique, ranging between 530–1,140 mm, to

Somalia, ranging between 250–375 mm per year. The island states receive more rainfall on average than the mainland states of eastern and southern Africa (FAO 2005a). The annual rainfall in Seychelles, Mauritius and Comoros, for example, ranges between 2,000–4,000 mm (FAO 2005a). In contrast, the maximum rainfall in the coastal regions of continental states such as South Africa, Mozambique, Tanzania and Kenya does not exceed 1,500 mm, and usually is in the range of 500 to 1,000 mm (FAO 2005a).

The rainfall seasons in the WIO are strongly influenced by monsoon winds. The northern part of Mozambique, Tanzania, Kenya and the southern parts of Somalia receive heavy, extended rains in the March through May period before the Southeast Monsoon sets in (FAO 2005a). Short rains are experienced in October through December in the same region during the Northeast Monsoon (AFRISCO 1994; Kitheka et al. 2004). The islands of the Seychelles receive heavy, extended rains during the Northeast Monsoon, while the rainfall pattern in the other island states is strongly influenced by the Southeast Monsoon (FAO 2005a).

The volume of river discharge into the Indian Ocean reflects the rainfall patterns in the region to a certain extent. Thus, rivers draining high rainfall areas exhibit relatively higher discharges (Alemaw and Chaoko 2006). In the northern parts of the WIO region (e.g. Somalia and Kenya), the estimated total annual river discharge is in the range 1.8–4.95 km^3/yr. The annual river discharge in the central and southern parts (e.g. Tanzania, Mozambique and South Africa) is in the range of 2.9–106 km^3 (Hatziolos et al. 1996; Hirji et al. 1996; FAO 2001; UNEP 2001). As a result, the southern parts of the WIO region (particularly Mozambique) are characterized by the presence of large estuaries supporting extensive mangrove forests (Taylor et al. 2003).

Geology and Geomorphology

In regard to its geological structure, the coastline of eastern Africa represents a passive continental margin from which continental fragments have separated and migrated across the adjoining oceanic crust over geological time, and creating what is now the Indian Ocean (Kairu and Nyandwi 2000). This 200-million-year process is reflected in the heterogeneity of the current WIO geological formations. The coastal sediments of Tanzania, Kenya and Mozambique, for example, vary in age from the Jurassic through the Cretaceous, to the Tertiary and Quaternary, being composed of both marine and terrestrial sedimentary rocks (see Kent et al. 1971). Some of the detached continental fragments comprise the granitic islands of the main Seychelles group and the island of Madagascar. The more recent outer islands of the Seychelles Archipelago (e.g., Aldabra, Cosmoledo) and the islands of Réunion, Comoros, Mauritius and Rodrigues are essentially of volcanic origin (Stoddart 1984).

This structural history has left the mainland states with generally narrow continental shelves (Ngusaru 1997), exceptions being the central parts of the coasts of Mozambique at Sofala, central Tanzania in the vicinity of Unguja and Mafia islands, the sedimentary river banks off the major rivers in the south (e.g., Maputo Bay in Mozambique and the Thukela Banks in South Africa). Similar wide continental shelves are found along western Madagascar. Although of different geological origin, the Seychelles Bank and Mascarene Plateau also represent extensive shelf areas (Kairu and Nyandwi 2000). The WIO region also harbours a variety of submerged geomorphologic features, including abyssal plains, basins, mid-ocean ridges, seamounts and ocean trenches, with some of the deep trenches ranging from 6,000 to 7,000 meters in depth.

Pleistocene coral limestone overlays older rock along much of the mainland coastline and on some of the islands, forming extensive coastal terraces, cliffs, and fringing intertidal platforms in some places (Arthurton 1992). The intertidal platforms, eroded from Pleistocene limestone cliffs, dominate the coastal geomorphology in much of the region, extending seawards generally from 100 to 2,000 meters. Their seaward edges form reef crests and offshore breaker zones. Terraces and platforms alike are incised by major creeks draining the hinterland, as at Dar es Salaam and Mombasa.

Watersheds, Hydrological Conditions and Estuaries

There are twelve main river basins within the WIO region (Fig. 2), including the largest in Madagascar. Table 1 provides an overview of the size of each river basin, the individual length of the main stem of each river, and values for mean annual precipitation (MAP) and mean annual runoff (MAR), the months of the highest and the lowest flows respectively, and the sediment load transported to each river mouth. Considerable spatial variation is exhibited for all these factors. Not shown is the fact that there also is considerable temporal variation in these variables in most of the basins, especially rainfall, runoff and sediment transport. Very few years can be considered 'average,' the norm being variability and change.

The freshwater flows from the various rivers have a profound effect on the marine ecosystems in the region, driving various ecological processes and providing nutrients for many biota (Kairu and Nyandwi 2000; Crossland et al. 2005). Rivers draining the central highlands, including the Maputo, Incomati, Limpopo, Save, Tana, Athi-Sabaki,

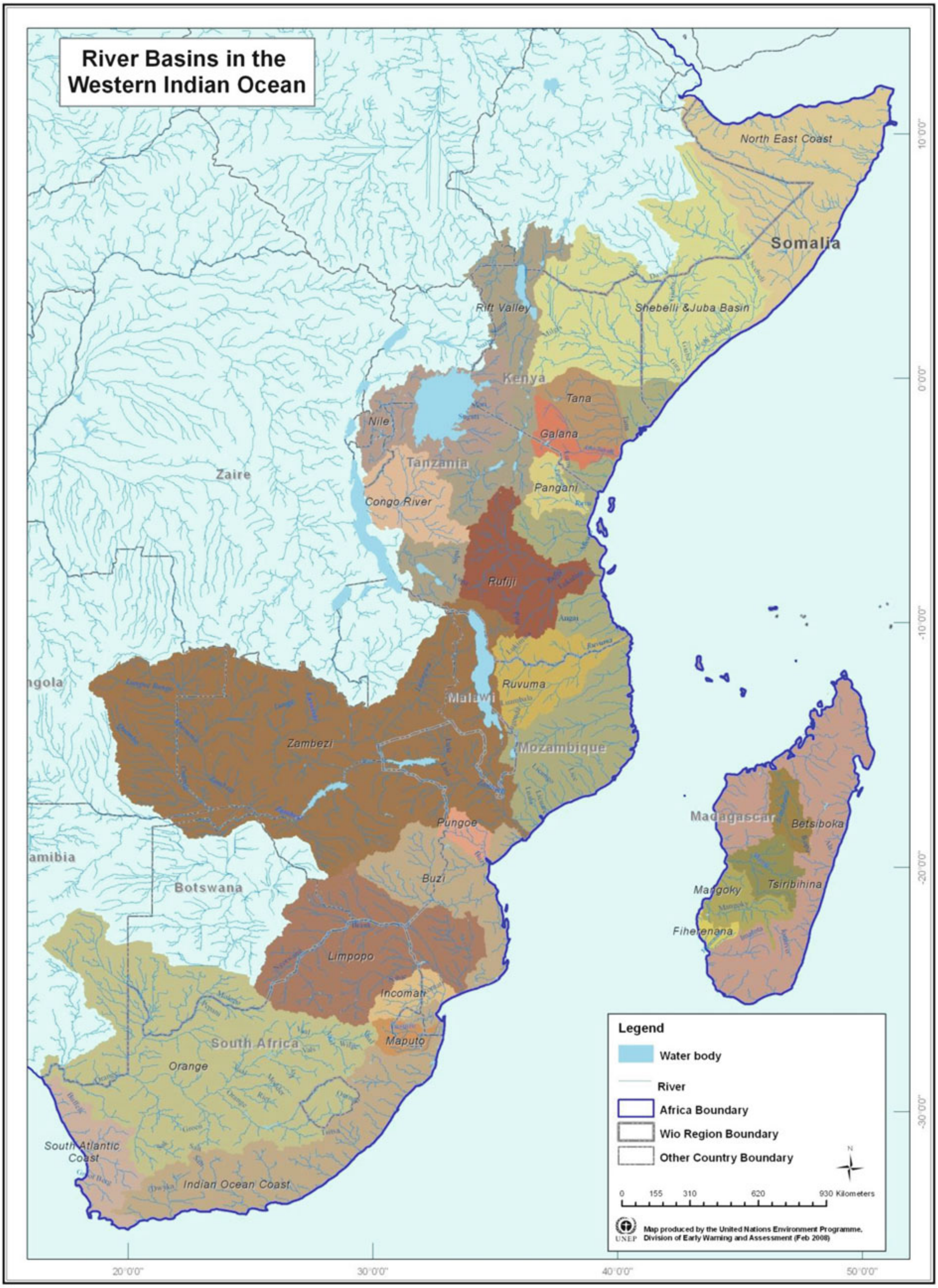

Fig. 2 Map of the main river basins in the WIO

Table 1 Overview of the main rivers in the WIO region

River	Area (km^2)	Length (km)	MAP[1] (mm)	MAR[2] (mm)	Average flow (mm^3/year)	Highest flow month	Lowest flow Month	Sediment Load (Mt/year)
Tana	126,828[a]	1,102[b]	566[a, c]	38[a]	7,200[d]	May[d]	Aug[d]	6.8[a]
Athi-Sabaki	69,930[e]	650[g]	585[a]	35[h]	2,302[a]	April[f]	Sept[i, f]	5.7[a]
	66,800[f]				1,539[f]			7.5–14.3[i, j]
Pangani	43,650[k]	432[k]	1,079[k, l]	20[m]	850[m]	May[m]	Sept[m]	No data
Rufiji	177,000	±600	1,000	No data	35,000[n] 30,000[o]	April[p, o]	Nov[p, o]	16.5[n]; 15–25[o]; 17[q]
Ruvuma	155,400[p, r]	800[p, s]	1,160[s]	96[t]	28,000[t, u]	Feb[t]	Aug[t]	No data
Zambezi	1,300,000[s]	2,650[s]	1,000[x]	67[t] 190[v]	106,000[t]	Feb[t]	Sept[t]	43[q]
	1,200,000[t]							22[v]
Pungwe	31,000[w]	395[w]	1,100[w]	115[t]	6,600[t]	Feb[w]	Oct[w]	No data
	29,500[t]							
Limpopo	415,500[y]	1,750[z]	530[v]	13[t]	5,200[t]	Feb[z]	Sep[z]	10[z]
	412,000[t]							34[q]
Incomati	46,800[ii, iii]	480[s]	736[ii]	46[t]	3,587[ii]	Feb[v]	Sep[v]	7[ii]
Maputo	28,500[t]	380[t]	630[t]	102[t]	2,900[t]	Feb[t]	Sept[t]	
Thukela	30,000 [vi]	405[viii]	840[vii]	133[vii]	3,800[vi]	Feb[vii]	Sept[vii]	9.3[vi]
					4,600[viii]			10.5[q]
Betsiboka	49,000 [ix]	525[ix]				Feb[x]	Sept[x]	

Sources: a. Kitheka et al. (2003a); b. Kitheka et al. (2003b); c. GOK (1979); d. Kitheka et al. (2004); e. Kitheka et al. (2003d); f. Fleitmann et al. (2007); g. UNEP (1998b); h. Kitheka et al. (2003c); i. van Katwijk et al. (1993); j. Watermeyer et al. (1981); k. PBWO/IUCN (2007); l. Røhr and Killingtvleit (2002); m. PBWO/IUCN (2006); n. Temple and Sundborg (1973); o. Shaghude (2004) citing Euroconsult (1980); p. Anon Tanzania (2006); q. Arthurton et al. (2002); r. GoT (2006); s. Pallet (1997); t. DNA (1994); u. Kaponda (2005); v. Hirji et al. (2002); w. Van der Zaag (2000); x. FAO (1997); y. Challenge Programme (2004); z. Louw and Gichuki (2003); ii. TPTC (2001); iii. Hoguane (2007); iv. UNEP (2005); v. Van der Zaag and Carmo Vaz (2003); vi. DWAF (2004a); vii. DWAF (2004b); viii. Forbes et al. (2002); ix. Shahin (2003); x. IWMI (2006)

[1] *MAP* mean annual precipitation

[2] *MAR* mean annual runoff

Rufiji, Zambezi and Ruvuma, discharge large volumes of siliclastic sediment to the sea (Kairu and Nyandwi 2000). In this regard, the zone of interaction between the freshwater and the saltwater ecosystems – the estuaries – is of particular significance.

According to the South African National Water Act (No. 36 of 1998), an estuary can be defined as "a partially or fully enclosed body of water, which is open to the sea permanently or periodically; and within which the sea water can be diluted, to an extent that is measurable, with freshwater drained from land" (RSA 1998). According to the DWAF (South Africa) report, "Methodology for the Determination of the Ecological Water Requirements for Estuaries," patterns of river inflow to estuaries manifest strong correlations with important hydrodynamic and sediment characteristics, including the state of the river mouth, amplitude of tidal variation, water circulation patterns and sediment deposition/erosion. The relationships between these characteristics and river inflows are generally difficult to interpret, however, because of the influence of the sea (i.e., the state of the tide and associated seawater intrusion). The manner in which these characteristics are influenced by river flow is often not the result of a single flow event, but rather that of characteristic flow patterns occurring over weeks or months. There also is a large buffer or delay-effect in estuaries between river inflow patterns and their effects on abiotic parameters (DWAF 2004c).

Many of the rivers terminate in important estuaries or deltas that serve as habitat and rich nursery and spawning grounds for various species of fish, crustaceans and other marine life. Table 2 provides an overview of the key ecological functions of the main estuaries and deltas in the WIO region, including some of their key ecological functions. Several of the estuaries in the WIO region are known to be experiencing stress due to land-based activities upstream, thereby being less able to provide the ecosystem services upon which communities depend (Arthurton et al. 2002; UNEP 2006). In addition to climatic variability and/or changes, the principal drivers of environmental change in basins in the region include agricultural development, urbanisation, deforestation, river damming and industrialisation (Crossland et al. 2005; UNEP 2006).

The small, island nations of Comoros, Seychelles, and those in the Mascarene Group (Mauritius, Réunion), have very small, usually seasonal rivers of low volumes and flow rates. These are not mentioned further in these discussions, with the countries with significant river basins being the focus of the remainder of this section.

Table 2 Summary of the features of the main estuaries and deltas of the WIO region

River	Key facts	Key ecological functions
Athi-Sabaki	The estuary at Malindi is small and narrow (0.58 km^2 and 2.5 km long), shallow, with an average depth of 2 m (Kitheka et al. 2004) and a small section is colonised by mangroves and associated plants. Accretion is associated with the deposition of high sediment load.	Habitat and nursery ground for shrimps and feeding ground for birds, (UNEP 1998a and Kitheka et al. 2004). Plays an important role in sustaining the productivity of Ungwana Bay (see above).
Betsiboka	The estuary is large, but shallow; highly deltaic and experiences significant tidal incursions during spring tide. Accretion associated with heavy deposition of sediments. Mangroves cover 420 km^2 (IWMI 2006)	Mangroves act as nursery and feeding grounds for shrimp, crab and finfish (Shahin 2003). Also, a source of building materials to the local communities.
Incomati	The estuary is of limited spatial extent, but with significant sea water intrusion. There are 5,000 ha of mangroves. Lower parts of the estuary are eroding.	Mangroves act as a nursery ground for fish and shrimp and provide building materials and charcoal to local communities.
Limpopo	Limpopo estuary is small, about 6 km in length (Louw and Gichuki 2003).	Nursery ground for fish and shrimp, provides building materials to local communities from limited mangroves (Louw and Gichuki 2003).
Maputo	Maputo Bay is 70,000 ha in extent and incorporates estuarine, mangrove and marine components (Hoguane et al. 2002).	Has a large mangrove forest and is important in terms of fisheries. It acts as a shrimp spawning ground (Hoguane et al. 2002; Arthurton et al. 2002).
Pangani	The estuary is about 3 km^2 in extent. Due to a reduction in sediment load, the estuary is eroding. There is also a large fringing mangrove forest.	Contains 753 ha of mangroves (Kijazi 2002). Also important for fishing (crabs and prawns) (PBWO/IUCN 2007).
Pungwe	The estuary is located 20 km north-west of the city of Beira (Van der Zaag 2000).	Used for aquaculture, targeting prawns; the farms also prevent saltwater intrusion into Beira's freshwater supply intake (Van der Zaag 2000).
Rufiji	Large delta area (65 km across, 23 km long and 1,200 km^2 in size) with 53,000 ha of mangroves (Richmond et al. 2002; Shaghude 2004).	Mangroves (largest estuarine forest in East Africa), fishing and aquaculture (Mwalyosi 2004; Shaghude 2004).
Ruvuma	Northern portion of 650 km^2 estuary declared a marine park – Mnazi Bay-Ruvuma Estuary Marine Park (Joint Water Commission. 2008).	Mangroves, seagrass beds, nursery ground for fish and shrimp (Francis et al. 2002; Richmond and Mohamed 2005).
Tana	Tana Delta consists of several estuaries such as Kipini (27 km^2), Mto Kilifi, Mto Moni and Mto Tana. The estuaries extend inland up to 10 km and are relatively deep with a mean depth of about 5 m. Accretion is limited and some sections of the delta are already eroding. The delta is colonized by mangroves (4,100 ha) and associated plants.	Large mangrove forests act as an important habitat and nursery ground for juvenile fish and shrimps (Munga et al. 2007). Plays an important role in sustaining the productivity of Ungwana Bay – Kenya's most productive coastal fishing ground.
Thukela	Size 0.6 km^2 during low flows (DWAF 2004b) with estimated axial length of 800 m, shore line length of 2 km, and a maximum width 350 m, with a channel width of 50 m, increasing to over 1,000 m during floods (DWAF 2004b)	Extensive areas of mud-flats interspersed with submerged aeolianite reef, providing significant fishing grounds and the only shallow water penaeid prawn trawling ground in South Africa (Forbes et al. 2002).
Zambezi	The delta is about 100 km long and 120 km wide at the coast, covering 15,000 km^2 (Pallet 1997; ZRA 1998; Chenje 2000) or 1.4 million ha (Turpie 2006).	Sustains rich offshore Sofala Bank with its fisheries, key nursery ground for fish and offshore shrimp resources (ZRA 1998; Masundire and Mackay 2002). Provides mangrove building materials to local communities.

Kenya The two river basins included in this study are the Tana and Athi-Sabaki. Both are medium-sized basins, seasonally flushed by rainfall mainly during the transitions between the Northeast and Southeast monsoons (Kitheka et al. 2004; Crossland et al. 2005). Both river basins have been subjected to diversions and changes in land use (UNEP 2006) since they originate in the highly populated and heavily cultivated Central Kenyan highlands (Kitheka et al. 2004; Dominik et al. 2007; WRI 2007). Although hydropower generated by dams in the Upper Tana Basin provides the principal source of electricity for the country (WRI 2007), dam construction has had a major influence on the river's downstream flow and physical characteristics, most notably by regulating water flows and decreasing the frequency and magnitude of flooding (IUCN 2003; UNEP 2006).

The Tana River Delta consists of four main estuaries – Kipini, Mto Kilifi, Mto Tana and Mto Moni (Kitheka et al. 2003b; Kitheka et al. 2004) – being Kenya's only major ocean delta (UNEP 1998a). The annual sediment load is currently estimated to be 6.8 x 10^6 tonnes $year^{-1}$ (Kitheka et al. 2004), which is lower than the estimated sediment load before the construction of dams in the Upper Tana Basin. The estuaries support artisanal and industrial fisheries, estimated to support around 50,000 people in 1991 (IUCN 2003). Construction of hydro-electric power (HEP) dams in the upper Tana Basin, however, has led to some changes in the river flow patterns in the lower Tana Basin. There has been a reduction in the surface area and longevity of flood-supported riverine forests, wetlands and mangrove areas, as well as the fish populations and diversity in the main river channel (Abuodha and Kairo 2001; Hoff et al. 2007; Kitheka et al. 2005). It is thought that additional dam construction will rapidly exacerbate this decline in fishing area and catch (Turpie 2006). A controversial sugar cane project that will cover several thousand hectares is being promoted by Nairobi-based business investors.

The Athi-Sabaki River comprises the second longest and the fourth largest catchment in Kenya, with an area that includes large urban centres such as Nairobi (UNEP 1998b; Kitheka et al. 2004). Urbanisation in the headwaters region has led to reduced infiltration of rainfall, causing rapid, but short-lived, high flows, and a much reduced base flow (van Katwijk et al. 1993; Snoussi et al. 2004). There are two main tributaries, namely the Tsavo and Athi rivers, which join to form the Sabaki River. The Athi River drains the lower parts of the Central Highlands of Kenya, including Nairobi, while the Tsavo receives flow from the slopes of Mount Kilimanjaro (Kitheka et al. 2004). The flow, terminating in the Indian Ocean north of Malindi, displays great seasonal and inter-annual variability (Kitheka et al. 2004). There is currently no dam on the river (Snoussi et al. 2004). Major land-use changes within the basin, however, combined with climatic variability, have already affected the river flow.

Malindi Bay is where the Athi-Sabaki River flows into the Indian Ocean, via the Sabaki estuary (Kitheka et al. 2004). This system is important in terms of the biodiversity and productivity of Malindi-Ungwana Bay, which supports both artisanal and industrial fisheries targeting prawns (UNEP 1998b). The estuary has experienced a large increase in sediment load, however, from an estimated 58,000 tonnes in the 1960s to a sediment load that ranged between 7.5 and 14.3 million tonnes in the 1980s (Watermeyer and Piesold 1981; van Katwijk et al. 1993). Recent studies by Kitheka et al. (2004) have estimated the present annual total sediment load for the Sabaki River to be 4 million tonnes. The Athi-Sabaki River also experiences a high variability in sediment load, partly governed by the rainfall patterns in the river basin. Rainfall variability affects river flows and the river capacity to transport sediments. The general increased sediment load of the Athi-Sabaki River has had a negative impact on coral reef ecosystems in Malindi Bay, particularly in Malindi Marine National Park and the Watamu Marine Reserve (van Katwijk et al. 1993; Fleitmann et al. 2007). One positive impact of the sediment accretion in the estuary, however, has been an increase in the area colonised by mangroves in the Sabaki Estuary (Kitheka et al. 2004).

Tanzania The Pangani River drains a transboundary river basin shared by Kenya and Tanzania, with the Ruvu, Weruweru, Kikuletwa, Rau and Kikafu Rivers being the main tributaries. Much of its catchment covers Tanzania, with the headwaters of the Pangani River located in the Kilimanjaro and Meru mountains (both in Tanzania), being fed by cloud-forest precipitation and snowmelt from the glaciers, respectively (PBWO/IUCN 2007; Hamerlynck et al. 2008.). The name Pangani is assumed after the confluence of Kikuletwa and Ruvu at Nyumba ya Mungu Dam. The river then successively flows across dry plains, through the extensive Kirua swamps, being finally joined by the Mkomazi at Korogwe and the Luengera before traversing the Pangani Falls before entering the Indian Ocean (Akitanda 2002).

The Pangani River flows into the Indian Ocean just south of the town of Pangani through the Pangani Estuary (PBWO/IUCN 2007). Fishing is an important activity in the estuary, particularly for crabs and prawns (PBWO/IUCN 2007). Local inhabitants also harvest aquatic plants such as reeds, sedges, mangroves and medicinal plants for household use or sale (Kijazi 2002). In the Pangani Basin Water Office (PBWO) State of the Basin Report, the estuary is described as being in a "poor condition" (PBWO/IUCN 2007). It has far fewer fish, birds and other animal species than comparable systems in Tanzania. Sea water is intruding ever further upstream as the river flow weakens because of upstream abstraction, which is eroding banks and affecting agriculture. Pollution associated with decomposing plants and fine silt from cultivated land, as introduced by the river, are resulting in oxygen levels so low for much of the year that most fish and other aquatic animals cannot survive (PBWO/IUCN 2007). A once-abundant estuarine fishery is now seriously depleted (PBWO/IUCN 2007).

The Umba River basin is another transboundary river catchment shared between Kenya and Tanzania. This river drains northeast and crosses the Tanzania-Kenya border bore entering the Indian Ocean through a huge mangrove system at Vanga in southern Kenya. The river's main catchment lies in the Usambara Mountains of northeast Tanzania. The flow of the Umba River is characterised by high seasonal variability, attributed to land-use changes (cultivation, deforestation, etc) and climatic variability within its upper catchment areas.

Tanzania's largest river basin is the Rufiji, being comprised of three sub-basins – the Great Ruaha, the Little Ruaha and the Kilombero. The Great Ruaha originates in the Paroto Mountains and Njombe highlands, where numerous rivers flow into the Usangu Plains and the vast Utengule swamps. It then traverses the Great Ruaha National Park plains, later joining the Little Ruaha before entering the Kisigo River at Mtera. It joins with the Kilombero River system after Mtera in the Kilombero Plainsm before entering the Rufiji River upstream of Stiegler's Gorge – the site of a proposed new hydropower dam (Richmond et al. 2002). The river presently is not dammed, although its resources are used via direct abstraction and fishing (Shaghude 2004). The river flows another 180 km from Stiegler's Gorge, across the floodplain and into the Rufiji Delta – flowing into the Indian Ocean opposite Mafia Island (Anon Tanzania 2006). Sediments carried by the river have resulted in accretion, causing a substantial seaward shift in the shoreline over the millennia (Mwalyosi 2004). Another important feature of the lower Rufiji floodplain is the presence of a permanent lake system comprising a total of 13 lakes (Hogan et al. 1999).

The Rufiji Delta is characterised by the presence of a huge expanse of mangrove forests (about 50 km^2) that play an important role in supporting the productivity of the coastal fisheries. The delta is formed via the splitting of the river into seven main channels, interwoven by lesser channels (Kajia 2000). The Rufiji Delta estuarine mangrove forests constitute around 46% of the total mangrove cover in Tanzania, and support an extensive marine food web (Mwalyosi 2004). The estuary serves as a nursery ground for shrimps, supporting a commercially important fishing industry, with around 80% of Tanzania's prawn catch coming from the Rufiji Delta and the area to its north (Mwalyosi 2004). Studies have shown the shrimp catch to be closely correlated with the extent of intertidal vegetation (particularly mangroves) and freshwater flows in the estuary (Mwalyosi 2004). The delta is dynamic in terms of flux – with the present trend being an increase in water flow to the northern channels and a decrease to the south (Richmond et al. 2002; Mwalyosi 2004). Dynamism in the system is driven by changes in river flows and sediment loads, via interplay between erosion and accretion. A change in this balance (e.g., through construction of a dam upstream), could have negative impacts on the delta ecosystems (Mwalyosi 2004).

Mozambique – Tanzania The Ruvuma River (Rovuma in Mozambique) forms the border between Tanzania and Mozambique for the final 650 km of its journey to the Indian Ocean (Pallet 1997). A small part (about 470 km^2) of the catchment is located in Malawi. After arising in the Matogoro Mountains in southeast Tanzania, the river flows across the Makonde Plateau before dropping to the coastal plain (DNA 1994; Anon Tanzania 2006; GoT 2006). The river basin includes ecologically important areas such as the Nyassa Nature Reserve and the Quirimbas National Park, both in northern Mozambique.

The Ruvuma estuary is famous for its beaches, mangroves and other tropical coastal marine resources. It is shared by Tanzania and Mozambique within the Mtwara Region and Cabo Delgado Province, respectively (Joint Water Commission. 2008), being a dominant feature in the region with the estuary covering 15% of the coastline. The coastline is generally comprised of a stable substrate, with deep sheltered bays exhibiting fishing and recreational potential. The Mnazi Bay-Ruvuma Estuary Marine Park (MBREMP) is on the Tanzanian side, while the Qurimbas National Park is found on the Mozambique side. There are plans to link up these two marine reserves, providing a contiguous protected habitat (Anon. Tanzania 2006). The estuary has a large mangrove forest, as well as important stretches of seagrass beds (Francis et al. 2002; Richmond and Mohamed 2005).

Mozambique The Zambezi River basin is the fourth largest in Africa after the Congo, Nile and the Niger. The river rises from the Kalene Hills in the North Western Province of Zambia, flowing south and then eastwards for some 2,650 kilometres to the Indian Ocean. The Zambezi River is an exemplary transboundary river system since it flows through nine riparian states before discharging into the Indian Ocean, namely Angola, Botswana, Democratic Republic of Congo (DRC), Malawi, Namibia, Tanzania, Zambia, Zimbabwe and Mozambique. The main stem of the river forms the southern border of Zambia with Namibia, Botswana and Zimbabwe, before flowing through Mozambique, where it discharges into the Indian Ocean. The Shire River, one of the largest tributaries of the Zambezi, drains Lake Nyassa (Malawi), which is shared by Malawi, Mozambique and Tanzania (Pallet 1997). There is no doubt that water resource developments have improved the economy of the riparian states. The construction and operation of large HEP dams, such as the Kariba and Cahora Bassa dams, however, have had downstream environmental impacts (Beilfuss 1999). Proposals for the development of these dams did not seriously consider environmental impacts (Brown and King 2002) and, as a result, adverse impacts of the dams were not effectively mitigated against. These impacts now include reduction in terrigenous sediment loads (thereby increasing downstream scouring), changes to the riverine habitats and flora due to reduced natural variability in streamflows, and destruction of estuarine habitat (Hirji et al. 2002).

The Zambezi River supports important functions in sustaining and maintaining the productivity of aquatic fauna and fisheries in the Indian Ocean directly off the Delta, through the transport of vital nutrients downstream and their discharge into the sea (ZRA 1998; Brown and King 2002). Coastal geography and the oceanic current patterns combine to restrict the availability of these nutrients to the nearshore marine environment along this stretch of the East Africa coastline (Hoguane 1997a, b). Subsequent to construction of the Cahora-Bassa dam in Mozambique, the flow regime at the delta has become much more constant – with higher low flows and lower high flows, and very few years of floods (Brown and King 2002). This has had a negative impact on fisheries, prawn catch and mangrove forests in Mozambique (ZRA 1998; Hirji et al. 2002). The delta floodplain areas have shrunk, and the reduced sediment load have led to accelerated coastal erosion and incision of the channel (Brown and King 2002). Studies carried out by Hoguane et al. (2002) indicate the northern part of the estuary, the Chinde outlet, is eroding by an average of 22 m/yr, while accretion is taking place at the southern outlet of Ponta Liberal at a rate of 58 m/yr (Anon. Mozambique 2007).

The Pungwe River originates on the Zimbabwe Highveld (Inyangani Mountain system) at an altitude of more than 1,000 meters, then traveling 395 kilometres eastwards into the Indian Ocean (van der Zaag 2000). The climate can be described as tropical savannah in the northern and eastern part of the basin, while the rest is characterised by a humid, subtropical climate (Maud 1980). The river enters the Indian Ocean through Mozambique at an estuary located about 20 kilometres northeast of the city of Beira, at Bué Maria. Although there is no up-country dam on the main stem of the river, a water pipeline transfers 22 million m^3 per year (60,273 m^3/day) of water from the Pungwe River to the Odzani catchment (a part of the Save River basin) to augment the water supply for the town of Mutare in Zimbabwe (Van der Zaag 2000).

The estuary of the Pungwe River and its discharge have critical environmental implications by restricting seawater intrusion upstream, which is crucial for Beira's freshwater supply. A flow of 10 m^3/s is considered minimal to safeguard the Beira freshwater intake (Van der Zaag 2000). The 10% low flow (i.e., the flow with a 10% chance of occurring; with a return period of 10 years) at Bué Maria has been established at 8.8 m^3/s (Zanting et al. 1994). The large amount of sediment discharged by this river minimises the effects of coastal erosion in the area (Anon. Mozambique 2007).

The Limpopo River drains parts of Botswana, South Africa and Zimbabwe, which are some of the most economically developed areas in the region, and flows into the Indian Ocean at the city of Xai-Xai in Mozambique (UNEP 2005). The river experiences high streamflow variability, characterised by flooding after intense rainfalls, and extreme low flows during periods of severe drought. This creates great hardship for rural communities that rely on rain-fed subsistence agriculture (Ashton et al. 2001). Three important tributaries join the Limpopo River in Mozambique: the Nuanedzi River in the north of the basin (rising entirely in Zimbabwe), joining the Limpopo after flowing for about 60 kilometres through Mozambique; the Changane River (rising close to the Zimbabwe border) that joins the Limpopo close to its mouth on the coast near Xai-Xai (SARDC 2003); and the Elephants River that joins the Limpopo after the Massingir Reservoir (Louw and Gichuki 2003). Wetlands comprise two important water resources in the Limpopo River basin within Mozambique, namely the swamp ecosystems downstream of the confluence of the Limpopo River with the Elephant River, and riverine floodplains extending along the Limpopo River near its approaches to its confluence with the Changane River (Brito et al. 2003). Although there are no large dams on the main stem of the river (the Massingir Dam being on the Elephants River tributary), the water resources of the basin are heavily utilised, mainly through direct abstraction (UNEP 2005). The basin supports large urban settlements such as Francistown, Gaborone, parts of Pretoria, Polokwane and Johannesburg – all contributing a heavy pollution load to the river (Earle et al. 2006).

Although the Limpopo River estuary is comparatively small, it plays an important role in supporting fisheries and providing a breeding ground for shrimp (Louw and Gichuki 2003). Sea water intrusion into the river is regularly measured up to 55 km upstream and up to 80 km during droughts (Anon. Mozambique 2007). Between 40,000 and 60,000 hectares of floodplain directly upstream of the estuary are extensively cultivated by local farmers (Louw and Gichuki 2003).

The Incomati River is shared by South Africa, Swaziland and Mozambique, with its six tributaries supporting a large variety of ecosystems, including important conservation areas such as the Kruger National Park. The Incomati's water resources are heavily exploited, and with increasing demand, they are becoming insufficient to meet water needs (Hoguane 2007). Population growth, and urban and industrial development, negatively affect the river basin, with an increasing demand for land and water (van der Zaag and Carmo Vaz 2003). The water quality has subsequently deteriorated (Hoguane 2007), and upstream impoundments and abstractions have changed the flow regime with negative effects for the estuarine ecosystem, exacerbated by mangrove harvesting for construction, charcoal and firewood (Van der Zaag and Carmo Vaz 2003). The estuary of the Incomati River extends approximately from Manhiça to where the river discharges into the Indian Ocean at Marracuene (Noble and Hemens 1978), comprising everal inter-linked habitats, including a long narrow peninsula (the Macaneta Peninsula) and a series of inter-riverine islands. This estuarine ecosystem is important for aquatic birds and Palaearctic migrants, and provides a variety of services such food security through harvesting and flood damage amelioration (TPTC 2001).

The Maputo River has two principal tributaries, namely the Usutu and the Pongola. Most of the catchment of the Usutu River lies in Swaziland, with the Pongola River being in South Africa. The river passes through the Lebombo Range within Mozambique, thereafter taking a northerly course until it discharges into Maputo Bay and the Indian Ocean. Although the Maputo River basin is small, in terms of size and discharge, it is one of the two largest rivers flowing into Maputo Bay along with the Incomati River (DNA 1994).

Maputo Bay supports one of Mozambique's most important fisheries (Arthurton et al. 2002; Anon. Mozambique 2007). The estuary and the adjacent mangrove forests serve as nursery grounds for fisheries, sustaining a considerable proportion of the local population and fishing industry, and contributing approximately 20% to the overall shrimp catch of Maputo Bay (Anon 2006). The high productivity of the Bay depends largely on the freshwater input from the Maputo River and the Incomati, as previously discussed (Hoguane 2007). This fishery is considered an environmental "hot spot," however, due to its continued degradation (Hoguane et al. 2002), which includes the estuary's mangrove forests, which have been significantly reduced by harvesting, particularly on Benguelene Island. A ban on mangrove harvesting has been enforced on the island since 1996 (TPTC 2001). Nevertheless, Maputo Bay is experiencing increasing stress from pollution emanating from agricultural return flows, industrial wastes and urban sewage (Hoguane et al. 2002).

South Africa The Thukela River catchment has notably steep gradients, extending from sea level to the Drakensberg Plateau, with peaks of over 3,000 meters within a distance of 180 kilometres from the coast (Forbes et al. 2002). The rainfall in the catchment is erratic, with years of prolonged drought in the central and lower catchment alternating with very wet periods. The main economic activities in the Thukela catchment are manufacturing industry, trade, transport and agriculture, the latter being an important source of livelihood, both in the form of commercial and subsistence agriculture (DWAF 2004a). Although most of the Thukela River system is comparatively undeveloped, there are some dams in the upper catchment and infrastructure for four inter-basin transfers of water out of the basin. The feasibility of further development through the Thukela Water Project, consisting of two large dams, has been positively assessed, although no decision regarding construction has yet been made.

Turbid conditions are frequently associated with Thukela River outflows, with extensive areas of muddy marine sediments contrasting with the rest of the South African east coast, and rendering the Thukela estuary and shelf a unique ecosystem in the South African context (Forbes et al. 2002). The offshore muddy sediments from the Thukela River mouth (the Thukela Banks) are interspersed by submerged aeolianite reef. The combination of the two habitats provides significant fishing grounds, and the only shallow water penaeid prawn trawling grounds in South Africa (Forbes et al. 2002). It is believed that extreme flood events transport large quantities of suspended sediment from the Thukela River, making extreme events significant in the sediment dynamics of the adjoining continental shelf (Forbes et al. 2002). Two sub-catchments that would be affected by the proposed dams that form part of the Thukela Water Project provide 40% of the total discharge and 25–30% of the total sediment load (Forbes et al. 2002). Although a DWAF (2004b) study suggests the sediment dynamics in the estuary would remain in a dynamic equilibrium even after the construction of the dams, this remains to be proven, especially as the mouth to the sea has closed in recent years.

Madagascar The Betsiboka River is one of the largest river systems in Madagascar, originating near Falaise de l'Angavo at 1,755 meter altitude (Shahin 2003). It is navigable for about 140 kilometres upstream, making it important for shipping and local transport. It flows to the northwest and empties into Bombetoka Bay forming a large delta. Major tributaries are the Mahajamba, Isandrano and Ikopa Rivers. A well developed floodplain containing some 150 small lakes is located in the lower course (IWMI 2006), including Amparihibe-South (12.5 km^2), Ambania (9.1 km^2), Amboromalandy (6.6 km^2) and Bondrony and Matsiabe (5.0 km^2 combined). The total lake area is 80 km^2. The Ikopa tributary basin contains large dams at Mantasoa and Tsiazompaniry.

The Betsiboka River estuary is one of the largest and most important in Madagascar, with distinctively red-coloured water caused by sediments emanating from highly eroded/degraded catchments associated with high rates of soil erosion, reaching up to 250 tonnes per hectare (IWMI 2006).

Coastal Ecosystems and Habitats

The WIO region comprises the western extremity of the tropical Indo-West Pacific, the world's largest marine biogeographic province (Ekman 1953; Sheppard 1987, 2000). The region is characterized by diverse coastal and marine ecosystems such as coral reefs, seagrass beds, mangroves, sandy beaches, sand dunes and terrestrial coastal forests.

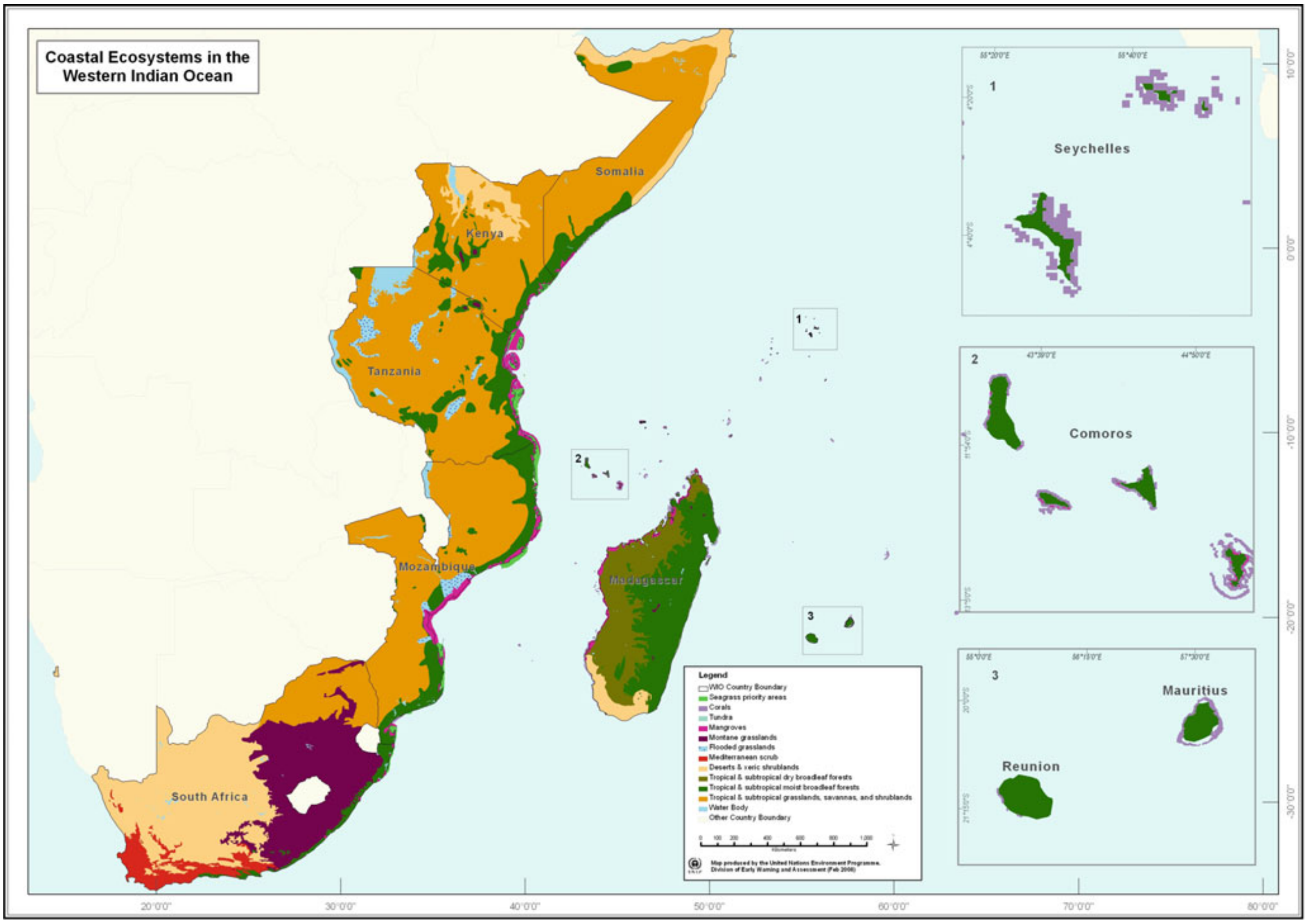

Fig. 3 Map displaying key ecosystems and habitats in the WIO region

Figure 3 provides a synopsis of the key habitats found in the WIO region, with a further description of each of the types of ecosystems presented in the following sections.

Coral Reefs

The coral reefs in the WIO cover a surface area of approximately 12,913 km^2. Most coral reefs in the region are fringing carbonate reefs found along the length of the coastline, particularly in areas with no river drainage into them. Atolls and patch reefs are common in the island states and the offshore islands along the East African continental margin. Coral reefs play an important role in the socio-economic well-being of the people in the WIO region, with many being dependent on them for work and subsistence. Coral reefs are probably the most biodiverse marine ecosystem in the WIO, exhibiting more than 300 coral species (see Table 3). While WIO coral reefs were severely affected by bleaching from elevated temperatures associated with the El Nino Southern Oscillation, it was followed by a measure of recovery (see CORDIO status reports by Souter et al. 2000; Souter and Lindén 2005). Nevertheless, this stress nevertheless must have adversely affected the coastal services provided by the coral reefs.

Table 3 Cover of coral reef and scleractinian coral species diversity by country in WIO

Country	Reef area (km^2)	Recorded no. coral species
Comoros	430	314
Kenya	630	237
Madagascar	2,230	315
Mauritius	870	161–294
Mozambique	1,860	194–314
Seychelles	1,690	206–310
Somalia	710	59–308
South Africa	~50	99
Tanzania	3,580	314

Source: Spalding et al. (2001)

Somalia Fringing reefs are well developed in the south and around the islands of the Bajuni Archipelago. North of the Archipelago, however, fossil carbonate reef structures are present, although they support reduced coral cover and biodiversity (Spalding et al. 2001). This has been attributed to the upwelling of cold water within the Somali Current during the Southeast Monsoon. Somalia has 710 km^2 of coral reef, accounting for 5.5% of the regional total. Despite this significant contribution to regional coral reef cover, very little is currently known about the state of Somalia's coral reef because of political instability, especially after the 1998 coral bleaching event in which the neighbouring countries suffered coral mortality of between 50–95% (Wilkinson 1999; Wilkinson et al. 1999).

Kenya Coral reefs cover a surface area of about 630 km^2, comprising 4.9% of the total regional reef area (Spalding et al. 2001). Better reef development is found in the fringing reefs in the southern part of the country (Spalding et al. 2001). The northern coast of Kenya contains patchy reefs within a system of barrier islands, mangroves and seagrass beds (Obura 2001). Reef development is reduced in the northern part of the Kenya coast, which contains large areas of mobile sediment and significant freshwater input from the Tana and Athi-Sabaki Rivers (Spalding et al. 2001). Fringing reefs also are found off Lamu Island and along many of the barrier islands to the north.

Tanzania This country has the largest area of coral reef in the region (3,580 km^2), with fringing reefs being found along the coast of most of mainland Tanzania and the three main offshore islands: Pemba, Unguja (Zanzibar) and Mafia (Spalding et al. 2001). In areas of high freshwater discharges, such as around the Rufiji Delta, there are breaks in the reef structure attributable to high sediment loads and reduced salinity (Spalding et al. 2001). The South Equatorial Current meets with the east African coast in southern Tanzania, rendering its reefs the most biodiverse in East Africa. Deflection of this current by the mainland further renders the Tanzanian coral reefs important in terms of connectivity with the other reef systems in East Africa, providing them with a source of larvae. Coral reefs are well-developed around the three main offshore islands, although the eastern shores of these islands have been reported to have a low cover of living hard coral, possibly attributable to high wave impacts combined with anchor damage from decades of artisanal fishing (Spalding et al. 2001).

Mozambique Coral reefs cover an area of about 1,860 km^2, comprising about 14.4% of the regional total, most being located within the provinces of Nampula and Cabo Delgado in northern Mozambique (Motta et al. 2002; Spalding et al. 2001). As a result of the remoteness of some of the northern areas, many Mozambican reefs have experienced less exploitation than elsewhere in East Africa. Though the country has the longest coastline on the east African coast, its coral reefs are not as prolific as might be expected. This is partly due to the fact that more than 24 rivers discharge into the Indian Ocean in the central part of the country (known as the 'swamp' coast), introducing sediment and reducing the salinity of coastal waters (Motta et al. 2002; Spalding et al. 2001).

South Africa South Africa's only coral reefs are found along the north coast of KwaZulu-Natal, covering a surface area of just under 50 km^2 (Schleyer and Celiers 2005a, b). The coral communities here represent the southernmost distribution of these biota on the African coast, reportedly characterized as high coral cover, alhtoughdominated by soft corals (Schleyer and Celliers 2002). The coral reefs of South Africa, however, are not true accretive reefs, since corals only grow as a thin cover over the limited sandstone substrata (Ramsay and Mason 1990; Ramsay 1996).

Seychelles Coral reefs cover a surface area of about 1,690 km^2, equivalent to 13.1% of the total coral reefs in the WIO region (Spalding et al. 2001). The granitic islands on the Seychelles Bank are surrounded by widespread, discontinuous fringing reefs (Spalding et al. 2001). The best developed reefs are found along the east coast of Mahe, and the west coast of Praslin, where the reef flats often reach 2 kilometres in width, and exhibit highly exposed, seaward algal ridges (Spalding et al. 2001). Granite reefs are common in the shallow waters around the inner islands, comprising corals growing over granitic boulders. These coral communities were shown to be better able to recover from coral bleaching events (Engelhardt et al. 2003; Payet 2005). The coralline islands to the south and southwest of the Seychelles Bank form a number of geographic groups, with the Amirantes Group being the largest and the Aldabra Groups the most distant (Spalding et al. 2001). The reefs in these outer islands are highly varied, and include true atolls, raised atolls, submerged or partially-submerged atolls and platform or bank structures with coral cover varying considerably between localities (Spalding et al. 2001). In the inner islands of the Seychelles Bank, the coral cover is very low following the 1989 El Niño event, and is taking a long time to recover (Payet 2005).

Comoros Archipelago The three islands are surrounded by fringing reefs, with the exception of the main island of Grande Comoro, an island which is still volcanically-active with steep and barren shores. Fringing reefs are restricted here to a only few parts of the coastline, mostly to the north and west (Spalding et al. 2001). Moheli Island has the most extensive reef system, with continuous fringing reefs all around the island (Spalding et al. 2001).

Madagascar Coral reefs are widely distributed along the west coast, with the following account of these reef being derived from Spalding et al. (2001), except where indicated. The reefs cover a surface area of about 2,230 km^2, equivalent to 17.3% of the regional total. Different types of reefs are present, including fringing reefs, barrier reefs, patch reefs and submerged coral banks and shoals (Cooke et al. 2000). Extensive fringing reefs off the west coast of Madagascar are located between 500 meters and a few kilometres offshore, being separated from the shore by generally shallow channels (Spalding et al. 2001). A notable feature is the well-developed barrier reef known as Grand Recif, which runs continuously for 18 kilometres. A fragmented barrier reef system is located in the area between Baie des Assassins and Morombe, and north of the Mangoky Delta. Along most of the central section of the west coast, there is no reef development, being probably attributable to terrigenous sediment discharges from rivers and large freshwater inputs. There is a series of raised banks on the outer edge of the continental shelf in the far north, forming a near-continuous ridge thought to be the remains of a large barrier reef system. The east coast has less-developed coral reefs.

Reefs on the east coast are not as well studied. Those located to the north and south of Tamatave, however, were surveyed by Schleyer and Celliers (2005a, b). Similar reef types to those found on the west coast were encountered, but appear to be severely degraded.

Mascarene Islands The Indian Ocean islands of Mauritius (including Rodrigues) and Réunion (France), located at the southern end of the Mascarene Ridge are all of volcanic origin. Réunion has the youngest reef in the region, while Mauritius is almost completely surrounded by fringing reefs with substantial lagoon and barrier reef development on the east and southwest coast (Spalding et al. 2001). In contrast, Rodrigues has a highly-developed reef structure, although a true barrier reef has not formed (Fenner et al. 2004). The outer reef slopes in both Rodrigues and Mauritius are reported to have a high coral cover (Ahamada et al. 2004). The islands of the Cargados Carajos Bank also are under the jurisdiction of Mauritius, being surrounded by large expanses of coral reefs (Spalding et al. 2001). Réunion has only a few fringing reefs, restricted to the leeward western shore, while the French-administered Iles Eparses, have reefs accounting for about 1.9% of the coral reefs in the region.

Mangrove Forests

Mangroves trees and shrubs are a common sight on sheltered areas of tropical and sub-tropical shorelines in many parts of the world. They represent valuable resources to local communities, particularly the coastal service of protecting the shoreline from erosion, and provision of building materials. From the perspective the latter, they have been subjected to over-exploitation and degradation in many areas.

Mangrove forests form fringing coastal vegetation between the spring-tide high mark and the mid-tide level. Thus, their roots are exposed to air at least during each tidal cycle. The latitudinal limits of mangroves are about 31°22′N and 38°20′S (Tomlison 1986; Spalding et al. 1997). Two main centres of mangrove diversity have been identified: the eastern group, which includes the Indo-West Pacific stretching from the central Pacific to the mainland coastline of the WIO, and the western group, which includes the mangroves of Atlantic-East Pacific, including those of West Africa, the Americas and the Caribbean Sea. The eastern group is more diverse, containing about five times the species diversity recorded in the western region (Duke 1992).

The total area of mangroves in the WIO is estimated to be 10,000 km^2 (Spalding et al. 1997), representing about 5.0% of the total global mangrove coverage. The best-developed mangroves in the region are found in the deltas of the Rufiji River (Tanzania), the Tana River (Kenya), the Zambezi and Limpopo Rivers (Mozambique), and along the west coast of Madagascar at Mahajanga, Nosy be and Hahavavy (Table 4).

Nine species of mangroves are commonly encountered in the WIO region, with the most common species being *Rhizophora mucronata* and *Ceriops tagal* (see Table 5). The most common mangrove-associated tree species occurring within the mangrove ecosystem are *Barringtonia asiatica*, *Barringtonia racemosa* and *Pemphis acidula* (Beentje and Bandeira 2007).

Table 4 Distribution of mangroves in the WIO

Country	Area (ha)[1]	Species	Main mangrove areas
Mozambique	390,500	9	Zambezi Delta
Madagascar	314,000	9	West coast at Mahajanga bay, Nosy Be, and Hahavavy
Tanzania	164,200	9	Rufiji Delta, Tanga, Kilwa, Pangani.
Kenya	51,600	9	Lamu Archipelago, Tana Delta
Seychelles	1,900	7	Aldabra Atoll
South Africa	667	6	St Lucia
Comoros	670	5	Grande Comoro, Moheli
Mauritius	106	2	Mathurin Bay, Rodrigues
Somalia	9,100	6	Juba/Shebele Estuary

Data source: [1]FAO (2005b), Beentje and Bandeira (2007), and Spalding et al. (1997)

Table 5 Species of mangroves in WIO

Family	Species
Avicenniaceae	*Avicennia marina* (Forsk.) Vierh.
Combretaceae	*Lumnitzera racemosa* Willd.
Meliaceae	*Xylocarpus granatum* König
	Xylocarpus moluccensis (Lamk.) Roem.
Rhizophoraceae	*Bruguiera gymnorrhiza* (L.) Lam.
	Ceriops tagal (Perr.) C.B. Robinson
	Rhizophora mucronata Lamk.
Sonneratiaceae	*Sonneratia alba* J. Smith
Sterculiaceae	*Heritiera littoralis* Dryand.

Sources: Macnae (1968) Semesi et al. (1999)

Table 6 Seagrasses species in WIO counties

Species	Comoros	Kenya	Madagascar	Mauritius	Mozambique	Seychelles	Somalia	South Africa	Tanzania
Zostera capensis		√			√			√	√
Thalassia hemprichii	√	√	√		√	√	√	√	√
Thallassodendron ciliatum	√	√	√	√	√	√			√
Syringodim isoetifolium	√	√		√	√	√	√		√
Halodule wrightii	√	√	√	√	√		√		√
Halodule uninervis	√	√	√	√	√	√	√		√
Halphila stipulacea	√	√	√	√	√	√	√		√
Halphila minor		√			√				√
Halophila ovalis	√	√	√	√	√		√	√	√
Enhalus acoroides		√			√	√			√
Cymodocea serrulata	√	√	√	√	√	√	√	√	√
Cymodocea rotundata	√	√	√		√	√	√	√	√
Ruppia maritime								√	

Sources: Bandeira (2000), Colloty (2000), Bandeira and Bjork (2001), Bandeira and Gell (2003), and Ochieng and Erftemeijer (2003)

Table 7 Important sites of seagrass beds in the WIO mainland (and area covered)

Country	Key location	Area (km^2)
Kenya	Gazi Bay	8.00
	Diane-Chale Lagoon	4.50
Tanzania	Chwaka Bay (Zanzibar)	100.00
Mozambique	Inhaca Island and Maputo Bay	80.00
	Mecúfi-Pemba	30.00
	Quirimbas Archipelago	45.00
South Africa	St. Lucia estuary	1.81

Seagrass Beds

Twelve seagrass species, comprising about a fifth of the world's total, occur in the WIO region (Bandeira and Bjork 2001; Gullström et al. 2002). These species are divided into three families; namely, *Zostera capensis* of the Zosteraceae; *Thalassia hepmrichii, Halophila ovalis, H. minor, H. stipulacea* and *Enhalus acoroides*, all Hydrocharitaceae, and *Cymodocea rotundata, C. serrulata, Halodule uninervis, H. wrightii, Syringodium isoetifolium* and *Thalassodendron ciliatum* of the Cymodoceaceae family. Kenya, Tanzania and Mozambique support the highest diversity of seagrasses (see Table 6). *Ruppia maritima*, which was recently defined as a seagrass (Short et al. 2001), also is a dominant species in southeastern Africa (Colloty 2000).

The principal WIO seagrass bed locations are illustrated in Table 7, while the following sections describe the main characteristics of seagrass bed distribution within the relevant countries.

Kenya Seagrass beds cover a surface are of about 33.6 km^2, with the most important sites located in the region between Lamu and Kiunga, Malindi, Mombasa, Gazi Bay (8 km^2), and Mida Creek and Diane-Chale lagoon (4.5 km^2) (Dahdouh-Guebas et al. 1999; Ochieng and Erftemeijer 2003). Twelve species of seagrass are present in Kenya. Some of the most common species are *Thalassodendon ciliatum*, *Halodule wrightii* and *Halophla minor* (see Obura 2001; Gullström et al. 2002). The pioneer seagrass association comprises *Halophila ovalis* and *Halodule wrightti*, while the climax vegetation of the intertidal zone is mainly composed of *Thallasia hemprichii* and some few macroalgae (e.g. *Halimeda opuntia*, *Gracilaria salicornia* and *G. corticata*). Monospecific patches of *Enhalus acoroides* also occur in deep water, while the lagoon is dominated by monospecific stands of *T. ciliatum* (Coppejans et al. 1992). In the north of Kenya (Kiunga Marine Reserve), eleven of the total number of species was identified in estuary, bay and reef habitats (McMahon and Waycott 2009). There has been a significant loss of seagrass along the coast, initially attributed to human-related activities, but now also to sea urchin population growth. In the Diane-Chale lagoon, for example, preliminary studies indicate that *T. ciliatum* beds experienced a loss of more than 50% of cover. These degraded sites also were found to have a density of the sea urchin *Tripneustes gratilla* of >37 individuals/m^2, while healthy sites had a density of 4 individuals/m^2 (Uku et al. 2005, 2007).

Tanzania Major seagrass areas include Pemba, Unguja and Mafia Islands (Ochieng and Erftemeijer 2001). One of the best-studied seagrass sites in Tanzania is Chwaka Bay, Unguja Island, Zanzibar (de la Torre e Castro and Ronnback 2004; Gultröm et al. 2006). Two types of seagrass habitats are found here; namely shallow beds in marine embayments, located some distance from coral reefs but adjacent to mangroves and mud flats, and shallow seagrass beds situated on the shallow continental shelf adjacent to coral reefs, but located far from mangroves and mud flats (Dorenbosch et al. 2005). Seagrasses are present in most of the tidal zone, but are more abundant in the western part of the bay. There are about 11 species, the dominant ones including *T. hemprichii*, *E. accoroides* and *T. ciliatum* (de la Torre e Castro and Ronnback 2004; Eklof et al. 2005). The island has become an important site for seaweed farming since 1990, which is reported to negatively affect seagrass beds (de la Torre e Castro and Ronnback 2004; Eklof et al. 2005).

Mozambique Seagrass beds are estimated to cover a total surface area of 439 km^2, with some 25 km^2 around Inhassoro and Bazaruto Island, 30 km^2 at Mecúfi-Pemba and 45 km^2 in the southern Quirimbas Archipelago (Bandeira and Gell 2003). The largest seagrass beds occur at Fernão Veloso, Quirimbas and Inhaca-Ponta do Ouro (Bandeira and Gell 2003). Pioneer species observed in Mozambique include *Halophila wrightii, H. ovalis* and *Cymodocea serrulata*. The first two species occur in exposed sandy areas close to the coastline (den Hartog 1970), whereas *C. serrulata* is a pioneer species in silted channels (Bandeira 2002). Seagrasses abound in the muddy and biogenic (calcareous) sediments of Mozambique, with the three dominant mixed-seagrass communities on the sandy substrata of southern Mozambique being comprised of *Thalassia hemprichii*, *Halodule wrightii, Zostera capensis, Thalassodendron ciliatum* and *C. serrulata* (Bandeira 1995). In contrast, the seagrass communities of the more northerly limestone areas are quite different, with seagrasses tending to occur intermingled with seaweeds species (Bandeira and António 1996). Macroalgae such as *Gracilaria salicornia*, *Halimeda* spp. and *Laurencia papillosa* occur mixed with *T. hemprichii*, and *Sargassum* spp. with *T. ciliatum* (Bandeira and António 1996; Bandeira 2000). Elsewhere, *Zostera capensis* and *Haludule wrightii* also form mixed beds (Bandeira 2000; Bandeira and Bjork 2001; Massingue and Bandeira 2005). *Enhalus acoroides, Halophila stipulacea* and *H. minor* were found only in northern Mozambique.

South Africa Seagrass beds, found only along the east coast of South Africa, cover about 7 km^2, with the largest concentration in the St. Lucia and Richards Bay estuaries. (Colloty 2000). Periodic droughts, however, severely impact both the extent and the species composition of the seagrass systems in South Africa. The species *Zostera capensis* is the most widespread and dominant of the seagrasses in the country, occurring mostly in estuaries from KwaZulu-Natal to the Western Cape. Other important locations containing seagrass species are the rocky promontories of KwaZulu-Natal, mostly dominated by *Thalassodendron ciliatum*, adapted to the rocky habitat, together with seaweeds (Barnabas 1991). *Ruppia maritima* is another dominant seagrass species occurring within estuaries, especially St Lucia (Short and Coles 2003).

Seychelles A total of eight seagrass species occur in the Seychelles Archipelago, mixed with more than 300 species of macroalgae. Although the total area covered is not known, *Cymodocea serrulata, Syringodium isoetifolium* and *Thalassia hemprichii* generally dominate the soft bottoms (Ingram and Dawson 2001). The clear waters of Seychelles have supported the deepest known seagrass distribution within WIO, with *Thalasodendron ciliatum* occurring down to a 33 meter depth (Titlyanov et al. 1995). The structure of the Aldabra Atoll differs considerably from some of the other island groups, since its coasts are built primarily of dead consolidated corals, and are steeply undercut with overhangs. Four seagrass species (*T. ciliatum*, *T. hemprichii*, *H. uninervis* and *S. isoetifolium*) and 119 algal species occur both on the reef slope and in the lagoon itself Some of the common algae genera in the atoll are *Halimeda, Turbinaria* and *Laurencia* (Kalugina-Gutnik et al. 1992). Mahé Island supports the highest recorded seagrass diversity in the archipelago (seven species), with *C. rotundata* inhabiting a narrow band along the shore, which is then replaced by *T. ciliatum* occupying the entire area exposed at low waters. Kalugina-Gutnik et al. (1992) report that algae may be associated with seagrasses, including *Sargassum cristaefolium*, *Gracilaria crassa*, *Cheilosporum spectabile*, *Jania* spp., *Hypnea* spp., *Laurencia parvipapillata, Amphiroa foliacea, Cladophoropsis sundanensis* and *Gelidiella acerosa*.

Comoros These islands, being located less than 400 kilometres east of the coastline of Mozambique, and sharing a similar climate, are likely to support seagrass meadows similar to those of northern Mozambique, with mixed seagrass species in intertidal areas, and sub-tidal seagrass species dominated by broad-leafed species such as *Thalassodendron ciliatum* (see Bandeira and Gell 2003).

Madagascar Little is known about the relative dominance of seagrass species, although it is likely that the species in southwest Madagascar are similar to those found in Mozambique. Most meadows are dominated by *Thalassodendron ciliatum* and *Thalassia hemprichii* (Bandeira and Gell 2003). Seaweeds also are a common feature in intertidal and sub-tidal seagrass areas of Madagascar (Rabesandratana 1996).

Mascarene Islands Seagrass beds cover a surface area of 5.5 km^2 and 64.9 km^2, respectively, around Mauritius and Rodrigues islands (Turner and Klaus 2005). The most abundant species in Mauritian lagoons is *Syringodium* *isoetifolium*, with other species present being *Thalassodendron ciliatum*, *Halophila ovalis*, *H. stipulacea*, *Halodule uninervis* and *Cymodocea serrulata* (Montaggioni and Faure 1980; Database of Marine Organisms of Mauritius 2007). Seagrass beds are found both as extensive beds of mixed species and monospecific stands, constituting natural habitats for a diverse group of organisms in these lagoons.

Fisheries Ecology

Only about 5% of the sea in the WIO is shallower than 200 meters, this being the zone of the seabed known as the continental shelf. The central coast of Tanzania (with the continental islands of Mafia and Unguja (Zanzibar)), the southcentral coastal section of Mozambique, including the Sofala Bank, parts of the west coast of Madagascar and the Seychelles Plateau that surround the main granitic islands of Mahe, Praslin and La Digue are the only areas with considerable expanses of continental shelf (Spalding et al. 2001). The remaining 95% of the region's ocean is deep, mostly between 500 and 4,000 meters. This deep water surrounds the oceanic islands of Mauritius, La Réunion, the east coast of Madagascar, all the Seychelles islands (excluding those on the Seychelles Plateau), most of the KwaZulu-Natal coast of South Africa, the northern coast of Mozambique extending into central Tanzania, and almost the entire coast of Kenya and Somalia. All are fringed by a very narrow continental shelf rarely wider than a few hundred meters.

Despite this relatively small continental shelf, the coastal zone of the WIO contains numerous marine ecosystems of vital importance to the productivity of the region, particularly its fisheries. Mangroves are located at the estuaries of most rivers, and seagrass beds flourish in shallow, soft bottom environments where the sun's rays are able to penetrate into the water column (Okemwa and Wakabi 1993; Gove 1995; Sheppard 2000). Coral reefs have developed on rocky fringes and hard surfaces in clear, well-oxygenated waters, notably along sediment-free coastlines, forming hundreds of kilometres of fringing reefs along the edge of the mainland and around the oceanic islands (Okemwa and Wakabi 1993; Gove 1995; Sheppard 2000). All these ecosystems, and the waters they are bathed in, support plankton communities that, in turn, feed diverse and numerous fish populations (Sheppard 2000).

Estimates vary, but the consensus based on relatively few studies estimates the overall fish diversity in the WIO region includes some 2,200 species, about 15% of the global total of marine fishes, with few oceans sharing similar ichthyofaunal richness (Smith and Heemstra 1986). This richness is due to the large variety of the region's habitats and oceanographic conditions (van der Elst et al. 2005). There also are zones of high endemism within the region, as well as unique groups of fish (van der Elst et al. 2005).

Of the 2,200 species of fish recorded so far in the region, plus the countless crustaceans, molluscs and echinoderms, several hundred are important to the five main fishery sectors; namely, those associated with coral reefs, the shrimp fisheries, the small pelagic species fishery, the larger pelagic species fishery, and the deepwater demersal fishery. Whether a fishery resource is exploited or not, and to what extent, depends on many factors, including shifts in demands for the resource or species, increasing fuel prices, and the technology available to exploit the resource. With this perspective, the basic ecological features of the five main WIO fisheries are described in the following section.

Coral Reef Fisheries The WIO coral reefs support thousands of species of animals and plants. Two main resources are of particular interest to humans along these shores, including fish and edible marine invertebrates (octopus, squid, lobsters, crabs, sea cucumbers) (Rajonson 1993; Semesi and Ngoile 1993). The collection or gleaning of marine invertebrates from reef crests, in shallow lagoons and seagrass areas is typically conducted on foot, especially during low tide (Darwall and Guard 2000; Horril et al. 2000). The majority of species taken are relatively immobile or sessile. Although harvesting may involve all members of a community, it tends to be the preserve of women and children (Darwall and Guard 2000; Horril et al. 2000). A variety of methods are employed, including simple hand collection of shellfish (lobsters and crabs) and sea cucumbers in baskets or small nets, the use of natural or synthetic poisons in pools or, in deeper water, the capture of lobster and octopus using spear guns and, in some places, SCUBA diving gear (Horril et al. 2000; Semesi and Ngoile 1993).

Most of the collected animals are consumed locally, although some are marketed internationally (Rajonson 1993). Sea cucumbers (bêche-de-mer) particularly are exported after preparation, mainly to markets in southeast Asia (Darwall and Guard 2000; Gabrie et al. 2000; Rajonson 1993; Semesi and Ngoile 1993). Many gastropods (marine snails) are eaten, although larger, more attractive species are specifically collected for their shells to be sold to tourists or exporters (Gabrie et al. 2000; Rajonson 1993; Semesi and Ngoile 1993).

The capture of fish on coral reefs involves the use of a variety of traditional and modern gear, targeting hundreds of species of fish, but mostly of small to medium-size (10–30 cm) belonging to the families Acanthuridae, Labridae, Lethrinidae, Lutjanidae, Mullidae, Scaridae, Serranidae and Siganidae (Darwall and Guard 2000; LaRoche and Ramanarivo 1995; Shah 1993). Many are caught using baited fish traps, with nets being important for the larger species fisheries (Darwall and Guard 2000). Goatfish (Mullidae) are caught in gill-nets set in lagoons and over seagrass beds, and also often actively corralled into the net by the fishers. Parrotfish (Scaridae) and some surgeonfish (Acanthuridae) also are caught in nets as they cross the reef crest at high tide to feed (Darwall and Guard 2000). Many species, particularly highly-valued groupers (Serranidae) and snappers (Lutjanidae), are fished using baited hook and line set from dugout sailing canoes, small dhows or other vessels at anchor on or around reefs (Darwall and Guard 2000).

Penaeid Shrimp Fisheries In WIO countries endowed with large river mouths and mangrove forests, nutrient-rich sediments support a productive soft sediment ecosystem dominated by benthic-feeding penaeid shrimps. These, in turn, support a coastal industrial fishery, notably in the estuarine fishing grounds of the Malindi-Ungwana Bay in Kenya, Rufiji and Wami Rivers in Tanzania, the Sofala Bank and Maputo Bay in Mozambique, the Thukela Banks in South Africa (Krantz et al. 1989; Nhwani 1988; Okemwa and Wakabi 1993; Turpie and Lamberth 2005), and off the west coast of Madagascar (UNEP/GPA and WIOMSA 2004). Estuarine penaeid shrimp fisheries are a particularly important contribution to the economy of Mozambique (UNEP 2006).

Mangroves and estuaries are a vital element in the life cycle of the shrimps, acting as a nursery ground for the juveniles (Crona and Rönnbäck 2005). The shrimp fisheries depend on the rivers supplying the estuaries with freshwater and nutrients brought to the coast from the hinterland. They extend hundreds of kilometres inland and drain vast areas in some cases (e.g., Rufiji and Zambesi rivers).

Fishing for penaeid shrimps is undertaken by a wide range of fishers and their gear, ranging from large industrial and smaller commercial trawlers (15–30 m in length) to artisanal fishermen using seine-nets, traps and scoop-nets, operating on foot or from small canoes (De Young 2006; Fennessy and Groeneveld 1997; FAO 2003). Many of the industrial vessels in the region are foreign-owned and operate under license. The catch is dominated by *Penaeus indicus, Penaeus monodon* and *Metapenaeus monoceros*, although there also is a significant finfish bycatch (Fennessy 1993; Fennessy 1994; Sookocheff and Muir 2006; Fennessy et al. 2008). Estimates from Tanzania and Kenya's Ungwana Bay suggest the weight of bycatch is nearly twice that of the shrimp, with the majority of the valuable finfish bycatch (commonly snappers, groupers, emperors and jacks) landed and not wasted (Anon 2006; Fennessy et al. 2003). However, since the by-catch also includes juveniles that inhabit the highly-productive estuarine areas, prawn fishing may threaten recruitment to finfish stocks (Fennessy 1994). Other fauna such as turtles also form a threatened bycatch, although their capture is being alleviated with the use of bycatch exclusion devices (Muir 2006).

Deepwater Demersal Fishery Beyond the depth of coral reefs, on the wide expanses of the Seychelles Plateau, the Saya de Malha Bank (Mauritius) and at the edge of the continental shelves of mainland countries and western Madagascar, a community of demersal fish are targeted in a deepwater fishery at depths from 100–200 m (De Sousa 1987; Samboo and Mauree 1987; Silva and Sousa 1987; Lablache et al. 1987). Various sharks (e.g., Charcharhinidae), groupers (Serranidae), jacks (Carangidae), and deep-water snappers (Lutjanidae)) live on the edges of the continental shelf and slopes, moving to and from shallower depths to feed and rest (Fischer and Bianchi 1984). The crimson jobfish (*Pristipomoides filamentous*) is a deepwater snapper and comprises the main target of a trial demersal fishery at Saya de Malha where the potential sustainable yield is estimated to be 567 kg/km^2 per annum (Grandcourt 2003). Although not well researched, the growth (and recruitment) rates are thought to be slow in members of this fish community, and it is widely agreed that fishing pressure may rapidly over-exploit this resource. Nevertheless, this deepwater resource in the Seychelles is an important contribution to the artisanal fishery (Robinson and Shroff 2004). Setting of gill-nets for such deepwater species in Tanzania yields sharks and other species, as well as a bycatch of coelacanth, which seriously threatens the local population of this rare species (Ribbink and Roberts 2006).

Small Pelagic Fish and Baitfish Fishery In the same manner that inshore waters along the continental margins of the WIO and the coast of Madagascar benefit from nutrients and sediments that nurture the above-noted mangrove-based shrimp fishery, large populations of small pelagic species feed on the rich plankton in the water column (Sheppard 2000). Thus, the same waters off estuaries that support the industrial shrimp fisheries yield schools of herrings, sardines and shads (Clupeidae), anchovies (Engraulidae), pony-fish (Leiognathidae), scads (Carangidae) and larger mackerel (Scombridae) (Fischer and Bianchi 1984). Cast-nets, beach-seines, gill-nets, scoop-nets and traditional and commercial purse-seines are used to land large quantities of these small pelagic species (De Sousa 1987; Ralison 1987; Silva and Sousa 1987; Lablache et al. 1987; Jiddawi and Öhman 2002). Purse-seine fishing in Tanzania is undertaken at night with lights to attract and aggregate schools of fish, much of which is consumed locally, fresh or dried (Jiddawi and Öhman 2002).

Large Pelagic Fishery The oceanic waters of the WIO region are the territory of migratory and shoaling schools of various species of tuna (notably yellowfin, skipjack, bigeye) of the family Scombridae, as well as other oceanic species, including dorado (Corphyaenidae), sailfish and marlin (Istiophoridae), and sharks (Fischer and Bianchi 1984). These fish are referred to collectively as 'large pelagic species,' restricted to warm water, with many known to migrate thousands of kilometres. They occur from the water surface to a depth of several hundred meters (Fischer and Bianchi 1984). Juvenile tuna venture inshore at certain times of the year to feed on small pelagic species (see above), while billfish roam the oceans feeding on pelagic squid, flying fish and juvenile tunas (Fischer and Bianchi 1984).

Stocks of tuna form the basis of two distinct fisheries, including a local, small-scale inshore fishery and an international, large-scale, highly mechanized industrial fishery (Kambona and Marashi 1996). There are two forms of the latter, including long-liners that deploy kilometres of baited hooks, and purse-seiners capable of rapidly setting 200 meter-deep nets around moving schools (Miyake et al. 2004). The countries with the largest EEZs, such as Seychelles and Mauritius, benefit the greatest from the pelagic fishery, while smaller mainland Africa countries like Kenya see little profit from this resource (De Young 2006).

Socio-Economic Values of the Coastal and Marine Environment

The coastal and marine ecosystems also provide essential livelihood and income sources for numerous coastal inhabitants, contributing to the growing economies of countries in the region. With a combined coastline exceeding 15,000 km (including those of the island states), and a total continental shelf area of about 450,000 km^2, (GEO Data Portal 2003), the economic value of the goods and services provided by the coastal and marine environment in the WIO region is enormous, with current conservative estimates being more than 25 billion US dollars annually (UNEP-Nairobi Convention Secretariat 2009; UNEP-Nairobi Convention Secretariat and WIOMSA 2015). The ecosystem service value of coral reefs in the WIO region alone is estimated to be greater than 7 billion US dollars per year, while that of mangroves is close to 9 billion US dollars per year.[1] It is estimated that the direct benefits obtained from coastal goods and services in South Africa, the largest economy in the region, are equivalent to about 35% of the country's gross domestic product (Department of Environmental Affairs and Tourism 2000).

Tourism is the largest source of income directly linked to the coastal and marine environment (Fig. 4). The region's

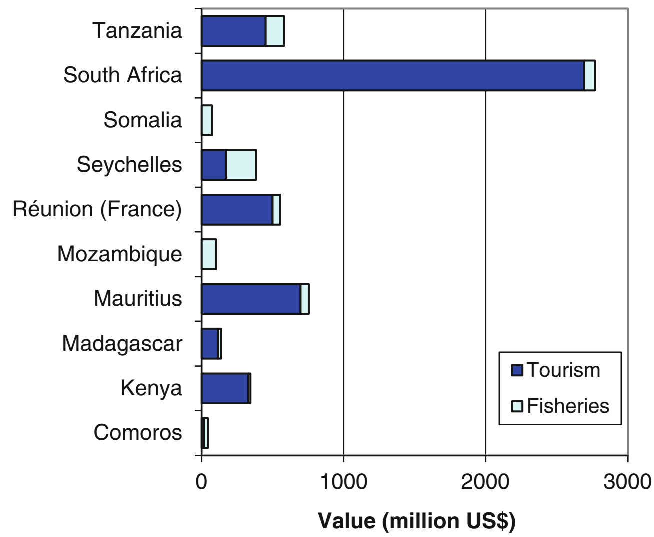

Fig. 4 Direct use values represented by the fishery and tourism sectors in the WIO region

beautiful sandy beaches, mangrove forests, lagoons and coral reefs attract over 20 million tourists from all over the world every year, injecting more than 6 billion US dollars per year into the economies of the WIO region countries.

The coastal and marine waters of the WIO, particularly its coastal waters, lagoons, estuaries and continental shelves, are important fishing grounds. The WIO region generates about 4.8% of the global fish catch, equivalent to about 4.5 million tonnes of fish per year (FAO 2007). Even this value is likely to be an underestimate because of the under-reporting of catches by some of the countries (Van der Elst et al. 2005). While not as productive as some other well-known fishing grounds in the world (particularly those associated with upwelling systems), the WIO fisheries sector is still of high importance in terms of food security, employment, and income generation for the growing coastal population, providing food and livelihoods for some 61 million coastal inhabitants. Furthermore, mangroves, seagrass meadows and coral reefs provide coastal protection, as well as food and shelter for fishes, crustaceans, molluscs and other organisms of immense ecological and commercial value.

References

Abuodha PAW, Kairo JG (2001) Human-induced stresses on mangrove swamps along the Kenyan coast. Hydrobiologia 458:255–265

AFRISCO (1994) Lake Chala water resources development project (Phase one). Project evaluation report to Coast Development Authority, Kenya. Afrisco Ltd., Israel

Ahamada S, Bijoux J, Bigot L, Cauvin B, Kooonjul M, Jean Maharavo J, Meunier S, Moine-Picard M, Quod J-P, Pierre-Louis R (2004) Status of the coral reefs of the South West Indian Ocean Island States. In: Wilkinson C (ed) Status of coral reefs of the world, vol 1. AIMS, Townsville, pp 189–211

[1] It is noted that these initial estimates originate from a basic economic assessment undertaken as part of the WIO-LaB Transboundary Diagnostic Analysis (see section 1.5), and it is believed that the actual value of the WIO marine and coastal environment may be significantly higher, as substantiated by recent estimates of the value of South Africa's coastal and marine environment, which exceeds 30 billion US$ annually (Department of Environmental Affairs and Tourism 2000).

Akitanda P (2002) South Kilimanjaro catchment forestry management strategies – perspectives and constraints for integrated water resources management in the Pangani River Basin. Paper presented at the international conference on Integrated River Basin management, Pangani River Basin, Moshi, Tanzania, 13–14 June 2001

Alemaw BF, Chaoko TR (2006) The 1950–1988 warm ENSO events and regional implications to river flow variability in Southern Africa. Water SA 32(4):459–464

Anon, Tanzania (2006) Prawn fishery performance for the year 2006. Ministry of Natural Resources and Tourism, Fisheries Division, Dar es Salaam

Anon, Mozambique (2007) Draft national status report on land-based activities, sources of pollution and pollutant levels in water and sediment: Mozambique. Report submitted to UNEP/WIO-LaB Project/Nairobi Convention Secretariat, Nairobi, Kenya, 24 pp

Arthurton R (1992) Beach erosion: case studies on the East African coast. In: Proceedings of Bordeaux, international convention on rational use of the coastal zone. IOC, Bordeaux, 91–95

Arthurton RS, Kremer HH, Odada E, Salomons W, Marshall Crossland JI (2002) African basins: LOICZ global change assessment and synthesis of rivers catchment – coastal sea interactions and human dimensions. LOICZ reports and studies no. 25. LOICZ, Texel, p 344

Ashton PJ, Love D, Mahachi H, Dirks PHGM (2001) An overview of the impact of mining and mineral processing operations on water resources and water quality in the Zambezi, Limpopo and Olifants catchments in southern Africa. Contract report to the mining, minerals and sustainable development (southern Africa) project, by CSIR-Environmentek, Pretoria, South Africa and Geology Department, Uni. of Zimbabwe, Harare, Zimbabwe. Report no. ENV-P-C 2001-042I xvi + 336 pp

Bandeira SO (1995) Marine botanical communities in southern Mozambique: seagrass and seaweed diversity and conservation. Ambio 24:506–509

Bandeira SO (2000) Diversity and ecology of seagrasses in Mozambique: emphasis on *Thalassodendron ciliatum* structure, dynamics, nutrients and genetic variability. PhD thesis, Göteborg University, Sweden

Bandeira SO (2002) Diversity and distribution of seagrasses around Inhaca Island Southern Mozambique. S Afr J Bot 68:191–198

Bandeira SO, António CM (1996) The intertidal distribution of seagrasses and seaweeds at Mecúfi Bay, northern Mozambique. In: Kuo J, Phillips RC, Walker DI, Kirjman H (eds) Seagrass biology: proceedings of an international workshop. University of Western Australia, Nedlands, pp 15–20

Bandeira SO, Bjork M (2001) Seagrass research in eastern Africa region: emphasis to diversity, ecology and ecophysiology. S Afr J Bot 67:420–425

Bandeira SO, Gell F (2003) The seagrasses of Mozambique and Southeastern Africa. In: Short F, Green E (eds) Seagrass atlas of the world. World Conservation Monitoring Centre. University of California press, 93–100 pp

Barnabas AD (1991) *Thalassodendron ciliatum* (Forsk.) den Hartog: root structure and histochemistry in relation to apoplastic transport. Aquat Bot 40:129–143

Beckley LE (1998) The Agulhas current ecosystem with particular reference to dispersal of fish larvae. In: Sherman K, Okemwa EN, Ntiba MJ (eds) Large marine ecosystems of the Indian Ocean: assessment sustainability, and management. Blackwell Science, London, pp 255–276

Beentje H, Bandeira SO (2007) Field guide to the mangrove trees of Africa and Madagascar. Royal Botanic Gardens, Kew, p 91

Beilfuss R (1999) Can this river be saved? rethinking Cahorra-Bassa could make a difference for dam-battered Zambezi. World Rivers Rev 14(1):8–11

Brown C, King J (2002) Environmental flows: requirements and assessments. In: Hirji R, Johnson P, Maro P, Chiuta TM (eds) Environmental sustainability in water resources management in Southern Africa. SADC, IUCN, SARDC, World Bank, Maseru/Harare/Washington, DC

Brito R, Munguambe P, Massingue F (2003) Wetlands of Mozambique – country report. Presented at IUCN and IWMI workshop in Maputo, Mozambique 2003

Carbone F, Accordi G (2000) The Indian Ocean coast of Somalia. Mar Pollut Bull 41(1–6):141–159

Challenge Programme for Water and Food Annual Report (2004) Limpopo Basin profile, IWMI, Colombo

Chenje M (ed) (2000) State of the environment Zambezi Basin 2000. SADC/IUCN/ZRA/SARDC, Maseru/Lusaka/Harare

Colloty BM (2000) Botanical importance of estuaries of the former Ciskei/Transkei region. PhD thesis, Department of Botany, University of Port Elizabeth

Cooke AJ, Ratomahenina O, Ranaivoson E, Razafindrainibe H (2000) Madagascar. In: Sheppard C (ed) Seas at the Millennium: an environmental evaluation, Ch. 2 60:1, 13–131

Coppejans E, Beeckman H, de Wit M (1992) Status, primary productivity and taxonomy of mangroves, seagrasses and algae. Hydrobiologia 247:59–75

Crona BI, Ronnback P (2005) Use of replanted mangroves as nursery grounds by shrimp communities in Gazi Bay, Kenya. Est Coast Shelf Sci 65:535–544

Crossland CJ, Baird D, Ducrotoy J-P, Lindeboom H (2005) The coastal zone – a domain of global interactions. Chap. 2. In: Crossland CJ, Krenner HH, Lindeboom HI, Marshall Crossland JI, Le Tisser MDA (eds) Coastal fluxes in the Anthropocene – the land-ocean interactions in the coastal zone. Project of the International Geosphere-Biosphere Programme Series. Springer, Berlin

Dahdouh-Guebas F, Coppejans E, van Speybroeck D (1999) Remote sensing and zonation of seagrasses and algae along the Kenyan coast. Hydrobiologia 400:63–73

Darwall WRT, Guard M (2000) Southern Tanzania. In: McClanahan TR, Sheppard. CR. Database

Database of Marine Organisms of Mauritius (2007). Mauritius Oceanography Institute [online]. http://moi.gov.mu/marinedb

de la Torre e Castro M, Ronnback P (2004) Links between humans and seagrasses – an example from tropical east Africa. Ocean Coast Manag 47:361–387

De Sousa T (1987) Summary of fisheries and resources information for Kenya. In: Sanders MJ, Sparre P, Venema SC (eds) Proceeding of the workshop on the assessment of the fishery resources in the southwest Indian Ocean, 14–25 September 1987, Albion. FAO/UNDP.RAF/79/065/WP/41/88/E. FAO, Rome, 277 pp

De Young C (ed) (2006) Review of the state of the world marine capture fisheries management: Indian Ocean, FAO fisheries technical paper 488. FAO, Rome, p 485

Den Hartog C (1970) The seagrasses of the world. North Holland Publication Co, Amsterdam, p 275

Department of Environmental Affairs and Tourism (DEAT) (2000) White paper for sustainable development in South Africa

DNA (1994) Actual situation of water resources management in Mozambique (Final Draft). Ministério da Construção e Águas, Maputo – Moçambique

Dominik F, Dunbar RB, McCulloch M, Mudelsee M, Vuille M, McClanahan TR, Cole JE, Eggins S (2007) East African soil erosion recorded in a 300 year old coral colony from Kenya. Geophys Res Lett 34:L04401. doi:10.1029/2006GL028525

Dorenbosch M, Groll MG, Christianen MJ, Nagelkerken I, Velde G (2005) Indo-Pacific seagrass beds contribute to fish density and diversity on adjacent coral reefs. Mar Ecol Prog Ser 302:63–76

Duke NC (1992) Mangrove floristics and biogeography. In: Robertson AI, Alongi DM (eds) Tropical mangrove ecosystems. American Geophysical Union, Washington, DC, pp 63–100

DWAF (2004a) Thukela water project decision support phase: reserve determination model. Department of Water Affairs and Forestry, Pretoria

DWAF (2004b) Internal strategic perspective: Thukela water management area – version 1. Department of Water Affairs and Forestry, Pretoria

DWAF (2004c) Methodology for the determination of the ecological water requirements for estuaries. Department of Water Affairs and Forestry, Pretoria, p 171

Earle A, Goldin J, Machiridza R, Malzbender D, Manzungu E, Mpho T (2006) Indigenous and institutional profile: Limpopo River Basin, Working paper no 112. International Water Management Institute, Colombo

Eklof JS, de la Torre Castro M, Adelskold L, Jiddawi NS, Kautsky S (2005) Differences in Macrofauna and seagrass assemblages in seagrass bed with and without seaweed farms. Estuar Coast Shelf Sci 63:385–396

Ekman S (1953) Zoogeography of the sea. Sidgwick and Jackson, London, p 417

Engelhardt U, Russell M, Wendling B (2003) Coral communities around the Seychelles Islands 1998–2002. In: Coral Reef Degradation in the Indian Ocean (CORDIO) – Stat Rep 2002. CORDIO: 212 pp

Euroconsult (1980) Identification study on the ecsological impacts of the Stiegler's Gorge power and flood control project, Part I, Summary, background and approach to the study

FAO (1997) Madagascar: FAO/WFP mission to assess the impact of cyclone Gretelle on food and agriculture. FAO/GIEWS special report on Madagascar 04/97

FAO (2001) A synthesis of flood warning and control for the Juba and Shabelle Rivers, Working paper FAO SOM/SSF/2001/12. FAO, Nairobi

FAO (2003) Fishery statistics: capture production. Volume 92/1, Year 2001, FAO

FAO (2005a) Irrigation in Africa. Available at: http://www.fao.org/docrep/w4347e/w4347ey.htm

FAO (2005b) Status and trends in mangrove area extend worldwide, Working paper no. 64. Forest resource division. FAO, Rome, p 89

FAO (2007) Fisheries statistics – capture production, vol 100/1, Fisheries series. FAO, Rome, p 539

Fenner D, Clark TH, Turner JR, Chapman B (2004) A checklist of the corals of the island state of Rodrigues, Mauritius. J Nat Hist 38:3091–3102

Fennessy ST (1993) An investigation of the ichthyofaunal bycatch of the Tugela Bank prawn trawlers. MSc thesis, University of Natal, Durban, 104 pp

Fennessy ST (1994) The impact of commercial prawn trawlers on linefish off the north coast of Natal, South Africa. S Afr J Mar Sci 14:287–296

Fennessy ST, Groeneveld JC (1997) A review of the offshore trawl fishery for crustaceans on the east coast of South Africa. Fish Manag Ecol 4:135–147

Fennessy ST, Mwatha GK, Thiele W (2003) Report of the regional workshop on approaches to reducing shrimp trawl bycatch in the Western Indian Ocean, 13–15 April 2003, Mombasa. FAO Fisheries Report 734. FAO, Rome, 49 pp

Fennessy ST, Vincent X, Budeba Y, Mueni EM, Gove DZ (2008) An update on initiatives to reduce prawn trawl bycatch in the West Indian Ocean. West Indian Ocean J Mar Sci 7(2):217–222

Fischer W, Bianchi G (1984) Species identification sheets for fishery purposes: West Indian Ocean, vol I–VI. FAO, Rome

Fleitmann DRB, Dunbar M, McCulloch M, Mudelsee M, Vuille M, McClanahan TR, Cole JE, Eggins S (2007) East African soil erosion recorded in a 300 year old coral colony from Kenya. Geophys Res Lett 34, L04401. doi:10.1029/2006GL028525

Forbes AT, Demetriades NT, Bosman C, Leuci R, Sinclair D, Smith AM, Fennessy S, Robertson WD, Sink K, Kruger A, Everett BI, Fielding PJ, Celliers L (2002) Impacts of proposed dams on the Thukela estuary and inshore marine environment, Kwa-Zulu-Natal, South Africa. In: Arthurton RS, Kremer HH, Odada E, Salomons W, Marshall Crossland JI (eds) African basins: LOICZ global change assessment and synthesis of river catchment-coastal sea interaction and human dimensions, LOICZ reports and studies no. 25. LOICZ, Texel, pp 168–173

Francis J, Wagner GM, Mvungi A, Ngwale J, Salema R (2002) Development and protection of the coastal and marine environment in sub-Saharan Africa: Tanzania national report phase 1: integrated problem analysis. Part of the GEF MSP Sub-Saharan Africa Project (GF/6010-0016): "Development and Protection of the Coastal and Marine Environment in Sub-Saharan Africa". UNEP, Nairobi. http://www.acops.org/African_Process/National_Reports.htm

Gabrie C, Vasseur P, Randriamiarana H, Maharavo J, Mara E (2000) The coral reefs of Madagascar. In: McClanahan TR, Sheppard CRC, Obura DO (eds) Coral reefs of the Indian Ocean – their ecology and conservation. Oxford University Press, Oxford, pp 411–444

GEO Data Portal (2003) Global environment outlook data portal. Retrieved January 2003, from http://geodata.grid.unep.ch/

GoK (1979) 1979–1983 development plan. Government of the Republic of Kenya, Government Press, Nairobi

GoT (2006) National land use framework plan, existing situation, trends and issues. The National Land use Planning Commission. Government of Tanzania, Dar es Salaam

Gove DZ (1995) The coastal zone of Mozambique. In: O Lindén (ed.) Workshop and policy conference on integrated coastal zone management in Eastern Africa including the Island States. Coastal Management Center (CMC), Conference proceedings, Metro Manila, Philippines, pp 251–273

Grandcourt EM (2003) The effect of intensive line fishing on the virgin biomass of a tropical deepwater snapper, the crimson jobfish – *Pristipomoides filamentosus*. Fish Bull 101:305–311

Gullström M, de la Castro Torre M, Bandeira SO, Björk M, Dahlberg M, Kautsky N, Rönnbäck P, Öhman MC (2002) Seagrass ecosystems in the Western Indian Ocean. Ambio 31:588–596

Hamerlynck O, Richmond MD, Mohamed A, Mwaitega SR (2008) Fish and invertebrate life histories and important fisheries of the Pangani River Basin, Tanzania final report. Pangani River Basin flow assessment – specialist study. IUCN/PBWO, 74 pp

Hatziolos M, Lundin CG, Alm A (1996) Africa: A framework for integrated coastal zone management. The World Bank, Washington, DC, p 150

Hirji R, Partorni FM, Rubin D (eds) (1996) Seminar on integrated water resources management in Kenya. In: Proceedings sponsored by government of The Netherlands and the World Bank. The Economic Development Institute of the World Bank, 227 pp

Hirji R, Johnson P, Maro P, Chiuta TM (eds) (2002) Environmental sustainability in water resources management in Southern Africa. SADC/IUCN/SARDC/World Bank, Maseru/Harare/Washington, DC

Hoff H, Noel S, Droogers P (2007) Water use and demand in the Tana basin: analysis using the water evaluation and planning tool (WEAP), Green water credits report 4. ISRIC – World Soil Information, Wageningen, p 38

Hogan AR, Nandi RXL, Mtiga MO, Chirwa EB, Kilonzo P, Peter J (1999) Selection of pilot villages: a report on the rapid appraisal methodology used and the selections made, together with eleven individual village profiles. REMP technical report no 2

Hoguane AM (1997a) Hydrodynamics, temperature and salinity in mangrove swamps in Mozambique. Ph.D. thesis, University of North Wales, Bangor, UK

Hoguane AM (1997b) Shrimp abundance and river runoff in Sofala Bank – the role of Zambezi. Presented at the workshop on sustainable development of the Cahora Bassa Dam and the Valley of Zambezi

Hoguane AM (2007) Diagnosis of Mozambique coastal zone. Revista de Gestão Costeira Integrada 7(1):69–82

Hoguane AM, Motta H, Lopes S, Menete Z (2002) Development and protection of the coastal and marine environment in sub-Saharan Africa: Mozambique national report phase 1: integrated problem analysis, 113 pp, March 2002. GEF MSP Sub-Saharan Africa Project (GF/6010-0016) Sponsored by GEF, UNEP, IOC-UNESCO, GPA and ACOPS

Horril JC, Kamkuru AT, Mgaya YD, Risk M (2000) Northern Tanzania and Zanzibar. In: McClanahan TR, Sheppard CRC, Obura DO (eds) Coral reefs of the Indian Ocean – their ecology and conservation. Oxford University Press, Oxford, pp 167–198

Hughes RH, Hughes JS, Bernacsek G (1992) A directory of African wetlands. http://ramsar.wetlands.org/ADirectoryofAfricaWetlands/tabid/824/Default.aspx. Accessed on 18 Feb 2009

Ingram JC, Dawson TP (2001) The impacts of a river effluent on a coastal seagrass habitat of Mahé, Seychelles. S Afr J Bot 67:483–487

IUCN (2003)Tana River, Kenya: integrating downstream values into hydropower planning. Case studies in wetland valuation, no. 6, May 2003. IUCN – The World Conservation Union. http://www.iucn.org/themes/wani/econ/CaseStudy06Tana.pdf

IWMI (2006) IWMI global wetland initiative – Madagascar country overview. www.iwmi.cgiar.org/wetlands/pdf/Africa/Region6/MADAGASCAR.pdf. Accessed on 13 Oct 2007

Jiddawi NS, Öhman MC (2002) Marine fisheries in Tanzania. Ambio 31(7):518–527

Joint Water Commission (2008) Rovuma/Ruvuma River Basin issues paper, final report, Republic of Mozambique and United Republic of Tanzania, 77pp

Kairu K, Nyandwi N (eds) (2000) Guidelines for the study of shoreline change in the Western Indian Ocean region, IOC manuals and guides no. 40. UNESCO, Paris, p 55

Kajia YS (2000) Assessment of the effects of rice farming in the mangrove forests of the Rufiji delta, mainland Tanzania. MSc. thesis in ecological marine management, Faculty of Science, Free University of Brussels (cited in Shagude 2004)

Kalugina-Gutnik A, Perestenko L, Titlyanova T (1992) Species composition, distribution and abundance of algae and seagrasses of the Seychelles Islands. Atoll Res Bull 359(5):1–68

Kambona JJ, Marashi SH (1996) Process for the establishment of the Indian Ocean Tuna Commission, FAO fisheries circular 913. FAO, Rome, p 53

Kaponda A (2005) Ruvuma River Basin: water resources development and management. Unpublished report for Ruvuma river and Southern Coast Basin

Kent PE, Hunt JA, Johnstone DW (1971) The geology and geophysics of coastal Tanzania, Geophysical 6. HMS Stationary Office, Institute of Geophysical Sciences, London, p 101

Kijazi AS (2002) Catchment and mangrove management programmes: activities and environmental concerns in the Pangani Basin. Report of a workshop held at the Kilimanjaro Crane Hotel, Moshi, Tanzania, 8–10 May 2002. December 2002, Nairobi, IUCN – EARO, pp 38–43

Kitheka JU, Nthenge P, Obiero M (2003a) Sabaki estuary sediment transport dynamics and resultant sediment fluxes in the monsoon-driven Malindi Bay. LOICZ-START AfriCAT report, Nov 2003, 58 pp

Kitheka JU, Obiero M, Nthenge P (2003b) Dynamics of sediment transport and exchange in the Tana Estuary in Kenya. Project report – KMFRI, Mombassa, 40 pp

Kitheka JU, Ochiewo J, Nthenge P, Obiero M (2003c) Coastal impacts of damming and water abstraction in the Tana and Athi-Sabaki river basins. LOICZ-START AfriCAT project report, Dec 2003, 70 pp

Kitheka JU, Ongwenyi GS, Mavuti KM (2003d) Fluxes and exchange of suspended sediments in tidal inlets draining a degraded mangrove forest in Kenya. Estuar Coast Shelf Sci 56:655–667

Kitheka JU, J Ochiewo, Nthenge P, Obiero M (2004) Land-use change and climatic variability in the Tana and Athi-Sabaki River Basins and associated impacts on coastal resources in Kenya. Paper for START-LOICZ AfriCAT workshop, 18–21 February 2004

Kitheka JU, Obiero M, Nthenge P (2005) River discharge, sediment transport and exchange in the Tana Estuary, Kenya. Estuar Coast Shelf Sci 63:455–468

Krantz L, Sorensen NK, Olesen J, Kotalova J (1989) The fisheries in Mozambique: a sector study. Fisheries Research Institute, Maputo, p 103

Lablache G, de Mossac GA, Shah NJ (1987) Summary description of the marine fisheries and resources for Seychelles. In: Sanders MJ, Sparre P, Venema SC (eds) Proceeding of the workshop on the assessment of the fishery resources in the southwest Indian Ocean, 14–25 September 1987, Albion Mauritius. FAO, Rome, p 277

LaRoche J, Ramanarivo N (1995) A preliminary survey of the artisanal fishery on coral reefs of the Tulear Region (southwest Madagascar). Coral Reefs 14:193–200

Louw A, Gichuki F (2003) Limpopo Basin profile: strategic research for enhancing agricultural water productivity. Challenge program on water and food. http://www.waterforfood.org

Macnae W (1968) A general account of the fauna and flora of mangrove swamps and forest in the Indo-West Pacific region. Adv Mar Biol 63:73–270

Massingue A, Bandeira SO (2005) Distribution of seagrasses and common seaweeds around Nampula Province (northern Mozambique) with emphasis on Mozambique Island. W Indian Ocean J Mar Sci 4(2):175–185

Masundire H, Mackay H (2002) The role and importance of aquatic ecosystems in water resources management. In: Hirji R, Johnson P, Maro P, Chiuta TM (eds) Environmental sustainability in water resources management in Southern Africa. SADC/IUCN/SARDC/World Bank, Maseru/Harare/Washington, DC

Maud RR (1980) In: Bruton MN, Cooper KH (eds) Studies on the ecology of Maputaland. Rhodes University & Wildlife Society/Transvaal Printers, Cape Town, pp 1–7

McMahon KM, Waycott M (2009) New record for *Halophila decipiens* (Ostenfeld) in Kenya based on morphological and molecular evidence. Aquat Bot 91(4):318–320

Miyake MP, Miyabi N, Nakano H (2004) Historical trends of tuna catches in the world, FAO fisheries technical paper 467. FAO, Rome, p 74

Montaggioni L, Faure G (1980) Les Récifs corallines des Mascareignes (Océan Indien). Coll. Trav. Univ. Fr. O, Indien, p 151

Motta H, Pereira MAM, Gonçalves M, Ridgway T, Schleyer MH (2002) Coral reef monitoring in Mozambique. II: 2000 report. MICOA/CORDIO/ORI/WWF. Maputo, Mozambique Coral Reef Management Programme, 31 pp

Muir C (2006) The use of TEDS to reduce turtle bycatch in the WIO region. In: Kambona JJ, Marashi SH (eds 1996). Process for the establishment of the Indian Ocean Tuna Commission. FAO Fisheries Circular 913. FAO, Rome, 53 pp

Munga D, S Mwangi J, Kamau MM, Nguli PO, Gwada LN, Daudi H, Ong'anda SM, Mwaguni HS, Massa M, Tole JM, Onyari J, Makopa A, Gachanja G, Opello A, Kheir M, Machua S (2007) National status report on land-based activities, sources of pollution and pollutant levels in water and sediment: Kenya. Unpublished report submitted to UNEP/WIO-LaB Project/Nairobi Convention Secretariat, Nairobi, Kenya, 56 pp

Mwalyosi RBB (2004) Management of the Rufiji Delta as a wetland. Institute of Resource Assessment, University of Dar es Salaam, Dar es Salaam

Ngusaru A (1997) Geological history. In: Richmond MD (ed) A field guide to seashores of Eastern Africa and the Western Indian Ocean Islands. SIDA, SAREC and University of Dar es Salaam, Dar es Salaam, 461 pp

Nhwani LB (1988) Summary of fisheries and resources information for mainland Tanzania. In: Sanders MJ, Sparre P, Venema SC (eds) Proceedings of the workshop on the assessment of the fishery resources in the Southwest Indian Ocean, 14–25 September 1987, Albion. FAO/UNDP: RAF/79/065/WP/41/88/E, FAO, Rome, 277 pp

Noble RG, Hemens J (1978) Inland water ecosystems in South Africa – a review of research needs. Council for Scientific and Industrial Research (CSIR), Pretoria, p 150

Obura DO (2001) Kenya. Mar Pollut Bull 42(8):1264–1278

Ochieng C, Erftemeijer PLA (2001) The status of seagrass ecosystems in Kenya and Tanzania. In: Draft paper submitted to the UNEP-WCMC Global Seagrass workshop, St Pete's Beach, Florida, 5-–9 November 2001, Unpublished Report

Ochieng CA, Erftemeijer PLA (2003) Seagrasses of Kenya and Tanzania. In: Green EP, Short FT (eds) World atlas of seagrasses. WCMC, Cambridge, pp 82–92

Okemwa EN (1998) Application of the large marine ecosystem concept to the Somali Current. In: Sherman K, Okemwa EN, Ntiba MJ (eds) Large marine ecosystems of the Indian Ocean: assessment, sustainability and management. Blackwell Science, London, pp 73–100

Okemwa E, Wakabi E (1993) Integrated coastal zone management of Kenya. In: Lindén O (ed) Proceedings of the workshop and policy conference on integrated coastal zone management in eastern Africa including the island states, 21–23 April 1993, Arusha. Coastal Management Center, Manila, pp 183–204

Pallet J (ed) (1997) Sharing water of Southern Africa. The Desert Research Foundation of Namibia, Windhoek

Payet R (2005) Research, assessment and management on the Mascarene Plateau: a large marine ecosystem perspective. Philos Trans R Soc 363:295–307

PBWO/IUCN (2006) Hydrology of the Pangani River Basin. PBWO/IUCN Eastern Africa Regional Program, Moshi/Nairobi

PBWO/IUCN (2007) Pangani River System – State of the Basin Report – 2007. PBWO/IUCN Eastern Africa Regional Program, Moshi/Nairobi

Rabesandratana RN (1996) Ecological distribution of seaweeds in two fringing coral reefs at Toliara (SW of Madagascar). In: Bjork M, Semesi AK, Pedersén M, Bergman B (eds) Current trends in marine botanical research in East Africa region. SIDA/SAREC, Uppsala, pp 141–161

Rajonson J (1993) Mangroves and coral reefs in Mauritius. In: LINDÉN O (ed) Proceedings of the workshop and policy conference on integrated coastal zone management in eastern Africa including the Island States, 21-23 April 1993, Arusha. Coastal Management Centre, Manila, pp 205–216

Ralison S (1987) Summary of fisheries and resources information for Madagascar. In: Sanders MJ, Sparre P, Venema SC (eds) Proceedings of the workshop on the assessment of the fishery resources in the Southwest Indian Ocean. Albion, Mauritius, 4–25 Sept 1987. RAF/79/065/WP/41/85/E, 277 pp

Ramsay PJ (1996) Quaternary marine geology of the Sodwana Bay continental shelf, northern KwaZulu-Natal. Bull Geol Sur S Afr 117:1–86

Ramsay PJ, Mason TR (1990) Development of a type zoning model for Zululand coral reefs, Sodwana Bay South Africa. J Coast Res 6:829–852

Ribbink AJ, Roberts M (2006) African coelacanth programme overview: an overview of the conference contributions. S Afr J Mar Sci 102:409–415

Richmond MD, Mohamed A (2005) Assessment of marine biodiversity, ecosystem health and resource status of intertidal (non-mangrove) and sub-tidal (non-coral) habitats in MBREMP. Final report. Samaki Consultants Ltd./Dar es Salaam, Tanzania for IUCN-EARO, Nairobi, p 61

Richmond MD, Wilson JDK, Mgaya YD, Le Vay L (2002) An analysis of smallholder opportunities in fisheries, coastal and related enterprises in the floodplain and delta areas of the Rufiji River, Tanzania. Rufiji Environment Management Project, IUCN, Dar es Salaam

Robinson J, Shroff J (2004) The fishing sector in Seychelles. An overview with an emphasis on artisanal fisheries. SFA. Republic Seychelles, Seychelles Med Dent J spec issue 7(1):52–56

Røhr PC, Killingtveit A (2002) Study of two catchments on the hillside of Mt. Kilimanjaro. In: Ngana JO (ed) Water resources management: the case of Pangani River Basin. Issues and approaches. Dar es Salaam University Press, Dar es Salaam, pp 211–231

RSA (1998) National Water Act – Act no 36 of 1998. ftp://ftp.hst.org.za/pubs/govdocs/acts/1998/act36.pdf

Samboo CR, Mauree D (1987) Summary of fisheries and resources information for Mauritius. In: Sanders MJ, Sparre P, Venema SC (eds) Proceeding of the workshop on the assessment of the fishery resources in the southwest Indian Ocean, 14–25 September 1987, Albion. FAO, Rome, p 277. FAO/UNDP.RAF/79/065/WP/41/88/E

SARDC (2003) Limpopo River Basin – Factsheet 1. SADC Water Sector. http://www.sardc.net/imercsa/Limpopo/pdf/Limpopo1.pdf. Accessed on 21 Sept 2006

Schleyer MH, Celliers L (2002) Biodiversity on Southern African coral reefs: what does the future hold? CORDIO annual report, 2002

Schleyer MH, Celliers L (2005a) An assessment of the coral reefs at Tamatave on the east coast of Madagascar for a proposed marine outfall: 2005 survey. Oceanographic Research Institute, Durban, p 18

Schleyer MH, Celliers L (2005b) Modelling reef zonation in the Greater St. Lucia Wetland Park, South Africa. Estuar Coast Shelf Sci 63(3):373–384

Semesi AK, Ngoile M (1993) Status of the coastal and marine environment in the United Republic of Tanzania. In: Lindén O (ed) Proceedings of the workshop and policy conference on integrated coastal zone management in eastern Africa including the Island States, 21–23 April 1993, Arusha. Coastal Management Center, Manila, pp 291–314

Semesi AK, Muruke MHS, Mgaya YD (1999) Introduction to the mangroves, seagrasses, seaweeds and coral reefs. In: Workshop proceedings on coastal resources of Bagamoyo District Tanzania, Bagamoyo, 18–19 December 1997

Shaghude YW (2004) Coastal impacts of water abstraction and impoundment by damming in Africa: the case of Rufiji river, Tanzania. Final report submitted to START, Washington, DC

Shah NJ (1993) The coastal zone of Seychelles. In: Lindén O (ed) Proceedings of the workshop and policy conference on integrated coastal zone management in eastern Africa including the Island States, 21–23 April 1993, Arusha. Coastal Management Center, Manila, pp 275–290

Shahin M (2003) Hydrology and water resources of Africa. Kluwer, New York

Sheppard CRC (1987) Biodiversity, productivity and system integrity. Mar Pollut Bull 34:680–681

Sheppard CRC (2000) Coral reefs of the Western Indian Ocean: an overview. In: McClanahan TR, Sheppard CRC, Obura DO (eds) Coral reefs of the Indian Ocean – their ecology and conservation. Oxford University Press, Oxford, pp 3–38

Short FT, Coles R (eds) (2003) Seagrass research methods. Elsevier, Amsterdam

Short FT, Coles RG, Pergent-martini C (2001) Global seagrass distribution. In: Short FT, Coles R (eds) Seagrass research methods. Elsevier Publishing, Amsterdam, pp 5–30

Silva C, Sousa MI (1987) Summary description of the marine fisheries and resources for Mozambique. In: Sanders MJ, Sparre P, Venema SC (eds) Proceeding of the workshop on the assessment of the fishery resources in the southwest Indian Ocean, 14–25 September 1987, Albion. FAO, Rome, p 277. FAO/UNDP.RAF/79/065/WP/41/88/E

Slingo J, Spencer H, Hoskings B, Berriford P, Black E (2005) The meteorology of the Western Indian Ocean and the influence of the East African highlands. Phil Trans R Soc A 363:25–42

Smith MM, Heemstra PC (1986) Tetraodontidae. In: Smith MM, Heemstra PC (eds) Smiths' sea fishes. Springer, Berlin, pp 894–903

Snoussi M, Kitheka J, Shaghude Y, Kane A (2004) Coastal impacts of damming and water abstraction in African catchments – a synthesis of Africa. START, Washington, DC

Sookocheff W, Muir C (2006) A literature review of marine fisheries bycatch data in the United Republic of Tanzania. In: Kiszka J, Muir C (eds) Incidental catch of non-targeted marine species in the Western Indian Ocean: problems and mitigation measures. Workshop proceeding, 13–15 November 2006, Mayotte, pp 92–99

Souter D, Lindén O (eds) (2005) Coral reef degradation in the Indian Ocean: status report 2000. CORDIO, Stockholm, p 285

Souter D, Obura D, Lindén O (eds) (2000) Coral reef degradation in the Indian Ocean: status report 2000. CORDIO, Stockholm, p 205

Spalding MD, Blasco F, Field CD (eds) (1997) World mangrove atlas. The International Society for Mangrove Ecosystems, Okinawa, p 178

Spalding MD, Ravilious C, Green EP (2001) World atlas of coral reefs. University of California Press, Berkeley, CA

Stoddart DR (1984) Biogeography and ecology of the Seychelles Islands, Monographia biologica v55. Junk, The Hague, p 691

Taylor M, Ravilious C, Green EP (2003) Mangroves of East Africa. UNEP, Cambridge, p 24

Temple PH, Sundborg A (1973) The Rufiji River, Tanzania: hydrology and sediment transport. In: Rapp A, Berry L, Temple P (eds) Studies on soil erosion and sedimentation in Tanzania, Research monograph no. 1. Institute of Resource Assessment and Land use Planning, Dar es Salaam, pp 345–368

Titlyanov E, Cherbagdy I, Kolmakov P (1995) Daily variations of primary production and dependence of photosynthesis onvan K irradiance in seaweeds and seagrass *Thalassodendron ciliatum* of the Seychelles Islands. Photossyntetica 31:101–115

Tomlison CB (1986) The botany of mangroves, Cambridge tropical biology series. Cambridge, University Press, Cambridge/New York

TPTC (2001) Joint Incomati Basin Study. Tripartite Permanent Technical Committee: Kingdom of Swaziland, Republic of Mozambique, Republic of South Africa

Turner J, Klaus R (2005) Coral reefs of the Mascarenes, Western Indian Ocean. Philos Trans R Soc A 363(1826):229–250

Turpie JK (2006) The valuation of riparian fisheries in southern and eastern Africa. Southern Waters, Cape Town

Turpie JK, Lamberth S (2005) Economic value of the Thukela Banks crustacean and line fish fisheries and the potential impacts of changes in the Thukela River flow. In: Turpie JK, Hosking SG (eds) Proceedings of a national workshop on resource economics in South Africa, 10–11 March 2005, Velddrif. Anchor Environmental Consultants CC, Rhodes Gift, pp 61–75

Uku J, Ndirangu S, Muhtama C, Kimathi A (2005) An evaluation of the effect of sea urchin herbivory in the Diani-Chale lagoon. Preliminary report no 1. A KMFRI/CORDIO collaborative project report, 8 pp

Uku J, Ndirangu S, Muhtama C, Kimathi A, Kilonzi J (2007) An evaluation of the effect of sea urchin herbivory in the Diani-Chale lagoon: recovery potential of the seagrass habitat. Preliminary report II. A KMFRI/CORDIO collaborative project report, 13 pp

UNEP (1998a) Eastern Africa atlas of coastal resources 1: Kenya. UNEP regional reports and studies. UNEP and BADC (Government of Belgium), Nairobi, p 119

UNEP (1998b) Overview of land-based sources and activities affecting marine, coastal and associated freshwater environment in the eastern Africa region. UNEP regional; seas reports and studies no 167

UNEP (2001) Eastern Africa atlas of coastal resources: Tanzania. UNEP and BADC (Government of Belgium), Nairobi, p 111

UNEP (2005) Hydropolitical vulnerability and resilience along international waters. United Nations Environment Programme, Nairobi

UNEP (2006) Our environment our wealth, Africa environment outlook 2. United Nations Environment Programme, Nairobi, p 542

UNEP/GPA, WIOMSA (2004) Regional overview of the physical alteration and habitat destruction (PADH) in the Western Indian Ocean. UNEP, Nairobi, p 74

UNEP/Nairobi Convention Secretariat and WIOMSA (2009) Transboundary diagnostic analysis of land-based sources and activities degrading the Western Indian Ocean ecosystems. UNEP, Nairobi

UNEP/Nairobi Convention Secretariat and WIOMSA (2015) The regional state of the coast report: Western Indian Ocean, UNEP and WIOMSA, Nairobi, Kenya, 546 pp

Van der Elst RP, Everett B, Jiddawi N, Mwatha G, Afonso PS, Boulle D (2005) Fish, fishers and fisheries of the West Indian Ocean: their diversity and status. A preliminary assessment. Philos Trans R Soc 363(1826):263–284

Van der Zaag P (2000) The Pungwe river basin. Webworld.unesco.org

Van der Zaag P, Carmo Vaz A (2003) Sharing the Incomati waters: cooperation and competition in the balance. Water Policy 5:349–368

Van Katwijk MM, Meier NF, van Loon R, van Hove EM, Giesen WBJT, van der Velde G, den Hartog C (1993) Sabaki River sediment load and coral stress: correlation between sediments and condition of the Malindi-Watamu reefs in Kenya (Indian-Ocean). Mar Biol 117:675–683

Watermeyer, Legge, Piesold, and Uhlman (1981) Athi river basin preinvestment study. For: TARDA (Tana and Athi- River Development Authority), Nairobi. Agrar and Hydrotechnik GmbH, Essen

Weller RA, Baumgartner MF, Josey SA, Fischer AS, Kindle JC (1998) Atmospheric forcing in the Arabian Sea during 1994–1995: observations and comparisons with climatology and models. Deep-Sea Res II Top Stud Oceanogr 45(10–11):1961–1999

Wilkinson CR (1999) Global and local threats to coral reef functioning and existence: review and prediction. Mar Freshw Res 50:867–878

Wilkinson CR, Lindén O, Cesar H, Hodgson G, Rubens J, Strong AE (1999) Ecological and socio-economic impacts of 1998 coral mortality in the Indian Ocean: an ENSO impact and a warning for future change? Ambio 28:188–196

WRI (2007) World Resources Institute. Climate analysis indicators tool. http://www.wri.org/climate/project_description2.cfm?pid=93

Zanting HA, Nacueve C, Novela B (1994) Salt intrusion in the Pungwe estuary. Conflict prevention and cooperation in international water resources handouts DNA. Maputo/Companhia de Aguas da Beira E. E., Beira: 1–3, 19, 29–32. http://unesdoc.unesco.org/images/0013/001333/133316e.pdf

ZRA (1998) Environmental Impact of Expanding Utilisation of Water Resources from the Basin. Sector Studies under ZACPLAN. Sector Study No. 7. Zambezi River Authority, Lusaka

Section I

Western Indian Ocean Estuaries, a Source of Life

Being rich nurseries and spawning grounds for many fish, crustaceans and other marine species, the West Indian Ocean (WIO) estuaries provide an important home for many sources of aquatic life; its aquatic habitats providing critical nursery grounds and habitats for numerous aquatic species, as well as being important feeding ground for birds. The extensive mangrove forests in some of the regional estuaries act as feeding grounds for shrimp, crab and finfish, as well as being important sources of building materials for the local communities. The WIO estuaries are also used for aquaculture, mainly targeting prawns, while their large mangrove forests play an important role in sustaining the productivity of deltas and bays, which represent the most productive coastal fishing ground in the region. There is an estimated 10,000 km^2 of mangroves in the WIO (Spalding et al. 1997), representing about 5.0 % of the total mangrove coverage in the world with the best developed mangroves in the region being located in the deltas of the Rufiji River (Tanzania), Tana River (Kenya), Zambezi and Limpopo Rivers (Mozambique), and along the west coast of Madagascar at Mahajanga, Nosy be and Hahavavy.

In providing these important functions, the WIO estuaries exhibit a diverse range of coastal and marine habitats and ecosystems, including coral reefs, sea grass beds, mangroves, sandy beaches, sand dunes and terrestrial coastal forests. It is clear that the rich and diverse marine ecosystems in the relatively small continental shelf area of the WIO coastal zone are fundamentally important to the sustainable productivity of the region as a whole, particularly its fisheries. Mangroves are located in the estuaries of most rivers, while sea grass beds flourish in shallow, soft bottom environments that receive sunlight. Coral reefs have developed on rocky fringes and hard surfaces in clear, well-oxygenated waters, most notably along sediment - free coastlines, forming hundreds of kilometres of fringing reefs along the edge of the mainland and around the oceanic islands. All these ecosystems and the waters they contain support plankton communities which, in turn, feed numerous diverse fish populations.

The following four papers provide a cross -cut description of some of the ecological richnesses of the WIO estuary ecosystems, with a specific focus on the ecosystem services they provide.

Species Composition of Fisheries Resources of the Tana and Sabaki Estuaries in the Malindi-Ungwana Bay, Kenya

Cosmas N. Munga, Edward Kimani, Renison K. Ruwa, and Ann Vanreusel

Abstract

For over 30 decades, the Sabaki and Tana estuaries of the Malindi-Ungwana Bay, Kenya have supported both the artisanal fishery and semi-industrial bottom trawl sectors. Currently these estuaries in the bay support over 3 000 artisanal fishers and a maximum acceptable fleet of four medium-sized trawlers. These sectors have exerted pressure on the fisheries resources of the bay and will continue to do so due to the increasing artisanal fishing effort. We describe the present status of the fisheries resources of the estuaries in the bay following shore-based catch assessments between 2009 and 2011, and shallow-water bottom trawl surveys in early 2011. These aimed to determine species composition, relative abundance and distribution patterns of the penaeid shrimps and associated trawl fish bycatches, and fish catches from the artisanal fishers. Five shrimp species: *Fenneropenaeus indicus*, *Penaeus monodon*, *Metapenaeus monoceros*, *Penaeus semisulcatus* and *Penaeus japonicus* were recorded. Distinct shrimp species composition existed between the two estuaries characterised by more abundant *F. indicus* in the Tana estuary, and more abundant *P. semisulcatus* in the Sabaki estuary. Bottom trawl fish bycatch species diversity was higher than for artisanal fish catches with a total of 223 and 177 species respectively. Shrimp total biomass and catch rates were significantly higher during the wet Southeast Monsoon (SEM) season than the dry Northeast Monsoon (NEM) season, and decreased as depth increased. On the other hand, trawl bycatch rates were significantly higher in inshore than offshore areas and distinct in composition but less differing between the seasons. Similarity in catch composition was evident between the artisanal catches and bottom trawl bycatches in the inshore areas. This similarity was attributed mainly to seven common and most abundant fish species targeted in artisanal fishery as well as these species made the highest bycatch proportion in the shrimp bottom trawls. Significantly smaller-sized individuals of these seven species occurred in trawl bycatches than in artisanal catches attributed to differences in gear selectivity. Implementation of the present shrimp fishery management plan, and continued monitoring of fish trawl bycatches will be crucial for the effective management of fisheries resources of the estuaries in the bay.

Keywords

Species composition • Semi-industrial bottom trawl • Penaeid shrimps • Fish bycatches • Artisanal catches • Tana • Sabaki • Kenya

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Introduction

The Kenyan coast runs in a south-westerly direction from the border with Somalia in the north at 1^0 41′S to 4° 40′S at the border with Tanzania in the south. It lies in the hot tropical region where the weather is influenced by the monsoon winds of the Indian Ocean. Climate and weather systems are dominated by the large scale pressure systems of the Western Indian Ocean (WIO) and the two monsoon seasons, the dry Northeast Monsoon (NEM) from October to March and the wet Southeast Monsoon (SEM) from April to September (McClanahan 1988). Kenya's coastal ecosystems occupy the western extremity of the tropical Indo-Pacific biogeographic region, and have been classified as part of the Coral Coast of the East African Marine Ecoregion (WWF 2004).

The coastal ecosystems are broadly classified into tropical rainforests, estuarine and nearshore areas, and the open sea (Government of Kenya 2008). These ecosystems include: mangrove swamps, coral reefs, seagrass beds, rocky shores, estuaries, beaches, mudflats, sand dunes and terrestrial habitats, all closely interlinked through various biotic and abiotic fluxes. A wide variety of fish and other marine organisms migrate between ecosystems for breeding, feeding and refugia. An almost continuous fringing reef dominates the inshore areas along the Kenyan coast, except in the Malindi-Ungwana Bay where the river systems of the Sabaki and Tana have created conditions of low salinity and high turbidity especially during the wet SEM season, with limited coral growth.

The distribution of these coastal ecosystems is also influenced by the coastal geology and oceanography. The interactions between the north-flowing East African Coastal Current (EACC) and the seasonal south-flowing Somali Current (SC) create a temperature gradient of warm to cool from south to north. This affects the productivity of the open sea ecosystems, resulting in the development of coral reefs in the cooler, nutrient-rich waters of the north, and extensive mangrove, seagrass and suspension-feeding communities towards the south. The rich biodiverse coastal ecosystems provide critical socio-economic and ecological services such as protection from storm surges, food, wood fuel, and livelihoods for the local communities. In the lower Sabaki and Tana River flood plains and oxbow lakes, subsistence fisheries of brackish and freshwater species of Protopteridae (lungfishes), Claridae (catfishes), Cichlidae (tilapines), Anguillidae (eels), and prawns (*Macrobrachium* sp.) is common. These vital coastal ecosystems are on the other hand, facing threats from ever increasing human pressure through tourism, industrial pollution, inshore overfishing, mangrove logging (Tychsen 2006), commercial salt mining and the ongoing offshore gas and oil exploration (pers. obs.).

Sustainable management of coastal artisanal fisheries in the tropics is challenging due to the multigear, multispecies and multifleet nature and the lack of adequate resources to conduct scientific studies, monitoring and enforcement (McClanahan and Mangi 2004). There is a growing awareness that reliable knowledge on trends in catch composition and selectivity of commonly used gear is important for management recommendations (Gobert 1994; McClanahan and Mangi 2004). The artisanal fishery is receiving increasing attention from scientists and environmental managers for both ecological and socio-economic reasons, including user conflicts, habitat destruction and stock depletion. The current climate change phenomenon has threatened species composition of reef-based fisheries as reef habitats are getting destroyed under unprecedented pressure (Cinner et al. 2009). In the Malindi-Ungwana Bay, artisanal fisheries is restricted to the inshore fishing grounds of <3 nautical miles (nm) off the Sabaki and Tana estuaries due to inability of the artisanal fishers to access offshore fishing grounds. These inshore fishing grounds, are also the main shallow water shrimp trawling grounds (Mwatha 2005; Munga et al. 2012; Munga et al. 2013) causing user conflict between the artisanal fishers and semi-industrial shrimp trawlers. The shrimp fishery management plan 2010 ensures sustainable shrimp exploitation by restricting trawl fishing grounds and limiting trawling effort as a way to reduce conflicts between the trawl and artisanal fishery sectors.

Recent estimates place the number of artisanal fishers in the bay at >3 000, with around 1 000 fishing crafts, ranging from dugout canoes used near the shore to large dhows for open sea fishing (Government of Kenya 2014). The number of artisanal fishers is expected to increase as a result of population growth. Catches of the artisanal fishery comprise a multispecies mix of demersal fishes (50% by weight), pelagic fishes (28%), and the rest 22% made up of sharks and rays, octopus and squid, shrimps, lobsters and crabs (Government of Kenya 2010; Munga et al. 2012, 2014). This species mix is typical of artisanal fisheries in the South West Indian Ocean (Jiddawi and Öhman 2002; van der Elst et al. 2005).

The semi-industrial bottom trawl fishery was initiated after a series of successful surveys undertaken by the Kenya Government, UNDP and FAO in early 1960s (Iversen 1984; Venema 1984; Saetersdal et al. 1993). Bottom trawling thereafter, continued for several decades, landing an average 400 tonnes of shrimps annually in the 1970s, 80s and 90s (Mwatha 2005). The government however, temporarily closed the shrimp trawl fishery in 2006, as a result of user conflicts between trawlers and artisanal fishers over declining catches, artisanal fishing gear destruction, perceived environmental degradation, and excessive trawl bycatches whose low market prices competed unfairly with artisanal catches (Fulanda et al. 2009, 2011; Munga et al. 2012). Trawl catches in the bay include shrimps and a large bycatch of fish, sharks, rays, crustaceans and other invertebrates (Fennessy et al. 2004, 2008). Although some of the bycatch is retained and sold, most has low commercial value, and is discarded overboard. The trawl fishery resumed in 2012 with a maximum of four trawlers but the spatio-temporal management strategy of Ungwana Bay remains under review.

This paper aims to investigate the spatial and temporal patterns in the composition of the bottom trawl catches (shrimps and bycatches) and of catches made by the artisanal fishers as well as to assess the extent of resource-use overlap between the two sectors in the bay. Factors taken into account in comparisons were seasons (SEM *versus* NEM) and fishing areas (inshore *versus* offshore). This information is important for the development of spatio-temporal fisheries management strategies to mitigate against resource-use conflict in the estuaries of the bay.

Material and Methods

The Study Area

The Malindi-Ungwana Bay comprises of the larger northward Ungwana Bay and the smaller southward Malindi Bay, and lies off the East African coast in the Western Indian Ocean (WIO) region (Fig. 1). The bay is located between the latitudes 2° 30′S and 3° 30′S, and the longitudes 40° 00′E and 41° 00′E and extends from Malindi through Ras Ngomeni in the south to Ras Shaka in the north covering a distance of about 200 km long. It encompasses the fishing grounds of Sabaki and Tana river estuaries. Administratively, the Malindi-Ungwana Bay is located within the two counties of Malindi and Tana River with populations of 281,600 and 181,000 respectively out of a population of about 3 million for the entire coastal area, about 8% of the Kenyan population (Government of Kenya 2002). The bay including the North Kenya Bank covers a total trawlable area of 10,994 km^2 against a total estimate of 19,120 km^2 of the entire Exclusive Economic Zone (Mutagyera 1984). The bay around the Tana outflow is shallow and extends between 8 and 32 nm. The mean depth at spring high tide is 12 m at 1.5 nm, and 18 m at 6.0 nm from the shore. The depth increases rapidly to 100 m after 7 nm from the shore. Near the Sabaki outflow, the offshore distance stretches between 3 and 5 nm, whereafter depth rapidly increases to 40 m (Kitheka et al. 2005; Kitheka 2013). Critical habitats along the bay include mangrove forests, patchy reefs, islets, sandy shores and tidal flats. The Sabaki estuary is also an Important Bird Area (IBA) as it hosts large visiting flocks of Madagascar pratincole, and important resting, roosting and feeding ground for gulls and terns (Tychsen 2006). Climate and weather systems are dominated by the large scale pressure systems of the WIO and the two monsoon seasons, the dry NEM and the wet SEM.

Bottom Trawl Data Collection for Shrimps and Fish Bycatches

Two bottom trawl surveys were conducted during January–February (NEM) and May–June 2011 (SEM) using a commercial shrimp trawler. A total of 41 and 36 tows were made roughly parallel to the shoreline in NEM and SEM seasons respectively, with total trawled areas of 546.4 nm^2 and 507.7 nm^2 each. All unwanted debris, plants and large organisms were first removed from catches, whereafter the remainder were sorted into fish and shrimp categories. Total catches of shrimps were weighed, a 2 kg sub-sample for large catches, and the entire catch for small catches, were frozen for species identification in a laboratory. The remainder of the fish component was divided into mixed equal portions, and one portion was randomly sampled after large fish were first removed and weighed separately. The total catch of each species from each tow was calculated by multiplying the sub-sample by a raising factor derived from the sub-sample to total shrimp or fish catch weight (see Stobutzki et al. 2001; Tonks et al. 2008). The FAO species identification sheets for the WIO (Fischer and Bianchi 1984) were used for shrimps and all fish were identified according to Smith and Heemstra (1998), Lieske and Myers (1994), and van der Elst (1981). Sub-samples from representative proportions of fish were weighed and measured using a fixed marked ruler on a flat board (TL, cm).

Data Collection for Artisanal Fish Catches

Shore-based artisanal catch assessments were undertaken between 2009 and 2011. In 2009 during the months of June, November and December; in 2010 (March, June and September), and in 2011 during the months of March, July and September along the three fishing areas of Malindi (39 assessments), Ngomeni (27 assessments) and Kipini (18 assessments) (Fig. 1) totalling to 49 shore visits and 84 samples covering both NEM and SEM seasons. Number of fishers and active fishing time (h) excluding navigation time to and from the fishing grounds were recorded. A total of 9,502 kg of fish was weighed during the entire exercise, and a sub-sample of 2,237 kg (24%) more than the recommended 10% representative proportion (Stobutzki et al. 2001; Tonks et al. 2008) was used for the enumeration of number of individuals per species, identification of species and TL measurements. Fish identification and measurements followed the same procedure as that for the trawl fish bycatches.

Data Analyses for Bottom Trawl Shrimps and Fish Bycatches

For the trawl surveys, shrimp and fish biomasses were calculated using the swept area method (Spar and Venema 1998).

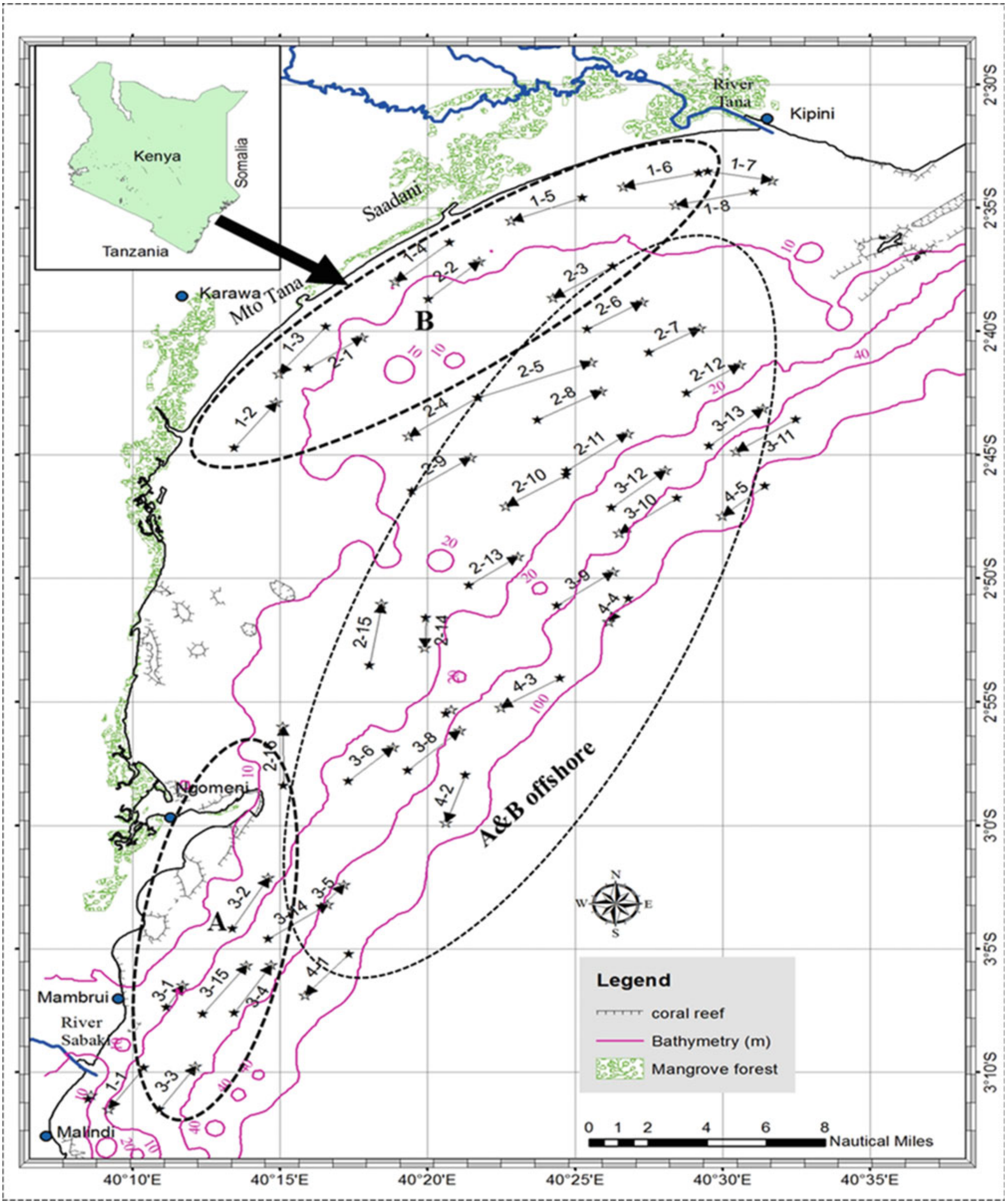

Fig. 1 A map of Malindi-Ungwana Bay, Kenya, showing the grouping of trawl stations at the Sabaki (*A*) and Tana (*B*) inshore area, and offshore (area *A & B*). Figures on the map indicate station number and depth stratum respectively. Station 1–2 means station No. 2 in depth stratum 1. Malindi, Ngomeni and Kipini were the fishing areas where artisanal catches were sampled

The swept area (a, nm^2) or 'effective path swept' for each tow was calculated as:

$$a = D \times h \times X$$

where D is the distance covered in nm ($D = 60 \times \sqrt{(Lat_1 - Lat_2)^2 + (Lon_1 + Lon_2)^2 \cos 0.5^2 (Lat_1 + Lat_2)}$), h is the length of the head-rope (m), and X is the fraction of the head-rope length equal to the width of the path swept by the trawl. The value of X was set at 0.5 in this study (Pauly 1980).

Shrimps and bycatch rates were calculated as catch (C, kg) divided by the time spent trawling (t, hrs) and converted to catch-per-unit-area (CPUA, kg/nm^2) by dividing by the swept area ($(C/t) / (a/t) = C/a$).

Total biomass (B, kg) was calculated from:

$$B = \frac{\left(\overline{C/a}\right) \times A}{X_1}$$

where C/a is the CPUA of all tows (kg/nm^2), A is the overall area under investigation (nm^2), and X_1 is the estimated proportion of shrimp or fish bycatch present in the area swept. We assumed that all shrimps would be captured, and that not all fish in the path of the tow would be captured (i.e. $X_1 = 1$ and $X_1 = 0.5$ respectively). The total biomass of shrimps and fish for the entire surveyed area (546.4 nm^2) was calculated from 41 tows made in the NEM season and 37 tows in the SEM season.

The multivariate non-metric multi-dimensional scaling (MDS) technique was used to identify whether geographical areas (inshore and offshore) or seasons (NEM and SEM) affected the community compositions of both trawl and artisanal catches based on Bray-Curtis similarity using PRIMER v6 (Clarke and Warwick 2001). The area and seasonal differences were further analysed by 2-way crossed ANOSIM with area and season as factors. The most influential species to the dissimilarity were identified using 2-way SIMPER. A 2-way ANOVA, followed by post hoc pair-wise comparison by Tukey HSD test, and test of homoscedascity of variance (Levene's test) was used to test for significant differences in trawl bycatch rates (kg/h) and biomass (CPUA) between areas and seasons. Catch rates for the artisanal fishery were defined as kg/fisher. h and 1-way ANOVA was used to test for significant differences between the seasons. Measures of diversity using species richness (S) and Shannon-Wiener diversity index (H') were calculated for fishes caught by shrimp trawlers and those caught by artisanal fishers, and these were compared between the two sectors for the two seasons using ANOVA. Sizes of fishes caught by shrimp trawlers were compared with those caught by artisanal fishers and differences tested using either the parametric 2-way ANOVA or non-parametric Kruskal-Wallis depending on homoscedascity of the data. All the parametric and non-parametric tests were done using STATISTICA v.

Results

Shrimp Distribution Patterns, Composition and Abundance

The MDS plots (Fig. 2) showed a distinct separation of shrimp species composition by geographical area and by depth, but not by season. Pair-wise comparison tests indicated species composition at 0–10 m depth differed significantly from those at 10–20 m and 20–40 m ($R = 0.337$; $p = 0.002$ and $R = 0.970$; $p = 0.001$ respectively), and that composition at 10–20 m differed from 20–40 m ($R = 0.248$; $p = 0.047$).

The difference in shrimp composition between areas was due to more abundant *P. semisulcatus* in area A (Sabaki; on average 82.2%), and more abundant *F. indicus* in area B (Tana; 52.8%; Table 1). By area, *P. semisulcatus* contributed the highest dissimilarity (36.6%) and *F. indicus* followed with 26.9%. The least contributing species to the dissimilarity were *M. monoceros, P. monodon* and *P. japonicus* (12.5%, 5.1% and 1.8% respectively). Two-way SIMPER analysis based on depth and season indicated that *F. indicus* was most abundant in 0–10 m (66.2%) and *P. semisulcatus* in 20–40 m depth (81.1%). Neither *F. indicus* nor *P. japonicus* were recorded at 20–40 m depth.

Seasonal differences in shrimp species composition were non-existent for *P. semisulcatus*, *F. indicus* and *P. japonicus,* but existed for more abundant *M. monoceros* during SEM and more abundant *P. monodon* during NEM (Table 1). The seasonal dissimilarity depended mostly on *F. indicus* (14.6%), followed by *M. monoceros* (11.8%) and *P. semisulcatus* (10.4%). *P. semisulcatus* contributed on average 90% (NEM) and 72% (SEM) by numbers to catches in the south of the bay, Malindi-Sabaki area A, followed by *M. monoceros* (6% in NEM and 25% in SEM). Five penaeid shrimp species were recorded in north of the bay, Tana area B in both seasons; *F. indicus* contributed 60% (NEM) and 48% (SEM), followed by *M. monoceros* (16% and 29%). *P. japonicus* was the least abundant, irrespective of area, depth or season.

The combined data for all shrimp species, including both seasons and all depths shallower than 40 m, indicated that shrimps were more abundant in the Tana area (3.76 kg/hr) than in the Malindi-Sabaki area (0.82 kg/hr). The overall shrimp catch rate and biomass during the SEM (6.17 kg/hr and 251 203 kg) were higher than during the NEM survey

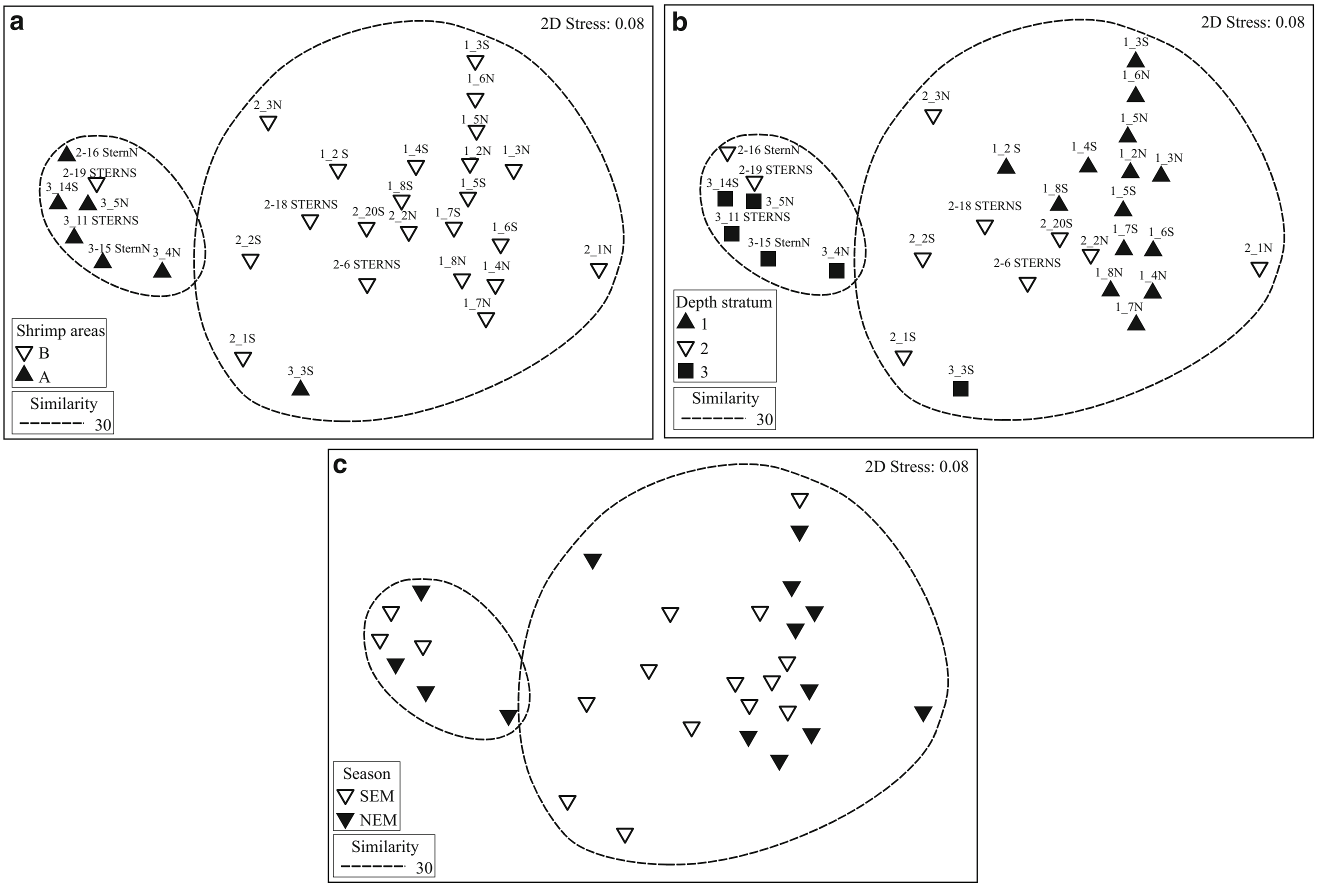

Fig. 2 Non-metric MDS plots (with indication of similarity levels of 30) showing the composition of shrimps by (**a**) area, (**b**) depth stratum and (**c**) season in the Malindi-Ungwana Bay, Kenya, based on shrimp species abundance for the combined Northeast Monsoon (NEM) and Southeast Monsoon (SEM) surveys

Table 1 Two-way SIMPER analysis: Shrimps species contributing to the dissimilarity in terms of abundance (%) by area (area A = Sabaki; area B = Tana) and by season (NEM = Northeast Monsoon survey; SEM = Southeast Monsoon survey) levels. The average dissimilarity was 82.9 and 45.7%, respectively

	Area analysis				Seasonal analysis			
	Abundance (avg. %)				Abundance (avg. %)			
Species	Area A	Area B	Dissim. (avg. %)	Contrib. (%)	NEM	SEM	Dissim. (avg. %)	Contrib. (%)
Penaeus semisulcatus	82.2	12.2	63.6	44.2	29.3	27.8	10.4	22.8
Fenneropenaeus indicus	0.0	52.8	26.9	32.4	42.6	38.7	14.6	31.9
Metapenaeus monoceros	13.9	23.4	12.5	15.0	13.3	28.1	11.8	25.8
Penaeus monodon	2.3	9.1	5.1	6.2	11.1	4.4	6.6	14.5
Penaeus japonicus	1.6	2.5	1.8	2.2	3.7	1.1	2.3	5.0

(1.45 kg/hr and 74 570 kg). In both surveys, biomass was greatest at the shallowest depth (0–10 m), and no shrimps were caught deeper than 40 m. Results of 2-way ANOVA indicated shrimp catch rates and biomass differed significantly between depths and seasons, and that the effect of the depth-season interaction was insignificant (<0.05; Table 2).

Comparison of Fish Composition Between Trawl Bycatches and Artisanal Catches

A total of 158 and 161 fish species from trawl bycatch against 90 and 148 species from artisanal catches were recorded during the NEM and SEM seasons respectively. Overall, trawl bycatches constituted a total of 223 species, and 177 species

Table 2 Results of 2-way ANOVA showing significant differences in shrimp catch rates (kg/h) and biomass (kg/nm^2) between seasons, depth strata and the interaction of season and depth stratum, in Malindi-Ungwana Bay, Kenya

			Catch rate (kg/h)		Biomass (kg/nm^2)	
Factors	Df	Error Df	F	p-value	F	p-value
Season	1	23	9.138	***0.006***	8.531	***0.008***
Depth stratum	2	23	4.397	***0.024***	3.872	***0.036***
Season × Depth stratum	2	23	1.748	0.197	1.670	0.210

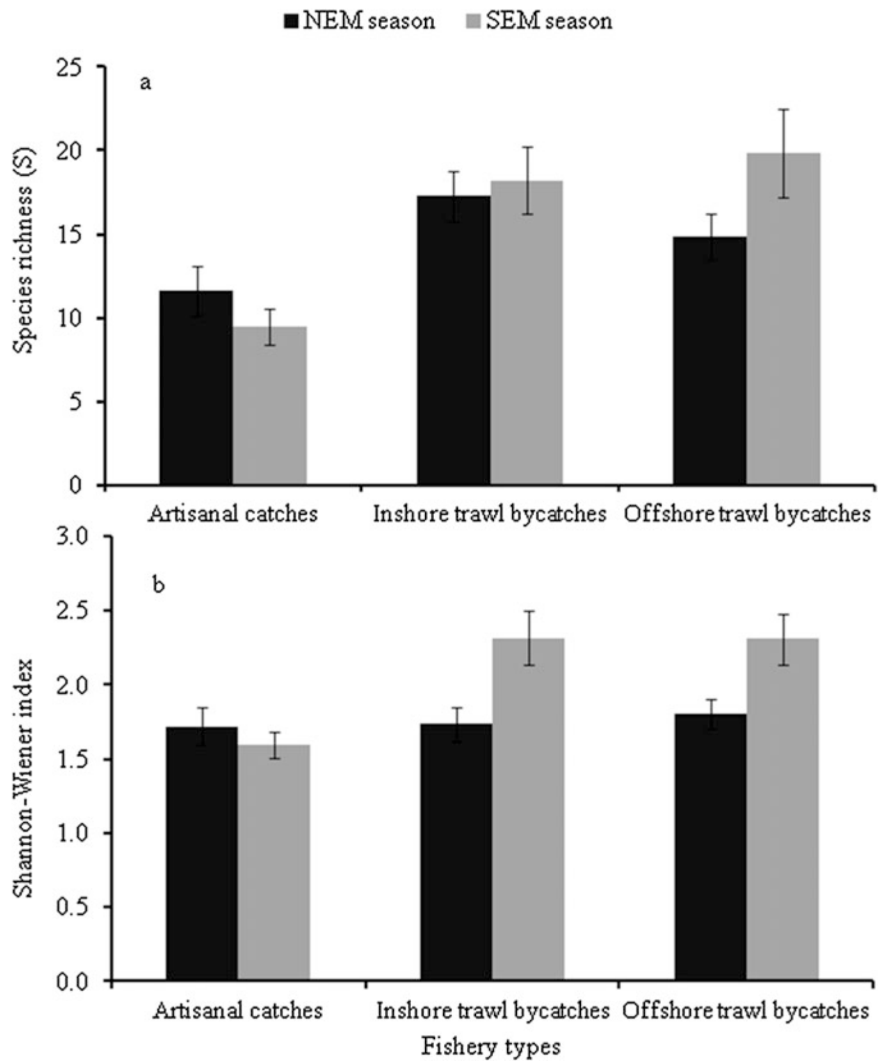

Fig. 3 Comparison of mean (±SE) species richness (**a**) and Shannon-Wiener diversity index (**b**) per sample between artisanal catches and trawl bycatches during the Northeast Monsoon (NEM) and Southeast Monsoon (SEM) seasons in Malindi-Ungwana Bay, Kenya

for the artisanal catches. Species richness (*S*) for artisanal catches was higher during the NEM season (on average 12 per sample) and lower during SEM season (on average 9 per sample), while for the inshore trawl bycatches, species richness was almost similar between the seasons (on average 18 and 17 per tow for the SEM and NEM respectively; Fig. 3a); and for the offshore bycatches, species richness was higher in SEM than NEM (on average 20 and 15 per tow respectively). The Shannon-Wiener diversity index (*H*') for the artisanal catches was almost similar between the seasons (on average 1.7 and 1.6 per sample for the NEM and SEM respectively), and for both the inshore and offshore bycatches, it was higher in SEM (on average 2.3 per tow each) and lower in NEM (on average 1.7 and 1.8 per tow respectively, Fig. 3b).

There was a significant difference in species richness between the fishery sectors (2-way ANOVA: $p < 0.05$) but not between seasons, nor was there a significant effect due to the interaction of fishery sector with season ($p > 0.05$, Table 3). Post hoc pair-wise comparison showed significantly higher species richness for the trawl bycatches in both seasons

Table 3 Results of 2-way ANOVA showing significant differences in finfish species richness between fishery types (trawl bycatches and artisanal catches) and significant differences in Shannon-Wiener diversity index between fishery types, seasons and the interaction of fishery type with season in the Malindi-Ungwana Bay, Kenya (p-value bold and italic are significant)

			Species richness (S)		Shannon-Wiener diversity index (H′)	
Factors	Df	Error Df	F	p-value	F	p-value
Area	2	149	14.718	***<0.001***	6.794	***0.002***
Season	1	149	0.834	0.363	8.178	***0.005***
Area × Season	3	149	2.726	0.069	5.089	***0.007***

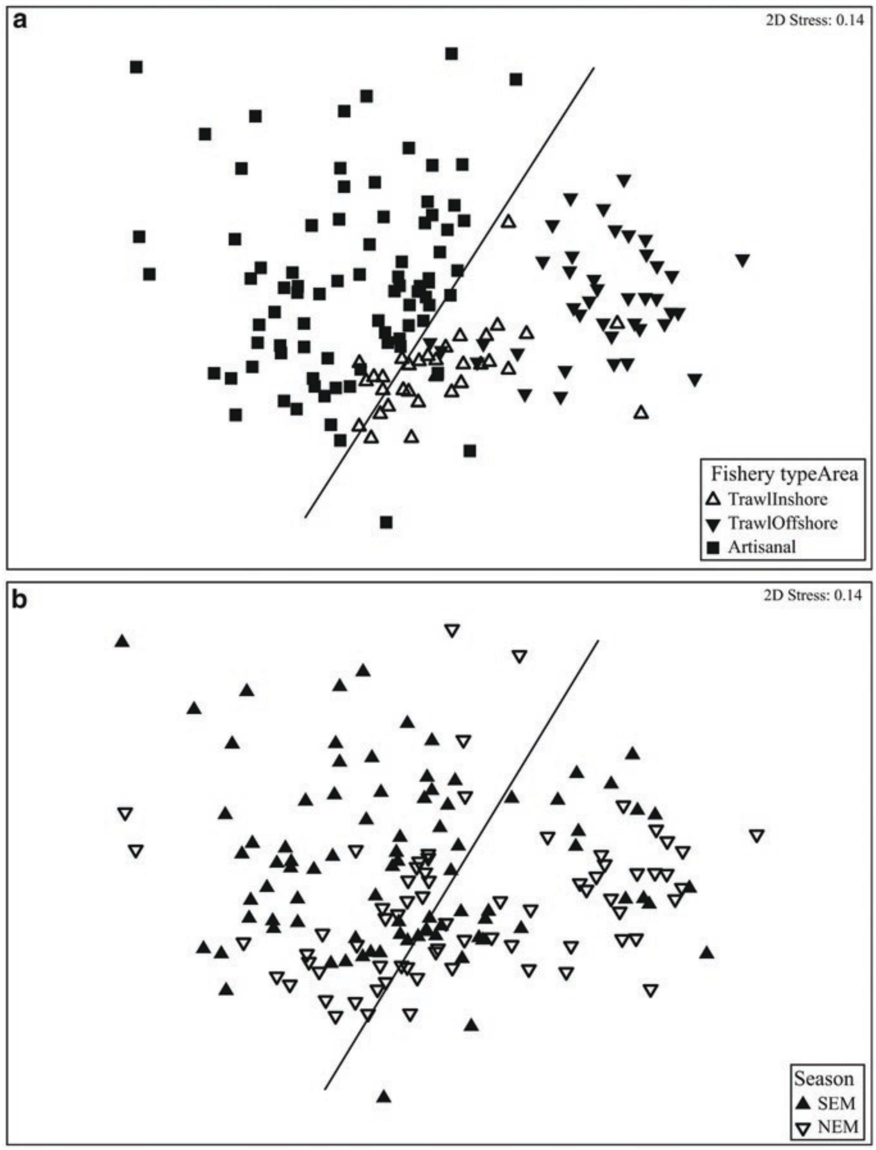

Fig. 4 Non-metric MDS plots showing the composition of fish catches in Ungwana Bay: (**a**) by fishing sector [artisanal, inshore and offshore trawl]; and (**b**) by season [NEM and SEM for combined artisanal and trawl bycatches]. The *dotted lines* separate artisanal from trawl bycatches

($p < 0.05$). The same test showed significant differences in Shannon-Wiener diversity index between the fishery sectors, seasons and a significant effect due to the interaction of fishery sector with season ($p < 0.05$, Table 3). Post hoc pair-wise comparison indicated significantly higher Shannon-Wiener diversity index for the inshore and offshore trawl bycatches during both seasons ($p < 0.05$).

The non-metric MDS plots (Fig. 4a) showed a distinct species composition between the artisanal catches and trawl bycatches (inshore and offshore), and to some extent between the seasons (Fig. 4b). There was a significant difference between the fishery sectors, and to a lesser extent between the seasons (2-way ANOSIM: $R = 0.317$; $p = 0.001$ and $R = 0.088$; $p = 0.003$ respectively). Pair-wise comparison tests showed the inshore trawl bycatches differed significantly from the offshore trawl bycatches (R = 0.631; p = 0.001), but not from the artisanal catches (R = 0.066; p = 0.090). Also the offshore trawl bycatches significantly differed from the artisanal catches (R = 0.460; p = 0.001). The differences in composition between offshore trawl bycatches and artisanal catches from the results of 2-way SIMPER was due to more abundant species of *Bothus mancus*, *Trachinocephalus myops*, *Callionymus gardineri* and *leiognathus lineolatus* in offshore trawl bycatches, and more abundant *Lobotes surinamensis*, *Lutjanus fulviflamma*, *Galeichthys feliceps*, *Psettodes erumei* and *Pellona ditchela* in artisanal catches. The seasonal differences in composition between the fishery sectors was due to more abundant *G. feliceps*, *P. ditchela*, *B. mancus*, *Thryssa vitrirostris* and *T. myops* in NEM, and more abundant *P. erumei* in the SEM season (2-way SIMPER analysis).

A total of seven common and most abundant fish species explained the similarity between inshore trawl bycatches *versus* artisanal catches (Table 4; 2-way SIMPER analysis). The relative abundance of these species was higher both in inshore trawl bycatches and artisanal catches than those in offshore trawl bycatches. Size comparison of these seven species between artisanal catches and trawl bycatches showed that except for *Lobotes surinamensis* all the species were significantly smaller in size for the trawl bycatches than for the artisanal catches ($p < 0.05$), while seasonal differences in sizes were only significant for *L. surinamensis* and *Leiognathus equulus* ($p > 0.05$, Table 5).

Table 4 Two-way SIMPER analysis: Species contributing most to similarity in terms of abundance (%) between inshore trawl bycatches (within similarity of 23.3%) and artisanal catches (within similarity of 9.3%) in Ungwana Bay, Kenya

Species	Average abundance	Average similarity	% contribution
Inshore trawl bycatches			
Galeichthys feliceps	14.65	5.27	22.59
Pellona ditchela	9.12	2.79	11.97
Johnius amblycephalus	6.68	1.95	8.35
Leiognathus equulus	3.54	1.30	5.57
Pomadasys maculatus	4.05	1.10	4.71
Otolithes ruber	2.36	0.84	3.61
Lobotes surinamensis	0.95	0.22	0.96
Artisanal catches			
Lobotes surinamensis	7.52	1.40	14.98
Galeichthys feliceps	5.20	0.80	8.61
Pellona ditchela	4.09	0.70	7.45
Otolithes ruber	3.90	0.58	6.23
Pomadasys maculatus	2.50	0.30	3.17
Leiognathus equulus	1.24	0.13	1.44
Johnius amblycephalus	1.15	0.12	1.33

Table 5 Mean total lengths (cm ± SE) of the most abundant artisanal target fish species which occurred in trawl bycatches during the Northeast Monsoon (NEM) and Southeast Monsoon (SEM) seasons in Malindi-Ungwana Bay, Kenya with trawl bycatches indicating significantly smaller individuals than those in artisanal catches ($p < 0.05$, bold and italic)

Species	Artisanal	Trawl	N/Error Df	Statistic	p-value
Galeichthys feliceps	39.8 ± 1.3	20.5 ± 0.3	357	227.171	***<0.001***
Johnius amblycephalus	14.4 ± 1.8	11.4 ± 2.2	228	51.819	***<0.001***
Pellona ditchela	14.8 ± 0.4	13.6 ± 0.1	787	8.272	***0.004***
Lobotes surinamensis	56.2 ± 0.9	55.1 ± 1.7	298	3.045	0.082
Otolithes ruber	24.3 ± 0.3	18.9 ± 0.2	380	165.400	***<0.001***
Leiognathus equulus	12.5 ± 0.2	13.3 ± 0.1	448	19.218	***<0.001***
Pomadasys maculatus	21.9 ± 0.6	12.9 ± 0.1	289	299.596	***<0.001***
	NEM season	SEM season			
Galeichthys feliceps	25.8 ± 0.7	24.4 ± 0.7	357	0.129	0.719
Johnius amblycephalus	11.9 ± 2.5	11.8 ± 2.2	228	0.960	0.328
Pellona ditchela	14.4 ± 0.4	14.0 ± 0.1	787	0.002	0.968
Lobotes surinamensis	59.4 ± 1.3	53.2 ± 1.0	298	12.823	***<0.001***
Otolithes ruber	21.4 ± 0.3	20.9 ± 0.3	380	1.093	0.296
Leiognathus equulus	13.4 ± 0.2	12.7 ± 0.1	448	13.349	***<0.001***
Pomadasys maculatus	17.1 ± 0.4	16.6 ± 0.5	289	2.857	0.910

Discussion

Shrimp Distribution Patterns, Composition and Abundance

The distribution of shallow-water penaeid shrimps was restricted to the nearshore areas of the Sabaki and Tana estuaries of the Malindi-Ungwana Bay, and no shrimps were caught further offshore (Fig. 1). The species composition and abundance patterns differed between these estuaries: all five shrimp species were recorded at the Tana estuary in both the NEM and SEM seasons, whereas only three species (*P. semisulcatus*, *M. monoceros* and *P. monodon*) were recorded at the Sabaki estuary during the SEM. *Fenneropenaeus indicus* was the most abundant species at the Tana estuary, coinciding with the more turbid environment. Turbid waters in Maputo Bay, Mozambique also coincided with areas of high *F. indicus* catches by commercial trawlers, and turbidity also affected the distribution of *F. indicus* and *M. monoceros* at Saco da Inhaca (Macia 2004). Juvenile *F. indicus* and *M. monoceros* inhabited turbid waters with reduced visibility to escape predators (Macia 2004; de Freitas 2011). *F. indicus* in the present study was not recorded in the less turbid and deeper Sabaki estuary.

Penaeus semisulcatus dominated shrimp catches in the Sabaki estuary, and previous studies from the Western Indian Ocean (WIO) region showed that this species prefers low turbidity, muddy substrates and deeper water, where it is often associated with sea grass meadows (Macia 2004; Forbes and Demetriades 2005; de Freitas 2011). *P. semisulcatus* is a naturally burrowing species during daytime, but feeds during the night when it can be fished more successfully (Hughes 1966; Vance et al. 1994; de Freitas 1986; 2011). Post-larval and young adult *P. semisulcatus* are often associated with submerged macrophytes, especially in estuarine backwaters, and adults prefer deeper waters (3–20 m) in large bays and offshore shelf areas (de Freitas 1986, 2011). Macia (2004) observed that *P. semisulcatus* preferred deeper water bays compared to *F. indicus*; our findings agree with this observation. *P. monodon*, *M. monoceros* and *P. japonicus* inhabited both Tana and Sabaki areas, suggesting that they have a broader tolerance to factors that may limit *F. indicus* distribution in the bay. Forbes and Demetriades (2005) also suggested that *M. monoceros* can inhabit diverse habitats, from areas with submerged macrophytes to deeper reaches of mangrove swamps in low salinity environments.

The relatively shallow depth associated with sandy bottom and high turbidity, especially during the SEM season, favoured the existence of higher shrimp biomass at the Tana, compared to the Sabaki estuary. Fulanda et al. (2011) and Munga et al. (2012) also reported higher shrimp catch rates at the Tana estuary during the SEM than NEM season, using longer term commercial bottom trawl data. Similar seasonal variation in shrimp catch rates were also reported for the Tanzanian commercial bottom trawl and artisanal fisheries (Semesi et al. 1998; Teikwa and Mgaya 2003).

Composition of Trawl Bycatches and Artisanal Catches

The Malindi-Ungwana Bay is sub-divided into use zones according to the shrimp fishery management plan 2010 (Fulanda et al. 2011; Government of Kenya 2014). The 0–3 nm is the inshore area designated for the artisanal fisheries exploitation and therefore, excludes shrimp trawling. The zone beyond 3 nm offshore is the area designated for bottom trawling activities. Before the trawling ban in 2006, trawlers contravened this zonation and trawled in the inshore area and therefore, resulted into user conflict with the artisanal fishery. In this present study we use the terms inshore and offshore areas to identify resource-use overlap between the two fishery sectors that could have necessitated the conflict. Both agriculture and fisheries are important food sectors within the Tana River County where the largest section of the Malindi-Ungwana Bay falls. Agriculture production in this county has over the years been affected by unfavourable weather patterns of long dry spells and flush floods. Artisanal fishing effort is therefore, expected to increase in the bay (Government of Kenya 2014), and the implementation and enforcement of the existing management plans are therefore crucial if conflict is to be reduced.

Catch rate and abundance comparisons between artisanal and trawl bycatch data are relative only, because of different methods of calculation, and of collecting data. Results indicated that trawl bycatch rates and biomass decreased from inshore to offshore areas of the estuaries in the bay. This pattern is reflected due to differences in depth distribution by individual fish species. Studies of differences in spatial distributions of bycatch species showed preferences for six species of elasmobranchs on Tugela Bank of South Africa (Fennessy 1994). Similarly, bycatches of flatfishes (Paralichthyidae) were distributed over a broader spatial range than the Pleuronectidae in the Gulf of California (Rabago-Quiroz et al. 2008). The bycatch of *Galeichthys feliceps, Pellona ditchela, Johnius amblycephalus, Leiognathus equulus, Pomadasys maculatus, Lobotes surinamensis* and *Otolithes ruber* were more abundant in the inshore area (Tana and Sabaki estuaries), where they also form a target catch of the artisanal fishery and therefore, the source of resource-use overlap with the bottom trawling. Conversely *Trachinocephalus* *myops* and *Bothus mancus*, less targeted in the artisanal fishery, were more abundant in offshore waters of these estuaries.

Apart from catching a high diversity of fish bycatch, tropical shrimp trawl fisheries are also associated with large volumes of bycatch that consist mostly of undersized and immature individuals. This present study was no exception. Six of the seven most common and abundant fish species were of significantly smaller-sized individuals in the inshore trawl bycatch than in the artisanal catches. With continued and intensive trawling especially in the inshore area, such affected bycatch species which are otherwise a target in the artisanal fishery, are possibly given less time to recruit before capture or may be totally depleted. This scenario may possibly explain why the Malindi-Ungwana Bay artisanal fishery after a long period of trawling activity before the trawling ban in 2006, had started experiencing reduced artisanal catches (Munga et al. 2012), and this needs to be considered for further management. In addition, the small-sized individuals of a majority of trawl bycatches especially in offshore, were composed of low commercial value species of *Bothus mancus, Callionymus gardineri, Aluteres monoceros* and *Apogon fasciatus* as found in the Gulf of California (Rabago-Quiroz et al. 2008). These authors reported the majority of trawl bycatch fish species sampled in a survey off the Gulf of California to be mostly small-sized individuals ranging between 6 – 18 cm in total length. Since tropical shrimp trawl bycatch species richness is high, coupled with many small-sized and juvenile individuals, there is a high risk of reduced species diversity and to some extent disappearance of certain species, as observed by Chong et al. (1987) when assessing the effects of a 1978 sustained ban on trawling in an Indonesian shrimp fishery. So far in the Sabaki and Tana estuaries of the Malindi-Ungwana Bay, no single study has established a complete disappearance of species due to the impact of trawling, but reduced catches in the artisanal fishery before the trawl ban in 2006 have been confirmed to some extent (Munga et al. 2012). In order to avoid this risk of biodiversity loss, emphasis on the use of effective Bycatch Reduction Devices (BRDs) that allow escape of small-sized and juveniles should be made mandatory in the estuaries of the bay. This is in addition to restriction of bottom trawling activity within the inshore area.

Conclusions

Shrimp abundance in the Tana and Sabaki estuaries of the Malindi-Ungwana Bay is concentrated near the outflows of the estuaries, and these two estuaries have distinct species compositions, with *F. indicus* dominating in the Tana estuary and *P. semisulcatus* in the Sabaki estuary. The total biomass of shrimps and fish bycatch decreased with increasing depth, and was higher during the wet SEM than the dry NEM season.

The inshore areas of the Sabaki and Tana estuaries of the bay, are accessible to an increasing number of artisanal fishers (Government of Kenya 2014), and are richer in fish abundance and diversity than the offshore areas. The most abundant and affected fish species which are also a target by the artisanal fishery were *G. feliceps, P. ditchela, J. amblycephalus, L. equulus, P. maculatus, L. surinamensis* and *O. ruber*. Coincidentally the inshore areas of these estuaries also harbour abundant shrimps (Munga et al. 2013), thereby confirming the existing resource-use conflict. Therefore, in order to avoid this conflict in the estuaries of the bay, the stipulated measures in the management plan of minimum trawling distance of ≥3 nm offshore, closed trawling season, and the mandatory use of BRDs and Turtle Excluder Devices (TEDs) should be emphasised, in addition to continued prohibition of night trawling in order to achieve sustainable utilisation of fisheries resources. Continued monitoring of fish trawl bycatch quantities and species diversity is however, recommended for both estuaries so as to get a clearer spatio-temporal pattern for effective management.

References

Chong KC, Dwponngo A, Ilyas S, Martosubroto P (1987) Some experiences and highlights of the Indonesian trawl ban: bioeconomics and socio-economics. Indo Pac Fish Comm RAPA Rep 10:458–477

Cinner JE, McClanahan TR, Graham NAJ, Pratchett MS, Wilson SK, Raina JB (2009) Gear-based fisheries management as a potential adaptive response to climate change and coral mortality. J Appl Ecol 46:724–732

Clarke KR, Warwick RM (2001) Change in marine communities: an approach to statistical analysis and interpretation, 2nd edn. PRIMER-E, Plymouth, UK, 172 p

de Freitas AJ (1986) Selection of nursery areas by six Southeast African Penaeidae. Estuar Coast Shelf Sci 23:901–908

de Freitas AJ (2011) The Penaeoidea of South Africa IV – the family Penaeidae: genus *Penaeus*. Oceanographic Research Institute investigational report no. 59, 125 p

Fennessy ST (1994) Incidental capture of elasmobranchs by commercial prawn trawlers on the Tugela Bank, Natal, South Africa. S Afr J Mar Sci 14:287–296

Fennessy ST, Mwatha GK, Thiele W (eds) (2004) Report of regional workshop on approaches to reducing shrimp trawl by-catch in the Western Indian Ocean, Mombasa, Kenya. FAO fisheries report 734. Food and Agriculture Organisation of the United Nations, Rome, 13–15 April 2003, 49 p

Fennessy ST, Vincent X, Budeba Y, Mueni E, Gove DZ (2008) An update on initiatives to reduce prawn trawl by-catch in the Western Indian Ocean. West Indian Ocean J Mar Sci 7(2):217–222

Fischer W, Bianchi G (1984) FAO species identification sheets for fishery purposes. Western Indian Ocean Fishing Area 51, vol V. FAO, Rome

Forbes AT, Demetriades NT (2005) A review of the commercial, shallow water penaeid prawn resource of South Africa: status, fisheries, aquaculture and management. Specialist report prepared for Ezemvelo KZN Wildlife, South Africa. Marine and Estuarine Research, 64 p

Fulanda B, Munga C, Ohtomi J, Osore M, Mugo R, Hossain MY (2009) The structure and evolution of the coastal migrant fishery of Kenya. Ocean Coast Manag 54:401–414

Fulanda B, Ohtomi J, Mueni E, Kimani E (2011) Fishery trends, resource-use and management system in the Ungwana Bay fishery, Kenya. Ocean Coast Manage 54:401–414

Gobert B (1994) Size structures of demersal catches in multispecies multigear tropical fishery. Fish Res 19:87–104

Government of Kenya (2002) Kenya 1999 population and housing census. Central Bureau of Statistics, Ministry of Finance and Planning, Nairobi

Government of Kenya (2008) Kenya state of the coast report: towards the integrated management of Kenya's coastal and marine resources. UNEP and NEMA, Nairobi, 90 p

Government of Kenya (2010) Fisheries annual statistical bulletin 2010. Ministry of Fisheries Development, Department of Fisheries, Nairobi, 56 p

Government of Kenya (2014) Marine waters fisheries frame survey 2014 report. Ministry of Fisheries Development, State Department of Fisheries, Nairobi, 88 p

Hughes DA (1966) Investigations of the "nursery areas" and habitat preferences of juvenile penaeid prawns in Mozambique. J Appl Ecol 3:349–354

Iversen SA (1984) Kenyan marine fish resources in waters deeper than 10 meters investigated by R/V "DR. FRIDTJOF NANSEN". In: Iversen SA, Myklevoll S (eds) Proceedings of the NORAD-Kenya seminar to review the marine fish stocks and fisheries in Kenya, Mombasa, Kenya, 13–15 March 1984

Jiddawi NS, Öhman MC (2002) Marine fisheries in Tanzania. Ambio 31:518–527

Kitheka JU (2013) River sediment supply, sedimentation and transport of the highly turbid sediment plume in Malindi Bay, Kenya. J Geogr Sci 23(3):465–489

Kitheka JU, Obiero M, Nthenge P (2005) River discharge, sediment transport and exchange in the Tana estuary, Kenya. Estuar Coast Shelf Sci 63:455–468

Lieske E, Myers R (1994) Collins pocket guide to coral reef fishes: Indo-pacific and Caribbean. Herper Collins Publisher, London, 400 p

Macia A (2004) Juvenile penaeid shrimp density, spatial distribution and size composition in four adjacent habitats within a mangrove-fringed bay on Inhaca Island, Mozambique. West Indian Ocean J Mar Sci 3:163–178

McClanahan TR (1988) Seasonality in East Africa's coastal waters. Mar Ecol Prog Ser 44:191–199

McClanahan TR, Mangi SC (2004) Gear-based management of a tropical artisanal fishery based on species selectivity and capture size. Fish Manage Ecol 11:51–60

Munga CN, Ndegwa S, Fulanda B, Manyala J, Kimani E, Ohtomi J, Vanreusel A (2012) Bottom shrimp trawling impacts on species distribution and fishery dynamics; Ungwana Bay fishery Kenya before and after the 2006 trawl ban. Fish Sci 78:209–219

Munga CN, Mwangi S, Ong'anda H, Ruwa R, Manyala J, Groeneveld JC, Kimani E, Vanreusel A (2013) Species composition, distribution patterns and population structure of penaeid shrimps in Malindi-Ungwana Bay, Kenya, based on experimental bottom trawl surveys. Fish Res 147:93–102

Munga CN, Omukoto JO, Kimani EN (2014) Propulsion-gear-based characterisation of artisanal fisheries in the Malindi-Ungwana Bay, Kenya and its use for fisheries management. Ocean Coast Manage 98:130–139

Mutagyera WB (1984) Distribution of some deep water prawn and lobster species in Kenya's waters. In: Iversen SA, Myklevoll S (eds) The proceedings of the NORAD-Kenya seminar to review the marine fish stocks and fisheries in Kenya, Mombasa, Kenya, 13–15 March 1984

Mwatha GK (2005) Stock assessment and population dynamics of penaeid prawns in the prawn trawling grounds of Malindi-Ungwana Bay: the challenges of managing the prawn fishery in Kenya, WIOMSA/MARG-I/2005-06, 21 p

Pauly D (1980) On the interrelationships between natural mortality, growth parameters and mean environmental temperature in 175 fish stocks. Journal du Conseil international pour l'Exploration de la Mer 39:175–192

Rabago-Quiroz CH, Lopez-Martinez J, Herrera-Valdivia E, Navare-Martinez MO, Rodriguez-Romero J (2008) Population dynamics and spatial distribution of flatfish species in shrimp trawl bycatch in the Gulf of California. Hidrobiologica 18(3):177–188

Saetersdal G, Bianchi G, Stromme T, Venema C (1993) The Dr. Fridtjof Nansen Programme 1975–1993. Investigations of fishery resources in developing countries. History of the programme and review of results. FAO fisheries technical paper no. 391. FAO, Rome, 434 p

Semesi AK, Mgaya YD, Muruke MHS, Francis J, Mtolera M, Msumi G (1998) Coastal resources utilisation and conservation issues in Bagamoyo. Ambio 27:635–644

Smith JLB, Heemstra R (1998) Smith's sea fishes, 4th edn. Valiant Publishing, Santon, 578 p

Spar P, Venema SC (1998) Introduction to tropical fish stock assessment: part 1 manual. FAO fisheries technical paper 306/1. FAO, Rome, 407 p

Stobutzki I, Miller M, Brewer D (2001) Sustainability of fishery bycatch: a process for assessing highly diverse and numerous bycatch. Environ Conserv 28:167–181

Teikwa ED, Mgaya YD (2003) Abundance and reproductive biology of the Penaeid prawns of Bagamoyo coastal waters, Tanzania. West Indian Ocean J Mar Sci 2:117–126

Tonks ML, Griffiths SP, Heales DS, Brewer DT, Dell Q (2008) Species composition and temporal variation of prawn trawl bycatch in the Joseph Bonaparte Gulf, northwesten Australia. Fish Res 89:276–293

Tychsen J (ed) (2006) KenSea. Environmental sensitivity atlas for coastal areas of Kenya. Geological Survey of Denmark and Greenland (GEUS), Copenhagen, 76 pp. ISBN 87-7871-191-6

van der Elst RP (1981) A guide to the common sea fishes of Southern African. Struik Fisher Publishers, Cape Town, 398 p

van der Elst RP, Everret BI, Jiddawi N, Mwatha G, Alfonso PS, Boulle D (2005) Fish, fishers and fisheries of the Western Indian Ocean; their diversity and status, a preliminary assessment. Philos Trans R Soc Lond A 363:263–284

Vance DJ, Heales DS, Loneragan NR (1994) Seasonal, diel and tidal variation in beam-trawl catches of juvenile grooved tiger prawns, *Penaeus semisulcatus* (Decapoda: Penaeidae), in the Embley River, north-eastern Gulf of Carpentaria, Australia. Aust J Mar Freshw Res 45:35–42

Venema SC (1984) Review of marine resource surveys in Kenyan waters. In: Iversen SA, Myklevoll S (eds) Proceedings of the NORAD-Kenya seminar to review the marine fish stocks and fisheries in Kenya, Mombasa, Kenya, 13–15 March 1984

WWF (2004) Eastern African Marine Ecoregion. Towards a western Indian Ocean dugong conservation strategy: the status of dugongs in the western Indian Ocean region and priority conservation actions. Dar es Salaam, Tanzania, WWF, 68 p

Are Peri-Urban Mangroves Vulnerable? An Assessment Through Litter Fall Studies

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Abstract

The productivity of an over-exploited and sewage polluted peri-urban mangrove was assessed through litter fall studies to establish vulnerability to human actions and climate change. Litter from three common mangrove species, *Rhizophora mucronata* Lam. (Rhizophoraceae), *Sonneratia alba* Sm. (Sonneratiaceae), and *Avicennia marina* (Forssk.) Vierh. (Avicenniaceae) were monitored over a period of two years. The mean annual litter fall was estimated at 12.16 ± 2.89 t $ha^{-1}yr^{-1}$. Litter fall was seasonal in both content and quantity, with high rates occurring in the dry North Easterly Monsoon (NEM) season, January-April (ca. 5.10 ± 1.36 g DW m^{-2} day^{-1}) and lower rates in the cool and wet South Easterly Monsoon (SEM) season, June-October (ca. 2.53 ± 0.47 g DW m^{-2} day^{-1}). Litter fall varied significantly between species, *R. mucronata* recording the highest annual rate (15.34 ± 3.34 t $ha^{-1}yr^{-1}$), with no significant difference between *A. marina* and *S. alba*, (11.44 ± 2.90 and 9.69 ± 5.26 t $ha^{-1}yr^{-1}$ respectively). Sewage exposure did not affect litter fall rates for all species, but affected leaf nutrient content as expressed by the leaf $\delta^{15}N$ signature. A strong correlation between leaf C:N ratio and leaf $\delta^{15}N$ signature was observed, indicating a more open N cycle, favouring $\delta^{15}N$ accumulation. Sewage exposure therefore does not necessarily translate into elevated productivity in mangroves, but causes alteration of leaf nutrient content depending on species. Prevailing climatic conditions however, may influence litter fall and thus phenology and health of the system. The vulnerability of the Tudor Creek mangroves to climate change may be ranked high due to low production (20%<) of reproductive materials and a die back of seaward fringe *S. alba* stand.

Keywords

Phenology • Peri-urban • Sewage • Vulnerability • $\delta^{15}N$ • Climate

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Introduction

In forests, litter fall from deciduous and evergreen trees is a mechanism by which nutrients are returned to the sediments. Leaves, which form 70% of the dry mass of all aboveground litter in forested ecosystems, are the most important litter component (O'Neill and DeAngelis 1981). In mangrove forests, leaf litter can account for 40–95% of the total pool of litter fall (e.g. Day et al. 1987; Wafar et al. 1997). According to Benner and Hodson (1985), this fraction of mangrove litter represents a relatively large, potentially labile source of organic matter to decomposer com munities. Therefore, intertidal mangrove forests form some of the most productive tropical coastal ecosystems. They are characterized by high productivity, high biomass and litter production compared to terrestrial ecosystems (Twilley et al. 1986; Saenger and Snedaker 1993; Bouillon et al. 2007b; Chen et al. 2009).

High rates of primary production and rapid decomposition of the mangrove litter was hypothesized to support a trophic link between mangrove ecosystems and adjacent habitats, providing an important food and energy source for a variety of aquatic organisms (Odum and Heald 1972). However, a number of studies using natural tracer techniques (e.g. stable isotopes, fatty acids) have failed to prove such a link (Bouillon et al. 2008; Nagelkerken et al. 2008). Although faunal communities may have a profound effect on litter dynamics (Lee 1998), the role of mangrove litter in sustaining epibenthic communities is often fairly limited and decreases when systems are more open with respect to material exchange with adjacent systems (Bouillon et al. 2004a, b). However, litter fall is a commonly measured functional aspect of mangrove forests world-wide. It is used as a proxy measure of net primary production, and dynamically reflects phenological events occurring in the canopy (Twilley et al. 1986; Duke 1990). Furthermore, litter fall is an important element in the calculation of energy and nutrient fluxes in mangrove ecosystems representing a modest proportion of the carbon fixed by trees (Li 1997; Wafar et al. 1997; Bouillon et al. 2008).

Although recognized as economically and ecologically important coastal biotopes, resilient to natural catastrophes, mangroves are rapidly disappearing due to anthropogenic impacts such as oil spills, overharvesting due to lumbering, removal for construction of fish and shrimp ponds, pollution and general urban development (Fortes 1988; Duke et al. 2007; Walters et al. 2008; Nagelkerken et al. 2008). Litter fall studies present a useful means in comparing mangrove communities, the possible export of materials into adjacent subtidal communities, and support of food webs (Mackey and Smail 1995). The level of litter fall in intertidal forests has been variably attributed to stress (Pool et al. 1977), habitat optimization (Saenger and Snedaker 1993), and tidal flushing (Twilley et al. 1986).

The peri-urban mangroves of Mombasa, situated within the Tudor Creek, are recipients of sewage-polluted rivers and flash-flood waters. These mangroves are used for sewage dumping, with possible risk to marine ecosystems and public health, though no specific structural attributes have been correlated with sewage exposure (Mohamed et al. 2009). Limited studies have shown that water quality in Creeks around Mombasa is poor with faecal coliform exceeding safe limits (Mwaguni and Munga 1997; Mwangi et al. 1999). The ever increasing human population in the coastal zone, with the lack of sewage treatment facilities, mangroves are likely to be more and more affected by raw sewage discharge. This threat of domestic wastewater discharge to sustainable coastal development is defined as a key global issue (UNEP/GPA 2001; UNEP-WCMC 2006). Additionally, siltation and overharvesting has influenced the structure and regeneration of the Tudor Creek mangroves (Mohamed et al. 2009). Considering the limited studies on peri-urban mangroves globally, this study estimates the annual litter fall and trends. From these estimates, we assess the vulnerability of the forest by establishing phenological trends and the impacts of sewage exposure and nutrient loading through leaf N and δ^{15}N composition. The δ^{15}N leaf signature has been utilised as an indicator of anthropogenic nutrient loading into mangrove marshes and has been recommended as a good biomonitor of N loading (Fry et al. 2000; Bouillon et al. 2008). Declines in mangrove condition can also be inferred from litter fall studies (Ellison 2012). When trees are stressed, they reduce flowering and fruiting, which is commonly 20 to 30 percent of productivity (Ellison 2012). It is envisaged that this may provide insights on the future of the forest, and its capacity to adopt to climate change.

Materials and Methods

Study Area

The floristic composition of Tudor Creek mangrove has been described by SPEK (1992), and more recently by Mohamed et al. (2009). The tidal dynamics were described by Nguli (2006). Briefly, Tudor Creek has a single narrow sinuous inlet with a mean depth of 20 m, that broadens out further inland to a central relatively shallow basin (5 m) fringed by a well developed mangrove forest mainly composed of *R. mucronata*, *A. marina* and *S. alba*. The basin has an area of 6.37 km^2 at low water spring and 22.35 km^2 at high water spring. Mangrove forests occupy 8 km^2 of the Creek. The forest resembles the fringing mangroves described by Lugo and Snedaker (1974), with strong inward tidal current during the high tides which reverses during ebb tides, attaining maximum tide velocities of 0.6–0.7 ms^{-1} (Nguli 2006),

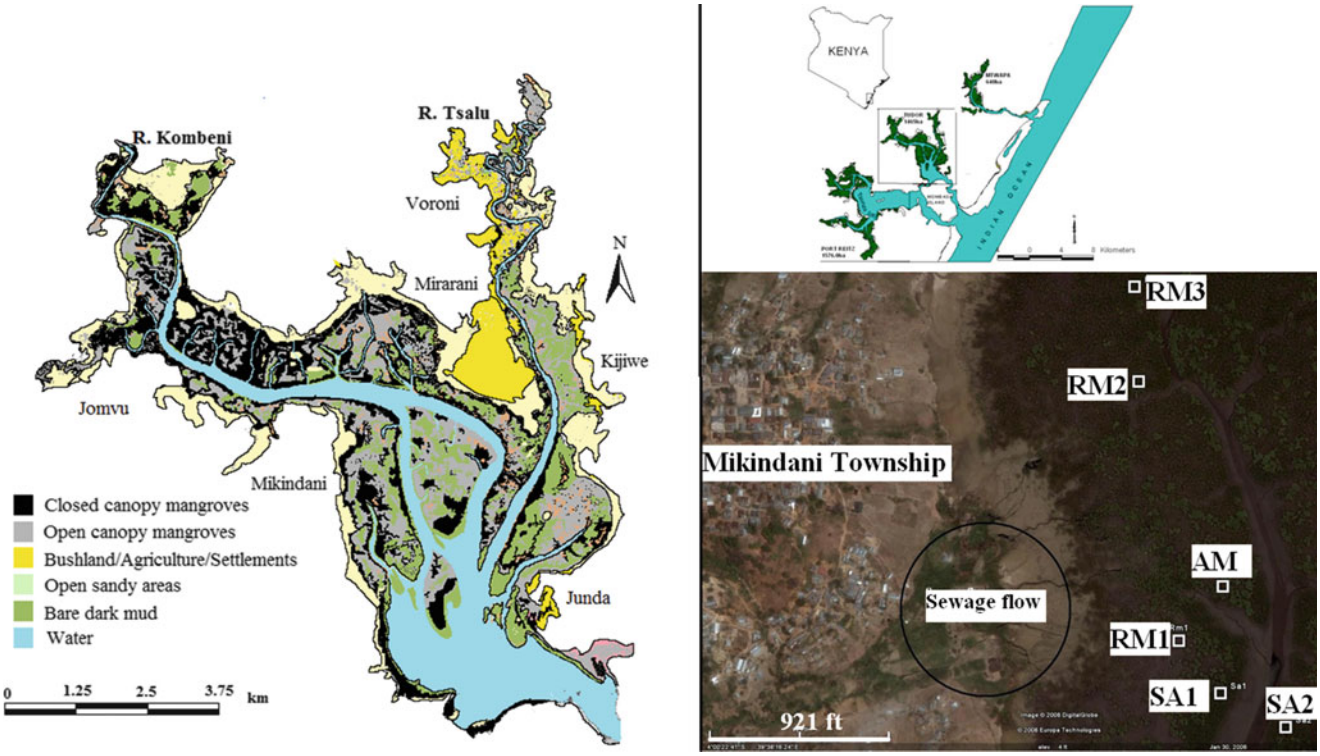

Fig. 1 Map showing the location of litter traps. *R. mucronata* plots are labelled as Rm1 for direct or full sewage exposure, and Rm2 and Rm3 for indirect exposures. *S. alba* is labelled as Sa1 for direct exposure and Sa2 for indirect exposure, and Am for A. marina (Map scale 1:25000)

and the dense, well-developed prop roots that accumulate large stocks of debris, with a spring tidal range of 3.5 m and a neap tidal range of 1.1 – 1.3 m (Fig. 1).

The mangroves of Tudor Creek are separated naturally by two main tidal Creeks, Kombeni and Tsalu, 4.5 and 3 km long respectively cutting through the mangroves connecting to the upstream rivers. The Creek is characterised by diurnal flooding, with a flushing time of 13 days that causes complete exchange of waters within the Creek (Nguli 2006). This study focuses on the Mikindani area, a township with about 67,164 people located in the west mainland of Mombasa district. In this area, raw sewage drains directly into the mangroves via an open surface flow system. The sewage runs through the mangrove forest in canals and has been discharged into the Tudor Creek waters for more than a decade. The mangroves are doused with sewage every tidal cycle, with the loading exponentially reducing with distance from the source. It is estimated that about 1,200 kg of nitrogen and 5.5 kg of phosphorus are discharged via sewage into the Mikindani system every day (PUMPSEA 2007).

The climate of Mombasa is influenced by the semi annual passage of the inter-tropical convergence zone (ITCZ) and the monsoons. The North Easterly Monsoon (NEM) occurs from December to March, while the South Easterly Monsoon (SEM) from May to October. Most of the rainfall occurs between the monsoons when convention is enhanced. The mean annual rainfall is 1,038 mm with the months of April, May and June recording the maximum. Average annual temperatures for the two seasons are 23.9 °C and 28.5 °C respectively (Fig. 2).

Litter Fall

Six selected plots measuring 20 m by 20 m were identified according to the sewage exposure gradient. Sites located at the point source were designated as direct exposed and further from the point source as indirect exposed (flushed by tidal flooding with dilution of sewage). Litter fall data were obtained from a total of 60 traps (10 traps per plot). Selected sites included three *R. mucronata* plots, two *S. alba* plots, and one *A. marina* plot. The number of plots was based on the rationale that *R. mucronata* was dominant and widely distributed, and like *S. alba,* occurred in both direct sewage exposed and indirect exposed sites, while *A. marina* was distributed mainly in direct sewage exposed sites. All plots were inundated daily during flood tides. Litter traps were made of round metal frames of 0.25 m^2 mouth area to which a conical fabric net (mesh size 2 mm) was attached. Traps were positioned above the high water level and emptied monthly. Litter was sorted into: leaves, flowers, wood (twigs, bark and debris), reproductive materials (propagules

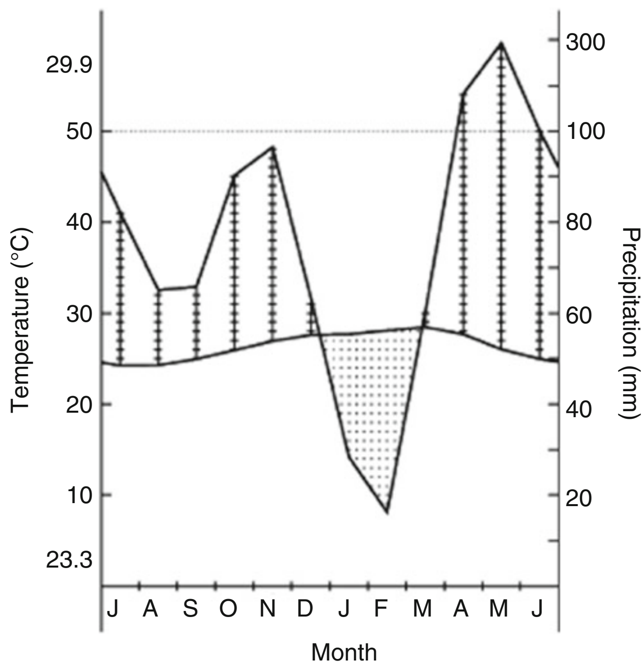

Fig. 2 Climate of Mombasa (Source Lieth et al. 1999)

and fruits). The sorted samples were then oven dried at 60 °C for 48 hours and the dry weight recorded.

Leaf $\delta^{15}N$, and C:N Ratio Analysis

Leaf litter samples (brown leaves) collected in April, June, August, and October 2006 were analysed for total carbon (TC), total nitrogen (TN) and $\delta^{15}N$ composition. Additional fresh (green) leaves were also collected directly from the trees, avoiding damaged leaves. To prepare samples for C and N analysis, dried samples were pre-treated with liquid nitrogen to make them brittle and ground to a fine consistency using a pestle and mortar. The carbon and nitrogen content of pre-weighed samples in pre-weighed tin cups were determined with a Thermo Finnigan Elemental Analyser flash 1112 connected to an Isotope Ratio Mass Spectrometer (Thermo Finnigan Delta) by a continuous flow interphase (Finnigan Conflo III). The relative abundance of the heavy and light stable isotopes of N are expressed as $\delta^{15}N$ values; i.e. in relative conventional standard N_2 for nitrogen. $\delta^{15}N$ values are calculated according to the formula:

$$\delta X(\text{in}\%) = \lfloor (R_{sample}/R_{standard}) - 1 \rfloor * 1000$$

where $X = {}^{15}N$ and $R = {}^{15}N/{}^{14}N$ or $X = {}^{13}C$ and $R = {}^{13}C/{}^{12}C$. Acetanilide (C = 71.03%, N = 10.36%) were used for standardisation in C:N ratio estimation. Internal sample reference material used for $\delta^{15}N$ and $\delta^{13}C$ was ammonium sulphate (IAEA-N1) and sugar (IAEA-CH-6) respectively.

The leaf resorption efficiency (RE) of N was calculated as: – $RE = 100(1 - N_{LL}/N_{GL})$ where N_{LL} is the leaf litter nutrient concentration and N_{GL} is the green leaf nutrient concentration.

Physicochemical Parameters

Sediment cores were sampled randomly in twenty plots along four transects laid from the landward to the seaward fringe, two transects under direct sewage exposure and two under indirect sewage exposure. In each plot, three sediment cores of diameter 6.4 cm and 30 cm depth were taken by hand, placed on ice (in a cool box) and transported to the laboratory for analysis immediately. The sediment cores were sectioned at 0–5 cm and 9–11 cm and 24–26 cm. Laboratory analysis included porewater analysis. The pore water was extracted with KCl solution under centrifugation (3000x g). The resulting pore water was then filtered through prewashed 0.45 μ Whatman GF/F microfiber filter and analysed for ammonium (NH_4^+), nitrate/nitrite (NO_X^-) and phosphates (PO_4^{3-}). The pore water collecting in the holes after sediment extraction was used to measure salinity with a hand held refractometre. For subsequent nutrient analysis after extraction, the conventional seawater analysis method by Parsons et al. (1984) was used. For estimation of organic carbon (%TOM), a pre-weighed sampled was dried to a constant weight at 70 °C and then combusted at 450 °C for 4 hours (ashing). The weight loss was then estimated as the percentage total organic matter (%TOM).

Data Analysis

Litter production was estimated per unit area based on the averages per litter trap per plot and the annual litter fall determined by summing the average masses per month per unit area. Differences in litter production between the 2 years (2005–2006 vs. 2006–2007), between species and among the 12 sampling months (nested within years) were tested by the nested 3 way analysis of variance (ANOVA). Multiple comparisons by the Tukey HSD test were used to assess differences between plots. Data on porewater nutrients were log transformed and analysed by a nested 2-way analysis of variance (factors, exposure, and transects (nested in exposure). The non-parametric Spearman-R was used to establish correlation between litter fall and environmental factors. All analyses and graphical presentation were done using STATISTICA 8.

Results

Physicochemical Parameters

There were significant seasonal variations in sediment pore water salinity ($F_{1, 18} = 376.95, p < 0.001$) spatial variations were however not significant in each season ($F_{2, 15} = 1.116$, $p = 0.38$). Salinity averaged 42‰ (±3) during the dry season and 29‰ (± 2) during the wet season. Sediment pore water NH_4^+ ($F_{1, 116} = 5.93$, $p < 0.02$), NO_x^- ($F_{1, 116} = 7.633$, $p = 0.007$), PO_4^{3-} ($F_{1, 116} = 15.18$, $p < 0.001$) and %TOM ($F_{1, 116} = 64.02$, $p < 0.001$) were significantly different for all sampled plots (Table 1). Directly sewage exposed plots had high NH_4^+ and NO_x^- levels and high %TOM content. However, multiple comparisons by Tukey HSD test reveal that PO_4^{3-} levels were significantly different in only one directly sewage exposed transect. Higher sediment pore water nutrients levels were observed in the landward and seaward fringe for directly exposed transect and decreasing from landward to seaward fringe for indirectly exposed transects, with a corresponding increase in %TOM content from landward to seaward fringe (3.8% ± 3.31 – 7.79% ± 1.11) (Table 2).

Litter Fall

Litter fall was observed throughout the year. The mean annual litter fall was estimated at 12 (± 3) t DW $ha^{-1}yr^{-1}$ for Tudor Creek. The highest litter fall rates occurred in the hot and dry NEM season (January-April – ca. 5 ± 1 g DW m^{-2} day^{-1}) which coincides with the onset of the rainy season. The lowest rates occurred in the cooler and wet SEM season (June-October – ca. 3 ± 0.5 g DW m^{-2} day^{-1}). On average, leaves formed the most important litter component at 8 ± 2 t DW $ha^{-1}yr^{-1}$, accounting for 68% of the total litter fall. Reproductive material (fruits and propagules; 2 ± 1 t DW $ha^{-1}yr^{-1}$), wood (0.9 ± 0.7 t DW $ha^{-1}yr^{-1}$) and flowers (0.97 ± 0.97 t DW $ha^{-1}yr^{-1}$) accounted for 16%, 8% and 8% of the mean annual litter respectively (Table 3).

Litter fall differed significantly between species and season (Tables 4 and 5). *R. mucronata* account for the highest annual rate (15.34 ± 3.34 t DW $ha^{-1}yr^{-1}$), while no significant differences were observed between *A. marina* and *S. alba* (11.44 ± 2.90 – 9.69 ± 5.26 t DW $ha^{-1}yr^{-1}$ respectively) (Tables 3 and 5). The effect of sewage on litter fall was not significant for *S. alba* and no clear effect pattern for *R. mucronata* (Table 5). However, the directly sewage exposed *R. mucronata* plots had significantly lower litter than one of the indirectly exposed plots. In the second year of study, a die back of *S. alba* receiving raw sewage occurred. This was observed after unusually high rainfall between August and December. The die back contributed to the lower litter fall for *S. alba* in the second year. The cause of die back is not obvious and may require a separate study. This die back may be associated with the infestation of the *S. alba* stand by insects. A similar die back of *S. alba* was observed in Gazi Bay (ca. 47 km south; Wang'ondu, University of Nairobi, personal communication).

Litter production displayed distinct seasonal fluctuations and differed significantly between years (Table 4 and Fig. 3). The seasonal patterns displayed significant correlation with climatic factors such as temperature, rainfall and relative humidity (Table 6). Both Leaf fall and production of reproductive materials were significantly correlated with temperature, rainfall, and relative humidity (Table 6). High temperatures favoured leaf fall, while rainfall and high relative humidity correlate with low leaf fall. The second year of study was characterised by high rainfall and temperatures.

Table 1 Average (± standard deviation) pore water nutrients concentrations (μM) and the %TOM from 30 cm sediment cores

Depth	0–5 cm				9–11 cm				24–26 cm			
Parameter	NH_4^+	NO_x^-	PO_4^{3-}	%TOM	NH_4^+	NO_x^-	PO_4^{3-}	%TOM	NH_4^+	NO_x^-	PO_4^{3-}	%TOM
Direct exposure	570 (± 354)	16(± 11)	2 (± 2)	5 (± 3)	543 (± 271)	22 (± 10)	1(± 0.9)	5 (± 4)	465 (± 359)	27 (± 14)	2 (± 1)	6 (± 4)
Indirect exposureo	417 (± 201)	9 (± 5)	2 (± 1)	2 (± 0. 9)	339 (± 153)	12 (± 10)	2 (± 1)	3 (± 2)	305 (± 190)	10 (± 12)	2 (± 1)	4 (± 2)

Table 2 A 2-way nested ANOVA showing significant differences in porewater nutrients concentrations between transects and exposure

	PO_4^{3-}			NO_x^-			NH_4^+			%TOM		
	Df	F	P	Df	F	P	df	F	P	Df	F	P
Transect (Exposure)	2	8.76	***0.0003***	2	1.99	0.14	2	3.37	***0.038***	2	7.59	***0.001***
Exposure	1	15.18	***0.0002***	1	7.63	***0.007***	1	5.93	***0.016***	1	64.02	***< 0.0001***
Error	116			116			116			116		

Table 3 Average (± standard deviation) annual litter fall (t DW ha-1 yr^{-1}) per species

	Leaves	Wood	Flowers	Reproductive material	Total (t $ha^{-1}yr^{-1}$)
A. marina	7.63 ± 0.29 (66.64)	0.62 ± 0.14 (5.46)	1.98 ± 2.01 (17.30)	1.21 ± 0.46 (10.60)	11.44 ± 2.90
R. mucronata[a]	9.08 ± 0.10 (74.32)	0.45 ± 0.57 (2.61)	0.58 ± 0.08 (6.78)	3.30 ± 0.19 (9.21)	13.42 ± 0.74
R. mucronata[b]	13.20 ± 2.29 (73.10)	0.33 ± 0.06 (2.11)	1.42 ± 0.93 (1.68)	4.09 ± 3.45 (9.90)	19.04 ± 0.53
R. mucronata[c]	9.17 ± 1.14 (70.73)	0.33 ± 0.10 (2.69)	0.62 ± 0.19 (2.12)	3.12 ± 1.10 (8.97)	13.25 ± 2.53
S. alba[a]	6.94 ± 6.10 (66.78)	1.66 ± 0.99 (15.97)	0.04 ± 0.06 (0.42)	1.75 ± 0.95 (16.82)	10.4 ± 8.10
S. alba[b]	6.77 ± 4.13 (69.88)	1.73 ± 0.58 (17.84)	0.04 ± 0.04 (0.43)	1.08 ± 0.95 (11.15)	9.69 ± 5.26
Average (t $ha^{-1}yr^{-1}$)	8.29 ± 1.94 (68.22)	0.91 ± 0.72 (7.48)	0.97 ± 0.97 (7.94)	1.93 ± 1.36 (15.89)	12.16 ± 2.89

Values in parenthesis indicate percentage of total litter per species
[a]Direct sewage exposure
[b]Indirect sewage exposure
[c]Indirect sewage exposure

Table 4 3 way nested ANOVA showing significant differences in productivity between months, year and species

	SS	Df	MS	F	P
Month (Year)	47.10	22	2.14	16.63	***<0.001***
Year	0.75	1	0.75	5.83	***0.016***
Species	11.81	2	5.91	45.88	***<0.001***
Error	186.95	1452	0.13		

Table 5 Tukey HSD test showing significance in differences between species and plots for total litter g DW m^{-2} $month^{-1}$

	Plot	(1) – 106	(2) – 199	(3) – 142	(4) – 101	(5) – 78	(6) – 170
1	*A. marina*[a]	–	***< 0.001***	0.18	0.99	0.48	***< 0.001***
2	*R. mucronata*[b]	***<0.001***	–	***0.005***	***< 0.001***	***< 0.001***	0.28
3	*R. mucronata*[a]	0.18	***0.005***	–	0.097	***0.001***	0.35
4	*S. alba*[a]	0.99	***< 0.001***	***0.097***	–	0.69	***< 0.001***
5	*S. alba*[b]	0.48	***< 0.001***	***0.001***	0.69	–	***< 0.001***
6	*R. mucronata*[c]	***<0.001***	0.28	0.35	***< 0.001***	***< 0.001***	–

Figures in bold italics show significant difference at $p < 0.05$
[a]Direct sewage exposure
[b]Indirect sewage exposure
[c]Indirect sewage exposure

R. mucronata produced flowers all year round. The main flowering event occurs between March and July. Propagules were produced between March and June. *S. alba* produced flowers between August and October, with minor flowering events between February and April. Fruiting occurred in July–January, and seeds produced biannually in November and April. *A. marina*, produced flowers between December and April, fruits between February and July and seeds between April and May. Woody materials did not correlate with any climatic factors and did not display any seasonal trends.

Leaf $\delta^{15}N$, $\delta^{13}C$ and C:N Ratio Analysis

The leaf total nitrogen (TN) and carbon (C) content and the C:N ratio for all sampled plots are shown in Table 7. The leaf TN content differed significantly by species ($F_{2, 50} = 38.5027$, $p < 0.001$) and mode of exposure ($F_{5, 47} = 14.8876$, $p < 0.001$). Leaf C did not differ significantly between plots ($F_{5, 47} = 0.80$, $p = 0.56$) and species ($F_{2, 50} = 0.313$, $p = 0.73$). The leaf TN and C did not vary significantly monthly ($F_{3, 49} = 0.33685$, $p = 0.78$) or by mode of sewage exposure ($F_{2, 50} = 1.4357$, $p = 0.25$). Leaves of *A. marina* and *S. alba* had higher TN content than *R. mucronata*. Directly exposed *R. mucronata* had higher leaf TN content (4.80 ± 1.09 mg/g DW) than indirectly exposed (3.35 ± 0.45 mg/g DW). The C:N ratio varied significantly between species. Direct sewage exposed *S. alba* did not differ from the indirectly sewage exposed *S. alba*. Differences were observed for directly and indirectly sewage exposed *R. mucronata*, with directly sewage exposed *R. mucronata* having a lower C:N ratio. The $\delta^{13}C$ levels measured were in the same range of values

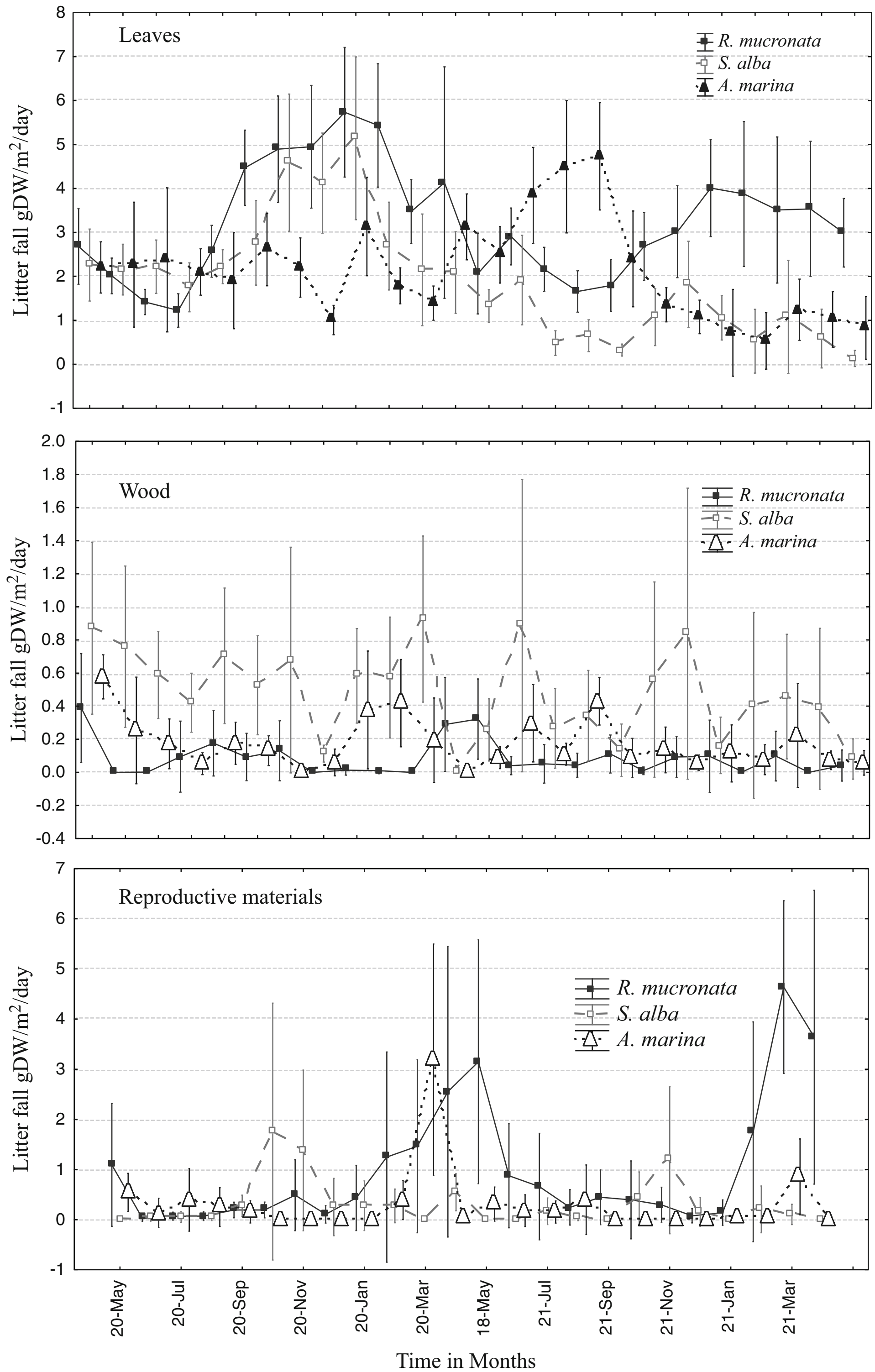

Fig. 3 Litter fall trends per part/category and species for two years

reported for mangrove leaves globally and ranged between -21‰ and -35‰, and is representative of C_3 terrestrial plants (Rao et al. 1994; Muzuka and Shunula 2006; Bouillon et al. 2008; Kristensen et al. 2008). Slight enrichment in $\delta^{13}C$ (less than 1‰) was generally observed for like species.

A slight elevation in leaf TN content was observed for direct sewage exposed *R. mucronata*, and *S. alba*. Species with higher C:N ratio had corresponding lower $\delta^{15}N$ signatures and vice versa. This indicates elevated leaf TN content in direct sewage exposed sites. lowers the C:N ratio (Fig. 4).

Table 6 Spearman Rank tests showing significant correlation between climatic factors and litter fall

Litter component	Climate factor	Spearman – R	t(N-2)	*P*
Leaves	Temperature	0.49	8.71	***<0.001***
	Rain	−0.21	−3.34	***<0.001***
	Relative Humidity	−0.28	−4.53	***<0.001***
Wood	Temperature	−0.09	−1.46	0.15
	Rain	0.05	0.84	0.4
	Relative Humidity	0.05	0.84	0.4
Flowers	Temperature	−0.35	−5.69	***<0.001***
	Rain	0.15	2.4	***0.02***
	Relative Humidity	0.17	2.63	***0.01***
Reproductive material	Temperature	0.12	1.8	0.07
	Rain	−0.05	−0.71	0.48
	Relative Humidity	−0.12	−1.88	0.06

Table 7 Total organic carbon (TOC) and total nitrogen (TN) (mg/g DW), $\delta^{15}N$, $\delta^{13}C$ and the C:N ratio for senescent leaves (brown leaves) sampled from the three mangrove species

Species	TN% (mg/g DW)	TOC% (mg/g DW)	$\delta^{15}N$‰	$\delta^{13}C$‰	C:N
A. marina[a]	0.86 ± 0.18 (8.60 ± 1.77)	47 ± 3 (468 ± 30)	6.46 ± 0.59	−27.53 ± 1.76	56 ± 9
R. mucronata[a]	0.48 ± 0.11 (4.80 ± 1.09)	48 ± 2 (477 ± 20)	4.73 ± 1.26	−28.37 ± 0.86	103 ± 19
R. mucronata[b]	0.39 ± 0.03 (3.85 ± 0.32)	48 ± 2 (476 ± 24)	3.31 ± 0.80	−27.46 ± 0.97	124 ± 12
R. mucronata[c]	0.33 ± 0.04 (3.35 ± 0.45)	45 ± 44.68 (449 ± 5)	3.45 ± 0.96	−27.90 ± 0.90	135 ± 8
S. alba[a]	1.16 ± 0.41 (11.59 ± 4.07)	44 ± 5 (443 ± 48)	7.03 ± 0.64	−28.76 ± 0.61	42 ± 12
S. alba[b]	1.14 ± 0.34 (11.43 ± 3.42)	44 ± 3 (443 ± 25)	6.62 ± 0.75	−28.18 ± 0.91	42 ± 13

Differences are made between direct and indirect sewage exposure
[a]Directly sewage exposure
[b]Indirectly sewage exposure
[c]Indirectly sewage exposure

Table 8 Total organic carbon (TOC) and total nitrogen (TN) (mg/g DW) and the C:N ratio for fresh leaves (green), and the Resorption Efficiency (RE) for the three mangrove species

Species	Data	Direct	Indirect	RE (%)
A. marina	C:N	23.53 ± 3.28	–	
	TN (mg/g dw)	19.77 ± 4.49	–	48 ± 24
	TOC (mg/g dw)	461 ± 109	–	
R. mucronata	C:N	44.29 ± 5.56	44.53 ± 7.08	
	TN (mg/g dw)	9.89 ± 0.07	9.54 ± 0.95	55 ± 23
	TOC (mg/g dw)	438 ± 52	420 ± 33	
S. alba	C:N	25.80 ± 5.20	28.56 ± 3.94	
	TN (mg/g dw)	21.00 ± 9.84	19.02 ± 6.41	53 ± 22
	TOC (mg/g dw)	518 ± 193	531 ± 144	

Discussion

Litter production for the mangroves of Tudor Creek is moderately within the upper global range reported for *R. mucronata*, *A. marina* and *S. alba* in different geographical ranges (Table 9). Values in the range 3 t DW ha^{-1} yr^{-1} (Phuket; Chansang and Poovachiranon 1985) to 16 t DW ha^{-1} yr^{-1} (Malaysia; Sasekumar and Loi 1983) have been reported for Rhizophoraceae, 8 t DW ha^{-1} yr^{-1} (Australia; Duke et al. 1981) to 17 t DW ha^{-1} yr^{-1} (India; Wafar et al. 1997) for *S. alba* and 3 t DW ha^{-1} yr^{-1} (Australia; Clarke 1994) to 16 t DW ha^{-1} yr^{-1} (Australia; Bunt 1995) for *A. marina*. Consistent with other studies, leaves were the major contributors of total litter. Saenger and Snedaker (1993) suggest that high litter fall indicates an optimal habitat, particularly with respect to reduced salinity, optimum climate and increased site fertility. Pool et al. (1977),

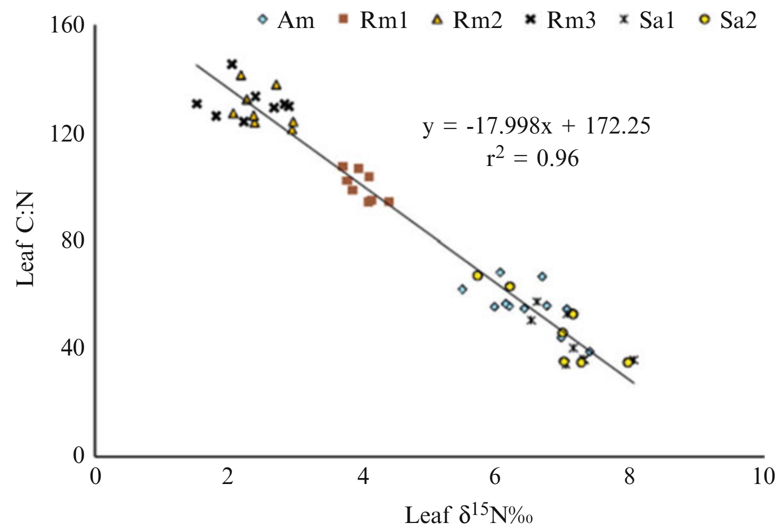

Fig. 4 Scatter plot of leaf C:N ratio versus leaf δ^{15}N‰ showing a significant correlation ($r^2 = 0.96$, $F = 1231.71$, $p < 0.05$)

Table 9 Litter production in mangroves from different parts of the world

Location	Latitude	Species	Litter fall (t DW ha^{-1} yr^{-1})	References
Kenya (Tudor creek)	4° S	*A. marina*	11	This study
		R. mucronata	15	
		S. alba	10	
Kenya (Gazi Bay)	4° S	*A. marina*	5	Wang'ondu et al. (in press)
		R. mucronata	11	
		S. alba	10	
Kenya (Mida Creek)	3° S	*R. mucronata*	16	Kairo (2001)
		C. tagal	23	
		A. marina	8	
Kenya (Gazi Bay)	4° S	*R. mucronata*	9	Slim et al. (1996)
		C. tagal	2	
China (Futian Nature National Reserve)	22° N	*Aegiceras corniculatum* and *Kandelia candel*	12	Tam et al. (1998)
South Africa (Wavecrest (Transkei))	21° S	Mixed forest	5	Steinke and Ward (1990)
India (Mandovi-Zuari Estuary)	15° N	*R. mucronata*	11	Wafar et al. (1997)
		S. alba	17	
New Guinea (Fly River Estuary)	8° S	Mixed forest	8–14	Twilley et al. (1992)
Malaysia (Matang mangal)	5° N	Mixed forest	4	Gong and Ong (1990)
Australia (Jervis Bay, NSW)	35° S	*A. marina*	3	Clarke (1994)
Australia (Embley River)	12° S	*A. marina*	6	Conacher et al. (1996)
Papua New Guinea (near Port Moresby)	9° S	*A. marina*	16	Bunt (1995)
Mexico (Teacapan-Ague Brava Lagoon)	22° N	Mixed forest	14	Flores-Verdugo et al. (1990)

however, argues that increasing stress increases leaf litter fall. The unique setting in this study however is the human disturbance and sewage pollution, as most litter fall studies have looked at productivity of mangroves under normal or pristine conditions.

Globally, litter fall from a number of mangrove forests has been observed to be variable, depending on tree height, latitude (Saenger and Snedaker 1993), climate (Wafar et al. 1997) and species specific traits (Woodroffe 1985). In our study, *R. mucronata* was more productive than *A. marina* and *S. alba*, despite the human disturbances through unregulated exploitation. Unregulated exploitation has resulted in predominant young vegetation especially for *R. mucronata* in Tudor Creek (Mohamed et al. 2009), partly explaining the high litter fall rates. Generally, young mangrove stands have been reported to produce large and high quality litter compared to older established stands (Nga et al. 2005; Ellis and Bell 2004). Mackey and Smail (1995) also observed that the maturity of mangrove forests and increase in canopy height slows organic cycling due to the accumulation of non-photosynthetic biomass by the trees, thus low productivity. The high yields can also be explained by the location of the site at a lower latitude, the tropical humid monsoon weather, specific growing conditions as a result of greater tidal activity and water turn-over within the Creek, coupled with the monsoon climate (Wafar et al. 1997). Conversely, as suggested by Pool et al. (1977)), stress from high levels of siltation, unregulated harvesting and sewage pollution (as reported by Mohamed et al. (2009)) may best explain the observed high rates of litter fall.

The seasonal variation in litter fall has featured prominently in earlier studies, but the underlying causes remain obscure. Seasonal variations in litter fall as observed in this study correspond to those reported by Slim et al. (1996), Wafar et al. (1997) and Ochieng and Erftemeijer (2002), where the maxima in litter fall are associated with the dry NEM months. The maxima litter fall being a result of water stress due to low or no rainfall, higher temperatures, characterised by high evapo-transpiration rates leading to higher salinities, complicating metabolism of transpiration and necessitating canopy thinning by leaf loss (Chen et al. 2009; Wafar et al. 1997; Tam et al. 1998; Slim et al. 1996; Eusse and Aide 1999). Our observations on phonological trends are in accord with observations made by Mackey and Smail (1995), that reproductive output is not affected by the age structure of the forest and, it can be seen that reproductive output per tree is greatest in the mature forest (Mackey and Smail 1995). Thus the reproductive output of the population is essentially a balance between tree density and size (Mackey and Smail 1995). Mangrove litter production and phenological cycles are thus highly linked to both anthropogenic effects (the unregulated harvesting of trees) and climatic factors and may be vulnerable to influences of global climate change. Based on the vulnerability scale developed by Ellison (2012), the productivity of the system may be ranked medium, implying sub-optimal conditions for the three mangrove species in Tudor Creek. The ranking is based on the proportion of reproductive materials, which was less than 20% of the total litter, except for *A. marina*. The observed die back of *S. alba* stand on the seaward fringe is indicative of stress and adds to the vulnerability of the forest to climate change. The recovery of such seaward fringe mangroves is more difficult. Annual differences in litter fall were significant but the short period of study provides insufficient data to establish the temporal variability particularly in relation to detecting anthropogenic changes.

The effect of sewage exposure on litter production had no effect on the quantity of litter. Tam et al. (1998) also reported no differences in litter fall between sewage receiving and control sites. In Tudor Creek however, the edaphic physicochemical conditions varied spatially, with high NH_4^+ and NO_x^- pore-water levels in and low PO_4^{3-} levels. Since mangroves are either phosphorus (P) limited (Feller 1995; Koch and Snedaker 1997) or differentially N or P limited across tidal gradients (Boto and Wellington 1983; Feller et al. 2003a, b; Krauss et al. 2008), our observations imply possible site variability in limiting nutrients. A study at the same site reported that sewage pollution has caused a higher benthic metabolism and more reduced sediment conditions (PUMPSEA 2007) shifting a healthy decomposing aerobic-anaerobic system of the mangrove to a complete anaerobic system, which is less efficient and slow in recycling nutrients. This may have affected tree growth and possibly productivity (Alongi 1994; Bouillon et al. 2007a; Holguin et al. 2001). Considering that Tudor Creek is a degraded mangrove (Mohamed et al. 2009), the success of structural recovery via growth and functional recovery via productivity depends on the preservation of benthic microbial communities and their geochemical environment (Alongi 1994; Holguin et al. 2001). Therefore, long term variable exposure to sewage, high NH_4^+ and NO_x^- supply, lower PO_4^{3-} supply, coupled with siltation effects (i.e. altered biogeochemical cycles), may account for the lack of differences in productivity between sites and further inhibits the recovery of the forest.

Carbon and Total Nitrogen Content

From our observations, leaf TN differed between species and location. These differences were observed between direct and indirect sewage exposed *R. mucronata* only. Directly exposed leaf samples had elevated TN compared to the indirectly exposed. This leaf TN enrichment is a consequence of the sediment nutrients levels (Dittmar et al. 2006; Wanek et al. 2007). This relationship is displayed by the leaf $\delta^{15}N$, which indicates nutrients dynamics within the Tudor Creek mangrove system are affected by inputs from raw domestic sewage and land use patterns by humans. Nutrient enrichment lowers the conservation of essential nutrients depending on mangrove species, resulting in nutrient enriched litter (Wanek et al. 2007). The leaf nutritive values, expressed as C:N ratio reported in this study are within the range of values reported globally (Twilley et al. 1986; Ellis et al. 2006; Krauss et al. 2008), though lower than values reported for Gazi Bay (Rao et al. 1994; Slim et al. 1996; Ochieng and Erftemeijer 2002). The C:N ratio estimates for Gazi Bay mangroves changed from 47.5 ± 21 in fresh leaves to 129 ± 60 in senescent leaves, indicating an approximate 64% total nitrogen RE by the mangroves. Similar values have been reported for nutrient deficient arid mangrove in Western Australia (Alongi et al. 2005). We observe that, in Tudor Creek, the C:N changed from 44.53 ± 7.08 to 124 ± 12 for indirectly exposed *R. mucronata* and 44.29 ± 5.56 to 103 ± 19 for direct exposed. The estimated RE were low ranging between 44% and 55% depending on species (Table 8).

The two indicators of mangrove ecosystem N cycling measured in this study – mangrove leaf %N and $\delta^{15}N$‰, were on average higher than reported for unpolluted sites globally (−1.32 – 2.789‰: Fry et al. 2000; Costanzo et al. 2003; Bouillon et al. 2008). Lower values have been reported for unpolluted sites in Tanzania (Muzuka and Shunula 2006) and Kenya (Gazi Bay ca. 47 km south; Marguillier et al. 1997). Mangroves of Gazi Bay had $\delta^{15}N$ ‰ values ranging between 0.76–2.18 (Marguillier et al. 1997). Within the study site, leaf $\delta^{15}N$ signatures for the directly sewage exposed leaf samples were slightly elevated than the indirectly exposed, though not significant ($F_{1, 42} = 2.0193$, $p = 0.16$). This may be due to the long term variable exposure to sewage in the entire site, with the stands near the point source (directly exposed) being continuously exposed. Between-site differences in plant isotopic composition may therefore be viewed as an expression of the differences in $\delta^{15}N$ levels in the source – the raw domestic sewage effluent (Cabana and Rasmussen 1996), and/or isotopic fractionation during uptake (Costanzo et al. 2003). The tidal flooding within the sites dilutes and distributes the sewage effluent, creating a characteristic gradient, resulting in high pore-water NH_4^+ and NO_x^- levels in '*directly sewage exposed*' sites compared to '*indirectly sewage exposed*' sites, setting the ecosystem baseline for $\delta^{15}N$. Isotopic fractionation occurs via plant uptake, microbial nitrification-denitrification, soil adsorption and volatilization, as the sewage flows sequentially first through farm lands, then through *R. mucronata* plots, and eventually through *S. alba* plots, where preferential microbial processing and uptake of the lighter ^{14}N results in $\delta^{15}N$ enriched effluent (Yoneyama et al. 1991; Pennock et al. 1996; Costanzo et al. 2003). Thus the lower leaf $\delta^{15}N$ values in *R. mucronata* leaves may indicate nutrient use efficiency for the species, known to possess leaves with lower nutrient content as an adaptation against herbivory (Alongi et al. 2005).

The absence of seasonal variation in mangrove leaf $\delta^{15}N$ signatures has also been reported by Costanzo et al. (2003). This is because leaf $\delta^{15}N$ signatures reflects the mangrove life history and the sediment N source, and is a representation of the long-term nitrogen supply and the edaphic conditions (Fry et al. 2000; Costanzo et al. 2003). Subsequently, the leaf $\delta^{15}N$ levels may reflect the mangrove sediments capacity to retain or immobilize nutrients in wastewater. However, whether mangrove trees contribute to phytoremediation depends on uptake and would require a separate study to establish (Chu et al. 1998; Ye et al. 2001; Feller et al. 2003a, b; Costanzo et al. 2003).

Conclusion

Mangrove ecosystems are complex and nutrients enrichment or sewage pollution may not always elicit similar or direct responses. The response may depend on the status of sediment microbial community, the structural condition of the forests and the presence of other disturbances (PUMPSEA 2007). Sewage pollution, as observed in this study and others, increases nutrient levels in the foliage of mangrove plants (see also Henley 1978; Boto and Wellington 1983; Clough et al. 1983). Considering that mangroves normally possess leaves with lower nutrient content as an adaptation against herbivory (Alongi et al. 2005), it is questionable therefore; to what extent this adaptation is compromised as a result of sewage pollution. Nutrients enrichment of mangrove litter may imply faster degradation of above ground litter. This presents a functional trait in mangroves that evokes considerable trophic links that directly impacts nutrient cycling. Since litter fall returns a greater fraction of nutrients from the tree canopy to the soil, eutrophication may markedly accelerate nutrient cycling via litter fall and decomposition and enhance organic matter provisioning to mangrove food webs (Wanek et al. 2007). C:N have an inverse relationship with litter degradation, litter degradation being strongly dependent on availability of nitrogen for microbial decomposers (Dittmar et al. 2006; Ellis et al. 2006). Therefore leaves enriched in N will potentially accelerate nutrient cycling via litter fall and decomposition in these ecosystems, enhancing organic matter provisioning to mangrove food webs (Wanek et al. 2007). Such nutrient enriched systems have been shown to be prone to shifts in fauna composition favouring the herbivore communities (Feller 1995; Bouillon et al. 2002). However, the compounding effects of the major limiting nutrients, N and P (Feller et al. 1999; Feller et al. 2003a, b; McKee et al. 2002), coupled with the status of the sediment microbial communities and presence of crabs, play an important role in nutrient dynamics and possibly influence vegetation responses such as growth and productivity. Therefore, the use of mangroves for sewage treatment should only be attempted with clear understanding of the ecosystem carrying capacity.

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References

Alongi DM (1994) The role of bacteria in nutrient recycling in tropical mangrove and other coastal benthic ecosystems. Hydrobiologia 285 (1–3):19–32

Alongi DM, Clough BF, Robertson AI (2005) Nutrient-use efficiency in arid-zone forests of the mangroves *Rhizophora stylosa* and *Avicennia marina*. Aquat Bot 82:121–131

Benner R, Hodson RE (1985) Microbial degradation of the leachable and lignocellulosic components of leaves and wood from Rhizophora mangle in a tropical mangrove swamp. Ma Ecol Prog Ser 23(3):221–230

Boto KG, Wellington JT (1983) Phosphorus and nitrogen nutritional status of a Northern Australian mangrove forest. Mar Ecol Prog Ser 11(1):63–69

Bouillon S, Raman AV, Dauby P, Dehairs F (2002) Carbon and nitrogen stable isotope ratios of subtidal benthic invertebrates in an estuarine mangrove ecosystem (Andhra Pradesh, India). Estuar Coast Shelf Sci 54:901–913

Bouillon S, Moens T, Koedam N, Dahdouh-Guebas F, Baeyens W, Dehairs F (2004a) Variability in the origin of carbon substrates for bacterial communities in mangrove sediments. FEMS Microbiol Ecol 49(2):171–179

Bouillon S, Moens T, Overmeer I, Koedam N, Dehairs F (2004b) Resource utilization patterns of epifauna from mangrove forests with contrasting inputs of local versus imported organic matter. Mar Ecol Prog Ser 278:77–88

Bouillon S, Middelburg JJ, Dehairs F, Borges AV, Abril G, Flindt MR, Ulomi S, Kristensen E (2007a) Importance of intertidal sediment processes and porewater exchange on the water column biogeochemistry in a pristine mangrove Creek (Ras Dege, Tanzania). Biogeosciences 4(3):311–322

Bouillon S, Dehairs F, Velimirov B, Abril G, Borges AV (2007b) Dynamics of organic and inorganic carbon across contiguous mangrove and seagrass systems (Gazi Bay, Kenya). J Geophys Res Biogeosci 112(G02018)

Bouillon S, Connolly RM, Lee SY (2008) Organic matter exchange and cycling in mangrove ecosystems: recent insights from stable isotope studies. J Sea Res 59(1–2):44–58

Bunt JS (1995) Continental scale patterns in mangrove litter fall. Hydrobiologia 295(1–3):135–140

Cabana G, Rasmussen JB (1996) Comparison of aquatic food chains using nitrogen isotopes. Proc Natl Acad Sci U S A 93(20):10844–10847

Chansang H, Poovachiranon S (1985) Fate of mangrove litter in a mangrove forest at Ko Yao Yai, southern Thailand. Estuaries 8: 106A

Chen L, Zan Q, Li M, Shen J, Liao W (2009) Litter dynamics and forest structure of the introduced *Sonneratia caseolaris* mangrove forest in Shenzhen, China. Estuar Coast Shelf Sci 85(2):241–246

Chu HY, Chen NC, Yeung MC, Tam NFY, Wong YS (1998) Tide-tank system simulating mangrove wetland for removal of nutrients and heavy metals from wastewater. Water Sci Technol 38(1):361–368

Clarke PJ (1994) Base-line studies of temperate mangrove growth and reproduction demographic and litter fall measures of leafing and flowering. Aust J Bot 42(1):37–48

Clough BF, Boto KG, Attiwill PM (1983) Mangrove and sewage: a re-evaluation. In: Teas HJ (ed) Biology and ecology of mangroves, vol 8, Tasks for vegetation science series. Dr W. Junk Publishers, Lancaster, pp 151–162

Conacher CA, Obrien C, Horrocks JL, Kenyon RK (1996) Litter production and accumulation in stressed mangrove communities in the Embley River Estuary, north-eastern Gulf of Carpentaria, Australia. Mar Freshw Res 47(5):737–743

Costanzo SD, O'Donohue MJ, Dennison WC (2003) Assessing the seasonal influence of sewage and agricultural nutrient inputs in a subtropical river estuary. Estuaries 26(4A):857–865

Day JW Jr, Conner WH, Ley-Lou F, Day RH, Navarro AM (1987) The productivity and composition of mangrove forests, Languna de Terminos. Mexico Aquat Bot 27:267–284

Dittmar T, Hertkorn N, Kattner G, Lara RJ (2006) Mangroves, a major source of dissolved organic carbon to the oceans. Glob Biogeochem Cycles 20. GB1012

Duke NC (1990) Phenological trends with latitude in the mangrove tree *Avicennia marina*. J Ecol 78(1):113–133

Duke NC, Bunt JS, Williams WT (1981) Mangrove litter fall in North-eastern Australia. 1. Annual totals by component in selected species. Aust J Bot 29(5):547–553

Duke NC, Meynecke JO, Dittmann S, Ellison AM, Anger K, Berger U, Cannicci S, Diele K, Ewel KC, Field CD, Koedam N, Lee SY, Marchand C, Nordhaus I, Dahdouh-Guebas F (2007) A world without mangroves? Science 317:41–42

Ellis WL, Bell SS (2004) Canopy gaps formed by mangrove trimming: an experimental test of impact on litter fall and standing litter stock in Southwest Florida (USA). J Exp Mar Biol Ecol 311:201–222

Ellis WL, Bowles JW, Erickson AA, Stafford N, Bell SS, Thomas M (2006) Alteration of the chemical composition of mangrove (*Laguncularia racemosa*) leaf litter fall by freeze damage. Estuar Coast Shelf Sci 68(1–2):363–371

Ellison JC (2012) Climate change vulnerability assessment and adaptation planning for mangrove systems. World Wildlife Fund (WWF), Washington, DC

Eusse AM, Aide TM (1999) Patterns of litter production across a salinity gradient in a *Pterocarpus officinalis* tropical wetland. Plan Ecol 145(2):307–315

Feller IC (1995) Effects of nutrient enrichment on growth and herbivory of dwarf red mangrove (Rhizophora mangle). Ecol Monogr 65(4):477–505

Feller IC, Whigham DF, O'Neill JP, McKee KL (1999) Effects of nutrient enrichment on within-stand cycling in a mangrove forest. Ecology 80(7):2193–2205

Feller IC, McKee KL, Whigham DF, O'Neill JP (2003a) Nitrogen vs. phosphorus limitation across an ecotonal gradient in a mangrove forest. Biogeochemistry 00:1–31

Feller IC, Whigham DF, McKee KL, Lovelock CE (2003b) Nitrogen limitation of growth and nutrient dynamics in a disturbed mangrove forest, Indian River Lagoon, Florida. Oecologia 134(3):405–414

Flores-Verdugo FJ, Gonzalez-Farias F, Ramirez-Flores O, Amezcua-Linares F, Yañez-Arancibia A, Alvarez-Rubio M, Day JW Jr (1990) Mangrove ecology, aquatic primary productivity, and fish community dynamics in the Teacapán-Agua Brava lagoon-estuarine system (Mexican Pacific). Estuaries 13:219–230

Fortes MD (1988) Mangrove and seagrass beds of East-Asia-Habitats under stress. Ambio 17(3):207–213

Fry B, Bern AL, Ross MS, Meeder JF (2000) δ^{15}N Studies of nitrogen use by the red mangrove, *Rhizophora mangle* L. in South Florida. Estuar Coast Shelf Sci 50(2):291–296

Gong WK, Ong JE (1990) Plant biomass and nutrient flux in a managed mangrove forest in Malaysia. Estuar Coast Shelf Sci 31(5):519–530

Henley DA (1978) An investigation of proposed effluent discharge into a tropical mangrove estuary. In: Proceeding of international conference on Water Pollution Control in Developing Countries. September 1978, Thailand, pp 43–64

Holguin G, Vazquez P, Bashan Y (2001) The role of sediment microorganisms in the productivity, conservation, and rehabilitation of mangrove ecosystems: an overview. Biol Fertil Soils 33(4):265–278

Kairo JG (2001) Ecology and restoration of mangrove systems in Kenya. Laboratory of General Botany and Nature Management (APNA). Free University of Brussels, Belgium

Koch MS, Snedaker SC (1997) Factors influencing *Rhizophora mangle* L. seedling development in Everglades carbonate soils. Aquat Bot 59(1–2):87–98

Krauss KW, Lovelock CE, McKee KL, López-Hoffman L, Ewe SM, Sousa WP (2008) Environmental drivers in mangrove establishment and early development: a review. Aquat Bot 89:105–127

Kristensen E, Bouillon S, Dittmar T, Marchand C (2008) Organic carbon dynamics in mangrove ecosystems: a review. Aquat Bot 89:201–219

Lee SY (1998) Ecological role of grapsid crabs in mangrove ecosystems: a review. Mar Freshw Res 49(4):335–343

Li MS (1997) Nutrient dynamics of a Futian mangrove forest in Shenzhen, South China. Estuar Coast Shelf Sci 45(4):463–472

Lieth H, Berlekamp J, Fuest S, Riediger S (1999) Climate diagrams of the world, CD-series: climate and biosphere. Blackhuys Publishers, Leiden

Lugo AE, Snedaker SC (1974) The ecology of mangroves. Annu Rev Ecol Syst 5:39–64

Mackey AP, Smail G (1995) Spatial and temporal variation in litter fall of Avicennia marina (Forssk.) Vierh in the Brisbane-River, Queensland, Australia. Aquat Bot 52(1–2):133–142

McKee KL, Feller IC, Popp M, Wanek W (2002) Mangrove isotopic (δ15N and δ13C) fractionation across a nitrogen vs. phosphorus limitation gradient. Ecology 83(4):1065–1075

Marguillier S, vander Velde G, Dehairs F, Hemminga MA, Rajagopal S (1997) Trophic relationships in an interlinked mangrove-seagrass ecosystem as traced by $\delta^{13}C$ and $\delta^{15}N$. Mar Ecol Prog Ser 151(1–3):115–121

Mohamed MOS, Neukermans G, Kairo JG, Dahdouh-Guebas F, Koedam N (2009) Mangrove forests in a peri-urban setting: the case of Mombasa (Kenya). Wetl Ecol Manage 17:243–255

Muzuka ANN, Shunula JP (2006) Stable isotope compositions of organic carbon and nitrogen of two mangrove stands along the Tanzanian coastal zone. Estuar Coast Shelf Sci 66(3–4):447–458

Mwaguni S, Munga D (1997) Land based sources and activities affecting the quality and uses of the marine coastal and associated freshwater environments along the Kenya coast. Coastal Development Authority, Mombasa

Mwangi S, Kirugara D, Osore M, Njoya J, Yobe A, Dzeha T (1999) Status of marine pollution in Mombasa Marine Park and Reserve and Mtwapa Creek. A technical report. Kenya Marine and Fisheries Research Institute, Government Chemist Department, Kenya Wildlife Service, Mombasa

Nagelkerken I, Blaber S, Bouillon S, Green P, Haywood M, Kirton LG, Meynecke JO, Pawlik J, Penrose HM, Sasekumar A, Somerfield PJ (2008) The habitat function of mangroves for terrestrial and marina fauna: a review. Aquat Bot 89:155–185

Nga BT, Tinh HQ, Tam DT, Scheffer M, Roijackers R (2005) Young mangrove stands produce a large and high quality litter input to aquatic systems. Wetl Ecol Manage 13:569–576

Nguli MM (2006) Water exchange and circulation in selected Kenyan Creeks. PhD dissertation, Earth Science Centre, Goteborg University, C78 2006

O'Neill RV, DeAngelis DL (1981) Comparative productivity and biomass relationships of forest ecosystems. In: Reichle DE (eds) Dynamic properties of forest ecosystems. Cambridge University Press, Cambridge, pp 411–450

Ochieng CA, Erftemeijer PLA (2002) Phenology, litter fall and nutrient resorption in *Avicennia marina* (Forsk.) Vierh in Gazi Bay, Kenya. Trees 16:167–171

Odum WE, Heald EJ (1972) Trophic analyses of an estuarine mangrove community. Bull Mar Sci 22(3):671

Parsons TR, Maita Y, Lalli CM (1984) A manual of chemical and biological methods for seawater analysis. Pergamon Press, Oxford, 173 pp

Pennock JR, Velinsky DJ, Ludlam JM, Sharp JH, Fogel ML (1996) Isotopic fractionation of ammonium and nitrate during uptake by *Skeletonema costatum*: implications for $\delta^{15}N$ dynamics under bloom conditions. Limnol Oceanogr 41:451–459

Pool DJ, Snedaker SC, Lugo AE (1977) Structure of mangrove morests in Florida, Puerto-Rico, Mexico, and Costa-Rica. BIOTR 9(3):195–212

PUMPSEA (2007) D5 – geographical differentiation of biogeochemical transformations in mangrove compartments exposed to selected sewage doses and compositions. Biological Institute from Southern Denmark University, Report for the Project: Peri-urban mangrove forests as filters and potential phytoremediators of domestic sewage in East Africa project. http://www.pumpsea.icat.fc.ul.pt/restricted.php

Rao RG, Woitchik AF, Goeyens L, van Riet A, Kazungu J, Dehairs F (1994) Carbon, nitrogen contents and stable carbon isotope abundance in mangrove leaves from an East African coastal lagoon (Kenya). Aquat Bot 47(2):175–183

Sasekumar A, Loi JJ (1983) Litter production in 3 mangrove forest zones in the Malay Peninsula. Aquat Bot 17(3–4):283–290

Saenger P, Snedaker SC (1993) Pantropical trends in mangrove aboveground biomass and annual litter fall. Oecologia 96(3):293–299

Slim FJ, Gwada PM, Kodjo M, Hemminga MA (1996) Biomass and litter fall of *Ceriops tagal* and *Rhizophora mucronata* in the mangrove forest of Gazi Bay, Kenya. Mar Freshw Res 47(8):999–1007

SPEK (1992) Mangrove forest of Mombasa. Technical Report by Society for Protection of Environment, Kenya

Steinke TD, Ward CJ (1990) Litter production by mangroves: III. Wavecrest (Transkei) with predictions for other Transkei Estuaries. S Afr J Bot 56(5):514–519

Tam NFY, Wong YS, Lan CY, Wang LN (1998) Litter production and decomposition in a subtropical mangrove swamp receiving wastewater. J Exp Mar Biol Ecol 226(1):1–18

Twilley RR, Lugo AE, Pattersonzucca C (1986) Litter production and turnover in basin mangrove forests in Southwest Florida. Ecology 67(3):670–683

Twilley RR, Chen RH, Hargis T (1992) Carbon sinks in mangroves and their implications to carbon budget of tropical coastal ecosystems. Water Air Soil Poll 64(1–2):265–288

UNEP/GPA (2001) Consultative meeting on municipal wastewater. Eastern Africa. Dar es Salaam, Tanzania, June 11–15 2001. UNEP, Eastern Afrian Action Plan/Regional coordinating unit, The Hague

UNEP-WCMC (2006) In the front line: shoreline protection and other ecosystem services from mangroves and coral reefs. UNEP-WCMC, Cambridge, 33 pp

Wafar S, Untawale AG, Wafar M (1997) Litter fall and energy flux in a mangrove ecosystem. Estuar Coast Shelf Sci 44(1):111–124

Walters BB, Rönnbäck P, Kovacs J, Crona B, Hussain S, Badola R, Primavera JH, Barbier EB, Dahdouh-Guebas F (2008) Ethnobiology, socio-economics and adaptive management of mangroves: a review. Aquat Bot 89:220–236

Wanek W, Hofmann J, Feller IC (2007) Canopy interactions of rainfall in an off-shore mangrove ecosystem dominated by *Rhizophora mangle* (Belize). J Hydrol 345:70–79

Woodroffe CD (1985) Studies of a mangrove basin, Tuff Crater, New-Zealand.1. Mangrove biomass and production of detritus. Estuar Coast Shelf Sci 20(3):265–280

Ye Y, Tam NFY, Wong YS (2001) Livestock wastewater treatment by a mangrove pot-cultivation system and the effect of salinity on the nutrient removal efficiency. Mar Pollut Bull 42(6):513–521

Yoneyama T, Omata T, Nakata S, Yazaki J (1991) Fractionation of nitrogen isotopes during the uptake and assimilation of ammonia by plants. Plant Cell Physiol 32:1211–1217

The Importance of the Thukela River Estuary, East Coast of South Africa, for the Biology of the Near-Shore Environment and Associated Human Aspects: A Review

Ander M. De Lecea and Rachel Cooper

Abstract

The Thukela is the largest river on the KwaZulu-Natal (KZN) coastline, located at the South-Western edge of the Indian Ocean. This coast and the KZN Bight, where the Thukela meets the ocean, is considered meso-oligotrophic with distinct sources of nutrients entering the system. These sources are a series of oceanographic phenomena, including an upwelling cell, and several estuaries, the largest of which is the Thukela River estuary. The shallow Thukela Bank, formed as a sediment plume just off the Thukela estuary mouth, is the major site of several fisheries, notably the prawn trawl and line fisheries. Riverine influence has long been thought to be important for this fishery, but oceanographic research has, until recently, suggested that the main ecosystem driver was the upwelling cell. However, recent studies have shown that the biology of the Bight is primarily maintained by riverine organic matter and nutrients, mainly from the Thukela River. This input has helped support subsistence, recreational and commercial fisheries in one of South Africa's most populated provinces. Despite the evidence of the Thukela's ecological importance for the marine environment, the possibility of increasing water abstraction from the Thukela catchment to meet the needs of a growing population has been considered in water-stressed South Africa. Policy makers will increasingly have to face trade-offs between water demands for human consumption and marine ecological functioning, which are likely to be complicated by uncertainty surrounding future climate change effects on the river and its associated marine ecosystems. This review examines the role played by the Thukela estuary, amongst other estuaries in the Bight, and assesses their overall importance for the area from an ecological and human perspective.

Keywords

Thukela River • Estuary • Fisheries • Biology • KwaZulu-Natal • South Africa • Western Indian Ocean

Introduction: Understanding South Africa's Estuaries

Studies have demonstrated that estuaries are of invaluable importance to humans supplying goods and services that can range from nursery grounds, fisheries, to recreational amenities (Costanza et al. 1997). Food production has been considered one the of most valuable goods provided by estuaries at \$520 ha^{-1} year^{-1} on average, with estuaries

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being valued at over \$22,000 ha^{-1} $year^{-1}$ (Costanza et al. 1997). The value of estuarine fisheries in South Africa alone was estimated at R433 million with a further contribution of R490 million to inshore fisheries in 2003 values (Lamberth and Turpie 2003). Thus for a water-stressed country like South Africa with competing social, environmental and economic objectives understanding the role of estuaries is an important consideration.

From its north-east border with Mozambique to the north-west border with Namibia, South Africa has about 3000 km of coastline, with 289 estuaries (Reddering and Rust 1990). As South Africa is a water stressed country, of the 289 estuaries only 37 (12.8%) are permanently open to the sea (Reddering and Rust 1990). Although each South African estuary is unique, together they share the same general characteristics: I) they have a small tidal prism, spring tides of 1.8 to 2.0 m and neap tides between 0.6 and 0.8 m; II) most are constrained from the sea for periods of time and III) most lack ebb-tidal deltas, while flood-tidal deltas are well-developed (Reddering and Rust 1990; Cooper 2001). Morphologically there are two main types of estuaries with these being divided a further six times (Table 1). Cooper et al. (1999), after considering climate, hinterland topography, coastal dynamics, sediment supply and coastal lithology, concluded that the estuaries could be divided in four broad coastal geomorphological zones or regions: I) Northern KwaZulu-Natal (the area of interest to this chapter); II) Southern KwaZulu-Natal and the former Transkei area in the Eastern Cape Province; III) the South Coast of the Eastern and Western Cape and IV) the Northern Coast of the Western Cape.

On South Africa's East Coast, where the Indian Ocean laps its shores, the Thukela is the largest river (Fig. 1). It is also one

Table 1 The morphological classification of estuaries of South Africa

Main classification	Sub-classification	Examples
Open estuary		
	Barred	Orange and Thukela Estuary
	Non-barred	Nahoon Estuary
	River-dominated	Mvoti Estuary
	Tide-dominated	Mtenty Estuary
Closed estuaries		
	Perched	Mdloti Estuary
	Non-perched	Seekoei Estuary

After: Cooper (2001)

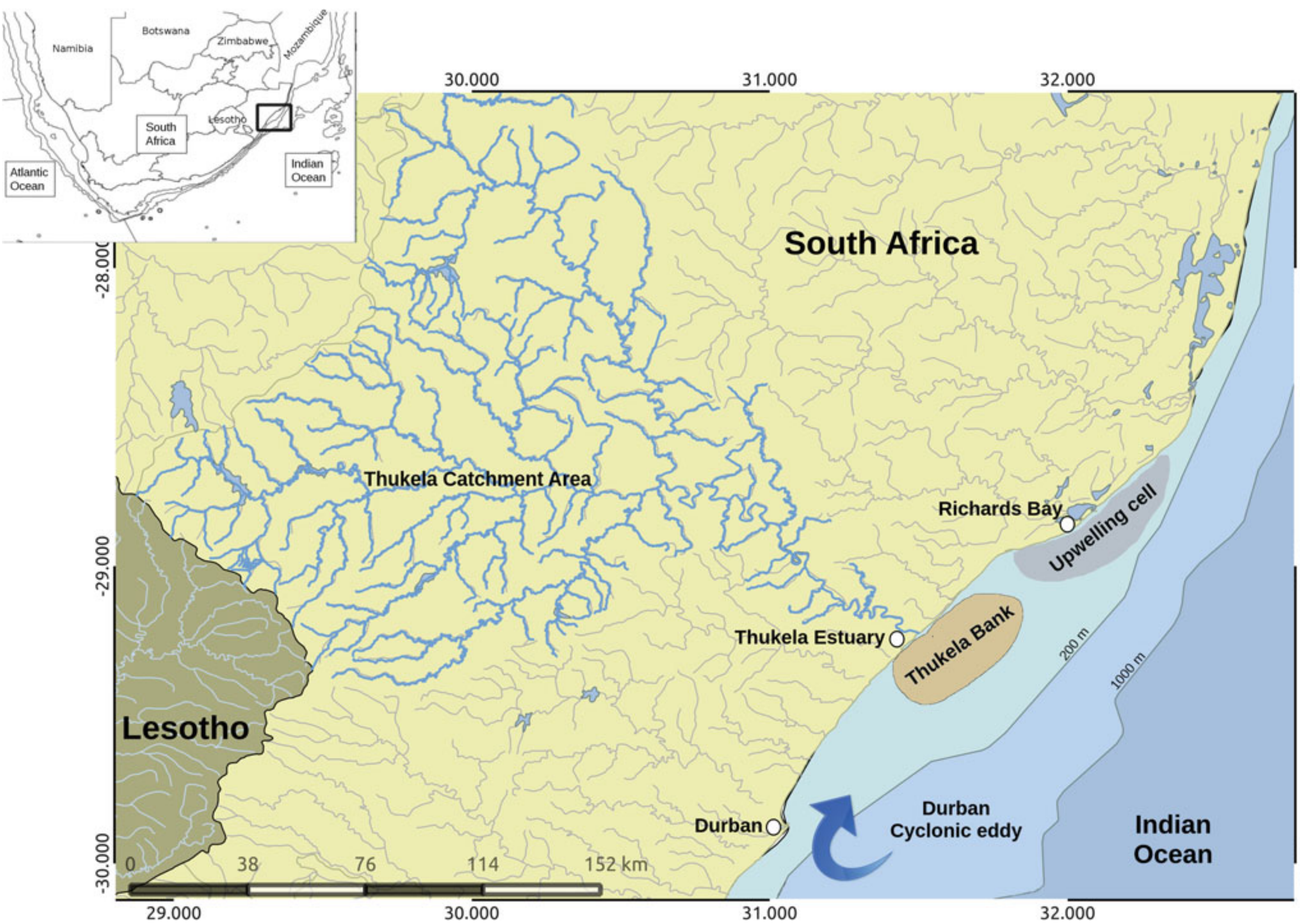

Fig. 1 Map of the KwaZulu-Natal province showing the Thukela River catchment area, estuary, and Bank (*middle*), the Cyclonic Eddy (*south*) and upwelling cell (*north*)

of the largest by volume in the country. As the focus of this book is Estuaries in the Western Indian Ocean, this chapter reviews the role played by the Thukela River/Estuary, a permanently open estuary in northern KwaZulu-Natal on the biology of the local marine system as well as role for people in the area. The chapter will examine the regional settings of the Thukela River and explore its role in the near-shore ecosystem, known as the Thukela Bank. It will also examine how this river supports not only the biology in the area, but also the commercial fisheries and an important recreational fishing industry along the east coast of South Africa. Finally, a brief look is taken at the role of the Thukela for humans in the region, including current trade-offs in uses of the river.

The Thukela River, Estuary and the Kwazulu-Natal Bight

It has been demonstrated that rivers play an important role in the near-shore neritic environment by shaping its ecology (Gillanders and Kingsford 2003). Terrigenous allochthonous material is one of the most important sources of nutrients for primary and secondary productivity in the neritic zone (Polis and Hurd 1996), and contributes to enhancing the overall productivity of these systems (Cloern 2001; Maslowski 2003). On the east coast of South Africa there are a total of 73 catchment basins that discharge their sediment loads directly or indirectly into the sea along the KwaZulu-Natal (KZN) coastline (Begg 1978). The greatest of these originate in particular from the Thukela River, the third largest river in southern Africa (Bosman et al. 2007).

The Thukela River, and some of its main tributaries, such as the Buffalo River, form in the Drakensberg Mountain Range in the south-west of KwaZulu-Natal where it borders with Lesotho at an altitude of some 3,000 m above sea level (Cooper 2001). According to the Department of Environmental Affairs (DEA 2001) the Thukela catchment area is 29,101 km^2 with 75% described as natural, 15% area used for agriculture, 8% considered degraded and approximately 1% of the catchment considered to be urban. The vegetation surrounding most of the catchment is comprised of a range of woodlands, coastal forest, montane forest, thicket and grasslands (Fairbanks and Benn 2000). Sugarcane monoculture is also present in the agriculturally-transformed catchment area (Dominy et al. 2001). On its path to the sea the Thukela is dammed seven times. At the historical time that these dams were built no environmental assessments were carried out on their effects, such as on river flow. However, possible future plans for damming the Thukela River have included 16% to 44% flow reduction from present (Lamberth et al. 2009).

The Thukela River eventually reaches the ocean at the KZN Bight (hence forth the “Bight”) (Fig. 1). The Bight

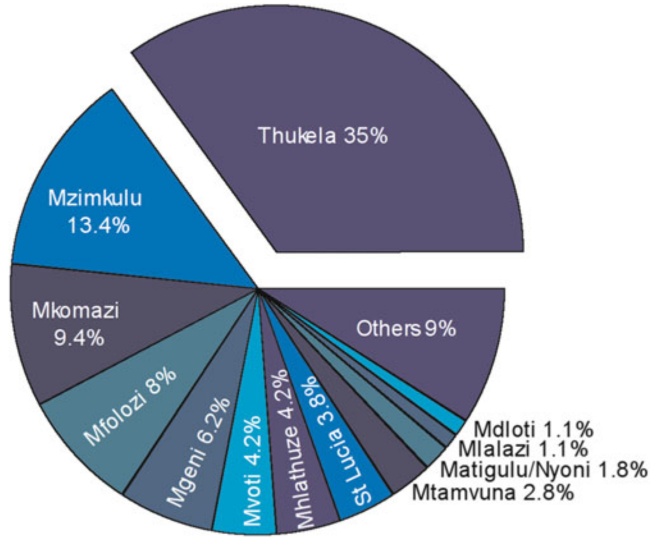

Fig. 2 Proportional contributions to mean annual run-off for all those rivers with greater than 1% of the total freshwater input to the KwaZulu-Natal coastline (Modified from Hutchings et al. 2010)

extends for ~160 km from St Lucia to an area just south of Durban on the East Coast of South Africa. It is about 50 km wide at its broadest point, offshore of the Thukela River (Lutjeharms and de Ruijter 1996). Three major processes, which introduce nutrients, have been suggested as possible drivers for primary production in the Bight: I) topographically-induced upwelling off Richards Bay (Meyer et al. 2002); II) a cyclonic lee-eddy off of the coast of Durban (Roberts et al. 2010); and III) a series of fluvially-induced processes dominated mainly by the Thukela River (de Lecea et al. 2013).

The main drivers of the fluvially-induced processes are the Thukela, Mzimkulu, Mkomazi, Mfolozi, Matigulu, Mlalazi and Mhlathuze rivers, amongst others (Fig. 2). Hutchings et al. (2010) estimated from their calculations that the total nitrogen (N) entering the system through these estuaries was on average 2,333 t year^{-1}, which appears minor in comparison to the authors estimated value for the upwelling cell of 289,154 t year^{-1} N. However, the upwelling N concentration and flux were calculated from the relatively limited nutrient studies of the Bight. Amongst the rivers the Thukela plays an important role. Its estuary is river-dominated and therefore small, allowing for most matter to pass through the mouth into the sea without being deposited, consequently forming the Thukela Bank and a large organic matter plume.

At present the Thukela River accounts for more than 35% of the freshwater entering the Bight with an annual run-off of 3,865 $\times$ 10^6 m^3 and a sediment output estimated at approximately 6.79 $\times$ 10^6 m^3 year^{-1} that is discharged directly into the Bight (Whitfield and Harrison 2003; Hutchings et al. 2010). The KZN region has well-defined wet and dry

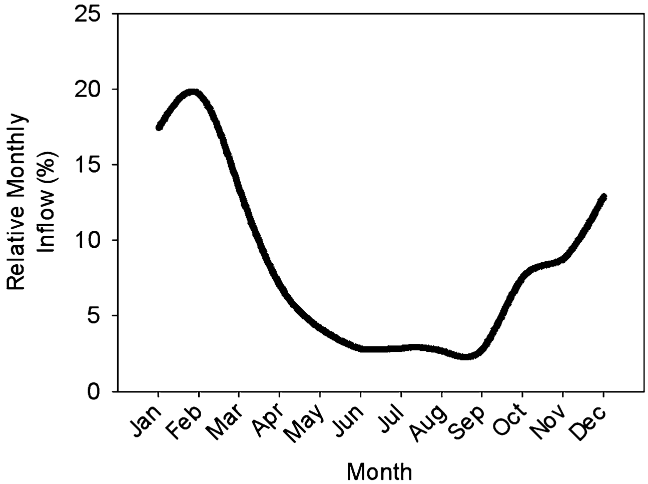

Fig. 3 The monthly freshwater inflow distribution for rivers in the KwaZulu-Natal coastline as a percentage of annual inflow. The "dry" winter months are May to September and the "wet" summer months are October to April (Modified from Hutchings et al. 2010)

seasons, with rainfall in the order of 1,000 to 1,200 mm year^{-1} (Day 1981). The height of the rainy season is in January with a mean monthly precipitation of 118 mm, while August at the peak of the dry season has a mean monthly precipitation of 39 mm (Hunter 1988). As expected, this increase in rainfall has a direct impact on the freshwater inflow into the Bight (Fig. 3). The Thukela River subsurface nutrient data shows that the river outflow extends approximately 25 km from the shore (Meyer et al. 2002).

The high sediment input from the Thukela River, aided by a complex current system over the shelf, has over time created the Thukela Bank (Bosman et al. 2007), a large mud bank important for fisheries (Lamberth and Turpie 2003). The Thukela Bank, covering an area of ~300 km^2, is located off the Thukela River towards the North-East part of the Bight, where it extends from 200 m to 16 km offshore (Fennessy and Groeneveld 1997). It is of extreme importance as it is the only near-shore area on the east coast of South Africa where prawn trawling is possible (Fennessy et al. 1994).

Despite the size and characteristics of the Thukela River, there are a series of oceanographic processes that were considered the main drivers of the Bight and hence the ecosystems of the Thukela Bank until recently. The Agulhas Current impinges upon the coast and strongly influences the oceanography of the generally very narrow continental shelf region in the area (de Ruijter et al. 1999). The shelf widens at the northern end of the Bight and local bathymetry induces upwelling (Lutjeharms et al. 2000). The area of active upwelling is known to be a point source of nutrients for the entire Bight. It has the highest primary productivity of the entire region when active. To the south of the Bight near Durban a cyclonic gyre is suggested to trap water for a period of time, circulating it within the shelf before reintroducing it into the Agulhas Current (Schumann 1982).

Table 2 Maximum nitrate levels and chlorophyll-a concentrations recorded n the KwaZulu-Natal Bight

Location	Nitrate concentration (μmol l^{-1})	Chl-*a* concentrations (mg m^{-3})	References
Durban Eddy	2.3	1.7–2.8	Carter and d'Aubrey (1988) and Barlow et al. (2008)
Central Bight region	1.01–1.86	2.1	Meyer et al. (2002) and de Lecea (2012)
Upwelling cell	Up to 16	1.5–3.2	Meyer et al. (2002) and Barlow et al. (2008)

In spite of several nutrient sources entering the system, the East Coast of South Africa, including the Thukela Bank, has been described as both oligotrophic and mesotrophic in nature (Bustamante et al. 1995; Barlow et al. 2013). The discrepancy in nutrient classification most likely arises from differences in the local conditions at the time of sampling. Studies on nutrient characteristics indicate that the concentrations of nitrate have generally ranged from 1.0 to 7.0 μmol l^{-1} across the Bight (Oliff 1973). However, this can vary with region and hydrographic phenomena, with levels up to 16 μmol l^{-1} having been recorded in the upwelling region (Table 2). Clearly, significant nutrient inputs to the Bight occur in the upwelling areas, which would conceivably enrich the otherwise oligotrophic water of the Bight. It is worth noting, however, that the samples in the Meyer et al. (2002) study were collected in July, the dry season and lowest river flow (Fig. 3) in KZN, when riverine input of nutrients is likely to be at its lowest. It is therefore unlikely that the authors captured the full importance of the river outflows for the system.

In terms of the question of the relative role of oceanographic versus riverine nutrient inputs for the ecosystem, recent studies have demonstrated that riverine input, especially that from the Thukela River, played a much more important role for the ecosystems of the Thukela Bank than previously thought (de Lecea et al. 2016; Omarjee 2012; de Lecea et al. 2013). These studies collected samples in a wet and dry season and found that during the wet season, and due to the upwelling cell not being active, the highest nutrient levels in the Bight were on the Thukela Bank (Fig. 4) (Omarjee 2012). The nutrient input through the upwelling cell is not permanent, and the upwelling does not occur at regular intervals (Hutchings et al. 2010). On the other hand, the nutrient input through the Thukela Estuary into the Thukela Bank system should be a constant input during the wet season for several months (as seen in Fig. 3). This is not the case during the dry season when nutrient levels have been found to be homogeneous throughout the Bight, except when upwelling occurs

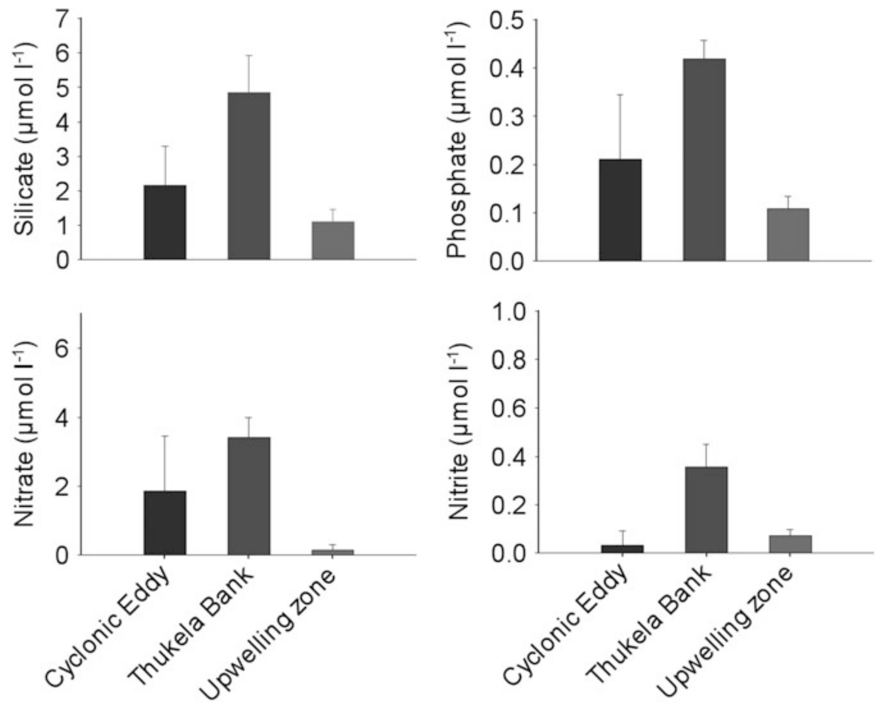

Fig. 4 Surface water nutrient levels from three different sites along the KwaZulu-Natal Bight during the wet summer season when the upwelling cell is not active. See Fig. 1 for locations of sites (Modified from de Lecea 2012)

(de Lecea 2012). Additionally, two recent studies on the impacts of the input of riverine organic matter on the ecosystem of the Bight (de Lecea et al. 2013), and the role played by the Thukela River for the Thukela Bank food-web (de Lecea et al. 2016), found that riverine organic matter was more important for the demersal than for the pelagic environment (de Lecea et al. 2015). The former mapped the sediment $\delta^{13}C$, $\delta^{15}N$, %C_{org}, %N and C:N ratios for an area 165 km long (north to south) and over 50 km wide at its widest point (east to west) and depths from ~ 20 m to ~180 m (Fig. 5) (de Lecea et al. 2013). The sediment isotopic data showed that although organic matter was well-mixed throughout the Bight in both seasons, riverine organic matter dominated most of the Bight. Only at the Bight's northern and southern edges did oceanic organic matter increase in importance for both the pelagic and demersal food-webs (de Lecea et al. 2015; de Lecea et al. 2013). Both studies found that the demersal ecosystem of not just the Thukela Bank, but the entire Bight, was driven by the organic matter entering the system through the KZN estuaries, mainly the Thukela Estuary (de Lecea et al. 2016; de Lecea et al. 2015; de Lecea et al. 2013).

The Biology of the Thukela Bank

Unlike the oceanography of the Bight, its biology, including that of the Thukela Bank, has received very little detailed research focus compared to South Africa's West Coast. Ayers and Scharler (2011), who produced a biological model for the Bight, described the area as data-poor. There are none-the-less a considerable number of studies looking into the biology of the Thukela Bank, mainly driven by research on the area's fisheries (Fennessy and Groeneveld 1997; Lamberth et al. 2009). Studies have suggested the important role of the local estuaries, mainly the Thukela estuary, on the biology of the Thukela Bank (Lamberth et al. 2009).

The low nutrient levels described earlier, impact the primary productivity of the region, and in cases when the upwelling cell is not active, the productivity in the area is low (Fig. 6a, b). Studies have indicated that the primary production of the Bight has a varied range (Table 2), but despite these large spatial differences, the primary productivity for the entire Bight has been described as low (Carter and Schleyer 1988; Bustamante et al. 1995; Omarjee 2012).

There are a paucity of studies on zooplankton within the Bight. The earliest record of a study on zooplankton in the Bight dates from the 1960's (Shipley and Zoutendyk 1964). Very little research has been conducted on zooplankton since then, making it difficult to obtain a clear picture on the role of the Thukela River for the zooplankton ecology. What is known, is that the Bight is an important location for larvae of non-planktonic species as well as purely planktonic species (Beckley et al. 2002). Hutchings et al. (2002) highlighted the important role played by the cyclonic eddy in trapping ichthyoplankton and eggs in the Bight, increasing their chance of survival, before being transported south. The authors also described the importance of the Bight in generating recruitment for the entire shelf region south of the Bight, by offering shelter from the Agulhas Current. In this regard, the Thukela plays an important role. Firstly, the

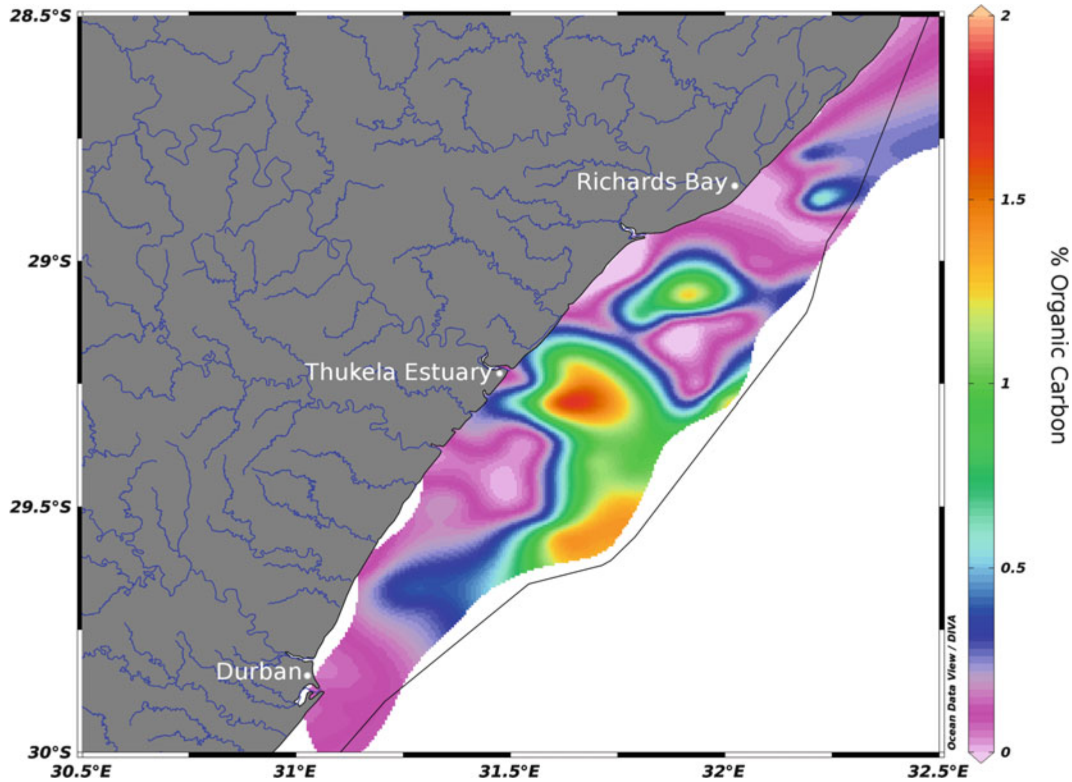

Fig. 5 Map showing the percentage organic carbon (%C_{org}) accumulated in the surface sediments of the KwaZulu-Natal Bight. Notice the funnel-shaped plume protruding off the Thukela Estuary (Modified from de Lecea et al 2013; see reference for stable isotope maps)

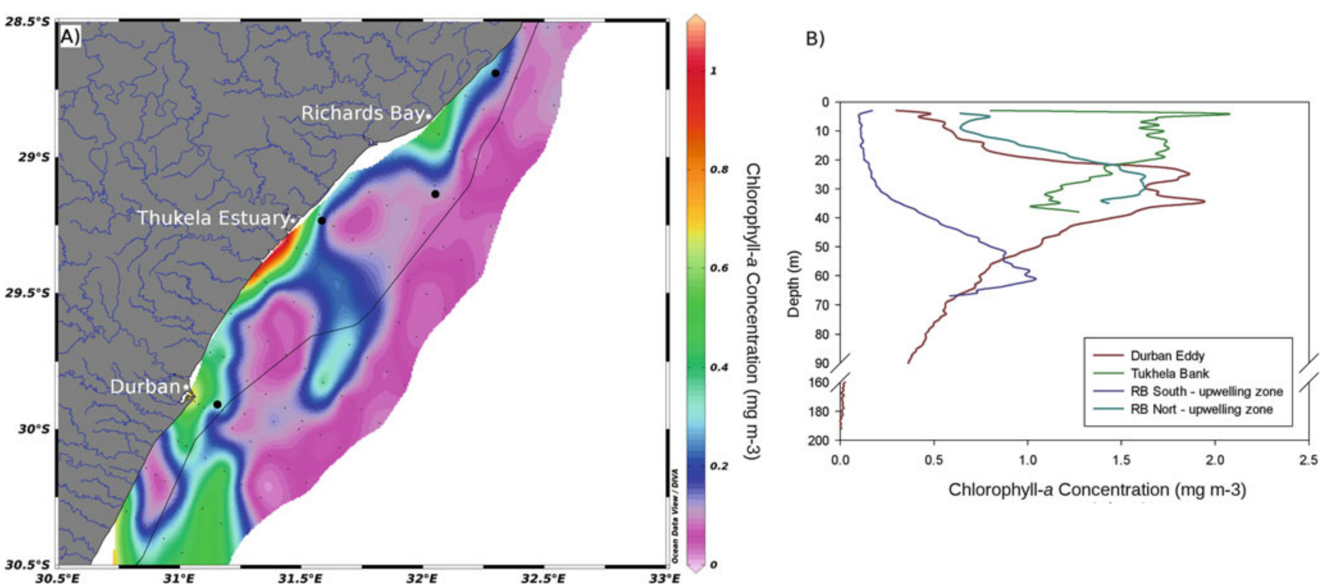

Fig. 6 (**a**) Map of the surface waters of the KwaZulu-Natal Bight showing increased chlorophyll-a concentrations off the Thukela Estuary during the wet season of 2010 (Modified from Omarjee 2012). (**b**) Chlorophyll-a concentrations with depth for four different locations. Richards Bay North and South are areas where the upwelling cell is known to occur, although not found at time of sampling (Modified from de Lecea 2012). Site locations marked with a *black dot* in map (**a**) and shown in Fig. 1

Thukela has generated the bank through deposition over time. Secondly, it provides nutrients that drive zooplankton food-webs for the Thukela Bank. A recent study indicated that terrestrial organic matter from the Thukela River drove the zooplankton food-web of the adjacent Bank (de Lecea et al. 2015). Additionally, there is evidence demonstrating that the organic matter from the Thukela Estuary plays an important role on the ecology of filter feeders along the coast (Porter et al. 2014).

Research has shown that an increase in freshwater flow in the Thukela Estuary, from the usual 5 to 30 $m^3\ s^{-1}$ to values greater than $>100\ m^3\ s^{-1}$, leads to a decline in fish species within the estuary from 31 to just 13 different taxa (Whitfield and Harrison 2003). Yet, increases in water input from the Thukela have been found to increase the prawn catches considerably (Turpie and Lamberth 2010) and it has been suggested that anthropologically-induced changes in water regime have the potential to affect linefishery catches (Lamberth et al. 2009). Evidence also indicates that the seasonal changes affect the ecology of the Thukela Bank. For example, *Panaeus indicus* dominates during the wet season and *Metapenaeus monoceros* dominates during the dry season (Demetriades and Forbes 1993). This could quite plausibly be linked to seasonal changes in river flow. Additionally, under high flow conditions the estuarine zone may be pushed out to sea (Whitfield 2005). This would reduce the already small chances of nutrients and organic matter getting trapped in the estuary itself, making them more available for the marine environment. Organic matter mainly from the Thukela and to a lesser extent from other rivers in the region supported the entire demersal food-web of the Bight and, especially the demersal food-web of the Thukela Bank (de Lecea et al. 2013). By extension, organic matter from the Thukela would be important for supporting the inshore commercial prawn trawl fishery. Reinforcing this, Ayers et al. (2013) demonstrated that prawn recruitment in the Thukela Bank was indirectly impacted by anthropogenic and environmental factors affecting the estuaries in the Bight.

The Thukela River Estuary and People

Fisheries

South African estuaries, including the Thukela, have been identified as important to a variety of estuarine fisheries and for marine inshore and shore-based fisheries, such as boat or shore-based linefisheries (Lamberth and Turpie 2003), in addition to the many other goods and services that they provide The Thukela provides a nursery area for resident and non-resident species, particularly marine fish taxa (Whitfield and Harrison 2003), and it provides a habitat for resident species. The Thukela also provides nutrient inputs and organic matter important for primary productivity and demersal ecosystems, as discussed earlier. Thus the Thukela and other estuarine systems in KZN are critical to the survival or prosperity of many exploited and unexploited species.

Many estuarine species (predominantly fish) are legally and illegally harvested for commercial, recreational and subsistence purposes in South Africa (Lamberth and Turpie 2003). In KZN 71 species of fish are harvested in estuaries. Greater abundance and diversity of fish is found in permanently open, large estuaries, like the Thukela (Lamberth and Turpie 2003). A variety of fishing activities takes place in the Thukela estuary, including illegal, largely subsistence gillnetting and seine-netting (Lamberth and Turpie 2003). Increasingly, though, gillnetting and seine-netting effort has been commercially driven by crime-syndicates selling poached fish to near and distant markets (Everett 2014). Linefishing and castnetting is commonplace in all KZN estuaries (Lamberth and Turpie 2003). Although by far the largest portion of fishers in KZN are recreationists, in the Thukela estuary mainly subsistence and not recreational boat-based (line-)fishing takes place, because of safety and accessibility concerns for recreational anglers. Instead recreational linefishing is more focused on marine areas nearby (Everett 2014).

These nearby marine ecosystems driven by or intrinsically linked to the Thukela system support not just the recreational linefishery. Rocky shore and sandy beach invertebrates are collected by local communities for subsistence. A commercial oyster fishery harvests oysters from the Thukela estuary mouth south to the Mzimkhulu estuary and from the intertidal down to shallow subtidal rocky reefs (Everett 2014). There are also the commercial, marine prawn trawl and offshore linefisheries (Fennessy and Groeneveld 1997; Dunlop and Mann 2013). The prawn trawl fishery includes an active offshore component along the edge of the continental shelf (600–1000 m), with a retained catch of 365 tons in 2012 (Everett 2014), and is likely too far out to be influenced by the Thukela. There is also an almost commercially extinct inshore component; the retained catch in 2012 was just 700 kg, caught on the one active vessel (Fig. 7; Everett 2014). The inshore trawling grounds trace the Thukela Bank and mud accumulation areas of other rivers of the region from 1–45 m depths and depend on freshwater input from estuaries (Fennessy and Groeneveld 1997; Turpie and Lamberth 2010). Given that these fisheries depend to an extent on the level of water output from the Thukela and other estuaries in the region, any changes to these flows could have important economic consequences for fisheries.

Estuaries add considerably to commercial fisheries value in terms of direct catches and through their contribution to inshore marine fisheries (Lamberth and Turpie 2003). The value of the Thukela estuary and its associated (ecosystem) goods and services has not been individually calculated. But Lamberth and Turpie (2003) estimated that the total estuarine fisheries catch for South Africa was 2,480 tons $year^{-1}$. Of this, 50% of the catch was attributed to seine and/or

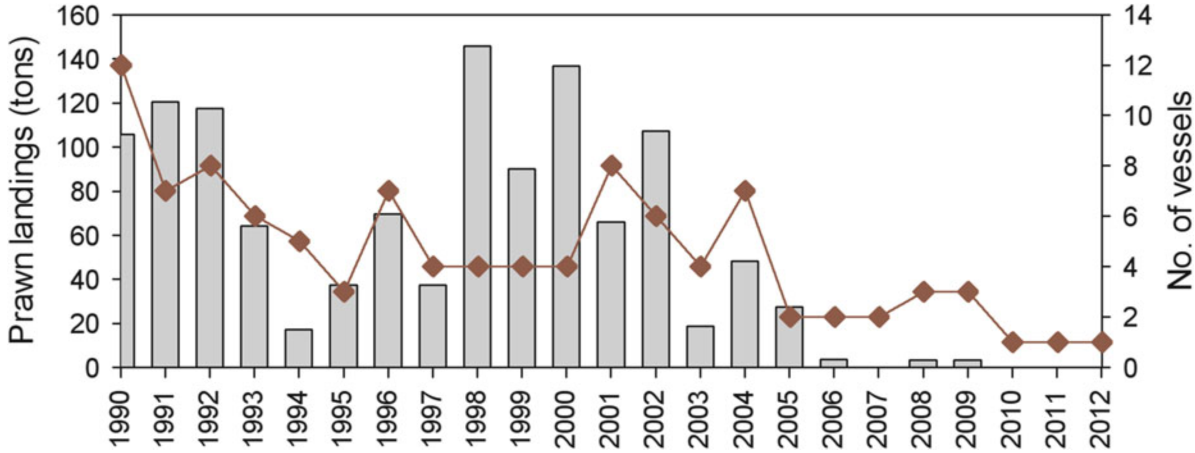

Fig. 7 Landings (*histogram*) and number of active vessels (*red diamonds*) for the inshore prawn trawl in KwaZulu-Natal for the period 1990 to 2012 (Modified from Everett 2014)

gillnet fisheries, 46% to recreational and 4% to traditional trap and spear fisheries. These same authors calculated the value of these estuarine fisheries to be R433 million, with a further contribution to the inshore marine fisheries of R490 million (Lamberth and Turpie 2003). For the Thukela crustacean (prawn) trawl for the 1992–2002 period, when both inshore and offshore components were active, catches averaged ~700 tonnes. Gross output was estimated to be R36.7 million per year with gross value added of R13.8 million; the inshore component accounted for R8.5 million of gross output (Turpie and Lamberth 2010). Looking at these figures, it is possible to appreciate that the dwindling of the inshore prawn trawl would have had a measurable economic impact for fisheries in the region.

The remaining commercial, recreational and subsistence fisheries also have important economic and social values. As an important recreational activity in KZN, fishing may generate local revenue related to boating, travel, equipment and bait (Beckley et al. 2008)), which is important to the economy of the area (Turpie and Lamberth 2010). Recreational shore-based angling in estuaries, for example, had as many as 76 000 angler visits, and represented an investment of R19 million in fishing equipment, bait and visits, in just two urban estuaries in 2000 (Pradervand et al. 2003). It follows that other recreational fishing activities that either take place in the Thukela estuary or which are dependent on the system ecologically, represent a significant monetary investment for the province. Most of this money flows through small, localised businesses in the area that supply this recreational industry. With numbers of up to 68 087 anglers for the shore-based linefishery and up to another 20 274 recreational and charter boat linefishers (Dunlop and Mann 2012; Dunlop and Mann 2013), fishing represents a significant recreational and subsistence opportunity for the province and must have a large social impact.

Subsistence fishing that depends directly or indirectly on the Thukela system is very important for local communities. Subsistence fishers in and around estuaries in KZN mostly fish to provide food for themselves and their families and, in the uncommon event of a surplus catch, to barter or sell their catch (Everett 2014). For those living in the Thukela region about 60% of their catch comes from estuarine systems (Clark et al. 2002). They tend to come from low-income or no-income households in rural areas, many of which (45.9%) spend more than 60% of their annual income on food (Clark et al. 2002; Branch et al. 2002). Unemployment and poverty are high among these individuals and education levels are low (Branch et al. 2002), leaving them with few choices. In 2002 there were estimated to be 1959 households in northern KZN (the Thukela region) engaged in subsistence fishing (Clark et al. 2002). These figures, while more than a decade old, are not likely to have decreased with continued high unemployment rates, poverty and a burgeoning population in South Africa. Everett (2014) suggests that current figures for the entire KZN province are around 2000 to 4000 'true' subsistence linefishers. There are also an increasing number of line fishermen who fish to sell their catch rather than eat it, an unknown number of illegal gill-netters, seine-netters and lobster fishermen and approximately 556 (or more) individuals harvesting intertidal invertebrates (Everett 2014). Because the catches are largely for personal consumption and the sector is informal, the value of these catches is hard to quantify. Economic estimates put the value of the subsistence linefishery in the region of R150 000 to R920 000 (South African Rands), R445 000 for the intertidal invertebrate fishery, and an unknown but likely high (a few million Rands) value for the heavily fished and unsustainable illegal gill and seine-net fisheries (Everett 2014). The true value of these subsistence fisheries though is measured in the ability for poor, vulnerable households to supplement their food supply and often provide much-need protein for their families, as is the case in many poorer countries and households worldwide (FAO 2014).

Due to the high population density of the KZN coastline there has been intense recreational and subsistence use of inshore species, leaving many invertebrate and fish species overexploited (Griffiths et al. 2010). This could jeopardize future food security, not to mention ecosystem health and biodiversity. Furthermore, estuarine subsistence fishers tend to be extremely localised (Lamberth and Turpie 2003). This means that they will be highly dependent on particular estuaries, making them vulnerable to any changes in these estuaries that might undermine an important food source. Environmental change is happening and will continue to happen in and around the Thukela. This change operates through (coastal) population growth and, along with it, development, pollution, water abstraction and increasing demands on the estuarine system for a variety of goods and services. Albeit that there are uncertainties surrounding climate change and the impacts it might have on hydrology of the Thukela, it is certain that changes will take place.

Such changes in freshwater flow through the Thukela estuary are quite significant for fisheries in the area, including the commercial and recreational boat-based linefishery, which largely takes place on the Thukela bank. Catches of this fishery have been correlated to freshwater output of estuaries in the region, of which over 40% of the mean annual runoff flows out of the Thukela estuary (Lamberth et al. 2009). And as a result, Lamberth et al. (2009) predict that future planned water abstractions on the Thukela (and other rivers in the region) are likely to reduce catches. Turpie and Lamberth (2010) further found that under various freshwater flow reduction scenarios (up to 44% reduction in flow in the worst case scenario) that while catches for the recreational linefishery remained largely unaffected, there were decreases in commercial linefishery catch volumes by 20% and value by 17%. The prawn trawl fishery also showed declines of 0.7–11% in catches (Turpie and Lamberth 2010). Turpie and Lamberth (2010) caution that there is uncertainty surrounding their predictions; some of the variablity in catches could not be accounted for. Climate change could further complicate matters through increased uncertainty, and there could be cumulative, long-term effects of reducing estuarine outflow through abstractions (Turpie and Lamberth 2010).

Water Abstractions

By volume the Thukela is the largest river in South Africa, has a catchment area of 29 036 km^{2}, supplies more than 40% of mean annual runoff to the KZN coastline (Taylor et al. 2001; Lamberth et al. 2009), and is the longest (316 km) river in KZN. But it is also amongst the most impounded rivers in the province, with more than 672 dams (Rivers-Moore et al. 2007). The seven major dams in the Thukela catchment: Spioenkop, Wagendrift, Ntshingwayo (previously Chelmsford), Craigieburn, Woodstock, Kilburn and Zaaihoek have a full supply capacity of 1164.9×10^6 m^3 (Taylor et al. 2001; DWAF 2004). These and other water abstractions on the Thukela course serve a variety of purposes, including for potable water, agricultural irrigation, industrial consumption, supply to a thermal power station and inter-basin transfers (IBTs). The most notable IBT is the Thukela-Vaal water transfer that provides water to Gauteng, South Africa's industrial and economic hub (Taylor et al. 2001; DWAF 2004). Future developments have been proposed for the Thukela, with some changes already recently implemented (Department of Water Affairs and Forestry (DWAF) website; Lamberth et al. 2009; DWS 2015).

South Africa is a water-stressed country, that is undergoing development and population growth. Demands for water are likely to become more pressing in the future. For the case of the Thukela this raises a potential conflict. The Thukela and to a lesser extent the other estuaries in KZN are important for ecological functioning and fisheries in the region, which represent important social and economic activities and a food source for the poor. As such, decisions on water abstraction and use in the region will have a major impact biologically, economically and socially. This raises the need for collaborative management between different governmental sectors: water, environment, fisheries and local government, where management objectives need to be clearly defined and trade-offs need to be identified. In a country where water is scarce and social and economic problems are not, it will be impossible to simultaneously maximise economic, social and ecological goals. However, well-informed management between government sectors will go a long way to ensuring a beneficial future for the Thukela and South Africa's other estuaries. This is especially important given uncertainties around future hydrology and environment in the face of global change.

Climate Change and the Thukela estuary

South African estuaries, their ecology and fish assemblages are vulnerable to climate change (James et al. 2013). Fisheries with an estuarine connection in South Africa, including those with estuarine-dependent target species, have been identified as those likely to be most severely impacted by climate change. Effects of climate change on physical factors already have uncertainty attached, but the feed-on effects on biological systems and fisheries have been speculative and uncertain at best (Clark 2006).

Nevertheless, some studies have taken the important step of examining potential consequences for estuaries like the Thukela. James et al. (2013) suggest that short-term effects on estuaries could emerge from increases in temperature, rainfall and the frequency and severity of coastal storms on the subtropical east coast of South Africa. Longer term changes may result from factors such as sea-level rise and ocean acidification (James et al. 2013). Increases in high rainfall events could potentially increase freshwater and sediment flow through estuaries. This could smother benthic communities and fish, increase estuarine turbidity and change community structures (James et al. 2013). Based on the findings summarised in the earlier section entitled *The Thukela River, Estuary and the KwaZulu-Natal Bight*, increased inflow of sediment and freshwater from the Thukela estuary in such high rainfall events could also significantly affect the size and function of Thukela Bank ecosystem, which it drives.

Recent research suggests that climate change will *likely* increase rainfall and river runoff in the region. Although, depending on modelling methods applied, rainfall and associated river runoff can also be shown to decrease. In short, there is some degree of uncertainty in river runoff in the area under future climate scenarios (Graham et al. 2011). Dealing with this uncertainty has important consequences for management of rivers and hence estuaries in the area, from human and environmental perspectives. In the case of reduced future rainfall there is the possibility that impoundments on water courses such as the Thukela could exacerbate environmental problems, e.g. reduced flow could impact the prawn fisheries of the Bight in combination with other effects of climate change. Reduced rainfall could also compound human problems, e.g. reduced surface run-off to fill impoundments while agriculture may require more piped water in the face of a drier climate. In the best case scenario there will be increased future rainfall and high rainfall events. Even so, unless this increase is extremely voluminous, the increased water abstraction on the Thukela (and a growing population) in future means that it is more likely that additional rainfall will inevitably be taken up as a windfall for human consumption. Anthropogenic change in freshwater output is therefore more likely to be seen acting through water abstraction on the Thukela estuary than through predicted increases in rainfall for the region. A high rainfall scenario will, of course, make meeting both human and natural system needs easier than a reduced rainfall scenario. Unpredictable or uncertain rainfall could be yet more problematic for management.

Climate change is also known to act synergystically on ecosystems along with other human-induced changes such as fishing (Harley et al. 2006). As biological systems, estuaries in South Africa are extremely vulnerable to climate change and any further decreases in fresh water entering these systems will have strong effects, such as longer and more frequent mouth closures, altered physico-chemical parameters in estuaries and decreased dilution of pollutants in estuaries (Clark 2006). There are several industries, including textile, paper, and sewerage treatment, discharging effluent into the Thukela as well as other rivers in the KZN coastline. Although the Thukela is at present at less risk than the others (Stryftombolas 2010), decreases in flow could change this. Climate induced changes to estuarine flow and potential estuarine mouth-closures will affect many migrant species as well as many marine commercial species that depend on the seasonal mouth openings of estuaries with increased rainfall and the biological communities of estuaries in general (Clark 2006).

Dealing with the uncertainty of outcomes from uncertain future climate scenarios, particularly rainfall, in relation to impact on rivers and, by extension, estuaries and the near-shore marine environment is likely to remain a major challenge for the region. Especially so, because of its direct and indirect effects on the associated ecological and human communities. Graham et al. (2011) state that consistent future climate projections for the Thukela River basin area do not exist, a common problem for developing countries. International efforts are under-way to improve regional data-sets, especially in Africa, which will allow for better regional projections of local hydrology (Graham et al. 2011). More research is needed to improve localised climate model projections, convert uncertainty to quantified risk estimations where possible, and determine potential effects of projections on natural and human systems in the Thukela area. This will allow for mitigation of or at least contingency plans for various undesirable effects on human and natural ecosystems, including the Thukela estuary and associated marine ecosystems. Reid and Vogel (2006) point out that if suitable responses and choices for adaptation to risks such as climate change are not taken, development in KZN could be subverted.

This of course presents the tricky situation for governance and management needing to balance human and ecological water requirements. On the one hand there is the growing, developing population with increasing demands for water. But on the other, there is legislation governing the maintenance of ecological integrity of estuaries by regulating the quantity of water that reaches them. Cyrus and MacKay (2007), discuss methods for determining the ecologically necessary quantity of water for the Thukela River. It will be much harder, though, to determine what quantities of water might be needed to maintain integrity of near-shore marine ecosystems. More complex still, would be determining what quantities of sediment and organic matter outflow are needed for proper ecological functioning. This is partly because of multiple human impacts affecting marine ecosystems in the region, and partly because the full effects of water abstraction on sediment and organic matter outflow will be difficult to determine.

It certainly will be complex for policy makers and managers to make decisions on trade-offs between providing water for human demands and marine ecological demands, when there is much uncertainty in requirements. Research that monitors the KZN estuarine ecosystems, including the Thukela, and near-shore marine ecosystem function, especially of the Thukela Bank, will be valuable in this regard as new water abstraction plans are implemented. Ideally, research would include concurrent monitoring of fisheries catches and estuarine flow, as suggested by Turpie and Lamberth (2010). Finally, the real challenge will be integrating this research into meaningful policy and implementation. In a review on estuarine water requirement determination, Adams (2014) succinctly put it that although good methods and frameworks exist from the South African example (and others), implementation is slow and will need stakeholder engagement, solid governance structures and adaptive management with ongoing monitoring and feedback.

Conclusion

This review set out to determine the role played by the Thukela for the biology of the near-shore marine environment as well as its role for people. The Thukela River is the longest and third largest river in South Africa. Its steep hinterland and small estuary area result in nutrients and sediment expulsion onto the continental shelf, forming the Thukela Bank. This mud bank used to support the inshore component of the only prawn fishery in South Africa, which is now almost commercially extinct. It also supports a commercially and recreationally important linefishery. The Thukela also has importance for the ecology and biology of the Bight. During the wet season it provides the nutrients and organic matter necessary to maintain a planktonic pelagic food-web on the Thukela Bank. It is also highly significant for the demersal ecosystem of the Bight, particularly the Thukela Bank, as the demersal food-web is maintained by the organic matter from the Thukela Estuary. Consequently, reductions in the freshwater outflow and sediment (with associated organic matter) load of the Thukela could have consequences not only for the estuarine system, but also for the marine ecosystems of the Bight.

Changes in flow through the Thukela estuary could result from climate change and water abstraction. In KZN, future climate scenarios remain somewhat uncertain, although an increase in rainfall and frequency for high rainfall events is possible. Nevertheless as a water stressed country with a growing, developing population, water abstraction from the Thukela will increase to meet the demands for human consumption, agriculture and industry. Projects that could reduce freshwater flow through the estuary have been suggested and in some cases approved for the Thukela catchment. Yet the impact that these could have on the Thukela Bank and estuarine ecology are not fully understood. As discussed in this chapter, changes in estuarine flow could likely have serious impacts on the local estuarine and Thukela Bank fisheries, as well as the ecology of the area. This would consequently affect the people depending on these resources, including subsistence, recreational and commercial fishers.

To enable policy makers and managers to make informed decisions on management of the Thukela system and particularly on water balance issues, a variety of ongoing research that builds on existing knowledge is needed. As discussed, this research should focus on clarifying regional future climate projections and associated impacts on the natural and human systems of the Thukela. Research that monitors the Thukela and near-shore marine ecosystems in relation to changes in estuarine outflow, sediment budgets, ecology and fisheries yields will be valuable for understanding the effects of abstraction and ecological demands.

References

Adams JB (2014) A review of methods and frameworks used to determine the environmental water requirements of estuaries. Hydrol Sci J 59(3):1–15

Ayers MJ, Scharler UM (2011) Use of sensitivity and comparative analyses in constructing plausible trophic mass-balance models of a data-limited marine ecosystem – The KwaZulu-Natal Bight, South Africa. J Mar Syst 88:298–311

Ayers MJ, Scharler UM, Fennessy ST (2013) Modelling ecosystem effects of reduced prawn recruitment on the Thukela Bank trawling grounds, South Africa, following nursery loss. Mar Ecol Prog Ser 479:143–161

Barlow R, Kyewalyanga M, Sessions H, van den Berg M, Morris T (2008) Phytoplankton pigments, functional types, and absorption properties in the Delagoa and Natal Bights of the Agulhas ecosystem. Estuar Coast Shelf Sci 80:201–211

Barlow R, Lamont T, Britz K, Sessions H (2013) Mechanisms of phytoplankton adaptation to environmental variability in a shelf ecosystem. Estuar Coast Shelf Sci 133:45–57

Beckley LE, Alexander H, Skelton PH (2002) Synoptic overview of marine ichthyology in South Africa. Mar Freshw Res 53:99–105

Beckley LE, Fennessy ST, Everett BI (2008) Few fish but many fishers: a case study of shore-based recreational angling in a major South African estuarine port. Afr J Mar Sci 30:11–24

Begg G (1978) The estuaries of Natal, Report 41. Natal Town and Regional Planning Commission, Pietermaritzburg

Bosman C, Uken R, Leuci R et al (2007) Shelf sediments off the Thukela River mouth: complex interaction between fluvial and oceanographic processes. S Afr J Sci 103:490–492

Branch GM, May J, Roberts B et al (2002) Case studies on the socio-economic characteristics and lifestyles of subsistence and informal fishers in South Africa. S Afr J Mar Sci 24:439–462

Bustamante RH, Branch GM, Eekhout S et al (1995) Gradients of intertidal primary productivity around the Coast of South Africa and their relationships with consumer biomass. Oecologia 102:189–201

Carter RA, d'Aubrey J (1988) Inorganic nutrients in Natal continental shelf waters. In: Schumann EH (ed) Coastal ocean studies off Natal, South Africa. Springer, Berlin, pp 131–151

Carter RA, Schleyer MH (1988) Plankton distributions in Natal Coastal waters. In: Schumann EH (ed) Coastal ocean studies off Natal, South Africa. Springer, Berlin/Heidelberg, pp 152–177

Clark BM (2006) Climate change: a looming challenge for fisheries management in southern Africa. Mar Policy 30:84–95

Clark BM, Hauck M, Harris JM et al (2002) Identification of subsistence fishers, fishing areas, resource use and activities along the South African coast. S Afr J Mar Sci 24:425–437

Cloern JE (2001) Our evolving conceptual model of the coastal eutrophication problem. Mar Ecol Prog Ser 210:223–253

Cooper JAG (2001) Geomorphological variability among microtidal estuaries from the wave-dominated South African coast. Geomorphology 40:99–122

Cooper A, Wright I, Mason T (1999) Geomorphology and sedimentology. Estuaries of South Africa. Cambridge University Press, Cambridge, pp 5–25

Costanza R, d'Arge R, de Groot R, Farber S, Grasso M, Hannon B, Naeem S, Limburg K, Paruelo J, O'Neill RV, Raskin R, Sutton P, van den Belt M (1997) The value of the world's ecosystem services and natural capital. Nature 387:253–259

Cyrus DP, MacKay CF (2007) The environmental reserve and its role in retaining the diversity of birds at the Thukela Estuary. Ostrich J Afr Ornithol 78:621–631

Day JH (1981) Estuarine ecology, with particular reference to Southern Africa. A. A. Balkema, Cape Town

DEA (Department of Environmental Affairs - South Africa) (2001) South African estuaries: Catchment landcover. http://www.environment.gov.za/soer/estuary/kzn.html. Access – 2014

De Lecea AM (2012) Isotopic ecosystem studies in the KwaZulu-Natal Bight. University of KwaZulu-Natal, Doctoral thesis

De Lecea AM, Fennessy ST, Smit AJ (2013) Processes controlling the benthic food-web of a mesotrophic bight (KawZulu-Natal, South Africa) revealed by stable isotope analysis. Mar Ecol Prog Ser 484:97–114. doi:10.3354/meps10311

de Lecea AM, Cooper R, Smit AJ (2015) Identifying the drivers of the pelagic ecosystem of an Oligotrophic Bight (Kwazulu-Natal, South Africa) using stable isotopes (δ C, δ N) and C:N ratio analyses. Mar Freshw Res. http://dx.doi.org/10.1071/MF15256

De Lecea AM, Smit AJ, Fennessy ST (2016) Riverine dominance of a near-shore marine demersal food-web: evidence from stable isotope and C:N ratio analysis. Afr J Mar Sci 38(1), (in Press)

De Ruijter WPM, van Leeuwen PJ, Lutjeharms JRE (1999) Generation and evolution of natal pulses: solitary meanders in the Agulhas Current. J Phys Oceanogr 29:3043–3056

Demetriades NT, Forbes AT (1993) Seasonal changes in the species composition of penaeid prawns on the Tugela Bank, Natal, South Africa. S Afr J Mar Sci 13:317–322

Dominy CS, Haynes RJ, Van Antwerpen R (2001) Long-term effects of sugarcane production on soil quality in the South Coast and Midlands areas of KwaZulu-Natal. Proc S Afr Sugar Technol Assoc 75:222–227

Dunlop S, Mann B (2012) An assessment of participation, catch and effort in the KwaZulu-Natal shore-based marine linefishery, with comments on management effectiveness. Afr J Mar Sci 34:479–496

Dunlop S, Mann B (2013) An assessment of participation, catch and effort in the offshore boat-based linefishery in KwaZulu-Natal, South Africa. Afr J Mar Sci 35:79–97

DWAF (2004) Internal strategic perspective: Thukela water management area. Department of Water Affairs and Forestry; Report No. P WMA 07/000/00/0304, South Africa

DWS (2015) Implementation and maintenance of the water reconciliation strategy for the KwaZulu-Natal Coastal Metropolitan areas. In: Department water sanitation – online. https://www.dwaf.gov.za/

Everett B (2014) Marine and estuarine fisheries along the KwaZulu-Natal coast: an inventory and brief description. No 11. South African Association for Marine Biological Research. Durban. ISBN: 978-0-620-64146-3

Fairbanks DHK, Benn GA (2000) Identifying regional landscapes for conservation planning: a case study from KwaZulu-Natal, South Africa. Landsc Urban Plan 50:237–257

FAO (2014) The state of world fisheries and aquaculture 2014. Food and Agriculture Organisation, Rome

Fennessy STT, Groeneveld JCC (1997) A review of the offshore trawl fishery for crustaceans on the east coast of South Africa. Fish Manage Ecol 4:135–147

Fennessy ST, Villacastin C, Field JG (1994) Distribution and seasonality of ichthyofauna associated with commercial prawn trawl catches on the Tugela Bank of Natal, South Africa. Fish Res 20:263–282

Gillanders B, Kingsford M (2003) Impact of changes in flow of freshwater on estuarine and open coastal habitats and the associated organisms. In: Gibson RN, Atkinson JA, Barnes M (eds) Oceanography and marine biology, an annual review, vol 40. Taylor and Francis, London, pp 233–309

Graham LP, Andersson L, Horan M et al (2011) Using multiple climate projections for assessing hydrological response to climate change in the Thukela River Basin, South Africa. Phys Chem Earth Parts A/B/C 36:727–735

Griffiths CL, Robinson TB, Lange L, Mead A (2010) Marine biodiversity in South Africa: an evaluation of current states of knowledge. PLoS One 5:e12008

Harley CDG, Hughes AR, Hultgren KM et al (2006) The impacts of climate change in coastal marine systems. Ecol Lett 9:228–241

Hunter IT (1988) Climate and weather off Natal. In: Schumann EH (ed) Coastal ocean studies off Natal, South Africa. Springer, Berlin/Heidelberg, pp 178–208

Hutchings L, Beckley LE, Griffiths MH et al (2002) Spawning on the edge: spawning grounds and nursery areas around the southern African coastline. Mar Freshw Res 53:307–318

Hutchings L, Morris T, van der Lingen CD et al (2010) Ecosystem considerations of the KwaZulu-Natal sardine run. Afr J Mar Sci 32:413–421

James NC, van Niekerk L, Whitfield AK et al (2013) Effects of climate change on South African estuaries and associated fish species. Clim Res 57:233–248

Lamberth SJ, Turpie JK (2003) The role of estuaries in South African fisheries: economic importance and management implications. Afr J Mar Sci 25:131–157

Lamberth SJ, Drapeau L, Branch GM (2009) The effects of altered freshwater inflows on catch rates of non-estuarine-dependent fish in a multispecies nearshore linefishery. Estuar Coast Shelf Sci 84:527–538

Lutjeharms JRE, de Ruijter WPM (1996) The influence of the Agulhas Current on the adjacent coastal ocean: possible impacts of climate change. J Mar Syst 7:321–336

Lutjeharms JRE, Valentine HR, Van Ballegooyen RC (2000) The hydrography and water masses of the Natal Bight, South Africa. Cont Shelf Res 20:1907–1939

Maslowski J (2003) Effects of trophic conditions on benthic macrofauna in the vicinity of the River Swina mouth (Pomeranian Bay; southern Baltic Sea). Oceanologia 45(1):41–52

Meyer AA, Lutjeharms JRE, de Villiers S (2002) The nutrient characteristics of the Natal Bight, South Africa. J Mar Syst 35:11–37

Oliff WD (1973) Chemistry and productivity at Richards Bay. CSIR NPR: Oceanography Division Contract Report. Vol. CFIS 37B. Durban

Omarjee A (2012) Phytoplankton studies in the KwaZulu-Natal Bight by. University of KwaZulu-Natal, MSc thesis

Polis GA, Hurd SD (1996) Linking marine and terrestrial food webs: allochthonous input from the ocean supports high secondary productivity on small islands and coastal land communities. Am Nat 147:396–423

Porter SN, Kaehler S, Branch GM, Sink KJ (2014) Riverine subsidies for inshore filter-feeder communities: potential influences on trophic patterns among bioregions. Mar Ecol Prog Ser 498:13–26

Pradervand P, Beckley LE, Mann BQ, Radebe PV (2003) Assessment of the linefishery in two urban estuarine systems in Kwazulu-natal, South Africa. Afr J Mar Sci 25:111–130

Reddering JSV, Rust IC (1990) Historical changes and sedimentary characteristics of southern African estuaries. S Afr J Sci 86:425–428

Reid P, Vogel C (2006) Living and responding to multiple stressors in South Africa—Glimpses from KwaZulu-Natal. Glob Environ Chang 16:195–206

Rivers-Moore NA, Goodman PS, Nkosi MR (2007) An assessment of the freshwater natural capital in KwaZulu-Natal for conservation planning. Water SA 33:665–674

Roberts MJ, van der Lingen CD, Whittle C, van den Berg M (2010) Shelf currents, lee-trapped and transient eddies on the inshore boundary of the Agulhas Current, South Africa: their relevance to the KwaZulu-Natal sardine run. Afr J Mar Sci 32:423–447

Schumann EH (1982) Inshore circulation of the Agulhas Current off Natal. J Mar Res 40:43–55

Shipley AM, Zoutendyk P (1964) Hydrographic and plankton collected in the south west Indian Ocean during the SCOR international Indian Ocean Expedition 1962–1963. University of Cape Town Oceanography Department, vol 3. Cape Town

Stryftombolas IC (2010) Ecotoxicological assessment of the impact of paper and pulp effluent on the lower Thukela River catchment, KwaZulu-Natal, South Africa and the toxicological assessment of similar effluent from two other mills. University of Johanesburg, MSc thesis

Taylor V, Schulze RE, Jewitt GPW et al (2001) Practical issues of HELP: examples from the Thukela Basin in South Africa. In: Proceedings of the AWRA/University of Dundee international specialty conference on globalization and water management. http://www.awra/org

Turpie JK, Lamberth SJ (2010) Characteristics and value of the Thukela Banks crustacean and linefish fisheries, and the potential impacts of changes in river flow. Afr J Mar Sci 32:613–624

Whitfield AK (2005) Fishes and freshwater in southern African estuaries – a review. Aquat Living Resour 18:275–289

Whitfield AK, Harrison TD (2003) River flow and fish abundance in a South African estuary. J Fish Biol 62:1467–1472

The Mangroves of Ambanja and Ambaro Bays, Northwest Madagascar: Historical Dynamics, Current Status and Deforestation Mitigation Strategy

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Abstract

Madagascar contains Africa's fourth largest extent of mangroves, representing approximately 2% of the global distribution. Since 1990, more than 20% of Madagascar's mangrove ecosystems have been heavily degraded or deforested due primarily to increased harvest for charcoal and timber and the expansion of agriculture and aquaculture. Anthropogenic-driven loss is particularly prominent in the north-western Ambanja and Ambaro Bays (AAB). At over 24,000 ha, AAB is one of Madagascar's largest mangrove ecosystems, including prominent estuaries fed by rivers and streams originating in the country's highest mountain range. Similar to the national rate, AAB has experienced approximately 20% loss since 1990, driven primarily by over-harvesting for charcoal and timber. Continued loss threatens the livelihoods and wellbeing of thousands of residents who rely on the many goods and services provided by a healthy, relatively intact mangrove ecosystem. To combat this loss, Blue Ventures (BV), in partnership with local communities and the University of Antananarivo, is working to protect, restore and encourage the sustainable use of mangroves. BVs' Blue Forests project aims to help maintain and diversify local livelihoods and to sustainably manage mangroves and their associated biodiversity in AAB, as well as throughout western Madagascar. This chapter provides an overview of the biophysical characteristics, historic dynamics and current status of the AAB mangrove ecosystem, and mitigation strategies being implemented through BVs' Blue Forests project.

Keywords

Madagascar • Mangrove • Ecosystem services • Carbon • Deforestation • Coastal • Charcoal

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Abbreviations

AAB	Ambanja-Ambaro Bays
BV	Blue Ventures
Dbh	diameter at breast height
GoM	Government of Madagascar
ODK	Open Data Kit
PES	Payments for Ecosystem Services
REDD+	Reduced Emissions from Deforestation and forest Degradation plus sustainable management of forests, conservation of forest carbon stocks and enhancement of forest carbon stocks
USGS	United States Geological Survey
VCS	Verified Carbon Standard

Introduction

Madagascar's 5,600 km^2 coastline comprises one of the most extensive shallow marine habitats in the Western Indian Ocean, including extensive estuarine areas. As of 2010, Madagascar contained approximately 2,100 km^2 of mangroves, Africa's fourth largest extent behind Nigeria, Guinea Bissau and Mozambique and approximately 2% of the global distribution (Giri and Muhlhausen 2008; Giri et al. 2011; Giri 2011). The tremendous and multi-faceted importance of these ecosystems is in their 'provisioning' (food (e.g., fisheries and aquaculture), fuel (e.g., wood), natural products (e.g., construction materials), medicine (e.g., traditional plant-based, ports and shipping), 'regulating' (carbon sequestration, shore-line stabilization, water filtration, storm and flood protection), 'supporting' (soil and sediment formation, nutrient cycling) and 'cultural' (tourism, recreation, education) services (Lau 2012). Though not as species rich as in other parts of the world, Madagascar's mangrove ecosystems comprise nine true mangrove species but also support unique flora and fauna; much of which is endangered or at-risk (Giri and Muhlhausen 2008; Nagelkerken et al. 2008). Despite their importance and in agreement with global trends (see Polidoro et al. 2010; Spalding et al. 2010), many of these ecosystems are rapidly being degraded and in some areas completely deforested from unsustainable logging for timber and charcoal, and from conversion for agri- and aquaculture. Erosion, sedimentation and siltation because of upstream deforestation, burning and agriculture also pose increasing threats.

Amongst Madagascar's mangrove ecosystems, anthropogenic loss is particularly prominent in the northwestern Ambanja-Ambaro Bay (AAB) complex (Jones et al. 2014). If degradation and conversion continue here unimpeded, many of the important services provided will be seriously compromised (Jones 2013). Until recently in AAB, as throughout Madagascar, comprehensive and effective management to curb or reduce degradation and deforestation have been absent. Here we present an overview of AAB with a focus on its mangrove ecosystem. We begin with the biophysical characteristics of this coastal area, followed by an overview of the mangrove ecosystem, its socio-economic importance, and how its use is governed. Next, we present historical dynamics within the mangrove ecosystem; which facilitated quantifying loss, characterizing ecological variability and estimating carbon stocks throughout the remaining distribution. The results of in-depth socio-economic analyses and participatory mapping are used to contextualize historical dynamics and current threats. With this context provided, we describe current mitigation strategies being implemented through Blue Ventures' (BV) Blue Forests project.

Study Approach and Methodology

Biophysical Overview

The study area for this chapter includes the combined marine and terrestrial extent of AAB in northwestern Madagascar, within a seven km buffer from the coast and centered at latitude 48° 30′ East, longitude 13° 26′ South (Fig. 1). AAB experiences a sub-humid tropical climate which is typified by a relatively dry and cool period from May-October and a comparatively hotter and wetter period from November-April (Rasolofo and Ramilijaona 2009). The wetter months are also influenced by periodic cyclones, with 1.3 reported each year between 1920 and 1972 (Cooke et al. 2000). The underlying geology in the region is predominantly characterized by alluvial and lake deposits. In terms of incoming freshwater, Ambanja bay is most influenced by the Sambirano River and its numerous outlets, with its headwaters in the Tsarantanana Massif mountain range to the southeast, including Maromokotro, Madagascar's highest peak at 2,876 m. The predominant rivers in Ambaro Bay include the Mananjeba, the Mahavavy and the Ifasy, all of which also originate in the Tsarantanana range. AAB is dominated by extensive mangroves, scattered seagrass meadows and coral reefs. Due to the influence and protection offered by close proximity to high mountains, AAB enjoys relatively calm winds, though still variable and at their strongest from September-December (Cooke et al. 2000; McKenna and Allen 2005). According to Giri (2011), AAB's network of mangroves comprises Madagascar's second most extensive mangrove ecosystem at >24,000 ha, surpassed only by Mahajamba Bay. AAB's mangrove ecosystem is influenced by semi-diurnal tidal ranges which vary between maximums of 3–3.5 m (Rasolofo

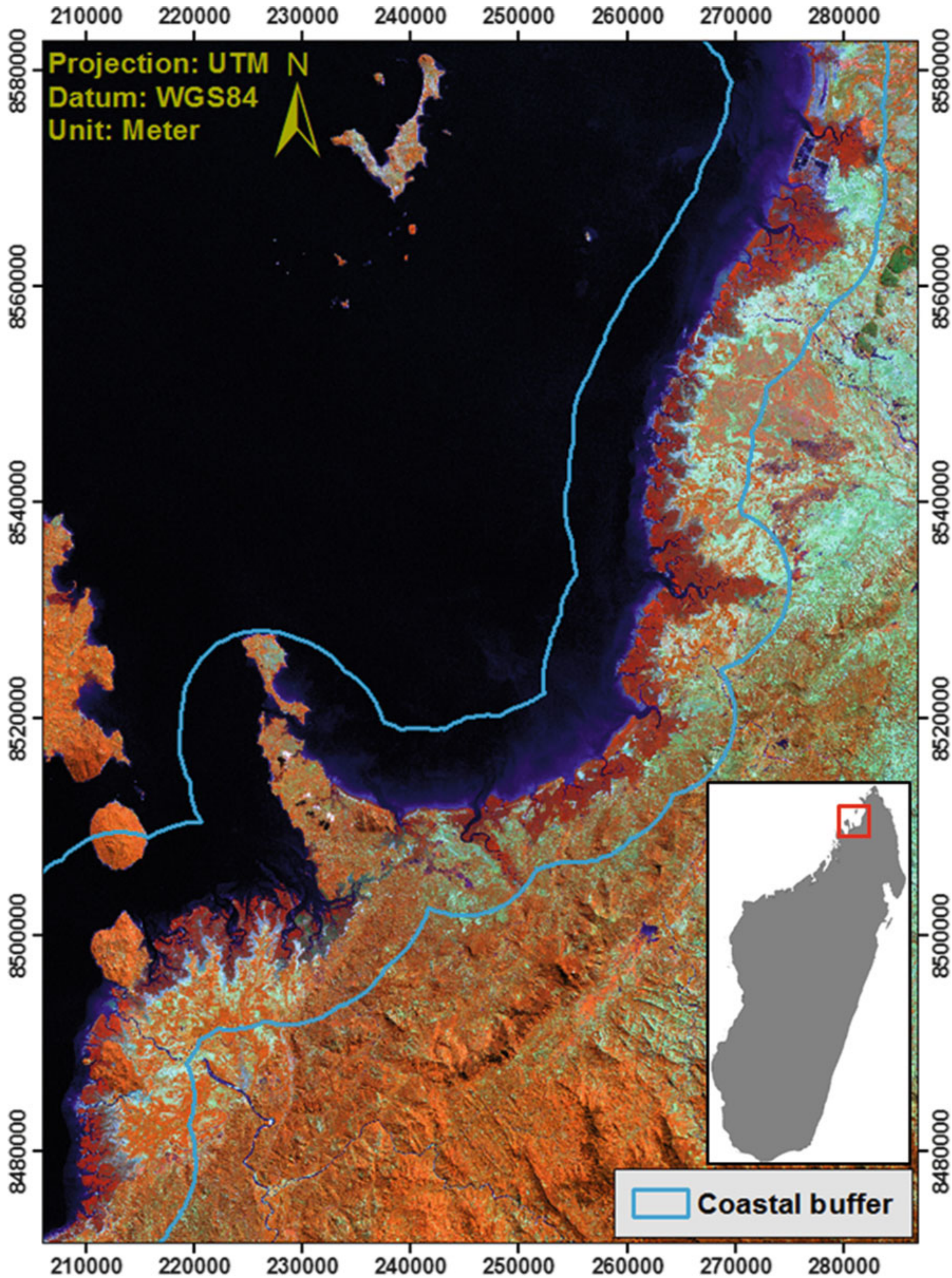

Fig. 1 The area of interest includes the combined marine and terrestrial extent of AAB in northwestern Madagascar, within a seven km buffer from the coast and centered at latitude 48° 30′ East, longitude 13° 26′ South. The background image is a false color composite of Landsat ETM+ bands 3 (*red*), 5 (short-wave infrared) and 4 (near-infrared), wherein mangrove vegetation appears primarily as *dark red*

and Ramilijaona 2009); though McKenna and Allen (2005) note a peak spring tide of 3.8 m. The comparative abundance of precipitation results in much higher stature mangroves here than in central-western and south-western Madagascar (Rasofolo 1997; Giri and Muhlhausen 2008). Ocean currents in AAB flow from a predominantly southerly direction, with mean annual sea surface temperature ranging from 22–28° C (McKenna and Allen 2005). Cooke et al. (2000) note that the warmest sea temperatures in Madagascar are found in the northwestern coast during the cyclone season. AAB and offshore waters host high marine and coastal biodiversity, amongst which many species are rare and endangered. Of particular note, AAB is considered one of the last refuges for dugongs (*Dugong dugon*) in the Western Indian Ocean (Laran et al. 2012). The terrestrial areas surrounding coastal AAB are dominated by a mixture of grass/bush-lands, agro-forest mosaics comprised of cash crops such as cacao, coffee and vanilla, and extensive agricultural lands producing mainly rice.

The Socio-Economic Importance of Mangrove Resources

Throughout coastal Madagascar, communities rely heavily on the goods and services provided by healthy, intact mangrove ecosystems. These coastal communities are also particularly vulnerable to climate change and the continued benefits provided by mangrove ecosystems are vital to their climate change preparedness. In AAB, due to a lack of governance, increasing population, and other factors contributing to economic pressure, the range, frequency and magnitude of mangrove resource exploitation continue to rise (Jones et al. 2014). In particular, mangroves supply coastal communities and nearby urban centers with the majority of their energy needs. To better understand the historic and current relationship between coastal communities and mangrove resources an array of socio-economic studies were undertaken. From April–May 2012, participative appraisals were carried out in the communes (i.e., municipalities) of Ambalahonko and Maherivaratra in the District of Ambanja, within or surrounding areas where deforestation was the highest from 2000–2010 in AAB, using focus group discussions (n = 40) and semi-directed key informant interviews (n = 30) to identify mangrove use and threats. From April–July 2013, community members in thirteen villages within these two communes were engaged in participatory mapping to define the current and historic distribution of mangrove types and uses by communities. Household surveys (n = 580) were also carried out to gain a detailed understanding of the socio-economic context of mangrove charcoal production. In October 2013, a consumer survey (n = 387) was conducted in AAB's three major urban areas (i.e., Ambanja, Hell Ville and Ambilobe) to assess the extent of mangrove charcoal consumption in urban centers. Lastly, from July–August 2014, a participative appraisal of socio-economic conditions, mangrove deforestation and degradation drivers as well as management association status was undertaken in thirty-five coastal villages located throughout 10 communes spread across AAB. All aforementioned activities followed methodologies outlined in Parts 1 and 2 of the *Social and Biodiversity Impact Assessment (SBIA) Manual for REDD+ Projects* (Richards 2011; Richards et al. 2011). For each identified agent of degradation and/or deforestation, research was designed to identify the factors that drive land-use, including variables which explain the quantity and location of deforestation and the underlying causes determining *why* these activities are occurring.

Resource Governance

Madagascar's mangrove ecosystems are governed by a complex legal framework falling under forestry, fishery and spatial management regulations. Regarding ownership, both their coastal setting and status as areas containing naturally grown forests qualify them as part of the public domain of the state (Government of Madagascar (GoM) 1997a, 2008). Mangrove ecosystems also qualify as sensitive areas, which are defined as having both specific value and sensitivity towards human activities and events (GoM 1997b). Despite this status of state ownership, there is limited private ownership in certain areas, which is primarily a legacy of the French colonial era. Due to the complex legal status, management options for mangroves are not easy to establish. While commercial exploitation is officially forbidden in all sensitive areas, including mangrove ecosystems (GoM 2000, 2014), local communities are permitted to extract forest products for self-consumption as part of their traditional user rights (GoM 2005). Throughout their distribution, the mangroves of AAB, while state owned, qualify for local management. This can be established through a 1996 law which allows for transferring the management of renewable natural resources from the state to local communities (GoM 1996). In regards to payments for ecosystem services (PES), such as forest carbon, because mangrove ecosystems are the property of the state, the state negotiates with different stakeholders on how to distribute revenue (Andriamahefazafy et al. in preparation).

Quantifying Historic Dynamics

Information on the contemporary and historic extent of mangroves is needed to quantify historic dynamics and can be extracted from remotely sensed data (e.g., satellite imagery). The use of satellite imagery to map and monitor mangrove ecosystems is now well-established (Heumann 2011; Kuezner et al. 2011). Of particular value, Landsat imagery, which is freely available and offers >40 years of coverage, has been utilized to quantify mangrove extent and locate loss in numerous studies (Heumann 2011; Kuezner et al. 2011). Using Landsat imagery, and based on methods described in Giri and Muhlhausen (2008), Chandra Giri and colleagues at the United States Geological Survey (USGS) produced national-level maps of single-class mangrove distribution for 1990, 2000 and 2010 (Giri 2011). These USGS maps were used as input to analyze historic mangrove dynamics in AAB from 1990–2010.

In addition to analyzing national-level maps to quantify mangrove dynamics, a suite of Landsat scenes providing coverage over only Ambanja bay and part of Ambaro bay for the years 2000, 2003, 2006, 2008, 2010 and 2013 were acquired from the USGS Earth Resources Observation Center (http://glovis.usgs.gov) and used for a localized assessment. Due to context gained through the analysis of the USGS national-level maps, Ambanja Bay and only a portion of Ambaro bay received focus in this detailed, localized, multi-date quantification of mangrove dynamics.

Ecological Characterization

In order to ecologically characterize and partition the mangroves of AAB, a Global Land Survey Landsat ETM+ scene providing coverage from June 2010 was acquired from the USGS Earth Resources Observation Center. As described in detail in Jones et al. (2014), this Landsat image was used to create a preliminary map of different mangrove types and surrounding land-cover categories. Fine spatial resolution satellite imagery viewable in Google Earth and existing national-level land-cover and mangrove maps were used to label the initial map. Mangrove class labels assume the presence of true mangroves, defined by Tomlinson (1986) as salt-tolerant halophytic trees and shrubs occurring entirely in tidal and inter-tidal areas. To improve upon the initial map and refine class labels, a field survey was conducted in February 2012, during which stratified random sampling was used to establish 22 100 × 100 m plots within mangrove and surrounding land-cover types. Within each mangrove plot, height, species dominance, stature, age, density, canopy-cover, micro-relief, tidal-inundation and the presence of disturbance were observed. Diameter at breast height (dbh), height and crown dimensions were recorded for representatives of each observed mangrove species. In addition, canopy-cover was quantified, and stumps, regeneration, litter and standing dead wood inventoried. Following the field campaign, additional reference areas for mangrove and surrounding land-cover classes were identified in finer spatial resolution satellite imagery viewable in Google Earth. A total of 71 100 × 100 m reference areas representing four mangrove types and 120 100 × 100 m reference areas representing seven surrounding land-cover types were delineated (Figs. 2 and 3). Reference areas were subsequently used as input in a supervised classification of the Landsat scene, resulting in a refined ecological characterization of AAB. The accuracy of the final map was assessed through standard practices, including tabulating a confusion matrix using a sub-set of classification reference areas held back for validation purposes, and through visual comparisons with existing data-sets and finer resolution imagery viewable in Google Earth. A detailed description of ecological characterization, mapped classes, mapping results and accuracy is provided in Jones et al. (2014).

Estimating Carbon Stocks

To estimate biomass and calculate carbon stocks, additional field surveys were carried out in April–May and August-September 2012. Based on mangrove strata defined using the supervised Landsat classification, a systematic stratified sampling design was employed. Within a network of 50 rectangular nested plots, measurements were taken in accordance with protocol proposed by Center for International Forestry Research (CIFOR) and outlined in Kauffman and Donato (2012). Tree measurements included species type, height, dbh and lead stem quality; from which plot-level biomass, density, stature and species dominance were determined. Both natural and anthropogenic degradation of each plot were calculated by inventorying and measuring the heights and diameters of dead trees and stumps. Soil depth was measured systematically throughout the plot and samples were extracted at plot-center using a soil corer at 0–15, 15–30, 30–50, 50–100 and 100–150 cm. Soil organic carbon stocks are reported up to 100 cm to facilitate comparison with similar published studies. For further details regarding these field surveys, please consult Kauffman and Donato (2012) and Jones et al. (2014).

Above-ground biomass was estimated using species-specific allometric equations relating diameter at breast height (dbh) to biomass; and for certain species, dbh and tree height to biomass (Fromard et al. 1998; Smith and Whelan 2006) (Table 1). The below-ground biomass of trees and the biomass of standing dead wood were calculated based on equations developed by Komiyama et al. (2005) and Kauffman and Donato (2012), respectively (Table 1). Carbon concentrations of 0.47 and 0.39 were used to convert biomass to carbon mass for above- and below-ground biomass respectively. Soil samples were sent to the Laboratoire des Radio Isotopes (LRI) in Antananarivo, Madagascar to determine organic carbon content. Soil organic carbon content was determined using a modified Walkley-Black

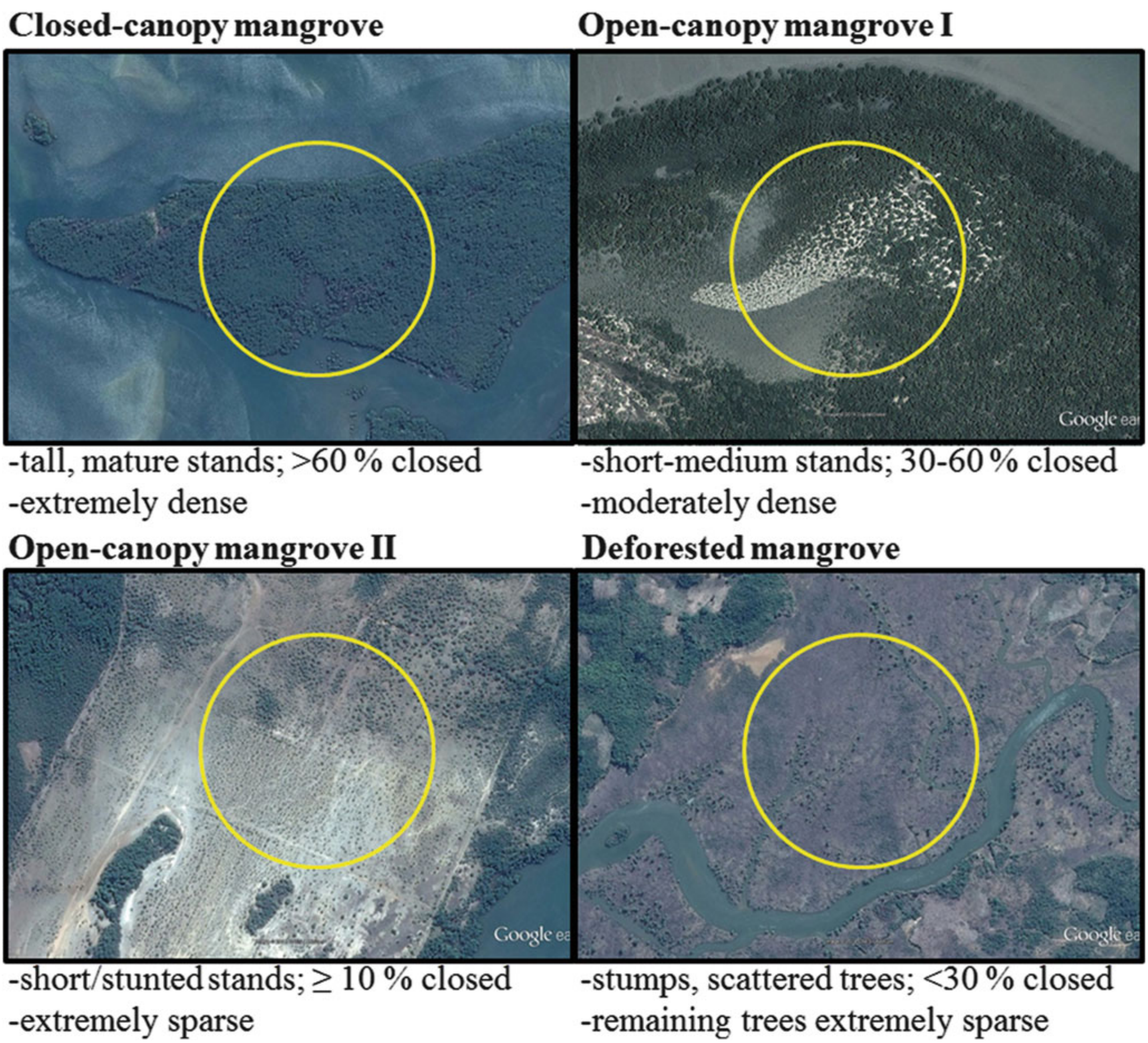

Fig. 2 Examples of four ecologically distinct mangrove types in AAB are shown within the center of *yellow circles* in each panel. The name of each class is at the *top* of each panel, with the class definition *below* each panel. Within each panel, the imagery employed to provide the class example is fine spatial resolution satellite imagery viewable in Google Earth

method, as described in Schumacher (2002), Mikhailova et al. (2003), De Vos et al. (2007) and Meersmans et al. (2009). For further details regarding carbon stock calculations, please consult Jones et al. (2014).

Results and Discussion

Mangrove Ecosystem Dynamics

The analysis of the USGS-produced national-level mangrove maps indicates a loss of 7,659 ha (23.7%), a gain of 995 ha (3.1%) and persistence of 24,669 ha (76.3%) of mangroves within AAB from 1990–2010 (Fig. 4). Most notably, from 2000–2010, AAB's mangroves decreased by 5,762 ha (18.7%) and increased by 1,104 ha (3.3%). While this is an important initial assessment of dynamics in AAB, localized applications of these national-level mangrove maps are limited in providing reliably detailed insight. The USGS-produced maps accurately represent taller, intact, mature stands, but tend to underrepresent comparatively lower stature and more open, highly degraded, and/or scrub stands, which can lead to an over-estimation of deforestation (Jones et al. 2014). In addition, these maps are approximately > five years old, providing temporal coverage only up until 2010 for a region which continues to experience increased and widespread mangrove

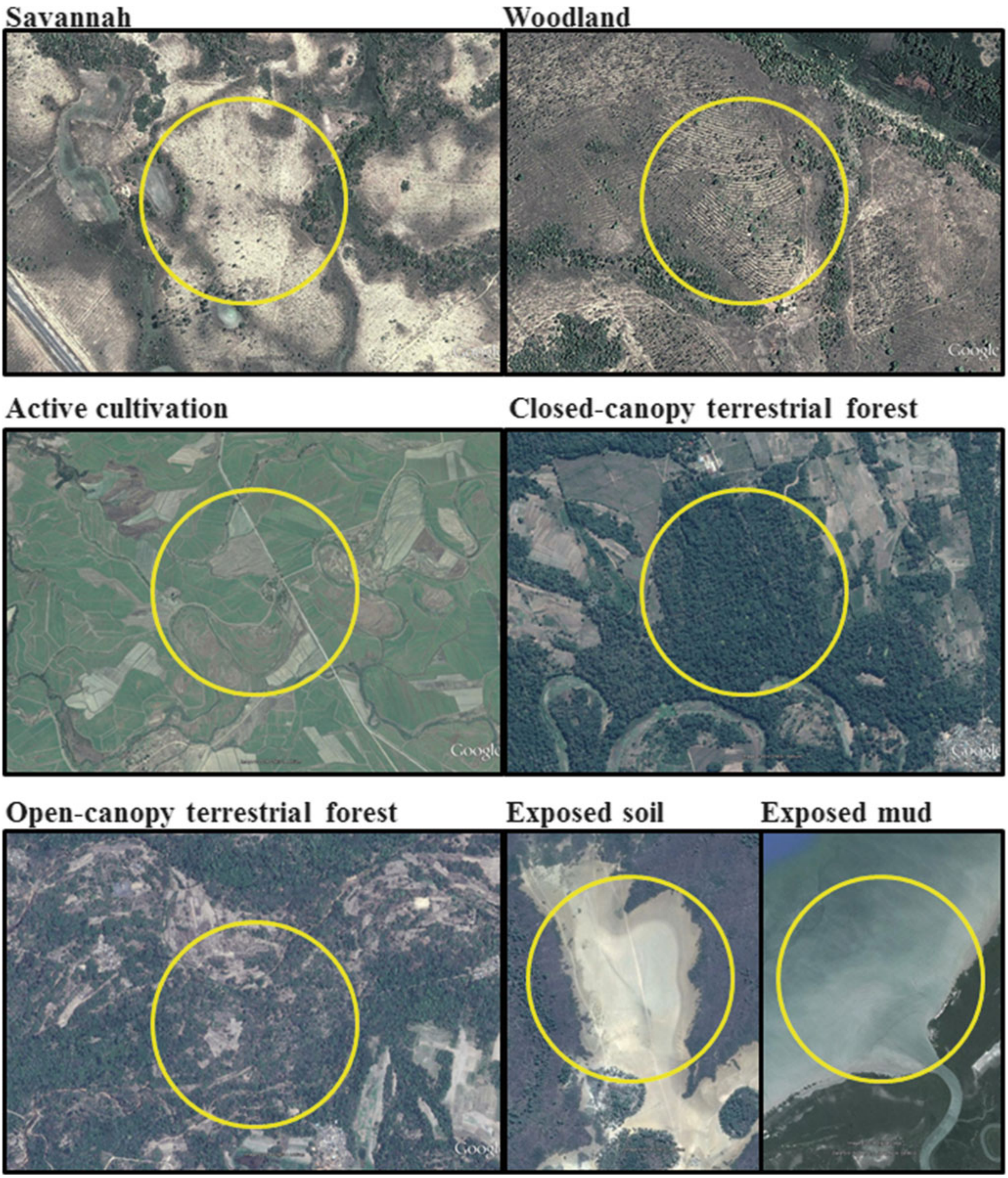

Fig. 3 Examples of seven ecologically distinct land-cover classes in AAB which surround mangrove types are shown within the center of *yellow circles* in each panel. The name of each class is at the top of each panel. Within each panel, the imagery employed to provide the class example is fine spatial resolution satellite imagery viewable in Google Earth

Table 1 Allometric equations used for calculating biomass

Live aboveground biomass (AGB)				
Species		Equation	ρ	Sources
Avicennia marina		$B = 0.1848 \times dbh^{2.3524}$	0.661	Dharmawan and Siregar (2008)
Bruguiera gymnorrizha	(leaves)	$B = 0.0679 \times dbh^{1.4914}$	0.741	Clough and Scott (1989)
	(stem)	$B = 0.0464 \times (dbh^2 \times H)^{0.94275} \times \rho$	0.741	Kauffman and Donato (2012), Chave et al. (2005) and Cole et al. (1999)
Ceriops tagal	(dbh: 2–18 cm)	$B = 10^{-0.7247} \times dbh^{2.3379}$	0.803	Clough and Scott (1989)
	(dbh: 18–25 cm)	$B = 10^{-0.494} \times dbh^{2.056}$	0.803	Comley and McGuinness (2005)
Heritiera littoralis	(leaves)	$B = 0.0679 \times dbh^{1.4914}$	1.074	Clough and Scott (1989)
	(stem)	$B = 0.0464 \times (dbh^2 \times H)^{0.94275} \times \rho$	1.074	Kauffman and Donato (2012), Chave et al. (2005) and Cole et al. (1999)
Lumnitzeria racemosa		$B = 0.0214 \times (dbh^2 \times H)^{1.05655} \times \rho$	0.565	Kauffman and Donato (2012), Chave et al. (2005) and Cole et al. (1999)
Rhizophora mucronata	(leaves)	$B = 0.0139 \times D^{2.1072}$	0.867	Clough and Scott (1989)
	(root)	$B = 0.0068 \times dbh^{3.1353}$	0.867	Clough and Scott (1989)
	(stem)	$B = 0.0311 \times (dbh^2 \times H)^{1.00741} \times \rho$	0.867	Kauffman and Donato (2012), Chave et al. (2005) and Cole et al. (1999)
Sonneratia alba		$B = 0.0825 \times (dbh^2 \times H)^{0.89966} \times \rho$	0.78	Kauffman and Donato (2012), Chave et al. (2005) and Cole et al. (1999)
Xylocarpus granatum		$B = 0.0830 \times (dbh^2 \times H)^{0.89806} \times \rho$	0.7	Kauffman and Donato (2012), Chave et al. (2005) and Cole et al. (1999)
Other equations				
Belowground biomass:		$B = 0.199 \times \rho^{0.899} \times dbh^{2.22}$		Komiyama et al. (2005)
Live and dead roots		*(where ρ = species-specific wood density, as above)*		
Dead tree: *Decay status 1*		$B = 0.975 \times AGB$		Kauffman and Donato (2012)
Dead tree: *Decay status 2*		$B = 0.8 \times AGB$		Kauffman and Donato (2012)

Species-specific wood density values (ρ) are derived from Simpson (1996). For standing dead trees, decay status 1 are recently dead with fine branches remaining; decay status 2 have some large branches but no small branches or twigs

B biomass (kg), *dbh* diameter at breast height (cm), *D* diameter (cm), D_{top} diameter at top of stem (cm), D_{base} diameter at base of stem (cm), *H* height (m), and ρ wood density (g cm^{-3})

deforestation. In contrast, the results of localized, multi-date quantification of mangrove dynamics provides more detailed insight into loss within Ambanja bay and part of Ambaro bay, indicating 1,653 ha of loss from 2000–2013 (Fig. 5).

While differing in the specific quantities, the results of both dynamics analyses highlight that mangrove deforestation in the region is concentrated on the peninsula that separates the two bays. While the localized analysis provides a more appropriate and reliable estimate of loss for AAB than the USGS results, it still falls short of accurately quantifying mangrove degradation. Reliably detecting deforestation using established methods and Landsat data is possible; however, accurately detecting subtler modification associated with degradation remains a vexing challenge. Sub-pixel changes in forest appearance are difficult to accurately measure in Landsat data and accurately quantifying degradation requires further research involving the incorporation of complimentary remotely sensed data sets, such as finer spatial resolution satellite imagery and structural information provided by light detection and ranging (LiDAR).

The results of the participative appraisals, participatory mapping and household surveys from 13 villages in two communes revealed that the primary driver of deforestation in AAB is over-exploitation for charcoal production (Fig. 6); whereas degradation is primarily driven by harvest for multiple timber products. Given the illegality, reliable data to accurately assess the extent of charcoal production are lacking; however, our results suggest that mangrove charcoal production is carried out partly (i.e., 31%) by recently settled (e.g., for less than one generation) migrant populations, but mostly (i.e., 69%) by long-established community members. Large-scale exploitation for timber follows a different pattern and is typically performed by unsettled migrants temporarily residing in camps within the mangroves. Consumer survey results further indicate that mangrove charcoal is the primary cooking fuel used by 37–46% of households in AAB's three major urban

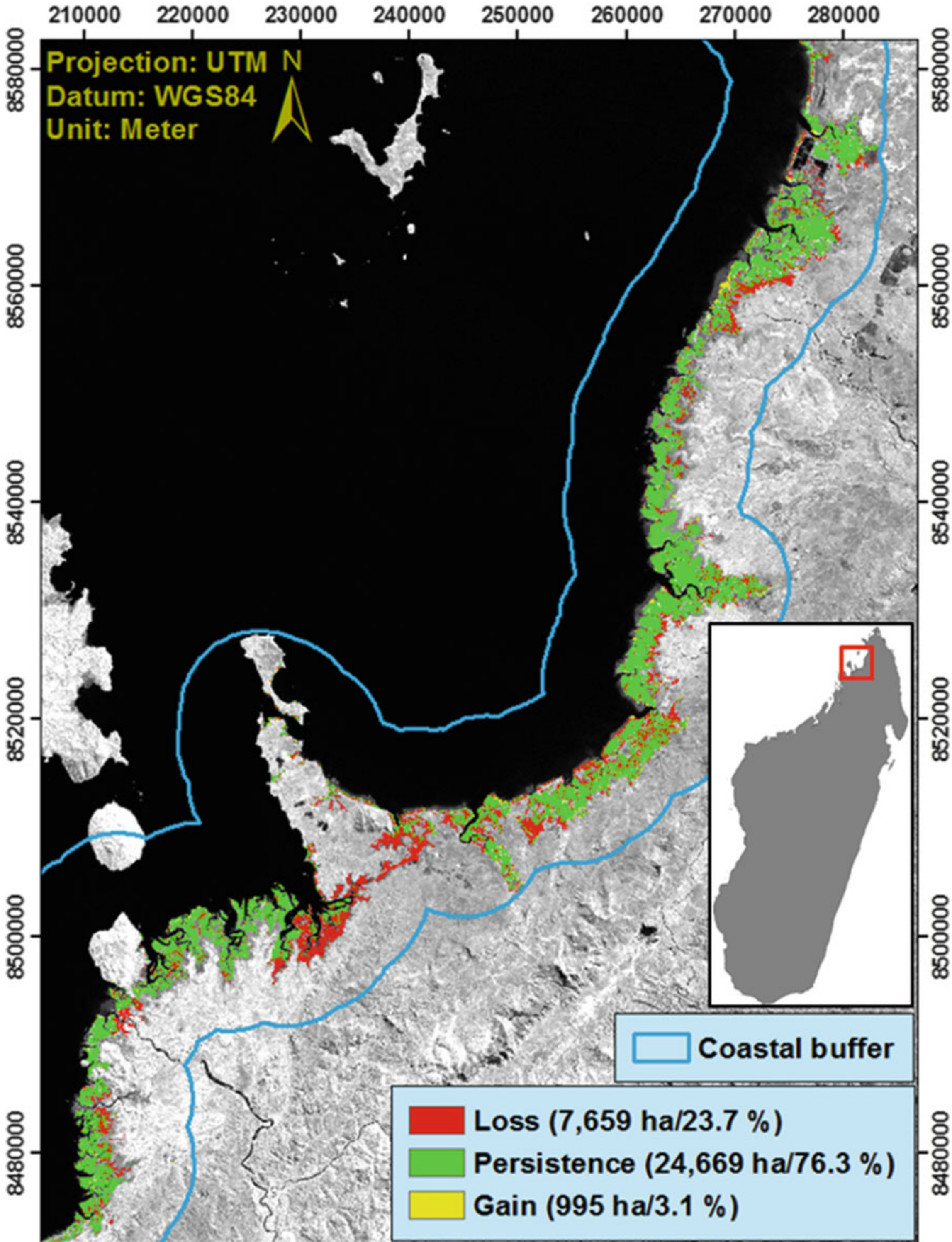

Fig. 4 The results of analysis of the USGS-produced national-level mangrove maps, showing mangrove dynamics for AAB from 1990–2010. The background image is Landsat ETM+ band 4 (near-infrared)

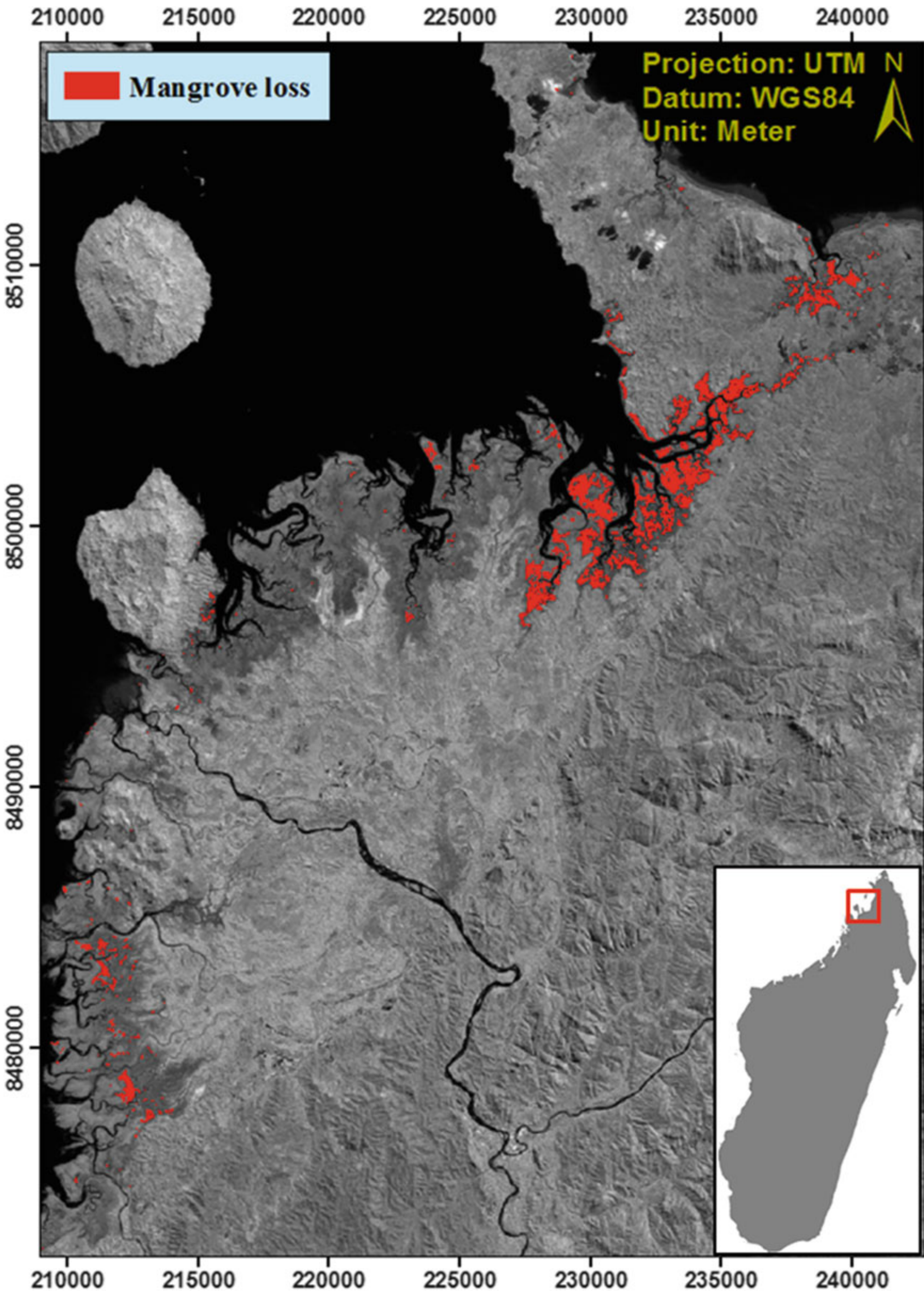

Fig. 5 The results of localized, multi-date quantification of mangrove deforestation, indicating an accumulative loss of 1,653 ha from 2000–2013. The background image is Landsat ETM+ band 4 (near-infrared)

Fig. 6 The primary driver of mangrove deforestation in AAB. These images show (**a**) deforested mangrove area (**b**) harvested mangrove trees (**c**) and charcoal production

areas (i.e., Ambanja, Hell Ville and Ambilobe). Based on deforested areas, it can be conservatively estimated that around 2,000 tons of mangrove charcoal were produced annually in AAB from 2000–2010. This represents an annual income of approximately 400 million Ariary, providing a primary source of revenue to around 300 local producers.

The participatory appraisal carried out in thirty-five villages across 10 communes throughout AAB indicated that the primary underlying causes of these agents (i.e., the migrants and local community members) and drivers (i.e., charcoal and/or timber production) are population growth because of increased births and migration; reduced fishery and agricultural production (largely associated with population growth); reduced employment opportunities throughout the region (e.g., the closure of several cocoa and sugar plantations; reduction of the shrimp industry; decline in tourism); and most importantly, a growing demand for charcoal and timber from urban markets, in particular the tourist hub of Nosy Be, where effective forest law enforcement makes charcoal or timber production almost impossible. Though not exhaustive, these results represent ten of the thirteen coastal communes of AAB, including all of the areas that have experienced significant mangrove losses since 1990.

The Current Distribution and Composition of Mangrove Ecosystems

Mapping indicated that as of 2010, AAB had 14,015 ha of closed-canopy, 26,192 ha of open-canopy I, 5,473 ha of open-canopy II, and nearly 1,000 ha of deforested mangroves (Fig. 7; descriptions of mapped classes in Fig. 2). As supported by historical dynamic analyses, deforestation is most prevalent on or near the peninsular base which separates Ambanja and Ambaro Bays. The measurements taken within reference areas and carbon plots facilitated summarizing the forest characteristics of each mapped mangrove class and further allowed partitioning in to sub-classes based on ecological properties (Fig. 8). Plots within the closed-canopy class typically contained stands dominated by *Rhizophora mucronata,* with high stature trees of variable density with well-formed canopies. In contrast, plots within the open-canopy I class were typically dominated by moderately-dense stands of medium stature *Ceriops tagal* and *Rhizophora mucronata* with relatively open canopies. Open-canopy II plots were typified by sparse, stunted, low stature, extremely open-canopied stands of *Avicennia marina*. The observations made in plots within these broad mangrove class types confirmed their ecological rationality (Jones et al. 2014).

Carbon Stock Estimates

Plot-level biomass estimates were converted and scaled to obtain hectare-level C stocks, reported in megagrammes per hectare (Mg ha^{-1}). Tall-stature closed-canopy mangroves had higher average above- and below-ground vegetation C values of 114.8 Mg ha^{-1} (±9.3 (i.e. standard error), n = 22)), compared to open-canopy I and open-canopy II mangroves which had average vegetation C values of 43.6 Mg ha^{-1} (±7.3, n = 24) and 19.3 Mg ha^{-1} (±4.6, n = 4), respectively (Table 2; Fig. 9a). This pattern between classes is reflected in organic C stocks in soil up to 100 cm in depth, with the highest soil C values in closed-canopy mangroves (309.9 Mg ha^{-1} (±19.4, n = 22)). Open-canopy I mangroves had average soil C values of 278.8 Mg ha^{-1} (±21.0, n = 24) and open-canopy II mangroves had the lowest average soil C values of 165.2 Mg ha^{-1} (±29.1, n = 4)) (Tables 2 and 3; Fig. 9b). The C stock estimates presented here represent a modification of those originally presented in Jones et al. (2014), based on a sub-set of plots (i.e., 50 vs. 55), reduced soil depth (i.e., to 100 cm), the re-calculation of C for *B. gymnorrizha* and *H. littoralis* through revised allometric equations, a revised C conversion factor, and the re-estimation of total C (above- and below-ground, including soil and SE.

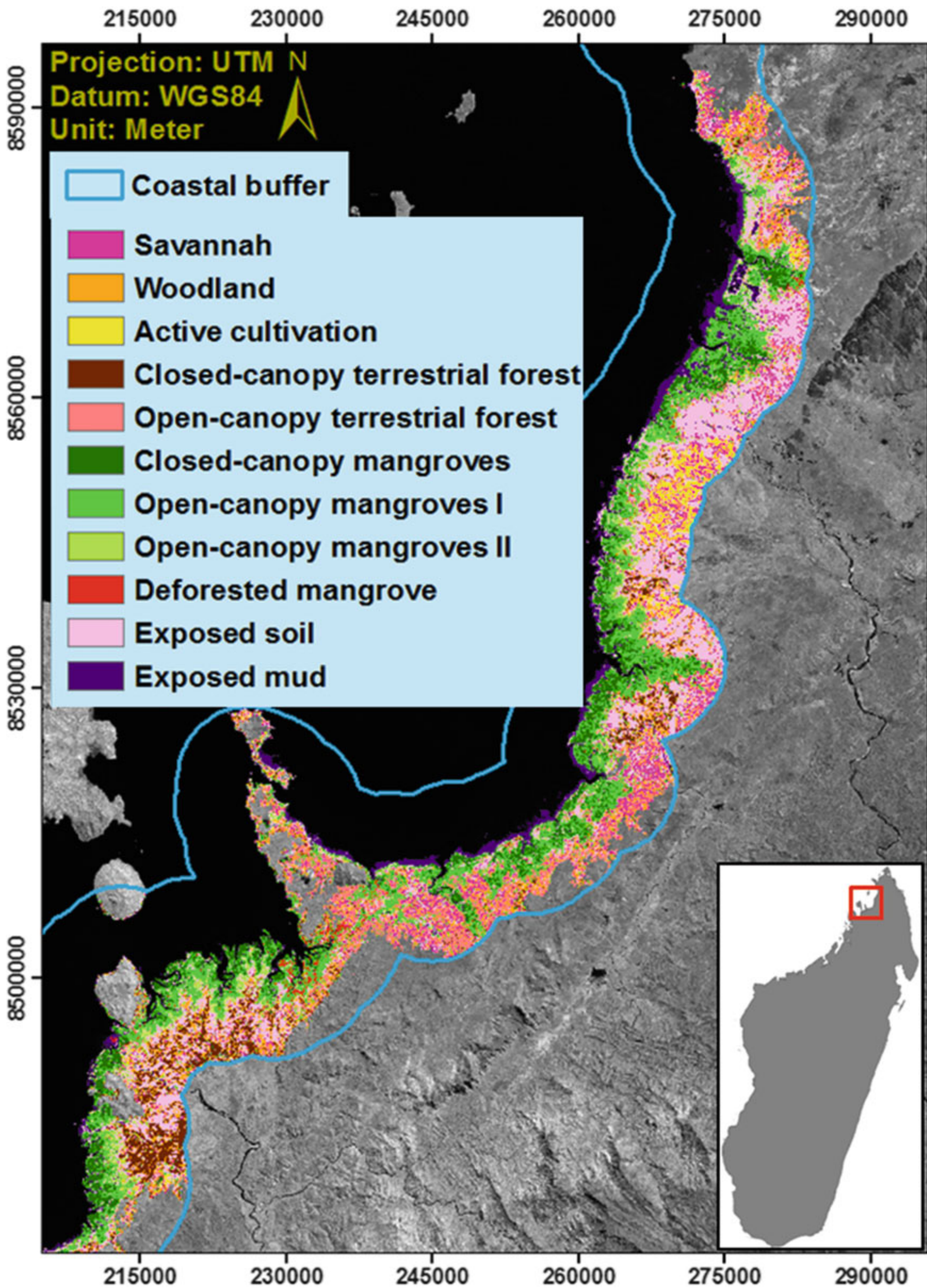

Fig. 7 Mapping results for mangroves and surrounding land-cover types in AAB, circa 2010, based on the classification of Landsat satellite imagery (Jones et al. 2014). The background image is Landsat ETM+ band 4 (near-infrared)

With comparatively few plots within the open-canopy II category, the establishment of additional plots is required for a more reliable and robust estimate of both vegetation and soil C for this category (Jones et al. 2014). In addition, our soil samples were analyzed using a modified Walkley-Black method, which is known to under-represent SOC

Fig. 8 Ecologically distinct mangrove sub-classes in AAB, characterized by dominant stature, canopy cover and species dominance. (**a** to **f**) Closed canopy mangroves: high stature stands of variable density with well-formed canopies >60% closed. (**g** to **k**) Open-canopy I mangroves: moderately-dense stands of medium stature with canopies 30–70% closed. (l) Open-canopy II mangroves: very sparse, stunted ecosystems, with canopies <30% closed

Table 2 Carbon stock estimates (± SE) of mangrove vegetation and soil up to 100 cm in depth in AAB

Mangrove class		Vegetation carbon	Soil organic carbon	Total carbon
Closed-canopy	*(N = 22)*	**114.8** *(9.3)*	**309.9** *(20.5)*	**424.7** *(20.5)*
Open-canopy I	*(N = 24)*	**43.6** *(7.3)*	**278.8** *(21.2)*	**322.4** *(21.2)*
Open-canopy II	*(N = 4)*	**19.3** *(4.6)*	**165.2** *(29.1)*	**184.5** *(31.6)*

Average carbon stock estimates (Mg ha^{-1}) are stratified by mangrove class, where N represents the number of plots per mangrove class for the vegetation and soil carbon pools, respectively

Table 3 Carbon stock estimates (± SE) of mangrove soil in AAB, up to 100 cm in depth

	Soil organic carbon content (Mg ha^{-1})					
Mangrove Class	Depth (cm):	0–15	15–30	30–50	50–100	Total
Closed-canopy $N = 22$		**43.0** *(3.1)*	**43.1** *(3.3)*	**61.1** *(3.1)*	**162.7** *(13.7)*	**309.9** *(19.4)*
Open-canopy I $N = 24$		**36.6** *(3.7)*	**49.3** *(5.1)*	**52.1** *(5.4)*	**140.8** *(16.3)*	**278.8** *(21.0)*
Open-canopy II $N = 4$		**12.1** *(0.7)*	**10.6** *(2.6)*	**22.5** *(7.4)*	**120.0** *(28.9)*	**165.2** *(29.1)*

Average per-hectare soil organic carbon estimates (Mg ha^{-1}) are stratified by mangrove class and soil depth. N represents the number of plots per mangrove class

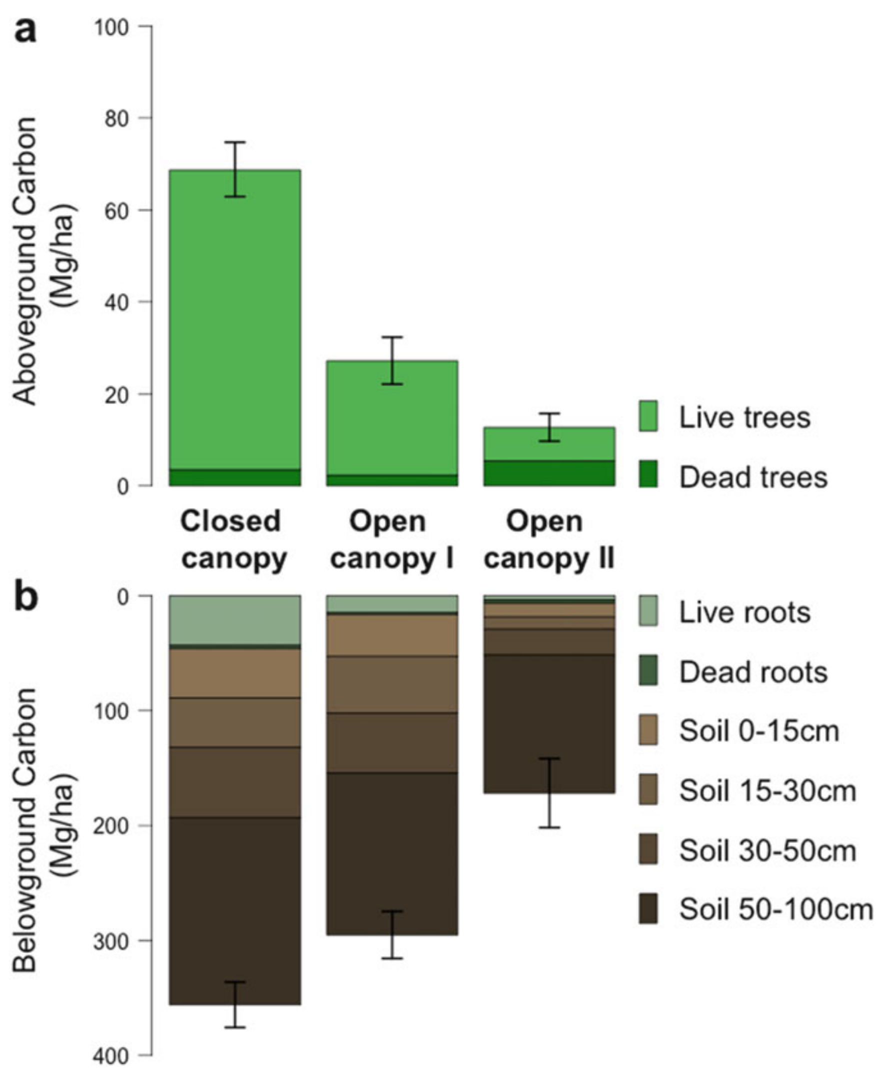

Fig. 9 (**a**) Aboveground vegetation carbon stocks and b belowground vegetation and soil carbon stocks; for each mangrove class in AAB, measured in Mg ha^{-1}. *Error bars* show ± SE of (**a**) total aboveground carbon and (**b**) total belowground carbon. Note that (**a**) and (**b**) use different scales

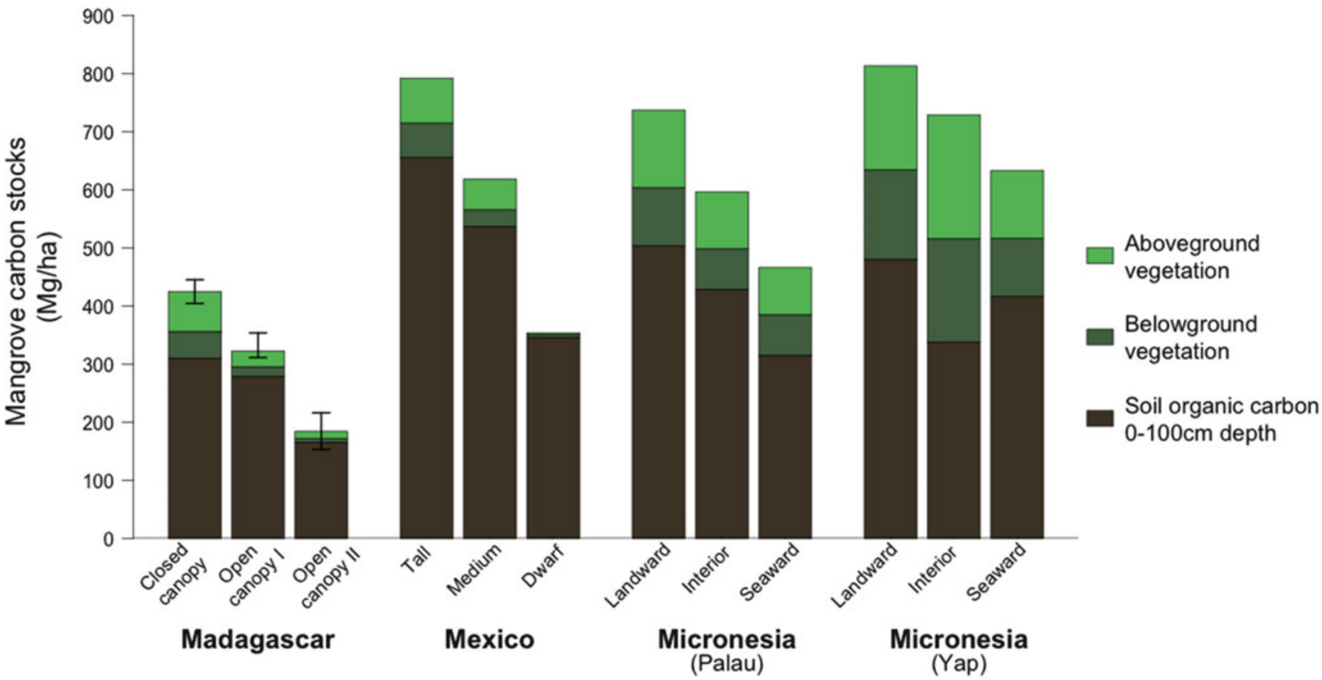

Fig. 10 Carbon stock estimates for AAB in Madagascar compared with published carbon stocks for different mangrove strata in Mexico (Adame et al. 2013) and two regions in Micronesia: Palau and Yap (Kauffman et al. 2011). Soil organic carbon comparisons are made up to a depth of 100 cm. *Error bars* for Madagascar show ± SE of total aboveground and belowground carbon stocks.

(Schumacher 2002; Mikhailova et al. 2003; De Vos et al. 2007; Meersmans et al. 2009). Bearing in mind the shortcomings of this method, a comparison of our results with those presented in other mangrove ecosystem studies indicates that our values are comparatively low (Fig. 10). However, as compared to terrestrial forest types, the initial estimates for the closed-canopy category support a growing body of evidence that mangroves are amongst the most carbon-dense forests in the tropics (Fig. 11) (Donato et al. 2011; Kauffman et al. 2011; Ray et al. 2011; Chen et al. 2012; Donato et al. 2012; Adame et al. 2013; Wang et al. 2013; Jones et al. 2014; Kauffman et al. 2014).

Mitigation Strategy

While the role of natural processes (e.g., cyclone damage; forest succession) in affecting changes in vegetation cover should not be overlooked, it is clear that AAB's mangroves are subject to substantial loss due to anthropogenic activities. If current loss continues unabated, critical ecosystem services could effectively collapse. The marine conservation NGO BV (www.blueventures.org) has a well-established presence in Madagascar, and a proven record of novel, holistic, locally-led conservation initiatives that directly improve livelihoods in some of the world's poorest coastal communities. Since 2012, the BV Blue Forests project has been working with coastal communities, partner NGOs, Malagasy and foreign Universities, and government bodies at all levels to establish and incentivize community-based sustainable management of mangroves in AAB. This initiative aims to build local resilience to climate change, improve, sustain and diversify local livelihoods and safeguard biodiversity throughout AAB.

Through scientific research and numerous partnerships, the Blue Forests project has been assessing the feasibility of community-centered mangrove REDD+ (Reduced Emissions from Deforestation and forest Degradation plus other measures to restore, enhance and conserve forest carbon stocks) and other payments for ecosystem services (PES) opportunities centered on locally-led conservation and sustainable use of AAB's mangrove ecosystem. Through the Blue Forests initiative, a team of 11 full time conservationists, field technicians, geospatial analysts, ecologists, socioeconomic scientists and community organizers are currently engaged with numerous communities. Most of this team is Malagasy. In addition to its staff, the Blue Forest project trains and supports Malagasy and foreign undergraduate and graduate students. Lastly and critically, the project places community members at the center of all activities.

In addition to the scientific work outlined in this chapter and Jones et al. (2014), in order to develop the conservation activities inherent to any REDD+ project, BV have been working closely with communities in AAB to empower them to manage their mangroves sustainably through:

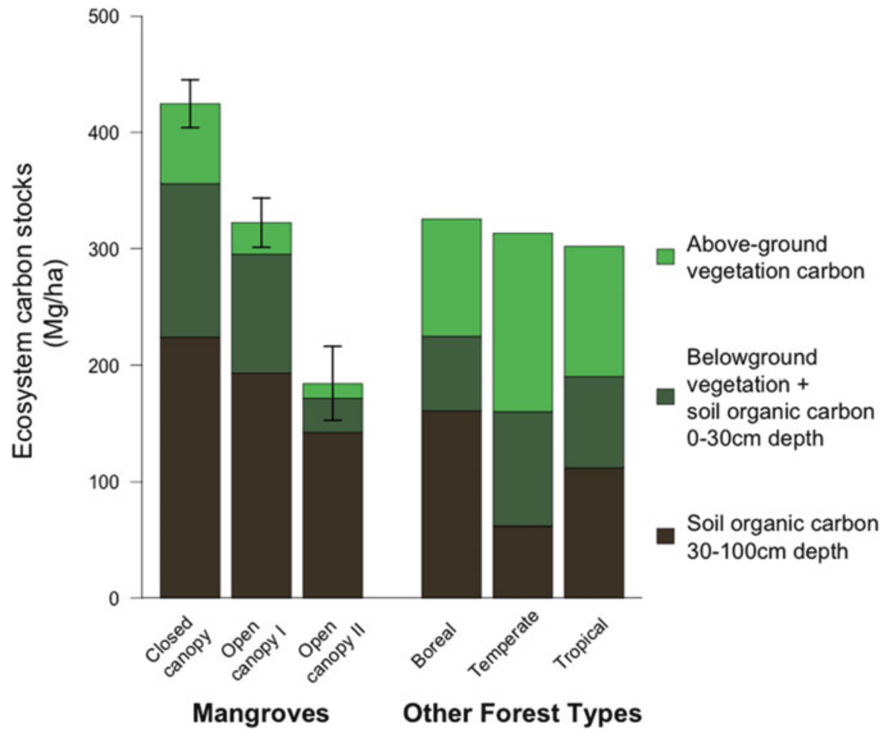

Fig. 11 Carbon stock estimates for AAB in Madagascar compared with published carbon stocks for different forest types. Carbon stocks for non-mangrove forest types are taken from Donato et al (2011). Soil organic carbon comparisons are made up to a depth of 100 cm. *Error bars* for AAB show ± SE of total aboveground and belowground carbon stocks.

- The establishment of legal natural resource management rights for community management associations;
- The development of mangrove plans for five management associations, using participatory mapping and field delineation;
- The capacity building of five community management associations in accounting, legal framework, forest inventory, tax collection on timber and mangrove patrolling;
- The involvement of community members in mangrove rehabilitation and fuelwood plantation programs;
- The development of livelihoods alternative to the exploitation of mangroves.

These activities and their results as presented in this chapter continue to be augmented in AAB in a number of ways:

Improving Above- and Below-Ground Carbon Stock Estimates

- The establishment of >50 additional carbon plots throughout the different dominant mangrove types and within areas that once were or once again could become mangroves;
- Improved soil carbon analysis in partnership with the Department of Forestry at the University of Antananarivo. New equipment acquired and personnel trained to carry out robust and reliable analysis of soil samples to determine soil organic carbon using alternative methods to Walkley-Black.

Mapping Mangrove Types and Their Deforestation and Modelling Change

- Developing and trialing community-based methods for carbon stock measurement and monitoring using Open Data Kit (ODK)
- Updating contemporary and historic mangrove and surrounding land-cover type distribution maps;
- Temporally stratifying deforested areas and collecting soil samples to understand the breakdown of carbon over time in relation to deforestation;
- Defining baseline deforestation;
- Projecting future deforestation scenarios through modelling based on historical changes and the results of deforestation and causal analysis;
- Estimating carbon stock changes related to different deforestation scenarios and the GHG emissions associated with each.

Strengthening/Supporting Community Management and Sustainable-Use Initiatives

- Helping community management associations establish functional community management contracts which transfer formal management rights, zones and responsibilities;
- Working to restore key ecosystem services (e.g., shoreline protection) through organization of and participation in mangrove restoration efforts;
- Integrating sustainable fisheries practices into community mangrove management plans;

- Strengthening forest and fishery administrations to enforce mangrove laws and support and monitor management effectiveness;
- Up-scaling and diversifying alternative fuelwood and timber plantations to supply local urban centers with sustainably sourced fuel-wood and timber;
- Helping to create alternative livelihoods (e.g., small-scale aquaculture; honey production; ecotourism).

Expanding Research Horizons

- Assessing the biodiversity impacts of continued versus altered land-use practices;
- Acquiring the equipment and establishing surface elevation tables to measure mangrove sediment dynamics and gain a true appreciation of the carbon cycle within the mangrove ecosystem of AAB.

All of these efforts are helping to pilot the application of an approved REDD+ methodology for AAB's mangroves. As all approved Verified Carbon Standard (VCS) REDD+ methodologies were designed for upland terrestrial forests, existing methodologies were compared based on how key methodological components (e.g., applicability conditions, project boundaries, baseline, leakage, monitoring, uncertainty, quantification of GHG emission reductions) apply to mangrove ecosystems. This comparison led to selecting the VM0009 version 2.1 "Methodology for Avoided Deforestation" for the pilot. More information on this particular methodology is available on the VCS website (www.v-c-s.org). By working through this approved methodology in its entirety and maintaining a constant dialogue with in-country government institutions, the Blue Forests project is contributing robust estimates of the greenhouse gas emission reductions achievable through mangrove REDD+ and helping integrate mangroves in to Madagascar's National REDD+ strategy. In addition to working through the application of an approved VCS REDD+ methodology, the project is also contributing to the development of new methodologies specific to the conservation of intact mangrove wetlands.

Of all their services, carbon sequestration continues to receive the most attention (Pendleton et al. 2012); however, establishing REDD+ or any carbon financing mechanism is not without challenges, including needing to secure traditional management and carbon rights, establish equitable cost and benefit sharing amongst stakeholders, avoid leakage, prove additionality, and combat the risk of non-permanence and liability. Realizing income from "blue" (i.e., marine) carbon is further complicated in that existing methodologies were designed for terrestrial forests, and many mangrove stands do not adhere to official forest definitions. Because REDD+ projects can take many years to establish, there are long-term uncertainties associated with carbon markets and policy. Mangrove ecosystems have multiple values (as noted by Alongi (2011), Grimsditch et al. (2012), Lau (2012) and Pendleton et al. (2012)), and any comprehensive mitigation strategy must also look beyond carbon to create shorter-term ways for coastal communities to gain from the sustainable management of mangrove fisheries and forest resources. As such, the Blue Forests project continues to research other avenues for the sustainable financing of community-led mangrove management, with a particular focus on fisheries value-chain improvement and PES. Through education outreach and peer-to-peer demonstrations, these efforts will be expanded to other mangrove communities in Madagascar. It is hoped that other coastal communities throughout Madagascar and beyond will also gain valuable understanding and may implement similar initiatives for revenue generation and improved management.

Conclusions

To date, across Madagascar, research and conservation initiatives have typically focused on areas of high biodiversity to the detriment of comparatively less biodiverse ecosystems, such as mangroves, and the critical services provided to their residents. Our research confirms that substantial deforestation is ongoing in AAB, which is amongst Madagascar's primary mangrove ecosystems. Since 2012, the BV Blue Forests project has been engaged with communities in AAB to combat this loss through helping leverage sustainable funds to improve management of mangroves and related resources. To date, the Blue Forests project has successfully quantified and contextualized loss in AAB, and through conservation, restoration and sustainable-use initiatives, including the first-ever piloting of approved VCS methodology to mangroves, is exploring myriad ways to help maintain and diversify local livelihoods and protect mangroves and associated biodiversity in AAB.

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References

Adame MF, Kauffman JB, Medina I, Gamboa JN, Torres O, Caamal JP, Reza M, Herrera-Silveira JA (2013) Carbon stocks of tropical coastal wetlands within the karstic landscape of the Mexican Caribbean. PLoS One 8:e56569

Alongi DM (2011) Carbon payments for mangrove conservation: ecosystem constraints and uncertainties of sequestration potential. Environ Sci Pol 14:462–470

Andriamahefazafy M, Carro A, England K, Ravaoarinorotsihoarana L (in preparation) Mangroves conservation in practice: policy and legal challenges of mangroves conservation and sustainable use in Madagascar, case studies of two mangrove projects

Chave J, Andalo C, Brown S, Cairns MA, Chambers JQ, Eamus D, Fölster H, Fromard F, Higuchi N, Kira T et al (2005) Tree allometry and improved estimation of carbon stocks and balance in tropical forests. Oecologia 145:87–99

Chen L, Zeng X, Tam NFY, Lu W, Luo Z, Du X, Wang J (2012) Comparing carbon sequestration and stand structure of monoculture and mixed mangrove plantations of Sonneratia caseolaris and S. apetala in Southern China. For Ecol Manage 284:222–229

Clough BF, Scott K (1989) Allometric relationships for estimating above-ground biomass in six mangrove species. For Ecol Manage 27:117–127

Cole TG, Ewel K, Devoe NN (1999) Structure of mangrove trees and forests in Micronesia. For Ecol Manag 117:95–109

Comley BWT, McGuinness KA (2005) Above- and below-ground biomass, and allometry, of four common northern Australian mangroves. Aust J Bot 53:431–436

Cooke A, Ratomahenina O, Ranaivoson E, Razafindrainibe H (2000) Chapter 60. Madagascar. In: Sheppard CRC (ed) Seas at the millennium: an environmental evaluation, vol 2, Regional chapters: the Indian Ocean to the Pacific. Pergamon, Amsterdam, pp 113–131

De Vos B, Letterns S, Muys B, Deckers JA (2007) Walkley-Black analysis of forest soil organic carbon: recovery, limitations and uncertainty. Soil Use Manag 23:221–229

Dharmawan IWS, Siregar CA (2008) Soil carbon and carbon estimation of Avicennia marina (Forsk.). Vierh. Stand at Ciasem, Purwakarta. J Penelit Hutan dan Konservasi Alam 5:317–328

Donato DC, Kauffman JB, Murdiyarso D, Kumianto S, Stidham M, Kanninen M (2011) Mangroves among the most carbon-rich forests in the tropics. Nat Geosci 4:293–297

Donato DC, Kauffman JB, Mackenzie RA, Ainsworth A, Pfleeger AZ (2012) Whole-island carbon stocks in the tropical Pacific: implications for mangrove conservation and upland restoration. J Environ Manage 97:89–96

Fromard F, Puig H, Mougin E, Marty G, Betoulle JL, Cadamuro L (1998) Structure, above-ground biomass and dynamics of mangrove ecosystems: new data from French Guiana. Oecologia 115:39–53

Giri C (2011) National-level mangrove cover data-sets for 1990, 2000 and 2010. United States Geological Survey, Sioux Falls

Giri C, Muhlhausen J (2008) Mangrove forest distributions and dynamics in Madagascar (1975–2005). Sensors 8:2104–2117

Giri C, Ochieng E, Tieszen LL, Zhu Z, Singh A, Loveland T, Masek J, Duke N (2011) Status and distribution of mangrove forests of the world using earth observation satellite data. Glob Ecol Biogeogr 20:154–159

Government of Madagascar (GoM) (1996) Law No. 96–025 of 30 September 1996, on the local management of renewable natural resources, Articles 1, 2

Government of Madagascar (GoM) (1997a) Decree No. 97–1200 of 2 October 1997, adopting the national policy of forestry, Line 15

Government of Madagascar (GoM) (1997b) Inter-ministerial order n°4355/97 on the definition of the sensitize areas, Articles 2, 3

Government of Madagascar (GoM) (2000) Order No. 12.704/2000 of 20 November, 2000, prohibiting any extractive activity of wood resources in sensitive areas

Government of Madagascar (GoM) (2005) Decree No. 2005–849 of 13 December 2005 revising the general conditions of application of Law No. 97–017 of 8 August 1997, revising forestry legislation, Chapter 4

Government of Madagascar (GoM) (2008) Law No. 2008–013 on the public domain, Government Gazette of 2008, Article 3(a)

Government of Madagascar (GoM) (2014) Order No. 32.100/2014 of 24 October, 2014, prohibiting any exploitation of mangrove wood on the national territory

Grimsditch G, Alder J, Nakamura T, Kenchington R, Tamelander J (2012) The blue carbon special edition—introduction and overview. Ocean Coast Manage 83:1–4

Heumann BW (2011) Satellite remote sensing of mangrove forests: recent advances and future opportunities. Prog Phys Geogr 35:87–108

Jones TG (2013) Shining a light on Madagascar's mangroves. Madag Conserv Dev 8:4–6

Jones TG, Rakoto Ratsimba H, Ravaoarinorotsihoarana L, Cripps G, Bey A (2014) Ecological variability and carbon stock estimates of mangrove ecosystems in northwestern Madagascar. Forests 5:177–205

Kauffman JB, Donato DC (2012) Protocols for the measurement, monitoring and reporting of structure, biomass and carbon stocks in mangrove forests, Working paper 86. CIFOR, Bogor

Kauffman JB, Heider C, Cole TG, Dwire KA, Donato DC (2011) Ecosystem carbon stocks of Micronesian mangrove forests. Wetlands 31:343–352

Kauffman JB, Heider C, Norfolk J, Payton F (2014) Carbon stocks of intact mangroves and carbon emissions arising from their conversion in the Dominican Republic. Ecol Appl 24:518–527

Komiyama A, Poungparn S, Kato S (2005) Common allometric equations for estimate the tree weight of mangroves. J Trop Ecol 21:471–477

Kuezner C, Bluemel A, Gebhardt S, Quoc TV, Dech S (2011) Remote sensing of mangrove ecosystems: a review. Remote Sens 3:878–928

Laran S, Van Canneyt O, Dorémus G, Mannocci L, Ridoux V, Watremez P (2012) Distribution et abondance de la mégafaune marine dans le sud-ouest de l'océan Indien tropical. REMMOA-Océan Indien. Rapport final pour l'Agence des Aires Marines Protégées, 168 p

Lau WWY (2012) Beyond carbon: conceptualizing payments for ecosystem services in blue forests on carbon and other marine and coastal ecosystem services. Ocean Coast Manage 2013(83):5–14

McKenna SA, Allen GR (eds) (2005) A rapid marine biodiversity assessment of Northwest Madagascar, Bulletin of the rapid assessment program 31. Conservation International, Washington, DC

Meersmans J, van Wesemael B, van Molle M (2009) Determining soil organic carbon for agricultural soils: a comparison between the Walkley & Black and the dry combustion methods (north Belgium). Soil Use Manage 25:346–353

Mikhailova EA, Noble RRP, Post CJ (2003) Comparison of soil organic carbon recovery by Walkley-Black and dry combustion methods in the Russian Chernozem. Commun Soil Sci Plant Anal 34:1853–1860

Nagelkerken I, Blaber SJ, Bouillon S, Green P, Haywood M, Kirton LG, Meynecke J-O, Pawlik J, Penrose HM, Sasekumar A et al (2008) The habit function of mangroves for terrestrial and marine fauna: a review. Aquat Bot 89:155–185

Pendleton L, Donato DC, Murray BC, Crooks S, Jenkins WA, Sifleet S, Craft C, Fourqurean JW, Kauffman JB, Marba N et al (2012)

Estimating global "blue carbon" emissions from conversion and degradation of vegetated coastal ecosystems. PLoS One 7:e43542

Polidoro BA, Carpenter KE, Collins L, Duke NC, Ellison AM, Ellison JC, Farnsworth EJ, Fernando ES, Kathiresan K, Koedam NE et al (2010) Mangrove extinction risk and geographic areas of global concern. PLoS One 5:e10095

Rasofolo MV (1997) Use of mangroves by traditional fishermen in Madagascar. Mangroves Salt Marshes 1:243–253

Rasolofo MV, Ramilijaona O (2009) Variability in the abundance and recruitment of Fenneropenaeus indicus and Metapenaeus monoceros postlarvae and juveniles in Ambaro Bay mangroves of Madagascar. Nat Faune 24:103–109

Ray R, Ganguly D, Chowdhury C, Dey M, Das S, Dutta MK, Mandal SK, Majumder N, De TK, Mukhopadhyay SK et al (2011) Carbon sequestration and annual increase of carbon stock in a mangrove forest. Atmos Environ 45:5016–5024

Richards M (2011) Social and Biodiversity Impact Assessment (SBIA) manual for REDD+ projects: part 2 – social impact assessment toolbox. Climate, Community & Biodiversity Alliance and Forest Trends with Rainforest Alliance and Fauna & Flora International, Washington, DC

Richards M, Panfil SN, Maldonado O (2011) Manuel sur l'évaluation des impacts sociaux et sur la biodiversité (EISB) pour les projets de REDD+: Première partie – Directives de base à l'intention des initiateurs de projets. Seconde version. Climate, Community & Biodiversity Alliance, Forest Trends, Rainforest Alliance et Fauna & Flora International. Washington, DC

Schumacher B (2002) Methods for the determination of Total Organic Carbon (TOC) in soils and sediments; Ecological Risk Assessment Support Center, Office of Research and Development. US Environmental Protection Agency, Washington, DC

Simpson WT (1996) Method to estimate Dry-Kiln schedules and species groupings: tropical and temperate hardwoods. US Department of Agriculture, Forest Service, Forest Products Laboratory, Washington, DC

Smith TJ III, Whelan KRT (2006) Development of allometric relations for three mangrove species in South Florida for use in the Greater Everglades Ecosystem restoration. Wetl Ecol Manage 14:409–419

Spalding M, Kainuma M, Collins L (2010) World atlas of mangroves. Earthscan, London

Tomlinson PB (1986) The botany of mangroves. Cambridge University Press, Melbourne

Wang G, Dongsheng G, Peart MR, Chen Y, Peng Y (2013) Ecosystem carbon stocks of mangrove forest in Yingluo Bay, Guangdon Province of South China. For Ecol Manage 310:539–546

Section II

Meanwhile, the Western Indian Ocean Estuaries Are Under Threat. . . . .

The West Indian Ocean/WIO region is one of the least ecologically-disturbed areas of ocean in the world. Nevertheless, several of its estuaries and deltas are experiencing serious stresses attributable to land-based activities taking place in the upstream areas draining into them. This situation is jeopardizing many of the ecosystem services they provide, including the well-being of the local communities's dependent upon them. In fact, the region's coastal and marine environment has been exhibiting significant signs of degradation over the last decade. A number of natural factors contribute to this degradation, including climate change/variability leading to coral bleaching, sea level rise, flooding, etc. However, there are also a range of pressures and drivers attributable to human activities, acting at different intensities and in various combinations, including agricultural development, urbanisation, deforestation, river damming and industrialisation.

Although the region's overall population density is not remarkably high, still more than 60 million people inhabit its coastal zones In fact, the WIO coastal zone hosts most major cities, harbours, industries and other socio-economic infrastructure in the region, a factor that is increasingly affecting the coastal and marine environment A major problem is the pollution from domestic, industrial and agricultural sources that is degrading water and sediment quality in some key spots within the WIO region. This pollution, in turn, is causing decreased biological diversity, human health problems, and reduced fish stocks and catches. Further, the increasing population pressures, combined with inadequate alternative resources being available to sustain the local populations, are resulting in unsustainable extraction of marine resources. Coastal habitats also have been converted to other uses in some areas, including agriculture, aquaculture, ports/harbours and urban settlements, which are destroying vital coastal habitats, including mangrove forests, sand dunes, sea grass beds and coral reefs. The physical alteration of the WIO coastline from erosion and accretion participate indeed to the loss of the natural coastal protection and regulation functions of coastal habitats.

One key WIO concern focuses on the interactions between river basins and their coastal-marine environment. Important transformations of the coastal and marine environment can be attributed to human activities and climatic variability occurring in the river basins that drain into them. The impacts of these human activities include impeded freshwater flows, terrigenous sediment and increased organic matter, which collectively have significantly altered the interactions between river systems and downstream coastal processes. It is clear that water quality degradation in some major river systems draining into the WIO estuaries, deltas and other coastal ecosystems is the results of nutrients and other pollutants from domestic sewage, industrial and agricultural chemicals from human activities in these upstream river systems.

The available evidence indicates alteration of river flows is the most common factor throughout the WIO region in regard to the severity of associated river issues. This problem can be traced to increased water abstraction, damming and land use changes that are subsequently altering the hydrological dynamics of the WIO estuarine region. There also are cases whereby sediment loading and water quality changes have resulted in severe impacts on the productivity of critical WIO coastal habitat, including mangroves, sea grass beds and coral reefs.

The following chapter provide an overview of some of the main cases of impacts related to human factors on estuaries and related coastal habitats in the WIO region.

Tana Delta and Sabaki Estuaries of Kenya: Freshwater and Sediment Input, Upstream Threats and Management Challenges

Johnson U. Kitheka and Kenneth M. Mavuti

Abstract

This study is focused on the determination of the extent to which changes in river freshwater and sediment input affects the sustainability of the Tana Delta and Sabaki estuaries in Kenya. The study involved the determination of river freshwater and sediment fluxes, as well as water exchange and sediment fluxes at the mouths of the two estuaries. The horizontal and vertical distributions of tidal current velocities, salinity and total suspended sediment concentrations (TSSC) within the estuaries enabled determination of the degree of stratification and the extent to which mixing of seawater and freshwater leads to the formation of the maximum turbidity zone (TMZ) in the two estuaries. The two estuaries are important for biodiversity conservation, sustainability of socio-economic livelihoods and provision of global environmental benefits. The study shows that the hydrologic dynamics controlling water circulation including the trapping and exchange of terrigenous sediments in the two estuaries is a function of the river discharge and tidal forcing. In the much smaller Sabaki estuary, there has been a reduction in freshwater input and an increase in sediment supply leading to heavy accretion. The shallow nature of the Sabaki estuary ensures reduced penetration of the semi-diurnal tidal wave into the estuary and seawater intrusion is restricted to 2.5 km of the estuary. On the other hand, there has been a substantial reduction in both freshwater input and sediment supply into the Tana Delta. This has led to deepening of the estuary channels with the result that tidal wave penetrates much deeper into the estuary and seawater intrudes up to 10 km inside the estuary. The tidal asymmetry in the two estuaries is characterized by ebb tidal flow dominance due to presence of mangrove forests, wide intertidal areas and freshwater input. This has resulted in net export of sediments out of the two estuaries. However, the cohesive clay sediments are trapped within the estuaries in mangrove forest wetlands and in sheltered intertidal areas that are now occupied by mudflats. The shallow Sabaki estuary experiences greater rates of water and sediment exchange as compared to the relatively deeper Tana estuary. The changes in freshwater and sediment supply into the two estuaries were attributed to landuse change, damming and climatic variability. The major impacts in both estuaries include high turbidity, heavy sedimentation, changes in beach morphology and degradation of the marine ecosystems such as the coral reefs and seagrass beds. In

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the Tana delta system, the impacts include high turbidity, alteration of the morphology of the delta, degradation of the mangrove forests, coastal erosion and sea water intrusion. The study notes that the proposed large-scale hydropower and irrigation projects in the Athi-Sabaki and Tana river basins have the potential of causing massive degradation of the two estuaries. The paper puts forward recommendations for sustainable management of the two estuaries in Kenya.

Keywords

Tana and Athi • Sabaki • River basins • Estuaries • Freshwater supply • Sediment input • Water exchange • Sediment trapping • Landuse change • Climate variability • Hydrology

Introduction

Tana Delta and Sabaki estuaries are two major estuaries found along the Kenya coast in East Africa. The two estuaries are important for biodiversity conservation and sustainability of livelihoods. The two estuaries are highly vulnerable to global climate variability because there are sustained by freshwater inflows which are depended on rainfall which in the recent past has shown significant variability due to climate change. The estuaries are also facing diverse range of anthropogenic stresses related to landuse change, damming and water abstraction. The natural and human-related stresses in the two estuaries are influencing the key drivers of estuarine ecosystem health such as river runoff and tidal seawater incursion. The stresses associated with land-based development activities in the river basins are causing more stresses to the two estuaries as compared to marine-based stresses. The impacts of climate variability are being exacerbated by over-exploitation of natural resources and land use change as a result of rapid population growth and expansion. The increasing magnitude of natural and anthropogenic stresses in the two systems makes it important that information is generated to guide decision–makers in the processes of approving development projects in the upstream areas including also the formulation of strategies for the protection of the two estuaries at the coast.

The two estuaries present interesting contracts in terms of their spatial extent, hydrology, geomorphology, land uses and pressures. Tana Delta has several distributaries and estuaries, making it much larger as compared to the Sabaki Estuary. The Tana Delta receives much greater river runoff on account of its origin from the high rainfall region of Central Kenya Highlands. In addition, the Delta has a much larger extent of mangrove forest including riverine forests along the upper reaches of the Tana river. The Sabaki estuary has a small patch of mangrove forest and lacks a well-formed riverine forest in its upper course. In terms of development, the Tana river basin has been subject of major development projects which includes hydropower development, water supply and irrigation schemes. There are few major development projects in the upper course of the Athi River with the exception of industrial development in Nairobi region. Both estuaries are considered important for biodiversity conservation and they also support livelihoods of coastal communities including the sustainability of the productivity of Kenya's most important fishing ground through the supply of freshwater, sediments and nutrients.

Previous studies on the two estuaries have focused mainly on estuarine hydrodynamics (Kitheka et al. 2003a, b, 2005). These studies have been critical in understanding key features of the two estuaries. Past studies in the two estuaries and Ungwana Bay have also elucidated on the seasonal changes in the movement of turbid water plume (Brakel 1984; McClanahan and Obura 1997; Kitheka (2013). There have also being several investigations on the impact on high sediment load on the biodiversity particularly on corals within the Malindi bay (McClanahan and Obura 1997). Most of the previous hydrological investigations have been focused on the upper Tana Basin, with few studies on the Athi-Sabaki river Basin (e.g. Ongwenyi 1983; Pacini et al. 1998; Maingi and Marsh 2001; Maingi and Marsh 2002). There is lack of information on the effects of hydrologic alteration of the rivers discharging into the two estuaries. This study attempts to fill this gap by examining the dynamics of freshwater and sediment input into the two estuaries and how changes in the magnitude of freshwater and sediment supply have influenced the two estuarine systems. The study also examines how changes in the magnitude of river freshwater and sediment input influences estuarine water exchange and the morphological structure of the two estuaries. The study is also important in understanding the impacts of global environmental change on tropical estuaries as it also examines the effects of climatic variability of freshwater input in the two estuaries. The study provides information that can be used to guide the processes for the formulation of strategies for sustainable management of the Tana and Athi-Sabaki estuaries in view of their importance for biodiversity conservation, provision of livelihoods and global environmental benefits such as carbon sequestration. The later is important in mitigating the effects of climate change.

Description of the Study Area

Tana Delta and Estuary

The Tana Delta (Longitude 2°30'0"S & Latitude 40°18'20" E) receives runoff from the Tana river- the largest and the most important river in Kenya (GOK 1979). The delta consists of several distributaries in which the river divides into several branches in the area south of Garsen (Fig. 1). The river drains from the Central Kenya highlands as shown in Fig. 2. The points of outflow of the Tana river have changed significantly in the past as evidenced by the present drainage pattern of the delta (cf. Ojany 1984; Kairu 1997). The current main branch is at Kipini (Photo 2). The Tana delta is broad and crescent shaped at the coast covering a distance of nearly 40 km from Kipini distributary to the east to Mto Kilifi distributary to the south. The width of the delta decreases at an exponential rate from 40 km at the coast to about 10 km near Garsen- a distance of about 60 km. The estuaries are fringed by mangrove forests within the tide affected zone and in the upper freshwater zone is found a large freshwater water swamp. Riverine forests are found along the river banks but these been cleared in most areas within the river flood plains and only scattered patches are remaining. The mangrove forest is the most important ecosystem at the delta with dominant mangrove species such as *Xylocarpus granatum*, *Avicennia marina*, *Rhizophora mucrunata* and *Ceriops tagal*. The estuary and the turbid coastal waters are an important nursery ground for prawns and numerous species of fish and crustaceans. The sandy shores along Ungwana are considered important breeding grounds of turtles (FAO 1981; UNEP 1998; KMFRI 2002). Rainfall in the basin varies from 1500 mm per annum in the upper Tana Basin to about 500 mm in the semi-arid lower Tana Delta region.

Athi-Sabaki Estuary

The Sabaki Estuary (Longitude: 3.2°S & Latitude: 40.15°E) is located about 10 km north of Malindi town in Kenya (Fig. 3). The estuary receives freshwater and terrigenous sediment load from a 70,000 km^2 Athi-Sabaki river basin (Fig. 4). The estuary is characterized by heavy sedimentation (Photo 1). The estuary is vital for marine biodiversity conservation as it is considered an important bird sanctuary. Thousands of birds of various species forage on the intertidal mudflats during low tide. The estuary and the nearby turbid coastal waters are also an important nursery ground for prawns and numerous species of fish and crustaceans, some of which are of commercial importance (Ruwa et al. 2001; KMFRI 2002). The sandy shores flanking either sides of the mouth of the estuary extending into Malindi Bay are important breeding grounds of turtles (FAO 1981; UNEP 1998). The estuary also has a small mangrove forest habitat. High sediment load of the river has led the heavy siltation of the estuary and the adjacent Malindi Bay (Abuodha 1998; Delft Hydraulics 1970; GOK-TARDA 1981a–c; Ongwenyi 1983; McClanaham and Young 1996; Munyao 2001; Blom et al. 1985; Brakel 1984). Rainfall in the basin ranges from 1200 mm.yr^{-1} in the upper regions to less than 750 mm.yr^{-1} in the lower Sabaki estuary region. The rainy season is in the period between March and May and also in the period between October and December (Ojany and Ogendo 1986).

Methodology

Determination of Freshwater Input

The monitoring of river freshwater input was undertaken at River Gauging Stations located at Malindi and Garsen (RGS 4G02) for two years (2001–2003). Measurements were carried out on a monthly basis using standard hydrological procedures involving determination of river flow velocities using the cross-sectional-area velocity approach as described in Linsley et al. (1988). The past records of the discharges of the two rivers were obtained from the Water Resources Management Authority (WARMA). The rainfall data for stations located in the basins were obtained from Kenya Meteorological Department. These were used to establish the relationship between rainfall and river discharges in the two basins.

Determination of River Sediment Input

The river total suspended sediment concentrations (TSSC) were determined through filtration using Whatman GF filters according to APHA (1992) methods. Water samples were drawn at the middle of the main channel of the river, using a Niskin Sampler. These were subsequently analysed at the Kenya Marine and Fisheries Research Institute (KMFRI) marine environmental studies laboratory in Mombasa.

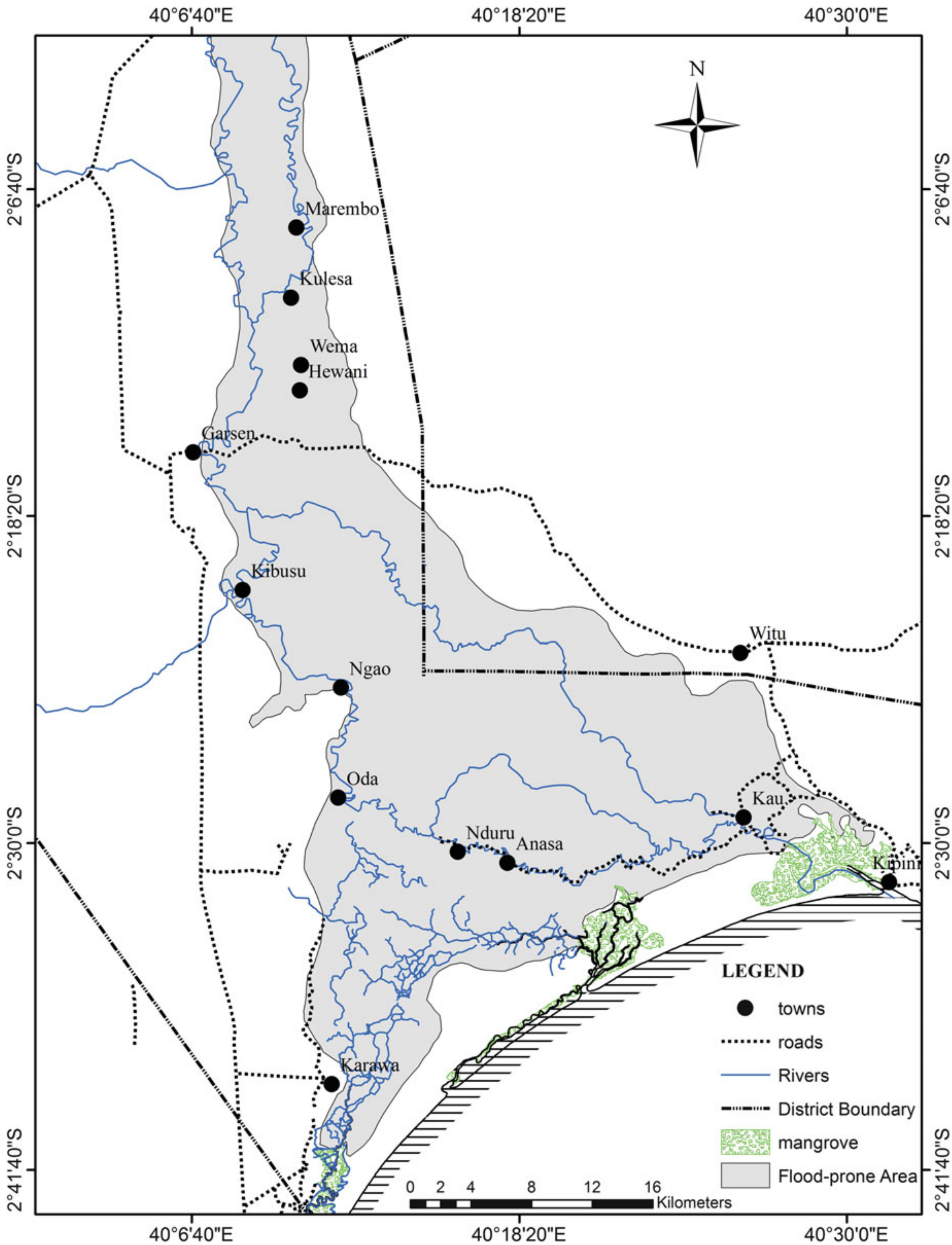

Fig. 1 The extent of the Tana Delta including the drainage network and mangrove forests

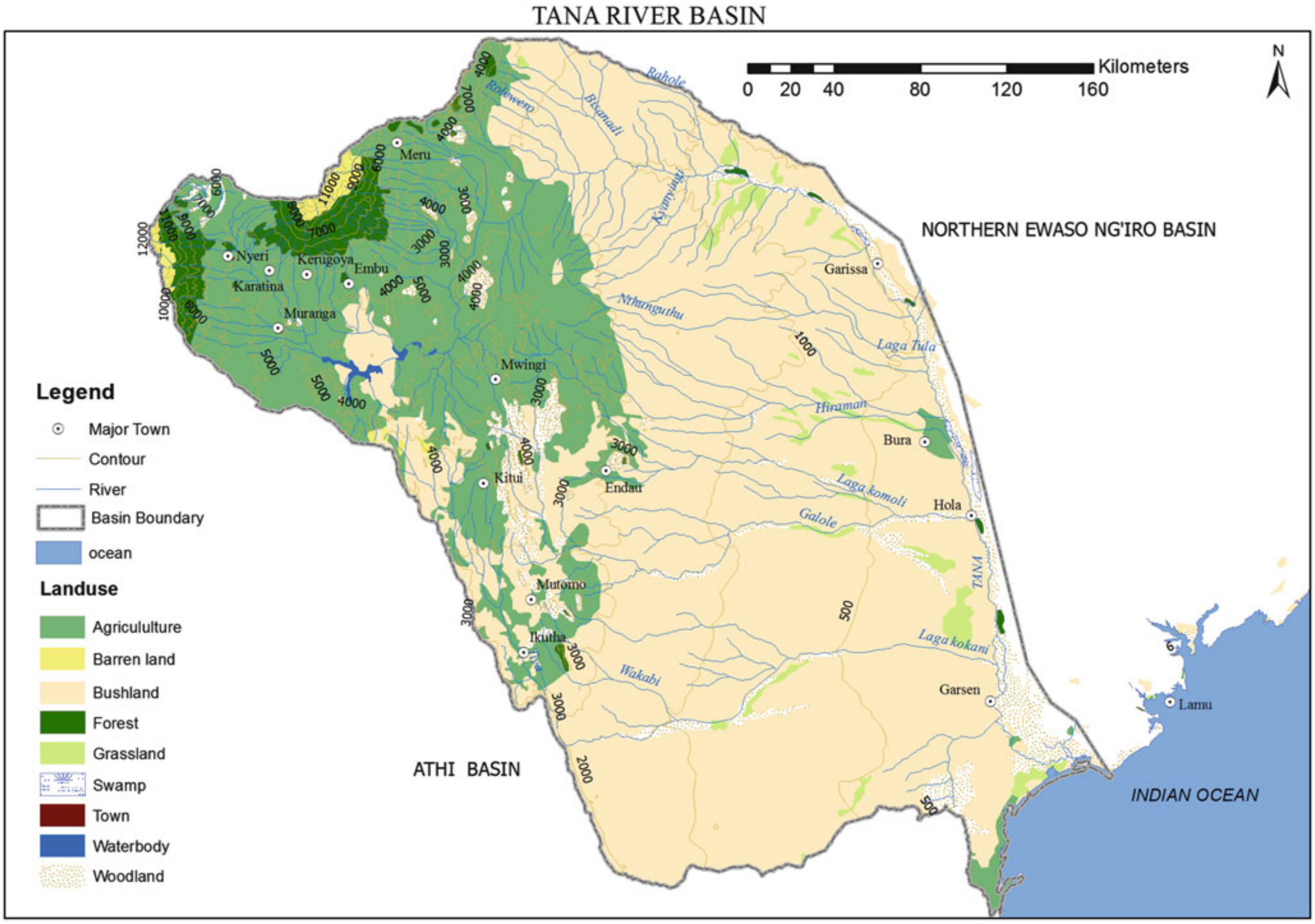

Fig. 2 The extent of the Tana River Basin in Kenya

Determination of Changes in River Freshwater Input

The study relied on past river discharge records archived by WARMA to determine the changes in the flow of the Athi and Tana river. In order to determine the changes of the Tana River, the past river discharge data for the Garsen river gauging station was split into pre-dam and post-dam periods. The pre-dam period was between 1951 and 1981 while the post-dam period was between 1982 and 1993. The pre-dam period was limited to the year 1981 because Masinga Dam which has had the most important impact on the river was commissioned in that year. The construction of the Masinga Dam in 1981 led to major impoundment of the river leading to significant impacts as compared to other relatively small dams that were constructed earlier before 1981. It is on this basis that we have used the construction of Masinga dam in 1981 as an important delineation of the period when significant impacts on the flow of the Tana river started being experienced. The past data for the Athi river for the period between 1952 and 1995 were also analysed to determine whether there has been any significant changes in the freshwater input at the Sabaki estuary. The most current river discharge records for the two rivers were unavailable partly as a result of the cessation of river discharge monitoring programme in 1998.

Using the past river discharge data, the daily and monthly river discharges were summed up and the mean and maximum flows were computed. The mean and maximum flows were plotted to establish key trends. The assessment of the impacts of water abstractions was based on the review of the findings of the past technical reports and development plans produced by the various consultancy companies for the Government of Kenya and the Tana and Athi Rivers Development Authority (TARDA) (e.g. ILACO 1971; GOK 1979; GOK TARDA 1982; GOK-JICA 1998).

Determination of Estuarine Water Circulation Dynamics

A field monitoring campaign was undertaken to establish the horizontal and vertical distribution of TSSC concentrations, ebb and flood tide current velocities and tidal elevations in

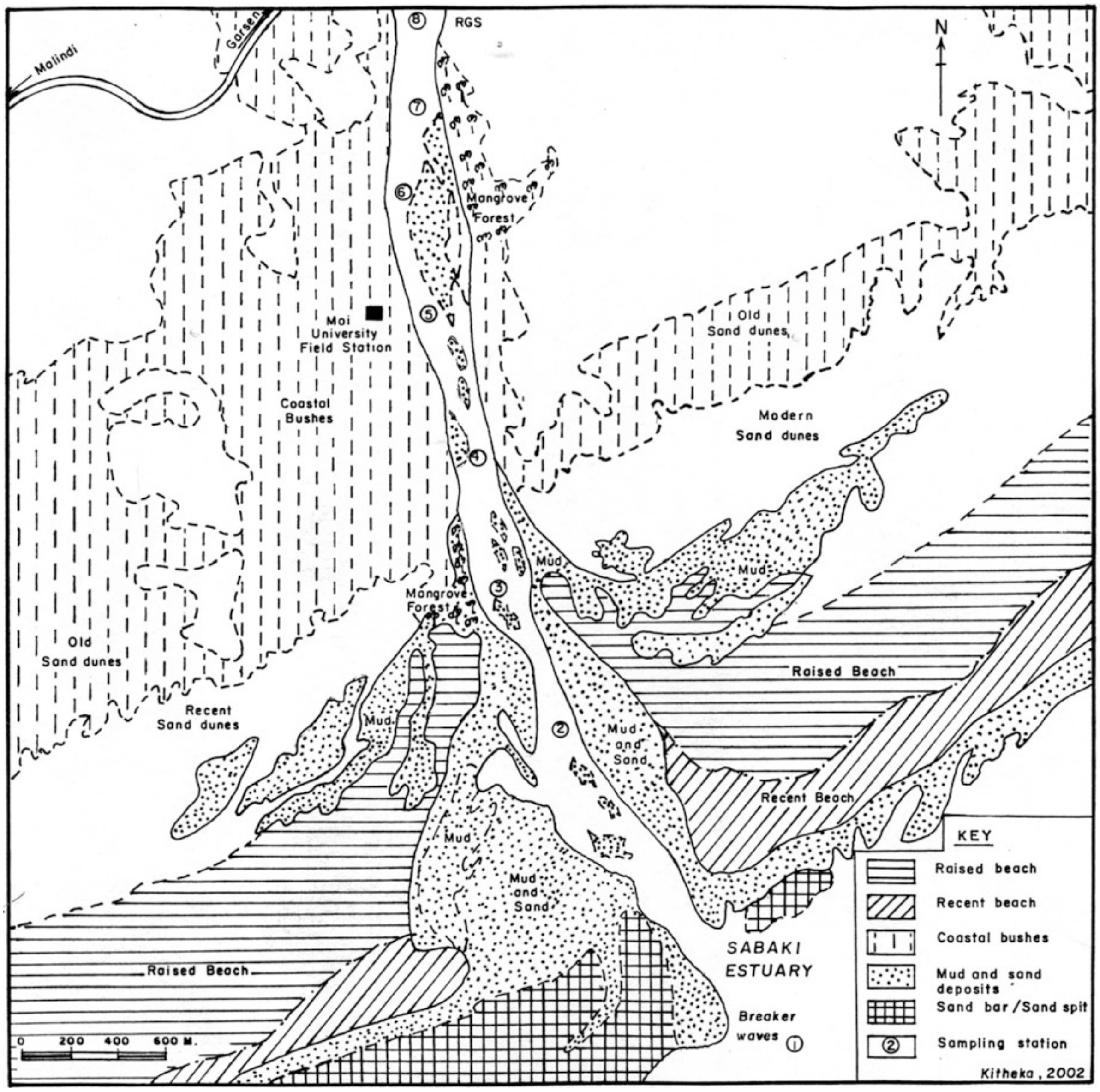

Fig. 3 Main physical features of the Sabaki Estuary and the immediate areas. Locations of sampling stations are also shown

the estuaries. Measurements of these parameters were undertaken in eight (8) sampling stations established in both Sabaki and Tana estuaries. For the Sabaki estuary, the sampling stations were located 0.5 km apart while in the Tana estuary they were located 1–1.5 km apart with stations 1 being located at the mouth of the estuary and stations 8 being located at the maximum limit of tidal excursion.

The TSS concentrations were determined using two approaches. The first approach involved the use of a turbidity sensor fitted on an Aanderaa Recording Current Meter (RCM-9) which was moored in different periods at stations 3 in the two estuaries. The turbidity sensor was programmed to log in turbidity at interval of 5 minutes. The second approach involved filtration of water-sediment mixture in the laboratory using Whatman GF filters according to APHA (1992) methods. In this method, water-sediment mixture samples at each station were drawn using a Niskin sampler at 0.2h, 0.4h, 0.6h and 0.8h, where h is the local water depth. Salinity in each of the sampling stations was measured at same vertical depths using a hand-held Aanderaa Salinometer and was expressed in terms of Practical Salinity Units (PSU). Tidal water levels were measured using divers' pressure gauges mounted on a RCM-9, which also logged in water level data at intervals of 5 minutes. Using a fast rubber dinghy, spot measurements of TSSC, salinity, tidal current velocities and water depths were carried out during high water (HW) in each of the 8 stations established in the Tana and Sabaki estuaries.

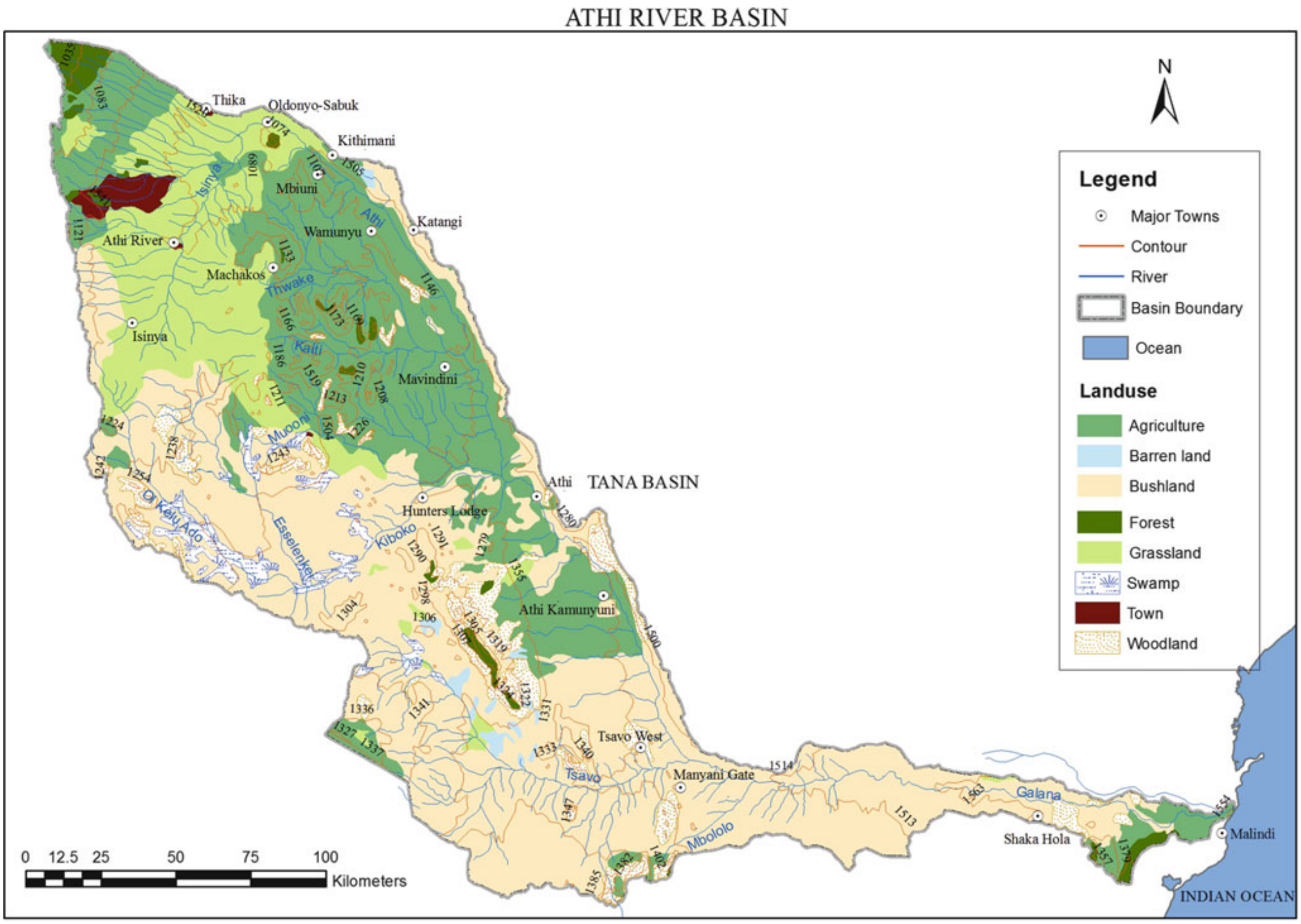

Fig. 4 The extent of the Athi-Sabaki River Basin in Kenya.

Photo 1 The mouth of the Sabaki Estuary showing heavy deposition of sediments and debris. The main channel and sand dunes are visibility on the far edge of the photograph

Photo 2 The Tana Estuary at Kipini showing the mangrove-fringed main channel and turbid water at high tide

Determination of the Coastal State Changes

The determination of coastal state changes was done through rapid field assessments. Field surveys were undertaken within the estuaries to determine the extent of sediment accretion and erosion, and the state of mangrove forests, among others. Observations were also made on the nature of sediment deposition within the mangrove forests, tidal channels, intertidal areas and along the shorelines. Sediment samples taken in these areas were subjected particle-size analysis using sediment sieves. Additional details on anticipated impacts were made through literature review of the findings of past studies.

Results

Input of Freshwater into the Estuaries

The freshwater input in the Tana Delta and Sabaki estuaries is important in controlling estuarine processes. The freshwater inputs are however highly variable with that of Sabaki estuary showing greater variability as compared to that of the Tana estuary. The river runoff in the upper zone of the Tana delta varied from 60 m^3s^{-1} in the driest period to 730 m^3s^{-1} in the wettest period. In the Sabaki estuary, the river runoff varied from 7 m^3s^{-1} in the driest period to 680 m^3s^{-1} in the wettest period (Figs. 5a, 5b, 6a and 6b). The river freshwater inputs in the Tana delta were in most cases >50 m^3s^{-1} as compared to the Sabaki estuary where it was low as 2 m^3s^{-1} during dry season. For the Sabaki estuary, the freshwater inputs < 70 m^3s^{-1} were more common they occurred 90% of the time. The high freshwater inputs > 150 m^3s^{-1} that are normally experienced during rainy season were less frequent in that they occurred in < 10% of the time. For the Tana delta, the freshwater inputs <300 m^3s^{-1} were more frequent. The mean annual freshwater discharge into the Sabaki and Tana Delta estuaries were estimated to be 2,300 × 10^6 $m^3.yr^{-1}$ and 4,700 × 10^6 $m^3.yr^{-1}$, respectively.

The input of freshwater in the two estuaries shows significant seasonal variability. There are in general two periods of high freshwater input that are usually separated by dry periods in which the river freshwater input is of the order 50 m^3s^{-1} and 2 m^3s^{-1} for the Tana Delta and Sabaki estuaries, respectively. The periods of high freshwater inputs are also characterised by high sediment supply (Figs. 6a and 6b). These are March-April-May-June period (MAMJ) during the South-East Monsoon and the October-November-December-January (ONDJ) period during the North-East monsoon. The river freshwater inputs during the MAMJ period are in general much higher than those during the ONDJ period. There is evidence of significant inter-annual variability of river freshwater inputs in the two estuaries which is related to rainfall in the river basins (cf. Ovuka and Lindqvist 2000). Figure 7 show the time-series plot of rainfall and the corresponding river discharges in the Tana delta. Figures 6a, b shows the inter-annual variability of the Athi-Sabaki river discharge.

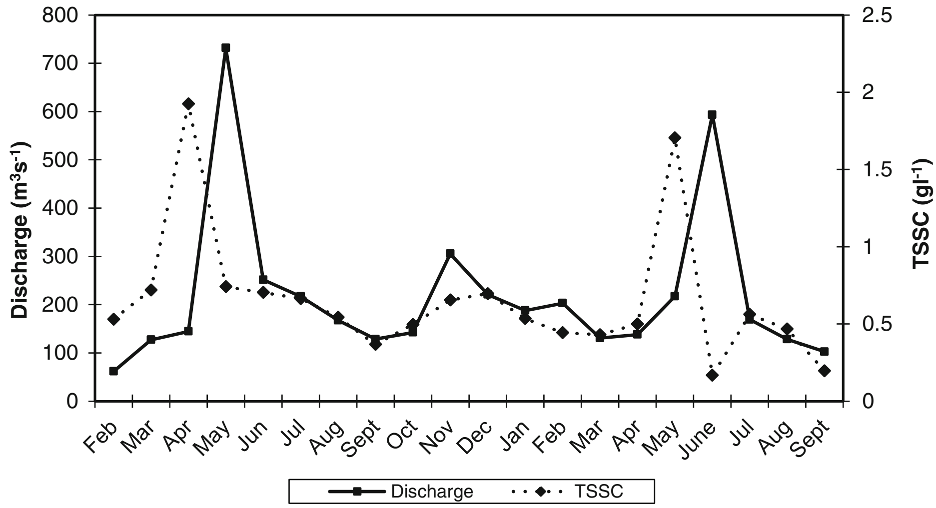

Fig. 5a Tana Delta freshwater input and TSSC in the period between January 2001 and September 2003

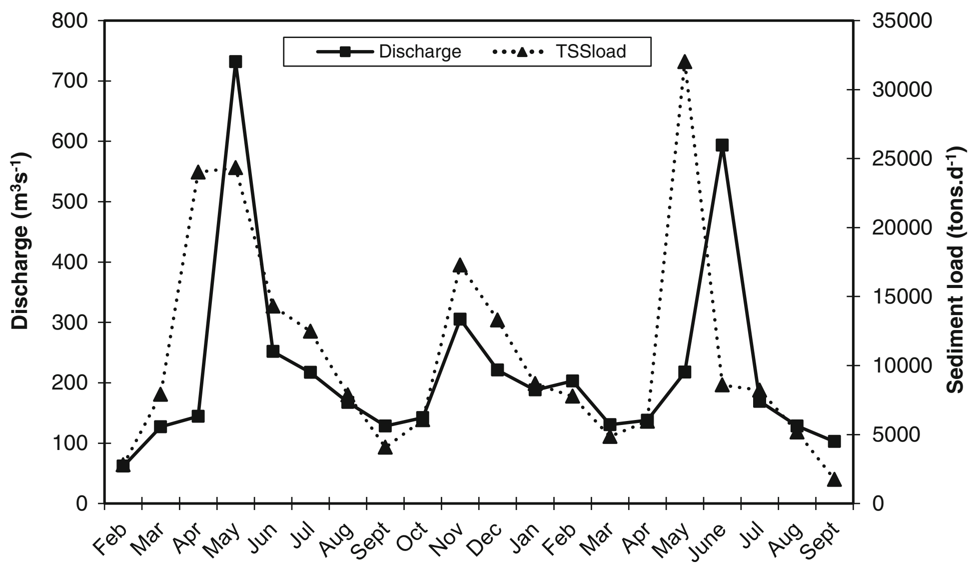

Fig. 5b Tana Delta freshwater input and sediment load in the period between January 2001 and September 2003

The Input of Sediment Load into the Estuaries

The river runoff into the Sabaki and Tana estuaries is characterized by highly turbidity as a result of high suspended sediment concentrations (TSSC). However, the Tana river was relatively less turbid as the TSSC varied from 0.53 to 1.5 g.l^{-1}. The Sabaki river was relatively more turbid as the TSSC ranged from 0.3 to 2.5 gl^{-1}. The peak river TSSC was measured in both rivers during the long rain season in the months of April and May. For both estuaries, the relationship between river runoff and river TSSC was such that at the beginning of the rainy season, the relatively low river discharges were characterized by high TSSC. There was a decline in TSSC as the rainy season progressed due to reduction of materials to be transported and also improvement of vegetation cover that retards soil erosion.

The total sediment load input into the Tana Delta was relatively lower than that of the Athi Sabaki as it ranged from 2,797 to 24,322 tons.day^{-1} while in the Sabaki estuary it ranged from 30 tons.day^{-1} to 133,000 tons.day^{-1}. The annual sediment load for the Tana river was estimated to be 6.8 $\times$ 10^{6} tons.yr^{-1} while that of the Sabaki was estimated to be 5.7 $\times$ 10^{6} ton.yr^{-1}. The peak sediment load in both rivers occurred in May during the South-East monsoon in a period when high flows were measured (Figs. 5a, 5b, 6a and 6b). There have been significant changes in the sediment loads of the two rivers based on data from previous studies. Ongwenyi (1983) estimated sediment load of the Tana river before the damming of the river to be 12 $\times$ 10^{6} tons.yr^{-1} which is much higher as compared to the sediment load of the Tana upstream of Masinga dam (8.5 $\times$ 10^{6} tons.yr-^{1}) (Otieno and Maingi 2002) which provides an indication of

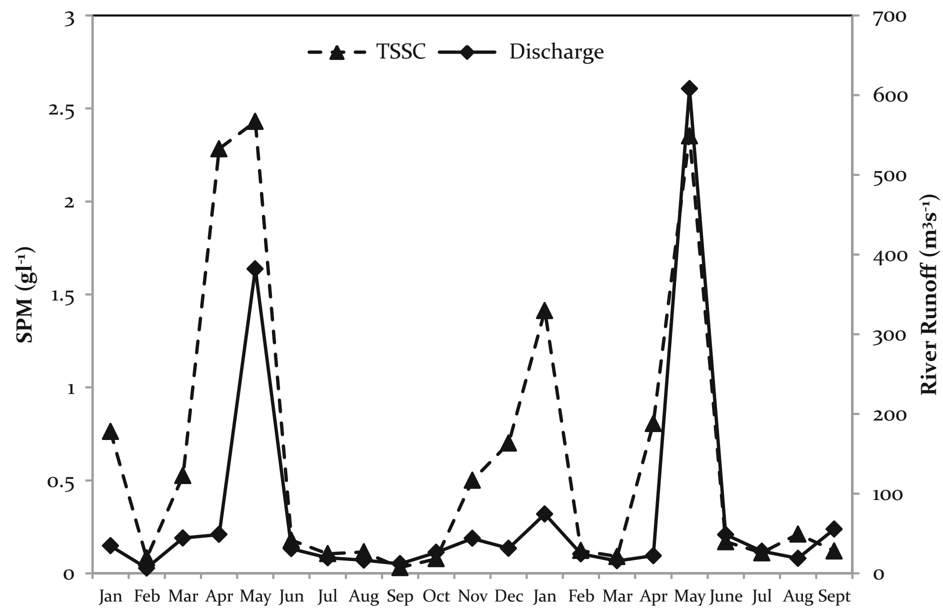

Fig. 6a The monthly averaged instantaneous river discharges and TSSC in the Athi Sabaki river between January 2001 and December 2003

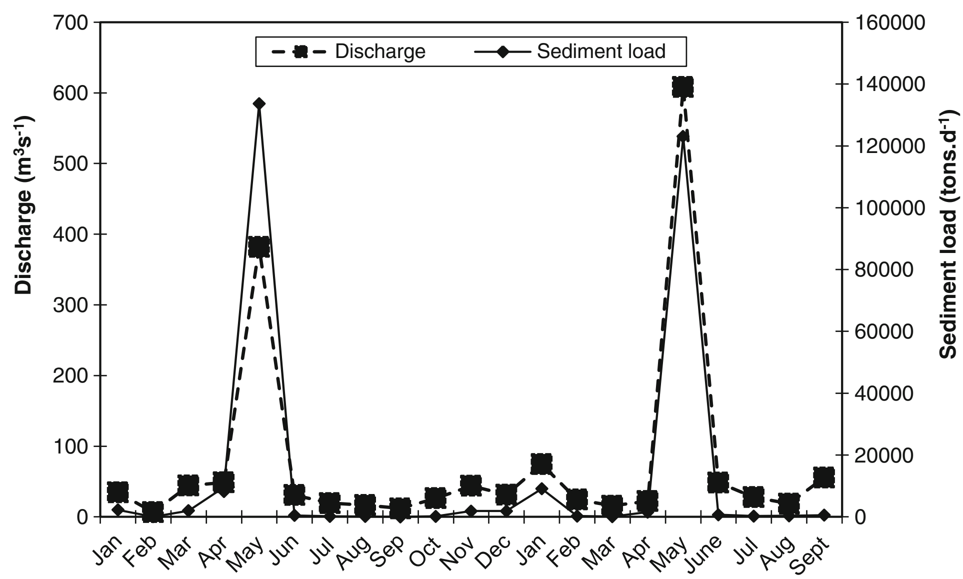

Fig. 6b The monthly averaged river discharges and sediment load in the Athi Sabaki river between January 2001 and December 2003

pre-damming sediment loads. This study found the sediment load of the Tana river at Garsen in 2003 to be 6.8×10^6 tons.yr^{-1} which is still much lower than that measured before the damming of the river. Assuming that these data are accurate, it can be argued that the sediment load of the Tana river has reduced by 30 to 60% as a result of damming of the river in its upper course (Gibb 1959; ILACO 1971). While there has been a reduction in sediment load of the Tana river, data shows that there has been a substantial increase in sediment load of the Athi -Sabaki river. In the pre-colonial periods, it was estimated that the sediment load of the Athi river was of the order 50,000 tons.yr^{-1} (Ongwenyi 1983), which is much lower than the sediment load of 5.7×10^6 ton.yr^{-1} measured in 2003. Assuming that the pre-colonial period sediment load data is accurate, it would mean that there has been almost 100% increase in sediment supply to the Sabaki estuary in the last 80 years.

Hydrologic Alterations of the River Tana and Athi-Sabaki

There have been significant changes in the hydrology of the two river systems as a result of landuse changes and

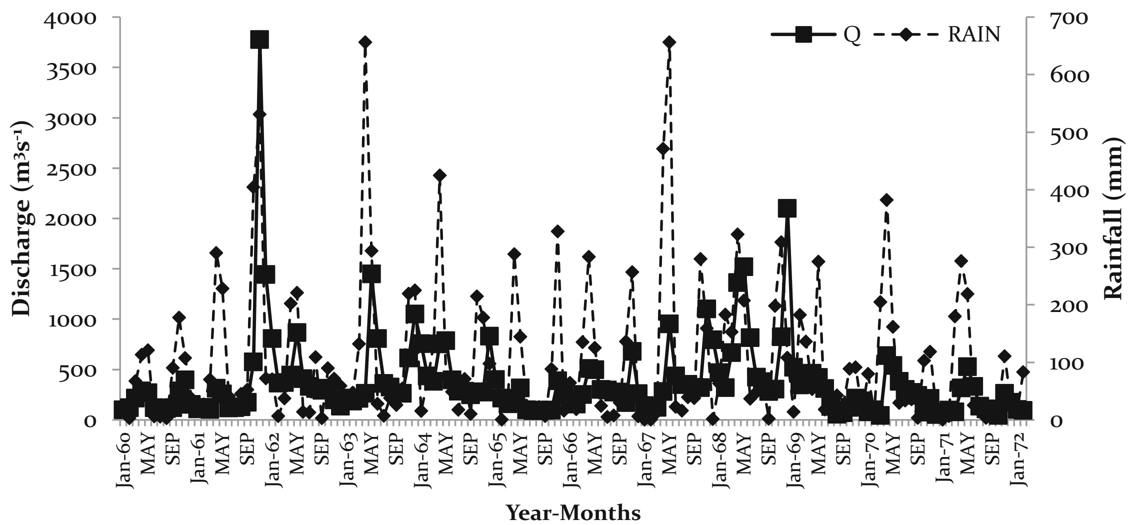

Fig. 7 The relationship between rainfall and Tana river discharge at Garsen in the period between 1960 and 1972

developments in the river basins. The damming of the Tana river by construction of Masinga dam in the Upper Tana Basin in 1981 effectively regulated the flow of the river downstream. As compared to other dams in the Upper Tana Basin, Masinga dam has had more significant impact on the flow of the Tana in view of its large spatial extent (surface area of 113 km^2) and volume (1,560 $\times$ 10^6 m^3). An attempt was made to establish whether there are significant changes in the maximum and mean flows of the river during both dry and wet seasons after and before construction of Masinga dam. Figure 7 show the mean and peak river freshwater input into the delta at Garsen before and after the construction of Masinga dam. The maximum river discharges in MAMJ and ONDJ period refers to the flood events during which there is high input of freshwater into the delta. However, the maximum river discharges during periods of relatively low river discharges (JFM and JAS) refers to the maximum streamflows during those specific periods. It can be seen that the magnitude of flood flows in the period before and after the construction of the dam are quite similar. There has however been a reduction in mean river freshwater input into the delta. There has also been a significant shift in the period of maximum river freshwater input into the delta during rainy season – in that they now occur about one month later. Before damming of the Tana River, the maximum river freshwater input into the Tana Delta occurred in the months of April and December. However, following the construction of Masinga dam, this has shifted to the period between May and June and also in November. It can also be seen that following damming of the river, the period of occurrence of maximum mean river discharge coincides with the period of maximum flood flows (Fig. 8).

In the case of the Athi river, available data indicates a significant progressive decline in the mean daily river discharges as well as the maximum daily discharges that occurs in the two rainy seasons (Figs. 9a and 9b). This significant reduction in river runoff in the upper course of the Athi river can be attributed to human related factors such as landuse change. The natural climate variability also seems to be important although there is a need for further investigation in this area. It is however important to note that in the past 100 years, there has been substantial changes in landuse in the Athi river basin as evidenced by extensive deforestation, overgrazing and expansion of settlements in the catchment areas of the river situated in Eastern and Central Kenya (Kiambu, Kajiado and Machakos Counties).

Salinity and Seawater Intrusion in the Estuaries

In both the Tana and Sabaki estuaries, the distribution of salinity and TSSC showed significant horizontal and vertical variation attributable to tidal incursion of seawater and river freshwater and sediment input (Figs. 10 and 11). The degree of salinity variation in both estuaries was more-or less similar despite differences in the magnitude of freshwater supply. The salinity in the Tana estuary (with greater freshwater input) however tended to fall to much lower levels of 0.02 PSU in the inner zones as compared to those in the Sabaki estuary which dropped to 0.2 PSU. The maximum salinities were largely similar in the two estuaries as they ranged between 34 and 35.5 PSU in the region fronting the ocean. The low salinities were experienced during low tide when freshwater supply was dominant and high salinities were experienced during high tide when the influx of seawater

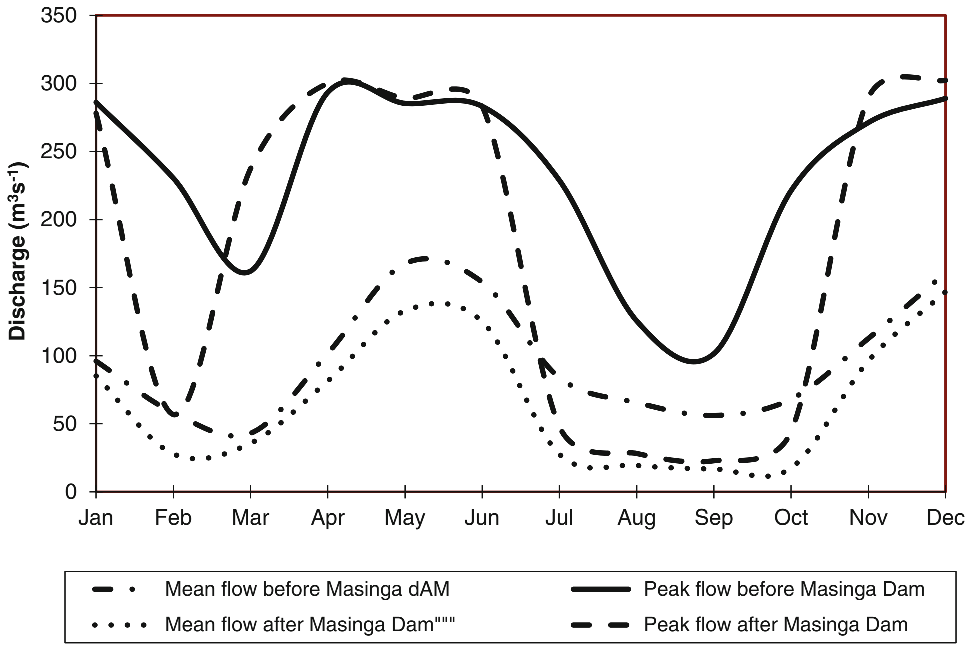

Fig. 8 The flow of the Tana at Garsen before (1942–1981) and after (1982–2000) construction of Masinga Dam

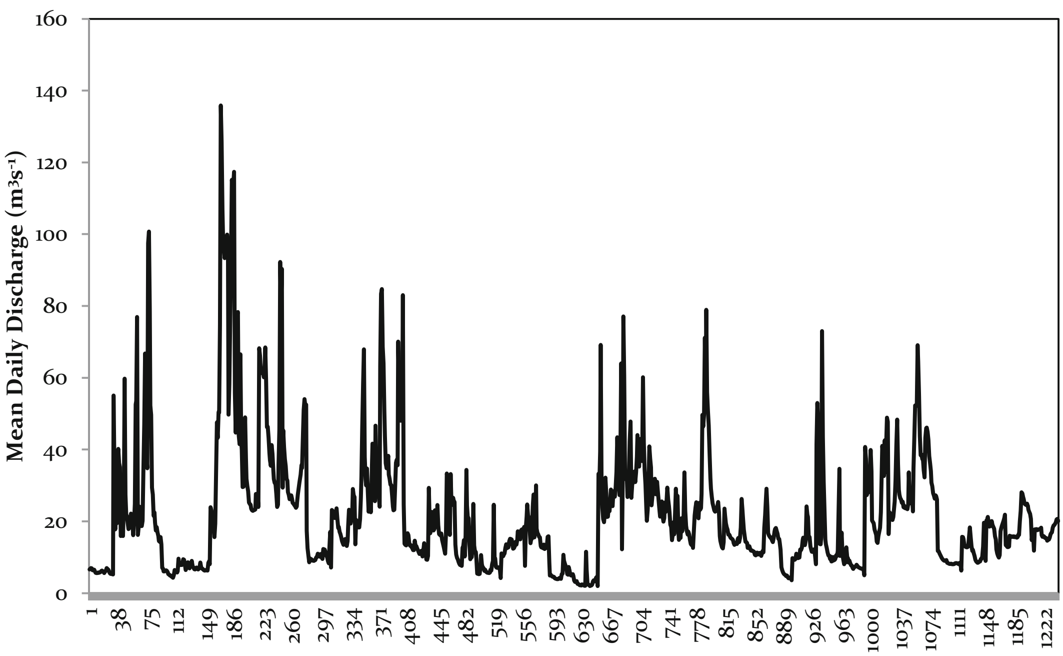

Fig. 9a The mean daily discharges of the Athi River at RGS 3DA02 in the period between 1952 and 1995. Horizontal axis shows the number of months

from the ocean was dominant. The data on salinity distribution indicated that there is a significant intrusion of seawater in both estuaries during high tide. However, saltwater intrusion was more prevalent in the much deeper Tana estuary where bottom water with salinity ranging between 5 and 10 PSU was found 10 km from the mouth of the estuary. In the Sabaki estuary, the saltwater intrusion was limited to only 2.5 km from the mouth of the estuary. In both estuaries, there is significant stratification in terms of both salinity and suspended sediments. Far much greater salinity stratification occurred in neap tide than in spring tide in the middle lower zones of the estuaries. During neap tide, the frontwater zones were partially well-mixed and during spring tide these zones were largely well-mixed. The middle zones of the two

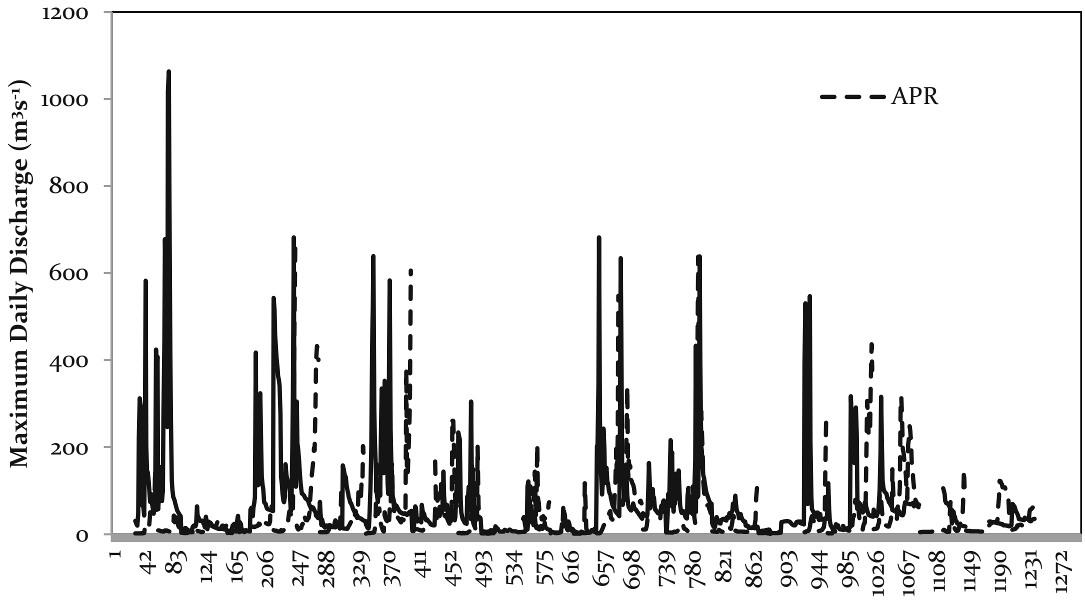

Fig. 9b The change in the maximum daily discharges experienced in April and June in the Athi River at RGS 3DA02 in the period between 1952 and 1995. Horizontal axis shows the number of months

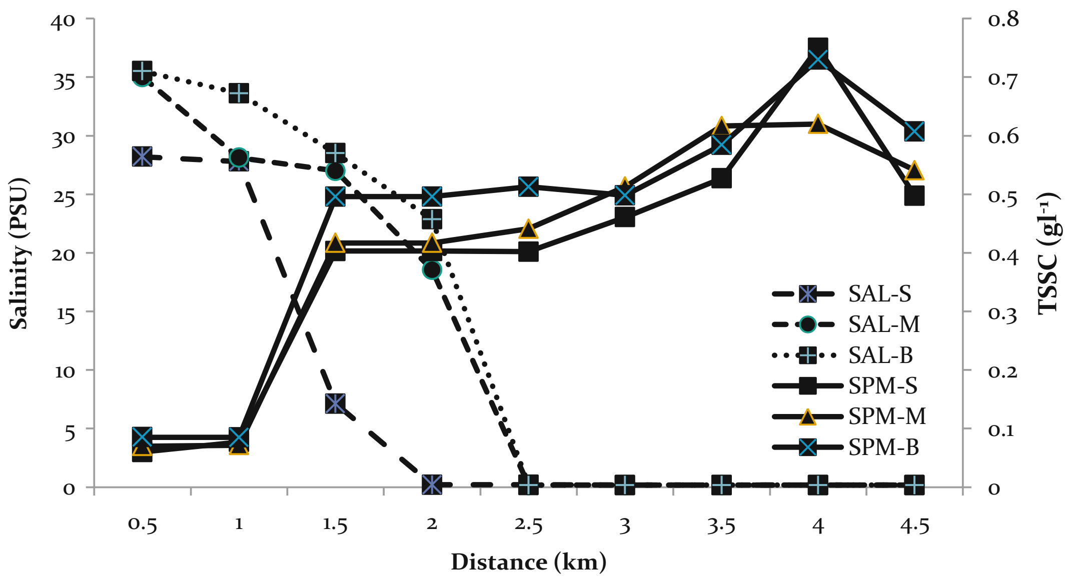

Fig. 10 The horizontal variations of depth-averaged salinity and TSSC in the Sabaki estuary at high tide during medium river discharge conditions. The 0.5 km is at the mouth of the estuary and 4.5 km is at the bridge.

estuaries tended to be stratified in neap tide but were partially well-mixed during spring tide. The riverine zones in both estuaries were largely well-mixed in both neap and spring tides. The differences in the degree of stratification were attributed to differences in the degree of mixing by the tidal currents in the down-estuary and mid-estuary regions and mixing by the river currents in the up-estuary regions.

TSSC and Turbidity Maximum Zones

In both estuaries, there was a region where the TSSC was relatively higher as compared to the up-estuary riverine zone and the down-estuary marine region (Figs. 10 and 11). This turbidity maximum zone (TMZ) was attributed to gravitational circulation at the seawater-freshwater interface and

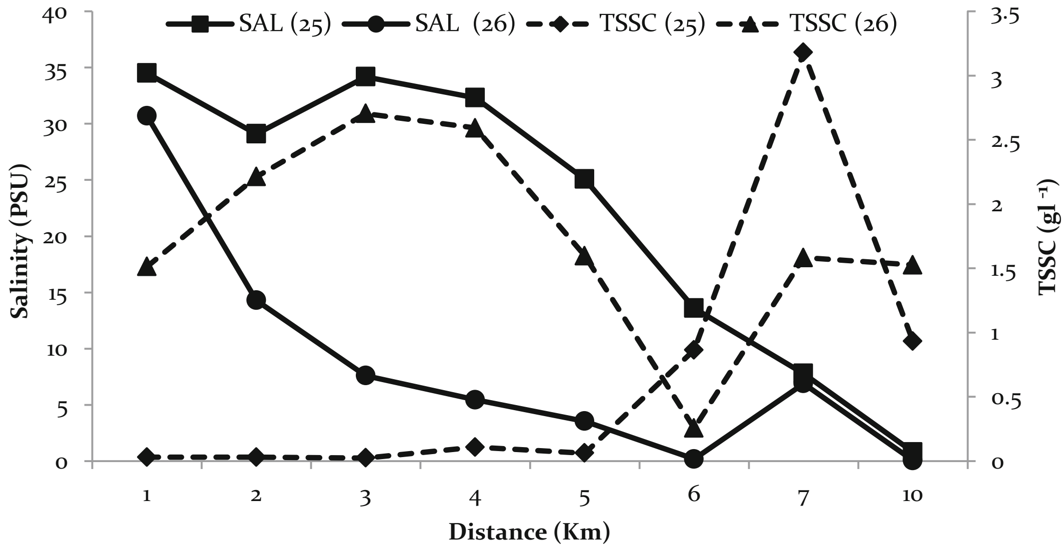

Fig. 11 The horizontal variations of depth-averaged salinity and TSSC at HW in the Tana Estuary at high tide (25) and low tide (26) during medium river discharge conditions. The 1 km is at the mouth of the estuary and 10 km is up-estuary at Ozi

sediment resuspension–deposition cycle driven by semi-diurnal tides (Kitheka et al. 2005). The levels of TSSC in the TMZ's of the two estuaries were however different. In the Tana estuary TSSC in the TMZ ranged from 1 and 3.5 g. l^{-1} while in the Sabaki estuary, TSSC in the TMZ were relatively higher as they ranged from 1.5 to 11 g.l^{-1} (see also Kitheka et al. 2003a, b). In both estuaries, the TMZ s were located at the lower middle zones of estuaries and there was a tendency for the TMZs to shift location depending on the phase of the tide. During ebb tide, the TMZ was advected down-estuary leading to trapping of cohesive clay sediments in the sheltered zones (with low current velocities) near the mouths of two estuaries. The cohesive clay sediments settling in these zones is responsible for creating mudflats that consist of sediment particles that were <38 μm. These are essentially clay mud. Within the main channels of the estuaries where velocities were relatively high, sediments were mainly fine sand with particle diameter of 125 μm.

Tidal Water Circulation, Sediment Import and Export in the Estuaries

The water circulation in the Sabaki and Tana estuaries is driven mainly by the tides and freshwater input and to a certain extent winds. The two estuaries experiences semi-diurnal tides that are largely asymmetrical with mean spring and neap tidal range of 3 m and 1.5 m, respectively along the coast. Within the Tana estuary, the region of tidal influence is 10 km while that in the Sabaki estuary is limited to 2.5 km. In both estuaries, the tidal currents exhibited a significant asymmetry with the ebb current velocities being relatively much stronger and of longer duration as compared to the flood tide currents in both neap and spring tides (cf. Kitheka et al. 2003a, b). In the Tana estuary, the flood periods were much shorter (4–5 hrs) as compared to ebb periods, which were longer (8–9 hrs). The peak ebb tide current velocities reached up to 0.87 m.s^{-1} in the Tana estuary while the flood ones reached up to 0.64 m.s^{-1}. The ebb tide currents were in general 30% greater than flood tide currents.

In the Sabaki estuary, the ebb tide current velocities attained a maximum of 0.70 m.s^{-1} in the middle region and lasted 6–7 hrs, while the flood tide current velocities attained a relatively lower peak of 0.30 m.s^{-1} and lasted 3 to 4 hours. The ebb tide currents in the Sabaki estuary were in general 50% greater than flood tide currents a manifestation of significant tidal current asymmetry. This asymmetry was attributed to channel geometry, presence of mangrove swamps, presence of intertidal areas and river freshwater input. Previous studies elsewhere have shown that tidal asymmetry is an important indicator of tidal pumping which is crucial in the export of sediments out of estuaries (cf. Wolanski et al. 1998a, b; Wolanski et al. 2001). The same is true in the case of the Tana and Sabaki estuaries. The variations in the width of the channels (w_{tide}) at station 3 as a function of tidal elevation (h_{tide}) were used to determine the channel cross-sectional area as a function of tidal elevation (A_{tide}). This was multiplied with the tide varying curent velocities (V_{tide}), to obtain the tidal discharges ($Q_{tide} = V_{tide}$ x A_{tide}) during flood and ebb periods (Tables 1 and 2). The tidal discharges (Q_{tide}) as a function of tidal elevation (h_{tide})

Table 1 Typical suspended sediment budget for the Tana Estuary

	Sources		Sinks	
	(kg.m^2.tidal cycle^{-1})	(tons.yr^{-1})	(kg.m^2.tidal cycle^{-1})	(tons.yr^{-1})
River supply	1.43	5.21×10^6		
Erosion (export)			0.075	0.55×10^6
Deposition (within the Estuary)			1.355	4.66×10^6
Total	**1.43**	**5.21×10^6**	**1.430**	**5.21×10^6**

Table 2 Typical suspended sediment budget for the Sabaki estuary

	Sources		Sinks	
	(kg.m^2.tidal cycle^{-1})	(tons.yr^{-1})	(kg.m^2.tidal cycle^{-1})	(tons.yr^{-1})
River supply	2.29	1.60×10^6		
Erosion (export)			0.22	0.19×10^6
Deposition (within the Estuary)			2.12	1.45×10^6
Total	**2.34**	**1.64×10^6**	**2.34**	**1.64×10^6**

Table 3 Mean spring tide water volumes and tidal prism at the Tana estuary

	Surface area (m^2)	Mean depth (m) high water in bracket	Volume at low tide (m^3)	Volume at high tide (m^3)
Estuary channel	3.82×10^6	5 (8)	19×10^6	30×10^6
Mangrove forest	23.2×10^6	0.3	0	7×10^6
Total	27.02×10^6		19×10^6	37×10^6
Tidal prism 18×10^6 (60%)				

were related to TSSC (C_{tide}) to determine the sediment fluxes ($Qsed_{tide} = Q_{tide}$ x C_{tide}) during ebb and flood periods. The ebb sediment fluxes were considered as the sediment export out of the estuary while the flood tide sediment fluxes were designated as sediment import into the estuaries. The difference between ebb and flood tide sediment fluxes was the net sediment export out of the estuary. When the flood sediment flux was greater than the ebb sediment flux, there was a net sediment import into the estuary causing deposition of sediments. In both estuaries, our results showed that although there is net export of sediments out of estuaries, there is accumulation of cohesive sediments in the two estuaries, as evidenced by presence of extensive mud flats in the inner zones of the two estuaries. Most of sediments exported out of the estuaries was silt and fine sand, with a much smaller proportion of clay sediments. This explains the presence of sandspits and sandbars at the mouth of the estuaries. The magnitude of the export and import of sediments is also influenced by the volume of water in the two estuaries that is exchanged per every tidal cycle (tidal prisms). Data shown in Tables 3 and 4 shows that in the Sabaki estuary almost 97% of the entire volume of water is exchanged after every tidal cycle, as compared to the Tana estuary where only about 60% is exchanged after every tidal cycle during spring tide.

Despite relatively high water exchange rates, there is net trapping of cohesive sediments in the mangrove forest wetlands. This was attributed to the reduction in the capacity of the ebb currents in the wetland to transport the cohesive sediments back to the main channel. The turbid flood tide water entering the mangrove wetlands has relatively high TSSC ranging between 0.2 and 2 gl^{-1} as compared to the TSSC in the ebb tide TSSC which was usually <0.02 gl^{-1}. The current velocities in the mangrove forest wetland were generally <0.10 ms^{-1} which is consistent to those reported in other similar systems (Wolanski et al. 2001; Kitheka et al. 2003a, b). These low current velocities were below the threshold for keeping the cohesive sediments in suspension resulting in rapid sedimentation in the wetland. This was evidenced by the presence of thick deposits of brown clay sediments in the mangrove forest. During periods of high river discharge and high TSSC, there was heavy sedimentation in the mangrove forest. There was evidence of smothering of mangrove pneumatophores in areas where sedimentation was high. These areas were characterized by stunted mangroves and in some areas dead mangroves were observed. The extent of smothering was more prevalent in the Sabaki estuary as compared to the Tana estuary which could be as a result of differences in the magnitude of sediment supply in the estuaries.

Table 4 Water volumes and tidal prism at the Sabaki estuary

	Surface area (m^2)	Mean depth (m) high water in bracket	Volume at low tide (m^3)	Volume at high tide (m^3)
Estuary channel	6.25×10^5	2 (5)	3.0×10^5	97.0×10^5
Mangrove forest	0.50×10^5	2.5	0	1.25×10^5
Total	6.75×10^6	2.25 (6)	3.0×10^6	98.25×10^5
Tidal prism 95×10^5 (97%)				

Discussions

Alterations of the Freshwater Input and Coastal State Changes in the Tana Delta

The supply of freshwater and terrigenous sediments into the Tana Delta estuaries has experienced significant changes in the recent past. This supply shows significant seasonal and inter-annual variability that can be attributed to a number of factors. These include landuse change, damming, water abstraction and natural climatic variability. While it is difficult to most conclusively attribute a particular factor to recent changes in river freshwater and sediment supply, this study has relied on available data to elucidate on the possible linkages. It is however important to note that rainfall variability in Central Kenya Highlands plays an important role in determining the magnitude of river freshwater discharge and subsequent, river sediment load. During periods of high magnitude rainfall in Central Kenya Highlands, there is usually a significant increase in freshwater supply and sediment load transported into the Tana Delta estuaries. The opposite is true during periods of low magnitude rainfall. This study including previous studies have shown that there is no significant change in the patterns of seasonal and inter-annual variability rainfall in Central Kenya Highlands (see Ovuka and Lindqvist 2000). Therefore, the decrease in sediment load in the Tana river, can be attributed to the anthropogenic influences. The discharge of the Tana river portrays rapid response to rainfall which is an indication of the low storage capacity of the reservoirs and degradation of the basin (cf. Brown et al. 1996; Pacini et al. 1998; Kauffman et al. 2007). The effects of abstraction of Tana river water for rural-urban water supply and irrigation schemes is equivalent to 11.5% of the mean flow of the Tana river with the total water abstraction of about 72×10^6 m^3 $month^{-1}$ for both rural-urban water supply and supply of water to irrigation schemes. The water abstraction has reduced the flow of the Tana by about 40 m^3s^{-1} which is equivalent to about 11% of the total flow of the river. This reduction in flow has much greater effects during periods of low flows in dry seasons. As compared to water abstraction, the construction of dams in the Upper Tana Basin has had major impacts in terms of modifying the flow of the Tana river by storing a substantial portion of the runoff in the reservoirs. This has significantly reduced the mean flows during both periods of high and low flows. The effect has been such that the current mean flows of the river are much lower during both wet and dry seasons. There has also been a reduction in the maximum monthly flows in periods of high and low flows. However, it is important to note that there does not seem to have been a major dampening of the river flow as evidenced by significant seasonal variability of the river (cf. Poff et al. 1997, 2007).

The reduction of the sediment load of the Tana river cannot be attributed to the reduction in the river discharge but to the presence of the Seven Folk Dams that traps most of the sediments emanating from the upper Tana basin (Maingi 1991; Schneider 2000). However, the river still has high sediment loads which is attributed to significant flow contributions from undammed tributaries of the river. It is thus expected the Mutonga/Grand Falls dam that is planned to be constructed downstream of Kiambere dam (GOK-JICA 1998) will trap most of the sediment load emanating from the presently undammed tributaries of the Tana river leading to major reduction in sediment supply to the Tana Delta.

Studies elsewhere have shown that alteration of flood flows and changes in low flows in the downstream sections of a river can induce major changes in the river morphology and ecosystem (cf. Basson 2013). Although the Tana river has deepened as a result of reduction in sediment load, the ecosystem impacts of the alteration of the Tana river system appeared moderate as compared to those observed in extensively dammed river systems. The moderate impacts of the Tana river is expected due to lack of significant dampening of the seasonal variability of the river due to small sizes of the reservoirs (cf. Poff et al. 2007). The major impacts observable in the Tana Delta include degradation of the riverine forest which now occurs in small patches along the river. Although the disappearance of the once extensive riverine forest can be attributed to reduction in maximum floods flows, it is also important to note that other contributors have a played a role. These include unregulated clearance of the forests for agriculture, settlement, timber and fuel wood. It is however important to emphasise that the reduction of the maximum flood flow following damming of the river must have also played a significant role in the degradation of the riverine forests, due to reduction in the influx of flood water into the far edges of the forest (see also Maingi and Marsh 2001, 2002).

Reservoirs are known to have a tendency of modifying the biogeochemical cycles of rivers by interrupting the flow of organic carbon and also by changing the nutrient and sediment load. This affects aquatic habitats by increasing habitat fragmentation (cf. Cushman 1985; Pringle et al. 2000; Friedl and Wuest 2002). The reduction in sediment load supply into the delta indicates a reduction in organic carbon, silicates as well as nutrients load since most of these are retained in the Upper Tana reservoirs (cf. Ittekkot et al. 2000). The retention of these materials in the reservoir has deprived the Tana Delta of much required nutrients and sediment materials and this has had significant effects on the downstream ecosystems, particularly on the productivity of marine ecosystems as demonstrated in changes in the productivity of Ungwana Bay fisheries (KMFRI 2002).

Field observations at mouth of the Tana Delta, showed significant erosion of the beaches. Although, coastal erosion can also be attributed to sea level rise, the reduction in sediment supply into the delta is considered important due to alteration of the sediment budget (cf. Ojany 1984; Kairu 1997). This is not unique in the Tana Delta since similar observations have been reported for the Zambezi Delta in Mozambique following construction of Cahora Bassa dam (see Beilfuss and dos Santos 2001; Basson 2013). The reduction in sediment supply to the delta also seems to be responsible for the deepening of the tidal channels fringing the mangrove forest ecosystem. These have deepened by 10 m as a result of increased channel-bed erosion due to reduced availability of sediments to be transported during ebb flows (Kitheka et al. 2003a, b). There was indication that this erosion of the channels is progressively extending into the mangrove forests ecosystem. There is also increased intrusion of seawater into the delta due to the reduction in river freshwater input and deepening of the main tidal channels. Seawater intrudes to more than 10 km into the delta at high tide. This has a potential of affecting aquatic ecosystem community structure, including contamination of freshwater supplies. Perhaps one of the major impacts of hydrologic alteration of the Tana river can be considered to be the reduction in flood-recession agriculture and loss of dry season pastures in the Tana Delta. This is due to the reduction in the frequency of influx of flood waters into the flood plains where flood-recession agriculture is practiced and where the dry-season pastures are usually found.

Alteration of Freshwater Input and Coastal State Changes in the Sabaki Estuary

There was evidence of progressive decline in the mean and maximum river discharges and an increase in sediment load of the Athi river. The Athi river runoff and sediment discharge exhibits significant seasonal and inter-annual variability and the long-term trend indicates a significant decline in freshwater supply to the Sabaki estuary. While the seasonal variability in river freshwater input can be attributed to the seasonal variations of rainfall, the long-term decline in runoff can only be attributed to the global climatic variability and landuse change. Although climatic variability is considered important, landuse change in the basin seems to have contributed to the decline. Previous studies have shown that there have been major landuse changes in the basin in the last 100–300 years (Fleitmann et al. (2007). These change have been attributed to the rapid increase in both human and livestock population in the catchment areas as well as massive destruction of forests, cultivation on steep slopes, overgrazing and burning of the fragile semi-arid lands vegetation (Edwards 1979; Denga et al. 2000). It is thus important to note that the effects of climatic variability or change are being intensified by landuse change. There is however a need for further research is this area. As compared to the Tana delta, the reduction in freshwater supply has not led to major intrusion of seawater into the Sabaki estuary. This is attributed to the shallow nature of the estuary with mean water depths of 3 m at HW which limits the propagation of the tidal wave into the estuary. In the much deeper Tana Estuary, the seawater intrudes almost 10 km inland.

The massive increase in sediment load of the Athi river (Ongwenyi 1983; Munyao 2001; Kitheka et al. 2003a, b) to levels above 5×10^6 ton.yr^{-1} can be attributed to land use change particularly the destruction of catchment areas through deforestation, cultivation, overgrazing and settlements. The impacts of climatic variability could be in terms of causing more aridity which combined with catchment degradation can trigger increased soil erosion. The effects of high sediment supply into the Sabaki estuary is evidenced by heavy accretion on the beaches and high turbidity. Beaches within the vicinity of the estuary have accreted by nearly 200 m (cf. Edwards 1979; Kairu 1997). This has also led to the siltation of Malindi harbour (Delft Hydraulics 1970; GOK-TARDA 1981; Ongwenyi 1983) and has also affected the coral reef ecosystem in the Malindi Marine National Park (Brakel 1984; Blom et al. 1985; van Katwijk et al. 1993; Obura 1995; McClanahan and Young 1996; Abuodha 1998).

Sediment Trapping in the Mangrove Wetlands

There is significant trapping of cohesive sediments in mangrove forest wetlands within both the Tana Delta and the Sabaki estuary, although this is more important in the later due to its large spatial extent.

This is due to the reduction in the capacity of the ebb tide currents in the mangrove forest wetland to transport the cohesive sediments back to the main channel. This reduction in sediment transport capacity is attributed to the low gradient and resistant to flow due to the presence of dense mangrove vegetation. The flow into the mangrove system during flood tide is generally driven by an increase in the horizontal pressure gradient occasioned by the increased water level in the main channel (cf. Wolanski et al. 2001). This results in rapid entry of turbid flood tide water with high TSSC (0.2–2 gl^{-1}) into the mangrove forest wetland. The flocculated cohesive sediments settle rapidly due to low flow velocities (<0.10 ms^{-1}) inside the wetland which are below the threshold for keeping sediments in suspension (Wolanski et al. 2001; Kitheka et al. 2003a, b). Between 50% and 80% of the cohesive sediments brought in by the flood tide are trapped inside the mangrove forest. This is important in the sense that it helps the wetland keep up with sea level rise.

The level of TSSC in the incoming flood tide water which is a function of river sediment supply has important implication on the magnitude of sedimentation in the mangrove forest wetland. High TSSC water during periods of high river sediment inputs results in more heavy sedimentation. This is detrimental to mangroves since pneumatophores are smothered leading to stunted growth and in some cases, complete death of mangroves. This was evident in the upper sections of the Sabaki estuary where only a small patch of old stunted mangroves were observed and also in the Tana Delta where there has been degradation of mangroves in some locations. The degradation of mangroves in the two estuaries cannot be attributed to the decline in freshwater input, but to the heavy sedimentation. This is also applicable in the case of the Tana delta because there is still a significant sediment supply into the delta (8×10^6 tons. yr^{-1}). Mangroves are able to extract freshwater from seawater and thus reduction in freshwater supply should not significantly stress them. Furthermore, there is still a significant input of freshwater in the two estuaries. It is however important to note mangroves in the Sabaki estuary are re-establishing in the lower sections of the estuary where mudflats have formed due to deposition of cohesive clay sediments at high tide. These mudflat areas coincides with the turbidity maximum zones (TMZ) where gravitational circulation and flocculation induces rapid settling of cohesive sediments in a null zone where the flood tide induced bottom current and the river induced surface current converges.

Seawater Intrusion

The saltwater intrusion in the Tana and Sabaki estuaries is attributed to the geometry of the estuary, the volume of freshwater input and the intensity of tidal incursion which is usually stronger during spring tide as compared to neap tide. Saltwater intrusion was more prevalent in the Tana estuary where water with salinity of up to 10 PSU was found 10 km inside the estuary. In the Sabaki estuary, the saltwater intrusion extended upto 2.5 km inside the estuary. The relatively greater intrusion of seawater in the Tana estuary was attributed to its relatively greater depth (up to 10 m) which allows rapid penetration of the tidal wave. However, in the case of the Sabaki estuary, relatively shallow depths (up to 6 m) induce greater frictional effect on the tidal wave. The differences are also due to the volume of freshwater entering the estuaries. The relatively greater volume of the Tana estuaries allows greater gravitational circulation with much greater saltwater-freshwater interface that results in stronger bottom flow of seawater. The reduction of freshwater and sediment inflow into the estuaries under the scenario of rising sea level would further deepen the tidal channels and cause more intrusion of seawater into the two estuaries. On the other hand, increased sediment supply as in the case of the Sabaki estuary will lead to increased accretion and shallow channels with less penetration of the tidal wave and hence reduced seawater intrusion.

Sediment Export and Trapping of Cohesive Sediments in the Estuaries

The import and export of sediments are important features of estuaries. There is net export of sediments in both the Tana and Sabaki estuaries. This is expected due to the strong tidal asymmetry that exists in both estuaries. The net export of sediments is partly evident by the sediment accretion on beaches found adjacent to the mouths of estuaries and also the presence of highly turbid sediment plume that moves along the coast depending on the direction of the prevailing monsoon winds (Brakel 1984; Kitheka 2013). The net export of sediments ensures that the estuaries do not fill up with the sediments. However, a substantial quantity of cohesive clay sediments is usually trapped within the estuaries due to processes associated with the TMZ. The magnitude of river freshwater input and sediment load supply has important implication on the magnitude of sediment export out of the estuaries. High river sediment supply results in more sediment export while reduction in river sediment supply leads to reduced sediment export. Where the river sediment supply is reduced significantly, the estuary compensates through increased channel erosion which deepens the tidal channels as in the case of the Tana delta estuaries.

At this juncture, it would be important to examine the anticipated impacts of major development projects that are planned for the Athi river and Tana river basins. These include large-scale irrigation projects such as Galana-Kulalu 1.7 million acre irrigation project and the planned construction of Thwake and Munyu dams in the upper Athi River Basin. In the Upper Tana Basin, a large multipurpose dam is planned to be constructed at Grand Falls to produce electricity and supply water for various purposes including irrigation. These projects are bound to have major impacts on the supply of freshwater and terrigenous sediments into the two estuaries. In case of the Sabaki, it is expected that there would be a major reduction in sediment supply which would lead to rapid erosion of the estuary. The damming of the river would decrease the freshwater input into the estuary and also reduce flood flows considerably. Although the dry season low flows would be augmented due to damming, the overall mean flow of the river would decrease due to reduction in the magnitude of flood flows. This will alter the geometric configuration of the estuary due accelerated channel deepening which will eventually lead to increased seawater intrusion. In case of the Tana delta estuaries, further damming of the Tana river will lead to major reduction of sediment supply to the delta and cut off completely the flood flows. The reduction in sediment supply will result in more accelerated erosion of the estuaries in the delta with potential to reduce the extent of mangroves. The loss of flood flows will spell doom to the remaining riverine forests and flood recession agriculture in the Tana Delta. The reduction in freshwater input will further increase saltwater intrusion in the Tana delta to more than 10 km inland impacting on agriculture, horticulture, pastures, freshwater supplies and fisheries for numerous villages in the Tana Delta. These impacts will severely limit the livelihoods of coastal communities leading to overexploitation and conflicts in the use of few available natural resources.

Conclusions

The Tana delta and the Sabaki estuaries are facing stresses that originate from their river basins. While the Tana delta has been impacted by the reduction of sediment supply due to upstream damming, the Sabaki estuary has been experiencing heavy sediment supply due to landuse change and climatic variability in the Athi River basin. Also, there is a decline in river runoff in both rivers as a result of damming, landuse change and climatic variability. The current impacts of the reduction in river freshwater input and sediment discharge in the Tana Delta are increased seawater intrusion, high turbidity, degradation of mangrove wetland and riverine forests (leading biodiversity loss) and beach erosion. In the Sabaki estuary, the major impacts of reduction in freshwater input and increase in sediment load are heavy accretion, degradation of coral reefs and mangroves and high turbidity.

The water circulation dynamics within the estuaries is a function of river freshwater input and tidal forcing and there is significant stratification of the water column in terms of salinity and TSSC. The tidal asymmetry characterized by much stronger ebb tide flows as compared to flood tide flows, results in the net export of sediments from the two estuaries. The cohesive sediments are trapped within the estuaries in intertidal areas and in the mangrove forest wetlands. The relatively coarse sand particles are exported out of the estuaries forming beaches, sandspits and sand bars at the mouth of the estuaries. Further changes in freshwater and sediment input in the two estuaries is expected to negatively impact the two estuaries with a high possibility of massive seawater intrusion, coastal erosion, degradation of mangroves and coral reef ecosystems, reduced nutrients levels, and decline in coastal fisheries. These will ultimately impact on the socio-economic livelihoods of coastal communities. There is therefore a need for an integrated management of the Tana and Athi River basins to maintain required balance of freshwater and sediment input into the estuaries. Large-scale development projects planned in the two basins have a huge potential of causing major alteration of the freshwater and sediment inputs into the two estuaries under the current scenario of high climatic variability and change. It is recommended that these projects be planned such that they take into consideration the potential impacts in the two estuaries. Where the anticipated impacts are major, these mega projects should be avoided altogether, otherwise they will spell doom to the future of the Tana and Sabaki estuaries.

References

Abuodha JOZ (1998) Geology, geomorphology, oceanography and meteorology of Malindi Bay. In: Hoorweg J (1998) (ed) Dunes, groundwater, mangroves and birdlife in Coastal Kenya, Coastal Ecology Series no 4, Moi University, pp 17-39

APHA (1992) Standard methods for the examination of water and wastewater-18th Edition. Greenber A, Clescer LS, Eaton AD (eds) American Public Health Association, American Water Works Association, Water Environment

Basson G (2013) Hydropower dams and fluvial morphological impacts – an African perspective. Department of Civil Engineering, University of Stellenbosch, South Africa, 18p

Beilfuss R, dos Santos D (2001) Patterns of hydrological change in the Zambezi Delta, Mozambique. Working paper No. 2, Program for the sustainable management of Cahora Bassa Dam and the Lower Zambezi Valley, 159p

Blom J, Hagen H van der, Hove E van, Katwijk M van, Loon R van, Meier R (1985) Decline of Malindi-Watamu Reef complex. Laboratory of Aquatic Ecology, Catholic University, 195p

Brakel WH (1984) Seasonal dynamics of suspended sediment plumes from the Tana and Sabaki Rivers, Kenya: analysis of Landsat Imagery. Remote Sens Environ 16:165–173

Brown T, Schneider H, Harper D (1996) Multi-scale estimates of erosion and sediment yields in the Upper Tana Basin, Kenya. In: Erosion and sediment yield: global and Regional perspectives. Proceedings of the Exeter Symposium, July 1996. IAHS Publ. no. 236, 1996

Cushman RM (1985) Review of ecological effects of rapidly varying flows downstream from hydroelectric facilities. N Am J Fish Manag 5:330–339

Delft Hydraulics (1970): Malindi Bay pollution II: Delft Hydraulic Laboratory, Report. R611

Denga FO, Ongwenyi GS, Kitheka JU (2000) Erosion and sedimentation problems in Arid and Semi-arid lands of Kenya. In: Gichuki FN, Mungai DN, Gachene CK, Thomas DB (eds) Land and water management in Kenya-towards sustainable land use-Proceedings of

the fourth national workshop, Kikuyu, 15–19 February 1993, pp 61–63
Edwards KA (1979) Regional contrasts in the rates of soil erosion and their significance to agricultural development in Kenya. In: Lal R, Greenland DJ (eds) Soil physical properties and crop production in the tropics. Wiley, London
FAO (1981) Offshore trawling survey in Kenya – project findings and recommendations. FAO, Rome
Fleitmann D, Dunbar RB, McCulloch M, Mudelsee M, Vuille M, McClanahan TR, Cole JE, Eggins S (2007) East African soil erosion recorded in a 300 year old coral colony from Kenya. Geophys Res Lett 34:5
Friedl G, Wuest A (2002) Disrupting biogeochemical cycles – consequences of damming. Aquat Sci 64(1):55–65
Gibb A (1959) Upper Tana catchment water resources survey-1958–1959. Report of the Kenya Government, Nairobi
GOK (1979) The National Water Master Plan. Stage I. Water Resources and Demands, Water Development Department/ Tippetts-Abett-McCarthy-Stratton, Nairobi, Kenya
GOK-TARDA (1981a) Athi River Basin: pre-investment study, preliminary Report. Agrar-UND Hdrotechnik Gmbh, Essen; Watermeyer, Legge, Piesold & Uhlmann, London Consulting Engineers, UK
GOK-TARDA (1981b) Athi River Basin: pre-investment study-Annexes 1-11. Agrar-UND Hdrotechnik Gmbh, Essen; Watermeyer, Legge, Piesold & Uhlmann, London Consulting Engineers, UK
GOK-TARDA (1981c) Athi River Basin: pre-investment study-Annexes 12-21. Agrar-UND Hdrotechnik Gmbh, Essen; Watermeyer, Legge, Piesold & Uhlmann, London Consulting Enginners, UK
GOK-JICA (1998) Feasibility study on the Mutonga/Grand Falls hydropower project. Final Report, vol III. Supporting Report (2), Environmental Impact Assessment, Nippon Koei Co. Ltd, Pasco International. Inc. Tokyo, Japan
GOK-TARDA (1982) Tana delta irrigation project-feasibility study. Vol III-Hydrology, river morphology and flood plain hydraulics. Haskoning-Royal Dutch Consulting Engineers and Architects, Netherlands and Mwenge International Associates Ltd, Nairobi, Kenya
ILACO (1971) Upper Tana catchment survey. ILACO, Arnhem
Ittekkot V, Humborg C, Schäfer P (2000) Hydrological alterations and marine biogeochemistry: a silicate issue? Bioscience 50 (9):776–782
Kairu KK (1997) Vulnerability of the Kenya shoreline to coastal instability. UNESCO-Kenya National Seminar on sustainable development, Mombasa, pp 13–25
Kauffman JH, Droogers P, Odada F, Macharia P, Gicheru P, Dijkshoorn JA (2007) Green and blue water resources and assessment of soil and water management scenarios using an integrated modelling framework. Green Water Credits Report 3. ISRIC – World Soil Information, Wageningen, 129p
Kitheka JU (2013) River sediment supply, sedimentation and transport of the highly turbid sediment plume in Malindi Bay. J Geogr Sci 23 (3):465–489
Kitheka JU, Nthenge P, Obiero M (2003a) Sabaki estuary sediment transport dynamics and resultant sediment fluxes in the monsoon-driven Malindi Bay, Kenya. KMFRI-LOICZ-START AfriCAT Project Technical Report, 53p
Kitheka JU, Nthenge P, Obiero M (2003b) Dynamics of sediment transport and exchange in the Tana estuary in Kenya. KMFRI-LOICZ-START AfriCAT Project Technical Report, 48p
Kitheka JU, Obiero M, Nthenge P (2005) River discharge, sediment transport and exchange in the Tana Estuary, Kenya. Estuar Coast Shelf Sci 63:455–468
KMFRI (2002) The current status of Ungwana Bay fishery. Report of the Kenya Marine and Fisheries Research Institute, Mombasa, Kenya
Linsley RK, Kohler MA, Paulhus JLH (1988) Hydrology for engineers. McGraw-Hill Book Company, New York, p 492
Maingi JK (1991) Sedimentation in Masinga Reservoir. Unpublished M.Sc. thesis, University of Nairobi, Kenya
Maingi JK, Marsh SE (2001) Assessment of environmental impacts of river basin development on the riverine forests of eastern Kenya using multi-temporal satellite data. Int J Remote Sens 22:2701–2729
Maingi JK, Marsh SE (2002) Quantifying hydrologic impacts following dam construction along the Tana River, Kenya. J Arid Environ 50:53–79
McClanaham TR, Young TP (1996) East African ecosystems and their conservation. Oxford University Press, New York, 100p
McClanahan TR, Obura D (1997) Sedimentation effects on shallow coral communities in Kenya. J Exp Mar Biol Ecol 209:103–122
Munyao TM (2001) Dispersion of the Sabaki sediments in the Indian Ocean. Unpublished D. Phil. thesis, School of Environmental Studies, Moi University, Kenya
Obura DO (1995) Environmental stress and life history strategies, a case study of corals and river sediment from Malindi, Kenya. PhD thesis, University of Miami, Florida, USA, pp 1-60
Ojany FF (1984) Kenya's shorelines. In: Schwarz ML, Bird ECF (eds) World shorelines. Downdan Hutchinson and Ross
Ojany FF, Ogendo RB (1986) Kenya: a study in physical and human geography. Longman, Nairobi, p 228
Ongwenyi GS (1983) Development of water resources. In: Ojany FF, Olson S (eds) The Kenyan Geographer-special issue on the Proceedings of the Nairobi workshop (4–8th August 1981)-Strategies for developing the resources of the semi-arid areas of Kenya, pp 36-47
Otieno FAO, Maingi SM (2000) Sedimentation problems of Masinga Reservoir. In: Gichuki FN, Mungai DN, Gachene CK, Thomas DB (eds) Land and water management in Kenya. Soil and Water Conservation Branch, Ministry of Agriculture and Rural Development, Nairobi
Ovuka M, Lindqvist S (2000) Rainfall variability in Murang'a district, Kenya: meteorological data and Farmers' perception. Geografiska Annaler Ser A, Phys Geogr 82(1):107–119
Pacini N, Harper D, Mavuti K (1998) Hydrological and ecological considerations in the management of a catchment controlled by a reservoir cascade: the Tana River, Kenya. In: Harper D, Brown T (eds) The sustainable management of tropical catchments. Wiley, Indianapolis, pp 239–258
Poff NL, Allan JD, Bain MB, Karr JR, Prestegaard KL, Richter BD, Sparks RE, Stromberg JC (1997) The natural flow regime: a paradigm for river conservation and restoration. BioScience 47:769–784
Poff NL, Olden JD, Merritt DM, Pepin DM (2007) Homogenization of regional river dynamics by dams and global biodiversity implications. PNAS 104(14):5732–5737
Pringle CM, Freeman MC, Freeman BJ (2000) Regional effects of hydrologic alterations on riverine macrobiota in the new world: tropical-temperate comparisons. Bioscience 50:807–823
Ruwa RK, Muthiga N, Esposito M, Zanre R, Mueni E, Muchiri M (2001) Report of the scientific information and conservation subcommittee-prawn fishery in Kenya. Marine Waters. Prawn Trawling Task Force, 8p
Schneider HM (2000) Sediment sources to Masinga Dam. In: Gichuki FN, Mungai DN, Gachene CK, Thomas DB (eds) Land and water management in Kenya-towards sustainable land use-Proceedings of the Fourth National Workshop, 15–19 February 1993, pp 47–54
UNEP (1998) East Africa atlas of coastal resources-Kenya. UNEP, Nairobi, 119

van Katwijk MM, Meier NF, van Loon R, van Hove EM, Giesen WBJT, van der Velde G, den Hartog C (1993) Sabaki River sediment load and coral stress—correlation between sediments and condition of the Malindi-Watamu reefs in Kenya (Indian-Ocean). Mar Biol 117:675–683

Wolanski E, Gibbs RJ, Spagnol S, King B, Brunskill G (1998a) Inorganic sediment budget in the mangrove-fringed Fly River Delta, Papua New Guinea. Mangrove Salt Marshes 2:85–98

Wolanski E, Nhan NH, Spagnol S (1998b) Sediment dynamics during low flow conditions in the Mekong River estuary, Vietnam. J Coast Res 14(2):472–482

Wolanski E, Mazda Y, Furukawa K, Ridd P, Kitheka JU, Spagnol S, Stieglitz T (2001) Water-circulation in mangroves and its implications for biodiversity. In: Wolanski E (ed) Oceanographic processes of Coral Reefs: physical and biological links in the Great Barrier Reefs. CRC Press, New York, 356p

Integrated Water Resources Management in a Changing Climate: The Implication of Anthropogenic Activities on the Tana and Athi/Sabaki Rivers Water System for Sustainable Development

Saeed Mwaguni, Renison K. Ruwa, Jacob Odhiambo-Ochiewo, and Melckzedec Osore

Abstract

This paper highlights the effects of damming and farming activities on the Tana and Athi/Sabaki Rivers' water system. The two activities are responsible for freshwater shortages, pollution, and habitat and community modification downstream. Damming has resulted in modification of the stream flow, and in changes in the water table; while nutrient loading and sedimentation from irrigated agriculture, human settlements and industrial activities upstream are responsible for the pollution effect. The resulting effects were evaluated using the Global International Waters Assessment scoping methodology backed with hindsight experiences of the team members in both environmental and socio-economic issues. The individual scores when averaged led to the following results: 1) the effect of damming the river system was scored at 3 in a scale of 1–3, signifying a severe impact as attested by the freshwater shortages experienced downstream; 2) the effect of nutrient loading and sedimentation, using the same scale, was score at 2, a score that exhibited a moderate pollution problem. The degree of impact from nutrient and sediment loading varied with seasons. It was localized and more pronounced in hot spots in the dry season; while in the wet period, it occurred throughout the system as flooding occurred; 3) reduced stream flow and pollution affected ecosystem functions, resulting in habitat and community modification, 4) the impact of global climate change was difficult to score at the river basin level. However, deforestation resulting from slash and burn agriculture and harvesting of wood to meet domestic energy needs were practiced in a large scale, reducing the action of forests as carbon sinks, while promoting the emission of carbon-dioxide, which contribute to global climate change. This study concluded that freshwater shortages and pollution were issues of major concern and they were causing socio-economic conflicts. To address the concerns, a proactive approach borrowing from the tenets of Integrated Water Resources Management, Ecosystem Based Management and Integrated Coastal Zone Planning and Management, need be adopted as the vehicles that would promote sustainable development in the river basin.

Keywords

Damming • Irrigated farming • Pollution • Habitat and community modification • Global change • Freshwater shortages

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Introduction

This paper highlights the effects of human activities on the Tana and Athi/Sabaki Rivers water system –specifically analyzing the effects resulting from damming and farming activities along the river banks. The issues of concern discussed in this paper are 1) *freshwater shortages*, resulting from stream flow modification due to damming of the river upstream to produce hydro-electricity, and 2) *pollution* of the water system due to nutrient loading and sedimentation from irrigated agriculture and discharges of effluent from both industrial establishments and domestic sources. The concerns arising from freshwater shortages and pollution of the water system, including their socio-economic effects were scoped for score and impact, taking into consideration the influences rapid economic growth, population increase and global change, which allowed for the prediction of the possible future outlook for the two concerns and the interventions needed in order to promote sustainable use of the water system.

Geology and Climatic Conditions

The Tana River traverses the Tana County, an area that is geologically composed of the Mesozoic peri-oceanic troughs that are block shaped mono-clinical structures made of thick folded and faulted Meso-Cenozoic strata (Carbone and Accordi 2000), that comprise two structural layers –the upper layer sedimentary rocks, and the lower metamorphic rocks. In addition, Jurassic deposits, mainly composed of sandstones and carbonates are found. There are also Karoo deposits of clay and sand, and Paleogene deposits of up to more than 3,000m thickness.

It is through this terrain that the Tana River, later joined by the perennial rivers Athi/Sabaki in its upper catchment, travel from its mouth in the flanks of Mount Kenya in the Tharaka County in the North, flowing east to Mbalambala and Garissa –forming the Lower Tana River Basin, then continues southwards to Kipini (located half way between Malindi and Lamu) to the East, before discharging its contents into the Indian Ocean. From its source to discharge, the river stretches a distance of 1012 km, (Saha 1982).

The profile of the Tana-Athi/Sabaki river system is given in Table 1, while the extent of its drainage basin, which covers an area of approximately 120,000 km^2 (Ogwenyi et al. 1993a), is shown in Fig. 1.

As is characteristic for the rest of Kenya, rainfall experienced in the Tana River Basin is bimodal with the major rainfall peak, occurring during the South East Monsoon (SEM) between April and June. The minor rainfall period peak occurs between October and December, a season where the North East Monsoons (NEM) predominate.

The rainfall distribution in the Tana River Basin is given in Fig. 2, while the characteristic conditions of climate and the Agro-ecological zones found within its confines are presented in Tables 2 and 3.

The Tana River is regularly replenished by a number of tributaries that have their headwaters on Mount Kenya. However, from Mbalambala to Kipini, the river flows through a semi-arid region, with characteristic hot and dry weather conditions where it is joined by the perennial rivers Talu, Hirimani, Galole and Tiva. Being dry for most of the year, their replenishment contribution to the River Tana is insufficient to counter the continuous loss of water through evaporation in the prevailing hot and dry weather conditions. Rainfall comes in two rain seasons. It decreases both towards the interior and northwards, resulting in increased aridity index. The rain increases southwards with a recording of 350 mm per annum registered at Garissa, 470 mm per annum at Hola, and up to over 1000 mm per annum at Kipini. As a result the lower Tana Belt experiences semi-arid to arid tropical climate with a bimodal pattern of rainfall, influenced by the monsoon winds (McClanahan and Young 1996; Carbone and Accordi 2000). The rainfall is generally low and erratic and with a mean average ranges of between 300mm–500mm.

Before entering the Indian Ocean, this river system branches off to a complex system of tidal creeks, flood plains, coastal lakes and mangrove swamps –collectively referred to as the Tana Delta. The Delta covers an area of approximately 1,300 km^2 behind a 50m high sand dune system that protects it from the wave action of the open sea in the Ugwana Bay (UNEP 1998). The mangrove cover supported by the Tana estuarine environment, is estimated to span an area of 39,825 Ha while that of the Athi//Sabaki is 13,155 Ha (Doute et al. 1981; UNEP 1998). The Tana Delta is characterized by wetlands of great potential for agriculture, seasonal grazing and attraction for tourism.

The river discharges large volumes of freshwater and sediment into the Indian Ocean annually. The distribution and median monthly discharges for the period 1941–1996 are given in the Fig. 2. The peaks in May and November

Table 1 Profile of the Tana-Athi/Sabaki River system

Drainage Basin	Size (km^2)	Mean annual run-off (m^3 $\times$ 10^6)	Length (km)	Source
Tana River	132,090	4,700	708	Mt Kenya & Aberdare ranges
Athi-Sabaki	69,930	1,295	(547)	

Source: Hartziolos et al. (1994), Hirji et al. (1996), UNEP (2001), Vanden Bossche and Bernacsek (1990), and FAO (2001)

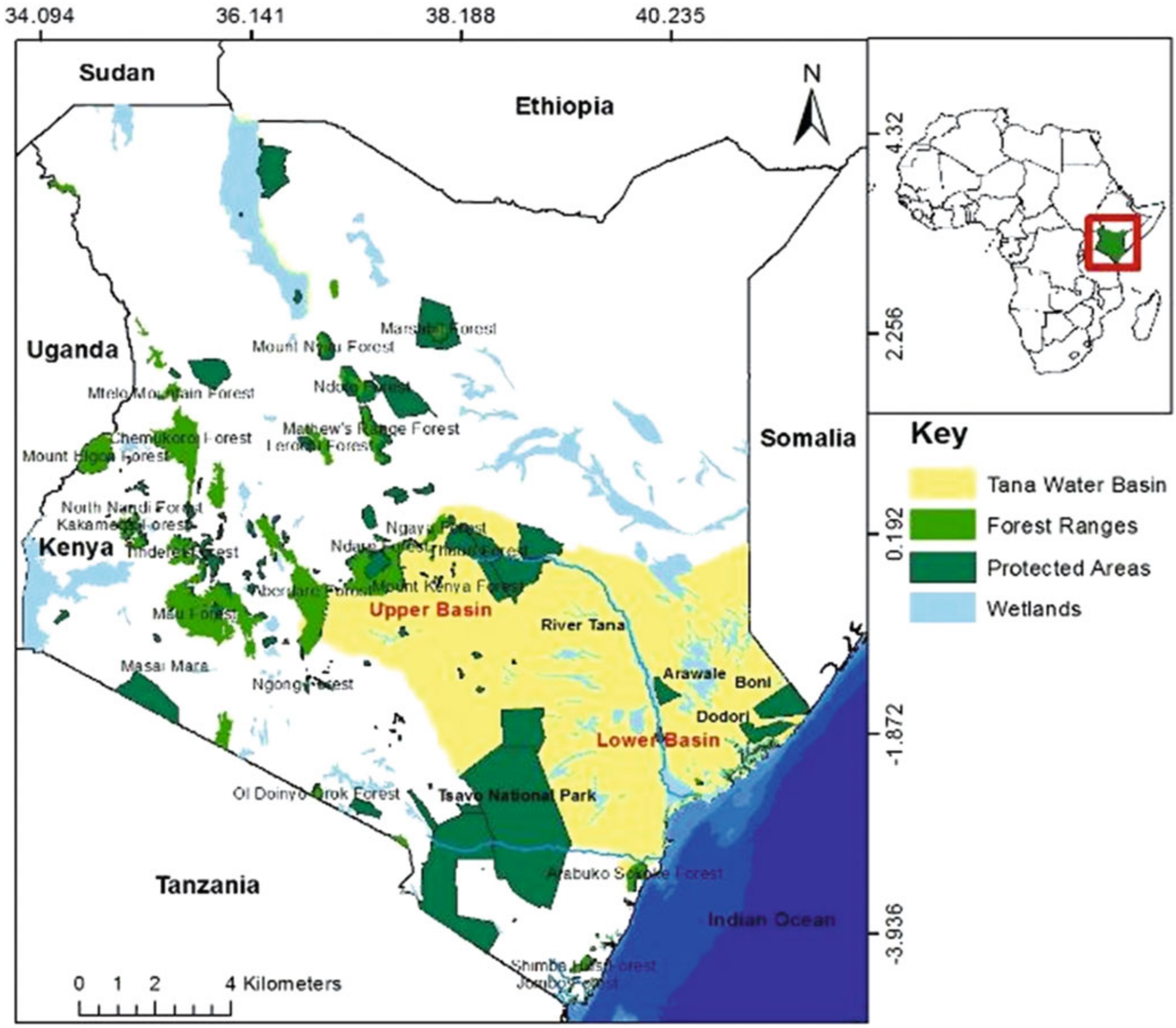


Fig. 1 Map of Kenya showing the Tana River, its Major Tributaries and Drainage Basin

correspond to the peak rainfall season during the long rains and the short rains.

The Tana and Athi/Sabaki Rivers system define the hydrology of the coastal area of the Tana River County. The rivers' drainage systems link the hinterland with the Indian Ocean and have a catchment that extends from the coastal hinterland into Mount Kenya and the Aberdare Ranges. The catchment ecosystem structure and function of the two rivers is dependent on rainfall and drainage process of water discharge, sediment transport, flooding, inundation and alluvial deposition (Brinson 1990), Fig. 3.

The rivers' water discharge therefore follows the bimodal precipitation pattern with the extent of their watershed environment defined by the dynamics of seasonality –where, as the rivers flow into the hot and arid areas, the tributaries decrease in volume of the water flow, and in some cases, even disappear completely due to evaporation and soil infiltration. Owing to the large volume of water from their sources, the Tana, Athi/Sabaki rivers system is generally perennial. As the rivers system recharge surrounding areas through floods and/or groundwater, an environment suitable for the development of unique ecosystems is provided. Against this background, the water-shed of the Tana-Athi/Sabaki rivers system has a flood plain of evergreen forest with riverine endemic *Populus ilicifolia* (Medley and Hughes 1996), which explains the existence of several areas of forest, woodland and grassland in the lower Tana.

The riverine forest starts from Mbalambala to Kipini. It extends 0.5–3.0 km on either side of the river. This forest is a remnant of the rainforest belt extending from the Congo Basin

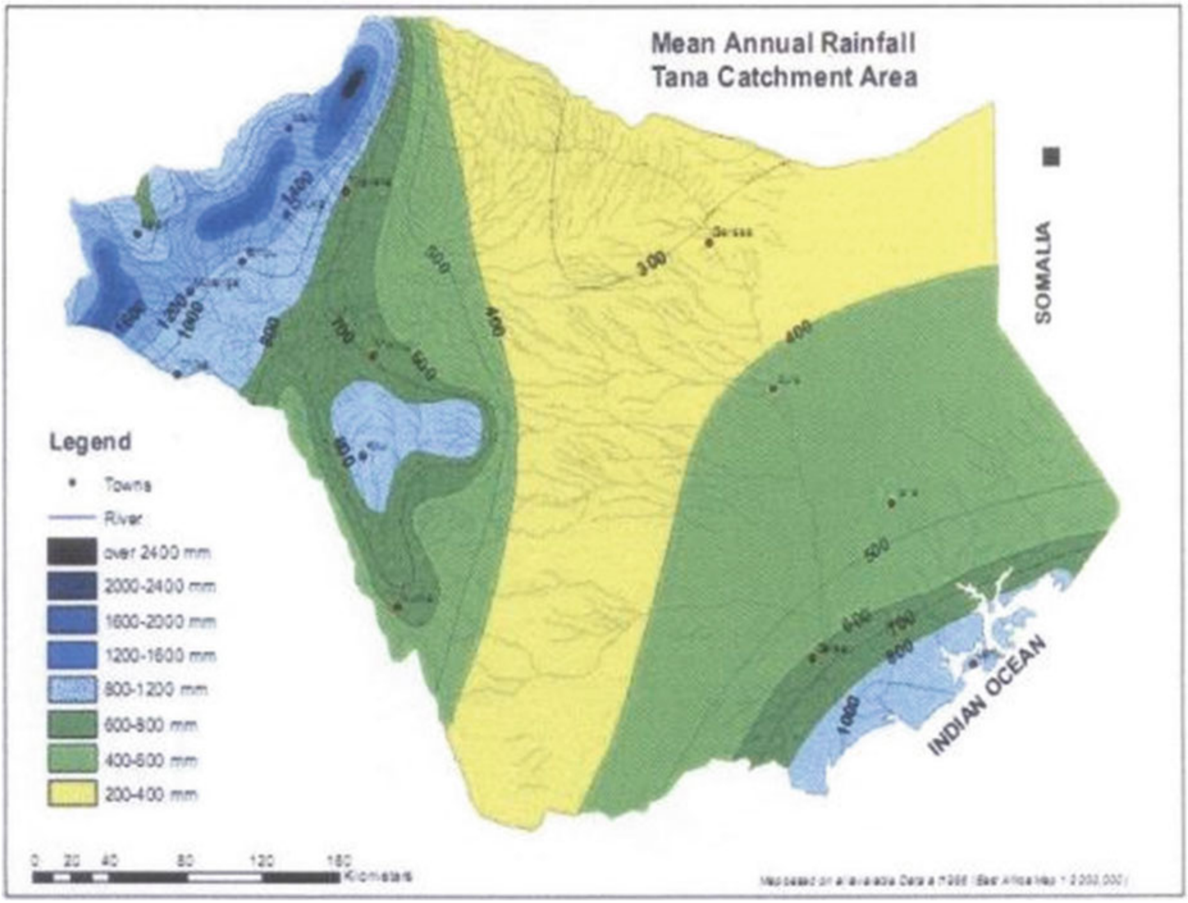


Fig. 2 Mean rainfall distribution in the Tana River Basin (Source: Kasper Lange 2014)

Table 2 Climatic conditions in the Tana River Basin

Zone	Ratio RF/Ep. %	Class	Rainfall mm	Potential evaporation, mm	Vegetation	Potential plant growth
I	>80	Humid	1100–2700	1200–2000	Moist forest	Very high
II	65–90	Sub-humid	1000–1600	1300–2100	Moist forest, dry forest	High
III	50–65	Semi-humid	800–1400	1450–2200	Dry forest, moist land	High to medium
IV	40–50	Semi-humid to Semi-dry	600–1100	1550–2200	Dry woodland, bush land	Medium
V	25–40	Semi-arid	450–900	1650–2300	Bush land	Medium to low
VI	15–25	Arid	300–550	1900–2400	Bush land, scrub land	Low
VII	<15	Very arid	150–350	2100–2500	Desert Scrub	Very low

Source: WARMA 2009: in Tana Delta Strategic ESA, GOK 2012
Key *RF* Rainfall, *Ep*. Evaporation

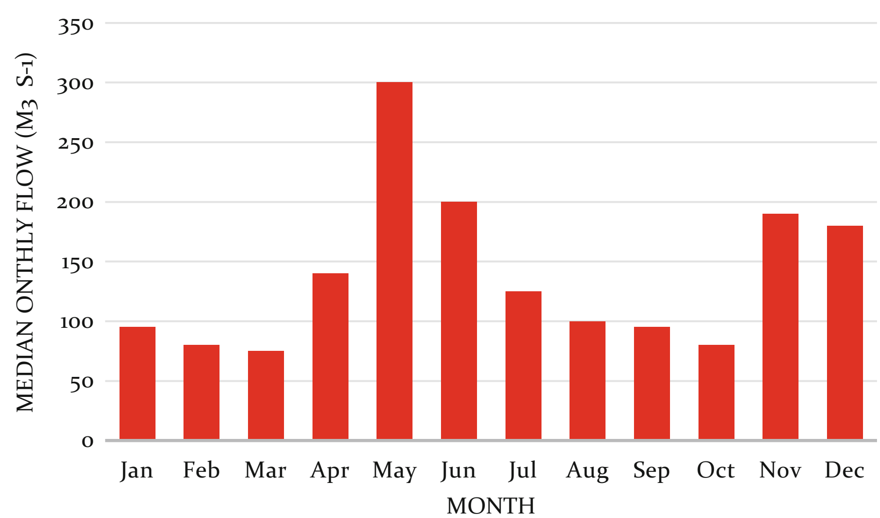
to the Eastern Coast of Africa (Livingstone 1975, adapted from Medley 1990). The extent of the forest in the river basin is determined by the depth of the water table, as it drops off significantly from the edge of the river (Marsh 1978; Hughes 1985). Away from the flood plain, the land is characterized by bush land, common in dry regions; scattered and dominated by thorny bushes. The riverine forest is unique because of its biodiversity. The Tana riverine forest houses the Tana River Red Colobus and the Tana River Mangabey –two endemic sub-species of the primate, giving the area a high conservation value. The composition, structure and the dynamics of the forests are influenced by the hydrological characteristics of the river system (Marsh 1978; Hughes 1988). The Tana riverine forest has been declining due to lack of regeneration, attributed to decreased peak flows (Marsh 1976; Hughes 1985; Medley 1990).

Table 3 The agro-ecological zones in the Tana Basin

AEZ	Zone	Characteristics and types of crops and livestock reared
Tropical Alpine	TA 0, I, II	No land use, National Park, Limited grazing
Upper Highland	UH 0, 1, 2, 3, 4	Very wet, important as catchment areas, bamboo thickets and forest reserves pyrethrum barley, oats, peas, radish, rapeseed, Irish potatoes, kohlrabi, celery, leeks, wheat, plums, pears, apples, pasture and forage, white clover, rye grass, grade dairy & beef cattle, merino sheep, 0.8–2 ha/LU
Lower Highland	LH 0, 1, 2, 3, 4, 5	Forests, long cropping season, good crop yield for lettuces, tea, kales, peas, cabbage, carrots, Irish potatoes, pyrethrum, hybrid maize, leeks. Napier grass, clover and grade dairy cows, black wattle, kikuyu grass, apples, pears, plums, avocadoes, wheat, barley, Rhodes grass, white clover, 0.5–6 ha/LU
Upper Midland Zones	UM 1, 2, 3, 4, 5, 6	Coffee-tea and sunflower zone. Crops grown includes; tea, coffee, Irish potatoes, tomatoes, cabbages, maize, bananas, beans, sun flower, tobacco, sweet potatoes, citrus, mangoes, pawpaw, avocadoes, passion fruits, cassava, yams, sugarcane, miraa, taro, fodder crops/legumes, 0.5–5 ha/LU
Lower Midland Zones	LM 3, 4, 5, 6	Cotton-livestock-millet zone. Crops grown include; cotton, dry land maize, sorghum, dolichos, ground nuts, tobacco, sweet potatoes, cassava, pineapples, cowpeas, chick peas, soya bean, pumpkins, green grams, mangoes, onions, castor, bulrush/proso/finger millet, pigeon peas, macadamia nuts, sisal, *leucaena leucocephala,* bana grass, zebra grass, sirato, 0.8–4.5 ha/LU
Lowland/Inner lowland	L 5, 6	Livestock millet zone-short cropping season. Crops grown include; bulrush/proso millet green grams, gourds, dry land composite maize, dwarf sorghum, sisal, castor, jojoba, cowpeas, bambara nuts, 2.5–5.5 ha/LU, game ranching with Oryx, gazelle and gerenuk
Coastal lowland	CL 3, 4, 5, 6	Coconut-cassava zone: composite maize, white sorghum, finger millet, bulrush millet, foxtail millet, green grams, sunflower, ground nuts, lima beans, cucumber, garlic, water melons, bixa, pepper, chilies, sisal, cashew nuts, pineapples, bananas, lemons, ground nuts, cow peas, cassava okra, Indian avocadoes, pawpaws, bocoboco, jatropha, rice, sugarcane, castor, sweetpotatoes, dolichos, gourds, eggplant, simsim, limes, oranges, mangoes, *leucaena leucocephala,* bana grass, fodder legumes, maram beans, buffel grass, siratro, centro, acacia, albida, 0.2–5 ha/LU, sclerophytic evergreen (infested with tsetse fly) bush land, seasonally flooded grasslands

Source: (WRMA 2009) in Tana Delta Strategic ESA, 2012

Fig. 3 Distribution of median m discharges for the River Tana 1941–1996: (Maingi and Marsh 2002)



Despite the large size of the Tana River County $-38{,}782\ km^2$, only 240,075 people live there. This gives a population density of only 6.2 people per km^2. Harsh environmental conditions have made the County sparsely populated. The Pokomo, the Orma and the Wardey are the three major communities living here. 72% of the population lives below the poverty line, and the poverty index is 62% (GoK 2009). Weather conditions determine the socio-economic activities taking place in the county and population mobility in the area is attributed mainly to nomadic pastoralism in response to drought and flood episodic events, trade movements and rural to urban migration of the population in search of employment opportunities.

The population of the Tana County, comprising the Tana River and Tana Delta districts grew by 40.9% between 1989–1999. Its growth in the period 1999 to 2009 was 32.7, indicating a decline of 8.2%. The decline in the population over the period may be attributed to several causes

Table 4 Population census figures Tana River District 1989–2009

1989	1999	2009	
District Pop: 128,426	District Pop: 180,901	Tana River: 143,411	Tana Delta: 96,664
Divisions:	Divisions	Divisions:	Divisions:
Galole 37,712	Galole 34,948	Galole 44,981	Garsen 43,346
Garsen 47,206	Garsen 51,592	Bura 31,786	Tarassa 33,741
Bura 25,035	Bura 28,848	Madogo 27,464	Kipini 19,577
Madogo 16,473	Madogo 21,731	Bangale 23,295	
	Bangale 14,853	Wenje 15,885	
	Wenje 12, 868		
	Kipini 16,243		

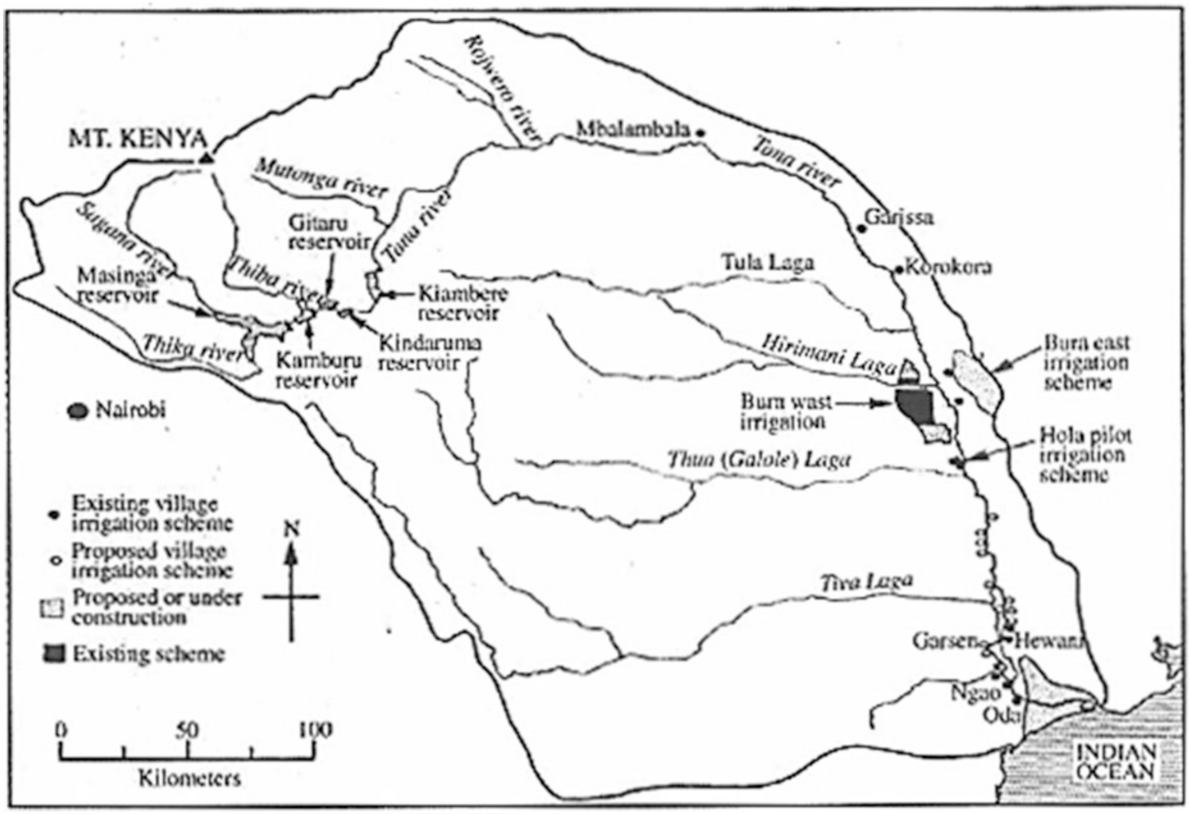


Fig. 4 Location of dams and irrigation schemes on the River Tana (Source: Maingi and Marsh 2002; Vanden Bossche and Bernacsek 1990; Maingi and Marsh 2001)

including insecurity, or people escaping the harsh environmental conditions to seek solace in other areas as shown in the information in Table 4.

The socio-economic wellbeing of the people residing in the Tana River Basin is dictated by availability of, and patterns of use of natural resources for a majority of people depend on it for support of their livelihoods. As a result, the waters of the Tana-Athi/Sabaki system are utilized for both consumptive and non-consumptive purposes. The consumptive uses include water abstraction for irrigated agriculture, for potable and domestic needs, for commercial uses and for industrial development purposes. The non-consumptive uses include transportation, exploitation for its aesthetic beauty in tourism and, damming for hydroelectric power production. The economic sectors supported by the river system include agriculture, fisheries, industry, manufacturing and services, tourism and transportation. The largest and most important uses of this water system remains to be commercial irrigation and hydro-electric power generation, with 5 dams so far having been built in the upper basin for the two purposes. There are also several irrigation schemes that support subsistence farming, Fig. 4.

Table 5 Profile of the dams in the Tana River water system

River	Reservoir	Volume (10^6 m^3)	Area (km^2)	Fishery (ty^{-1})
Tana	Masinga (1981)	1,560	120	480
	Kamburu (1975)	156	15	–
	Gitaru (1978)	20	3.1	–
	Kindaruma (1968)	16	2.4	–
	Kiambere (1988)	535	25	50

Kindaruma, Kamburu and Gitaru were the first to be built following each other. These three dams had small reservoirs, leaving the Tana River, essentially unregulated (Hughes 1985).

Masinga dam –the largest so far, stored more water, and had a capacity to generate 40 MW of electrical power. It is sited upstream the other dams to serve as the regulating reservoir, besides –improving electrical power generation during the dry season, increase irrigation potential in the lower Tana basin; thereby increasing the utilization of dry season flows in the upper parts of the river. The Kiambere Dam, which took four years to complete had an installed hydro-power generation capacity standing at 140MW. The profile of the dams in this water system, which is found in the river Tana alone, is as shown in Table. 5.

The agricultural practice within the river basin is dominated by peasant farming, mostly for subsistence crop production and animal husbandry. These activities form the mainstay of the economy of the rural people. Pastoralism is practiced by the Orma and Wardey while the Pokomo engage in crop production. The farming practice is rotational with slash and burn method being applied for land clearance. Due to the large population in the Tana Basin, agricultural land is scarce, resulting in short fallow periods and the parcels of land to cultivated more frequently. In the long-run, this has led to poor yields due to loss of soil fertility from the frequent use of the land. To counter the scarcity of parcels of land for farming, deforestation has been taking place to increase the size of land available for agriculture. This has in turn led to desertification in the upper stretches of land that is already poorly vegetated, due to the semi-arid and arid environments, making farming in the new acquired lands difficult to sustain.

Apart from the subsistence farming, there is irrigated agriculture, which is practiced mainly for commercial crop production. Though the potential for irrigated agriculture is not established, it is t known that the full potential is yet to be exploited. Since large scale irrigated farming and livestock production activities use large amounts of fertilizers and pesticides to enhance productivity, their effects on the water quality in the river system cannot be ignored. Needless to say, however, despite the efforts in crop and animal production, the lower Tana catchment is still a food deficient area with incidences of undernourishment reported (Source: GoK 2009).

High sediment loading into the river system, partly attributable to poor land use practices in the upper catchments, is responsible for the high discharge rate of sediment into the ocean, threatening the sustainability of the marine environment, its resources and livelihoods. Similarly, the high concentration of silt –pouring in, undermines the aesthetic beauty of the marine environment, rendering it unattractive for recreational uses, including tourism in the areas where large sediment and silt discharges occur.

The damming of a river, for whatever purpose, results in the formation of a lake behind the dam, and the consequent alteration of the natural flow of that river downstream. Dams on a river therefore cause impacts to both upstream and downstream environments. Freshwater unavailability in time and space is a major constraint to economic development. Its availability, or otherwise within the lower Tana Basin has been a source of conflict, pitying pastoralists against farming communities. The long periods of dry weather and frequent flooding in the river basins have thus posed challenges in terms of general security and food insecurity. It is against this background that the effects of damming and irrigated farming on the Tana-Athi/Sabaki River water system were studied in this research. The investigations determined the impacts arising from freshwater shortages downstream due to the reduced river flow regime and from pollution on the water system due to anthropogenic activities; establishing the level of these concerns –through scoring and ranking, as baseline for predicting the future situation in a global changing environment, where effects are exacerbated by the impacts of climate change; and to draw conclusions from the situation for recommendations that would lead to efficient utilization of the water system in order to realize sustainable development goals.

Materials and Methods

The study was accomplished through the Global International Waters Assessment methodology (UNEP 2006) that provided a holistic and comparable assessment of the world's trans-boundary aquatic resources. The assessment incorporated both environmental and socio-economic factors and recognized the inextricable links between freshwater and the marine environment. As the method dictated, the work was achieved through an interactive process, guided by a Methods Task Team, comprising of experts on water science, environment and socio-economics backgrounds.

Though the GIWA assessment focused on the impacts of five pre-defined concerns of freshwater shortages, pollution, habitat and community modification, overfishing and other threats to aquatic living resources, and global change, including evaluation of the impacts arising from the said concerns. This paper concentrated on pollution and freshwater shortages, including their future look as the major concerns based on their scores, which were derived as the summation of contributions from both environmental and socio-economic inputs on a scale of 1–3 as determined through expert hindsight/knowledge, integrating the environmental and socio-economic data, to determine the severity of the impacts from each of the two concerns evaluated, including their constituent issues. Two participatory meetings were held with experts in the fields being studied. During the workshops, preliminary analyses based on collective knowledge and experience were performed. The results were then substantiated with the best available information to feed in the report.

The GIWA methodology is divided into four logical steps: 1) scaling to describe the geographical extent of the study area; 2) scoping to identify and prioritize problems based on the magnitude of their impact on the environment and human communities; 3) Causal chain analysis to define root causes of the problems; and 4) Policy option analysis to assess the various policy options that address the root causes of the problems in order to reverse negative trends in the aquatic environment. The latter two were pursued only the extent where it provided the information necessary for this study.

Scoping assessed the severity of environmental and socio-economic impacts caused by each of the two concerns and their constituent issues in order to prioritize the most important issues for remedial action. The issues were evaluated using a standard scoring system involving a four point scale where, 0 = no impact recorded; 1 = slight impact; 2 = moderate impact; and 3 = severe impact. Once each issue had been scored, it was weighted according to the relative contribution it made to the overall environmental impact of concern and a weighted average score for each of the concerns calculated.

The socio-economic impacts were also assessed for each concern not issue. These impacts were grouped into economic impacts, health impacts and other social and community impacts. For each of these categories, the size, degree and frequency of the impact was evaluated and a weighted average score calculated for the overall socio-economic impact of the concern. To ensure applicability and sustainability of the mitigation options for the concerns, scoping not only analyzed the current impacts of the concerns, but also their future predicted outlook according to the most likely scenarios of demographic, economic, technological and other changes with potential influence on the water environment.

In order to identify top priority concerns, a final overall score based on the present and future scores of the environment and socio-economic impacts of each of the prioritized concerns was calculated, and analyzed further in the Casual Chain Analysis and Policy Option Analysis. The assessment recognized that the concerns interact with each other, and therefore the links between the concerns were highlighted to enable the identification of the interventions measures that would yield the greatest benefits for the environment and society.

Results and Discussions

The results of the freshwater shortages and pollution concerns to give a picture of the present situation, and using these results to make predictions for the future outlook as extrapolated from this baseline and, taking into account the influence of a globally changing environment.

Freshwater Shortage

Alteration of the rivers regimes due to anthropogenic activities has impacted both surface and ground water supplies with the same effects applying to changes in the water table. Similarly, modification of the rivers basins has had significant effects on the natural resources allied to the rivers course with consequences to the economy in general as well as to the social and welfare of people of the Tana River County. The effects of arising from freshwater shortages as a result of modification of stream flow, pollution of existing supplies and changes in the water table are presented as the baseline information for rating the score and impact resulting there-from.

Modification of Stream Flow

The main anthropogenic activity that has directly contributed to the modification of the Tana and Athi/Sabaki River basin stream flow is the damming the river Tana for (i) hydroelectric power generation and (ii) irrigated agriculture in both rivers. The two activities are elaborated on below.

(i) Damming on the Tana River System

The Tana River has been dammed for multi–purpose uses. However, with an estimated potential of generating 960 MW of hydro-power, damming for this purpose is the force driving the construction. With a total of five dams, already built, the appetite is still there as the desired capacity

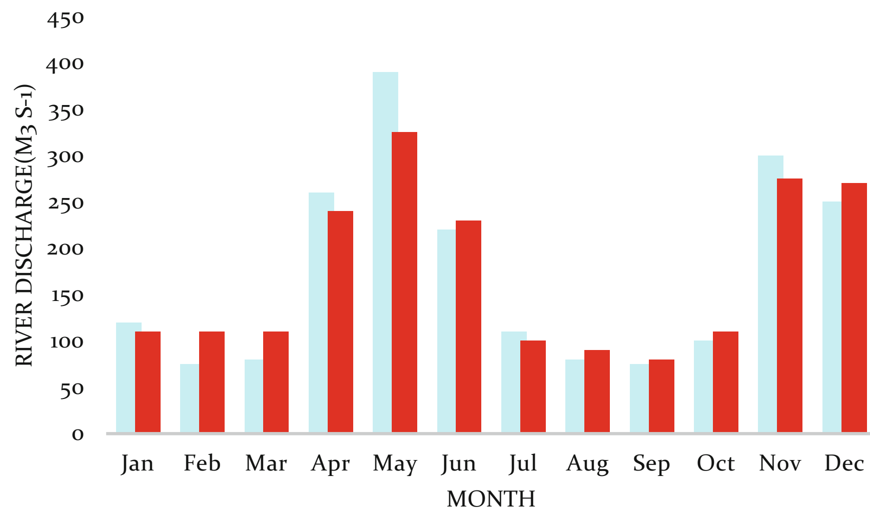


Fig. 5 Mean monthly flows for the Tana River for pre and post-dam periods

has not yet been met from those already built. The effects of the dams on the river both upstream and downstream are set to increase as potential sites planned for future development have been identified at Mutonga (Grand Falls), Usueni, Adamson's falls and Kora. Apart from hydropower generation, Masinga and Kiambere, which have large reservoirs were developed with capacities meant for river flow regulation to tame flooding downstream. The effects on the river flow regime before and after the construction of the dams, analyzed by Maingi and Marsh 2001) contribute to the discussion.

Maingi and Marsh 2001; through analysis of rainfall distribution, using data from Meru, Embu, Muranga and Nairobi, segregated into pre-and post-dam periods using both parametric and non-parametric tests to compare if there was statistically significant differences in means, medians, and distribution of rainfall between the two periods, established no such significant differences, confirming that any reductions observed in river discharge between the two periods are solely attributed to dam construction.

The river discharge volume is dictated by the bi-annual rainfall pattern experienced in the catchment. This results in the biannual floods, with peaks in May – (the long rain), and in November, (–the short rain) season. The May river flows are generally higher and less variable (Maingi and Marsh 2001), an indication of more rainfall during this period than that in November, which may be more variable. The lowest river flows occur in February–March and September-–October, corresponding to the end of the dry weather in the upper river catchment, and the lower Tana River floodplain.

High rainfall signify high daily discharges as exemplified by high river flows observed throughout East Africa in the 1960s, which were attributed to higher than average rainfall observed in that period. Analysis of the flood frequency from data on daily river discharge pre and post-dam periods over 5, 10 and 20 year periods, have indicated the pre-dam discharges calculated from their medians were significantly greater ($p < 0.01$) than the post-dam discharges, and the discharges from the two periods were also significantly different ($p < 0.01$) Fig. 5.

Consequently, it can be generalized that while the dams have augment the minimum river flows, their major impact have been the significant reduction of peak floods in May and November (Maingi and Marsh 2001). This observation is consistent with the studies of Petts (1984) and Toner and Keddy (1997). Of particular note is that May, the month of the highest discharge during the year, experiences the largest reduction in flow, while the flows for April, June, and July see virtually no change in median flows since the dam constructions. Reduced peak flows in many have the implication of reduced flood plain area, with effects on irrigated farming downstream.

(ii) Irrigated agriculture and Livestock Keeping

The area under irrigated agriculture is about 54,676 Ha. (JICA/GOK 1992) encompassing 30,148 Ha of private development and 24,528 of Government irrigation development schemes. The basin has a potential of 132,000 ha for irrigation development. The National Irrigation Board (NIB) has Mwea Paddy Rice Scheme and Tana River Scheme for cotton and subsistence cereals. The Tana and Athi River Development Authority (TARDA) manages Masinga Horticulture Irrigation Project (12,000 ha) and Kiambere Fruit Irrigation Project (150 ha). The Bura Irrigation Project, initially to cover 6,700 ha on completion, has 2500 ha.

In upper Tana, private irrigation schemes, coffee, horticulture and floriculture are intensively practiced in Muranga, Thika, Lower Kiambu and Nyeri using sprinkler irrigation. Along the main Tana River, rice, bananas and fruit irrigation schemes exist. Other irrigation sites on Tana River include

Muka Mukuu settlement and Irrigation Scheme, a development of 500 ha of coffee and fruit trees as well as rehabilitation of a 1,200 ha sisal plantation. Flood recession agriculture is practiced on the Tana Delta, especially during and immediately after heavy rains when floods reach lower Tana River, overflow the banks, and enter previously prepared depressions. These depression or basins are closed to return floodwater when floods recede. The basins are then planted with rice, green-grams and other vegetables. Sometimes these agricultural activities are interrupted by retention of floodwater in hydroelectric dams located upstream (Bobotti 1996).

The downstream meandering of River Tana caused by sediment loading led to a shift in the river coarse in the late 1980’s, impacting negatively on Bura Irrigation Scheme. Abstraction of the water from the River Tana to meet the domestic water supply needs of Nairobi and other urban centers is also associated with modification of stream flow and consequent freshwater shortages downstream.

In the lower Tana basin, livestock farming is the main socio-economic activity. It is practiced by the Orma and the Wardei. This is so because a major part of this area is suitable for livestock farming. Small scale crop production along the river flood plain is practiced by the Pokomo who grow maize, bananas and mangoes. Other socio-economic activities that support livelihood include fishing and small scale businesses. The Tana County has potential for rice production and eco-tourism development. The per-capita herd size of animals is 8.8 for the Orma and 13.7 for the Wardei (Nema 2009). The mean number of cattle held by a house hold is 12. Bearing in mind that the household size is about 14 individuals, then there are as many people as there are cattle in the area. Movement of cattle in the area resonates with the rain seasons. Hence the movement of livestock in and outside the delta corresponds to the rainfall pattern. Cattle move into the delta from January to March, after which they move into the plains in anticipation of the rain in April. They stay in the pains until June, when the herds head back to the delta, staying there until October. With the on-set of the short rains in this month, the cattle head back to the plains, remaining there until the end of the year. Thus the migration to and out of the delta in search of pasture sustains livestock keeping in the area. However, the movement of animals creates a number of problems, including resource use conflicts when the cattle graze on the crops belonging to the agrarian Pokomo tribe. The movement of the herds also exposes them to the tsetse fly menace and risk from trypanosomiasis, risk with the loss of many animals. Movement of the animals is also associated with diseases such as rinderpest and the foot and mouth disease. The Tana River being a source of water for the animals has also led to conflict at watering points along the river and to other watering points, provided through wells away from the river. The watering points are thus a source of conflict over water among the two pastoral communities and also from the farming Pokomo community.

A livestock population census, 2006 showed that there were 370,000 cattle, 280,500 sheep, 369,000 goats, 58,000 camels and 19,600 sheep in Tana River County. Challenges met in maintaining such huge livestock population include the problem of extended drought, rainfall failure, predation particularly on sheep, and farmer/pastoralist conflicts. This problem is compounded by the influx of additional livestock from Ijara and Garissa Counties. Key environmental issues associated with livestock include persistent drought, denying the animals water for drinking, influx and movement of animals and livestock, overgrazing in the Tana Delta, human conflicts, trampling, overgrazing and lack of sanitation in livestock markets/holding grounds, livestock/wildlife conflict at watering points and soil erosion, inadequate water supplies for both human and livestock and water user conflicts between farmers and pastoralists.

Another effect resulting from the large herd of animals is habitat loss/deterioration, pollution from increased turbidity in the water due to erosion of de-vegetated land, reduced farmland, loss of water through evapotranspiration etc.

The demand for water in the lower Tana River has caused excessive water abstraction in the river, resulting in salt water intrusion upstream as has been reported in some rice irrigated schemes of the Tana Delta, leading to reduced productivity, (Nema 2009). The appetite for water upstream, salt water intrusion and pollution contribute to water unavailability for some intended uses, causing water scarcity, and fueling the conflicts in resources use with dire downstream consequences.

Pollution of Existing Supplies

Pollution of existing water supply is another cause for freshwater shortages downstream. Field information shows that the upstream has severe soil erosion, leading to runoff of soil debris into the river. This is hauled as sediments within the channel down to the reservoirs. The reservoirs or hydropower settlement dams have been filling up with these sediments, reducing –capacity of the reservoirs, transparency of the water, and silting of the turbines. Sediments cause pollution of water in these rivers, limiting access of its waters to people for domestic use. (Dubois et al 1985; Delft Hydraulics Laboratory 1970a, b; Kitheka 2002) have reported that suspended sediments in the Tana are a course of pollution in these water bodies.

Besides, sewage and industrial effluents from the major cities and towns upstream are catchment areas that contribute to pollution of the water systems with effluents discharged from both Nairobi and Thika (Njuguna 1978; Mwaguni 2009) contributing to the pollution concern. Analysis of water from the Tana River, during the rainy season,

Sharing Water with Animals

Animals away from farmlands

Man/Animals waiting for their turn to water

Herder invading a road searching for pasture

Drinking under guard from farmland

waiting for her turn as goats drink

Shell well scoped to provide water

Animals waiting for their turn to drink

Photos depicting conflicts in resource use has yielded coliform counts of above 100/100ml and E. coli values greater than 10/100ml, which is indicative of pollution by sewage, (Mwaguni 2008). Thus, poor management of effluents from human settlements has effects on both surface and groundwater as it renders water sources unavailable for human needs through the effects pollution causes, which in turn, results in freshwater shortage as the consequence. The city of Nairobi with approximately 3.4 million people is traversed by the Nairobi River that joins the Athi River downstream; the urban centers of Thika, Murang'a and Nyeri, located within the catchment of the Tana River and other smaller trade centers also contribute to pollution, deteriorating the quality of water for domestic uses. Water pollution results from contaminated storm water discharges, poor disposal of urban and industrial wastes that leach or seep and only eventually end up into the water system.

Changes in the Water Table

Change in the water table is another cause for freshwater shortages. Uneven distribution of water in both time and space makes balancing of water demand and supply quite a difficult challenge. Competition for available water by various uses is high and in order to alleviate the water problem, the government has intensified extraction of groundwater through drilling of boreholes. More than 4,500 have been drilled in the Tana River basin to access groundwater as an alternative source of potable supplies. Majority of the boreholes now require rehabilitation due to non-yielding of the water resource. Consequently, new boreholes have to be dug deeper in order to access the water (GOK 2002).

(i) Rating for freshwater Shortage

Environmental aspects and the socio-economic characteristics of the Tana-Athi/Sabaki Rivers system coupled with specific reviews on the effects of modification of stream flow, pollution of existing water supplies and changes on the water table provided the basis for the justification of the scores on the identified effects. The scores were then averaged to give the present rating for freshwater shortages. Considering the existing socio-economic activities and projecting for future growth, the future effects were predicted and the impacts rated. This gave the overall time and impact averaged score for freshwater shortage in the Tana-Athi/Sabaki River system. Information on environmental conditions and socio-economic activities was required to satisfy or justify the criteria for scoring as indicated in the GIWA methodology.

Natural environmental conditions such as semi-arid to arid conditions, sedimentation and siltation are recognized as the factors that can cause modification of stream flow, promote pollution of existing supplies or affect the position of the underground water-table. Besides the natural factors, anthropogenic activities can exacerbate the problem. When these are taken into account, the use of water for hydroelectric power production, irrigation, domestic consumption, commercial and industrial establishments and for pastoralism –a huge demand is exerted on the water system of these two rivers. The detailed score tables for the Tana-Athi-Sabaki Drainage Basin where scoping environmental impacts under present and future conditions can be seen give an overall score of 3, indicating that the basin is negatively impacted by changes in water table, modification of stream flow and pollution, Table 6.

Table 6 Present and future impact of freshwater shortage, scores and calculated overall time and impact averaged scores based on team scores

Concern	Issue	Score
I. Freshwater shortage	1. Modification of stream flow	3
	2. Pollution of existing supplies	3
	3. Change in the water table	2
	Present impact averaged score	2.67
	Future impact score	3
	Overall time and impact averaged score	2.96

NB: The prediction is that the future impact scenario for each issue will be extreme, and worse

(ii) Rating for Socio-economic Impacts of Freshwater Shortages

Freshwater scarcity will be chronic as the population continues to grow and socio-economic activities expand. This will impose negative impacts on health, reduction in food production, aggravating food security; further spiraling effects on health are anticipated with increased urban growth and discharge of effluents from urban areas, with potable water becoming even scarcer, and worsening with time. Increased damming is geared to harness water for irrigation activities to increase crop production. However, most crops are for commercial production and are not directly related to enhancement of food crops for consumption.

Clear management of the freshwater resources for harmonious use in the various sectors of agriculture industry, hydropower etc. that are using water resources is lacking or inadequate. This makes future socio-economic scenario difficult to gauge (Table 7). Water shortages cause extensive migrations, especially among pastoral communities, contributing towards the distribution of diseases over wider areas. It also leads to poor water quality and other social problems such as resource use conflicts between the migratory pastoralists and sedentary crop farming communities. With time, in the vast rural areas of the Tana River County where these activities are mainstay lifetime enterprises, water scarcity poses real threat to life. Table 7 gives the team scoping for socio-economic impacts under present and future conditions for Tana-Athi/Sabaki Drainage Basin under the freshwater shortage concern.

Table 7 Freshwater shortages socio-economic impacts –the Present and Future Outlook

Economic impacts		Health impacts		Other social and community impacts	
Present	Future	Present	Future	Present	Future
3	3	3	3	3	3

Shortage of freshwater has built an age-old confrontations between pastoralists and crop farmers arising from conflicts over grazing areas and watering points although irrigation schemes have assisted to reduce these conflicts. Increased food production has contributed to food security and assisted in alleviation of malnutrition. Provision of piped water from surface and groundwater has also assisted in combating water borne diseases.

Although the full potential for both hydropower and irrigation have not been fully exploited, frequent occurrences of long periods of drought have constrained achieving it. During dry spells water levels in the dams are reduced to levels where hydropower production is curtailed; similarly, reduced water flow in the rivers has effect on irrigation, resulting in crop failure. The impact of saline water on farming has only been reported in the Tana Delta rice farms.

Pollution

Pollution is another issue of concern in the Tana and Athi/ Sabaki Rivers' water system. The sources of pollution include agricultural based activities along the major rivers, human settlements and industrial activities. The causes of pollution considered in detail in the study include eutrophication, sediment loading, solid wastes and human effluents. The effects these cause on socio-economic activities have also been analyzed.

Eutrophication

Nutrient loading is a major problem in the basins of the river system because, about 85% of the population in these catchments practice subsistence and commercial farming, involving mixed crop and livestock husbandry, and irrigated farming, confined mostly to schemes run by public and private agencies. Smallholder farmers hardly use chemicals to improve their crop yields. However, their land use patterns through slash and burn agriculture, overgrazing and nomadic pastoralism, including farming on the fringes of the rivers, contribute to increased eutrophication. Eutrophication occurs through soil erosion, animal and domestic wastes, suspended solids and solid wastes generated in the farms and human settlements, getting discharged through runoff from the basin catchment into the rivers and down to the marine environment, mostly during the short, but high intensity and highly erosive rainfall seasons. The effects of eutrophication are discussed subsequently.

Sediment Loading

Approximately 16.2–21.3 million tonnes of sediments are deposited into the ocean annually from the Tana and Athi/ Sabaki river system. River Tana discharges about 7 million tonnes, while the Athi/Sabaki discharges 9.2–14.3 million tonnes. The main cause for this high sediment load, as stated earlier is bad farming management systems in smallholder subsistence agriculture in the drought prone areas (Dubois et al 1985; Kitheka 2002). Historical data shows that the river Tana, according to data recorded at Garsen, contributed 1,661 tonnes per day of sediment in the month of October, 1979; 2,304 tonnes per day in August, 1980; and 3,387 tonnes per day in August, 1982. On the other hand, the Sabaki River, with data recorded at the Baricho water treatment works intake gave the following results; October 1979, 74 tonnes per day; May/July/November 1980, 3,769 tonnes per day; June–October 1981, 846 tonnes per day; January—March 1982, 195 tonnes per day; June/August 1984, 111 tonnes per day; and in August 1994 at the river mouth, 143 tonnes per day, (Kazungu et al. 2001) Though recent data is unavailable, owing to poor monitoring, the volume of sediment discharge should be increasing with increased population and socio-economic activities. As a consequence of the high loads of suspended solids discharged through the Sabaki estuary there is an absence of mangrove vegetation, unlike in other estuaries along the Kenya coast. The coral ecosystem extending into the Malindi National Marine Park and Reserve has been negatively impacted (McClanahan and Obura 1995) as evidenced by the shadowing of corals. Similarly, as a result of the high sediment discharges and deposition seagrass communities have been impacted on negatively resulting in a reduction of species diversity (Wakibia 1996). Also beach accretion is dominant, such that beach hotels have lost their beach frontage and, due to the nature of the river sediment being deposited (brown sand and silt) the aesthetic value of the beach along the Malindi Bay has been reduced considerably making it less attractive to the development of tourism, (Kazungu et al. 2001).

More intensive agriculture is undertaken using irrigation most especially along the Tana River which accounts for 87% of the total irrigated agricultural land. Since irrigation farming is for commercial crop varieties makes use of pesticides and fertilizers to enhance production, it is common that pesticide and fertilizers carried by the sediments cause pollution in the water courses but monitoring studies on the environmental impacts arising from their use is lacking. Examples of the agrochemicals used, and which are of concern include pyrethrins, dimethoate, fungicides, furadan, paraquet, glycophosphate, Atrazine, 2, 4-D, herbicides (Tordon IOI, Hyvar X), Acaricides (Triatix, Stelladone); while the fertilizers include phosphate and muriate of Potash, (Mwaguni and Munga 1997).

Solid and Microbiological Pollution

Urban settlements are regarded as pollution hot spots in the basin because these are places where generation and concentration of industrial and domestic effluent and wastes is greatest in comparison with rural areas. In most urban settlements, water borne and water related diseases account for 50% of all the diseases. Water borne diseases accounted for 30%, Mwaguni 2013). Growth of urbanization has been phenomenal from 1969 reaching its peak in 1989. During this period, there was proliferation of trading centers, which became the genesis of urban centers. Their growth rate was approximately 2.1 percent per year (CBS 1989). Established urban centers such as municipalities or cities have continued to show high growth rates. Ruwa and Mwaguni (2002) observed urban areas generated large quantities solid waste with only a fraction of it being collected. Similarly, domestic wastes and effluents in the urban centers are largely handled on-site. Crude management of solid wastes, including human and industrial effluents means these remain to get washed into the marine environment through surface run-off, or through storm-water drainage. Leachate and seepage from landfills enhance pollution. This observation is supported by the high levels of disease causing pathogens are detected in the creeks along Kenya's coastline (Mwaguni 2002). Human waste discharges into the ocean has also led to high productivity in the creeks around Mombasa and in the Tana-Athi-ASabaki River basins, Osore et al. (1997).

Industrial effluents discharges coupled with domestic waste disposal are most significant in Nairobi and the urban areas in the Tana-Athi/Sabaki water catchment. These wastes are disposed of untreated largely due poor infrastructure and poor provision of services. This problem is compounded by a situation where the country does not have any regular monitoring programs to sound warnings on the effects of such a practice on human health and socio-economic activities.

Environmental Impacts of Pollution

Pollution affects ecosystems in many forms, and leads to debilitation of fauna and flora. Aquatic dependent resources, especially fish and mangroves are particularly impacted by pollution. The intricate relationship between economic activities and pollution, which hurt the bounty for man's survival, has been explicitly shown in the analysis of Tana-Athi/Sabaki basin. On the basis of the reviewed information, using the GIWA Methodology, the scoring for pollution due to environmental issues is shown in Table 8.

From Table 8, the average scores for suspended solids and solid wastes were higher in rivers than for other issues being evaluated. This was attributed to increased sediment

Table 8 Present and future Environmental Impacts on major systems

Concern 2: Pollution	Issue	Score
	4. Microbiological	2
	5. Eutrophication	2
	6. Chemical	2
	7. Suspended solids	3
	8. Solid wastes	3
	9. Thermal	–
	10. Radionuclide	–
	11. Spills	1
	Present impact averaged scores	2.17
	Future impact averaged scores	2.17
	Overall time and impact averaged scores	2.89

and silt loads enhanced by erosion of soils caused by the poor agricultural practices of the peasant farmers, and from increased deforestation, where the land was being cleared to create new farm lands. Similarly, waste disposal from urban areas, contributed to sediment loading into the rivers and in the shoreline. Thus, the Tana-Athi/Sabaki basin is being impacted as a result of increased anthropogenic activities and urbanization, with the latter being the major source of micro-bacteria, nutrients and chemical pollution, magnified in localized "hotspot" areas of pollution downstream. The assessment of the present and future averaged score of 2 for micro-biological pollution, eutrophication from nutrients and chemicals was judged to be a moderate impact. However, the overall time and impact average score of 3 obtained for these issues, indicate an outlook of severe impact in the future.

Socio-Economic Impacts of Pollution

The economic activities that support the welfare of households are in themselves the cause of pollution. The effect of pollution is water quality degradation, loss of aesthetic value of environment, provision of habitat for disease causing pests, increased turbidity in the water courses caused by silt, obnoxious ordours in water, resulting from putrefying organic wastes, all of which in turn discourage on water activities and water scarcity for potable uses. The discharge and deposition of silt or sediments on coral reefs reduces species diversity through the gradual death of coral polyps, resulting in negative impacts on nature tourism. Collectively, the loss of aesthetic value of a destination reflects badly on the tourism trade, an industry which contributes approximately 60% of Kenya's total national revenue, either directly or indirectly.

The effects of agrochemicals discharge emanate from the strong toxicity of these substances. As such, these have the potential to destroy flora and fauna, reducing biodiversity and aesthetic value of the impacted environments. The

Table 9 Team for Tana-Athi-Sabaki Drainage Basin (TAS) from the pollution issue of concern

Economic impacts		Health impacts		Other social and community impacts	
Present	Future	Present	Future	Present	Future
3	3	3	3	3	3

consumption of fish or other aquatic species from areas contaminated with agrochemicals may cause serious public health problems. The discharge of human wastes is an issue of major concern. Though documented information on such effects is lacking for the Tana-Athi/Sabaki water system, the sporadic outbreaks of cholera in Tana River County, is as a result of such discharges. Reported cases elsewhere, such as those by Mwaguni (2002) for Mombasa indicate the gravity of human waste discharges into water bodies as exemplified by the waterborne infections of cholera, dysentery, diarrhea, eye diseases, dermal, and intestinal worms. The effects of pollution were rated in terms of economic, health and other social and community impacts for the present and then extrapolated into the future.

Rating Socio-Economic Impacts of Pollution

The results of rating of the impacts from the pollution concern for the Tana-Athi-Sabaki drainage basin are presented in Table 9, giving the Teams scooping for socio-economic impacts under present and future conditions.

The Tana-Athi-Sabaki basin residents face severe socio-economic impacts associated with pollution from the different sources cited in the table above. This is especially so, because residents depend solely on water from this river system for purposes. Pollution in the marine environment, resulting from river discharges is significant around urban areas where it is common for raw sewage and garbage to be disposed of in water bodies. This makes land based-activities the main sources of pollution into ocean environment. The effects of sediment in the marine environment has been detected on the browning of coral reefs, turbidity in the water, affecting its aesthetic value for tourism, McClanahan and Obura 1995. Water Polluted with human waste is a source of water borne diseases, impacting on socio-economic activities of the people as the affected individuals seek medical treatment (Mwaguni 2009). The lost hours lost seeking treatment contribute to loss in productivity time.

Future Outlook and Global Change for the Two Concerns

It is difficult to consider the effects of global change at the drainage basin level, however, impacts associated with changes in hydrological cycle and ocean circulation, have manifested themselves in the occurrences of cyclic events drought and flooding. Global change is associated with anthropogenic increases of carbon dioxide in the atmosphere. Such increase sea temperatures, leading to among other things, sea level rise, and decreased capacity of the oceans as carbon sinks. Activities that contribute to this include deforestation, combustion of fossil and wood fuels, slash-and-burn agriculture, etc. all of which are major activities undertaken in the Tana-Athi/Sabaki rivers' water basins, a situation that is promote global change.

The effects of global change have manifested themselves in the droughts and floods phenomena, which is not new in Kenya with their characteristics -intensity, duration and spatial extent - varying from one event to the other. The frequency of occurrence and severity of these events have been increasing over time. For example, the frequency of drought increased from once in every 10 years in 1970s, to once in every 5 years in 1980s, once in every 2–3 years in 1990s and every year since 2000 (Howden 2009). The occurrences of droughts and flood are shown in Table 10.

Flood and drought events contribute to water scarcity both directly and indirectly through the agent of pollution. Droughts and Floods destroy livelihood sources, and also severely undermine the resilience of the people living in the affected areas (KRCS 2012). In some arid and semi-arid counties, pastoralists have lost more than half of their livestock to droughts in the past ten years, contributing to over 60% of the inhabitants living below the poverty line (Grunewald et al. 2006).

The effects of droughts and floods have been devastating in a number of ways. Floods inundates arable lands, destroying crops, livestock and property; droughts cause deficiency in rainfall, resulting in freshwater shortages, compounding the problem of stream flow and availability of the resource with consequences to agricultural activities, including the rearing of livestock. In Kenya, the worst drought to have occurred in the last one hundred years, was experienced in years 1999–2001. This drought occurrence killed about 60–70% of livestock in the arid and semi-arid areas, caused massive crop failures, drying up of water resources, severe environmental degradation and loss of goods and services. Global change will thus exacerbate both freshwater shortages and spread of pollution in the entire Tana River water system, and export the pollution problem into the marine environment through the water media.

Scoping for Global Change from Environmental Impacts

As stated, due to its scale in terms of time and space the concern for global change cannot be assessed at the basin level. However, considering the environmental parameters associated with global change such as deforestation rates, use of wood fuel, and slash-and-burn agriculture, temperature change is a major concern in the river basins as

Table 10 Recent hydro-meteorological disasters and their impacts in Kenya

Year(s)	Disaster type	Area of occurrence	Impacts
2011	Drought	Garrisa, Isiolo, Wajir, Mandera, Mombasa, Marsabit, Nairobi, Turkana, Samburu,	4.3 million people in dire need of food
2010	Floods	Budalangi, Tana River, Turkana	73 people killed, 14,585 people affected
2009	Droughts	Widespread	70–90 % loss of livestock by Masai pastoralists
2008	Floods	Rift Valley, Ktale Transnzoia, Machakos, Kibwezi	2398 people affected
	Mudslides	Pokot Central	11 people dead
2007–2008	Drought	Widespread	4.4 Million people affected, 2.6 Million at risk of starvation, up to 70 % loss in livestock among pastoral communities
2006	Drought	Widespread	3.5 Million in need of food, 40 dead, 40 % cattle, 27 % sheep, 17 % goats dead
	Flood	Widespread	7 dead, 3,500 displaced
		Isiolo	3,000 people displaced
2005	Storm	Merti-Isiolo	4,000 people cut off between Merti-Isiolo
	Drought	Widespread	2.5 Million people close to starvation, declaration of national disaster
2004	Drought	Widespread	About 3 million people in need of relief food for 8 months up to March, 2005
	Flood/ Landslides	Nyeri, Othaya, Kahuri	5 people dead
2003	Floods	Nyanza, Western, Tana River basin	60,000 people affected
2002	Landslides	Meru-central, Muranga, Nandi	2,000 people affected
	Floods	Busia, Nyanza, Tana River Basin	150,000 people affected
1999–2001	Drought	Widespread	4.4 million affected
1997–1998	El-Nino Floods	Widespread	1.5million affected
1995–1996	Drought	Widespread	2.0 million people affected, declared a national disaster
1991–1992	Drought	N/Eastern, Eastern, Rift Valley, Coast	1.5million affected
1985	Floods	Nyanza	10,000 people affected
1983–1984	Drought	Widespread	200,000 people affected
1980	Drought	Widespread	40,000 people affected

Source: Huho and Kasonei (2014)

exemplified by periodic droughts and flooding, Table 10. As such all issues and impacts associated with freshwater shortages and pollution will worsen.

Socio-Economic Impacts Due to Global Change

The socio-economic impacts due to global change could similarly not be analyzed from the GIWA method on a basin scale. However, such impacts include the observed extreme changes in weather pattern, which cause food insecurity and disease outbreaks that were noted in the water basin. With such a scenario, it is anticipated that future impacts due to global change will adversely affect the socio-economic fabric of society in the basin. As shown in Table 10, loss of property has resulted from droughts and floods. The frequency of such occurrence have also been shown to be increasing, a situation that will undermine livelihood in areas already suffering from extreme whether events.

Conclusions and Recommendations

Freshwater Shortages

Conclusions

(i) Both dam construction and farming activities upstream have been identified as the major causes of freshwater shortages in the Tana River County. Damming is associated with reduced stream flow, change in water courses, changes in water table and general water scarcity downstream while farming is associated with water pollution in the entire river courses.

(ii) There is still unexploited potential for increasing hydropower production and irrigation in the basin with an estimated 155,100 Ha available for irrigated farming, depending on the availability of ample supplies of water (GOK/JICA 1992). A situation, which could lead into more dams being built and more irrigation schemes started.

(iii) Arid and semi-arid areas are significant source of beef and other livestock products. However, owing to conflicts between farming and pastoral communities, both socio-economic activities cannot be developed sustainably to their full potential.

(iv) Pollution of water at watering points where livestock and people compete for this resource is common. This scenario is a common cause of freshwater shortages in that polluted water supplies are not suitable and therefore unavailable for human use, creating the water scarcity situation in the process.

(v) Although there have been efforts to provide piped water there is still a disparity in its provision between rural and urban areas. The proportion of people with access to safe drinking water within reasonable distances remains a pipe dream in the rural areas despite huge investment in the sector. This situation replicates itself in Tana-Athi/Sabaki catchment areas with consequences of waterborne diseases, causing disorders to both human and livestock health.

Recommendations

(i) There is need to plan for the development of the Tana River Basin and its resources in an integrated way, taking into account of all the issues before development decisions are made. Such an undertaking will realize the importance of water conservation and promotion of its quality for the preservation of ecosystem functions downstream.

(ii) Long term monitoring of the activities taking place in the basin should be undertaken and integrated with the monitoring of the total natural resource base of this area to generate plans that guide utilization of the basins resources for sustainable development.

(iii) Water use conflicts are bound to increase in scope and frequency. Consequently, there is need to consider water as an economic good to prevent wasteful use of the resource and, making it available to more users in order to reduce the conflicts among the sectors that compete for this resource.

(iv) There is need for adequate planning for both pastoral and farming activities to reduce the completion of water by these two entities.

(v) Strengthen and enforce land use regulations.

Pollution

Conclusions

(i) Deposition of suspended solids on the coral system of the Malindi Marine Park and Reserve results in shadowing of the coral ecosystem with the consequent loss of the aesthetic value of this system making it less attractive to tourists.

(ii) Decline in biodiversity in the two major habitats i.e. the coral and seagrass meadows, with the disappearance of seagrass species in the Malindi Bay probably due to suspended solid loading having been reported.

(iii) Sediment deposition and beach accretion in the Malindi Bay have resulted in the loss of beach frontage to some beach hotel establishments resulting in loss of tourism revenues.

(iv) There is potential negative impact to fisheries productivity as the major habitats, are impacted by suspended solids. This will translate to decreased fish catches, undermining food security and earnings from fishing.

(v) Health impacts associated with pollution are high due to micro organic/disease causing organisms and water related disease such as malaria.

Recommendations

(i) Promotion of good agricultural practices along river basins and marginal areas

(ii) Enforce the treatment of sewage and industrial effluents before they discharged to the waterways.

(iii) Promote the polluter pays principle to address pollution from agrochemicals.

(iv) Address the root causes of population pressure, institutional governance and macro-economic policies.

Global Change

Conclusions

(i) Unusual patterns of unpredictable climatic extremities in occurrence and magnitudes of drought and floods have been observed in the region besides coastline erosion and coral bleaching. The loss of livelihood options, starvation, deaths, and losses of livestock associated with the extreme weather patterns and their associated effects have been established.

(ii) Activities such as deforestation, combustion of fossil fuels, savannah burning, combustion of wood fuel, burning of refuse etc., take place in the area and these make a contribution to the problem of global change.

(iii) The socio-economic losses associated with extreme droughts and floods such as risk to life, damage to infrastructure and environmental degradation are common in the area.

(iv) Associated with climate change are health problems caused by enhanced opportunistic occurrence of diseases due to the extreme climatic fluctuations and malnutrition, resulting from food insecurity.

Recommendations

(i) As it has been established that anthropogenic activities contribute to global change, with the cyclic events of droughts and floods now predictable, proactive strategies should be put in place to mitigate against, or develop resilience of such impacts.

(ii) Secondly, there is also a need for coordinate actions among countries in mitigation or adaptation to climate change since the problem is global in nature. This notwithstanding, local intervention measures should also be pursued

(iii) There is need to build capacity for monitoring, control and surveillance of activities that contribute to global change to assist understanding of global change to be able to minimize the effects it causes. Monitoring for drought and flooding undertaken as a matter of policy.

References

Bobotti OK (1996) Tana river basin management – a case study. In: Hirji R, Partoni FM, Rubin D (eds) Proceedings of seminar on integrated water resources management in Kenya. Sponsored by government of Netherlands & the World Bank. The Economic Development Institute of World Bank, 227p

Brinson MM (1990) Riverine forests. In: Lugo AE, Brinson M, Brown S (eds) Forested wetlands. Ecosystems of the World, vol 15. Elsevier, Amsterdam, pp 87–114

Carbone F, Accordi G (2000) The Indian Ocean coast of Somalia. Mar Pollut Bull 41(1–6):141–159

CBS (1989) Kenya Population census 1989, Analytical Report vol III, Population dynamics of Kenya, Ministry of Planning and National Development. Nairobi, Government of Kenya

Delft Hydraulics Laboratory (1970a) Malindi Bay Pollution, Part 2: field measurements and recommendations. Report submitted to the Ministry of Agriculture, Kenya. The Netherlands, 11p

Delft Hydraulics Laboratory (1970b) Malindi Bay Pollution, Part 2: field measurements and recommendations. Report submitted to the Ministry of Agriculture, Kenya. The Netherlands, 11p

Doute R, Ochanda N, Epp H (1981) A forest inventory of Kenya using remote sensing techniques. Kenya Rangeland Ecological Monitoring Unit Report, No. 30. Nairobi, 130P

Dubois R, Berry L, Ford R (1985) Catchment land use and its implication for the coastal resources conservation. Programme for International Development. Clark University, Worcester, Massachusetts, USA, 65p

FAO (2001) A synthesis of flood warning and control for the Juba and Shabelle Rivers. FAO SOM/SSF/2001/12. Working Paper. FAO, Nairobi

GOK (2002) Country strategy on the integrated water resource. Management Ministry of Environment and Natural Resources, Nairobi (Draft Paper), 84p

GOK (2009) Kenya National Population and Housing Census, Central Bureau of Statistics, Government Printer, Nairobi

GOK (2012) Tana delta strategic environmental assessment scoping report, 46pp

GOK-JICA (1992) National water masterplan. Government of Kenya (GOK) and Japanese International Cooperation Agency (JICA), Nairobi, Kenya

Government of Kenya (GoK) (2002) National Development Plan, Ministry of Planning, Nairobi, Kenya

Grunewald F, Robins K, Odicoh A, Nicholson N (2009) Kenya. RTE mission final report

Hatziolos M, Lundin CG, Alm A (1994) Africa: a framework for integrated coastal zone management. The World Bank, Washington, DC, 150p

Hirji R, Partorni FM, Rubin D (eds) (1996) Seminar on integrated water resources management in Kenya. Proceedings sponsored by government of the Netherlands and the World Bank. The Economic Development Institute of the World Bank, 227p

Howden D (2009) The great drought in East Africa available at www.infiniteunknown.net/2009/10/3

Hughes FMR (1985) The Tana River floodplain forest in Kenya: ecology and impact of development. PhD thesis, University of Cambridge

Hughes FMR (1988) The environmental impact of river basin development: whose responsibility? Cambridge African monographs. African Studies Center, University of Cambridge, Cambridge

Huho JM, Kasonei RC (2014) Understanding extreme climate events for economic development in Kenya. IOSR J Environ Sci Toxicol Food Technol 8(2):14–24

JICA/GOK (1992) National Water Masterplan. Ministry of Land Reclamation, Regional and Water Development, Nairobi, Kenya. Japanese International Cooperation Agency and Government of Kenya

Kazungu JM, Munga D, Mwaguni SM, Ociewo J (2001) Kenya country report GEF-MSP Sub-Sahara Africa Project, (GF/6010-0016) for the development and protection of the coastal and marine environment, National report. Phase 1: Integrated problem analysis, 70pp

Kitheka JU (2002) Dry season sediment fluxes in the frontwater zone of the mangrove-fringed Mwache creek, Kenya. In: Arthurton RS, Kremer HH, Odada E, Salomons W, Crossland JIM (eds) African Basins: LOICZ Global change assessment and synthesis of river catchment – coastal area interaction and human dimensions. LOICZ Reports and Studies No.25ii+344 pages. LOICZ, Texel, pp 194–202

KRCS (2012) Drought 2011: how Kenya responded. https://www.kenyaredcross.org/PDF/K4K/Drought

Lange K (2014) Hydrological consequences of proposed interventions in the Tana Basin, Kenya, A water Balance study. M.Sc., thesis, Utrecht University, 96pp

Livingstone DA (1975) Late quaternary climate change in Africa. Annu Rev Ecol Syst 6:249–280

Marsh CW (1976) A management plan for the Tana River Game Reserve. Report to the Kenya Department of Wildlife Conservation and Management. Nairobi

Marsh CW (1978) Tree phenology in a gallery forest on the Tana River, Kenya. East Afr Agric For J 43:305–316

Maingi JK, Marsh SE (2001) Assessment of environmental impacts of river basin development on the riverine forests of eastern Kenya using multi-temporal satellite data. Int J Remote Sens 22:2701–2729

Maingi JK, Marsh SE (2002) Quantifying hydrologic impacts following dam construction along the Tana River, Kenya. J Arid Environ 50:53–79

McClanahan TR, Obura D (1995) Status of Kenya coral reefs. Coast Manag 23:57–76

McClanahan TR, Young TP (eds) (1996) East African ecosystems and their conservation. Oxford University Press, New York, 452p

Medley KE (1990) Forest ecology and conservation in the Tana River National Primate Reserve, Kenya. PhD dissertation, Michigan State University

Medley KE, Hughes FMR (1996) Riverine forests. In: McClanahan TR, Young TP and Their Conservation. Oxford University Press, New York/Oxford, pp 361–383, 452p

Mwaguni SM (2002) The cost and benefits associated with addressing the sewage problem affecting the coastal marine and associated freshwater environment with special focus on the town of Mombasa on the Kenyan Coast. A contribution to the implementation in East Africa of the UNEP/GPA Strategic Action Plan on Sewage, 51p

Mwaguni SM (2008) The effects of tourism and related human settlements on the conservation of coastal water resources: a case study of Nyali-Bamburi-Chale. PhD thesis, University of Nairobi, 227 pp

Mwaguni SM (2013) Urbanization and groundwater pollution in Mombasa-Kenya: do we have to transfer the problem due to construction of the Dongo-Kundu by-Pass? J Environ Earth Sci 3(4):2013

Mwaguni (2009) Effects of tourism and related settlements on the conservation of coastal water resources, Unpublished PhD thesis, University of Nairobi, 227pp

Mwaguni S, Munga D (1997) Land based sources and activities affecting the quality and uses of the marine coastal and associated freshwater environments along the Kenyan Coast. National Report, Republic of Kenya, 56p

Nema (2009) Tana River District National Environmental Plan 2009–2013, 57pp

Njuguna SG (1978) A study of the effects of pollution on a tropical river in Kenya. MSc. thesis, University of Nairobi, Kenya, 202pp

Ogwenyi GS, Denga FGO, Abwao P, Kitheka JU (1993a) Impacts of floods and drought on the development of water resources of Kenya: case studies of the Nyando and Tana catchments. Hydrol Warm Regions 216:117–123

Osore MKW, Daro N, Tacks MLM (1997) The effect of rainfall and tidal rhythm on the community structure and abundance of the zooplankton of the Gazi Bay, Kenya. Hydrobiologia 356:117–126

Petts GE (1984) Impounded rivers: perspectives for ecological management. Wiley, Chichester, 326 pp

Ruwa RK, Mwaguni SM (2002) Waste collection, handling and treatment system: Port of Mombasa environment impact appraisal. A report to the East African Marine & Environmental Management Company Limited, Mombasa, Kenya, 36p

Saha SK (1982) Irrigation planning in the Tana Basin of Kenya. Water Supply Manag 6:261–279

Toner M, Keddy P (1997) River hydrology and riparian wetlands: a predictive model for ecological assembly. Ecol Appl 7:236–246

UNEP (1998) Eastern Africa Atlas of coastal resources 1: Kenya. UNEP and BADC (Government of Belgium), 119p

UNEP (2001) Eastern Africa atlas of coastal resources: Tanzania. UNEP and BADC (Government of Belgium), 111p

UNEP (2006) GIWA, Challenges to international waters –regional assessment in a global perspective

Vanden Bossche JP, Bernacsek GM (1990) Sourcebook for the inland fishery resources of Africa, vol 1. CIFA Technical Report No.18.1. FAO, Rome, 240 p

Wakibia JG (1996) The potential human induced impacts on the Kenya Seagrass, UNESCO Reports in Marine Science No. 66

WRMA (2009) Tana water catchment management strategy

The Socioeconomic Causes and Impacts of Modification of Tana River Flow Regime

Jacob Odhiambo-Ochiewo, Renison K. Ruwa, Melckzedec Osore, Diana Mutiso, and Saeed Mwaguni

Abstract

The flow regime of River Tana has been altered by a number of socioeconomic activities upstream. Modification of the flow regime of the river has had significant effects on the natural resources allied to the river, its economy and the general welfare of people. The study aimed at establishing the social and economic causes and impacts of modification of the Tana River flow regime. Results indicated that the main causes of modification of River Tana flow regime include hydropower generation, irrigated agriculture, abstraction of water for domestic and industrial use, and pastoralism. The demand for hydropower and food has also increased following the 35% increase in population between 1999 and 2009. The socio-economic impacts of modification of River Tana flow regime have been manifested in freshwater shortage downstream particularly during drought periods. Low levels of water flow in the river during the long drought periods put considerable constraints on irrigation activities. The freshwater shortages have further resulted in confrontations between pastoralists and crop farmers over grazing areas and watering points leading to loss of lives. Waterborne health disorders arise from unsafe water sources and affect human communities. It is recommended that investments should be channeled towards water treatment programs for the population in this area. There is need to come up with appropriate integrated resource management for sustainable utilization and management of water and land resources in the basin. The arid areas still remain the most significant source of beef and other livestock products hence there is need to provide water services to reduce conflicts with other demanding sectors especially competition for watering points.

Keywords

Socioeconomic activities • Modification of river flow • Hydropower generation • Irrigated agriculture • Freshwater shortage • Demand

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Introduction

River Tana is the longest river in Kenya with a total length of approximately 1000 km and catchment area of 100 000 km^2 (IUCN 2003). The river has its source in the eastern slopes of the Aberdares and southern slopes of Mount Kenya (Maingi and Marsh 2002). It flows through semi-arid and arid regions of Kenya as it flows to the Indian Ocean. In its lower course, the river meanders through alluvial floodplains and forms a delta as it enters the sea. The long-term and short-term flooding of River Tana strongly influences the ecology of the flood plain as well as the formation and distribution of wetlands. Traditionally, River Tana floods twice a year; in May–June and November–December. These floods resulted in deposition of a fertile layer of silt on the floodplain. The lower sections of River Tana support unique wetland ecosystems such as riverine forests, oxbow lakes, floodplain grasslands, mangrove forest, sand dunes and coastal waters. The riverine forest along the River Tana is an isolated remnant of a continuous rainforest belt that extended between the Congo Basin and Kenya Coast during the moister periods of the Pleistocene (Terer et. al. 2004; Susanne 2004; Allison and Badjeck 2004).

River Tana and its associated wetlands provide livelihoods, income, and socio-cultural benefits to the local people. It also harbours threatened species of primates, fish (De Vos et al. 2002), birds (Bennun and Njoroge 1999) and plants. The local people have traditionally exploited and developed strong connections with the wetland resources within the river basin. Traditional land-use practices of small-scale agriculture, pastoralism and fishing have maintained the ecological balance of the Lower Tana for thousands of years. However, the rapid population growth has resulted into forest fragmentation and use of improper farming practices such as bush burning and shifting cultivation (Wieczkowski 2002).

Like the rest of the world where the natural flow of majority of rivers has been substantially altered through dam construction as observed by Maingi and Marsh (2002), the flow regime of River Tana has been altered by damming and other human activities. Five multi-purpose dams have been constructed on the upper reaches of the river namely: Kindaruma dam (1968), Kamburu dam (1975), Gitaru dam (1978), Masinga dam (1981) and Kiambere dam (1988). A new multi-purpose dam has also been proposed for construction at Mutonga-Grand Falls just below the first five dams. Dam construction has affected the river's downstream flow and physical characteristics, as it regulates water-flow and decreases the frequency and magnitude of flooding (IUCN 2003). Modification of the river's flow regime has had significant effects on the natural resources allied to the river, its economy and the general welfare of people. The study aimed at establishing the social and economic causes and impacts of modification of the Tana River flow regime.

Socioeconomic Impacts

In this section we discuss both negative and positive socioeconomic impacts of modification of the flow regime of River Tana. The socioeconomic impacts include negative impacts on livelihoods and food security, increased conflicts, displacement of people from their traditional homes, scarcity of potable water (freshwater), and disruption of social organization, among others.

Negative Impacts

Before the construction of the five multipurpose dams, River Tana used to flood its banks twice a year. However, since 1989 when the last dam was commissioned, flooding has decreased dramatically in both volume and frequency (IUCN 2003) with resultant socioeconomic consequences. The poor indigenous people and other ethnic minorities downstream bear the disproportionate share of these social and economic impacts without gaining any commensurate share of the economic benefits. This confirms the observation by Beck et al. (2012) that river damming mostly affect marginalized people whose livelihoods entirely depend on riverine resources. The socio-economic impacts of modification of River Tana flow regime have been manifested in freshwater shortage downstream particularly during drought periods. Low levels of water flow in the river during the long drought periods put considerable constraints on traditional economic activities.

Conflicts Due to Limited Dry Season Pasture and Watering Points

Over 240,000 people who reside in Tana River County (Republic of Kenya 2010) are engaged in agriculture, livestock keeping and fishing in areas that are adjacent to the River and the Tana Delta. Most of these people depend on the river's flooding regime for their livelihood activities. Approximately 2.5 million livestock, including over a million cattle, depend on River Tana's floodplain grasslands and water bodies for dry season pasture and water (CADP 1991). During prolonged droughts there is influx of cattle from neighboring counties as pastoralists migrate to the Tana delta in search of pasture and water. The decrease in the flooding regime caused by construction of the existing dams has limited dry season pasture and watering points to the area that is directly adjacent to the river. This has disrupted the traditional patterns of transhumance (the seasonal movement of people with their livestock between rainy season and dry season pastures), increased grazing pressure, and resulted in intensified confrontations between pastoralists and floodplain agriculturalists (Goldson 1994) over grazing areas and watering points. The conflicts sometimes lead to loss of life particularly at the Tana delta. In August 2012, 52 people were killed in ethnic violence in Tana River County between agriculturalists and pastoralists (http://en.wikipedia.org/wiki/Tana_River_County).

Reduction in Floodplain Agriculture

Over 115,000 people depend on flood recession and riverbank farming along the river and at the delta (Emerton 2003). These farmers depend on floodwater to irrigate their crops and on the depositions of fertile alluvial soil. As is the case in many countries in Africa where dam construction has led to the disruption of many traditional production systems including flood recession agriculture (Maingi and Marsh 2002), the agricultural activities in the Tana River basin are interrupted by retention of floodwater in hydroelectric power dams that are located upstream (Hirji et al. 1996). The dam construction has regulated the volume of water that flow downstream, controlled floods and reduced floodplain agriculture. It is likely that after the construction of the sixth dam which is planned at Mutonga-Grand Falls, the farmers will limit their farming to riverbanks only. The reduction in farming area may adversely impact on agricultural production and food security, resulting loss of income and poverty among the local communities.

Reduced Fish Landings from Ox-Bow Lakes

As already highlighted, the natural flooding regime of River Tana has been disrupted by dam construction and other anthropogenic activities which occur upstream. Communities downstream at the river delta have reported noticeable decline in river size as well as decline in the sizes of oxbow lakes (Terer et al. 2004). The ox-bow lakes are important for fish production. A big population also depends on fishing as a source of income and household protein. River Tana, its delta and estuary area support both subsistence and commercial fisheries, providing the main livelihood for more than 50,000 people (Nippon Koei 1998; CADP 1991). Modification of the river flow regime has therefore negatively impacted on the fish production from the ox-bow lakes. It is thought that additional dam construction will rapidly exacerbate this decline in fishing area and catch (Mavuti 1994).

Displacement of People and Shortage of Fresh Water for Domestic Use

The construction of the five dams in the upper reaches of River Tana displaced communities from their traditional homes and impacted negatively on their human wellbeing. Displacement of communities led to the weakening of social networks and other traditional systems that supported human wellbeing. Socially, the displacement separated families and clans that traditional lived together in one place enjoying a common sense of belonging and social networks. Psychologically, this separation of families and clans caused some trauma to the affected people.

The displacement also led to loss of access to the river water which served as a common property resource for the local communities. As highlighted earlier, the local communities traditionally depended on the river water for their livelihoods. It was expected that besides compensation for their land, the project would provide basic things such as supply of potable water to support the displaced populations in the areas where they built their new homes. This study has however revealed that the benefits from damming have not trickled down to those communities whose living conditions were affected by the project. The displaced communities obtained land far away from the river which is their main source of potable water. Consequently, during droughts the women and children have to walk longer distances to collect fresh water for domestic use. This has further impacted negatively on those economic activities that rely on labour provided by women. The population within the Tana basin that can access safe drinking water within reasonable distances has continued to decline. About 78% of the households in Tana River County depend on fresh water supplies from the river, pond or dam and spring or well while only 11% have access to piped water. Waterborne health disorders arise from unsafe water sources and affect human communities.

Disruption of Traditional Norms and Values

Social interactions between the local communities and migrants from other areas who worked in the dams and power stations during the construction phase as well as those who are currently working in the power stations have both positive and negative impacts on the traditional norms and values of local communities. The dam construction resulted in the establishment of temporary towns or areas of high population concentrations with intricate relationships that diluted the traditional norms and values. These temporary towns eventually fizzled out after completion of the dams.

Public Safety

Public safety concern is a problem and it may include accidents from sudden peak releases downstream in the dry season. Dam construction led to relocation of households as well as a permanent loss of arable land and hence reduction of agricultural production.

Positive Impacts

Provision of Employment to the Local People

Despite the negative impacts, modification of stream flow has also provided benefits (Doyle et al. 2000). Some members of the local population particularly from the upper reaches of the river obtained employment as construction workers during dam construction. This employment was however only available for a limited period during dam construction since after the completion of dam construction, the labour was no-longer required. Some members of the local population also got employed by the Kenya Electricity Generating Company (KenGen) while others were employed as security guards. In addition, dam construction created market for local produce and provided opportunities for the growth of small-scale businesses that further provided self employment or supplementary source of income and livelihood to the local population. Although the market eventually reduced after the construction phase, the employees of KenGen who remained behind to work in the power stations have continued to provide a market for some local produce. Expanded market for local produce has similarly been reported elsewhere by Canter (1985).

Improved Transport Infrastructure

Transport infrastructure has been improved in the upper course of River Tana to support hydropower generation activities at the five dams. This road network has improved transport for the local people who are now able to commute without difficulties.

Fish Production from the Tana River Dams

The fish production in River Tana averages about 1000 MT annually and is an important source of protein, income and employment to the local communities. While fish production from the ox-bow lakes at the river delta is negatively affected by modification of stream flow, the dams have become important sources of fish upstream. The main types of fish are Tilapia spp. (50%), Common carp (29.5%) and Clarias (20.3%). The other fish include Eels, Barbus, Labeo and Moromyrids which contribute less than 0.5% of the total fish landings. Despite fisheries being an old occupation within the Tana River basin, commercial fisheries in dams began in 1981 at Masinga dam, while that of Kamburu and Kiambere dams began in 1988 and 1991 respectively (Emerton 2005). There is however no fishing activity at Gitaru and Kindaruma dams because the two are heavily infested by crocodiles.

Generation of Electricity

Hydropower is a clean and cheap source of energy. It is associated with minimal production of dangerous gases, limited solid waste, and one of the cheapest sources of electricity compared to other sources such as thermal plants. Experience gained from using electricity generated from thermal plants during years when prolonged droughts occurred showed that the price of thermal electricity always doubled that of hydropower.

Causes of Modification

The socioeconomic impacts of modification of the flow regime of River Tana have been caused by a number of immediate and underlying root causes (Fig. 1). The analysis of both immediate and root causes is critically important because the socioeconomic impacts could only be effectively tackled by addressing the respective root causes.

Immediate Causes of Modification

The main immediate causes of modification of River Tana flow regime include hydropower generation to meet the growing demand for energy and water abstraction for irrigation and domestic supplies. Hydropower generation currently produces 481 megawatts (MW) but has a potential of 960 MW. Irrigated agriculture currently covers 54,676 ha with a potential of 132,000 ha. Water is also abstracted for domestic and industrial use as well as for livestock.

Hydropower Generation

Since the mid 1960's River Tana has been heavily dammed upstream for multi-purpose activities, though hydropower generation has always been the main driving force for the construction of these dams. Five major dams have been constructed on the river with the aim of harnessing water for hydroelectric power generation. These dams are Kamburu, Kiambere, Gitaru, Masinga, and Kindaruma (VandenBossche and Bernacsek 1990; Boboti 1996). Total hydropower potential for the river is estimated at 960 MW compared to the present output of 481 MW (GOK/JICA, 1992). A new hydropower scheme has been proposed for construction at Mutonga-Grand Falls on the Tana River below the five existing dams. In recent years, hydroelectric power generation has been affected due to low levels of water in the dams caused by droughts. Apart from hydropower generation, the two large reservoirs at Masinga and Kiambere were developed with capacities meant for river flow regulation.

In addition, there are minor dams in the upper catchment for public water supplies, namely Sasummwa, Thika, and Ruiru. Other smaller dams have also been constructed for purposes of irrigating coffee, horticulture, and floriculture. Despite the economic benefits associated with damming, local and downstream communities suffer from its negative

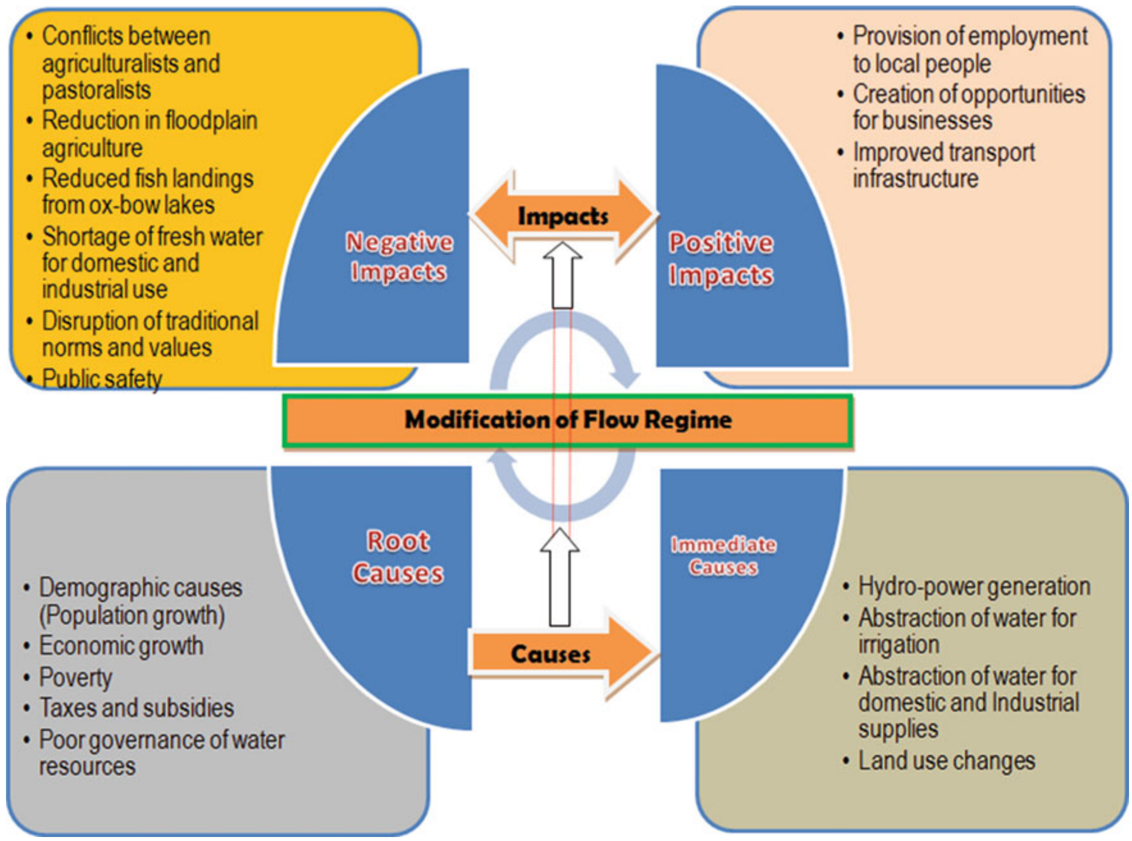

Fig. 1 Diagrammatic summary of the analysis of the causes of modification of River Tana flow regime

consequences. Damming had a major influence on the river's downstream flow and physical characteristics, most notably through regulating waterflow and decreasing the frequency and magnitudeof flooding (IUCN 2003).Families who were displaced by the hydroelectricity generation project were given minimum compensation yet this project altered their livelihoods (GOK-JICA 1992).

Irrigation Agriculture

Water is abstracted from River Tana to satisfy irrigation needs. The River Tana basin has a potential of 132,000 ha for irrigation development (Agwata 2005). Public irrigation schemes located within the basin are partially or entirely managed by quasi government organizations particularly the National Irrigation Board (NIB), the Tana-Athi Rivers Development Authority (TARDA), the Agricultural Development Corporation (ADC) and the Bura Management. The area presently under irrigated agriculture is about 54,676 ha. (GOK/JICA 1992; Agwata 2005) encompassing 30,148 ha of private development and 24,528 of Government irrigation development schemes.

The irrigation project by TARDA as well as Bura irrigation and settlement programme and the Hola irrigation scheme have been dormant for some time (Republic of Kenya 1997, 1999). With the planned revival of these major irrigation schemes as stated in the strategies of the Ministry of Regional Development (GOK 2003), abstraction of water for irrigation is likely to increase tremendously with Tana Delta Irrigation scheme being a major abstractor of water from the Tana River (GOK-TARDA 1982a, b, c). This would further have significant negative impacts on the downstream communities as well as on the marine environment since the type of irrigation being practiced does not promote efficient use of fresh water. While abstraction for domestic use has not been regulated in the rural areas, failure to regulate large-scale abstraction for irrigation is a serious problem considering the quantity of water involved.

In the upper course of the river there are private irrigation schemes using sprinkler irrigation. Small-scale schemes are based along Thanantu, Rubungazi, Thuchi, Thiba, Sagana, Thika, Yatta Canal, and Chania sub basins. The schemes are gravity-fed except lower Tana River where pumps or furrow and basin type irrigation methods are used. The minor irrigation schemes have been established by groups of farmers and are covering a total of 1,483 ha. The projects being implemented in the catchment area are highly intensive requiring substantial amounts of agro-chemicals to enhance the yields (CBS 1998).

Abstraction of Water for Domestic and Industrial Use

Abstraction of water for various purposes is a significant immediate cause of fresh water shortage and modification of river flow in the Tana basin. Management weaknesses with regard to large-scale water abstraction are evident. The Ministry of Water Resources Management and Development in Kenya is responsible for regulating water abstraction (GOK-JICA 1992; GOK 2003). Water is also abstracted to meet the high demand for freshwater for domestic and industrial uses in Nairobi City. To achieve this, the following dams have been constructed in the upper reaches of the river.

- Thika dam located at Ndakaini in Thika sub-county was completed in 1994 with a capacity of 77 million m^3 and linked to Chania River by 4 km tunnel
- Sasumwa dam located at Njambini, Nyandarua was constructed in two phases. The first stage was completed in 1955 and the second completed in 1968. The dam has storage capacity of 15.9 million m^3 and is on Sasumua River but receives Kiburu and Chania waters. The dam has a pipeline to Kabete 60 km. Currently the dam yields 52,80m^3/day but has a design yield of 59,000m^3/day.

Land Use Changes

Over the years, land use patterns have been shifting from the more traditional and sustainable agricultural practices and forest harvesting to less environmentally friendly practices. This shift is being attributed to increased population that exerts pressure on the scarce arable land. The rapid population increase in the Tana basin has led to increased demand for agricultural land with consequent unsustainable expansion of agricultural activities and human settlements in the catchment areas. Nationally, there is increasing concern over the rate at which forests and other vegetation that hold soil on the slopes are being cleared to pave way for agriculture and human settlement on the one hand, and to provide timber and fuelwood that are in high demand. The Tana River basin extends over areas of different agricultural potential with some areas being categorized as high, medium and low potential. According to GOK (1979 and 2003), the changes in land use practices are exemplified by decrease in the size of forestland due to conversion of forests to farmland for the cultivation of tea, coffee, and maize and for human settlement. It has been observed that in the 1920s, coffee and tea were grown on 10,000 and 5,000 ha respectively while at present; coffee and tea cover 500,000 and 150,000 ha respectively (Odingo 1971; GOK 1979, 1984, 1994; McMaster 1969; Othieno 1989). In addition, there is increased cultivation on steep slopes and riverbanks without applying basic soil conservation measures, and encroaching into the forest lands and other marginal areas. In the recent past, there has been extensive destruction of forests on the main water catchment areas of the Aberdare (Nyandarua) ranges and Ngong Hills. These changes in land use have led to serious soil erosion resulting in increased sediment loads in rivers. Similarly, there is the problem of overgrazing by the pastoralists who keep large herds of cattle in a fragile semi-arid environment, particularly in lower parts of the river basin.

Root Causes of Modification

Six root causes have been identified and classified under three broad categories as demographic, economic, and governance causes.

Demographic Causes

Demographic factors particularly high population growth and urbanization exerts pressure in the freshwater from the River Tana. This high population growth is due to both natural population growth and in-migration in the river basin. Major urban areas in Kenya fall within this river basin. A number of other significant towns such as Thika and Hola also exist. The growth in economic activities in these urban areas requires increased energy supply. High population has led to high demand for energy because economic activities depend heavily on energy supplies most of which is derived from hydropower. The Government of Kenya has since the mid 1960's constructed reservoirs on River Tana for this type of energy. These reservoirs impact negatively on downstream population especially during the dry seasons, as water has to be retained in the dams for generation of hydropower. In addition, rapid population leads to high dependency ratio thereby worsening the incidence of poverty. A large and increasing youthful population needs basic education and intensive health care thereby limiting the scope for investing in other productive activities. As a result, employment opportunities are not created and it is increasingly becoming more difficult to deal with serious environmental issues as population continues encroaching on the water catchments, forests, and marginal lands (Republic of Kenya 1997).

There is rising concern that rapid population growth, traditional forest clearing and shifting cultivation have resulted in fragmentation of forest despite the fact that traditional land use practices have maintained the ecological function of the River Tana in the past (Allison and Badjeck 2004).

Economic Causes

The economic root causes of modification of stream flow include economic growth in 1970s and 1980s and political instability, market forces of demand and supply, poverty, taxes and subsidies, and governance.

Economic Growth

Economic growth alongside changes in the political environment within the East African Community (EAC) which did not guarantee continued supply of energy from the Owen falls of Uganda, led to increased demand for a more secure source. Hence, the seven forks hydroelectric power generation project was established in the River Tana. At present, the domestically generated electricity plays a key role in satisfying commercial energy needs in the cities and urban centres. So far, the hydroelectric power provides 72 percent of the domestically generated electricity (World Bank 2003). It is to be noted however that Kenya is not still self sufficient in domestic generated electricity and there are frequent unplanned power shortages (GOK 2003). Similarly, to meet the growing national food requirements, there is increasing attention on expansion of irrigated agriculture (GOK 2003). At the same time, the Kenya government has been faced with the challenge of achieving the desired high economic growth rate, and irrigation has been viewed as a means of boosting the agricultural productivity. The NIB's Mwea rice irrigation scheme, TARDA rice irrigation scheme, and the Bura and Hola cotton irrigation schemes were established along the River Tana by the government towards achieving the targets for agricultural sector growth. Even though these public irrigation schemes have become dormant with the exception of Mwea rice scheme, investments of various magnitudes have been established targeting the lucrative horticulture industry. This industry has in turn boosted the Kenyan economy through foreign exchange earnings, mainly derived from external export market opportunities. However, the current water usage rates are inefficient due to low fees where such fees exist. As a result, crops like rice, which require large volumes of water (flood irrigation techniques) are grown with a water wasting technique instead of promoting irrigation of water efficient crops.

Poverty

Poverty manifests itself in many forms in the Tana River basin. It can be seen in small-scale farmers eking for survival on small parcels of land that can hardly provide the necessary agricultural output for sustenance, town dwellers that live in informal settlements without basic provisions such as water and survive from hand to mouth mostly based on daily contractual odd jobs. Poverty includes inadequacy of income and deprivation of basic needs and rights, and lack of access to productive assets as well as to social infrastructure and markets.

The 1997 welfare monitoring survey in Kenya estimated the absolute poverty line at Ksh.1, 239 (USD 15) per person per month and Ksh.2, 648 (US D 33) for rural and urban areas respectively (Republic of Kenya 2003). Because of the dehumanizing conditions of extreme poverty, the United Nations in its millennium resolutions agreed to place every effort to free the poor out of poverty (United Nations 2000). Development assistance is therefore to be given to countries that are genuinely making an effort to apply their resources to poverty reduction. Today, about 60% of the population living within this subsystem lives below this poverty line. As a result, many people are forced to engage in unsustainable farming practices, encroach into the water catchments, indulge in excessive use of fuel wood, and engage in unsafe sewage disposal (Republic of Kenya 2001). These activities have negatively affected the environment making the survival difficult. So far, there is increased soil erosion with sediments being transported downstream where the impacts are severe. Consequently, this has also resulted in over-exploitation of land and pollution of water resources. Water resources have been depleted, fuel wood exhausted and more time is now being spent to fetch these items from long distances.

Taxes and Subsidies

Even though the decision to invest in the generation of hydropower was driven by the changes in the political climate within the East African Community, the Kenya Government also saw this as an opportunity to generate the much needed revenue since the demand for power was guaranteed. As a result, the government gains through direct revenue paid to it as the leading shareholder and from the taxes collected from the corporation and users of electricity. The occasional provision of subsidies to the electricity generating company has also ensured sustained generation of hydropower. Currently, there are plans to expand hydropower generation in the river basin. This plan would only be implemented if the government of Kenya gives a subsidy to the Kenya Power Generation Company. The implementation of this plan would involve the establishment of new dams on the upper course of River Tana. One important site that has been identified for the construction of one more hydropower generation dam is at Mutonga/Grand Falls (Hirji et al. 1996; GOK 1979; GOK-TARDA 1982c). This may further impact adversely on the downstream communities.

Governance Causes

Poor Governance of Water Resources

Water resources in Kenya are managed under the Water Act of 2002 which deals with the conservation and controlled use of water resources. This Act provides for water-user rights which promote water abstraction for irrigation, domestic and industrial supplies. This legislation tackles issues pertaining to water resources management, water and sewerage development, and institutional framework and financing of the sector. Since this legislation promotes abstraction of large volumes of water for irrigation and other uses, this abstraction adversely impacts the poor rural farmers, pastoralists, and downstream fishermen.

Also relevant to river and river-basin management is the Tana and Athi Rivers Development Authority Act (Cap 443), which provides for the establishment of the Tana and Athi Rivers Development Authority (TARDA) to advise on the institution and coordination of development projects in the Tana River and Athi River basins and related matters. This includes the planning and development of the two rivers' basins and resources. However, the legal status of estuaries and deltas remains controversial, as they cut across several jurisdictions (riparian, forest, marine, coastal zone) and harbor abundant resources. There is little effective legal protection except under protected-area or forest-reserve regulations. The 1971 Ramsar Convention could be a primary instrument for the conservation of these ecosystems at the national level. In this regard, an application for appropriate Ramsar designation of the Tana Delta is under preparation. On the other hand, all intertidal zones are public, meaning that no beaches in Kenya are privately owned. However there is no single or specific legal instrument that relates to beaches, representing a weakness in the law (Government of Kenya 2008).

Conclusions

Modification of the flow regime of River Tana has both negative and positive impacts. These socio-economic impacts include conflicts between agriculturalists and pastoralists due to limited dry season pasture and watering points that lead to influx of large herds of cattle at the river delta with the pastoralists occasionally letting their cattle graze on crops in the farms. This has often resulted in serious confrontations and deaths. Floodplain agriculture has also reduced due to retention of floodwater in hydroelectric dams that are located upstream. Reduction in floodplain agriculture may adversely impact on agricultural output and food security in the Tana River basin. Furthermore, communities that live downstream have reported noticeable decline in river size as well as sizes of ox-bow lakes. This has had negative impacts on fish production from the ox-bow lakes and it is expected that additional dam construction will rapidly exacerbate this decline in fishing area as well as fish landings.

The immediate causes of modification of the flow regime of River Tana were damming for hydroelectric power generation, and abstraction of water for irrigation and public water supply. Behind these immediate causes are six root-causes that have been classified in to three broad categories namely: demographic, economic and governance causes. The demographic causes include high population growth rate and urbanization whereas the economic causes include instability in economic growth, poverty, taxes and subsidies. Governance causes include poor resource allocation and inadequate enforcement of policies and legislations.

The multiple impacts of modification of stream flow have tremendous effects on the welfare of people and therefore need to be addressed. It is recommended that investments should be channeled towards water treatment programs for the population in this area. A more long term monitoring study is needed on the rainfall patterns and stream flows which should also be integrated with a total natural resource monitoring of the basin to come up with appropriate integrated resource management of the basin for sustainable utilization and management of water and land resources.

Although there have been efforts to provide piped water from rivers and boreholes to various communities, there is still disparity in the provision of water in the rural and urban areas, with the urban areas being better off than the rural areas. A more proactive approach is required to bring water closer to the rural communities and at an affordable price. The arid areas still remain the most significant source of beef and other livestock products hence there is need to provide water services to reduce conflicts with other demanding sectors especially competition for watering points.

References

Agwata JF (2005) Water resources utilization, conflicts and interventions in the Tana basin of Kenya. FWU 3:13–23

Allison EH, Badjeck MC (2004) Livelihoods, local knowledge and the integration of economic development and conservation concerns in the Lower Tana River Basin. Hydrobiologia 527:19–23

Beck MW, Claassen AH, Hundt PJ (2012) Environmental and livelihood impacts of dams: common lessons across development gradients that challenge sustainability. Int J River Basin Manage 10:73–92. doi:10.1080/15715124.2012.656133

Bennun LA, Njoroge P (1999) Important bird areas in Kenya. The East African Natural History Society, Nairobi

CADP (1991) Tana River Wetlands Survey, Coast ASAL Development Programme, Ministry for the Reclamation and Development of Arid and Semi-Arid Lands and Water Resources, Mombasa

Canter L (1985) Environmental impact of water resources projects. Lewis Publishers, Michigan

Central Bureau of Statistics (CBS) (1998) Poverty in Kenya, first report Vol. 1. Incidence and depth of poverty. Office of the Vice President and Ministry of Planning and National Development, Government of Kenya, Nairobi

De Vos LD, Oyugi W, Nyingi DW, Nguku JK (2002) Assessment of fishes and fisheries in Tana River National Primate Reserve and its environs, Kenya. In: Gichuki NN, Oyieke HA, Handa C, Rotich DC, Dahiye Y, Mwangi S (eds) Research and monitoring of biodiversity in Tana River National Primate Reserve. GEF/World Bank – Tana River Conservation Project, National Museums of Kenya, Nairobi, pp 42–57

Doyle MW, Stanley EH, Luebke MA, Harbor JM (2000) Dam removal: physical, biological, and societal considerations. In: Hotchkiss RH, Glade N (eds) Proceedings of the 2000 joint conference on water resources engineering and water resources planning and

management, 30 July–2 August 2000. American Society of Civil Engineers, Minneapolis, pp 1–10
Emerton L (2003) Tana River, Kenya: integrating downstream values into hydropower planning. Case studies in Wetland valuation no. 6. IUCN, Nairobi, Kenya. http://www.iucn.org/themes/wani/publications/econ/CaseStudy06Tana.pdf (18/4/07)
Emerton L (2005) Values and rewards: counting and capturing ecosystem water services for sustainable development. IUCN water, Nature and Economics Technical Paper No. 1, IUCN - The World Conservation Union, Ecosystem and Livelihoods Group Asia, Colombo
Goldson J (1994) A study of farming systems in the middle and lower Tana Basin with special reference to their water requirements. Report prepared by Acropolis Kenya Ltd for Nippon Koei, Nairobi
GOK (1979) 1979–1983 development plan. Government Press, Nairobi
GOK (1984) 1984–1988 development plan. Government Press, Nairobi
GOK (1994) 1994–1996 development plan. Government Press, Nairobi
GOK (2003) Economic recovery strategy for wealth and employment creation 2003–2007. Government Printers, Nairobi
GOK-JICA (1992) National water masterplan. Government of Kenya (GOK) and Japanese International Cooperation Agency (JICA), Nairobi
GOK-TARDA (1982a) Tana delta irrigation project-feasibility study, vol III. Haskoning-Royal Dutch Consulting Engineers and Architects, Netherlands and Mwenge International Associates Ltd, Nairobi
GOK-TARDA (1982b) Tana Delta irrigation project-feasibility study, vol III – Hydrology, river morphology and flood plain hydraulics. Haskoning-Royal Dutch Consulting Engineers and Architects, Netherlands and Mwenge International Associates Ltd, Nairobi
GOK-TARDA (1982c) Tana Delta irrigation project-feasibility study, vol IV – Annex 5: irrigation, drainage and flood control. Haskoning-Royal Dutch Consulting Engineers and Architects, Netherlands and Mwenge International Associates Ltd, Nairobi
Government of Kenya (2008) Kenya State of the Coast Report: towards the integrated managementof Kenya's coastal and marine resources. UNEP and NEMA, Nairobi, 90 pp
Hirji R, Partoni FM, Rubin D (eds) (1996) Proceedings of the seminar on integrated water resources management in Kenya. The Economic Development Institute of the World Bank, Washington, DC, pp 15–129
IUCN (2003) Tana River, Kenya: integrating downstream values into hydropower planning. Case studies in wetland valuation #6. Tana River, Kenya
Maingi JK, Marsh SE (2002) Quantifying hydrologic impacts following dam construction along the Tana River, Kenya. J Arid Environ 50:53–79
Mavuti K (1994) Limnology and fisheries of the lower Tana River. Report prepared by Acropolis Kenya Ltd for NipponKoei, Nairobi
McMaster DN (1969) East Africa: influences and trends in land use. 204-263. In Prothero RM (ed) A geography of Africa: regional essays on fundamental characteristics, issues and problems. Routledge and Kegan Paul, London
Nippon Koei (1998) Executive summary for environmental assessment: feasibility study on Mutonga/Grandfalls Hydropower Project. Report prepared for JICA and Ministry of Energy, Kenya
Odingo RS (1971) The Kenya highlands: land use and agricultural development. East Africa Publishing house, Nairobi
Othieno C (1989) Research contributions to the development of the Kenya Tea Industry. In AAS Environmental crisis in Africa: scientific response. 57-69 NDP-1997-2001
Republic of Kenya (1997) The Tana River District Development Plan 1997–2001. Government Printers, Nairobi
Republic of Kenya (1999) Sessional Paper No. 1 of 1999 on National Policy on Water Resources Management and Development. Ministry of Water Resources, Nairobi
Republic of Kenya (2001) Poverty reduction strategy paper for the period 2001–2004. Government Printers, Nairobi
Republic of Kenya (2003) Geographic dimensions of well-being in Kenya – where are the poor? vol I. Central Bureau of Statistics, Nairobi
Republic of Kenya (2010) 2009 Kenya population and housing census, Volume IA Population Distribution by Administrative Units. Kenya National Bureau of Statistics, 206 pp
Susanne S-K (2004) The rationale of socio-economic research for the successful protection and use of wetlands: the example of participatory management approaches. Hydrobiologia 527:15–17
Terer TGG, Ndiritu N, Gichuki N (2004) Socio-economic values and traditional strategies of managing wetland resources in Lower Tana River, Kenya. Hydrobiologia 527:3–15
United Nations (2000) Resolution adopted by the General Assembly – United Nations millennium declaration. www.un.org/millenium/declaration/ares552e.pdf
VandenBossche JP, Bernacsek GM (1990) Sourcebook for the inland fishery resources of Africa, vol 1. CIFA technical report no. 18. 1. FAO, Rome,, 240 p
Wieczkowski J (2002) Changing its way. Swara 25(1):36–38
World Bank (2003) Towards a water-secure Kenya – water resource sector memorandum, vol 2. Unpublished working paper

Hydrodynamic Modelling on Transport, Dispersion and Deposition of Suspended Particulate Matter in Pangani Estuary, Tanzania

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Abstract

The present study was formulated with the aim of using MIKE 21 software in studying the hydrodynamic regime of the Pangani estuary. Water level, river discharge and wind drag force were used as hydrodynamic forcing factors during the model set up. The data set for the model (i.e. water level, tidal current winds and river discharge) were collected in Pangani estuary during the field campaigns conducted from December 2010 and August 2011. The results indicated that the tidal currents were relatively sluggish (0–0.05 m/s) in the beginning of model simulation. The ebb currents were established from 2 to 7 hours; originating from the inner part of the estuary tended to flow radially (Eastwards, Northwards and Southwards) soon after reaching the river mouth. The radial flow pattern of the ebb tidal currents seemed to be influenced by the funnel shape of the estuary. The flood tidal currents were established after 7 hours. The flood tidal phase started earlier on the southern part of the river mouth compared to the northern and tended to become more intensive on the northern part than on the southern part of the estuary. The currents pattern observed were influencing the transport and deposition of Suspended Particulate Matter (SPM). The maximum deposition of SPM preferentially occurred about 3 km north and south of the estuary mouth and the minimum deposition occurred in the middle of the estuary mouth. The deposition of SPM was highest during the southeast monsoon relative to the northeast monsoon. Approximately 872.6 kg/m^2/year of SPM were brought into the estuary. This implies that, in the long term, the SPM deposition along the river mouth will significantly change the Pangani hydrodynamic regime, from its present condition. Also infilling of navigational channel and alteration of the ecosystems is imminent. Urgent actions are required to minimize the generation of SPM within the Pangani river basin.

Keywords

Hydrodynamic modelling • Suspended particulate matter • Pangani estuary

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Introduction

The use of hydrodynamic models have been successfully applied in investigating engineering and environmental problems; including bathymetric charting (Long 2009), prediction of tides (Kregting and Elsaber 2014) transport and dispersion of pollutants (Gerritsen et al. 2000; Zhen-Gang 2008), prediction of oil dispersion pathway (Gritfull et al. 2009), eutrophication, assessing flood risk (Ghimire 2013; Timbadiya et al. 2013) and transport of suspended particulate matter (SPM) (Comerma et al. 2002; Hu et al. 2009). Due to their wide range of applicability, hydrodynamic models are therefore considered as useful tools for studying dispersion and transport of SPM in complex coastal systems (Kregting and Elsaber 2014). Estuaries which are defined as "semi-enclosed coastal water bodies which have free connections with the open seas and within which the salinity of the water is measurably different from the salinity of the open sea (Carmeron and Pritchard 1963; Cuff and Tomczak 1983; Priya et al. 2012)" are considered to be hydrodynamically complex coastal systems.

Estuaries are typically characterized by their strong gradients of water surface level, salinity, water temperature and suspended sediment concentration (Duck and Wewetzer 2001; Priya et al. 2012; Jay et al. 2014). Estuaries act as a transition zone between the upland wetlands and the ocean, and they are important nursery and feeding grounds for very large number of marine species (Bilgili et al. 2005; Rostamkhani 2015). Due to the increase in development pressure in coastal zones, estuaries have been acting as direct repositories of discharged contaminants and SPM (Bilgili et al. 2005; Silva et al. 2015). Unfortunately, the estuaries are not capable of assimilating SPM indefinitely. Therefore, predicting the transport and fate of pollutants in the estuaries is considered to be a challenging task (James 2002; Gleizona et al. 2003; Lopes et al. 2005; Gisen et al. 2015).

The coastal drainage system of Tanzania, which occupies 20% of the Tanzania land area is drained by several river networks discharging their water to the Indian Ocean (ASCLME 2012). The coastal rivers contribute about 50% of the total surface run off (Francis et al. 2001). Pangani river, with a mean annual discharge of 27 m^3/s is the fifth largest river along the coast of Tanzania (ASCLME 2012). The other four largest rivers along the Tanzanian coastal drainage system (Welcomme 1972; Hafslund 1980; Francis et al. 2001) with their mean annual discharges shown in brackets are: the Rufiji (900–1133 m^3/s), the Ruvuma (475 m^3/s), the Ruvu (63 m^3/s) and the Wami (63 m^3/s). Estuaries are among the major prominent features located at the mouths of these rivers (ASCLME 2012).

Climate changes and anthropogenic activities related to demand for water for irrigation and hydropower developments are considered to have significantly reduced the fresh water discharges of the above rivers at least during the last 50–60 years (IUCN 2003; Shaghude 2006; Duvail and Hammerlynck 2007; ASCLME 2012). Apart from the irrigation and hydropower developments, livestock developments and land use changes on the upper catchments of the above rivers have also contributed to the degradation of the river basins, with corresponding reduction in the fresh water discharges of the rivers. In some of the rivers such as the Pangani, Wami and Ruvu, the situation is considered to be critical with multipliable socio-economic conflicts and potential ecological and environmental impacts at the coast (IUCN 2003; Shaghude 2006).

Currently, there is insufficient information on how these estuaries respond to the changes resulting from the rivers upstream anthropogenic activities and climate changes. Particularly, there is a general lack of information on the pattern of transport of the discharged SPM to the estuaries and the Indian Ocean which boarders these estuaries. There is also a general lack of information on how the transport and dispersion pattern of the SPM discharged to the estuaries is related to the changing monsoon wind pattern as well as the tidal forcing from the Indian Ocean.

Although hydrodynamic modeling software such as ROMS, MIKE 21 (HD) and MIKE 3 are increasingly becoming commonly in predicting the hydrodynamic conditions of estuaries and bays (Zacharias and Gianni 2008; Webster et al. 2014), their usage in Tanzania is still at an infant stage. Noteworthy applications of ROMS in Tanzania include few studies on the Zanzibar Channel (Mayorga-Adame 2007; Garcia-Reyes et al. 2009) and the recent study conducted along the coast of Tanzania (Mahongo and Shaghude 2014). The applications of MIKE 21 in Tanzania include the study of hydrodynamics in estuaries (Mrema 2012) and determination of pollution dispersion pattern due to anthropogenic activities.

In view of the above presented examples on the usage of ROMS and MIKE 21 software in modeling coastal hydrodynamic processes in Tanzania coastal waters, it is becoming obvious that the availability of hydrodynamic modeling software like ROMS, MIKE 21 (HD) and MIKE 3 in Tanzania has opened a new path of studying hydrodynamics of coastal waters, and estuaries are not exceptions. The present study was therefore formulated with the aim of using MIKE 21 software in studying the hydrodynamic processes of the Pangani estuary and filling the knowledge gaps on how the changing monsoon pattern and tidal forcing influence the transport and dispersion pattern of the SPM discharge. The specific objectives of the study were: (i) to determine the magnitude and direction of tidal currents in the estuary (ii) to identify the weak current zone which in turn enhance the deposition of SPM (iii) to examine the influence of tidal current on erosion, transport and deposition of SPM in the estuary.

Methodology

Study Area Description

The Pangani River Basin, with an area of about 43,000 km^3 is one of the largest River basins along the Tanzania Indian Ocean drainage system (Fig. 1). Geomorphologically, the basin falls under two geomorphologic units, namely, 1- the highlands which rises from 1000–2000 m above sea level, characterized by rich volcanic soils abundant rainfall (1200–2000 mm per year), high biodiversity, intensive cultivation, urbanization and highly populated rural areas, with annual growth rates of >4%, and 2- the lowlands (also known as the "*Maasai steppe*") consisting of low sloping terrain, (generally less than 1000 m above sea level) descending to the coastal plain and characterized by relatively low rainfall (<500 mm per year) and low species biodiversity scattered (IUCN 2003). The Pangani Estuary which is located on the downstream part of the Basin is funnel shaped and meso-tidal, where the mean tidal range is about 3.9 meters (Sotthewes 2008). The river mouth of the estuary lies between Zanzibar channel to the South and the Pemba channel to the North. The climate of the entire Pangani coastal area is warm and humid with mean day-time temperature, ranging from 22 °C to 30 °C and an average precipitation rate of 1100 mm/year (UNEP 1998).

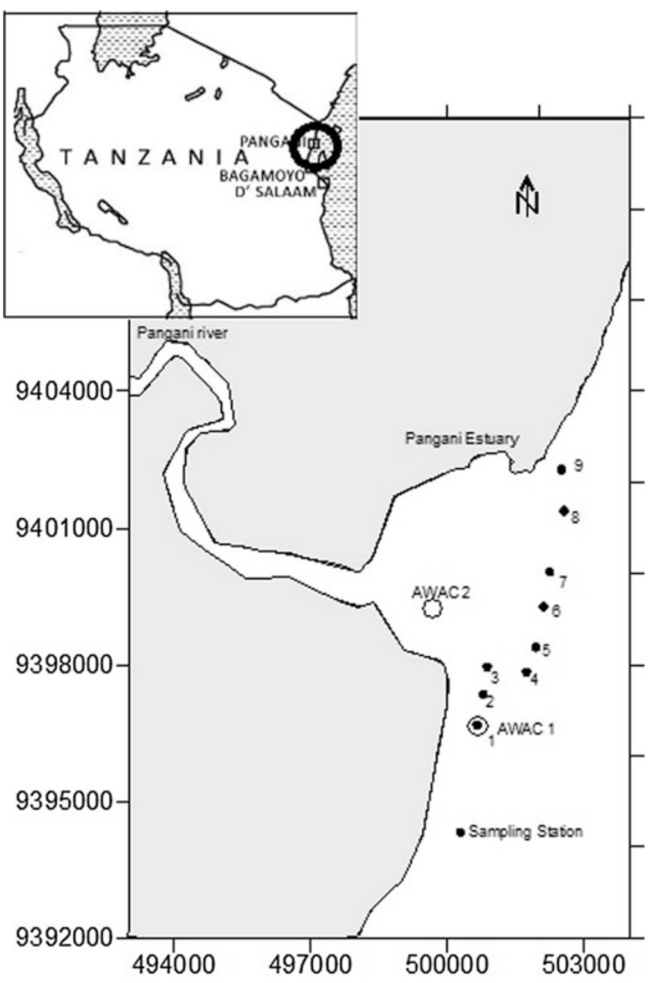

Fig. 1 A map of the Pangani estuary, showing the sampling stations for the SPM fluxes across the river mouth

The entire coastal of Tanzania is affected by the monsoon winds whose direction changes on seasonal basis and Pangani estuary is not an exception (Mahongo et al. 2012). According to Mahongo et al. (2012), the monsoon winds system, reverses in direction from SE (March to September) to NE (October to March). The former winds are associated with long rains and stronger ocean currents of up to 2 m/s, while the later are associated with short rains and weaker ocean currents (Newell 1959). From June to September or early October the winds are strongest and often stormy and the precipitation rates are minimal (Shaghude 2006; Mahongo et al. 2012). Wind speeds are usually weaker with highest variability in direction between October and December. The wind velocity data from Tanzania Meteorological Agency (TMA) located about 52 km north of the Pangani estuary shows that the wind speeds vary between 1.5 m/s and 8 m/s (average 5 m/s) during the period of the NE monsoon (Mahongo et al. 2012). However, during the SE monsoon season, the winds are generally stronger with an average of 8 m/s (UNEP 1998).

Sampling and Measurements of SPM Fluxes

Sampling and measurements of SPM fluxes were achieved through deployment of sediment traps for a period of one year, spanning from December 2010 to November 2011. The traps were constructed from local flowerpots and PVC pipes; with the internal diameter of about 100 mm and height of 500 mm, giving the aspect ratio of 1:5. The design of sediment traps were purposely for trapping SPM vertically from the sea surface. The mouth of each trap was placed at least 50 cm from the seafloor to trap vertical flux particles (English et al. 1994; Muzuka et al. 2010). Steel bars were used to fix the traps to the sea floor to avoid displacement by currents and waves. Funnels were placed at the top of each trap to retain the trapped SPM fluxes and to avoid the re-suspension fluxes due to tidal currents and to avoid intrusion by living organisms.

Nine sampling stations were established across the river mouth from South to North (numbered from 1 to 9 in Fig. 1) to enclose the entire river mouth but also to ensure that both the influence of SE and NE monsoons are covered. The selection of sampling stations took into consideration of the impacts of flood-ebb tidal phase variability along the main axis of the river. Sediment traps were retrieved and re-deployed on monthly basis. Retrieved SPM in traps, were carefully poured into the clean containers and then transported to the Institute of Marine Sciences (IMS) laboratory for further quantification and analysis.

In the laboratory, the samples from the sediment traps were transferred into plastic containers and left standing for a period of one week; allowing for fine particles to settle. The water in the container was removed using a syringe, to avoid sediment re-suspension. The water left in the containers, was allowed to evaporate under room temperature conditions. The sediments were finally oven-dried to constant weights at a temperature of 60°C. The dried samples, were weighed and then net sediment fluxes were determined according to English et al. 1994; Muzuka et al. 2010 (Eq. 1).

$$F = \frac{W_s - W_e}{A_{st} \times t} \tag{1}$$

Where: F = sediment flux; W_s = the weight of container with sediment; W_e = the weight of the empty container; A_{st} = Cross-section area of the sediment trap; t = time duration between sediments traps deployment and retrieval.

Measurements of SPM Concentration

Water samples for determination of SPM concentration were collected at 9 sampling stations along the longitudinal axis of the estuary (Fig. 1). The sampling was conducted between January and June, 2011 to encompass both dry and rain seasons. The samples were stored in plastic bottles and transported to the Institute of Marine Sciences Laboratory for further analysis. The SPM concentration in mg/l was determined gravimetrically following the procedures outlined by Strickland and Persons (1972). Water samples were filtered through pre-weighed 0.45 μm GF/F filter. Thereafter, the filters were rinsed with distilled water to remove salts and then dried to constant weight.

Measurements of Waves and Tidal Currents Velocities

Measurements of waves and tidal currents velocities were determine using a self-recording Acoustic Wave and Current (AWAC) profiler between December 2010 and April 2011. The profiler had a pressure sensor used to measure the tidal elevations. The AWAC was moored at two locations (station 1 and 2; Fig. 1). The AWAC was positioned at about 8 m water depth and the sensors were kept at an elevation of about 0.5 m above the seabed. The profiler was programmed to record the current velocity profiles after every 10 min. while the vertical current velocity profiles were recorded at 2 m intervals from the top of the sensor to the sea surface. The wave data was recorded at 1 hour intervals.

Hydrodynamic Model Set-Up

Hydrodynamic model was carried out using MIKE 21; where the bottom shear stress, wind shear stress, barometric pressure gradients, Coriolis force, momentum dispersion, evaporation, flooding and wetting were considered as forcing functions. For the purpose of model set up, the model domain was divided into several smaller model domains of velocity and momentum. The vertical velocity domain was calculated using continuity Eq. (2). Horizontal momentum in both x and y direction were calculated using momentum Eqs. (3) and (4), respectively. These Eqs. (2, 3, and 4) were integrated in the vertical direction to describe the flow and water level variation as described by Zacharias and Gianni (2008). For numerical stability, the model used an alternate direction implicit scheme in accordance with Duarte (2008).

$$\frac{\partial \xi}{\partial t} + \frac{\partial p}{\partial x} + \frac{\partial q}{\partial y} = 0 \tag{2}$$

$$\begin{aligned}&\frac{\partial p}{\partial t} + \frac{\partial}{\partial x}\left(\frac{p^2}{h}\right) + \partial\left(\frac{pq}{h}\right) + gh\frac{\partial \xi}{\partial x} + \frac{gp\sqrt{p^2+g^2}}{c^2h^2}\\ &\quad - \frac{1}{\rho_w}\left[\frac{\partial}{\partial x}(h\tau_{xx}) + \frac{\partial}{\partial y}\left(h\tau_{xy}\right)\right] - \Omega q - fVV_x\\ &\quad + \frac{h}{\rho_w}\frac{\partial}{\partial x}(pa) = 0\end{aligned} \tag{3}$$

$$\begin{aligned}&\frac{\partial p}{\partial t} + \frac{\partial}{\partial y}\left(\frac{q^2}{h}\right) + \partial\left(\frac{pq}{h}\right) + gh\frac{\partial \xi}{\partial x} + \frac{gq\sqrt{p^2+g^2}}{c^2h^2}\\ &\quad - \frac{1}{\rho_w}\left[\frac{\partial}{\partial y}(h\tau) + \frac{\partial}{\partial y}\left(h\tau_{xy}\right)\right] - \Omega q - fVV_y\\ &\quad + \frac{h}{\rho_w}\frac{\partial}{\partial y}(pa) = 0\end{aligned} \tag{4}$$

Where: τ_{xx}, τ_{xy}, τ_{yy} are components of effective shear stress; V, V_x, V_y are wind speed and components in x and y direction (m/s); Ω (x,y) is a Coriolis force, latitude dependent (m); ρa is atmospheric pressure (kg m^{-1} s^{-1}) is a density of water (kg/m^3); f(V) is a wind friction factor, C is Chezy resistance (m $^{1/2}$/s); g is acceleration due to gravity (m^2/s); t is time (s); p, q is flux densities in x and y direction (m^3/s/m); h is water depth (m), ζ is a surface elevation (m). The effective shear stress (τxx, τxy, τyy) in the momentum equations contains fluxes due to turbulence, vertical integration and sub-grid scale fluctuations. The formulation of the eddy viscosity in the equations was implemented based on the velocity as described in Eq. (5).

$$\text{Eddy viscosity} = \frac{\partial}{\partial x}\left\{h.E\frac{\partial u}{\partial x}\right\} + \frac{\partial}{\partial y}\left\{h.E\frac{\partial u}{\partial y}\right\} \tag{5}$$

Where u is the velocity (m/s) in the x- direction and h is the water depth (m) and E is the eddy viscosity coefficient. The eddy viscosity coefficient was specified using a time varying function of the local gradient in a velocity. The eddy viscosity coefficient (E) was calculated using Eq. (6) that puts into consideration of the Smagorinsky (1963) concept.

$$E = c_s{}^2\Delta^2\sqrt{\left(\frac{\partial u}{\partial x}\right)^2 + \frac{1}{2}\left(\frac{\partial u}{\partial y} + \frac{\partial v}{\partial x}\right) + \frac{\partial v}{\partial y}} \tag{6}$$

Where u, v are depth-averaged velocity components in the x and y-direction Δ is the grid spacing and c_s is a constant chosen in the interval of 0.25 to 1.0. The driving force due to wind was calculated from the following quadratic Eq. (7):

$$C_w\frac{\rho_{air}}{\rho_{water}}W^2 \tag{7}$$

Where: C_w is the wind friction coefficient ρ is the density and W is the wind velocity above the sea surface. Normally a wind friction coefficient of 0.0026 provides good results for moderate and strong wind in the open sea. For weak winds, however smaller coefficients can be used (Zacharias and Gianni 2008).

The bed resistance was estimated using Chezy number which is also a function of Manning number. The Chezy number and Manning number were estimated using Eqs. (8 and 9):

$$\text{Manning Number} = \frac{g.u.|u|}{C^2} \tag{8}$$

Where g is the acceleration due to gravity, u is velocity and C is the Chezy number. The manning number was converted to Chezy number using Eq. 9.

$$C = m.h^{1/6} \tag{9}$$

Where, m is the manning number. The units of Chezy (C) and Manning numbers (m) are m$^{1/2}$/s and m$^{1/3}$/s. respectively.

Table 1 Summary of parameters (and their values) used for the model set up.

Parameters	Values
Initial water level	0.23 m
Time steps	100 sec.
Number of Time steps	4500
Flood and ebb	Ebbing depth 0.2 m , Flooding depth 0.3 m
Wind friction coefficient	0.0026
Eddy viscosity	Smagorinsky formulation velocity based constant 0.5
Bed resistance	Manning number coefficient 32

Model Dataset

Water level, wind speed and direction and tidal currents were considered in the model simulation. These dataset were collected during the field campaigns conducted between December 2010 and August 2011. Data on wind speed and direction were obtained from the Tanzania Meteorological Agency (TMA) at Tanga station; the closest meteorological station is approximate 50 km to Pangani estuary. The water levels were recorded at each end of the open boundary; established in South, North, West and East of the estuary. The bathymetry setting of the model was accomplished by importing the navigational chart data, supplemented with echo soundings field measurements. During the echo-sounding survey, spot depth measurements were collected simultaneously with their corresponding geographic locations; recorded using a hand-held GPS. Processing of the bathymetric data included tidal corrections, followed by inspection of the actual bathymetric data. The true depths (corrected depths) were ranged from 1.5 to 20m. The data covered both upstream and downstream parts of the estuary described by the width of 20000 m and the length of 16000 m to ensure a uniform flow pattern at the model boundaries. Other parameters used for model set up are summarised in Table 1.

Model Calibration

The simulated tidal currents and water elevation in each open boundary were extracted and compared with their corresponding measured data. The model calibration in the present study involved three variables: bed coefficient, coefficient of eddy viscosity and the coefficient of wind friction. The variables were regulated to ensure that the simulated data agreed with the measured data. The eddy viscosity was regulated by selecting various values from a set of in built generated values, starting with a default value of 0.5 m^2/s. As the eddy viscosity approached 1.2 m^2/s, the simulated data showed a good agreement with the measured data. The bed coefficient was estimated using Manning numbers, which were assigned values ranging between 20 $m^{1/3}/s$ and 40 $m^{1/3}/s$ (The range recommended by DHI 2007). The wind friction coefficient was included in the model calibration as a constant value of 0.0026. The correlation between the simulated and the measured data was tested by linear regression analysis while their amplitude adjustment (Eq. 10) was quantified by the Root Mean Square (RMS).

$$\mathrm{RMS} = \left\{ \frac{1}{\mathrm{N}} \sum_{i=1}^{N} \left[\zeta_{\mathrm{obs}}(\mathrm{ti}) - \zeta_{\mathrm{model}}(\mathrm{ti}) \right]^2 \right\}^{1/2} \qquad (10)$$

$\zeta_{obs}(t)$ and $\zeta_{model}(t)$ are the observed and the modeled current velocity, N is the number of time series.

Results

Temporal and Spatial Variation of SPM Fluxes

The deposition of SPM preferentially occurred on either side (North or South) of the longitudinal axis of the river (Figs. 2, 3, 4, 5, 6, and 7), with minimum deposition observed at the middle parts of the river mouth, particularly at stations 4, 5, 6 and 7 (Figs. 2, 3, 4 and 5). The deposition of SPM increased on either side of the river mouth; forming a parabolic curve pattern with a maximum values occurring at a distance of about 3.3 km North /South of the central axis (Fig. 2). The observed pattern of SPM fluxes and its deposition from December 2010 to August 2011 revealed that in December 2010, the SPM fluxes were minimum (66 $gm^{-2}d^{-1}$) at the central part of the river mouth and increasing gradually with distance from the central axis on either side of the river mouth with a slight bias to the northern side ($R^2 = 0.79$, at confidence interval of 95%). The maximum SPM fluxes observed in December was approximately 1500 $gm^{-2}d^{-1}$ which occurred at station 9. In January 2011, the SPM fluxes resembled the pattern observed in December ($R^2 = 0.83$), but with the fluxes reduced from two to four times the fluxes observed in December (Fig. 3).

In February 2011, the SPM fluxes maintained the parabolic pattern ($R^2 = 0.82$) shown during the previous months but with a stronger bias towards the south (Fig. 4). The highest deposition of SPM fluxes (1800 $gm^{-2}d^{-1}$) were observed at station 2 located on the southern side of the river mouth and then decreased southward from station 2 to 9. Similarly in April 2011, the SPM fluxes maintained the parabolic pattern ($R^2 = 0.77$) shown by the previous months but with lower particles settling over most of the stations proximal to the main axis of the river mouth (Stations 3, 4,5,6,7 and 8) (Fig. 5). In June 2011, the SPM fluxes maintain the parabolic curve ($R^2 = 0.55$), with an almost symmetrical pattern about the central axis (Fig. 6). Some particles settled at the station located proximal to the middle of the river mouth (Stations 5 and 6); creating a central peak in addition to the other two peaks (Fig. 7). The site of highest deposition shifted from south in February to north in August. However the amount of

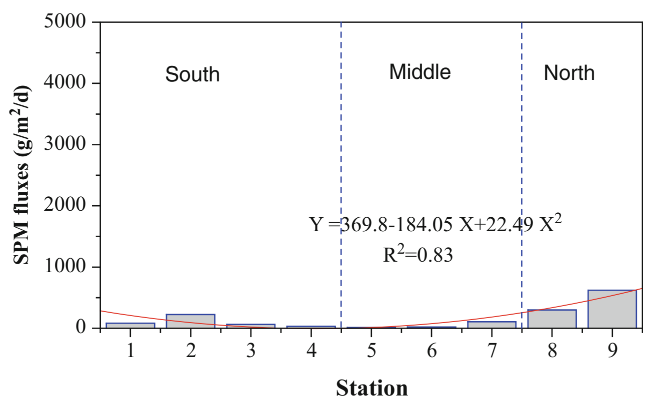

Fig. 3 Spatial variation of SPM fluxes observed on January 2011

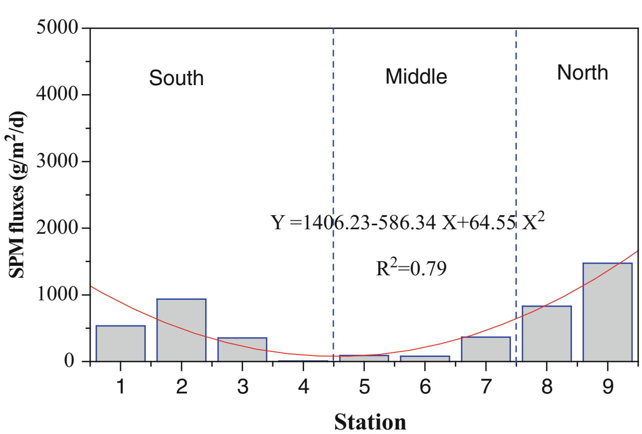

Fig. 2 Spatial variation of SPM fluxes observed in December 2010

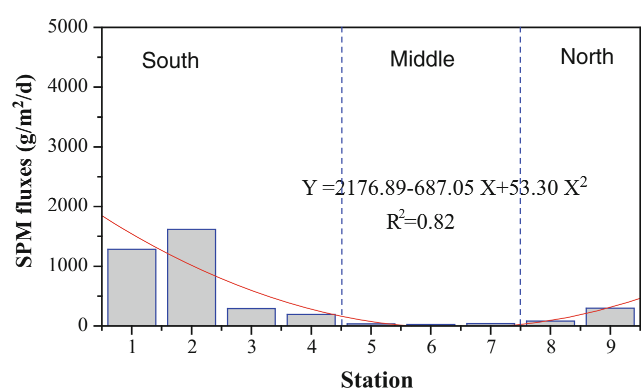

Fig. 4 Spatial variation of SPM fluxes observed on February 2011

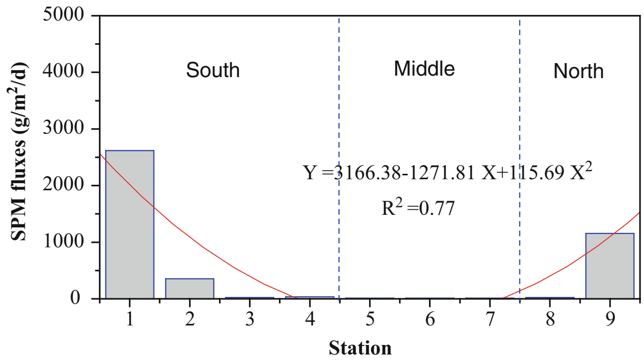

Fig. 5 Spatial variation of SPM fluxes observed on April 2011

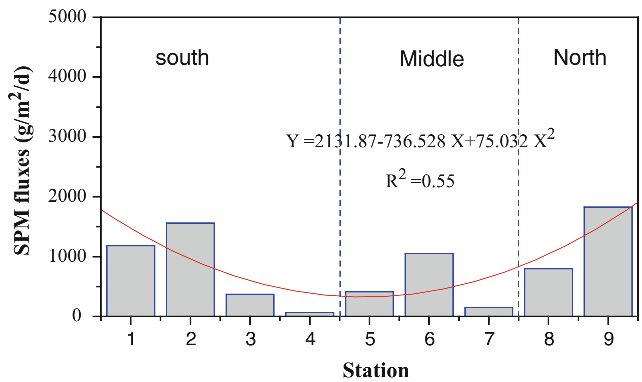

Fig. 6 Spatial variation of SPM fluxes observed on June 2011

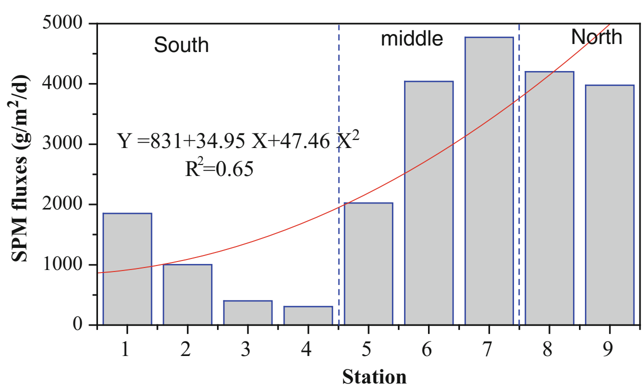

Fig. 7 Spatial variation of SPM fluxes observed on August 2011

SPM deposited in the northern part was relatively higher than in the southern part.

Comparison of Spatial Variation of SPM Between SE and NE Monsoon Wind

The SPM fluxes on the sampling stations (1–9) were compared between the NE and SE monsoon seasons (Fig. 8). In both sides of the estuary (northern and southern), the SPM fluxes were higher during the SE monsoon season compared to the NE monsoon season. The SPM fluxes on the southern parts of the estuary (Station 3 and 4) did not show major variations between the two seasons. Station 6, 7, 8 and 9, (located in the north of the estuary) received extremely SPM fluxes during the SE monsoon season than during the NE monsoon season (Fig. 8).

Tides and Waves and Mean Currents on the Pangani Estuary

Analyses of the tidal currents data revealed that tides are semi-diurnal with maximum spring tidal range of about 3.5 m and maximum neap tidal range of about 3.0 m. Analyses of the pattern of the mean current direction is summarized in Rose diagrams (Fig. 9). The results show that the mean currents on the River mouth changed with season. During January to March (representing the NE monsoon season) the mean currents flowed southwards (Fig. 9). During April (representing the SE monsoon season) the mean currents flowed northwards (Fig. 9). Moreover the mean currents during February and March were more variable (with relatively more components) than during January and April.

The monsoon induced waves in the Pangani estuary recorded from January to March are presented in Fig. 10. From January to nearly the end of February and from the 7th to 11th March, the wave heights were relatively high (>1.6m). From the end of February to 6th March the wave heights were relatively low (0.4 to 1.4m). The wave direction from January to April represented in Rose diagram, indicate that the main direction of waves was consistently southeasterly (Fig. 11). Analysis of the data on wave spectra (involving the direction and the frequency of waves) from January to April, 2011 revealed significant monthly variations (Fig. 12). In January the highest frequency of the wave occurred between 100° to 150° while in February there were two peaks, all centered between 100° and 150° (Fig. 12). The first peak was located at 0.1Hz while the second was located at about 0.3 Hz. In March there were two prominent peaks both located at 0.13Hz but in different directions (100° and 270°). In April the wave spectrum was characterized by two prominent peaks, the first one was highly variable in direction (0–360°) but centred at 0.1 Hz (Fig. 12). The second peak was centred at 0.3 Hz with a general direction of 100°.

River Discharge and SPM Concentration

The results of river discharge and SPM concentration are presented in Fig. 13. The results showed that minimum river

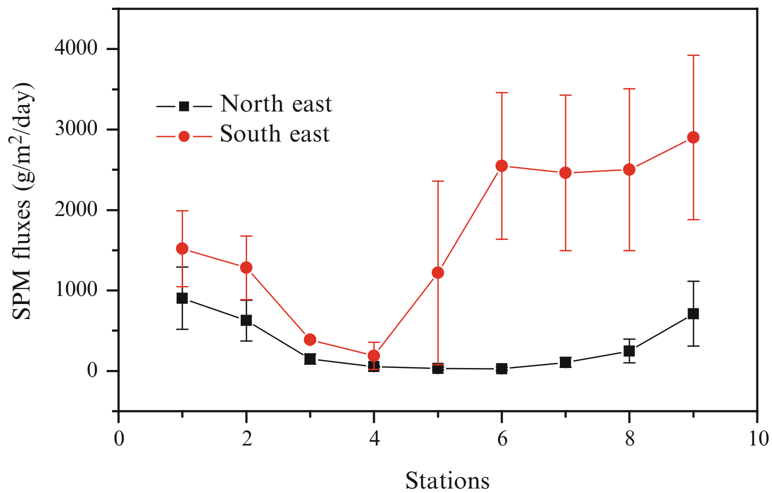

Fig. 8 Comparison of spatial variation of SPM fluxes during SE and NE monsoon

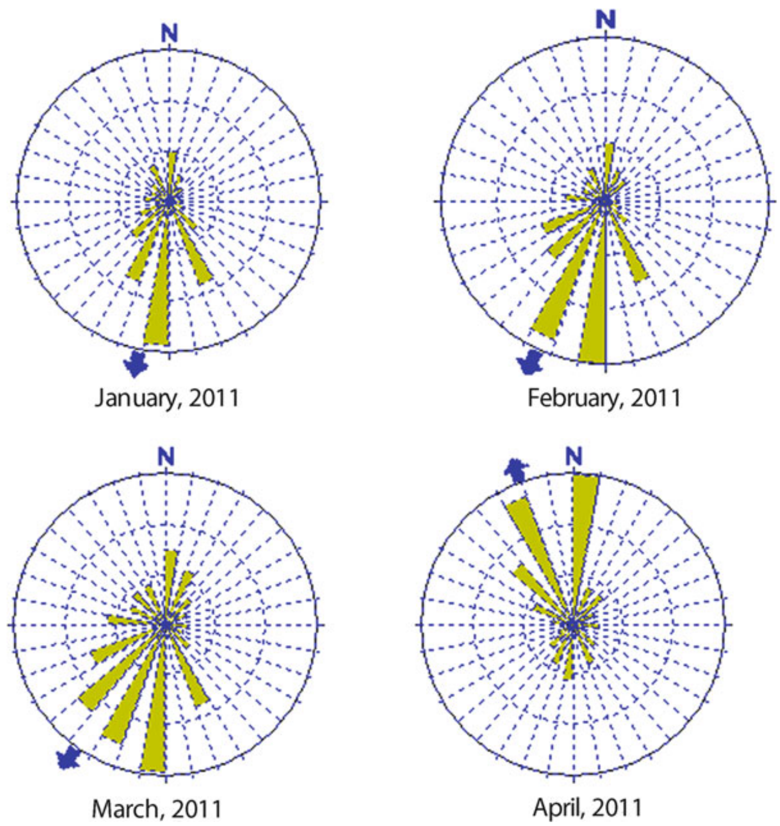

Fig. 9 Rose diagrams showing the direction of Currents from January to April 2011

discharge occurred between December and February (20 m^3/s) while maximum discharge (97 m^3/s) occurred in May (Fig. 13a). The SPM concentration in the estuary showed similar trends with that of river discharge data (Fig. 13b). The correlation between the river discharge and SPM concentration showed positive correlation relationship with $R^2 = 0.75$ at a confidence interval of 95% (Fig. 13c). The SPM was negatively correlated with time especially during April to December ($R^2 = 0.85$; Fig. 13d).

Hydrodynamic Model

Simulated and Measured Tidal Currents

The linear regression between predicted and observed currents for both boundaries (upstream and downstream boundaries of the estuary) showed strong positive correlation; where R^2 for the calibrated data were 0.91 and 0.92, respectively (Figs. 14 and 15). The RMS was generally low and satisfactory in both boundaries, ranging from 0.003 to 0.006.

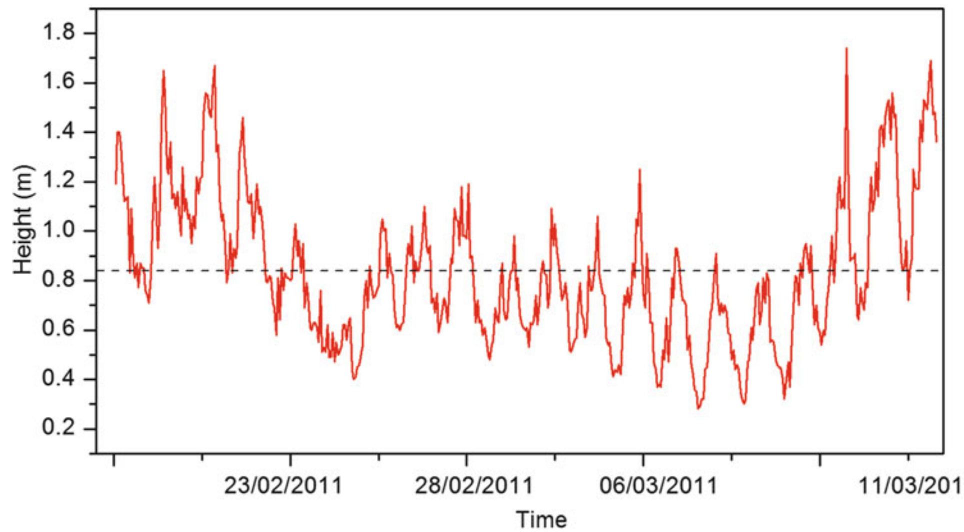

Fig. 10 Time series of wave height in the Pangani estuary: February–March 2011

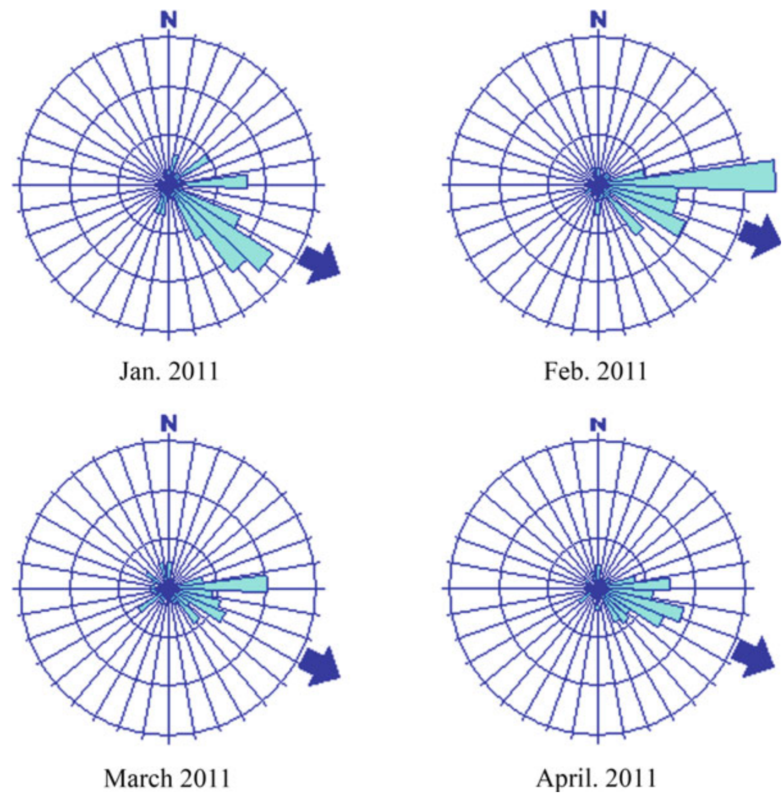

Fig. 11 The main wave direction recorded from January to April 2011

Simulation of the Tidal Currents in the Pangani Estuary

Initially the model simulation results showed relatively sluggish tidal currents (0–0.05 m/s) within the entire estuary (with the exception of areas located further offshore and further upstream parts of the river mouth) where the currents were relatively stronger; greater than 0.05 m/s. During the ebb tidal phase, the ebb currents from upstream side of the river mouth flowed seaward. However as they approached the river mouth, the ebb currents flowed radially, eastwards, northward and southwards up to at least the first two hours (Fig. 16a, b). During the third and fourth hour, the ebb currents became well established and became relatively stronger than during the first two hours. During this time span, highest intensification of ebb currents were observed along the coastal section on the northern parts of the river mouth. The maximum ebb currents occurred in the middle parts of the estuary, but further offshore and further upstream (towards the inner part of the estuary), the ebb currents became relatively weaker (Fig. 17a, b).

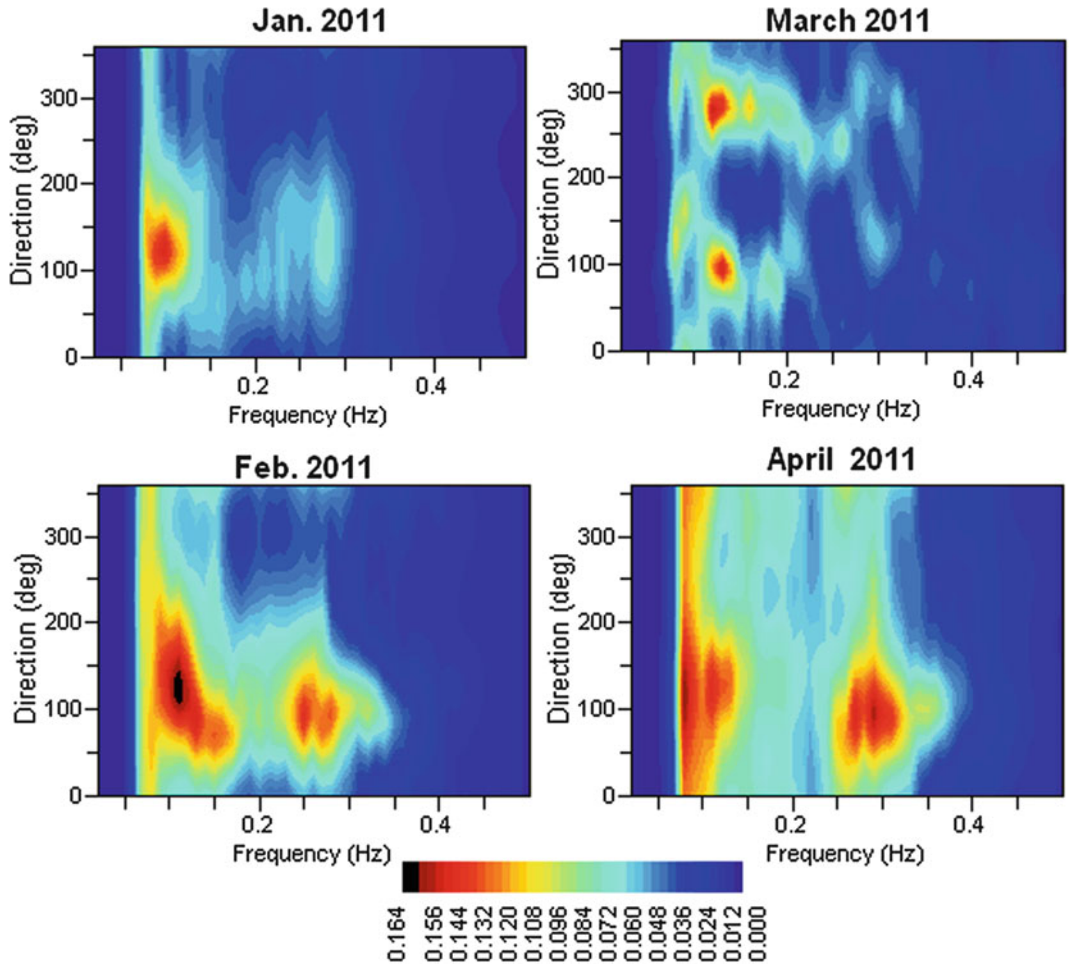

Fig. 12 Wave Spectrum from January to April showing the direction and frequencies

During the 5th and 6th hour of the model simulation, highest ebb currents were observed on the entire width of the river mouth, followed by relatively weaker currents further inshore and further offshore. During this time, a patch of flood tidal currents originating from the southern part of the river mouth, flowing northward were observed (Fig. 18a, b). The flood tidal currents became well established (particularly near the river mouth), during the 7th and 8th hour of the model simulation. The observed pattern of flood tidal currents suggest that the flooding tidal phase first starts on the southern part of the river mouth before it starts on the northern part of the river mouth (Fig. 19a, b). The highly intensified northward flood currents seemed to be more intensive on the northern parts (adjacent to the Pangani Town) than on the southern parts of the estuary opening (adjacent to Bweni village) (Fig. 19a, b). During the 9th and 10th hours, the flood currents were still highly intensified on the river mouth; flowing northward but became more sluggish on the outer parts of the estuary (Fig 20a). By the end of the 10th hour, the flood tidal currents became more intensified on the southern half of the river mouth compared to the northern half of the river mouth (Fig. 20b).

During the last two hours (11th and 12th) of the model simulation, the strong flood currents on the river mouth became extended further southwards and the northward flowing currents on the outer parts of the estuary were dominated by sluggish currents. Towards the last hour, the northward flowing currents were relatively stronger on the near shore parts of the estuary (with the exception of the estuarine mouth) and sluggish tidal currents were observed further offshore and on the inner parts of the estuary (Figs 21a, b.)

Variation of SPM Fluxes during the Northeast Mooson Season

The SPM fluxes within the Pangani estuary was determined during the NE monsoon season (January and April). The results indicated that, SPM fluxes were relatively higher in

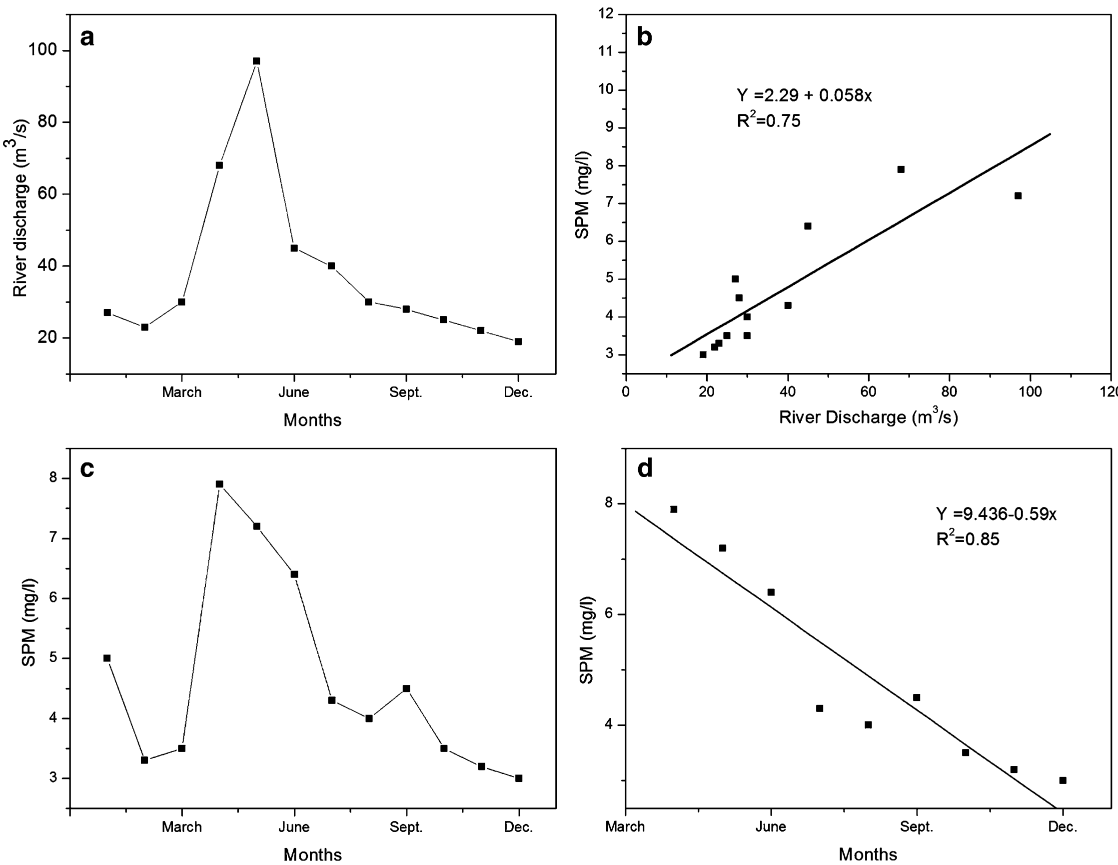

Fig. 13 (**a**) Monthly discharge of Pangani River (**b**) Correlation between River discharge and SPM concentration (**c**) Monthly SPM concentration (**d**) Correlation of SPM concentration as decreased from April to December

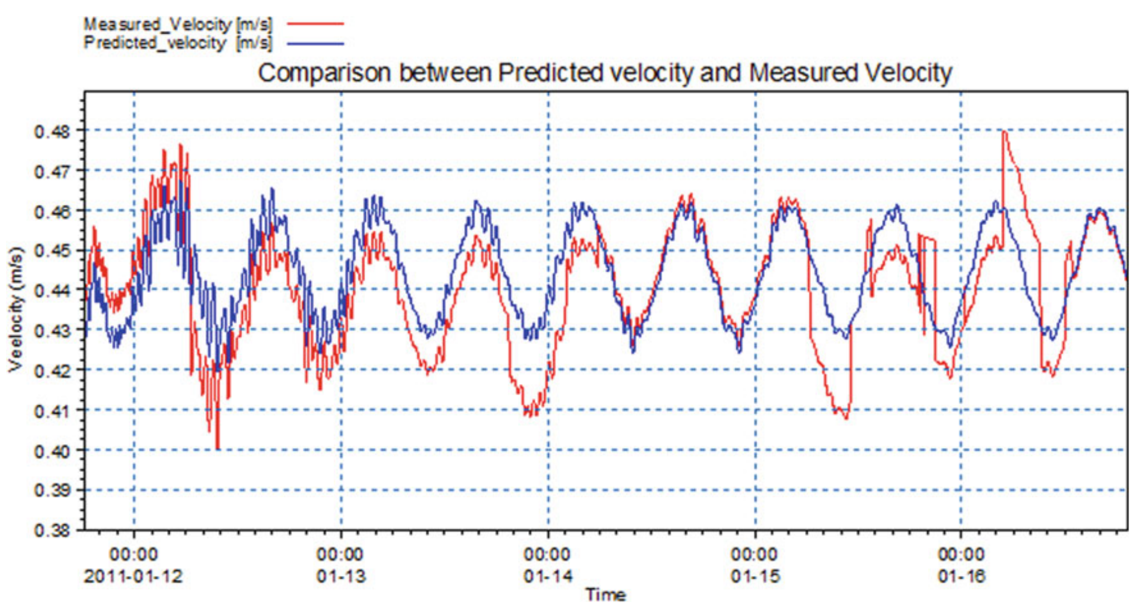

Fig. 14 Computed and Measured Current velocity in upstream after calibration procedures

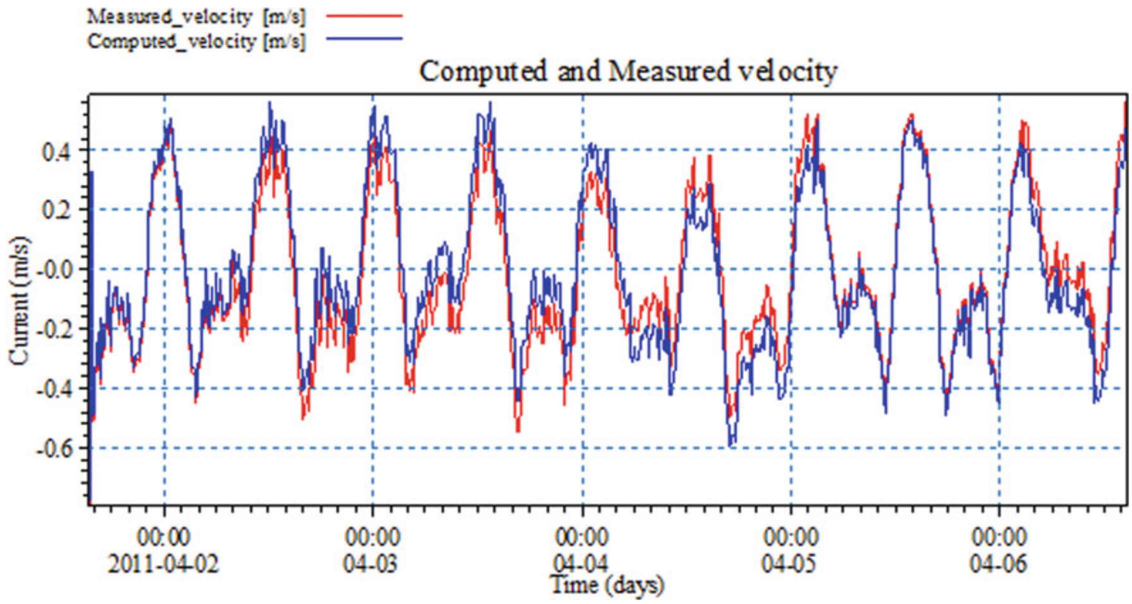

Fig. 15 Computed and Measured Current velocity in downstream after calibration procedures

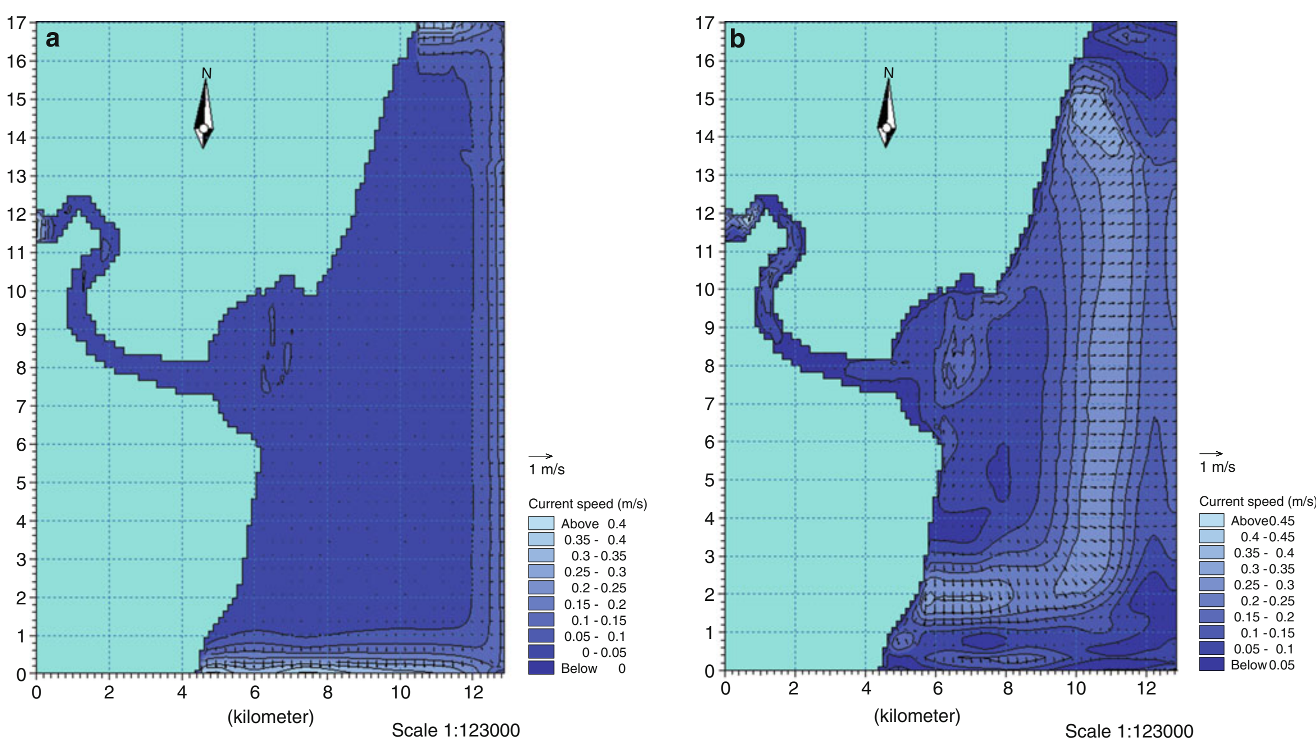

Fig. 16 The simulation of ebb currents in the Pangani estuary (**a**) After 1 h (**b**) After 2 h

April than January (Fig. 22a, b). Deposition of SPM on the estuary varied from one trap to another with preferential deposition on the northern parts of the estuary. The SPM fluxes in January and April ranged from 50 g/m^2/d to 1804 g/m^2/d, with a peak at the river mouth, however during April the peak extended further northward. During both periods (January and April), the deposition of SPM, indicate that the main direction of SPM transport was northward (Fig. 22).

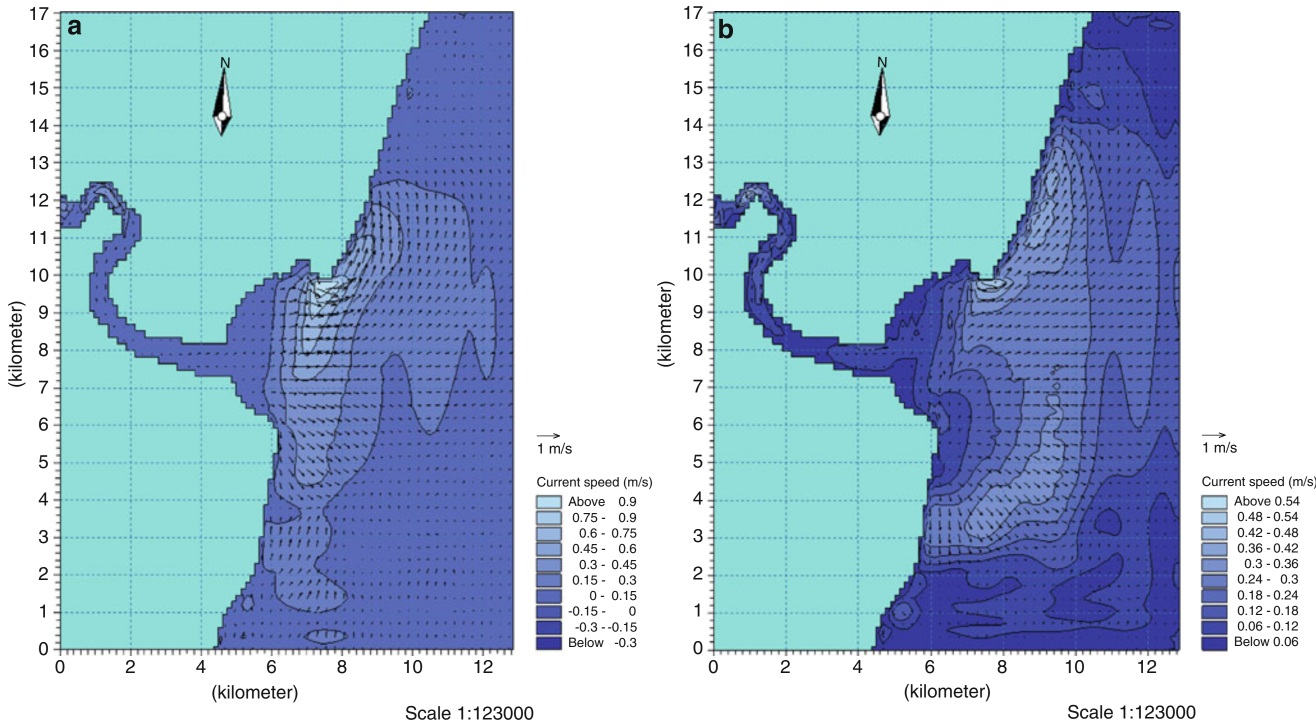

Fig. 17 The simulation of ebb currents in the Pangani estuary (**a**) After 3 h (**b**) After 4 h

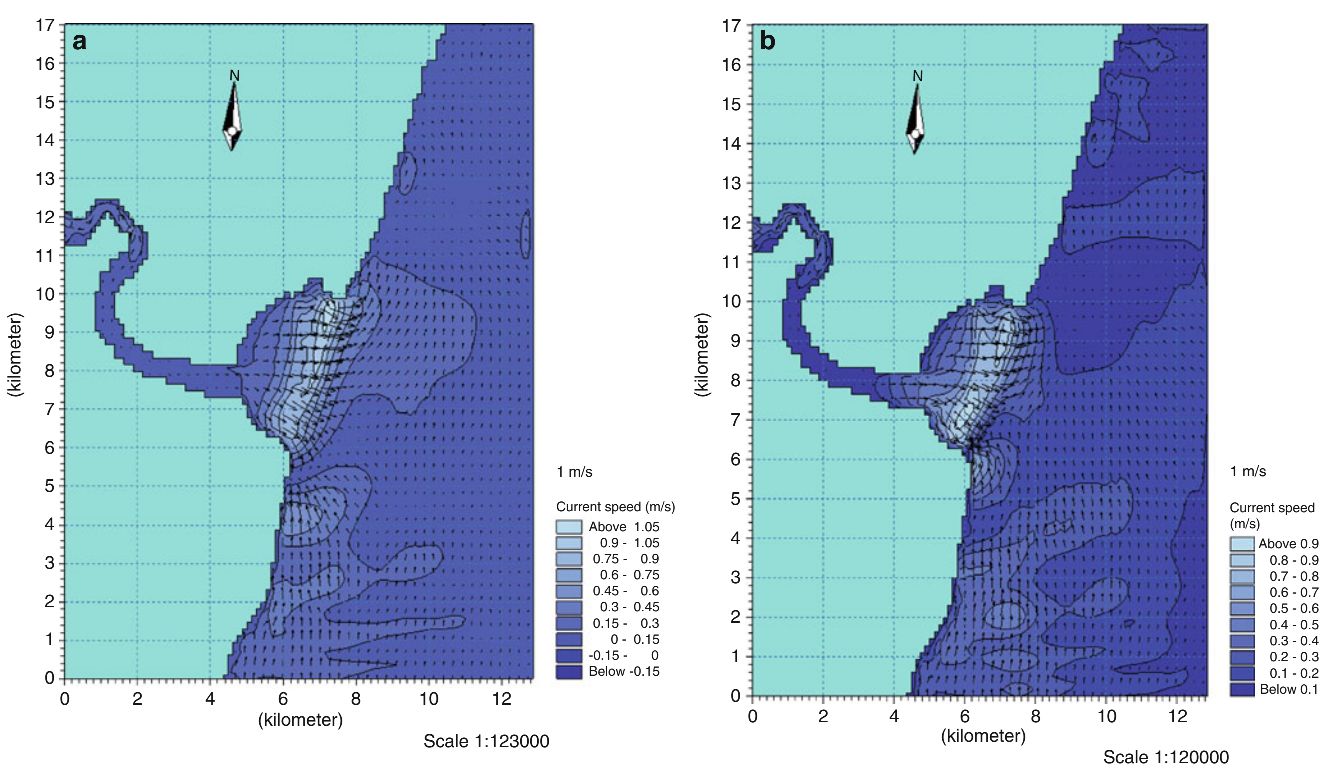

Fig. 18 The simulation of ebb currents in the Pangani estuary (**a**) After 5 h (**b**) After 6 h

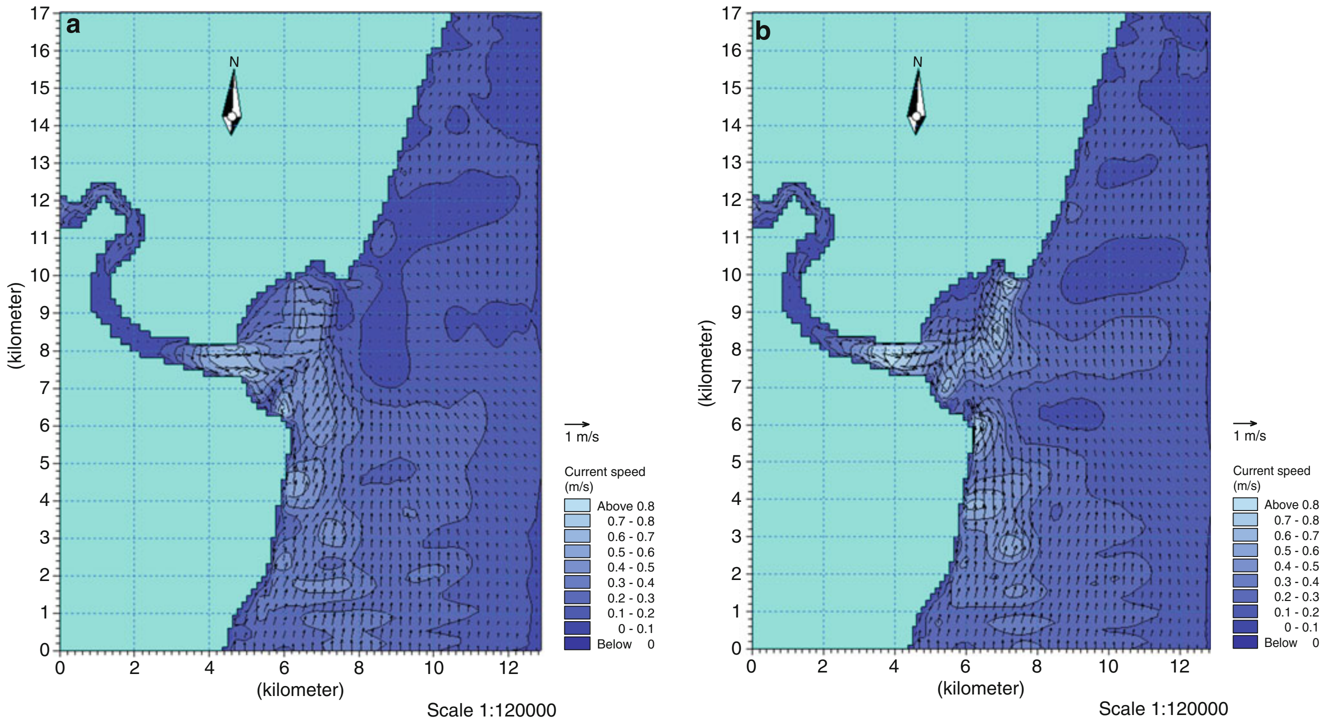

Fig. 19 The simulation of flood currents in the Pangani estuary (**a**) After 7 h (**b**) After 8 h

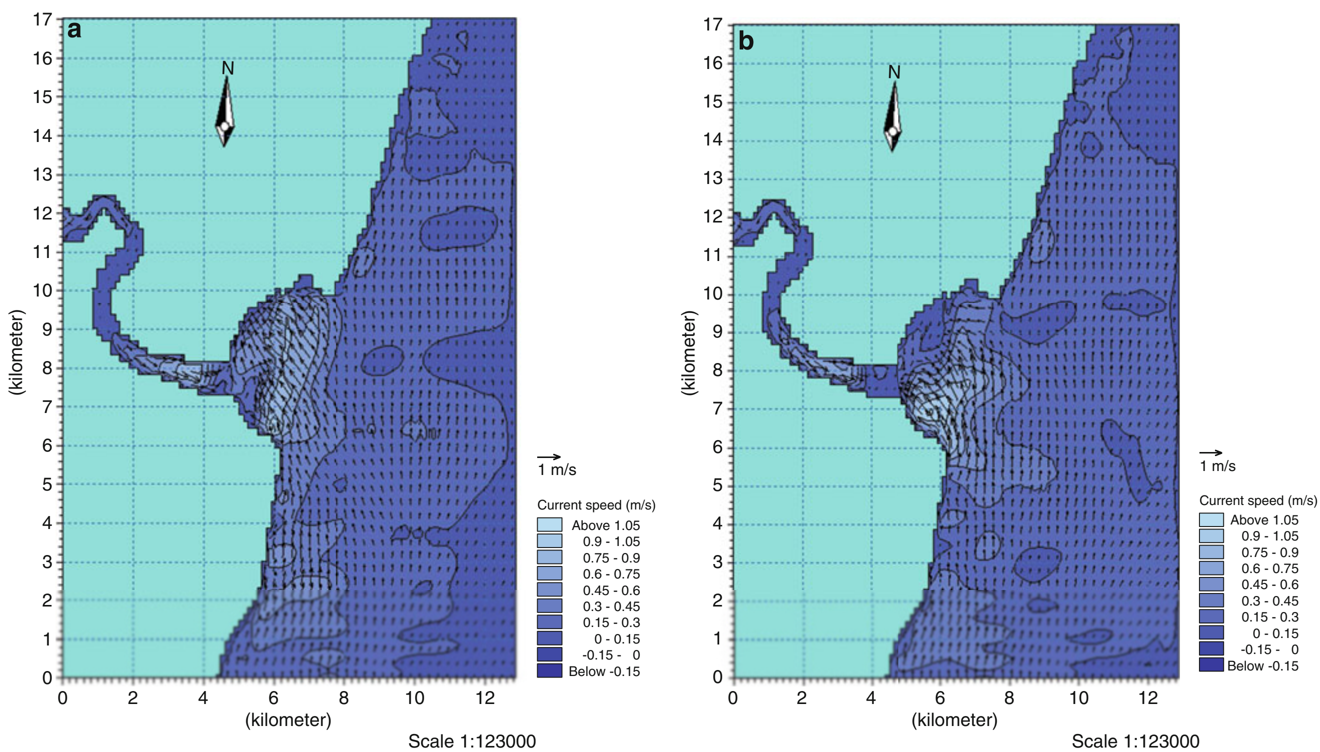

Fig. 20 The simulation of flood currents in the Pangani estuary (**a**) After 9 h (**b**) After 10 h

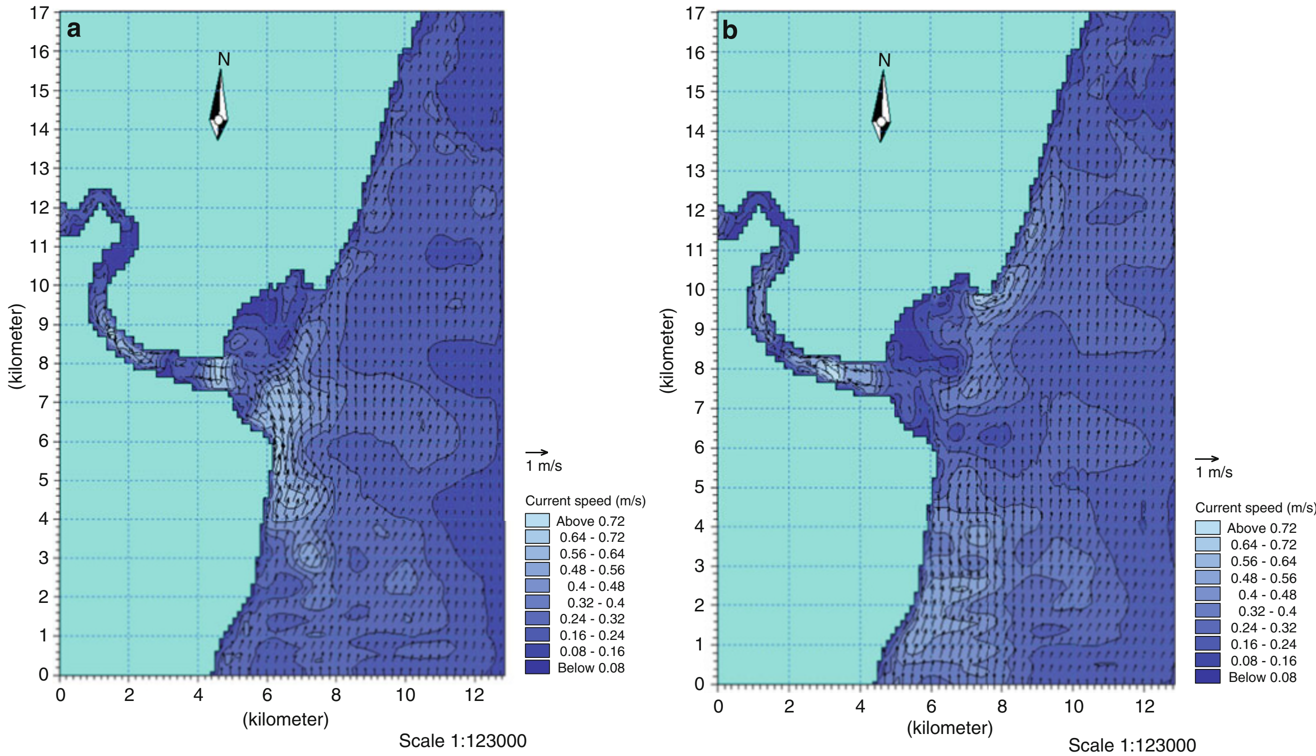

Fig. 21 The simulation of flood currents in the Pangani estuary (**a**) After 11 h (**b**) After 12 h

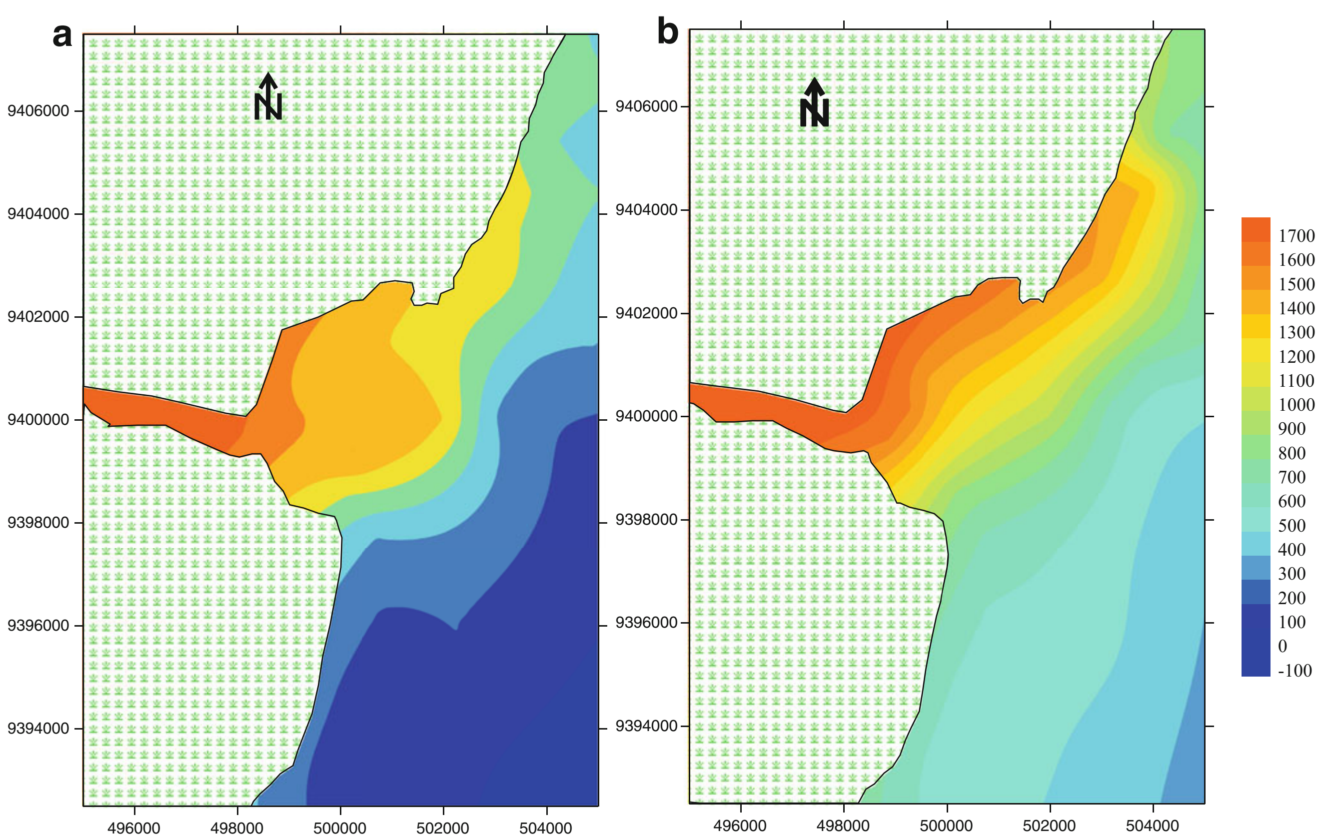

Fig. 22 Contour maps showing the SPM deposition inferred from the sediment traps collections during (**a**) January and (**b**) April

Discussion

Spatial and Temporal Variation of SPM Fluxes in the Pangani Estuary

The results presented in Figs. 2, 3, 4, 5, 6, and 7 revealed that the deposition of SPM increased gradually on either side of the River mouth forming a parabolic curve pattern with a minimum value at the middle of the River mouth. The observed pattern has not been reported elsewhere in Tanzania. The observed pattern of SPM fluxes could be attributed to several factors including, change of currents direction in response to monsoon winds and variation in Pangani River flow discharges. The presented results on SPM fluxes on the Pangani Estuary are consistent with other studies conducted along the coast of Tanzania, which show that, sediments transport along the coast are influenced by the seasonal monsoon wind pattern (Muzuka and Shaghude 2000; Shaghude et al. 2003; Mahongo et al. 2012). The observed high SPM fluxes in the southern part of the river mouth during the NE were attributed to the fact that during the NE monsoon season sediments are preferentially transported southward. In response of the changing wind pattern from NE to SE, the SPM fluxes also shifted to the northern parts of the river mouth during the SE monsoon season. In the middle part of the estuary, the SPM deposition were relatively low and this was attributed to the strong tidal and river flow currents which is maximum in the middle parts of the River mouth. The current speed observed on the middle part of the river mouth was as high as 0.5 m/s during ebb phases. The currents of this magnitude are strong enough to prohibit the deposition of fine grained particles in estuaries and coastal water (Liu et al. 2009; Muzuka et al. 2010; Silva et al. 2015).

SPM fluxes were relatively higher on the northern part of the river mouth than on the southern part. This is because SE monsoon winds are relatively stronger compared to the NE monsoon winds (UNEP 1998; Mahongo et al. 2012). Similar Pattern of deposition of SPM have been reported in the southern North sea and Mandovi estuary (India) where the influences of maximum turbidity shifted from one part of the estuary to another depending on the seasonality pattern of the winds. According to Bailey et al. (2011); the SPM concentration changes with direction of winds; increasing in the direction of winds and decreasing in opposite direction of the winds.

River Discharge and SPM Concentration

The variation of Pangani River discharge observed for the whole period of this study is related to the seasonal pattern of rainfall variability. Pangani River Basin is characterized by a bi-modal rainfall regime with short rain period from October to December (with a peak in November) and long rain season spanning from March to May with peak in April (Kabanda and Jurry 1999; Zorita and Tilya 2002; Kijazi and Reason 2009; Mahongo and Shaghude 2014). The Pangani river discharge data shows that the peak river discharges occur in May. The observed time lag between the precipitations tend to be infiltrated through the thick vegetation cover on the mountains before they could be drained to the Pangani River. According to Kabanda and Jurry (1999) and McSweeney et al. (1998), it is predicted that precipitations on the Pangani River Basin will decrease under the influence of the global climate change, which will in turn affect the River sediment delivery to the beach. This could lead to starvation of the beach in terms of sediment supply, thus resulting into enhanced shoreline erosion.

Hydrodynamic Regime of the Pangani Estuary

Simulation of Tidal Currents

The hydrodynamic regime of the Pangani estuary is considered to be influenced by three natural forcing namely; the tidal currents, freshwater outflows from the Pangani river and the monsoon wind generated currents. The simulated tidal currents during the ebb tidal phase (Figs. 17 and 18), tended to flow radially, that is both eastwards, northwards and southwards up to at least during the first four hours. The radial flow pattern could be attributed to the funnel shape of the estuary. This is because the funneling effects enhance the magnitude and the direction of the tidal currents (Rao et al. 2011). On the upstream side of the estuary, the cross sectional area of the estuary is narrower compared to the estuarine mouth. Along the narrower parts of the estuary, the water would tend to flow at high speeds which would be reduced further downstream on the wider parts of the estuary, at the estuary mouth. The funnel like morphology of the estuary would therefore tend to induce a radial flow pattern towards the open sea.

The model simulation results showed that highest intensification of tidal currents were confined within a narrow zone located on the river mouth, while weaker tidal currents occurred further offshore and further upstream of the river mouth. Moreover tidal currents tended to be stronger on the northern coastal section of the river mouth adjacent to Pangani Town and Kigombe village than on the southern coastal section of the river mouth adjacent to Bweni village. The observed model simulated results are consistent with the observed pattern of shoreline erosion at Pangani (Shaghude 2004). However the study of Shaghude (2004) attributed most of the observed shoreline changes (particularly coastal erosion) to monsoonal wind generated waves. The present study revealed that apart from waves, the tidal currents were also among the major contributing factors to the observed shoreline changes at Pangani. The present model simulated results further revealed that flooding of the estuary started from the southern parts before the northern parts, suggesting that there is a tidal phase difference between the southern parts of the estuary and the northern parts. This could probably be attributed to the observed difference of the bathymetry between the northern coastal section and southern coastal section of the investigated area.

The Dominant Forcing of the Pangani Estuary Hydrodynamics and SPM Fluxes

Deposition of SPM on the Pangani estuary preferentially occurred on the northern parts of the estuary than on the southern parts (Fig. 22). The observed low deposition of SPM on the southern parts of the estuary relative to the northern parts is probably due to the fact that the northward transport of SPM fluxes is more favored than the southward transport due to the dominance of the SE monsoon winds over the NE monsoon winds along the coast of Tanzania (Muzuka and Shaghude 2000; Shaghude et al. 2003). The relationship between SPM deposition and the hydrodynamic regime of estuaries has been studied by Wolanski et al. (1996), Uncles and Stephens (1997) and Azevedo et al. (2010). The results of the present study are consistent with the observations reported at Douro estuary where the change of estuarine hydrodynamics affected the dispersion of SPM originated from runoff (Azevedo et al. 2010). The estuarine hydrodynamics involve, the tidal currents, river discharge and wind regime; all of these affects the transport and dispersion pattern of SPM (Burchard 2008; Webster et al. 2014).

Although the model simulation in this study was based on the tidal currents, other studies (e.g. Flemer and Clamp 2006; Lane et al. 2007; Burchard 2008; Liu et al. 2009; Azevedo et al. 2010) showed that the variation of the river discharge also affects the deposition and dispersion of SPM in estuary. For instance Azevedo et al. (2010) showed that stable flow of the river discharge seem to be more effective on the dispersion of SPM compared to unstable flow. However, in the case of the Pangani estuary, the River forcing is relatively low, especially during the dry season. During the peak of the long rain season (in April) the River forcing cannot be neglected. The wind forcing is another important factor influencing the deposition of SPM. Lane et al. (2007) showed that the wind forcing tend to re-suspend the deposited SPM, bringing them back to the water column. In the present study, the observed preferential deposition of SPM on the northern parts of the estuary are being attributed to the dominance of the SE monsoon winds over the NE monsoon winds.

The Relationship between the Pangani Estuary Hydrodynamic Regime and Shoreline Changes and Deposition of Sediments

The funnel shaped nature of the Pangani Estuary, its orientation and the absence of a well-defined delta (depo-centre) are among the most important physical attributes characterizing the hydrodynamic regime of the Pangani estuary. The simulated model results revealed that the tidal currents were highly intensified proximal to the river mouth. Such hydrodynamic conditions would tend to accelerate shoreline erosion over time and would inhibit formation of the delta on the river mouth. Furthermore, the study of wind climate regime along the coast of Tanzania show that the coastal winds have intensified during the last three decade (Mahongo et al. 2012; Shaghude et al. 2012) suggesting that the wave climate regime have also changed over the last three decades. The intensification of waves could also inhibit the formation of a delta at the Pangani estuary. Finally, upstream anthropogenic activities associated with abstraction of river water by irrigation and impoundment of river water at Nyumba ya Mungu dam are also considered to have significant reduced the river sediment loads being discharged to the Pangani estuary. There is therefore no doubt that the present sea bottom morphology of the Pangani Estuary is being controlled by the hydrodynamic regime of the Pangani estuary.

The ebb currents seem to play a significant role in transporting the sediments brought by the Pangani River. The simulated model results revealed that the currents during the ebb tidal phase were very strong and with a general northward flow, approximately parallel to the shoreline. These currents are considered to play a significant role in shore erosion of the northern banks of the Pangani River mouth. In addition to tidal currents, Shaghude (2006) noted that the shore erosion on the northern banks of the Pangani River mouth was highly attributed by waves particularly during the NE monsoon season. The wind generated waves may also influence sediment re-suspension (Capo et al. 2006; Rostamkhani 2015).

The Long-Term Implication of the Observed Hydrodynamic Regime of the Pangani Estuary

The SPM fluxes per year at each station were estimated to determine the average deposition of SPM in the estuary (Fig. 23). The results indicated that approximate 872.6 kg/m^2/year of SPM are being deposited to the estuary, of which 394 kg/m2/year are being deposited northwards (comprising stations 6 to 9), while 292 kg/m^2/year are being deposited southwards (comprising stations 1 to 3) and 96.6 kg/m^2/year (comprising stations 4 and 5) are being deposited at the middle of the river mouth.

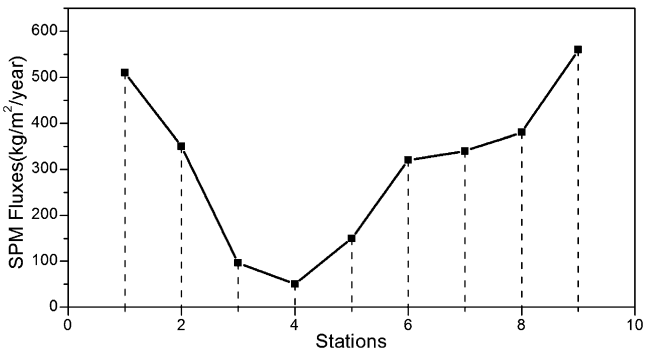

Fig. 23 SPM fluxes deposition per year in station 1–9 across the estuary mouth

The presented results suggest that in the long term, the SPM deposition pattern in the estuary mouth will significantly change the Pangani river mouth hydrodynamics regime from its present condition. The deposition of SPM fluxes which shows preferential deposition on the northern parts (station 6 to 9) would results to significant changes of the northern parts of the estuarine bottom topography. This could potentially lead to more turbulence conditions on the northern parts of the estuary as the incoming waves would tend to break as they approach the shallow estuarine topography. Marine vessels calling to the Pangani harbour (Town) could only access the harbour from the southern parts of the estuary. Thus if the current SPM deposition would persist for a long time marine vessels calling to Pangani harbour (Pangani Town) or marine vessels leaving the Pangani harbour must take the necessary precautions to ensure maximum safety.

Conclusion

The deposition of SPM in the Pangani Estuary showed a parabolic curve depositional pattern with maximum deposition located at about 3.3 km north or south of the central axis of the Estuary, and the minimum deposition occurring at the centre of the Estuary. The maximum SPM deposition occurred between May to August and the minimum deposition occurred from December to February. The spatial and temporal variation of SPM observed in Pangani Estuary was influenced by several factors including, monsoon wind, river discharge and tidal currents. The SPM deposition observed north or south of the Estuary was related with the seasonal pattern of the monsoon winds. Although deposition of SPM occurred at both the southern and northern parts of the estuary mouth, preferential deposition occurred on the northern parts, and this was attributed to the fact that the SE monsoon winds are stronger than NE monsoon winds (Newell 1959; Mahongo et al. 2012). The minimum deposition of SPM observed at the centre of the estuary was influenced by a combination of tidal current and river discharge. Along the central axis of the estuary, water flows at relatively high speeds, thereby prohibiting the deposition of SPM. The study also have revealed that, approximately 872.6 kg/m^2/year of SPM fluxes are being brought to the estuary: 394 kg/m^2/year being transported northward, 292 kg/m^2/year being transported southward and 96.6 kg/m^2/year at the centre of the estuary. It implies that for a long run, the SPM deposition in the Pangani estuary will change the hydrodynamic regime from its present condition.

In addition, the model simulation showed that the strongest tidal currents in the Pangani Estuary occurred on the northern coastal section of the river mouth adjacent to Pangani Town and Kigombe village compared to the southern coastal section of the river mouth adjacent to Bweni village. The flooding tidal currents from the northern parts of the river mouth were lagging behind the flooding tidal current from the southern parts of the river mouth. The ebb tidal currents that originated from the inner part of the estuary tended to flow radially soon after reaching the river mouth. The radial flow pattern of the tidal currents seemed to be influenced by the funnel shape nature of the estuary. The intensification of the tidal currents on the river mouth tended to intensify shoreline retreat and inhibited the formation of a delta off the Pangani river mouth. Apart from the observed natural factors the hydrodynamic regime of the Pangani estuary are also influenced by the upstream anthropogenic activities associated with abstraction of river water by irrigation and impoundment of river water at Nyumba ya Mungu dam.

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References

ASCLME (2012) National marine ecosystem diagnostic analysis (MEDA): Agulhas and Somali Current Large Marine Ecosystems Project. Available at http://asclme.org/report2013/national%20MEDIA/mozambique/_MozambiqueMEDAProof.pdf

Azevedo IC, Bordalo AA, Duarte PM (2010) Influence of river discharge patterns on the hydrodynamics and potential contaminant dispersion in the Douro estuary (Portugal). Water Res 44:3133–3146

Bailey A, Chase TN, Cassano JJ, Noone D (2011) Changing temperature Inversion characteristics in the U.S Southwest and relationship to largescale atmospheric circulation. J Appl Meteor Climatol 50:1307–1323

Bilgili M, Prochi JA, Lynch DR, Smith WK, Smith MR (2005) Estuary or ocean exchange and tidal mixing in a Gulf of Marine Estuary, a Lagranging modeling study. Estuar Coast Shelf Sci 65:607–624

Burchard H (2008) Combined effects of wind tide and horizontal density gradients on stratification in estuaries and coastal seas. J Phys Oceanogr 39:2117–2136
Capo S, Sottolichio A, Brenon I, Gastaing P, Ferry L (2006) Morphology, hydrography and sediments dynamics in mangrove estuary: the Konkoure estuary. Guinea Mar Geol 230:199–215
Carmeron WM, Pritchard D (1963) Estuaries. In: Hill MN (ed) The sea, vol 2. Wiley, New York, pp 306–324
Comerma E, Espino M, Sánchez-Arcilla A, González M (2002) Forecasting oil pollution in harbour engineering. In: Proceedings of the 29th international conference on coastal engineering. American Society of Civil Engineers (ASCE), pp 1242–1253
Cuff WR, Tomczak M (1983) Synthesis and modelling of intermittent estuaries. Springer, Berlin/Heidelberg/New York
DHI (2007) MIKE 21: sediment transport and morphological modelling – user guide. DHI Water and Environment, Hørsholm, p 388
Duck RW, Wewetzer SFK (2001) Impact of frontal system on estuarine sediment and pollutants dynamics. Sci Total Environ 266:23–31
Duvail S, Hammerlynck O (2007) Rufiji river flood: plague or blessing? Int J Biometeorol 52:32–42
Duarte P (2008) A three dimensional hydodynamic model implemented with ecodynamo Available at: https://bdigital.ufp.pt/dspace/handle/10284/834
English S, Wilkinson C, Baker V (1994) Survey manual for tropical marine resources. Australian Institute of Marine Sciences, Townsville, p 358
Flemer DA, Clamp MA (2006) What is the future given nutrient over-enrichment, freshwater diversion and low flows. Mar Pollut Bull 52:247–258
Francis J, Wagner GM, Mvungi A, Ngwale J, Sallema R (2001) Tanzania national report, phase I: integrated problem analysis. Unpublished report: GEF MSP Sub-Saharan Africa Project on Development and Protection of the Coastal and Marine Environment in Sub-Saharan Africa, 60 pp
Garcia-Reyes M, Mayorga-Adame G, Moulton MR, Nadeau PC (2009) Modelling of the Zanzibar channel. Unpublished report, Institute of Marine Sciences, Zanzibar., 13 p
Gerritsen H, Robert JV, Theo VK, Andrew L, Boon JG (2000) Suspended sediment modeling in a shelf sea (North Sea). Coast Eng 41:317–352
Ghimire S (2013) Application of 2D hydrodynamic model for assessing flood risk from extreme storm events. Climate 1:148–162
Gisen JIA, Savenije HHG, Njzink RC (2015) Revised predictive equations for salt intrusion modelling in estuaries. Hydrol Earth Syst Sci Discuss 12:739–770
Gleizona P, Punta AG, Lyonsb MG (2003) Modelling hydrodynamics and sediment flux within a macrotidal estuary: problems and solutions. Sci Total Environ 314–316:589–597
Gritfull M, Fontán A, Ferrer L, Manuel M, Espino GM (2009) 3D hydrodynamic characterisation of a meso-tidal harbour: the case of Bilbao (northern Spain). Coast Eng 56:907–918
Hafslund AS (1980) Stiegler's Gorge power and flood control development. Project planning, Unpublished report volume 1 Oslo Report to RUBANDA
Hu KL, Ding PX, Wang ZB, Yang SL (2009) A 2D/3D hydrodynamic and sediment transport model for the Yangtze Estuary, China. J Mar Syst 77:114–136
IUCN (2003) Eastern African Programme. The Pangani River basin: a situation analysis, The World Conservation Union; ScanHouse Press Ltd, Nairobi. p 104
James ID (2002) Modelling pollution dispersion, the ecosystem and water quality in coastal water: a review. Environ Model Software 91:99–164
Jay DA, Leffler K, Diefenderfer HL, Borde AB (2014) Tidal-fluvial and Estuarine processes in the lower Columbia River: I, a long-channel water level variations, Pacific ocean to Bonneville Dam. Estuar Coasts 38:415–433. doi:10.1007/s12237-014-9819-0
Kabanda TA, Jurry MR (1999) Inter-annual variability of short rains over northern Tanzania. Climate Res 13:231–241
Kijazi AL, Reason CJC (2009) Analysis of the 2006 floods over Northern Tanzania. Int J Climatol 29:955–970
Kregting L, Elsaber B (2014) A hydrodynamic modelling framework for Strangford Lough Part 1: tidal model. J Mar Sci Eng 2:46–65
Lane RR, Day JW, Marx BD, Reyes E, Hyfield E, Day JN (2007) The effects of riverine discharge on temperature salinity suspended sediments and chlorophyll a in a Mississippi delta estuary measured using a flow-through system. Estuar Coast Shelf Sci 74:145–154
Liu WC, Chen WB, Cheng RT, Hsu MH, Kuo AY (2009) Modeling the influence of river discharge on salt intrusion and residual circulation in Danshuei River estuary, Taiwan. Cont Shelf Res 27:900–921
Long JW (2009) Modeling shallow-water hydrodynamics: rotations, rips, and rivers: a dissertation submitted to Oregon State University
Lopes JF, Dias JM, Cardoso AC, Silva CIV (2005) The water quality of Ria de Aveiro lagoon, Portugal: from the observations to the implementation of a numerical model. Mar Environ Res 60:594–628
Mahongo SB, Shaghude YW (2014) Modelling the dynamic of the Tanzanian Coastal water. J Oceanogr Mar Sci 5:1–7
Mahongo SB, Francis J, Osima SE (2012) Wind patterns of coastal Tanzania: their variability and trends. West Indian Ocean J Mar Sci 10:107–120
Mayorga-Adame CG (2007) Ocean circulation of the Zanzibar channel: a modeling approach. IMS, Zanzibar, p 8
McSweeney C, New M, Lizeano G (1998) Tanzania model projection of climate changes: report UNDP climate change country Profile. http://country-profiles.Geog.ox.ac.uk
Mrema LE (2012) Characteristics of water circulation in Mnazi Bay, Tanzania. Msc. thesis, University of Dar es Salaam
Muzuka ANN, Shaghude YW (2000) Grain size distribution along Msasani beach, north of Dar es Salaam habour. J Afr Earth Sci 30:417–426
Muzuka ANN, Dubi AM, Muhando CA, Shaghude YW (2010) Impact of hydrographic parameters and seasonal variation in sediment fluxes on coral status at Chumbe and Bawe reefs, Zanzibar, Tanzania. Estuar Coast Shelf Sci 89:137–144
Newell BS (1959) The hydrography of British East African coastal waters. Col Off Fish Pubs Lond 12:1–18
Priya KL, Jegathambel P, James EJ (2012) Hydrodynamic modelling of estuaries-A-state-of-art. Int J Environ Sci 3:223–240
Rao PV, Shynu R, Kessarkar PM, Sundar D, Michael GS, Navekar T, Blossom V, Mehra P (2011) Suspended sediment dynamics on a scale in the Mandovi and Zuari estuaries, Central west coast of India. Estuar Coast Shelf Sci 91:78–86
Rostamkhani HM (2015) A novel approach to flow and sediment transport estimation in estuaries and bays. http://pdxscholar.library.pdx.edu/open_acess_etds
Smagorinsky J (1963) General circulation experiment with the primitive equations. Monthly Weather Rev 91:99–164
Shaghude YW (2004) Shore Morphology and sediment characteristics south of Pangani River, coastal Tanzania. West Indian Ocean J Mar Sci 3:93–104
Shaghude YW (2006) Review of water resource exploitation and land use pressure in the Pangani river basin. West Indian Ocean J Mar Sci 5:195–207
Shaghude YW, Wannas KO, Mahongo SB (2003) Biogenic assemblage and hydrodynamic setting of the tidally dominated reef platform sediments of Zanzibar channel. West Indian Ocean J Mar Sci 1:107–116
Shaghude YW, Mburu JW, Nyandwi N, Magori C, Ochiewo J, Sanga I, Arthurton R (2012) Beach sand supply and transport at Kunduchi in Tanzania and Bamburi in Kenya. West Ind Ocean J Mar Sci 11:135–154

Silva PD, Lisboa PV, Fernandes EH (2015) Changes on fine sediments dynamic of the Port of Rio Grande expansion. Adv Geosci 39:123–127

Sotthewes W (2008) Forcing the salinity distribution in the Pangani estuary. Final report, Delft University of Technology, p 85. Available at www.citg.tudelft.nl/fileadmin/Faculteit/.../Sothewes_2008.pdf

Strickland JDH, Parsons TR (1972) A practical handbook of seawater analysis. Bulletin 167, 2nd ed. Fisheries Research Board of Canada, Ottawa, 310 p

Tanaka A, Deleersnijder E (1996) Three-dimensional Island Wakes in the field, Laboratory experiments and numerical models. Continental Shelf Res 16:1437–1452

Timbadiya PV, Patel PL, Porey PD (2013) One dimensional hydrodynamic modelling of flooding and stage hydrographs in the lower Tapi River in India. Curr Sci 106:708–716

Uncles RJ, Stephens JA (1997) The freshwater-saltwater interface and its relationship to the turbidity maximum in the Famar estuary, United Kingdom. Estuaries 16:126–141

UNEP (1998) Maziwi Island off Pangani (Tanzania) history of its destruction and possible causes: UNEP regional seas report and studies number 139. Available at http://iwlearn.net/publications/regional-seas-reports-and-studies-no-139/view

Webster T, McGuigan K, Collins K, McDonald C (2014) Integrated river and coastal hydrodynamic flood risk mapping of the LaHave river estuary and town of Bridgewater, Nova Scotia, Canada. Water 6:517–547

Welcomme RL (1972) The inland waters of Africa. Les eaux interieures d'Afrique-CIFA teach pap/Doc. Tech. CPCA 1:117p

Wolanski E, Asaeda T, Tanaka A, Deleersnijder E (1996) Three-dimensional island wakes in the field, Laboratory experiments and numerical models. Contl Shelf Res 16:1437–1452

Zacharias I, Gianni A (2008) Hydrodynamic and dispersion modeling as a tool for restoration of coastal ecosystems. Application to a re-flooded lagoon. Environ Model Software 23:751–767

Zheng-Gang J (2008) Hydrodynamic and water quality: modelling rivers, lakes and estuaries. Wiley, Hoboken, 702 pp

Zorita E, Tilya FF (2002) Rainfall in Northern Tanzania in March- May Season (long rains) and its links to large scale climate forcing. Climate Res 20:31–40

Tanzanian Reef Building Corals May Succumb to Bleaching Events: Evidences from Coral-*Symbiodinium* Symbioses

Leonard Jones Chauka

Abstract

Coral reefs are among the most vulnerable ecosystems to current trends of climate change. Most of the reef systems along the coast of Tanzania have remained severely damaged following the 1997/1998 El-Niño that caused a massive coral bleaching, resulting into a wide spread of coral death. It is important therefore to find out/establish whether reef building corals develop adaptations to current trends of climate change so as to prioritise their conservation. There are evidences that coral-*Symbiodinium*-symbioses develop adaptation to current trends of climate change. This review therefore was meant to compare coral-*Symbiodinium* symbioses that occur along the Tanzanian coast with those occurring in others parts of the world. Like in most parts of the world, reef building corals along the Tanzanian coast are dominated by *Symbiodinium* clade C3 which is both thermal and irradiance intolerant. In the Tanzanian coast, coral genera that in most part of the world have been found to host clade D, the *Symbiodinium* type whose distribution is correlated with warmer environment, host other *Symbiodinium* clades. Unlike in most part of the world, most of Tanzania's reef building corals lack polymorphic symbioses, a phenomenon that is hypothetically believed to render environmental tolerance to the holobiont. This is probably due to low seasonal variation in both temperature and solar radiations. Thus, Tanzanian corals become less advantaged in terms of impacts that may be associated with current trends of climate change.

Keywords

Tanzanian coast • Reef building coral • Estuarine • Coral bleaching • *Symbiodinium* types • Climate change

Introduction

Coral reefs, often referred to as 'the rain forests of the ocean' due to their high biodiversity (Reaka-Kudla 1995), play a key role in the functioning of tropical coastal ecosystems through their ability to shelter a huge diversity of sessile and free-living organisms. Although coral reefs cover only less than 0.5% of the world's sea floor (Spalding and Grenfell 1997), they provide critically important goods and services to over 500 million people worldwide, mainly through fisheries and tourism industries (Moberg and Folke 1999).

Regarding fishery, coral reefs provide suitable reproductive spawning habitats and nursery ground to most fish species found in mangroves and seagrass beds (Lugendo et al. 2005; Mumbi 2006; Kimirei et al. 2011). There is scientific consensus showing mangroves, estuarine (zone between rivers) and sea grass ecosystems to significantly contribute to coral reefs fish populations (Verweij et al. 2008; Kimirei et al. 2013). This is possible through the ability of mangroves and seagrass to provide shelter against predators, nursery and feeding grounds to juveniles

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of many coral reef fishes (Lugendo et al. 2006, 2007; Verweij et al. 2008; Kimirei et al. 2013). For example, studies which were conducted in the Caribbean show about 17 of reef fish species to associate with estuaries and mangrove ecosystems (Nagelkerken et al. 2002; Dorenbosch et al. 2005; Mateo et al. 2010; Kimirei et al. 2011). In fact, in islands where there are no estuaries and or mangroves, those species were not found in adjacent reefs. Likewise, recent studies which were conducted in the Western Indian Ocean (WIO) region show mangroves and seagrass beds to replenish coral reef fish populations where sub-adult and adult fishes from mangroves and seagrass beds migrate to coral reefs for spawning (Kimirei et al. 2013). In Tanzania, mangroves occur on the sheltered shores of deltas, alongside river estuaries, and in creeks where fine-grained sediment (silt and clay) is abundant in the upper part of the inter-tidal zone (Wang et al. 2003). Along the Tanzania shoreline, well established mangroves with high fishery potential are found in extensive estuaries of Ruvuma, Fufiji and Ruvu estuaries. As indicated above, all these estuaries together with others which are smaller contribute significantly in improving coral reef fisheries in Tanzania which give employment to a good number of coastal dwellers.

The mangroves also help to reduce the amount of sediments going into other coastal ecosystems such as seagrass and coral reefs. Since the two ecosystems are intricately dependent, destruction of one would mean ecological disturbance of the other.

Furthermore, coral reefs provide coastal protection and stabilization by reducing wave energy and mitigating both routine erosion and damage from waves associated with small and moderate storm events. Current trends of climate change are accompanied by global warming, sea level rise and increased storm intensity (Hoegh-Guldberg 1999; Wilkinson 2002). Therefore, coastal protection will increasingly become more important. Coral reef ecosystems also may represent a biological source for new drugs and biochemicals that need to be explored. However, coral reefs are known to be fragile because they support a large number of delicate invertebrates that are susceptible to human-induced disturbances (Reaka-Kudla 1995). Thus, continued provision of these goods and services is threatened by recent increase in physical, chemical and biological stresses on corals (Hughes, 1994). It is estimated that such stresses would cause a decline of 40% to 60% of the world's coral reefs over the next 50 years if appropriate and corrective measures are not taken (Hoegh-Guldberg 1999; Wilkinson 2002; Hughes et al. 2003).

The survival of coral reefs ecosystem worldwide is threatened by coral bleaching which is a manifestation of current trends of climate change accompanied by global warming. Coral bleaching is a general term referring to the disappearance of the coloured pigments because of either degradation of algal pigments and/or partial to total loss of their *Symbiodinium* population to the extent of revealing the white coral skeleton (Glynn 1993; Hoegh-Guldberg 1999, Douglas 2003). Like other Cnidarians, scleractinian corals host photosynthetic dinoflagellates belonging to the genus *Symbiodinium*, commonly called zooxanthellae, within the vacuoles of their gastrodemal cells. These dinoflagellates are known to largely contribute to corals' metabolic demand in nutrient poor (oligotrophic) tropical environments (Muscatine 1990). About 40% of photosynthetic products in the *Symbiodinium* cell are translocated to coral tissues to meet up to over 90% of the coral's energetic demands (Johnson 2011). Due to high coral energetic reliance on photosynthetic products from *Symbiodinium*, bleaching impairs reproduction, growth rates, and general immunity of the corals (Hoegh-Guldberg 1999; Baird and Marshall 2002; Hughes et al 2003). In a situation where factors that cause bleaching are severe and persist for a long time, coral mortality occurs (Brown 1997; Douglas 2003).

Major factors that cause coral bleaching include sea surface temperature rise (Saxby et al. 2003, Barron et al. 2010) and/or drop (Hoegh-Guldberg and Fine. 2005), and UV radiation (Brown et al. 1994; Brown 1997; Dunne and Brown 2001; Winters et al. 2003). In fact, major coral bleaching events that pose the greatest management challenges to coral reef ecosystems worldwide are correlated with *El Nino and* Southern Oscillation (ENSO) events which are accompanied by both higher SST and solar radiation (Fitt and Warner 1995; Hoegh-Guldberg 1999; Lough 2000; Hueerkamp et al. 2001). Moreover, factors such as bacterial infection (Rosenberg and Falkovitz 2004), lowered salinity, and pollution (Glynn 1993) have also been documented to cause small scale bleaching events.

During the 1997/1998 worldwide coral bleaching event up to 90% reduction in coral cover in some places along the Tanzanian coast was recorded (Muhando 1999, Wilkinson et al. 1999; Lindahl et al. 2001). For example, Misali and Tutia reefs in Pemba and Mafia Islands respectively were the most affected when compared with other reefs. About 90% of these reefs suffered coral mortality following the 1997/1998 bleaching event. Other reefs such as Chumbe and Kwale in Zanzibar and Mikindani in Mtwara experienced coral mortality that ranged from 30% to 50%. Corals at Bawe, Changuu and Chapwani in Zanzibar, and Pangavin and Mbudya in Dar es Salaam were the least affected (mortality ranged from 11% to 20%). By 2007, most reefs have not recovered except very few (Muthiga et al. 2008).

Coral-*Symbiodinium* Symbioses in Relation to Bleaching Resistances

Differences in bleaching susceptibilities among coral species are well documented (Wilkinson et al. 1999; Marshall and Baird 2000; Lindahl et al. 2001; Hueerkamp et al. 2001;

Garpe and Öhman 2003; Sampayo et al. 2007). Such differences can be in terms of geographical area (Goreau and Hayes 1995; Hueerkamp et al. 2001), time (Berkelmans and Willis 1999), species, and individual colony history (Marshall and Baird 2000; Sampayo et al. 2007). Studies show *Acropora* and *Pocillopora* to be among the most susceptible genera (Hoeg-Guldberg and Salvat 1995; Marshall and Baird 2000; Lindahl et al. 2001). The Tanzanian coast is dominated by *Acropora* and *Porites* (Obura et al. 2004) which are susceptible and tolerant to bleaching, respectively. Although susceptibility of holobiont (coral and symbiont) to bleaching depends on both partners in symbioses, this review focuses on *Symbiodinium* (symbiont) as one of important inducer of tolerance to bleaching.

Although the host contribute significantly to bleaching susceptibility of the holobiont (Baird et al. 2009; Fitt et al. 2009), different *Symbiodinium* species hosted by the corals largely explain the observed difference in bleaching susceptibility among coral species (Warner et al. 1996; 2006; Bhagooli and Hidaka 2003; Iglesias-Prieto et al. 2004). This is due to the fact that, different species of *Symbiodinium* differ in terms of their thermal and irradiance tolerances (Kinzie et al. 2001; Bhagooli and Hidaka 2003). Such differences in thermal tolerance among *Symbiodinium* species is attributed to differences in photochemical responses (Winters et al. 2003; Bhagooli and Hidaka 2003; Iglesias-Prieto et al. 2004; Warner et al. 2006; Frade et al. 2008). *Symbiodinium* clade D is believed to be relatively more thermally tolerant due to its presence in marine habitats with chronically high temperatures (Fabricius et al. 2004; Glynn et al. 2001) and environmental disturbances (Toller et al. 2001; Berkelmans and van Oppen 2006; Rowan 2004; Baker et al. 2004; Stat and Gates 2011). A study that monitored seasonal variation in *Symbiodinium* types found an increase in the frequencies of clade D during hotter seasons (Chen et al. 2005). Basing on the fact that major bleaching events are correlated with rise in sea surface temperature, clade D is considered to be among the *Symbiodinium* types that can help corals to survive the current trends of climate change.

Distribution of *Symbiodinium* Clade D in Tanzanian Corals

Coral may acquire *Symbiodinium* cells including those belonging to clade D either vertically or horizontally. In vertical (also known as maternal) transmission strategy, *Symbiodinium* cells are present in the eggs or brooded planulae larvae prior to release from the parent (Trench 1987). However, in horizontal transmission strategy, released eggs and larvae do not have *Symbiodinium* cells. These juvenile corals acquire *Symbiodinium* cells from the ambient environment (Trench 1987). Most corals in most reefs around the world have been found to have potential of hosting *Symbiodinium* clade D (Oliver and Palumbi 2010; Ghavam Mostafavi et al. 2007). One should be cautious though, when reviewing diversity and distribution of *Symbiodinium* in reef building corals. This is due the employment of molecular approaches that have been found to over-estimate diversity of *Symbiodinium* in the coral host (Thornhill et al. 2007; Sampayo et al. 2009). These include those involving the cloning of PCR products and the use of real time PCR in the estimation of *Symbiodinium* diversity. To avoid such over estimation, only previous studies whose approaches in analysis of *Symbiodinium* types do not lead to overestimation of diversity of *Symbiodinium* types in the coral tissue were included in Tables 1 and 2. Therefore, only the study that involved Restriction Fragment Length Polymorphism (RFPL), Single Strand Conformational Polymorphism (SSCP) and Denaturing Gradient Gel Electrophoresis (DGGE) were included in Tables 1 and 2. These approaches have ability to detect *Symbiodinium* types in the coral tissue that are actually responsible for the physiology of the holobiont.

As presented in Table 1, a range of coral genera from fast growing branching (*Acropora*, *Stylophora*, *Seriatopora* and *Pocillopora*) to slow-growing massive corals (*Montastrea*), encrusting (*Montipora*) and solitary (e.g., *Fungia*) corals have been found to host *Symbiodinium* clade D globally (Table 1). In fact, in the Persian Gulf and Iran where sea surface temperatures are extremely high with very high seasonal fluctuations, *Porites* was found to establish symbiosis with *Symbiodinium* clade D (Mostafavi et al 2007). In other parts of the world, *Porites* has maintained symbiosis with *Symbiodinium* C15 type, probably because the symbiont has been found to be both thermo and irradiance tolerant (Fitt et al. 2009). However, in Tanzania only *Seriatopora*, *Galaxea* and *Acropora* in a few reefs have been found to host *Symbiodinium* clade D (LaJeunesse et al. 2010; Chauka 2012). Unlike in other regions where a large number of coral genera have been found to host *Symbiodinium* clade D, Tanzanian reef building corals are dominated by C3u and C3z (Chauka 2012), which are both thermal and irradiance intolerant (Chauka et al. in press).

Clade D *Symbiodinium* is present in higher abundance on some reefs than on others. In most cases, such difference is attributed to differences in exposure to sea surface temperature. For example, Oliver and Palumbi (2009) found higher abundance of *Symbiodinium* clade D in *Acropora* samples collected from back-reef lagoons in American Samoa. The sea surface temperature in these reefs was found to be higher when compared with other places in American Samoa. Other factors that can cause differences in distribution of *Symbiodinium* clade D in a particular region include turbidity and history of coral bleaching. In Tanzania, high turbidity seems to play a significant role in the distribution and abundance of

Table 1 Examples of coral genera which have been found to host Symbiodinium clade D in different part of the world and compared with the same genera in Tanzanian coast

Coral genera hosting Clade D Symbiodinium	Place observed with reference	Presence/Absence in Tanzania
Pocillopora	Carribean (Iglesias-Prieto et al. 2004), North Indian Ocean (LaJeunesse et al. 2010); Great Barrier Reef (Stat et al. 2009)	No (LaJeunesse et al. 2010; Chauka 2012)
Astreopora	North Indian Ocean (LaJeunesse et al. 2010)	No (LaJeunesse et al. 2010; Chauka 2012)
Montipora	Hawaii (LaJeunesse et al. 2004a, b, c), North Indian Ocean (LaJeunesse et al. 2010)	No (LaJeunesse et al. 2010; Chauka 2012)
Echinopora	North Indian Ocean (LaJeunesse et al. 2010)	No (LaJeunesse et al. 2010; Chauka 2012)
Favia	North Indian Ocean (LaJeunesse et al. 2010); Persian Gulf, Iran (Mostafavi et al. 2007)	No (LaJeunesse et al. 2010; Chauka 2012)
Goniopora	North Indian Ocean (LaJeunesse et al. 2010)	No (LaJeunesse et al. 2010; Chauka 2012)
Pavona	North Indian Ocean (LaJeunesse et al. 2010); Persian Gulf, Iran (Mostafavi et al. 2007)	No (LaJeunesse et al. 2010; Chauka 2012)
Seriatopora	North Indian Ocean (LaJeunesse et al. 2010)	Yes (LaJeunesse et al. 2010; Chauka 2012)
Galaxea	Kenya (Visram and Douglas, 2006), North Indian Ocean (LaJeunesse et al. 2010), Curaçao (LaJeunesse et al. 2004a, b, c)	Rare (LaJeunesse et al. 2010; Chauka 2012)
Goniastrea	North Indian Ocean (LaJeunesse et al. 2010); Great Barrier reef (Stat et al. 2009), Caribbean (LaJeunesse et al. 2004a, b, c)	No (LaJeunesse et al. 2010; Chauka 2012)
Montastrea	Belize, Caribbean (Toller et al. 2001; Garren et al. 2006), North Indian Ocean (LaJeunesse et al. 2010)	No (LaJeunesse et al. 2010; Chauka 2012)
Favites	Penghu Islands (the Pescadores) Taiwan and from Hong Kong (Chen et al. 2005)	No (LaJeunesse et al. 2010; Chauka 2012)
Platygyra	North Indian Ocean (LaJeunesse et al. 2010); Persian Gulf, Iran (Mostafavi et al. 2007)	No (LaJeunesse et al. 2010; Chauka 2012)
Porites	Persian Gulf, Iran (Mostafavi et al. 2007) Palauan reefs	No (LaJeunesse et al. 2010; Chauka 2012)

Symbiodinium clade D. Most reefs in Zanzibar are exposed to similar sea surface temperature (Chauka 2012). However, clade D was found in higher abundance in Chapwani and Changuu reefs, the most turbid reefs, while in Bawe and Munemba Clade D was absent (LaJeunesse et al. 2010; Chauka 2012). Likewise, clade D was sampled in Mbudya (39014′52.45″E, 6039′33.76″S), Dar es Salaam; a place where turbidity was found to be high. In a very close reef; Pangavin (39°14′17.77″E, 6°40′27.37″S) where turbidity is low, clade D was not sampled. In fact, low mortality in Mbudya reef, Dar es Salaam following the1998 bleaching event is attributed to high turbidity that might have prevented higher solar radiations reaching the surface of the coral and not the presence of *Symbiodinium* clade D (Chaukaet al. in press). This is due to the fact that corals species that were not found to host *Symbiodinium* clade D which are known to be bleaching susceptible such as *Seriatopora*, were not severely affected in Mbudya, Dar es Salaam as compared in other reefs (Muhando 1999).

With regard to bleaching history, several studies have shown increases in the abundance of clade D *Symbiodinium* in corals following bleaching events (Rowan 2004; Chen et al. 2005; Jones et al. 2008). In fact observation of clade D seems to be a manifestation of coral recovering from a bleaching event and therefore represents survival strategies (Stat and Gates 2011). In Tanzania, *Symbiodinium* clade D is found in Acropora samples collected from Kitutia (39°45′6.16″E, 8°1′1.87″S) reef, located off the Mafia Island. Such encounter can be explained by history of bleaching events (Chauka et al. in press). Therefore, only reefs off Mafia Island which have been experiencing seasonal bleaching indicate some development of adaptations to bleaching. Generally, most coral genera that have been found to host *Symbiodinium* clade D in other parts of the world have not been found to host clade D on the Tanzanian coast. The inability of most reef building corals species along the Tanzanian coast to host clade D *Symbiodinium* type that has been possible in other part of the world is an indication of low adaptation to climate change.

Endosymbiotic Flexibility

Flexibility in coral-*Symbiodinium* association is hypothetically believed to enable corals to adapt to climate change (Baker 2003). According to Rowan and Knowlton (1995),

Table 2 Examples of coral genera which have been found to form symbiosis with more than one Symbiodinium types as compared with symbiosis occurring in Tanzanian corals-Symbiodinium symbioses

Coral genera	Polymorphic symbioses observed in different parts of the world	*Symbiodinium* types found to associate with mentioned coral genus in Tanzania
Acropora	C3u, C3z, C101, C94, D1-4, D2 (LaJeunesse et al. 2010)	C3u, C3z, C109a, C109b, C115 and D1a (LaJeunesse et al. 2010; Chauka 2012)
Fungia	C3u, D1 (LaJeunesse et al. 2010)	C3u (LaJeunesse et al. 2010; Chauka 2012)
Platygyra	C3u, C101a, C101, D1, D1-4 (LaJeunesse et al. 2010)	C3u (LaJeunesse et al. 2010; Chauka 2012)
Symphyllia	C3u, D1 (LaJeunesse et al. 2010)	C3u (LaJeunesse et al. 2010; Chauka 2012)
Favites	C3u, C3z, C101, D1, D1-4 (LaJeunesse et al. 2010)	C3u (LaJeunesse et al. 2010; Chauka 2012)
Pavona	C3u, D1 (LaJeunesse et al. 2010)	C3u (LaJeunesse et al. 2010; Chauka 2012)
Montastraea	C3u, C101, D1-4 (LaJeunesse et al. 2010; Fabricius et al. 2004)	C3z (LaJeunesse et al. 2010; Chauka 2012)
Goniopora	C1, C3u, D1, D4-1, D5 (LaJeunesse et al. 2010)	C1b-s (LaJeunesse et al. 2010; Chauka 2012)
Poccilopora	C1c, C1d, C1-d, C1d-t, C42a, D1, D5 (LaJeunesse et al. 2010)	C1h (LaJeunesse et al. 2010; Chauka 2012)
Montipora	C15, C110, C26a, D5 (LaJeunesse et al. 2010)	C17, C17a (LaJeunesse et al. 2010; Chauka 2012)
Porites	C15, C114 (LaJeunesse et al. 2010), C & D	C15 (LaJeunesse et al. 2010; Chauka 2012)
Stylophora	C1, C8a (Fitt et al. 2009)	C105a (LaJeunesse et al. 2010; Chauka 2012)

Uppercase letters indicate clade while numbers represent ITS-2 DGGE type

some corals have the ability to get rid of their stress-intolerant *Symbiodinium* and let their tissues be dominated by stress-tolerant *Symbiodinium* type. Empirical data by Baker et al. (2004) shows a possibility of secondary acquisition of symbionts when populations of *Symbiodinium* type in the respective coral tissue are very low. Moreover, recent works show that most bleaching susceptible corals have been found to form symbiosis with different *Symbiodinium* types depending on the environment they are exposed to (Putnam et al. 2012). Likewise, a recent study shows specificity in coral-*Symbiodinium* symbioses to be rare than previously suggested (Silverstein et al. 2012). In most corals species that have shown flexibility in making symbiosis with different *Symbiodinium* partners, shuffling of *Symbiodinium* populations to maximise their survival potential is also possible. In other part of the world, a number of coral genera including *Acropora*, *Platygyra, Goniopora, Favites*, *Pocillopora*, *Symphyllia* are the most flexible in terms of partnership with different *Symbiodinium* types (Table 2). However, except *Acropora*, these genera form symbiosis with single *Symbiodinium* type on the Tanzanian coast (Table 2).

On the Tanzanian coast, the genus *Acropora* was found to establish symbiosis with six different *Symbiodinium* types (Table 2). In most cases however, local environment differences, mainly light micro-environments, played a role in such polymorphic symbioses. For example, in Zanzibar, samples of *Acropora* collected in very turbid reefs were found to establish symbiosis with many symbiont types than in reefs that were clean. Coral-*Symbiodinium* polymorphic symbioses in most part of the world seem to be associated with harsh environment including temperature (Mostafavi et al. 2007; LaJeunesse et al. 2010; Stat and Gates 2011). For example, reef corals in Andaman Sea and North-eastern Indian Ocean, where SSTs are extremely higher; most coral genera were found to establish symbiosis with multiple *Symbiodinium* types (LaJeunesse et al. 2010). Probably exposure of corals colonies to extremely higher SSTs and solar radiations that largely fluctuate seasonally has induced development of adaptations that caused co-existence of different *Symbiodinium* types in the same coral tissue. Genus *Porites* has been found to maintain its C15 *Symbiodinium* type in most part of the world (LaJeunesse et al. 2003, 2004a, 2004b; Stat et al. 2006). However, in the Persian Gulf in Iran where SSTs are extremely high, *Porites* has been able to establish symbiosis with multiple *Symbiodinium* types including clade D (Mostafavi et al. 2007). Inability of most Tanzanian reefs corals to associate with multiple *Symbiodinium* types including thermo tolerant symbionts reflect relatively stable and low temperatures and solar radiations as compared to other reefs that experience seasonal fluctuation of temperature and solar radiations. In case of events that are accompanied by sudden change rise in SSTs and solar radiation, the reef building corals may be severely affected by bleaching compared to those which has experienced such event over long time and be able to establish multiple symbioses.

Concluding Remarks and Future Work

Current predictions show expected rise in temperature of up to 3°C per century (IPCC 2001; Hoegh-Guldberg et al. 2007). Unless appropriate measures are put in place, such a rise is expected to increase frequencies of coral bleaching events which could lead to significant reduction in coral reefs worldwide in this century (Hughes 1994;

Hughes et al. 2003; Hoegh-Guldberg 1999; Hoegh-Guldberg et al. 2007). With evidences provided in this review, Tanzanian coral-*Symbiodinium* symbioses do not favour survival of corals especially in the context rise in sea surface temperatures. Survival rate of corals is also reduced by extremely high anthropogenic pressures from overfishing, pollution, and sedimentation. Because of this, protection and restoration are recommended as management options that will increase survival of Tanzanian coral reefs ecosystem. Such increased survival rate may give room for coral species to develop functioning adaptations strategies before they succumb to elevated sea surface temperature and solar radiations. On one hand, protection would lead to reduction in direct anthropogenic threats from destructive fishing and increase in herbivore fishes that are ecologically important in controlling alga pressure on corals. On the other hand, restoration may selectively lead to increase in bleaching resistant coral species for the survival of ecosystem. This work will not only benefit the corals themselves, but the mangroves, estuaries and see grass ecosystems because of their ecological interdependence.

The review shows that, most Tanzanian corals maintain their *Symbiodinium* types across the country. Probably, the reefs across the country are genetically connected. Reef building corals genetic connectivity study will provide insights on reasons for stability in coral-*Symbiodinium* symbioses along the Tanzanian coast. Likewise, such study will reveal the reefs whose coral populations are affected by others and those reefs whose regenerations depend on local coral population. To help in the survival of those reefs whose regeneration depend on local coral population because their genetical isolation, local management is the best options. This review paper therefore recommend a comprehensive reefs genetic connectivity study along the Tanzanian coast is necessary to provide information on how reefs are connected along the Tanzania coast.

References

Baird AH, Marshall PA (2002) Mortality, growth and reproduction in scleractinian corals following bleaching on the Great Barrier Reef. Mar Ecol Prog Ser 237:133–141

Baird AH, Bhagooli R, Ralph PJ, Takahashi S (2009) Coral bleaching: the role of the host. TREE 24:16–20

Baker AC (2003) Flexibility and specificity in coral-algal symbiosis: diversity, ecology, and biogeography of Symbiodinium. Ann Rev Ecol Evol Syst 34:661–689

Baker AC, Starger C, McClanahan T, Glynn P (2004) Corals' adaptive response to climate change. Nature 430:741

Barron MG, McGill CJ, Courtney LA, Marcovich DT (2010) Experimental bleaching of a reef-building coral using a simplified recirculating laboratory exposure system. J Mar Biol 41:51–67

Berkelmans R, van Oppen MJH (2006) The role of zooxanthellae in the thermal tolerance of corals: a 'nugget of hope' for coral reefs in an era of climate change. Proc R Soc B 273:2305–2312

Berkelmans R, Willis BL (1999) Seasonal and local spatial patterns in the upper thermal limits of corals on the inshore central Great Barrier Reef. Coral Reefs 18:219–228

Bhagooli R, Hidaka M (2003) Comparison of stress susceptibility of hospite and isolated zooxanthellae among five coral species. J Exp Mar Biol 291:181–197

Brown BE (1997) Coral bleaching: causes and consequences. Coral Reefs 16:129–138

Brown BE, Dunne RP, Scoyn TP, LeTissier MDA (1994) Solar damage in intertidal corals. Mar Ecol Prog Ser 105:30–43

Chauka LJ (2012) Diversity of the symbiotic alga *Symbiodinium* in Tanzanian Scleractinian corals. West Indian Ocean J Mar Sci 11:67–76

Chauka LJ, Steinert G, Mtolera MSP (in press) Tanzanian local environmental differences and bleaching histories' influence on diversity and distribution of Symbiodinium in reef building corals. Afr J Mar Sci

Chen CA, Wang J, Fang L, Yang Y (2005) Fluctuating algal symbiont communities in Acroporapalifera (Scleractinia; Acroporidae) from Taiwan. Mar Ecol Prog Ser 295:113–121

Dorenbosch M, Grol MGG, Christianen MJA, Nagelkerken I, van der Velde G (2005) Indo-Pacific seagrass beds and mangroves contribute to fish density and diversity on adjacent coral reefs. Mar Ecol Prog Ser 302:63–76

Douglas AE (2003) Coral bleaching-how and why? Mar Pollut Bull 46:385–392

Dunne RP, Brown BE (2001) The influence of solar radiation on bleaching of shallow water reef corals in the Andaman Sea, 1993–1998. Coral Reefs 20:201–210

Fabricius KE, Mieog JC, Colin PL, Idip D, van Oppen MJH (2004) Identity and diversity of coral endosymbionts (zooxanthellae) from three Palauan reefs with contrasting bleaching, temperature and shading histories. Mol Ecol 13:2445–2458

Fitt WK, Warner ME (1995) Bleaching patterns of four species of Caribbean reef corals. Biol Bull 189(3):298–307

Fitt WK, Gates RD, Hoegh-Guldberg et al (2009) Response of two species of Indo-Pacific corals, *Porites cylindrica* and *Stylophora pistillata*, to short-term thermal stress: the host does matter in determining the tolerance of corals to bleaching. J Exp Mar Biol Ecol 373:102–110

Frade PR, Englebert N, Faria J, Visser PM, Bak RPM (2008) Distribution and photobiology of *Symbiodinium* types in different light environments for three colour morphs of the coral Madracispharensis: is there more to it than total irradiance? Coral Reefs 27:913–925

Garpe KC, Öhman MC (2003) Coral and fish distribution patterns in Mafia Island Marine Park, Tanzania: fish-habitat interactions. Hydrobiologia 498:191–211

Garren M, Walsh SM, Caccone A, Knowlton N (2006) Patterns of association between Symbiodinium and members of the Montastraea annularis species complex on spatial scales ranging from within colonies to between geographic regions. Coral Reefs 25:503–512

Glynn PW (1993) Coral-reef bleaching-ecological perspectives. Coral Reefs 12:1–17

Glynn PW, Maté JL, Baker AC, Calderón MO (2001) Coral bleaching and mortality in Panama and Ecuador during the 1997–1998 El Nin˜o-Southern Oscillation event, spatial/temporal patterns and comparisons with the 1982–1983 event. Bull Mar Sci 69:79–109

Goreau TF, Hayes RL (1995) Coral reef bleaching in the South Central Pacific during 1994. Global Coral Reef Alliance, Chappaqua

Hoeg-Guldberg O, Salvat B (1995) Periodic mass bleaching of reef corals along the outer reef slope in Moorea, French Polynesia. Mar Ecol Prog Ser 121:181–190

Hoegh-Guldberg O (1999) Climate change, coral bleaching and the future of the world's coral reefs. Mar Freshw Res 50:839–866

Hoegh-Guldberg O, Fine M (2005) Coral bleaching following wintry weather. Limnol Oceanogr 50(1):265–271

Hoegh-Guldberg O, Mumby PJ, Hooten AJ et al (2007) Coral reefs under rapid climate change and ocean acidification. Science 318:1737–1742

Hueerkamp C, Glynn PW, D'Croz L, Maté JL, Colley SB (2001) Bleaching and recovery of five eastern Pacific corals in an El Niño-related temperature experiment. Mar Sci Bull 69(1):215–237

Hughes T (1994) Catastrophes, phase shifts, and large-scale degradation of Caribbean coral reefs. Science 265:1547–1551

Hughes TP, Baird A, Bellwood H et al (2003) Climate change, human impacts, and the resilience of coral reefs. Science 301:929–933

Iglesias-Prieto R, Beltran VH, LaJeunesse TC, Reyesbonilia H, Thome PE (2004) Different algal symbionts explain the vertical distribution of dominant reef corals in the eastern Pacific. Proc R Soc Lond B 271:1757–1763

IPCC (2001) Impacts, Adaptation, and Vulnerability – Contribution of Working Group II to the IPCC Third Assessment Report, ed. J.J. McCarthy, O.F. Canziani, N.A. Leary, D.J. Dokken, K.S. White. Cambridge University Press, Cambridge 1032p

Johnson MD (2011) The acquisition of phototrophy: adaptive strategies of hosting endosymbionts and organelles. Photosynth Res 107:117–132

Jones AM, Berkelmans R, Van Oppen MJH, Mieog JC, Sinclair W (2008) A community change in the algal endosymbionts of a scleractinian coral following a natural bleaching event: field evidence of acclimatization. Proc R Soc Lond B 275:1359–1365

Kimirei IA, Nagelkerken I, Griffioen B, Wagner C, Mgaya YD (2011) Ontogenetic habitat use by mangrove/seagrass-associated coral reef fishes shows flexibility in time and space. Estuar Coast Shelf Sci 92 (1):47–58. doi:10.1016/j.ecss.2010.12.016

Kimirei IA, Nagelkerken I, Mgaya YD, Huijbers CM (2013) The mangrove nursery paradigm revisited: otolith stable isotopes support nursery-to-reef movements by Indo-Pacific fishes. PloS One 8 (6):e66320. doi:10.1371/journal.pone.0066320

Kinzie RAIII, Takamaya R, Santos SA, Coffon MA (2001) The adaptive bleaching hypothesis: experimental tests of critical assumptions. Biol Bull 200:51–58

LaJeunesse TC, Loh WKW, van Woesik R, Hoegh-Guldberg O, Schmidt GW, Fitt WK (2003) Low symbiont diversity in southern Great Barrier Reef corals, relative to those of the Caribbean. Limnol Oceanogr 48:2046–2054

LaJeunesse TC, Bhagooli R, Hidaka M, deVantier L, Done T, Schmidt GW, Fitt WK, Hoegh-Guldberg O (2004a) Closely related *Symbiodinium* spp. differs in relative dominance in coral reef host communities cross environmental, latitudinal and biogeographic gradients. Mar Ecol Prog Ser 284:147–161

LaJeunesse TC, Thornhill DJ, Cox E, Stanton F, Fitt WK, Schmidt GW (2004b) High diversity and host specificity observed among symbiotic dinoflagellates in reef coral communities from Hawaii. Coral Reefs 23:596–603

LaJeunesse TC, Pettay DT, Sampayo EM, Phongsuwan N, Brown B, Obura D, Hoegh-Guldberg O, Fitt WK (2010) Long-standing environmental conditions, geographic isolation and host–symbiont specificity influence the relative ecological dominance and genetic diversification of coral endosymbionts in the genus *Symbiodinium*. J Biogeogr 37:785–800

Lindahl U, Öhman MC, Schelten CK (2001) The 1997/1998 mass mortality of corals: effects on fish communities on a Tanzanian coral reef. Mar Pollut Bull 42:127–131

Lough JM (2000) 1997–98: unprecedented thermal stress to coral reefs. Geophys Res Lett 27:3901–3904

Lugendo BR, Pronker A, Cornelissen I, de Groene A, Nagelkerken I, Dorenbosch M, van der Velde G, Mgaya YD (2005) Habitat utilisation by juveniles of commercially important fish species in a marine embayment in Zanzibar, Tanzania. Aquat Living Resour 18(2):149–158

Lugendo BR, Nagelkerken I, van der Velde G, Mgaya YD (2006) The importance of mangroves, mud and sand flats, and seagrass beds as feeding areas for juvenile fishes in Chwaka Bay, Zanzibar: gut content and stable isotope analyses. J Fish Biol 69 (6):1639–1661

Lugendo BR, Nagelkerken I, Kruitwagen G, van der Velde G, Mgaya YD (2007) Relative importance of mangrove as feeding habitats for fishes: a comparison between mangrove habitats with different settings. Bull Mar Sci 80(3):497–512

Marshall PA, Baird AH (2000) Bleaching of corals on Great Barrier Reef: differential susceptibilities among taxa. Coral Reefs 19:155–167

Mateo I, Durbin EG, Appeldoorn RS, Adams AJ, Juanes F, Kingsley R, Swart P, Durant D (2010) Role of mangroves as nurseries for French grunt Haemulon flavolineatum and schoolmaster Lutjanus apodus assessed by otolith elemental fingerprints. Mar Ecol Prog Ser 402:197–212

Moberg F, Folke C (1999) Ecological goods and services of coral reef ecosystems. Ecol Econ 29:215–233

Mostafavi PG, Fatemi SMR, Shahhosseiny MH, Hoegh-Guldberg O, Loh WKW (2007) Predominance of clade D Symbiodinium in shallow-water reef-building corals off Kish and Larak Islands (Persian Gulf, Iran). Mar Biol 153:25–34

Muhando C (1999) Assessment of the extent of damage, socio-economics effects, mitigation and recovery of coral reefs in Tanzania. In: Linden O, Sporrong N (eds) Coral reef degradation in the Indian Ocean: status report and project presentation. CORDIO, Townsville, Australia, pp 43–47

Mumbi PJ (2006) Connectivity of reef fish between mangroves and coral reefs: algorithms for the design of marine reserves. Biol Conserv 128:215–222

Muscatine L (1990) The role of symbiosis algae in carbon and energy flux in coral reefs. In: Dubinsky Z (ed) Ecosystems of the world: coral reefs. Elsevier, Amsterdam, pp 75–87

Muthiga N, Costa A, Motta H, Muhando C, Mwaipopo R, Schleyer M (2008) Status of coral reefs in East Africa: Kenya, Tanzania, Mozambique and South Africa. In: Wilkinson C (ed) Status of coral reefs of the world: 2008. Global Coral Reef Monitoring Network and Reef and Rainforest Research Center, Townsville, pp 91–104

Nagelkerken I, Roberts CM, van der Velde G, Dorenbosch M, van Riel MC, Cocheret de la Morinière E, Nienhuis PH (2002) How important are mangroves and seagrass beds for coral-reef fish? The nursery hypothesis tested on an island scale. Mar Ecol Prog Ser 244:299–305

Obura D, Church J, Daniels C, Kalombo H, Schleyer M, Mohammed MS (2004) In: Wilkinson C (ed) Status of coral reefs in East Africa 2004: Kenya, Tanzania, Mozambique and South Africa. Status of Coral Reefs of the World, pp 171–188

Oliver TA, Palumbi SR (2009) Distributions of stress-resistant coral symbionts match environmental patterns at local but not regional scales. Mar Ecol Prog Ser 93–103

Oliver TA, Palumbi SR (2011) Many corals host thermally resistant symbionts in high-temperature habitat. Coral Reefs 30:241–250

Putnam HM, Stat M, Pochon X, Gates R (2012) Endosymbiotic flexibility associates with environmental sensitivity in scleractinian corals. Proc R Soc B 279:4352–4361

Reaka-Kudla ML (1995) The global biodiversity of coral reefs: a comparison with rainforests. Limnol Oceanogr 48:2046–2054

Rosenberg EL, Falkovitz (2004) The *vibrio shiloi*/*Oculina patagonica* model system of coral bleaching. Ann Rev Microbiol 58:143–159

Rowan R (2004) Coral bleaching: thermal adaptation in reef coral symbionts. Nature 430:742

Rowan R, Knowlton N (1995) Intraspecific diversity and ecological zonation in coral-algal symbiosis. Proc Natl Acad Sci USA 92: 2850–2853

Sampayo EM, Franceschinis L, Hoegh-Guldberg O, Dove S (2007) Niche partitioning of symbiotic dinoflagellates. Mol Ecol 16:3721–3733

Sampayo EM, Dove S, LaJeunesse TC (2009) Cohesive molecular genetic data delineate species diversity in the dinoflagellates genus Symbiodinium. Mol Ecol 18:500–519

Saxby T, Dennison WC, Hoegh-Guldberg O (2003) Photosynthetic responses of the coral *Montipora digitata* to cold temperature stress. Mar Ecol Prog Ser 248:85–97

Silverstein RH, Correa AMS, Baker AC (2012) Specificity is rarely absolute in coral–algal symbiosis: implications for coral response to climate change. Proc R Soc B 279:2609–2618

Spalding MD, Grenfell AM (1997) New estimates of global and regional coral reef areas. Coral Reefs 16:225–230

Stat M, Gates RD (2011) Clade D *Symbiodinium* in scleractinian corals: a “nugget” of hope, a selfish opportunist, an ominous sign, or all of the above? J Mar Biol 730:1–9

Stat M, Cartera D, Hoegh-Guldberg O (2006) The evolutionary history of *Symbiodinium* and scleractinian hosts. Symbiosis, diversity, and the effect of climate change. Perspect Plant Ecol Evol Syst 8:23–43

Stat M, Loh WKW, LaJeunesse TC, Hoegh-Guldberg O, Carter DA (2009) Stability of coral–endosymbiont associations during and after a thermal stress event in the southern Great Barrier Reef. Coral Reefs 28:709–713

Thornhill DJ, LaJeunesse TC, Santos SR (2007) Measuring rDNA diversity in eukaryotic microbial systems: how intragenomic variation, pseudogenes, and PCR artefacts confound biodiversity estimates. Mol Ecol 16:5326–5340

Toller WW, Rowan R, Knowlton N (2001) Zooxanthellae of the Montastraeaannularis species complex: patterns of distribution of four taxa of Symbiodinium on different reefs and across depths. Biol Bull 201:348–359

Trench RK (1987) Dinoflagellates in non-parasitic symbioses. In: Taylor F (ed) The biology of dinoflagellates. Blackwell, Oxford, pp 530–570

Verweij MC, Nagelkerken I, Hans I, Ruseler SM, Mason PRD (2008) Seagrass nurseries contribute to coral reef fish populations. Limnol Oceanogr 53(4):1540–1547

Visram S, Douglas AE (2006) Molecular diversity of symbiotic algae (zooxanthellae) in scleractinian corals of Kenya. Coral Reefs 25:172–176

Wang Y, Ngusaru A, Tobey J, Makota V, Bonynge G, Nugranad J, Traber N, Hale L, Bowen R (2003) Remote sensing of mangrove change along the Tanzania coast. Mar Geod 26:35–38

Warner ME, Fitt WK, Schmidt GW (1996) The effects of elevated temperature on the photosynthetic efficiency of zooxanthellae in hospite from four different species of reef coral: a novel approach. Plant Cell Environ 19:291–299

Warner ME, LaJeunesse TC, Robison JE, Thur RM (2006) The ecological distribution and comparative photobiology of symbiotic dinoflagellates from reef corals in Belize: potential implications for coral bleaching. Limnol Oceanogr 51:1887–1897

Wilkinson C (2002) Status of coral reefs of the world. Australian Institute of Marine Sciences, Townsville, Australia 378 pp

Wilkinson C, Linden O, Cesar H, Hodgson G, Rubens J, Strong AE (1999) Ecological and socio-economic impacts of 1998 coral mortality in the Indian Ocean: an ENSO impact and a warning for future change? Ambio 28:188–196

Winters G, Loya Y, Röttgers R, Beer S (2003) Photoinhibition in shallow-water colonies of the coral *Stylophora pistillata* as measured in situ. Limnol Oceanogr 48:1388–1393

Estuarine Environmental and Socio-Economic Impacts Associated with Upland Agricultural Irrigation and Hydropower Developments: The Case of Rufiji and Pangani Estuaries, Tanzania

Yohana W. Shaghude

Abstract

The present paper reviews the anthropogenic pressure related with freshwater resource availability and its exploitation through irrigation and damming in the Rufiji and Pangani river basins and their resulting impacts on the Rufiji and Pangani estuaries and adjacent coastal and marine habitats, located along the southern and northern coast of Tanzania, respectively. In the Rufiji River Basin, water abstraction for the existing and planned irrigation and hydropower developments have been estimated at 12,788 million m^3, which is 45% of the total available 28,382 million m^3 water in the Basin. In the Pangani River Basin, water abstraction for the existing irrigation and hydropower developments have been estimated at 810 million m^3 which is 90% of the total available water (900 million m^3) in the Basin. The reduced river flow due to irrigation and hydropower developments are considered to have significant environmental impacts on the estuarine and adjacent coastal and marine habitats. The environmental impacts are more severe during the dry season than during the rainy season. It is also noted that the two basins are generally at different levels of water stress, with the Pangani Basin being under more critical water stress compared to the Rufiji Basin.

Keywords

Irrigation • Agriculture • Hydropower • Rufiji River • Pangani River • Estuarine • Coastal and marine habitats

Introduction

Water availability is increasingly becoming one of the major issues of concern in many global forums (WCD 2000; GWSP 2005; Shaghude 2006; UNFPA 2011). The ever increasing population growth around the world is contrasted with the dwindling trend of water availability (UNFPA 2011). Since 1970's to present, the per capita water availability has declined by 37%, where the observed drastic changes on the global water system are generally attributed to climate and anthropogenic influences (Shaghude 2006). Water abstraction from rivers and its impoundment by damming are among the leading anthropogenic activities on the global scale, exerting drastic changes on the global water system (WCD 2000; GWSP 2005; Shaghude 2006).

The global hydrologic cycle deposits about 113 m^{12} of water on the world's continents and islands in the form of rain or snow annually (Valls 1999). Out of this, 64% (i.e. 72 m^{12}) evaporates back into the atmosphere, leaving only 36% (about 41 m^{12}) for human use and sustainability of various ecological and environmental functions. The distribution of renewable fresh water on the surface of the earth is generally un-even. While, some countries are enjoying luxurious amount of fresh water availability, others are experiencing critical fresh water deficit (Parish et al. 2012).

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The critical limits of fresh water availability are therefore not at the global level but at regional, national or local levels (Valls 1999).

Tanzania's annual per capita water availability is estimated at 89 m^9 (89 km^3), which is equivalent to 2,700 m^3 per person per year (URT 2002). Based on the projected population of 59.8 million by the year 2025, the annual per capita water availability will be reduced by 45% to about 1,500 m^3. Considering that per capita water availability of less than 1700 m^3 is regarded as water stress category (Parish et al. 2012), it implies that Tanzania will soon experience a water stress situation (URT 2002). With the declining trend in per capita water availability, the drive for socio-economic developments may lead to stiff competition in water use between different sectors, resulting to biased distribution of water among the socio-economic and environmental sectors.

Damming is considered to influence major reforms in the Tanzania's energy sector. The hydropower potential estimated to be 4.5 GW but currently only utilizing 12% of this total (Mdemu and Magayane 2005), is considered to be a viable solution for boosting Tanzania's national energy capacity. The hydropower sources being among the major renewable energy sources is considered to play a major role in speeding up Tanzania's national strategy for achieving rural electrification and reduce the current trend of environmental degradation associated with the widespread usage of biomass fuel, comprising of mainly fuel wood and charcoal (URT 1997).Unsustainable use of biomass fuel will not only lead to increased land degradation, but will also affect the water resource availability which will in turn affect people's livelihoods (Shaghude 2006). In Tanzania, the biomass fuel accounts for approximately 90% of the final energy consumption (URT 1992).

The Agriculture sector in Tanzania which employs 80% of the nation's work-force contributes 45% of Tanzania's GDP and about 30% of its export earnings (URT 2009). Despite of its importance, the sector has been highly affected by inadequacy, seasonality and unreliability of rainfall and periodic droughts (URT 2009). Irrigation agriculture is therefore considered to influence major reforms in the Tanzania's agricultural sector, with potential to provide protection measures against the past experience of rainfall uncertainty, periodic droughts. Furthermore, irrigation agriculture is considered as a viable solution for stabilising crop production and assurance of household food security and poverty reduction (URT 2009).

In Tanzania, both hydropower and agriculture activities are located on the upstream parts of the river basins. Thus, by virtue of their location and also their importance in the Tanzania socio-economic reforms, the existing policy reforms allow them to withdraw considerable amount of the available water resource to boost agriculture through irrigation. However, the environmental and socio-economic impacts losses occurring on the coastal areas including the estuaries and the adjacent coastal and marine habitats have generally been neglected (Machibya et al. 2003; Mdemu and Magayane 2005; IUCN 2007). The present paper therefore reviews the socio-economic pressure associated with river abstraction through irrigation and its impoundment by damming and the resulting impacts on the estuaries and the adjacent coastal and marine habitats, with specific examples drawn from Rufiji and Pangani river basins, two of the largest river basins in Tanzania.

Rufiji River Basin

The Rufiji River Basin (Fig. 1) located on the southern parts of Tanzania, with a total area of about 179, 000 km^2 is the largest river basin in Tanzania. It consists of four main catchments, namely the Great Ruaha (84,000 km^2), Kilombero (26,000 km^2), Luwegu (40,000 km^2), which are collectively known as the Upper Rufiji (Shaghude 2004a). The three catchments of the Upper Rufiji merge on the lower parts of the basin and boarders with the Lower Rufiji catchment (19,215 km^2). The rainfall pattern is unimodal, characterized by a continuous rainfall, with rain season starting from October/ November to May, with a peak in April (Shaghude et al. 2008) The mean annual rainfall on the Upper Rufiji catchments is up to 1600 mm and between 500–700 mm on the Lower Rufiji (Kashaigili et al. 2006; Shaghude et al. 2008) The irrigation and hydropower developments being discussed in the present study are located on two of these catchments, namely the Great Ruaha and the Lower Rufiji (Shaghude 2004a).

The three Tributaries of the Upper Rufiji, Luwegu, Kilombero and Great Ruaha, supply respectively 18%, 62% and 15% of the total inflow to the Rufiji River (Hafslund 1980; RUBADA 2001). The Rufiji River which starts at the confluence of the Luwegu and Kilombero flows for about 100 km to the Stiegler's Gorge (Shaghude 2004a), where the largest hydropower dam in Tanzania, planned to produce 2300 KW is at the fore front of the National development agenda. Along with the Stiegler's Gorge dam, there is another dam (the Mtera dam) which is currently operational. Mtera dam, commissioned in 1980 and located on the Great Ruaha River has a storage capacity of 3,200 million m^3. The Dam is currently the largest dam in Tanzania and is a storage reservoir for the power plant at Mtera (80 MW) and another power plant, Kidatu (200 MW) located further downstream (Fig. 1).

Along with the hydropower developments, agricultural irrigation is being promoted by government authorities as part of the major national development agenda which is seeking for optimization of agricultural productivity for attaining national food security (RUBADA 2001; Hakiardhi

Fig. 1 Location of the Rufiji and Pangani River Basins in Tanzania (Note also that the four catchments of the Rufiji River Basin (the Great Ruaha, Kilombero, Luwegu and Lower Rufiji) and the Usangu Plain along the Great Ruaha catchment are indicated)

2011). In line with this, two major irrigation developments have been introduced on the Rufiji River Basin. The first one located at the Usangu plain of the Great Ruaha catchment (Fig. 1) has been in existence for more than four decades, but the second one, located on the Lower Rufiji Flood Plain (LRFP) of the Lower Rufiji catchment has only been recently introduced (RUBADA 2001; Hamerlynck et al. 2010; Hakiardhi 2011).

Pangani River Basin

The Pangani River Basin (Fig. 1), located on the northeastern part of Tanzania and with an estimated area of 43,000 km^2 (IUCN 2003; Turpie et al. 2003) is the largest river basin on the northern parts of Tanzania. The Pangani River receives its fresh water from two tributaries, namely the Ruvu and the Kikuletwa, both of which flow to the Nyumba ya Mungu Dam (NYM, Fig. 2) whose out flow eventually flow to the Indian Ocean.Unlike the Rufiji River Basin which is characterized by a unimodal rainfall pattern, the Pangani River Basin is characterized by a bimodal rainfall pattern (Kabanda and Jury 1999) with two rainfall seasons, namely: the short rainy season, spanning from October to December (with peak in November) and the long rainy season, spanning from March to May (with peak in April/May). The annual sediment deposited at NYM due to erosion from the upper catchments has been estimated at 411,000 tonnes and the estimated sediments deposited in the Dam since its commissioning in 1968 to 2010 have been reported to amount to 13 million tonnes (Ndomba 2010).

The Basin constitutes of two units, namely, the highlands and the lowlands also called the "*Maasai steppe*". The highlands, which comprise of steeply sloping mountain

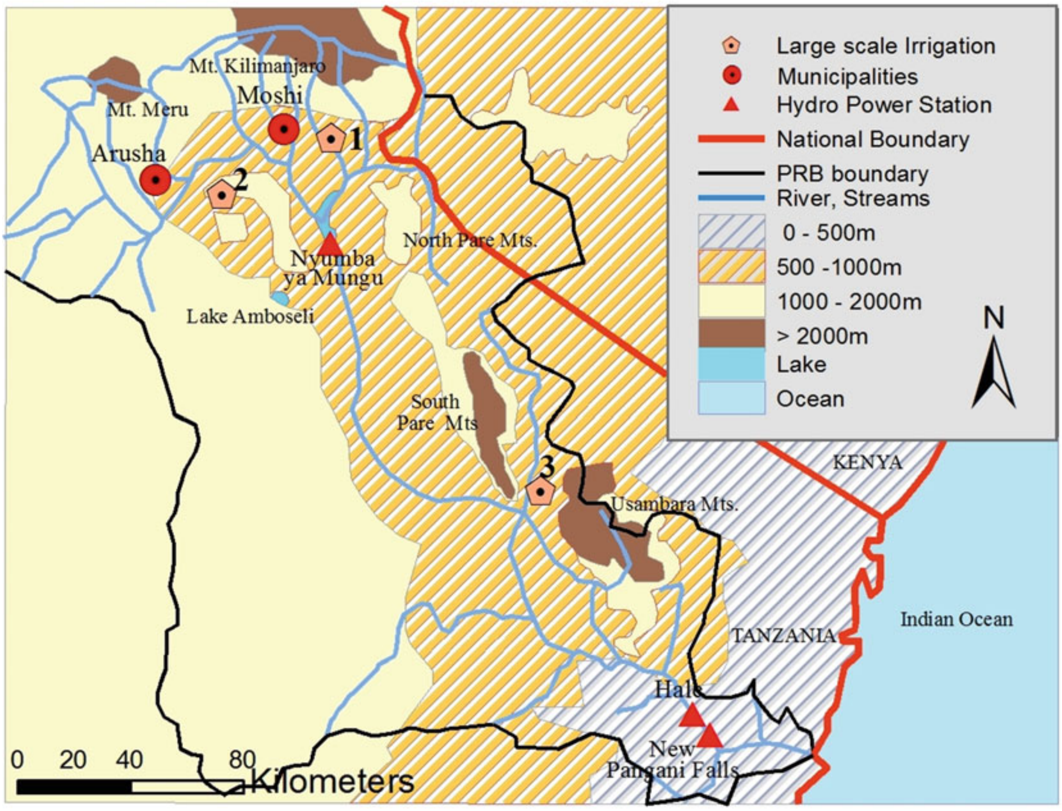


Fig. 2 Location of the Pangani River Basin (in Tanzania), its boundary and important basin features, namely, main river tributaries, mountain ranges, lakes/reservoirs, locations of hydropower stations and major Large-scale irrigation schemes (*1* = Lower Moshi, 2 = Ndungu and *3* = TPC. Observe the Kikuletwa and Ruvu tributaries feeding the Nyumba ya Mungu (NYM) reservoir from northwest and northeast, respectively (Source: Shaghude (2006))

terrain rising from 1,000 m to over 2,000 m above sea level (Fig. 2), are characterized by abundant rainfall (1,200-2,000 mm), high biodiversity, intensive cultivation, urbanization and densely populated rural areas, holding more than 80% of the basin's inhabitants (IUCN 2003; Shaghude 2006) and with annual population growth rates of up to 4%. The lowlands, which comprise of low sloping terrain generally below 1,000 m descending to the coastal plain (Fig. 2), receive relatively low rainfall (<500 mm per year) and is characterized by smaller settlement areas and low population growth rates of only up to 2% (Shaghude 2006).

The Nyumba ya Mungu Dam (Fig. 2) located at the confluence of the two main tributaries of the Pangani River, the Ruvu and the Kikuletwa is one of the largest dams in Tanzania. The dam has a total surface area of 15,000 ha, a maximum water depth of 40 m and a storage capacity of 875 million m^3 (about a quarter of the storage capacity of the Mtera Dam) is a storage reservoir for the power plant at Nyumba ya Mungu (8 MW) and two other power plants, Hale (21 MW) and New Pangani Falls (66 MW) located further downstream (Fig. 2).

Beside the river impoundment through damming and hydropower plants, the Pangani river water is also withdrawn to support various irrigation developments on the Basin (Fig. 2). Historically, two types of irrigation systems have evolved on the Pangani River Basin (IUCN 2003; Shaghude 2006), namely, large-scale irrigation systems (Fig. 2) located exclusively in the lowland areas where inadequate rainfall requires the farmers to irrigate their farms to ensure optimum productivity, and small-scale traditional irrigation systems, typically found in the highland areas where irrigation is taken as a safeguard measure to maximize productivity of the small farms (resulting from population pressure). Most of these small farms are usually cultivated during both the rain season (where the farmers utilize the rains for sustaining farming productivity) and the dry season (where the farmers use the irrigation water for sustaining farming productivity).

Irrigation Water Demands and Demographic Changes on the Rufiji River Basin

The major irrigation developments on the Rufiji River Basin are located on the Usangu plain of the Great Ruaha catchment of the Upper Rufiji River Basin and the LRFP of the

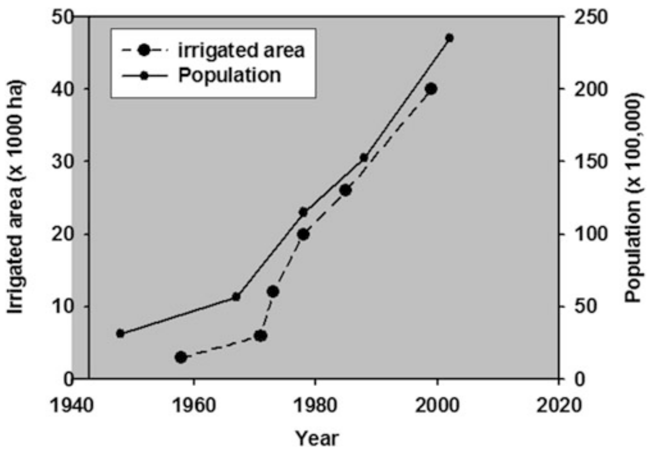


Fig. 3 Total irrigation land area (from 1948 to 1999) and the population of the Usangu plain (from 1958 to 2002) both increasing at approximately a steady growth rate with time (Data source: SMUWC (2001))

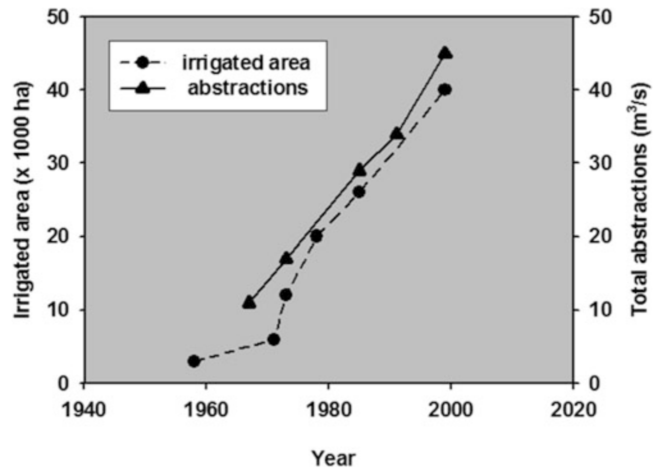


Fig. 4 Total land area under irrigation and total volume of water abstracted from the Great Ruaha River on the Usangu Catchment. Note that both the irrigated land area and the volume of water abstraction have been increasing at approximately a steady growth rate with time (Data source: (SMUWC (2001))

Lower Rufiji catchment. Studies conducted on the Great Ruaha River catchment revealed that to a large extent, the demographic pressure on the Usangu plain was driven by the presence of water and fertile land on the Usangu plain alluvial fans. The population on the Usangu plain is reported to have increased to more than six fold (Fig. 3) from 31,353 people in 1948 to 234908 in 2002 (SMUWC 2001; Sosovele and Ngwale 2002). The fastest growth rate during the late 1970s and early 1980s (Fig. 3) had been largely attributed to immigration of people from outside the Usangu plain rather than natural growth due to birth rates (Sosovele and Ngwale 2002).

Parallel with the demographic changes, the irrigated land area on the plain is reported to have grown from about 10,000 ha in 1970's to over 40,000 ha during the turn of the 21st century (SMUWC 2001; Fig. 4). Sosovele and Ngwale (2002) noted that although in principle the water abstractions on the Usangu plain were usually registered, illegal water abstractions had also expanded alongside with the irrigation expansion, exacerbating the problem of water in the Great Ruaha River.

The irrigated farms on the plain constitute of both large-scale and small-scale irrigation schemes Sosovele and Ngwale (2002). The large-scale irrigation schemes are owned and managed by the National Agriculture and Food Cooperation (NAFCO), which include the Mbarali, Kapunga and Madibira rice farms (Fig. 5). The smallholder's irrigation schemes are by contrast owned by the local communities. The large-scale irrigation schemes have the most efficient water regulating and distribution systems (SMUWC 2001). However, despite of their efficiency these systems continue to take water from the river during the dry season (even though they do not have dry season rice farming), for the domestic use by the villagers located on the downstream of the farms. During the transportation of water to these downstream villagers, large amounts of water are spilled onto the fields within the farms, where there is no crop, thereby contributing to significant water wastage (SMUWC 2001).

In smallholder farms, the irrigation systems are organized into irrigation blocks, where each irrigation system is served by one inlet, and water is passed from one field to another within the block. Farmers in these blocks cultivate rice at different phases and, could also use different varieties of rice (which have different growing times and water requirements), leading to "mosaic" irrigation practice (SMUWC 2001). This farming practice is again reported to contribute to significant wastage of water due to the fact that water is at times allowed to pass through a rice block which is already harvested (and therefore in no need of water). The system is also associated with significant water leakage when the water is distributed to the different fields. On the other hand, the improved smallholder farms use concrete weirs and water leakage is relatively lower. However, these improvements lead to improved access to water, which effectively promote the mosaic pattern, as the water is available over a relatively longer duration. The estimated mean annual water demand for both large-scale and small holder rice farms (including the water losses) for one ha of rice had been estimated at 31,325 m^3/ha (SMUWC 2001). The estimated water usage per hectare of rice are consistent with the estimates of 1.01/s/ha discussed by Magayane et al (2003). With the estimated irrigable area of 45,000 ha the annual irrigation water demand is estimated at 1400 million m^3.

As the for the Lower Rufiji catchment, demographic changes have been characterized by a relatively slower

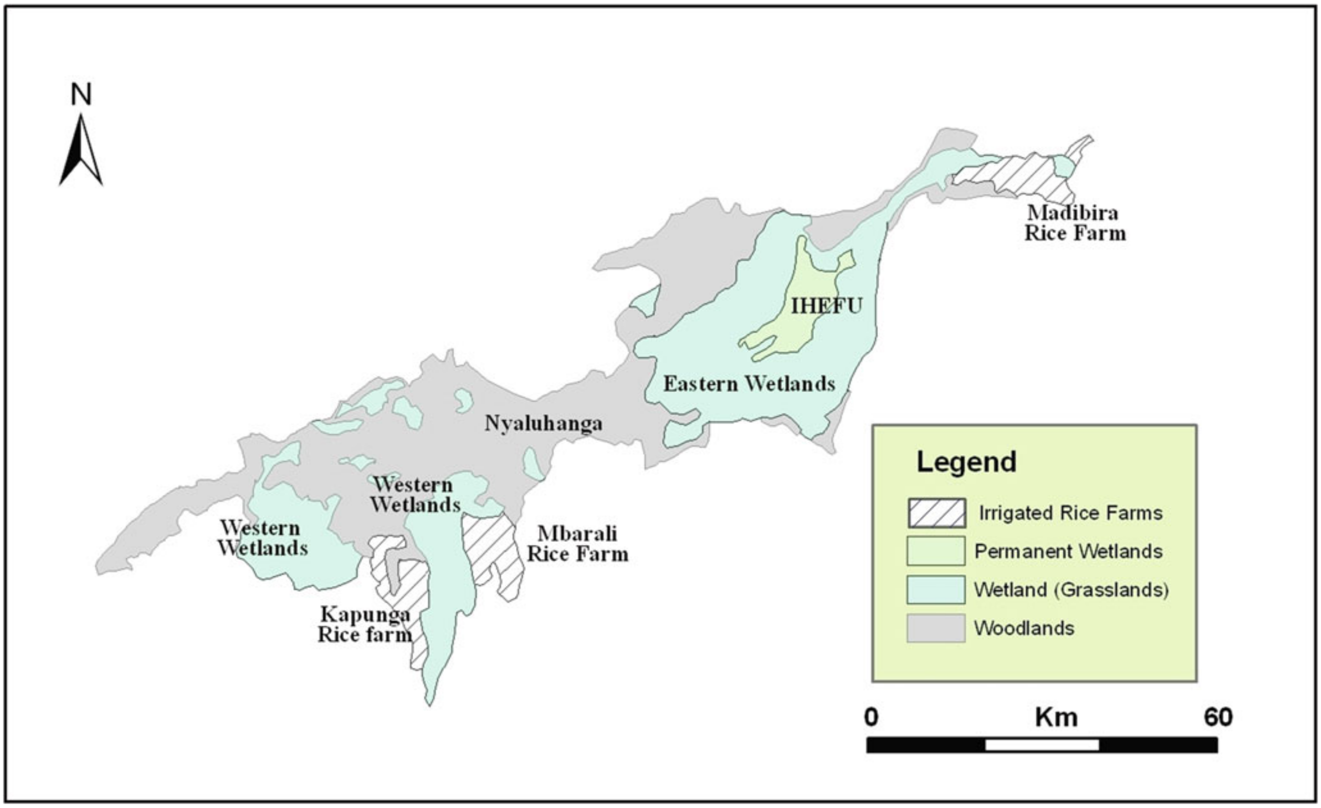
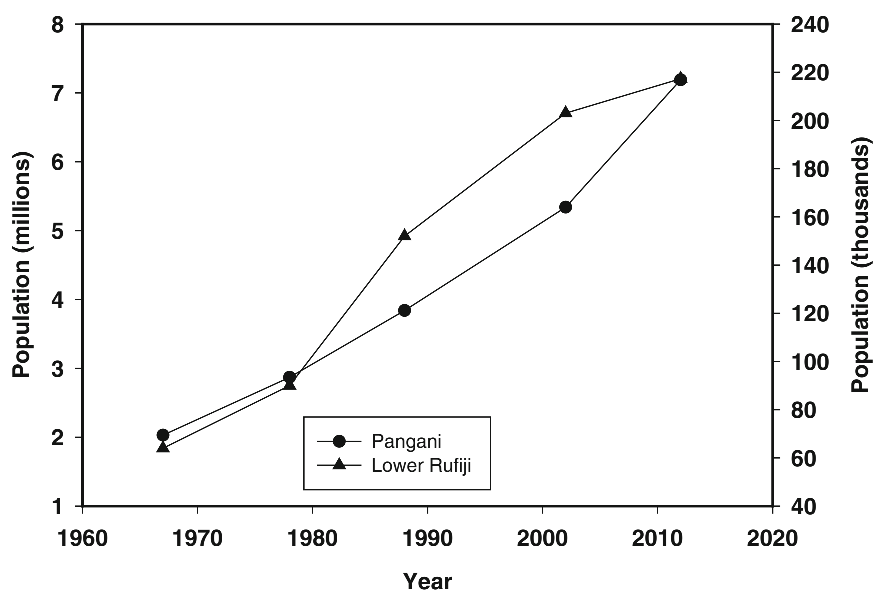


Fig 5 The Major land covers of the Usangu Plain (Modified after SMUWC (2001))

Fig. 6 Population statistics on the Lower Rufiji catchment and the Pangani River Basin (1967 to 2012). Note that the population of the Pangani River Basin is represented by four administrative Regions (Arusha, Kilimanjaro, Manyara and Tanga) (Source: URT (1988, 2006, 2013))



population growth rate between 1967 and 1977 and 2002 and 2012 and a relatively higher growth rate between 1977 and 2002 (Fig. 6), with the population growing from 64,000 people in 1967 to 217,274 people in 2012. Unlike the Great Ruaha catchment where irrigation agriculture has existed for several decades, irrigation agriculture on the Lower Rufiji is only slowly evolving. Historically, the agriculture of the Lower Rufiji and the people's livelihoods on the Lower Rufiji had strongly depended on the regular flooding of the Rufiji River, where the regular flooding of the river brought nutrients to the soil, which in turn sustained the cultivation of various agricultural crops, such as maize, rice, millet and peas along the flood plain which is up to 20 km wide (Bantje 1979; Bantje 1980; Ochieng 2002; Duvail and Hammerlynck 2007). The natural (wild) flooding of the Rufiji often compensated for the shortage of rainfall characterizing the climate of the Lower Rufiji (Shaghude 2004a). The wild floods of the Rufiji also helped to control vermin and created opportunity for second crop season (Bantje, 1980). Apart from sustaining the agricultural productivity of the Lower Rufiji catchment, the regular flooding of the Rufiji also helped to sustain the fish production on the system of lakes located along the LRFP and sustained the production of the mangroves in the Rufiji Delta. Thus, the agriculture system of the Lower Rufiji had been primarily flood dependent and less rainfall dependent. It was only after the villagization policy of 1973 when the local communities of the LRFP were forced to move to the less flood prone areas and less fertile areas located further upland of the flood plain that the local communities were compelled to practice rainfall dependent agriculture (Ochieng 2002).

Currently the national and local government authorities and other global institutions such as FAO are pushing forward policy strategies that are in favour of developing large scale irrigation schemes along the LRFP to optimize agricultural productivity of the LRFP which has been dwindling over the years. Within the framework of the planned policy strategy, donor agencies are being sought for constructing the biggest dam in Tanzania at Stiegler's Gorge along the upper reaches of the Rufiji River (Hamerlynck et al. 2010; Hakiardhi 2011; Fig. 1). The dam, designed in the 1970s, has three main objectives, firstly, hydropower production for speeding up the national plan for rural electrification, secondly, irrigated agriculture on at least 85,000 hectares of the LRFH (Hamerlynck et al. 2010), and thirdly, flood control along the LRFP (RUBADA 1981; Shaghude et al. 2004). Assuming that the Stiegler's Dam will eventually be constructed and assuming also that the water consumption pattern for the envisaged 85,000 ha allocated for rice irrigation scheme of the Lower Rufiji will have similar water consumption pattern as that discussed for Usangu plain (31,104 m^3/ha/yr), the anticipated water demand is being estimated at 2,644 million m^3 per year (Fig. 7).

Irrigation Water Demands and Demographic Changes on the Pangani River Basin

Studies conducted on the Pangani River Basin revealed that the population distribution on the Basin has been influenced by both the natural climatic and weather conditions as well as the soil conditions (IUCN 2003; Shaghude 2006). The highlands, characterized by volcanic rich soils and abundant rainfall, have a high population density (700–1000 people per km^2). The lowlands, by contrast which are characterized by semi-arid climate and higher likelihood of exposure to natural calamities such as floods, have a low population density of less than 50 people per km^2 (Shaghude 2006). Farming intensity in the highlands was further increased by the advent of foreigners and plantation farming of mainly coffee. On the Kilimanjaro highlands, the average landholding in 1943 was about 1.2 ha per household, but has decreased to an average landholding of 0.6 ha per household by early 2000's (IUCN 2003; Shaghude 2006). Lowland landholdings farm during the early 2000's were estimated at 10.5 ha per household (IUCN 2003). As plot sizes on the highlands approach the limit of viability to provide a livelihood, and as soils become exhausted by excessive cropping and irrigation, more and more people from the highlands are compelled to seek farmlands on the lowland areas, leading to intensification of the landuse pressure on the lowlands as well (IUCN 2003; Shaghude 2006). The population statistics from 1967 to 2012 show that the Basin population increased from 2,034,256 people in 1967 to 6,804,733 in 2012, with a more or less steady growth rate (Fig. 6).

Irrigation on the highlands is primarily characterized by small-scale traditional irrigational systems, where the driving force for irrigation had been to maximize farming yields of the farmer's small farm fields. Most of these small farms are usually cultivated during both the rainy season (where the farmers utilize the rains for farming productivity) and the dry season (where the farmers use the irrigation water for farming productivity (Shaghude 2006). The traditional irrigation system on the highlands, which consists of extensive furrow networks, dates back to pre-colonial times (IUCN 2003; Shaghude 2006). During the colonial times, water losses in these furrow networks were up to 85%, but in the course of time, management initiatives had been taking place to improve the water losses by installing control gates at specific locations and lining the furrows (to minimize water seepages) along the furrow networks. Such measures had aimed at minimizing the water losses to between 20 to 50% (IUCN 2003).

Unlike the highlands which are dominated by small scale traditional irrigation networks, the lowlands by contrast are dominated by large scale irrigation systems where the main driver for irrigation had been to maximize farming yields of the farms which are exposed to poor rain conditions. According to the Ministry of agriculture and food security irrigation department, the land area under traditional irrigation for the Arusha, Kilimanjaro, Tanga and Manyara (whose boundary are slightly larger than the boundary of the Pangani River Basin) increased from 15,115 ha in 1992 to 200,331 ha by the year 2003, while the land area under large scale irrigation grew from 3520 ha in 1992 to 26576 ha by the year 2003 (URT 2008), suggesting that the land area

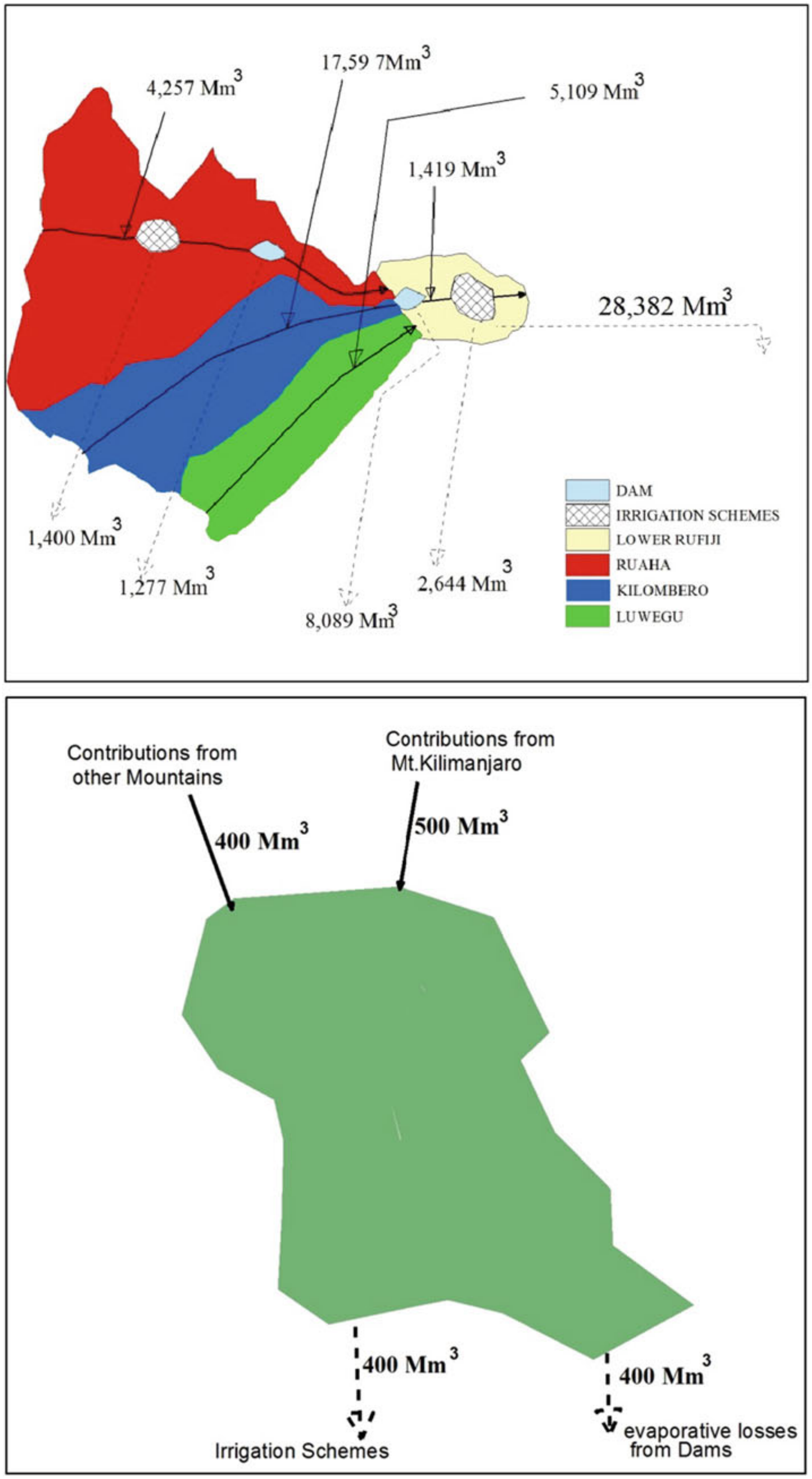


Fig. 7 Schematic representation of the Rufiji (*Top*) and Pangani (*Bottom*) river basins water budgets (Modified after Shaghude (2006). Note: Mm^3 = Million m^3)

under traditional irrigation and large scale irrigation had been growing to more than ten times and more than 8 times, respectively within ten years. Other studies (IUCN 2007) show that major crops which consume large amount of water in the basin include: coffee (1,000 m^3/ha/year), sugarcane (12,000 -17,000 m^3/ha/year), and flower industry (18,500 m^3/ha/year). The irrigation abstraction systems (both traditional and large scale) are estimated to use at least 400 million m^2 of water per annum (Fig. 7; IUCN 2003; Shaghude 2006).

The Hydropower Water Demand at Rufiji and Pangani River Basins

Apart from irrigation abstractions, the other major water abstractions from the two River basins are associated with the hydropower developments. Significant water losses on the Rufiji River Basin would be associated with the evaporative losses along the two major dams, the operating dam along the Great Ruaha catchment (the Mtera) and the planned dam (the Stiegler's Gorge) along the Lower Rufiji catchment (Fig. 7). The abstractions from Kidatu power plant are here neglected. This is because, although the water abstracted to run the turbines could be high, the water eventually returns to the river and the water usage in this sense is considered to be non-consumptive (URT 2002; Shaghude 2006). Evaporative losses in the existing Tanzanian large dams are considered to be well above 30% of the total inflow (URT 2002).

The Rufiji River mean monthly discharges to the Indian Ocean varies from month to month with lowest discharges of between 200–400 m^3/s during the dry season between June and November and highest discharges of between 500–1900 m^3/s during the rain season between December and May (Vorosmarty et al. 2004). The bank full discharges varies along the River, but is usually in the range between 2000–3000 m^3/s. The mean annual discharges is estimated at 900 m^3/s, which is equivalent to 28,382 million m^3 per year (Hafslund 1980; Shaghude 2004a; ASCLME 2012). The total inflow to Mtera dam from the Great Ruaha River (which contribute 15% of the total Rufiji discharges) could therefore be estimated at 4257 million (Fig. 7), while the total inflow to the Stiegler's Gorge would be the sum of the flows of the three tributaries Luwegu, Kilombero and Great Ruaha, each of which contribute 18%, 62% and 15%, respectively (that is, 5109 million m^3, 17,597 million m^3 and 4,257 million m^3, respectively), which sum to 26,963 million m^3 per year. Since the evaporative losses accounts to at least 30% of the total inflow URT 2002), the evaporative losses at Mtera and Stiegler's Gorge dams can be estimated at about 1277 million m^3 and 8089 million m^3, respectively (Fig. 7). The suspended sediment concentration in the river is usually in the range between 100 and 1000 mg/l, with typical values for the dry and wet season being 150 and 500 mg/l. respectively (Shaghude 2004a). The annual sediment discharge had been estimated at 15 – 25 million tons (VHL 1979; Shaghude 2004a).

As for the Pangani River Basin, the mean monthly River discharges to the Indian Ocean are also variable during the year with discharges ranging between 20–40 m3/s from July to March and discharges ranging between 50 – 100 during April to June (Pamba, 2015). Studies on available water resources in the Basin show that rainfall on Mount Kilimanjaro provides 1,600 million m^3 of fresh water annually (Hemp 2005), but only 500 million m^3 flow to the river networks or percolates to the groundwater storage systems. Other studies report that the rainfall on Mount Kilimanjaro provides about 55% of the surface and underground water storage in the Basin (IUCN 2003; Shaghude 2006).The contribution of other mountains to the surface and groundwater resources in the basin is estimated at 45% (equivalent to 400 million m3 per year), suggesting that total available water in the basin is about 900 million m^3 per year. The total inflow to the Nyumba ya Mungu dam have been estimated at 43.37 m^3/s (TANESCO 1994, cited in IUCN 2003) suggesting that the evaporative losses in Nyumba ya Mungu dam could be estimated at 13.0 m3/s, which is equivalent to 410 million m^3 per year.

Discussion

The water budgets of the Rufiji and Pangani River basins are summarized in Fig. 7. In the Rufiji River Basin, the mean annual Rufiji River discharge to the Indian Ocean which is estimated at 28,382 million m^3 is derived from the four main catchments, the Luwegu (5109 million m^3), Kilombero (17,596.8 million m3), Great Ruaha (4,257 million m^3) and Lower Rufiji (1,419 million m^3). From the Figure it is evident that main water abstraction or losses due to river impoundment by damming at Mtera (1,277 million m^3) and Stieglers Gorge (8,089 million m^3) and through irrigation on the Great Ruaha (1,400 million m^3) and Lower Rufiji (2,022 million m^3) would reduce the Rufiji flow by 45% from 28382 million m^3/year to 15,594 million m^3/year which is equivalent to 494 m^3/s. Considering that the River flow during the rainy season may exceed 2,500 m/s while during the dry season the River flow may be as low as 300 m/s (VHL 1979), it is evident that the reduced flow under the influence of irrigation and hydropower developments would have significant impacts on the estuarine, delta and the adjacent marine environments especially during the dry season. In the Pangani River basin, the presented results show that the River abstraction by irrigation and its impoundment by damming have reduced the total available water on the

Pangani River Basin by about 90%, suggesting that the coastal and marine environments (including the Pangani estuary) of the Pangani River Basin are more exposed to more severe water stressed conditions than the coastal and marine environments of the Rufiji.

Currently, the sea water intrusion along the most active deltaic braches of the Rufiji river delta is about 25 km inland (Mwalyosi 1988). Further river abstraction due to the planned irrigation and hydropower developments would lead to further intrusion of sea water with potential ecological and socio-economic impacts. Recent studies conducted on the Delta reveal that within a period of twenty years (1991–2011) at least 7000 ha of mangrove forest have been lost and most of the losses have been attributed to farmers invasion in search of better rice farms in the delta as their original farming areas are increasingly being invaded by salt water intrusion (Peter 2013).

In the Pangani River Basin, salt water intrusion and salinization of coastal soil have also been reported as issues of major socio-economic and ecological concern (Shaghude 2004b, 2006). According to Shaghude (2004b) and Shaghude (2006), 70 years ago crocodiles used to be common at Kimu, located about 4 km upstream of the Pangani estuary. However, in the course of time crocodiles are reported to have migrated further upstream to Kumba Mtoni (10–12 km) from the Pangani River mouth, in response to the changing brackish estuarine conditions which are not tolerable to crocodiles. The results of Shaghude (2004b) and Shaghude (2006) are consistent with the recent findings of Pamba (2015) which quantified the extent of salinity intrusion upstream of the Pangani River mouth by measuring the surface salinity values on seasonal basis. The study of Pamba (2015) noted that highest saline water intrusion was highest in December and January (the months during which the Pangani River flow is lowest) where the saline water intrusion extended 15 km from the River mouth. Apart from the upstream migration of species the saline water intrusion upstream has also been accompanied with changes in underground water quality and salinisation of coastal soils both with considerable socio-economic impacts (Sotthewes 2008; IUCN 2003).

The presented results for Rufiji and Pangani river basins are consistent with results discussed in other river basins in Africa. In the Tana River Basin for instance, changes due to the pressures of damming and water abstractions on the upstream parts of the river catchment is paralleled with significant reduction of the freshwater to the coast, with multipliable socio-economic impacts. According to Snoussi et al. (2007), more than one million people have been depending on the Tana River's flooding regime for their livelihoods, which include agriculture, fisheries of the lower basin and delta and prawn fisheries of the adjacent Ungwana Bay amongst others. However, the damming of the river is reported to have degraded the above socio-economic activities. In the Senegal River Basin, damming of the River at Manantali on the upper reaches and Diama on the lower reaches has been linked with observed salinization of estuarine soils, intrusion of seawater into estuarine groundwater tables with multipliable impacts on the agricultural productivity (Snoussi et al. 2007).

In addition to the above water stress related impacts (generally related with the upstream irrigation activities), other eastuarine/coastal environmental impacts of major concern are related with the sediment deficit (which are generally linked with the upstream damming) on the coastal areas of the two river basins. In the Rufiji River Basin, the delta morphology with its prolongation seawards towards the Mafia Channel had been attributed to the long-term depsotion of the sediments load derived from the Rufiji River during the Holocene (Euroconsult 1980). However, currently there are no evidence of Delta prolongation and the Delta is considered to be experiencing some minor erosion (Ochieng 2002). In the Pangani River Basin, coastal erosion on the immediate north of the Pangani River mouth has been reported as one of the major environmental issue of concern (Shaghude 2004b, 2006) and upstream damming at NYM had been considered to be among the major causative factors of the observed erosion north of the Pangani River mouth (Shaghude 2004b, 2006).

Apart from the presented socio-economic pressure on the water resources of the Rufiji and Pangani river basins, it is important to note that the climatic pressure on the two river basins is also different (being less severe on the Rufiji than on the Pangani). Direct evidence for climatic pressure on the two basins is derived mainly from the historical pattern of rainfall and temperature records. With regards to the Rufiji River Basin, trend analyses on historical rainfall data revealed increasing rainfall trends for the catchments on the Upper Rufiji but with no clear evidence of increasing or decreasing rainfall trends for the Lower Rufiji catchment (Shaghude et al. 2004). As for the Pangani River Basin, the historical pattern of rainfall data from four meteorological weather station located on the upper parts of the Basin revealed declining rainfall trends of between 2.5-12 mm per year at least during the last 60–100 years; suggesting a decrease of at least 250 mm of total annual rainfall per century (Shaghude 2006).

The results of Shaghude et al. (2004) are consistent with those discussed earlier by Mwandosya et al. (1998) and Agrawala et al. (2003) who used General Circulation Model (GCM) to compare the baseline climatic pattern of Tanzania with the expected climate pattern due to doubling of CO_2 which is anticipated at the beginning of the next century (IPCC 2001). The two studies project that, doubling of CO_2 in the atmosphere as, predicted by IPCC (2001), and would be associated with significant changes in the mean annual temperature, where it is projected that the mean annual temperature would increase by about 2.2°C throughout the country. The change in the mean annual temperature would in turn lead to an increase in rainfall in the south-eastern highlands of Tanzania, which includes the Upper Rufiji basin and a decrease in rainfall in the north-eastern highlands, which include the upper catchments of the Pangani River Basin. Thus, while the climatic pressure may potentially reduce the coastal impacts on the available water resource on the Rufiji River Basin due to the socio-economic pressure, the climatic pressure may potentially further increase the water stress on the Pangani River Basin.

Concluding Remarks and Recommendations

In view of the presented results, it is apparent that the socio-economic pressure associated with agricultural irrigation and water impoundments by damming for hydropower generation on the Rufiji and Pangani river basins are withdrawing considerable amount of water from the basins, leading to various levels of environmental pressure/threats on the estuaries and adjacent coastal habitats. In the Rufiji River Basin, the environmental pressure is not currently very critical. However, the anticipated imminent developments, associated with the addition of another large dam (the Stiegler's Gorge Dam) and another large-scale irrigation scheme (the LRFP irrigation scheme) will potentially lead to considerable reduction of the flow of the Rufiji River, especially during the dry season, which is currently estimated at 300 m^3/s. The reduced dry season discharges will enhance the salinity intrusion along the Rufiji delta distributary channels with considerable potential ecological and socio-economic impacts on the Rufiji River estuary, the Delta and the LRFP.

It is therefore recommended that the planned future developments on the Rufiji Basin be undertaken in a holistic approach (an approach which considers the national socio-economic benefits, environmental integrity as well as the local community's livelihoods). While the planned large scale Dam at Stiegler's Gorge is expected to boost the Tanzania energy capacity (RUBADA 1981; RUBADA 2001) a number of studies have been doubting as to whether the Dam will be economically feasibile to the local communities of Rufiji who had been depending on the flooding of the Rufiji for ages (Bantje 1979; Shaghude 2004a; Duvail and Hammerlynck 2007; Hamerlynck et al. 2010) and environmentally sustainable to the estuarine, coastal and marine habitats located on the lower parts of the Basin.

Historically, the flooding of the Rufiji regularly nourished the LRFP with nutrients and boosted the agricultural productivity (Bantje 1979; Shaghude 2004a; Duvail and Hammerlynck 2007; Hamerlynck et al. 2010). Regular flooding of the Rufiji had also historically sustained the Rufiji delta, estuary and the associated mangrove ecological system (Mwalyosi 1988). The anticipated damming at Stiegler's Gorge will lead to further reduction in Rufiji river flow to the Indian Ocean and sediments discharges to the sea, which in turn will negatively affect the estuarine and adjacent coastal and marine environments in the following manner: 1-The reduced river flow would increase the average salinity seaward of the delta with potential to change the nearshore ecological system of the Mafia channel, 2- Further upstream damming would lead to reduction in the River sediments discharge to the sea, with potential to inhibit delta formation and enhance coastal erosion and deepening of the tidal channels. The estuary and the adjacent coastal and marine areas sustain important natural resources such as fish and mangroves which in turn support the livelihoods of the local communities.

In the Pangani River Basin, abstraction of River water by irrigation and its impoundment by damming collectively withdraw at least 90% of the total available water resource making the Pangani River Basin under critical water stress condition (IUCN 2003; IUCN 2009). The dwindling trends of Pangani river flow to the sea and the reduced sediment load to the sea due to the upstream socio-economic activities have been paralleled with increased magnitude of salt water intrusion on the Pangani estuary, river bank erosion, salinization of coastal soils and coastal erosion (Shaghude 2004a; Shaghude 2006; PBWO/IUCN 2007; Sotthewes 2008).

Recognizing that water has increasingly been a scarce resource in the Pangani River Basin (IUCN 2003; IUCN 2006), the Pangani Basin Water Board (PBWB) was established in 1991 for the purpose of improving the water management in the Basin by formulating and coordinating a participatory and holistic approach of managing the water resource in the basin. After its establishment, a number of water management initiatives have been undertaken, namely: 1- Allocation of water rights and 2- Improvements of irrigation efficiency in the traditional furrow networks.

Water right policy was initiated to control water abstraction in the Basin. Under this policy, water users were obliged to hold water rights issued by the Pangani Board Water Office, the cost of which depended on the type of the water use (Mujwahuzi 2001; Shaghude 2006). However, it is reported that, the policy did not get fully support particularly from the traditional irrigation water users who were not fully involved in the policy formulation process and subsequent water resource management (Mujwahuzi 2001; Shaghude 2006). This had led to increasing number and magnitude of illegal water abstractions relative to the legal abstraction (Mujwahuzi 2001; IUCN 2003; Shaghude 2006). Turpie et al. (2003) and others (e.g. PBWO/IUCN 2007) noted that, although water supply in the basin primarily depended on precipitation from mounts Kilimanjaro, Meru and other highland areas of the basin, management of the resource need to be taken in a holistic approach over the entire catchment such that it involves all stakeholders. The water allocations set by the PBWB have been influenced by the government interests and underscored the local community participation in water resource management (Mujwahuzi, 2001). The environmentalists also noted that the natural environment has not been considered as a consumer of water and as such it did not receive direct water allocations from the PBWB (Turpie et al. 2003).

Due to severity of the water stress problem in the Pangani River Basin, measures to improve the irrigation water efficiency from 20–50% which had been taken by the PBWB should only be considered as temporary measures but long-term management options aiming at either further reduction of water losses from traditional irrigation farrow networks, application of modern water storage techniques or/and improvement in the catchment water retaining capacity need to be considered. Measures for minimizing water losses from the traditional furrow networks could involve conversion of the furrow networks to pipelines, which would get rid of both water leakages by infiltration and evaporation along the transport path.

Another management option which is being recommended for the Pangani River Basin is the application of modern innovative methods of water storage and recovery; Aquifer Storage and Recovery (ASR), which involves the recharge and storage of water in an artificial aquifer during rainy season and the recovery of the stored water when it is required (Pyne 1995; Vecchioli 1999; Shaghude 2006; Sheng and Zhao 2014). The aquifer essentially functions as a water bank, where deposits are made in time of surplus, typically during rainy season, and withdrawals are made when the available water is relatively less abundant, particularly during the dry season (Sheng and Zhao 2014).

The other management option recommended for the Pangani River Basin include, instituting management measures that would lead to improved basin and system level and its water retaining capacity (WCD 2000). This could include measures such as promotion of afforestation initiatives that are sustainable to the environment and discourage those which degrade the environments (e.g. afforestation of land covers using scrub, deciduous hard wood, eucalyptus and other similar vegetation), promotion of better farming practices as outlined by Critchlev (1991)) amongst others.

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References

Agrawala S, Mochleo A, Hemp A, Van Aalst M, Hetz S, Smith J, Meena H, Mwakifwamba S. M, Hyero T, Mwaipopo OU (2003) Development of climate change in Tanzania: focus on mount Kilimanjaro. Environment Directorate Development Cooperation, Working party on global and structural policies. http://www.oecd.org/env/cc/21058838.pdf. Accessed on 24 May 2015

ASCLME (2012) National marine ecosystems diagnostic analysis, Tanzania. Contribution to the Agulhas and Somali Current Large Marine Ecosystems Project (supported by UNDP with GEF grant financing), 92p

Bantje H (1979) The Rufiji agriculture system: impacts of rainfall, flood and settlements. BRALUP, University of Dar es Salaam research paper no. 62

Bantje H (1980) Floods and famines: a study of food shortages in Rufiji district. BRALUP, University of Dar es Salaam, research paper no. 63

Critchlev W (1991) Looking after our land, soil and water conservation in dryland Africa. Oxfarm, London, 84p

Duvail S, Hammerlynck O (2007) The Rufiji River flood: plague or blessing? Int J Biometeorol 52:33–42

Euroconsult (1980) Identification study on the ecological impacts of the Stiegler's Gorge power and flood control project, Part III, Analysis of physical and ecological impacts. Consultancy Report submitted to Rufiji Basin Development Authority (RUBADA)

GWSP (2005) The global water system project: science framework and implementation of activities. Earth System Science Partnership, 76p. UNT Digital Library. Available at http://digital.library.unt.edu/ark:/67531/mettadc11879/. Accessed on 24 May 2015

Hafslund AS (1980) Stiegler's Gorge power and flood control development. Project planning report, vol 1. Consultant report submitted to Rufiji Basin Development Authority (RUBADA), 136p

Hakiardhi (2011) Land grabbing in a post investment period and popular reaction in the Rufiji River Basin: a research report by Hakiardhi 2011, 44p

Hamerlynck O, Duvail S, Hoag H, Yanda P Jean, Paul JL (2010) The large scale irrigation potential of the lower Rufiji floodplain: reality or persistent myth?

Hemp A (2005) Climate change-driven forest fires marginalize the impact of ice cap wasting on Kilimanjaro. Glob Chang Biol 11:1013–1023

IPCC (2001) Climate change, 2001: impacts, adaptation and vulnerability. Contribution of Working Group II to the third assessment report of the Intergovernmental Panel on Climate Change published for the Intergovernmental Panel on Climate Change, Cambridge University Press, ix + 1005 pp

IUCN (2003) Eastern African Programme 2003. The Pangani river basin: a situation analysis, IUCN, Regional Office, Nairobi, xvi + 104 pp

IUCN (2006) Hydrology and system analysis Vol. 1 of 2, the hydrology of the Pangani River basin report 1. Pangani River Basin Flow Assessment Initiative, Moshi, 62 pp. https://cmsdata.iucn.org/downloads/hydrology_and_system_analysis_volume_1.pdf. Accessed 24 May 2015

IUCN (2007) Socio-economic baseline assessment report: the role of river systems in household livelihoods Pangani River basin flow assessment, Moshi. International Union for Conservation of nature and Pangani Basin Water Board, 58pp. Available at https://cmsdata.iucn.org/downloads/socio-economic_assessment.pdf. Accessed on 24 May 2015

IUCN (2009) IUCN Eastern and Southern Africa Programme 2009. The Pangani river basin situation analysis, 2nd edn, IUCN Regional Office Nairobi, xii + 82 pp. https://portals.iucn.org/library/efiles/documents/2009-073.pdf. Accessed 24 May 2015

Kabanda TA, Jury MR (1999) Inter-annual variability of short rains over northern Tanzania. Clim Res 13:231–241
Kashaigili JJ, McCartney MP, Mahoo HF, Lankford BA, Mbilinyi BP, Yawson DK, Tumbo SD (2006) Use of a hydrological model for environmental management of the Usangu wetlands, Tanzania. Colombo, Sri Lanka: International Water Management Institute. 48p. (IWMI Research Report 104). Available at http://www.iwmi.cgir.org/Publications/IWMI_Research_Reports/PDF/pub104/RR104.pdf. Accessed on 24 May 2015
Machibya M, Lankford B, Maloo HF (2003) Real or imagined water competition? The case of rice irrigation in the Usangu basin and Mtera/Kidatu hydropower, Tanzania. In: Proceedings of Hydro Africa 2003 international conference in hydropower, 17–19 November 2003, Arusha, Tanzania. http://r4d.dfid.gov.uk/PDF/Outputs/Water/R8064Ruaha10-Real_or_Imagined_Competition-paper.pdf. Accessed on 24 May 2015
Magayane M, Lankford B, Mahoo HF (2003) Real or imagined competition? The case of irrigation and hydropower in Usangu plains. In: Proceedings of Hydro Africa 2003 international conference in hydropower, Arusha Tanzania, 17–19 November 2003, 10 pp
Mdemu MV, Magayane MD (2005) Comparison assessment on water use and damage between modern and traditional rice irrigation schemes: case of Usangu Basin, Tanzania. Int J Environ Res Public Health 2:335–342
Mujwahuzi MR (2001) Water use conflicts in the Pangani Basin. In: Ngana JO (ed), Water resources management in the Pangani River Basin: challenges and opportunities. Dar es Salaam University Press, Dar es Salaam, pp 128–137. https://cmsdata.iucn.org/downloads/hydrology_and_system_analysis_volume_1.pdf. Accessed 24 May 2015
Mwalyosi RBB (1988) Environmental impacts of the proposed Stiegler's Gorge hydropower project, Tanzania. Environ Conserv 15:250–254
Mwandosya MJ, Nyenzi BS, Luhanga ML (1998) The assessment of vulnerability and adaptation to climate change impacts in Tanzania. The Centre for Energy, Environment, Science and Technology (CEEST), Dar-es-Salaam
Ndomba PM (2010) Modelling of sedimentation upstream of Nyumba ya Mungu Reservoir in Pangani River Basin. Nile Basin Water Sci Eng J 3:1–38
Ochieng CA (2002) Research masterplan for the Rufiji floodplain and delta 2003–2013. REMP technical report no. 28
Pamba S (2015) Transport and dispersion pattern of suspended particulate matter in estuaries of Tanzania: a case of Pangani estuary. PhD (Marine Sciences) thesis, University of Dar es Salaam, xi + 212 pp
Parish ES, Kodra E, Steinhaeuser K, Ganguly AR (2012) Estimating future global per capita water availability based on changes in climate and population. Comput Geosci 42:79–86
Peter L (2013) Assessment of the status of mangrove vegetation and their degradation in Rufiji delta, Tanzania. Masters of Philosophy degree thesis submitted to the University of Ghana, 150p. http://ugspace.ug.edu.gh/bitstream/handle/123456789/5367/Leonard%20Peter_Assessment%20of%20the%20Status%20of%20Mangrove%20Vegetation%20and%20their%20Degradation%20in%20Rufiji%20Delta%20in%20Tanzania_2013.pdf?sequence=1. Accessed on 24 May 2015
Pyne RDG (1995) Groundwater recharge and wells: a guide to aquifer storage and recovery in Florida. CRC Press, Gainesville, 620p, 27p
RUBADA (1981) Rufiji basin development authority: irrigated agriculture development in the Lower Rufiji valley. Annexes 1–8, 147p. RUBADA
RUBADA (2001) Development of the Stiegler's Gorge multi purpose project for hydropower, agriculture and flood control. Rufiji Basin Development Authority
Shaghude YW (2004a) Coastal impacts of water abstraction and impoundment in Africa: The case of Rufiji River. Final report submitted to START, 75p. Available at: http://www.oceandocs.org/bitstream/handle/1834/187/coastal.pdf;jsessionid=D47D91F56B727998D9A7152A51BC12A3?sequence=1. Accessed on 24 May 2015
Shaghude YW (2004b) Shore morphology and sediment characteristics, south of Pangani river, coastal Tanzania. Western Indian Ocean J Mar Sci 3:93–104
Shaghude YW (2006) Review of water resource exploitation and landuse pressure in the Pangani river basin. Western Indian Ocean J Mar Sci 5:195–207
Shaghude YW, Lobora AL, Mayunga JS (2004) Quantification of water abstration and landuse changes in Lower Rufiji catchment. In: YW Shaghude (ed) Coastal impacts of water abstraction and impoundment in Africa: the case of Rufiji river. Final report submitted to START, pp 47–62
Shaghude EE, Lobora YW, Mpeta AL, Matitu EJ, Matari MR (2008) Inter-linkanges between socio-economic dynamics and climatic variability on Lower Rufiji catchment. In: Arthurton R, Le Tissier M, Snoussi J, Kitheka J, Shaghude Y, Kane A, Flöser G, Kremer H (eds) LOICZ – Global change assessment and synthesis of river catchment – Coastal Sea Interactions and Human Dimensions in Africa, LOICZ reports and studies no. 30. LOICZ IPO, Geesthacht, pp 83–105
Sheng Z, Zhao X (2014) Special issue on Managed Aquifer Recharge: powerful management tool for meeting water resources challenges. J Hydrol Eng 20. doi: 10.1061/(ASCE)HE.1943-55840001139. Available at http://ascelibrary.org/doi/abs/10.1061/(ASCE)HE
SMUWC (2001) Sustainable management of Usangu water and its catchment: final report water resources. UK Department of International Development and Ministry of Water and Livestock Development, Dar es Salaam, 209p
Snoussi M, Kitheka J, Shaghude YW, Kane A, Arthurton R, Le Tissier M, Virji H (2007) Downstream and coastal impacts of damming and water abstraction in Africa. Environ Manag 39:587–600
Sosovele H, Ngwale JJ (2002) Socio-economic root causes of the loss of biodiversity in the Ruaha catchment area. Report submitted to WWF, Tanzania, March 2003, vi + 48 pp
Sotthewes W (2008) Forcing the salinity distribution in the Pangani estuary. Final report, Delft University of Technology, pp 81–85
TANESCO (1994) Environmental impact assessment of the redevelopment of hydro electric power station at Pangani Falls: baseline studies: mitigation of adverse impacts and monitoring the attainment of the intended mitigation. IVO – NORPLAN Joint Venture
Turpie JK, Ngaga YM, Karanja FK (2003) A preliminary economic assessment of the water resources of the Pangani River Basin, Tanzania: economic value, incentives for sustainable use and mechanisms for financing management. Submitted to IUCN Eastern Africa Regional Office and Pangani Basin Water Office, November 2003, xii + 100 pp. https://www.mysciencework.com/publication/show/1715832/preliminary-economic-assessment-of-water-resources-of-the-pangani-river-basin-tanzania-economic-values-and-incentives. Accessed 24 May 2015
UNFPA (2011) The state of the world population 2011: people and possibilities in a world of 7 billion. United Nations Development Fund, New York, USA, 126 p. Available at http://www.unfpa.org/publications/state-world-population-2011. Accessed on 24 May 2015
URT (1988) United Republic of Tanzania – 1988 population census, National profile, The population of Tanzania, the analytical report. Bureau of statistics, President's Office, Planning Commission, Dar es Salaam, iv + 195 pp. http://www.nbs.go.tz/tnada/index.php/catalog/13. Accessed May 2015
URT (1992) United Republic of Tanzania – The energy policy of Tanzania. Ministry of water, energy and minerals, April 1992. Printed by the Government Printer, Dar es Salaam, v + 31 pp.

https://mem.go.tz/.../0001_17022013_National_Energy_Policy_2003.pd. Accessed May 2015

URT (1997) United Republic of Tanzania: natural resources aspects of sustainable development in the United Republic of Tanzania. Document provided by the Government of the United Republic of Tanzania to the 5th session of the United Nations Commission on Sustainable development. http://www.un.org/esa/agenda21/natlinfo/countr/tanzania/natur.html. Accessed on 24 May 2015

URT (2002) United Republic of Tanzania – National Water Policy, July 2002. Ministry of Water and Livestock Development, 49 pp. www.egov.go.tz/egov_uploads/documents/Water_Policy_en.pdf. Accessed May 2015

URT (2006) United Republic of Tanzania – Tanzania census 2002: analytical report, Vol. X. National Bureau of Statistics, Ministry of Planning Economy and Empowerment, Dar es Salaam, August 2006, xiv + 193 pp. http://www.nbs.go.tz/nbs/takwimu/references/2002popcensus.pdf. Accessed May 2015

URT (2008) Ministry of Agriculture Food Security and Cooperatives final report: Agriculture Sector Review and Public Expenditure Review 2008/09, November 2008. http://www.agriculture.go.tz/publications/english%20docs/Agricultutre%20Sector%20Review%20and%20Public%20Expenditure%20Review%20200809.pdf. Accessed May 2015

URT (2009) United Republic of Tanzania. National Irrigation policy. Ministry of Water and irrigation, Dar es Salaam, March 2009, 70p

URT (2013) United Republic of Tanzania, 2012 population and housing census: population distribution by administrative areas. National Bureau of Statistics, Ministry of Finance, Dar es Salaam and Office of Chief Government Statistician, President's Office, Finance, Economy and Development Planning, Zanzibar. March 2013, xix + 235 pp. http://www.nbs.go.tz/nbs/takwimu/census2012/Migration_and_Urbanisation_Monograph.pdf. Accessed May 2015

Valls JX (1999) Carrying capacity in East Sub-Saharan Africa: a multi-level integrated assessment and a sustainable development approach. Doctoral thesis, UPC (Universitat Polytecnica de Catalunya), 330p. http://www.tesisenxarxa.net/TESIS_UPC/AVAILABLE/

Vecchioli PGJ (1999) Aquifer storage and recovery: is it South Florida's solution for regional waterStorage? South Florida Restoration Science Forum, May 1999. http://sofia.usgs.gov/sfrsf/presentations/vecchioli.html. Accessed on 24 May 2015

VHL (1979) Report on the hydraulic studies in the Lower Rufiji River, vol 1, Main report submitted to Rufiji Basin Development Authority (RUBADA), 136p

Vorosmarty CJ, Fekete BM, Tucker BA (2004) Monthly mean river discharge at gauging station Stiegeler's Gorge doi:10.1594/PANGAEA.219175 Related to: Vörösmarty Charles J, Fekete BM, Tucker BA (1998) Global river discharge, 1807–1991, V 1.1 (RivDIS). Data set. Available on-line at: http://www.daac.ornl.gov from Oak Ridge National Laboratory Distributed Active Archive Center, Oak Ridge, Tennessee, U.S.A., http://www-eosdis.ornl.gov/RIVDIS/rivdis.html Accessed on 24 May 2015

WCD (2000) Dams and development: a new framework for decision making. The report of the world commission on dams, Nov. 2000. Earthscan Publication Ltd, London/Sterling

The Rufiji Estuary: Climate Change, Anthropogenic Pressures, Vulnerability Assessment and Adaptive Management Strategies

Greg M. Wagner and Rose Sallema-Mtui

Abstract

The Rufiji Estuary supports the largest mangrove area (48,030 ha) in Tanzania, which, together with nearby seagrass beds, numerous coral reefs and islands, form an interacting seascape that provides invaluable ecological services to the Western Indian Ocean. For centuries, the Rufiji resources have been utilized sustainably by coastal people for food, firewood and building materials. However, in recent decades, the Estuary has been severely affected by a complexity of both climate change factors and anthropogenic pressures. A WWF/GEF Coastal Resilience to Climate Change Project was conducted between 2007 and 2009, involving repeated quantitative ecological plot assessment, rapid assessment, remote sensing (based on satellite images taken in 1991 and 2000) and recording indigenous knowledge through social science methods. Change analysis showed several definite trends. Along exposed seaward edges (not sheltered by reefs or islands), there was drastic erosion and mangrove loss attributed to increased wave activity and sea level rise, with over 25 m of coastline being lost along sections of the estuary over a two-year period. Along other seaward edges, some accretion and mangrove growth occurred due to being sheltered from wave action and having adequate sediment load from Bumba channel, which has received the main flow of Rufiji since the shift of the river in 1978. The 1998 El-Nino flooding event caused drastic mangrove loss in riverine areas. On several landward edges of the delta, migration of mangroves inland was detected. Overall, however, losses of mangroves in the Rufiji were clearly greater than gains. Anthropogenic pressures include deforestation in the watershed area and large scale cutting of mangroves for timber and conversion to rice farms. Recommended mitigation and adaptive management strategies include the establishment of buffer zones, mangrove planting, awareness raising and enforcement.

Keywords

Rufiji Estuary • Seascape • DPSIR analytical framework • Climate change • Socioeconomic factors • Anthropogenic Pressures • Environmental state changes • Vulnerability assessment • Impacts on human wellbeing • Mitigation • Adaptive management strategies

Introduction

Background and Objectives of the Study

The Rufiji Estuary is almost entirely covered by mangroves, thus a study of this Estuary must primarily focus on the vast mangrove ecosystems it supports. The major objective of

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this study was to detect changes in the Rufiji mangroves over time in relation to climate change factors, to assess their vulnerability to such factors and to come up with adaptive management strategies. However, in doing so, it became obviously important to have the additional objective of investigating anthropogenic pressures which have also lead to definite state changes in the mangrove ecosystems during recent decades. Moreover, climate change and anthropogenic pressures interact in the way that they affect the environment. During the investigation, however, effort was made to understand and differentiate between changes caused by these two types of forces. Thus, the adaptive management strategies developed are aimed at increasing the resilience and resistance of mangroves and associated ecosystems to climate change as well as at enhancing environmental awareness in the human communities and promoting sustainable resource utilization.

Global climate change is currently one of the greatest challenges to our environment and society. Among the most at-risk ecosystems are mangroves (McLeod and Salm 2006). Coral reefs, which are often associated with mangroves, are probably even more at risk and, in fact, have already been substantially affected (Grimsditch and Salm 2006). As a consequence, the ecological interactions that normally occur between these two types of ecosystems are also at risk, thus further compounding and increasing the severity of climate change effects. The challenge for resource managers today is to adequately assess the vulnerability of mangroves and associated ecosystems to ongoing climate change and to develop integrated adaptive management plans that will increase the capacity of both the natural environment and human environment to cope with predicted changes. This involves making necessary changes in human-ecosystem interactions.

Ironically, mangrove ecosystems are, at once, major victims of climate change and a potential mitigating factor in the fight against it. Since mangroves make a substantial contribution to carbon storage, mangrove conservation and restoration can play an important role in reducing greenhouse gas emissions (Alongi 2014). Mangroves are both an important atmospheric carbon sink and an essential source of oceanic carbon (Duke et al. 2007). Mangroves have 50 times greater carbon sequestration potential than other tropical forests (Sandilyan and Kathiresan 2012). Donato et al. (2011) found that mangrove soils in the Indo-Pacific ranged in depth from 0.5 m to over 3 m and accounted for 49-98% of carbon stored in these systems. According to Alongi (2014), globally mangroves contribute 10–15% (24 Tg C y^{-1}) of all coastal sediment carbon storage, though they occupy only 0.5% of the coastal area.

This chapter on the Rufiji Estuary highlights just some of the main findings of a study that was part of a project on "Coastal Resilience to Climate Change: Developing a Generalizable Method for Assessing Vulnerability and Adaptation of Mangroves and Associated Ecosystems". It was a Global Environment Facility (GEF) Medium-Sized Project implemented under United Nations Environment Program (UNEP), with management and co-funding being provided by World Wide Fund for Nature (WWF). This GEF/WWF Project was undertaken in Tanzania, Fiji and Cameroon. The findings documented in this chapter are based on two reports submitted to WWF. One report (Wagner 2008) described the results of a baseline study conducted in 2007 on the mangrove ecosystems of Rufiji, Mafia and Kilwa in Tanzania. The second report (Wagner and Sallema-Mtui 2010) documented a repeat study and change analysis undertaken in 2009 in the same areas. However, this chapter focuses on the Rufiji Estuary, making only minor reference to Mafia and Kilwa.

Driver-Pressure-State-Impact-Response Analytical Framework

This chapter examines the cause-and-effect relationships between ocean-climate variables, mangrove ecosystems and human society. It applies the UNEP Human-Environment Interaction Analytical Approach (UNEP 2006), which is built on the Driver-Pressure-State-Impact-Response (DPSIR) Framework (Organisation for Economic Co-operation and Development 1993). According to this analytical approach, the term 'drivers' refers to the root causes of change, including socio-economic factors such as rapid population growth, poverty and inadequate education/environmental awareness, as well as largely natural factors such as global climate change. The term 'pressures' refers to the immediate causes of environmental change, which encompass human activities that directly affect the environment (e.g., harvesting mangroves and conversion of mangrove forests to rice farms) and climate-driven pressures such as sea level rise and increased wave activity. 'State' refers to the state or status of the natural environment as well as trends in that state. 'Impacts' refers to the effects on human wellbeing as a result of state changes in ecosystem goods and services available to human populations. 'Responses', most of which are management interventions, refer to ways in which governments, non-governmental organizations, the private sector and local communities react to, or deal with, changes in the state of the environment and their impacts on society (Wagner 2007). In scientific reports, the term 'impacts' is often used when referring to the 'effects' of certain factors on the environment. However, in this chapter, the above definitions of terms are applied.

Vulnerability of Mangroves and Associated Ecosystems to Climate Change

Global climate change is often considered as a natural driver of change, though there is clear evidence that it has been exacerbated by anthropogenic activities (Clark and Webster 2003; Arthurton et al. 2007). As a driver, it puts a number of pressures on mangroves and other coastal ecosystems, primarily increased/fluctuating temperatures (in both air and water), increased levels of CO_2 in the atmosphere and hydrosphere, changes in precipitation sometimes accompanied by flooding, sea level rise, as well as extreme events in the atmosphere and hydrosphere, particularly storms, hurricanes, cyclones, and El-Nino/Southern Oscillation (ENSO) patterns, all of which cause increased wave action (Field 1995; McLeod and Salm 2006). Projected increases in temperature and levels of CO_2 are likely to enhance mangrove photosynthesis and growth rates and, in fact, may cause mangroves to move poleward (UNEP 1994; Ellison 2005), increasing their area of coverage. However, the other climate change factors mentioned above may have profound negative effects on mangroves.

Climate change and related factors subject many coastal areas to increasing risks, including coastal erosion, seawater and floodwater inundation of low-lying coastal areas, changes in sediment budgets and rising water tables (IPCC 2007). Coastal wetlands, including salt marshes and mangroves, are projected to be negatively affected by sea level rise, especially where they are constrained on their landward side or starved of sediments. This calls for various adaptive measures that integrate protection and conservation of coastal and marine ecosystems (Sallema and Mtui 2008).

Evidence shows that there has been substantially increasing frequency, intensity and duration of tropical storms since the 1970s and this trend is likely to accelerate (IPCC 2007). As a result, it is predicted that there will be higher waves, stronger storm surges and increased frequency of high water events, which when combined with increased sea level, will affect mangrove health and species composition and will result in mangrove destruction. Changes in salinity, inundation and sediment budgets will affect mangrove physiology (McLeod and Salm 2006).

According to Gilman et al. (2008), relative sea level rise is a great threat to mangroves. During the 20th century, global mean sea level rose at a rate of about 1.7 mm/yr (IPCC 2007). According to State of the Climate 2013, the rate increased to 3.2 mm/yr in 2013 (Merrifield et al. 2014). For the 21st century, mean sea level projections range from 90 to 880 mm, according to Houghton et al. (2001) and from 210 to 500 mm, according to IPCC (2007). Although sea level rise poses a significant threat to mangrove ecosystems, they can adapt and survive if it occurs slowly (Ellison and Stoddart 1991) and if space for migration exists (McLeod and Salm 2006). This is due to the fact that mangroves have suitable adaptations, particularly aerial and prop roots, which allow them to grow vertically upward. Moreover, mangrove ecosystems are capable of undergoing rapid processes such as decomposition and soil formation in order to keep up with shifting or changing conditions. In deltas, the rate of accretion of sediments often exceeds any projected rise in sea level. However, in specific local situations, mangrove migration may be hindered by the presence of roads, urbanization, agricultural fields or unsuitable topography, e.g., steep slopes (McLeod and Salm 2006). Sea level rise will have the greatest negative impact on mangroves where there is net lowering in sediment elevation and where there is limited area for landward migration (Gilman et al. 2008).

A whole range of environmental factors may affect the response of mangroves to sea level rise. These include substrate type, coastal processes, local tectonics, freshwater availability, sediment load, salinity, climatic variability, as well as local tidal and ocean currents. The size and shape of watersheds and land use upland will also affect the ability of mangroves to cope with climate change. Sea level rise is expected to reduce mangrove area coverage and species diversity on small islands with sediment limited environments (McLeod and Salm 2006).

Coral reefs, often associated with mangroves, are probably even more vulnerable to predicted climate change, particularly in that they bleach rapidly and dramatically in response to a rise in sea surface temperature of only 1 or 2°C (Grimsditch and Salm 2006). Other climate-related changes that can affect coral reefs include increases in CO_2 levels, changes in salinity, increased sedimentation due to flooding and runoff, sea level rise and extreme events leading to strong wave action. Increased CO_2 levels in the hydrosphere lead to ocean acidification, which interferes with calcium carbonate formation. Unlike mangroves, effects on coral reefs due to climate change have already been substantial. During the 1997/1998 El Nino Southern Oscillation (ENSO) event, a severe global bleaching event substantially affected coral reefs in 50 countries. The worst affected ocean area was the Western Indian Ocean where 30% regional mortality occurred (Obura 2005).

Degradation of coral reefs can, in turn, have significant effects on mangrove ecosystems in terms of reduced shelter from wave action, changes in ocean and tidal current patterns as well as disturbance of food chains and migration patterns of organisms that move between mangroves and coral reefs. Likewise, degradation of mangroves may have significant effects on coral reefs (Wagner 2007) due to increased sediment load flowing into the ocean, general reduction in water quality in the coastal zone and, again, disturbance to migrating organisms.

Mangroves of Tanzania

According to remote-sensing data, the total area covered by mangroves in mainland Tanzania dropped only slightly from 109,593 ha in 1990 (Semesi 1991) to 108,138 ha in 2000 (Wang et al. 2003). There are ten species of mangroves found in mainland Tanzania (including Mafia Island). Eight of these species are most commonly reported, namely, *Avicennia marina*, *Bruguiera gymnorrhiza*, *Ceriops tagal*, *Heritiera littoralis*, *Lumnitzera racemosa*, *Rhizophora mucronata*, *Sonneratia alba* and *Xylocarpus granatum*, all of which are found in the Rufiji Estuary.

Vulnerability of mangrove ecosystems to climate change can be considerably influenced by human-caused degradation. There are a number of human pressures on mangrove ecosystems in Tanzania that have had substantial negative effects (Wagner et al. 2003; Wagner et al. 2004; Wagner 2007). A causal chain analysis by Francis et al. (2001) revealed that the major pressures causing the loss and modification of mangrove ecosystems were the over-harvesting of mangroves (thought to account for 46% of the degradation) for firewood, charcoal-making, building poles and boat-making; as well as clear-cutting (30%) for aquaculture, agriculture, solar salt works, urbanization, as well as road and hotel construction. Other human pressures include dragging seine nets under mangrove canopies (Akwilapo 2001; Wagner et al. 2001) and dynamite fishing in estuaries (Wagner et al. 2004).

The major drivers or root socio-economic causes of the human pressures mentioned above are rapid population growth along the coast, poverty, inadequate education and environmental awareness, inadequate enforcement, the policy of open access, and increasing demands for marine and coastal resources (Wagner et al. 2003; Wagner 2007).

Background Information on the Rufiji Estuary Study Area

The Rufiji Estuary and surrounding area forms a seascape (Fig. 1) consisting of a variety of interacting coastal and marine ecosystems that support diverse resources, upon which human life in the area has depended for centuries. The Rufiji Estuary supports by far the greatest mangrove area in Tanzania, with a coverage of 49,799 ha in 1990 (Semesi 1991) which dropped slightly to 48,030 ha in 2000 (Wang et al. 2003), thus comprising almost half of the total mangrove area of Tanzania. Very large basal area of over 1200 cm^2 per 25-m^2 plot has been recorded in the Estuary (Wagner et al. 2003). Just offshore from this huge delta is Mafia Island, which is surrounded by numerous coral reefs. Towards the southern end of the Rufiji Estuary extends the Songo Songo Archipelago of coral reefs and small islands (from Ukuza southward in Fig. 1), which supports a rich diversity of marine life (Wagner 2004). Profuse seagrass beds are found intermingled amongst the coral reefs, islands and mangroves in this vast seascape.

Mgaya (2004) wrote a literature review on the vulnerability of mangroves of the Rufiji Delta and adjacent coral reefs to climate change, which also presented adaptive strategies which could be considered for the area.

Mangrove forests are considered as critical habitats with great ecological and socio-economic value. They are keystone ecosystems in that they have high productivity, producing large quantities of organic matter that serves as food for many organisms living both within and outside the forest, since much of the organic matter produced is exported to other areas of the marine environment (Wagner et al. 2003). The mangroves of the Rufiji Estuary serve as feeding, breeding and nursery grounds for a great variety of invertebrates and fish that inhabit the offshore reefs and seagrass beds around Mafia Island, as well as the open waters of the Western Indian Ocean. In addition, mangroves filter river water and facilitate the settlement of sediments, which would otherwise be detrimental to surrounding seagrass beds and coral reefs. Mangroves also play an important role in stabilizing the coastline, thus preventing shoreline erosion. Without the extensive mangrove coverage of the Rufiji Estuary, it is unlikely that the offshore coral reefs would have developed and survived.

From time immemorial, the mangroves of the Rufiji delta have been utilized sustainably as a source of firewood, charcoal, building poles, materials of boat construction, tannin and traditional medicines. However, in recent decades, these same uses have become threats and pressures due to increased population around the area and the commercialization of such products by selling them to the huge population of Dar es Salaam city located only about 100 km to the north.

The Rufiji River is the largest river in Tanzania and its catchment area drains 20% of the country. Tides reach about 25 km upstream (Kajia 2000). The delta floods twice yearly during the inter-monsoonal rains, i.e., November-December and March-May. According to Semesi (1991), the mean discharge of the river ranges from 202 m^3/sec in October, at the end of the long dry season, to 2,290 m^3/sec in April, during the principal rainy season.

The Rufiji Delta mangroves cover basically three areas. North Rufiji has by far the largest area of mangroves, Central Rufiji has a very small area of mangroves, and South Rufiji has about half the mangrove area of the North (Fig. 1). Over past decades (and probably centuries), the Rufiji Delta has undergone many changes due to natural, dynamic processes. Floods have created new channels, old channels have been blocked by sedimentation, and sand banks have formed causing diversions. One aspect that is important in

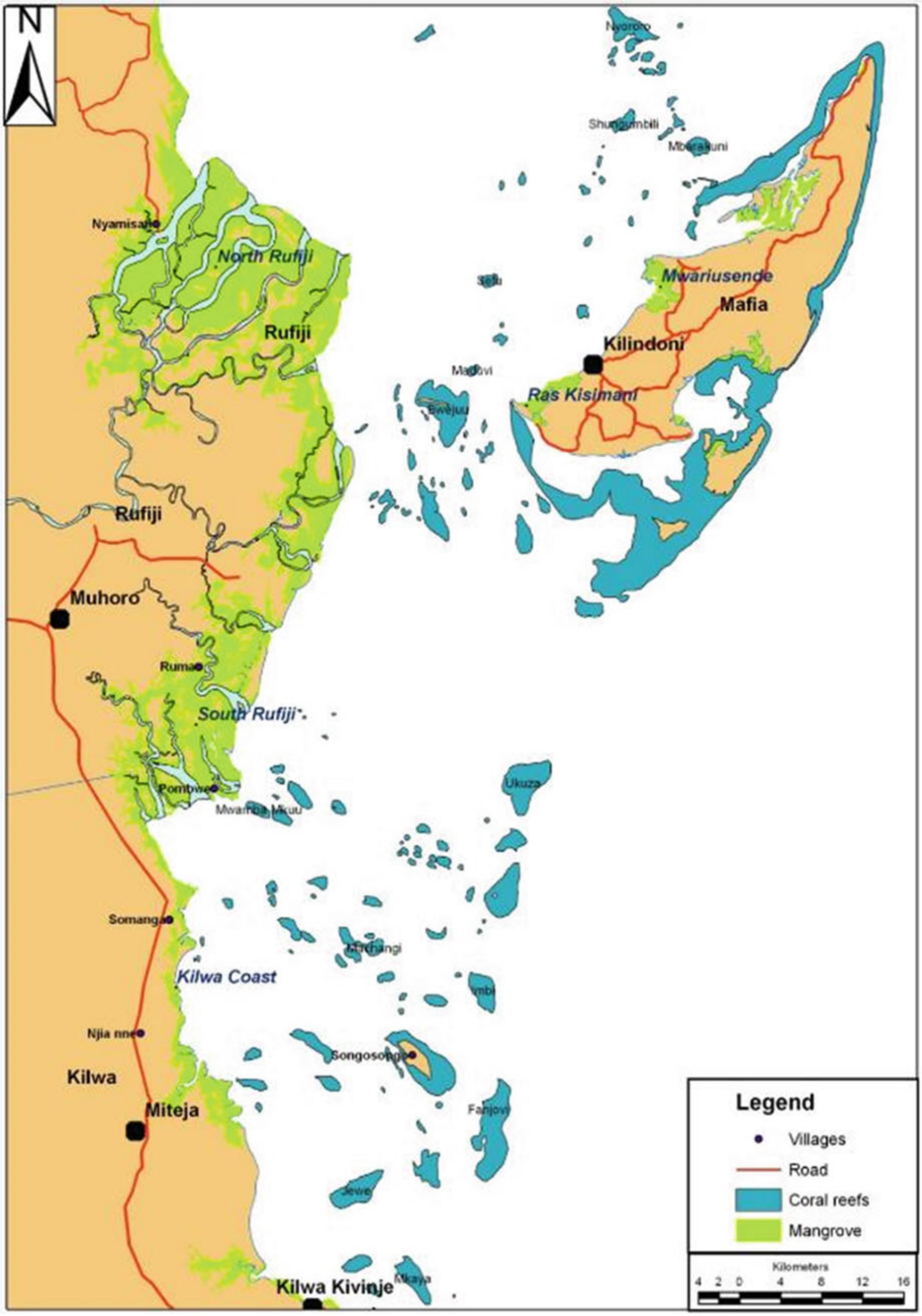

Fig. 1 Map of the study area showing North Rufiji, South Rufij, Mafia, the northern Kilwa coast and offshore reefs

understanding changes in the Rufiji Delta is that, before 1978, most of the river flow was towards the South Rufiji; whereas, since 1978, most of the river flow has been diverted to the North Rufiji. This is due to natural, dynamic processes of the Rufiji River that created a new inlet into the delta and brought more river flow northward (Kajia 2000). Within the North Delta, most of the water now flows through the Bumba River (Semesi 1991).

There are about 43 islands in the Rufiji Delta, with many of them being sparsely inhabited. In 1988, the population within the delta was 26,583, with a growth rate of 2% (Kajia 2000). The delta people have traditionally survived through fishing and farming various crops. In past times, they used to migrate upstream from the delta during the rainy season in order to farm rice. However, during the villagization campaign in Tanzania from 1969 to 1973, they began to clear mangrove areas in order to establish farms (Semesi 1991).

In Tanzania, the East African Coastal Current flows northward throughout the year. There are two main seasons, the northern monsoon (November – March) and southern monsoon (April – October), with periods of rainfall in between. Of the two monsoon periods, the southern monsoon is characterized by cooler air temperatures, higher wind speeds, rougher seas and a stronger East African Coastal Current which reaches a speed of 4 knots (McClanahan 1988). Tides rise and fall twice daily. The mean spring tidal range is 3.3 m, while the mean neap tidal range is 1.13 m (Stoddart 1971).

The research reported in this chapter may be the most comprehensive field study of the Rufiji mangroves conducted to date focussing on assessment of the vulnerability of the Estuary to the climate change factors and recommending specific adaptive strategies.

Methodology

In this research, a number of natural science and social science methods were utilized, which complemented and reinforced each other. Moreover, for several of these methods a participatory monitoring approach was implemented (Wagner 2005), which had the advantages of being able to collect large amounts of data in a short time and of having a profound positive impact on the participants.

Study Areas, Sampling Sites, and Research Design

This study, in its entirety, was conducted in the Rufiji Estuary, Mafia Island and the northern coast of Kilwa. However, the findings documented in this chapter focus on the two major areas of the Rufiji Estuary, the North Delta (Figs. 2 and 3) and the South Delta (Fig. 4). Since the total area involved was vast and since this study focused on determining vulnerability of mangroves to climate change, sampling sites were strategically established in three major mangrove zones/areas most likely to be affected by climate change factors such as sea level rise, increased wave activity and flooding. These were seaward edges, saline flats/landward edges (representing upper mangrove zones), and river channel edges. A paired research design was applied in the methods described below, with observations being recorded in 2007 and repeated in 2009 in the same sites and plots.

Capacity Building and Training

An important part of this project was capacity building. Throughout the fieldwork during both 2007 and 2009, a team of about 15 people were involved at any given time, including research assistants, staff members of government offices and the Mangrove Management Project as well as many villagers resident in the area. As the core of the team moved to new areas, new villagers were trained and involved. Thus a total of over 70 people were trained during the course of this project. This approach helped to build capacity and facilitated the collection of large amounts of data.

Repeated Quantitative Ecological Plot Assessment

The transect line permanent plots method (English et al. 1997) was conducted in 2007 (baseline) and 2009 (repeat study) in the same permanent plots (each 5 m x 5 m). Within each plot, all mangroves were identified to species level and counted according to three maturity categories: seedling (height < 1 m), sapling (height > 1 m, girth < 4 cm) and tree (girth > 4 cm). Girth at breast height (GBH, standardized at 1.3 m above ground) was measured for all trees and saplings. Stumps were also counted as an indication of cutting pressure. This provided detailed ecological data such as mangrove species composition, basal area, density, regeneration capacity, and stump density by species.

In the larger project, this plots method was carried out in 20 strategically-selected sites, with three subsites per site and 8 plots per subsite, making a total of 480 plots. However, this chapter focuses on the 14 sites established in Rufiji, which were designated as either NR (for North Rufiji) or SR (for South Rufij), followed by a number (1, 2, 3, etc.) For example, NR1 stands for North Rufiji Site Number 1. The three subsites in each site were numbered as SS1, SS2 and

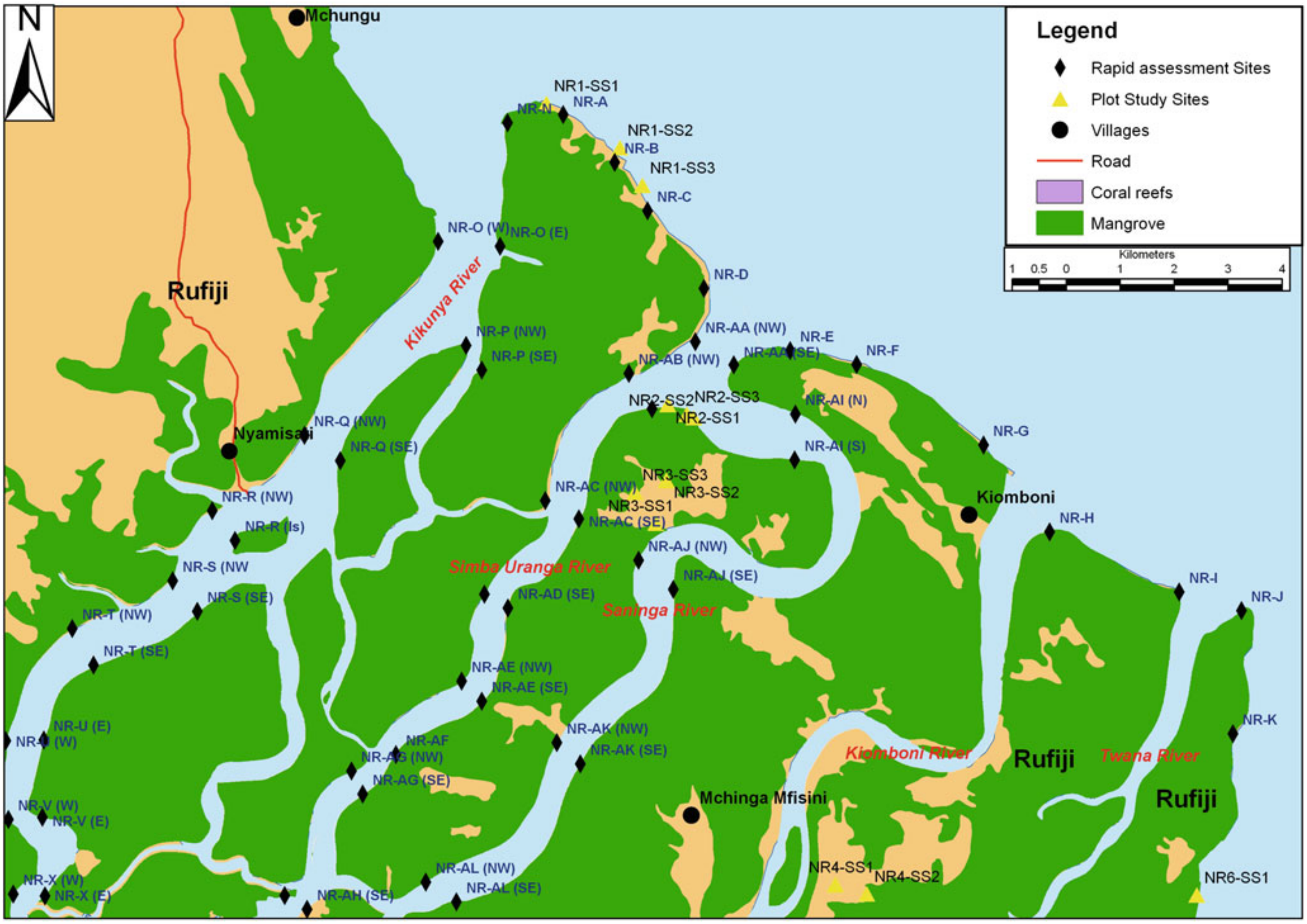

Fig. 2 Map of the northern part of North Rufiji showing rapid assessment and plot-method study sites

SS3, thus giving the designations shown in Figs. 2–4. In each subsite, plots were established in two rows parallel to the edge of the mangrove forest, one row being exactly along the edge and the second row being 20 m into the forest.

Change analysis for the two-year period from 2007 to 2009 was performed statistically using paired-sample hypothesis tests. Either the parametric paired-sample t-test (when the differences between paired observations were normally distributed or lognormal) or the nonparametric Wilcoxon paired-sample test was applied in each case.

Rapid Assessment

A rapid assessment technique (Wagner et al. 2004), which allowed for wide coverage of the study area, was used to obtain quasi-quantitative data on relative mangrove density, height, health, presence of seedlings, presence of stumps, overall aesthetic value and erosion/accretion. In the larger project, there were 58 rapid assessment sites observed in 2007, 46 of these being in the Rufiji Estuary. Many, but not all, of these sites were re-visited in 2009 and observations were again recorded for comparison. These sites are designated as either NR (for North Rufiji) or SR (for South Rufij), followed by a letter (A, B, C, etc.) All data were recorded on an ordinal scale of 0-5, where 0 = nil, 1 = very low (or very poor), 2 = low (or poor), 3 = moderate, 4 = high (or good) and 5 = very high (or very good).

Remote Sensing: Changes in Case Study Areas Between 1991 and 2000

Changes in four selected case study areas in North Rufiji were analyzed based on two landsat satellite images taken 9 years apart during the same season (May 29, 1991 and June 30, 2000). These case studies were erosion/loss of mangroves along the seaward edge of Simba Uranga Island, bordered by Kikunya and Simba Uranga rivers (Fig. 2); accretion along the seaward edge of Twana Island, just north of Bumba River (Fig. 3); invasion of mangroves into Saninga saline flat (Site NR3, Fig. 2); and loss of mangroves due to rice farming at the upper end of Kikunya River (Site NR5, Fig. 3).

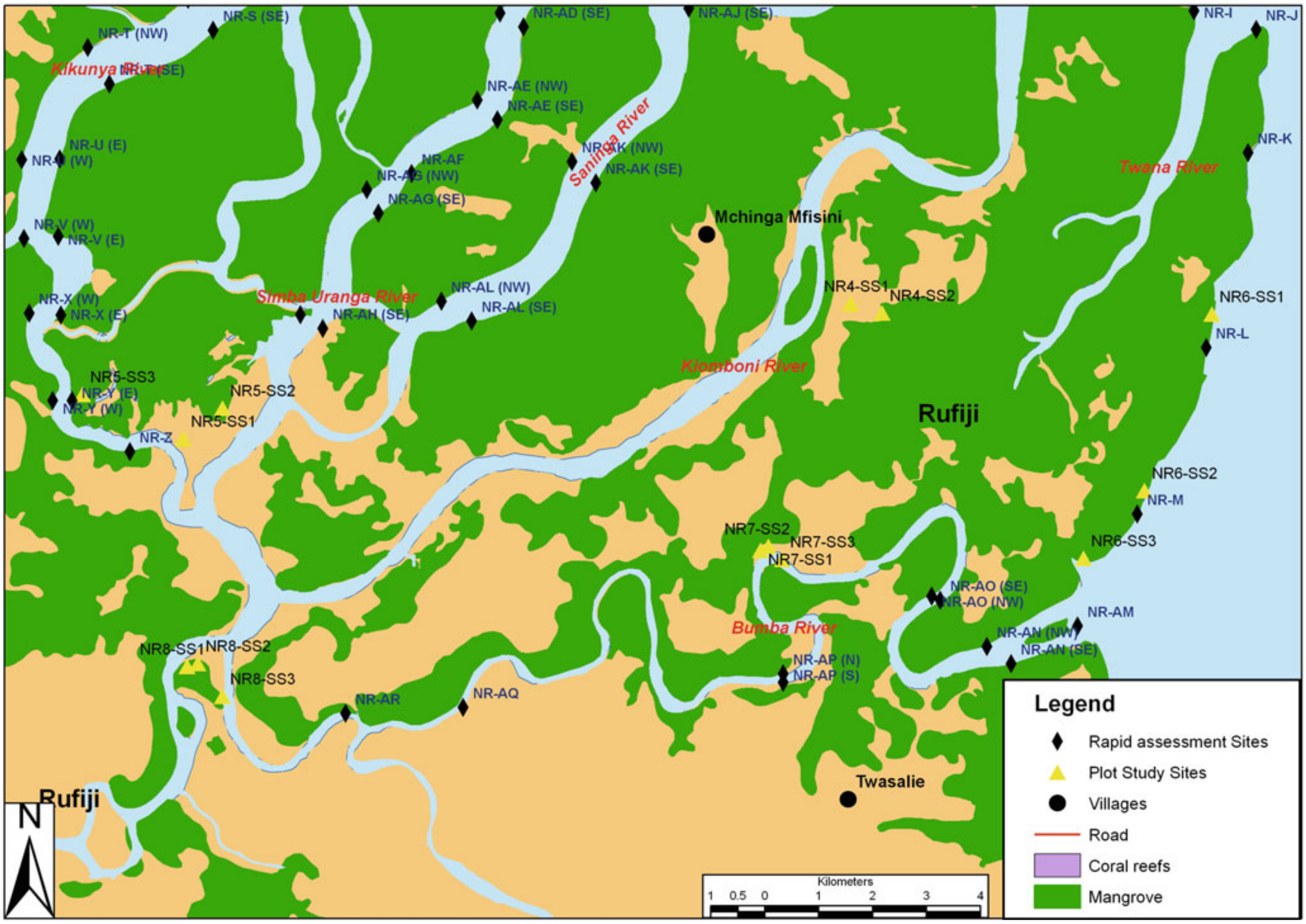

Fig. 3 Map of the southern part of North Rufiji showing rapid assessment and plot-method study sites

Repeated Use of Social Science Methods to Obtain Indigenous Knowledge

In both 2007 and 2009, focus group discussions were held with local residents/resource users and key informants were interviewed. This provided indigenous knowledge and oral history of the changes that had taken place in the Rufiji Estuary over past years and decades as well as the probable causes of those changes. Since most of those involved had lived in the area since childhood and had high dependence on the natural resources, their knowledge was considered very valuable and quite accurate. In rating most of the anthropogenic pressures, the perceptions of the villagers were recorded on an ordinal scale of 0-5, where 0 = nil, 1 = very low, 2 = low, 3 = moderate, 4 = high and 5 = very high.

Results and Discussion

This results section is organized according to the three major mangrove zones/areas focussed on in this study, namely, seaward edges, saline flats/landward edges representing upper mangrove zones and river channel edges. In presenting this analysis, these zones were further categorized according to the main trends shown or the pressures observed.

Seaward Edges Where Erosion Is Greater than Accretion

Northern Part of North Rufiji: Simba Uranga Island

Ecological plot assessment Site NR1 was located along the seaward edge of Simba Uranga Island, which is bordered by Kikunya and Simba Uranga rivers (Fig. 2). Two of its three subsites, i.e., NR1-SS1 and NR1-SS3, were completely eroded away by wave action between 2007 and 2009, with all mangroves being lost. Therefore, with all subsites combined, there was a very drastic and extremely significant decrease in mangrove basal area (Table 1). Subite NR1-SS2 analyzed separately showed no significant change in mangrove basal area, due to influx of sediment from a small, coastal stream entering the ocean just to the south, which offset the erosion. In Tanzania, ocean currents are

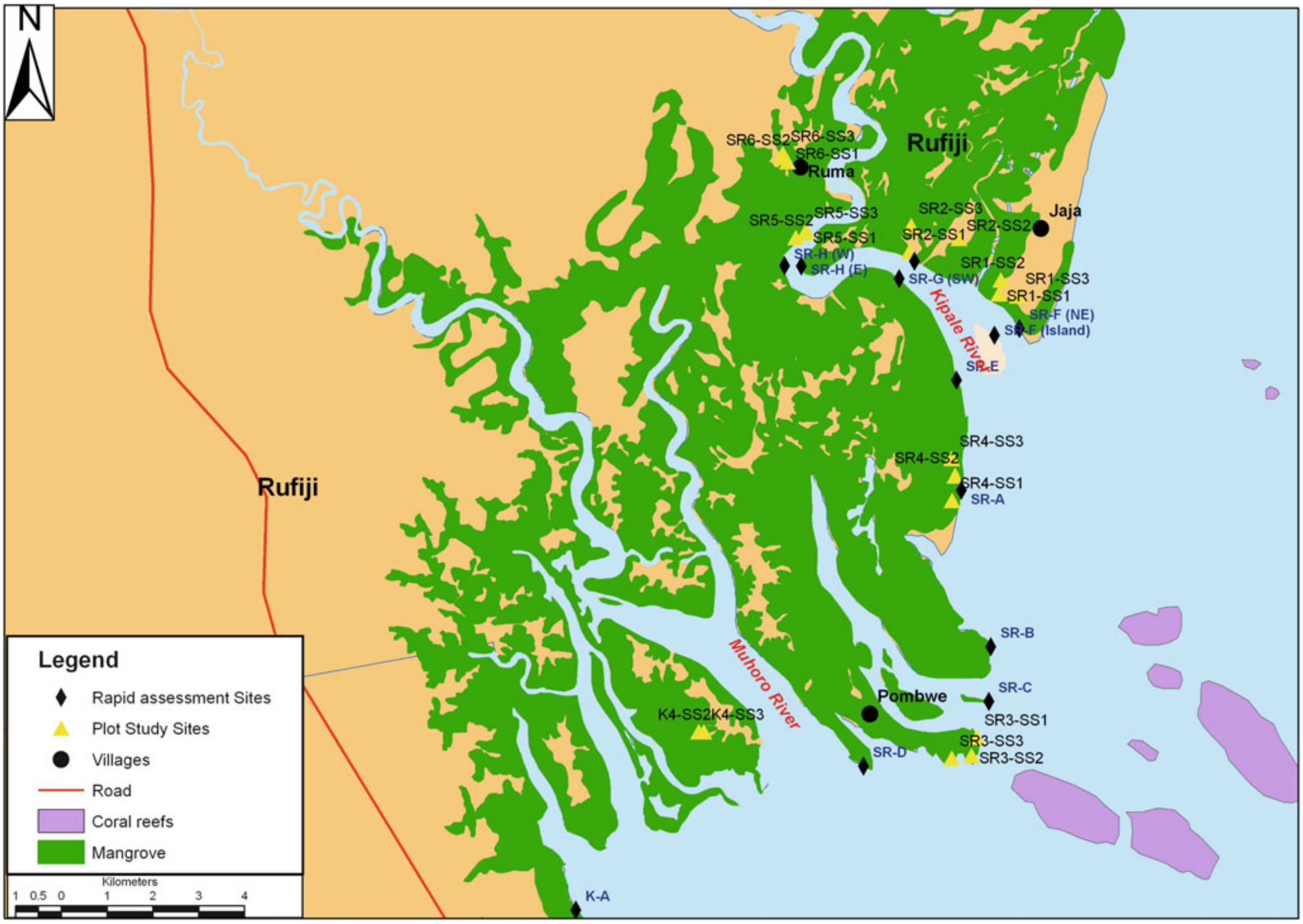

Fig. 4 Map of South Rufiji showing rapid assessment and plot-method study sites

Table 1 Paired statistical comparisons of 2007 and 2009 mangrove basal area in quantitative ecological plot assessment sites

Comparison	Test statistic	DF/n	P-value	Conclusion	Significance
NR1-all subsites	(T-) = 14.000*	n = 21	P = 0.0008	2007 > 2009	e.s.
NR1-SS2	(T+) = 0.000*	n = 5	P = 0.2500	No change	n.s.
SR4-all subsites	(T-) = 1.000*	n = 10	p = 0.0039	2007 > 2009	v.s.
NR6-all subsites	(T-) = 12.000*	n = 16	p = 0.0171	2009 > 2007	s.
NR3-all subsites	(T+) = 29.000*	n = 19	p = 0.0120	2009 > 2007	s.
NR4-all subsites	t = 2.571#	DF = 13	p = 0.0232	2009 > 2007	s.
SR2-all subsites	(T+) =5.000*	n = 21	p = 0.0002	2009 > 2007	e.s.
SR2-SS2	t = 0.8700#	DF = 6	p = 0.4177	No change	n.s
SR1-all subsites	T = 3.627#	DF = 16	p = 0.0023	2009 > 2007	v.s.
SR1-SS1	T = 1.421#	DF = 6	p = 0.2051	No change	n.s.
SR6-all subsites	t = 2.407#	DF = 17	p = 0.0277	2009 > 2007	s.
NR5-all subsites	(T+) =18.000*	n = 16	p = 0.0151	2009 > 2007	s.
SR5-all subsites	(T+) =11.000*	n = 16	p = 0.0017	2009 > 2007	v.s.
NR7-all subsites	(T+) =32.000*	n = 16	p = 0.0654	2009 > 2007	m.s.
NR8-all subsites	t = 3.630#L	DF = 17	p = 0.0021	2007 > 2009	v.s.

Paired-sample t test (performed on original data)
#L Paired-sample t test (performed on log transformed data)
* Wilcoxon paired-sample test
e.s. = extremely significant
v.s. = very significant
s. = significant
m.s. = marginally significant
n.s. = not significant

predominantly northward, thus the movement of sediments is in that direction.

In addition to the ecological plot sampling, direct observation of Simba Uranga Island was undertaken by walking with villagers from the area along its entire 5-km seaward edge, from Kikunya River to Simba Uranga River, including rapid assessment Sites NR-A and NR-D. It was evident that an almost uniform strip of coastline had been eroded away, accompanied by mangrove loss (observed as exposed roots and fallen trees, mainly *Rhizophora mucronata*, Photograph 1), between 2007 and 2009. Using the plot sites as a guide (two rows of plots 20 m apart), it was estimated that at least 25 m had been lost along most of the coastline over the two-year period.

Remote sensing change analysis (1991 to 2000) of Simba Uranga Island fully backed up the above field observations. The red strip in Fig. 5, which occurs all along the seaward edge and the mouths of the river channels, indicates the area

Photograph 1 Long stretch of coastline at Simba Uranga Island showing drastic erosion and mangrove loss (mainly *Rhizophora mucronata*)

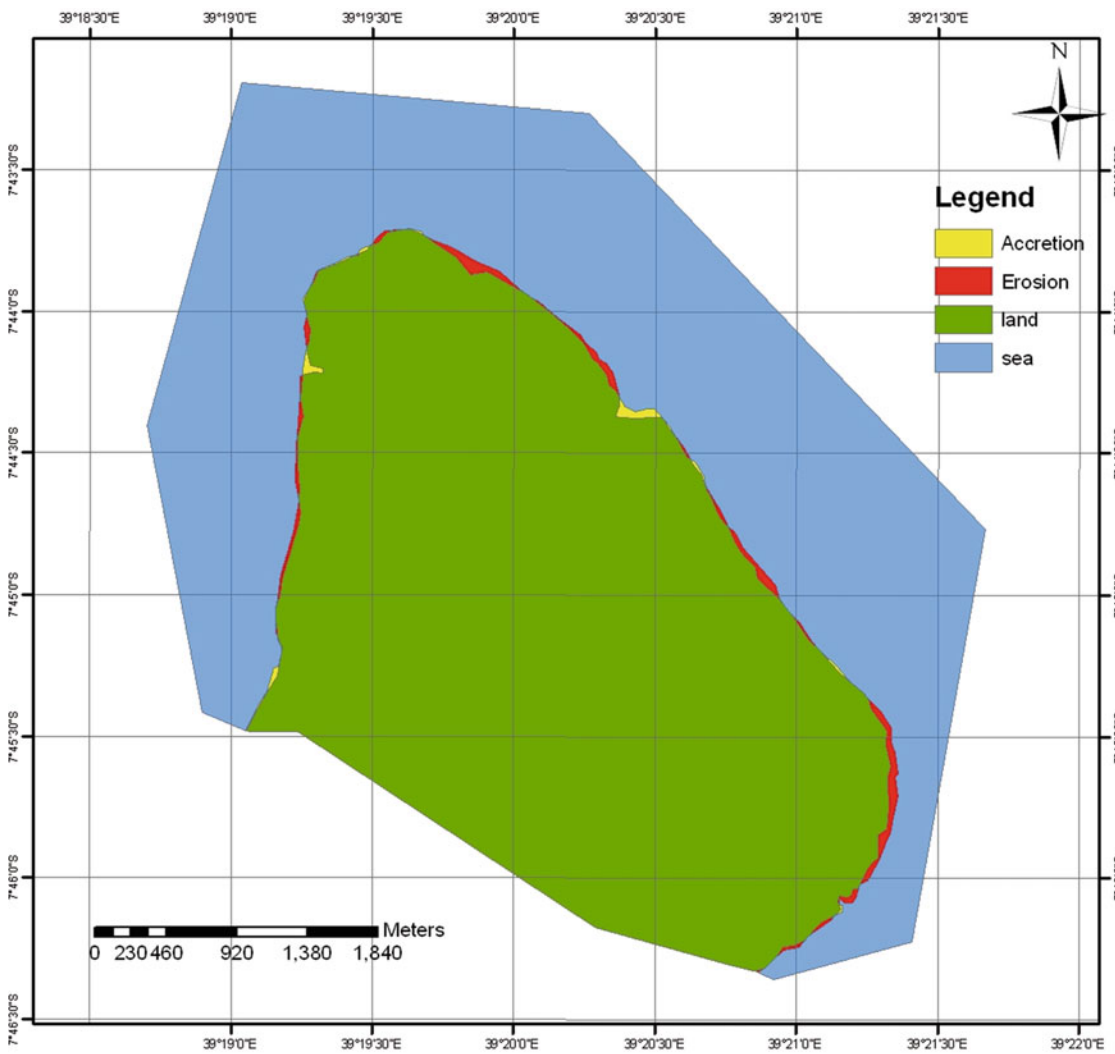

Fig. 5 Remote sensing change analysis of Simba Uranga mangroves over a 9-year period based on Landsat images taken on May 29, 1991 and June 30, 2000

Table 2 Area of mangroves/land, sea, erosion, accretion and net loss of mangroves on the seaward edge of Simba Uranga (based on remote sensing, 1991 to 2000, Fig. 5; all units = m^2)

	May 29,1991	June 30, 2000
Mangroves/land	12,132,981	11,991,427
Sea	9,824,841	9,966,395
Erosion and loss of mangroves/land	186,372	
Accretion and growth of new mangroves	44,818	
Net loss of mangroves	141,554	

of mangroves/land that were lost to erosion over the 9-year period (totalling 186,372 m^2, Table 2). (Compare the image in Fig. 5 with the seaward part of Simba Uranga Island shown in Fig. 2 in order to find locations.) The yellow patches in Fig. 5 (totalling 44,818 m^2) indicate small areas where accretion occurred due to small stream input, in particular, the yellow patch (at about the midpoint of the seaward edge) which is just to the south of Site NR1-SS2. Thus, the net loss of mangroves was 141,554 m^2.

Indigenous knowledge, obtained from focus group discussions and key informants, further strengthened the findings of the field observations and remote sensing change analysis. According to the villagers, erosion occurred in past decades at a slow rate, but during the past 10 years the rate had increased greatly along almost the entire seaward edge of Simba Uranga Island. In the vicinity of Subsite NR1-SS1, about 50 m had eroded away over the previous two years. At Subsite NR1-SS3, about 25 m had eroded (which coincided exactly with the loss of the plots). They attributed this accelerated erosion to waves, winds and currents, which have become unpredictable and increasingly severe, particularly during the southern monsoon period (April to October), with the strongest winds and wave action occurring in July and August. Also according to residents of Simba Uranga Village, located on the coast, sea level appeared to be rising because the highest spring tides (during March) were getting closer to their village, year by year. Moreover, the shoreline was being eroded away and some houses may soon be lost. Anthropogenic pressures in the area were very low (1.0 on a 0-5 scale) for all mangrove uses and changed little over the two-year period.

Thus, combining information from the ecological plot sampling, direct observation and indigenous knowledge, a very conservative estimate is that at least 25 m had eroded away between 2007 and 2009 along about 3.5 km of the 5.0-km seaward edge of Simba Uranga Island. This represents a loss of mangroves of 43,750 m^2 per year, which is a greatly accelerated rate of loss compared to the 20,708 m^2 loss per year over the 9-year period from 1991 to 2000 indicated by the remote sensing change analysis.

South Rufiji: Pombwe Island and Northwards

At Pombwe Island (bordered by Kipale and Muhoro Rivers), Site SR4 was established for ecological plot assessment (Fig. 4). Results showed that there was a very drastic and statistically significant decrease in mangrove basal area from 2007 to 2009 (Table 1). The main reason for substantial decrease in mangrove basal area and density, was the piling up of coarse sand along the coast, which smoothers roots, buries the base of the mangroves and kills them (Photograph 2). Clear evidence of this trend is that many of the plots at Site SR4 had been completely buried in sand and the mangroves had died by 2009. Judging by the plots and direct observation while walking along much of the coast, a strip of several meters of mangroves along the Pombwe coast had been killed over the two-year period.

However, the overall phenomenon is a dual process of deposition followed by erosion, which takes place along the entire northern and central coastline of Pombwe Island south of rapid assessment Site SR-E (Fig. 4). This is a unique phenomenon that does not occur along other parts of the Rufiji Delta. According to indigenous knowledge, this dual process, which also extends north of Kipale River, takes place particularly during the southern monsoon period when storm surges occur at the time of high spring tides. A strip of approximately 500 m has been lost by erosion along this coastline since 1970 due to increased wave activity and rising sea level and the rate of mangrove loss has accelerated over the past 5 years. The villagers explained that the causal factor is that this stretch of coastline is exposed to strong wave action due to a gap between Mafia Island and the northern end of the Songo Songo Archipelago of patch reefs (e.g. Mwamba Mkuu) and islands (e.g. Okuza Island) (Fig. 1). Year after year, the accreting sand bank moves further inland, burying and killing mangroves, while the seaward side of the sand bank is eroded away, along with the dead mangroves. This dual process was well understood by the villagers and was supported by our field observations.

Compared to natural pressures, anthropogenic pressures on the mangroves in this area were quite low. According to villagers' perceptions, cutting mangroves for domestic firewood was rated as a very low threat (1 on a 0-5 scale) and cutting for building poles was rated as a low threat (2). Cutting mangroves, particularly *Sonneratia alba*, for lime-making increased from a very low threat to a low threat from 2007 to 2009. In this area, instead of using corals for making lime as building material, villagers use the gastropod *Terebralia palustris*. To make lime, they pile up alternate layers of mangrove wood and gastropods to form a kiln. If this practice accelerates, it could become detrimental to mangroves because this gastropod is an important detritus feeder that regulates nutrient cycling in the mangrove ecosystems.

Photograph 2 Long stretch of coastline at Pombwe Island with mangroves (mainly *Sonneratia alba*) being lost by sand deposition followed by erosion

Seaward Edges Where Accretion Is Greater than (or Balances) Erosion

North Rufiji: Twana Island

Site NR6, situated along the seaward edge of Twana Island, just north of Bumba River mouth (Fig. 3), had fairly high mangrove basal area (over 400 cm^2/25-m^2 plot) during both 2007 and 2009. With all subsites combined, there was a small, but statistically significant increase in basal area over the two-year period (Table 1). Anthropogenic pressures were low.

Remote sensing change analysis also clearly showed that accretion has been occurring in the area. The map in Fig. 6 indicates areas in yellow where accretion and new growth of mangroves occurred between 1991 and 2000. Areas in red, which are almost undetectable by comparison, indicate erosion. The long, narrow island in the mouth of Bumba River, between rapid assessment Sites NR-AM and NR-AN (Fig. 3) was 0.9 km long in 2009 and had been colonized by a dense stand of mangroves dominated by *Sonneratia alba*. However, it is apparent that this island only formed in recent decades, since it was not there when aerial photographs were taken in 1989 for the production of the Mangrove Management Plan maps (Semesi 1991).

The general trend of accretion and new growth of mangroves along the southern and central part of Twana Island is probably mainly due to the fact that it is relatively sheltered from strong wave action. Twana Island is adjacent to the southwest end of Mafia Island, which juts out into the channel and is only 19.5 km east of Bumba River mouth (Fig. 1). Moreover, there are a number of smaller islands (e.g. Bwejuu) and coral reefs (Maduvi) between southern Mafia and Twana Island.

Another contributing factor is that there is evidence that Bumba River has carried the highest sediment load amongst the channels in Rufiji since the 1978 diversion of the main flow of Rufiji from the South Delta to the North Delta (Kajia, 2000). However, thorough investigation of sediment loads in these channels is required to verify this. Though Bumba is quite a narrow river, it is deep, fast-flowing and turbid. Once

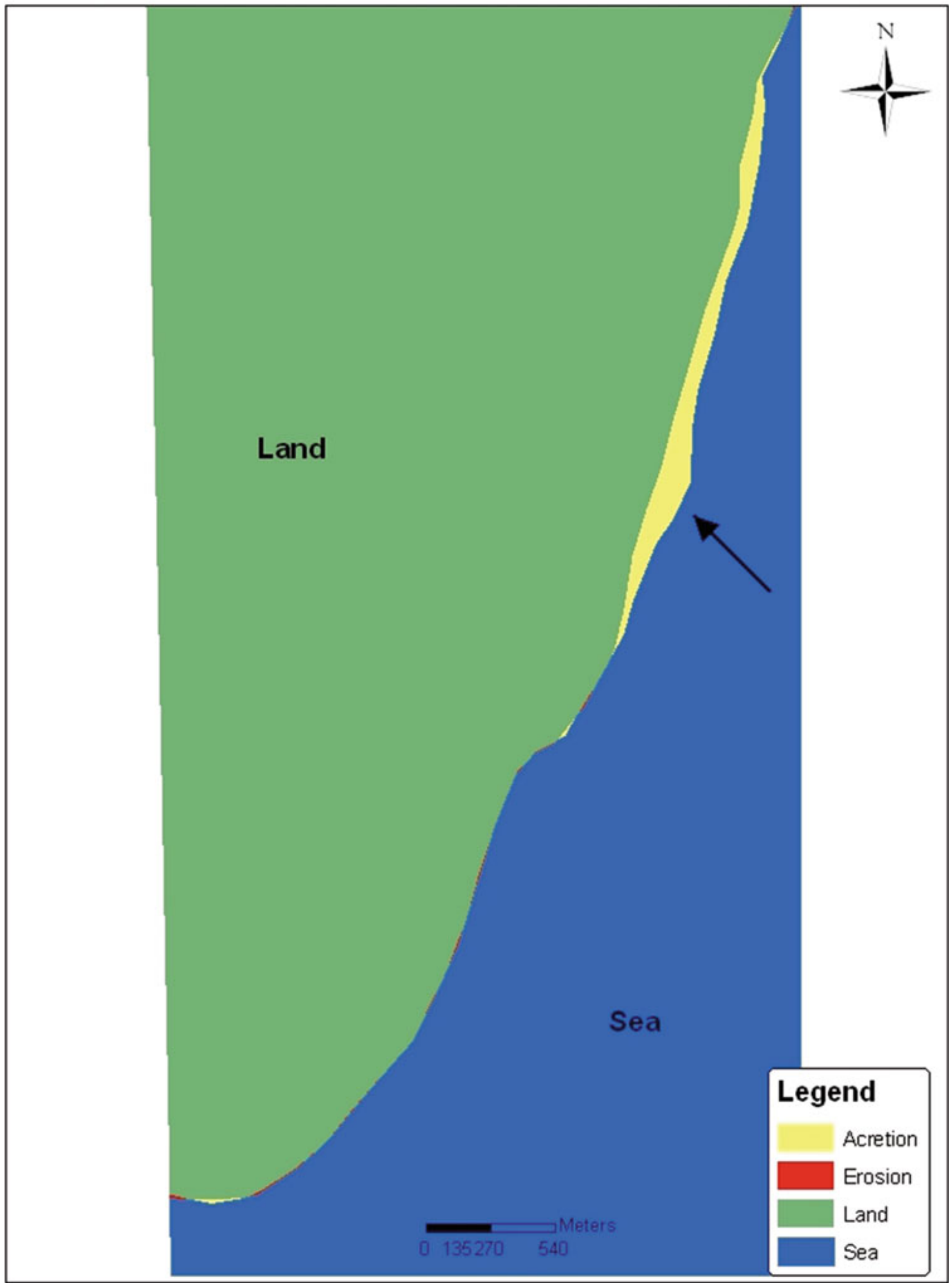

Fig. 6 Remote sensing change analysis of the seaward edge of Twana Island, north of Bumba River over a 9-year period, based on Landsat images taken in 1991 and 2000

at sea, the sediments are carried northward by the prevailing northward currents and deposited along the coast of Twana Island.

Moreover, the sediment load of Bumba River seems to have been exacerbated by climate change factors and anthropogenic pressures in the Bumba River watershed area. During the 1998 El-Nino, large areas of *Heritiera littoralis* and other species died due to flooding over an extended period of time. Also, large areas of mangroves have been converted to rice farms. Thus, without the binding force of mangrove roots, large quantities of soil have been carried by runoff into Bumba River and deposited along the coast. Degradation of the watershed basin of Bumba River is further documented in section "River channel edges affected by anthropogenic pressures and El-Nino flooding". According to focus group discussions, however, anthropogenic pressures along the seaward edge of Twana were low.

Observation of rapid assessment sites between Twana in the south and Simba Uranga in the north (NR-E, NR-F, NR-G, NR-H, NR-I and NR-J; Fig. 2) indicated slight accretion in some and slight erosion in others. According to indigenous knowledge, this seaward edge experienced erosion before 1978. However, since the shift in the main flow of the Rufiji in that year, there has been a slow rate of accretion and mangrove growth along some parts of this coastline. Therefore, coastal erosion in this area has been partly offset by the influx of sediment from Bumba River.

South Rufij: Pombwe South

Observations at rapid assessment Site SR-B (near Pombwe Village) and southward (Fig. 4) indicated that accretion and new growth of mangroves was occurring and this trend was supported by indigenous knowledge. The accretion is likely due to the sheltering effect of offshore patch reefs and islands at the northern end of Songo Songo Archipelago (Fig. 1).

Saline Flats That Are Entirely or Largely Surrounded by Mangroves

North Rufiji: Saninga Saline Flat

Site NR3 (Fig. 2) was established around the periphery of a saline flat located at Saninga that was completely surrounded by mangroves. Though this saline flat (as well as Sites NR4 and SR2 mentioned below) is found in the lower part of the Rufiji Delta, the forest surrounding it represents the upper mangrove zone since it is located at the upper reaches of tidal inundation at high tide. With all subsites combined in the analysis, there was a small, but statistically significant increase in mangrove basal area from 2007 to 2009 (Table 1). The plot data also showed increase in sapling and seedling density (particularly *Avicennia marina*), indicating that there was colonization and overall increase in mangrove abundance. Stump density was low (1.5 – 2 stumps/25-m^2 plot) and did not change significantly over the two-year period (Wilcoxon paired-sample test: (T+) = 2.500, n = 20, P = 0.1875).

Remote sensing analysis also indicated that mangroves were migrating into the saline flat. The green strip around the saline flat shown in Fig. 7 indicates mangrove invasion (90,296 m^2), while the yellow indicates mangrove loss/degradation (30,335 m^2). Thus, the net increase of mangroves between 1991 and 2000 was 59,961 m^2, which is 7.1% of the remaining saline area (844,151 m^2).

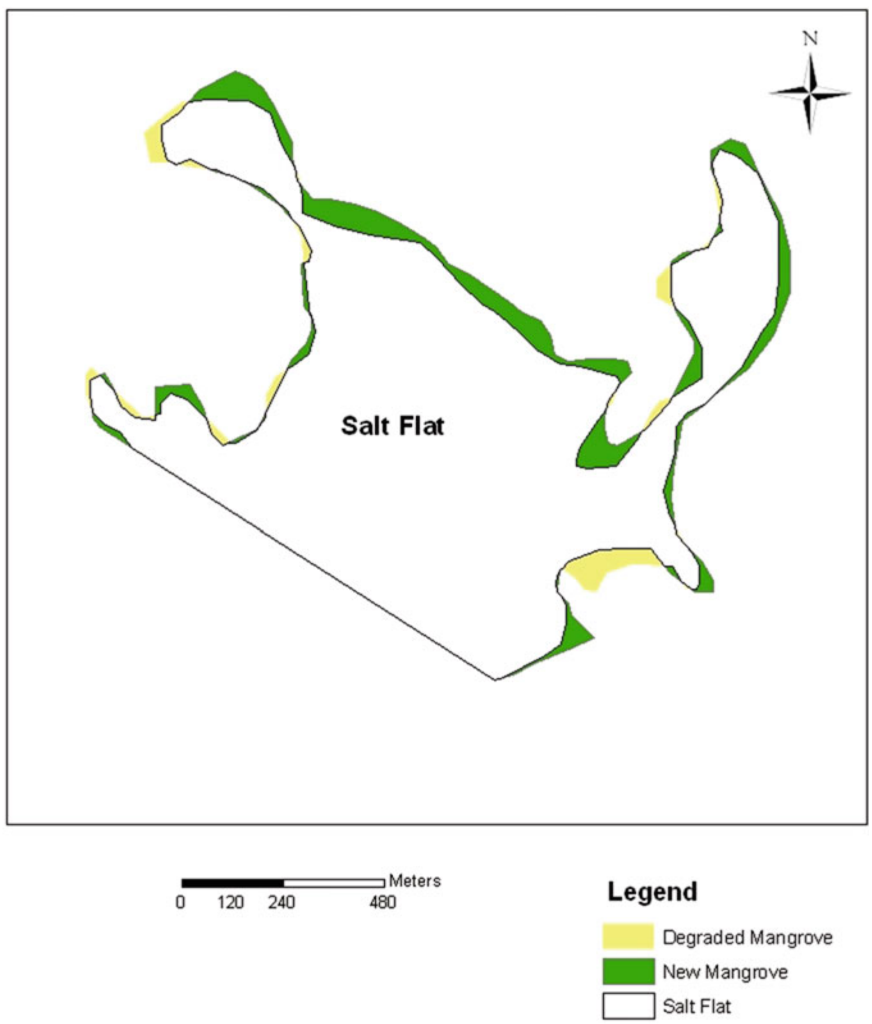

Fig. 7 Remote sensing change analysis of Saninga saline flat showing mangrove invasion and loss between 1991 to 2000

North Rufiji: Kilame Saline Flat

Site NR4 was established around the periphery of a saline flat located at Kilame near Mchinga Mfisini Village along Kiomboni River (Figs. 2 and 3). With all subsites combined in the analysis, there was a small, but statistically significant increase in mangrove basal area from 2007 to 2009 (Table 1). Tree and sapling density also increased.

South Rufiji: Ulaya Saline Flat

Site SR2 was located on the northern side of Kipale River (Fig. 4). With all subsites combined, there was an extremely significant increase in mangrove basal area from 2007 to 2009 (Table 1). However, SR2-SS2 when analyzed separately showed no significant change due to relatively high cutting pressure there, as indicated by a stump density of 6.0 stumps/25-m^2 plot in 2009. There was a very significant increase in stump density over the two-year period (Wilcoxon paired-sample test: (T+) = 3.000, n = 21, P = 0.0012). *Ceriops tagal* increased considerably in tree density and seedling density, while saplings of this species more than doubled over the two-year period. *Avicennia marina* and *Rhizophora mucronata* saplings, which did not occur in 2007, had invaded in small numbers by 2009. According to villagers' perceptions, sea water was increasingly inundating this saline flat, accompanied by mangrove invasion.

Landward Mangrove Edges and/or Saline Areas Little Affected by Human Pressures

South Rufiji: Jaja Landward Edge/Saline Flat

Site SR1was established along the edge of terrestrial vegetation and saline areas on the northern side of Kipale River mouth in South Rufiji (Fig. 4). With all subsites combined, there was a very significant increase in mangrove basal area from 2007 to 2009 (Table 1). Tree and sapling density increased for both *Avicennia marina* and *Ceriops tagal*. However, in Subsite SR1-SS1 (analyzed separately), there was no significant change over time since it bordered terrestrial grass where the ground was slightly raised, making seawater intrusion impossible. Stump density was low (approximately 1 stump/25-m^2 plot) and did not change significantly over two years (Wilcoxon paired-sample test: (T+) = 0.000, n = 17, P = 0.2500).

South Rufiji: Ruma Darajani Landward Edge/Saline Flat

Site SR6 was established along the landward edge of mangroves near Ruma village (Fig. 4). There was a significant increase in mangrove basal area from 2007 to 2009 (Table 1), which occurred in the adjacent saline flat area, not the terrestrial grassland where there was a slight rise in elevation. Stump density (primarily *Ceriops tagal*) was very low and did not change over two years (Wilcoxon paired-sample test: (T+) = 6.000, n = 18, p = 0.3125). The increase in mangrove abundance was backed up by focus groups discussions with villagers who perceived that there was increasing inundation of seawater and invasion of mangroves.

Landward Mangrove Edges Substantially Affected by Human Pressures

North Rufiji: Mawanda Area

Site NR5 (Fig. 3) had formerly been a dense mangrove forest dominated by *Heritiera littoralis*. Large areas of the forest had been cleared and converted to rice farms. There has also been ongoing harvesting of *H. littoralis* due to its popularity for timber production. Thus, Site NR5 is along the periphery of human-created open areas. Some rice farms are still in operation, while others have been abandoned and terrestrial grass has grown up.

In this particular site, effects of human pressures were primarily assessed by remote sensing, while plot assessment was mainly used to assess any possible re-invasion of mangroves into the cleared areas. Thus, plots with substantial ongoing cutting pressure were omitted from the analysis, which showed that there was a small, but statistically significant increase in basal area from 2007 to 2009 (Table 1). There was very high basal area during both years (over 800 cm^2/25-m^2 plot) due to the presence of trees with large girth at breast height. Stump density increased significantly over the two years (paired-sample t test performed on log transformed data: t = 2.394, DF = 20, P = 0.0266). Ongoing anthropogenic pressure was also observed around the site. The open area was being further expanded to establish more rice farms, with more *Heritiera littoralis* trees being cut.

Remote sensing indicated great loss in area coverage of dense mangroves and great increase in open/agricultural land, mainly rice farms, between 1991 and 2000 (Fig. 8, Table 3). There was a slight increase in sparse mangroves as some of the dense mangrove areas were thinned by harvesting, for timber production and building poles. The greater area coverage of open/agricultural land in 2009 probably included areas under rice at the time as well as rice farms that had been abandoned.

According to indigenous knowledge, there had been considerable deposition of sediments along the river banks and river bed near Site NR5 and similar landward areas, such that the depth of the river was decreasing. In addition, sediments were building up on the land. Thus, there is likely to be less and less seawater reaching the area during high tides, with gradual conversion of the mangrove forest to a terrestrial environment. Anthropogenic pressures were high, but did not change between 2007 and 2009. The greatest pressure was rice farming, rated as very high (5 on a 0-5 scale), during both years, followed by cutting for timber production and building poles, rated as 3 both years.

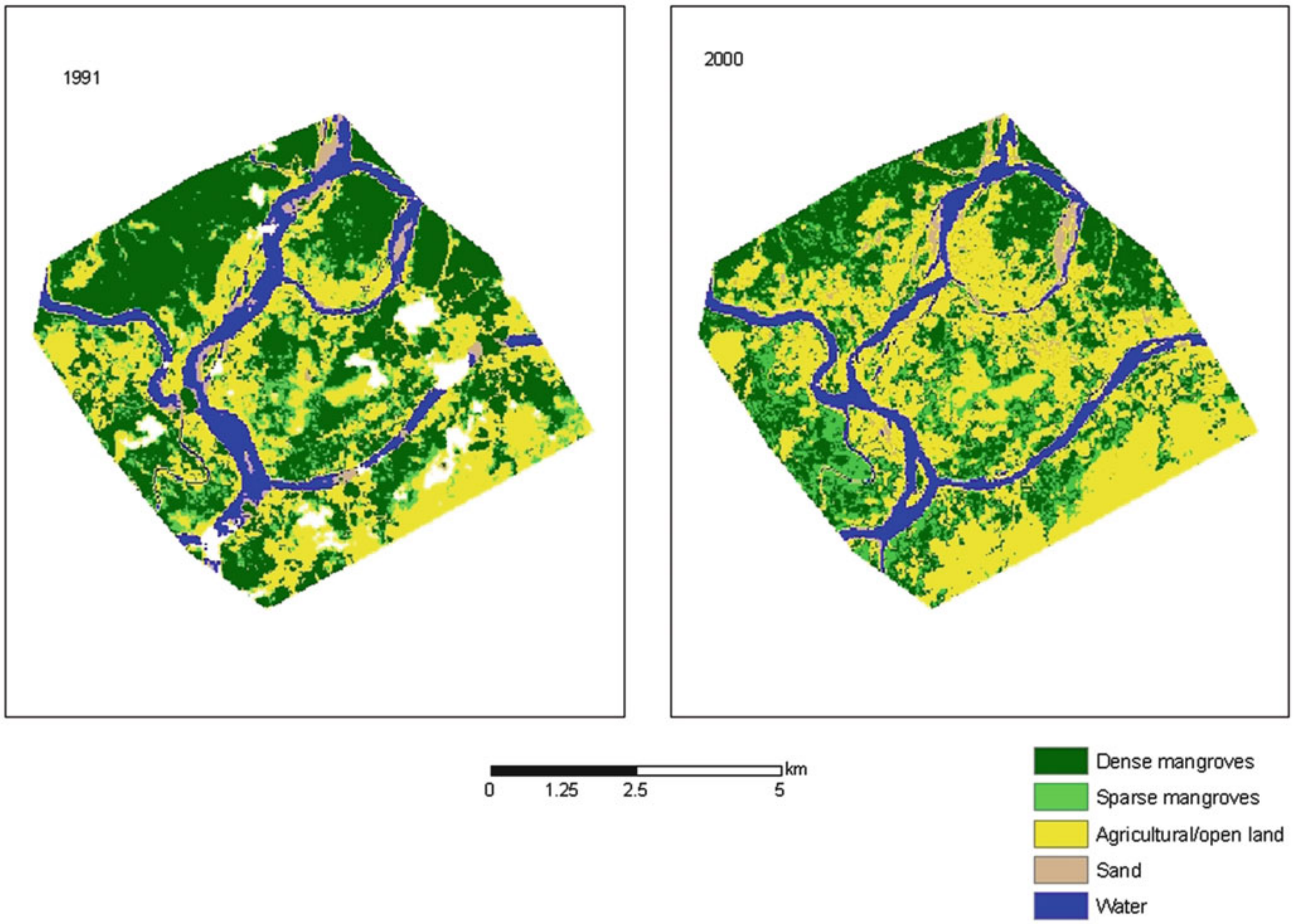

Fig. 8 Remote sensing change analysis of Mawanda area from 1991 to 2000

Table 3 Changes in area coverage of dense mangroves, sparse mangroves, agricultural/open land, water and sand in Mawanda area between 1991 and 2000 (remote sensing analysis, Fig. 8)

	1991	2000	
	Area in hectares	Area in hectares	Change
Dense mangroves	2,564	1,461	Great loss
Sparse mangroves	382	610	Slight increase
Agricultural/open land	1,096	2,113	Great increase
Water	479	434	Little change
Sand	117	251	Slight increase

River Channel Edges Mainly Influenced by Natural Delta Dynamics

North Rufiji: Daru Along Simba Uranga River Edge

For Site NR2, which was located at Daru where Simba Uranga and Saninga River channels converge (Fig. 2), it was not possible to make any comparisons between 2007 and 2009 using the ecological plot method since sedimentation covered some of the beacons so plots could not be re-traced. From visual observations, it appeared that changes along the edge of the river in this area were mainly due to natural dynamics of the estuary. There was general accretion of sediments, likely due to its situation on the inner side of the bend in the river. According to the villagers, anthropogenic pressures were relatively low. The main use of mangroves was for building poles and boat-making, both rated as low (2 on a 0–5 scale).

South Rufiji: Ruma South Along Kipale River Edge

Site SR5, on the bank of Kipale River, just south of Ruma village (Fig. 4), had very high basal area. There was a very significant increase in mangrove basal area from 2007 to 2009 (Table 1). Amongst the South Rufiji sites, stump density in this site was quite high (5.8 stumps/25-m^2 plot) and showed a marginally significant increase over the two-year period (paired-sample t test: $t = 2.034$, $DF = 17$, $P = 0.0579$). This cutting pressure was undoubtedly related to the proximity of the site to Ruma Village. Some portions of plots along the river bank had eroded away during the two years due to natural dynamics, since this site is located on the outer side of the river bend where erosion occurs due to centrifugal force of the water. Indeed, accretion and new mangrove growth was observed on the opposite side of the river. Despite high cutting pressure and some erosion, the increasing high basal area was likely due to the presence of rich mangrove mud supporting rapid mangrove growth.

River Channel Edges Affected by Anthropogenic Pressures and El-Nino Flooding

North Rufiji: Middle Bumba River Edge

Site NR7 was located along middle Bumba River (Fig. 3) though it represents an upper mangrove zone in terms of tidal inundation. At the time of conducting this study, mangroves formed only a narrow band along the river, beyond which were large areas where mangroves had been previously converted to rice farms. Also in a large area around this site, many mangroves, particularly *Heritiera littoralis*, had died during the extended floods that occurred as a result of the 1998 El-Nino. The mangroves did not appear to be healthy. Nevertheless, there was a marginally significant increase in mangrove basal area from 2007 to 2009 (Table 1). Stump density showed a marginally significant increase (paired-sample t test performed on log transformed data: $t = 1.916$, $DF = 19$, $P = 0.0705$).

North Rufiji: Upper Bumba River Edge

Site NR8 (Fig. 3) and surrounding areas have been subjected to heavy human pressures in recent decades and also experienced substantial loss of mangroves, particularly *Heritiera littoralis*, during the 1998 El-Nino floods. There was a very significant drop in basal area between 2007 and 2009 (Table 1), which can be related to the extremely significant increase in stump density over the two-year (Wilcoxon paired-sample test: $(T+) = 0.000$, $n = 18$, $P = 0.0005$). In many of the plots, mangroves had been completely cleared and rice planted by 2009. This site was almost a monospecific stand of *H. littoralis*, with traces of *Xylocarpus granatum*. According to indigenous knowledge, cutting mangroves (particularly *Heritiera littoralis*) to produce timber was rated as very high (5) during both 2007 and 2009. Large planks are cut and sold in other areas such as Dar es Salaam. Cutting for building poles and boat-making was rated as 2 during 2007 and 3 during 2009. Sedimentation in the forest was rated as having a threat level of 4.

Conclusions: Change Analysis and Vulnerability

Overall change analysis and vulnerability are presented in the flow charts in Figs. 9 and 10 for various parts of the Rufiji Estuary, categorized by mangrove zones, trends shown, and levels of anthropogenic pressures. Seaward edges proved to be particularly strategic for the investigation of possible state changes and vulnerability related to climate change factors, since these edges are open to forces of the sea. Moreover, changes due to natural pressures could be isolated from those due to human pressures since seaward edges are little affected by the latter. Thus, where changes were drastic, beyond just natural delta dynamics, this study provided strong evidence of the effects of climate change factors.

Seaward Edges Where Erosion Is Greater than Accretion

This study provided strong evidence that the exposed seaward edges of the Rufiji estuary, such as Simba Uranga and Pombwe islands, are very vulnerable to climate change factors (Fig. 9). Sea level rise alone would likely cause only minor and very gradual pressure on the seaward edges. However, when combined with greater wave activity and increasing frequency and severity of storm surges and high water events, these climate change factors appear to have caused drastic, large-scale coastal erosion and mangrove loss. These exposed seaward edges have already shown high vulnerability to climate change factors and, as climate change continues to escalate, they are likely to become increasingly vulnerable.

Both Simba Uranga and Pombwe islands are highly vulnerable due to their exposure to wide sections of open ocean, with no islands or coral reefs to provide protection from strong wave action. Moreover, the rate of erosion is accelerating over time. Erosion primarily occurs on a large scale during storm surge events that coincide with high spring tides during the strong winds of the southern monsoon season, particularly in July and August. Despite the similar

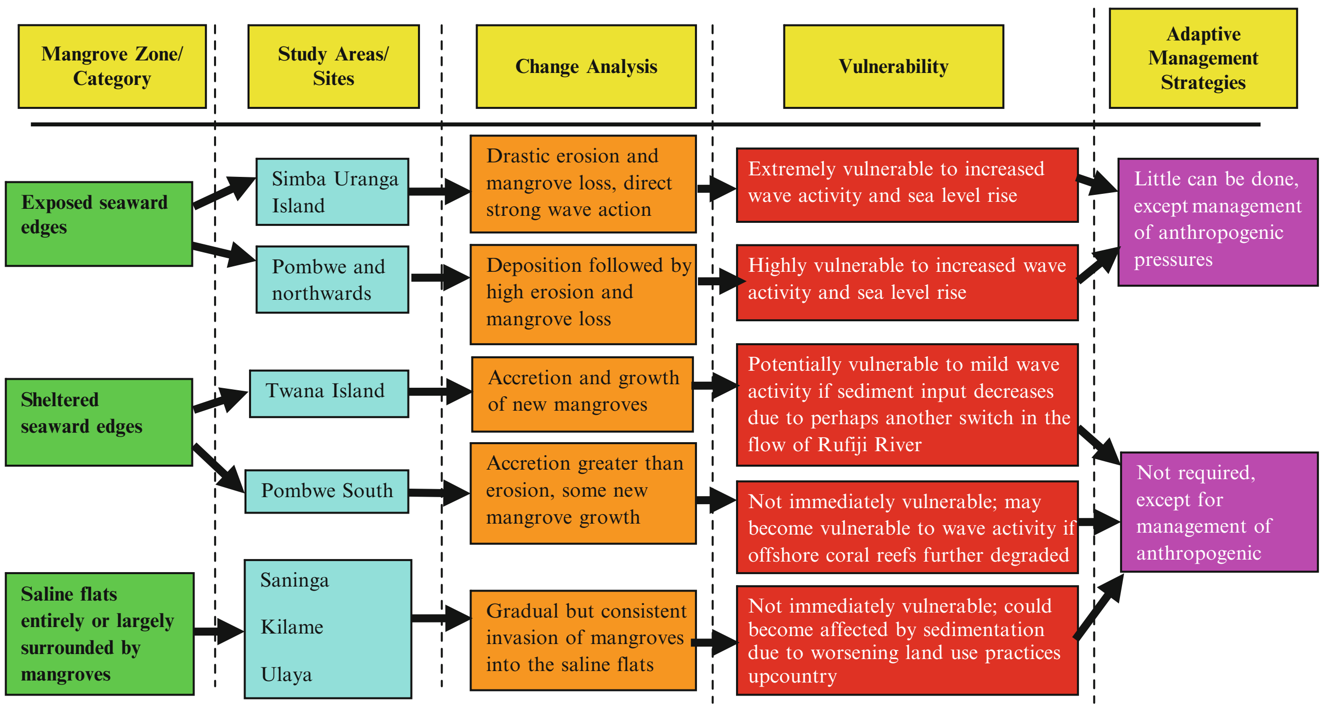

Fig. 9 Flow chart summarizing change analysis, vulnerability and adaptive management strategies for exposed seaward edges, sheltered seaward edges and saline flats entirely or largely surrounded by mangroves

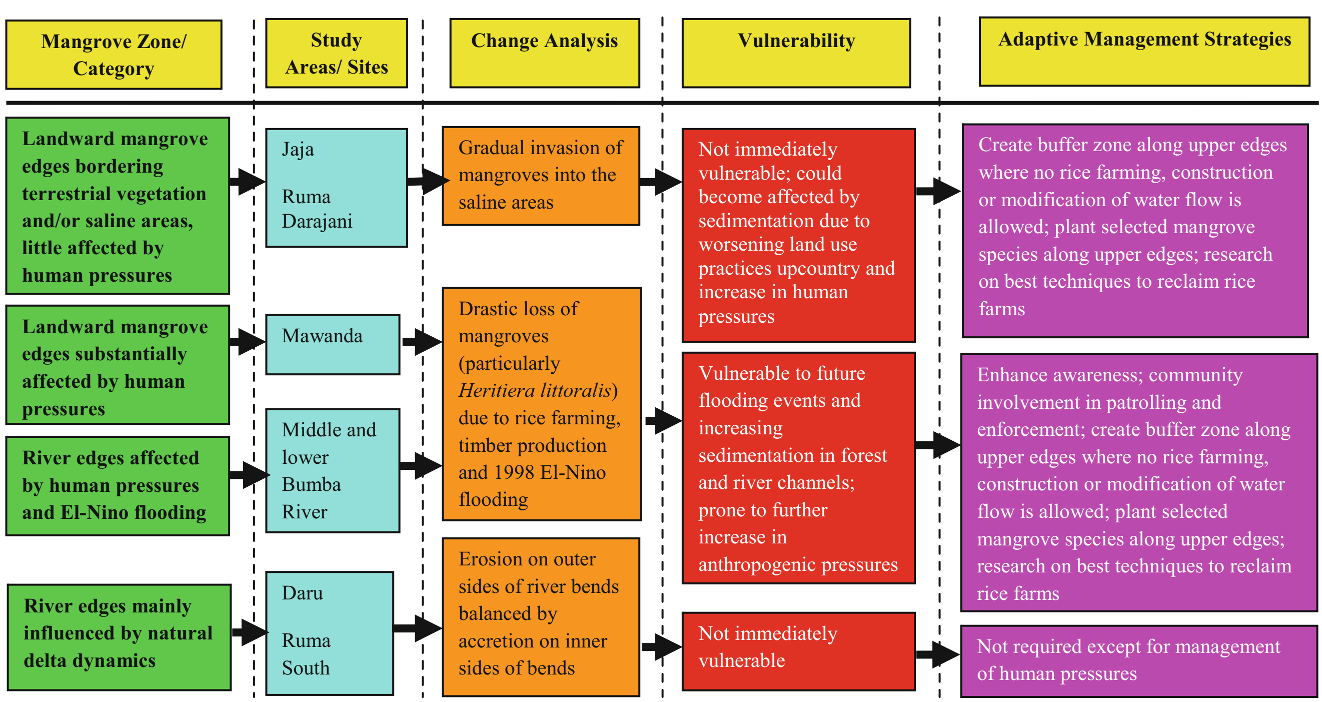

Fig. 10 Flow chart summarizing change analysis, vulnerability and adaptive management strategies for landward edges bordering terrestrial vegetation without and with human pressures as well as river edges affected by human pressures and El-Nino flooding and river edges influenced by natural dynamics

overall trend of erosion along Simba Uranga and Pombwe seaward edges, the two islands are subjected to different coastal processes. Along Simba Uranga, there is strong direct wave action that erodes the shoreline, immediately sweeping trees away, leaving steep banks of eroded mangrove mud and exposed mangrove roots. By contrast, along Pombwe Island, initially waves carry coarse sand from the shallow bottom and deposit it along the coast, killing the mangroves. Further wave action then erodes the sand bank away along with the dead mangroves.

Seaward Edges Where Accretion Is Greater than (or Balances) Erosion

Along the seaward edges that are relatively sheltered from wave action by offshore islands and reefs, such as Twana and Pombwe south, accretion was observed to be greater than (or at least balanced) erosion. Twana also receives high sediment load from Bumba River. Thus, such seaward edges are not immediately vulnerable to climate change (Fig. 9), but could become so if the offshore coral reefs are further degraded and/or sediment load into the ocean decreases, for example, due to changes in rainfall patterns or perhaps another shift in the flow of Rufiji River.

Saline Flats That Are Entirely or Largely Surrounded by Mangroves

Saline flats that are entirely or largely surrounded by mangroves, such as at Saninga, Kilame and Ulaya, make the best areas to observe effects of sea level rise and possible upward invasion of mangroves. This is because anthropogenic pressures are generally low and there is adequate space for mangrove colonization. Moreover, neither erosion nor excessive sedimentation takes place in such sites as evidenced by the fact that the tops of the beacons put in place to mark plots in this study were found to be approximately the same level above the sediment after two years.

All three saline flats showed invasion of mangroves into the flats (Fig. 9), as indicated by increase in both basal area and density of trees, saplings and seedlings. Though this study provides strong evidence that this trend is likely due to sea level rise, further studies on the inundation of tides into these flats should be conducted to confirm this. These saline flats are not immediately vulnerable to climate change factors. However, they could be negatively affected in the event of increased sedimentation due to poor agricultural practices in the Rufiji watershed area.

Landward Mangrove Edges and/or Saline Areas Little Affected by Human Pressures

At Jaja and Ruma Darajani, both ecological plot sampling and indigenous knowledge indicated that mangroves are invading into the saline areas, but not into the areas consisting of terrestrial vegetation, since along such edges the ground raised up slightly. Such areas are not immediately vulnerable to climate change factors (Fig. 10). However, they could potentially become affected by sedimentation and are prone to future increase in anthropogenic pressures since they are both located near human settlements.

Landward Mangrove Edges Substantially Affected by Human Pressures

Although Mawanda was the only site in this category that was studied in detail, it is typical of other areas in upper Rufiji. Thus, the conclusions drawn here can be projected to many other landward edges where anthropogenic pressures are substantial, particularly because of their suitability for the establishment of rice farms. Once rice crops have been grown in a certain area for a few years, it is no longer suitable for rice, so the area is abandoned and new mangrove areas are converted to rice farms. Mangroves do not easily re-establish in abandoned rice fields since the soil properties change. Therefore, this practice of clearing and abandoning has lead to large scale loss of mangroves. A negative spinoff from rice farming is that the pesticides applied kill mangrove macrofauna, which are important in the process of decomposition and nutrient recycling in the forests. Thus, ecosystem balance is disturbed.

Heritiera littoralis is often the mangrove species that is hardest hit by the establishment of rice farms (Fig. 10). *H. littoralis* zones are most easily converted to rice since this species grows in areas that are very near to a freshwater environment, with only occasional inundation of slightly saline water. Moreover, *H. littoralis* is a tall, straight tree with a large girth that is excellent for producing timber which can be exported to other areas like Dar es Salaam city and sold for a good price. On top of these cutting pressures, *H. littoralis* is the species that was hardest hit by the 1998 El-Nino flooding event, which affected many upper mangrove zones in Rufiji.

The increased sedimentation in river channels and in the forests themselves, as communicated by the villagers, is likely due to rampant deforestation and poor agricultural practices upcountry in the Rufiji watershed area and in the delta itself. This trend could hinder adequate influx of both seawater and freshwater and thus potentially result in large-scale die off of mangroves in the long term.

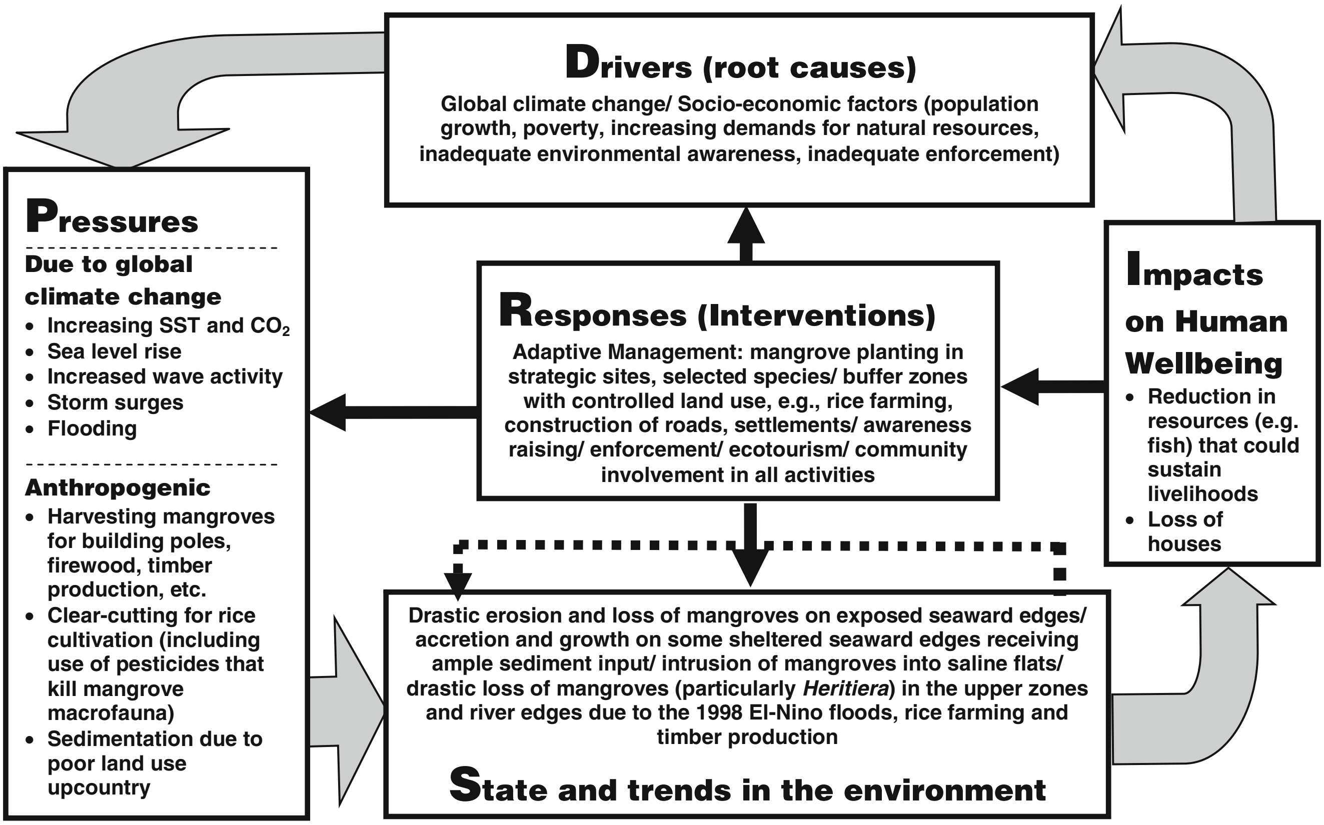

Fig. 11 Analysis of drivers, pressures, state changes, impacts and responses related to Rufiji Estuary

Thus, Mawanda and many other similar areas are extremely vulnerable (Fig. 10) due to the combined pressures of conversion to rice farms, large scale cutting for timber production, increasing sedimentation and future flooding events that are likely to occur due to the trend towards irregularity in rainfall patterns. Not only the forest area in general is vulnerable, but *Heritiera littoralis* is potentially in danger of local extinction in North Rufiji.

River Channel Edges

Along some river edges, such as Daru and Ruma South, mangroves are mainly influenced by natural delta dynamics and thus are not immediately vulnerable to climate change (Fig. 10). However, many other river edges, particularly in upper zones, have been substantially affected by anthropogenic pressures, such as was observed at Middle Bumba and Upper Bumba River sites. Most of the discussion above on landward mangrove edges affected by human pressures also applies here, since these areas are similarly suitable for conversion to rice farms and the production of timber, particularly from *Heritiera littoralis*. In addition, however, degradation along such river edges was drastically exacerbated by the 1998 El-Nino flooding. These factors have resulted in changes in the soil environment, such that the remaining mangroves exhibit poor health. Such areas are highly vulnerable (Fig. 10) to future flooding events, further human pressures and increasing sedimentation.

Overall Change Analysis, Vulnerability, and the DPSIR Cycle in Rufiji Estuary

Analysis of the Driver-Pressure-State-Impact-Response (DPSIR) cycle in Rufiji, as documented in this chapter, is summarized in Fig. 11. The greatest state changes over time were observed in seaward edges exposed to strong wave action where the drastic rate of erosion and loss of mangroves can be attributed to sea level rise combined with increased wave activity and storm surges during peak periods of the year. This study also indicates significant inland invasion of mangroves, likely a result of sea level rise. Overall however, the gains appear to be much smaller than losses (Figs 9 and 10). Thus, over an extended period, the Rufiji Estuary mangroves are likely to become increasingly vulnerable to loss and the delta may subsequently become shorter. Rigorous adaptive management, as recommended in the last section of this chapter, are required in order to facilitate landward migration of mangroves to keep up with sea level rise and to restore large areas already lost due to human influence.

Impacts on human wellbeing (Fig. 11), as a result of state changes in the environment, include a reduction in resources (ecosystem goods and services) that could otherwise help to sustain livelihoods. The mangroves in the Rufiji cover vast areas and the mangrove resources available are still considerable. However, if the current pressures and state changes continue at the present pace, they will have serious repercussions on human wellbeing in the years soon to come. A specific impact that is imminent is that houses will be lost along seaward edges that are eroding rapidly. In the case of Simba Uranga village, the entire settlement may have to be re-located elsewhere within a few years.

There is a reciprocal cause-and-effect relationship between poverty and environmental degradation (Wagner 2007). That is, poverty engenders destructive or inappropriate resource use patterns, which lead to environmental degradation and, in turn, environmental degradation generates further poverty (Moffat et al. 1998). This is particularly true where people have a strong dependence on marine and coastal resources (Wagner 2007), such as in the Rufiji Estuary. According to the DPSIR Analytical Approach (UNEP 2006), unless there are strong, appropriate responses to changes caused by both climate change factors and anthropogenic pressures, the impacts that environmental state changes have on human wellbeing will feed forward in the cycle and increase the tenacity of drivers such as poverty, inadequate environmental awareness/education and inadequate enforcement (Fig. 11). Details of the recommended adaptive management strategies are provided in section "Recommendations for mitigation and adaptive management".

Comparative Vulnerability of Other Mangrove Areas in the World

Comparing mangrove sites in the three countries (Tanzania, Cameroon, and Fiji) that participated in the WWF "Coastal Resilience to Climate Change" project, Douala Estuary (Cameroon) appeared to have the highest vulnerability due to low tidal range, non-climate pressures and loss of the seaward edge, while Tikina Wai (Fiji) was also vulnerable due to a subsiding coastal edge and a low tidal range (Ellison 2015). Though erosion was severe on some coastal edges of the Rufiji, comparatively it showed some resilience to climate change due to an uplifting coastline and a very high tidal range. The vastness of the mangrove area contributes to its resilience. In Cameroon, satellite imagery showed an overall mangrove loss of only 5% from 1975 to 2007 (Ellison and Zouh 2012), though there was loss of up to 3 m/year along two thirds of the seaward edge. Moreover, there was 89% loss of one offshore mangrove island, due to a combination of climate change factors and human pressures. The future of mangroves in Fiji looks grim. Mangroves on some low-lying islands lacking rivers carrying sediment are the most sensitive to sea level rise (Ellison and Fiu 2010). On some other islands, human settlements on landward edges preclude potential mangrove migration.

By contrast, the Rufiji mangroves form a strip of many kilometres (up to 25 km) along the coast. In addition, the nearly flat, low-lying flood plains on the upper side of the mangroves provide the possibility for mangrove migration. Thus, there is appropriate, adequate space (if protected through creating butter zones) for upward shift of the mangrove area. However, mangroves around Mafia Island, just offshore from the Rufiji, and some forests along the Kilwa coast south of the Rufiji have similar restrictions to some of the Pacific islands in that they cover a coastal strip of only a few meters, above which there is a steep slope that precludes any possible upward shift. Thus, such mangrove strips are likely to be eliminated in the event of substantial sea level rise.

Trends in mangroves vary in different parts of the world. According to a study at Gazi Bay, Kenya, a sea level rise of 48 cm by 2100 is surprisingly predicted to result in an increase in the dominant mangroves, such as *Rhizophora mucronata* and *Ceriops tagal*, due to the physiographic features of the area that allow for colonization of new areas (Di Nitto et al. 2014). In a vulnerability assessment of mangroves in Guangxi (China), applying the high-level IPCC A1FI scenario of 0.59 cm/year projected sea level rise (IPCC 2007), Li et al. (2015) predicted that 25.8 and 37.3% of mangroves would be within areas of low vulnerability by the 2030s and the 2050s, respectively; and that 23.9 and 13.4% of mangroves would be within areas of low and moderate vulnerability, respectively, in the 2100s.

Depending on physiographic features and other factors, projected mangrove losses in the Pacific Islands by 2100 vary considerably (Gilman et al. 2006b), ranging from no loss being predicted in the Solomon Islands to 23, 657 ha (58%) of the mangroves being lost in Fiji. Studies on some high Pacific islands (Krauss et al. 2010) showed that susceptibilities of mangroves to sea level rise were not always bleak, but varied according to hydrogeomorphics, with fringe, riverine, and interior settings showing elevation changes of -1.30, 0.46, and 1.56 mm/year, respectively, over a period of 6½ years. In Papua New Guinea, where human pressures have been minimal, analysis of aerial photography and satellite imagery from 1973 to 2002, indicated that there was a gross loss of 7,191 ha (primarily along seaward edges) and a gross gain of 6,199 ha (associated with rivers and landward edges), resulting in a net loss of 992 ha (Shearman 2010). In Western Samoa, Gilman et al. (2007) observed that, in three sites, the seaward edges of mangroves migrated landwards by 25, 64, and 72 mm/year over four decades, which was 12 to 37 times the observed rate of relative sea level rise during the same time period. Two of the sites showed overall reduction in mangroves, with high correlation between changes in seaward mangrove edges and changes in relative sea level. They predicted that, by 2100, there may be a reduction in mangrove area of up to 50% in these three Western Samoan sites and of 12% in the Pacific Islands in general.

Sea level rise is likely to pose the highest threat to atolls and low-lying islands around the world. For example, two-thirds of the land area in the Pacific islands of Kiribati are less than 2 m above mean sea level (Donner and Webber 2014) and the Indian Ocean Maldives are just 1-2 m above sea level (Jagtap et al. 2008). With projected sea level rise of up to 1 m or more by 2099 (Ellison 2012), this means that most or all of such islands will be under water by the end of the century. Of course, in such cases, mangrove loss is undoubtedly of lesser concern compared with the potential loss of human settlements and even entire small nations, unless drastic mitigative measures are taken.

Recommendations on Methodology, Adaptive Management and Further Research

Lessons for a Generalizable Methodology for Change Analysis and Vulnerability Assessment

Lessons which can be learned from the mangrove change analysis and vulnerability assessment reported in this chapter include the following:

- Using multiple methods for data collection, including both natural science methods (e.g., ecological plot assessment and remote sensing) and social science methods (to obtain indigenous knowledge), greatly adds to the breadth of the information obtained and each method helps to cross-check and reinforce the other. Indigenous knowledge proved to be very reliable and provided oral history of changes going back much farther in time than the natural science methods could. It also pointed to likely causal factors of change.
- The participatory monitoring approach (Wagner 2005) applied in this study proved to have many benefits, including easily obtaining site specific indigenous knowledge, greatly increasing manpower for the collection of large amounts of data with the limited resources of time and funds, enhancing the environmental awareness of the community, and ensuring the re-tracing of sites and plots easily during the repeat study. Sometimes it was difficult to find plots with the GPS, so it became a standing joke during the fieldwork that the indigenous GPSs (meaning the local villagers) were much more accurate than the scientific ones, since the villagers could invariably take the team directly to the permanent plots.
- The strategic sampling of seaward and landward edges of the mangroves proved to be very efficient since this study focussed on determining the effects of climate change. In vast mangrove areas, such as the Rufiji Estuary, random sampling of the whole area would be very time consuming and much of the information provided would not be directly related to the effects of climate change.
- The paired-sample research design using permanent plots and subsequent paired-sample data analysis proved to be a very effective and statistically powerful way of detecting even small changes in the mangroves over the very short two-year period of the study. Without the paired design, the huge spatial variation inherent in mangrove vegetation would have overridden the incremental changes over time and likely no statistically significant changes would have been shown in any of the sites, except in certain seaward edges where changes were drastic. Just as one example to demonstrate this important point, using the t-test for two independent samples to analyze data for Site SR1 would have resulted in $t = 0.369$, $P = 0.7147$ (a completely insignificant result) instead of $t = 3.628$, $P = 0.0023$ (a highly significant result) as provided by the paired-sample t-test. Strong pairing between observations in this site was indicated by a very high correlation coefficient ($r = 0.9898$).
- Marking plots with beacons such that the tops were consistently 2 cm above the surface of the substrate was very useful for the detection of any possible erosion or sedimentation over time.

It should be noted that WWF's "Coastal Resilience to Climate Change" project, conducted in Tanzania, Cameroon and Fiji, resulted in the development of a manual, put together by Ellison (2012), which presents a generalized methodology for climate change vulnerability assessment and adaptive management for mangrove ecosystems.

Recommendations for Mitigation and Adaptive Management

Gilman et al. (2006a) provided very detailed guidelines for institutional capacity building and adaptive management strategies to address the effects of climate change on mangroves. The mitigation responses or adaptive management interventions required to reduce, stop or perhaps even reverse, some of the present negative trends in the Rufiji mangrove ecosystems are given specifically for each category of mangrove area in Figs. 9 and 10 and summarized in Fig. 11.

A comprehensive environmental management package that is required to address the numerous and complex issues related to the Rufiji Estuary should be such that responses are targeted, either separately or in interaction, towards all components of the DPSIR cycle (Fig. 11), that is, drivers, pressures and environmental state changes, which will in turn have positive impacts on human wellbeing. Climate change and its associated pressures are global issues and thus interventions directed towards these are, for the most part, beyond the scope of this study. An exception may be replanting Rufiji mangroves in degraded areas, which may play a small part in returning some of the carbon from the atmosphere to the biosphere.

Responses directed at the socio-economic drivers (which at the same time usually target impacts on human wellbeing) should particularly include poverty reduction. According to Moffat et al. (1998), the vicious cycle of poverty, ignorance and environmental degradation can only be broken through responses or interventions that put the wellbeing of local communities at the forefront. An important strategy that can be implemented in the Rufiji Estuary is to take steps to empower local residents of the area to develop alternative sources of income and to diversify livelihoods.

Responses targeted at eliminating or controlling anthropogenic pressures, particularly excessive mangrove harvesting, timber production and rice farming, are essential in order to mitigate further loss of mangroves. Though this is largely the duty of the government, it requires the collaboration of all supporting agencies. Community involvement in patrolling and enforcement is essential. The reduction of anthropogenic pressures has been shown to be important in increasing mangrove ecosystem resilience to climate change by ensuring the dispersal and survival of propagules. Digital terrain modelling (DTM) in Gazi Bay (Kenya) showed that anthropogenic pressures may greatly hinder propagule dispersal, primarily due to the loss of aerial root masses, which are important in providing stranding areas for propagules (Di Nitto et al. 2008).

Responses targeted at the state of the environment are extremely important. Restoration of forest areas previously lost and mangrove planting on the upward edges of mangroves increases the resistance and resilience of mangrove ecosystems to climate change factors and facilitates the landward migration of mangroves to keep up with sea level rise. A yearly increase of 3.2 mm vertically, the current estimated rate of global mean sea level rise (Merrifield et al. 2014), can correspond to a horizontal seawater inundation of several meters into low-lying areas such as the Rufiji, which are almost completely flat.

Specific adaptive management strategies that could be effective in the Rufiji Estuary are as follows:

- Setting up buffer zones (McLeod and Salm 2006) along the upper mangrove edges, which should be protected from human influence, in order to provide appropriate, adequate space for mangrove migration inland to keep up with projected sea level rise;
- Planting mangroves along upper edges of the forests, within the buffer zones, to facilitate upward mangrove invasion, including experimenting to determine which species establish the best;
- Planting mangroves inwards into saline flats, particularly *Avicennia marina* and *Ceriops tagal*;
- Protecting *Heritiera littoralis* and replanting it in many parts of the Rufiji delta where this species has been lost due to the 1998 El-Nino flooding event, rice farming, and timber production;
- Mangrove restoration in areas such as Mwanda where there has been drastic loss of mangroves due to timber production and rice farming;
- Controlling the use of pesticides in rice farms near mangrove forests and investigating alternatives;
- Pilot testing a scheme for sustainable utilization of mangroves whereby local inhabitants, either as individuals or groups, are allocated plots, within which they would replant mangroves and would be allowed to harvest at a sustainable rate, both for their own use and for sale as a source of livelihood;
- Developing ecotourism in the Rufiji Estuary, which has great potential due to the high aesthetic value of the forest, by involving the local resources users in order to improve their economic wellbeing and to motivate them to protect and conserve the mangroves (Wagner, 2007).

Recommendations for Further Research

While conducting this study, it became apparent that research needs to be carried out to investigate a number of issues which could help to further analyze changes in the Rufiji Estuary and to improve adaptive management. These are as follows:

- To determine changes in sea level in relation to changes in mangrove substrate level due to sedimentation or erosion in the upper reaches of the Rufiji Delta, such as at Mawanda, where sediments appear to be building up in the mangrove forests;
- To conduct topographic surveys in strategically selected areas in order to facilitate proper identification of buffer zones into which mangroves may migrate to keep pace with sea level rise;

- To investigate the dynamics of selected saline flats that are surrounded by mangroves (e.g., Saninga), including monitoring changes over time in the inundation of tides and sediment load;
- To conduct studies on abandoned rice farms in order to investigate the best techniques required to restore them to healthy mangrove forests;
- To assess the comparative sediment loads of various channels of North and South Rufiji and the dynamics of changes in these loads over seasons and years;
- To monitor populations of *Terebralia palustris* in areas where it is collected for making lime in order to determine appropriate exploitation rates;
- To carry out remote sensing change analysis for all areas in Rufiji Estuary and surrounding coastal strips to determine overall trends in mangrove area coverage and changes in specific areas.

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References

Akwilapo FD (2001) The distribution and abundance of mangrove species and associated macrobenthos in ecosystems with varied anthropogenic degradation. M.Sc. thesis, University of Dar es Salaam, 111 p

Alongi DM (2014) Carbon cycling and storage in mangrove forests. Ann Rev Mar Sci 6:195–219

Arthurton R, Barker S, Rast W, Huber M, Alder J, Chilton J, Gaddis E, Pietersen K, Zöckler C, Al-Droubi A, Dyhr-Nielsen M, Finlayson M, Fortnam M, Kirk E, Heileman S, Rieu-Clark A, Schäfer M, Snoussi M, Tang LD, Tharme R, Vadas R, Wagner G (2007) Chapter 4: water. In: UNEP, fourth global environmental outlook for development (GEO-4). UNEP, Nairobi, pp 115–156

Clark CO, Webster PJ (2003) Interdecadal variability of the relationship between the Indian Ocean zonal mode and East African coastal rainfall anomalies. J Climate 16(3):548–554

Di Nitto D, Dahdouh-Guebas F, Kairo JG, Decleir H, Koedam N (2008) Digital terrain modelling to investigate the effects of sea level rise on mangrove propagule establishment. Mar Ecol Prog Ser 356:175–188

Di Nitto D, Neukermans G, Koedam N, Defever H, Pattyn F, Kairo JG, Dahdouh-Guebas F (2014) Mangroves facing climate change: landward migration potential in response to projected scenarios of sea level rise. Biogeosciences 11:857–871

Donato DC, Kauffman JB, Murdiyarso D, Kurnianto S, Stidham M, Kanninen M (2011) Mangroves among the most carbon-rich forests in the tropics. Nat Geosci 4:293–297

Donner SD, Webber S (2014) Obstacles to climate change adaptation decisions: a case study of sea-level rise and coastal protection measures in Kiribati. Sustain Sci 9:331–345

Duke NC, Meynecke JO, Dittmann S, Ellison AM, Anger K, Berger U, Cannicci S, Diele K, Ewel KC, Field CD et al (2007) A world without mangroves? Science 317:41–42

Ellison JC (2005) Impacts on mangrove ecosystems. The great greenhouse gamble: a conference on the impacts of climate change on biodiversity and natural resource management: conference proceedings, Sydney, NSW, EJ

Ellison JC (2012) Climate change vulnerability assessment and adaptation planning for mangrove systems. World Wildlife Fund (WWF), Washington, DC, 130 p

Ellison JC (2015) Vulnerability assessment of mangroves to climate change and sea-level rise impacts. Wetl Ecol Manage 23:115–137

Ellison JC, Fiu M (2010) Vulnerability of Fiji's mangroves and associated coral reefs to climate change: a review. WWF South Pacific Program, Suva, 50 p

Ellison JC, Stoddart DR (1991) Mangrove ecosystem collapse during predicted sea-level rise: Holocene analogues and implications. J Coast Res 7:151–165

Ellison JC, Zouh I (2012) Vulnerability to climate change of mangroves: assessment from Cameroon, Central Africa. Biology 1:617–638

English S, Wilkinson C, Baker V (1997) Survey manual for tropical marine resources, ASEAN Australia marine project, living coastal resources, 2nd edn. Australian Institute of Marine Sciences, Townsville, 368 p

Field CD (1995) Impacts of expected climate change on mangroves. Hydrobiologia 295(1–3):75–81

Francis J, Wagner GM, Mvungi A, Ngwale J, Sallema R (2001) Tanzania national report, phase I: integrated problem analysis. GEF-MSP sub-Saharan Africa project on "development and protection of the coastal and marine environment in sub-Saharan Africa", UNEP, Nairobi, 60 p

Gilman E, Ellison JC, Jungblat V, VanLavieren H, Wilson L, Areki F, Brighouse G, Bungitak J, Dus E, Henry M, Sauni I Jr, Kilman M, Matthews E, Teariki-Ruatu N, Tukia S, Yuknavage K (2006a) Adapting to Pacific Island mangrove responses to sea level rise and other climate change effects. Climate Res 32:161–176

Gilman E, Van Lavieren H, Ellison JC, Jungblut V, Wilson L, Areki F, Brighouse G, Bungitak J, Dus E, Henry M, Sauni Jr I, Kilman M, Matthews E, Teariki-Ruatu N, Tukia T, Yuknavage K (2006b) Pacific island mangroves in a changing climate and rising sea. UNEP regional seas reports and studies no. 179. United Nations Environment Programme, regional seas programme, Nairobi, Kenya, 58 p

Gilman E, Ellison JC, Coleman R (2007) Assessment of mangrove response to projected relative sea-level rise and recent historical reconstruction of shoreline position, American Samoa. Environ Monit Assess 124:105–130

Gilman EL, Ellison J, Duke NC, Field C (2008) Threats to mangroves from climate change and adaptation options. Aquat Bot 89:237–250

Grimsditch GD, Salm RV (2006) Coral reef resilience and resistance to bleaching, IUCN, Gland, Switzerland, 52 p

Houghton J, Ding Y, Griggs D, Noguer M, van der Linden P, Dai X, Maskell K, Johnson C (eds) (2001) Climate change 2001: the scientific basis, published for IPCC. Cambridge University Press, Cambridge, 881 p

IPCC (2007) Climate change 2007: impacts, adaptation and vulnerability. Working group II contribution to the IPCC fourth assessment report. Summary to policymakers. Available online: http://www.ipcc.ch

Jagtap TG, Kulkarni VA, Verlekar XN (2008) Vulnerability and adaptation of ecologically sensitive mangrove habitats to the changing climate. In: Proceedings of conference on marine problems and specific solutions, COMPASS2008, June 15–18, 2008, Maldives, pp 15–18

Kajia YS (2000) Assessment of the effects of rice farming in the mangrove forests of the Rufiji Delta (mainland Tanzania). M.Sc. thesis, Vrije Universiteit Brussel, Brussels, 82 p

Krauss KW, Cahoon DR, Allen JA, Ewel KC, Lynch JC, Cormier N (2010) Surface elevation change and susceptibility of different mangrove zones to sea-level rise on Pacific high islands of Micronesia. Ecosystems 13:129–143

Li S, Meng X, Ge Z, Zhang L (2015) Vulnerability assessment of the coastal mangrove ecosystems in Guangxi, China, to sea-level rise. Reg Environ Chang 15:265–275

McClanahan TR (1988) Seasonality in East Africa's coastal waters. Mar Ecol Prog Ser 44:191–199

McLeod E, Salm RV (2006) Managing mangroves for resilience to climate change. IUCN, Gland, 64 p

Merrifield MA, Thompson P, Leuliette E, Nerem RS, Hamlington B, Chambers DP, Mitchum GT, McInnes K, Marra JJ, Menéndez M, Sweet W (2014) Global oceans: sea level variability and change [in "State of the climate in 2013"]. Bull Am Meteorol Soc 95(7): S71–S73

Mgaya YD (2004) The vulnerability of mangrove and adjacent coral reef ecosystems to climate change in the Rufiji Delta. Report submitted to WWF, The Eastern African marine ecoregion programme, Dar es Salaam, 27 p

Moffat D, Ngoile M, Linden O, Francis J (1998) The reality of the stomach: coastal management at the local level in eastern Africa. Ambio 27(8):590–598

Obura D (2005) Resilience and climate change: lessons from coral reefs and bleaching in the Western Indian Ocean. Estuar Coast Mar Sci 63:353–372

Organisation for Economic Co-operation and Development (1993) OCED core set of indicators for environmental performance reviews. OECD environment monographs no. 83. OCED, Paris, 39 p

Sallema RE, Mtui GYS (2008) Review: adaptation technologies and legal instruments to address climate change impacts to coastal and marine resources in Tanzania. Afr J Environ Sci Tech 2(9):239–248

Sandilyan S, Kathiresan K (2012) Mangrove conservation: a global perspective. Biodivers Conserv 21:3523–3542

Semesi AK (1991) Management plan for the mangrove ecosystem of mainland Tanzania, vol 11. Mangrove Management Plan for All Coastal Districts. Ministry of Tourism, Natural Resources and Environment, Forest and Beekeeping Division, Catchment Forest Project, Dar es Salaam

Shearman PL (2010) Recent change in the extent of mangroves in the northern Gulf of Papua, Papua New Guinea. AMBIO 39:181–189

Stoddart DR (1971) Environment and history in Indian Ocean reef morphology. Symp Zool Soc Lond 28:3–38

UNEP (1994) Assessment and monitoring of climatic change impacts on mangrove ecosystems. UNEP regional seas reports and studies no. 154. UNEP, Nairobi

UNEP (2006) Country environmental profile information system. UNEP. http://countryprofiles.unep.org/profiles/ER/profile/state-of-the-environment

Wagner GM (2004) Coral reefs and their management in Tanzania. Western Indian Ocean J Mar Sci 3(2):227–243

Wagner GM (2005) Participatory monitoring of changes in coastal and marine biodiversity. Indian J Mar Sci 34(1):136–146

Wagner GM (2007) The Dar es Salaam seascape: a case study of an environmental management 'hotspot'. Western Indian Ocean J Mar Sci 6(1):85–109

Wagner GM (2008) Baseline assessment of the mangrove ecosystems of Rufiji, Mafia and Kilwa (Tanzania) for vulnerability to climate change, a WWF US/GEF coastal resilience to climate change project, WWF Tanzania Country Office, Dar es Salaam, 115 p

Wagner GM, Sallema-Mtui R (2010) Change analysis of Rufiji-Mafia-Kilwa mangroves (Tanzania) in relation to climate change factors and anthropogenic pressures, a WWF US/GEF coastal resilience to climate change project, WWF Tanzania Country Office, Dar es Salaam, 147 p

Wagner GM, Mgaya YD, Akwilapo FD, Ngowo RG, Sekadende BC, Allen A, Price N, Zollet EA, Mackentley N (2001) Restoration of coral reef and mangrove ecosystems at Kunduchi and Mbweni, Dar es Salaam, with community participation. In: Richmond MD, Francis J (eds) Marine science development in Tanzania and Eastern Africa. Proceedings of the 20th anniversary conference on advances in marine sciences in Tanzania, 28 June–1 July 1999, Zanzibar, Tanzania. Institute of Marine Sciences (IMS) and Western Indian Ocean Marine Science Association (WIOMSA), Zanzibar, pp 467–488

Wagner GM, Makota V, Sallema R (2003) Mangrove forests. In: Tanzania state of the coast report 2003: the national ICM strategy and prospects for poverty reduction. Tanzania Coastal Management Partnership (TCMP), Dar es Salaam, pp 5–11

Wagner GM, Akwilapo FD, Mrosso S, Ulomi S, Masinde R (2004) Assessment of marine biodiversity, ecosystem health and resource status in mangrove forests in Mnazi Bay-Ruvuma Estuary marine park. IUCN Eastern Africa programme, Nairobi, 106 p

Wang YQ, Ngusaru A, Tobey J, Makota V, Bonynge G, Nugranad J, Traber M, Hale L, Bowen R (2003) Remote sensing of mangrove change along the Tanzania Coast. Mar Geod 26(1–2):1–14

Small Estuarine and Non-Estuarine Mangrove Ecosystems of Tanzania: Overlooked Coastal Habitats?

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Abstract

Small estuaries and non-estuarine habitats harboring mangroves are very important ecosystems which provide important ecosystem goods and services; such as provision of ecological niches for juvenile fishes and invertebrates, enhances fisheries, and in biodiversity conservation. Similar to large estuaries, they are highly perturbed which threatens their existence. This chapter uses beach seine, underwater visual census, and stable isotope data to discuss the importance of and threats to small estuaries and non-estuarine mangroves found in Dar es Salaam, Bagamoyo and Zanzibar, Tanzania. For example, mangroves of Kunduchi (Dar es Salaam) and Mbegani (Bagamoyo) which harbour predominantly higher densities of juveniles ($\leq$10 cm) of two economically important species—*Lutjanus fulviflamma* and *Lethrinus harak*—than adjacent coral reefs. Evidence suggests further that the Kunduchi mangroves replenish fish populations on adjacent coral reefs; where over 90% and 29% of adult *L. fulviflamma* and *L. harak* individuals, respectively, have been identified to have lived in the mangroves as juveniles. In terms of habitat utilization by different size classes of fish, five of the 13 species (*Lethrinus lentjan*, *L. variegatus*, *Pelates quadrilineatus*, *Siganus sutor* and *Sphyraena barracuda*) found in Chwaka Bay (Zanzibar) were found as small-sized individuals in shallow and turbid mangrove areas with large juveniles and sub-adults in adjacent seagrass beds. The non-estuarine mangroves of Kunduchi and those of Mtoni estuary (Dar es Salaam) are subjected to pollution from urban activities. For example, stable isotope data of fishes indicate elevated levels of nitrogen in these mangroves with highest levels ($\delta^{15}N = 15.2 \pm 0.2$) recorded in Mtoni estuary. In view of their importance and threats they face, these ecosystems require attention similar to large estuaries. If the current degradation rate of these 'overlooked' but equally important ecosystems continues, they may be declared 'functionally disappeared' in a few decades.

Keywords

Small estuaries • Non-estuarine mangroves • Mangrove ecosystem • Ecosystem goods and services • Ecosystem values • Biodiversity conservation • Tanzania • Pollution • Seagrass beds • Fish populations • Kunduchi • Mbegani • Mtoni estuary • Chwaka Bay

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Introduction

Small estuaries and non-estuarine habitats harboring mangroves are very important ecosystems that, like many other large estuarine ecosystems, provide important and myriad functions and services. These incclude provision of ecological niches for juvenile fishes and invertebrates; prevention of coastal erosion, acts as buffers for pollutants, and sequester carbon dioxide (Alongi 2002; Costanza et al. 1997; Diop et al. 2002; Kimirei et al. 2011; Kimirei et al. 2013; Lugendo et al. 2006; Nagelkerken 2009; Nagelkerken et al. 2008; Dittmar et al. 2006). Hence they are equally important in enhancing fisheries and biodiversity conservation (Blaber 2009; Kimirei et al. 2013). They also provide food, firewood and medicine to human beings (Blaber 2009).

These mangroves do not occur in isolation; they are connected to, and contribute to and/or sustain other coastal habitats (i.e. seagrass beds, coral reefs, algal beds, and mudflats) through transport of dissolved (DOC) and particulate (POC) organic carbon, through fish movement and trapping of sediments which would otherwise smoother the seagrasses and coral reefs. Similar to large estuaries, these ecosystems are faced by natural and anthropogenic perturbations that threaten their existence and ability to provide goods and services. Anthropogenic stressors such as clear-cutting (Mwandya et al. 2009) and waste disposal (De Wolf and Rashid 2008; De Wolf et al. 2001; Kruitwagen et al. 2006; Machiwa 1992) have resulted into shifts in mangrove species' dominance (Wagner 2005; Wagner 2007), and deformities and reduction of growth rates in some resident fishes (Kruitwagen et al. 2006). Poor land use practices, increase in urbanization, high population growth, and increased demand for food and other resources from these ecosystems will surely exacerbate perturbation of not only large estuarine but also small and many other non-estuarine mangroves in Tanzania. And with the changing climate and anticipated sea level rise (Kebede et al. 2010; Dittmar et al. 2006; Pethick and Spencer 1990) these mangroves may lose both their monetary and functional values leading to being declared functionally extinct (Duke et al. 2007; Polidoro et al. 2010). Worse still, small estuarine and non-estuarine mangroves are often more prone to such disturbances as most of them are small in size. It is under such circumstances that the value of small estuaries and non-estuarine mangroves needs attention just like any other large estuarine mangroves in Tanzania and the WIO region for that matter.

This chapter uses five mangrove forests (Mtoni Kijichi—representing small estuarine habitat, Kunduchi, Mbegani and Nunge—representing small non-estuarine mangrove forests, and Chwaka Bay—representing large non-estuarine mangrove forest) (see Fig. 1) as case studies to present the importance of, and threats to, these habitats.

Mangrove Species, Coverage and Distribution in Tanzania

The term 'mangrove' includes trees, shrubs, palms or ground ferns generally exceeding half a meter in height, and which normally grow above mean sea level in the intertidal zone of marine coastal environments, or along estuarine margins (Duke 2006). Mangroves are typical plants of the upper intertidal zone in areas dominated by fine-grained silt and clay sediments.

Along the coastline of Tanzania mangroves occur in patches on gently sloping shores, sheltered bays, lagoons and creeks, river deltas and estuaries, and on leeward sides of peninsulas and islands (Mangora et al. unpublished). Mangroves cover in the United Republic of Tanzania is estimated to range between 127,200 and 133,500 ha (FAO 2007; Griffith 1949, 1950; MTNRE 1991; Semesi 1992); the latter figure being officially considered the total mangrove coverage in the country (115,500 ha on the Tanzania mainland, and 18000 ha on Zanzibar) (Mangora et al. unpublished). Large coverage of mangroves are found along deltas and estuaries of major rivers such as Pangani, Wami, Ruvu, Rufiji and Ruvuma, with the Rufiji Delta having the largest stand of mangroves, on average, in Tanzania and Eastern Africa (51,000 ha; see Table 1a) (Mangora 2011; Semesi 1992; Wang et al. 2003). In Zanzibar, well-developed mangroves occur on Pemba Island (Ngoile and Shunula 1992; Wang et al. 2003). Of the five mangrove sites that this study draws its conclusions from, only Chwaka holds about 1.6% of the total mangrove area in Tanzania; the rest of the sites combined hold about 0.5% (Table 1b). A total of ten mangrove species belonging to six families occur in Tanzania, 9 of which occur in Tanzania mainland (Ngoile and Shunula 1992; Wang et al. 2003) (Table 2), with *Rhizophora mucronata*, *Ceriops tagal* and *Avicennia marina* dominating and *Xylocarpus moluccensis* being the rarest (Wang et al. 2003).

Functions and Ecological Services Provided by Se & Ne Mangroves

Nursery Function, Habitat Connectivity, and Ontogenetic Migration

Tropical and subtropical estuaries are acknowledged worldwide to provide important direct and indirect ecosystem services to multitude of communities in their reach (Costanza et al. 1997; Gladstone 2009; Rönnbäck 1999). These habitats are highly productive and ecologically important that they have been a subject of scientific debate for decades (Blaber 2009; Faunce and Layman 2009). Mangroves, which are the most dominant features of tropical

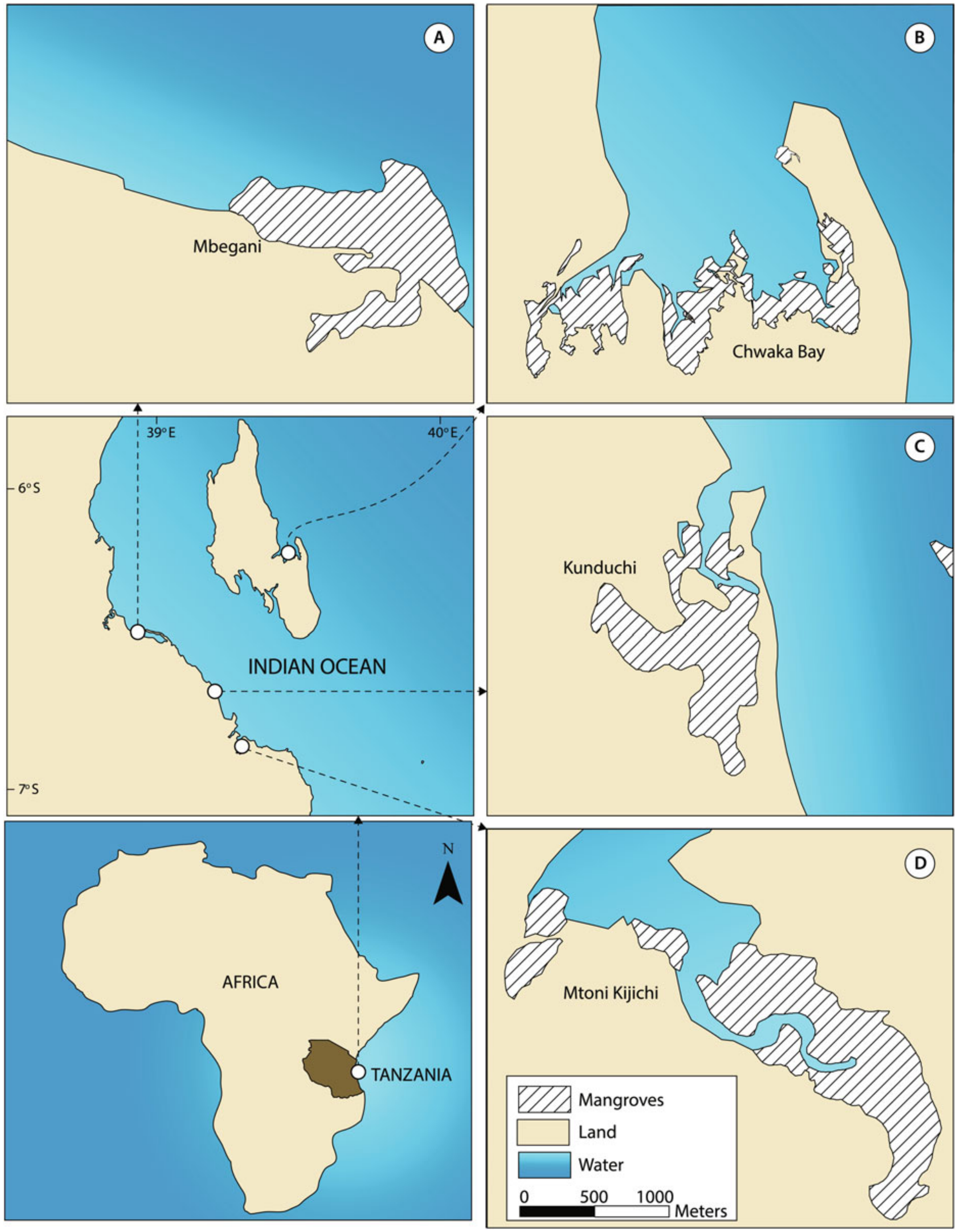

Fig. 1 Locations of non-estuarine (**a–c**) and small estuarine (**d**) mangroves. The Nunge mangroves could not be included in the map due to its small size and proximity to the Ruvu estuary

Table 1 (**a**) Comparison of mangrove areas (hectares) between 1990, 2000, and 2007 for the coastal districts of Tanzania mainland and one district in Zanzibar and (**b**) area coverage of mangrove forests in four study sites and percentage of every stand to an estimated 1,338 km^2 (133,838 ha) area of mangrove in Tanzania

(A)		Mangrove area (Hectares)[a]			
District/Region	Site	1990	2000	2007	Mean
Tanga and Muheza		9,217	9,313	11,227	9,919
Pangani		3,799	3,879	1,847	3,175
Bagamoyo	Ruvu, Mbegani, Nunge	5,039	5,051	5,709	5,266
Dar es Salaam	Mtoni, Kunduchi	2,494	2,516	816	1,942
Kisarawe		4,159	4,092	6,018	4,756
Rufiji		49,799	48,030	55,198	51,009
Kilwa		21,826	21,755	21,156	21,579
Lindi		4,034	9,458	5,311	6,268
Mtwara		9,226	9,458	9,656	9,447
Zanzibar central	Chwaka			2,135	2,135
Total (excluding Chwaka)		109,593	113,552	116,936	113,360

(B)	Mangrove area[b]		
Mangrove site	Km^2	Hectares	%
Chwaka Bay	21.32	2132.21	1.59
Mbegani	2.04	203.89	0.15
Kunduchi	0.65	64.87	0.05
Mtoni-Kijichi	4.04	404.05	0.30

[a]Data for 1990 and 2000 from Tanzania State of the Coast Report 2003, and 2007 from FAO shapefile dataset
[b]Data from FAO (2007)

Table 2 Mangrove tree species found in Tanzania and their presence in different mangrove sites

Scientific name and authority	Family	Presence
Avicennia marina (Forssk.) Vierh	Acanthaceae	15
Lumnitzera racemosa Willd	Combretaceae	4
Pemphis acidula Forst[a]	Lythraceae	
Sonneratia alba J. Smith	Lythraceae	1–5
Heritiera littoralis Dryand	Malvaceae	
Xylocarpus granatum König	Meliaceae	1–5
Xylocarpus moluccensis (Lamk.) Roem	Meliaceiae	
Bruguiera gymnorrhiza (L.) Lamk	Rhizophoraceae	
Ceriops tagal (Perr.) C.B. Robinson	Rhizophoraceae	2,4
Rhizophora mucronata Lamk	Rhizophoraceae	1–5

The mangrove sites are numerically coded as 1 = Chwaka Bay; 2 = Kunduchi; 3 = Mbegani; 4 = Mtoni Kijichi; 5 Nunge
[a]Inclusion of *Pemphis acidula* as one of mangrove species is still debated, the inclusion of the species in this list therefore should be treated with caution

estuaries, often provide a supporting function to many fish, invertebrates and other species—such as birds and amphibians (Nagelkerken et al. 2008). For example, a number of commercially important tropical fish species are associated with mangroves and seagrass beds as their primary juvenile and nursery habitats (Kanai et al. 2014; Kimirei et al. 2011; Kimirei et al. 2013; Nagelkerken 2009; Nagelkerken et al. 2008). The juveniles of fishes that settle into these habitats migrate back to their natal locations during sub-adult and adult stages—most often into deep-water habitats such as mudflats and coral reefs (Kimirei et al. 2011) for reproduction (see Fig. 2 for illustration of the processes involved). Fishes that migrate from juvenile to adult habitats with age/growth are termed as 'ontogenetic shifters' (Adams and Ebersole 2009).

Evidences in the Western Indian Ocean indicate that many coral reef species actually migrate or rather shift habitats as they grow (Dorenbosch et al. 2005; Kimirei et al. 2011; Lugendo et al. 2006). It has also been shown that vegetated shallow water habitats—estuarine or non-estuarine environment—harbour high densities of juveniles of the ontogenetic habitat shifters (Dorenbosch et al. 2005; Kimirei et al. 2011; Lugendo et al. 2006; Lugendo et al. 2005; Mwandya et al. 2010; Mwandya et al. 2009; Berkström et al. 2013)—supporting the nursery function of these habitats. For example, significantly higher relative densities of juveniles (≤10 cm length) of four fish species—*Lethrinus harak*, *Lethrinus lentjan*, *Lutjanus fulviflamma*, and *Siganus sutor*—in shallow-(mangroves and seagrass beds) than in deep-water habitats (mudflats or coral reefs), and an opposite pattern for adults (>15 cm) of the same species have been reported in the Kunduchi and Mbegani areas (see Fig. 3). Similarly, Mwandya et al. (2009) found that >71% of the fish populations sampled in three mangrove creek systems—Mbegani, Kunduchi, and Nunge—were juveniles (Table 3). Both Mwandya et al. (2009) and Kimirei et al. (2011) studies were

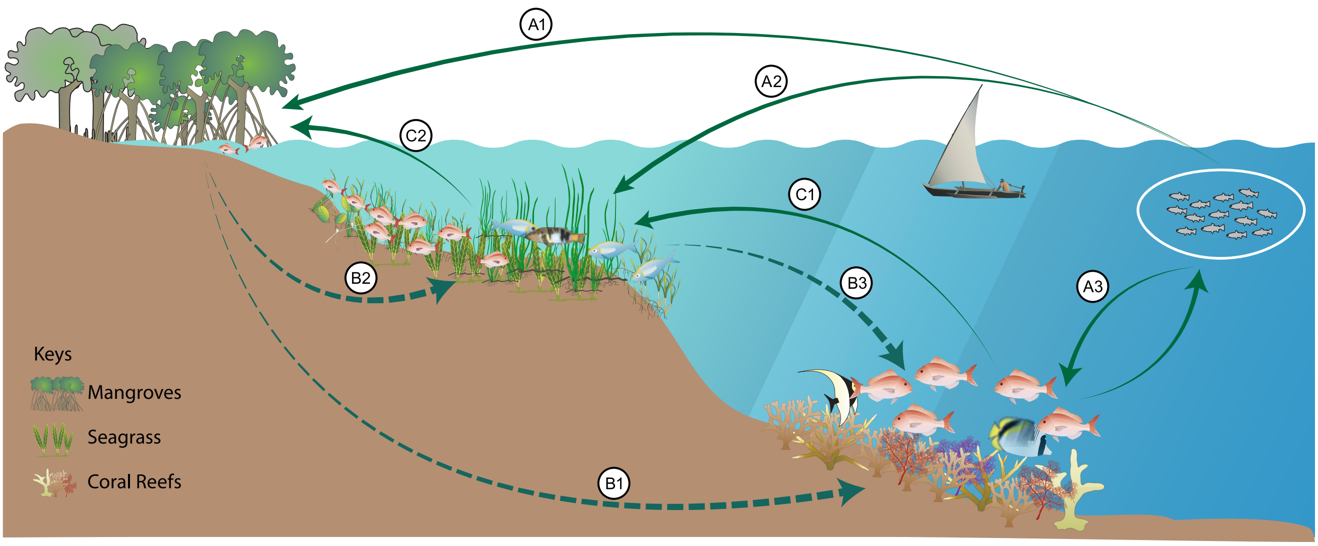

Fig. 2 Illustration of the nursery function of mangroves. A_{1-3} illustrates settlement movements; B_{1-3} illustrates ontogenetic migration of sub-adults to adult habitats for reproduction; and C_{1-2} indicate daily or tidal (C_2) feeding movements among habitats

conducted in non-estuarine mangroves on Mainland Tanzania. Some of these mangroves are under high anthropogenic pressure, especially clear-cutting for salt making and hotel construction (e.g. Kunduchi and Nunge). As a consequence, mangrove habitats with anthropogenic stressors had much reduced fish densities than considerably pristine ones; where clear-cut sites had much lower densities (i.e. <25 ind.100 m^{-1}) than in mangrove-fringed sections (>60 ind.100 m^{-1}) of the same or in relatively undisturbed stands (>50 ind.100 m^{-1}) (Mwandya et al. 2009).

Similar habitat use results have been reported for similar mangrove types in Zanzibar (Dorenbosch et al. 2005; Dorenbosch et al. 2006; Lugendo et al. 2007a; Lugendo et al. 2006; Lugendo et al. 2005; Dorenbosch et al. 2004b) and Mafia Islands (Berkström et al. 2013; Dorenbosch et al. 2006; Kamukuru and Mgaya 2004, 2005). Lugendo et al. (2005) studied habitat use of 13 fish species in the Chwaka Bay, Zanzibar, and found that five (i.e. *Lethrinus lentjan*, *L. variegatus*, *Pelates quadrilineatus*, *Siganus sutor* and *Sphyraena barracuda*) out of the 13 species studied were predominantly present in the turbid mangrove areas as juveniles while their sub-adults were found in adjacent seagrass within the same embayment, and attributed these results to ontogenetic migration. Both net sampling and underwater visual censuses results showed a heterogeneous population size structure where seagrass beds acted as corridors for migrating fishes (Lugendo et al. 2005). In a similar study and within the same mangrove system, Lugendo et al. (2007a) found that >50% of the 150 species found in the Chwaka mangroves were juveniles of commercially important fish species (Table 3); further supporting the nursery function of small estuaries and non-estuarine mangroves.

Although the importance of mangroves as primary sources of carbon to higher trophic levels seems to be limited, stable carbon isotopes from Chwaka Bay revealed that mangrove ecosystems form important feeding grounds for seven fish species in the bay (see Table 4). The findings of this study further revealed that mangrove habitat do not function in isolation; some fish that were caught from nearby habitats such as mud/sand flats and seagrass ecosystems displayed depleted $\delta^{13}C$ values which indicate that they also fed from the mangrove ecosystem (Table 5). These results indicate overlaps in food sources of the different groups of fish, which can be explained by either daily tidal migrations or recent ontogenetic migrations (Dorenbosch et al. 2004b; Lugendo et al. 2006; Unsworth et al. 2007; Unsworth et al. 2008). The relative importance of mangrove ecosystems as feeding habitats is however influenced by mangrove setting whereby mangrove-lined creeks seems to be more important feeding areas than fringing mangroves (Lugendo et al. 2007a).

However, not all shallow water habitats containing mangrove show spatio-temporal consistent functions—at least not for all species (Kimirei et al. 2011; Kimirei et al. 2015). This function also varies between areas with or without nearby mangroves or seagrass beds (Dorenbosch et al. 2004a; Nagelkerken et al. 2001). Mangrove-lined creeks, for example, seem to provide more time for fishes to stay and feed compared to the fringing mangroves, which are accessed only during high tide (Lugendo et al. 2007a). Both large and small estuaries and non-estuarine mangroves may harbor high densities of predators, therefore decimating the values of these coastal habitats to juvenile fishes (Baker and Sheaves 2008, 2009; Dorenbosch et al. 2009). Abundances of certain ‘ontogenetic shifters’—whose

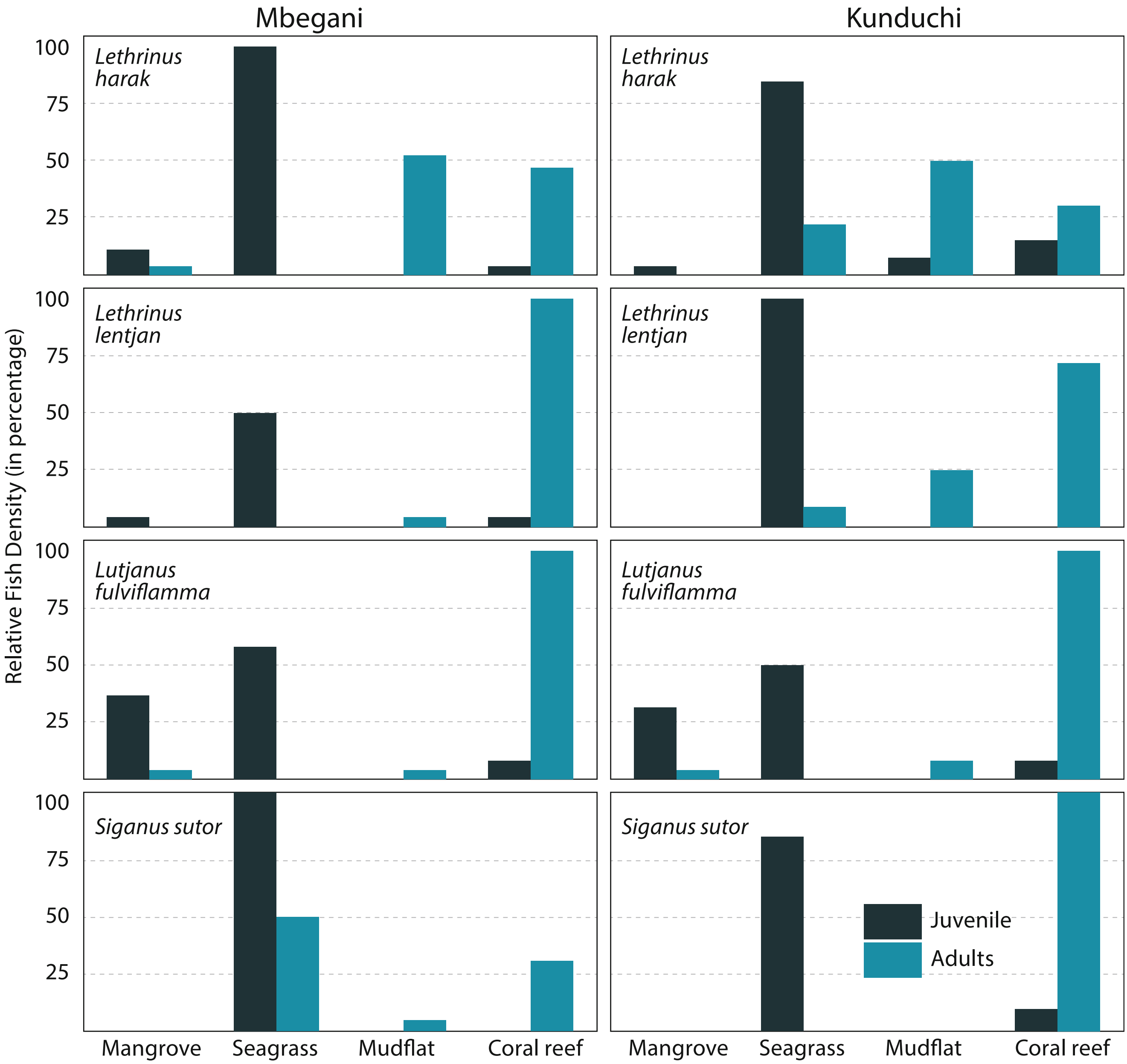

Fig. 3 Differences in relative fish densities of four fish species among coastal habitats at two sampling locations (Mbegani and Kunduchi) from underwater visual censuses (Synthesized from Kimirei et al. 2011)

juvenile life stages depend largely on shallow coastal habitats (Adams and Ebersole 2009)—can be very much reduced in areas with absence of these habitats (Dorenbosch et al. 2004a; Nagelkerken et al. 2001). Also, juveniles and adults of some species show considerable spatio-temporal flexibility in habitat use; which indicate that conservation efforts need to factor in these patterns (Kimirei et al. 2011).

Although fish species clearly depend on estuarine and or non-estuarine habitats during juvenile life stages, this does not imply that all these habitats function equally in their nursery potential and especially in enhancing fisheries in areas where they are found (Faunce and Layman 2009; Gillanders et al. 2003; Huijbers et al. 2013; Kimirei et al. 2011; Nagelkerken and van der Velde 2004). Sometimes these functions are concealed by habitat degradation, and fragmentation for that matter. Although several studies have revealed high juvenile densities at many shallow water sites harboring mangroves or seagrass meadows (Kimirei et al. 2011; Lugendo et al. 2007a; Lugendo et al. 2005; Mwandya et al. 2009), some spatio-temporal variations exist which may lead to different conclusions. For example, Kimirei et al. (2011) found that 94% of *Lethrinus harak* juveniles occupied seagrass beds adjacent to the Mbegani mangroves—using the mangroves at high tide, while in the

Table 3 Percent habitat use of juvenile fishes in non-estuarine and small mangroves and adjacent seagrass

		Juvenile fish density (%)			
References	#Species	Kunduchi	Mbegani	Nunge	Chwaka
Lugendo et al. (2005)	13	–	–	–	83
Lugendo et al. (2007a)	53	–	–	–	76
Mwandya et al. (2009)	63	74	71	75	–
Kimirei et al. (2011)	4	32	41	–	–

Table 4 Carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) stable isotope values of different fish species, used to demonstrate the importance of mangroves as feeding habitats

Species	*N*	$\delta^{13}C$ (mean ± SD)	$\delta^{15}N$ (mean ± SD)
Gerres filamentosus	10	−21.4 ± 0.8	8.3 ± 0.3
Gerres oyena	10	−19.4 ± 1.0	7.5 ± 0.5
Lethrinus lentjan	10	−21.8 ± 0.8	8 ± 0.3
Lutjanus fulviflamma	7	−20.6 ± 1.0	8.5 ± 0.4
Monodactylus argenteus	14	−22.5 ± 1.1	8.1 ± 0.4
Sphyraena barracuda	7	−20.1 ± 1.1	9.6 ± 0.4
Zenarchopterus dispar	18	−22.7 ± 0.3	8.1 ± 0.4

Source: Data synthesized from Lugendo et al. (2006)
N sample size

Kunduchi area, juveniles of this species were mainly found in the seagrass beds of the Bongoyo and Mbudya Islands. The mangroves at Kunduchi are somewhat separated from the nearby seagrass beds. The connection to and therefore the use of these mangroves by fishes during high tide may be limited, which may decimate their function as important nursery habitats. On the contrary, however, the Mbegani mangroves are directly linked to the adjacent seagrass bed—which actually extends into the mangroves—forming a perfect habitat mosaic important for swift habitat connectivity. This is a classic example of how spatial heterogeneity could actually disguise the importance of some habitats.

In a broader perspective, some studies have even concluded that mangroves in the Indo-Pacific and Western Atlantic regions do not enhance reef fisheries due to very few species overlap between the two habitats (Blaber et al. 1985; Laroche et al. 1997; Ley et al. 1999; Thollot and Kulbicki 1988). However, this is only true if the analysis is done on a species assemblage scale (Kimirei et al. 2013). When we view this from an individual species perspective, especially commercial species, a different picture emerges which somewhat contradicts previous assertions. For example, the Kunduchi mangroves, which are not only heavily degraded by encroaching humans in search for construction sites, and have high tidal differences (about 4 m)—which lessens inundation time—but also have a weak connection to nearby seagrass beds due to man-made dykes and sand accretion, have recently been shown to significantly enhance

Table 5 Carbon stable isotope ($\delta^{13}C$) values of fish species from different habitats in Chwaka Bay indicating carbon sources for fishes caught from different habitats

Species	N	$\delta^{13}C$
Gerres filamentosus		
Mangrove creeks	5	−21.2 ± 0.4
Mangrove channel	5	−21.6 ± 0.5
Mud/sand flats	4	−19.2 ± 1.0
Lethrinus lentjan		
Mangrove channel	10	−21.8 ± 0.3
Mud/sand flats	9	−19.3 ± 0.7
Chwaka seagrass beds	10	−12.3 ± 0.2
Marumbi seagrass beds	6	−12.0 ± 0.4
Siganus sutor		
Mud/sand flats	7	−22.8 ± 0.5
Chwaka seagrass beds	5	−19.5 ± 0.7
Marumbi seagrass beds	26	−16.1 ± 0.5

Source: Data synthesized from Lugendo et al. (2006)
Shaded values indicate depleted values indicating utilization of mangrove habitats by fish from respective habitats. *N* sample size

Table 6 Comparison of classification success (%) and juvenile habitat contribution to the adult fish population between the Western Indian Ocean (WIO) and the Caribbean

Species	Region	*N*	Classification success (%)	Juvenile habitat (%) MG	SG	CR	References
L. harak	WIO	25	73	29	53	18	Kimirei et al. (2013)
L. fulviflamma	WIO	21	61	99	1	0	Kimirei et al. (2013)
L. apodus	Caribbean	21	81	99	1	n.c	Mateo et al. (2010)
H. flavolineatum	Caribbean	24	86	63	37	n.c	Mateo et al. (2010)

Classification was based on multi-elemental signatures and $\delta^{13}C$ and $\delta^{18}O$ for Caribbean and $\delta^{13}C$ and $\delta^{18}O$ for the WIO region
n.c not considered as a potential juvenile habitat. *MG* mangrove; *SG* seagrass; *CR* coral reef; *L. apodus* Lutjanus apodus; *H. flavolineatum* Haemulon flavolineatum

populations of some fish species on adjacent coral reefs (Kimirei et al. 2013). While 29% of adult *Lethrinus harak*, caught on reefs in Kunduchi, had spent their juvenile life stage in the Kunduchi mangroves, almost all (99%) *Lutjanus fulviflamma* individuals had passed through mangroves (Kimirei et al. 2013). Kimirei et al. (2013) used otoliths stable carbon and oxygen isotopes to elucidate the contribution of mangroves and seagrass beds in enhancing coral reef fish populations and supporting habitat connectivity and ontogenetic migrations of fishes. These results compare well with studies elsewhere where mangroves are permanently available (Huijbers et al. 2013; Mateo et al. 2010) (see Table 6). *L. harak* and *L. fulviflamma* are highly targeted commercial fish species in coastal Tanzania, and form a bulk of hand-line catches from nearshore reefs. This shows how even small or fragmented mangroves can render important ecosystem services, and as such should not be subjected to indiscriminate clearing for hotel construction or salt-making nor be a subject of waste dumping.

The studies conducted in either small or non-estuarine mangroves in Tanzania, support ontogenetic habitat shifts from shallow to deep-water habitats, the pattern that has been shown to enhance fish population in nearby reef habitats and fisheries in these tropical waters (Dorenbosch et al. 2005; Kimirei et al. 2013). Similar to large estuaries, their presence or absence, and degradation or fragmentation, can make an audibly clear ecological statement. For example, Dorenbosch et al. (2005) found that 32 (>80%) of the 36 fish species recorded along the mangrove and seagrass lined coast of Tanzania were lacking from the coral reefs on the Grande Comoros Islands. They further found that 25 species (~70%) could not be observed or were much less on reefs far from mangroves or seagrass beds in Tanzania; that is in comparison with adjacent reefs with such habitats. These results, therefore, indicate that habitat fragmentation can have strong ecological consequences, such as turning source habitats into sinks (Schreiber and Kelton 2005) and thus call for inclusion of habitat mosaic (Sheaves 2009) which can also include sink habitat patches into conservation efforts.

Cues and Habitat Use by Juvenile Fishes

Studies conducted in Tanzanian shallow coastal waters have shown that small estuarine and non-estuarine mangroves function as nursery habitats for fish and invertebrates (see Sect. 1.2.1). Fish larvae do not just drift blindly and settle by chance; they actively swim and make choices of habitats into which they can settle, maximize their survival and grow to sizes they can compete better in their next habitat. Therefore, there must be important factors or cues guiding fish larvae and juveniles in choosing habitats into which to settle. The question is how mangroves—estuarine (large and small) and non-estuarine alike—support settlement of larvae and juvenile fishes? Are there evidences to actually support settlement of reef fishes into mangroves and other vegetated habitats in Tanzania and the WIO region in general? Studies conducted elsewhere—outside the WIO—and in laboratories hypothesize that early recruits are able to locate suitable microhabitats during settling (Hixon 1991; Jones 1991). For settlement-stage fish, the ability to locate suitable microhabitats is crucial to increase the chances of survival. However, larvae depend on reliable and widely available environmental and chemical cues to orient and navigate to appropriate microhabitats. Small estuarine and non-estuarine mangrove habitats have been proven to play an important role in terms of provision of visual, chemical and sound cues to early recruits just like other mangrove habitats.

For example, structural complexity of the mangrove roots provides an important cue for the recruits to settle into mangrove habitats. Settlers may detect conspecifics or heterospecifics present in the microhabitat when they are in close proximity (Igulu et al. 2011). While conspecifics may give assurance on the quality of the microhabitat, the heterospecifics' presence may provide a clue of safety (Lecchini et al. 2007; Igulu et al. 2011). However, vision is mostly important in environments where water transparency is high (McCormick and Manassa 2008), such as in coral reefs or in non-estuarine mangroves. Small estuarine and non-estuarine mangroves offer this important property.

Moreover, settlers may use olfaction to locate suitable habitats. The use of olfaction cue by fish depends on the direction of water currents, source of juveniles/propagules and the distance from the source of the signal/cue; which means, therefore, that the olfactory signal/cue must travel with water movement. The signals may become weaker as the distance increases from the source of the cue up-current (Atema et al. 2002), probably due to dilution of the cue. In East Africa where tidal amplitude and currents are large, olfaction cues from mangroves can be transported miles away offshore. Evidence suggest that despite the effect of currents and tides on the transportation of olfactory cues and the enormous diversity of cues in the water column, fish are able to detect these natural chemicals at concentrations of parts per billion (Belanger et al. 2006). Recent findings from Kunduchi indicate that early settlers are able to distinguish olfaction cues of mangroves origin from those of seagrass, coral reef, and individuals from the same or different species (Igulu et al. 2013a). This evidence not only points to the importance of the small estuarine and non-estuarine mangroves as nursery grounds for juvenile reef fishes but also as a source of olfaction cues to early settlers. In general, cues from small and/or non-estuarine mangroves can be used in orientation and navigation by early settlers, just like would large estuarine mangroves. It is evident therefore that the presence of mangroves in nearshore environments, where juvenile fishes can settle and grow to sizes capable of competing and surviving their next habitats, is key to perpetuation of fish species and the livelihoods they support.

Energy Flow

Small estuaries and non-estuarine mangroves have received increasing attention from studies aiming at elucidating the underlying mechanisms of ecosystem connectivity (Mwandya et al. 2010; Dorenbosch et al. 2005; Mumby et al. 2004). This is because they form an ideal model system due to the juxtaposition of highly productive ecosystems such as seagrass beds and coral reefs. Although these systems can thrive in isolation, it has been observed that where they occur close to each other, they subsidize productivity in adjacent ecosystems (Igulu et al. 2013b; Bouillon et al. 2008; Kimirei et al. 2013), show higher species richness at their interfaces (Dorenbosch et al. 2005), and show strong ecological linkages through tidal and diurnal migration by decapods and fishes (Sheaves and Molony 2000; Dorenbosch et al. 2004a). The interplay of tidal flow speed, hydrology of the system, the spatial and temporal distribution of predators and prey, the presence of aquatic vegetation, and difference in fish body size and fish species, amongst other things, result in a highly complex degree of connectedness among coastal habitats (Igulu et al. 2013b).

Tidal and diurnal migrations by motile fauna are common in coastal ecosystems, and include entering as well as exiting inshore habitats during different times of the day or at different tidal amplitudes (Boström et al. 2011). Tidal and diurnal inter-habitat migrations in mangroves are highly structured in time and space (Krumme et al. 2004; Verweij and Nagelkerken 2007). They are thought to be driven by changes in feeding opportunities or temporal changes in predation risk related to time of day or tidal stage (Laegdsgaard and Johnson 2001; Verweij et al. 2006; Hammerschlag et al. 2010). While this is happening, there is exchange of organic matter between mangroves and adjacent habitats. Among other mechanisms, the exchange of organic matter can occur through tidal currents or inter habitat feeding by motile fauna. Organic matter transported through tidal currents may vary from one ecosystem to the other depending on local settings. In general, mangrove fringes can be separated into those that are permanently inundated with continuous access to motile animals and those that are only available at high tide (Nagelkerken 2009).

Demonstrating the overall importance of mangroves is a good way to support the conservation and management of mangroves in light of the current degradation rates (Duke et al. 2007; FAO 2007). The importance of mangrove habitat could be greater if the energy produced by photosynthetic organisms (where sunlight energy is converted into organic carbon) in the mangrove habitat either directly or indirectly contributes to species living in other interlinked habitats (Lee 1995). However, there are variations in the utilization of carbon sources among species of macro-invertebrates and fishes from interlinked mangrove and adjacent habitats. In Mtoni estuary, where seagrass beds and coral reef are distances away from the mangroves, contribution of mangroves carbon to adjacent mudflats is little (Kruitwagen et al. 2010) despite large tidal amplitude in the area. Same observation were made in Chwaka (Lugendo et al. 2006) and Kunduchi mangroves (Igulu et al. 2013b). In Kunduchi where a holistic approach to examine carbon flow from individual species to the entire fish community was conducted, findings suggest that there is little dependency to mangroves carbon (Igulu et al. 2013b) (see Table 7).

Table 7 Average percentage mangrove carbon contribution to the diet of fish species collected from mangrove and adjacent habitat

Mangrove forest	N	Contribution (%)	References
Chwaka Bay	9	39 ± 32	Lugendo et al. (2006)
Kunduchi	10	22 ± 26	Igulu et al. (2013b)
Mbegani	17	72 ± 12	Lugendo et al. (2007b)
Mtoni-Kijichi	28	26 ± 15	Kruitwagen et al. (2010)
Nunge	7	29 ± 22	Lugendo et al. (2007b)

Mangrove contribution is based on Isosource Mixing Model (Phillips et al. 2005). *N* number of fish species studied

Furthermore, mangroves carbon contribution to individual species range from 19–30% and 12–45% for short and long term bases respectively, depending on the feeding mode of a that particular species (Igulu et al. 2013b). Overall dependence level on mangrove carbon to the entire fish community was 38% on average (Igulu et al. 2013b). Although this may suggest little direct trophic linkages between mangrove-derived organic carbon and coastal fisheries in the tropics, it is important to note that this function may vary on a spatio-temporal basis; and that degradation and fragmentation may further reduce this figure. This is particularly true for small and non-estuarine mangroves which are less protected, prone to encroachment and perturbation and normally equated to their monetary value than their ecological function. The above findings suggest that the ecological function of the mangrove habitats could be more complex than originally thought (Kimirei et al. 2015); perhaps its function is far more than energy provision, but provision of refugee to juvenile fish and invertebrate species.

Small Estuaries and Non-Estuarine Mangroves as Regulators of Ocean Pollution and Global Warming

Mangrove forests act as natural filters—cleaning wastewaters that pass through them before they enter the oceans. They remove sediments which would otherwise clog other ecosystems—such as seagrass beds and coral reefs. Small estuarine and non-estuarine mangroves in Dar es Salaam for example, have been shown to remove pollutants (nutrients and heavy metals) from the water column (Kruitwagen et al. 2008; Mremi and Machiwa 2003; Mrutu et al. 2013), thereby regulating the amount of pollutants entering the Indian Ocean. For example, Kruitwagen et al. (2008) collected sediment samples at different locations within the Mtoni mangroves and found that the concentration of heavy metal in the sediments decreased rapidly over a short distance downstream, while it increased towards the source of pollution—the Kizinga Stream where the Karibu textile mill is located. Similarly, in the Msimbazi River, Mrutu et al. (2013) found that the concentration of heavy metals decreased with distance out of the mangroves towards the Indian Ocean. The rapid decrease in metal concentrations can be related to the regulating function of mangrove ecosystems. The sediments in mangroves are anaerobic, reduced, and have high abundance of sulphate-reducing bacteria (Lyimo et al. 2002; Kruitwagen et al. 2008); the conditions which facilitate the trapping of heavy metals into the mangrove sediments by forming metal-sulphide complexes as the metal rich water passes through the mangrove sediment (Kruitwagen et al. 2008). It is also, generally accepted that mangrove ecosystems sequester carbon dioxide (CO_2). The CO_2 sequestration regulates the amount available in the atmosphere thereby reducing the effect of global warming. Of the 10.4 billion tons of CO_2 taken out of the atmosphere every year by forests, mangroves sequester about 26 million tons of it annually (Dittmar et al. 2006). This is another important regulating function that all mangroves—regardless of their size and nature—should be credited for; and one that should drive us all to conserve the mangroves, especially now when climate change and global warming are projected to cause more damages to the coastal communities.

Threats to Small Estuaries and Non-Estuarine Mangroves

The small estuarine and non-estuarine mangroves that are found in close proximity to cities and towns are easily accessible by different users and are subjected to a multitude of stressors which if not moderated may cause not only continued loss but extinction of some if not all these important habitats. The fact that these habitats are often small in size makes them even more vulnerable since they can easily be degraded and actually ignored because they can easily be equated to their monetary value than their functional importance. The small estuarine and non-estuarine mangroves of Tanzania are threatened by cutting for various causes and purposes, coastal development and construction, urban and industrial pollution, and climate change, to mention but a few (see Fig. 4).

Cutting of Mangroves

It has been established through interviews that mangroves are cut for a number of causes including firewood (25%), clearing for building sites (19%), construction poles (16%), and charcoal-making (%) (Wagner 2007; Wagner et al. 1999). The Kunduchi mangrove is highly fragmented and studies show that the density of mature trees, saplings and seedlings per m^2 declined from 0.26, 0.68, and 2.76 respectively in 1996 to 0.13 mature trees, 0.27 saplings and 2.62 seedlings per m^2 in 2003 (Wagner 2007). Similar declines were shown at Mbweni where selective cutting of mangroves has led to the disappearance of some species, for example *Rhizophora mucronata* and *Sonneratia alba* (Wagner et al. 2001). At Mtoni mangroves moreover, a drop in mangrove basal area from 23 to 7 $cm.m^{-2}$ and a corresponding fast increase in stamps was noted by Akwilapo (2001). There are no current data on the basal area at Kunduchi and Mbweni mangroves. However, Wagner (2005) shows that the basal area at Kunduchi and Mbweni were 64 and 85 cm2/25 m^2 respectively; while the large estuarine mangroves of Rufiji had a basal area of

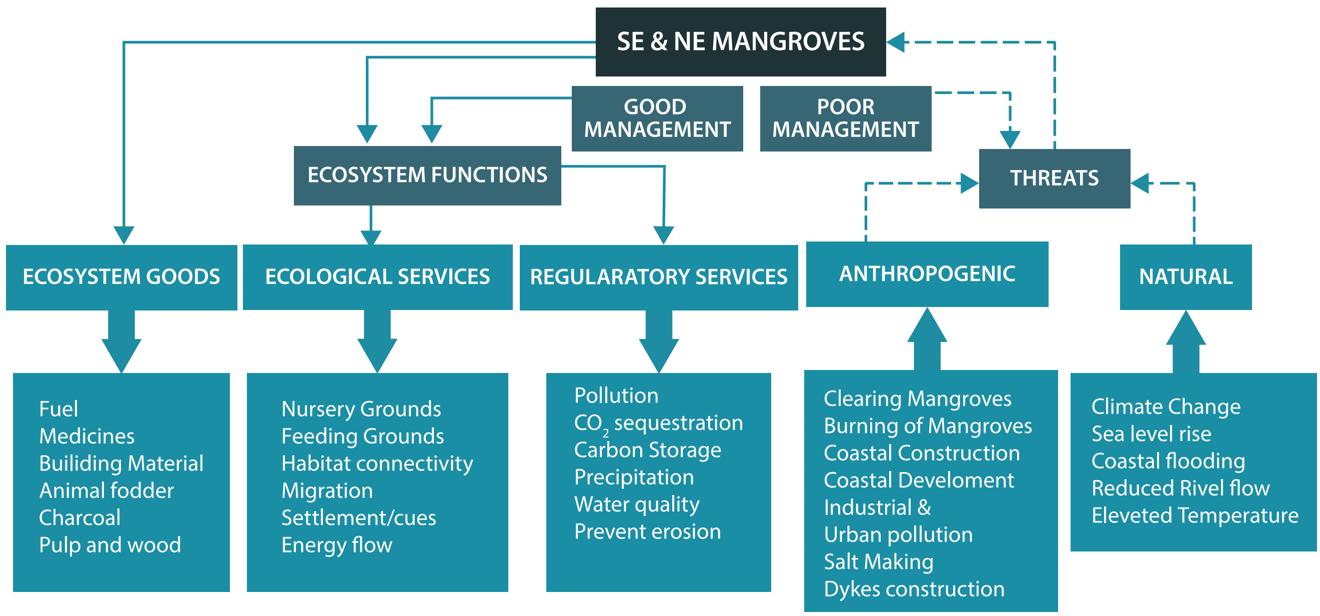

Fig. 4 A schematic representation of the different ecosystem goods and services provided by SE and NE mangroves and anthropogenic and natural threats to the SE and NE mangroves in Tanzania

Table 8 Mangrove area and the rate at which it has changed between 2000 and 2007

Region/District	Site	Mangrove area (ha)			
		2000	2007	Change	Rate (Area/year)
Bagamoyo	Ruvu, Mbegani, Nunge	5051	5709	94	13
Dar es Salaam	Mtoni, Kunduchi	2516	816	−1700	−243

1250 cm^2 in a 25 m^2 mangrove plot and between 631 and 753 $cm^2/25\ m^2$ for large estuarine mangrove of Ruvu in Bagamoyo. As a consequence, the mangrove areas at Mtoni and Kunduchi in Dar es Salaam region declined from 2516 ha to 816 ha between 2000 and 2007. That is to say that in less than a decade, mangrove areas at Mtoni and Kunduchi were about one third of their size in 2000 (Table 8). The commercial city of Dar es Salaam therefore losses about 243 hectares of mangrove every year while only 13 hectares are added annually at the Ruvu, Mbegani and Nunge mangroves combined (Table 8). The distance from Dar es Salaam City was used as an explanation for high basal areas far away from Dar es Salaam indicating that anthropogenic disturbances from the city are important factors affecting the mangroves.

Mangrove cutting and smuggling to Zanzibar for firewood and making of doors is a rampant business in Bagamoyo. As a common practice, mangrove logging is conducted stealthily deep into the mangrove forests to avoid the law enforcers' hands. Other reasons behind the mangrove cutting are clearance for agriculture, salt production and aquaculture (e.g. at Kilongawima and Kunduchi in Dar es Salaam), charcoal production, building poles and digging of the polychaete worms used as fish bait (Semesi et al. 1998). Hotels have also cleared mangroves to make room for more attractive tourist beaches and expand their hotels.

Coastal Construction and Development

The Kunduchi beach is among the most dynamic beaches whose shoreline has changed due to beach erosion (Makota et al. 2004). In order to combat beach erosion, hotel owners have built groynes or dykes along the shore—which trap sands over time—in order to protect their properties from being eroded away or to enlarge their beach areas. The construction of dykes (groynes) poses threats to not only the small non-estuarine mangrove at Kunduchi, but also to the Kunduchi fish landing beach/fishing village—located within the Kunduchi mangrove creek, which may be lost in a near future if dyke construction and more felling of mangroves by hotel owners will not stop. The dykes are diverting the normal watercourses thereby affecting normal functioning of the mangroves. The narrowing of the entrance into the Kunduchi Manyema Creek is a direct result of sand accumulation caused by dykes (see Plate 1).

Plate 1 Showing the level of coastal development between 2007 (*left*) and 2013 (*Right*); notice the increase in sands within the creek and dykes in the 2013 image

Urban and Industrial Pollution

Large cities and urban centers like Dar es Salaam, produce large quantities of both solid and liquid wastes—which are of both domestic and industrial origin (De Wolf and Rashid 2008; Machiwa 1992; Mremi and Machiwa 2003)—on a daily basis; but with an inefficient waste treatment infrastructure (De Wolf et al. 2001). Most dumping sites are located close to the coast, and in most cases, the wastes are dumped directly into rivers that drain through mangroves (Mrutu et al. 2013; Bwathondi et al. 1991; Machiwa 1992; TCMP 2001). The use of mangrove areas as dumping sites for both solid and liquid waste may have dramatic effects especially to small estuarine mangroves due to low dilution factor as a result of low water volume in them. The discharge of liquid waste in Mtoni estuarine mangroves from the nearby industries, for example, has resulted into dramatic changes in physico-chemical as well as biotic component of the habitat. For example, Kruitwagen et al. (2006) found that growth and morphology of barred mudskippers (*Periophthalmus argentilineatus*) from the Mtoni estuary were severely affected by urban and industrial pollution. Fishes from the Mtoni mangroves were small in size, had left eye deformed, and highly enriched tissue $\delta^{15}N$ ratios ($\delta^{15}N < 12‰$) (Kruitwagen et al. 2006) and $\delta^{15}N = 15.2 \pm 0.2‰$ for *Apogon* sp. (Lugendo et al. 2007b)—indicative of anthropogenic pollution from the Karibu Textile mill which pours its effluents directly into the estuary (Kruitwagen et al. 2008). Similarly, the Kunduchi mangrove creek is showing increasing nutrients pollution. While the $\delta^{15}N$ values ranged between 8 and 9‰ in 2005 (Kruitwagen et al. 2006), nitrogen enrichment in the mangroves of Kunduchi had increased by a magnitude to $\delta^{15}N$ values of 10‰ in 2010 (Lugendo et al. unpublished data).

Heavy metal studies also show that polluted mangroves have high metal concentrations, which are accumulating in soft tissues of organisms (De Wolf and Rashid 2008; De Wolf et al. 2001; Mremi and Machiwa 2003; Mshana and Sekadende 2014). Pollutants from the Dar es Salaam City are discharged directly into the ocean thereby affecting the littoral communities, especially organisms in mangroves where most of the pollutants are dumped (Machiwa 1992; Mremi and Machiwa 2003). For example, Periwinkles sampled in polluted mangroves in Dar es Salaam were found to have high tissue metal concentrations and were smaller in size and weighed less than their conspecifics in unpolluted mangroves (De Wolf and Rashid 2008). Both mangrove roots and crabs from Mtoni and Msimbazi mangroves have high metal contamination than elsewhere (Mremi and Machiwa 2003). It has also been shown that sediments in mangroves away from the Dar es Salaam City have significantly lower metal contamination than those within (Machiwa 1992; Mremi and Machiwa 2003). Mangroves are known to buffer metal and nutrients pollution; but as more pollutants are pumped into them and clearing and fragmentation are accelerated, their buffering capacity may reach a breaking point—where any addition pollutant input may not be buffered anymore. Furthermore, as coastal cities and their populations grow; the needs for human habitats, tourism and mangrove clearance increase, the mangrove ecosystems and their services will consequently diminish opening up for more coastal erosion, pollution and health hazards, if such mangroves are not given a special consideration in conservation efforts.

Climate Change Impacts on Mangrove Ecosystems

Knowledge of the direct impacts of climate change on mangroves in Tanzania is limited (Pethick and Spencer 1990). However, climate change related factors such as sea level rise, excessive flooding and changes in hydrological regime have been reported to affect both fringing and estuarine mangroves in Tanzania (Diop et al. 2002; Erftemeuer and Hamerlynck 2005). Analysis of a series of 1999 aerial photographs from the Rufiji Delta indicated a loss of about 117 ha of mangrove forest due to massive tree mortality following El Nino floods (Erftemeuer and Hamerlynck 2005). Experimental studies shows that mangroves cannot tolerate prolonged inundation of saline waters (Mangora and Shalli 2012, 2014)—indicating that sites that are extensively over-washed by tidal highs are likely to face poor regeneration, possibly due to significant decline in photosynthetic rates (Mangora and Shalli 2014). It is anticipated that climate change related factors which result into prolonged inundation of mangroves (e.g. through sea-level rise or flooding from heavy precipitation) may have dramatic effects on mangrove ecosystems, adding to the already existing threats to mangrove forests arising from overexploitation and coastal development. Climate change mediated precipitation on-land may come with excessive sediment into the mangroves as a result of land-ward erosion. The sediment flooding into the mangroves are normally transported out of the system by periodic tidal flushing (Kruitwagen et al. 2008). If the sediments were to remain in the mangroves, probably as a result of coastal works to prevent beach erosion, the mangroves would have to suffer from clogging and would gradually drown as the sea level rises. Of course the level of inundation and sediment accretion will depend on the rate at which the sea level will rise as a result of climate and non-climate change factors. Another scenario would be if rise in global temperatures result into droughts which may cause reduced river flows into the SE and NE and large mangrove ecosystems. It follows therefore that the mangrove wetlands might lose both their monetary and ecosystem values (Kebede et al. 2010). Apart from contributing over 10% of the global organic carbon to the world's oceans, mangroves also act as carbon sinks, sequestering about 26 million tons of carbon annually (Dittmar et al. 2006). Nonetheless, a sensible action towards protection of the mangrove ecosystem—whether in estuarine or non-estuarine environment—and the services they provide will be to let the natural system take its own course.

Management Issues in Small Estuaries and Non-Estuarine Mangroves

Protection of small estuarine and non-estuarine mangrove habitats in Tanzania is governed by similar management regime as that for large estuarine mangroves, and is provided hereafter. The background information on the management of mangroves in Tanzania is detailed in de Lacerda (2002) and Mangora et al. (unpublished data). Mangrove forests in Tanzania have been declared forest reserves since the colonial era; however, until now there is no legislation specific to mangrove management. Instead, management of mangroves in Tanzania is included in the management of terrestrial forests, and often more than one legislation and/or institutions are involved.

In Tanzania mainland, the forest sector is managed by the Ministry of Natural Resources and Tourism under the Tanzania Forest Service (TFS)—an agency that oversees the practical management of forest resources, and the Forestry and Beekeeping Division (FBD), which oversees the policy and legislative formulations for the forest sector. The primary legal frameworks that govern forest sector include the National Forestry Policy of 1998 and Forest Act of 2002. Other accompanying legislation includes the National Environmental Policy of 1997 and the Environmental Management Act of 2004, National Land Policy of 1997 and the Land Act of 1999, Village Land Act of 1999 and Land Use Planning Act of 2007, National Beekeeping Policy of 1998 and Beekeeping Act of 2002. In Zanzibar, the Department of Forest and Non-Renewable Natural Resources (DFNRNR) under the Ministry of Agriculture and Natural Resources manage the forest sector. The principal legal framework that governs the forest sector is the National Forest Policy for Zanzibar of 1999, and the Forest Resources Management and Conservation Act of 1996. Other associated legislation includes the National Environmental Policy for Zanzibar of 1992 among others. Moreover, several other institutions including those responsible for fisheries, wildlife, agriculture, ports, lands and minerals are involved in the management of mangroves. Down the ladder, the local governments (District and Village/Shehia Councils) have stake in local area forest resource management (Mangora et al. unpublished).

Effective conservation of mangroves is guided by the management plan. Tanzania mainland developed her mangrove management plan in 1991 (Semesi 1991). This plan has not been adequately implemented due to inadequate financial and human resources (Mangora 2011, Mangora et al. unpublished). Zanzibar developed her mangrove management plan between 2008 and 2009. Furthermore, there are a number of other initiatives that compliment mangrove management in the country. Examples of such include inclusion of mangrove forests as part of Marine Protected Areas (MPA), and the designation of some areas containing mangroves as Ramsar Sites e.g. Rufiji-Mafia-Kilwa. Community participation is another very important component in the management of mangroves in Tanzania. NGOs and CBOs in the country are active and most take part in the management of mangroves, including mangrove restoration initiatives, which are among the common management initiatives in the country.

The Missing Link

It is evident that legislation for forest and hence mangrove management between the Tanzania mainland and Zanzibar are different, which creates a slim chance for collaborative measures especially when it comes to enforcement of legislation in the two sides of the country. The fact that the management of forests and consequently that of mangroves are guided by more than one legislation and/or institution also poses an extra dimension to the management challenges. More often it tends to create institutional conflicts and jurisdictional contradictions across the different state departments (Mangora et al. unpublished). The current management plan for Tanzania mainland is obsolete and needs revision to incorporate new and emerging institutional and socio-economic changes and challenges (Mangora et al. unpublished).

In view of the above, therefore, we contest that the mangrove management plan be revised, if possible for all mangrove blocks of the country and not in piecemeal. Another important step would be declaration or inclusion of all mangroves, which are within or near urban areas, regardless of their nature or size into marine reserves. This is because these reserves can actually protect marine populations; protect species and ecosystems in places threatened by human disturbances, as long as laws and regulations are enforced to the letter.

Summary and Future Direction

The provision of ecosystem goods and services by small estuaries (SE) and non-estuarine (NE) mangroves in Tanzania cannot be ignored. The SE and NE provide a supporting function by serving as nursery habitats for many fish and invertebrate species. The nursery function of these habitats has been documented in Tanzania, based on juvenile fish densities—using both underwater visual census and net sampling (Dorenbosch et al. 2005; Kimirei et al. 2011; Lugendo et al. 2005; Mwandya et al. 2009); and conclusive evidence about this function has been provided using otoliths carbon and oxygen stable isotopes (Kimirei et al. 2013). However, disturbances in these mangroves significantly decimates fish abundance and diversity (Mwandya et al. 2009). Also experimental studies show that these habitats play an important role in guiding fish larvae and juveniles in choosing habitats into which to settle (Igulu et al. 2011; Igulu et al. 2013a). An important fact in light of this chapter is that most of these studies were conducted in small estuaries (Mtoni Kijichi) and/or non-estuarine mangrove (Kunduchi, Mbegani, Nunge, and Chwaka), both on mainland Tanzania and Zanzibar. It follows therefore that the habitats in question are as important as would be for large estuaries; and the results indicate—unequivocally—therefore that small estuarine and non-estuarine mangroves are important supporters of coastal livelihoods—through the nursery function and ontogenetic migration of fishes—in Tanzania.

Nevertheless, the SE and NE mangroves are facing threats from cutting of mangroves, coastal construction and development which is partly due to urbanization and tourism, pollution and climate change effects. Despite that these mangroves perform a regulating function by immobilizing most pollutants (Kruitwagen et al. 2008; Mrutu et al. 2013), and sequestering CO_2; they are cleared and or their functions obstructed by our actions (see Fig. 4). All mangrove forests are reserved in Tanzania (FAO 2007; Semesi 1991); however, they are among the most poorly managed coastal resources in Tanzania, despite the existence of several management instruments (Mangora 2011). While good management of these important habitats means continued provision of the different goods and services to the coastal communities, poor management jeopardizes their ecological functions and ecosystem goods and services (Fig. 4). Proper management of these systems will have to interface with proper and better land-use practices and planning upstream.

It is recommended that the existing management plan should be strictly enforced and that offenders should be reprimanded accordingly; that the management plan should be revised and updated for all mangrove blocks in Tanzania; that special attention in enforcing laws be given to mangrove stands found in or near urban areas or be included in the existing or new marine reserves; that restoration efforts involving the respective surrounding communities should be either started or intensified. There are success stories of restoration efforts which involved communities (e.g. Mbweni, Kunduchi and Tanga (TCMP 2001; Wagner et al. 2001)—they can be used as best practices or models for more restoration and protection works; that clearing for hotel construction and dumping of wastes should be stopped; and coastal construction (e.g. groynes) and development (e.g. hotel construction) should allow the mangroves function to take their natural course by observing existing laws and regulations–such as observance of the 60 m buffer zone as stipulated in Section 57(1) of the National Environmental Management Act of 2004. Moreover, the consumptive human use of stream and river waters (e.g. potable water supply for domestic use, livestock, and irrigation) that can significantly reduce freshwater flows into the mangrove ecosystems should be regulated; otherwise they can alter both the hydrological and ecological connectivity in these habitats. There is also a need to quantify the actual amount of CO_2 that is being sequestered by the SE and NE mangrove in Tanzania and compare it with data from larger estuarine mangroves. Also, a study to elucidate the contribution of larger mangrove ecosystems in Tanzania to coral reef fisheries will need to be conducted in order to attribute the right share of SE and NE to fisheries and conservation.

Overlooking the ecosystem values that mangroves provide, regardless of their size and nature, whether small and/or non-estuarine, jeopardizes conservation outcomes. As the human population continues to swell, demand for ecosystem goods and services accrued from mangroves increases, and conservation needs continue to manifest, we need even small fragments of coastal habitats to remain intact. It is by doing so and actually conserving a mosaic of these and other habitats that restoration of degraded mangroves and the services they provide can be accomplished.

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References

Adams AJ, Ebersole JP (2009) Mechanisms affecting recruitment patterns of fish and decapods in tropical coastal ecosystems. In: Nagelkerken I (ed) Ecological connectivity among tropical coastal ecosystems. Springer, Dordrecht, pp 185–228

Akwilapo FD (2001) The distribution and abundance of mangrove species and associated macrobenthos in ecosystems with varied anthropogenic degradation. M.Sc. thesis, University of Dar es Salaam. 111p,

Alongi DM (2002) Present state and future of the world's mangrove forests. Environ Conserv 29(3):331–349

Atema J, Kingsford MJ, Gerlach G (2002) Larval reef fish could use odour for detection, retention and orientation to reefs. Mar Ecol Prog Ser 241:151–160

Baker R, Sheaves M (2008) Refugees or ravenous predators: detecting predation on new recruits to tropical estuarine nurseries. Wetl Ecol Manage 17(4):317–330

Baker R, Sheaves M (2009) Overlooked small and juvenile piscivores dominate shallow-water estuarine "refuges" in tropical Australia. Estuar Coast Shelf Sci 85(4):618–626

Belanger RM, Corkum LD, Li WM, Zielinski BS (2006) Olfactory sensory input increases gill ventilation in male round gobies (*Neogobius melanostomus*) during exposure to steroids. Comp Biochem Phys A 144(2):196–202

Berkström C, Jörgensen TL, Hellström M (2013) Ecological connectivity and niche differentiation between two closely related fish species in the mangrove-seagrass-coral reef continuum. Mar Ecol Prog Ser 477:201–215

Blaber SJM (2009) Relationships between tropical coastal habitats and (offshore) fisheries. In: Nagelkerken I (ed) Ecological connectivity among tropical coastal ecosystems. Springer, Dordrecht, pp 533–564

Blaber SJM, Young JW, Dunning MC (1985) Community structure and zoogeographic affinities of the coastal fishes of the Dampier region of north-western Australia. Aust J Mar Freshw Res 36:247–266

Boström C, Pittman SJ, Simenstad C, Kneib RT (2011) Seascape ecology of coastal biogenic habitats: advances, gaps, and challenges. Mar Ecol Prog Ser 427:191–217

Bouillon S, Borges AV, Castañeda-Moya E, Diele K, Dittmar T, Duke NC, Kristensen E, Lee SY, Marchand C, Middelburg JJ, Rivera-Monroy VH, Smith TJ, Twilley RR (2008) Mangrove production and carbon sinks: A revision of global budget estimates. Global Biogeochemical Cycles 22 (2):1–12 GB2013, doi:10.1029/2007gb003052

Bwathondi POJ, Nkotagu HH, Mkuula S (1991) Pollution of the Msimbazi valley. Report commissioned by National Environmental Management Council (NEMC) Dar es Salaam, Tanzania

Costanza R, d'Arge R, de Groot R, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'Neill RV, Paruelo J, Raskin RG, Sutton P, van den Belt M (1997) The value of the world's ecosystem services and natural capital. Nature 387:253–260

de Lacerda DL (2002) Mangrove ecosystems: function and management. Springer, New York

De Wolf H, Rashid R (2008) Heavy metal accumulation in Littoraria scabra along polluted and pristine mangrove areas of Tanzania. Environ Pollut 152(3):636–643

De Wolf H, Ulomi SA, Backeljau T, Pratap HB, Blust R (2001) Heavy metal levels in the sediments of four Dar es Salaam mangroves: Accumulation in, and effect on the morpology of the periwinkle, *Littoraria scabra* (Mullusca: Gastropoda). Environ Int 26:243–249

Diop ES, Goldon C, Semesi AK AS, Diallo A, Guisse A, Diouf M, Ayivor JS (2002) Mangroves of Africa. In: de Lacerda DL (ed) Mangrove ecosystems: function and management. Springer, Berlin, pp 63–121

Dittmar T, Hertkorn N, Kattner G, Lara RJ (2006) Mangroves, a major source of dissolved organic carbon to the oceans. Global Biogeochem Cycles 20(1):GB1012. doi:10.1029/2005GB002570

Dorenbosch M, van Riel MC, Nagelkerken I, van der Velde G (2004a) The relationship of reef fish densities to the proximity of mangrove and seagrass nurseries. Estuar Coast Shelf Sci 60(1):37–48

Dorenbosch M, Verweij MC, Nagelkerken I, Jiddawi NS (2004b) Homing and daytime tidal movements of juvenile snappers (Lutjanidae) between shallow-water nursery habitats in Zanzibar, western Indian Ocean. Environ Biol Fishes 70:203–209

Dorenbosch M, Grol MGG, Christianen MJA, Nagelkerken I, van der Velde G (2005) Indo-Pacific seagrass beds and mangroves contribute to fish density and diversity on adjacent coral reefs. Mar Ecol Prog Ser 302:63–76

Dorenbosch M, Grol MGG, Nagelkerken I, van der Velde G (2006) Seagrass beds and mangroves as potential nurseries for the threatened Indo-Pacific humphead wrasse, *Cheilinus undulatus* and Caribbean rainbow parrotfish, *Scarus guacamaia*. Biol Conserv 129(2):277–282

Dorenbosch M, Grol MGG, de Groene A, van der Velde G, Nagelkerken I (2009) Piscivore assemblages and predation pressure affect relative safety of some back-reef habitats for juvenile fish in a Caribbean bay. Mar Ecol Prog Ser 379:181–196

Duke N (2006) Australia's mangroves. The authoritative guide to australia's mangrove plants. University of Queensland, St. Lucia, 200 pp. ISBN: 9780646461960

Duke NC, Meynecke JO, Dittmann S, Ellison AM, Anger K, Berger U, Cannicci S, Diele K, Ewel KC, Field CD, Koedam N, Lee SY, Marchand C, Nordhaus I, Dahdouh-Guebas F (2007) A world without mangroves. Science 317(5834):41–42

Erftemeuer PLA, Hamerlynck O (2005) Die-back of the mangrove Heritiera littoralis dryand, in the Rufiji Delta (Tanzania) following El Nino floods. J Coal Res SI(42) (42):228–235.

FAO (2007) The world's mangroves 1980–2005. A thematic study prepared in the framework of the Global forest resources assessment. FAO, Rome. 78 pp

Faunce CH, Layman CA (2009) Sources of variation that affect perceived nursery function of mangroves. In: Nagelkerken I (ed) Ecological connectivity among tropical coastal ecosystems. Springer, Dordrecht, pp 401–424

Gillanders BM, Able KW, Brown JA, Eggleston DB, Sheridan PF (2003) Evidence of connectivity between juvenile and adult habitats for mobile marine fauna: an important component of nurseries. Mar Ecol Prog Ser 247:281–295

Gladstone W (2009) Conservation and management of tropical coastal ecosystems. In: Nagelkerken I (ed) Ecological connectivity among tropical coastal ecosystems. Springer, Dordrecht, pp 565–605

Griffith AL (1949) Reconnaissance report on the forest problems of the Zanzibar Protectorate. The Government Printer, Zanzibar, 50 pp

Griffith AL (1950) Working scheme for the mangroves of the Zanzibar Protectorate. The Government Printer, Zanzibar, 42 pp

Hammerschlag N, Morgan AB, Serafy JE (2010) Relative predation risk for fishes along a subtropical mangrove-seagrass ecotone. Mar Ecol Prog Ser 401:259–267

Hixon MA (1991) Predation as a process structuring coral reef fish communities. In: Sale PF (ed) The ecology of fishes on coral reef. Academic, San Diego, pp 475–508

Huijbers CM, Nagelkerken I, Debrot AO, Jongejans E (2013) Geographic coupling of juvenile and adult habitat shapes spatial population dynamics of a coral reef fish. Ecology 94(8):1859–1870

Igulu MM, Nagelkerken I, Fraaije R, van Hintum R, Ligtenberg H, Mgaya YD (2011) The potential role of visual cues for microhabitat selection during the early life phase of a coral reef fish (Lutjanus fulviflamma). J Exp Mar Biol Ecol 401(1–2):118–125

Igulu MM, Nagelkerken I, van der Beek M, Schippers M, van Eck R, Mgaya YD (2013a) Orientation from open water to settlement habitats by coral reef fish: behavioral flexibility in the use of multiple reliable cues. Mar Ecol Prog Ser 493:243–257

Igulu MM, Nagelkerken I, van der Velde G, Mgaya YD (2013b) Mangrove Fish production Is largely fuelled by external food sources: a stable isotope analysis of fishes at the individual, species, and community levels from across the globe. Ecosystems 16(7):1336–1352

Jones GP (1991) Postrecruitment processes in the ecology of coral reef fish populations: a multifactorial perspective. In: Sale PF (ed) The ecology of fishes on coral reefs. Academic, San Diego, pp 221–238

Kamukuru AT, Mgaya YD (2004) The food and feeding habits of blackspot snapper, Lutjanus fulviflamma (Pisces: Lutjanidae) in shallow waters of Mafia Island, Tanzania. Afr J Ecol 42:49–58

Kamukuru AT, Mgaya YD (2005) Effects of exploitation on reproductive capacity of black-spot snapper, *Lutjanus fulviflamma* (Pisces: Lutjanidae) in Mafia Island, Tanzania. Afr J Ecol 42:270–280

Kanai T, Nanjo K, Yamane K, Amano Y, Kohno H, Watanabe Y, Sano M (2014) Utilization patterns of estuarine and marine habitats by the halfbeak Zenarchopterus dunckeri at Iriomote Island, southern Japan, evaluated from otolith microchemistry. Fish Sci 80 (6):1231–1239

Kebede AS, Brown S, Nicholls RJ (2010) Synthesis report: The implications of climate change and sea-level rise in Tanzania – The coastal zones. University of Southampton, Southampton, UK

Kimirei IA, Nagelkerken I, Griffioen B, Wagner C, Mgaya YD (2011) Ontogenetic habitat use by mangrove/seagrass-associated coral reef fishes shows flexibility in time and space. Estuar Coast Shelf Sci 92 (1):47–58

Kimirei IA, Nagelkerken I, Mgaya YD, Huijbers CM (2013) The mangrove nursery paradigm revisited: otolith stable isotopes support nursery-to-reef movements by Indo-Pacific fishes. PLoS One 8 (6):e66320

Kimirei IA, Nagelkerken I, Slooter N, Gonzalez ET, Huijbers CM, Mgaya YD, Rypel AL (2015) Demography of fish populations reveals new challenges in appraising juvenile habitat values. Mar Ecol Prog Ser 518:225–237

Kruitwagen G, Hecht T, Pratap HB, Wendelaar Bonga SE (2006) Changes in morphology and growth of the mudskipper (*Periophthalmus argentilineatus*) associated with coastal pollution. Mar Biol 149(2):201–211

Kruitwagen G, Pratap HB, Covaci A, Wendelaar Bonga SE (2008) Status of pollution in mangrove ecosystems along the coast of Tanzania. Mar Pollut Bull 56(5):1022–1031

Kruitwagen G, Nagelkerken I, Lugendo BR, Mgaya YD, Bonga SEW (2010) Importance of different carbon sources for macroinvertebrates and fishes of an interlinked mangrove–mudflat ecosystem (Tanzania). Estuar Coast Shelf Sci 88:464–472

Krumme U, Saint-Paul U, Rosenthal H (2004) Tidal and diel changes in the structure of a nekton assemblage in small intertidal mangrove creeks in northern Brazil. Aquat Living Resour 17(02):215–229

Laegdsgaard P, Johnson C (2001) Why do juvenile fish utilise mangrove habitats? J Exp Mar Biol Ecol 257:229–253

Laroche J, Baran E, Rasoanandrasana NB (1997) Temporal patterns in a fish assemblage of a semiarid mangrove zone in Madagascar. J Fish Biol 51:3–20

Lecchini D, Planes S, Galzin R (2007) The influence of habitat characteristics and conspecifics on attraction and survival of coral reef fish juveniles. J Exp Mar Biol Ecol 341(1):85–90

Lee SY (1995) Mangrove outwelling—a review. Hydrobiologia 295 (1):203–212

Ley JA, Mclvor CC, Montague CL (1999) Fishes in mangrove proproot habitats of northeastern Florida Bay: distinct assemblages across an estuarine gradient. Estuar Coast Shelf Sci 48:701–723

Lugendo BR, Pronker A, Cornelissen I, de Groene A, Nagelkerken I, Dorenbosch M, van der Velde G, Mgaya YD (2005) Habitat utilisation by juveniles of commercially important fish species in a marine embayment in Zanzibar, Tanzania. Aquat Living Res 18 (2):149–158

Lugendo BR, Nagelkerken I, van der Velde G, Mgaya YD (2006) The importance of mangroves, mud and sand flats, and seagrass beds as feeding areas for juvenile fishes in Chwaka Bay, Zanzibar: gut content and stable isotope analyses. J Fish Biol 69(6):1639–1661

Lugendo BR, Nagelkerken I, Jiddawi NS, Mgaya YD, van der Velde G (2007a) Fish community composition of a tropical nonestuarine embayment in Zanzibar, Tanzania. Fish Sci 73:1213–1223

Lugendo BR, Nagelkerken I, Kruitwagen G, van der Velde G, Mgaya YD (2007b) Relative importance of mangrove as feeding habitats for fishes: a comparison between mangrove habitats with different settings. Bull Mar Sci 80(3):497–512

Lyimo JT, Pol A, Op den Camp HJM (2002) Sulfate reduction and methanogenesis in sediments of Mtoni Mangrove Forest, Tanzania. Ambio 31(7–8):614–616

Machiwa JF (1992) Heavy metal content in coastal sediments off Dar Es Salaam, Tanzania. Environ Int 18:409–415

Makota V, Sallema R, Mahika C (2004) Monitoring shoreline change using remote sensing and GIS: a case study of Kunduchi Area, Tanzania. Western Indian Ocean J Mar Sci 3(1):1–10

Mangora MM (2011) Poverty and institutional management stand-off: a restoration and conservation dilemma for mangrove forests of Tanzania. Wetl Ecol Manage 19(6):533–543

Mangora MM, Shalli MS (2012) Socio-economic profiles of communities adjacent to Tanga marine reserve systems, Tanzania: key ingredients to general management planning. Curr Res J Soc Sci 4(2):141–149

Mangora MM, Shalli MS (2014) Sacred mangrove forests: who bears the pride? In: Behnassi M, Shahid SA, Mintz-Habib N (eds) Science, policy and politics of modern agricultural system. Springer, Dordrecht, pp 291–305

Mateo I, Durbin EG, Appeldoorn RS, Adams AJ, Juanes F, Kingsley R, Swart P, Durant D (2010) Role of mangroves as nurseries for French grunt *Haemulon flavolineatum* and schoolmaster *Lutjanus apodus* assessed by otolith elemental fingerprints. Mar Ecol Prog Ser 402:197–212

McCormick MI, Manassa R (2008) Predation risk assessment by olfactory and visual cues in a coral reef fish. Coral Reefs 27 (1):105–113

Mremi SD, Machiwa JF (2003) Heavy metal contamination of mangrove sediments and the associated biota in Dar es Salaam, Tanzania. Tanzan J Sci 29(1):61–76

Mrutu A, Nkotagu HH, Luhilo GB (2013) Spatial distribution of heavy metals in Msimbazi River mangrove sediments in Dar es Salaam coastal zone, Tanzania. Int J Environ Sci 3(5): 1641–1655

Mshana G, Sekadende BC (2014) Assessment of heavy metal pollution in *Octopus cyanea* in the coastal waters of Tanzania. J Health Pollut 4(6):10–17

MTNRE (1991) Management plan for the mangrove ecosystem of mainland Tanzania, vols 1–10. Ministry of Tourism, Natural Resources and Environment, Forest and Beekeeping Division, Catchment Forest Project, Dar es Salaam

Mumby PJ, Edwards AJ, Arias-Gonzalez JE, Lindeman KC, Blackwell PG, Gall A, Gorczynska MI, Harborne AR, Pescod CL, Renken H, Wabnitz CCC, Llewellyn G (2004) Mangroves enhance the biomass of coral reef fish communities in the Caribbean. Nature 427(6974):533–536

Mwandya AW, Gullström M, Öhman MC, Andersson MH, Mgaya YD (2009) Fish assemblages in Tanzanian mangrove creek systems influenced by solar salt farm constructions. Estuar Coast Shelf Sci 82(2):193–200

Mwandya AW, Gullström M, Andersson MH, Öhman MC, Mgaya YD, Bryceson I (2010) Spatial and seasonal variations of fish assemblages in mangrove creek systems in Zanzibar (Tanzania). Estuar Coast Shelf Sci 89(4):277–286

Nagelkerken I (2009) Evaluation of nursery function of mangroves and seagrass beds for tropical decapods and reef fishes: patterns and underlying mechanisms. In: Nagelkerken I (ed) Ecological connectivity among tropical coastal ecosystems. Springer, Dordrecht, pp 357–400

Nagelkerken I, van der Velde G (2004) Are Caribbean mangroves important feeding grounds for juvenile reef fish from adjacent seagrass beds? Mar Ecol Prog Ser 274:143–151

Nagelkerken I, Kleijnen S, Klop T, van den Brand RACJ, Cocheret de la Moriniére E, van der Velde G (2001) Dependence of Caribbean reef fishes on mangroves and seagrass beds as nursery habitats: a comparison of fish faunas between bays with and without mangroves/seagrass beds. Mar Ecol Prog Ser 214:225–235

Nagelkerken I, Blaber SJM, Bouillon S, Green P, Haywood M, Kirton LG, Meynecke JO, Pawlik J, Penrose HM, Sasekumar A, Somerfield PJ (2008) The habitat function of mangroves for terrestrial and marine fauna: a review. Aquat Bot 89(2):155–185

Ngoile MAK, Shunula JP (1992) Ngoile and Shunula 1992 Status and exploitation of the mangrove and associated fishery resources in Zanzibar. Hydrobiologia 247:229–234

Pethick J, Spencer T (1990) Mangrove response to sea level rise: the Rufiji delta, Tanzania. Interim report. Frontier Tanzania. The Society for Environmental Exploration. London, UK, 8 p

Phillips DL, Newsome SD, Gregg JW (2005) Combining sources in stable isotope mixing models: alternative methods. Oecologia 144(4):520–527

Polidoro BA, Carpenter KE, Collins L, Duke NC, Ellison AM, Ellison JC, Farnsworth EJ, Fernando ES, Kathiresan K, Koedam NE, Livingstone SR, Miyagi T, Moore GE, Ngoc Nam V, Ong JE, Primavera JH, Salmo SG, Sanciangco JC, Sukardjo S, Wang Y, Yong JW (2010) The loss of species: mangrove extinction risk and geographic areas of global concern. PLoS One 5(4):e10095

Rönnbäck P (1999) The ecological basis for economic value of seafood production supported by mangrove ecosystems. Ecol Econ 29:235–252

Schreiber SJ, Kelton M (2005) Sink habitats can alter ecological outcomes for competing species. J Anim Ecol 74(6):995–1004

Semesi AK (1991) Management plan for the mangrove ecosystem of mainland Tanzania, vol 1–10. Ministry of Tourism, Natural Resources and Environment, Forest and Bee-keeping Division. Dar es Salaam

Semesi AK (1992) Developing management plans for the mangrove forest reserves of mainland Tanzania. Hydrobiologia 247:1–10

Semesi AK, Mgaya YD, Muruke MHS, Msumi G, Francis J, Mtolera M (1998) Coastal resource utilization and conservation issues in Bagamoyo, Tanzania. Ambio 27:635–644

Sheaves M (2009) Consequences of ecological connectivity: the coastal ecosystem mosaic. Mar Ecol Prog Ser 391:107–115

Sheaves M, Molony B (2000) Short-circuit in the mangrove food chain. Mar Ecol Prog Ser 199:97–109

TCMP (2001) Tanzania state of the coast 2001: people and the environment. Working document 5059 TCMP. Tanzania Coastal Management Partnership, Dar es Salaam, Tanzania.

Thollot P, Kulbicki M (1988) Overlap between the fish fauna inventories of coral reefs, soft bottoms and mangroves in Saint-Vincent Bay (New Caledonia). Paper presented at the sixth international coral reef symposium, Townsville

Unsworth RKF, Bell JJ, Smith DJ (2007) Tidal fish connectivity of reef and sea grass habitats in the Indo-Pacific. J Mar Biol Assoc UK 87 (05):1287–1296

Unsworth RKF, De León PS, Garrard SL, Jompa J, Smith DJ, Bell JJ (2008) High connectivity of Indo-Pacific seagrass fish assemblages with mangrove and coral reef habitats. Mar Ecol Prog Ser 353:213–224

Verweij MC, Nagelkerken I (2007) Short and long-term movement and site fidelity of juvenile Haemulidae in back-reef habitats of a Caribbean embayment. Hydrobiologia 592:257–270

Verweij MC, Nagelkerken I, Wartenbergh SLJ, Pen IR, van der Velde G (2006) Caribbean mangroves and seagrass beds as daytime feeding habitats for juvenile French grunts, Haemulon flavolineatum. Mar Biol 149(6):1291–1299

Wagner GM (2005) Participatory monitoring of changes in coastal and marine biodiversity. Indian J Mar Sci 34(1):136–146

Wagner GM (2007) The Dar es Salaam seascape: a case study of an environmental management 'Hotspot'. Western Indian Ocean J Mar Sci 6(1):85–109

Wagner GM, Mgaya YD, Akwilapo FD, Ngowo RG, Sekadende BC, Allen A, Price N, Zollet EA, Mackentley N (1999) Restoration of coral reef and mangrove ecosystem at Kunduchi and Mbweni, Dar es Salaam, with community participation. In: Howell KM, Semesi AK (eds) Coastal resources of Bagamoyo Disctrict, Tanzania, Bagamoyo, 1999. Faculty of Science, University of Dar es Salaam, pp 467–488.

Wagner GM, Mgaya YD, Akwilapo FD, Ngowo RG, Sekadende BC, Allen A, Price N, Zollet EA, Mackentley N (2001) Restoration of coral reef and mangrove ecosystems at Kunduchi and Mbweni, Dar es Salaam, with community participation. pp 467–488. In: Richmond MD, Francis J (eds) Marine science development in Tanzania and Eastern Africa. 28 June–1 July 1999. Zanzibar, Tanzania, Zanzibar, Tanzania, 2001. IMS and WIOMSA, pp 467–488.

Wang Y, Bonynge G, Nugranad J, Traber M, Ngusaru A, Tobey J, Hale L, Bowen R, Makota V (2003) Remote sensing of mangrove change along the Tanzania Coast. Mar Geod 26:35–48

Limpopo Estuary Mangrove Transformation, Rehabilitation and Management

Salomão Bandeira and Henriques Balidy

Abstract

Limpopo River is among the most flood impacted rivers in southern and eastern Africa, having been flooded recurrently in many years. This paper deals with a successfully mangrove rehabilitation initiative in this river after floods in the year 2000 that devastated the estuarine mangrove. The flooding from upstream increased the width of the river from around 200 meters to several kilometers, drowning the mangrove forests for around 45 days, and causing sediment transformation and mangrove forest degradation, uprooting and dieback. The mean annual discharge of Limpopo River is usually 170 m^3/s. However, during the floods of 2000, the river exhibited a peak upstream flash of 16 515 m^3/s. River basin property and livelihoods were affected. The mangrove forest at Limpopo was, from the 1980's, known to cover only 387 ha, even though the Limpopo is the second largest river in Mozambique (after Zambezi). However, new recent research revealed a historical mangrove cover of 928 ha, of which 382 ha (41.2%) are quite pristine and 546 ha (58.8%) degraded. This prompted a mangrove rehabilitation in 2010 of with the species *Avicennia marina, Bruguiera gymnorhiza, Ceriops tagal*, *Rhizophora mucronata* and *Xylocarpus granatum, with* 26.3 ha replanted with 94 453 seedlings out of 168 367 produced in the nursery (74% survival rate). The Centre for Sustainable development of the Coastal Zones (CDZ-ZC) is the focus institution in mangrove rehabilitation process, which provided the initial training on nursery and replantation techniques. This chapter provides also insights of community engagement in the management of the Limpopo estuarine mangroves.

Keywords

Upstream floods • Mangrove degradation • Community participation • Nursery • Replantation • Eastern Africa

Introduction

Mangrove rehabilitation is currently a reality in several parts of the world (see Kairo et al. 2001; Primavera et al. 2011; Randy et al. 2015). The best examples of mangrove restauration in Africa are those from Gazi in Kenya (Kairo et al. 2008; Bosire et al. 2008). Mozambique is known to

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have among the largest mangrove forests in Africa (Barbosa et al. 2001; Fatoyinbo et al. 2008; Spalding et al. 2010), although anthropogenic deforestation was high in or near population centers (e.g. Macamo et al. 2014), but with few examples of mangrove replantation, a highlight of the UNEP-funded initiative at Lumbo, northern Mozambique (http://www.dlist-asclme.org/gallery/mangrove-plantation-near-lumbo-mozambique, 2015). The country is cyclone- and flood–prone, given the numerous cyclones and floods that recurrently hit Mozambique on a yearly basis (Mavume et al. 2014; Artur and Hilhorst 2012; Sitoe et al. 2015).

This study documents mangrove restoration at the Limpopo estuary. The Limpopo River is highly prone to floods, with its catchment occasionally influenced by tropical cyclones that can produce significant quantities of rainfall and cause phenomenal floods (WMO 2012, http://en.wikipedia.org/wiki/Limpopo_River, 2014). The Limpopo basin is one of the largest in southern and eastern Africa and, when flooded, may impact extensive areas, including its estuary. Data compiled by the Mozambique National Institute for Disaster Management indicated there has been severe flooding in the lower Limpopo basin over the past 50 yers, especially in 1955, 1967, 1972, 1975, 1977, 1981 and the 2000 flood, which was considered the worst flooding since 1848 (ARA-Sul 2002; INGC et al. 2003). The 2000 floods caused some 640 recorded deaths and more than 2 million people affected. The main highway road near the Limpopo estuary also was closed for several weeks, all being events reported and documented in the national and international media (WMO 2012). The recent 2013 flooding also had a high degree of severity, affecting about 380 693 people, with 154 816 displaced, 45 fatalities and 26 accommodation centres put in place within the Limpopo basin. Furthermore, the 2013 floods destroyed the infrastructure of several communities (Ministério da Planificação e Desenvolvimento 2013).

The Limpopo River (Fig. 1), with a length of 1750 km, and a basin of around 408 000 km^2, originates in the

Fig. 1 Geographical setting of the lower Limpopo river basin (Source: INGC et al. 2003; Limpopoark.com 2015)

southern African country of Botswana, runs over South African northern border with Botswana, then through Zimbabwe before flowing through southern Mozambique to the Indian Ocean at Zongoene, near Xai-Xai town (WMO 2012), located approximately 200 km north of the city of Maputo, Mozambique's capital. Its mean annual discharge is 170 m^3/s, at its mouth (Nakayama 2003). Extensive floods, such as those of the year 2000, increased the river width from around 200 m up to 10 km near the river mouth, causing mangrove forest degradation, uprooting and dieback via up 45 days of continuous flooding of both living and uprooted plants (https://en.wikipedia.org/wiki/2000_Mozambique_flood 2014; Vasco Mula, local community leader in personal communication). The flooding of the mangrove forests at Limpopo is related to the extensive upstream rains in its basin (Vaz 2000).

This study focuses on the impacts of flooding on the estuarine mangrove forests at the Limpopo River Estuary. It provides the background that enabled the mangrove replantation programme, the development of the nursery and restoration, as well as the engagement of the local communities in the mangrove restoration.

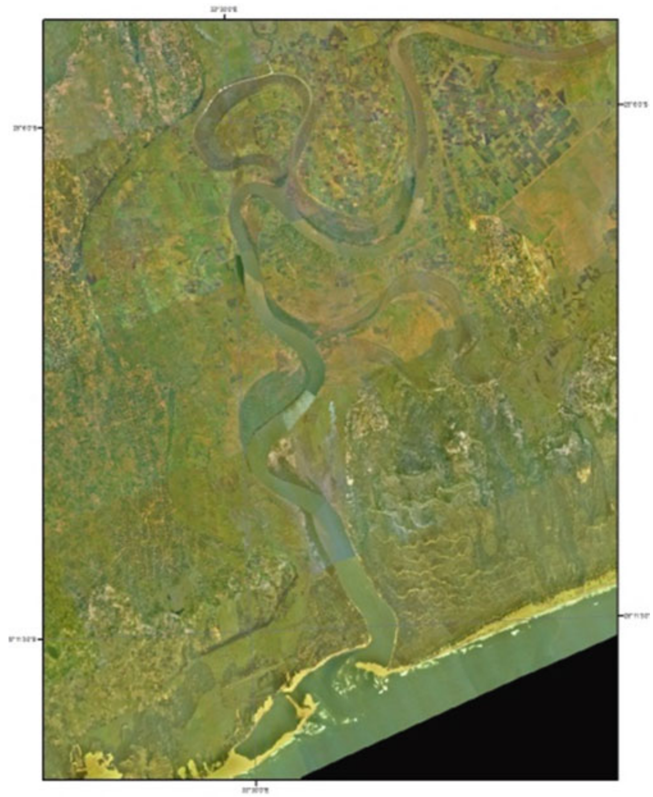

Fig. 2 Aerial photograph taken in 2003 and showing the Limpopo River estuary (Scale: 1:5000; Source: CDS-ZC)

Material and Methods

Mangrove Mapping

The aerial photography analysis of 2003 (three years after the 2000 floods), with a resolution of 1: 10000 (Fig. 2), as well as the field data collection (with "Garmin GPS V"), facilitated a preliminary processing and analysis with "Arc View" 3.2 extension "Image Analysis" 4 version 1.1. Field sampling was based on the collection of geographical coordinates, mangrove forest contours and boundaries of current and past occurrences, and the past occurrence information supported by observation and unstructured questionnaire targeting known boundaries of past mangrove forests in the estuary. Satellite image or aerial photographs describe the spatial information collected through remote sensing in detail, allowing interpretation of the final maps using the "Arc GIS 9". The cartographic database of DINAGECA 1997 was only used to better understand land use and land reclamation in the Limpopo estuary. The technical procedures used for the interpretation of aerial photograph complies with the manual "Arc View 3.2, Image Analysis 1.1" (ESRI 1998) and "Arc GIS 9.0" (http://www.virginiaview.net/education).

Field sampling was done to allow comparison of aerial photography data and the CENACARTA database of 1997. This was carried out through direct on-the-ground observations in the areas in which mangroves occurred, allowing a more precise location of the spots spatial coverage. The existence and degradation of mangroves in these locations was confirmed in the field, and their geographic coordinates recorded with the aid of the "Garmin GPS V." These geographic coordinates were the starting points for image analyses, using "remote sensing" techniques. Spots contours were outlined using ground truthing, walking in the countryside, correlating local and distinctive features with GPS, and using the "route 7" activated (methodology derived from interpretation of the functions of the "Garmin GPS V") function. This function allows drawing the contours of a march by converting the information into geographic coordinates. The contours drawn with the "GPS" in the field were superimposed on the aerial photograph, illustrating the actual boundaries of mangrove patches.

The map of the mangrove current situation was produced from manual and automatic scanning of the spots with distinctive characteristics, using "Arc View GIS 3.2." The aerial photography analysis was done with the extension "Image Analysis" 8, supervised classification (automatic scanning). This methodology also was used by Turner and Klaus (1996) to map the occurrence of seagrass in "Le Morne lagoon." Mauritius. The "unsupervised" classification (manual scanning) was used to correct the automatic scanning errors. Automatic classification describes the probability of finding a “pixel” at a point as belonging to a given class.

In a “supervised” classification, the following steps are considered:

1. Decide which classes to separate in the image. At this stage, the classes are further classified into information such as water, mangroves in good condition, degraded mangroves, agricultural areas, human settlements and sand dunes.
2. Select the “pixels” representative for each class/image sampling. These training sets or sampling can be established on the basis of field work (readings of “GPS”), maps or aerial photographs, or even color composites of satellite images. The “pixels” sampling can sometimes position themselves in the boundary areas of two classes and be a “pixels” mixture, which should be avoided since they would describe the characteristics of two classes.
3. Using the training data to classify the image with the chosen classifier, allowing assignment of the characteristics of each “pixel,” as the possible pre-determined classes and probability data. While the analysis to identify about 1% of “pixels” aerial photography was necessary in step 2, the classifier will identify and classify the entire image.
4. The production output results in the form of thematic charts, tabular data or digital data, which quantify and summarize the classification.
5. Proceed technically according to the manual of instructions “GIS ArcView Image Analysis” (ESRI 1998) and “Arc View GIS 3.2” ESRI (1996). The layout and printing of the maps was done in the “Arc Map GIS 9.0”.

Mangrove Rehabilitation

Mangrove restauration started with the involvement of The Centre for Sustainable Development of the Coastal Zones (CDS-ZC), headquartered at the outskirts of Xai-Xai town, close to Limpopo estuary. CDS-ZC was created in the 1990s, a few years before the flood of 2000, the most severe flood ever to hit the Limpopo. CDS-ZC is an autonomous institution under the Ministry of Land, Environment and Rural Development (MITADER), with overall mandate of providing technical assistance to local communities and promoting marine and coastal resources conservation.

The hydrological restoration, concerning the depth, duration and frequency of tidal flooding, was a key process for the success of mangrove restoration for the Limpopo estuary. Sedimentation also was evident, affecting extensive areas that became unsuitable for mangrove forest growth since hydrological channels were destroyed. A topography survey was first carried out, aimed at identifying the levels along the irrigation channel transect, to the subsequent correction in the excavation process. This topographic survey involved using a topographic level, a mira, tape measure and GPS. An initial area of 10 ha was identified for mangrove rehabilitation near the mouth of the Limpopo River.

The process of conservation and rehabilitation of mangrove estuary of the Limpopo estuary commenced in 2007. A local community leader (Mr. Vasco Mula) expressed interest on behalf of the entire Zongoene community and, given the CDS-ZC mandate for technical assistance to local communities and promoting of resources conservation, they began a baseline study and mangrove mapping in 2008 aimed at better understanding the mangrove state of conservation.

Results

Limpopo River Transformation and Impact on Mangroves

The Limpop estuary mangrove forest was known to cover only 387 ha in the 1980s, as published in the first mangrove change detection analysis for Mozambique (Saket and Mathusse 1994; Barbosa et al. 2001). Very recent mapping, however, revealed surprising results. The calculated new total historical mangrove area of the Limpopo estuary is 928 ha, of which only 382 ha (41%) are semi-intact mangrove forests and 546 ha (59%) are heavily degraded (Fig. 3). This degradation is attributed mainly to recurrent flooding related to those of the 2000’s that downed several trees, and enabled sedimentation and obstruction of mangrove inundation channels at the Limpopo estuary.

The susceptibility of the Lower Limpopo area to flooding is attributable to the topographic nature of extensive low land area and intense localized rainfall, which exhibits a peak flash of 16 515 m^3/s and 8 000 m^3/s in the 2000 and 2013 floods, respectively, all recorded at upstream river (Ministério da Planificação e Desenvolvimento 2013). There were no accounts of seastorms associated with the downstream river. The hydrometric measure of the peak flood at Xai-Xai (some 7–10 km upstream of mangroves) was 10 m in the 2000 flood and 6 m in 2013. The 2000 floods drowned all the mangroves, whereas the 2013 flood only drowned thse mangrove seedlings up to 1 m in height.

The Limpopo River estuary comprises 4 mangrove species: *Avicennia marina, Bruguiera gymnorhyza, Heritiera littoralis* and *Rhizophora mucronata. Ceriops tagal* and *Xylocarpus granatum* existed before the 2000 flooding (as denoted by the observed stumps).

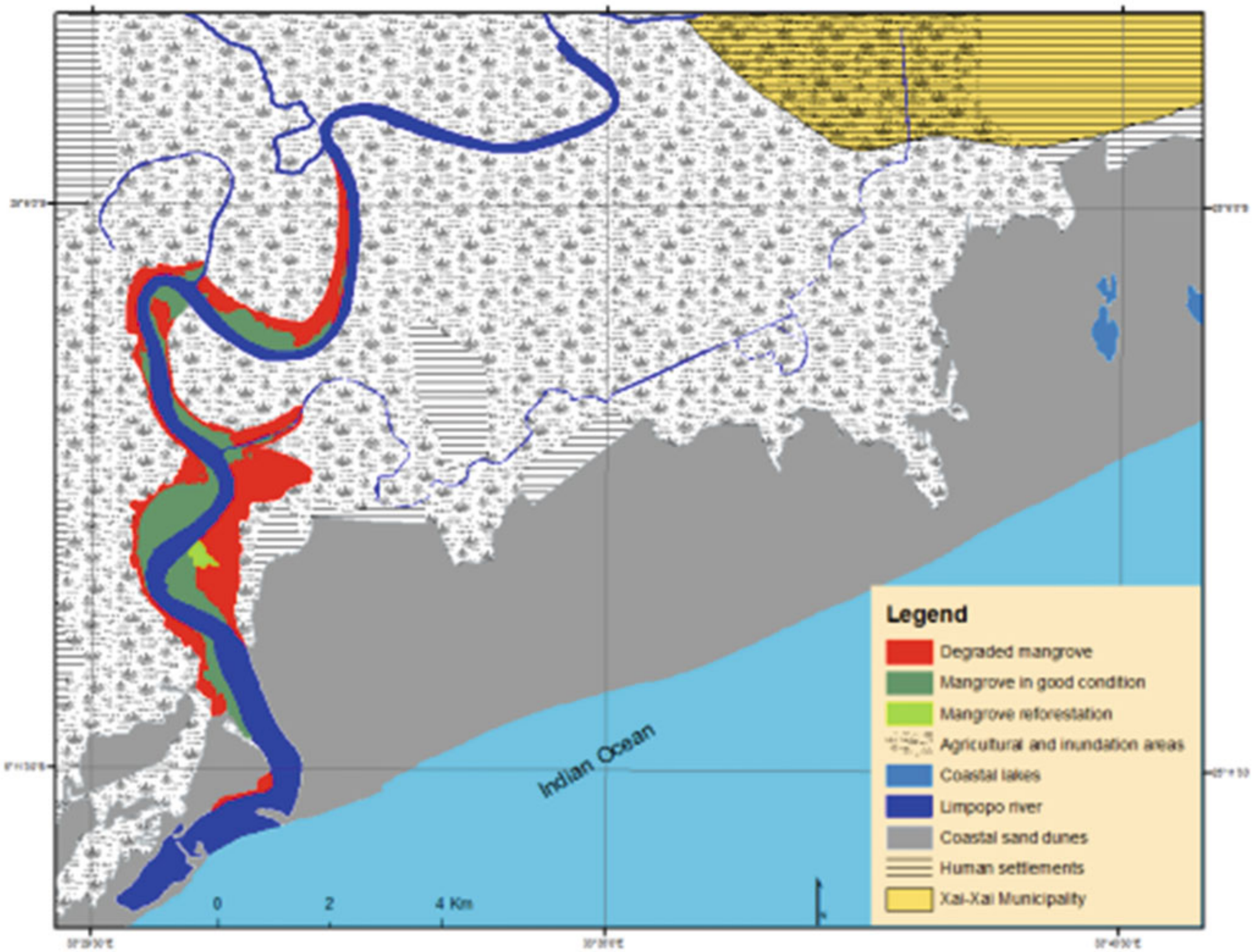

Fig. 3 Limpopo River estuary depicting well-kept, degraded and reforested mangrove areas

Each species of mangrove is adapted to a different level of substrate. The zonation of mangrove species allowed appropriate allocation of a zone for each of the species. For example, *Rhizophora mucronata* tolerates less variation in salinity, therefore preferring creeks and regular inundations. *Avicennia marina* tolerates fluctuations of salinities, both higher and lower. The other species generally occur between the inner and outer species.

Community Engagement

CDS-ZC highlighted the importance of restoring the Limpopo estuary mangroves for the following main reasons:

– Existence, at that time, of extensive damage to the Limpopo estuary mangrove forest due to the year 2000 extensive flooding that impacted not just the estuary, but also flooded several villages in the Limpopo valley also within the estuary.
– Community leaders at the estuary expressed interest in regaining their lost livelihoods after being wiped out by the 2000 floods. Fisherman and other livelihoods depending on mangrove habitats were impacted mainly those involved with fish, shrimp, crabs and wood products. The community leader of Zongoene village at the estuary (Vasco Mula) related the 45 days continuous flooding that drowned mangrove forest as well as destroying part of Zongoene village.
– The location of the CDS-ZC, some 20 km to the estuary was also decisive. The experience of seeing the flooding affecting the villages and populations centres adjacent to estuary (e.g., Zongoene), as well as affecting the Xai-Xai downtown located some 7 km to the mangrove forests, extensive livelihoods loss and displaced coastal communities, apart from losing their hopes, catapulted the initiative of revitalizing the lost mangrove forest at

Photo A Opening irrigation channel in preparation of the mangrove restauration in the Limpopo estuary (Photo: Henrique Balidy)

Limpopo River estuary. The replantation initiative had strong support because of the existence nearby of the flood disaster and a research institution (CDS-ZC).

A multisectorial and multidisciplinary team involving CDS-ZC, Provincial Directorate of Agriculture staff, and community from Mahielene village (located nearby the Limpopo Estuary) initially comprised 59 people (70% being women) initiated with the restoration of water creeks. They first opened a channel that covered the transect of 360 m and creek width with 2 m (Photo A). In general, the rehabilitation of the mangrove area comprised relatively uniform topography with an average difference of 1 m, leaving the mouth of the channel opening into the replantation area. These conditions are conducive to good irrigation or flooding of the area at Limpopo.

To develop the forest nursery, four community people chosen by the community were trained at the local school (theoretical lessons), as well as at the existing provincial nursery (practical lessons). The subjects discussed were as follows: substrate preparation, filling plastic bags, watering, transplanting and cultivation. A one hectare area near the village of Mahielene and Limpopo estuary was identified for locating the community nursery. Agricultural inputs such as watering cans, plastic pots, machetes, hoes, plastic buckets and boots were given to the community. CDS-ZC promoted environmental awareness and community campaigns for sensitization about mangrove conservation and replantation, culminating with the establishment of a community-led nursery in 2010. This initiative was later integrated in a five-year general government program plan which, in its objective number 3 of climatic changes, called for the restoration of degraded coastal ecosystems to create resilience at the community level. Additional funding (from PASA "Programa de Apoio ao Sector do Ambiente," a support program of the environmental sector) was annually allocated to ensure continuing mangrove forests rehabilitation at the Limpopo River estuary. The mangrove seedlings were taken to the location of their final planting at least six months after their stay in the nursery. The planting process was done with massive involvement of local community, through planting campaigns as a way to instill community interest in the defense of their environment and resources. A total of about 200 people from the community participated in several campaigns. As an incentive, each member of the community participating in the campaign received 60 Meticais (equivalent to 2 $US) for every 120 seedlings planted.

Nursery Development and Mangrove Rehabilitation

12 500 seedlings were produced by 2010, marking the start of the replantation process. The number of seedlings increased steadily over the years, totaling up to 168 367 in 2013 (Fig. 4). CDS-ZC had to seek propagules at the Incomati River estuary, located about 200 km south of Limpopo River, because of a scarcity of propagule seeds in the vicinity of the replanted area (Limpopo). The replanted mangrove species were *Avicennia marina, Bruguiera*

gymnorhiza, *Ceriops tagal*, *Rhizophora mucronata* and *Xylocarpus granatum*. An associated mangrove, *Thespesia populnea*, also was replanted. CDS-ZC decided to replant *C. tagal* and *X. granatum,* which were eradicated from the estuary after the 2000 floods, in order to reconstitute the previous mangrove forests diversity in Limpopo estuary. The Incomati River estuary (outskirts of Maputo city) was the donor mangrove for the propagules of *B. gymnorhiza, C. tagal* and *R. mucronata*, and seeds of *Xylocarpus granatum,* as well as the associated *T. populnea*. Until 2013 26,3 ha were replanted at the year 2000 flood-ravaged Limpopo River flood-ravaged areas, with 94.453 seedlings out of 168.367 produced in the nursery. An image of the nursery is provided in Photo B. The full nursery capacity is 100 000 seedlings. The overall seedlings survival rate was over 74% (Table 1). The 2013 floods, which inundated extensive areas, impacted the replantation effort by killing seedlings of 1 m height or less.

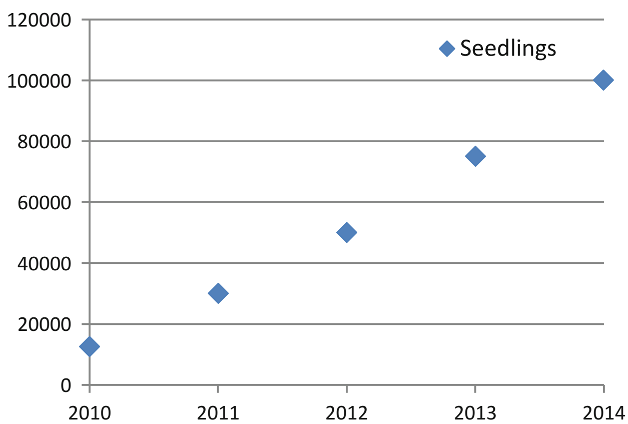

Fig. 4 Number of seedlings over the years in the nursery

Out of the total seedlings produced, most were *Avicennia marina*, followed by *Ceriops tagal, Rhizophora mucronata* (Table 2). The evolution of the area replanted over the years is shown in Fig. 5. It was estimated that of the 26.3 ha replanted in the Limpopo River estuary since 2010, 22.68 ha were replanted and restored in 2014. A contrasting image of the 'before' and 'after' replantation section in this estuary is illustrated in Photo C.

Mangrove Management

The Ministry for Coordination of Environmental Affairs (MICOA), renamed The Ministry for Land, Environment and Rural Development (MITADER) in early-2015 is the focus institution for mangrove rehabilitation. It also is a legislative and focal point for legislative instruments pertaining to mangroves, such as the upcoming Strategy and National Action Plan for Mangrove Management.

Photo B Nursery section with seedlings of Rhizophora mucronata (Photo: Salomão Bandeira)

Table 1 Total mangrove seedlings produced, replanted and survival rates over the years

Item	Year				
	2010	2011	2012	2013	Total
Seedlings produced in the nursery	12.500	29.200	51.667	75.000	**168.367**
Seedlings replanted to the field	10.800	25.200	37.453	21.000	**94.453**
Survival rate and percentage of planted	8.640 (80%)	11.340 (45%)	35.500 (94.8%)	14.700 (70%)	**70.140 (74.3%)**

Table 2 Raised seedlings per species at Limpopo estuary nursery

Species	Year				Total Nº species
	2010	2011	2012	2013	
Avicennia marina	9.621	18.752	14.404	39.526	**82.303**
Ceriops tagal	1.420	6.364	21.375	28.303	**57.462**
Bruguiera gymnorhiza	60	420	496	542	**1.518**
Rizophora mucronata	1.399	2.612	12.456	5.873	**22.340**
Thespesia populnea	0	632	1.620	0	**2.252**
Xylocarpus granatum	0	420	1.316	756	**2.492**
Total	**12.500**	**29.200**	**51.667**	**75.000**	**168.367**

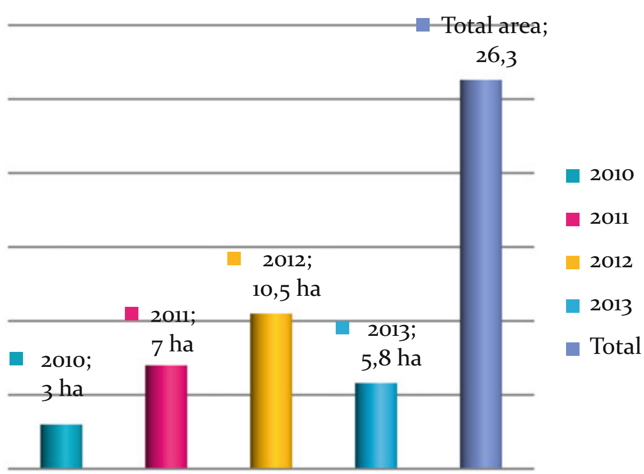

Fig. 5 Evolution of mangrove replanted area at Limpopo River estuary

The local communities at Zongoene have shown their appreciation for the mangrove forest restoration project. The ten employed community workers continue to conduct their daily activities of raising the nursery, conducting replantation and supervision of the tidal waters pathways to the replanted mangroves, and protecting seedlings away from grazers (mainly the crab *Neosarmatium meinerti)*. This is a daily activity from July until March at the latest, until the number of seedlings for replantation are sufficiently mature (i.e., those with at least 6 months resident time at the nursery). The routine involves checking tide inundation, the condition of the opened creek, the health of the replanted mangrove leaves, mangrove growth, and the emergence of new leaves. Dead seedlings are immediately replaced. Maintenance to the creeks and water ways is conducted annually, although the opening of a main creek in 2010 has negated the need for maintenance at the present time. The lower part of the mangrove seedling stems have been covered by reed (*Phragmites spp*.) stalks or stems to prevent grazing crabs from eating and destroying the seedlings. There is a need for more frequent checking (several times weekly) for newly-planted seedlings (up to 1 m height). Nursery growth monitoring is done weekly, beginning when the seedling emit the first sheet. The height is measured by first taking the length from the stem base to the tip of the plant (i.e., the growth of the main stem node). For each species, on a weekly basis, 200 seedlings chosen randomly and monitored for growth. Weekly growth measurements are taken during a six-month period, which is the minimum residence time of seedlings in the nursery. Watering occurs twice daily, using tidal water. Seedling mortality also is conducted daily, through verification of dead plants at each site of the nursery. These are excluded from the site and registered according to the instructions of the nursery files.

The replantation process has gauged natural regeneration, especially of *Avicennia marina*, the most common mangrove species at the Limpopo River estuary. Although regeneration measurements were limited, preliminary sampling indicated that *A. marina* corresponded to over 80% of existing mangrove species after replantation began (Balidy et al. 2005). Photo D depicts thriving *A. marina*, first replanted, but already exhibiting natural regeneration. Project monitoring at the nursery and at the replanted area has been conducted successfully several times. Replanting failures were noted at the beginning of the replantation effort, mainly because of the grazing of seedlings by crabs. As noted above, this problem was solved by covering the seedling stems with reeds. More recently, the recurrent floods at Limpopo,

Photo C Before and after replantation (Photo: Henriques Balidy)

Photo D Replanted Avicennia marina (Photo: Henriques Balidy)

especially in January 2013, were problematic, causing the loss of approximately 1 ha of recently-replanted juveniles (of up to 1 m height) because of about 15 days of continuous submersion of the seedlings.

CDS-ZC, together with other entities, are introducing alternative activities to ensure the sustainability of mangrove rehabilitation. These include fisheries farming and mangrove honey production, among others. The Limpopo River estuary community believe that being fully trained on the degraded mangrove rehabilitation process, as well as because of their own initiative, their request for the establishment of a community center for the training of degraded mangrove rehabilitation was justified. They also believed that Limpopo's rehabilitation success should be expanded to other regions of Mozambique where mangrove degradation have yet to be addressed with appropriate community engagement.

Discussion

Although mangrove transformation due to natural and antropogenic causes have been documented elsewhere (Giri et al. 2008), there are few examples of the impacts on mangroves attributable to upstream flooding, in contrast to those related to tsunamis (Alongi 2008; Yanagisawa et al. 2009). The year 2000 floods to Limpopo estuary resulted from upstream torrential rains in neighboring countries, a scenario documented for other rivers in Mozambique, such as the Zambezi. The mangrove forests at Limpopo exhibited the diversity and zonation patterns of mangrove stands in southern Mozambique, comprising basically five species, and dominated by *Avicennia marina*. (Paula et al. 2014), *Avicennia marina* being common in the inner and outer parts of the estuary, *Rhizophora mucronata* in the creeks where salinity is stable, and other species such as *Ceriops tagal* and *Bruguiera gymnorhiza* occurring intermingled between the first two species. This pattern may occur, although less pronounced, in other regions of eastern Africa, such as northern Mozambique and Tanzania. (Bandeira et al. 2009). As per the monitoring conducted after the 2000 floods, it appears the mangrove species *Xylocarpus granatum* and *Ceriops tagal* had disappeared from the estuary. The decision to replant these species was based on the objective of regenerating the pre-2000 mangrove vegetation at Limpopo. Mangroves also were replanted elsewhere in a new area not previously naturally colonized by mangroves, specifically in the Nahoon River mount in South Africa, which appears to be the southernmost location of mangroves in the western Indian ocean (Colloty 2001; Beentje and Bandeira 2007).

The flooding of the Limpopo estuary, and the process of community engagement in mangrove replantation and management marks the beginning of a common understanding of a broader estuarine system that previously lacked a community management structure. Misfortunes such as flooding events on mangroves might trigger action towards protection of mangroves habitats and livelihoods, as was observed in social-ecological research carried out in Cameroon (Munji et al. 2014). Mangroves account for a number of sustainable goods and services, as documented (Kirui 2013), some of which can be assessed with sustainable practices. Utilizing relevant perceptions and responses indicated impacts of such events as flooding of mangrove forests may poses challenges for sustainable management (Munji et al. 2014). Nevertheless, is also can facilitate identification of pertinent needs, challenges and opportunities to inform and guide effective decision-making towards appropriate community-based management strategies directed to natural resources such as mangroves (Datta et al. 2012; Munji et al. 2014).

Conclusion

The mangrove replantation activity has been a successful means for community efforts directed toward their main source of livelihoods in the Limpopo River estuary. A major success was opening of the mangrove creek, which enabled replantation and viable tidal inundation of the areas affected by the flood of the year 2000. Protection of young seedlings prevented major grazing by the crab pest (*Neosarmatium meinerti*). CDS-ZC and local communities engaged in this replantation, forming a bond and forum for better understanding of the relation of the community with nature in the estuary and in its surroundings. More study and research is still needed, including documentation of the impacts of such events on people's livelihoods, which has yet to be carried out. The structural dynamics of the natural and replanted mangroves also need to be documented in a manner similar to the analyses carried out for Kenyan replanted mangroves.

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References

Alongi DM (2008) Mangrove forests: resilience, protection from tsunamis and responses to global climate change. Estuar Coast Shelf Sci 76:1–13

ARA-Sul – Administração Regional das Águas do Sul (2002) Mapas de privisão da bacia do Limpopo. Versão final, Maputo (in Portuguese)

Artur L, Hilhorst D (2012) Everyday realities of climate change adaptation in Mozambique. Glob Environ Chang 22:529–536

Balidy HJ, Sitoe A, Menomussanga M, Pires PL (2005) Avaliação dos níveis de corte, composição específica e regeneração natural de mangal no Sul de Moçambique. Technical report. CDS-ZC, Xai-Xai, 20 pp (in Portuguese)

Bandeira SO, Macamo CCF, Kairo JG, Amade F, Jidawi N, Paula J (2009) Evaluation of mangrove structure and condition in two transboundary areas in the Western Indian Ocean. Aquat Conserv Mar Freshw Ecosyst 19(1):46–55

Barbosa FMA, Cuambe CC, Bandeira SO (2001) Status and distribution of mangroves in Mozambique. S Afr J Bot 67:393–398

Beentje H, Bandeira S (2007) A field guide to the mangrove trees of Africa and Madagascar. Royal Botanic Gardens, Kew, 91 pp

Bosire JO, Dahdouh-Guebas F, Walton M, Crona BI, Lewis RR III, Field C, Kairo JG, Koedam N (2008) Review functionality of restored mangroves: a review. Aquat Bot 89:251–259

Colloty BM (2001) Botanical importance in the former Ciskei7Transkei region. PhD thesis, University of Port Elizabeth

Datta D, Chattopadhyay RN, Guha P (2012) Community based mangrove management: a review on status and sustainability. J Environ Manage 107:84–95

ESRI – Environmental Systems Research Institute (1996) Using arc view GIS. The geographic information system for everyone TM. 350 pp, USA

ESRI – Environmental Systems Research Institute (1998) Using arc view GIS mage analysis. 208 pp, USA

Fatoyinbo T, Simard M, Washington-Allen L, Shugart H (2008) Landscape extent, height, biomass, and carbon estimation of Mozambique's mangrove forests with Landsat ETM+ and shuttle radar topography mission elevation data. J Geophys Res, 113:13pp. doi:10.1029/2007JG000551

Giri C, Zhu Z, Tieszen L, Singh A, Gillete S, Kelmelis A (2008) Mangrove forest distributions and dynamics (1975–2005) of the tsunami-affected region of Asia. J Biogeogr 35:519–528

INGC, UEM, FEWS Net Mind (2003) Atlas para Preparação e Resposta Contra Desastres na Bacia do Limpopo, Maputo (in Portuguese)

Kairo JG, Dahdouh-Guebas F, Bosire J, Koedam N (2001) Restoration and management of mangrove systems – a lesson for and from the East African region. S Afr J Bot 67:383–389

Kairo JG, Lang'at JKS, Dahdouh-Guebas F, Bosire J, Karachi M (2008) Structural development and productivity of replanted mangrove plantations in Kenya. For Ecol Manage 255:2670–2677

Kirui B (2013) Importance and dynamics of the mangroves in Kenya. Devs Earth Surf Process 16:193–198

Macamo C, Balidy H, Bandeira S (2014) Incomati mangrove deforestation. In: Bandeira S, Paula J (eds) The Maputo Bay ecosystem. WIOMSA, Zanzibar Town, pp 127–130

Mavume A, Pinto I, Massuanganhe E (2014) Potential climate change impacts on Maputo Bay. In: Bandeira S, Paula J (eds) The Maputo Bay ecosystem. WIOMSA, Zanzibar Town, pp 383–397

Ministério da Planificação e Desenvolvimento (2013) Diagnóstico Preliminar e Acções de Reconstrução Pós-Calamidades (2013). Report, Maputo, 49 pp (in Portuguese)

Munji CA, Bele MY, Idinoba ME, Sonwa DJ (2014) Floods and mangrove forests, friends or foes? Perceptions of relationships and risks in Cameroon coastal mangroves. Estuar Coast Shelf Sci 140:67–75

Nakayama M (2003) International waters in Southern Africa (Water Resources Management and Policy). United Nations University Press, USA, 318 pp. ISBN 9280810774

Paula J, Macamo C, Bandeira S (2014) Mangroves of Maputo Bay. In: Bandeira S, Paula J (eds) The Maputo Bay ecosystem. WIOMSA, Zanzibar Town, pp 109–126

Primavera JH, Rollon RN, Samson MS (2011) The pressing challenges of mangrove rehabilitation: pond reversion and coastal protection. Treatise Estuar Coal Sci 10:217–244

Randy AF, Hutomo M, Purnama H (2015) Collaborative efforts on mangrove restoration in Sedari village, Karawang district, West Java province. Procedia Environ Sci 23:48–57

Saket M, Mathusse R (1994) Study for the determination of the rate of deforestation of the mangrove vegetation in Mozambique. FAO/PNUD/MOZ/92/013. Unidade de Inventário Florestal Departamento de Florestas, DNFFB. Min da Agricultura, Maputo (in Portuguese)

Sitoe SR, Risberg J, Norström E, Snowball I, Holmgren K, Achimo M, Mugabe J (2015) Paleo-environment and flooding of the Limpopo River-plain, Mozambique, between c. AD 1200–2000 Catena 126: 105–116

Spalding M, Kainuma M, Collins L (2010) World atlas of mangroves. A collaborative project of ITTO, ISME, FAO, UNEP-WCMC, UNESCO-MAB, UNU – INWEH and TNC. Earthscan, London, 319 pp

Turner JR, Klaus R (1996) Remote sensing of coastal habitats. In: M. Richmond (ed) A Field Guide to the Seashores of Eastern African and the Western Indian Ocean Islands. Sida/SAREC-UDSM, Dar Es Salaam, pp 44–45

Vaz AC (2000) Coping with floods – the experience of Mozambique. WARFSA/WaterNet symposium: sustainable use of water resources, Maputo, 1–2 November 2000

WMO – World Meteorological Organization (2012) Limpopo river basin. A proposal to improve the floods forecasting and early warning system

Yanagisawa H, Koshimura S, Goto K, Miyagi T, Imamura F, Ruangrassamee A, Tanavud C (2009) The reduction effects of mangrove forest on a tsunami based on field surveys at Pakarang Cape, Thailand and numerical analysis. Estuar Coast Shelf Sci 81:27–37

Section III

However, Efforts Underway to Protect Estuaries Are a Source of Hope. . . . .

Coastal and marine ecosystems represent essential income and livelihood sources for numerous coastal inhabitants, significantly contributing to the growing economies of the countries in the WIO region. The current (and conservative) estimate of the economic value of the goods and services provided by the WIO coastal and marine environment exceed 25 billion US dollars annually, considering its combined coastline of more than 15,000 km, including those of the island states, and a total continental shelf area of about 450,00 km^2 (UNEP 2009). Indeed, the value of the ecosystem services provided by coral reefs alone in the WIO region is estimated to exceed 7 billion US dollars per year, with mangroves adding nearly 9 billion US dollars in ecosystem services per year. In fact, the direct benefits obtained from coastal goods and services in South Africa alone, the largest economy in the region, is estimated to be equivalent to about 35% of the country's gross domestic product.

The region's beautiful sandy beaches mangrove forests, lagoons and coral reefs make tourism the largest income source directly linked to the WIO's coastal and marine environment. They attract more than 20 million tourists from all over the world each year injecting more than 6 billion US dollars into the economies of the WIO region countries each year, sustaining the livelihoods of those dependent on this activity for their well-being.

As previously highlighted in Section 1, the WIO region's marine waters, particularly its coastal waters, lagoons, estuaries and continented shelves, are very important and lucrative fishing grounds Indeed, the WIO region generates about 4.5 million tonnes of fish per year, equivalent to about 4.8% of the total global fish catch. It is suggested that this value actually may be an underestimate because of the under-reporting of fish catches by some WIO countries. Although not as productive as some better-known fishing grounds in the world, particularly those associated with upwelling systems, the WIO fisheries sector nevertheless continues to be of major importance in terms of food security, employment and income generation for the growing coastal population, providing food and livelihoods to about 61 million WIO coastal inhabitants.

In light of the overwhelming values of the WIO estuarine and related coastal ecosystems, governance and management of these critical assets are a critical factor in maintaining the valuable functions and services that they represent. Fortunately, the latest decade has seen an increasing attention to the importance of the marine and coastal environment of the region. This book itself is a display of some of the important scientific work that has and is being undertaken by scientists from the region and beyond. Also Government increasingly put attention to the matters at hand, with key bodies such as the Nairobi Convention, the south Western Indian Ocean Fisheries Commission, the Indian Ocean Commission, gaining maturity and strength.

The final chapters of this book provide a cross-cut of some of the systems and intervention being undertaken to conserve and manage the important estuarine and coastal ecosystems of the WIO region, including the river basins leading in to them.

Re-thinking Estuarine Ecosystem Governance in the WIO Region

Akunga Momanyi

Abstract

It is undisputable that estuarine ecosystems are a vitally important component of the rich environmental and cultural heritage of the Western Indian Ocean (WIO). Estuaries are extremely important in providing cultural, provisioning and regulatory services.

There are a myriad of governance frameworks and structures at the global, regional, national and even local levels that affect the WIO region: generally on coastal and marine environmental protection, and less so on estuaries as such. There are remarkable examples at the regional level, including the 1987 Zambezi River System Agreement and its SADC Protocol and the Zambezi River Commission which together "represent the most ambitious approach to environmental protection of river basins in the developing world," and "which exemplify the potential of common management in addressing environmental problems." Other examples include the Inkomati River Basin and the Limpopo River Basin frameworks. It is apparent that regional river basin frameworks treat estuaries as part of international water courses or river basin systems.

As the case studies from Kenya, Mozambique and South Africa have demonstrated, the governance and institutional setting for coastal and marine environmental protection, including estuarine ecosystems, is quite robust. All the three countries have constitutional, legislative and policy frameworks that broadly address matters related to coastal and marine environmental protection generally, albeit with lack of specificity on estuarine ecosystem governance, with the remarkable exception of South Africa's Integrated Coastal Management Act No 24 of 2008 which dedicates its Chapter 4 to estuaries. In this regard, South Africa provides a model worthy of emulation across the WIO region.

However, it is apparent from this study that the various international and national legal, institutional and policy frameworks concerning the governance of estuarine ecosystems in the region are generally inadequate, uncoordinated and weak. This scenario poses threats and challenges to the estuarine ecosystems necessitating a rethinking of the frameworks with a view to reforms. The key threats and challenges include: land-based and sea-based sources and activities causing pollution and degradation, policy and legislative inadequacies, limited institutional capacities, inadequate awareness, inadequate financial resources and mechanisms, as well as poor knowledge management, and the sheer complexity of some of the regimes established. On the positive side, opportunities include better understanding of the various causes and impacts of pollution and degradation of the coastal and marine environment as borne by the numerous scientific and technical studies; promising climate change regulation and mitigation interventions; and better legal, policy

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and institutional frameworks which emphasize integration, ecosystem or basin wide approaches and sustainable development.

Key recommendations towards better protection of estuarine ecosystems include further studies and reviews on estuarine ecosystems and their complex interactions with other ecosystems and their governance arrangements particularly in the context of land-sea interaction. Others include reviews of relevant international and national legal and other regulatory regimes to address better specificity on ecosystem governance.

Keywords

Estuarine ecosystems • Estuarine ecosystem governance • International water law • International watercourses governance • Shared water resources • Riparian states • Equitable utilization • Common management • Community of interests • Common management institutions • Regional river basin frameworks • Estuarine management plan • Land/sea interaction

Introduction

The estuaries of this region are extremely important in providing cultural (recreational, spiritual, etc.), provisioning (food, timber etc.) and regulatory (flood protection, climate regulation, etc.) services that are at the core of both the functioning of the entire ecosystem and the livelihoods of the over 60 million coastal habitants of this region.

There have been numerous studies on various aspects of the environment and ecosystems of the WIO region over the years, including on governance frameworks and structures.[1] However, there has not been specific focus on estuaries as such. And yet it is arguable that the various international and national legal, institutional and policy frameworks concerning the governance of estuarine ecosystems in the region are generally inadequate, uncoordinated and weak. This scenario poses threats and challenges to the estuarine ecosystems necessitating a re-thinking of the frameworks with a view to reforms.

This chapter proposes to review the governance and institutional setting of the estuarine ecosystems in selected countries: Kenya, Mozambique, and South Africa. Taking a case study approach, it covers relevant global and regional legal, institutional and policy frameworks, including inter governmental agreements, river basin governance frameworks, as well as national governance frameworks in the selected countries. The chapter will also present a brief synthesis of threats, challenges and opportunities facing the governance of estuarine ecosystems in the WIO region, as well as key recommendations towards better protection of this vital ecosystem.

Governance and Institutional Setting: International Perspectives and Frameworks

Estuarine ecosystem governance is fundamentally part of international and national watercourses governance, underpinned by international and national water law. Some of the key problems and challenges confronting international water law include questions of access and allocation of water resources, environmental protection and sustainable use of water resources. Water supply is already seriously inadequate in parts of Africa and central Asia; desertification, pollution, irrigation use; construction of massive dams for hydro-electric purposes, and other factors and uses have caused a situation where many rivers and lakes no longer support a natural ecosystem, leading to the loss of wetlands, swamps and other natural habitats for wildlife, some of which are associated with estuarine ecosystems.[2] These may be regarded as largely anthropogenic problems of

[1] See, for example, UNEP/Nairobi Convention Secretariat, 2009. Transboundary Diagnostic Analysis of Land Based Sources and Activities Affecting the Western Indian Ocean Coastal and Marine Environment, UNEP Nairobi, Kenya 378p; AFD/IDDRI (2011): Sick Seas and Oceans and the Challenges of Combating Land Based Pollution and Degradation: The Example of Western Indian Ocean Region, in A Planet for Life, (2011); UNEP, 2009: Regional Synthesis Report On Legal, Regulatory and Institutional Frameworks in the WIO Region. UNEP/GEF WIO-LaB Project Report. *114p UNEP/GEF/WIO-LaB/LTR/2009;* UNEP, 2009: Regional Synthesis Report on Ratification and Implementation of International Environmental Conventions relevant to Land based Activities/Sources of Pollution of the Coastal and Marine Environment of the WIO Region, UNEP/GEF WIO-LaB Project Report. *77p UNEP/GEF/WIO-LaB/LTR/2009;* UNEP/GPA Government of Kenya/NEMA, 2009: State of Coast Report 2009; 2004: "Physical Alteration and Destruction of Habitats in the Marine and Coastal Environment of Eastern Africa: Legal and Institutional Issues.

[2] Birnie, P.W, Boyle A.E, Redgwell, *International Law and the Environment,* Third Edition, (2009) Oxford University Press, p 535.

poor management and inadequate governance, and they are worsened by effects of climate change on fresh water supply, with the melting of mountain glaciers in all continents and changing rainfall patterns already posing a real threat to the continued flow of major rivers.[3]

Unfortunately, historically international water law has not been particularly concerned with the foregoing problems as its main focus has been traditionally on the rules and principles for allocating water supply in international water sources between upstream and downstream states, with only incidental treatment of environmental and sustainability issues.[4] Fortunately, there have been serious efforts in recent years to address these shortcomings and to give international water law a broader ecological perspective within a legal framework more attuned to sustainability and water shortage.[5] Even then, it is true that international law largely does not address fresh water resources, unless they are part of an international water course,[6] or cause marine pollution, or supply problems have become so severe as to import human rights issues.[7]

On international water courses governance, according to *Birnie, Boyle and Redgewell*, the management of international watercourses through regional cooperation provides the most comprehensive basis for environmental protection and pollution control.[8] These leading authors also discuss the principles and approaches of allocation of shared water resources. These include the now largely discredited territorial sovereignty principle, which favors upstream states at the expense of downstream riparian states. According to this principle, also known as the Harmon doctrine, states enjoy absolute sovereignty over water within their territory and are free to do as they please with those waters, including extracting as much as necessary, or altering their quality, regardless of the effect this has on the use or supply of water in downstream or contiguous states.[9] Another equally discredited or questionable principle is the so-called principle of absolute territorial integrity or riparian rights. According to this theory, the lower riparian state would have the right to a full flow of water of natural quality, and interference with the natural flow by the upstream state would thus require the consent of the lower riparian.[10]

The third and most widely endorsed theory treats international watercourses as shared resources, subject to equitable utilization by riparian states.[11] Although the notion of "shared resources" is still controversial, nevertheless the theory of equitable utilization rests on a foundation of shared sovereignty, which entails a balance of interests, which accommodates the needs and uses of each state, and does not entail equal division.[12] The basic principle of equitable utilization enjoys substantial support in judicial decisions,[13] state practice, and international codifications[14] as one encapsulating a "community of interests".[15]

The principle of equitable utilization lead naturally to the fourth principle or theory, called "common management".[16] According to *Birnie, Boyle and Redgwell*, "common management is the logical combination of the idea that watercourse basins are most efficiently managed as an integrated whole, and the need to find effective institutional machinery to secure cooperation on environmental, social and economic objectives".[17] Common management represents a community of interest approach which goes beyond the allocation of equitable rights and opens up the possibility of integrating development and international regulation of the watercourse environment,[18] and is generally consistent with the basin or

[3] Ibid, p 536.

[4] Ibid.

[5] ibid.

[6] "International water course" may be simply defined as including rivers, lakes, or ground water resources shared by two or more states. Such water courses will normally either form or straddle an international boundary, or in the case of rivers, they may flow through a succession of states (See Birnie, P.W, Boyle A.E, and Redgwell, *ibid*, p 536, citing McCaffrey, *The Law of International Watercourses* (2nd ed, Oxford, 2007), chapter 2. Moreover, though not completely settled, the preferred geographical scope of watercourses is the basin approach. According to the Commentary to the 1966 Helsinki Rules, "The drainage basin is an indivisible hydrologic unit which requires comprehensive consideration in order to effect maximum utilization and development of any portion of its waters".(ILA, Helsinki Rules on the Uses of the Waters of International Rivers, Report of 52nd Conf (1966) 485). See also ILA, Berlin Rules on Water Resources ("Berlin Rules"), Report of 71st Conf (2004) 344; Teclaff, *The River Basin in History and Law* (The Hague, 1967).

[7] Birnie, P.W, Boyle A.E, Redgwell, *International Law and the Environment,* Third Edition, (2009) Oxford University Press, p 536.

[8] Ibid, p 572.

[9] Ibid p 540.

[10] Ibid, p 541.

[11] McCaffrey, *The Law of International Watercourses* (2nd ed, Oxford, 2007), ch. 10, cited favorably by Birnie, P.W, Boyle A.E, Redgwell, ibid. p 541.

[12] Birnie, P.W, Boyle A.E, Redgwell, *International Law and the Environment,* Third Edition, (2009) Oxford University Press, p 542.

[13] For example, *Territorial Jurisdiction of the International Commission of the River Oder Case*, PCIJ Ser A No 23 (1929); *Diversion of Water from the Meuse Case,* PCIJ Ser A/B No 70 (1937); *Lac Lanoux Arbitration*, 24 ILR (1957) 101.

[14] For example, 1997 Convention on Non Navigational Uses of International Watercourses, *36 ILM* (1997) 719, 27 EPL (1997) 233, Articles 5, 6;1992 Convention on the Protection and Use of Transboundary Watercourses and Lakes (Helsinki), *B&B Docs,* 345, Article 2; ILA, Berlin Rules on Water Resources ("Berlin Rules"), Report of 71st Conf (2004), 344, Article 12.

[15] Birnie, P.W, Boyle A.E, Redgwell, *International Law and the Environment,* Third Edition, (2009) Oxford University Press, p 542.

[16] Ibid, p 544.

[17] Ibid, citing Schwebel, II YbILC (1982) Pt 1, 76, para 70.

[18] Ibid, citing, *inter alia*, McCaffrey, *The Law of International Watercourses* (2nd ed, Oxford, 2007), p 147–70; Benvenisti, Sharing Transboundary Resources (Cambridge 2002).

hydrologic system approach which is preferred by modern state practice.[19] This invariably includes estuarine ecosystems.

The common management model is usually characterized by the creation of international institutions in which all riparian states cooperate in formulating and implementing policies for the development and use of a watercourse.[20] African examples include the Zambezi Intergovernmental Monitoring and Coordinating Committee, established under the Zambezi River System Agreement[21]; the River Niger Commission,[22] and the Permanent Joint Technical Commission for Nile Waters,[23] among others. Common management institutions, such as the foregoing examples, have become the basis for environmental regulation and sustainable development of a number of international watercourses.[24] Both the 1992 UNECE Watercourses Convention[25] and the 1997 UN International Watercourses Convention[26] have included provisions on common management institutions, although in different terms. While cooperation by riparian states in joint management institutions is not obligatory as a matter of prevailing general international law, it nevertheless may be regarded as a 'principle of progressive international law".[27]

In the WIO region, the 1985 Nairobi Convention framework,[28] and various river basin organizations[29] provide the platform for regional cooperation for water course management, including estuarine ecosystem governance.

Environmental Protection of Estuarine Ecosystems in the WIO Region

At the international environmental level, there is a recognition that "responsibility for action to protect and enhance the environment rests primarily with Governments and, in the first instance, can be exercised more effectively at the national and regional levels."[30] Moreover, with respect to international water courses, which may be classified as part of "environmental problems of broad international significance" the United Nations (UN) system has competence,[31] primarily through its various sectoral institutions. There are also international legal instruments which address water courses governance, the main one being the 1997 International Watercourses Convention,[32] which came into force in August 2014. As estuarine ecosystems environments in the WIO typically import the land –sea interaction,[33] the 1982 UN Convention on the Law of the Sea,[34] which is the framework law of the seas and oceans, also comes into play.

With respect to international watercourses ecosystems in the WIO region, including the Zambezi River and the Incomati River Basins, the 1997 International Watercourses Convention has relevant provisions on environmental protection and preservation, and particularly Article 20 (protection and preservation of ecosystems), Article 21 (prevention, reduction and control of pollution), Article 22(introduction of alien or new species), Article 23 (protection and preservation of the marine environment), and Article 24 (management).

Article 20 provides as follows:

> "Watercourse states shall, individually and, where appropriate, jointly, protect and preserve the ecosystems of international watercourses"

[19] Birnie, P.W, Boyle A.E, Redgwell, *International Law and the Environment,* Third Edition, (2009) Oxford University Press, p 544.

[20] Ibid., p 544.

[21] 1987 Botswana-Mozambique-Tanzania-Zambia-Zimbabwe Agreement. *27 ILM* (1988).

[22] 1963 Act Regarding Navigation and Economic Cooperation between the States of the Niger Basin, in Ruster and Simma, International Protection of the Environment (New York, 1977) xi, 5633.

[23] 1959 Agreement Between the UAR and the Republic of the Sudan for the Full Utilization of Nile Waters, and the 1960 Protocol Establishing Permanent Joint Technical Committee, in UN, *Legislative Texts and Treaty Provisions Concerning the Utilization of International Rivers for Purposes Other Than Navigation*, UN Doc ST/LEG/Ser B/12, 143 ff.

[24] Birnie, P.W, Boyle A.E, Redgwell, *International Law and the Environment,* Third Edition, (2009) Oxford University Press, p 545.

[25] 1992 Convention on the Protection and Use of Transboundary Watercourses and Lakes (Helsinki), *B&B Docs,* 345.

[26] 1997 Convention on Non Navigational Uses of International Watercourses, *36 ILM* (1997) 719, 27 EPL (1997) 233, Articles 8, 24.

[27] Birnie, P.W, Boyle A.E, Redgwell, *International Law and the Environment,* Third Edition, (2009) Oxford University Press, p 546.

[28] Convention for the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region (1985 Nairobi Convention); Protocol concerning Protected Areas and Wild Flora and Fauna in the Eastern African Region; Protocol Concerning Cooperation in Combating Marine Pollution in Cases of Emergency in the Eastern African Region 21 June 1985, IELMT 985; Protocol on Land Based Sources and Activities in the Western Indian Ocean (2010 LBSA Protocol). For the 2010 LBSA Protocol and 2010 Amended Nairobi Convention, see *www.unep.org/nairobiconvention/*

[29] For example, the Zambezi River and Incomati River Basins discussed, *infra*.

[30] UN General Assembly Resolution 2997(XXVII) on Establishment of Institutional and Financial Arrangements for International Environmental Cooperation (1972).

[31] Ibid.

[32] Convention on the Non-Navigational Uses of International Watercourses, *36 ILM* (1997)719; 27 EPL (1997) 233.

[33] The out flow of a river-where it transforms from being a freshwater system via an estuary into the saltwater marine environment, with all the cumulative impacts. In the WIO region, six key land based sectors account for estuarine ecosystem environmental impacts including alteration of river flow, degradation of water quality and alterations in river sediment loads: urbanization, agriculture and forestry, industry, mining, transportation, and energy production.

[34] 1982 UN Convention on the Law of the Sea (Montego Bay) Misc 11 (1983) CMND 8941;21 ILM (1982) 1261.

Article 21 obliges states to take measures to individually or jointly prevent, reduce and control pollution of international watercourses,[35] while on its part, Article 22 obliges states to prevent the introduction of alien or new species into international watercourses which may have detrimental effects to the ecosystems of the watercourse and which result in significant harm to other watercourse states.

Article 23 is of particular importance as it seeks to address the land- sea interaction. It provides as follows:

> "Watercourse States shall, individually and, where appropriate, in cooperation with other states, take all measures with respect to an international watercourse that are necessary to protect and preserve the marine environment, including estuaries, taking into account generally accepted international rules and standards".

On its part, Article 24 provides an obligation for States to enter into consultations "concerning the management[36] of an international watercourse, which may include the establishment of a joint management mechanism"

It is remarkable that Article 23 cited above is explicit in its mention of estuarine ecosystems as part of the marine and coastal environment. This provision and others in the 1997 International Watercourses Convention were apparently inspired by similar provisions in the 1982 UN Convention on the Law of the Sea.[37] The latter convention under Article 192 creates a general obligation on States "to protect and preserve the marine environment." Article 194 of the 1982 UN Convention on the Law of the Sea is detailed and generic on measures to prevent, reduce and control pollution of the marine environment. It provides *inter-alia* as follows:

Article 194(1) "States shall take, individually or jointly as appropriate, all measures consistent with this Convention that are necessary to prevent, reduce and control pollution of the marine environment from any source..."

Article 194(2) "States shall take all measures necessary to ensure that activities under their jurisdiction or control are so conducted as not to cause damage by pollution to other states and their environment, and that pollution arising from incidents or activities under their jurisdiction or control does not spread beyond the areas where they exercise sovereign rights in accordance with this Convention."

Article 194 (3) "The measures taken pursuant to this Part shall deal with all sources of pollution."

Thus, all sources and activities causing marine and coastal pollution or environmental degradation, whether land based or sea based, are contemplated under the 1982 UN Convention on the Law of the Sea, as are transboundary pollution problems. Apart from land based sources and activities, which are more prevalent in estuarine ecosystems in the WIO region, sea based sources and activities, such as dumping, pollution from vessels, pollution from or through the atmosphere, installations and related activities are also covered.[38] Of immediate relevance is Article 207 which states as follows in part:

Article 207(1) "States shall adopt laws and regulations to prevent, reduce and control pollution of the marine environment from land based sources, including rivers, estuaries, pipelines and outfall structures, taking into account internationally agreed rules, standards and recommended practices and procedures."

Article 207 (2) "States shall take other measures as may be necessary to prevent, reduce and control such pollution.

Article 207(3) "States shall endeavor to harmonize their policies in this connection at the appropriate regional level".

Thus estuarine ecosystems are also specifically mentioned, alongside rivers and others in connection with prevention of pollution from land based sources. The obligation is for States to adopt relevant laws and regulations to tackle land based sources, including through appropriate regional arrangements. In the WIO region, apart from national legal and other regulatory frameworks, which we shall review in the next section, this provision (Article 207) has arguably inspired the development of the 1985 Nairobi Convention framework, including the adoption of a Protocol on Land Based Sources and Activities (LBSA) in 2010.[39]

The 1971 Wetlands (Ramsar) Convention,[40] is the world's foremost international agreement for the protection and wise use of wetlands, and requires national commitment to the sustainable management of designated wetland sites.[41] The declaration of several estuaries along the WIO coastline

[35] Article 21(1) of the 1997 International Watercourses Convention defines "pollution of an international watercourse" as "any detrimental alteration in the composition or quality of the waters of an international watercourse which results directly or indirectly from human conduct".

[36] Article 24 (2) of the 1997 International Watercourses Convention defines management to include "(a) planning the sustainable development of an international watercourse and providing for the implementation of any plans adopted; and (b) otherwise promoting the rational and optimal utilization, protection and control of the watercourse".

[37] See, generally Part XII of the 1982 UN Convention on the Law of the Sea, and particularly Article 192, Article 194, Article 196 and Article 207.

[38] Ibid, Articles 207, 208, 209, 210, 211, and 212.

[39] Convention for the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region (1985 Nairobi Convention); Protocol concerning Protected Areas and Wild Flora and Fauna in the Eastern African Region; Protocol Concerning Cooperation in Combating Marine Pollution in Cases of Emergency in the Eastern African Region 21 June 1985, IELMT 985; Protocol on Land Based Sources and Activities in the Western Indian Ocean (2010 LBSA Protocol). For the 2010 LBSA Protocol and 2010 Amended Nairobi Convention, see *www.unep.org/nairobiconvention/*

[40] Convention on Wetlands of International Importance (Ramsar) 996 UNTS 245; 11 ILM (1972) 963.

[41] See https://www.savingcranes.org/lower-zambezi-valley-and-delta-program.html, accessed on 12 May 2015.

as Ramsar sites of international importance also imposes various conservation and management duties and responsibilities on the states and other stakeholders.

The Tana River Delta was in 2012 declared as Africa's newest Ramsar site. The Tana Delta forms an area of rich biodiversity for sea species including fish and prawns, five species of marine turtles. There are a host of terrestrial animals such as the African Elephant, Tana Mangabey, Tana River Red Colobus, and White Collared monkey. In addition to more than 600 plant species, the Tana Delta is a home for many bird species and is a critical transit point for migratory water birds such as waders, gulls and terns.[42]

In October 2003, the Government of Mozambique declared the Marromeu Complex of the Zambezi Delta as the first Wetland of International Importance in Mozambique under the 1971 Ramsar Convention.[43] In addition to designating the site, the Government of Mozambique stipulated that an integrated management plan must be developed to ensure the sustainable development of the Marromeu Complex in accordance with the spirit of the 1971 Ramsar Convention.[44]

In South Africa, there are several Ramsar sites along or near the East Coast. These include the Kosi Bay, which is located south of Mozambique on the East Coast, approximately 470 km north of Durban, and which has been described as "the most wonderful aquarium and gorgeous aviary."[45] The Kosi Bay is composed of four interconnected lakes subject to tidal influence, an estuarine channel, and three extensive swamps. Fresh water is derived from three permanent rivers. Principal habitats include swamp and mangrove forest, reedbeds, dune systems with associated woodland, and coastal grassland. The site supports a diverse bottom-dwelling invertebrate fauna (30 species) and a rich fish fauna, including eight endangered species. Several birds, mammals, butterflies, and plants are endemic, threatened or endangered. Large areas of swamp forest have been subjected to non-sustainable slash and burn cultivation practices. Human activities include subsistence farming and fishing.[46]

Other examples of Ramsar sites include the Natal Drakensberg Park in Eastern Kwazulu, along the border between South Africa and the Kingdom of Lesotho; and the Turtle Beaches/coral reefs of Tonga land near Mtubtuba. The beaches are situated south of the Kosi Bay estuary mouth. Finally there is the St Lucia system, in the East Coast in Kwazulu Natal approximately 200 km north of Durban.[47]

The Nairobi Convention Framework

The 2010 Amended Nairobi Convention, though not in force, has carried most of the provisions of the 1985 Nairobi Convention.[48] Among others, it has provisions on pollution from various sources, such as from ships (Article 5), pollution caused by dumping (Article 6), pollution from land based sources and activities (Article 7), and pollution from sea bed activities, and pollution resulting from transboundary movement of hazardous wastes. Of particular significance is Article 7, modeled on Article 207 0f the 1982 UN Convention on the Law of the Sea. It (Article 7) states as follows:

Article 7 " The Contracting Parties shall endeavor to take all appropriate measures to prevent, reduce and combat pollution of the Convention area caused by coastal disposal or by discharges emanating from rivers, estuaries, coastal establishments, outfall structures, or any other land based sources and activities within their territories"

The 2010 Amended Nairobi Convention defines the "Convention area" to include the WIO estuarine ecosystems: it "shall comprise the riparian, marine and coastal environment including the watershed of the Contracting Parties to this Convention. The extent of the watershed and of the coastal environment to be included within the Convention area shall be indicated in each protocol to this Convention, taking into account the objectives of the protocol concerned".[49]

Moreover, in defining "pollution" the 2010 Amended Nairobi Convention makes explicit reference to "river flows... estuaries..."[50] Among the key general provisions is one to the effect that contracting parties "may enter into bilateral or multilateral agreements, including regional or sub-regional agreements, for the protection and management of the marine and coastal environment of the Convention area".[51] Article 4 on general obligations provides for an overall obligation of the contracting parties to "prevent,

[42] See http://wwf.panda.org/?206813/Tana-River-Delta-Ramsar-Site-Status-a-Plus-for-Coastal-East-Afric, accessed 12 May 2015.

[43] https://www.savingcranes.org/lower-zambezi-valley-and-delta-program.html, accessed 12 May 2015.

[44] Ibid.

[45] Campbell, G.G. (1948). Two expeditions to Tongaland. **African wildlife, 2**(3): 53–56, cited in http://www.ewisa.co.za/misc/wetlands/defaultwetKZNkosibay.htm, accessed on 12 May 2015.

[46] See http://ramsar.rgis.ch/cda/en/ramsar-documents-list-anno-southafrica/main/ramsar/1-31-218%5E16187_4000_0_, accessed on 12 May 2015.

[47] http://www.ewisa.co.za/misc/wetlands/defaultwetKZNkosibay.htm, accessed 12 May 2015.

[48] Ibid.

[49] Ibid (Amended 2010 Nairobi Convention), Article 2.

[50] Ibid.

[51] Ibid, Article 3(1)

reduce and combat pollution of the Convention area and to ensure sound environment management of natural resources. . .”

The 2010 LBSA Protocol to the Nairobi Convention has very detailed provisions regarding measures to control, reduce or prevent land based sources and activities, including pollution and physical alterations and destruction of habitats (PADH).[52]

Clearly, the estuarine ecosystems in the WIO region, as part of the wider marine and coastal environment, have a strong regional legal framework for protection and sustainable use.

Regional River Basin Frameworks

River basin agreements primarily provide for two categories of shared watercourse institutions:

- Shared water commissions are essentially advisory bodies providing a forum for notification, consultation and negotiation; for coordinating responses to emergencies; for collecting data; and for setting water-quality targets and standards.
- River basin authorities go further in that they have specific powers granted to them by parties to the shared water agreement.[53]

Among the numerous commissions that govern transboundary waters in the WIO perhaps the best known is the Zambezi River Basin Commission, which is founded on the 1987 Zambezi River System Agreement.[54] It administers the Zambezi River System Agreement to which Botswana, Democratic Republic of the Congo (DRC), Mozambique, Tanzania, Zambia and Zimbabwe are signatories. Apart from the 1987 Zambezi River System Agreement, there is the Protocol on Shared Watercourse Systems in the Southern Africa Development Community (SADC Protocol),[55] which is closely modeled on the 1997 Watercourses Convention. This is perhaps because the 1987 Zambezi River System Agreement preceded the 1997 Watercourses Convention. It has been held that the 1987 Zambezi River System Agreement and its SADC Protocol “represent the most ambitious approach to environmental protection of river basins in the developing world.”[56] They “exemplify the potential of common management in addressing environmental problems”.[57] The 1987 Zambezi River System Agreement provides a comprehensive environmental management programme for the Zambezi River. Under its framework, an Intergovernmental Monitoring and Coordinating Committee provide policy guidance, oversee implementation and evaluate results.[58] On its part, the SADC Protocol represents an attempt to address sustainable development of Southern Africa’s watercourses on a regional basis.[59] Under its framework, the 2002 Incomati River Basin Agreement was the first one to be adopted.[60] And finally there is the **Zambezi River Commission** (ZAMCOM), whose Agreement was signed by most of the Zambezi riparian states.

Among river basin organizations in the WIO region with a mandate to develop and execute joint projects is the Incomati/Komati Basin Water Authority (South Africa, Mozambique and Swaziland). The others are Pangani Basin Water Office (Tanzania), Rufiji Basin Development Authority (Tanzania), Tana and Athi River Development Authority (Kenya) and Zambezi River Authority (Mozambique, Zambia and Zimbabwe). These organizations operate according to a clearly defined mandate[61] for a specific purpose such as shared dam construction or operation, hydropower generation and irrigation. They do not engage in interstate negotiations or formulate policy.

The Limpopo Basin Commission has the following members: Botswana, Mozambique, South Africa and Zimbabwe. Apparently the Limpopo commission has been in the process of being converted into an authority vested with a measure of executive authority donated by the member states which cede some of their sovereign rights.[62]

The foregoing exposition demonstrates that as for international watercourses in the WIO, there is a fairly elaborate legal and institutional framework that seeks to address a range of environmental protection and sustainable use imperatives, including WIO estuarine ecosystems. However, like for the Nairobi Convention

[52] see *www.unep.org/nairobiconvention/*

[53] UNEP/Nairobi Convention Secretariat, 2009. Transboundary Diagnostic Analysis of Land Based Sources and Activities Affecting the Western Indian Ocean Coastal and Marine Environment, UNEP Nairobi, Kenya 378p, p 241.

[54] 1987 Botswana-Mozambique-Tanzania-Zambia-Zimbabwe Agreement. *27 ILM* (1988).

[55] See 40 ILM (2001), 317.

[56] Birnie, P.W, Boyle A.E, Redgwell, *International Law and the Environment*, Third Edition, (2009) Oxford University Press, p 579.

[57] Ibid.

[58] Ibid.

[59] Ibid.

[60] Ibid.

[61] Malzbender and Earle (2007).

[62] UNEP/Nairobi Convention Secretariat, 2009. Transboundary Diagnostic Analysis of Land Based Sources and Activities Affecting the Western Indian Ocean Coastal and Marine Environment, UNEP Nairobi, Kenya 378p, p 242. See also Birnie Boyle and Redgewell (2009); Malzbender and Earle (2007).

framework, this regime does not specify or isolate estuaries but rather treats them as part of international watercourses or river basin systems.

Governance and Institutional Setting: National Frameworks in Selected WIO Countries

National legal, policy and institutional frameworks would be addressed in this section on a case study basis.

Kenya

- **Case Studies: Tana and Athi-Sabaki River Basins**

Both the Tana and Athi Sabaki are national rivers, and not international water courses. It has been reported recently that since the development of the hydro power dams on the Tana River there has been a 56% decline in sediment load to the Tana Delta, leading to erosion of beaches along the Tana Delta/Ungwana Bay and loss of mangroves and wetlands in the delta. The flow regime is apparently highly variable, both year to year and seasonally. Apparently also, although there has been an increase in the use of agro-chemicals in the Tana Basin, expected to lead to increased nutrient levels, the chemical water quality at the delta has not reflected major deterioration at the Tana Delta.[63]

On its part, the Athi Sabaki River has over the past five to six decades experienced a major increase in sediment load at the estuary, attributed largely to poor land use patterns in the catchment with various forms of intensive agriculture leading to soil erosion. Consequences include the siltation of Malindi Bay, deposition of sedimentary matter and debris on the beaches, and degradation of nearby coral reefs and sea grass beds.[64]

Kenya's recently adopted and promulgated Constitution 2010 has provisions on the protection of the environment. Its promulgation fundamentally changed the legal landscape for environmental conservation, management and dispute resolution mechanisms and processes in Kenya. The major highlights of the Constitution implicitly domesticate the requirements of coastal and marine environmental protection, and include principles of environmental law, access to environmental justice and obligations of the state with regard to international legal commitments.[65]

The Constitution of Kenya (2010) reinforces the importance of natural resources and the environment. Chapter 5, on Environment and Natural Resources, contains principles and obligations on the environment; protection and conservation of the environment; enforcement of environmental rights; the use and development of natural resources; agreements relating to natural resources; and environmental legislation. The constitution also provides for the establishment of an environment and land court to address legal disputes related to environmental and land resources and processes.[66]

Kenya's newly devolved system of government calls for collaboration between national and county administrations. The central government has jurisdiction over the use of international waters and water resources, marine navigation, and the protection of the environment and natural resources including fishing and water. The county government is responsible for fisheries and implementing national policies, among others.[67]

The 1999 Environmental Management and Coordination Act (1999 EMCA)[68] is the framework environmental law, and it came into operation in 2002. However, only Section 55 deals specifically with coastal zone issues, including estuarine governance. Under the 1999 EMCA an area of the sea may be declared to be a protected coastal zone. On institutional arrangements, the National Environment Council (NEC), established by Section 4(1) of the EMCA, undertakes policy formulation and provides direction for the purposes of the 1999 EMCA. The National Environment Management Authority (NEMA) was established under the Act as the principal environmental agency to implement policy. It became operational in 2002.

Section 55 mandates NEMA, in consultation with the relevant agencies, to prepare a survey of the coastal zone and an Integrated Coastal Zone Management (ICZM) Policy to encourage effective methods for managing and protecting the marine and coastal environment and its river basins and estuaries. The EMCA imposes stringent penalties for pollution and hazardous-waste dumping. Prosecutions are rare, and regulations envisaged under section 55(6) for the prevention, reduction and control of pollution have yet to be issued.

[63] UNEP/Nairobi Convention Secretariat, 2009. Transboundary Diagnostic Analysis of Land Based Sources and Activities Affecting the Western Indian Ocean Coastal and Marine Environment, UNEP Nairobi, Kenya 378p, p 182.

[64] Ibid.

[65] See generally, Chapter Five of the Constitution of Kenya 2010, and particularly Articles 69,70, 71 and 72.

[66] Ibid, see Article 162 (2) (b) of the Constitution of Kenya 2010.

[67] Ibid, generally Chapter 10, particularly Articles 185, 186, and 187 and Schedule Four of the Constitution of Kenya 2010.

[68] The Environmental Management and Coordination Act No 8 of 1999. This framework law is currently under review to, inter alia, align it to the Constitution of Kenya 2010.

The provisions of Section 55(7) relate directly to Kenya's obligations under UNEP's 1995 Global Programme of Action on Protection of the Marine Environment from Land Based Activities (GPA)[69] for the protection of the coastal environment from land-based sources. It mandates the minister to issue regulations to control pollution in rivers and estuaries from pipeline and outfall structures in vessels, aircraft and other engines used in the coastal zones. In addition, Environmental Impact Assessment (EIA) and Audit Regulations (2003) require the inclusion of environmental management plans in all EIA reports.

The Lakes and Rivers Act Chapter 409 is fairly old legislation, and does not quite address integrated coastal zone and river basin management. It may require review to include the foregoing issues, and so as to be in line with the provisions of section 42 of the 1999 EMCA which expressly outlaws the excavation, damming, pollution, interference of rivers, lakes and dams, unless on authority of the director General of NEMA upon an EIA study. The Act is rather superficial and does not specifically prohibit pollution of rivers or estuarine ecosystems as such from land based sources or activities.

On its part, the Water Act (2002) gives the minister responsible for water powers to gazette catchment areas as protected areas. It outlaws actions that degrade the quality of water in rivers, obviously including estuarine ecosystems. The Water Act 2002 establishes catchment area advisory committees (CAACs), working under the Water Resources Management Authority (WRMA), also established under the Act.

The committees are appointed by the Minister responsible for Water & Irrigation according to Section 16 (3) of the Water Act, 2002. There are six committees each with a membership of up to 15 and representative of the six catchment areas in the country. Their role is to advise the WRMA at the regional level in the appropriate regions for which they are appointed on matters concerning water resources conservation, use and apportionment; grant, adjustment, cancellation or variation of permits; and issues related to water resources management.

WRMA is required by law to enlist the support of CAACs to effectively involve community members and other stakeholders in the implementation of activities meant to protect and conserve water catchment areas. Thus is introduced the concept of multi-level governance of the water catchments in Kenya, and in particular river ecosystems, with direct implications for river flows arriving at estuarine ecosystems downstream. The committees comprise of: ministry representatives or public bodies responsible for matters relating to water resources; representatives of regional development authorities and local authorities falling partly or fully in the catchment; representatives of the business community; representatives of non-governmental organizations (NGOs) involved in water resources management programs and other persons who have demonstrated competence in matters relating to water resources management.[70]

Estuaries have been identified as some of the primary sources or recipients of pollution of coastal zones. Untreated sewage and wastewater discharged into the sea is one of the principal sources of pollution in the coastal zone. NEMA has recently gazetted wastewater regulations.[71]

The Wildlife Conservation and Management Act (2013) broadly provides for the protection of vulnerable ecosystems along the coastal zone through marine protected areas (MPAs) managed by KWS. This invariably includes estuarine ecosystems. This Act mainly focuses on terrestrial wildlife resources, and less so on marine living resources or sea based ecosystems.

Closely associated with wildlife legislation is the Forest Act (2005), which is a substantial improvement on preceding legislation, namely the Wildlife (Conservation and Management) Act cap 376. The Forest Act 2005 established KFS and encourages private-sector and community participation in the management of forests. Mangrove areas and coastal forests, including *kaya* forests, are recognized as areas requiring better management. Community forest associations, enshrined in the Act, engage with the government on sustainable management. The Act prohibits dumping waste in the mangroves. Previous studies have shown a severe loss of vegetation and reduced fish populations from sewage sludge, oil spills and other urban waste.[72]

In lieu of basin wide river commissions which are characteristic for international river courses in the WIO region and elsewhere, the Tana and Athi Rivers Development Authority (TARDA) is the national responsible authority for the Tana and Athi River Basins. TARDA was established under an Act of Parliament, chapter 443 in 1974 (revised 1991). The functions and responsibilities of the Authority are spelt out in TARDA Act chapter 443, Laws of Kenya.

Broadly, the functions encompass integrated planning and coordination of all development projects within the Tana and Athi River Basins and specifically to implement any projects for the purpose of utilization and protection of water and soils of the area covered by its jurisdiction.

[69] See, www.gpa.unep.org

[70] http://www.wrma.or.ke/index.php/77-news/228-wrma-conducts-3rd-catchment-area-advisory-committee-induction.html, accessed 15 May 2015.

[71] UNEP/Nairobi Convention Secretariat (2010) Regional Synthesis Report on the Review of the Policy, Legal and Institutional frameworks in the Western Indian Ocean (WIO) Region, UNEP, Nairobi Kenya 99p, p 14.

[72] Ibid, p 13.

TARDA's area of jurisdiction covers approximately 138,000 km^2, comprising 100,000 km^2 of the Tana Basin and 38,000 km^2 of the Athi Basin. This includes most of the former Central province, the southern districts of the former Eastern province, the riverine portion of the former North Eastern Province along the Tana River (Garissa and Ijara Districts) and parts of the former Coast province where both the Tana and Athi rivers drain into the Indian Ocean.[73]

TARDA is divided into three zones namely:-

1. Upper Zone – Catchment areas of Mt Kenya and Aberdares region to Lower reaches of Murang'a county
2. Middle Zone – Catchment areas in the lower reaches of Murang'a up to Garissa
3. Lower Zone – Catchments areas below Garissa town, the Sabaki area and the coastal region[74]

The Authority is mandated with many functions, which include to advise the Government generally and the Ministries set out in the schedule in particular on all matters affecting the development of the area of jurisdiction including the apportionment of resources; to draw up and keep up to date, a long-range development plan for the area of jurisdiction; and to initiate such studies, and to carry out such surveys of the Area as it may consider necessary, and to assess alternative demands within the Area on the resources thereof, including electricity power generation, irrigation, wildlife, land and other resources and, to recommend economic priorities.

Other functions include: to co-ordinate the various studies of, and schemes within the Area, so that human, water, animal, land and other resources are utilized to the best advantage, and to monitor the design and execution of planned projects within the area; to effect a programme of monitoring of the performance of projects so as to improve that performance and establish responsibility therefore and to improve future planning; to ensure close co-operation between all agencies concerned with the abstraction and use of water; to collect, assemble and correlate all such data related to the use of water and other resources as may be necessary for efficient forward planning; to maintain a liaison between the Government, the private sector and foreign agencies in the matter of the development of the area of jurisdiction with a view to limiting the duplication of effort and to assuring the best use of technical resources; and to cause the construction of any works necessary for the protection and utilization of the water and soils of the area of jurisdiction.[75]

The primary mandates of the TARDA are therefore largely developmental and less so environmental. Fortunately they cover the entire basins of the Tana and Athi rivers.

The Coast Development Authority Act (1990) established the Coast Development Authority to coordinate development projects in the coastal zone and in the EEZ. Its institutional mandate definitely covers estuarine ecosystems in the coastal zone, but perhaps in duplication or overlapping with TARDA. It covers most of the upstream areas connected with land-based sources and activities, up to the outer limits of the coastal zone inland.

Other relevant legislations include The Kenya Maritime Authority Act (2006) is reinforced by the Merchant Shipping Act (2009) and any other legislation relating to the maritime sector. It established the Kenya Maritime Authority, which advises the government on legislative and other measures for implementing international conventions, protocols and agreements. It also safeguards the marine environment from pollution and responds to marine environment incidents.

Others are the Fertilizers and Animal Foodstuffs (Amendment) Bill (2013) which will regulate the use of POPs; the Public Health Act,(1986, revised 2012) Cap 242; the Pharmacy and Poisons Act,(1957) Cap 244; and the Narcotic Drugs and Psychotropic Substance Control (1994). However these laws may need to be amended to provide for the reduction and elimination of POPs containing dioxins, furans, hexaclorobenzene, and polycyclic aromatic hydrocarbons.

On policy, there are a number of relevant instruments relevant to estuarine ecosystem governance in Kenya generally.

The National Oceans and Fisheries Policy (2008) is rooted in the provisions of the 1982 UN Convention on the Law of the Sea, the Maritime Zones Act (1989), Section 5 and the Presidential Proclamation of June 2005. It affirms Kenya's sovereignty over the exploration, exploitation, conservation and management of ocean resources. It focuses on resource management in territorial waters and the EEZ. It addresses most aspects of fisheries management and development, including environmental conservation, regional cooperation, research, surveillance and monitoring, social responsibility and governance. The preparation of specific fishery management plans is given high priority, including in estuarine ecosystems, but certain regulations need to be adapted to allow for these plans to be effective.[76] This policy

[73] http://www.tarda.co.ke/about/

[74] Ibid.

[75] Ibid, The TARDA Act chapter 443 Laws of Kenya. (This law and other similar development authority legislations are under review).

[76] Samoilys M, Osuka K, Maina GW (2011) Opportunities and challenges of current legislation for effective conservation in the Tana Delta: Pate Island coast of Kenya, CORDIO Status Report (Mombasa, Kenya). p. 5–9.

and the ICZM Policy and Action Plan (2010–2014) have similar objectives and should be harmonized to avoid duplication.

The ICZM Policy brings together all those involved in the development, management and use of the coastal zone within a framework that facilitates the coordination and integration of activities and decision-making processes. The ICZM Action Plan is arguably a first for Kenya as it protects fragile ecosystems while pursuing sustainable development. Its thematic areas are integrated planning and coordination; sustainable economic development; conservation of coastal and marine environment; environmental risks and management of shoreline change; capacity building, information and public participation; and implementation through institutional and legal frameworks.

The Draft National Environment Policy (2012) seeks to align sectoral policy with the EMCA. It is a framework for integrating environmental considerations into sectoral policies, development plans and decision-making processes and for regional and international cooperation in environmental management. It calls for sustainable management of terrestrial and aquatic resources to raise the livelihoods and standard of living for coastal communities.

The Draft Wetland Policy (2009) recognizes the economic importance of coastal, marine and inland wetlands and proposes stringent measures to counter the (primarily human) threat to their long term sustainability. Its integrated approach complements other sector policies and fulfils Kenya's obligations under the 1971 Ramsar Convention[77] and other multilateral environmental agreements and protocols. Unfortunately, even with the anticipation of a policy framework, no clear legal framework governs wetland conservation and management. Different aspects are handled by different agencies, including Kenya Wildlife Services (KWS), Kenya Forest Services (KFS), National Environment Management Authority (NEMA), the State Department of Fisheries, water sector institutions, regional development authorities and communities[78]

The National Land Policy (2009) underpins a system of land administration and management that allows all citizens to gain access to land and to use it. It calls for the equitable and environmentally sustainable use of land resources and requires policies, regulations and laws to be aligned with the EMCA. Its guidelines for formulating land use and management practices take into account the fragile nature of the coastal zone, including estuarine ecosystems. As land use has major implications for the coastal and marine environment, reform in land tenure is imperative for achieving the goals set out in the Land Policy and the ICZM Action Plan among others.

The Regional Development Authorities Policy (2007), which underpins TARDA and other regional development authorities, calls for equitable socio-economic development through the sustainable use of natural resources by formulation of integrated regional development plans in consultation with all those involved, closure of gaps in regional resource mapping and attraction of resource-based investment that benefit communities. The policy is the framework for streamlining and strengthening the Coast Development Authority (CDA) and TARDA in coastal zone development and management.[79]

There are also national over arching policy instruments which address the themes of national socio-economic development and poverty reduction and transformation. They include the National Poverty Reduction Plan (1999–2015); the Poverty Reduction Strategy Paper launched in 2001; Economic Recovery Strategy (ERS) 2003–2007), and Kenya's Vision 2030. The latter cites environmental degradation as a cause of poverty and argues for environmental protection.

Mozambique

- **Case studies : Zambezi, Incomati, and Limpopo**

The Zambezi, Incomati and Limpopo rivers suffer flow alterations and sediment loading alongside other rivers in the WIO region.[80] As transboundary river basins, their governance has already been covered above. However, it is of interest what provisions exist in national legislation in Mozambique, and how they relate to the transboundary river basin regimes discussed elsewhere above.

However, some relevant science and factual detail is appropriate. For Zambezi River, the construction of the Kariba and the Cahora-Bassa dams has had the dual impact of reducing sediment load and reducing the seasonal variability of flows on the Zambezi Delta.[81] It has been reported that "in the case of the Zambezi River the construction of Cahora Bassa Dam in Mozambique has had a

[77] Convention on Wetlands of International Importance (Ramsar) 996 UNTS 245;11 ILM (1972) 963.

[78] Samoilys M, Osuka K, Maina GW (2011) Opportunities and challenges of current legislation for effective conservation in the Tana Delta: Pate Island coast of Kenya, CORDIO Status Report (Mombasa, Kenya). p. 5–9.

[79] Ibid.

[80] UNEP/Nairobi Convention Secretariat, 2009. Transboundary Diagnostic Analysis of Land Based Sources and Activities Affecting the Western Indian Ocean Coastal and Marine Environment, UNEP Nairobi, Kenya 378p, p 193.

[81] Ibid, p 183.

significant impact on fisheries, particularly along the Sofala Bank at the river mouth."[82] Other impacts include a large reduction in the extent of the mangrove forests and reduced sediment loads which in turn lead to increased coastal erosion.[83]

The flow regime of the Incomati River has been significantly altered due to the heavy utilization of its water resources. Water quality has also reduced in some areas, but flow alteration accounts for the biggest impact on the system and the estuary.[84] The estuary has suffered from reduced freshwater inflows and sediment deposition, leading to salt intrusion and sediment deficit, and consequently erosion.[85]

The Limpopo River suffers heavy water use by the four basin states (Botswana, South Africa, Mozambique and Zimbabwe). Agriculture (large and small scale), mining, industry, energy production and urban water use are all significant water users of this large river.[86] The large number of dams in the basin (estimated at over 40), coupled with direct abstractions has significantly reduced the annual flow of the river, reducing the potential of the river to absorb pollutants and contributing to poor water quality, increased seawater intrusion inland that affects agriculture, and the spread of sediment in the near shore sea.[87]

Article 90 of the Constitution of Mozambique establishes the right to live in a balanced environment and invests the state, local authorities and environmental associations with responsibility for its care and protection.[88]

Colonial-era legal instruments are gradually being replaced, but much has yet to be done.

Mozambique has a number of legal instruments on environmental protection, but enforcement and compliance are generally weak.[89] Portugal enacted the Decree Law no. 495 (1973) to protect coastal and marine environments in its then overseas provinces, which included Mozambique. It prohibits, except by special licence, the disposal of pollutants in areas under the jurisdiction of the maritime authority.

The Sea Law (1996) sanctions conservation of marine areas and species by creating marine national parks, marine nature reserves and marine protected areas. It is consistent with the 1982 UN Convention on the Law of the Sea, which Mozambique has ratified. It has far greater regulatory competence than the Environment Law to protect and preserve the maritime and coastal environment.[90]

The Environment Law (1997) defines the legal basis for sustainable management of the environment by the public and private sectors. It does not include specific provisions for coastal and marine environmental protection, but it is an instrument for enacting relevant regulations.

The Water Law provides for a system of devolved water resources management thereby offering better protection to river ecosystems upstream all the way to the coastal estuarine ecosystems. Thus river basin governance is devolved to the basins and community and other stakeholder participation is enhanced. In terms of the geographical distribution, five regions have been identified, bringing together contiguous river basins and they will be administered by the Regional Water Administration (ARAs), set up by the water law:

- ARA – South – includes all the basins south of the Save, and the Save river basin itself.
- ARA – Centre – covers all the basins between the Save and Zambezi basins.
- ARA – Zambezi – corresponds to the Zambezi river basin.
- ARA – Centre-North – covers the Zambezi basin as far as Lúrio river, including the Lurio basin.
- ARA – North – Covers all the basin north of the Lurio basin.[91]

The National Strategy for Sustainable Development derives from the 2002 World Summit for Sustainable Development and integrates recommendations from the Johannesburg Plan of Implementation into the national agenda. It is an important national initiative rooted in local knowledge, local ideas, local expertise and local solutions. The priority areas are biodiversity conservation, land degradation, health, education, agriculture, water, energy and technology transfer. The National Council for Sustainable Development and the Council of Ministers are responsible

[82] Ibid.

[83] Ibid.

[84] Ibid, p 184.

[85] Ibid.

[86] Ibid.

[87] Ibid.

[88] Mazivila R (2009) National report on legal, regulatory and institutional framework for land-based sources and activities management in Mozambique. Unpublished report submitted to UNEP/WIO-LaB Project/Nairobi Convention Secretariat, Nairobi, Kenya. 72 pp.

[89] UNEP/Nairobi Convention Secretariat (2010) Regional Synthesis Report on the Review of the Policy, Legal and Institutional frameworks in the Western Indian Ocean (WIO) Region, UNEP, Nairobi Kenya 99p, p 21.

[90] Mazivila R (2009) National report on legal, regulatory and institutional framework for land-based sources and activities management in Mozambique. Unpublished report submitted to UNEP/WIO-LaB Project/Nairobi Convention Secretariat, Nairobi, Kenya. 72 pp.

[91] Taucale Fransisco (2012): Water Resources of Mozambique and the Situation of the Shared Rivers, in www.unep.org/…/fransisco.doc, accessed on 15 May 2015.

for its implementation. Once approved, it will be incorporated into all sectoral plans.[92]

The National Environmental Policy is the principal planning instrument for the environment sector. It calls on the state to provide incentives for the sustainable use of natural resources[93] It integrates environmental issues into economic planning, recognizes the role of the communities in environmental management and monitoring, and acknowledges a role for the private sector in managing the environment. It also defines the strategy that provides the framework for the Ministry for Coordination of Environmental Affairs (MICOA) and recommends multi-sectoral coordination.[94]

The Strategic Plan for the Environmental Sector (2005–2015) combines the Action Plan for the Fight against Drought and Desertification, the Strategy for Urban Environment Management, the Coastal Zone Management Strategy, the Strategy and Action Plan Controlling the Fight against Soil Erosion, the Strategy to Combat Deforestation and Burning, the Urban Solid Wastes Integrated Management Strategy, the Hazardous Wastes Management Strategy, the Biodiversity Strategy, and the Action Plan for Biodiversity Conservation.[95] Its priority areas are biodiversity conservation, land degradation, health, education, agriculture, water, energy and technology transfer.

Policy instruments relevant to environmental management (not restricted to coastal areas) are the National Action Plan to Combat Desertification and Drought; the National Forests and Wildlife Policy and Strategy; the National Tourism Policy and Strategy; the National Fisheries Policy; the National Land Policy; the Agrarian Policy; the National Water Policy; and the Strategy and Action Plan for Biodiversity Conservation in Mozambique.[96] Others are the Energy Policy and Strategy (1998); the National Environmental Policy (1995); the Policy for Disaster Management (1999); the National Policy for Land Use Planning (1996); the National Action Program for Adaption to Climate Change; the Policy (1996) and Strategy (2006) for Meteorology Development; and the Conservation Policy and Implementation Strategy (2009).

A good example of a catchment based management system in Mozambique is the Marromeu Complex of the Zambezi Delta. In December 2003 the Museum of Natural History and International Crane Foundation were approached by Mozambique's Zambezi Valley Planning Authority *(Gabinete do Plano de Desenvolvimento da Região do Zambeze)* and the Ministry for Coordination of Environmental Affairs *(Ministério para a Coordenção da Acção Ambiental)* to coordinate the first integrated management plan for the newly designated Marromeu Complex of the Zambezi Delta Ramsar site. The project was to be implemented through collaboration with formal and emergent community leaders, management authorities (National, Provincial and District decision-makers) and stakeholders (including safari operators, sugar plant managers, foresters, aid organizations, prawn industry representatives, small farmers, subsistence fishers, others), with emphasis on participatory management in natural resource decision-making. It requires the integration of wetland conservation and wise use with broader river basin management objectives as defined by the Zambezi Basin Action Plan. Through this process, the project would offer a tremendous opportunity to promote the sustainable use of natural resources while protecting and restoring the ecological integrity of the Zambezi Delta. The project also offered the opportunity to advance integrated river basin management for the entire Zambezi River basin.[97]

South Africa

- **Case study: Thukela River**

The Thukela River is not transboundary. The Thukela and its estuary which had until recently remained relatively stable, has suffered reduced flows and enhanced sediment loads from increased number of dams, inter-basin transfer infrastructure such as pipelines and canals.[98] The effect of the dams on floods and sediment dynamics at the estuary has over the past decade or so become an issue of increasing concern, with the estuary now closing for days at a time, whereas historically the system was permanently open to the sea.[99] There are also increasing challenges of declining water quality, including at the estuary especially when flow rates are reduced.[100] Reduced flow is also likely to accentuate the pollution problems that are increasingly being experienced.

The Constitution of South Africa has some provisions relevant to the regulation of the marine and coastal environment, which includes estuarine ecosystems. The Bill of

[92] Ibid.

[93] Ibid.

[94] Ibid.

[95] Ibid.

[96] Ibid.

[97] https://www.savingcranes.org/lower-zambezi-valley-and-delta-program.html, accessed 12 May 2015.

[98] UNEP/Nairobi Convention Secretariat, 2009. Transboundary Diagnostic Analysis of Land Based Sources and Activities Affecting the Western Indian Ocean Coastal and Marine Environment, UNEP Nairobi, Kenya 378p, p 189.

[99] Ibid.

[100] Ibid.

Rights in the constitution includes environmental rights (section 24). This right includes having an environment that is not harmful to their health or well-being and to have the environment protected including through pollution prevention and ecological degradation, promotion of conservation and securing ecologically sustainable development. In addition Chapter 3 of the Constitution provides for Cooperative Government, including the framework for the national, provincial and local government. This is a fairly complex system of national and environmental governance which seeks to demarcate the respective legislative and executive mandates of the three levels of government in South Africa.[101]

South Africa currently has two environmental framework laws, namely the Environment Conservation Act 73 of 1989 (ECA) which was enacted prior to the transition from apartheid to democracy, and the National Environmental Management Act 107 of 1998 (NEMA). Many of the provisions in the 1989 ECA have been subsumed in the 1998 NEMA, and in due time all of 1989 ECA will be subsumed in the 1998 NEMA. The latter is primarily concerned with cooperative environmental governance. In section 2(4) the Act lays down national environmental management principles, including for the protection of marine and coastal area management. These include the principle that sensitive, vulnerable, highly dynamic or stressed ecosystems, such as coastal shores, estuaries, wetlands and similar systems require specific attention in management and planning procedures, especially when they are subject to significant human resource usage and development pressure.[102] The Act also provides for the public trust doctrine, to the effect that “the environment is held in public trust for the people, the beneficial use of environmental resources must serve the public interest and the environment must be protected as the people’s common heritage”(Section 2(4) (o)).

Perhaps the most important legislation for the coastal ecosystems including estuaries is the National Environmental Management: Integrated Coastal Management Act No 24 of 2008, which repealed most of the Sea shore Act 21 of 1935, and the Dumping at Sea Act No 73 of 1980. The Integrated Coastal Management Act has detailed provisions on the protection of the coastal and marine environment, including by domestication of South Africa’s international obligations concerning the coastal and marine environment. It defines “coastal waters” to include estuaries. Chapter 4 of the Act (sections 33, 34) deals with estuaries specifically. The Act obliges the establishment of national estuarine management protocol (section 33) and estuarine management plans (section 34) respectively. According to the Act, estuaries must be managed in a coordinated and efficient manner and in accordance with a national estuarine management protocol, to be established by the responsible Minister in consultation with the minister responsible for water resources. The protocol must articulate strategic vision and objectives for achieving integrated management of estuaries as well as setting standards for management of estuaries. Moreover, there shall be established an estuarine management plan following a public consultation/participation process, consistently with the national estuarine management protocol, as well the national coastal management programme. The latter could be cascaded and adapted by the provincial and municipal levels of government.

There is no doubt that the Integrated Coastal Management Act No 28 0f 2008 is ground breaking and transformative in the management and protection of the coastal and marine environment generally, and for estuarine ecosystems in the present context. Although relatively new, its impact and influence could well go beyond South Africa to the rest of the WIO region and beyond. As noted above, it makes provision for both national level and provincial and municipal/local governance of the coastal and marine environment.

The National Environmental Management Biodiversity Act 10 of 2004 provides for the management and conservation of South Africa’s biodiversity and its components; the protection of species and ecosystems that warrant national protection, including marine and coastal protected areas; the sustainable use of indigenous biological resources among others.

Other laws include the Disaster Management Act 57 of 2002; Maritime Zones Act 15 of 1994;; the Water Services Act 108 of 1997; Fertilizers, Farm Feeds, Agricultural Remedies and Stock remedies Act 36 of 1983; National Environmental Management Protected Areas Act 57 of 2003; and Conservation of Agricultural Resources Act 43 of 1983.[103]

Under the National Water Act 36 of 1998, the Minister of Water and Sanitation Affairs in 2012 established nine Catchment Management Agencies (CMAs) in South Africa. The CMAs will play a critical role in managing the country’s scarce water resources, including facilitating stakeholder input into the management of water resources. The establishment of the CMAs assists in achieving full implementation of the National

[101] UNEP/Nairobi Convention Secretariat (2010) Regional Synthesis Report on the Review of the Policy, Legal and Institutional frameworks in the Western Indian Ocean (WIO) Region, UNEP, Nairobi Kenya 99p, p 27.

[102] Ibid.

[103] Ibid, p 28.

Water Act 1998 and to maintaining the sustainable use of the nation's water resources in line with the national development imperatives of government. Until the establishment of the CMAs, the delegation of water management functions to the catchment level had only been partially implemented since the promulgation of the National Water Act in 1998. The Minister decided to reduce the number of CMAs to nine from the original proposal of 19 CMAs, due to a number of reasons including the technical capacity required to staff CMAs, and the challenges such a large number of institutions would pose to the Department of Water and Sanitation Affairs in regulating their performance. The Minister is also required to ensure that stakeholders in the various water management areas are engaged in the process of establishing the CMAs as part of democratization of water management in the country.[104]

On institutions, the Department of Environmental Affairs administers the Integrated Coastal Management Act No 24 of 2008, the Environment Conservation Act 73 of 1989, as well as the National Environmental Management Act 108 of 1998. Its Marine Pollution Division is responsible for various aspects of marine pollution, including clean-up of spills once they hit the sea or sea-shore. The Department of Environmental Affairs has at its disposal a number of vessels and aircraft to enforce the various laws which it administers.[105]

The Department of Water and Sanitation Affairs administers the National Water Act 36 of 1998 and Water Services Act 1997. The Department's Chief Directorate has responsibility for: water use and conservation and water quality management. The latter in turn has four relevant sub-directorates: urban development and agriculture, mines; waste management; and industries. The Directorate of Water Quality is responsible for water quality generally and thus for pollution of the marine environment from land-based sources, including point sources (for example, effluent pipelines out to sea) and non-point sources (for example, seepage)[106]

The Department of Minerals and Energy administers the Mineral and Petroleum Development Act 28 of 2004. It grants prospecting and mining authorizations to mine whether terrestrially or off-shore. These authorizations could include conditions relating to pollution of coastal and marine waters.[107]

The four coastal provinces administer certain legislation assigned to them. For example the administration of most the provisions retained in the repealed Sea-shore Act 21 of 1935 has been assigned to the coastal provinces. The administration of land based pollution at provincial level is not straightforward for two reasons. First because as far as the two key national departments, Department of Environmental Affairs and Department of Water and Sanitation Affairs are concerned, the former has provincial offices in the (coastal) provinces while the latter does not, only regional national offices. Secondly, the 'place' of the provincial departments of environmental affairs is not consistent in the various provinces. Thus the location of the provincial departments of environmental affairs in the four coastal provinces is as follows:

– KwaZulu-Natal – Department of Traditional and Environmental Affairs;
– Eastern Cape – Department of Economic Affairs, Environment and Tourism;
– Western Cape – Department of Environment and Development Planning;
– Northern Cape – Department of Agriculture, Nature Conservation and Land Reform; and Department of Health and Welfare and Environmental Affairs

Coastal local authorities play an important role in the administration and monitoring of marine pollution rules and regulations of their respective coastlines. Many of the provisions of the Sea Shore Act described above have been delegated to coastal local authorities.[108]

The key relevant policy instruments include the *1998 White Paper on an Environmental Management policy for South Africa* which underpinned the development and enactment of the National Environmental Management Act (NEMA) No 107 of 1998. The other policy instrument is the 2000 *White Paper for Sustainable Coastal Development in South Africa*, which underpinned the development of the Integrated Coastal Management Act No 24 of 2008.[109]

[104] See https://www.dwa.gov.za/Communications/PressReleases/2012/Media%20release%20Catchment%20Management%20Agencies%20March%20%20%2030%202012.pdf

[105] UNEP/Nairobi Convention Secretariat (2010) Regional Synthesis Report on the Review of the Policy, Legal and Institutional frameworks in the Western Indian Ocean (WIO) Region, UNEP, Nairobi Kenya 99p, p 55.

[106] Ibid, p 56.

[107] Ibid.

[108] Ibid.

[109] Ibid, p 47.

Threats, Challenges and Opportunities Facing Estuarine Ecosystem Governance in the WIO Region

In spite of the fairly elaborate governance frameworks described above, estuarine ecosystems in particular, and coastal and marine environment generally face threats and challenges.

Environmental problems and threats afflicting estuarine ecosystems are mainly land based, and to some extent sea based where the land-sea interaction exists. Estuarine ecosystems naturally become downstream recipients of river pollution and nutrient loading, which mainly originates from industrial effluent, agricultural runoff, or domestic sewage discharge.[110] The environmental impacts and consequences in the WIO estuarine ecosystems include increased salinisation of agricultural land, modification of ecosystem community structure and dynamics, loss of wetlands due to lack of flooding, decreased productivity, changes in habitats of terrestrial fauna, and reduction of available natural living resources (fauna and flora). Others are loss of biodiversity, reduced assimilative and dilution capacity of freshwater inflows leading to water quality degeneration, and water quality degeneration due to increased pollutant loads,[111] and climate change among others.

The main problems and issues related to governance of coastal and marine environment of the WIO Region include: policy and legislative inadequacies, limited institutional capacities, inadequate awareness, inadequate financial resources and mechanisms, as well as poor knowledge management.[112]

Some of the key challenges include better management of international rivers, management of dams and reservoirs in the various river basins, the multiplicity of legal, policy and institutional arrangements for ecosystem governance, even as most of the instruments are not specific to estuaries; and the sheer complexity of some of the regimes established. It is noteworthy that there are several international rivers in the WIO region, and, as discussed above, there are governance frameworks established. However, the effective implementation and enforcement of the international frameworks, together with the corresponding national frameworks continues to be challenging. The multiple uses of upstream waters in all these rivers makes the implementation and enforcement of environmental and sustainable development goals naturally difficult. As noted in previous sections above, there is no shortage of river basin uses or of legal, policy and institutional arrangements.

Some of the emerging opportunities for better governance of the coastal and marine environment of the WIO region include better understanding of the various causes and impacts of pollution and degradation of the coastal and marine environment as borne by the numerous scientific and technical studies[113]; promising climate change regulation and mitigation interventions; and better legal, policy and institutional frameworks which emphasize integration, ecosystem or basin wide approaches, and sustainable development. Arguably the most illustrious in the WIO region is South Africa's Integrated Coastal Management Act No 24 of 2008. There is also increasing realization at the policy making level both in the WIO region and individual countries of the importance of linking socio-economic development to environmental well being across all sectors, including downstream such as coastal and marine areas.

Conclusions and Key Recommendations

Conclusions

From the foregoing discussion, it is apparent that estuarine ecosystems are vitally important but fragile ecosystems. They provide cultural, provisioning and regulatory services for local communities, countries and the entire region. There is also adequate scientific and technical understanding of the various estuarine ecosystems in the region, including

[110] Birnie, P.W, Boyle A.E, Redgwell, *International Law and the Environment,* Third Edition, (2009) Oxford University Press, p 547; UNEP/Nairobi Convention Secretariat, 2009. Transboundary Diagnostic Analysis of Land Based Sources and Activities Affecting the Western Indian Ocean Coastal and Marine Environment, UNEP Nairobi, Kenya 378p, p178.

[111] Ibid (UNEP/Nairobi Convention Secretariat, 2009. Transboundary Diagnostic Analysis of Land Based Sources and Activities Affecting the Western Indian Ocean Coastal and Marine Environment, UNEP Nairobi, Kenya 378p, p195, 196).

[112] Ibid.

[113] See, for example, UNEP/Nairobi Convention Secretariat, 2009. Transboundary Diagnostic Analysis of Land Based Sources and Activities Affecting the Western Indian Ocean Coastal and Marine Environment, UNEP Nairobi, Kenya 378p; AFD/IDDRI (2011): Sick Seas and Oceans and the Challenges of Combating Land Based Pollution and Degradation: The Example of Western Indian Ocean Region, in A Planet for Life, (2011); UNEP, 2009: Regional Synthesis Report On Legal, Regulatory and Institutional Frameworks in the WIO Region. UNEP/GEF WIO-LaB Project Report. *114p UNEP/GEF/WIO-LaB/LTR/2009;* UNEP, 2009: Regional Synthesis Report on Ratification and Implementation of International Environmental Conventions relevant to Land based Activities/Sources of Pollution of the Coastal and Marine Environment of the WIO Region, UNEP/GEF WIO-LaB Project Report. *77p UNEP/GEF/WIO-LaB/LTR/2009;* UNEP/GPA Government of Kenya/NEMA, 2009: State of Coast Report 2009; 2004: "Physical Alteration and Destruction of Habitats in the Marine and Coastal Environment of Eastern Africa: Legal and Institutional Issues

governance frameworks that affect them. However, studies specific to estuarine ecosystem governance are not abundant.

The global, regional, national and local legal, policy and institutional arrangements abound, but they generally fall short of effective protection of the estuarine ecosystems alongside the rest of the coastal and marine environment. Remarkably, they are hardly specific on estuaries as such, with the exception of South Africa's Integrated Coastal Management Act No 24 of 2008, which dedicates its chapter 4 to estuaries. From the case studies including Kenya, Mozambique and South Africa, it is clear that more could be done to make national frameworks more specific, integrated and effective. In this regard South Africa provides a model worthy considering for replication in the other countries of the WIO region. This should help address the various threats and challenges facing the coastal and marine environment of the WIO region, including the estuarine ecosystems. It is time to re-think estuarine ecosystem governance in the WIO region.

Recommendations

i. There should be further studies and reviews on estuarine ecosystems and their complex interactions with other ecosystems, and their governance arrangements particularly in the context of land-sea interaction. The governance studies should include both international and national frameworks concurrently or separately, and these studies should together provide further justification and momentum for rethinking ecosystem governance in the WIO region.
ii. International legal, policy and institutional frameworks, and particularly regional and sub-regional frameworks, should be reviewed to address lack of specificity on estuarine ecosystems and to improve their governance arrangements. This is more so considering that estuarine ecosystems usually traverse the land –sea interaction and are affected by both land based and sea based sources and activities causing pollution and degradation.
iii. National legal, policy and institutional frameworks, including regional or local frameworks, should also be reviewed to address better specificity, integration and coordination where these attributes are lacking. The relevant environmental and resource use/sector laws and policies should be more specific and integrated on estuarine ecosystems, and the responsible institutions and agencies should be better integrated and coordinated.

References

Abuodha PAW, Kairo JG (2001) Human induced stresses on mangrove swamps along the Kenyan coast. Hydrobiologia 458:255–265

Africa Biodiversity Collaborative Group (2012) Climate change in the Western Indian Ocean: a situation assessment and policy considerations, Washington, D.C: USAID-Bureau for Africa, 97 pp

Birnie P, Boyle A, Redgwell C (2009) International law and the environment, 3rd edn. Oxford University Press, New York

Cinner JE, Wamukota A, Randriamahazo H, Rabearisoa A (2009) Toward institutions for community-based management of inshore marine resources in the Western Indian Ocean. Mar Policy 33:489–496

Kimball L (1996) Treaty implementation: scientific and technical advice enters a new stage. ASIL Studies in Transnational Legal Policy no. 28, 1996, Washington DC, pp 99–100

Malzbender D, Earle A (2007) The impact and implications of the adoption of the 1997 UN Watercourse Conventions for Countries in Southern Africa. Africa Centre for Water Research, Cape Town, South Africa, 244 pp

Mazivila R (2009) National report on legal, regulatory and institutional framework for land-based sources and activities management in Mozambique. Unpublished report submitted to UNEP/WIO-LaB Project/Nairobi Convention Secretariat, Nairobi, Kenya, 72 pp

Mgaya YD, Juma S (2001) Integrated coastal zone management in Tanzania. In: Voabil C, Engdahl S (eds) The voyage from Seychelles to Maputo: successes and failures of integrated coastal zone management in Eastern Africa and Island States 1996–2001, vol I, SEACAM. Maputo, Mozambique, pp 123–149

Obura D (2001) Kenya. Mar Pollut Bull 42:1264–1278

Samoilys M, Osuka K, Maina GW (2011) Opportunities and challenges of current legislation for effective conservation in the Tana Delta: Pate Island coast of Kenya, CORDIO status report (Mombasa, Kenya), pp 5–9

Sands P (1994) Principles of international environmental law, vol 1. Cambridge University Press, Cambridge, pp 81–82

Sands PH (2007) The evolution of international environmental law. In: Bodansk D, Brunnee J, Hey E (eds) The Oxford handbook of international environmental law. Oxford University Press, Oxford/New York, pp 30–43

Sands P, Klein P (2001) Bowett's law of international institutions, 5th edn. Sweet & Maxwell, London, pp 84–85

UNEP (2002) Global environment outlook 3: *past, present and future perspectives*. UNEP, Earthscan Publications Ltd, London. Sterling, VA

UNEP (2006) African Environmental Outlook 2: *Our environment, Our Wealth*, UNEP, Progress Press Ltd, Malta

UNEP (2007) Global Environment Outlook 4: environment for development. Progress Press Ltd, Valletta, Malta

UNEP/Nairobi Convention Secretariat (2009a) Trans-boundary diagnostic analysis of land-based sources and activities affecting the Western Indian Ocean coastal and marine environment. UNEP, Nairobi, pp 226–240

UNEP/Nairobi Convention Secretariat and WIOMSA (2009b) Regional review of policy, legal and institutional frameworks for addressing land-based sources and activities in the Western Indian Ocean region. UNEP, Nairobi, pp 10–56

WIOMSA (2011) Climate change impacts in coastal and marine areas of the Western Indian Ocean region: an assessment of problems, solutions, and strategic options for promoting climate resilient development in the WIO Region. WIOMSA, Zanzibar, 161 pp

World Resources 2000–2001: People and ecosystems: the fraying web of life, World Resources Institute, Washington, D.C.

Mapping Factors That Contribute to Coral Reef Resilience Using In situ and Satellite Data in East Africa

Denis Macharia, Gabriel Grimsditch, Ameer Abdulla, and David Obura

Abstract

Understanding factors that promote coral reef resilience to climatic and anthropogenic stressors is required in order to develop methods and decision support systems to establish resilient Marine Protected Areas (MPAs) and other marine managed areas. This study presents an analysis of coral reef resilience factors obtained from a rapid assessment of coral reefs in 5 locations along the East African coast. The sites span more than 600Km along the coastline of Kenya and Tanzania and are subjected to varying environmental conditions. The study also attempts to present an approach to mapping reef resilience factors and their integration into planning or decision support tools that inform management actions. The analysis revealed that coral reef resilience is highly influenced by biological and anthropogenic factors. Highly resilient reefs were found in areas with high scores for biological factors and low anthropogenic activities. It also revealed that areas with higher cumulative thermal stress and lower levels of pollution from terrestrial sources had higher overall resilience; whilst terrestrial pollution was a major limiting factor on coral reef resilience in the region. Interestingly, the results reveal that reefs with higher resilience are also found in more populated areas compared to reefs in marginal areas that were found to have relatively lower resilience scores. Although this correlation is the weakest compared to other correlations, it could imply that coral reefs found in these highly populated areas are at risk of degradation in the future. However, it is noteworthy that these reefs were those that are within already established MPAs. In order to anticipate and plan for future likelihood of degradation of these reefs, results from this study are proposed to assist in identifying and prioritizing alternative reefs for conservation and management away from such high population density areas. We conclude that incorporating coral reef resilience factors into decision support tools such as GIS can inform management actions aimed at conserving reef ecosystems.

Keywords

Climate change • Coral reef resilience • Marine spatial planning • Geographic Information Systems • Marine Protected Areas

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DOI 10.1007/978-3-319-25370-1_16

Introduction

Coral reefs are critically important for the ecosystem goods and services they provide to many maritime tropical and subtropical nations (Moberg and Folke 1999). Yet coral reefs are in serious decline; an estimated 30% are already severely damaged, and close to 60% may be lost by 2030 (Wilkinson 2002). There is widespread agreement that increasing coastal populations and projected increases in sea temperatures will continue to increase pressures to coral reefs, and that the need for effective coral reef management has never been greater (Hoegh-Guldberg 1999; Hughes et al. 2003; Buddemeier et al. 2004; McClanahan 2002). Resilient coral reefs -those that have the capacity to maintain dominance of hard corals and/or morphological diversity - need to be identified and prioritized for conservation. The natural resilience of coral reefs, which maintains them in a coral dominated state, is being undermined by stresses associated with human activities in the water and on land. Unmanaged, these stresses have the potential to act in synergy with climate change to functionally destroy many coral reefs and shift them to less diverse and productive states dominated by algae or suspension feeding invertebrates. Coral reefs are under pressure from a variety of human activities, including catchment uses that result in degraded water quality, unsustainable and destructive fishing, and coastal development. These local pressures act to reduce the resilience of the system, undermining its ability to cope with climate change, and lowering the threshold for the shift from coral-dominated phase to other phases. Increasingly, policy-makers, conservationists, scientists and the broader community are calling for management actions to restore and maintain the resilience of coral reefs to climate change, and thus avoid worst-case scenarios (Obura and Grimsditch 2009). Coral reef management efforts that maintain and increase coral reef resilience will play a critical role in determining the future of coral reefs by allowing ecosystems to adapt and adjust before irreversible damage occurs or to recover from mortality events (Hansen et al. 2003). Emerging theories and new multi-disciplinary approaches point to the importance of assessing and actively managing resilience; that is, the extent to which ecosystems can absorb recurrent natural and human perturbations and continue to regenerate without slowly degrading or unexpectedly flipping into alternate states (Folke et al. 2004; Berkes et al. 2003). These efforts are those that will facilitate the ability of systems to maintain key functions and processes in the face of stresses or pressures by either resisting or adapting to change (Nyström and Folke 2001; Holling et al. 1995). Sustaining this capacity requires improved protection of coral reef resilience (Nyström et al. 2000).

Factors that influence coral reef resilience can be grouped into three categories: (1) biological factors (2) physical factors, and (3) anthropogenic factors. Each of these categories includes attributes that can strengthen resistance, survival, and recovery from mass bleaching as well as recovery from other types of disturbances. Maintaining social–ecological resilience and successfully managing the delivery of ecosystem goods and services requires an ability to detect and react to ecological feedbacks (Berkes et al. 2003). Identification of areas that have historically had high resilience to bleaching provides the basis for a network of refugia to underpin resilience-based management of the reef ecosystem. Refugia serve as a seed bank to facilitate the recovery of areas with lower natural resilience, and will play a central role in networks of protected areas designed to maximize ecosystem resilience. Marine spatial planning and management (MSPM) offers a strategic way of improving decision-making and delivering an ecosystem-based approach to managing human activities in the marine environment (Ehler and Douvere 2007). Efforts are underway to inform conservation and management through identification of sites with high resistance to change and recovery from disturbance (Maynard et al. 2010; Obura and Grimsditch 2009). For example, the IUCN has developed a protocol for assessing coral reef resilience in this way to define management priorities (Obura and Grimsditch 2009). Although such site selection processes are crucial to the spatial management of coral reef resilience, empirical criteria to support these decisions are few. It is therefore critical that the scientific community develops resilience selection criteria based on the current state of knowledge and identifies key research priorities for future study (McClanahan et al. 2012).

Despite gaps between resilience theory and field observations, the rapid rate of climate change disturbance has elevated demand for immediate solutions and management intervention for coral reef ecosystems (Hoegh-Guldberg and Bruno 2010; Donner 2009). By examining coral reef responses to disturbance across a range of past oceanographic and management conditions, site selection criteria for coral reef resilience can be developed that reflect how known disturbances and local environmental conditions have shaped present reef communities (Donner 2009). Relevant conditions may include a range of physical factors such as reef hydrographic conditions and connectivity (Graham et al. 2011; McClanahan 2008); biological factors such as coral diversity, disease, and herbivory (Cheal et al. 2010); and habitat factors such as nutrients inputs, habitat complexity, and human impacts (Wilson et al. 2006; McClanahan et al. 2011). However, having a wide range of physical and biological factors alone is not sufficient to develop sound resilience selection criteria. Factors must also be supported by science with substantial empirical evidence, weighted by the strength of the evidence linking factors to resistance and recovery (McClanahan et al. 2012).

Two approaches have been applied to set management and conservation priorities for supporting the natural resilience of coral reefs: measure as many variables as possible and select sites with the best set of positive characteristics; or measure a feasible set of factors with scientific support for promoting resilience. While the former approach has been applied recently (Obura and Grimsditch 2009), a recent study concludes that the latter approach will lead to greater adoption and success in supporting coral reef resilience because it adopts a reduced set of factors that are both manageable and defensible, and therefore more likely to be implemented (McClanahan et al. 2012).

Coral reef resilience can be integrated into MSP since it is important in determining the conservation value of a reef area or a set of reef areas. When analyzed together with other biophysical data such as satellite measured environmental data, coral reef resilience factors can help the MSP process by identifying priority areas for conservation. In such an approach, it is critical to first create a ‘resilience layer’ that represents the health of a reef site, zone or location. There exist gaps in effective techniques that can provide detailed and ecologically relevant spatial information on coral reef resilience across broad and structurally complex geographical areas. However, advances in spatial technologies such as GIS, remote sensing and spatial modeling show great potential to address this challenge (Maina et al. 2008; Rowlands et al. 2012; Knudby et al. 2013, 2014).

This paper has two parts. First, we present a multivariate analysis of resilience and mapping of coral reef resilience across five locations; exploring the variations between sites within each of these locations and the latitudinal gradient of resilience across the locations. The second part explores the use of these products coupled with satellite measured environmental data in conducting marine spatial and conservation planning analyses.

Methods

Study Sites

A total of 57 sites from five locations across two countries were surveyed for coral reef resilience indicators (Fig. 1). We used this basic field dataset, and derived variables from environmental data from satellite observations and modeled population density data.

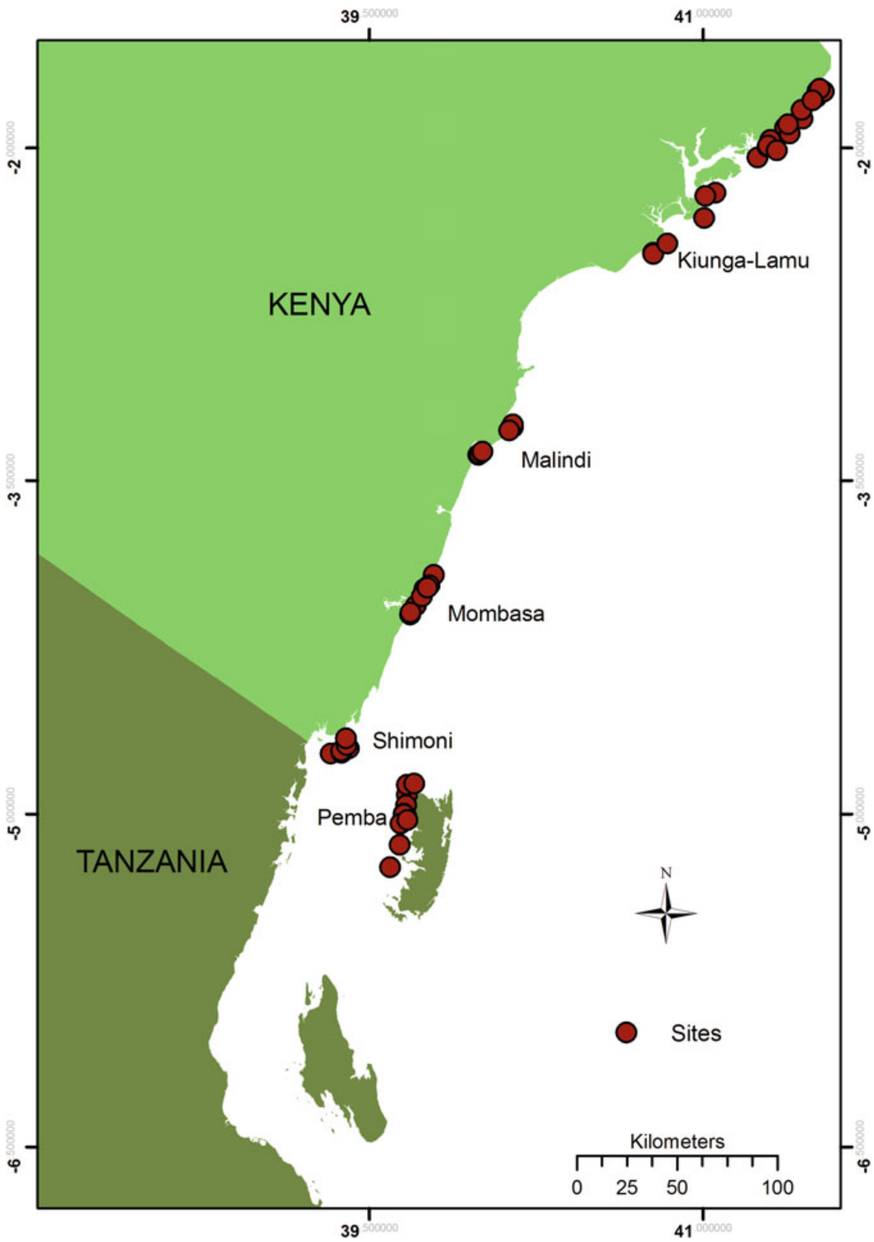
Selection of Resilience Indicators

A total of 61 resilience indicators (variables) were analysed and from these a subset of 34 indicators selected for further analysis and later aggregated into 14 factors. Resilience indicators data collection followed the IUCN Rapid Assessment methodology described in Obura and Grimsditch 2009. In their study, McClanahan et al. (2012) did a rigorous assessment of factors promoting coral reef resilience based on their perceived importance, empirical evidence, and feasibility of measurement. They developed empirical selection criteria for prioritizing coral reef management and conservation in the face of climate change. These criteria sought to identify reefs with the greatest resilience to climate disturbance so that local managers can support the persistence of local coral reef ecosystems. In their study, they also identify key research priorities for coral reef resilience, based on levels of perceived importance and areas of debate within the coral reef scientific community. They conclude that their evaluation supports the concept that, despite high ecological complexity, relatively few strong variables can be important in influencing ecosystem dynamics. Results from their study identified 11 principal factors that influence the resistance and/or recovery of coral reef ecosystems to climate-driven disturbances – and are also feasible to assess from local field observations at relatively fine spatial scales. These include aspects of the coral fauna (presence of stress-resistant coral species, diversity of coral species, high levels of coral recruitment, and absence of coral disease) and competition for space (low presence of microalgae) as well as moderators of competition (herbivore biomass), the physical environment (high annual temperature variability, low nutrient and sediment levels), and direct human impacts (physical impacts and fishing pressure). These 11 factors can thus act as resilience indicators and may function as a list of mapping targets that combined have the potential to characterize the resilience of a coral reef ecosystem to climate-driven disturbance.

Indicators were selected on the basis of the criteria above and in addition, the following;

I. Their relevance to science in terms of explaining or quantifying aspects of resilience, and with a specific reference to bleaching. Some indicators have only an indirect influence on resilience or resistance to bleaching while others are primary variables. In this case the latter had more weight than the former.
II. Their reliability in terms of measurement in the context of the rapid assessment methodology; some indicators are more reliably or directly measured than others, e.g. percent cover of substrate is very reliable thus had more weight compared to assessment of turbidity based on secondary data, knowledge of the site and/or single-observations thus lesser weight.
III. Their relevance to management; direct relevance to management at the local level, or the degree to which decisions about management of a site are generally (or can be) influenced by the indicator.
IV. Indicators that can be altered by managers; a manager can directly change the value of the indicator through management actions. Some important indicators, such as currents, cannot be changed by a manager, while some otherwise low-importance indicators (e.g., soft coral), can be.

Fig. 1 Map showing study sites



In addition to these criteria, an attempt was made to have at least 2 indicators for each factor, so that information from multiple indicators is used to calculate the level of that factor.

Resilience Analyses and Classification

Analysis of resilience was first done by aggregating individual variables/indicators into 14 factors depending on their relevance to each of the factors. In the second aggregation, factors were grouped into three broad aggregate components of resilience (Biological, Physical and Anthropogenic). The third aggregation averaged the three components in order to derive a single overall resilience index for each site. Aggregation indicators into factors, factors into resilience components and resilience components into overall resilience index assumed equal weights to the component of resilience being measured. (Fig. 2).

For each of these aggregation levels, the minimum and the maximum values were identified and lower, mid and upper thirds calculated from these to classify resilience into three classes (High, Medium and Low). The formula to calculate the terciles is as shown below;

$$\text{1st tercile} = (X_{max} - X_{min})/3 + X_{min}$$
$$\text{2nd tercile} = (X_{max} - X_{min})/3 + \text{1st tercile}$$

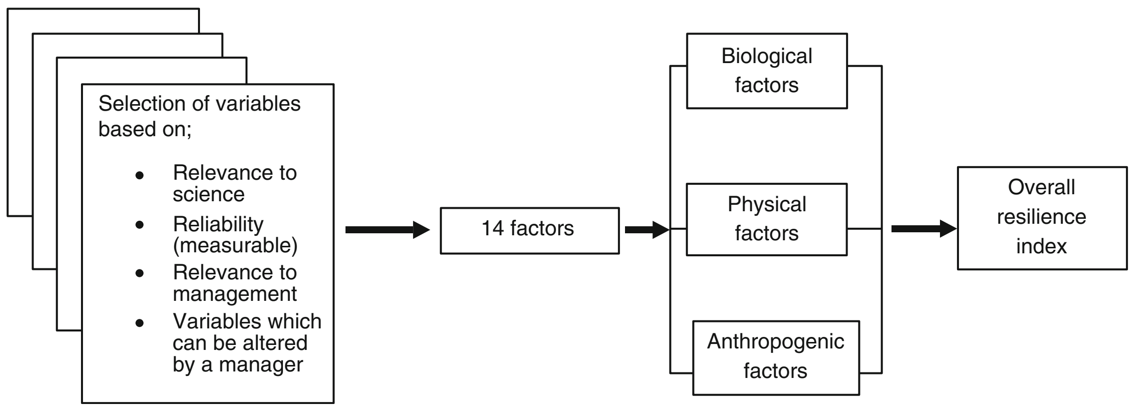


Fig. 2 This figure shows the aggregation levels of resilience indicators

Where,

X_{max} is the maximum value at each level and,
X_{min} is the minimum value at each level.

From the formula above, values between the minimum value and 1st tercile value were classified as 'Low', those between 1st and 2nd tercile values classified as 'Medium' and those between 2nd tercile and maximum values classified as 'High'. The resultant classified tables were imported to PRIMER and multivariate statistics performed. Cluster and multi-dimensional scaling statistics were performed on this data to reveal patterns and relationship between variables and components of resilience being measured (biological, physical, anthropogenic and overall resilience).

Mapping Resilience Factors

All sites were geo-referenced and mapping done for the three resilience components and overall resilience. Sites were coded as either highly, averagely or least resilient using the above classification.

Environmental Factors

Sea Surface Temperature (SST)

Sea surface temperature (SST) is a critical physical attribute of coastal ecological systems. SST is believed to be the most dominant factor causing coral bleaching and mortality (McClanahan et al. 2007). Sea surface temperature data products at global and regional scales are well established (Robinson and Donlon 2000).

We used SST products from AquaMODIS accessible from the oceancolour website in scaled Hierarchical Data Format (HDF). This dataset contains composites at approximately 4-km and 9-Km resolution from July, 2002 to present. Monthly and annual daytime SST composites were downloaded at 4-km resolution for 2003–2010. Data processing included the retrieval, sub setting the files to the study area, masking, back-scaling and offsetting the data from 16 bit unsigned integer format to geophysical units using the following equation:

$$\mathrm{SST}\ (^{\circ}\mathrm{C}) = 7.17185\mathrm{E} - 4 \times \mathrm{byte\ \ value} - 2$$

Long term average (LTA) mean and maximum SST composites were calculated for the entire analysis period. Site-level LTA mean and maximum SST was calculated for all the study sites by extracting site values from the LTA composites using the point statistics function in ESRI ArcGIS. We also calculated site thermal stress by subtracting the LTA maximum SST for the whole time series from the monthly '8-yr LTA' and summing up all positive anomalies from 2003 to 2010. Since thermal stress thresholds can vary from one location to another, we assumed that even the slightest positive deviation from 'normal' conditions can cause bleaching. Anomalies below the LTA Maximum SST were ignored since these affected the zero cut-off thresholds.

Chromophoric (Colored) Dissolved Organic Matter (CDOM)

Oceanic satellite observations in the visible and near-infrared bands allow for the measurement of a variety of ocean color information including phytoplankton chlorophyll-a, total suspended matter (TSM), and colored dissolved organic matter (CDOM) (Wang et al. 2010). For modeling purposes, ocean waters are commonly described as being of Case I or case II types (Morel and Prieur 1977; Morel and Be'langer 2006). The former type are those waters whose optical properties are determined primarily by phytoplankton and related colored dissolved organic matter (CDOM) and detritus degradation products; while the later represents the turbid coastal zones influenced by land drainage or sediment re-suspension, with optical properties mainly influenced by CDOM of terrestrial origin, mineral particles, various suspended sediments, urban discharges and industrial wastes (Morel and Prieur 1977).

Level 3 Chromophoric (colored) Dissolved Organic Matter (CDOM) index composite for the period 2003–2010 was downloaded in HDF format, data retrieved, back-scaled and subset to the area of interest and used as a proxy for turbidity. Back-scaling of the byte values to geophysical measurements (index) used the following equation:

$$\text{CDOM index} = 1^{*}\text{byte values}$$

In this analysis CDOM is used as a proxy for turbidity and nutrient input into the reefs.

Population Density

Global population density grid for 2010 was downloaded from the Socio-Economic Data and Applications Center (SEDAC) GPWv3 website http://sedac.ciesin.columbia.edu at 1/4 degree resolution. This dataset renders global population data at the scale and extent required to demonstrate the spatial relationship of human populations and the environment across the globe. Population data estimates are provided for 1990, 1995, and 2000, and projected to 2005, 2010, and 2015. The projected grids were produced in collaboration with the United Nations Food and Agriculture Programme (FAO) as Population Count and Density Grid Future Estimates. The global grid was subset to the area of interest and population density statistics calculated for all the sites.

Main Results

Overall Relative Resilience by Aggregating Resilience Indicators

When aggregating resilience indicators, on average Mombasa had the highest resilience score (2.93) on a scale of 1–5, followed by Shimoni (2.87), Pemba (2.74), Malindi (2.73) and Kiunga-Lamu having the lowest score at 2.54 (Fig. 3).

When all sites were analyzed together, six sites (Pe12, Mo7, Mo8, Sh1, Sh4 and Sh5) had high resilience scores and were classified as highly resilient sites (See Appendix for site names). 28 sites scored lowly while 23 had average resilience scores (Fig. 4). A cluster analysis done on the aggregate factors (biological, physical and anthropogenic) showed three clear clustering of sites: the highly resilient, the averagely resilient and the least resilient.

While a multi-dimensional scaling (Figs. 5 and 6.) showed that overall resilience was positively correlated with each of the three aggregate factors, it was relatively stronger with biological factors and weakest with anthropogenic factors. The results also show that highly resilient sites were in areas with low anthropogenic pressure i.e., sites whose scores for anthropogenic resilience were high. These were observed as those sites with regulated use, mostly those within MPAs or those that are difficult to access. These sites also scored highly-averagely for biological factors and averagely for physical factors.

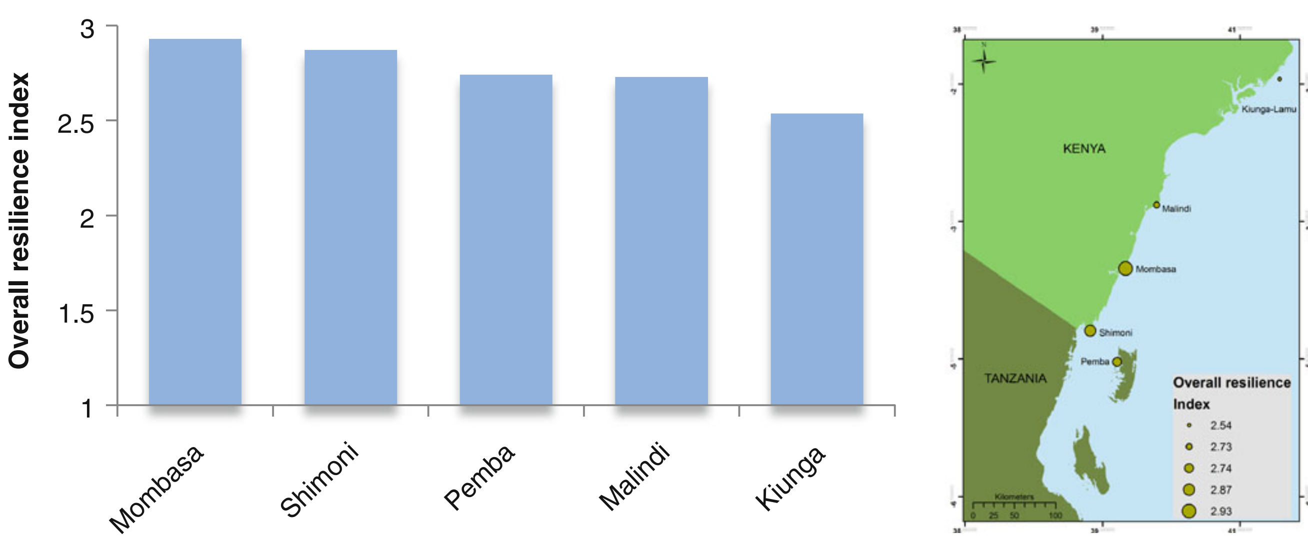


Fig. 3 Overall resilience scores averaged by locations (*left*) and map showing the latitudinal gradient of the scores (*right*).

Kenya_Pemba	Classification								
Site no.	overall	anthr	biol	phys	Site no.	overall	anthr	biol	phys
Pe12	High	High	High	Medium	Pe6	Low	Low	Medium	Medium
Mo7	High	High	Medium	Medium	Pe8	Low	Low	Medium	Medium
Mo8	High	High	Medium	Medium	Pe9	Low	Low	Low	Medium
Sh1	High	High	Medium	Medium	Pe10	Low	Low	Medium	Medium
Sh4	High	High	Medium	Medium	Pe11	Low	Low	Low	Medium
Sh5	High	High	Medium	Medium	Mo2	Low	Low	Low	Medium
Pe1	Medium	Low	Medium	Medium	Mo4	Low	Low	Medium	Medium
Pe2	Medium	Low	Medium	Medium	Sh3	Low	Low	Medium	Medium
Pe3	Medium	Low	Medium	Medium	Sh6	Low	Low	Medium	Medium
Pe4	Medium	Low	Medium	Medium	Sh7	Low	Low	Medium	Low
Pe5	Medium	Low	Medium	Medium	Ki1	Low	Medium	Medium	Medium
Pe7	Medium	Low	Medium	Medium	Ki2	Low	Medium	Medium	Medium
Pe13	Medium	Low	Medium	Medium	Ki3	Low	Low	Medium	Medium
Mo1	Medium	Low	Medium	Medium	Ki4	Low	Medium	Medium	Medium
Mo3	Medium	Medium	Medium	Medium	Ki7	Low	Medium	Medium	Low
Mo5	Medium	Low	Medium	Medium	Ki8	Low	Low	Medium	Low
Mo6	Medium	Low	Medium	Medium	Ki9	Low	Medium	Medium	Low
Sh2	Medium	Low	High	Medium	Ki10	Low	Medium	Medium	Low
Ki5	Medium	Medium	Medium	Medium	Ki11	Low	Medium	Medium	Medium
Ki6	Medium	Medium	Medium	Low	Ki12	Low	Medium	Medium	Low
Ki14	Medium	Low	Medium	Medium	Ki13	Low	Low	Low	Medium
Ki15	Medium	Medium	Medium	Medium	La1	Low	Medium	Medium	Low
La2	Medium	Medium	Medium	Medium	La3	Low	Medium	Low	Low
La6	Medium	Low	Medium	Medium	La4	Low	Medium	Medium	Medium
Ma1	Medium	Medium	Medium	Medium	La5	Low	Low	Medium	Low
Ma4	Medium	Medium	Medium	Medium	Ma2	Low	Medium	Medium	Medium
Ma5	Medium	Medium	Medium	Low	Ma3	Low	Low	Medium	Medium
Ma6	Medium	Medium	Medium	Medium	Ma8	Low	Medium	Medium	Medium
Ma7	Medium	Medium	Medium	Medium					

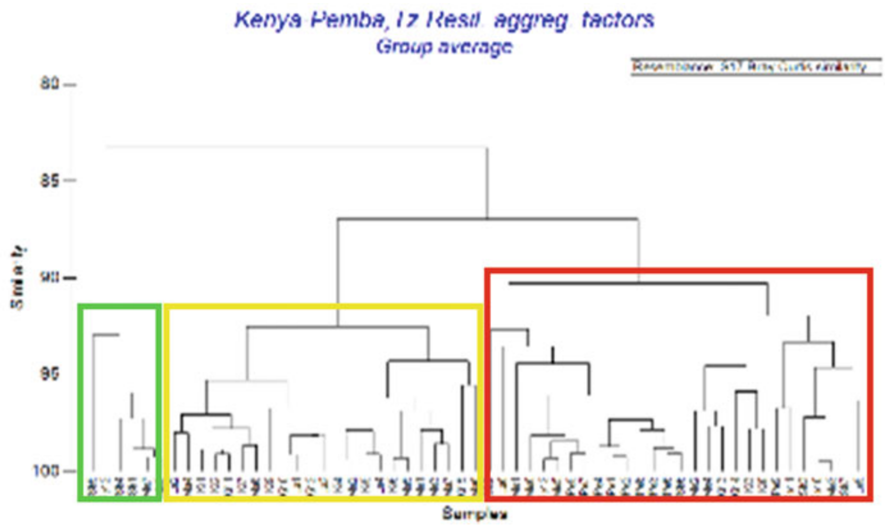


Fig. 4 Table (*left*) shows the classification of resilience scores for different factors and a cluster analysis (*right*) showing the three clear groupings of sites.

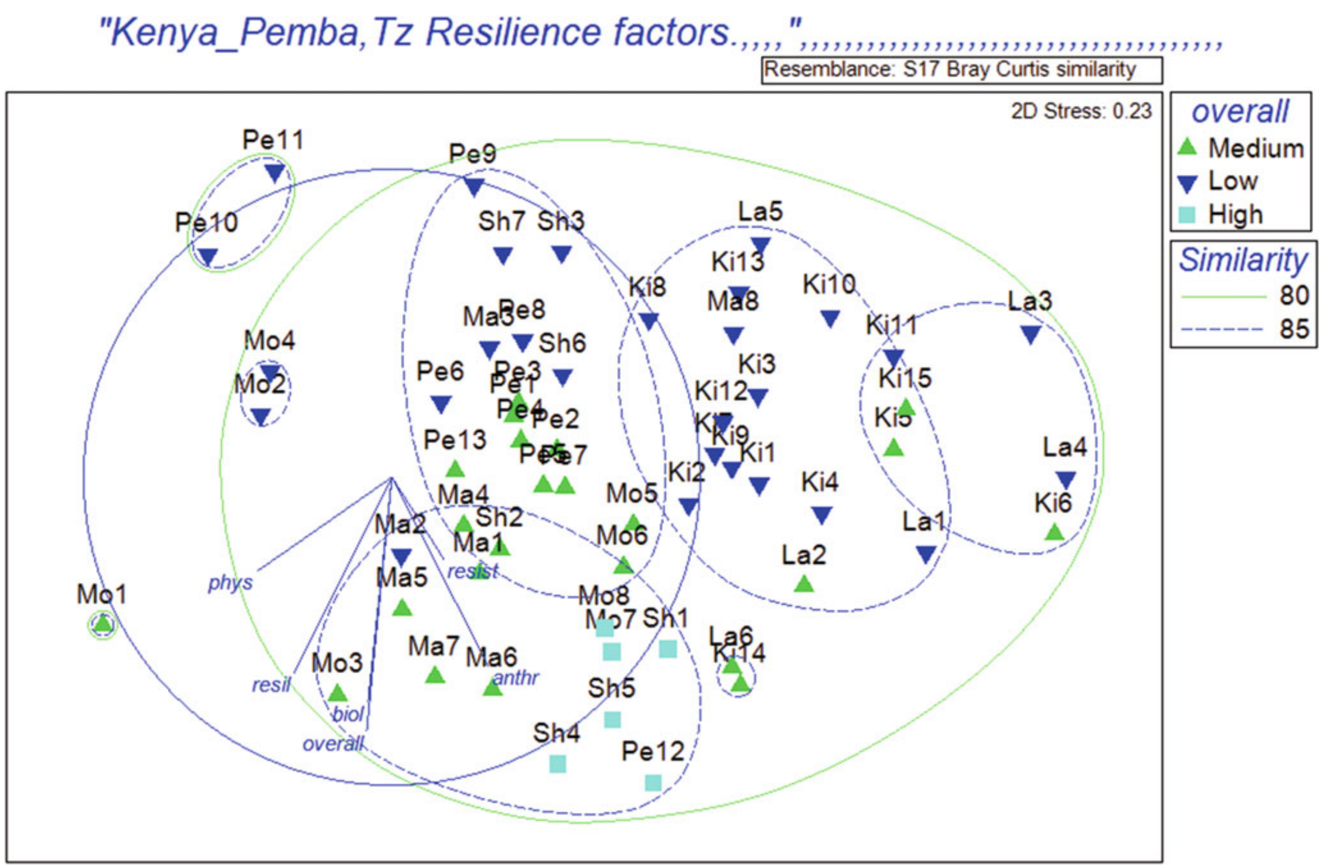


Fig. 5 Multi-Dimensional Scaling of overall coral resilience. The trajectories show the correlations between the aggregate resilience factors and the overall coral resilience

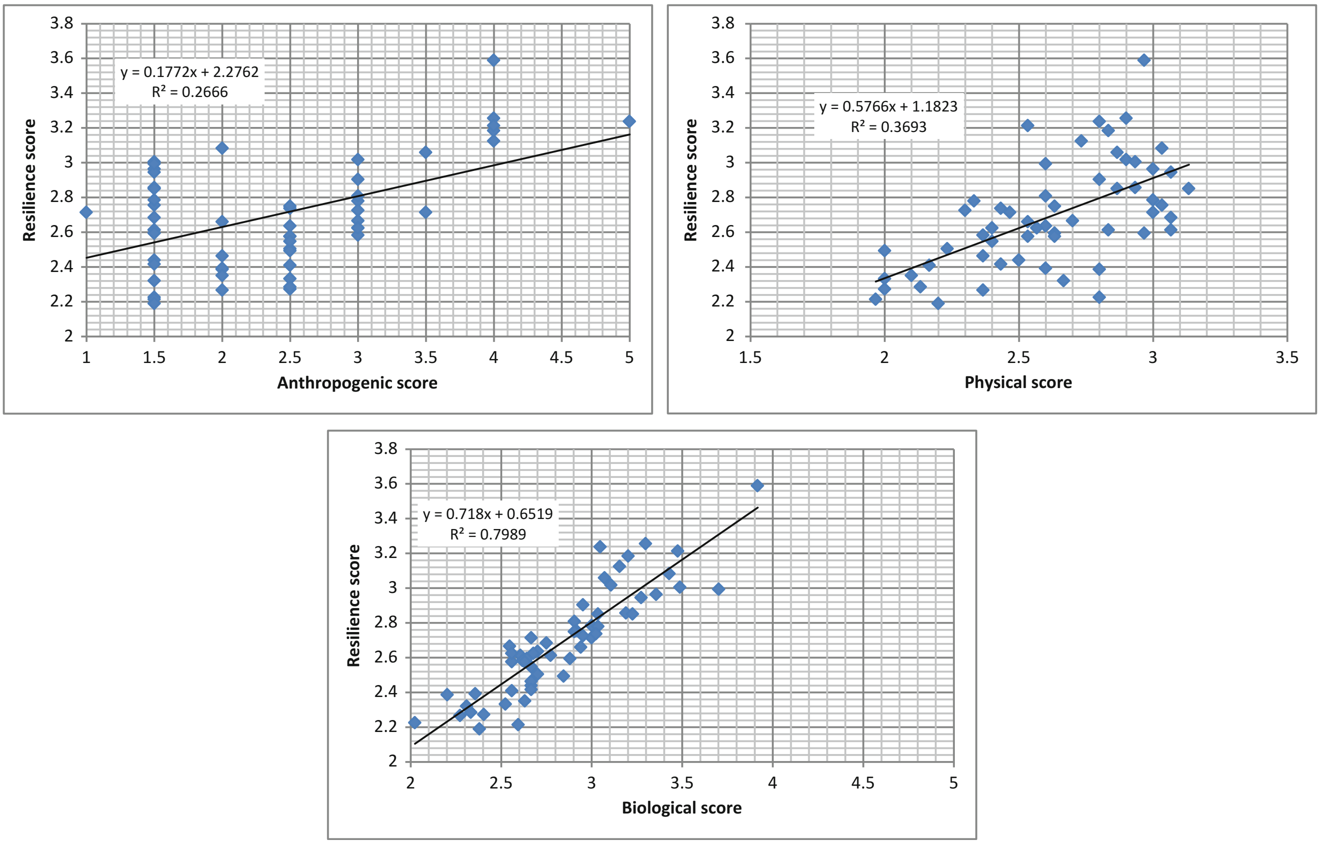


Fig. 6 Graphs showing the correlation between various resilience factors and overall resilience cores

Environmental Data

Sea Surface Temperature

Results showed that cumulative SST anomaly (thermal stress) and climatological mean SST were positively correlated with overall resilience. However, climatological maximum SST was negatively correlated with overall resilience (Figs. 7 and 8). Although these correlations are rather weak, the positive correlation with cumulative SST anomalies could possibly be attributed to the fact that reefs that are exposed to warmer conditions develop some degree of acclimatization due to these exposures (Grimsditch et al. 2010).

Chromophoric Dissolved Organic Matter

Our results showed a negative correlation with overall resilience meaning that sites where there high turbidity was observed as a result of nutrient influx from terrestrial sources had low resilience scores. Turbidity is a limiting factor to the ecosystem since it hinders the circulation of oxygen for living organisms and also prevents penetration of light essential for photosynthesis into the water column. Increases in the nutrient influx into reef waters may cause a switch in dominance from corals to various forms of frondose and filamentous algae and bioeroding sponges (Smith et al. 1981; Rose and Risk 1985; Cuet et al. 1988; Bell 1992; Risk et al. 1995; Lapointe et al. 1997).

Population Density

Population density was negatively correlated with anthropogenic factors i.e., as the score for anthropogenic factors increased (high resilience), the population density also increased (Fig. 9).

Resilience Classification and Mapping

Overall Resilience

Six sites were classified as highly resilient, 23 as averagely resilient and 28 as least resilient. The classification system applied here produced consistent results with the cluster analysis. Most of the highly resilient sites were under full protection with an exception of the two sites in Mombasa (Mo7 and Mo8), which were partly protected.

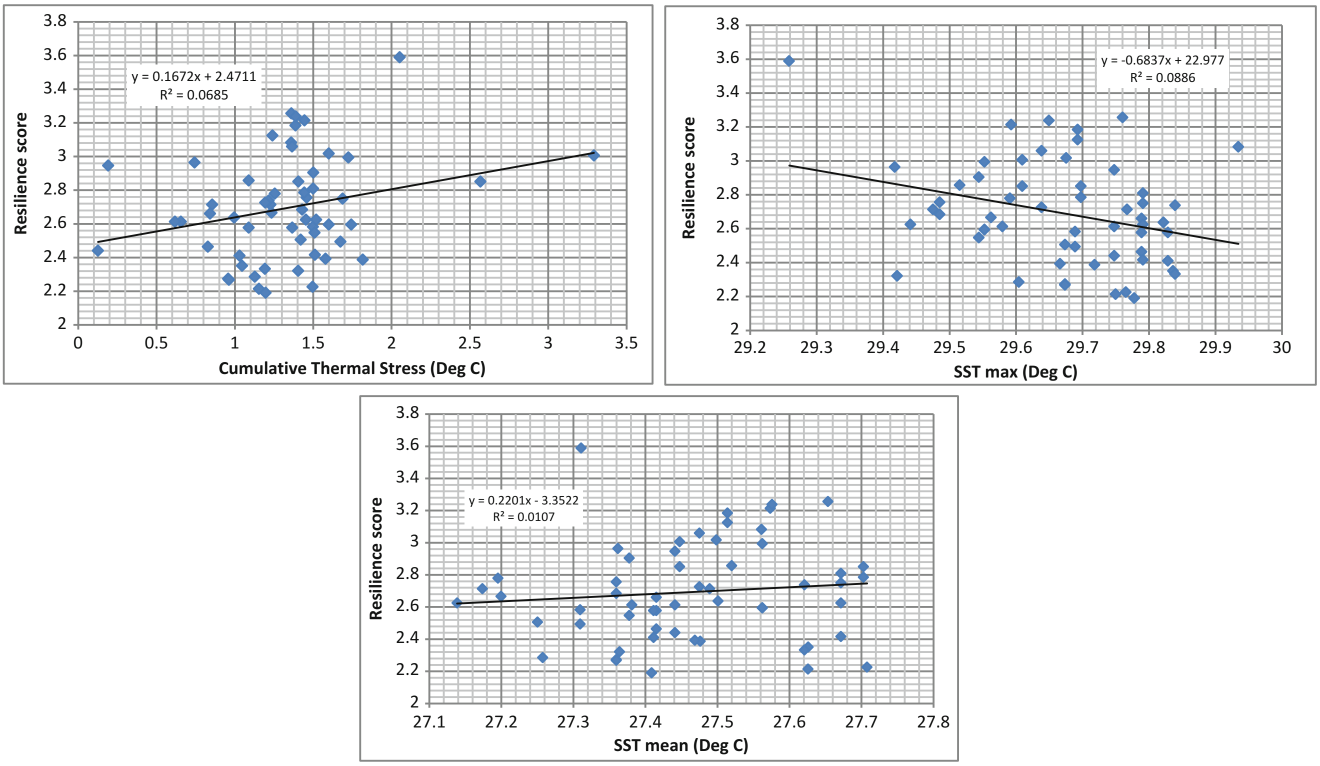


Fig. 7 Graphs showing the correlation of overall resilience with SST variables

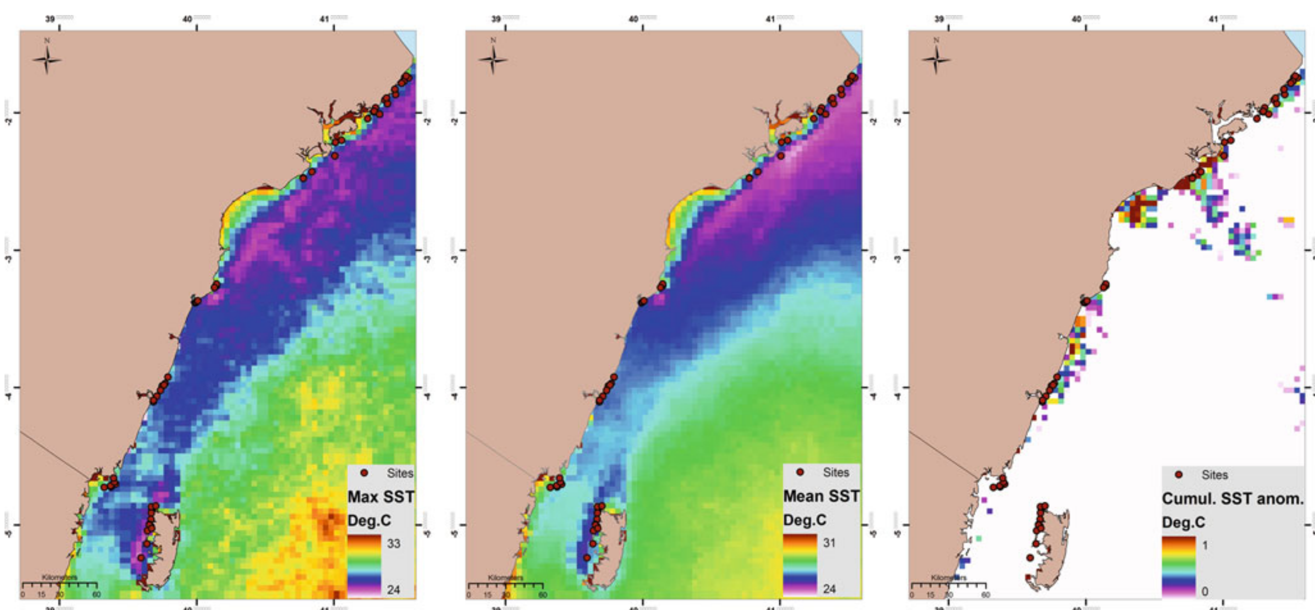


Fig. 8 Maps showing the distribution of SST variables

Most of the least resilient sites were under no protection or partly protected. The exception to this was two sites in Malindi (Ma2 and Ma8), which had full protection. It is important to note that these two sites had the lowest overall resilience scores with all the factors scoring averagely (Figs. 4 and 10).

Aggregate Factors Maps

Biological Index

Two sites (Sh2 and Pe12) were classified as having high biological resilience, 50 as average while 5 had low scores. It's important to note that sites whose scores for

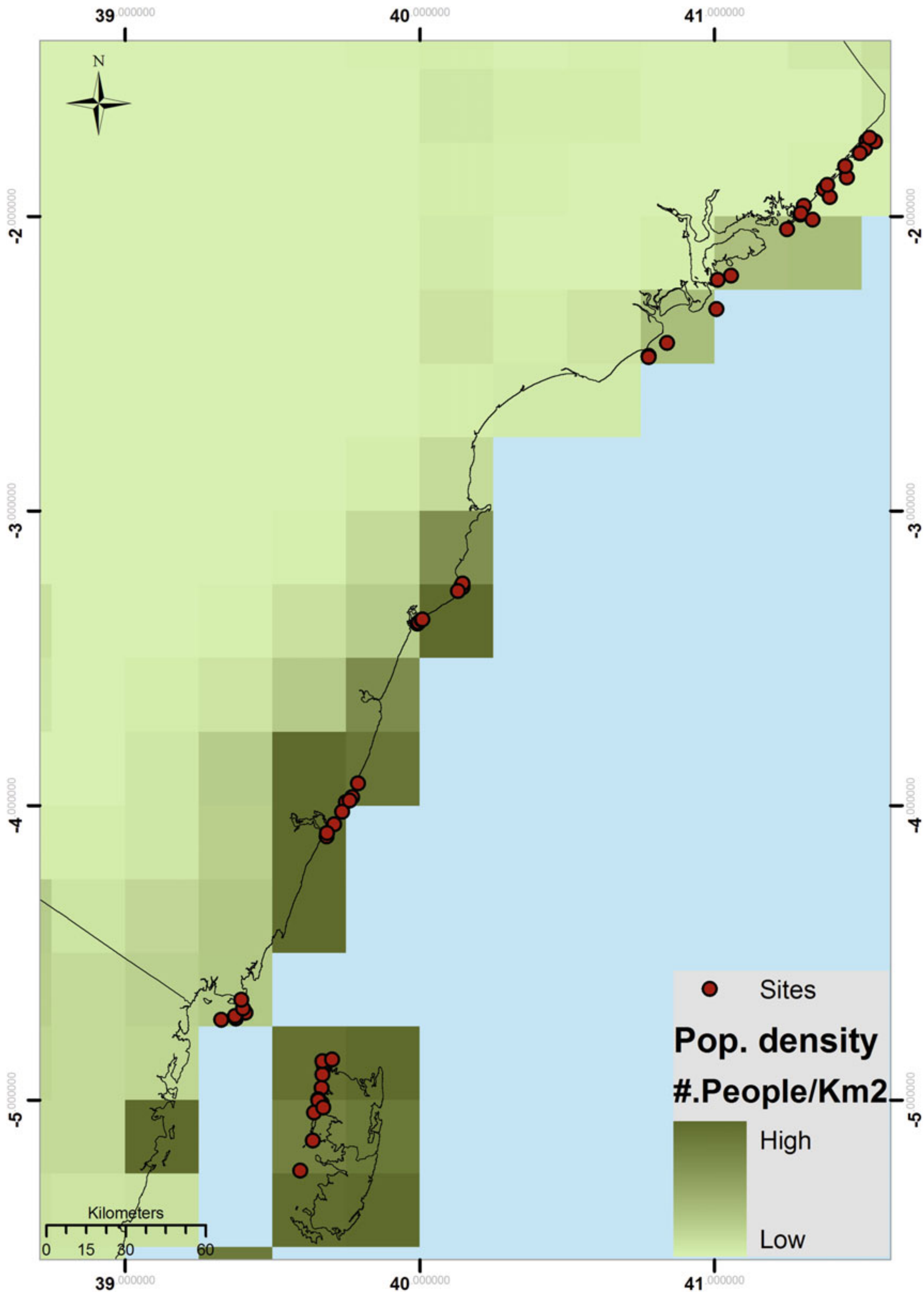


Fig. 9 Map showing the population density (*right*).

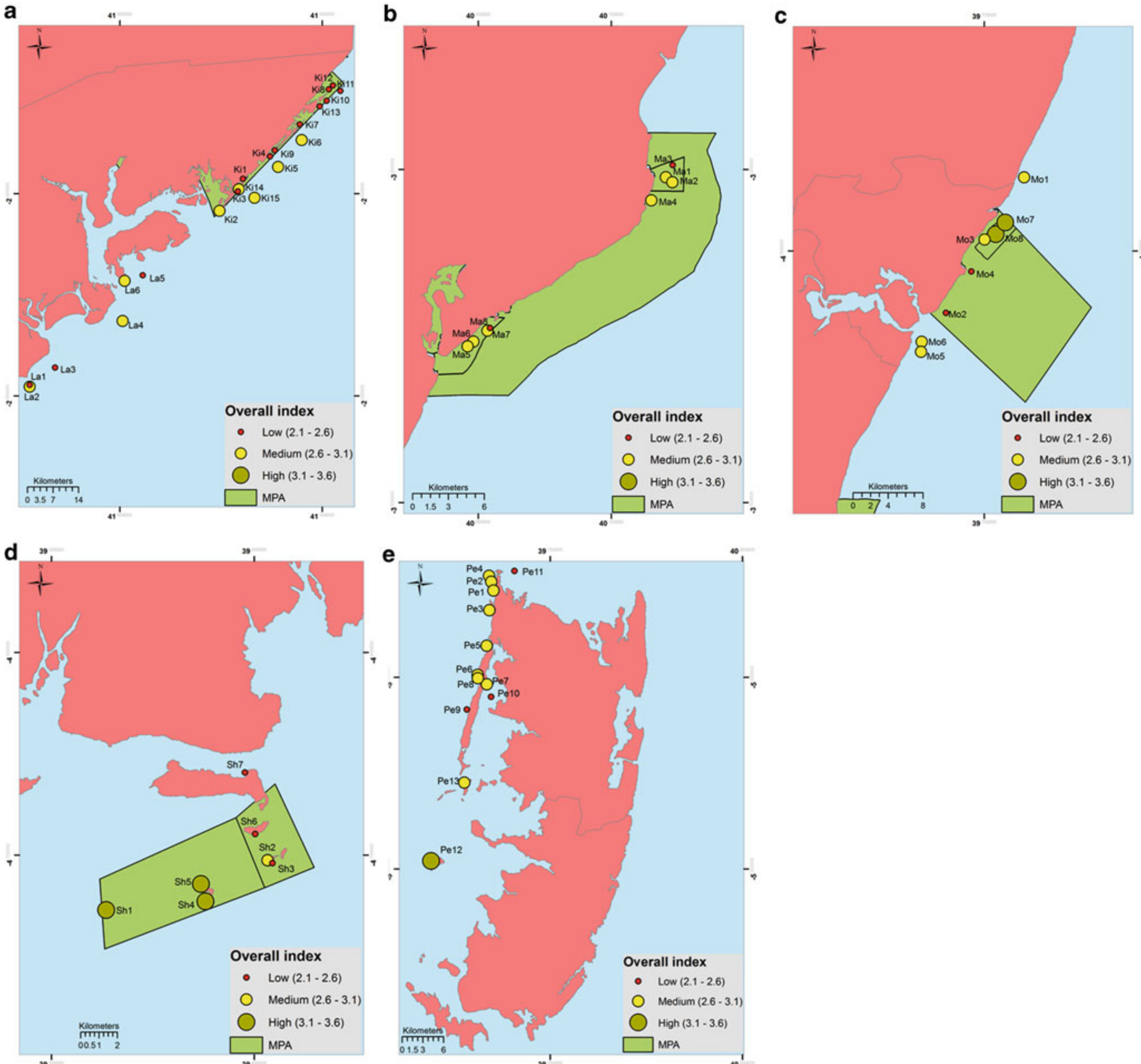


Fig. 10 Maps showing the distribution of overall coral health in the study locations; **A** (Kiunga-Lamu), **B** (Malindi), **C** (Mombasa), **D** (Shimoni) and **E** (Pemba)

biological factors were high were within MPAs. Sh2 is a site within the reserve while Pe12 is a site within a MPA (Fig. 11).

Physical Index

Sites scored between average (46 sites) and low for physical factors (Fig. 12). Most sites that had low scores for overall coral health scored relatively higher than some sites that had high overall coral health.

Anthropogenic Index

Six sites scored highly for anthropogenic factors (meaning there is low anthropogenic activity) while the rest of the sites were closely split between average and low scores (Fig. 13). Anthropogenic factors influenced the overall score for coral health as all of the sites that had high scores for anthropogenic factors had high-average scores for biological and physical factors. Most of the sites found in high population density areas and those outside of conservation areas had

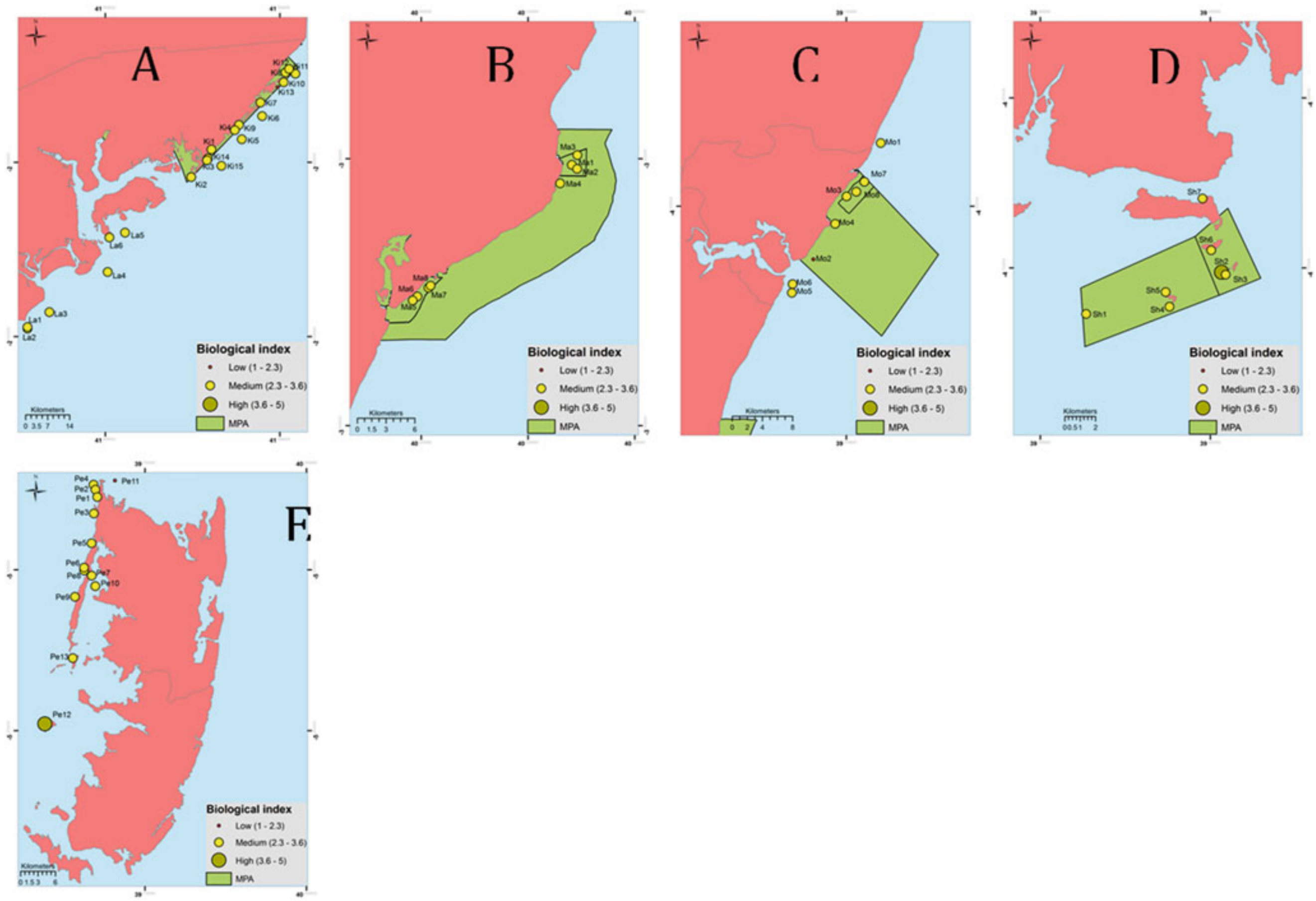


Fig. 11 Maps showing the distribution of biological factors resilience index in the study locations; **A** (Kiunga-Lamu), **B** (Malindi), **C** (Mombasa), **D** (Shimoni) and **E** (Pemba)

low scores for anthropogenic factors and low-average scores for overall coral health.

Discussion

Resilience Factors

This study has revealed that biological and anthropogenic factors had the greatest influence on the overall health of the studied reefs. McClanahan et al. (2012) also achieved similar results. Other studies (Mora et al. 2011; Sandin et al. 2008) show that the most resilient reefs are expected to be those with high fish and coral diversity; and few human impacts. Reefs which scored medium-high for biological factors and medium-high (medium-low anthropogenic activity) scores for anthropogenic factors had higher overall resilience than those that had low scores for the two factors. These aggregate factors seemed to explain much of the variability in overall health than individual disaggregated factors. This could be explained by the fact that two or more factors working independently could have a much smaller effect to explain variability in overall health than when these factors work synergistically. It was also observed that reefs found in protected areas or areas under some form of regulation had scored relatively higher for overall resilience and had low anthropogenic influence. Exceptions to this are reefs off the northern Kenyan coast; in Malindi and Lamu. Malindi reefs scored averagely for overall health although most of these were inside MPAs. These reefs are under increasing nutrient input and sedimentation from Sabaki River associated with an increase in land uses that promote soil loss (Dunne 1979; Finn 1983). McClanahan and Obura (1997) argued that this influence however had only a minor effect on the corals over a 7 year period of their study and that the total cover of hard coral remained nearly the same despite periods of three months or more when the water was brown and turbid. They observed that the largest change over the study period was a shift in the species composition of the corals such that those species more tolerant of the increased nutrient and sediment conditions increased and less tolerant species decreased. During the

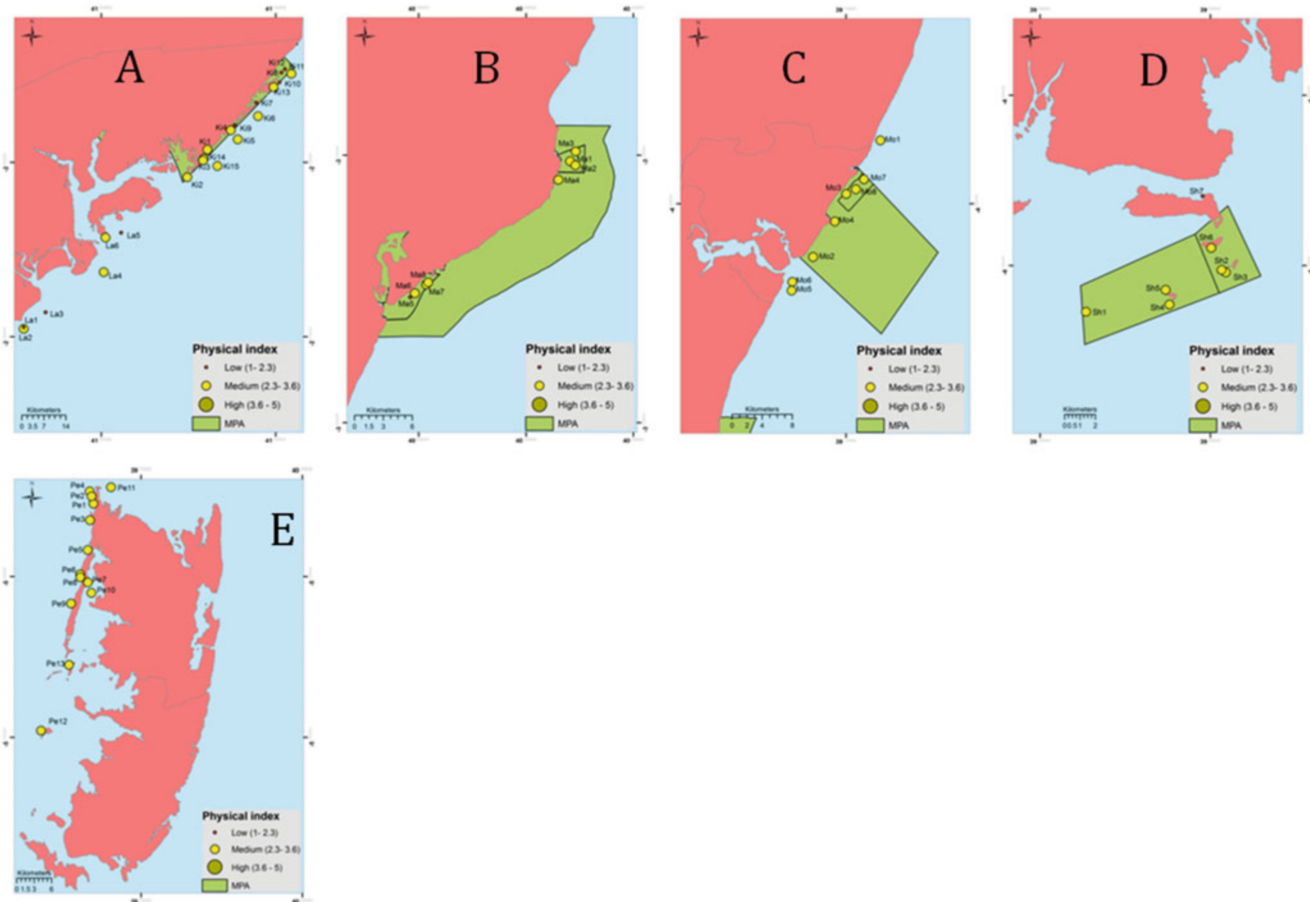


Fig. 12 Maps showing the distribution of physical factors resilience index in the study locations; **A** (Kiunga-Lamu), **B** (Malindi), **C** (Mombasa), **D** (Shimoni) and **E** (Pemba)

1998 mass coral bleaching event, most of Kenyan reefs had high mortality where the main casualties were the marine protected areas (McClanahan et al. 2001). Malindi reefs were the greatest casualties as coral cover declined from 35–45% (pre-bleaching) to 10-20% after the bleaching event. This cover had further reduced to 5% in 2004 (Lambo and Ormond 2006). Coral cover in reefs in the south of Kenya such as Mombasa and Kisite suffered the lowest losses (McClanahan et al. 2007). The findings in these papers indicate that when multiple stressors are actively compounding each other in an ecosystem rather than counteracting each other, then the resilience of such an ecosystem is greatly reduced and this could possibly explain why the resilience of the reefs in Malindi was relatively lower than that of reefs in other protected areas in the study locations.

Environmental Stress

The number and extent of environmental stresses can influence the resilience of coral reefs. This occurs because stresses can act in multiplicative rather than additive ways or because the sum of two stresses can exceed a threshold that a single stress would not reach by itself. Consequently, two or more stresses or disturbances working independently may have a much smaller effect than the two factors working together. Inhibitory synergies can also occur when one factor would have an effect except that a second factor is nullifying this influence (McClanahan et al. 2002). In the study, the results revealed that cumulative heat stress and long term average mean SST was positively correlated with overall coral reef resilience. When corals are exposed to higher than normal thermal environment, they may develop acclimation and adaptation characteristics to resist bleaching events. These coral reefs would therefore be able to maintain their coral cover over time even after anomalous heat stress episodes and thus maintain their resilience. Most of the coral reefs identified as resilient in this study, particularly those in the south (Mombasa and Kisite) showed better recovery following the 1998 coral bleaching event. In their paper, McClanahan et al. 2007 indicate that mean SST and variability increase southwards from Somalia to Comoros. This could probably indicate why southern reefs have higher

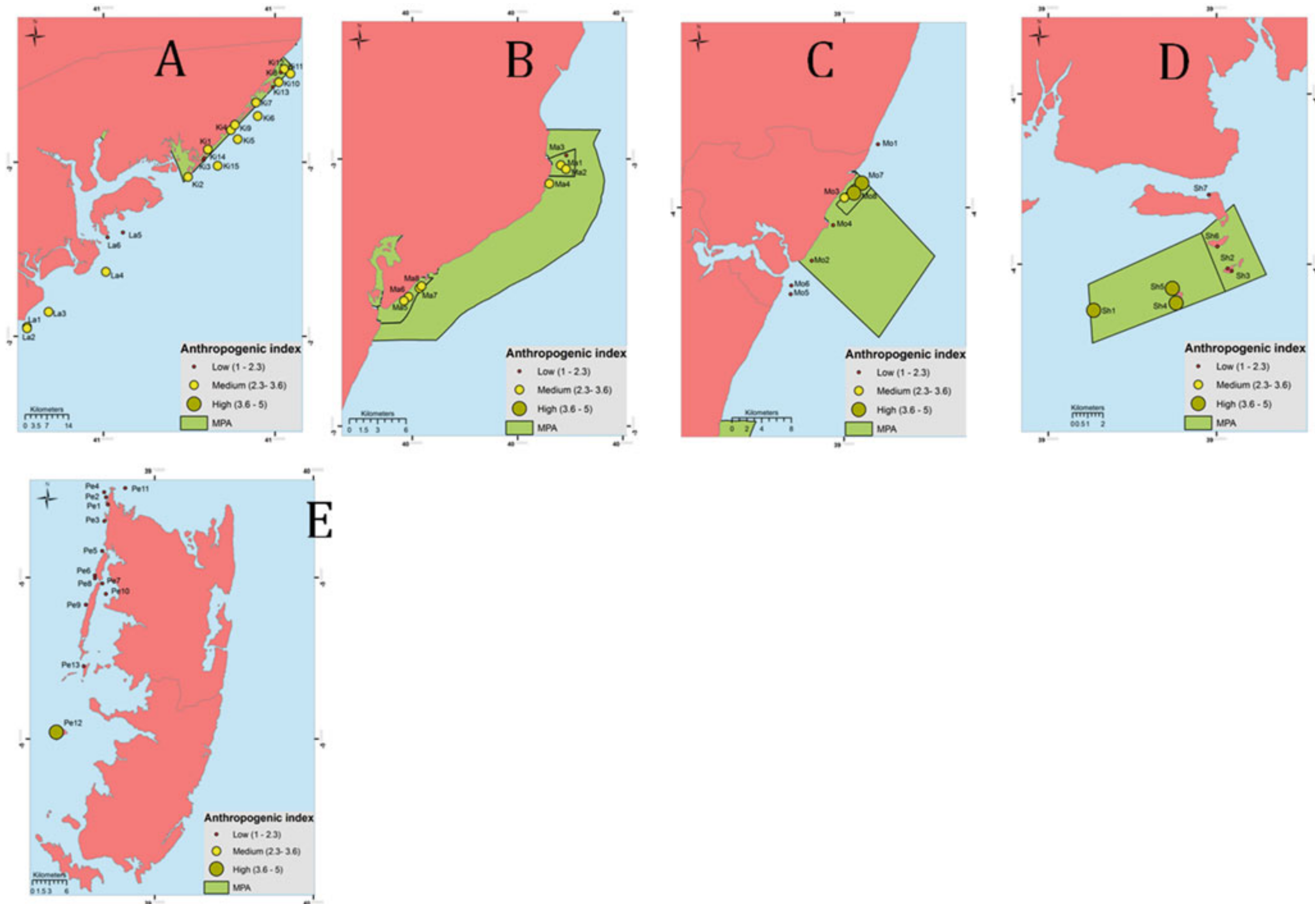


Fig. 13 Maps showing the distribution of anthropogenic factors index in the study locations; **A** (Kiunga-Lamu), **B** (Malindi), **C** (Mombasa), **D** (Shimoni) and **E** (Pemba)

relative resilience due to the exposures with warmer temperatures and their relatively higher biodiversity compared to their counterparts in the north. In the Western Indian Ocean, coral locations exposed to high reinforcing stress correspond to those areas with high river runoff and sedimentation (Maina et al. 2011). These locations are exposed to moderate radiation stress but overall are severely exposed to high reinforcing effect of water quality from highland runoff. Local management of the coastal watershed in these areas is expected to shift the overall exposure towards lower severity grades (Maina et al. 2011).

Turbidity and Sedimentation

Terrigenous sediments carried onto reefs as a result of increased soil erosion largely promote the dominance of turf or macro-algae. Elevated nutrients and organic matter can increase internal bioeroders of reef substratum and a mixture of filamentous algae. Coral reefs urgently need to be managed with a view to strengthening their resilience to the increased frequency and intensity of these pressures. In this study, colored dissolved organic matter of terrestrial origin was negatively correlated with biological factors and overall coral health. This means that its presence on coral reefs beyond certain thresholds only reduces the coral reefs' resilience. The addition of nutrients such as nitrogen and phosphorus, has the potential to increase the growth rates of some fast-growing algae and heterotrophic invertebrates (Littler et al. 1991; LaPointe et al. 1997). Though to some degree turbidity can be important in screening corals from direct UV radiation, we find that overall it is a limiting factor to the overall resilience of the ecosystem. Similarly, other studies such as McClanahan and Obura (1997) and Maina et al. (2011) found that sedimentation from terrigenous sources had a negative cumulative impact on the resilience of the reefs and only reinforced the stresses within these reefs.

Population Density

The largest human influences on coral reefs that are potentially manageable on a local scale are fishing, pollution, and sedimentation. These factors are frequently associated with human population densities (Ginsburg 1994). As human population densities increase in the agricultural communities of watersheds, agricultural labour becomes less limiting, greater partitioning of labour can occur among the various production systems, and a guild of full-time fishers can develop. Fishing becomes less a subsistence activity and more a commercial activity in that fishers will attempt to catch far beyond their household requirements. As human populations increase, fishing can become one of the few ways to meet deficits and earn extra money to support impoverished people (McClanahan et al. 2002).

Results from this study indicate that anthropogenic factors and population density were positively correlated with overall coral reef resilience. As population grows, the pressure on the ecosystem to provide the goods and services is also exacerbated. For example, due to the rise in number of people living in an area, there are more and more people fishing an already diminishing resource. This leads to unsustainable extraction of key functional groups leading to a collapse of the ecosystem. The positive correlation with overall resilience could probably indicate that people focus their efforts on 'good' reefs where the reefs continue providing for goods and services. A tour operator will take their clients to snorkel and a fisher will deploy their fishing gear on a reef where there are 'good' fish. This could partly explain the higher population density in areas with good reefs. It is also true that most of the reefs that scored highly for overall resilience were those within main towns/cities where population is relatively higher than marginal locations and no-take zones/MPAs are established. Most of the highly resilient reefs were found in protected areas and all of these MPAs were established in highly populated areas. Examples are Mombasa, Shimoni and Malindi.

Conclusions and Recommendations

Overall results from this study show that factors that promote resilience could be incorporated into practices for managing coral reefs to promote their ecological services into the future. Most of the MPAs in the region were set up in haphazard fashion with little or no factual scientific evidence for placing of park boundaries (McClanahan 1999). However the importance of incorporating coral reef resilience factors into the management of coral reefs is becoming increasingly in a changing climate. This paper presents some insights into such an approach by providing results from a rapid coral reef resilience assessment exercise and presenting tools that can be used to inform management. The study has revealed interesting patterns in relation to factors that influence resilience but more studies are needed. Such study should include more sites placed geographically in between those studied here to increase the resolution of the data. We believe that the more diverse and numerous the number of sites in such a study, the stronger the results. Alternative sources of higher resolution environmental data should be considered. For example, higher resolution SST data would improve our understanding on the influences of within-sites/reefs/reef zones thermal stress and temperature variability.

The strongest negative influence on coral reef resilience we found in this study was dissolved organic matter of terrestrial origin. Terrestrial pollution and nutrients beyond certain thresholds can reduces coral reef resilience by promoting growth of algae and suspension feeding invertebrates over hard coral. Though to some degree turbidity can be important in screening corals from direct UV radiation, we find that overall it is a limiting factor to the overall resilience of the ecosystem and that it potentially interacts synergistically with other stressors to reduce resilience (McClanahan and Obura 1997; Maina et al. 2011).

Interestingly the results yield a weak positive correlation between population density and overall health, although it should be noted that the population data used were obtained from a globally modeled dataset with very coarse resolution. This result should not be considered conclusive and more research needs to be conducted to explore further the opportunities and challenges that human population densities can bring to coral reef management. It should be noted with caution that this result alone cannot be relied upon to conclude that the number of people living in an area determines the resilience of the coral reef but that it should be considered together with other anthropogenic, ecological and physical factors. Collection of primary data on the factors considered in the IUCN methodology (Obura and Grimsditch 2009) should be primarily emphasized and modeled data can be used to compliment such data. The patterns displayed in this study could possibly mean that MPAs that harbour highly resilient sites are found in areas with high human population density. The relatively lower resilience scores of coral reefs in the Kiunga-Lamu area are also dependent on a range of oceanographic and biological factors and not just population density, and these coral reefs are considered marginal and occur in less populated areas. This result however presents us with an opportunity to note that the fact that higher populations are around resilient reefs presents the risk of eroding the resilience of these coral reefs in the future. There is thus a need to identify and/or prioritize the conservation and management of more sites; particularly those located away from such high population density areas. In such a setting, a spatial conservation planning exercise is necessary to identify these areas and recommendations thereof to managers with solid justifications on such prioritization.

Mapping of coral reef resilience factors presents a holistic picture of the variability in coral health between reefs in a site, between sites within a location and also between locations along a coastline. It also presents a quick look of what factors influence the overall resilience of a reef. An overlay of this with other environmental data can present managers with various scenarios to what reefs or sites should be prioritized for conservation and also gives factual evidences to support such prioritization. Ecological models can be used to integrate relevant geographic datasets such as coral reef resilience, thermal stress, benthic primary production etc. in a process that combines the effect all these variables into metrics that can reflect coral reef resilience.

It is important to note that while advances in remotely sensed technology for mapping coral reefs exist globally, such technology is often lacking in East Africa with the only available global coral reef maps being those from the Millennium Coral Reef Mapping Project (http://oceancolor.gsfc.nasa.gov/cgi/landsat.pl). These maps are however unvalidated and their usage is thus limited to research and with limited value for conservation planning purposes. Future studies in coral reef resilience for management in the region could include the creation of more detailed maps of coral reef habitats from satellite images and validation of the same using geo-referenced data on coral reef resilience factors.

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Appendix

Table showing the site names

Country	Location	Site	Site code
Kenya	Mombasa	Kanamai	Mo1
Kenya	Mombasa	Nyali	Mo2
Kenya	Mombasa	Mombasa Coral Garden	Mo3
Kenya	Mombasa	Ras Iwatine	Mo4
Kenya	Mombasa	Shelly	Mo5
Kenya	Mombasa	Likoni	Mo6
Kenya	Mombasa	Shark Point	Mo7
Kenya	Mombasa	Kasa	Mo8
Kenya	Shimoni	Mako Kokwe	Sh1
Kenya	Shimoni	Upper Mpunguti Inner	Sh2
Kenya	Shimoni	Upper Mpunguti Outer	Sh3
Kenya	Shimoni	Kisite Inner	Sh4
Kenya	Shimoni	Kisite Dive Point	Sh5
Kenya	Shimoni	Lower Mpunguti	Sh6
Kenya	Shimoni	Mkwiro	Sh7
Kenya	Kiwaiyu	Mkokoni	Ki1
Kenya	Kiwaiyu	Shimo la Tewa	Ki2
Kenya	Kiwaiyu	Mike's Outer	Ki3
Kenya	Kiwaiyu	Mlango wa Hindi	Ki4
Kenya	Kiwaiyu	Chongo cha Muhindi	Ki5
Kenya	Kiunga	Mongo Shariff	Ki6
Kenya	Kiunga	Kui	Ki7
Kenya	Kiunga	Bomani	Ki8
Kenya	Kiunga	Chole	Ki9
Kenya	Kiunga	Mwamba Mkuu	Ki10
Kenya	Kiunga	Chongo cha Boso	Ki11
Kenya	Kiunga	Boso	Ki12
Kenya	Kiunga	Kijiweni	Ki13
Kenya	Kiwaiyu	Mike's Inner	Ki14
Kenya	Kiwaiyu	Chongo cha Chano	Ki15
Kenya	Lamu	Tenewi North	La1
Kenya	Lamu	Tenewi South	La2
Kenya	Lamu	Kinyika	La3
Kenya	Lamu	Mwamba Kitau	La4
Kenya	Lamu	Pezali Rock	La5
Kenya	Lamu	Iweni	La6
Kenya	Malindi	Malindi New Coral Gardens	Ma1
Kenya	Malindi	Malindi Old Coral Gardens	Ma2
Kenya	Malindi	Mayungu	Ma3
Kenya	Malindi	North Reef	Ma4
Kenya	Malindi	Watamu Coral Garden	Ma5
Kenya	Malindi	Richard Bennett	Ma6
Kenya	Malindi	Turtle Reef	Ma7
Kenya	Malindi	Pothole	Ma8
Tanzania	Pemba	Misali	Pe01
Tanzania	Pemba	Manta	Pe02
Tanzania	Pemba	Fundo Gap	Pe03
Tanzania	Pemba	Shimba	Pe04
Tanzania	Pemba	Kokota	Pe05
Tanzania	Pemba	Mandela	Pe06
Tanzania	Pemba	Swiss	Pe07
Tanzania	Pemba	Msuka Bay	Pe08
Tanzania	Pemba	The Hole	Pe09
Tanzania	Pemba	Njao Gap	Pe10
Tanzania	Pemba	Fundo Outer	Pe11
Tanzania	Pemba	Fundo Lagoon	Pe12
Tanzania	Pemba	Paradise	Pe13

References

Bell PRF (1992) Eutrophication and coral reefs: some examples in the Great Barrier Reef lagoon. Water Res 26:553–568

Berkes F et al (eds) (2003) Navigating social–ecological systems: building resilience for complexity and change. Cambridge University Press, Cambridge

Buddemeier RW, Kleypas JA, Aronson R (2004) Coral reefs and global climate change: potential contributions of climate change to stresses

on coral reef ecosystems. Prepared for the Pew Center on Global Climate Change, Arlington
Cheal AJ, MacNeil MA, Cripps E, Emslie MJ, Jonker M et al (2010) Coralmacroalgalphase shifts or reef resilience: links with diversity and functional roles of herbivorous fishes on the Great Barrier Reef. Coral Reefs 29:1005–1015
Cuet P, Naim O, Faure G, Conan JY (1988) Nutrient-rich groundwater impact on benthic communities of La Salina fringing reef (Reunion Island, Indian Ocean): preliminary results. In: Proceedings of the sixth international coral reef symposium, vol 2, pp 207–212
Donner SD (2009) Coping with commitment: projected thermal stress on coralreefs under different future scenarios. PLoS One 4:e5712
Dunne T (1979) Sediment yield and land use in tropical catchments. J Hydrol 42:281–300
Ehler C, DouEvere F (2007) Visions for a sea change. Report of the First International Workshop on Marine Spatial Planning, Intergovernmental Oceanographic Commission and man and the Biosphere Programme, IOC manual no. 48, IOCAM Dossier, no. 4, Paris
Finn D (1983) Land use and abuse in the East African region. Ambio 12:296–301
Folke C et al (2004) Regime shifts, resilience and biodiversity in ecosystem management. Annu Rev Ecol Syst 35:557–581
Ginsburg NR (1994) Proceedings of the colloquium on global aspects of coral reefs health, hazards and history. University of Miami, Rosensteil School of Marine and Atmospheric Science, Miami
Graham NAJ, Nash KL, Kool JT (2011) Coral reef recovery dynamics in a changing world. Coral Reefs 30:283–294
Grimsditch G, Mwaura J, Amiyo N (2010) The effect of habitat of coral bleaching responses in Kenya. Ambio 39(4):295–304
Hansen LJ, Biringer JL, Hoffmann JR (2003) BUYING TIME: a user's manual for building resistance and resilience to climate change in natural systems. WWF, Washington, DC
Hoegh-Guldberg O (1999) Climate change, coral bleaching and the future of the world's coral reefs. Mar Freshw Res 50:839–866
Hoegh-Guldberg O, Bruno JF (2010) The impact of climate change on the world's marine ecosystems. Science 328:1423–1428
Holling CS, Schindler DW, Walker BW, Roughgarden J (1995) Biodiversity in the functioning of ecosystems: an ecological synthesis. In: Perrings CA, Maler K-G, Folke C, Holling CS, Jansson B-O (eds) Biodiversity loss, ecological and economical issues. Cambridge University Press, Cambridge
Hughes TP, Baird AH, Bellwood DR, Card M, Connolly SR, Folke C, Grosberg R, Hoegh-Guldberg O, Jackson JBC, Kleypas J, Lough JM, Marshall P, Nystroem M, Palumbi SR, Pandolfi JM, Rosen B, Roughgarden J (2003) Climate change, human impacts, and the resilience of coral reefs. Science 301:929–933
Knudby A, Jupiter S, Roelfsema C, Lyons M, Phinn S (2013) Mapping coral reef resilience indicators using field and remotely sensed data. Remote Sens 5(3):1311–1334
Knudby A, Pittman SJ, Maina J, Rowlands G (2014) Remote sensing and modeling of coral reef resilience. In Finkl CW, Makowski C (eds) Remote sensing and modeling: advances in coastal and marine resources, Coastal research library 9. Springer, Cham. doi:10.1007/978-3-319-06326-3_5
Lambo AL, Ormond RFG (2006) Continued post-bleaching decline and changed benthic community of a Kenyan coral reef. Mar Pollut Bull 52(2006):1617–1624
Lapointe BE, Littler MM, Littler DS (1997) Macroalgal overgrowth of fringing coral reefs at discovery Bay, Jamaica: bottom-up versus top-down control. In: Proceedings of the eighth international coral reef symposium (Panama City, 1997), vol 1. Smithsonian Tropical Research Institute, Panama City, Panama, pp 927–932
Littler MM, Littler DS, Titlyanov EA (1991) Comparisons of N- and P-limited productivity between high granitic islands versus low carbonate atolls in the Seychelles Archipelago: a test of the relative-dominance paradigm. Coral Reefs 10:199–209
Maina J, Venus V, McClanahan TR, Ateweberhan M (2008) Modelling susceptibility of coral reefs to environmental stress using remote sensing data and GIS models in the western Indian Ocean. Ecol Model 212:180–199
Maina J, McClanahan TR, Venus V, Ateweberhan M, Madin J (2011) Global gradients of coral exposure to environmental stresses and implications for local management. PLoS One 6(8):e23064. doi:10.1371/journal.pone.0023064
Maynard JA, Marshall PA, Johnson JE, Harman S (2010) Building resilience into practical conservation: identifying local management responses to global climate change in the southern Great Barrier Reef. Coral Reefs 29:381–391
McClanahan TR (1999) Is there a future for coral reef parks in poor tropical countries? Coral Reefs 18:321–325
McClanahan TR (2002) The near future of coral reefs. Environ Conserv 29:460–483
McClanahan TR (2008) Response of the coral reef benthos and herbivory to fishery closure management and the 1998 ENSO disturbance. Oecologia 155:169–177
McClanahan TR, Obura D (1997) Sedimentation effects on shallow coral communities in Kenya. J Exp Mar Biol Ecol 209:103–122
McClanahan TR, Muthiga NA, Mangi S (2001) Coral and algal changes after the 1998 coral bleaching: interaction with reef management and herbivores on Kenyan reefs. Coral Reefs 19:380–391
McClanahan TR, Cokos B, Sala E (2002) Algal growth and species composition under experimental control of herbivory, phosphorus and coral abundance in Glovers Reef, Belize. Mar Pollut Bull 44:441–451
McClanahan TR, Ateweberhan M, Muhando C, Maina J, Mohammed SM (2007) Effects of climate and seawater temperature variation on coral bleaching and mortality. Ecol Monogr 77:503–525
McClanahan TR, Maina JM, Muthiga NA (2011) Associations between climate stress and coral reef diversity in the Western Indian Ocean. Glob Chang Biol 17:2023–2032
McClanahan TR, Donner SD, Maynard JA, MacNeil MA, Graham NAJ et al (2012) Prioritizing Key resilience indicators to support coral reef management in a changing climate. PLoS One 7(8):e42884. doi:10.1371/journal.pone.0042884
Moberg F, Folke C (1999) Ecological goods and services of coral reef ecosystems. Ecol Econ 29:215
Mora C, Aburto-Oropeza O, Ayala Bocos A, Ayotte PM, Banks S et al (2011) Global human footprint on the linkage between biodiversity and ecosystem functioning in reef fishes. PLoS Biol 9:e1000606
Morel A, Be'langer S (2006) Improved detection of turbid waters from ocean colour sensors information. Remote Sens Environ 102:237–249
Morel A, Prieur L (1977) Analysis of variations in ocean color. Limnol Oceanogr 22:709–722
Nyström M, Folke C (2001) Spatial resilience of coral reefs. Ecosystems 4:406–417
Nyström M et al (2000) Coral reef disturbance and resilience in a human-dominated environment. Trends Ecol Evol 15:413–417
Obura DO, Grimsditch G (2009) Resilience assessment of coral reefs–Assessment protocol for coral reefs, focusing on coral bleaching and thermal stress. IUCN Working Group on Climate Change and Coral Reefs. IUCN, Gland, Switzerland, 70 pages
Risk MJ, Sammarco PW, Edinger EN (1995) Bioerosion in *Acropora* across the continental shelf of the Great Barrier Reef. Coral Reefs 14:79–86
Robinson IS, Donlon JC (2000) Global measurement of sea surface temperature: some new perspective. Oceanic Sci 234:111–115
Rose CS, Risk MJ (1985) Increase in *Cliona delitrix* infestation of *Montastrea cavernosa* heads on an organically polluted portion of the Grand Cayman fringing reef. Mar Ecol 6:345–363
Rowlands G, Purkis S, Riegl B, Metsamaa L, Bruckner A, Renaud P (2012) Satellite imaging coral reef resilience at regional scale. A case-study from Saudi Arabia. Mar Pollut Bull 64(6):1222–1237

Sandin SA, Smith JE, DeMartini EE, Dinsdale EA, Donner SD et al (2008) Baselines and degradation of coral reefs in the Northern Line Islands. PLoS One 3:e1548

Smith SV, Kimmerer WJ, Laws EA, Brock RE, Walsh TW (1981) Kaneohe Bay sewage diversion experiment: perspectives on ecosystem responses to nutritional perturbation. Pac Sci 35:279–402

Wang H, Hladik CM, Huang W, Milla K, Edmiston L et al (2010) Detecting the spatial and temporal variability of chlorophyll-a concentration and total suspended solids in Apalachicola Bay, Florida using MODIS imagery. Int J Remote Sens 3:439–453

Wilkinson C (ed) (2002) Status of coral reefs of the world. Australian Institute of Marine Science, Townsville

Wilson SK, Graham NAJ, Pratchett MS, Jones GP, Polunin NVC (2006) Multiple disturbances and the global degradation of coral reefs: are reef fishes at risk or resilient? Glob Chang Biol 12:2220–2234

Coral Recruitment and Coral Reef Resilience on Pemba Island, Tanzania

Gabriel Grimsditch, Jerker Tamelander, Jelvas Mwaura, Monica Zavagli, Yukari Takata, and Tanausu Gomez

Abstract

This study explores the patterns of coral recruitment, survivorship and resilience on the coral reefs of the west coast of Pemba Island, Tanzania. The results show that recovery from the 1998 mass coral bleaching event has been patchy, with great variation in coral cover among sites, and a generally high macroalgal cover. Sites with low coral recovery were found to exhibit higher numbers of coral recruits but lower survivorship, implying that larval supply is not impeding recovery but rather that local stressors are reducing coral reef resilience. The main stressors observed were predation by *Acanthaster plancii*, overfishing and use of destructive fishing methods (including dynamite fishing). *A. plancii* predation was shown to negatively correlate with coral recruit survivorship, implying that it is a potential cause of failure of corals to reach adult sizes. Fish surveys showed that Pemba is being overfished, with the vast majority of fish observed less than 10 cm in length, and only 4 individuals larger than 40 cm recorded throughout the whole survey. It is recommended that the two major, and potentially synergistic, stressors of *A. plancii* predation and overfishing/destructive fishing are addressed in order to avoid loss of Pemba's coral reefs. Land-ocean connections are also explored in this context.

Keywords

Coral recruitment • Resilience • Pemba

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Introduction

The island of Pemba lies 50 km off the northern Tanzanian Coast in the Indian Ocean, and forms part of the Zanzibar Archipelago. The climate is tropical and can be broadly divided into two monsoon periods, the northeast monsoon ('kaskazi') with trade winds blowing from the northeast between December and April, and the southwest monsoon ('kusi') with trade winds blowing from the southwest between May and November. The Northeast monsoon is generally characterized by lower wind speeds, calmer seas and higher sea surface temperatures. The doldrums at the end of the northeast monsoon is the usual coral bleaching period in this region. The southwest monsoon is generally characterized by higher wind speeds, rougher seas and lower water temperature (Newell 1959).

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The shoreline consists of stretches of sandy beach interspersed with low limestone cliffs and headlands. Offshore, there are shallow fringing reef flats which drop off rapidly into the ~2,000 metre deep Pemba Channel. There are several smaller islands in the area to the west of Pemba, creating tidal channels between them and sheltered lagoon-like areas behind them. The west coast of Pemba is more protected from oceanic swell and generally exhibits lower wave energy than the east coast, which is more exposed to oceanic swell and where wave energy is generally higher. The tidal range for Pemba is relatively large, with the neighbouring Zanzibar Island (Unguja) harbour exhibiting a mean spring tidal range of 3.9 m (Nhnyete and Mahongo 2007). The islands (from north to south) to the west of Pemba include Njao, Fundo (the largest of the islands), Uvinje and Misali. Coral reefs surround the islands and are present in the tidal channels, lagoons, bays and fringing the western edges of the islands (Grimsditch et al. 2009).

The coral reefs of Pemba are not in the centre of coral diversity for the Western Indian Ocean, which lies in the northern Mozambique channel and from which diversity decreases radially (Obura 2012). However, on the gradient of diversity they are among the ‘second tier’ of the most diverse coral reefs in East Africa and locally extremely important. The local human population relies on them heavily for food security and income from fishing, aquaculture and increasingly SCUBA diving tourism. However, these corals are also vulnerable to bleaching caused by warming sea surface temperatures. In 1998 they bleached heavily and live coral cover decreased from 54% average around the island to 12% in 1999 (Obura 2002). In one particular site on Misali, a study found that live coral cover decreased from 74% to 17% during 1998 (Muhando 2003). In the 5 years after bleaching, the corals around Pemba recovered modestly up to an average of 16% live cover in 2002 with variable recovery around the island (Obura 2002). Overfishing and the use destructive fishing methods are additional threats to the corals of Pemba. Most fishers in the area have low incomes and use traditional fishing boats such as outrigger and dugout sailed canoes with hand lines, beach seines and fish traps. The fishers cannot usually access offshore areas due to their boats’ constraints, so most fishing is carried out relatively close to shore and the nearby reefs are thus intensely fished. Beach seines, gill nets, and dynamite fishing are typical of the destructive methods in the area that cause significant damage to the coral reef structure and populations. Furthermore, population outbreaks of the corallivorous *Acanthaster plancii* have occurred on Pemba’s reefs in the last decade (Obura et al. 2004). Although this organism is a natural part of the coral reef ecosystem, population outbreaks can cause severe and widespread coral mortality. These drivers of coral mortality can reduce coral reef resilience over time, and lead to trajectories of ecological degradation that can have negative socioeconomic implications as ecosystem services associated with coral reefs (for example shoreline protection, food security from fisheries, biodiversity values, recreational values and medicinal values) are reduced and eventually lost. Understanding the factors affecting coral mortality and coral reef recovery or resilience is important for improving the management of coral reefs. In particular, it is important to understand whether coral recruitment is a limiting factor in coral reef recovery or whether mortality (and which causes of mortality) are hindering coral reef recovery.

Coral reef resilience can be defined as ‘following mortality of corals, the ability of the reef community to maintain or restore structure and function and remain in an equivalent ‘phase’ as before the coral mortality’ (Obura and Grimsditch 2009). Mortality could occur, for example, due to bleaching, *A. plancii* predation and/or destructive fishing practices. Coral reef resilience is dependent on a myriad of ecological, physical, oceanographic and anthropogenic factors and the complex interactions between them (Obura 2005). Because of the complex nature of these interactions, it is challenging to determine which factors are the most important for driving resilience. One important factor is coral recruitment that replenishes coral populations after a mortality event, and this paper will present an analysis of spatial patterns for coral recruitment for selected sites along the west coast of Pemba Island, linking recruitment patterns to potential resilience of the sites surveyed and potential drivers of coral reef degradation. The potential for land-ocean connections to affect coral recruitment in Pemba is also discussed.

Materials and Methods

Study sites – Surveys were conducted in February 2009 on 13 sites along the fringing reefs along the west coast of Pemba Island. Table 1 names, describes and locates the sites surveyed, as well as the survey depth. Sites sampled could broadly be categorized into three geomorphological types: sheltered bays (The Hole, Fundo Lagoon and Msuka Bay), fringing reef slopes (Simba Wall, Paradise, Swiss, Mandela, Fundo Outer and Misali) and tidal channels (Njao Gap, Manta, Fundo Inner and Kokota). The depth of the surveys varied according to site in order to sample the most representative areas of the reef ecosystem.

General sampling design – Fieldwork followed the methods outlined in the ‘Resilience Assessment for Coral Reefs’ protocol (Obura and Grimsditch 2009) and is described below.

Benthic surveys – Coral size classes were measured along two or three 25 × 1 m transects per site counting all corals larger than 10 cm, identifying them to genus level and placing them in appropriate size classes according to

Table 1 Shows the sites, sampling depths, geographical position and habitat for the sites surveyed for this study

Site name	Sampling depth (m)	Lat. (S)	Long. (E)	Site description
The Hole	6	4.88720	39.67632	Sheltered bay
Simba Wall	9	4.87575	39.67349	Fringing reef slope
Paradise	13	4.91282	39.67093	Fringing reef slope
Swiss	18	4.86786	39.67046	Fringing reef slope
Njao Gap	10	4.95911	39.66748	Tidal channel
Mandela	8	4.99694	39.65576	Fringing reef slope
Manta	10	5.00146	39.65607	Tidal channel
Fundo Inner	9	5.00924	39.66755	Tidal channel
Fundo Outer	10	5.04207	39.64161	Fringing reef slope
Fundo Lagoon	3	5.02569	39.67309	Sheltered bay
Msuka Bay	4	4.86164	39.7036	Sheltered bay
Misali	9	5.23958	39.5952	Fringing reef slope
Kokota	11	5.1374	39.63824	Tidal channel

diameter (10–20 cm, 20–40 cm, 40–80 cm, 80–160 cm, 160–320 cm and >320 cm). Corals smaller than 10 cm were measured in six 1 m^2 quadrats placed along the same transects. These smaller corals were also divided into size classes (0–2.5 cm, 2.5–5 cm and 5–10 cm). Size class data were collected for a restricted set of commonly occurring coral genera representing a range from low to high bleaching susceptibility as described in the literature for the region (McClanahan 2004; Obura 2001). The genera sampled were grouped into 4 distinct groups: Susceptible to bleaching stress (*Acropora*, *Pocillopora*, *Stylophora*, *Seriatopora* and *Montipora*), Resistant (*Porites* massive and *Pavona*), Moderate tolerance Faviidae (*Favia, Favites*, *Leptastrea*, *Echinopora* and *Platygyra*) and Moderate tolerance non-Faviidae (*Galaxea*, *Porites* branching, *Lobophyllia*, *Fungia*, *Hydnophora* and *Coscinarea*). Incidence of coral disease, predation or bleaching were also recorded along the transects and presented as a percentage of all colonies in the belt.

Fish surveys – Sampling was combined in one long swim, to maximize sampling of the large mobile fish (e.g. bumphead parrotfish), with 3 replicate transects for density estimates of fish. The long swim consisted of a 20 minute timed swim at a standardized swimming speed parallel to the reef axis. The area sampled was approximately 10 m on either side of the observer and only the largest size classes of key genera were recorded to genus level. Three transects were then undertaken using 50 × 5 m belt transects for the remainder of target families (Acanthuridae, Scaridae, Kyphosidae, Siganidae, Serranidae, Haemulidae and Mullidae), and all fish observed were recorded to genus level, their length estimated and placed into appropriate 5 cm size classes (5–10 cm, 10–15 cm, 15–20 cm, 20–25 cm, 25–30 cm, etc). Herbivorous fish were classified into different functional groups depending on their feeding modes and preferred diet (large excavators, small excavators, scrapers, grazers and browsers; Green and Bellwood 2009). Predatory fish which are commercially important and good indicators of fishing pressure were also surveyed.

Results and Discussion

The average live hard coral cover around the island was found to be 23%, with large variations from 86% in the Misali (a no-take area) to only 3% and 5% in highly degraded sites such as Paradise and Fundo Outer, respectively (Fig. 1). The overall trend shows further recovery from the 1998 bleaching event compared to the 16% live cover recorded in 2002; however, the figures are highly variable and specific comparisons for sites are often not possible (Obura 2002). One site that can be compared directly to previous literature is Misali, where live coral cover was recorded to 17% shortly after the 1998 bleaching (Muhando 2003), and was recorded to 86% in 2009. Coral reef conditions were highly variable, with some sites (Misali, Mandela and Manta) being dominated by hard coral, whilst others (Paradise and Fundo Outer) were dominated by rubble and turf algae. In total, 47 hard coral genera were recorded, with Misali having the highest diversity with 42 genera recorded and Paradise having the lowest diversity with 23 genera recorded (Fig. 2). *Acropora*, massive *Porites* and *Ecninopora* dominated the hard coral cover, accounting for 46% of coral area, while *Pocillopora* were by far the most numerous colonies, accounting for 24% of all coral colonies (Fig. 3). This corresponds to the life history strategies of the genera. *Pocillopora* is an early colonizer that reproduces quickly and colonizes disturbed environments but does not grow to a large size or old age compared to other genera. *Acropora* is fast-growing, susceptible to bleaching and typical for an undisturbed reef. At the time of the survey, only Misali and Mandela were dominated by *Acropora*, whilst the rest of the sites were dominated by more stress-resistant genera such as the slow-growing massive *Porites* (Fig. 4).

Coral recruitment is considered an important characteristic for driving ecological resilience of coral reef systems by allowing coral populations to replenish after a mortality event (Lukoschek et al. 2013; Nyström et al. 2008). This not only encompasses a plentiful larval supply and the successful settling of larvae at a site, but also post settlement

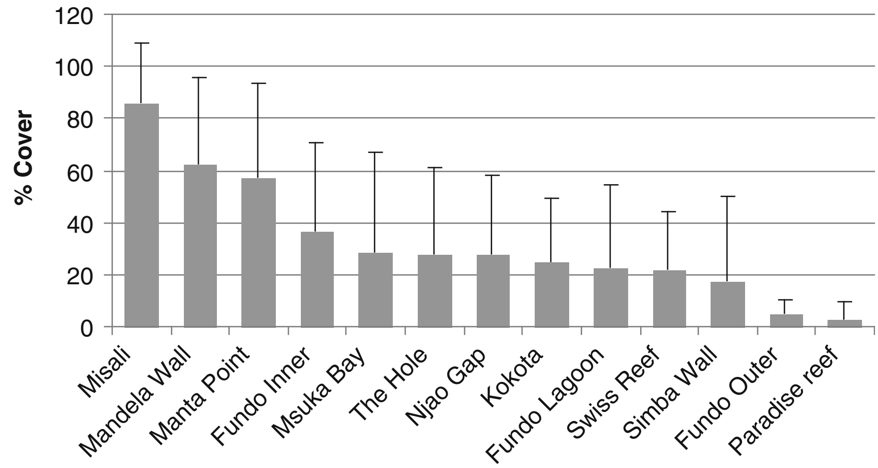

Fig. 1 Depicts live hard coral cover per site. Coral cover varied greatly from a high cover of 86% in Misali (no-take zone) to low cover of 3% in Paradise Reef and 5% in Fundo Outer, two highly degraded sites

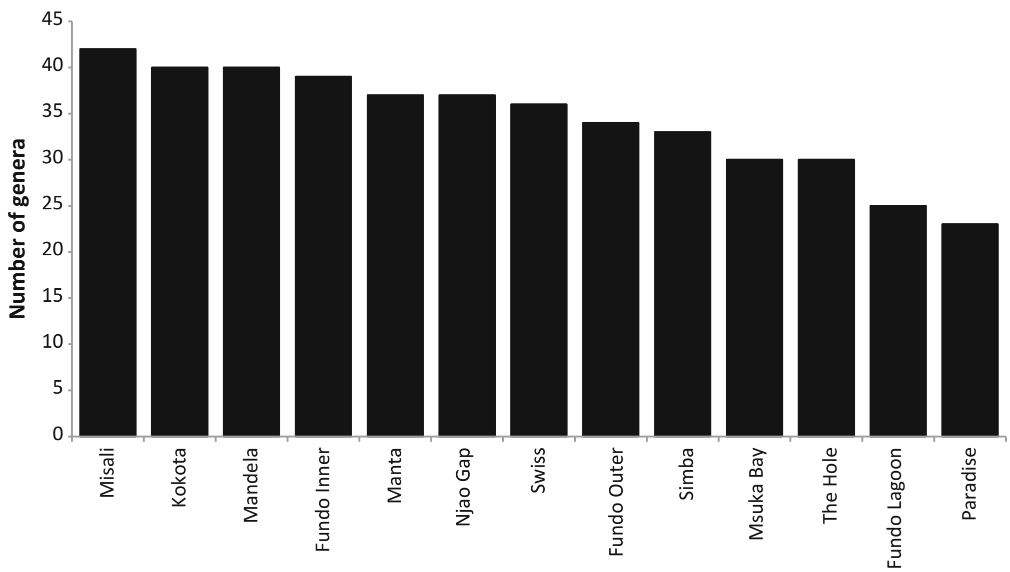

Fig. 2 Depicts coral diversity by site. The total number of coral genera found was 47, with a maximum of 42 found in Misali and a minimum of 23 in Paradise. Fundo Lagoon, The Hole and Msuka Bay also had relatively low diversity

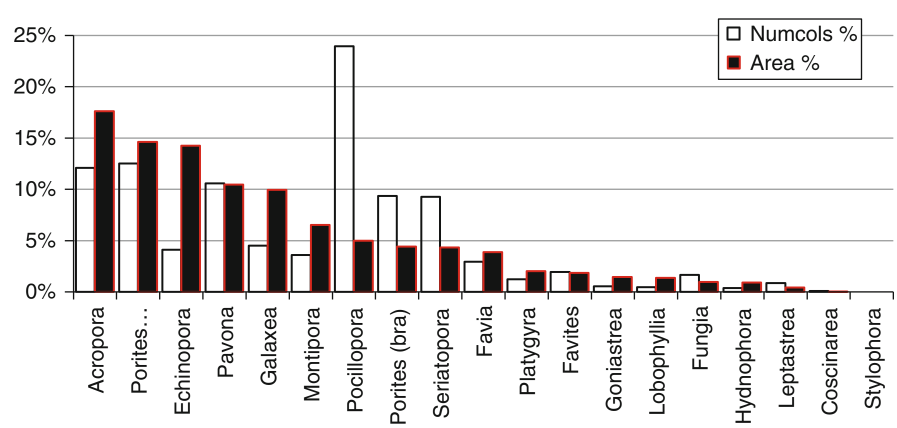

Fig. 3 Depicts the number of colonies and area covered by genus. Acropora, Porites massive and Ecninopora dominate coral cover, accounting for 46% of the total coral area, while Pocillopora is by far the most numerous genus, accounting for 24% of all coral colonies

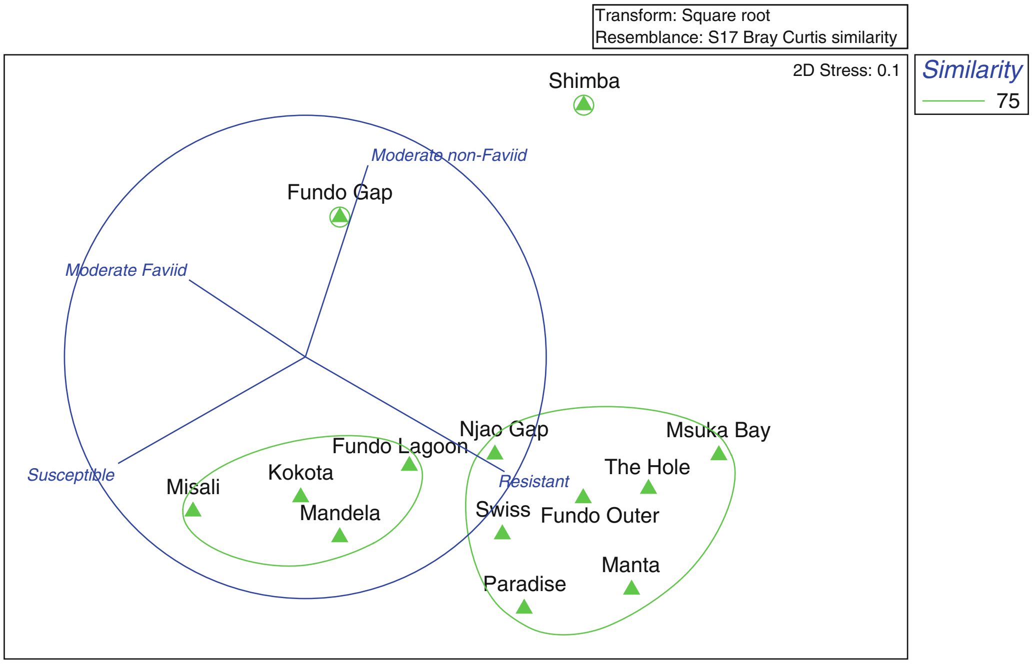

Fig. 4 Depicts genus susceptibility by site. Genera were categorized into groups depending on their bleaching responses. Four groups were identified – Susceptible (Acropora, Pocillopora, Stylophora, Seriatopora and Montipora), Resistant (Porites massive and Pavona), Moderate tolerance Faviidae (Favia, Favites, Leptastrea, Echinopora and Platygyra) and Moderate tolerance non-Faviidae (Galaxea, Porites branching, Lobophyllia, Fungia, Hydnophora and Coscinarea). The proportion of total coral cover occupied by each bleaching response group was calculated, and sites were compared using Multi-Dimensional Scaling analysis

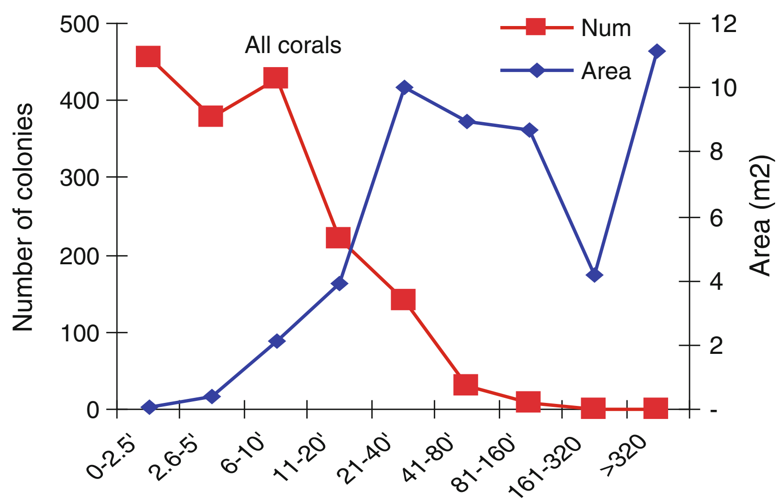

Fig. 5 Depicts overall size class distribution for all coral colonies recorded. The distribution of size classes is shown by number of colonies, and by area of colonies for all size classes. On average, there were 1665 colonies in an area of 100 m^2, corresponding to 49.5 m^2 of coral colony surface. The dominant size classes by area, were >320 cm, 21–40 cm, 41–80 cm and 81–160 cm

survival and growth into juveniles and then adults (Ho and Dai 2014; Martinez and Abelson 2013). Coral size class distributions can be indicative of the history of mortality of reefs' coral populations (Zvuloni et al. 2008). Large-scale coral mortality events caused by bleaching or other factors are known to cause reduced fecundity and recruitment in coral populations (Hoegh-Guldberg 1999). Periods of mortality could thus be reflected in the size class structure of Pemba's coral reef community. This study found that the coral size class distribution of Pemba's reefs showed reduced numbers of corals sized 2.5 to 5 cm and 1.6 to 3.2 m (Fig. 5). The coral size class analysis was conducted both including and excluding *Pocillopora* colonies in order to ascertain the influence on coral size distribution exerted by *Pocillopora* given that it is the genus with the highest number of colonies and could disproportionately drive patterns; however, the same pattern was apparent even excluding *Pocillopora* from the analysis. The dip in the

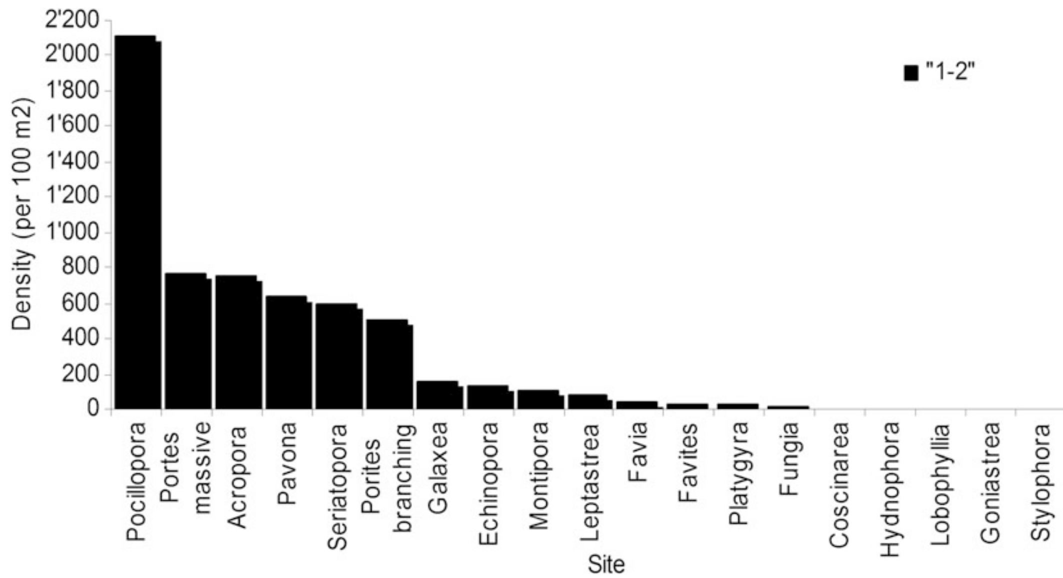

Fig. 6 Shows that recruitment by genus. Recruitment is largely dominated by Pocillopora, followed by Acropora, Porites massive, Pavona, Seriatopora and Porites branching

population of corals sized 1.6 to 3.2 m is probably indicative of a past mass mortality event, possibly the 1998 bleaching event, which is known to have greatly affected Pemba's coral populations (Muhando 2003). However, the dip in the population of corals sized 2.5 to 5 cm indicates more recent failures in recruitment and survivorship, and is probably related to recent 'pulse' stress events such as bleaching and *A. plancii* outbreaks or more ongoing 'press' mortality and stress from destructive fishing practices (Visram et al. 2008; Kayal et al. 2012; Fox and Caldwell 2006). This failure in recent recruitment and survivorship indicates that these stressors could further affect future recovery of Pemba's coral populations. Overall, coral recruitment is largely dominated by *Pocillopora* (Fig. 6), an early colonizer that can reproduce asexually via fragmentation as well as sexually through spawning and production of larvae.

Recruitment patterns also vary spatially, giving indications of recovery potential at different sites and for different reef types. We observed that in most sites the number of small recruits (0–2.5 cm) was healthy and comparable to results of other studies on recruitment in the region (Tamelander 2002); however, post-settlement mortality is not allowing recruits to survive into adults in degraded sites. It was found that coral recruitment was generally higher at degraded sites than in sites with higher live coral cover. A correlation analysis between coral cover and recruitment found a weak negative correlation between the two variables ($r = -0.16$), indicating that sites with higher coral cover can display lower recruitment. The sites with the highest recruitment rates were Simba, Fundo Outer and Paradise (670, 641 and 623 corals sized 0–2.5 cm per 100 m^2 per site, respectively), but these sites had the lowest coral cover (18%, 5% and 3%, respectively) and were instead dominated by turf algae. The sites with the highest live coral cover, i.e. Misali, Manta or Mandela (86%, 59% and 62% respectively) were all found to have relatively low recruitment rates (385, 418 and 572 corals sized 0–2.5 cm per 100 m^2, respectively), possibly because less suitable substrate was available for coral larvae to colonize as the area has already being occupied by larger corals (Fig. 7). This indicates that the low coral cover in the identified degraded sites (Simba, Fundo Outer and Paradise) was not due to a lack of larval supply and recruitment but rather to a local stress on the site that can have caused mortality of young corals and not allowing them to grow into adults.

Two sites stand out as having relatively low recruitment; Msuka Bay and Fundo Lagoon (Fig. 7), two sheltered and shallow sites with 58 and 85 corals sized 0–2.5 cm per 100 m^2, respectively. This is possibly due to their geographical positioning and substrata. Msuka Bay is located at the north of the island, not well connected to the dominant currents in the area, and is characterized by high wave energy and dominated by *Sargassum* (35% *Sargassum* cover recorded) where it is difficult for coral recruits to settle and grow. Fundo Lagoon is located in a sheltered bay area, and although there is tidal exchange of water, it appears that coral larvae do not successfully settle there. The case of Fundo Lagoon is interesting given that the tidal flushing and exchange is high, so it could be assumed that coral larvae do reach the site. However, Fundo Lagoon was observed to have the highest sediment- and turbidity load of all sites surveyed, and this could be related to lower settlement and survival of small recruits.

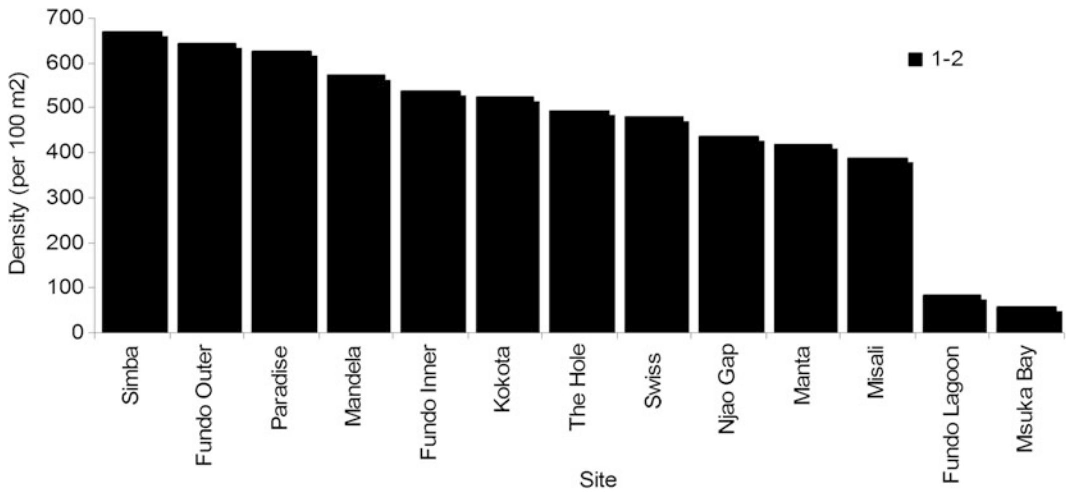

Fig. 7 Depicts coral recruitment. Simba, Fundo Outer, Paradise and Mandela had the highest coral recruitment (0–2.5 cm size of corals) despite having lower coral cover than sites such as Misali, Mandela and Manta

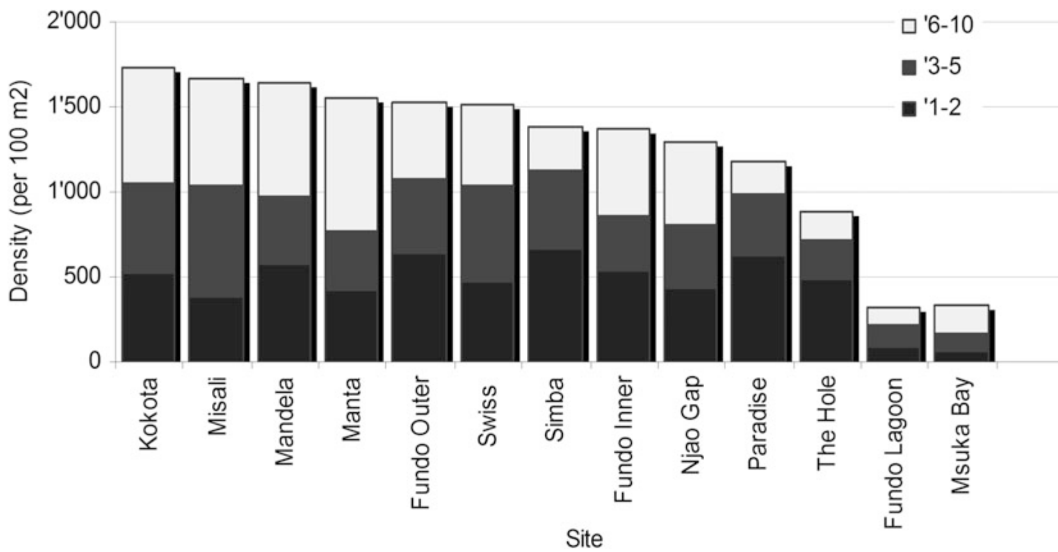

Fig. 8 Depicts the density of smaller corals at different size classes (0–2.5 cm, 2.5-5 cm and 5–10 cm)

The findings suggest that in degraded sites such as Paradise, Simba, Fundo Lagoon and Fundo Outer, lack of coral recruitment is not driving the lack of recovery, but rather that poor survivorship of coral recruits does not allow the reef to regain coral cover and diversity. This finding is further accentuated when we examine the fate of the recruits as they grow older. Although recruitment and the number of corals smaller than 2.5 cm may be higher in degraded sites such as Paradise, Simba and Fundo Outer, the data show that recruit survivorship was much lower than in other sites, meaning that the number of corals sized 2.5–5 cm and 5–10 cm decreases drastically in these degraded sites. If this trend continues, fewer and fewer corals attain larger sizes at these sites (Fig. 8). Figure 9 shows the mortality rates of small corals from the 0–2.5 cm size class to the 5–10 cm size class. Degraded sites such as Paradise, Simba and Fundo Outer exhibited high mortality rates (69%, 62% and 31% respectively), while most other sites did not display such high rates of recruit mortality and instead high survival rates were observed. It is thus apparent that local stress factors and mortality of small corals are eroding resilience in the degraded sites, rather than a lack of larval supply.

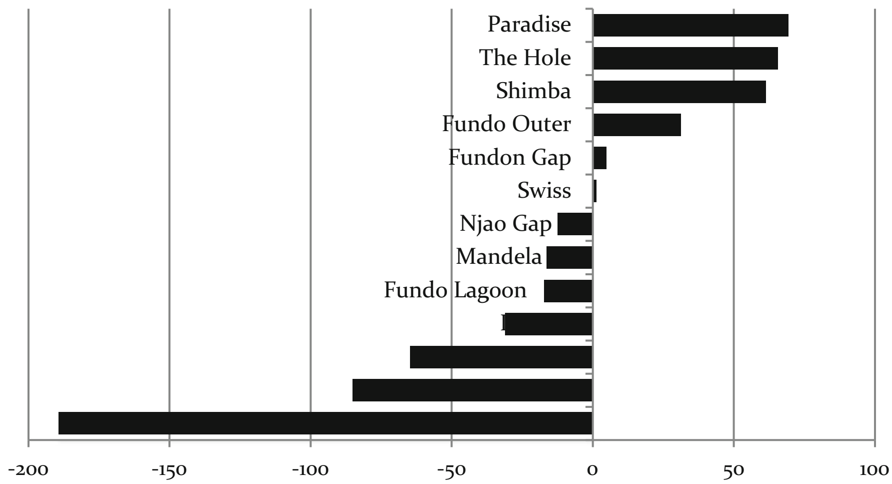

Fig. 9 Depicts the mortality rates (%) of small corals in different sites. Positive numbers mean high mortality, while negative numbers mean low mortality and high survivorship. The figure presents mortality rates from the 0–2.5 cm size class to the 5–10 cm size class

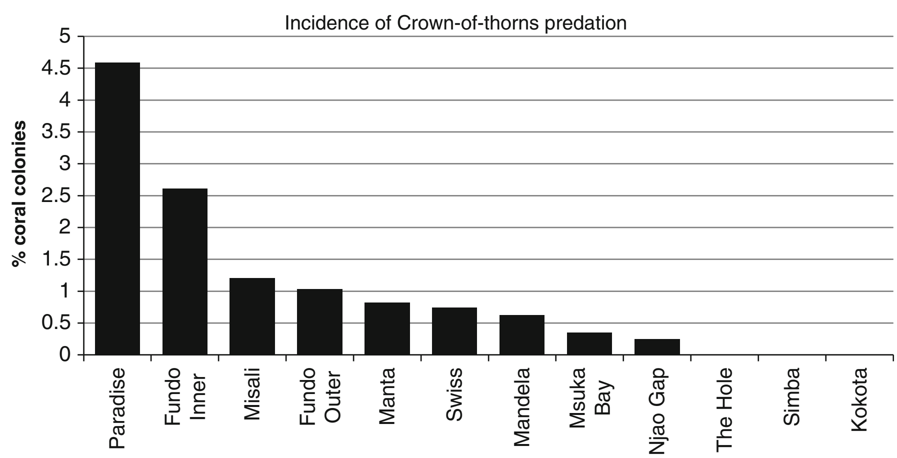

Fig. 10 Shows predation on corals by crown-of-thorns starfish. Paradise had the highest incidence of crown-of-thorns starfish feeding scars observed on coral colonies, with 5% of colonies being predated

Plausible explanations include predation by *A. plancii*, overfishing, destructive fishing and/or further bleaching events since 1998.

At Paradise the high incidence of *A. plancii* predation scars on corals (5% of all colonies surveyed were predated compared to 2.5% at Fundo Inner or 1.2% at Misali, the two sites with the second and third highest incidences of coral predation; Fig. 10) indicates that the corallivores are one cause of coral recruit mortality. An *A. plancii* outbreak was observed at Fundo Inner, with over 50 individuals sighted in one dive. It is unknown what the trigger for population outbreaks is. Removal of predators of *A. plancii* (e.g. triton shells or triggerfish) through overfishing, improved survival of larvae due to land-based nutrient inputs and increasing sea-surface temperature have all been postulated as potential triggers of *A. plancii* outbreaks (Moran 1986; Uthicke et al. 2009). Although no outbreak was observed at Paradise, it still exhibited the highest percentage of feeding scars. Paradise had the lowest coral cover of all sites, and the coral population comprised mostly of smaller sized corals (no corals larger than 40 cm in diameter were observed, and total coral area mostly consisted of colonies of 11–20 cm in size). These small corals were being predated opportunistically by *A. plancii*, showing that even in the absence of an outbreak, the corallivores were causing mortality of coral recruits and exerting pressure on the coral population. A correlation analysis between survivorship of smaller coral size classes and *A. plancii* predation across all sites showed a weak negative correlation between survivorship of the 0–2.5 cm coral size class and *A. plancii* predation ($r = -0.16$), while a moderate negative correlation between survivorship of the 2.5-5 cm coral size class and *A. plancii* predation ($r = -0.32$) was found. *A. plancii* predation thus does appear to be driving mortality of smaller corals to some extent, and more so when corals attain sizes larger than 5 cm.

Another major cause of ecological degradation in Pemba is overfishing and destructive fishing. The data presented

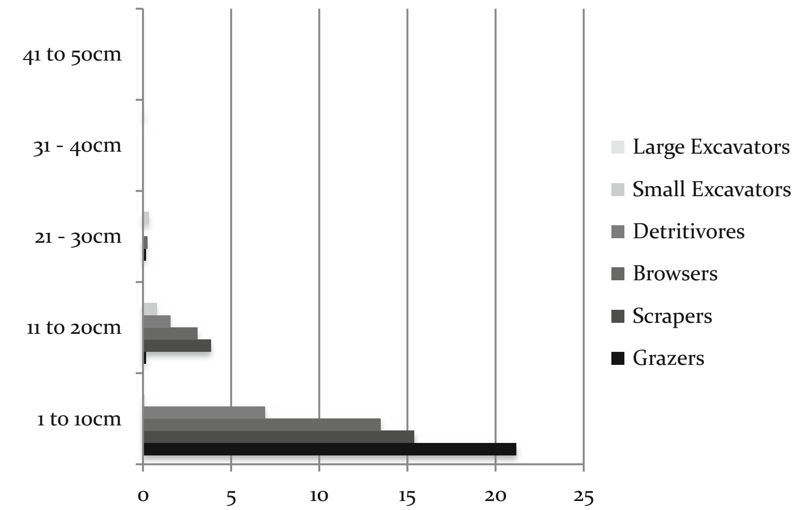

Fig. 11 Shows the overall abundance of herbivores (separated into functional groups) in Pemba according to size class. The vast majority of herbivorous fish on Pemba seen were <10 cm long, indicating overfishing. The absence of large fish is ubiquitous across predators and herbivorous functional groups

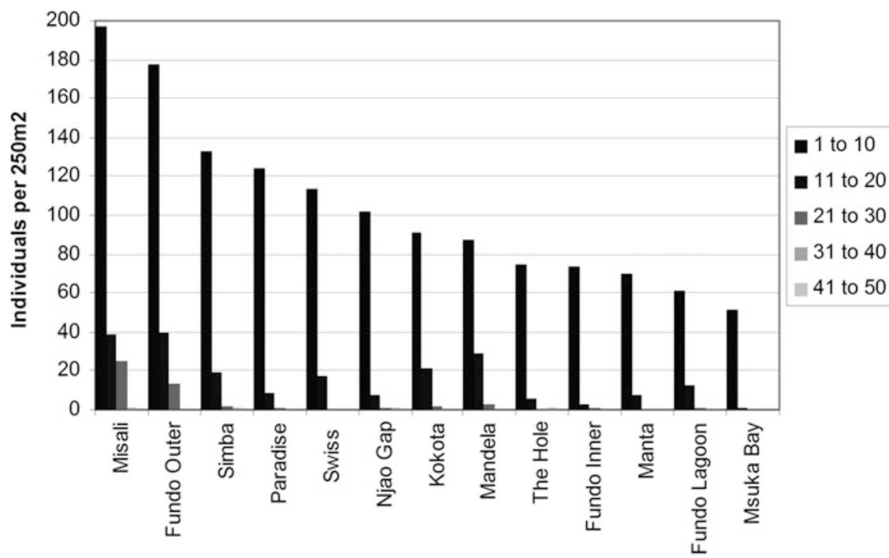

Fig. 12 Shows fish abundance for all species of fish (herbivores and predators) recorded. Misali, which is a no-take zone, had the highest number of large bodied fish, showing that management can impact fish populations

indicates that the fish populations in Pemba are being harvested aggressively. Fish populations in Pemba varied greatly among sites surveyed, from over 250 individuals of all species per 250 m^2 observed in Misali to 50 individuals per 250 m^2 observed in Msuka Bay for all species measured. These fish abundance numbers are comparable to studies from Kenya with the lower numbers comparable to non-protected areas and the higher numbers comparable to protected areas (Munga et al. 2012). Small-bodied herbivorous Acanthuridae and Scaridae were the most common fish observed. Very few (<5 individuals per 250 m^2) were large excavators or commercially valuable Serranidae, Haemulidae or Mullidae, and no sharks were seen; this indicates the overfishing of large bodied predators, herbivores and commercially valuable species. Within the herbivorous functional groups, grazers and scrapers were the most abundant, followed by browsers, while small excavators (<35 cm) were twice as abundant as large excavators. *Bolbometopon* spp., which is a large excavator, was absent (Fig. 11). Again, this is an indication of overfishing, as large bodied excavators are usually among the first to disappear on an overfished reef. Misali, the no-take zone, exhibited the highest fish densities (Fig. 12) but overall the vast majority of fish observed on Pemba were <10 cm long, and only four individuals larger than 40 cm were recorded during the entire survey (Fig. 11). Herbivorous fish are important in regulating algal-coral dynamics, and healthy herbivore populations are important for coral reef resilience as algae can outcompete coral recruits and reduce their survivorship (West and Salm 2003). Evidence of frequent and/or continuous destructive fishing was also evident from anecdotal visual observations. Illegal beach seines have routinely being used in the area and dynamite blasts were regularly heard during dives. These indiscriminate destructive methods not only destroy coral habitat but also remove sexually immature juveniles as well as rarer species. At this point, however, there is insufficient data to correlate overfishing and destructive fishing pressure with coral recruitment and survivorship, and more targeted studies are needed in this area for Pemba. Nevertheless, Paradise and Fundo Outer serve as warnings of what currently 'healthy' sites could look like if ecological resilience continues to be eroded through destructive practices.

Land-based sources of pollution were not observed to have major impacts on Pemba's coral reefs. There is very little coastal development, with only few hotels and fishing villages on the coast. However, nutrients from the seaweed farms that are common in Pemba could be contributing to algal growth. Macroalgal cover was found to be high, with an average of 9% of the sites covered by macroalgae (mostly *Dictyota*, *Cyanophyta*, *Sargassum* and *Jania*), and an average of 28% turf algae cover. Lack of herbivorous fish and grazing is probably the major cause of the high macroalgae cover, with some influence from nutrients; however, more data is needed to understand the cause-effect relationship. Higher nutrient levels have also been linked to *A. plancii* outbreaks due to improved larval survival (Brodie et al. 2005), but there is no evidence of this in Pemba and more research is needed.

Despite the degradation observed in many sites, we have continued to observe live coral cover increasing compared to other surveys since 1998, indicating that some coral populations on parts of the island are able to recover. For example, Misali is currently a no-take zone that has been offered higher protection than other sites surveyed, and it has exhibited significantly healthier populations of both coral and fish. Currently, degraded sites such as Paradise and Fundo Outer could, in theory, also recover to the extent of other sites if the stressors currently causing mortality of recruits and juveniles are identified and, if possible, eliminated. Some of the major stressors found to be driving mortality of coral recruits were *A. plancii* predation, the use of destructive fishing methods and overfishing. Dynamite fishing is illegal in Tanzania with a penalty of at least a 5-year prison term according to the Tanzania Fisheries Act of 2003, yet there was frequent evidence of its use in Pemba. Fishing activity using gill nets and beach seine was also observed in supposedly protected areas. We recommend that these anthropogenic stressors should be monitored and managed more closely in order to avoid further degradation of the coral reefs of Pemba.

Acknowledgements We would like to acknowledge the generous support of the Marine Conservation Unit of the Department of Fisheries and Marine Resources of the Revolutionary Government of Zanzibar (http://mcu.go.tz/), the Manta Lodge Resort (http://www.themantaresort.com/), the MacArthur Foundation (http://www.macfound.org/), the IUCN Climate Change and Coral Reefs Working Group (http://www.iucn.org/cccr/) and Coastal Oceans Research and Development in the Indian Ocean (http://www.cordioea.org/) for their generous support and allowing this study to happen.

References

Brodie J, Fabricius K, De'ath G, Okaji K (2005) Are increased nutrient inputs responsible for more outbreaks of crown-of-thorns starfish? An appraisal of evidence. Mar Pollut Bull 51:266–278

Fox H, Caldwell R (2006) Recovery from blast fishing on coral reefs: a tale of two scales. Ecol Appl 16:1631–1635

Green AL, Bellwood DR (2009) Monitoring functional groups of herbivorous reef fishes as indicators of coral reef resilience – a practical guide for coral reef managers in the Asia Pacific region, IUCN working group on climate change and coral reefs. IUCN, Gland, 70 p

Grimsditch G, Tamelander J, Mwaura J, Zavagli M, Takata Y, Gomez T (2009) Coral reef resilience assessment of the Pemba channel conservation area, Tanzania. IUCN, Gland, 40 pp

Ho M-J, Dai C-F (2014) Coral recruitment and subtropical coral community at Yenliao Bay, northern Taiwan. Zool Stud 53:5

Hoegh-Guldberg O (1999) Climate change, coral bleaching and the future of the world's coral reefs. Mar Freshw Res 50:839–866

Kayal M, Vercelloni J, Lison de Loma T, Bosserelle P, Chancerelle Y et al (2012) Predator crown-of-Thorns Starfish (*Acanthaster planci*) outbreak, mass mortality of corals, and cascading effects on reef fish and benthic communities. PLoS One 7:e47363

Lukoschek V, Cross P, Torda G, Zimmerman R, Willis B (2013) The importance of coral larval recruitment for the recovery of reefs impacted by Cyclone Yasi in the Central Great Barrier Reef. PLoS One 8:e65363

Martinez S, Abelson A (2013) Coral recruitment: the critical role of early post-settlement survival. ICES J Mar Sci 70:1294–1298

McClanahan T (2004) The relationship between bleaching and mortality of common corals. Mar Biol 144:1239–1245

Moran P (1986) The Acanthaster phenomenon. Oceanogr Mar Biol 24:379–480

Muhando C (2003) Trends in reef benthos after the 1998 coral bleaching event in Mafia Island Marine Park, Tanzania. West Indian Ocean J Mar Sci 1: 43–52

Munga C, Mohamed M, Amiyo N, Dahdou-Guebas F, Obura D, Vanreusel A (2012) Status of coral reef fish communities within Mombasa Marine Protected Area, Kenya, more than a decade after establishment. West Indian Ocean J Mar Sci 10:169–184

Newell BS (1959) The hydrography of the British East African coastal waters. Part II, Fishery Publications Colonial Office Number 12. H.M.S.O, London, 18 pp

Nhnyete I, Mahongo S (2007) National report of the United Republic of Tanzania on sea level measurements. Government of Tanzania, Tanzania

Nyström M, Graham NAJ, Lokrantz J, Norström A (2008) Capturing the cornerstones of coral reef resilience: linking theory to practice. Coral Reefs 27:795–809

Obura DO (2001) Differential bleaching and mortality of eastern African corals. Marine science development in Tanzania and Eastern Africa. Proceedings of 20th anniversary conference on advances in marine science in Tanzania, IMS/WIOMSA, Zanzibar

Obura D (2002) Status of coral reefs in Eastern Africa: Kenya, Tanzania, Mozambique and South Africa. Australian Institute of Marine Science, Townsville

Obura D (2005) Resilience and climate change: lessons from coral reefs and bleaching in the Western Indian Ocean. Estuar Coast Shelf Sci 63:353–372

Obura D (2012) The diversity and biogeography of Western Indian Ocean reef-building corals. PLoS One 7:e45013

Obura DO, Grimsditch G (2009) Resilience Assessment of coral reefs – Assessment protocol for coral reefs, focusing on coral bleaching and thermal stress, IUCN working group on climate change and coral reefs. IUCN, Gland, 70 p

Obura D, Church J, Daniels C, Kalombo H, Schleyer M, Suleiman M (2004) Status of coral reefs in East Africa 2004: Kenya, Tanzania, Mozambique and South Africa. Coral reefs of the world 2004 1. - Australian Institute of Marine Science

Tamelander J (2002) Coral recruitment following a mass mortality event. Ambio 31:551–557

Uthicke S, Schaffelke B, Byrne M (2009) A boom-bust phylum? Ecological and evolutionary consequences of density variations in echinoderms. Ecol Monogr 79

Visram S, Mwaura J, Obura D (2008) Assessing coral community recovery from coral bleaching by recruitment in two reserves in Kenya. West Indian Ocean J Mar Sci 6:199–205

West JM, Salm R (2003) Resistance and resilience to coral bleaching: implications for coral reef conservation and management. Conserv Biol 17:956–967

Zvuloni A, Artzy-Randrup Y, Stone L, van Woesik R, Loya Y (2008) Ecological size-frequency distributions: how to prevent and correct biases in spatial sampling. Limnol Oceanogr Methods 6:144–153

The Hydrodynamics of the Incomati Estuary – An Alternative Approach to Estimate the Minimum Environmental Flow

Antonio Mubango Hoguane and Maria Helena Paulo Antonio

Abstract

The Incomati Estuary, with a length of about 30 km, a width of about 1.5 km, and an average depth of 10 m, is located in the Southeastern part of Africa, north of Maputo Bay, between the Latitudes 24° 00'S and 26° 30'S and Longitudes 29°30′ E and 33° 15′ E. The Incomati River basin as a whole has an area of 46,426 km^2, of which 28,745 km^2 is located in South Africa, 2,786 km^2 in Swaziland and 14,856 km^2 in Mozambique. There are several dams upstream, mainly used to supply irrigation water. The river discharge is 200–400 m^3 s^{-1}, corresponding to a river discharge of about 700–1000 Mm^3, with peak and lower flows during the wet season (November to March) and dry season or winter (June-August), respectively. The morphology of the estuary is characterized by meanders, islands and sand banks, fringed by mangroves. The tides are semidiurnal, with amplitudes varying from 0.5 to 3.5 m during the spring and neap tides, respectively. The predominant winds are trade winds, with average speed of 4 m s^{-1}. The estuary plays an important ecological role and sustains important socioeconomic activities. It is a nursery ground for important fisheries, a tourism attraction, and hosts significant agriculture and livestock activities. The sustainability of the natural resources and the activities in the estuary is threatened by the effects of the dams. The three basin countries agreed in 1991, during the "Piggs Peak Agreement," to the maintenance of a minimum environmental flow of 2 m^3 s^{-1}, which has been violated several times by South Africa. The present work investigates how the hydrodynamics of the estuary, which controls most of the ecological processes, is influenced by the river discharges. Tides, currents and water masses in the estuary are described, based on historical hydrodynamic data. A simple 1-D, depth and width integrated, tidal hydrodynamic model forced by the observed tides at the mouth, was used to reproduce the hydrodynamics of the estuary. In addition, an analytical solution of the advection–diffusion of salt is applied to reproduce the salt intrusion. The model was then used to determine the extent of salt intrusion as a function of the river discharge, and so used to estimate the minimum environmental flow required for a healthy estuarine ecosystem in Incomati. The results indicated the estuary shifts, from a stratified condition in the wet season to vertically-mixed in the dry season. The tidal wave takes about 30 minutes to propagate from the mouth to the head of the estuary. The current varied between 0 and 1.2 m s^{-1}, mostly driven by tides. The density driven circulation is weak, with a density gradient of about 0.10 kg m^{-3} per km, and a density-driven velocity of about 0.04 m s^{-1}. The hydrodynamic model applied explains 87% of the current observed in the estuary,

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meaning the currents were mostly tidal driven, with the remaining 13% attributed to other factors such as density gradient and winds. The minimum recommended river flow required to prevent the intrusion of salt water to a distance 20 km upstream is 20 $m^3 s^{-1}$, against the 2 $m^3 s^{-1}$ value in the Piggs Peak Agreement. The results of this study may contribute to a better understanding of the dynamics of the ecosystems in the Incomati Estuary, and in the management of the river for the health of the downstream ecosystems.

Keywords

Incomati Estuary • Salt intrusion • Tides • Currents • Density gradient • Hydrodynamic model

Introduction

Mozambique is a low riparian country, vulnerable to the negative effects of activities that take place in the upper riparian area, characterised mainly by a reduced river flow and pollutant inflow. This situation is particularly critical, since surface water is the main water source in Mozambique, and 60% of the available surface water in the territory is from the international river basins. The reduced flow in Mozambique is mainly attributable to intensive use in the neighbouring countries. The reduced flow from the hinterland is more critical in the central and southern part of the country. It is estimated that South Africa, Swaziland and Zimbabwe now abstract about 40% to 60% of the border flow (van der Zaag and Carmo Vaz 2003; Hoguane et al. 2002).

The Incomati River, which is shared with Mozambique, Swaziland and South Africa, is intensively used for irrigation in South Africa. van Eekelen et al. (2015) stated that streamflow reduction due to forest plantations in South Africa is rather high, being about 2 to 3 times more than that allowed in the country's agreements on this shared river course. Saraiva Okello et al. (2015), analysing a long series (60 years) of rainfall and streamflow data of the Incomati River basin, concluded that land use and flow regulation are the main drivers of temporal changes in streamflow, rather than climatic forces.

Water is continuously being restricted upstream, regardless of the severe implications the reduced river flow causes in the downstream ecology. The reduced flow often has a detrimental effect on downstream ecology. It causes deterioration of downstream water quality and upstream salt intrusion, with consequences for the ecosystems and for the agriculture. The reduced flow is often associated with the reduced sediment inflow rate (Morris and Fan 1998), which may trigger erosion processes in the river mouth (McCully 1996) and affect the estuarine and coastal ecosystems. Dams may prevent natural upstream and downstream migration of fish species such as salmon and trout animals (Mann and Plummer 2000). The changes in freshwater instream flow brought by the dams may change the salinity distribution in the estuary can subsequently cause changes in the species composition, distribution, abundance, and health of fish and invertebrate (Alber 2002). LeMarie et al. (2006) used remote sensing techniques to identify and quantify mangrove forests in the estuary, observing that mangroves areas were significantly decreasing by 25% to 40%, with the tree biomass densities also declining. They attributed the observed changes to overharvesting of mangrove woods, natural rainfall trends, and modifications of the river flow regime. Hoguane (2004) reported the occurrence of erosion in Macaneta in the estuary, which can be attributed to the sediment deficit caused by the dams. Furthermore, according to Hoguane et al. (2002), because they change the natural regimen of the river, the dams introduce chronic stresses in marine organisms.

Until now, the management of river water in the Incomati basin focused on meeting the needs of the industries and the highly-developed agriculture in the upstream country, as well as guaranteeing the minimum water requirements for domestic use in the lower riparian country. Agriculture development in Mozambique is far inferior to that of South Africa, and the issue of ecological river flows had seldom been brought into negotiations, partly because its ecological values are often difficult to convert into a cash value.

Shared rivers can be the focus of transboundary conflicts if the issues of concern for every involved country are not adequately addressed. Aware of the downstream damage caused by the upstream obstruction of water, the three countries sharing the Incomati Rivers signed the Piggs Peak Agreement in 1991. This agreement established, among other items, the maintenance of a minimum runoff of 2 m^3s^{-1} for the lower riparian country (Saraiva Okello et al. 2015; TPTC 2010; Kistin and Ashton 2008). However, the scientific or technical reasoning for setting this agreed minimum runoff is not clear. On the other hand, in order to enable fruitful discussions that could lead to a mutual benefit during negotiations of the use of shared rivers, it is important that any arguments or discussions be founded on a scientific basis and evidence. In the case of Mozambique, this evidence is often absent because of a lack of expertise and material resources. Accordingly, the present study focuses on evaluating the extent to which the reduced flow of the Incomati River affects the estuarine environment, using salt (a conservative substance) as an indicator. The hydrodynamics of the estuary are described, and an advection – diffusion model of salt forced by ocean waters and river flows is applied to estimate the extent of salt intrusion as a function of the river runoff. The river runoff required to limit the salt intrusion to a critical length is then estimated. This estimated value is taken as a broad indication of the minimum environmental flows for the Incomati Estuary. It is hoped this study will provide some scientific guidance to the managers and technical experts of the two countries in their negotiations regarding the rational use of the Incomati River as a shared resource for the mutual benefit of the two countries.

General Description of the Incomati Estuary

Geographical Location

The Incomati Estuary, is located at Latitudes 25° 43′ S and 25° 53′ S and Longitudes 32° 41′ E and 32° 44′ E (Fig. 1), and discharges in the northern part of Maputo Bay. The estuary is about 40–50 km long, and meanders within the coastal plain, separated from the ocean by a narrow sand dune, a manifestation of the sluggish flow of the river. The sides of the estuary gradually converge upstream, resulting in a funnel-type shape. The surface area of the mouth during high water is about 9,000 m^2, with a slope factor of about 0.1 km^{-1}. There are some islands and sand dunes located toward the mouth.

Geology, Climate and River Flow Regime

The main geological features are an alluvial plain flat that stretches out along the river, with the coastal area being dominated by the yellowish or reddish coastal sand dunes and the sandy inner plains of the most recent formations of the Quaternary. There are considerable groundwater sources, with a recharge rate of 29–150 Mm^3 per annum within the coastal plain and in the Aeolian sands (Tauacale et al. 2007), most of it attributable to infiltration of rain through surface runoff.

The general climate in the Incomati River Basin varies from tropical warm/hot humid climate in the Mozambique coastal plain, to cool dry in the Transvaal Plateau in South Africa. The average annual rainfall is about 735 mm, increasing from east to west, with the average annual evaporation being about 1,900 mm, decreasing from east to west (Leestemaker 2000; JIBS 2001). The climate of the Incomati

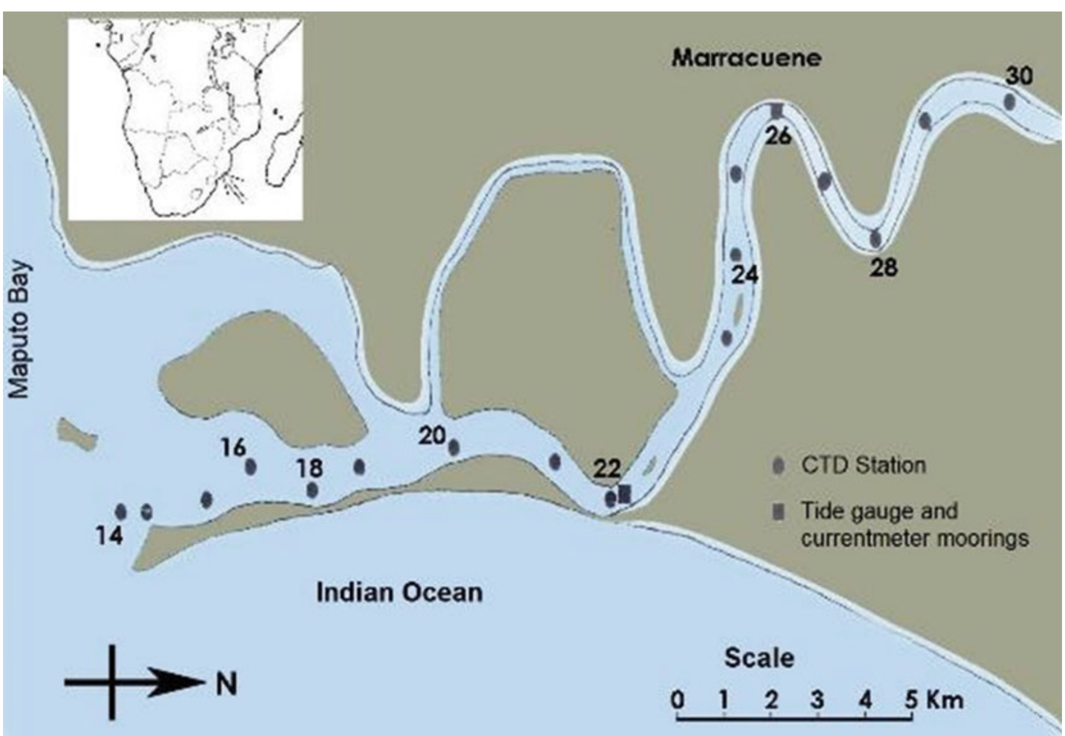

Fig. 1 The Incomati Estuary

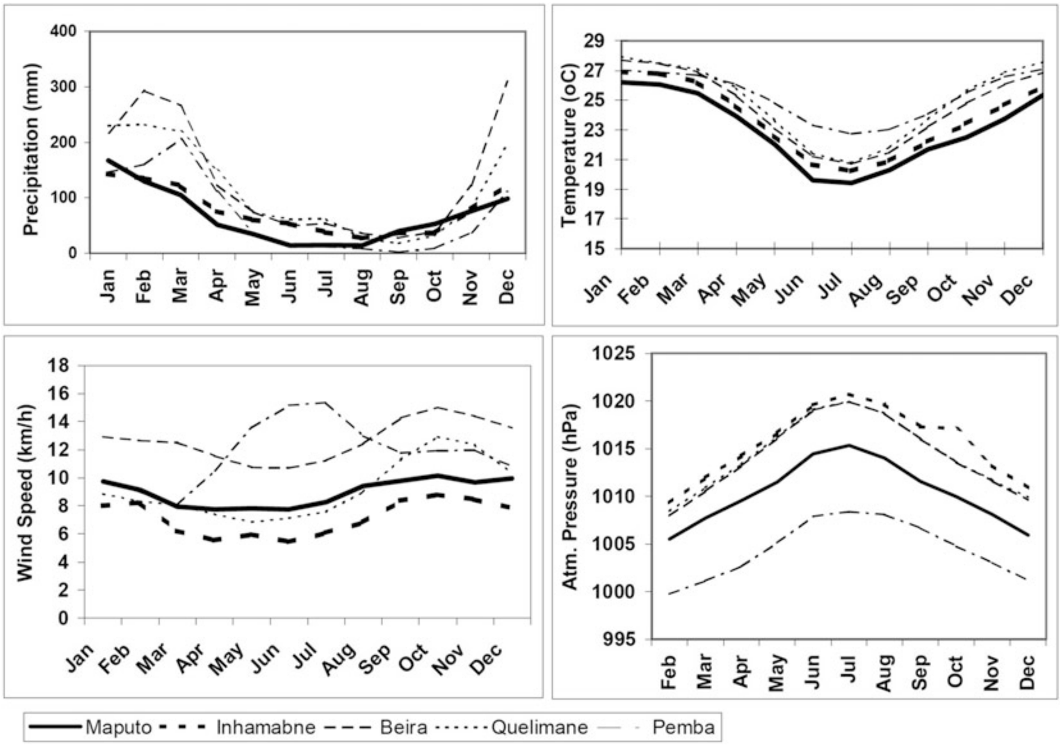

Fig. 2 Mean monthly averaged meteorological parameters over 30 years (from 1966 to 1996) from port town of Mozambique. Key towns are located sequentially along the coast from the Maputo in the South through to Pember in the North (Hoguane 1999)

Estuary can be quantified by figure from Maputo. Figure 2 presents the mean monthly averaged meteorological parameters from 1966 to 1996 along the Mozambican coast sequentially from Maputo in the South, through to Pemba in the North (Hoguane 1999). The climate of the Estuary is tropical humid (AW) with a dry winter, according to the Koopen classification. The mean monthly air temperature varies between 19 and 26°C observed in winter (June to July) and summer (January and February), respectively. The annual rainfall varies between 800 and 1000 mm. The rainfall occurs mostly from November to February, being equivalent to about 25 to 35% of the potential evapotranspiration; hence, the evaporation exceeds the precipitation.

Hydrological Characteristics of the Basin – River Flow

The hydrological year is October-September, with about 80% of all runoff occurring during the rainy season months of November-April. Thus, the Incomati River has high flows during the wet season from November to March, and relatively low flows in the dry season from April to October. On average, 60 to 80% of the mean annual flow occurs in only a few months of the year (DNA 2000). As a consequence, extreme floods and droughts occur in the basin, particularly in Mozambique, due to the fact it is located in the lower part of the basin. It suffers water shortages as a result of flow restriction in the upper riparian countries during the dry season, and floods as a result of water releases in the upper riparian countries during the rainy season. The most recent devastating floods occurred in the year 2000, and had severe impacts on agriculture and infrastructures, as well as causing a significant loss of lives (van Ogtrop et al. 2005; Carmo Vaz and Lopes Pereira 2000). Figure 3 presents the average monthly river discharge over the period 1953–1979. Variations in discharges from year to year are significant, with a coefficient of variation of around 50-65% (Smithers et al. 2001). Floods and long drought periods also occur. Figure 4 shows the time series of the monthly Incomati River runoff at Magude runoff gauge station, indicating there was a long drought from the mid-80’s to mid-90’s, with less than 200 $m^3\ s^{-1}$ during the peak flows, to nearly zero flow during the dry season. The extreme floods of the year 2000 exceeded 1,000 $m^3\ s^{-1}$. The volume of surface and groundwater resources of the Incomati Basin estimated by the Joint Incomati Basin Study (JIBS 2001) is 3,587 Mm^3 per annum (Table 1). The estimated total water use in 2002 was about 1,800 Mm^3 per annum. Thus, the total consumptive water use represents about 50% of the total water volume in the basin. This level of water consumption is high, frequently leading to water shortages, given the high flow variability, both within

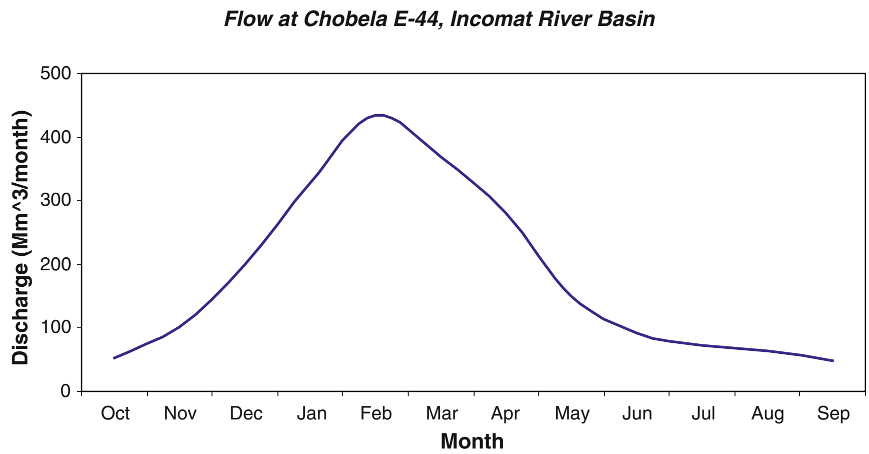

Fig. 3 Mean monthly river discharge of the Incomati river at Chobela Station, Mozambique, E-44; 1953–1979

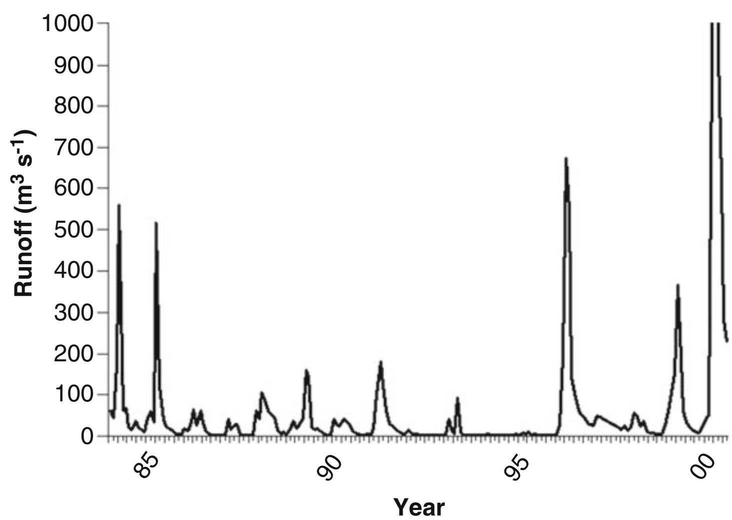

Fig. 4 Time series of the Incomati River runoff from October 1983 to September 2000, at Magude, Mozambique (Hoguane et al. 2002)

Table 1 Water generation in the Incomati River Basin, by Country (Mm^3 per annum)

Country	Catchment area Mm^2	Catchment area %	Virgin discharge Mm^3	Virgin discharge %
South Africa	28,556	61	2937	82
Swaziland	2545	5	479	13
Mozambique	15647	33	171	5
Total	46748	100	3587	100

Source: JIBS (2001)

and between years (Carmo Vaz and van der Zaag 2002; van der Zaag and Carmo Vaz 2003).

Classification of the Estuary

The annual evaporation in the Incomati Estuary exceeds the annual freshwater input, therefore being classified as a negative estuary according to Pritchard (1967). Based on geomorphology and Pritchard (1967) classification, the Incomati Estuary, located in a coastal flood plain, may be classified as a lagoon-type or bar-built since it is separated from the ocean by a sand dune and presents a spit at the mouth. Further, based on its water circulation, the Incomati Estuary may be classified as a mixed to partially-mixed estuary (Dyer 1997) because of reduced freshwater inputs and its shallowness, which facilitates its effective mixing by tides. In extreme flood condition, however, similar to that of 2000, a salt wedge may develop. And considering the salt intrusion curve type, the Incomati Estuary switches between types 1 and 2 during the wet and dry seasons, respectively (Vassele 2005) (Fig. 5).

Vegetation

Estuarine vegetation is dominated by: (i) Ubperenifólia forest to subdecídua, with fringes of sub-humido forests, composed of *Afzelia quasensis, Fcus* spp., *Brachylaena, richilia emetica, Albizzia adianthifolia, Garcinia livingstonei, Strychnos* sp.,

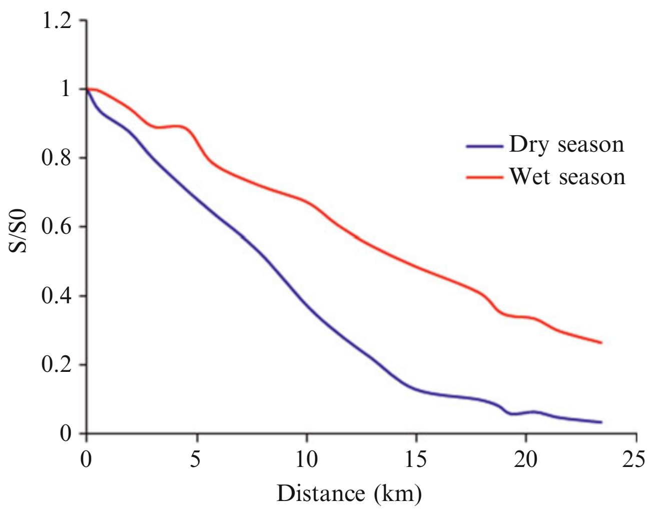

Fig. 5 Salt intrusion curves for Incomati River (Vassele 2005)

Conopharyngia elegans, Vernonia senegalensis, Tecomaria capensis, Acalypha spp., on reddish sandy soils; (ii) Secondary forests of *Sclerocarya caffra, Trichilia emeica, Albizzia verticolor, A. Adiantipholia, Alfzelia quazensis, Strychnos* spp. *Trema guineesis, Combretum, Conopharyngia elegans, Vernonia Snegalensis, Tecomaria capensis, Clematis orientalis*, highly modified by antropogenic influence with exotic species such as *Mangifera indica* e *Anacardium occidentale*, on reddish and rough sandy soils; and (iii) Hydrophytes formations composed by *Phoenix reclinata, Syzygium cordatum, Voacanga,* com *Phragnites, Cyperus, Juncus, Pteridium aquilidum,* etc., on dark, organic rich and hydro-morphological soils. The estuary exhibits extensive mangrove forests, some still in a pristine state, while others are heavily degraded. There are six main mangroves species, with *Avicennia marina* comprising 77% of the total mangrove species, *Phragmites australis* covering about 37% of the area, and *Rhizophora mucronata* and *Ceriops tagal* together comprising 18% of the species density (Bandeira et al. 2004). The area covered by the mangroves in 2003 was estimated to be about 4451 ha (Carmo Vaz and van der Zaag 2003; Bandeira et al. 2004). The evolution of the mangrove vegetation cover estimated by LeMarie et al. (2006) over a period of about 20 years (1984–2003) indicated a decreased mangrove coverage of about 25 to 40% and, according to Bandeira et al. (2004), the mangrove deforestation rate was estimated at 0.4% per year. The main cause of deforestation is overharvesting for firewood and building, attributable to increasing population pressures.

The Ecological Importance of the Incomati Estuary

The Incomati Estuary is a sanctuary for breeding colonies of aquatic birds, provides resources, and services the local bird populations. It also is a nursery ground for many species of fish and crustacean (Sengo et al. 2005). Macia and Menomussanga (2004) studied the flux of shrimp larvae through the Incomati Estuary, and concluded the inflow density of larvae increases with season and with tidal cycle, being higher during winter and between neap and spring tides. Juveniles and post-larvae of shrimp are more abundant in non-degraded mangroves, compared to degraded mangroves and estuarine channel. Degraded mangroves support less species density and biomass than non-degraded mangroves (Monteiro and Marchand 2009; Macia and Menomussanga 2004). LeMarie et al. (2006) reported the reduction of the mangrove area and mangrove tree and biomass density, attributing it partially to the modifications of the river flow regime by dams. The Incomati Estuary contributes approximately 20% of the overall shrimp catch in Maputo Bay (Anon 2001), estimated to be about 200 tons (Sousa 1990; Dengo and Govender 1998). According to Leestemaker (2000), there are about 24 freshwater and estuarine fish species of significant social and commercial value in the lower Incomati River and in Lake Chuáli, one of the large natural lakes in the lower Incomati. The fishery sector in Maputo Bay currently employs about 6,000 fishermen, of which about half are engaged in shrimp fishery (IIP 2000).

Streamflow and Climate Change in Incomati River Basin

According to Saraiva Okello et al. (2015), there is no evidence of the effects of climate changes in the Incomati River basin. Recent studies on the impacts of global climate change for southeastern Africa, however, predict a decline in annual rainfall (Fanta et al. 2001), which is likely to strongly influence the availability of water resources. Love et al. (2010) examined trends for various climate parameters, including rainfall and river discharges in the Limpopo Basin, a basin adjacent to the Incomati River basin, and observed that both the rainfall and discharge have notably declined since 1980, in terms of total annual water resources and temporal water availability of water. Further, the annual rainfall was found to be negatively correlated to the El Niño – Southern Oscillation index. Climate change projections also indicate the occurrence of erratic rainfall, severe drought and severe floods, which would potentially alter the river runoff intensity and regime, with subsequent ecological impacts (Shongwe et al. 2009). On the other hand, water scarcity attributable to climate changes, coupled with envisaged population pressures, and the consequent demand for land and water usage, would result in increased water abstractions, via the dams, for agriculture, domestic and industrial purposes (Rockström et al. 2009; Warburton et al. 2010, 2012). As a result, the salt intrusion, along with the associated ecological impacts, will be aggravated.

Water Masses and Hydrodynamics

The Use of Salinity as Indicator of Water Quality in the Estuaries

Water salinity is frequently used as a water quality indicator for estuaries, being directly related to freshwater flows (Peñas et al. 2013). Many estuaine living resources require water of a particular salinity and sufficient dissolved oxygen. Thus, water mixing and flushing are required for oxygen and nutrient renewal, creating an adequate salinity gradient, and stimulation of several bio-chemical processes and, hence, for water quality improvement (Powell et al. 2002). Salinity can reduce, or even prohibit, crop production (Mass 1990), which is of particular interest for the lower Incomati River basin, where agriculture and livestock activities occur about 20 km upstream from the river mouth. Stachelek and Dunton (2013) examined fluctuations in the abundance of selected salt marsh plants, finding that *S. alterniflora* flourished under low salinity conditions, and decline rapidly under salinities exceeding 25. They then used this information to estimate the ecological freshwater requirement. Peñas et al. (2013) developed a methodology for determining the environmental flow regime based on salinity distribution in the estuary.

Salinity Profiles in the Incomati Estuary

Figures 6 and 7 illustrate the longitudinal salinity distribution along the Incomati Estuary, from the river mouth up to about 25 km upstream. The estuary shifted from being partially-mixed in the southern summer or wet season (Fig. 6), to vertical mixed during the southern winter or dry season (Fig. 7). In the case of extreme floods, like that in 2000, it becomes stratified and a salt wedge can develop. The river discharge was about 922 m^3s^{-1} in March 2000, with the salt intruding about 6 km, and the river discharge was about 250 m^3s^{-1} in June, with the salt intruding about 9 km.

Salt Intrusion in the Incomati Estuary

Salt intrusion in the Incomati Estuary occurs as a direct consequence of the reduced river runoff. According to Gonzalez and Serraventosa (1999) and Hoguane (2002), salt intrudes between 40 km upstream to over 80 km, in the Incomati River estuary. The impacts of the salt intrusion in the estuarine environment are diverse. Species less tolerant

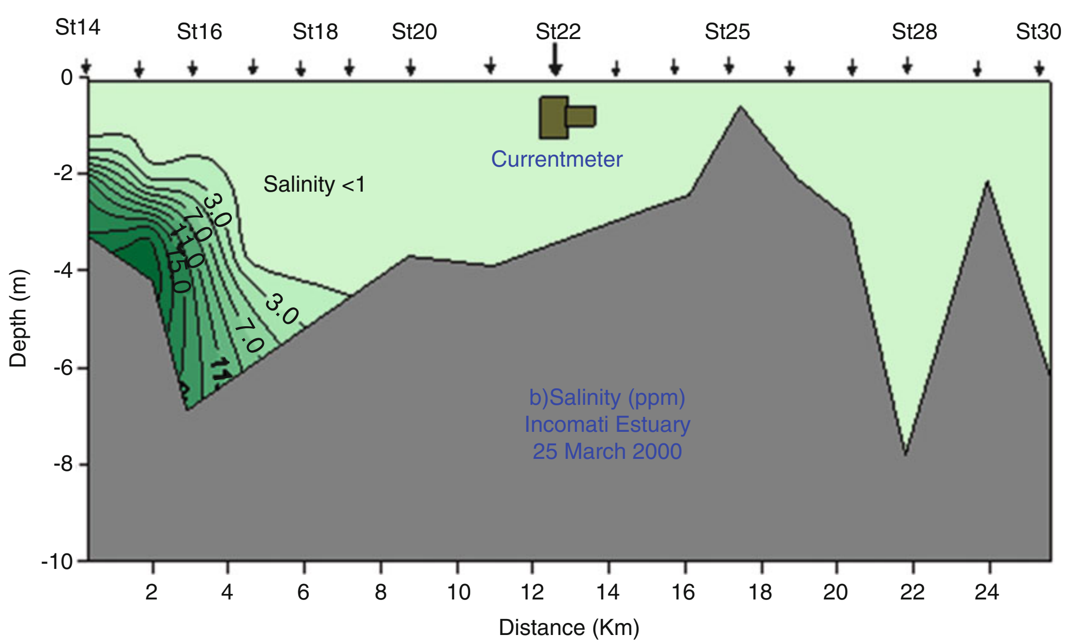

Fig. 6 Longitudinal salinity profile of the Incomati Estuary, during winter or dry season, June 2002

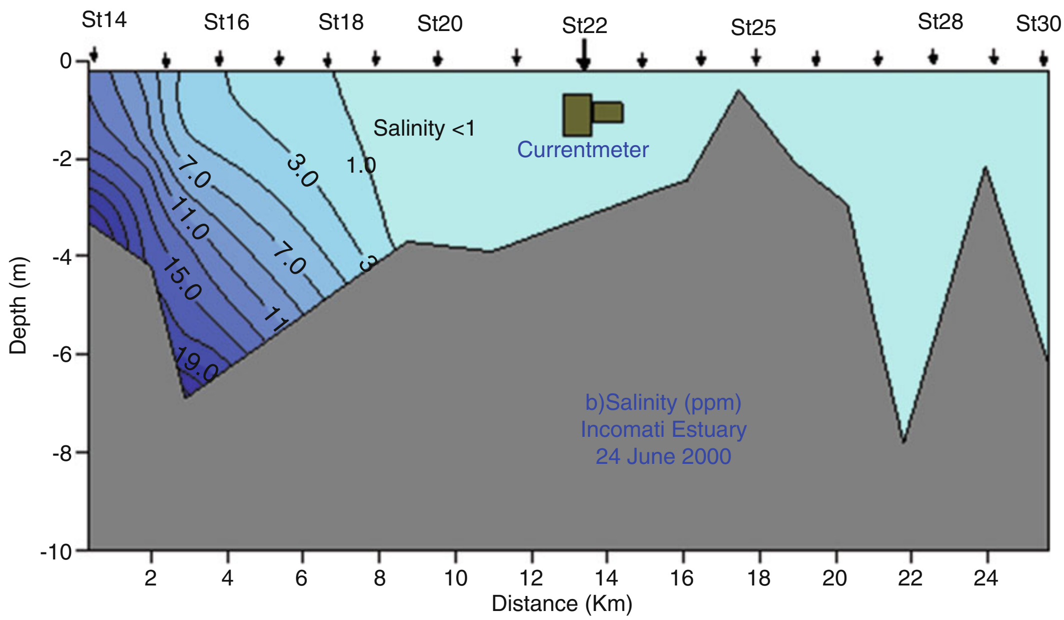

Fig. 7 Longitudinal salinity profile of the Incomati Estuary, during winter or dry season, March 2002

to salt content would be highly affected and forced to migrate, resulting in changes in the estuarine species composition. Increased salinity threatens the normal growth and survival of the plants and animals living in mangrove swamps. Gundry et al. (1981) pointed out that, although salt is important for growth and survival of mangrove plants, an extreme high salinity would retard even the most resistant species. According to Hoguane (2002), there are farmers located about 20 km upstream from the mouth of the Incomati River that cultivate land and graze their livestock in the river margins and banks, meaning they would be negatively impacted by the salt intrusion. Brockway et al. (2006) developed a salt intrusion model based on the advection diffusion equation, as follows:

$$\frac{dS}{dx} = -S\frac{R_0}{ak_x} \tag{1}$$

Where S = salinity; x = distance upstream from the mouth; R_0 = river runoff; kx = tidal diffusion coefficient; and a = a cross-sectional area, decaying exponentially upstream as follows:

$$a = a_0 e^{-\beta x} \tag{2}$$

Where a_0 = cross-sectional area at the mouth; and β = the decaying coefficient. The solution is given by:

$$\ln\left(\frac{S}{S_0}\right) = -\frac{R_0}{\beta a_0 k_x}\left(e^{\beta x} - 1\right) \tag{3}$$

Based on this model, the runoffs for salt intrusion lengths of different salinity values were simulated (Table 2). Considering the maximum salinity tolerance of maize, for example, which according to Sadat-Noori et al. 2008 is less than 0.5, the model result indicated the minimum runoff required to keep the salt intrusion below 20 km upstream was about 20 m^3 s^{-1} (Fig. 8).

Response of the Incomati Estuary to Flood Events

The response of the Incomati Estuary to changes in river discharge can be explored with the salt intrusion model, as presented in Table 2 and Fig. 8. After a strong flood event, the entire estuarine zone may be dominated by freshwater, and a sharp salt wedge frontal structure may develop at the estuary mouth (Fig. 6). With a reduced freshwater input, the front moves upriver and the stratification is destroyed. A mixed or partially-mixed water column system with a fairly constant horizontal density gradient in the lower estuary is then formed (Fig. 7).

Hydrodynamics

The hydrodynamics of the Incomati Estuary were examined by a simple one-dimensional tide model, as described by Bowen and Pinless (1977), as follows:

Table 2 River runoff (in $m^3\ s^{-1}$) for given values of salinity and salt intrusion extent (km) in the Incomati Estuary

	L (km)											
S	2	4	6	8	10	12	14	16	18	20	22	24
34	0	0	0	0	0	0	0	0	0	0	0	0
32	9	4	3	2	1	1	1	1	0	0	0	0
30	19	9	5	3	2	2	1	1	1	1	1	0
28	30	14	8	5	4	3	2	2	1	1	1	1
26	41	19	11	7	5	4	3	2	2	1	1	1
24	54	24	14	10	7	5	4	3	2	2	1	1
22	67	30	18	12	9	6	5	4	3	2	2	1
20	82	37	22	15	11	8	6	5	4	3	2	2
18	98	44	26	18	13	9	7	6	4	3	3	2
16	116	52	31	21	15	11	8	7	5	4	3	3
14	137	62	37	25	18	13	10	8	6	5	4	3
12	161	72	43	29	21	15	12	9	7	6	4	4
10	189	85	51	34	24	18	14	11	8	7	5	4
8	224	101	60	40	29	21	16	13	10	8	6	5
6	268	121	72	48	35	26	19	15	12	9	7	6
4	331	149	89	60	43	32	24	19	14	11	9	7
2	438	197	118	79	56	42	32	25	19	15	12	10

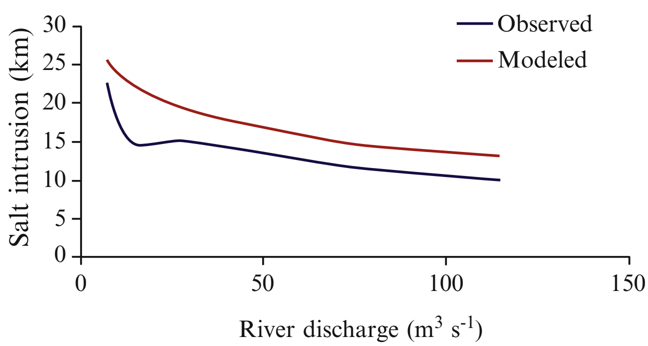

Fig. 8 Extension of the salt intrusion in the Incomati Estuary, observed and modeled, as a function of river discharge

$$\frac{\partial \eta}{\partial t} = -\frac{1}{B}\frac{\partial (Au)}{\partial x} \text{ where } A = B(D+\eta) \tag{4}$$

Eq. of Continuity

and

$$\frac{\partial u}{\partial t} = -g\frac{\partial \eta}{\partial x} - \frac{ku|u|}{(D+\eta)^{4/3}} \qquad \text{Eq. of Motion} \tag{5}$$

Where A = estuary cross sectional area; k = bottom friction coefficient (10^{-3}); B = width of estuary; D = mean water depth; η = height of the surface above the mean level; and u = current speed, being positive during the flood. Thus, (D + η) is the total water height. The depth (D) and width (B) of the estuary were considered to decay exponentially with the distance from the mouth to the head of the estuary, and fitted with the topographic data obtained from Google map and the echo sounder. Thus, the depth expression used in the model was as follows:

$$D(i) = 15\exp(-0.0001 * i * (dx + 2)) \tag{6}$$

and the width expression used was as follows:

$$B(i) = 164\exp(-0.0001 * i * dx) \tag{7}$$

Where i =1:1:n; n = number of elevation points set to n = 30; and dx = grid space set to dx = 1000 m.

The equations were solved numerically, forced by the tides at the mouth, as follows:

$$\eta = \sum_1^8 H_i \cos(\omega_i t + G_i) \tag{8}$$

Where H = amplitude; w = frequency; and G = the phase for given tidal constituent. Four most significant tidal constituents: M2, S2, K1 and O1 provided by the institute for Hydrography and Aid to Navigation (INAHINA) were considered.

Figure 9 illustrates the time series of tides and tidal currents modeled at the mouth for the spring and neap tides. The tides are semi-diurnal, with amplitudes of about 1.5 m and 0.5 m during the spring and neap tides, respectively. The tidal currents varied from about 1.25 to 0.5 m s^{-1} during the spring and neap tides, respectively. Figure 10 illustrates the water velocity at 7 points along the estuary.

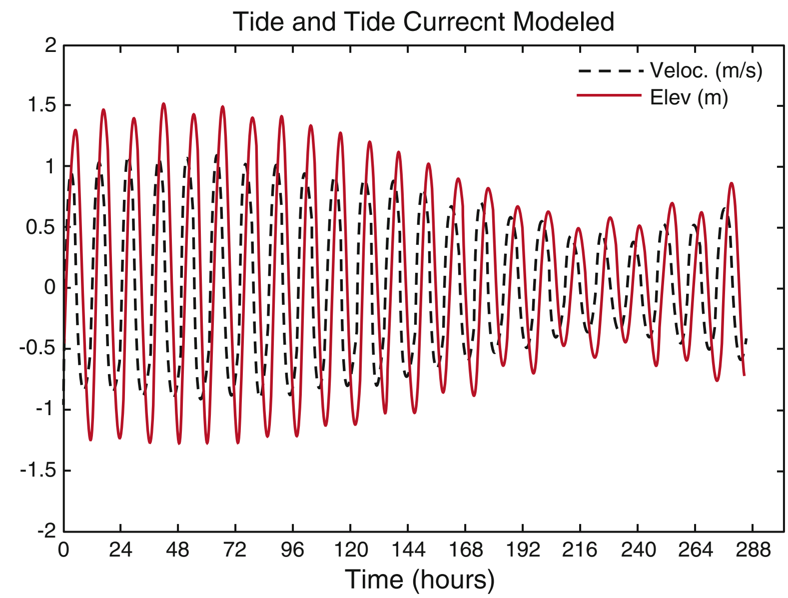

Fig. 9 Tides and tidal currents modeled at the mouth of the Incomati Estuary

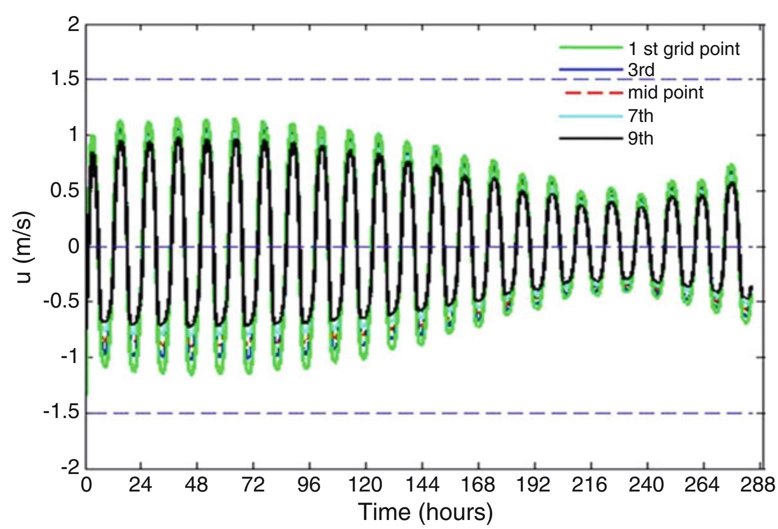

Fig. 10 Tidal currents modeled at 7 positions along the Incomati Estuary

The speed reduces as one moves upstream, from about 1.25 m s^{-1} at the mouth to about 0.25 m s^{-1} at the head.

Concluding Remarks

The regimen of the Incomati River is highly seasonal, with high runoff during the wet season, and almost virtually no runoff during the dry season. Thus, the water masses are dominated by ocean water during the dry season, and freshwater during the rainy season. The water column is mixed during the dry season, and partially mixed to stratified during the wet season. Salt intrusion occurs frequently during the dry season and can reach up to 30 km upstream. Based in the salinity model, the river runoff required to keep salt intrusion to less than 20 km upstream is 20 m^3 s^{-1}, compared with the 2 m^3 s^{-1} for the environmental flow set in the Piggs Peak Agreement. The hydrodynamics are mostly controlled by the tides, and the density-driven circulation can be significant during the wet season. Knowledge of the hydrodynamics, and particularly the salt intrusion model could be used in the management of the Incomati River, which is shared with three neighbouring countries, and is highly obstructed and heavily-used upstream. The river runoff needed to contain the salt intrusion proposed in the present work could serve as a basis for further negotiation of water use and sharing between the neighbouring countries.

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References

Alber M (2002) A conceptual model of Estuarine freshwater inflow management. Estuaries 25(6B):1246–1261. ISSN: 0160-8347. doi:10.1007/BF02692221. Available online at: http://www.jstor.org/stable/1352855

Anon (2001) Transboundary water management as an international public good – development financing 2000. A report prepared for the Ministry for Foreign Affairs of Sweden. Department for International Development Cooperation, Ministry for Foreign Affairs/ Norstedts, Stockholm. ISBN: 91-7496-250-7. http://www.odi.org/sites/odi.org.uk/files/odi-assets/publications-opinion-files/3870.pdf

Bandeira S, Balidy HJ, Macia A, Barbosa FMA, Fumo O, Onguane E, Amade F (2004) The actual condition of the mangrove forest of Incomati Estuary. A Catchment2Coast partner annual report, Eduardo Mondlane University, Maputo

Bowen AJ, Pinless SJ (1977) The response of an estuary to the closure of a mobile barrier; Richmond barrier on the upper Thames Estuary. Estuar Coast Mar Sci 5(2):197–208. doi:10.1016/0302-3524(77)90016-0. Available online at: http://www.sciencedirect.com/science/article/pii/0302352477900160

Brockway R, Bowers D, Hoguane A, Dove V, Vassele V (2006) A note on salt intrusion in funnel shaped estuaries: applications to the Incomati Estuary, Mozambique. Estuar Coast Shelf Sci 66:1–5. doi:10.1016/j.ecss.2005.07.014

Carmo Vaz A, Lopes Pereira A (2000) The Incomati and Limpopo international river basins, a view from downstream. Water Policy 2(1–2):99–112. doi:10.1016/S1366-7017(99)00024-0

Carmo Vaz A, van der Zaag P (2002) Sharing the Incomati waters: cooperation and competition in the balance. Sharing the Incomati waters, p 71, Shared Rivers. Technical Report, Maputo/Harare, December 2002

Carmo Vaz A, van der Zaag P (2003) Sharing the Incomati waters: cooperation and competition in the balance, IHP technical documents in hydrology—PCCP series no. 14. UNESCO, Paris, p 102

Dengo A, Govender A (1998) Pesca semi-industrial e artesanal de camarão na Baía de Maputo.IIP. Maputo

DNA (2000) Cheias do ano hidrológico 1999/2000. Seminário Técnico. Maputo, 30–31 Maio, 2000

Dyer KR (1997) Estuaries. A physical introduction, 2nd edn. Wiley, Chichester, p 195

Fanta B, Zaake BT, Kachroo RK (2001) A study of variability of annual river flow of the southern African region. Hydrol Sci J 46:513–524. doi:10.1080/02626660109492847

Gonzalez R, e Serraventosa V (1999) Moçambique e o protocolo dos rios partilhados da SADC – Constatações, questões e recomendações. DNA, comunicação interna, Maputo

Gundry J, Drennan M, Bejark P (1981) Salinity control in relation to pre-abscission propagule development in Avicennia marina (Forsk) Verh. Electron Microsc Soc S Afr Proc 11:155–156

Hoguane AM (1999) Sea level measurement and analysis in the Western Indian Ocean. National report: Mozambique, UNESCO IOC, Maputo

Hoguane AM (2002) Salt intrusion in Incomati river. Proceedings of the II-national conference on integrated coastal zone management. Proceedings of the II national conference on Coastal Zone Research. Maputo, 27–29 September 2000. CDS-ZC/MICOA, SECAM/UEM, 127 p

Hoguane AM (2004) Application of remote sensing and Geographic Information Systems (GIS) on integrated management of ecosystems and water resources in Mozambique. UNESCO Chair of marine sciences and oceanography technical report. Eduardo Mondlane University, Maputo, 18 p. Available online at: http://portal.unesco.org/ci/en/files/18496/11121061961Moz_Final Report

Hoguane AM, Motta H, Lopes S, Menete Z (2002) Mozambique national report on integrated problem analysis. Development and protection of the coastal and marine environment in sub-Saharan Africa. GEF MSP Sub-Saharan Africa Project (GF/6010-0016)

IIP (2000) Relatório sobre estado dos recursos: camarão da Baía de Maputo. IIP. Maputo. Instituto de Investigação Pesqueira. Relatório sobre estado dos recursos: camarão da Baía de Maputo. IIP, Maputo

JIBS (2001) Joint Incomati basin study phase 2. Consultec in association with BKS Acres. Final draft, April 2001, Maputo

Kistin EJ, Ashton PJ (2008) Adapting to change in transboundary rivers: an analysis of treaty flexibility on the Orange-Senqu River Basin. Int J Water Resour Dev 24(3):385–400. doi:10.1080/07900620802127325

Leestemaker JH (2000) Revisão da Literatura Relevante sobre a Saúde do Rio Incomati e seu Estuário. In: Proceedings of the workshop on “Contribution to Coastal Zone Management in Mozambique”, pp 39–66. Eduardo Mondlane University, Maputo, Mozambique

LeMarie M, van der Zaag P, Menting G, Baquete E, Schotanus D (2006) The use of remote sensing for monitoring environmental indicators: the case of the Incomati estuary, Mozambique. Phys Chem Earth Parts A/B/C 31(15–16):857–863. doi:10.1016/j.pce.2006.08.023. Avalable online at: http://www.sciencedirect.com/science/article/pii/S1474706506001574

Love D, Uhlenbrook S, Twomlow S, van Der Zaag P (2010) Changing hydroclimatic and discharge patterns in the northern Limpopo Basin, Zimbabwe Water SA 36:335–350. Available online at: http://www.wrc.org.za

Macia A, Menomussanga M (2004) Abundance and seasonality of juvenile shrimp in the Incomati estuary. A Catchment2Coast partner annual report, Eduardo Mondlane University, Maputo

Mann CC, Plummer ML (2000) Can science rescue salmon? Sci New Ser 289(5480):716–719. doi:10.1126/science.289.5480.716

Mass EV (1990) Crop salt tolerance. In: Tanji KK (ed) Agricultural salinity assessment and management manual. ASCE, New York, pp 262–304

McCully P (1996) Silenced rivers: the ecology and politics of large dams. Zed Books, London, p 432. ISBN 1-85649-902-2

Monteiro PMS, Marchand M (eds) (2009) Catchment2Coast. A systems approach to coupled river-coastal ecosystem science and management. Deltares selected series 02. ISBN-10: 1607500302. ISBN-13: 978-1607500308, 75 p

Morris G, Fan J (1998) Reservoir sedimentation handbook: design and management of dams, reservoirs, and watersheds for sustainable use. McGraw-Hill Publishers, New York, p 848. doi:10.1036/007043302X. ISBN 007043302X

Peñas FJ, Juanes JA, Galván C, Medina R, Castanedo S, Álvarez C, Bárcena JF (2013) Estimating minimum environmental flow requirements for well-mixed estuaries in Spain. Estuar Coast Shelf Sci 134(2013):138–149. http://dx.doi.org/10.1016/j.ecss.2013.05.020

Powell GL, Masumoto J, Brock DA (2002) Methods for determining minimum freshwater inflow needs of Texas bays and estuaries. Estuaries 25:1262–2174

Pritchard DW (1967) What is an estuary: a physical viewpoint. Estuar Washington Am Assess Adv Sci 83:3–5

Rockström J, Falkenmark M, Karlberg L, Hoff H, Rost S, Gerten D (2009) Future water availability for global food production: the potential of green water for increasing resilience to global change. Water Resour Res 45:W00A12. doi:10.1029/2007wr006767

Sadat-Noori SA, Mottaghi S, Lotfifar O (2008) Salinity tolerance of Maize in Embryo and adult stage. Am Eur J Agric Environ Sci 3(5):717–725. ISSN 1818-6769

Saraiva Okello AML, Masih I, Uhlenbrook S, Jewitt GPW, van der Zaag P, Riddell E (2015) Drivers of spatial and temporal variability of streamflow in the Incomati River basin. Hydrol Earth Syst Sci 19:657–673. doi:10.5194/hess-19-657-2015. Available online at: www.hydrol-earth-syst-sci.net/19/657/2015/

Sengo DJ, Kachapila A, van der Zaag P, Mul M, Nkomo S (2005) Valuing environmental water pulses into the Incomati estuary: key to achieving equitable and sustainable utilisation of transboundary waters. Phys Chem Earth Parts A/B/C 30(11–16):648–657. doi:10.1016/j.pce.2005.08.004. Available online at: http://www.sciencedirect.com/science/article/pii/S1474706505000756

Shongwe ME, van Oldenborgh GJ, van den Hurk BJJM, de Boer B, Coelho CAS, van Aalst MK (2009) Projected changes in mean and extreme precipitation in Africa under global warming. Part I. S Afr J Clim 22:3819–3837. doi:10.1175/2009jcli2317.1

Smithers JC, Schulze RE, Pike A, Jewitt GPW (2001) A hydrological perspective of the February 2000 floods; a case study in the Sabie river catchment. Water SA 27(3):325–332

Sousa MI (1990) Informação sobre a fauna acompanhante do camarão da baia de Maputo de Janeiro /80 a Agosto /81. Rel. No 14 I.I.P. Maputo 30 pp

Stachelek J, Dunton KD (2013) Freshwater inflow requirements for the Nueces Delta, Texas: Spartina alterniflora as an indicator of ecosystem condition. Texas Water J 4(2):62–73. ISSN 2160-5319

Tauacale FPI, Amida M, Tomazina S, Marcio M (2007) Perfil Ambiental da bacia do Incomati em Moçambique. Departamento de Geografia, Universidade Eduardo Mondlane, Maputo

TPTC (2010) Tripartite Permanent Technical Committee (TPTC) between, Moçambique, South Africa, Swaziland, PRIMA: IAAP 3: consultancy services for integrated water resources management. Baseline evaluation and scoping report: Part C. Report No.: IAAP 3: 03C–2010. Prepared for TPTC by Aurecon, Pretoria, 2010

Tryckeri AB, Stockholm (2001) ISBN: 91-7496-250-7. http://www.odi.org/sites/odi.org.uk/files/odi-assets/publications-opinion-files/3870.pdf

Van der Zaag P, Carmo Vaz A (2003) Sharing the Incomati waters: cooperation and competition in the balance. Water Policy 5 (4):349–368. IWA publishing. ISSN: 1366-7017

Van Eekelen MW, Bastiaanssen WGM, Jarmain C, Jackson B, Fereira F, van der Zaag P, Saraiva Okello AML, Bosch J, Dyeh P, Bastidas-Obando E, Dost RJJ, Luxemburg WMJ (2015) A novel approach to estimate direct and indirect water withdrawals from satellite measurements: a case study from the Incomati basin. Agric Ecosyst Environ 200:126–142. doi:10.1016/j.agee.2014.10.023

Van Ogtrop FF, Hoekstra AY, Frank van der Meulen F (2005) Flood management in the lower Incomati River basin, Mozambique: two alternatives. J Am Water Resour Assoc 41 (3):607–619. doi:10.1111/j.1752-1688.2005.tb03758.x

Vassele VJ (2005) Modelo de Intrusão Salina no Estuário do Incomáti Baseado na Influência Geométrica e Topográfica do Estuário. Tese de Licenciatura, Departamento de Física, Universidade Eduardo Mondlane, Maputo, 46 p

Warburton ML, Schulze RE, Jewitt GPW (2010) Confirmation of *ACRU* model results for applications in land use and climate change studies. Hydrol Earth Syst Sci 14:2399–2414. doi:10.5194/hess-14-2399-2010

Warburton ML, Schulze RE, Jewitt GPW (2012) Hydrological impacts of land use change in three diverse South African catchments. J Hydrol 414–415:118–135. doi:10.1016/j.jhydrol.2011.10.028

Estuary Management in South Africa – An Overview of the Challenges and Progress Made to Date

Pierre de Villiers

Abstract

The need for estuary management and estuary management plans in South Africa has resulted from the encroachment of human settlements and activities into the estuarine environment. The advantages of this environment to humans are that it provides freshwater, is relatively safe from coastal storms and provides a myriad of ecosystem goods and services. Estuarine systems are diverse and have thus resulted in the involvement of a diverse set of Government Departments and stakeholders in their use and management. The South African Government has developed the Integrated Coastal Management Act in an attempt to facilitate cooperative governance in this field. The C.A.P.E. Estuaries Programme was a cooperative initiative aimed at testing the implementation of this Act in the field of estuary management in an inclusive, transparent manner. Valuable lessons were learned and systems and processes that were developed were embedded into Local, Provincial and National Government planning documents. The continued involvement of all spheres of Government and other stakeholders is required for this field to become a successful show case for South Africa. Lessons from global success stories should be incorporated into best practices into the future.

Keywords

Estuary management • Freshwater • Ecosystems • Stakeholders • Lessons learned • Best practice

Background

The natural environment has evolved over millions of years and will continue to do so. The human population forms part of this process, however in order to understand and manage the impact of humanity on our globe it has been deemed wise to develop fields of specialisation and the associated management fields. To this end, teams of experienced individuals all over the world have been tasked to develop sets of norms and standards or regulations to assist with the management of each field. In South Africa this process has resulted in the setting up of specific government departments to address each field of specialization. While this works for the managers of each field it does not address the required integrated nature of all natural and cultural systems that exist. As in nature, no one field can exist without the other.

In South Africa, estuaries and the overall management thereof have historically been included within the management mandate of the Department of Water Affairs due to the fact that estuaries form the receiving environment for freshwater flowing out of the catchments.This department manages the freshwater resources of the country in terms of the National Water Act (Act No. 36 of 1998). However, this legislation and its associated best practice guidelines do not address the water resource once fresh and ocean water are mixed together. It also makes minimal provision for the

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management of freshwater flow into estuaries, or for the management of estuary mouths.

The estuarine Reserve Determination process is designed to identify the freshwater environmental flow requirements as well as the basic human needs in each catchment, including the estuary. This was possibly the only process that catered to some extent for the freshwater flows required to sustain estuarine ecosystems. In terms of the National Water Act, the freshwater resource must be protected in order to sustain the environment and to provide for basic human needs. In 1999, in order to achieve these goals, the Department of Water Affairs began to develop methods to determine the environmental flow requirements of catchments and estuaries. This was known as the Ecological Reserve (DWA 2010). Specific water quality and quantity criteria were set and the associated monitoring requirements were identified. A limited number of studies have since been conducted to determine preliminary Ecological Water Requirements (DWAF 2008). This data then contributes towards the overall Catchment Classification process. To date, however, very few reserve determination processes have been approved and implemented by the Department of Water Affairs.

Linked to the opening of estuary mouths and the critical contact between the catchment and the ocean, is the management of the marine living resources that enter estuaries from the ocean. Various estuary-dependent marine species and estuarine species with an oceanic life cycle phase use estuaries as developmental nodes for their offspring and depend upon the opening of estuary mouths for completion of life cycles. The Department of Fisheries was set up to manage marine species according to the Marine Living Resources Act (Act No. 18 of 1998). However, the legislation at that time did not include the management of estuary mouth conditions to facilitate the completion of the life cycles of rare or commercially important species. The Department of Fisheries was then subsumed in the broader Department of Agriculture, Forestry and Fisheries, whose primary mandate is to manage food security in the country. The Department of Environmental Affairs is responsible for the conservation of biodiversity as well as sustainable development, but it only addresses the management of rare species or ecosystems and the control of development as part of the Environmental Impact Assessment and Authorisation process. In essence, biodiversity issues have mainly been addressed by managing the use of species or access to sensitive areas in line with the National Environmental Management Act (Act No. 107 of 1998), the National Environmental Management: Biodiversity Act (Act No. 10 of 2004) and the National Environmental Management: Protected Areas Act (Act No. 57 of 2003).In general while there have been various global management initiatives aimed at addressing specific species within estuaries, e.g. Kairo et al. 2001, very few initiatives exist that are aimed at the overall management of an estuary which in essence should ensure the effective management of the various habitats and individual species, e.g. Tamar Estuaries Consultative Forum www.plymouth.gov.uk/tecf and TIDE www.tide-project.eu. The latter examples were initiated as a result of extreme pressure on estuarine systems and their associated environmental services.

Towards Integrated Coastal Management

Over 300 estuaries exist along the South African coastline, which in itself is a complex ecosystem (Fig. 1). The coastal environment abuts two of the 64 Large Marine Ecosystems of the world, namely the Benguela Current and the Agulhas Somali Current Large Marine Ecosystems (NOAA 2013).

Estuaries form vital links between catchments and the ocean. They have great value in that they provide natural resources that can be utilized, they provide areas of safe settlement and they provide valuable recreational areas. These various values are generally summed up and described as ecosystem goods and services. The need to understand how these systems function and in so doing maximize the benefits to humanity and the environment, is critical. The management of these estuarine systems needs to be integrated into the management of the associated catchments and coastal zones Fig. 2.

The steady decline of estuary condition was an issue that was identified by several specialists, including Heydorn and Tinley (1980) as far back as the 1980s. Teams of scientists met to discuss how to integrate the various fields of science mentioned above, while at the same time improving the management of estuaries and associated species by each relevant Government Department. Several initiatives took place over time. An example is the Eastern Cape Estuary Management Programme (MacKenzie and Hay 2002). However, little success was achieved due to the fact that there was no clear supporting legislation and no lead Department. The Integrated Coastal Management Act (Act No. 24 of 2008) was the culmination of these discussions and initiatives. The South African Government recognised the need for a specific coastal management field due to the specialisation required to manage this field. In addition, the need to integrate the management mandates of several existing government departments was identified as a priority action. Innovative thinking and planning was needed in order to develop a truly integrated act that recognised the existence and facilitated the implementation of other legislation pertaining to aspects of estuarine management. This led to a process, spanning decades, that culminated in the development of the Integrated Coastal Management Act.

This act includes a section that deals specifically with estuaries and estuary management. This was a major breakthrough in the field of estuary management. In support of this

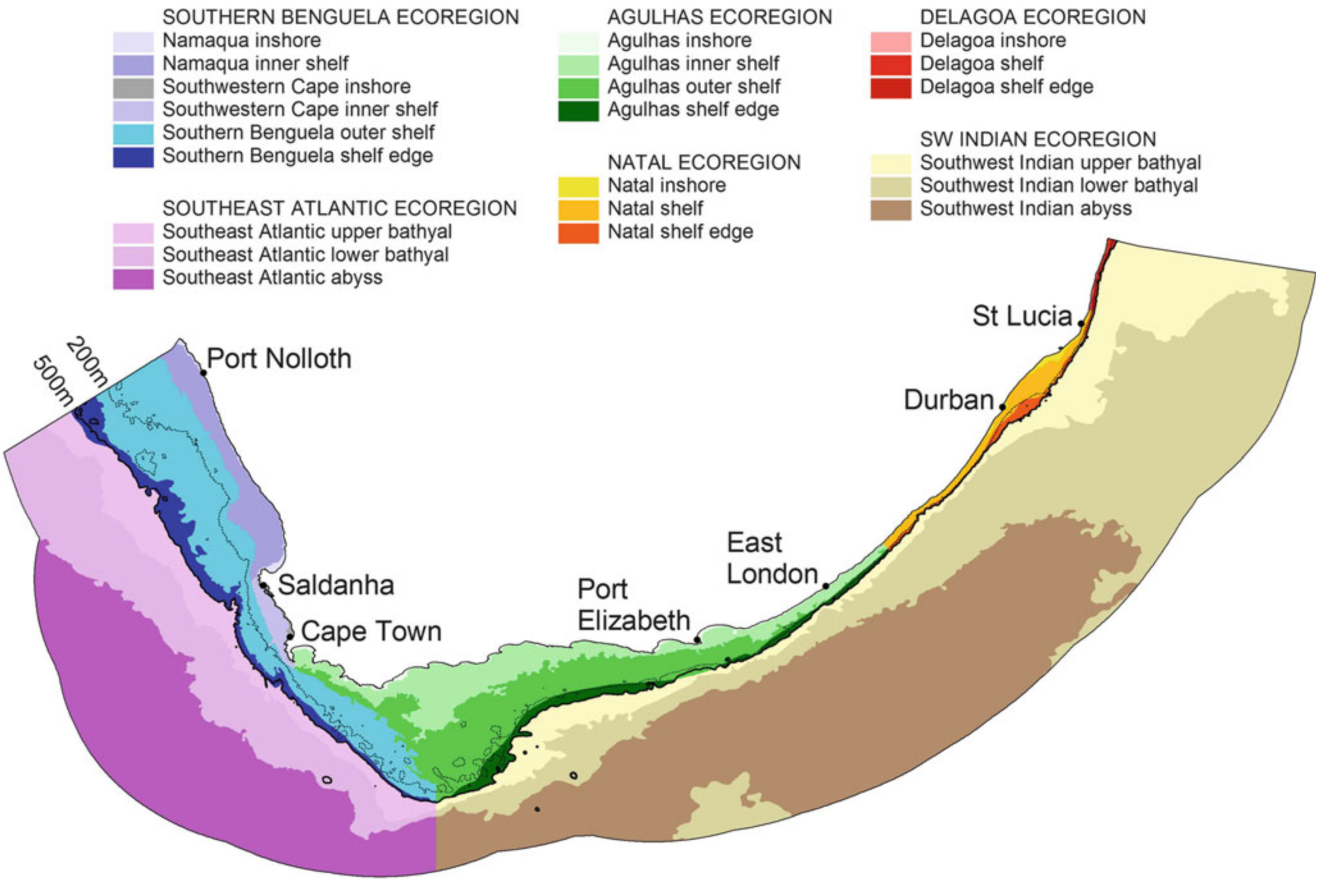
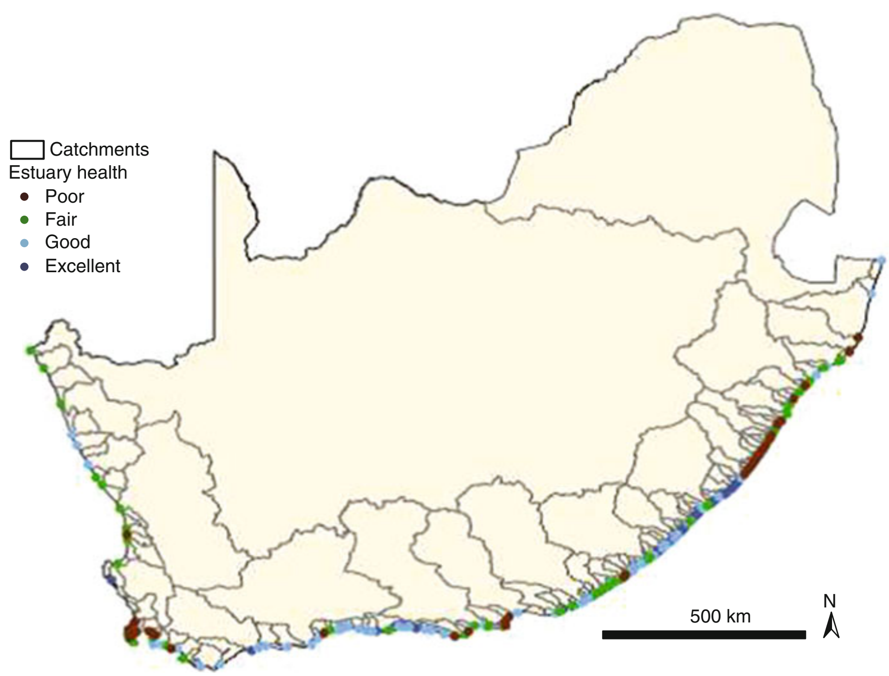


Fig. 1 The complexity of the coastal ecosystem that exists offshore of South Africa (From Sink et al. 2012). Over 300 estuaries flow into the coastal environment, creating regional and estuary-specific systems and processes that must be taken into consideration when deciding upon management strategies and plans for coastal regions and estuaries

Fig. 2 The health status of South Africa's estuaries was determined as part of the National Biodiversity Assessment (Sink et al. 2012). This assessment found that 17% of estuaries were in an excellent state and 41% were in a good state. About 35% were in a fair state and 7% were in a poor state (Sink et al. 2012)



developmental process, the C.A.P.E. Estuaries Programme was initiated in 2005 (CSIR 2006). The programme, the first of its kind, aimed to set up a team that assessed and tested the systems and processes listed in the draft Integrated Coastal Management Bill.

C.A.P.E. Estuaries Programme as the Beginning

The C.A.P.E. Estuaries Programme formed one of the sub-programmes of the broader programme, "Cape Action for People and the Environment" (C.A.P.E.). The latter is a programme of the South African Government which, with support from international donors, aims to protect the rich biological heritage of the Cape Floristic Region. C.A.P.E. seeks to "unleash the economic potential of land and marine resources through focused investment in development of key resources, while conserving nature and ensuring that all people benefit" (www.capeaction.org.za).

The initial aims of C.A.P.E. Estuaries Programme were to set up a management team to drive the programme, to set up a process aimed at developing estuary management plans in South Africa and to develop stakeholder awareness and support for the programme. In support of the planning process, a Generic Framework for Estuary Management Plans Document (CSIR 2007) (to be used as the starter document for the National Estuary Management Protocol [Government Gazette No. 36432, 10 May 2013]) and an Estuarine Conservation Planning Document, were developed (Turpie and Clark 2007). A Programme Coordinator was employed on a contractual basis to drive this process, and this was critical for the success of the process. CEP operated within South Africa's Cape Floristic Region. The Cape Floristic Region is one of the world's 18 biodiversity hotspots and is a global priority for conservation action (UNESCO 2004). It was chosen for CEP because there was funding available within the broader C.A.P.E. Programme for this region, and because the Western Cape Province had the capacity to drive the programme. Although most of the Cape Floristic Region falls in the Western Cape Province, it also includes some estuaries from the Eastern Cape Province.

In order to address the effective management of this integrated field, a Task Team of appropriate representatives from different Government Departments was set up. This became known as the C.A.P.E. Estuaries Programme Task Team. The management of the Task Team was carried out by the coordinator, who was based at CapeNature. CapeNature is an implementing agent for the Provincial Department of Environmental Affairs in the Western Cape Province. Task Team members were made aware of the broader C.A.P.E. Estuaries Programme process and the relevance of the management mandate of their own department to this process was explained. However, no members were allowed to prioritize the management mandates of their departments. The lead Departments participated during the early stages of this Programme, with additional Departments joining the Task Team as the Programme gained momentum and started addressing estuary specific issues. For example, the South African National Biodiversity Institute joined later, with the aim of addressing estuary rehabilitation. The Task Team coordinator then reported to funders, Coastal Committees and stakeholders with regards to Task Team systems and processes, and decisions taken at the Task Team meetings. The Task Team still meets on a quarterly basis. Frequent meetings are necessary because broader management issues as well as estuary specific issues must often be dealt with as a matter of urgency. Government and stakeholders will only support a Programme that provides solutions to identified problems.

With regards to decision-making and general technical support it, was understood from the outset of CEP that all decisions should be based on the best available scientific data or evidence. With this in mind, a Technical (Scientific) Working Group was set up, including specialists who had been identified at the initial regional stakeholder workshop held in 2005 (CSIR 2006). This group continues to provide ongoing scientific data and advice to the Task Team. The principle of specialist input is critical, especially when implementing teams meet with stakeholders during the estuary management plan developmental process.

Initially, due to the fact that this was a new process in South Africa, there was minimal capacity to develop estuary management plans within the lead Departments. Funding was therefore allocated to cover the costs of registered service providers who developed the estuary management plans. In 2006 there were very few qualified service providers in this field in South Arica, so CEP initiated a process aimed at facilitating the development of additional qualified service providers in this field. This was an important process, since management plans for each estuary in the country would soon become a legislative requirement. The Task Team members and coordinator provided support to the initial service providers employed. An integrated team involved with the development of an estuary management plan therefore consisted of service providers and experienced officials. It was understood that while service providers could develop estuary management plans, they could not address questions and discussions around management mandates of Government Departments. It was essential that officials from these Departments attended all stakeholder engagements, so that issues could be addressed immediately and were not left to "simmer". Minutes were kept from all engagements so that progress could be tracked.

Estuary Management Plans

The CAPE Estuaries Programme Task Team, constituted based on decisions taken at the initial Regional Stakeholder Workshop (http://www.saeon – estuaries management site in prep), was used to decide on priority estuaries for which management plans should be developed. The decision was based on information contained in the C.A.P.E. Estuaries Conservation Plan (Turpie and Clark 2007). Aspects such as biodiversity value, as well as social and economic values and goods and services were considered. The aim was also to spread the selection of estuaries along the Cape Floristic Region coastline. This would expose the participants to different estuary-specific issues and would also provide a broad range of stakeholders along the coastline with an opportunity to engage with estuary management planning processes. The final choice of the initial six estuary management plans was based on available funding as well as available capacity. The case studies were the Olifants, Klein, Heuningnes, Breede, Knysna and Gamtoos estuaries (Fig. 3). Funding was provided by the C.A.P.E. Programme. As mentioned earlier, individual service providers were employed to develop these estuary management plans. While the procurement of each service provider followed the appropriate systems and processes of the C.A.P.E. Programme and of CapeNature, the Task Team members were also included in the evaluation panel and the final decision-making process. This resulted in full support for the process from all participating Departments.

The Generic Framework for Estuary Management Plans (CSIR 2007) provided the broad guidelines that needed to be followed by each team of service providers when developing an estuary management plan. This included structure and content. The associated stakeholder engagement process had to be in line with the stipulations in Chapter Five of the Integrated Coastal Management Act regarding effective stakeholder engagement. The processes had to be advertised to the broader stakeholder groups, including appropriate Government Departments, at each estuary. Also, good communication channels had to be maintained for the duration of the process. It was accepted that an added bonus would be for stakeholders to continue to participate in an inclusive and

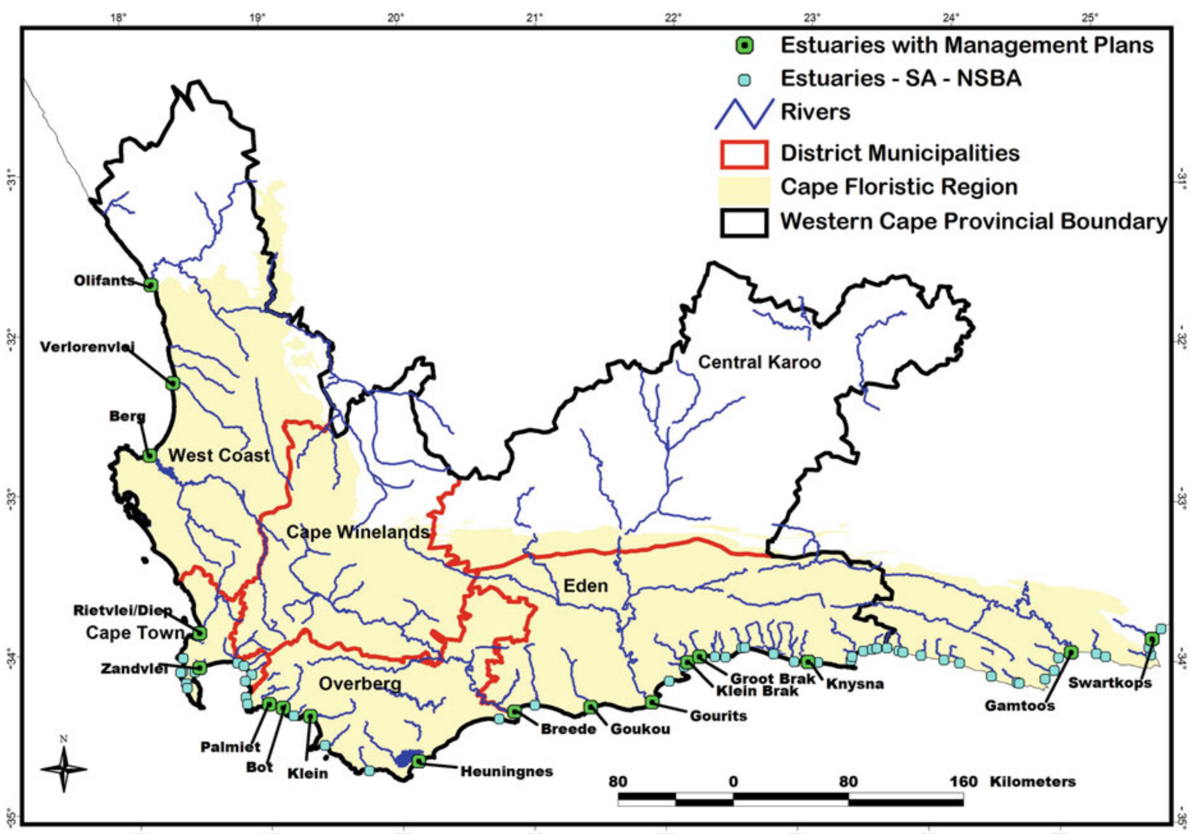


Fig. 3 CAPE Estuaries Programme was responsible for the development of management plans for estuaries along the coastline of the Cape Floristic Region (*shaded area*). The initial six plans were developed for the Olifants, Klein, Heuningnes, Breede, Knysna and Gamtoos estuaries. The work was carried out in two Provinces in South Africa, namely the Western Cape (areas outlined in *black*) and the Eastern Cape (C.A.P.E. Estuaries Programme)

transparent implementation process once the plans had been developed and adopted. Based on the required structure, content and stakeholder participation, a basic process was designed and implemented (see below). This was not restrictive but was rather seen as a guide, because each estuary and its associated issues are different, requiring the planning team to implement an adaptive management approach depending on individual circumstances.

The broad principles implemented included, but were not restricted to, the following:

- Selection of service providers wasbased on skill set, experience and location
- An initial meeting was arranged with Government Department officials responsible for aspects of the management of each estuary, to identify any major issues and to propose a clear way forward.
- A first broad stakeholder meeting was arranged at a venue close to the estuary to facilitate the participation of local people. The aim of this meeting was to introduce the team and process to stakeholders, and to discuss the need for specific information that would be included in a Situation Assessment. This could include local knowledge. At the meeting, an attempt was made to identify any critical stakeholders who were absent. A draft Situation Assessment was then tabled and served as a starter document for the process to focus upon.
- A second stakeholder meeting was arranged approximately three months later, to provide the team with an opportunity to engage with the broader stakeholder group and to finalize the Situation Assessment. Stakeholders were requested to comment on the draft document and to add information.
- A third stakeholder meeting was required to develop a vision and identify management objectives that would be used to achieve the vision. All stakeholders had to agree on the vision and objectives. This meeting took place within a month of the second meeting to maintain momentum and address the areas of conflict. The estuary advisory forum/body was structured at this meeting. Ideally, a call for nominations to serve on the forum was sent out after the meeting. The forum consisted of representatives of all stakeholder groupings and relevant Government Departments. Formal nomination forms were sent to all stakeholders. The foundations for the forum and the implementation phase of the plan need to be laid during the developmental phase.
- At a fourth stakeholder meeting, the draft estuary management plan was presented. This normally took place within three to four months after the vision and objectives had been approved. This meeting was used to address any outstanding issues and to edit the plan. The estuary management forum was formally constituted.
- At a final stakeholder meeting, the plan was presented to the Management Authority (as defined in the National Estuary Management Protocol [2013]) to process. While the initial focus of the Programme was to develop Estuary Management Plans for estuaries along the Cape Floristic Region coastline, this was later broadened to include the entire South African coastline. To date in excess of 26 estuary management plans have been developed in this way (Fig. 4).

Those management plans that are being implemented successfully all have estuary management forums in place. Without effective stakeholder engagement and support, estuary management plans will not be adopted or implemented. Without creating a common vision and a local management team or forum, local issues will result in unresolved conflict. While scientific rigor is essential with regards to the content of the Situation Assessment and Estuary Management Plan, a transparent and inclusive stakeholder process is also of the utmost importance. People need to be provided with documented proof of the process that was followed.

Once the estuary management plan has been approved by all stakeholders, it must be submitted to the Management Authority for adoption. The National Estuary Management Protocol (Government Gazette No. 36432, 10 May 2013) acts as a guide to which Government Department is mandated to be the Management Authority. The Estuary Management Plan should be considered as step one in the overall estuary management process. The plan provides the framework and priorities which will guide an implementation planning strategy (see below).It is essential that the estuary management strategy is adaptive. The estuarine environment is ever-changing, and new lessons are constantly being learnt in both the national and global arena, e.g. Tamar Estuaries Consultative Forum (www.plymouth.gov.uk/tecf). The Management Authority must be responsive on both counts.

An implementation strategy, in the form of action tables linked to responsible Departments, is included in each estuary management plan. In order to succeed, this strategy needs the support of the Management Authority and Estuary Management Forum representatives, as they form the team responsible for the implementation of the management plan. An important aspect to bear in mind when developing an implementation plan is the need for funding and capacity. Ideally, these aspects will be included in the budgeting and Human Resources processes of the Management Authority and all other mandated Departments, and the process will be supported by the Provincial and National Lead Agents. There should also be planning for additional support in the event that there are funding and capacity issues at a local level.

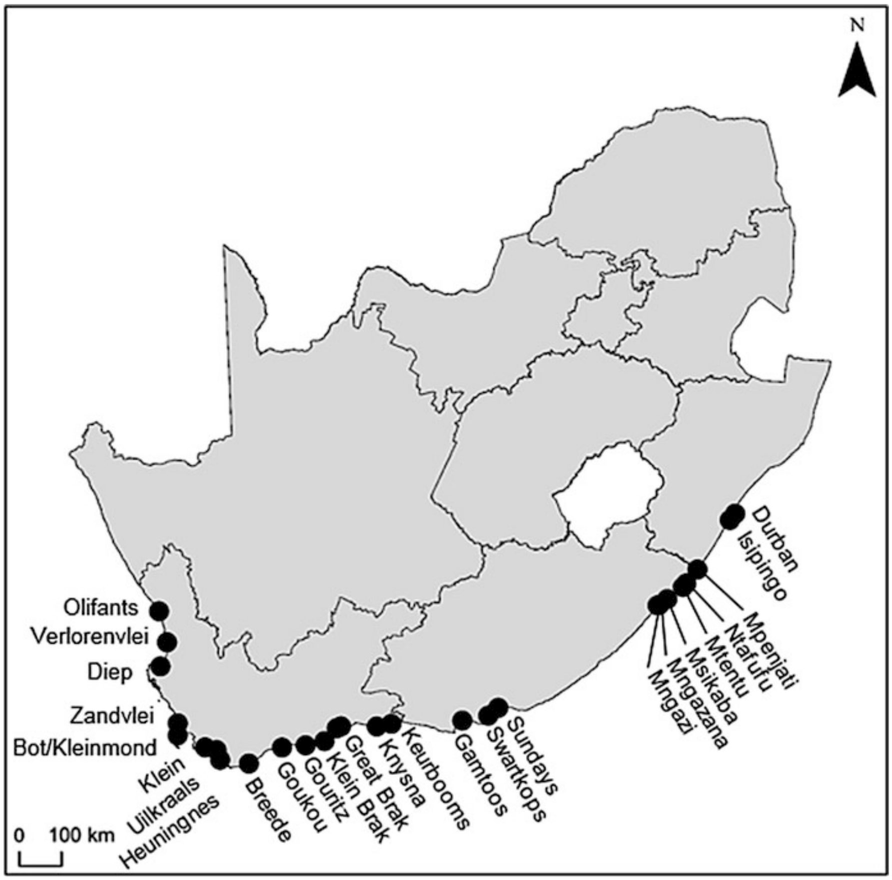


Fig. 4 Management plans have been developed for estuaries along the entire South African coastline, based on the systems and processes developed within the CAPE Estuaries Programme and legislated for in terms of the National Estuarine Management Protocol (Government Gazette No. 36432, 10 May 2013)

Towards Implementation – Funding, Technical and Institutional Support

Estuary management is a long term process and should not be project base. During the initial phase of C.A.P.E. Estuaries Programme, funding was provided by the C.A.P.E. programme as well as the National Department of Marine and Coastal Management (later to become Department of Environmental Affairs). This funding covered the running costs of the Programme, the three initial support documents, the first six estuary management plans and 50% of the next five co-funded estuary management plans. It also covered two estuary management plan review processes. The Task Team understood that the C.A.P.E. and National government funding would not continue in the long term. Plans had to be made to develop funding strategies to ensure the sustainability of the programme in the long term. The governance of any such programme is as important as the technical documents and stakeholder engagement. Given the complexity of estuary management and the diverse sectors involved, the governance aspect of this particular field required a focussed effort. Managers needed to think differently about their approach and the integration of the different Departmental mandates. Egos had to be put aside to achieve the final goal – effective estuary management in South Africa.

In essence, the CEP Task Team needed to continually evaluate funding streams, looking not only to national Departments such as Department of Water Affairs, but also to Institutions such as Catchment Management Agencies (CMAs), which were able to focus on freshwater issues such as Reserve Determination (National Water Act, 1998) processes and provide support to the various planning processes. Local Government Departments were also approached for funding, given that many of the estuaries existed within the Municipal areas. Funding applications were submitted to funding bodies such as the Green Trust, World Wildlife Fund, Wildlife Society of South Africa and Birdlife South Africa amongst. These proposals were submitted by C.A.P.E. Estuaries Programme on behalf of Local or District Municipalities.

Possible funders were identified and their mandates were assessed for compatibility with CEP. The co-ordinator of the CEP Task Team then developed funding proposals, in association with relevant Government Department officials. If the funders approved a proposal, a Memorandum of Understanding or Service Level Agreement was developed between CapeNature and the funding body. CapeNature was the responsible agent in each case, as the Task Team coordinator was contractually based in CapeNature. This process has ensured ongoing funding up until 2014. This process included developing a novel co-funding option which enabled funders with different mandates to fund specific aspects of the plan.

Upon the publication of the National Estuarine Management Protocol (Government Gazette No. 36432, 10 May 2013, www.environment.gov.za), National Department of Environmental Affairs ceased to fund CEP and funding had to be sourced from mandated Departments and NGOs instead. This process has now been initiated, mainly by embedding CEP into the Provincial estuary management structures in the Western Cape Province. A Western Cape Estuaries Programme has been created. If successful, this concept can be implemented by other coastal Provinces in South Africa. The Western Cape Estuaries Programme acts as an advisory committee to the Provincial Coastal Committee, which has been formally constituted in terms of the Integrated Coastal Management Act. The national flavour of the CEP Task Team will be incorporated into the newly constituted National Estuaries Subcommittee, an ad hoc advisory body for the formally constituted National Working Group Eight Committee. The latter committee acts as a National Coastal Committee in terms of the Integrated Coastal Management Act. Estuary management is now formally embedded in the management of National, Provincial and Municipal Coastal Committees, thus achieving one of C.A.P.E. Estuaries Programme's long term goals.

In order to structure the development of estuary management plan standards, CEP organised two estuary management plan review processes. These were reviewed over time, and eventually formed the basis for National review standards and processes. This work fed into the National Department of Environmental Affairs Estuary Management Plan review process that was initiated in 2014. Twenty six Estuary Management Plans that had been developed since 2006 have been reviewed and are being appropriately edited. They will soon be ready for adoption. The adoption process will need to be refined over time, as the plans involve multi-stakeholder engagement and mandates. A National Guideline for the development and implementation of estuary management plans has also been developed in 2015 (www.environment.gov.za).

Estuaries as Centres for Multi-stakeholder Engagement

The Integrated Coastal Management Act (2008) aims to facilitate the mandates of other Departments and to guide integrated management processes. In South Africa, there is a mass of legislation that applies to estuaries at all levels of Government, i.e. – Local, District, Provincial, National and International. Estuarine systems are the receiving environments of freshwater from catchments. Freshwater transports valuable sediment and nutrients, as well as pollutants, and the amount and quality of water entering estuaries is thus affected by land- and water-use in catchments. An estuary management plan needs to address these and other issues, without trespassing on the mandates of other Government Departments. True cooperative governance is required to achieve the desired management outcomes.

In cases where user conflict is inevitable, well designed zonation maps can be created depicting use zones. In some cases, trade-offs may be required. This should be achieved through a formal scientific process that includes stakeholders and is conducted in a professional and transparent manner. The estuary management forum members need to form part of the decision making process and the Management Authority should provide staff and equipment to ensure effective implementation which will in turn ensure ongoing provision of estuarine goods and services.

The estuary management plan is not a once-off initiative, but a document that must be revised every five years. This forms part of an adaptive management cycle that engages stakeholders on an ongoing basis and incorporates the findings of estuary monitoring and evaluation. The constant engagement of all stakeholders results in a continual educational process. Each user or interest group is continually sensitized to the issues of the other groups and this tempers any conflict that arises. Estuary management forums or committees have been successfully used to achieve this goal. Quarterly or biannual meetings regularly inform stakeholders with the latest progress and information. Results of research and monitoring programmes can be presented to the forum.

Members of forums can act as the eyes and ears of the appropriate Government Departments. This is important in a country where capacity is cited as the reason for non-delivery of services in many sectors. Forum members can also help to secure private sector funding to assist with the development of estuary management plans. In specific cases, forum members may assist Department officials in achieving their mandated outcomes, as has been the case in the estuary monitoring programme of the Department of Water Affairs. Estuary forums have successfully been used to drive the implementation of management plans. They help to keep participants motivated and focussed on priority issues. They also act as sounding boards for policy developers and decision makers.

Estuaries Form an Integral Part of the Overall Coastal Management Process

The integration of estuary management plans into Coastal Programmes, Coastal Committees and broader coastal planning is essential. To this end, C.A.P.E. Estuaries Programme assisted Local, Provincial and National Government Departments that are involved in coastal management to incorporate estuary management into their broader processes. Estuary-related issues were formally included in appropriately structured Coastal Committee agendas and Coastal Programme documents, thereby focussing attention on estuaries as important ecosystems, the management of which must be formally addressed. This facilitated the recognition of Estuary Management Plans in all local, provincial and national spatial planning processes, the associated development planning systems and processes and protected area planning and management processes. The estuary management plans had to be structured appropriately, to facilitate their inclusion in the integration processes.

Each estuary management plan included an implementation plan that was reported on to the Management Authority, the Municipal Coastal Committees and Provincial Coastal Committees, allowing for implementation to be audited. The Provincial Coastal Committee in the Western Cape Province developed an estuary reporting template to guide this process, resulting in a consistent reporting format and process for estuaries. The understanding was that if the Management Authority addressed each issue in the template, then the estuary was being effectively managed. This process must now be expanded and possibly embedded in a system such as the Protected Area Management Effectiveness Tracking Tool. The effectiveness of the Management Effectiveness Tracking Tool can be tested for those estuaries that fall within the management mandate of Conservation Departments. This is a priority for the year 2015.

At a local level the implementation of an estuary management plan will require a management team. The Integrated Coastal Management Act is relatively new and imposes new regulations on existing Management Authorities with little or no estuary management capacity. Estuary management forums comprising representatives from local stakeholder groups as well as representatives from appropriate Government Departments were set up to assist these Management Authorities. Forums helped create a cooperative approach to the implementation of estuarine management plans. In many cases, it allowed local stakeholders to assist Government officials with the implementation of their mandated activities at specific estuaries. Furthermore, inter-Governmental teams were empowered to address specific management problems that officials from single Departments were not able to do due to capacity or funding constraints. The estuary management planning process needs to be managed in such a way that it facilitates the development of trust and cooperation between stakeholders, thereby allowing these integrated teams of Government officials and members of the public to develop over time. Estuary Management Forums were set up to act as advisory committees for the appropriate Municipal Coastal Committees. This was accepted by the three Municipal Coastal Committees (West Coast, Overberg and Eden) in the Western Cape Province, and provided a management framework in which Estuary Management Forums could operate.

Funding to cover running costs of forums still remains a challenge. Various funding streams do exist in South Africa, and many of these focus on job creation. Managers need to think creatively in order to harness these funding streams in support of management forums. Other South African legislation also includes the need for and the management of stakeholder forums, e.g. National Water Act(Act no. 36 of 1998) and Catchment Management Forums, National Environmental Management: Protected Area Act(Act no. 57 of 2003) and Protected Area Advisory Committees and lessons learned can be taken from the management of these committees.

While it is understood that each estuary is different and therefore has its own specific management issues, there are also issues that are common to several estuaries. Specific mouth and reed management protocols have been developed to assist Management Authorities to address these critical aspects of estuary management. A relatively new issue that has been identified at a national level is the need for guidelines for sand mining in the estuarine zone.

Monitoring to Make a Difference

In addition to the possible application of the proposed long term Management Effectiveness Tracking Tool, each estuarine management plan identifies priority monitoring programmes. In each case, the Department or Institution best suited to carry out this monitoring has been identified. This is a relatively new field in estuary management, so capacity must be developed over time. With this in mind the C.A.P.E. Estuaries Programme and its partners have set up several broad cooperative monitoring programmes and an associated database.

Department of Water Affairs has effectively set up and implemented a new estuary monitoring programme which is aligned to the aims of the C.A.P.E. Estuaries Programme. Cooperation between this Department and

Estuary Management Forums is the cornerstone of its success. While the department provides funding for equipment and running costs, members (public or private) of the forums carry out the sampling and courier samples to local laboratories. The department presents the results of analyses at forum meetings. All data are curated on the department's national database. In addition, the South African Environmental Observation Node is developing a web based National Estuary Database(http//: saeon.gov.za) that will be used to house all monitoring data and official documents.

To help develop monitoring and research capacity in South Africa, CEP and partners developed an accredited Estuary Management Training Course, funded by Department of Environmental Affairs and implemented by Nelson Mandela Metropolitan University in the Eastern Cape Province. This course has since been rolled out to all Provinces in South Africa.

There are now sufficient data available for the South African Government to develop a "State of the Estuaries" report that can be used to assess the state of the nation's estuaries each year. The 2011 National Biodiversity Assessment of (Driver et al. 2012) was the first such assessment that considered estuaries, and that linked the state of estuaries to the state of catchments. Since then, significant progress has being made in understanding how catchment, estuaries and the ocean interact. Furthermore, an understanding of the nature of the required management interventions in order to maintain the functionality of these interactions is now emerging at all levels of Government in South Africa.

Conclusions

In South Africa, a structured process to address estuary management has been developed within the legislative framework of the Integrated Coastal Management Act. The process has been tested on a range of estuary ecosystem types and for estuaries with an array of social and economic conditions. The embedding of the specific estuary management plans and management forums into the broader coastal management programmes and committees has also successfully been tested under varying conditions. The legislation and management structures have been found to be robust and effective and have facilitated the implementation of management interventions identified in the management plans. Continued creative thinking and management is needed in order to take advantage of existing and new funding streams. Ongoing capacity building within Management Authorities needs to take place.

The ongoing need to develop new estuary management plans must be prioritized. These will secure estuarine ecosystem services into the future. Funds should be allocated for this purpose. A formal process is required to develop scientific and management capacity within the field of estuary management in South Africa. Linking this to the developing databases, a National "State of Estuaries" report can be published each year with the aim of identifying progress as well as gaps. South Africa needs to build on the momentum created within the field of estuary management by the C.A.P.E. Estuaries Programme.

Acknowledgements I would like to take this opportunity to acknowledge the contributions made by all those involved in the setting up and implementing of the C.A.P.E. Estuaries Programme. Without their contribution this programme would have simply been "a nice idea". Thanks to Marienne de Villiers, Janine Adams and Rikki de Villiers (map) for advice and comments on the text.

References

CSIR (2006) C.A.P.E. Estuaries Programme. Regional management workshop proceedings, 27–28 September 2005. Report submitted to the C.A.P.E. Estuaries Programme, 29 pp

CSIR (2007) C.A.P.E. Estuaries Programme. Proposed generic framework for estuarine management plans. Report submitted to the C.A.P.E. Estuaries Programme, 43 pp + appendices

Department of Water Affairs (DWA) (2010) Resource directed measures for protection of water resources: methods for the determination of the ecological reserve for estuaries. Water resource protection and assessment policy implementation process. Version 3, Pretoria

Department of Water Affairs and Forestry (DWAF) (2008) Water resource protection and assessment policy implementation process. Resource directed measures for protection of water resources: methodology for the determination of the ecological water requirements for estuaries. Version 2, Pretoria

Driver A, Sink KJ, Nel JN, Holness S, Van Niekerk L, Daniels F, Jonas Z, Majiedt PA, Harris L, Maze K (2012) National biodiversity assessment 2011: an assessment of South Africa's biodiversity and ecosystems. Synthesis report. South African National Biodiversity Institute and Department of Environmental Affairs, Pretoria

Heydorn AEF, Tinley KL (1980) Estuaries of the Cape. Part 1. Synopsis of the Cape Coast, CSIR research report 380. CSIR, Stellenbosch

Kairo JG, Dahdouh-Guebas F, Bosire J, Koedam N (2001) Restoration and management of mangrove systems—a lesson for and from the East African region. S Afr J Bot 67:383–389

MacKenzie M, Hay D (2002) Eastern Cape Estuaries Management Programme. Report on Phase 1 of the Eastern Cape Estuaries Management Programme, July 1998–December 2000. WRC report no. 1018/1/01. ISBN: 1 86845 847 4

NOAA (2013) Ecosystem science: large marine ecosystems. National Oceanic and atmospheric administration. http://www.st.nmfs.noaa.gov/ecosystems/lme/index. Accessed Oct 2014

Sink K, Holness S, Harris L, Majiedt P, Atkinson L, Robinson T, Kirkman S, Hutchings L, Leslie R, Lamberth S, Kerwath S, von der Heyden S, Lombard A, Attwood C, Branch G, Fairweather T, Taljaard S, Weerts S, Cowley P, Awad A, Halpern B, Grantham H, Wolf T (2012) National biodiversity assessment 2011: technical report. Volume 4: marine and coastal component. South African National Biodiversity Institute, Pretoria, p 325

Tamar Estuaries Consultative Forum. www.plymouth.gov.uk/tecf. October 2012

TIDE Tidal River Development. www.tide-project.eu. August 2011

Turpie JK, Clark BM (2007) The health status, conservation importance, and economic value of temperate South African estuaries and development of a regional conservation plan. Report to CapeNature

UNESCO (2004) Cape Floral Region protected areas. Description. http://whc.unesco.org/en/list/1007. Accessed Oct 2014

Future Outlook: The Importance of a Science and Ecosystem-Based Approach for the Sustainable Management of Estuaries and Their Life-Supporting Ecosystem Services in the Western Indian Ocean

Salif Diop, John Machiwa, and Peter Scheren

Abstract

The need, but also the momentum for improved management of the coastal ecosystems of the Western Indian Ocean (WIO), including its estuaries, is increasing. Initiatives by Governments of the Mainland and Island States of the WIO region to develop and enact appropriate environmental laws and putting relevant regulations in place are highly commendable. The main challenge now involves the enforcement of the same, while a critical need remains to raise more awareness among the general public on issues related to the environmental health and sustainability of the WIO coastal and marine environment. In light of the scarcity of reliable information and data on marine and coastal ecosystems, there also is a critical need for more research to decipher the complex, and often subtle, relationships between land and ocean at the WIO coastal zone. Scientific approaches and research may therefore provide an important basis for better decision making related to the sustainable management of the WIO coastal zone and its important resources.

Keywords

Ecosystem approach • Estuary management • Scientific approach • Ecosystem health • Western Indian Ocean

Introduction

Although the WIO region is still one of the least ecologically disturbed areas of global ocean relative to other regions, it is becoming increasingly threatened; indeed, the region's coastal and marine environment has begun exhibiting significant signs of degradation over the last decade, attributable to both natural factors (climate change/variability leading to coral bleaching, sea level rise, flooding, etc.) as well as a variety of anthropogenic activities, all acting at different intensities and in varying combinations, as exemplified in the preceding chapters.

Current Status

The WIO coastal zone region is the site of most major cities, harbours, industries and other socio-economic infrastructure increasingly affecting the marine environment. Although the population density of the region as a whole is not remarkably high, more than 60 million people inhabit the coastal zones of the region. Pressures associated with urbanization are most marked in the mainland states, containing such major cities as Mombasa (Kenya), Dar es Salaam (Tanzania), Maputo

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(Mozambique) and Durban (South Africa), each of which support populations of 2 to 4 million people.

Because of increasing population pressures, combined with a lack of alternative resources to sustain the local populations, resource extraction is becoming unsustainable. In some areas, for example, coastal habitats have been converted to other uses such as agriculture, aquaculture, ports/harbours and urban settlements. Such developments are leading to the destruction of important coastal habitats such as mangrove forests, sand dunes, seagrass beds and coral reefs, as well as physically altering the coastline (via erosion and accretion) because of the loss of the natural coastal protection and regulation functions of these habitats. Further, over-fishing and unsustainable fishing practices (dynamite fishing, the use of drag-nets, etc.) have resulted in declining fisheries resources and fish harvests in many areas (UNEP/Nairobi Convention Secretariat 2009), as well as causing further destruction of coastal habitats such as coral reefs (see chapter "Coral Recruitment and Coral Reef Resilience on Pemba Island, Tanzania").

The impacts of such activities are most evident in the interactions between river basins and the downstream coastal-marine environment (LOICZ 2002). As demonstrated in preceding chapters, the impacts of human activities on the river basins have altered the nature of the interactions between river systems and coastal processes throughout much of the WIO region; the most prominent examples of these impacts appear to be associated with changed flow regimes (see chapters "Integrated Water Resources Management in a Changing Climate: The Implication of Anthropogenic Activities on the Tana and Athi/Sabaki Rivers Water System for Sustainable Development", "Estuarine Environmental and Socio-economic Impacts Associated with Upland Agricultural Irrigation and Hydropower Developments: The Case of Rufiji and Pangani Estuaries, Tanzania" and "The Hydrodynamics of the Incomati Estuary – An Alternative Approach to Estimate the Minimum Environmental Flow") and sediment fluxes (see e.g., chapters "Tana Delta and Sabaki Estuaries of Kenya: Freshwater and Sediment Input, Upstream Threats and Management Challenges" and "Hydrodynamic Modelling on Transport, Dispersion and Deposition of Suspended Particulate Matter in Pangani Estuary, Tanzania"). Also, nutrients and pollutants from domestic sewage, industrial effluents and agricultural chemicals have caused significant water quality degradation in some of the major river systems, including the Tana and Athi-Sabaki rivers in Kenya (see chapter "Integrated Water Resources Management in a Changing Climate: The Implication of Anthropogenic Activities on the Tana and Athi/Sabaki Rivers Water System for Sustainable Development"), the Pangani river in Tanzania and the Maputo river in Mozambique (UNEP/Nairobi Convention Secretariat, ACWR and WIOMSA 2009).

An overview of river-induced coastal and marine degradation 'hotspots' for the 15 main river basins in the WIO region, as identified in a recent regional assessment of river-coast interactions (UNEP/Nairobi Convention Secretariat, ACWR and WIOMSA 2009) is presented in Table 1.

Based on this overview, it may be concluded that alteration of river flows is the most common issue throughout the WIO region, in terms of the severity of problems associated with these river systems, essentially as a result of increased water abstractions (for urban and rural water supply schemes and irrigation), river damming (for hydroelectric power generation and irrigation), and land use changes that alter hydrologic dynamics. There also are cases where changes in sediment loading (increases or decreases) and water quality have had severe impacts on the productivity of such critical coastal habitats as mangroves, seagrass beds and coral reefs. The most affected main river basins include:

- Pangani (Kenya and Tanzania);
- Athi-Sabaki (Kenya);
- Incomati (South Africa, Swaziland and Mozambique);
- Zambezi (Angola, Botswana, Democratic Republic of Congo (DRC), Malawi, Namibia, Tanzania, Zambia, Zimbabwe and Mozambique); and
- Betsiboka (Madagascar).

In addition to these main rivers, numerous smaller rivers found throughout the WIO region have not escaped human influences, although the extent of alteration strongly varies with the latter.

The socio-economic consequences of these changes to coastal and estuarine ecosystems are fundamentally important in the WIO region. They range from direct impacts such as increased coastal vulnerabilities to climate change and other natural causes, including erosion, reduced fish catches, declining water quality impeding access to safe water for drinking and other uses, and lost touristic and recreational potential, to more indirect impacts such as the loss of carbon sequestration potential, visual amenities and cultural values (see among others, chapter "The Socioeconomic Causes and Impacts of Modification of Tana River flow Regime"). This is an important consideration, given that the current (conservative) estimate of the economic value of the ecosystem goods and services provided by the WIO coastal and marine environment exceeds 25 billion US dollars annually (UNEP/Nairobi Convention Secretariat 2009), with tourism and fisheries being the two main income earners for the region's inhabitants.

Table 1 Hot spots of river-induced coastal and marine degradation in the WIO region (after UNEP/Nairobi Convention Secretariat, ACWR and WIOMSA 2009)

COUNTRY[a]	HOT SPOT	TRANSBOUNDARY PROBLEM			
		Alteration in river flow	Degradation of water quality	Increase in sediment load	Decrease in sediment load
Kenya	Tana	■	■		■
	Athi-Sabaki	■	■	□	
Madagascar	Betsiboka	■		□	
	Tsiribihina	■		■	
	Mangoky	■		■	
	Fiherenana	■		■	
Mozambique	Zambezi	□			□
	Pungue	■			■
	Limpopo	□			
	Incomati	□			□
	Maputo		■		
South Africa	Thukela	■			■
Tanzania	Pangani	□	□	■	
	Rufiji		■		
	Ruvuma				

[a]Refers to country of river outflow
■ Issues of 'critical' concern

Future Outlook

In view of these extremely important values, the effective and timely governance and management of these assets is a critical factor in maintaining the valuable functions and services they provide to humanity. Fortunately, the latest decade has witnessed increasing attention being paid to the importance of the marine and coastal environment of the WIO region. Indeed, this book is itself a display of some of the important scientific work that has been, and is being, undertaken by scientists from the WIO region and beyond. Furthermore, as described in chapters "Re-Thinking Estuarine Ecosystem Governance in the WIO Region" and "Estuary Management in South Africa – An Overview of the Challenges and Progress Made to Date", Governments are increasingly devoting attention to the matters at hand, including national Government systems, River Basin Organisations, and other key regional bodies (e.g., Nairobi Convention) gaining maturity, capabilities and influence.

As demonstrated again in this book, as in previous editions of Estuaries of the World, estuarine and other coastal ecosystems represent a critical link between the land and the sea. They are complex, they are biologically productive and they are important, both for human existence and environmental sustainability, whether in Africa or other regions of the world. Completion of this book devoted to ESTUARIES, as a LIFELINE OF ECOSYSTEM SERVICES IN THE WESTERN INDIAN OCEAN, therefore, serves an important purpose. Firstly, it is geared towards increasing knowledge of the structure and functions of estuarine ecosystems in the region, highlighting the life-supporting ecosystem goods and services they provide to humanity. Secondly, it aims to enhance networking and to increase the capacities of young scientists working on African coastal zones, while establishing research communities through extensive networks. As information and data are fundamental to improved monitoring and management of estuarine and other coastal ecosystems, the development and mechanisms for long-term exchange of data and information, including the maintenance of the systems and networks to support such is essential. In this regard, the challenge ahead of us is to constitute and support viable centers, such as WIOMSA (West Indian Ocean Marine Science Association), in maintaining the necessary systems and networks for quality information exchange to allow for better informed decision-making, including their linkages with other regions of the African continent.

References

LOICZ (2002) African basins: global change assessment and synthesis of River Catchment-Coastal sea interaction and human dimension. LOICZ reports & studies no. 25. LOICZ, Texel, ii + 344 pp

UNEP/Nairobi Convention Secretariat (2009) Strategic action programme for the protection of the coastal and marine environment of the Western Indian Ocean from land-based sources and activities, Nairobi, 164 pp

UNEP/Nairobi Convention Secretariat, ACWR and WIOMSA (2009) Regional overview and assessment of River-coast interactions in the Western Indian Ocean Region. UNEP, Nairobi

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