WESTERN INDIAN OCEAN

CRITICAL HABITATS OUTLOOK

Towards achievement of the Sustainable Development Goals

UNEP

Nairobi Convention

gef

WIOMSA Coasts Ocean and People

WESTERN INDIAN OCEAN MARINE SCIENCE ASSOCIATION

SWEDEN

WESTERN INDIAN OCEAN

CRITICAL HABITATS OUTLOOK

Towards achievement of the Global Biodiversity Framework Targets

Published by the United Nations Environment Programme/Nairobi Convention Secretariat.

Nairobi Convention Secretariat
United Nations Environment Programme
United Nations Avenue, Gigiri
PO Box 47074
Nairobi, Kenya
Tel: +254 (0)20 7621250/2025/1270
Fax: +254 (0)20 7623203
Email: nairobi.convention@unep.org

Coordinators for the preparation of the Critical Habitats Outlook: Jared Bosire, Dixon Waruinge and Julius Francis ??
Editor: José Paula
Copy editor: Matthew Richmond
Editorial assistant: Mariana Cravo
Maps: Marco Nunes Correia
Layout: Desiré Pelser | Earth & Oceans Developments

Cover: STILL OUTSTANDING feedback and changes on COVER_PROOF 1_dd. 03 March 2023
For citation purposes this document may be cited as:
UNEP-Nairobi Convention and WIOMSA. 2024. *Western Indian Ocean Critical Habitats Outlook: Towards achievement of the Global Biodiversity Framework Targets.* UNEP and WIOMSA, Nairobi, Kenya, 434 pp.

ISBN: 978-9976-5619-3-7

EDITOR
José Paula

COPY EDITOR
Matthew Richmond

AUTHORS
Janine Adams, Vanessa Adams, Andrea Angel, Rose Antipa, João Azevedo E Silva, Salomão Bandeira, Rhett Bennett, Dave Van Beuningen, Jared Bosire, Peter Chadwick, Mariana Cravo, Stephanie D'agata, Tammy Davies, Sean Fennessy, Luisa Fontoura, Julius Francis, Paul Gacheru, Majambo Gamoyo, Victoria Graham, Johan Groeneveld, Sabrina Guduff, José Guerreiro, Issufo Halo, Vera Horigue, James Kairo, Blandina Lugendo, Célia Macamo, Joseph Maina, Timothy McClanahan, Daudi Msangameno, Nyawira Muthiga, Doris Mutta, Remy Oddenyo, José Paula, Rivo Rabarisoa, Anusha Rajkaran, Matthew Richmond, Michael Schleyer, Aurélie Spadone, Vikash Tatayah, Arthur Tuda, Dixon Waruinge, Chemuku Wekesa

FOREWORD

STILL OUTSTANDING

EXECUTIVE SUMMARY

The *WIO Critical Habitats Outlook* is one of the main outputs of Component A of the Global Environment Facility (GEF) funded project '*Implementation of the Strategic Action Programme for the protection of the Western Indian Ocean from land-based sources and activities (WIOSAP)*', designed to respond to sustainable development goal (SDG) 14.2 and 14.5; and related Aichi Biodiversity targets, besides promoting various management approaches and tools.

It further intends to address conservation challenges from the previous decade and reverse the decline in biodiversity, an aspiration of the Convention on Biological Diversity's Kunming Montreal Global Biodiversity Framework (GBF). The GBF encompasses four goals and 23 targets designed to reduce species extinction through sustainable management of biodiversity and associated ecosystem services, protection of genetic resources and traditional knowledge, and provision of financing mechanisms and capacity for implementing necessary policies. In particular, Target 3 envisages the conservation of at least 30 per cent of degraded terrestrial, inland water and coastal/marine ecosystems by 2030.

Together with the regional *MPA Outlook*, the *Critical Habitats Outlook* initiative proposes to inform policy-making about enhanced coastal and marine conservation in the Western Indian Ocean (WIO) region, with a wider aim at supporting Contracting Parties to meet their obligations under the GBF Targets, and corresponding former SDG and Aichi targets.

The development of both Critical Habitats Outlook and MPA Outlook drew strongly on an earlier partnership project between the Nairobi Convention and WIOMSA, namely the production in 2015 of the *Regional State of the Coast Report: Western Indian Ocean*. The Nairobi Convention, through relevant COP decisions, gave oversight in the production of the *Outlooks* and managed inter-governmental coordination and national validation as well through the GEF-funded WIOSAP project. WIOMSA provided a solid connection throughout the scientific, academic, and technical communities across the WIO region, playing a pivotal coordination role in the process and contributing financially to the initiative through the Marine Science for Management (MASMA) Programme.

The general purpose of the *Critical Habitats Outlook* is to evaluate the most important and critical marine and coastal habitats of the WIO region, and in particular, to:

- describe the most important and critical habitats of the WIO and the relevance of their associated biodiversity;
- review the socio-economic use and dependence by coastal human communities on the WIO marine habitats;
- highlight gaps regarding the scientific knowledge of WIO marine habitats;
- review the current levels of protection of the WIO marine habitats and identify areas and opportunities for increasing protection; and
- develop alternative scenarios for the future protection of the marine habitats in the WIO.

The environmental setup

The WIO region encompasses tropical and subtropical regions that support rich biological diversity nature along the mainland countries of Somalia, Kenya, Tanzania, Mozambique and South Africa, and vast oceanic areas surrounding the island states of Madagascar, Seychelles, Comoros, Mauritius, and the French Territories of La Reunion. The complexity and wide geographical span of the WIO region create environmental gradients and contrasts, providing the basis for compartmentalization and regionalization based on different criteria and classification schemes (see Chapter 2).

The north of the WIO is strongly influenced by the monsoon regime of the Arabian Sea, which pulses seasonally and triggers coastal upwelling and associated biological productivity. In the central WIO, the main equatorial current meets the African continent and splits into two major currents along the continental coastal waters to the north and south. In the Mozambican Channel, the current moves southward through complex systems of gyres that meet the Agulhas Current and transports energy to higher latitudes in the southern hemisphere. The vastness of the WIO and its complex oceanographic dynamics (see Chapter 3) create a biophysical mosaic of coastal and offshore environments that spread from temperate to tropical habitats of diverse nature.

Broad threats and pressures affecting the WIO region

Threats to the environment in the WIO can be broadly categorized as those which are natural, for example, episodic events (cyclones, tsunamis, floods) and anthropogenic or human in cause, for example, exploitation (direct and indirect), habitat destruction (land 'reclamation', urbanization, dredging, mining and oil/gas extraction), pollution (point and diffuse sources) and climate change (including ocean acidification and sea-level rise). The 2015 Western Indian Ocean Transboundary Diagnostic Analysis (WIO TDA) presents a detailed regional analysis of these threats. The strong linkages between land-based activities and nearshore marine ecosystems and associated socio-economics demand that marine resource management evolves to consider human activities on land (see Chapter 4). The complexity of processes linking basin land-use change to changes in coastal ecosystems hinders effective integrated land-sea planning. Overcoming this complexity can be facilitated through efforts to integrate models from the drivers of land-use change to management responses for marine ecosystems. Among other pressures, urbanization and forest conversion for agriculture continue to alter hydrological processes and regimes within coastal catchments.

These processes underpin land-sea connectivity and all ecological functions and water quality outcomes directly linked to the health of the adjacent marine environment. Adopting source-to-sea and integrated land-sea planning approaches will help mitigate the impact of upstream processes on the coastal/marine environment.

WIO critical habitats and associated biodiversity

The core of the *Critical Habitats Outlook* is a presentation of the main habitats of the WIO, particularly those that may be considered critical in terms of biodiversity and ecological functioning and, as such, provide the most ecosystem services to human populations. Appropriate definitions, vulnerability and conservation categories are contextualized throughout this volume (see Chapter 5). The chapters include descriptions of habitats, relevant taxa such as marine birds and threatened species, and morphological features that are not obviously included within specific habitats but constitute complex environments of great importance for biodiversity, such as seamounts or small islands.

Rocky outcrops and sedimentary formations create a diversity of coastal configurations along the coastlines of the WIO countries (see Chapter 6). Ecologically, these highly variable habitats are important areas on the coast-sea interface, providing a multitude of microhabitats and niches for organisms, including breeding and nursery areas for many species. They also serve as important feeding and foraging grounds for both terrestrial and marine animals. Due to their accessibility, rocky outcrops and sedimentary coast resources are intensively used as a source for coastal livelihoods in the WIO, providing a major source of income for artisanal subsistence and food security in the region. Several phenomena and activities threaten nearshore habitats in the WIO region, affecting their ecological productivity, integrity and, by extension, livelihoods and economies. Estimation of the cover area of these nearshore habitats is necessary, as well as increasing their protection by incorporating additional areas into the existing marine protected areas (MPAs) and locally managed marine areas (LMMAs) within each national jurisdiction.

Mangrove forests are widespread in the WIO (see Chapter 7), and their ecological importance extends from coastal protection to biodiversity maintenance and from mitigation to adaptation to climate-induced changes. Mangrove forests sustain extensive fisheries in addition to being directly used, mainly as building material and firewood. Mangroves can store higher amounts of carbon than that accumulated by other terrestrial vegetation systems and sustain tangible livelihoods, including ecotourism, while supporting some of the largest fisheries in the region. Anthropogenic threats to mangroves include habitat destruction for land reclamation and over-exploitation of their resources. Global phenomena also impact mangrove forests and contribute to their degradation, such as sea-level rise and extreme events like storm surges and floods.

National agendas on mangroves should be re-visited so that they are mainstreamed with global platforms such as the main targets of the SDGs, including the incorporation of these blue carbon ecosystems in countries' nationally determined contributions (NDCs) under the Paris 2015 Agreement. Some information gaps still need to be addressed, such as mapping forests and vulnerable areas and assessing threats at local scales. Integrating the wider society, both at local and country levels, will help improve the steering of the discussion on tackling the wider mangrove management challenges in the WIO. In view of local degradation and deforestation rates, the WIO countries must also strategize the implementation of mangrove restoration programs involving local communities, thus contributing to the UN Decade of Ecosystem Restoration. Countries should also take advantage of the carbon financing opportunities to raise the profile of mangroves and generate resources for community development and conservation.

Seagrass meadows are distributed along the coastlines of the WIO mainland and the Island States. In most countries of the region, seagrass beds often occur in close connection with coral reefs and mangroves. Seagrasses form key components of marine ecosystems but have received limited scientific attention. Comprehensive mapping of seagrass beds has not yet been achieved for most countries in the region, and hence the total seagrass coverage in the WIO region is not fully understood (see Chapter 8). Seagrasses are one of the most productive aquatic ecosystems in the world, supporting productivity through the recycling of nutrients and carbon. Seagrasses also stabilize sediment, thereby reducing coastal erosion and strengthening coastal protection. Seagrasses also provide many important ecosystem services through support to fisheries and tourism industries, reliant on the ability of healthy seagrass beds to support finfish, shellfish, and other fishery-related products.

Most threats to seagrasses result from human activities, though natural causes also account for seagrass loss in the region. Important anthropogenic threats are eutrophication due to excessive nutrient input into coastal waters, sedimentation originating from various sources, and physical destruction related to water-based leisure activities. Information regarding the status of seagrass beds within the WIO is largely lacking. There is inadequate protection of seagrass habitats in the WIO region, and hence there is a need to identify priority areas for conservation as well as opportunities that can be used to enhance seagrass protection. Mechanisms should be implemented at the regional level to ensure regional collaboration and joint actions for the conservation of seagrass ecosystems, including restoration programs.

Salt marshes are typically temperate coastal habitats and, in the WIO, occur mainly on temperate South African shores (see Chapter 9). These productive ecosystems are important for carbon storage, water purification, flood control, refugia, and habitat for other organisms. Salt marsh plants are also increasingly used for human consumption. Threats to salt marshes include sea-level rise at the seaward interface and coastal development at the land interface. The latter include land reclamation for agriculture, seawater evaporation ponds for salt production, shellfish or fish farming ponds or livestock production that restrict tidal exchange and promote the establishment of invasive species. There is a degree of protection in South Africa of the larger salt marshes and some degree in the legislation of other WIO countries, but overall, there is a need for better attention and research to fill gaps of knowledge regarding the distribution and condition of salt marshes in WIO countries.

Coral reefs fringe most shorelines in the WIO (see Chapter 10), supporting a wide range of goods and ecosystem services and generating many benefits for local and national economies. These include the provision of seafood and other resources that are important for the livelihoods of coastal communities. Coral reefs also provide regulatory services such as beach replenishment and coastal protection. Coral reefs further support important revenues in tourism, fisheries and trade. Coral reefs are connected to and interact with adjacent coastal and marine ecosystems such as mangroves and seagrass beds that contribute to the integrated seascape ecological functioning. WIO coral reefs are threatened by multiple factors, the three main ones being climate-associated disturbances, fishing, and the interrelated factors of nutrient pollution and sedimentation caused by human influences on land. The threat intensity is patchy in space and time. Coral reefs can experience one, all three, or all possible combinations of these degrading forces.

MPAs are the most implemented area-based tools in the WIO for coral reefs. In many cases, however, the design of MPAs did not consider marine zoning considerations such as representativeness (ecological and biodiversity), adequacy (size), and irreplaceability. The second most common area-based approach is co-management, which gives local communities a voice in conservation through a decentralized management model focusing on fisheries. More policy, institutional and funding support is required to strengthen co-management and other effective conservation measures (OECMs). Co-management approaches provide an opportunity to contribute towards the GBF 30x30 target.

Although countries of the WIO have invested in many programs and initiatives to protect and manage coral reefs, a more concerted effort is urgently needed because coral reefs are in imminent danger due to climate change disturbances, fishing pressure and the drive for coastal development to accommodate expanding populations. The adoption and expansion of Blue Economy policies may also be a double-edged sword, that strives to increase marine-based revenue but does not necessarily safeguard the marine environment in a sustainable way. In some cases, Blue Economy initiatives may result in simply adding pressure to heavily exploited marine resources, exacerbating the threats.

Estuaries are the transitional aquatic systems between the freshwater and marine environments and are among the most productive natural systems in the world. These systems export sediments, nutrients and organic matter to the continental shelf, enhancing coastal productivity. They often form complex ecosystems that include critical habitats such as mangroves, seagrass beds, salt marshes and extensive tidal flats. Due to their characteristics, estuaries have historically attracted the settlement of human communities, creating socio-ecological systems that have developed into most of the world's largest coastal cities. Multiple stressors threaten the natural balance of WIO estuaries (see Chapter 11). Sea-level rise impacts low-lying estuarine land, and floods from extreme events induce erosion and mangrove destruction. Further human-induced alterations at catchment scales, such as damming and water abstraction, as well as intensive agriculture and alterations of vegetation cover, put pressure on the natural ecological balance. Widespread pollution and habitat destruction through land reclamation contribute to the degradation of estuaries and the natural habitats and resources they contain. Protection for WIO estuaries is provided by international agreements on shared watersheds and further promoted by wetland conventions. Conservation of estuaries is complex because it includes the activities within the estuarine system and the upstream land-use activities. Thus, there is a need to integrate the management of the catchment. Another issue is that estuaries are very diverse regarding hydrological and ecological regimes, further impacted by diverse anthropogenic stressors, leading to the need for individually based management and action plans.

The offshore habitats and the deep-sea constitute most of the WIO and are largely unknown (see Chapter 12), particularly the benthic fauna from shelf sediments and deeper seabed. The threats to the vast offshore areas and the deep-sea can be broadly grouped into three categories – extraction of resources (renewable and non-renewable), contamination and pollution, and climate change, but also include unsuitable governance, economic factors, insufficient financial resources, a lack of knowledge and diverse pressures resulting from population growth, especially in the coastal zone. Shipping traffic in the region is also related to the regional economy and extraction of resources and is associated with increased pollution, ship strikes on cetaceans, and the spreading of invasive species from ballast water and fouling. Due to the vastness of the offshore areas, there is a need to prioritize conservation areas, but the immediate difficulty is that the majority remain under-explored, and information is lacking.

There are mechanisms in place for the declaration of protected areas within state-exclusive economic zones (EEZs) and a process to declare international MPAs, and there is also a need for effective management of existing protected areas in offshore habitats in the WIO. The Nairobi Convention, in collaboration with the Maritime Technology Cooperation Center (Africa International Maritime Organization (IMO) Partner), the Council for Scientific and Industrial Research (South Africa), WIOMSA and Macquarie University have supported the region to develop a toolkit for green port development in response to a Convention COP 8 Decision due to the huge ecological footprint ports have on the environment.

Over the last seven years, the number of marine species listed in the IUCN Red List that occur in the WIO increased (see Chapter 13), and the conservation of threatened species necessitates the conservation of their primary habitats. Among threatened species, there are various taxonomic groups, from invertebrates and fish (including iconic species such as the Coelacanth) to sea turtles and marine mammals. Threats to specific taxa depend on the species and its biology, ecology and distribution, but most ecosystems and species are prone to the impacts of global threats derived from climate change, pollution and widespread environmental degradation. Other threats include over-fishing and the ornamental species trade.

Appropriate management through integrated coastal zone management (ICZM) provides the best framework to protect vulnerable, threatened critical habitats, such as seagrass beds and coral reefs, and through these, protect many other species that depend on the habitats. Other specific measures must be adapted to individual taxa/species and their conservation requirements, and MPAs and community-managed areas are among the protection measures currently utilized in the WIO.

The WIO region habitats support a high diversity of seabird and coastal birds, including several endemic and near-endemic species (see Chapter 14). Seabird populations in the WIO are considered a fraction of the historical estimates, and many colonies have become extinct or greatly reduced in size. Seabirds are useful indicators for identifying priority sites for conservation, and their distributions can provide surrogates for biodiversity hotspots in marine spatial planning. Important Bird and Biodiversity Areas (IBAs) have been identified within the WIO. Seabirds face threats when nesting on land, including predation by invasive species (particularly rats), harvesting and human disturbance, and when feeding at sea, threatened by fisheries activities, both through depletion of food sources and mortality as bycatch. General conservation actions that are required include: conserving a network of sites (IBAs) across the WIO; removal of predatory, alien and invasive species from seabird breeding areas, feeding and/or aggregation; control of unsustainable harvesting; integrating bird conservation into ICZM and marine spatial planning; reduction of bycatch; and maintenance of long-term monitoring.

Seamounts and ridges are recognized as significant habitats for a wide diversity of species and are considered hotspots of biodiversity, have high endemism and attract a range of oceanic predators, including seabirds, whales and sharks (see Chapter 15). Seamounts and ridges are potentially impacted mainly by non-sustainable fisheries and seabed mining, especially considering that many of these habitats are located in international waters. The generalized lack of information regarding these systems creates enormous difficulty in assessing threats and specific protection measures. There is an urgent need to explore and survey these ecosystems to complete the picture of the biodiversity and productivity associated with the Indian Ocean. Efforts should be made to extend the geographical coverage of regional areas beyond national jurisdiction (ABNJs) and MPAs beyond national jurisdictions. Where relevant, promoting the establishment of Ecologically or Biologically Significant Marine Areas (EBSAs) may contribute to developing the conservation momentum for such sites.

The small islands of the WIO have a high diversity of country designations and vary in size from relatively large landmasses to small, isolated coral atolls widely scattered across large ocean spaces (see Chapter 16). Together, they have been identified as one of the world's biodiversity hotspots. As countries within the WIO intensify their efforts to achieve sustainable ocean economies, this places an increasing burden on the diverse ecosystems and biodiversity of the region's islands and atolls. Mounting resource utilization, habitat degradation, tourism and development, alien invasive species, pollution, and climate change, all negatively impact these already fragile systems. Some islands have already been afforded formal protection, with one site (Aldabra Special Reserve) listed under UNESCO World Heritage status. However, far more conservation effort is needed to ensure the preservation of these biodiversity hotspots through the additional proclamation of MPAs and through ensuring that those currently under formal protection are effectively managed.

The WIO coastal forests comprise small and fragmented patches, which are host to high biological diversity of global significance (see Chapter 17). They provide a diversity of ecosystem services directly and indirectly linked to the livelihoods of coastal communities. Hence, coastal forests are of significant environmental and socio-economic importance and critical for the long-term survival of the region's economy. The forests reduce soil erosion and mitigate potential harmful discharge into the coastal waters of the Indian Ocean that could lead to the degradation of nearshore marine habitats. Trends in coastal forest cover show a general decline characterized by fragmentation. Promoting the conservation of the coastal forests should be contextualized under a framework that involves a balance between the environment, society, development and conservation strategies. Additionally, forest conservation should be integrated into river basin and catchment management.

The high seas comprise ecosystems that support ecologically important functions and livelihoods and are critical migration routes that maintain biodiversity globally. Ocean connectivity is critical for the persistence of marine life and the vast benefits that derive from it. Regional scale connectivity patterns in the WIO demonstrate the potential of using oceanographic modelling to estimate functional connectivity among zones of maritime jurisdictions (see Chapter 18). Threats to connectivity are of a global nature and include unsustainable fisheries and uncontrolled shipping. Knowledge of large-scale connectivity patterns is essential for managing the oceans, both within and outside areas of national jurisdictions. Furthermore, studies on the feasibility, options, and scenarios for establishing MPAs in ABNJ are necessary. This may involve partnerships with global organizations, such as the IMO and the United Nations Convention on the Law of the Sea (UNCLOS), to facilitate identifying and designating particularly sensitive sea areas (PSSAs).

The need for marine protection measures

Most MPAs in the WIO predominantly protect critical coastal habitats, including mangroves, seagrass beds and coral reefs. However, most existing MPAs across the region are not managed effectively due primarily to inadequate capacity and poor enforcement and compliance, as described in detail for each of the region's countries in the *MPA Outlook*. National-level assessments suggest a disparity in implementation efforts of area-based management tools (ABMTs), with most countries indicating shortfalls. Despite the regional approach to conservation policy implementation under regional mechanisms, eg the UNEP-Nairobi Convention, international commitments require implementation at the national level and are reported as such. However, the continuous nature of biodiversity and the socio-ecological interdependence requires regionwide transboundary cooperation for biodiversity conservation to address representativeness, connectivity, and socio-economic benefits at the regional level. The evaluation of conservation policy outcomes needs to focus on the quality of conservation efforts from the perspective of socio-economic benefits, threats and the condition of biodiversity and associated habitats.

The evaluation of the WIO region's progress on a range of Aichi targets (and corresponding GBF targets) revealed overall considerable efforts in protected area coverage (see Chapter 20), but in terms of habitat quality and representation, climate change exposure, and the placement of protected areas relative to functionally connected areas, significant effort is still required at both country and regional levels. Towards implementing the post-2020 GBF, regional and national goals must be discussed and aligned with the GBF as part of developing the WIO roadmap for marine conservation and sustainable development. An outcome of this roadmap could be a Biodiversity Framework for the WIO that can provide strategies for regional implementation of the GBF. The *WIO MPA Outlook* and its sister *Critical Habitats Outlook* could provide important foundational references to inform this regional framework.

Establishing MPAs requires significant resources, technical expertise, and social capital among stakeholders, especially government institutions. Therefore, scaling up to form a regional Marine Protected Areas Network (MPAN) would require countries to formulate concrete plans to develop their own national MPANs. A national MPAN implies that the MPAs with a national jurisdiction are designed and located (and retrofitted if necessary) to maximize representativeness, connectivity, replication and redundancy. Moreover, national governments should coordinate with neighbouring states to create synergies, address boundary disputes, and align development priorities with increasing the effectiveness of the regional network. To establish a regional MPAN in the WIO, a systematic framework is necessary to organize and coordinate efforts (see Chapter 21). This systematic framework could be described as two major work streams. The first work stream sets targets for the individual WIO states to accomplish within their EEZ. In contrast, the second work stream requires concerted efforts by the WIO states to create a functional regional network. The Joint Management Area (JMA) between Mauritius and Seychelles provides an example.

A governance structure that allows transboundary arrangements should be in place to facilitate transboundary conservation. In terms of management, protecting 30 per cent of biodiversity features is one of the key targets of the GBF. OECMS in the region, which have evolved, such as LMMAs, will have a significant role in managing conservation areas in the region and globally. A multi-objective approach that achieves the targets while considering ecological and socio-economic benefits, among other uses, is necessary for considering management strategies. Maintaining biodiversity for the future requires that all habitats are adequately protected and represented in MPA networks. Their placement in space should be strategic to avoid or minimize a myriad of threats affecting critical habitats and ecosystems, including climate change and direct human threats. Replicating and optimizing conservation area selection for ecosystem persistence would also require protecting ecological processes such as connectivity among critical habitats. Other considerations are protecting threatened species, genetic diversity, and climate resilience. Socio-economic considerations are an integral part of selecting conservation areas in the region, from the perspective of resource use and minimizing threats. Conservation areas should be spatially configured to avoid present and future coastal developments to minimize conflict and promote compliance and effectiveness. Minimizing human pressure can be achieved by favouring protection in areas with lower human intervention. The scenario also includes socio-economic considerations to identify locations that can provide maximum socio-economic benefits for small-scale fisheries, and also includes promoting sustainable ecotourism, maintaining long-term sustainability of fish stocks, and prioritizing the protection of culturally significant areas.

To be effective, conservation of marine habitats and biodiversity in the WIO region should encompass different levels (see Chapter 22): (i) a common understanding of ocean governance; (ii) a sound scientific knowledge of ecological systems and their biodiversity and ecological processes, including connectivity among critical habitats; (iii) implementation of effective conservation measures, management and financial capacities; and (iv) consideration of a complex socio-ecological and adaptive approach, decreasing conflicts with local communities and inefficiencies in the management of protected areas.

José Paula
Editor

ACKNOWLEDGEMENTS

STILL OUTSTANDING

LIST OF CONTRIBUTORS

Janine ADAMS
Institute of Coastal and Marine Research, Nelson Mandela University
Gqeberha, South Africa

Vanessa ADAMS
School of Technology, Environments and Design, University of Tasmania
Hobart, Australia

Andrea ANGEL
BirdLife South Africa
ohannesburg, South Africa

Rose ANTIPA
Independent Consultant
Nairobi, Kenya

João AZEVEDO E SILVA
Faculty of Sciences, University of Lisbon
Lisbon, Portugal

Salomão BANDEIRA
Department of Biological Sciences, Eduardo Mondlane University
Maputo, Mozambique

Rhett BENNETT
Wildlife Conservation Society, Western Indian Ocean Sharks and Rays Conservation Program
Grahamstown, South Africa

Dave van BEUNINGEN
Wildlife Conservation Society, Western Indian Ocean Sharks and Rays Conservation Program
Cape Town, South Africa

Jared BOSIRE
Nairobi Convention, United Nation Environment Programme (UNEP)
Nairobi, Kenya

Peter CHADWICK
African Conservation Management Services
Cannon Rocks, South Africa

Mariana CRAVO
Faculty of Sciences of University of Lisbon
Lisbon, Portugal

Stephanie D'AGATA
School of Natural Sciences, faculty of Science and Engineering, Macquarie University
Sydney, Australia

Tammy DAVIES
BirdLife International, The David Attenborough Building
Cambridge, UK

Sean FENNESSY
Oceanographic Research Institute (ORI)
Durban, South Africa

Luisa FONTOURA
School of Natural Sciences, faculty of Science and Engineering, Macquarie University
Sydney, Australia

Julius FRANCIS
School of Aquatic Sciences and Fisheries (SoAF), University of Dar es Salaam
Zanzibar, Tanzania

Paul GACHERU
Nature Kenya, Museum Hill Off-Kipande Road
Nairobi, Kenya

Majambo GAMOYO
Coastal and Marine Resources Development
Mombasa, Kenya

Victoria GRAHAM
School of Natural Sciences, Faculty of Science and Engineering, Macquarie University
Sydney, Australia

Johan GROENEVELD
Oceanographic Research Institute (ORI)
Durban, South Africa

Sabrina GUDUFF
Global Marine and Polar Programme, IUCN
Gland, Switzerland

José GUERREIRO
MARE - Marine and Environmental Sciences Centre, Faculty of Sciences, University of Lisbon
Lisbon, Portugal

Issufo HALO
Department of Forestry, Fisheries and the Environment (DFFE), Oceans and Coasts Research Branch
Cape Town, South Africa

Vera HORIGUE
School of Natural Sciences, faculty of Science and Engineering, Macquarie University
Sydney, Australia

James KAIRO
Kenya Marine and Fisheries Research Institute
Mombasa, Kenya

Blandina LUGENDO
School of Aquatic Sciences and Fisheries Technology, University of Dar es Salaam
Dar es Salaam, Tanzania

Célia MACAMO
Department of Biological Sciences, Eduardo Mondlane University
Maputo, Mozambique

Joseph MAINA
School of Natural Sciences, faculty of Science and Engineering, Macquarie University
Sydney, Australia

Timothy McCLANAHAN
Wildlife Conservation Society, Marine Programs
New York, USA

Daudi MSANGAMENO
Institute of Marine Sciences, University of Dar es Salaam
Zanzibar, Tanzania

Nyawira MUTHIGA
Wildlife Conservation Society, Marine Programs
Mombasa, Kenya

Doris MUTTA
African Forest Forum
Nairobi, Kenya

Remy ODDENYO
Wildlife Conservation Society, Marine Programs
Mombasa, Kenya

José PAULA
MARE - Marine and Environmental Sciences Centre, Faculty of Sciences, University of Lisbon
Lisbon, Portugal

Rivo RABARISOA
Asity Madagascar
Antananarivo, Madagascar

Anusha RAJKARAN
Department of Biodiversity and Conservation Biology, University of the Western Cape
Bellville, South Africa

Matthew RICHMOND
RSK International Projects Group
Porto, Portugal

Michael SCHLEYER
Oceanographic Research Institute (ORI)
Durban, South Africa

Aurélie SPADONE
Ocean Team, Centre for Conservation Action, IUCN
Gland, Switzerland

Vikash TATAYAH
Mauritian Wildlife Foundation
Vacoas, Mauritius

Arthur TUDA
Western Indian Ocean Marine Science Association (WIOMSA)
Zanzibar, Tanzania

Dixon WARUINGE
Nairobi Convention, United Nation Environment Programme (UNEP)
Nairobi, Kenya

Chemuku WEKESA
Kenya Forestry Research Institute
Wundanyi, Kenya

CONTENTS

ABBREVIATIONS

AAC	Antarctic Circumpolar Current
AAIW	Antarctic Intermediate Waters
AAMW	Australasian Mediterranean Sea Waters
ABES	Africa Blue Economy Strategy
ABMTs	Area-Based Management Tools
ABNJ	Areas Beyond National Jurisdiction
ABW	Antarctic Bottom Water
AC	Agulhas Current
ACAP	Agreement on the Conservation of Albatrosses and Petrels
AEWA	Agreement on the Conservation of African-Eurasian Migratory Waterbirds
AMCEN	African Ministerial Conference on the Environment
APEI	Areas of Particular Environmental Interest
APMs	Associated Protective Measures
ARC	Agulhas Return Current
ASB	Arabian Sea Basin
ASB	Annual Severe Bleaching
ASCLME	Agulhas and Somali Currents Large Marine Ecosystems
ASHSW	Arabian Sea High-Salinity Water
ASLOW	Arabian Sea Low Oxygen Waters
ASS	Aragonite Saturation State
ASW	Arabian Sea Water
ATF	BirdLife's Albatross Task Force
AU	African Union
BBNJ	Marine Biodiversity of Areas Beyond National Jurisdiction
BBW	Bengal Bay Water
BIOT	British Indian Ocean Territory
BLM	Boundary Lenght Modifier
BMU	Beach Management Unit
BPA	Benthic Protected Areas
BYCAM	Bycatch Assessment and Mitigation in the WIO fisheries project
CBD	Convention on Biological Diversity
CBO	Community-Based Organisation
CCAMLR	Commission for the Conservation of Antarctic Marine Living Resources
CCSBT	Commission for the Conservation of Southern Bluefin Tuna
CDW	Circumpolar Deep Waters
CEMZA	Combined Exclusive Maritime Zone of Africa
CESM	Community Earth System Model
CFAs	Community Forest Associations
CIB	Central Indian Basin
CIR	Central Indian Ridge
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
CLCS	Commission on the Limits of the Continental Shelf
CLP	Chagos-Lacadive Plateau
CLR	Chagos-Laccadive Ridge
CLS	Collecte Localis Satellites
CMA	Community Managed Areas
CMS	Convention on the Conservation of Migratory Species of Wild Animals (also known as Bonn Convention)
CNES	National Centre for Spatial Studies
CO_2	Carbon dioxide

COP	Conference of the Parties
CORDIO	Coastal Oceans Research and Development in the Indian Ocean
CR	Critically Endangered
CRTF	Coral Reef Task Force
CSIR	Council for Scientific and Industrial Research
CSIRO	Commonwealth Scientific and Industrial Research Organisation
CZB	Crozet Basin
DD	Data Deficient
DEA	Department of Environment Affairs
DOALOS	Division for Ocean Affairs and the Law of the Sea
DSCC	Deep Sea Conservation Coalition
DVR	Davie Ridge
EACC	East African Coastal Current
EBSAs	Ecologically or Biologically Sensitive Area
EEZ	Exclusive Economic Zone
EFZ	Estuarine Functional Zone
EN	Endagared
ENSO	El-Niño Southern Oscillation
ESA	Endangered Species Act
FAO	Food and Agriculture Organisation
GBF	Global Biodiversity Framework
GBR	Great Barrier Reef
GDP	Gross Domestic Product
GEBCO	Global Earth Bathymetric Chart of the Oceans
GEF	Global Environment Facility
GLS	Geolocator
GOBI	Global Ocean Biodiversity Initiative
GOODS	Global Open Oceans and Deep Seabed
GPS	Global Positioning System
GRSE	Grand River South East
HP	Hydroelectrical Power
IAS	Invasive Alien Species
IBA	Important Bird Area
ICCAT	International Commission for the Conservation of Atlantic Tunas
ICM	Integrated Coastal Management
ICW	Indian Central Waters
ICZM	Integrated Coastal Zone Management
IDW	Indian Deep Water
IFC	International Finance Corporation
IMMA	Important Marine Mammals Areas
IMO	International Maritime Organization
IOB	Indian Ocean Basin-wide
IOC-UNESCO	Intergovernmental Oceanographic Commission of the United Nations Educational, Scientific and Cultural Organization
IOD	Indian Ocean Dipoles
IOTC	Indian Ocean Tuna Commission
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
ISA	International Seabed Authority
ITCZ	Intertropical Convergence Zone
ITF	Indonesian Throughflow
IUCN	International Union for Conservation of Nature
IUU	Illegal, Unregulated and Unreported
KBA	Key Biodiversity Area

KMFRI	Kenya Marine and Fisheries Research Institute
LBSA	Land Based Sources and Activities
LBSP	Land Based Sources of Pollution
LC	Least concern
LME	Large Marine Ecosystem
LMMA	Locally Managed Marine Area
LNG	Liquefied Natural Gas
m.a.s.l.	meters above sea level
MAB	Man and the Biosphere
MADT	Map of Absolute Dynamic Topography
MARPOL	International Convention for the Prevention of Pollution from Ships
MASMA	Marine Science for Management Programme
MASP	Mascarene Plateau
MDB	Madagascar Basin
MDR	Madagascar Ridge
MEA	Millennium Ecosystem Assessment
MEA	Management Effectiveness Assessment
MEPC	Marine Environment Protection Committee
MIHARI	Madagascar's Locally Managed Marine Area Network
MOR	Mid-Oceanic Ridges
MPA	Marine Protected Area
MSB	Mascarene Basin
MSC	Marine Stewardship Council
MSP	Marine Spatial Planning
MZB	Mozambique Basin
MZR	Mozambique Ridge
NBA	South African National Biodiversity Act
NDCs	Nationally Determined Contribution
NDVI	Normalized Difference Vegetation Index
NECC	North Equatorial Countercurrent
NEMC	Northeast Madagascar Current
NGO	Non-Governmental Organization
NIDW	Northern Indian Deep Water
NMC	North Monsoon Current
NT	Near Threatened
OA	Ocean Acidification
OECMs	Other Effective Area-based Conservation Measures
PGW	Persian Gulf Waters
PLD	Pelagic Larval Duration
PrepCom	Preparatory Committee
PSEPA	Primeiras and Segundas Environmental Protected Area
PSSA	Particularly Sensitive Sea Area
PTT	Platform Transmitter Terminals
RCoE	Rovuma Centre of Endemism
RFB	Regional Fisheries Bodie
RFMO	Regional Fisheries Management Organization
RLE	Red List of Ecosystems
ROMS	Regional Ocean Modelling Systems
RSCR-2015	Regional State of the Coast Report, 2015
RSW	Red Sea Water
SADSTIA	South African Deep Sea Trawl Industry Association
SAMW	SubAntartic Mode Water
SANBI	South African National Biodiversity Institute
SCOW	Scatterometer Climatology of Ocean Winds

SDG	Sustainable Development Goal
SEAFO	South East Atlantic Fisheries Organization
SEC	South Equatorial Current
SECC	South Equatorial Countercurrent
SEIR	South-East Indian Ridge
SEISAMW	South-East Indian SubAntarctic Mode Water
SEMC	South East Madagascar Current
SGD	Submarine Groundwater Discharge
SICC	South Indian Ocean Countercurrent
SIDS	Small Island Developing States
SIDW	Southern Indian Deep Water
SIODFA	South Indian Ocean Deepwater Fisheries Association
SIODFA	Southern Indian Ocean Deep Sea Fishers Association
SIOFA	South Indian Ocean Fisheries Agreement
SLA	Sea Level Anomalies
SMB	Somali Basin
SPAMI	Specially Protected Area of Mediterranean Importance
SSA	Sargasso Sea Alliance
SSH	Sea Surface Height
SSP	Shared Socioeconomic Pathway
SST	Sea Surface Temperature
STSW	Subtropical Surface Water
SWIOFP	South West Indian Ocean Fisheries Project
SWIR	Southwest Indian Ocean Ridge
SWIR	Southwest Indian Ridge
SWMC	Southwest Monsoon Current
TBCA	Transboundary Conservation Area
TSW	Tropical Surface Water
UN	United Nations
UNCLOS	United Nations Convention on the Law of the Sea
UNEA	United Nations Environment Assembly
UNEP	United Nations Environment Programme
UNEP-WCMC	UN Environment World Conservation Monitoring Centre
UNESCO	United Nations Educational, Scientific and Cultural Organization
UNFCCC	United Nations Framework Convention on Climate Change
UNGA	United Nations General Assembly
VME	Vulnerable Marine Ecosystem
VU	Vulnerable
WDPA	World Database on Protected Areas
WHC	World Heritage Convention
WIO	Western Indian Ocean
WIOMSA	Western Indian Ocean Marine Science Association
WIO-MTTF	Western Indian Ocean Marine Turtle Task Force
WIO-SAP	Strategic Action Programme for the protection of the Western Indian Ocean
WMMPR	Watamu/Malindi Marine Parks and Reserve
WOA	World Ocean Atlas Database
WSA	Watershed Approach
WTBs	Wet Tropics Basins
WWF	World Wildlife Fund

PART I: PURPOSE AND APPROACH

José Paula

PURPOSE

The Nairobi Convention for the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region is a partnership between governments, civil society and the private sector, working towards a prosperous Western Indian Ocean (WIO) region with healthy rivers, coasts and oceans. The Convention addresses this vision by providing a platform for regional cooperation, coordination, and collaborative actions, that enable the Contracting Parties to benefit from the critical resources and expertise from a wide range of stakeholders and interest groups. Ultimately it contributes to solving common problems across the WIO coastal and marine environments (UNEP/Nairobi Convention, 2010).

The Nairobi Convention is part of the United Nations Environment Programme (UNEP) Regional Seas Programme, that was first signed in 1985. It entered into force in 1996, aiming to promote the sustainable management and use of the marine and coastal environment to address the accelerating widespread degradation of the world's marine and coastal areas. The Nairobi Convention promotes the participation of countries that share the WIO for the protection of their shared marine environment. The Contracting Parties (Comoros, France, Kenya, Madagascar, Republic of Mauritius, Mozambique, Seychelles, Somalia, the Republic of South Africa and the United Republic of Tanzania) to the Convention are part of more than 143 countries that participate in 18 Regional Seas initiatives (UNEP/Nairobi Convention, 2010).

The Work Programme of the Nairobi Convention focuses on priorities of the WIO region governments. It is implemented through various collaborative projects, which have significant impact on marine related activities in the WIO region, such as capacity building, management, coordination, and legal aspects, while maintaining momentum for the implementation of the Nairobi Convention and its protocols. Additional support is also provided to the Contracting Parties in delivering towards relevant global commitments such as those of the 2030 Sustainable Development Agenda.

The Project funded by the Global Environment Facility (GEF), "*Implementation of the Strategic Action Programme for the protection of the Western Indian Ocean from land-based sources and activities (WIO-SAP)*", from December 2016, permits the Nairobi Convention to support the Contracting Parties towards the delivery of the United Nations 2030 Sustainable Development Agenda in general and in particular the Sustainable Development Goal (SDG) 14 "*Life below Water*" with special focus on Targets 14.2 and 14.5.

Target 14.2 calls for the sustainable management and protection of marine and coastal ecosystems to avoid significant adverse impacts, including by strengthening their resilience, and take action for their restoration, to achieve healthy and productive oceans by 2020, while Target 14.5 stated that by 2020, countries should conserve at least 10 per cent of coastal and marine areas, consistent with national and international law and based on best available scientific information (United Nations, 2015). Unlike most SDG targets, which have a target year of 2030, this indicator was set to be achieved by 2020. However, despite some countries having met this goal, the majority, including most countries within the WIO region, are still far from achieving the 10 per cent of conservation (UNEP/Nairobi Convention, WIOMSA, 2021).

The WIO-SAP Project addresses priority conservation issues within the WIO region as agreed by Contracting Parties through the following components:

Component A:
Sustainable management of critical habitats
focuses on the protection, restoration and management of critical coastal habitats and ecosystems, recognizing the enormous value of healthy critical coastal and marine habitats for the future well-being of people in the WIO region.

Component B:
Improved water quality
focuses on the need for the WIO region's water quality to attain international standards by the year 2035.

Component C:
Sustainable management of river flows
aims at promoting wise management of river basins in the region through implementation of a suite of activities aimed at building the capacity for environmental flows assessment and application.

Component D:
Governance and regional collaboration
focus on strengthening governance and awareness in the WIO region with a view to facilitating sustainable management of critical coastal ecosystems and habitats.

The production of the *Critical Habitats Outlook* relates to Component A, designed to respond to a number of SDG 14 and related Aichi Biodiversity targets. The Component promotes various approaches and tools, including: ecosystem restoration, management planning and implementation, marine spatial planning, ecosystem valuation and development of various supporting guidelines.

The regional *Critical Habitats Outlook* is one of the main outputs of this initiative, and together with the regional *MPA Outlook* is intended to inform policy-making with regard to enhanced coastal and marine conservation in the region. These two publications feed into a third publication on recommendations for strengthening marine conservation in the WIO region, aimed at supporting contracting parties to meet their obligations under SDG Targets 14.2 and 14.5 and Aichi Target 11, besides other SDGs.

The Nairobi Convention and the Western Indian Ocean Marine Science Association (WIOMSA), have been engaged in close collaboration for many years. This partnership was the scientific and technical basis to produce both *Critical Habitats* and *MPA Outlooks*. WIOMSA provides a solid connection throughout the scientific, academic, and technical communities across the WIO region, which allowed it to play a pivotal coordination role in the process leading to the preparation of these two *Outlook* volumes.

WIOMSA was thus involved from the beginning of the process and contributed financially to the initiative as it is in line with the Marine Science for Management (MASMA) Programme, whose main goal is "*to establish and operationalise a regional science to policy platform by 2022 that generates knowledge, builds capacity, mobilises resources, and shares scientific and policy-relevant knowledge to assist the WIO region to deliver on the 2030 Agenda for oceans, islands and coasts, and climate change*."

Under this Programme, WIOMSA is working with the Nairobi Convention and national/regional partners to set up a baseline for at least four SDG 14 targets (including 14.2 and 14.5) and track progress over time.

The development of both *Critical Habitats* and *MPA Outlooks* drew strongly on an earlier partnership project between the Nairobi Convention and WIOMSA, namely the production, in 2015, of the *Regional State of the Coast Report: Western Indian Ocean* (UNEP, Nairobi Convention and WIOMSA, 2015).

The general purpose of the *Critical Habitats Outlook* is to evaluate the most important and critical marine and coastal habitats of the WIO region, and in particular to:

- describe the most important and critical habitats of the WIO and the relevance of their associated biodiversity;
- review the socio-economic usage and dependence of coastal human communities on the WIO marine habitats;
- highlight gaps regarding the scientific knowledge of WIO marine habitats;
- review the current levels of protection of the WIO marine habitats and identify areas and opportunities for increasing protection;
- develop alternative scenarios for the future protection of the marine habitats in the WIO.

This *Critical Habitats Outlook* will contribute to a larger process involving the *MPA Outlook* for the region, and the final *Outlook* volume on recommendations for the available future strategic options, including achieving the targets based on the identification of critical habitats that require protection. The link between the *Critical Habitats Outlook* and the *MPA Outlook* is that it advances knowledge on critical habitats and evaluates gaps that need to be addressed to improve conservation throughout the region. This included the extensive offshore and deep-sea areas that are not well represented in current conservation schemes.

The *Critical Habitats Outlook* further intends to promote conservation of the marine and coastal habitats throughout the region, encouraging the scientific community, stakeholders and decision-makers to engage in the shared responsibility of sustainable development for the benefit of human populations throughout the region.

STRUCTURE

The *Critical Habitats Outlook* is organized in four parts, as follows:

Part I

Part I (Chapter 1) contextualizes the volume and intends firstly to inform on its purpose and structure. It also presents the methodological approaches used, namely the process of developing the *Critical Habitats Outlook* and assembling the data for the descriptions and evaluation. This part finally discusses the challenges and limitations of accessing data related to coastal and marine environments in the WIO, thus enumerating the limitations of the volume.

Part II

Part II presents the broader regional and wider contexts that account for the regional characteristics of the WIO marine environments and their main habitat types. Firstly, Chapter 2 addresses how the WIO is compartmented with respect to the bioregional classification schemes, from ecoregions to large provinces, from coastal to offshore areas, and the relationship with major geographical and environmental gradients.

The main topographic and oceanographic features are summarized in Chapter 3, as constituting the basis for the distribution of habitat types, their productivity and regional specificity. Chapter 4 discusses regional land-based interactions between the coastal and marine environment. Finally, Chapter 5 presents the classification of habitats and their health status as per the IUCN general categories, as the basic criteria used throughout the volume.

Part III

Part III constitutes the main core of the volume by presenting the main habitats of the WIO, in particular those that may be considered as critical in terms of biodiversity and ecological functioning, and as such, provide the most ecosystem services to human populations. The chapters include descriptions of habitats, but also relevant taxa such as marine birds and threatened species, as well as morphological features that are not obviously included within specific habitats but constitute environments of great importance for biodiversity, such as seamounts or small islands.

In detail, this part describes rocky and sandy shores (Chapter 6), mangrove forests (Chapter 7), seagrass meadows (Chapter 8), temperate salt marshes (Chapter 9), coral reefs (Chapter 10), estuarine zones (Chapter 11), offshore and deep-sea (Chapter 12), threatened species throughout the region (Chapter 13), marine birds (Chapter 14), seamounts and ridges (Chapter 15), small islands and atolls (Chapter 16), coastal forests (Chapter 17), and marine and coastal connectivity (Chapter 18). Chapter 19 then highlights the most relevant features of the habitat chapters.

Part IV

Part IV summarizes and the *Critical Habitats Outlook*, by addressing global cross-cutting issues relevant for effective protection of critical habitats in the WIO region. Chapter 20 describes past achievements and lessons learned from past actions and contextualizes the possible roadmap for biodiversity conservation in the WIO. Chapter 21 draws a scenario for effective conservation of marine biodiversity in the WIO, aiming at ensuring protection of ecological systems and fisheries. Finally, Chapter 22 summarizes the volume and highlights the main priority areas for ongoing and future marine conservation in the region.

METHODOLOGY

The process for the development of the *Critical Habitats Outlook* was initiated in close articulation with the *MPA Outlook* in a scoping workshop in Victoria, Mahé, Seychelles in June 2017, and specifically through a preparatory workshop held at Mombasa, Kenya, in June 2018. The definitions and categorizations of key concepts agreed at the *MPA Outlook* Scoping Workshop in the Seychelles were used in the *Critical Habitats Outlook* (UNEP/Nairobi Convention and WIOMSA, 2021). Critical habitats are defined according to the IUCN Key Biodiversity Areas (IUCN, 2015):

"*Critical habitats provide important functions (eg species refugia, commercially important species, and uniqueness); they have a representativity of species, processes, functions; and they have connectivity both within the ecosystem and externally.*"

Chapter 5 in this volume further contextualizes the concepts of critical habitat according to the more rigorous definition of the International Finance Corporation (IFC) of the World Bank:

"Critical habitats are areas with high biodiversity value, including a) habitat of significant importance to Critically Endangered and/or Endangered species; b) habitat of significant importance to endemic and/or restricted-range species; c) habitat supporting globally significant concentrations of migratory species and/or congregatory species; d) highly threatened and/or unique ecosystems; and/or e) areas associated with key evolutionary processes."

The habitat assessment followed the broad categories of IUCN (Bland et al., 2017), and the broad habitat classifications were used as in the *Regional State of the Coast Report* (UNEP/Nairobi Convention and WIOMSA, 2015).

Preparation of the *Critical Habitats Outlook* was initiated with the selection of the editor and selection of the authors. The Nairobi Convention Secretariat and WIOMSA issued a "Call for Expression of Interest" to be authors of the *Critical Habitats Outlook*. Based on the CVs of the candidate authors, their publishing and reporting record, and their availability to fully engage with the process, the lead authors were selected to develop the chapters. Lead authors then proposed to engage with a few co-authors where relevant for specific expertise and timely completion of the chapters.

Following the appointment of the editor and authors for the *Critical Habitats Outlook*, an Authors' Workshop was held in Mauritius, in 8–9 October 2018, where most authors presented preliminary outlines of their chapters based on the framework developed at the scoping workshop, refined and augmented by guidelines and thorough discussions with WIOMSA and the editor. Possible topics for case studies and potential authors for these were also identified.

Additionally, the meeting set the basic layout for the contributions, both in terms of their type contents and structure, as well as detail, size and writing standards. The different parts of the *Outlook* required specific approaches but the core chapters on critical habitats were guided to bear similar contents, such as background, importance, threats, existing protection, priority options for conservation, and recommendations, if applicable.

The draft chapters entered then a process of blind peer-review, with at least two reviewers for each contribution, as in standard scientific editorial processes. The authors revised their manuscripts according to the comments from reviewers, and editorial check was made at draft, review, and revision processes by the editor. Final accepted chapter versions were then copyedited for consistency and standardization of language, and a final editorial check was made prior to the layout phase. Maps and graphic elements were standardized for volume consistency.

LIMITATIONS

The *Critical Habitats Outlook* is a synthesis of the complex marine environments and their distribution in the wide geographical span of the WIO region. At such scale, it is impossible to avoid limitations and gaps that constrain the information gathered.

Limited information on geographical areas

There is an asymmetry of the information available in different countries and regions. While there is more information in more developed areas and in the vicinity of the major coastal urban centres, other more remote areas are poorly studied and may induce bias on the overall analysis. The large coastal and oceanic areas of Somalia have little information available due to the local security condtions, and constitute a shadow on the existing knowledge concerning the status of marine environments in the WIO.

Gaps of knowledge

Much has been achieved during the past decades regarding our knowledge of the marine environments and critical habitats of the WIO region. However, while many gaps exist in the vast geographical regions of the WIO, offshore and deep-sea habitats are very poorly known, and very limited information is currently available. A significant part of the information has been generated through environmental assessments related to oil and gas exploitation, which is partially not publicly available.

Inaccessibility of information

Much regional scientific information is also in the form of grey literature, such as unpublished theses, institutional reports, and other non-publicly available formats, despite the rapid development of science and its dissemination in the WIO region during recent years.

These limitations and gaps constrain our understanding of regional habitats and the ecological processes that are established within and among them and the diverse regional marine biodiversity they support. Nevertheless, this volume provided an opportunity to update and complete the analysis made for the *Regional State of the Coast Report* (UNEP/Nairobi Convention and WIOMSA, 2015), and complements the synthesis on marine conservation provided by the *MPA Outlook* (UNEP/Nairobi Convention and WIOMSA, 2021).

REFERENCES

Bland, L.M., Keith, D.A., Miller, R.M., Murray, N.J. and Rodríguez, J.P. (eds.) (2017). Guidelines for the application of IUCN Red List of Ecosystems Categories and Criteria, Version 1.1. IUCN, 99 pp. Gland, Switzerland.

IUCN (2015). *IUCN Standard for the Identification of KeyBiodiversity Areas*, Version 1.0. First edition. IUCN. Gland, Switzerland.

UNEP/Nairobi Convention (2010). A*mended Nairobi Convention for the Protection, Management and Development of the Marine and Coastal Environment of the Western Indian Ocean*. Nairobi, Kenya.

UNEP/Nairobi Convention and WIOMSA (2021). *Western Indian Ocean Marine Protected Areas Outlook: Towards achievement of the Global Biodiversity Framework Targets*. UNEP and WIOMSA, Nairobi, Kenya, 298 pp.

UNEP/Nairobi Convention and WIOMSA (2015). *The Regional State of the Coast Report: Western Indian Ocean*. UNEP and WIOMSA, Nairobi, Kenya, 546 pp.

United Nations (2015). [Website] (available at https://sustainabledevelopment.un.org/). Accessed 20 September 2022.

PART 2: CONTEXT OF THIS OUTLOOK

CRITICAL HABITATS

GLOBAL AND REGIONAL CONTEXT

José Paula, João Azevedo e Silva and Mariana Cravo

Artisanal fishing boat, Zanzibar. © Rahim Saggaf

BACKGROUND

The geographic region of the Western Indian Ocean (WIO) comprises the African continental coast from eastern Somalia (excluding Socotra Island) to the tip of South Africa, and east to the Chagos Archipelago and Mascarene Plateau. The limits are thus set by the consistency of pure geographical units but also modulated by political boundaries (Fig. 1) mindful that several disputed territories still exist in the region. The WIO region includes the territorial waters and exclusive economic zones (EEZs) of 11 regional countries and a large area of open sea.

The WIO region spans a wide latitudinal range. In the north, the Somalia region is influenced by the strong monsoon regime of the Arabian Sea in the northern Indian Ocean, which pulses seasonally driven by winds and reversing currents. This system triggers coastal upwelling and associated biological productivity. In the southern temperate regime of the tip of South Africa, the south-flowing Agulhas current transports intertropical energy influencing local climate and marine habitats, before diverging eastward.

The WIO region encompasses tropical and subtropical regions of diverse nature, rich stretches of coast along the mainland countries of Somalia, Kenya, Tanzania, Mozambique and South Africa, and vast oceanic areas surrounding the island states of Madagascar, Seychelles, Comoros, Mauritius and the French Territories.

The large latitudinal span of the WIO region creates a basal gradient of environmental conditions, which include a wide inter-tropical subregion from Somalia to southern Mozambique, and a temperate subregion in southern Africa. It further presents a long continental coast subjected to the input of significant water basins of the eastern African continent (see Chapter 11), and a complex system of islands, from large islands such as Madagascar to numerous smaller islands and seamounts (see Chapter 15).

Major climatic and oceanographic systems further create environmental compartmentalization and dynamic processes (see Chapter 3) that modulate biodiversity and productivity across the region. Major current systems significantly influence ecological processes and biodiversity, such as the Agulhas current flowing to the southern ocean, and the equatorial westwards current splitting into north and southern branches near the African coast (see Chapter 12).

Major coastal tropical habitats flourish in the region, such as mangroves (see Chapter 7), seagrasses (see Chapter 8) and coral reefs (see Chapter 10). These habitats, in particular in coastal zones with significant runoff, intermingle and form complex seascape mosaics that act as integrated ecological units. To the south, the mangroves progressively become residual and give place to temperate habitats such as salt marshes (see Chapter 9). Throughout the region, rocky outcrops and sedimentary formations create a diversity of coastal configurations, including extensive sand dune systems on the southern area between Mozambique and eastern South Africa (see Chapter 6).

The biological diversity of the region is high and includes charismatic taxa of conservation interest, such as marine mammals, birds and other organisms listed on the IUCN Red List (see Chapter 13). Many populations of marine birds rely on the extensive network of small islands for their nesting activities (see Chapters 14 and 16).

The content of this chapter contextualizes the geographical span of the WIO region and provides a brief overview of the ecological compartmentalization according to the most relevant sources. The rationale of the approaches for the different classification schemes is presented, and these are described for the geographical context of the WIO. For the coastal environments, the focus was on the WWF Marine Ecoregions (Spalding et al., 2007), the OneEarth Bioregions 2020 (One Earth, 2020), and the Large Marine Ecosystems (LMEs) (LEARN, 2017). Other classifications provide divisions for the oceanic areas, such as the Longhurst Provinces (Longhurst, 2007) for the pelagic ocean, the Bioregions of the Indian Ocean (Dunstand et al., 2020) that address both the pelagic and the benthic zones, and the Global Open Oceans and Deep Seabed (GOODS) biogeographic classification (UNESCO, 2009).

Albeit with different conceptual and methodological approaches, the classification schemes mentioned above provide a system of ocean division that relies on an underlying concept of 'bioregion'.

CONCEPT OF BIOREGION

Bioregions are areas that contain ecologically distinct content, being relatively homogeneous within, and distinct compared to other bioregions (Hill et al., 2020). This biological and physical partitioning of the ocean geographic space is based on the spatial distribution of

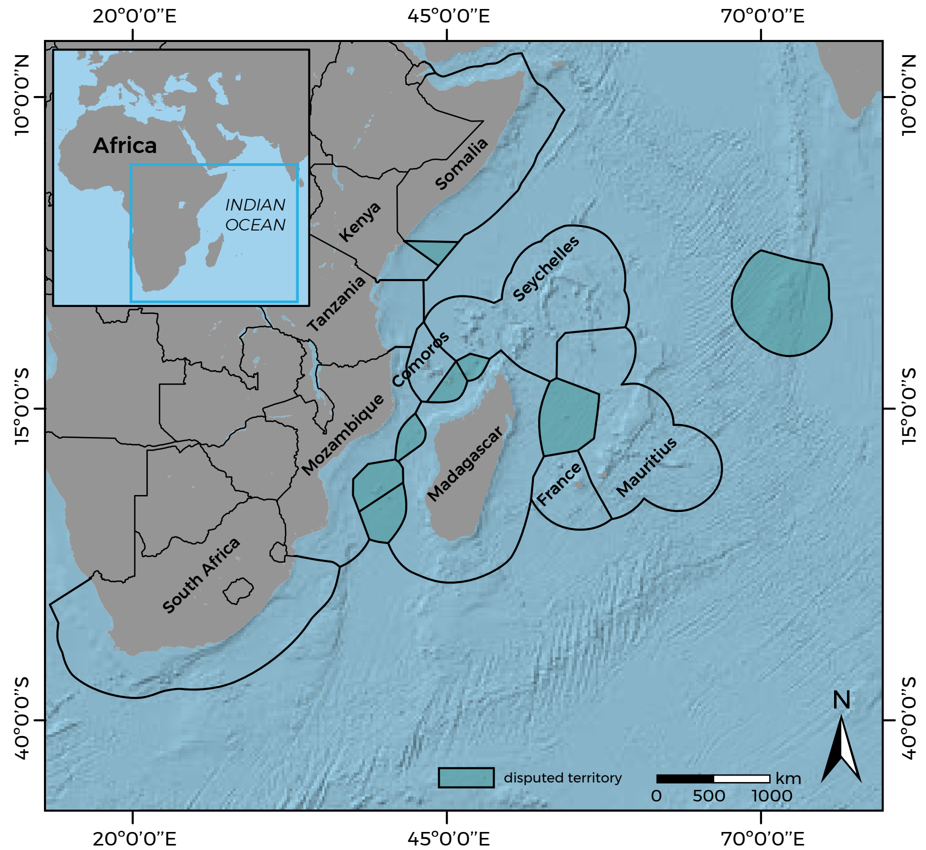


Figure 1: Geographical context of the WIO region. Latitudinally the region spans from the tip of the African continent at Somalia (excluding Socotra Island) to the southern tip of the Cape of Good Hope in South Africa. Longitudinally it includes the Indian Ocean to the EEZ of Mauritius including the Chagos Archipelago. The lines correspond to EEZ limits. Those under dispute are shown darker. Note that the EEZs from countries not part of the WIO region are excluded from the map.

multiple species, communities, ecosystems, or other biological characteristics (Woolley et al., 2019). Marine biogeography is also based on the concept that organisms are associated with specific regions separated from each other by physical and biological barriers (Porter et al., 2013). Therefore, a bioregion is a geographical area defined not by political boundaries but by ecological systems (One Earth, 2020).

Bioregionalization of the oceans is a process that aims to divide oceanic areas into distinct spatial regions based on abiotic and biotic information (Godet et al., 2020; Spalding et al., 2007). Mapping the ocean into bioregions has long been an important tool for evolutionary studies on conservation planning (Spalding et al., 2007). However, these methodological strategies are more advanced in the terrestrial environment. Some publications that have attempted to use biogeographic regionalization in global marine conservation planning are qualitative and have highlighted the lack of an adequate global classification (Spalding et al., 2007). The growing knowledge of the last few decades has provided essential means to understand how oceanic and ecological processes may be best partitioned, as Wendt et al (2018) describe:

"*Classifications typically assess spatial patterns in generalized environmental characteristics such as structural features of habitat, ecological function and processes, and physical features such as water characteristics and seabed topography to select relatively homogeneous regions with respect to habitat and associated biological community characteristics. These are refined with direct knowledge or inferred understanding of the patterns of species and communities, driven by processes of dispersal, isolation and evolution. Using such data and, often, literature reviews, experts aim to ensure, also, that biologically unique features, found in distinct basins and water bodies, are also captured in the classification.*"

The awareness and necessity of management of the ocean resources beyond EEZs has been increasing. In part this is due to the fact that bioregionalization act as natural frameworks used by regional fisheries management organizations (Spalding et al., 2007; Hill et al., 2020).

COASTAL CLASSIFICATIONS

The Marine Ecoregions of the World (MEOW)

Spalding et al. (2007) present the global biogeographic system for coastal and shelf areas known as the Marine Ecoregions of the World (MEOW). It provides a comprehensive and fine scale coverage based solely on biodiversity criteria (UNESCO, 2009). This classification has consistent spatial scales and incorporates the full spectrum of habitats found across shelves, focusing on coastal and shelf waters, combining benthic and shelf pelagic (neritic) biotas, and does not consider the open ocean and deep benthic environments (Spalding et al., 2007; UNESCO, 2009; WWF, 2007). MEOW represents broad-scale patterns of species and communities in the ocean and was designed as a tool for planning conservation across a range of scales and assessing conservation efforts and gaps worldwide (WWF, 2007). This project was led by WWF and The Nature Conservancy, with broad input from a working group representing key NGOs, academic and intergovernmental conservation partners (WWF, 2007).

The MEOW system has its limits extending up to 370 kilometres (200 nautical miles) offshore (or to the 200 m isobath, where this lies further offshore), but the principal focus of this classification was the benthos above 200 m and the overlying water column. The authors mention "that beyond 200 m, other biogeographic patterns will increasingly predominate, altering or hiding the patterns represented by the system proposed" (Spalding et al., 2007).

The MEOW biogeographic system is a hierarchical and nested system based on taxonomic configurations, influenced by evolutionary history, patterns of dispersal, and isolation. As described by Spalding et al. (2007), the nested system of MEOW is divided into three categories of bioregions:

- **Realms** (12): the system's largest spatial units. Very large regions of coastal, benthic, or pelagic ocean across which biotas are internally coherent at higher taxonomic levels, as a result of a shared and unique evolutionary history.
- **Provinces** (62): nested within the realms, these are large areas defined by the presence of distinct biotas that have at least some cohesion over evolutionary time frames.
- **Ecoregions** (232): the smallest-scale units in the MEOW system, being areas of relatively homogeneous species composition, clearly distinct from adjacent systems.

In many areas, the scale at which provinces may be conceived is similar to that of the detailed spatial units used in global systems such as Briggs's provinces, Longhurst's biogeochemical provinces, and LMEs (Spalding et al., 2007).

The WIO region encompasses two biogeographic realms (Western Indo-Pacific and Temperate Southern Africa), four provinces (Somali/Arabian, Western Indian Ocean, Central Indian Ocean Islands and Agulhas), and 13 ecoregions (Central Somali Coast, Northern Monsoon Current Coast, East African Coral Coast, Seychelles, Cargados

Seaweed farming at Zanzibar Island. © José Paula

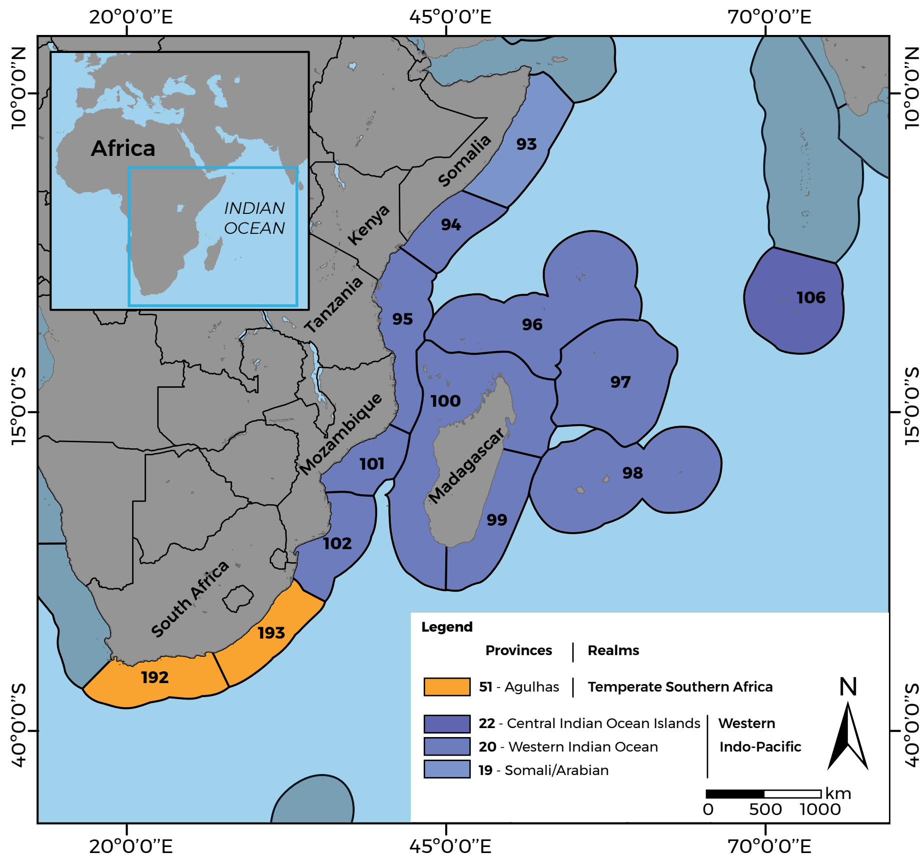


Figure 2: The MEOW classification on the WIO region.

Carajos/Tromelin Island, Mascarene Islands, south-east Madagascar, Western and Northern Madagascar, Bight of Sofala/Swamp Coast, Delagoa, Chagos, Agulhas bank, and Natal), as shown in Fig. 2.

The MEOW classification is considered to be an important reference for marine conservation planning due to its practicality, level of detail and for adopting a nested hierarchy that utilize systems that are already widely adopted (eg the Nature Conservancy's system) and fit closely within large-scale systems and other regional systems (Spalding et al., 2007). Two major international conservation agencies (the Nature Conservancy and WWF) use this system as the basis for marine conservation planning on coastal and shelf areas.

Obura (2012) demonstrated that, when considering the biogeography and connectivity of reef-building corals in the Western Indian Ocean MEOW province, a different arrangement of the ecoregions therein contained could be considered. Obura (2012) proposes to extend the WIO province to the north to include the Central Somali ecoregion and to the east to include the Chagos ecoregion. An additional proposal was to rearrange the internal WIO ecoregions to mirror the diversity and distribution of coral communities.

OneEarth Bioregions 2020

The nested biogeographical system Bioregions 2020 builds upon 844 terrestrial ecoregion divisions (Dinerstein et al. 2017) to delineate 185 discrete bioregions organized within the world's major biogeographical realms (One Earth, 2020). Similar to Spalding et al. (2007), the system of Bioregions 2020 is a hierarchical and nested system, where subrealms cluster bioregions into a more familiar geographical taxonomy (One Earth, 2020). Hierarchically, the system is divided into realms, subrealms, bioregions and finally ecoregions. Although this bioregional framework is built upon terrestrial ecoregion divisions, it integrates all three types of ecoregions – terrestrial, freshwater, and marine – into a cohesive system (One Earth, 2020).

The extension of these bioregions derives from the extension of bioregions with coastal borders to the corresponding country's Exclusive Economic Zone (EEZ) boundaries. While EEZs are administrative, not biological boundaries, they do provide an ecosystem constraint as most fishing and industrial activities occur within the EEZs. The One Earth Bioregions 2020 system in some cases resorts to the marine provinces of the MEOW classification to define a bioregion (One Earth, 2020).

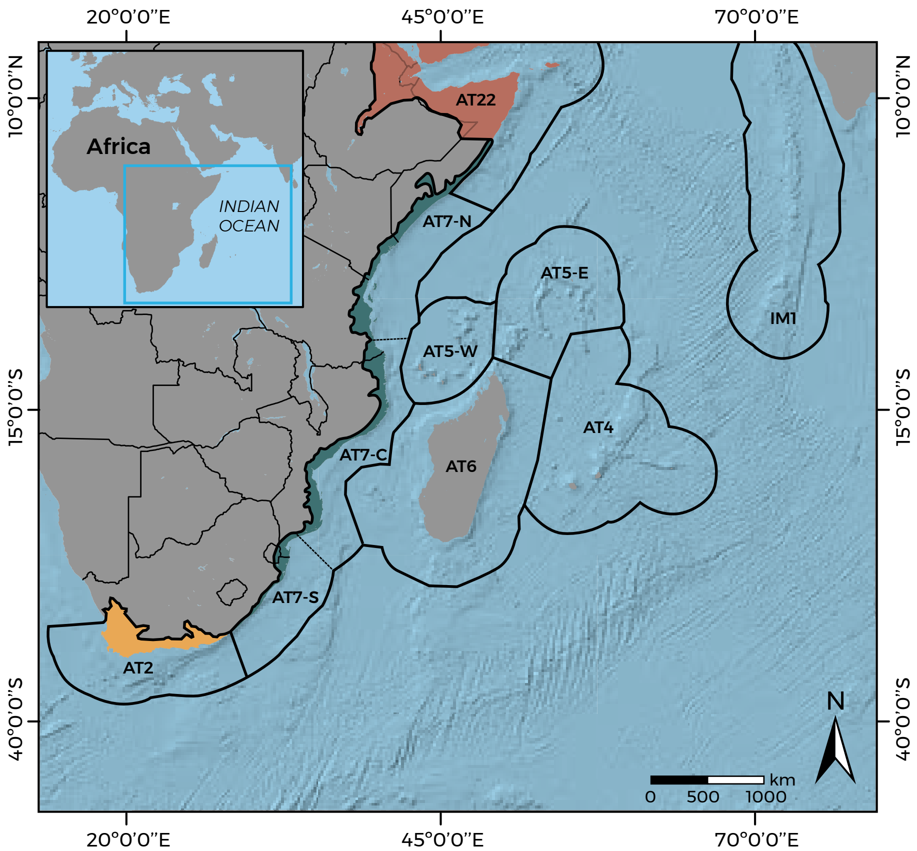


Figure 3: OneEarth Bioregions 2020 of the WIO region. AT4 – Mascarene Tropical Forest Islands, AT5 – Seychelles and Comoros Tropical Islands, AT6 – Madagascar Island, AT7 – East African Coastal Forests, AT22 – South Red Sea and Gulf of Alden Coastal Drylands, AT2 – South African Cape Shrublands Mountain Forests, IM1 – Central Indian Ocean Islands . Adapted from One Earth (2020).

The 185 bioregions that comprise the Bioregions 2020 system are organized in line with the world's major biogeographical realms, that roughly correspond to the major continental masses, further subdivided to coincide with climatic zones (One Earth, 2020).

These realm divisions (14 in total) provide the overarching content framework for the One Earth website[1] (One Earth, 2020). Of these, two realms, four sub-realms, and eight bioregions are applied to the WIO (see Fig. 3).

The criteria used in the definition of the 185 bioregions across all realms include large-scale geological structures or climatological zones and biome types (One Earth, 2020). The marine bioregions are demarcated using the EEZ boundary lines, as these waters are often heavily fished, or experience impacts from activity on adjacent land areas. In some cases, marine provinces are used to articulate a bioregion (Spalding et al. 2007).

Large Marine Ecosystems

The system of Large Marine Ecosystems (LMEs) represents an expert-derived system, with considerable input from fisheries scientist Ken Sherman (eg Sherman and Alexander, 1989; Hempel and Sherman, 2003; Sherman et al., 2005). LMEs are widely adopted by international organizations, and focus on productivity and oceanographic processes, but omit substantial areas of islands and oceanic areas of the Indian Ocean.

The system includes large areas (200 000 km^2 or greater) that are characterized by distinct bathymetry, hydrography, productivity, and trophic interactions, and represent 95 percent (64 LMEs) of the world's fish catch annually (Duda and Sherman, 2002). It represents a concept intended to promote solutions for transboundary management issues (fisheries, pollution, habitat restoration, productivity, socio-economics, and governance) in view of widespread degradation of natural habitats[2].

[1] www.oneearth.org/bioregions-2020/

[2] www.lme.noaa.gov/Portal/

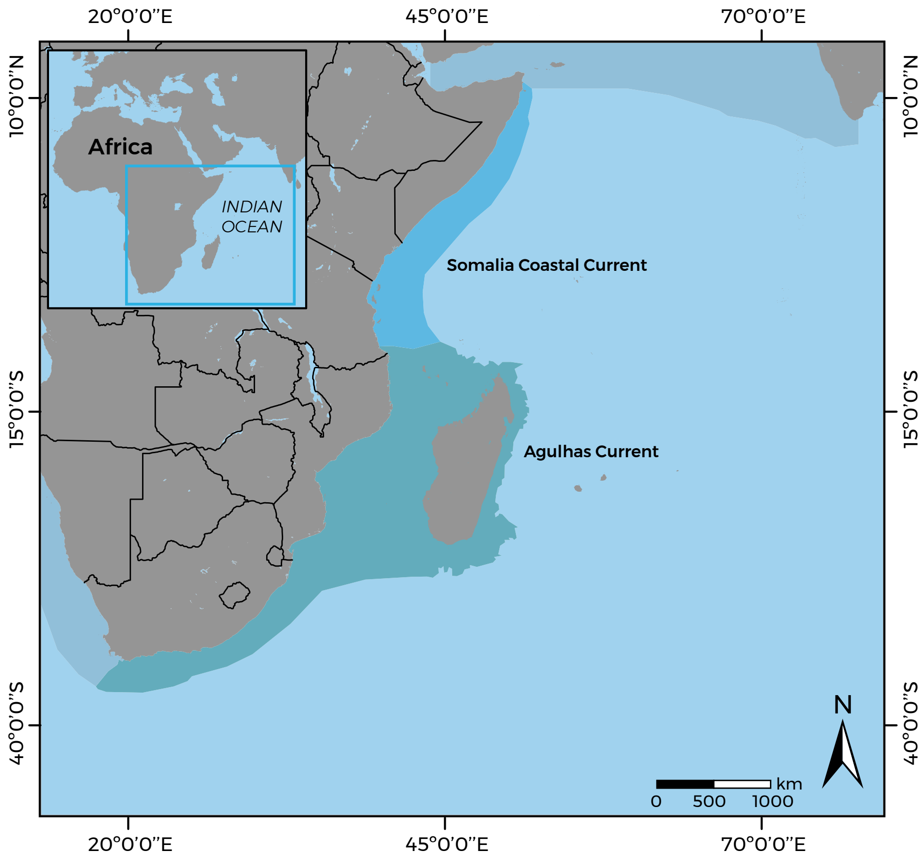


Figure 4: LMEs of the WIO region. The system mainly includes the coastal zone of the Eastern African continent and extends to Madagascar. It excluded most of the waters of WIO island states, such as the Seychelles, Mascanere or Chagos archipelagos. Adapted from the ILearn platform (https://iwlearn.net/iw-projects/basins/lmes/).

The coverage of the LMEs extends from river basins and estuaries to the seaward boundaries of continental shelves and the outer margins of the major current systems (UNESCO, 2009). However, the oceanic pelagic and deep-sea areas beyond national jurisdiction are not included. Many island systems are also excluded, such as the Chagos and Mascarene archipelagos in the WIO region (Fig. 4). The boundaries of LMEs have been set by a multicriteria approach including biological and geopolitical considerations.

According to LEARN (2017) the LME system can constitute a regional framework and science-informed process that contributes to the 2030 Agenda for Sustainable Development, in particular Sustainable Development Goal 14. The Global Environmental Facility LME:LEARN project was developed as a base for LME activities, and was implemented by the United Nations Development Programme (UNEP) and managed by the Intergovernmental Oceanographic Commission of the United Nations Educational, Scientific and Cultural Organization (IOC-UNESCO and UNEP, 2016).

OCEANIC CLASSIFICATIONS

Ecological geography of the sea

Longhurst (2007) presents the ocean divided into four biomes which are subdivided into biogeochemical provinces. This approach is based on a detailed assembly of oceanographic factors. The results represent a very comprehensive segmentation of the pelagic biota for the open ocean system but are of limited utility in coastal waters (Spalding et al., 2007). The boundaries of these provinces are not permanent in space and time but are dynamic and move under seasonal and interannual changes (Longhurst, 2007).

The approach presents four primary biomes: Polar biome, Westerlies biome, Trades biome, and Coastal biome. These were defined based on six models to predict pelagic production mechanisms regarding only the upper part of the ocean and are present in every major ocean basin (Longhurst, 2007). The biogeochemical provinces, presented as secondary compartments, take into

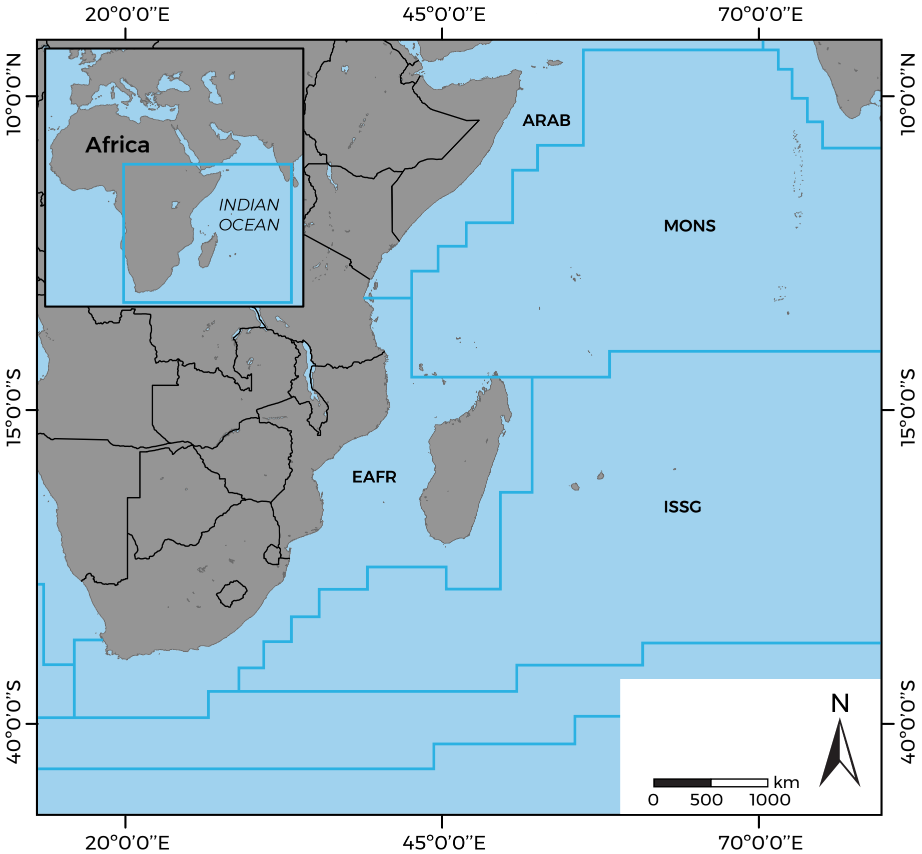


Figure 5: The Longhurst biogeochemical provinces of the WIO region. ARAB – Northwest Arabian Sea Upwelling Province, MONS – Indian Monsoon Gyres Province, EAFR – Eastern Africa Coastal Province, ISSG – Indian South Subtropical Gyres Province. Adapted from Longhurst (2007).

consideration more variables resulting in a more detailed division of the ocean.

Focusing on the WIO region, Longhurst (2007) delineates four provinces included in two biomes: the Indian Ocean Trade Wind Biome, and the Indian Ocean Coastal Biome (see Fig. 5). In the first, there are two provinces:

- **Indian Monsoon Gyres Province (MONS)**, extending from the hydrochemical front located at 10°S, northward to the offshore limits of the coastal provinces. This province also includes the central Bay of Bengal and the southern part of the Arabian Sea. The reversing circulations of the two monsoon gyres of the northern India Ocean give name to this province. It has low salinity and high surface temperatures, benefiting from an extension westward of the warm-water pool of the Pacific Ocean.

- **Indian South Subtropical Gyres Province (ISSG)**, extending from the hydrochemical front at ~10°S, northward to the Subtropical Convergence at ~40°S. The Australian coastal boundary at the outer edge of the Leeuwin Current sets the eastern margin. The western limit is the outer edge of the Agulhas and East Madagascar Currents. The Nazareth, Saya de Malha and Seychelles Banks constitute wide areas of flat shallow topography (<200m deep) that spread along the Mauritius-Seychelles Ridge. The zonal thermocline ridge, located at about 10°S is the limit of the westward flow of the low-salinity South Equatorial Current (SEC) on its southern border, and of the eastward flow of the South Equatorial Counter-current (SECC) along its northern limit.

The Indian Ocean Coastal Biome has also two provinces within the WIO region:

- **Northwest Arabian Sea Upwelling Province (ARAB)**, including the coastal areas from central Kenya (2°S) to Pakistan, and the north-west Arabian Sea. Processes on both flanks of the Southwest Monsoon Jet become a single system, such as the region off the coasts of Somalia and Oman. The African and Arabian coasts of this province have very narrow (<5–10 km) shelves adjacent to steep continental slopes. The reversing monsoon wind stress stimulates the circulation and mixed layer dynamics, which result in upwelling and algal blooms during the boreal summer.

- **Eastern Africa Coastal Province (EAFR)**, comprising the coastal boundary of the Indian Ocean from Zanzibar (5°S) to the Cape of Good Hope, including the Mozambique Channel and the east coast of Madagascar, down to the Agulhas current retroflexion south of Africa. The continental shelves along most of the eastern coast of Africa are rather narrow with a steep slope. In the wider bight (15–24°S) along the coast of Mozambique, the shelf is wider so that at Beira the break of slope is almost 150 km offshore. The greatest area of shelf is the flat triangular Agulhas Bank. Processes over the continental shelves are everywhere directly influenced by local patterns of weather and wind stress at the sea surface. In this province, these factors respond to the seasonal, latitudinal progression of the atmospheric intertropical convergence zone (ITCZ).

Dunstan bioregions of the Indian Ocean

This system presents sub-regional bioregionalizations for the Indian Ocean. It combines approaches developed by CSIRO, in Australia, used throughout the Indian and Pacific Oceans to derive a single combined bioregionalization (Dunstan et al., 2020). The project draws on experience from CSIRO, the Global Ocean Biodiversity Initiative (GOBI) partners, and other collaborators. The main method comprised an expert workshops and novel statistical analysis of the physical and biological data (Dunstan et al., 2020). The system attempts to reduce complexity of classification by harmonizing systems with the perspective of providing support for the management of both areas under national jurisdictions as well as areas beyond national jurisdiction (ABNJ).

The classification considers both the epipelagic and benthic regions and provinces. In the WIO region, the included epipelagic regions are: Western Tropical Indian Ocean, Northern Central Indian Ocean, Mozambique Channel and the Indian Ocean Subtropical Gyre. The benthic regions included in the WIO region are: Western Indian Ocean, and Southern Indian Ocean. The provinces within these regions are presented in Table 1.

OTHER BIOGEOGRAPHIC APPROACHES

The WIO region roughly corresponds to the Area 51 of the FAO major fishing areas, set for fisheries statistical purposes. The areas have arbitrary boundaries which include criteria such as the natural boundaries and divisions of the oceans, boundaries of adjacent statistical fisheries bodies, existing practices, national boundaries, a longitude/latitude grid system, distribution of marine fauna, and the distribution of resources and environmental conditions. The system is set to facilitate statistical

Table 1: The six regions within the Dunstan WIO bioregion and associated provinces for both for both epipelagic and benthic environments, for the considered geographical context of the WIO region.

TYPE	REGION	PROVINCES
Epipelagic	Western Tropical Indian Ocean	Western Tropical Indian Ocean
	Northern Central Indian Ocean	Northern Central Indian Ocean
	Mozambique Channel	Mozambique Channel
	Indian Ocean Subtropical Gyre	Northern Subtropical Gyre, Southern Subtropical Gyre, Indian Atlantic Transition, Northern Subtropical Gyre Transition, Southern Subtropical Gyre Transition
Benthic	Western Indian Ocean	Central Western Indian Ridge, Coral Island Ridge Bathyal, Chagos Shelf, Arabian Sea Abyss, Somali Abyss, Somali Bathyal, Somali Shelf, East Africa Coral Shelf, Sofala Shelf, North and West Seychelles Shelf, Mascarene plateau shelf, Mascarene plateau bathyal, Mascarene Islands shelf, North Madagascar Shelf, South Mozambique Channer Bathyal, West Madagascar Shelf, East Madagascar Shelf, East Madagascar Bathyal, South Madagascar Shelf, Madagascar Plateau Bathyal, Madagascar Plateau Bathyal, Mascarene and Madagascar Basis, Delagoa Shelf, Natal Shelf
	Southern Indian Ocean	Agulhas Bank South, Southwest Indian Bathyal, Southwest Indian Abyss, Del Cano Southern Shelf, Del Cano Southern Ridge

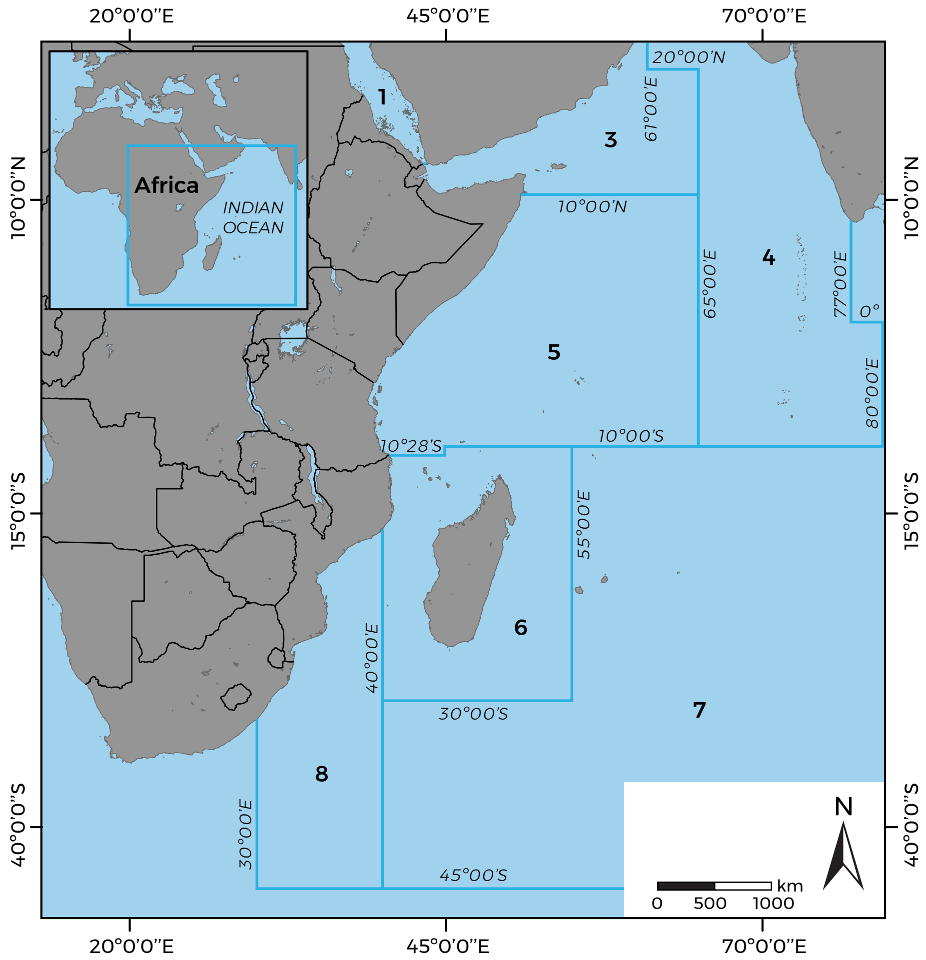


Figure 6: The Area 51 of the FAO major fishing areas, corresponding to the Western Indian Ocean. Note that Area 51 is much larger latitudinally than the WIO region, by including part of the southern ocean in subareas 7 and 8, and also including in the norther Indian Ocean the Red Sea, the Gulf, and the Arabian Sea in subareas 1-4. Conversely, Area 51 e slightly smaller longitudinally to the west, by not including the southern coast of South Africa to the Cape. Adapted from FAO (2022).

comparisons by aligning as much as possible with other major classification schemes, in particular those of competent fisheries commissions (FAO, 2022). Area 51 includes some subareas that are not contained in the geographical context of the WIO region under this *Outlook* (see Fig. 6). These are the Red Sea (subarea 51.1), the Gulf (subarea 51.2) and the Western Arabian Sea (subarea 51.3). This later subarea is partially included due to its southern boundary made by projection of rounded parallel degree limits. Subarea 51.4 (Eastern Arabian Sea and Laccadives) has a long latitudinal span and also only partially contained in the WIO region. Subareas 51.5 (Somalia, Kenya and Tanzania) and 51.6 (Madagascar and Mozambique Channel) are at the core of the WIO region. Subareas 51.7 (Oceanic) and 51.8 (Mozambique) extend deeply into the Southern Ocean, and the southern portion of the South African coast to the Cape is excluded from the WIO, and instead included on the Major Fishing Area 47 of the southern Atlantic Ocean.

Other systems can be drawn, and researchers for instance interested in particular depth zones or specific taxa have provided different classifications, or proposed alteration to the major schemes. The fact is that although the different classification schemes can provide coherent compartmentalization according to environmental zones and biological communities contained therein, specific management targets may require other approaches.

Briggs biogeographic provinces (Briggs and Bowen, 2012) were designed to refine early recognized marine divisions and are based essentially on distribution of reef fishes. Also based on the reef fishes, Kulbicki et al. (2013) have provided results of comparisons with the ecoregions system. The WIO region is distinct from adjacent provinces, and when compared to the MEOW classification some similar conclusion to the analysis by Obura (2012) can be seen, such as for instance the links to Chagos ecoregion to the east.

Analysis of coral species distribution shows that historical processes, habitat heterogeneity and species colonization ability account for more of the present-day biogeographical patterns of corals than explanations based on the contemporary distribution of reefs or environmental conditions (Keith et al., 2013). The WIO region results in a cohesive coral province including the Chagos and Maldives archipelagos when defining the divisions based on co-occurrence of multiple species range boundaries. However, when analyzing the geological and environmental classification, the discontinuities seem to split the region into several units.

The Chagos region seems to be a transitional zone. Briggs and Bowen (2012) include Chagos as part of the Indo-Polynesian Province, but according to Winterbottom and Anderson (1997) and Gaither et al. (2011) the archipelago has faunal affinities with both the Indo-Polynesian Province and the WIO Province.

Most classifications refer to coastal and shallow water. When we consider depth zones, current patterns and geographical influence vary and will result in different ocean divisions. On the other hand, addressing different taxonomic groups may also give different results.

The biogeographic classification of the oceans is of primary importance for the management of marine resources and facilitates our understanding of complex natural processes. Management action leading to effective protection measures is based on bioregionalization systems (Woolley et al., 2019). Bioregions are thus of key importance for promoting spatial management options (Hill et al., 2020). The spatial classification of the WIO region can be made according to specific needs, such as fisheries management, biodiversity conservation or others. No single system provides all the answers for decision-making.

CONCLUSION

The geographical context of the WIO region extends from Somalia to the southern tip of South Africa and from the eastern African coast to the central Indian Ocean, including the Chagos Archipelago. The region comprises wide environmental gradients and hosts all major tropical ecosystems and important marine biodiversity and resources. It further includes temperate habitats in the southern African zone.

Several regionalization classification systems have been developed for the WIO to facilitate ocean study, management, and governance. These classifications were based in different contexts and used diverse criteria to meet sectorial needs. The coastal regions were divided using biophysical characteristics (eg MEOW, OneEarth Bioregions 2020) or fishing areas (eg LMEs, EEZs). Oceanic approaches used biogeochemical characteristics (eg Longhurst provinces), analysis of physical and biological data (Dunstan et al., 2020) or fisheries management areas (eg FAO major fishing areas). Bioregionalization systems allow the study of the oceans with partitioned and objective views, but natural processes do not obey political or other artificial man-made boundaries. Integrated transboundary approaches are thus needed to reach effective regional scale management in the WIO.

REFERENCES

Briggs, J.C. and Bowen, B.W. (2012). A realignment of marine biogeographic provinces with particular reference to fish distributions. *J. Biogeogr*. 39, pp. 12–30. https://doi:10.1111/j.1365-2699.2011.02613.x

Dinerstein, E., Olson, D., Joshi, A., Vynne, C., Burgess, N.D., Wikramanayake, E., Hahn, N., … Saleem, M. (2017). An Ecoregion-Based Approach to Protecting Half the Terrestrial Realm. *BioScience* 67(6), pp. 534–545. https://doi.org/10.1093/biosci/bix014

Duda, A.M. and Sherman, K. (2002). A new imperative for improving management of large marine ecosystems. *Ocean Coast. Manag.* 45(11–12), pp. 797–833. https://doi.org/10.1016/S0964-5691(02)00107-2

Dunstan, P.K., Hayes, D., Woolley, S.N.C., Bernawis, L., Foster, S.D., Khani, E., Walton, R., … Molony, B. (2020). *Bioregions of the Indian Ocean*. CSIRO, Australia

FAO (2022). Indian Ocean, Western (Major Fishing Area 51). Fisheries and Aquaculture Division [online]. Rome. https://www.fao.org/fishery/en/area/51/en

Gaither, M.R., Bowen, B.W., Bordenave, T.-R., Rocha, L.A., Newman, S.J., Gomez, J.A., van Herwerden, L. and Craig, M.T. (2011) Phylogeography of the reef fish Cephalopholis argus (Epinephelidae) indicates Pleistocene isolation across the Indo-Pacific Barrier with contemporary overlap in the Coral Triangle. *BMC Evol. Biol.* 11(1), pp. 1–16. https://doi.org/10.1186/1471-2148-11-189

Godet, C., Robuchon, M., Leroy, B., Cotté, C., Baudena, A., Da Silva, O., Fabri-Ruiz, S., Lo Monaco, C., Sergi, S. and Koubbi, P. (2020). Matching zooplankton abundance and environment in the South Indian Ocean and Southern Ocean. *Deep-Sea Res. I: Oceanogr. Res. Pap.* 163, 103347. https://doi.org/10.1016/j.dsr.2020.103347

Hempel, G. and Sherman, K. (eds.) (2003). *Large Marine Ecosystems of the World: Trends in Exploitation, Protection, and Research*. Amsterdam: Elsevier

Hill, N., Woolley, S.N.C., Foster, S., Dunstan, P.K., McKinlay, J., Ovaskainen, O. and Johnson, C. (2020). Determining marine bioregions: A comparison of quantitative approaches. *Methods Ecol. Evol.* 11(10), pp. 1258–1272. https://doi.org/10.1111/2041-210X.13447

IOC-UNESCO and UNEP (2016). Large Marine Ecosystems: Status and Trends. United Nations Environment Programme (UNEP), Nairobi

Keith, S.A., Baird, A.H., Hughes, T.P., Madin, J.S. and Connolly, S.R. (2013). Faunal breaks and species composition of Indo-Pacific corals: the role of plate tectonics, environment and habitat distribution. *Proc. R. Soc. B: Biol. Sci.* 280(1763). https://doi.org/10.1098/rspb.2013.0818

Kulbicki, M., Parravicini, V., Bellwood, D.R., Arias-Gonzàlez, E., Chabanet, P., Floeter, S.R., Friedlander, A., McPherson, J., Myers, R.E., Vigliola, L. and Mouillot, D. (2013). Global Biogeography of Reef Fishes: A Hierarchical Quantitative Delineation of Regions. *PLoS ONE* 8(12), e81847. https://doi.org/10.1371/journal.pone.0081847

LEARN (2017). The Large Marine Ecosystem approach: An Engine for Achieving SDG 14. GEF LME:LEARN. Paris, France

Longhurst, A. (2007). *Ecological Geography of the Sea*. Burlington: Academic Press, 2nd Edition, 542 pp.

Obura, D. (2012). The diversity and biogeography of the Western Indian Ocean reef building corals. *PLOS One* 9(7), e45013

One Earth (2020). Bioregions 2020. One Earth. https://www.oneearth.org/bioregions-2020/. Accessed on 28/3/2022.

Porter, S.N., Branch, G.M. and Sink, K.J. (2013). Biogeographic patterns on shallow subtidal reefs in the western Indian Ocean. *Mar. Biol*. 160(5), pp. 1271–1283. https://doi.org/10.1007/s00227-013-2179-7

Sherman, K. and Alexander, L.M. (1989). Biomass Yields and Geography of Large Marine Ecosystems. Boulder (CO): Westview Press

Sherman, K., Sissenwine, M., Christensen, V., Duda, A., Hempel, G., Ibe, C., Levin, S., Lluch-Belda, D., Matishov, G., McGlade, J., O'Toole, M., Seitzinger, S., Serra, R., Skjoldal, H.-R., Tang, Q., Thulin, J., Vanderweerd, V. and Zwanenburg, K. (2005). A global movement toward an ecosystem approach to management of marine resources. *Mar. Ecol. Prog. Ser.* 300, pp. 275–279

Spalding, M.D., Fox, H.E., Allen, G.R., Davidson, N., Ferdaña, Z.A., Finlayson, M., Halpern, B.S., Jorge, M.A., Lombana, A., Lourie, S.A., Martin, K.D., McManus, E., Molnar, J., Recchia, C. A. and Robertson, J. (2007). Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. *BioScience* 57(7), pp. 573–583. https://doi.org/10.1641/B570707

UNESCO (2009). Global Open Oceans and Deep Seabed (GOODS) – biogeographic classification (Vol. 4, Issue 1). IOC Technical Series, 84. UNESCO-IOC, Paris

Wendt, H., Wendt, H., Beger, M., Sullivan, J., LeGrand, J., Davey, K., Yakub, N., Kirmani, S., Grice, H., Mason, C., Raubani, J., Lewis, A., Jupiter, S., Hughes, A., Ceccarelli, D. and Fernandes, L. (2018). Marine Bioregions of Tonga. MACBIO (GIZ/IUCN/SPREP), 56 pp. Suva, Fiji

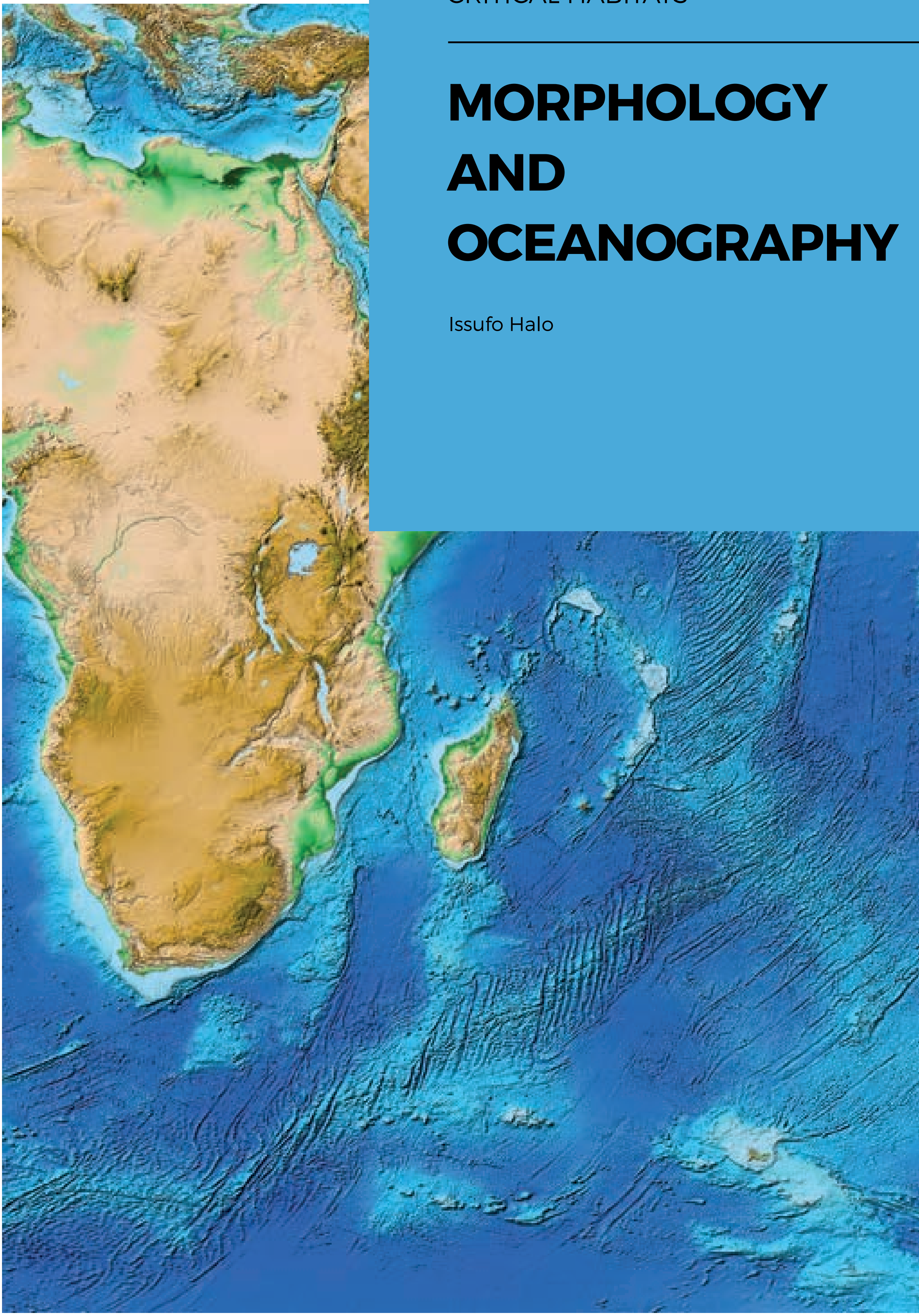
Winterbottom, R. and Anderson, R.C. (1997). A revised checklist of the epipelagic and shore fishes of the Chagos Archipelago, central Indian Ocean. *Ichthyological Bulletin* 66, pp. 1–28

Woolley, S.N.C., Foster, S.D., Bax, N.J., Currie, J.C., Dunn, D.C., Hansen, C., Hill, N., O'Hara, T.D., Ovaskainen, O., Sayre, R., Vanhatalo, J.P. and Dunstan, P.K. (2019). Bioregions in Marine Environments: Combining Biological and Environmental Data for Management and Scientific Understanding. *BioScience* 70(1), pp. 48–59. https://doi.org/10.1093/biosci/biz133

WWF (2007). Marine Ecoregions of the World: A Bioregionalization of Coastal and Shelf Areas. World Wildlife Fund (WWF) https://www.worldwildlife.org/publications/marine-ecoregions-of-the-world-a-bioregionalization-of-coastal-and-shelf-areas. Accessed on 28/3/2022

MORPHOLOGY AND OCEANOGRAPHY

Issufo Halo



Relief model of the WIO surface integrating land topography and ocean bathymetry.
© Amante and Eakins / National Geophysical Data Center, NOAA (2009)

INTRODUCTION

On a global scale, analysis of a histogram of the Earth's solid surface (which accounts for land topography, and ocean bathymetry) by means of a hypsographic curve, shows that the continents are several hundred meters above the sea-level, while the oceans are about 4300 m below. A great deal of geological processes across several time and space scales are known to drive the land and seafloor morphology of the earth's surface (Parson and Evans, 2005). In some cases, these forcings, both in the vertical and horizontal directions, have resulted in unique morphologies, eg upwarping and down-dropping along the fracture zone of the East Africa tectonic plate (Pepper and Everhart, 1963). Massive uplifting land topography on the reams of the Western Indian Ocean (WIO) countries plays an important role in modulating various geologic processes (Oettli and Camberlin, 2005) and services across multi-disciplinary sciences (UMLP, 2016), eg historical migration of the hominids over the past eight million years (Myr), agricultural practices (Sepulchre et al., 2006) as well as the evolution of the atmosphere, ocean and ecological systems across various trophic levels (Spencer et al., 2005).

It has been hypothesized that the uplifting of the eastern Africa land topography has induced a significant abrupt re-structuring of atmospheric circulation with significant impact on moisture transports and precipitation (Sepulchre et al., 2006; Jung et al., 2016). Studies suggest that the eastern Africa topography has an influence on seasonal rainfall distribution (Oettli and Camberlin, 2005; Yang et al., 2015). The runoff becomes one of the most important physical mechanisms responsible for the export and deposition of land-based material into the sea, thus shaping the coastline and seafloor configuration (Moore et al., 2009; Partridge et al., 2010; Fenta et al., 2020).

The seafloor morphology and bathymetric relief (eg islands, ridges, banks, abysses, canyons) in the WIO region (Fig. 1a) influence the hydrographic and dynamical behaviour of dominant oceanographic processes, such as the intensity of mean currents; filaments, fronts, eddy generation, eddy propagation, pathways and decay of vortex structures (eg meanders, rings and eddies); hotspot formation of internal waves; ocean-atmosphere flux exchanges, vertical mixing, mass/volume transports, and the development of upwelling/downwelling events, etc. (Matano et al., 1999; Ansorge and Lutjeharms, 2003; Parson and Evans, 2005; Spencer et al., 2005; Penven et al., 2006; Lutjeharms, 2006; Read and Pollard, 2017; Pollard and Read, 2017). Many of these processes, either isolated or combined, have a strong impact on the composition, state, and functioning of the regional ecosystems (Partridge et al., 2010; Barlow et al., 2014). A case study to highlight this fact has been presented in this chapter.

Therefore, in this chapter the aim is to present a detailed and comprehensive description of the characteristics of the land and ocean bottom topography of the WIO (Fig. 1b–c), based on the best available scientific information of the region, assessed through published material. Among many, an important study to highlight in this chapter is the extensive work conducted and published by an anonymous professor of the University of Minnesota Library (hereafter UMPL), Minneapolis, USA, cited as UMLP (2016).

In addition to the review, wherever appropriate, the description is complemented by analysing recent observed datasets freely available through the worldwide web, such as global earth bathymetric chart of the oceans (GEBCO)[1], global mean dynamic topography of the oceans (CNES–CLS09 MDT), satellite altimetry derived maps of absolute dynamic topography (MADT)[2] and their derived fields of geostrophic currents, satellite windstress fields derived from Scatterometer Climatology of Ocean Winds (SCOW)[3], and ocean-atmosphere surface density fluxes computed from long-term hydrographic thermal and haline properties derived from the world ocean atlas database (WOA)[4]. These allow us to capitalize on relevant oceanographic processes such as mesoscale variability, eddies, and hydrological properties, somehow linked to the prominent bathymetric configurations that have a significant impact on shaping the WIO's habitat systems, classified as "critical".

LAND MORPHOLOGY

Continental mainland morphology

Fig. 1b–c shows a three-dimensional view of the Earth's solid surface (land-ocean), derived from GEBCO dataset, mapped for the WIO, here defined as the region spanning from 10–80°E and 50°S–30°N. For the purpose of detailed visualization of the topographic features, Fig. 1 is presented in two different three-dimensional perspectives (Fig. 1a, b). From south to north, the land topography

1. https://www.gebco.net/data_and_products/ridded_bathymetry_data/
2. http://marine.copernicus.eu/
3. http://numbat.coas.oregonstate.edu/scow/
4. https://www.nodc.noaa.gov/OC5/woa18/woa18data.html/

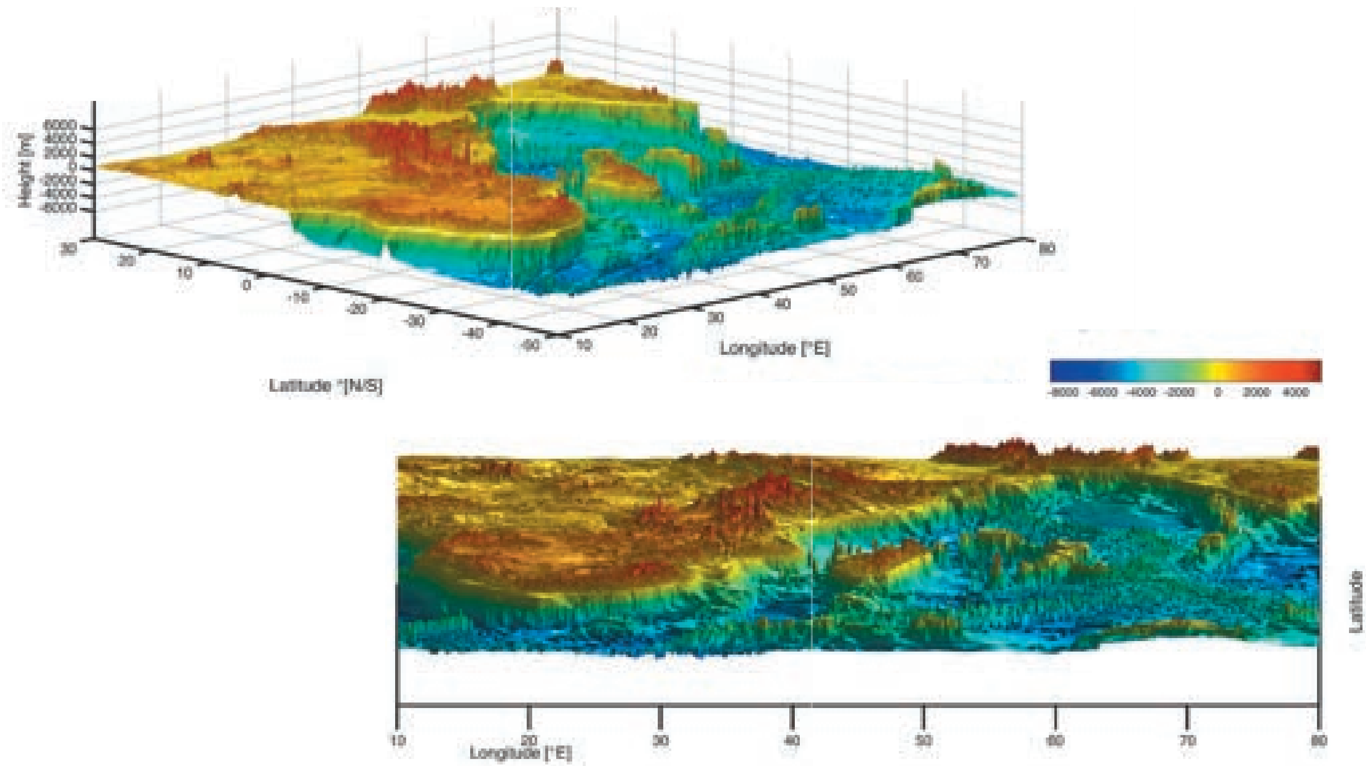


Figure 1 a and b: Land-ocean bed morphology of the WIO region as derived from the GEBCO dataset. For better visualization, the topographic features in this panel have been reconstructed in a 3D frame, rotated by 5° Azimuthal angle and 60° elevation, using MATLAB software, then displayed in a horizontal flat plane. Note that the longitude and latitude values are not shown due to the rotational effects. The reader may find more informative to compare this plate with that shown in Fig. 5 with portrayal of the ocean currents. The background positive (negative) colours show the land (seafloor) topographic domains. The corresponding colour bar is scaled in meters.

CASE STUDY

Indian Ocean climate variability

The tropical Indian Ocean forms one of the major parts of the largest warm pool on Earth. It is not surprising that its interaction with the overlying atmosphere plays a crucial role in influencing the climate system at regional and global scales.

As unprecedented, the tropical WIO is warming at a faster rate than any other in the world's ocean. Severe catastrophic flooding/drought events over the WIO rim's countries are directly linked with dynamic climate modes of oceanic variability such as Indian Ocean Dipole and El-Niño Southern Oscillation (ENSO). During the worst ENSO of the century in 1997–1998, the ENSO driven flooding/droughts were reported to have caused thousands of deaths and misplaced hundreds of thousands of people in the WIO region.

The need for better prediction of Indian Ocean climate variability is essential. Interestingly, coral records suggest a strong linkage between the WIO sea surface temperature and the ENSO. A case study by Zinke et al. (2008) based on analysis of coral *γ18Oseawater* derived from both coupled ratio between chemical elements Strontium (Sr) and Calcium (Ca), ie Sr/Ca and salinity measurements, rainfall and rate of precipitation minus evaporation in *Porites* of Mayotte corals, in Comoros Archipelagos between 1881 and 1994 has enabled reconstruction of the WIO hydrological historical data. The results reveal that the balance between precipitation and evaporation rates varies on timescales of five to six years and 18–25 years. High and low oceanic surface temperatures are found to be linked with positive and negative *γ18Oseawater*. It also has been found that negative freshwater balance at Mayotte island are linked with warm ENSO events.

The study highlights the importance and synergies between the physical environmental forcing on critical habitats of the WIO region on climatological timescales. It also reinforces the need for much denser network of *γ18Oseawater* reconstruction for a better assessment of spatial patterns of hydrological conditions in the region.

starts first by depicting the South African continental morphology (Fig. 1a–b), known to have a higher surface elevation that covers more than 40 per cent of the total land surface area of about 1.22 million km^2, and the mean altitude is about 1200 m (Bond, 1979). The surface land encompasses three main regions: (i) marginal region with a width ranging between 80 and 240 km in the east, and between 60–80 km in the west; (ii) the interior plateau, which separates from the marginal regions by means of the Great Escarpment; and (iii) the Kalahari Basin (Kruger, 1983). The land upslopes from west to east towards the Drakensberg Mountains, where the tallest mountain is the Injasuti Mountain with an altitude of about 3408 m, near the border with Lesotho (Moore et al., 2009; Partridge et al., 2010). From Drakensberg the terrain downslopes eastward towards the Indian Ocean (Fig. 1c), passing through the hills and narrow coastal plain in the valleys of KwaZulu-Natal (Fig. 1a). The country's total coastline is about 2798 km, and the climate is predominantly semi-arid and subtropical along the east coast, with an average precipitation of about 495 mm year^{-1} (Partridge et al., 2010).

Extending northward along the coast bordering the Indian Ocean, the political boundary separates South Africa from Mozambique (Fig. 1a). Mozambique has a surface area of nearly 800 000 km^2 (Cabral et al., 2017) and the stretched coastline is about 2800 km long (Palalane et al., 2016). The terrain is characterized mostly by low coastal plains and in the central interior it is elevated (Fig. 1b–c). The plateau lies in the north-west of the country marked by a range of mountains in the western part. The climate is tropical to subtropical, and the average rainfall is about 1032 mm year^{-1} (Partridge et al., 2010).

To the north of Mozambique lies the United Republic of Tanzania (Fig. 1a), with a surface area of about 945 090 km^2. Like in Mozambique, the terrain is also variable, characterized by coastal plains, plateaus in the centre, and highlands in the northern and southern parts of the country (Fig. 1b–c). The country hosts the highest mountain in Africa, the Kilimanjaro, which has an altitude of about 5894.83 m (UMLP, 2016; Fenta et al., 2020), and is located close to the border with Kenya. The climate varies, being of tropical nature in the coast and temperate in the highlands. The average rainfall is estimated at 1071 mm year^{-1} (Westeberg and Christiansson, 1999; NationMaster, 2018).

To the north of Tanzania along the Indian Ocean lies Kenya (Fig. 1a), which encompasses a surface area of about 569 250 km^2, with a relatively shorter coastline of about 536 km (Westeberg and Christiansson, 1999; Fenta et al., 2020). The surface land is characterized by low plains that ascend to the central highlands (Fig. 1b-c), which are then bisected by the Great Rift Valley, while the plateaus lie to the western side. The country hosts the second highest mountain in Africa, the Mount Kenya, which has an altitude of about 5199.28 m, and is located to the north of the Nairobi capital city, near the Equator. The climate varies from tropical along the coast to arid in the central interior. Both Kilimanjaro in Tanzania and Mount Kenyan are located to the east of the East Rift Valley and are characterized by inactive volcanoes and have permanent snowing regime at their summits (UMLP, 2016). They are providers of freshwater to the low-land surrounding areas. Nevertheless, there are also some active volcanoes in some relatively low range latitude mountains such as the Ol-Doinyo Lengai (Westeberg and Christiansson, 1999).

Somalia is located to the north of Kenya (Fig. 1a), occupying a surface area of about 637 650 km^2, and a coastline of about 3025 km (Leslie, 1991), which is the longest along the eastern African continent. The land topography is mostly flat with an undulating plateau rising to high elevations in the north. Leslie (1991) indicates that the central and southern parts of Somalia are mostly plain and plateau. This feature is contrasted with that in the north, which is mountainous, whereby some peaks can reach more than 2000 m above sea level (Fig. 1b–c). The climate is predominantly desert, with strong influence of the north-east and south-west monsoons, spanning from December to February and May to October, respectively. It is moderate in the north and hot in the south, typical of the south-west monsoon season. During the north-east monsoon the climate is torrid in the north and hot in the south. During the transitional phases of the monsoons the climate is characterized by irregular rainfall events with hot and humid periods. Past studies revealed that larger parts of the country receive less than 300 mm year^{-1}. The overall annual average of the rains is 2330 mm year^{-1} (NationMaster, 2018).

Islands morphology

To the east of the African mainland lie the several island states of the WIO (Fig. 1a). The island of Madagascar is the largest (it being the world's fourth-largest), and is located at about 415 km away from the African mainland (Fig. 1a). It is thought that the island broke away from the main continent more than 160 million years ago (Ma), thus developing its unique environmental characteristics. The island is about 1400 m above sea level and its surface area is about 587 040 km^2, and the coastline is about 4828 km long. The land morphology is characterized by narrow coastal plain and high plateaus and mountains in the centre (UMLP, 2016). The summit of the Tsaratanana massif, known as the Maromokotro is about 2876 m in altitude, and the tip of the Ankaratra massif, known as the Tsiafajavanova is about 2643 m. The climate is tropical along the coast, temperate inland, and arid in the south. The average rainfall is about 1513 mm year^{-1} (NationMaster, 2018). Tropical rain forests are located on the eastern edge on the windward side of the island, whereas the western side is characterized by rain shadow effect, receiving lower precipitation rates.

Madagascar is surrounded by several independent island states to the north, namely, Comoros, Seychelles and to the east by Reunion and Mauritius Islands (Fig. 1a). The Comoros is an archipelago located at the northern entrance of the Mozambique Channel, between Madagascar and northern Mozambique (Fig. 1a). The archipelago hosts four main islands, Grande Comore, Mohéli, Anjouan, and Mayotte (Fig. 2e). The distance between the first two islands is about 40 km, and between the last two is about 80 km (Goodman et al., 2010). Its surface area is about 2230 km^2, and an extension of coastline of about 340 km. The islands are oceanic of volcanic origin (Harris and Rocha, 2009), varying from steep mountains to low hills in the interior. The climate is marine tropical with rainy seasons between the months of November and May (Harris and Rocha, 2009) and the average rainfall is about 900 mm year^{-1} (NationMaster, 2018). Mayotte Island in the archipelago is the closest to Madagascar by 300 km (Fig. 2e), and it has a surface area of about 37 km^2, and a coastline of 185.2 km. The terrain is bumping with deep ravines and ancient volcanic peaks. It is characterized by a tropical, marine, hot, humid rainy climate during the north-east monsoon (November–May), while cooler-dry season dominates during the south-west monsoon.

To the north of Madagascar lies the Seychelles (Fig. 1a, Fig. 2h), occupying a surface area of about 460 km^2, and a coastline of about 491 km long. The largest islands are characterized by a granitic narrow coastal strip, being rocky and hilly. Many of the smaller islands of the Seychelles are coral flats and elevated reefs. The climate is tropical marine. During the southeast monsoon the climate is humid and cool, from late May to

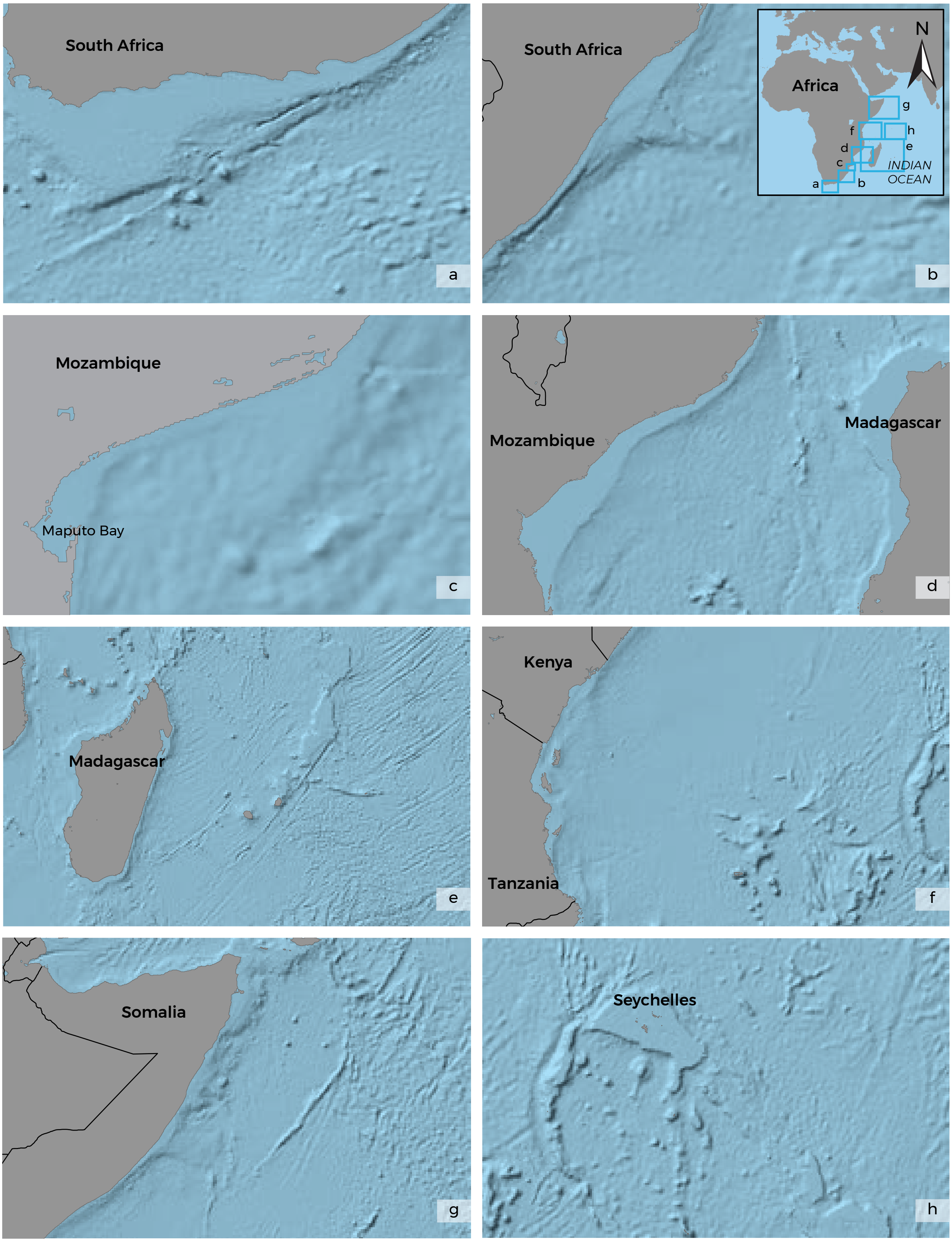


Figure 2: Aerial view of the seabed morphology showing continental margins and main deep ocean basins in the WIO. Special attention is made for the following features: (a) Agulhas Bank; (b) Natal Bight; (c) Delagoa Bight; (d) Sofala Bank and western Madagascar Bank; (e) Comoros archipelagos, Madagascar southern and eastern continental shelves, Mauritius and La-Reunion Islands, Mascarene Plateau; (f) Tanzanian and Kenyan continental shelves, including Mafia, Zanzibar and Pemba Islands; (g) Somali continental shelf; (h) Seychelles Island and its bank.

September, while it is warmer during the north-west monsoon, between March and May. The average rainfall is about 2330 mm year^{-1} (NationMaster, 2018).

To the east of Madagascar lies the volcanic Island of Reunion (Fig. 1a, Fig. 2e). The coastline has an extension of about 207 km long, and the land covers about 120 km^2. The terrain is predominantly rugged and mountainous, with fertile lowlands along the coast. Maximum altitude is the mount Piton des Neiges with an elevation of about 3069 m. The climate is tropical but gets moderated with altitude. It is cool and dry between May and November, but hot and rainy between November and April.

Further east of Reunion Island lies Mauritius Island (Fig. 1a, Fig. 2e). It covers a surface area of about 2040 km^2, and a coastline length of about 177 km. It is characterized by small coastal plains and discontinuous mountains that encircle a central plateau. The climate is tropical, modified by the south-east trade winds. It has warm and dry winters between May and November, and hot, wet and humid during summer, which is extended between December and April (NationMaster, 2018). The smaller sister island of Rodrigues lies further to the east.

Seafloor morphology

The morphology of the world's ocean seafloor topography is divided in three oceanic provinces, namely, (i) continental margin, (ii) deep ocean-basins (abyssal plains), and (iii) mid-ocean ridges. While limited and scant, scientific studies to-date conducted in the WIO region provide a great deal of detail from which it is possible to infer that the seafloor topography of this region is unique (Fig. 1). Some of the peculiarities have been highlighted in several published material, and include: (i) it is composed by all types of tectonic plate boundaries of active and fossil composition, (ii) it encompasses some of the most deep-reaching fracture zones (Parson and Evans, 2005); (iii) it is dominated by an array of several mid-oceanic ridges (Fig. 1), which have nearly a meridional orientation (Defant, 1961; Tomczak and Godfrey, 1994); with (iv) some of the thickest sedimentary settlements of the world's ocean basins (Parson and Evans, 2005).

CONTINENTAL MARGINS

The continental margin is the domain that stretches from the littoral up to the end of the continental rise (or beginning of the deep ocean basin). It comprises three important bathymetric features, namely, (i) continental shelf, (ii) continental slope and (iii) continental rise. The boundaries of the deep ocean basins in the WIO region are generally of non-volcanic origins, with passive rift geometries, floored extensively by moderate sediment cover. They are characterized by continental shelves of variable forms and dimensions (Parson and Evans, 2005). In the following sections are presented an account of some of the most prominent features of the WIO continental shelves.

Continental shelves, banks and bights

The continental shelf is an integral part of the continental margins, which extends from the littoral to the location where its profile depth steepens abruptly (shelf-break). The shelves are commonly characterized by gentle slopes, and the isobaths of 200 m contour is generally referred as their average extension. They encompass banks, bights, bigots, etc., and are generally floored by coastal rocks and sediments eroded from the inner-land environment. In the WIO region, adjacent to the African continent, one of the most prominent continental shelf is the Agulhas Bank, located directly to the south of South Africa (Fig. 2a). The second after this is the Sofala Bank, in central Mozambique (Fig. 2d). Apart from these, elsewhere the continental shelves of the WIO region are relatively narrow strips (Pepper and Everhart, 1963).

South African south and south-eastern continental shelves

The Agulhas Bank is a broad, near-triangular bathymetric protrusion of the continental shelf, lying at the southern tip of Africa (Fig. 2a). The Bank extents southward, from the coast to a maximum distance of about 300 km. It has an area of about 80 000 km^2 (Lutjeharms and Cooper, 1996), and is largely dominated by broad platforms of rocks believed to be from Palaeozoic age, mixed with sediments of Tertiary and more recent ages (Pepper and Everhart, 1963). The morphology of the Bank suggests that this feature may have been affected by tectonic forces that thrust inland from the sea, and impacted the inland system, to form narrow east-west folds of Cape Mountains (Pepper and Everhart, 1963). The topography of the Agulhas Bank can be divided in two distinct domains: to the west and to the east of the Cape Agulhas. The edge of the continental margin to the west of Cape Agulhas descends to depth of about 4000 m towards the seafloor. The upper margin is divided in three distinct morphological zones, namely, the inner-shelf; middle shelf; and the upper slope. The inner-shelf is rocky, with widths varying between 16–32 km between Cape Point and near Cape Agulhas, respectively. The middle shelf is gentle ~ 1 m km^{-1}, but it steepens at the outer shelf towards the shelf-break. The shelf-edge varies from 200 to 400 m between south of Cape Agulhas and Cape Point respectively. The upper slope is characterized by a steeper average gradient of about 40 m km^{-1}. It encompasses some submarine canyons below the shelf-break. The Agulhas Bank is surrounded by important large-scale oceanographic features, namely, the warm Agulhas Current along the south and east coast of South Africa, which derives its water from the Indian Ocean, and the cold Benguela Current along the west coast, which derives its water from the Atlantic Ocean (Lutjeharms, 2006).

Along the east coast of South Africa, the continental shelf varies only between 3 and 16 km in width, stretching nearly in a sinuous fashion for about 724 km, between Port Elizabeth and Durban (Fig. 2b). In Durban the shelf widens to nearly 45 km and form the Natal Bight (Fig. 2b). The wider shelf lies seaward of two trending coastal faults. After the Bight, the continental shelf narrows again to about 6 km, heading north toward Maputo in Mozambique (Fig. 2c) nearly in a straight line, parallel with the inland Lebombo Mountain chain located 80 km from the coast (Pepper and Everhart, 1963), especially between north of Cape St. Lucia and Maputo Bay.

Mozambican continental shelf

In Maputo Bay (Fig. 2c) the shelf widens again and shallows forming one of the largest coastal indentations in the south-west Indian Ocean, the Delagoa Bight (Lamont et al., 2010; Lutjeharms and da Silva, 1988), of about 64 km width (Pepper and Everhart, 1963). Pepper and Everhart (1963) attributed the greater widths of the Bight to the accumulation of sediments deposited by a combination of two opposing currents at the bight, characterized by an intense southward flow field offshore and a weaker north-easterly flow inshore. Recent studies however suggest that the Bight hosts a semi-permanent cyclonic eddy, termed the Delagoa Bight eddy (Cossa et al., 2016; Lamont et al., 2010; Lutjeharms and da Silva, 1988). From the Bay the coast runs northward in a concave trajectory over a sandy ground toward Ponta de Barra, along a shelf which varies from about 16–74 km in width – the latter possibly related to deposition and accumulation of sediments driven by a system of currents including an offshore strong southward flow and a relatively weaker northward inner-shore flow around the Maputo Bay.

From Ponta da Barra the coast progresses northward for about 217 km to the latitude of 22°S, near the Bazaruto Archipelago. From Ponta da Barra to Bazaruto the shelf width narrows from about 16 km to 6 km at the offshore side of the island. Both at Ponta da Barra and south of Bazaruto, coastal embayments are formed, likely from the same geological process (Pepper and Everhart, 1963). From Bazaruto the continental shelf runs north-westwards toward the broad shelf of Sofala Bank (Fig. 2d). There are shoal areas, likely to be coral reefs within the Bank, probably built up from sediments derived from several river streams flowing into the Indian Ocean (Fig. 2d). In this area, at the mouth of the rivers, the shelf widens to about 145 km near Beira. From the Zambezi Delta (Fig. 2d) to the narrows of the Mozambique Channel near 17°S, along an extension of about 322 km, the continental shelf varies from 64 km to 18 km. Across the Bank, transverse geological fractures, or faults on the seafloor off the coast and on the continental slope are evident, forming deep canyons (eg Zambezi canyon). Zambezi canyon extends south-eastward toward the Mozambique Basin (Fig. 1a). As characterized by Pepper and Everhart (1963), from the narrows of the Channel to the northern end of Mozambique (Rovuma River), the continental shelf is relatively narrow and irregular, ranging in width from 3 km to 24 km.

Tanzanian continental shelf

From the Rovuma estuary to about 8°S lies a stretch of about 209 km along the coast of Tanzanian mainland (Fig. 2f), characterized by several cliffs and reefs. The narrow continental shelf closely follows the deep indentation of the coastline, and its width may not even exceed 5 km on average (Pepper and Everhart, 1963). However, around 8°S deep horizontal fractures across the seabed toward the coast and through the continental shelf are present. In the region neighbouring the fracture zone the continental shelf widens out considerably, extending in a north-east orientation around Mafia Island (southernmost island of the Zanzibar archipelagos). From northern Mafia Island, the continental shelf tapers from about 72 km to 6 km along the coral-rich stretch towards the Tanzanian mainland in a north-west direction. It is thought that sediment deposition from several rivers outflowing from the mainland to the west of Mafia have caused a seaward extension of the continental shelf in this area (Pepper and Everhart, 1963).

From about 7°S to Dar es Salaam the continental shelf is narrow, but then it widens again in a north-east orientation, progressing closely along the east coast of Zanzibar Island (Fig. 2f). Subsequently it bends on a north-west direction approaching the mainland at the opposite side of the Pemba Island (northernmost island of the Zanzibar

Archipelago). Several small-scale indentations are present along the western edge of the Pemba coastline. Excluding its northern and parts of the western sector, the continental shelf of Pemba Island is very narrow. Excluding its northern entrance, the transect between Pemba and Tanzania mainland reveals a continental slope that descends to depth of between 549 m and 732 m across the entire length of the Island. Whereas westward along a stretch of about 26 km the seafloor is flat, but it consistently upslopes toward the mainland continental shelf, where it attains a width of about 10 km. The stretch from the northern Pangani River in Tanzania towards the Tanzania-Kenya border (Fig. 2f), as well as the section between Dar es Salaam and Mombasa present geological faults of complex nature and sedimentations that appears to explain the origins and movement of the whole set of islands of the Zanzibar Archipelago (Fig. 2f) (Pepper and Everhart, 1963).

Kenyan and Somalia continental shelves

From the north-east coast of Kenya (Fig. 2f) to Somalia (Fig. 2g) the coast is generally very narrow, and predominantly rocky. There are few indentations and narrow beaches. The north-eastward continental shelf from Kenya to Horn of Africa in Somalia varies in width from place to place (Fig. 2g), between 19 and 60 km. The widest area is north of Watamu where the North Kenya Bank has an offshore extension of the continental shelf that extends 60 km from Ungwana Bay.

Madagascar continental shelf

Directly east of Mozambique, roughly at a minimum distance of about 700 km lies the island of Madagascar (Fig. 2e), with a narrow continental shelf overall of about 24 km in width (Fig. 2e). In localized places however, it reaches a width of about 80.5 km on the south coast, and 161 km on the north-west coast at St. Andre. Madagascar has a nearly straight and narrow coastline along the east, with some places (north-east part) exhibiting an almost complete absence of a continental shelf (Fig. 2e). The edges of the narrow continental shelf of the east coast are geological fault scarps, with a slope that descends to depth of about 1829 m. In the western sector, the shelf extending from Morondava to Cape St. Andre hosts a series of small banks of corals reefs. The widest shelf surface, possibly harbouring some granites is located on the west coast, close to the Juan de Nova Island (Fig. 2d). From Cape St. Andre to Cape Amber the shelf average width is about 48 km, nevertheless it broadens further to the west of the Cape.

Seychelles Bank

Seychelles Bank is a bathymetric feature of Precambrian granite basement, which lies in the western equatorial Indian Ocean, located on the northern extension of the Mascarene Plateau (Fig. 2h). The Bank is confined between 4–6°S and 54–57°E, and is shallow with a nearly flat-topped shape at about 50 m depth below the sea-surface. It is characterized by an oval shape with a dimension of 400 km by 200 km long (Yossi, 1988).

The slope edges of the Bank are steep, descending to depths of more than 3000 m, with the exception of the south-western and south-eastern sectors, which descend to depths of about 2000 m and 1500 m respectively (Yossi, 1988). To the south-west, the Bank is connected to the Amirante Arc by a 2000 m deep saddle depth, while to the south-west it gets separated from the remaining Mascarene Plateau by another saddle also of about 2000 m depth.

MID-OCEAN RIDGE SYSTEM

Mid-Oceanic Ridges are prominent, long extensions of mountain-like structures formed by tectonic plates in response to the convective processes in the mantle underneath the oceanic crust, creating magma material where the tectonic plates have collided and moved apart. In the Indian Ocean, such topographic relief is reflected by a system of three different ridges: Central Indian Ocean Ridge (CIR, Fig. 3a), Southwest Indian Ocean Ridge (SWIR, Fig. 3b), and Southeast Indian Ocean Ridge (SEIR, Fig. 3c).

The ridge system is about 7000 km long, and its western and eastern flanks rise from the seafloor for about 3000 m (Parson and Evans, 2005). Along this length, the water depth is very variable. At its shallowest point it has been documented by Parson and Evans (2005) as being 1500 m, at about 66°9'57"E, 17°30'22"S. Whereas at its deepest point, it is the deepest in the world oceans, being about 5600 m around the SWIR.

It is a system of ridges that divides the Indian Ocean seafloor into three different tectonic plates, separating (i) the Antarctica and Indian-Australian plate, (ii) the Arabian plate, and (iii) the East African plate, through three actively spreading plate boundaries. The ridge system meets at a triple point known as the Rodrigues Triple Junction (~ 25°33S, 70°E) nearly in the centre of the Indian Ocean. A brief characterization of these ridges is presented on the next pages.

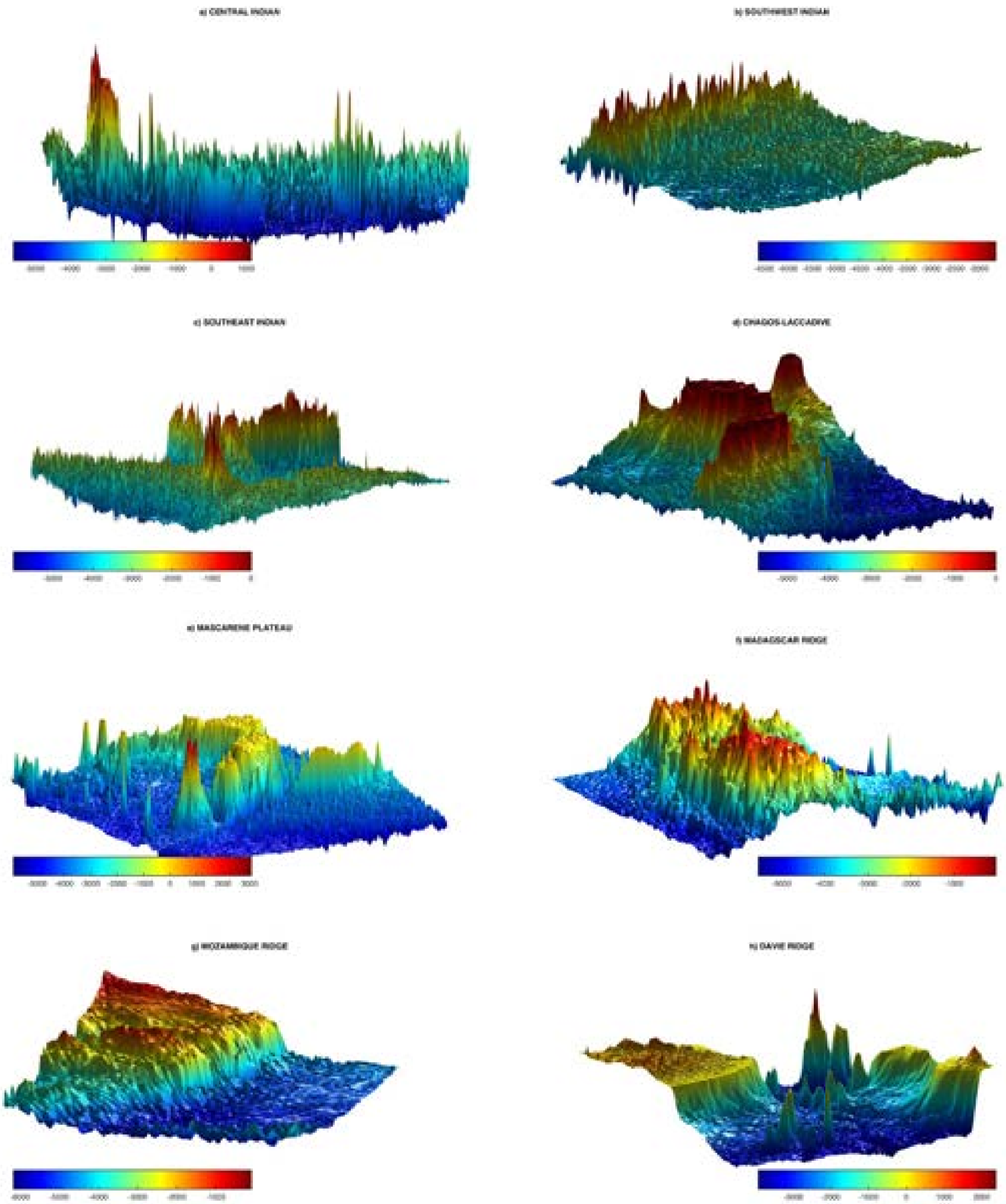


Figure 3: Morphological profiles of the major oceanic ridges in the WIO region, discussed in this chapter. See the titles for better reference. Notice that the colour bar is not fixed for all the panels. It is not meant to compare them as in some cases the islands and parts of the continent are shown, whereas in other cases it is not. Positive (negative) in the colour bar indicates land elevation (ocean depth), respectively, expressed in meters. For more information, to compare each ridge shown here, seethe display presented in Fig. 1 and Fig. 5 for better geographic information, and their normal orientation in relation to the coordinate systems.

Central Indian Ocean Ridge

The CIR is a very long feature that stretches from the Rodrigues Triple Junction, going northward nearly on a meridional direction (Fig. 3a). After crossing the equator, it bends slightly to run along a north-westward direction into the Arabian Sea, where it joins the Carlsberg Ridge (Spencer et al., 2005). New oceanic crusts are known to be created along this ridge. Inspections on the segmentation and morphology of CIR using multibeam bathymetry and magnetic data over a stretch between 3°S and 11°S (Raju et al., 2012) suggests that CIR is slowly spreading with an average full spreading rate between 26 and 38 mm year^{-1}. On the stretch nearly between 20°S and the Equator, it reaches its highest elevation running through the remote volcanic islands of New Amsterdam and Saint Paul (Defant, 1961).

While isolated from human interferences it has been documented that these islands are under threat from invasive alien fauna, flora and pollution brought about by the oceanic current system. As indicated before, the southernmost point of the CIR is at Triple Junction. From this point, running in a southwestward direction lies the SWIR.

Southwest Indian Ridge

The SWIR is a prominent plate boundary that separates the African continent from Antarctica (Fig. 3b), over 100 Ma, and has an extension of over 77 000 km, between Rodrigues Triple Junction and Bouvet Triple Junction at 55°S, 0.5°W (Bernard et al., 2005). It has been distinguished as the deepest ridge in the world ocean basins, and one that has an ultra-slow spreading rate, estimated in 15 mm year^{-1} (Patriat et al., 1997). It consists of magmatic spreading segments alternated by amagmatic segments, obliquely oriented (Baines et al., 2007). The Ridge is also characterized by several transverse fractures of enormous width and depths range (Parson and Evans, 2005). It has been indicated that strong evidence exists supporting that the SWIR can be divided in two parts: between 33–35°E and 35–43°E, exhibiting different spreading rates. From the Triple Junction running south-eastward lies the SEIR, presented below.

Southeast Indian Ridge

Extending from the Rodrigues Triple Junction to Macquarie Triple Junction (63°S, 165°E), the SEIR (Fig. 3c) stretches for about 6000 km (Graham et al., 1999). The SEIR has been described as the fastest spreading ridge when compared to the SWIR and CIR (Royer and Schlich, 1988). Small et al. (1999) indicate its spreading rate ranges between 59–75 km Ma^{-1}. It has been also inferred that this Ridge has the spreading memory of the movements of the Antarctica continent in relation to Australia and India since the late Cretaceous (Royer and Schlich, 1988). The SEIR is comprised of a variety of distinct morphology and segmentations. It is a primary place for basaltic magmatism, transporting the Indian Ocean Isotope in-print which is determined by its relatively high ratio 87Sr/86Sr, 207Pb/206Pb, and 208Pb/206Pb (Graham et al., 1999). SEIR runs through a rough topographic path, near islands (eg Kerguelen-Heard, Amsterdam, St. Paul), ridges (eg Ninety-East Ridge) and plateaus (eg Kerguelen).

As seen from Fig. 1, apart from the Mid-Ocean Ridge System, the WIO region is also further complexed by several other ridge structures. Along the meridional extension of the CIR there are two outlying ridges: to the east, being the Chagos-Laccadive Ridge (CLR) (Fig. 3d), and to the west, the Mascarene Plateau (MASP) (Fig. 3e), of which the descriptions are presented next.

Chagos-Laccadive Ridge

CLR is a notorious aseismic volcanic ridge in the northern Indian Ocean (Fig. 3d) that stretches along a north–south distance of about 2000 km (Parson and Evans, 2005). Starting from the south-west Indian continental shelf, it runs south, carrying the Chagos Archipelago, passing through the Maldives and Laccadive, thus providing to the Ridge three extending blocks, termed in the literature as the Southern, Middle and Northern blocks, which represent the Chagos, Maldives and Laccadive blocks respectively (Nair et al., 2013).

The CLR is characterized by several fracture zones along its margins, with a north-south orientation, comprising an array of segments of different origins (Avraham and Bunce, 1977), of which some parts are volcanic while others are of continental origin rifted from India. It is important to note that the literature fails to present a conclusive agreement on the actual origin of this Ridge, with several authors suggesting different processes and contrasting explanations. The CLR is described as being asymmetric due to its eastern flank being steeper than the western one. The average depth of the CLR is known to be less than 1000 m, comprising of coral atolls and volcanic islands (Nair et al., 2013). Observational drilling data supports the possibility that the northern portion of the CLR is thicker than a normal oceanic crust. This being true, then it confirms that it has originated from a hotspot locus (Nair et al., 2013).

Mascarene Plateau

The MASP has been referred in the literature as one of the most prominent shallow bathymetric features of the Indian Ocean (Parson and Evans, 2005). The MASP is crescent-shaped, concaved to the east, stretching between 4°S–20°S, between the islands of Seychelles and the Mauritius respectively, covering a distance of about 2000 km (Fig. 3e). It is located nearly across a zonal stretch lying within 54°E and 63°E. The description presented by Defant (1961) infers that it comprises two shallow branches: the north-west branch that hosts the Seychelles (Fig. 2h) and the Amirante Islands, of continental origin (Seychelles Archipelago), and the other branch to the south-west which is of volcanic origin that hosts the volcanic islands of Mauritius and Reunion (Fig. 2e).

The MASP supports several shallow banks and shoals, fractured by deep-reaching channels (~ 1000–1500 m). The water depth over the banks can be as shallow as 20 m (New et al., 2005). They are floored by corals, and occasionally can out-crop the sea-surface to form small scale islands (New et al., 2007). The flanks of the plateau in some cases descend abruptly to depths of about 2000–3000 m, and in other cases it gently descends to depths of about 4000 m. New et al. (2007) indicates that between the Seychelles and the Saya de Malha Bank exists a 400 km long topographic ridge below 1000–1500 m depth. Furthermore, New et al. (2007) also identify a narrow gap of 100 km between the banks of Saya de Malha and Nazareth, located between 12°S and 13°S, deepening to over 1000 m.

In addition, the description by New et al. (2007) also reveals the existence of a shallow and narrow channel near 16°S, of about 200 m deep and less than 50 km wide between the Nazareth and Cargados-Carajos Banks. Between the latter and Mauritius also exist two deep and narrow channels of about 2000–3000 m and of about 50 km width respectively, along a meridional stretch between 18°S and 19°S. It is expected that this complex bathymetric configuration has strong implications for the hydrodynamic nature of the ocean circulation in the area.

Madagascar Ridge

A comprehensive characterization of the topographic structure of the Madagascar Ridge (MDR) can be found in the study by Goslin et al. (1980), from which this section is based. The MDR is an elongated topographic barrier, southward oriented (Fig. 3f), which stretches directly from the southern tip of Madagascar at about 26°S, and terminates at about 36°S (Fig. 1a), where it collides against the SWIR, covering a meridional distance of about 1300 km. It is extensively characterized by a localized complex assemblage of Precambrian to Holocene continental igneous and sedimentary rocks. The geometry of the MDR suggests it has a maximum width of about 750 km at latitude of 32°S (Goslin et al., 1980).

Interestingly, the latitude of 32°S also sets the mark that divides the MDR in two distinct domains: to the north, more precisely to the north of 31°S, it is characterized by small sediment-filled pockets between a number of basaltic sediment highs. In this part, the western sector of the MDR is delineated by large-scale normal faults. Whereas to the east, the late Cretaceous fracture zones of the Madagascar Basin advances with deep penetration into the MDR. To the South of 32°S, it is characterized by an extensive region of thick undeformed sediments across the central part of the MDR. The maximum depths in the centre of the plateau ranges between 1500 and 2000 m, excluding localized seamounts or shoals where a minimum depth of 20 m has been observed over the Walter Shoal, situated on the southwestern portion of the ridge. Both the western and eastern flanks of the MDR are steep and descend to depths below 5000 m into the Mozambique and Madagascar Basins, respectively (Fig. 1a).

Mozambique Ridge

Details about the morphological characteristics and geological processes (magmatism and volcanism) occurring in the Mozambique Ridge (MZR) and its Basin has been published by Köning and Jokat (2010). The MZR is an elongated feature, north-south oriented, of about 1300 km long (Fig. 3g). It lies roughly parallel to the south-east coast of Africa (Fig. 1a), between the parallels of 25°S and 35°S, and meridians of 34°E and 36°E (Maia et al., 1990). The MZR is bound to the east by a deep, nearly linear scarp that steeply descends into the Mozambique Basin, for over 5000 m. Whereas to the west the flanks of the MZR descend gently onto the Transkei Basin, also known as the Natal Valley (Fig. 1a, Fig. 2b). The MZR has been recently reclassified as being of oceanic origin (Köning and Jokat, 2010), and being comprised of several bathymetric plateaus which rise from the seafloor to heights of about 3500 m (Köning and Jokat, 2010), leading to a shallow water depth over the MZR crest.

Contrary to former conceptual descriptions, the tectonic model results presented by Köning and Jokat (2010) suggest that the present location (off south-east Africa) of the MZR has been determined by long-lasting volcanic activities during the initial phases of separation between the African continent and Antarctica (Fig. 1a).

Davie Ridge

The Davie Ridge (DVR) is a shallow bathymetric feature located in the Mozambique Channel, between Mozambique to the west and Madagascar Island to the east (Fig. 3h). The DVR runs in a north-south orientation and is about 1200 km long (Müller, 2017). The water level above the DVR is variable, where shallowest depth observed is 20 m over the Mount Saint Lazare. The DVR shows an asymmetric profile, with its western flank progressing more steeply than its eastern flank, which shows a relatively gentle incline. This north-south orientation has a strong impact on the hydrodynamic regime of the bottom water circulation and sediment transports within the Channel (Wiles, 2014). The DVR has been described as a curvilinear fracture zone that facilitated the southward drift of Madagascar away from the African main continent between the late Jurassic and early Cretaceous geological scales (Bassias, 1992). It separates two important oceanic basins: Mozambique Basin to the south and Somalia Basin to the north (Fig. 1a).

DEEP OCEAN CHANNELS AND OCEAN BASINS

The seafloor topography of the WIO hosts seven major deep ocean basins namely, Arabian Sea Basin (ASB), Somali Basin (SMB), Central Indian Basin (CIB), Mascarene Basin (MSB), Madagascar Basin (MDB), Mozambique Basin (MZB), and Crozet Basin (CZB). Formed by distinct geologic processes (origins), each has a different geologic pattern (Fig. 1a).

Mozambique Channel

The Channel lies between the coast of Mozambique on the west side and the west coast of Madagascar to the east (Fig. 1a, Fig. 2d). The northern end of the Channel extends from the Rovuma estuary (10°28'S and 40°26'E) to Ras-Habu, at the northern point of Comoro Islands (Fig. 1a, Fig. 2e), to the northern tip of Madagascar at Cape Amber (11°57'S, 49°17'E). The southern boundary stretches from Ponta do Ouro on the southern Mozambique mainland (26°53'S, 32°56'E) to Cape Sainte-Marie, at the southern tip of Madagascar (Fig. 2e).

Arabian Basin

The Arabian Basin is a submarine basin of the southern Arabian Sea (Fig. 1a), which rises toward the submerged Carlsberg Ridge to the south, the Maldive Islands to the south-east, India and Pakistan to the north-east, Iran to the north, and the Arabian Peninsula to the west. The Basin has a maximum depth of 5875 m and is separated by the Carlsberg Ridge from the deeper Somali Basin to the south and west. The sill depth between the Arabian and Somali Basins is 3000 m. The floor of the basin, except along the south-eastern edge, is covered by sediment deposited by the Indus River in the form of a large alluvial fan. The northern Arabia Sea is dated to 40 Myr.

Somali Basin

The Somali Basin is a submarine basin located in the south-western sector of the Arabian Sea, in the tropical north-west Indian Ocean, to the east of Somalia (Fig. 1a). The Carlsberg Ridge separates the Basin from the shallower Arabian Basin to the north-east. The Basin connects with the Mascarene and Madagascar Basins to the south, with sill depths of more than 3600 m. The deepest sections of the Basin are about 5100 m, and the seafloor is dated to 66 Myr (Parson and Evans, 2005).

Mascarene Basin

On its northern part, the Mascarene Basin lies between Madagascar and the Seychelles Plateau in the north-west Indian Ocean (Fig. 1a). At its north-western-most part, it is defined by the complex ridge trench Amirante Arc (Masson, 1984). To the south it connects to the Madagascar Basin. The Basin has depths of about 5000 m, with deep floor areas aged to 76 Myr (Parson and Evans, 2005).

Madagascar Basin

The Madagascar Basin lies to the east of the Mozambique Basin (Fig. 1a), and is bounded by the south-eastern coast of Madagascar and the MDR in the western side, and to the east by the Mid-Indian Ridge. In the north it is limited by the MASP, while to the south it is limited by the SWIR. The seafloor is relatively younger than that of the Mozambique Basin, being dated to 75 Myr (Parson and Evans, 2005) and characterized by sandy silt sediments.

Mozambique Basin

The Mozambique Basin is located between South Africa and the Mozambique Ridge to the west and to the east by Madagascar and the MDR (Fig. 1a). The Basin is relatively shallower in the north, at the southern entrance of the Mozambique Channel, and deeper in the south. Maximum depth of the Basin reaches beyond 5000 m at its southern entrance. Its bottom floor is dated to 110 Myr (Parson and Evans, 2005).

OCEANIC AND ATMOSPHERIC PROCESSES IN THE WESTERN INDIAN OCEAN

The ocean and atmosphere form a dynamic coupled-system, with a strong interaction between them through the upper boundary layer turbulent fluxes. The processes in each of these environments cannot be fully understood if they are regarded separately as independent entities.

Large-scale atmospheric circulation

The atmospheric circulation in the WIO, and by extension the whole Indian Ocean, is unique when compared against that of the Atlantic and Pacific Oceans. Fig. 3 (Halo and Raj, 2020) shows wind stress and windstress-curl and monthly climatology for the WIO. The product is gridded on ¼°x¼° mesh over the global ocean. It is known to resolve/reproduce small scale (mesoscale) features otherwise impossible in other wind products such as NECP re-analysis.

In the map (Fig. 4), positive (negative) windstress curl in the northern hemisphere represents the Ekman suction (pumping) phenomena. In the southern hemisphere the opposite holds. Ekman suction suggests water is pushed upward in the base of the Ekman layer, resulting in an upwelling event due to divergence at the sea-surface. Conversely, Ekman pumping implies water being pushed downward from the base of the Ekman layer, resulting on downwelling event through the water column, due to convergence at the sea-surface. The upward and downward movement of the water column is generally translated into the movement of the thermocline, which in the WIO region is located at depths identified by the isotherm of 20°C.

Looking at the windstress in Fig. 4, it is evident that in the WIO region, a marked contrast between the southern and northern Indian Ocean surface wind patterns is observed at latitude of 10°S. To the south, the winds are predominantly easterly trade winds almost all year around, while to the north of that latitude the winds vary strongly seasonally due to the Indian monsoons (Schott et al., 2009).

A noticeable change of the windstress direction during the full course of the year is evident to the south of Madagascar, where winds change from north-easterly to easterly (Fig. 4). Former studies in the region have attributed this forcing as partially contributing to upwelling events in the region (Lutjeharms and Machu, 2000; Machu et al., 2002; Ho et al., 2004; Ramanantsoa et al., 2018). In the central Mozambique Channel, the winds are predominantly south-easterly all year around, but considerable changes are observed on the strength of the windstress curl. It is mostly positive from July to November (Fig. 4g–k), and more negative from January to March (Fig. 4a–c).

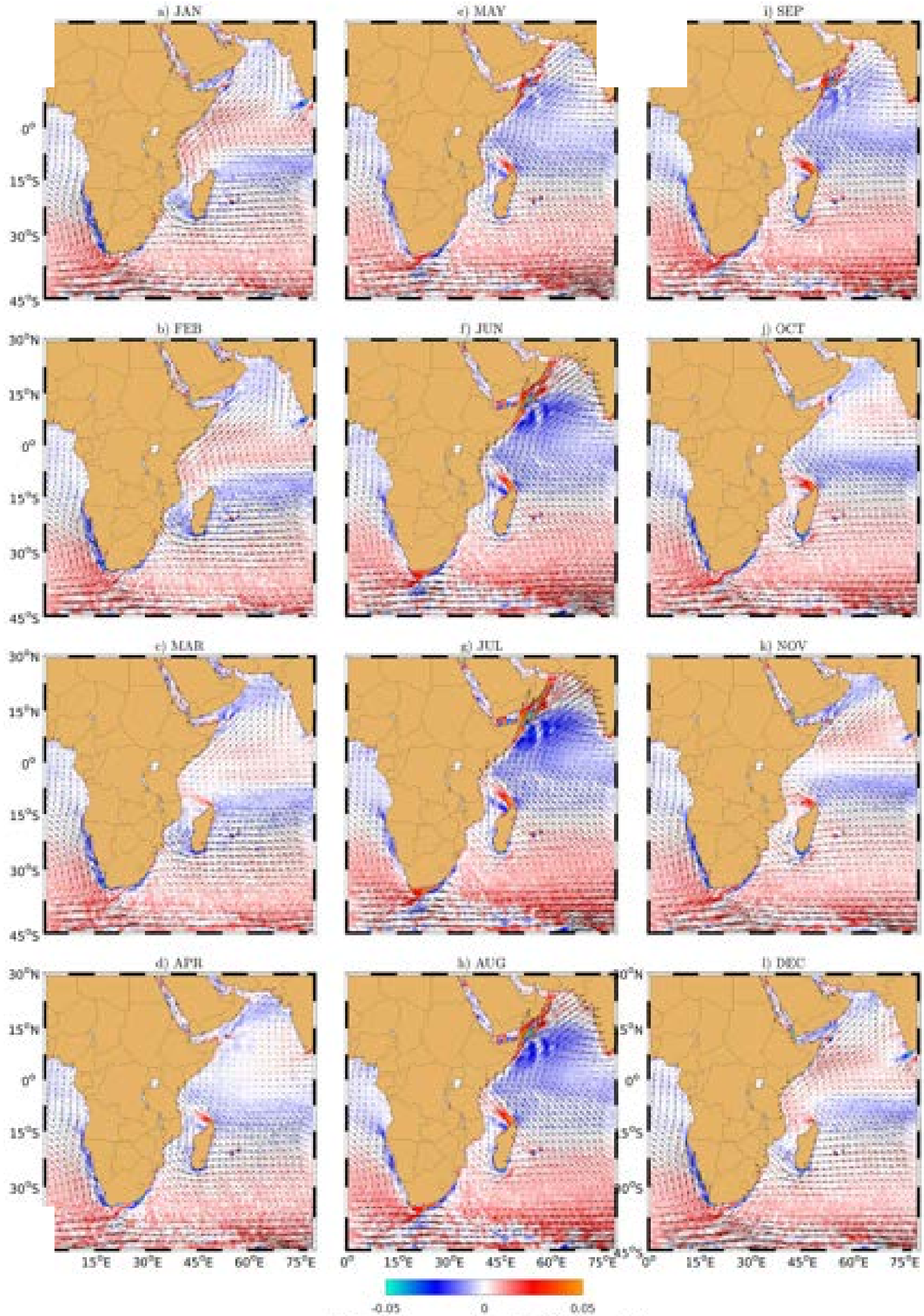
Monsoons

Monsoons are seasonally reversing winds dominant in the north Indian Ocean, caused by the different warming rates of the earth's surface between the Indian Ocean and the highlands of India's interior region. During the months of November to March (Fig. 4k–c), the winds blow from the India landmass to the sea (north-east monsoon, north Indian Ocean winter monsoon), and from May to September (Fig. 4e–i) they blow from the ocean toward the Asian continent (south-west monsoon). The months of April (Fig. 4d) and October (Fig. 4j) are the period of the transitions between the monsoons and are known as inter-monsoons. In addition, the equatorial trade winds are exceptionally weak, and unsustained.

Peculiarities in the proximities of the Equator are further revealed from the fact that the near equatorial winds have their easterly component during the late winter to/or beginning spring only, while the westerlies are semi-annual during the transitional phase of the two monsoons (Schott et al., 2009). During these transitions, strong eastward equatorial surface jets known as Wyrtki Jets strike. This provides significant contribution to the onset of Indian Ocean climate mode of variability, such as Indian Ocean Dipoles (IOD).

Large-scale oceanic circulation

The large-scale surface circulation in the WIO is shown in Fig. 5 and Fig. 6. The former schematic suggests the direction of the flow field, and the latter indicates the intensity (speed) of the currents, computed from CNES-CLS9 dataset. The circulation pattern includes a complex system of currents and counter-currents dominating both



Windstress Curl (N. m^{-3})

Figure 4: Monthly climatologies (January to December) of the windstress (vectors, expressed in N m^{-2}) and winstress-curl ($\nabla \times \tau$) (colours, expressed in N m^{-3}) derived from the Scatterometer Climatology of Ocean Winds (SCOW), described by Risien and Chelton (2008). Notice the influence of the monsoons expressed through the reversal of the wind directions between January (a) and July (g). Figure extracted from Halo and Raj (2020).

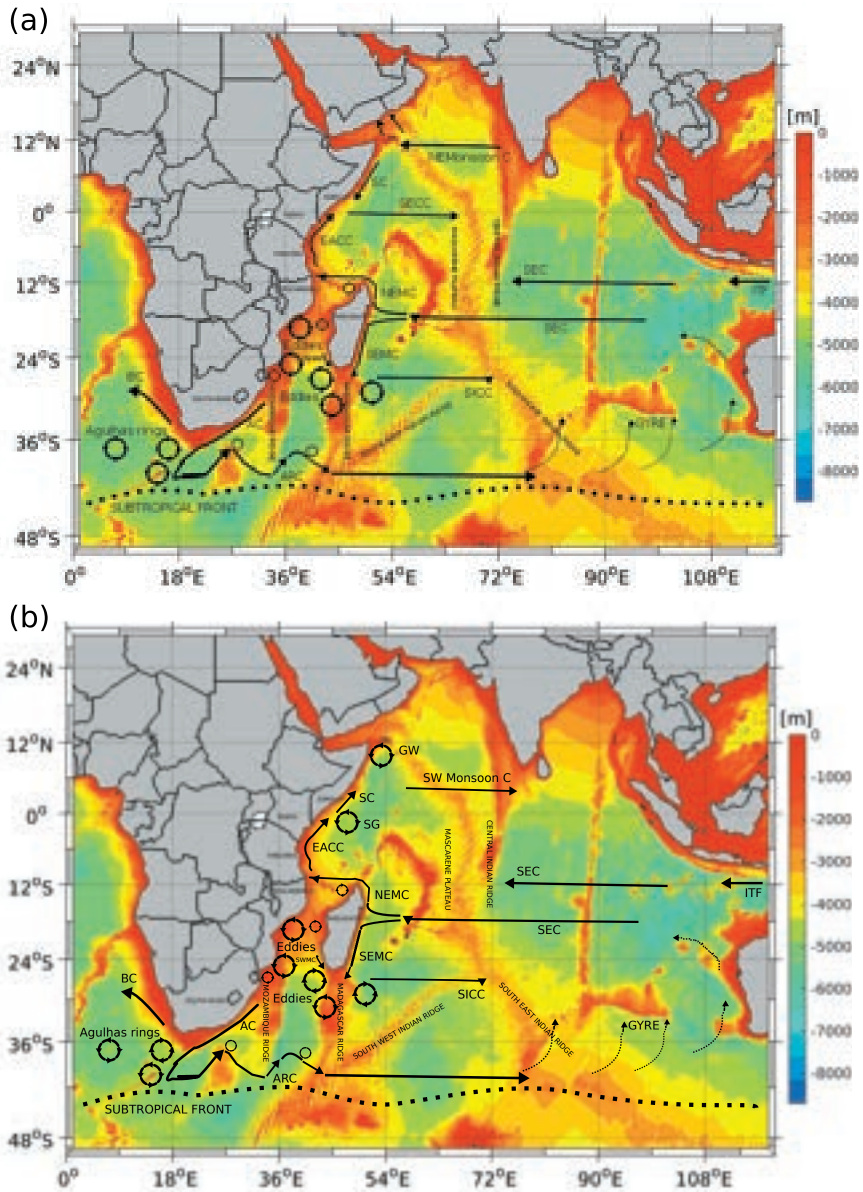


Figure 5: Schematic of the surface currents in the Indian Ocean during different phases of the monsoonal wind: a) Circulation during the south-west monsoon, and b) During the north-east monsoon. The background colour shows the seafloor topography, with main bathymetric features indicated by their respective labels.

Courtesy: Cedras et al., 2020; Halo and Raj (2020), adapted from Schott et al., 2009.

For more information to interpret Fig. 5, see Fig. 6 below where the intensity of the currents has been estimated with long-term oceanographic data.

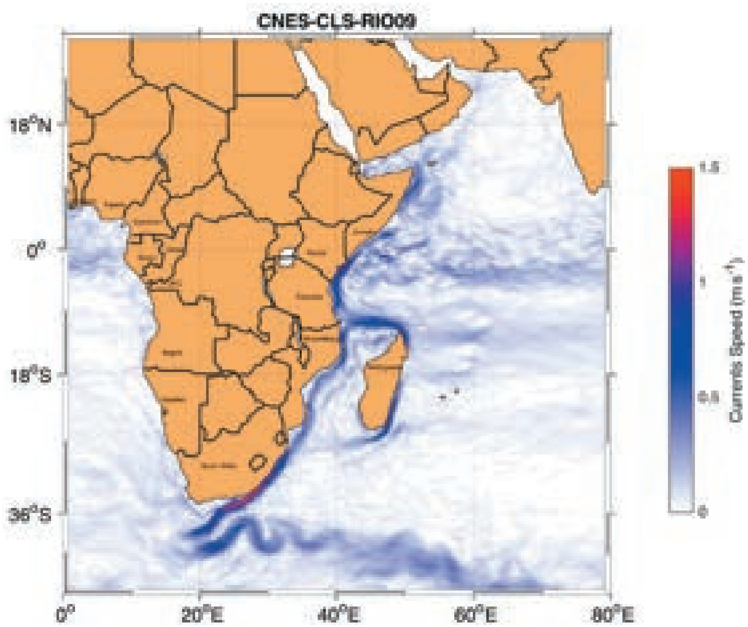


Figure 6: Magnitude of the mean surface currents (speed in m s^{-1}) in the WIO, based on Mean Dynamic Topography dataset (CNES-CLS09). It reveals strong intensification of the boundary currents, with the peak in the core of the Agulhas Current on the eastern flank of the Agulhas Bank and eastern Cape province. Note the main oceanographic features of the circulation.

For more information to interpret Fig. 6, see the schematic portray in Fig. 5 above.

the northern and southern hemispheres and are strongly influenced by the atmospheric wind system forcing. The main current structure is the westward South Equatorial Current (SEC), that originates its waters from the western Pacific Ocean in the form of the Indonesian Throughflow (ITF) (Fig. 5) and the recirculation of the subtropical anticyclonic gyre of the south Indian Ocean (Stramma and Lutjeharms, 1997). The SEC propagates westward carrying slightly more than 50–55 Sv of volume transport between 10°S–20°S, before it starts to split-off into two branches when crossing through the confines of the gaps between the banks of the Mascarene Plateau (New et al., 2007). On reaching the east coast of Madagascar, near 17°S (Fig. 5), the branches of the SEC are fully decoupled into two opposing branches parallel to the Madagascar coast, and receive their individual identities, namely, South East Madagascar Current (SEMC) propagating southward carrying a volume of about 20 Sv (Schott et al., 1988), and Northeast Madagascar Current (NEMC) due north, transporting about 30 Sv (Chapman et al., 2003; Schott and McCreary, 2001; Schott et al., 2009). The latter, after passing the northern tip of Madagascar, creates horizontal shear at its southern edge interacting with the island's topography (to be discussed later). The bulk of the NEMC added to the northern remains of the SEC propagate westward toward Africa's mainland along the latitude of 12°S.

On reaching the African continent, the flow splits into another two branches (Fig. 5) with contrasting and distinct dynamical patterns over space and time-scales. The southward branch carries about 18 Sv of the volume transport and propagates discontinuously through the whole length of the Mozambique Channel in the form of mesoscale eddies (De Ruijter et al., 2002; Ridderinkhof and De Ruijter, 2003; Halo et al., 2014a). Meanwhile, the northward branch transports about 15 Sv of the water volume and flows continuously as East African Coastal Current (EACC) along the coasts of Tanzania and Kenya (Schott et al., 1988). The former merges with the flow spawned at the southern termination of the SEMC, at the southern tip of Madagascar, in the southern Mozambique Basin (Stramma and Lutjeharms, 1997). Subsequently, they become important sources of water and variability to the Agulhas Current (Lutjeharms, 2006).

The Agulhas Current (AC) is one of the most intense western boundary currents of the global ocean, which transports on average about 70 Sv, and propagates with an average speed of more than 1.5 m s^{-1} (Lutjeharms, 2006). The vigorous nature of the AC in relation to all other currents in the WIO region also can be perceived from Fig. 6. The AC terminates at the south-western most tip of the Agulhas Bank, where it revolves on itself and flows back into the south-west Indian Ocean in form of the Agulhas Return Current (ARC) (Fig. 5 and Fig. 6), carrying about 55 Sv of water transport (Lutjeharms and Ansorge, 2001), which recirculates back into the southern Indian Ocean anticyclonic subtropical gyre.

The EACC along the coast of Kenya supplies the Somali Current (SC). However, this supplement is conditional, ie seasonally (monsoon) dependent (Tomczak and Godfrey, 1994; Schott and McCreary, 2001; Schott et al., 2009). During the south-west monsoon, the EACC feeds the SC, and the flow progresses northward, crossing the Equator, and later at about 4°N, it bends eastward and propagates as the North Equatorial Countercurrent (NECC), that feeds the westward Southwest Monsoon Current (SWMC) in the Arabian Sea. Whereas during the north-east monsoon, the Somali Current propagates southward along the coast, and confronts the EACC at about 2°S–4°S. This confluence of the currents induces an eastward flow that supplies water toward the South Equatorial Countercurrent (SECC). SECC is known to flow eastward all year around (this feature also can be traced in Fig. 5), simply exchanging its position with reference to the water column (deepening into the sub-surface during the south-west Monsoons, thus becoming an equatorial undercurrent). During this period the monsoon current in the Arabian Sea propagates westward as the North Monsoon Current (NMC).

In the southern hemisphere, to the south-east of Madagascar near the latitude of 25°S, an intriguing shallow South Indian Ocean Countercurrent (SICC) flows eastward transporting about 21 Sv (Siedler et al., 2006; Palastanga et al., 2007; Siedler et al., 2009). This flow was discovered almost one decade ago only. More recently, to the south-west coast of Madagascar another current has been uncovered. This current flows southward parallel to the coast, and it has been termed South-West Madagascar Current (Ramanantsoa et al., 2018).

Perhaps the most important aspect to highlight in Fig. 6 is the importance of incorporating the long-term maps of mesoscale dynamic topography derived from satellite altimetry. The data portrays features of the circulation known to be in geostrophic balance. They are computed from the horizontal components of the momentum equation of the fluid dynamics, whereby the Coriolis force (associated with the rotation of the earth) is in dynamic balance with the pressure gradient force, while all the other terms of the equation are assumed to play a negligible effect. From the patterns shown in Fig. 6, it is evident that most of the above-described surface features (Fig. 5) of the circulation in the WIO are in geostrophic balance. It is also interesting to highlight the presence of mesoscale features (eg Great-Whirl eddy off northern Somali coast; Mozambique Channel eddies and Agulhas rings, though with a relative weaker imprint; and the equatorial waves) in this large-scale time-averaged dataset of the mean flow.

OCEANIC MODES OF VARIABILITY

The most dominant modes of climate variability in the Indian Ocean are the monsoons, the Indian Ocean Dipole (IOD), the Indian Ocean Basin-wide (IOB) warming and the El-Niño Southern Oscillation (ENSO) (Saji et al., 2006).

El-Niño Southern Oscillation

ENSO is a natural phenomenon characterized by strong warming and cooling events of the sea-surface on the eastern equatorial Pacific Ocean and changes in the zonal pressure gradient on the western equatorial Pacific, which gets phased with the annual cycle of the Pacific SST anomalies. It shifts the Pacific atmospheric convection eastwards, while intensifying at the central-eastern equatorial region. This shift modifies atmospheric circulation remotely in the tropics and extra-tropics through an atmospheric wave adjustment mechanism. This anomalous warming enters the equatorial Indian Ocean and reaches the WIO (Fig. 7a–c). Thus, the WIO regions warms up during the ENSO periods, with maximum temperatures being observed from March to May. However, there are different mechanisms responsible for this warming. The leading mechanism in the tropical south-west Indian Ocean differs from that observed in other regions of the basin. For example, in the tropical south-west Indian Ocean processes within the ocean interior dominate, while in the rest of the Basin surface fluxes dominate (Schott et al., 2009). ENSO plays a critical role on the degradation of coral population and the hydrology in the WIO region (see the case study presented earlier in this chapter).

Anomalous atmospheric perturbations in the form of anticyclonic wind stress curl in the tropical east Indian Ocean (due to changes of the atmospheric Walker circulation), excites a downwelling of Rossby waves that propagate westward. Upon their arrival in the tropical south-west Indian Ocean (after many months), they force both deepening of the thermocline and warming of the sea-surface. In the WIO region, and by extension for the whole Indian Ocean, the most severe ENSO event occurred in 1997 (Westerberg and Christiansson, 1999). The WIO region experienced severe rainfall and flooding events which caused many deaths and displacement of thousands of people, while the eastern Indian Ocean experienced severe drought, which led to many fires, causing likewise loses (see the case study presented earlier in this chapter).

Indian Ocean Dipole and the Indian Ocean Basin-wide (IOB) warming

In response to the westerly equatorial wind forcing, the oceans respond by triggering accelerated flows in only a few days (jets). In the equatorial Indian Ocean, these occur during the transitional phases of the monsoons. These Wyrtki Jets (Wyrtki, 1973) move warm equatorial surface waters eastward, pilling-up in the eastern Indian Ocean, resulting in increased sea-level and thickness of the mixed layer in the ocean interior, as it deepens the thermocline. However, in the WIO, it generates cooling events due to the onset of a shallower thermocline. Thus, these jets become key role players contributing to the onset of the IOD, by virtue of weakening or eliminating the upwelling along the coast of Sumatra (Schott et al., 2009).

IOD is a natural coupled ocean-atmosphere event that usually develops during the month of June, and reaches its maximum peak in October, caused by a strong seasonal variability of the monsoonal winds that favours the occurrence of Bjerknes feedback in the eastern Indian Ocean during the summer and fall seasons, which occasionally sets the developments of oceanic-atmospheric anomalies of similar nature to the La-Niña phenomenon (Schott et al., 2009). The cooling of the zonal equatorial gradient of sea-surface temperature is coupled with the shoaling of the thermocline. With the development of the IOD, a zonal east-west dipole of anomalous rainfall strikes the tropical Indian Ocean. IOD is characterized by a strong increase in rainfall events in the WIO region. The IOD also influences the intensity of the ocean currents depending on its positive or negative phase translated by a weakening or strengthening, respectively (Palastanga et al., 2006). During the IOD positive phase in 1994 and 1997 the measured ocean currents in the WIO, namely, the SEC, NEMC, SEMC, EACC and the eddy field in the Mozambique had weakened, whereas during its negative phase in 1996 and 1998 they had intensified (Palastanga et al., 2006).

The IOD, IOB and ENSO events are depicted in Fig. 7. The IOB is not further discussed here as its signal can be damped by the amplitude of the ENSO variability. Fig. 7 was extracted from the work by Wieners et al. (2019) and constructed from a set of observational datasets

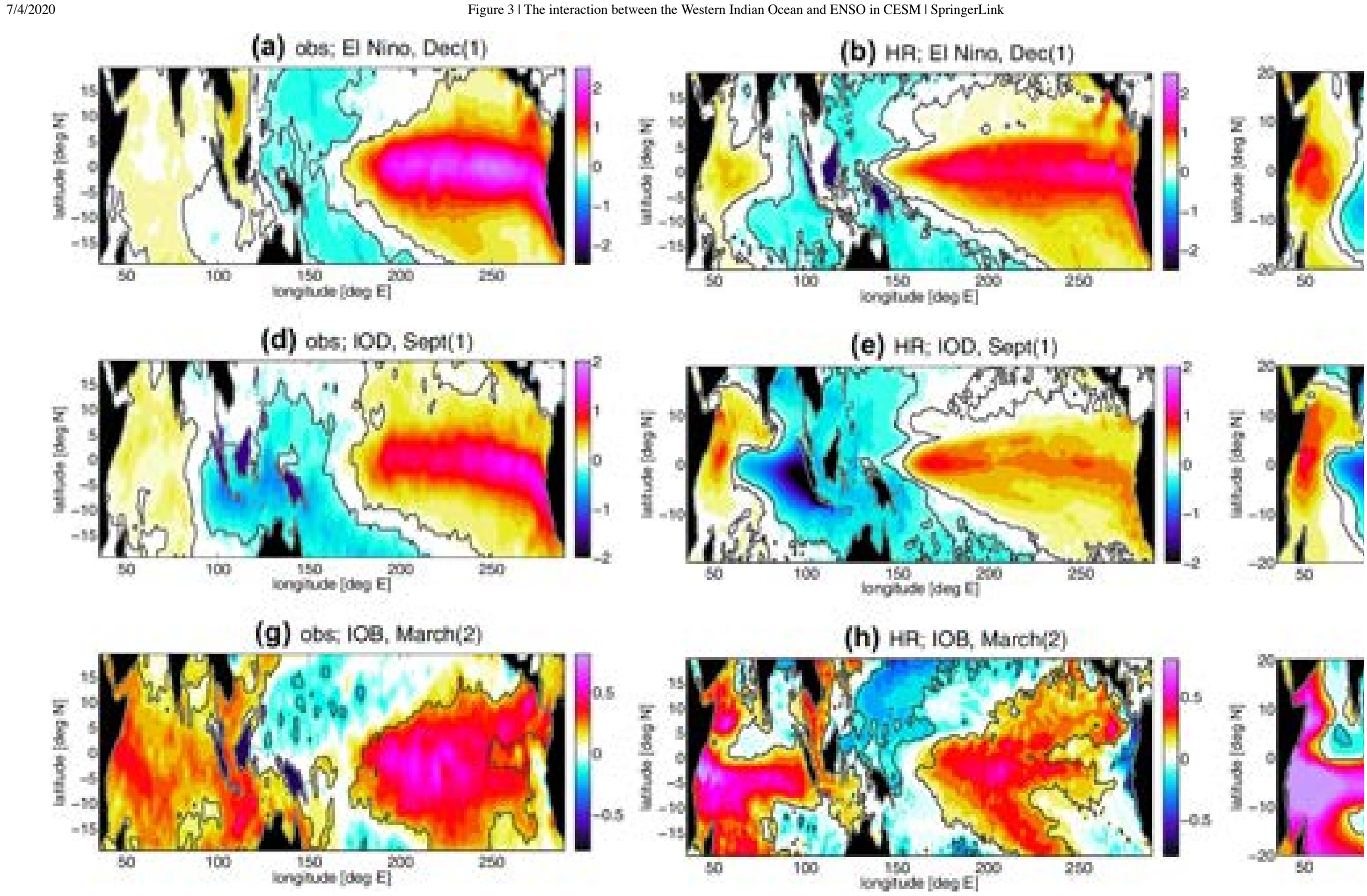


Figure 7: Anomalies of sea surface temperature composites El Niño (a–c), Indian Ocean Dipole (d–f) and Indian Ocean Basin (g–i) in the Pacific and Indian Ocean Basin-wide. The left panels represent the phenomenon observed, and the middle panels shows their respective simulation in a numerical model at higher resolution (0.1°) grid cell, and the right panels at relatively lower resolution (1°) grid cell. Note that only specific months (December, September and March) have been used, representing the stronger ENSO and positive IOD phase.

Figure extracted from Wieners et al., 2019.

(Fig. 7, left panels) and numerical simulations by the climate model the Community Earth System Model (CESM), in two experiments with different grid resolutions for the oceans and atmosphere. The higher resolution (HR) is at 0.1° grid cell (Fig. 7, middle panels), and the lower resolution (LR) is at 1° grid cell (Fig. 7, right panels). The top panels (Fig. 7a–c) represent the peak of the ENSO, occurring during the month of December, computed from composite SST anomalies. Noticeable are the warm anomaly (in the Pacific Ocean) and the cold anomaly (eastern Indian Ocean) tongues. Figure 7, middle row (Fig. 7d–f) also shows the IOD event during its positive phase (September). This plot has been computed from composite anomalies, also from observations, and HR and LR model simulations. The bottom panels represent the IOB distribution (Fig. 7g–i).

Overall, Fig. 7 highlights the level of interactions of the co-occurrence between ENSO and the positive phase of IOD events, and vice-versa (Wieners et al., 2019).

Mesoscale variability

Monthly climatological maps of sea level anomalies (SLA) for 24 years of observation, 1993–2016, over the full annual period (January–December) are displayed in Fig. 8. The daily data was gridded in a regular grid of ¼°x¼°, across the global ocean. The dataset is produced by the French institutions: National Centre for Spatial Studies (CNES), Collecte Localis Satellites (CLS) and freely distributed on-line via the Copernicus Marine Services[5].

The patterns in Fig. 8 reflect a strong seasonality, with significant regional and local variabilities. Furthermore, the differences of the geometrical structures represented by the closed contours of SLA are also remarkable. Larger and predominantly zonally elongated features dominate the north-west Indian Ocean, while more circular geometric structures are dominant in the south-west Indian

5. http: //marine.copernicus.eu/

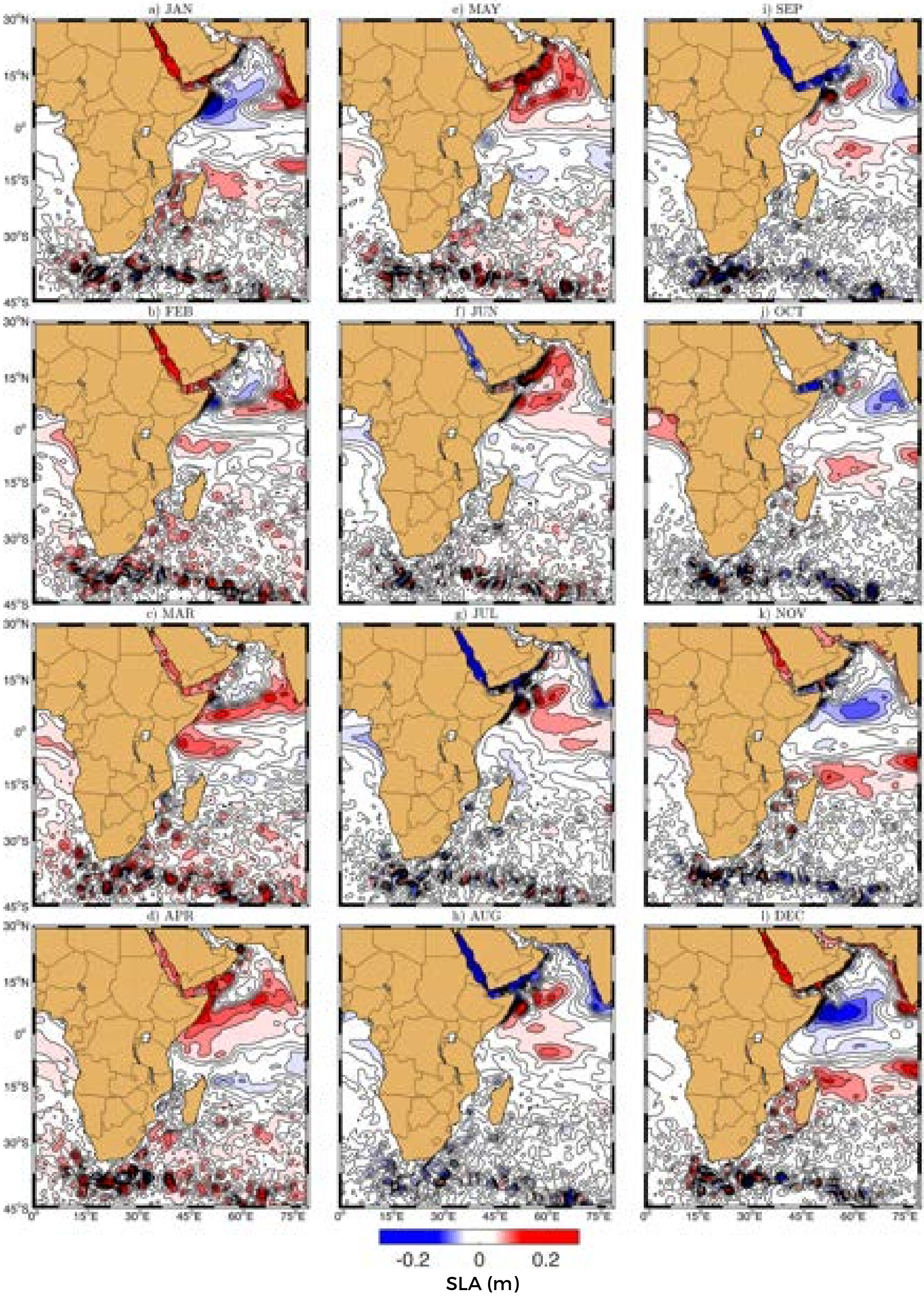


Figure 8: Monthly mean anomalies of the sea surface height (SLA) throughout the annual cycle. The dataset used is the satellite derived altimetric delayed time product for the period starting in 1993 to 2016. The dataset was downloaded from https://resources.marine.copernicus.eu/

Ocean. Overall, the northern Indian Ocean suggest stronger seasonal differences than the southern hemisphere. The same is true within the northern hemisphere itself between the eastern and western boundaries of the Arabian Basin.

On climatological timescales, from November to February, sea level falls within the north-east monsoon period, during which a remarkable larger and intense lower (negative) SLA is positioned along the northern coast of Somali (Fig. 8k, l–a), exactly where the Great Whirl eddy develops and resides (see Fig. 6). This pattern shifts to a strongly higher (positive) SLA from May to September (Fig. 8e–i) when the winds are from south-east (see Fig. 4). These patterns are consistent with positive windstress curl adjacent to the northern Somali coast which results in surface divergence and Ekman driven upwelling during the north-east monsoon, which in turn gives way to surface convergence thus downwelling during the south-east monsoons.

A remarkable feature worthy of highlighting in Fig. 8, in the greater Agulhas system, is the seasonality of the strength of the SLA. Higher SLA have been observed from November to March (Fig. 8k–c), and lower SLA have been observed from April to October (Fig. 8d–j).

Today it has been well established that the dominant patterns observed in Fig. 8 represent both westward propagating mesoscale eddies (Schouten et al., 2002; Quartly et al., 2006) and planetary Rossby waves (Schouten et al., 2002; De Ruijter et al., 2005; Palastanga et al., 2007), whereby the former prevails the most (Quartly et al., 2006).

Eddies

Mesoscale oceanic eddies are turbulent circular rotating flows in the ocean, characterized by a typical time-scale of about 10–90 days, and typical space-scale between 10–500 km (Robinson, 1983). They can be generated by different physical processes, such as barotropic and baroclinic instabilities of the flow field. In fact, eddies are ubiquitous features in the global ocean. They are the most vigorous mesoscale processes. Many can be found in the south-west Indian Ocean, especially to the south of about 10°S.

Attempt to map eddies in the WIO Basin as a whole, a domain which combines three large Marine Ecosystems (LME), namely the Agulhas Current LME, the Somali Coastal Current LME and the Red Sea LME was conducted recently by Halo and Raj (2020), using 20 years (1993–2012) of satellite altimetry dataset. By applying an automatic eddy detection algorithm as described by Halo et al. (2014a), eddies were identified and tracked in time and space from 1 January 1993 to 31 December 2012, on a daily basis. Their generation sites and trajectories are shown in Fig. 9 (Halo and Raj, 2020). The red colour indicates anticlockwise rotating eddies and blue clockwise rotating. Only eddies with a lifetime equal and greater than 90 days have been presented. The bold dots indicate the generation sites and the end of the trajectory lines indicate their sites of decay.

Statistical census of the eddy field conducted by Halo and Raj (2020) reveals different spatial/temporal distribution patterns between the north-west and south-west Indian Ocean sectors, separated by a strong eddy desert region between 12°S and 3°N. Many mesoscale structures in this latitude band have relatively short lifespans, less than three months. Geometrical patterns of sea level anomalies in such a band (Fig. 8) suggest their identity as baroclinic Rossby waves (Halo and Raj, 2020).

Overall, more cyclonic than anticyclonic eddies were found, and all tracked structures exhibited a predominant westward and south-westward propagation, which were heavily impacted by the seabed morphology, continental land masses, islands and bathymetric ridges. These highlight the role that bottom topography plays in influencing oceanographic circulation processes. The eddy trajectories (Fig. 9) strongly suggest an effective inter-basin telecommunication, which could potentially favour connectivity pathways of oceanic materials. The eddies also play a noticeable role on the distribution of surface chlorophyll, especially in coastal upwelling-dominated areas (see Fig. 10).

Given their strong nonlinear characteristics (Halo et al., 2014b; Halo and Raj, 2020) and their ability to circulate through different ocean basins in the region, it is thus expected that these eddies are important vectors of biological connectivity between different ecosystems within the Agulhas Somali Currents LME region.

When the SEC or NEMC passes at the northern tip of Madagascar some eddies are formed by barotropic instability (Biastoch et al., 1999; Halo, 2012). This is a physical mechanism or process whereby the energy from the mean currents are converted into a kinetic turbulent energy. Similar processes take place in the narrows of the Mozambique Channel (De Ruijter et al., 2002; Schouten et al., 2003). On average, four to six eddies are generated and pass through the Mozambique Channel each year. They are highly energetic, and propagate through the full length of the Channel, with average speed of about

EDDIES TRACKS – OVER 30 DAYS

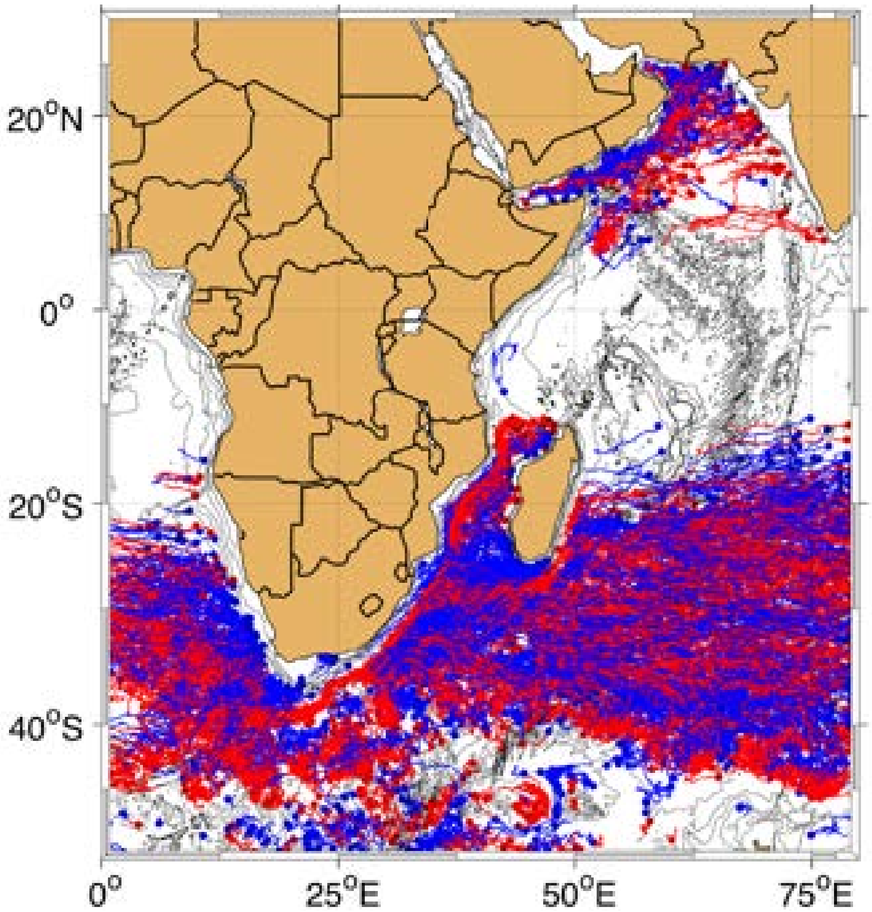


Figure 9: Eddy generation sites, trajectories and decay. The eddies were identified using the automatic algorithm presented by Halo and Raj (2020). The dataset used is the satellite derived altimetry product absolute dynamic topography for the period starting in 1993 to 2012. The dataset was downloaded from AVISO website. Clockwise rotating eddies are shown in blue, and the anticlockwise rotating are shown in red. The background grey contours represent the seafloor topography.

3.5 km day^{-1}. They rotate anticlockwise (anticyclonic) and are relatively warmer than their homologous counterparts that rotate clockwise (cyclonic). Hydrographic measurements across some of the anticyclonic eddies in the Mozambique Channel revealed that they are about 400 km wide, and reach the seafloor, in some cases deeper than 3000 m (De Ruijter et al., 2002; Ridderinkhof and De Ruijter, 2003). Similarly, at the southern tip of Madagascar, eddies and dipoles are also formed at the southern termination of the SEMC (De Ruijter et al., 2004; Quartly et al., 2006; Ridderinkhof et al., 2013; Halo et al., 2014b).

The Mozambique Channel eddies and dipoles interact among themselves, at times merging or decaying in the southern Mozambique Channel (De Ruijter et al., 2004; Quartly and Srokosz, 2004). Nevertheless, they move downstream with the Agulhas Current. During their southward excursion they occasionally interact with the main current, causing instabilities on the patch of the Agulhas Current (Lutjeharms, 2006), causing the current to move slightly offshore, thus generating a large cyclonic meander inshore of the Agulhas Current, termed the Natal Pulses (Lutjeharms and da Silva, 1988). The process is more frequently observed in the Natal Bight, because of the change of the continental shelf observed in the region.

Several cyclonic eddy structures have been observed along the south-east coast of South Africa, inshore of the Agulhas Currents, such as the Durban eddies and break-away eddies in Algoa Bay (Gustella and Roberts, 2016). Eddies also are formed further downstream at the Agulhas Retroflection region (Fig. 9). These Agulhas rings transport heat and salt into the south Atlantic Ocean, where it is thought they play a crucial role influencing the climate (Biastoch et al., 2009; Beal et al., 2011). Several eddies and meanders are also formed along the path of

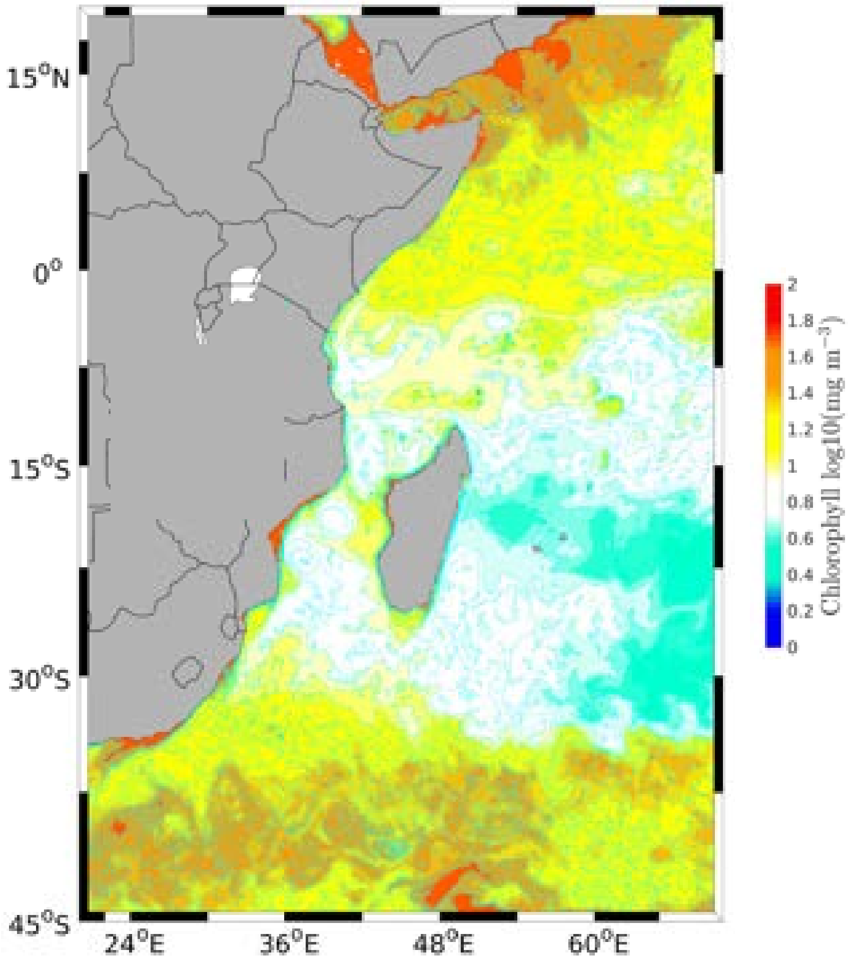


Figure 10: Composite satellite imagery of chlorophyll concentration in the WIO for November 2016. Note enhanced chlorophyll concentrations mirroring main oceanographic features of the circulation, especially in the Arabian Sea, Red Sea, Somali Coast, Sofala Bank and in the Delagoa Bight off Mozambique, along the north-west and south-east coasts of Madagascar, and along the Subtropical Front.

the Agulhas Return Current, formed either from the interaction of the current with the seafloor topography, and/or from the strong meridional thermal gradient across the subtropical front. Some eddies are also formed from baroclinic instabilities.

Along the coasts of Tanzania, Kenya and Somalia, the mesoscale oceanic variability is dominated by relatively fewer eddy structures (Fig. 9), but with a strong presence, especially during the south-west monsoon. During this period the SC that received water supply from the EACC turns offshore after crossing the 4°S, and remnants of the flow re-circulate around the Equator and form the eddy-like structure termed the Southern Gyre (see also Fig. 6). The gyre is relatively large (about 400 km wide) and shallow (about 100–300 m deep), and has been reported to have its first appearance in the upper ocean in early June, being triggered by instabilities of the northward flowing Somali Current, during the south-west monsoon.

According to Gamoyo et al. (2017), the northward migration of the Southern Gyre is intensified by the arrival of downwelling energy pulses from the large-scale Rossby-waves coming from the far flanks of the east Indian Ocean. A closer look into Fig. 10 also reveals higher chlorophyll concentration around or/within the Southern Gyre.

Further north, another structure is formed, termed the Great Whirl (see Fig. 5 and Fig. 6). Even more north a third mesoscale feature is formed in Socotra, termed the Socotra eddy (Bruce and Beatty, 1985; Schott et al., 2009). It has been indicated that the Socotra eddy is persistently observed in many summer monsoon regimes, north-east of Socotra (Schott et al., 2009). Average climatology of the SSH anomalies presented in Fig. 8, as well as the map of eddy detection generation sites in Fig. 9 seems to corroborate this hypothesis.

COASTAL SHELF DYNAMICS AND UPWELLING EVENTS

Because of shallow bottom topography, strong local wind forcing, river discharges and tides, the oceanography of the coastal shelf systems are to some extent different from that of the open ocean. Nevertheless, they are strongly connected. Some examples are presented below to help describe the main features.

In Fig. 10 a satellite-derived composite map of chlorophyll concentrations in the WIO region for the month of November 2016 depicts a typical scenario that highlights the synergy between physical forcing of the circulation and its associated biological response in the form of upwelling events. After their formation in the narrows of the Channel, the Mozambique Channel eddies and rings propagate southward parallel to the western boundary of the Channel (Halo et al., 2014a). Because of their large horizontal scale, along the Sofala Bank, located immediately to the south of the narrows of the Channel (see reference to Sofala Bank in Fig. 2d), where the shelf is broad and shallow, their influence on the shelf circulation is very strong (Fig. 10). Recently it has been investigated within a numerical modelling framework of the Regional Ocean Modelling Systems (ROMS). It has been found that when a mesoscale cyclonic eddy is present at the coast, a shelf current is observed following northward. Whereas when an anticyclonic eddy is present, the shelf current changes its direction and propagates southward. Therefore, the shelf current is controlled by the offshore coastal flow (regular train of mesoscale eddies) (Malauene et al., 2018). It is likely that this shelf current is the same current associated to the Zambezi River plumes investigated by Nehama and Reason (2015).

Tides also have a strong influence on the shelf dynamics over the Sofala Bank (Fig. 2d). Modelling studies by Nehama and Reason (2015) and Chevane et al. (2016) have shown that tides interacting with the Bank are important drivers of strong vertical mixing, able to bring cold deep waters to the surface. Another predominant physical process occurring in Sofala Bank are internal waves. The bank has been identified as a hotspot for these types of waves. The Mozambique Channel eddies also have a strong influence on the onset of the Delagoa Bight eddy (Cossa et al., 2016), off Maputo Bay (Fig. 2c).

Downstream of the Mozambique Channel, as the Agulhas Current propagates south-westward along the East Coast of South Africa, parallel to the coast and attached to the continental slope (Fig. 6), it induces strong shear instabilities that generate small-scale clockwise rotating vortices at the inshore edge of the Current, because of the lateral friction. A typical example of this type of process is the formation of the Natal Pulse in the Natal Bight (Lutjeharms and Roberts, 1988), along the coast of Durban, where the continental shelf widens (see Fig. 2b). Because of surface divergence occurring within the clockwise rotating vortices, waters are upwelled towards the sea-surface onto the shelf. These cold, nutrient-rich waters (Gustella and Roberts, 2016) provide the ingredients required for phytoplankton growth (Lamont et al., 2014), thus enhancing blooms of biological primary productivity (Fig. 10).

Processes similar to the Natal Pulses have been documented in the region, and given other attributes such as Durban Eddies, Durban Break-Away eddies (Gustella and Roberts, 2016). The dynamics of these events occasionally become responsible for flooding the Agulhas Bank with cold waters. In addition, local winds on the western edge of the Agulhas Bank have been also identified as playing a contributing role in the generation of coastal upwelling events in the region (Goschen et al., 2015; Roberts and Nieuwenhuys, 2016). Cross-shelf dynamics driven by topographic-induced upwelling events in the region also contribute towards high chlorophyll concentrations observed in Fig. 10, along the east coast of South Africa.

Upstream of the Mozambique Channel the continental shelf along the coasts of Tanzania and Kenya is generally a narrow strip which exhibits relatively different morphology and width (Fig. 2f). As earlier discussed in this chapter, the shelf is relatively narrow in the south and wider in the north. Along the coasts of Tanzania and Kenya (Fig. 2f), the on-shelf circulation has not been investigated extensively due to limited data records at desirable spatial and temporal resolution.

Efforts to generate scientific information on the area comes from few numerical solutions of circulation models. Thus, reliable knowledge of the inner-shelf circulation is scant. Studies such as those by Manyilizu et al. (2014) and Shigalla and Shaghude (2014) have addressed primarily the dynamics along the path of the main core structure of the EACC, which appears to run relatively far offshore. The most comprehensive study about the channels and inner-shelf circulation for these regions can be assessed from the work by Nyandwi (2013), Zavala-Garay et al. (2015) and that of Mayorga-Adame et al. (2016). The latter was based on an inter-annual simulation using ROMS, with a horizontal grid resolution of 4 km, especially configured for the coasts of Tanzania and Kenya for a period spanning from 2000 to 2007. The studies indicate that on approaching the shelf, the geostrophic circulation enters into an accelerating mode as it follows the isobaths. In contrast, on the shelf (considered here as the inner 100 m isobath) the circulation weakens as the flow experiences friction due to the shallow bathymetry that slows down the deep flows, as well as the presence of the chain of islands (Pemba, Zanzibar and Mafia). The shelf circulation appears to be sensitive to the spatial and temporal scale variability of the main large-scale oceanographic feature, the EACC.

The EACC is known to flow northward all year (Newell, 1959; Schott et al., 1988; Swallow et al., 1998; Shigalla and Shaghude, 2014), nevertheless it experiences modes of weaker and stronger regimes linked to the seasonal variation of the atmospheric wind field, expressed in terms of monsoonal winds (see Fig. 4). During the south-east monsoon and the two periods of the reversal of the monsoons (inter-monsoons), spanning from April to November, the EACC is stronger (with characterizing velocities greater than 0.85 m s^{-1}), so is the shelf circulation (Mayorga-Adame et al., 2016). On the other hand, during the north-east monsoon, extending from December to March, the EACC is weaker (characteristic velocities below 0.75 m s^{-1}), due to the opposite direction of the surface wind forcing. Consequently, the northward shelf circulation between the Mafia and Zanzibar Islands is blocked (Mayorga-Adame et al., 2016), as the effect of the bathymetry becomes more pronounced (ie shallow channels). The topographic effect on the circulation patterns during the north-east monsoon is well perceived at the different southern entrances of the channels (ie southern gaps between the islands and the African mainland), as in-situ observations have revealed different current speeds at different water depths.

The effects of the chain of the island system along the coasts of Tanzania and Kenya (Fig. 2f) on the coastal circulation depends on the intensity of the EACC (see Fig. 6), the geometry of the coastline (Fig. 2f) and the depth of the channels between the islands and the mainland (Mayorga-Adame et al., 2016). The interaction of the EACC with the shallow bed topography and the island's coastlines are important mechanisms driving the reversal of portions of the northward EACC at the northern entrances of the channels. These southward flows within the channels appear to exhibit different behaviours: in the northernmost Island of Pemba this southward oriented shelf current is weaker, but with a persistent nature all year around. Along the Zanzibar Island the flow is relatively stronger, reaching the strongest velocities between December and March, of which is the period of the north-east monsoon. The current-topography/coastline interactions are also driving mechanisms leading to the formation of small-scale oceanic eddies which rotates clockwise/anticlockwise, depending on the morphology of the coast. These locally generated eddies (see Fig. 9) have been described as exhibiting both permanent to semi-permanent regimes. At the southern entrances of the channels these eddies can block the intrusion of the EACC into the channels (Mayorga-Adame et al., 2016). These eddies also are the drivers of localized upwelling phenomena (Fig. 10), that occasionally cause cooling events over the continental shelf (Mayorga-Adame et al., 2016).

Further north, along the EACC, another mesoscale feature observed and recently inspected from numerical solution of ROMS in the climatological configuration set by Gamoyo et al. (2017) along the coast of Kenya and Somalia is the Southern Gyre (see Fig. 5b). It appears to be formed in early June near the surface and deepens between 100–300 m below the surface, as a result of instability in the northward-flowing Somali Current. The gyre has a mean diameter of about 400 km, and retains cool and fresh waters in its interior, derived from the SEC (Gamoyo et al., 2017).

WATER MASSES IN WIO REGION

Oceanic water masses are usually classified on the basis of their vertical distribution throughout the water column and can be clustered in an upward direction as: surface, intermediate, deep, bottom and abyssal waters (Defant, 1961). Because of its relatively small geometry, the Indian Ocean has a complex upper water mass structure, mostly due to several factors such as: its enclosure by the Asian continent at the subtropics (Fig. 5), the regime of the monsoonal winds (Fig. 4) which control the dynamics of the currents on the upper layers in the northern Indian Ocean (Fig. 5, Fig. 6, Fig. 8), and the unbalanced rates of precipitation/evaporation between the eastern and WIO (Wyrtki, 1971; Tomczak and Godfrey, 1994; Schott et al., 2009).

Figure 11 shows monthly means (January to December) of air-sea density fluxes, computed from haline and thermal fluxes at the sea surface (Howe, 2008). It depicts oceanic density gains (losses) portrayed by the positive (negative) fluxes on climatological time scales. Positive (negative) fluxes are indicative of cooling (heating) of the sea surface. Comparison between these fluxes and monthly mean climatology of SLA (Fig. 8), suggests a weakening pattern of SLA signals in the greater Agulhas system during the periods of strong positive density fluxes, indicative of strong surface cooling events, especially observed between May (Fig. 11e) and August (Fig. 11h).

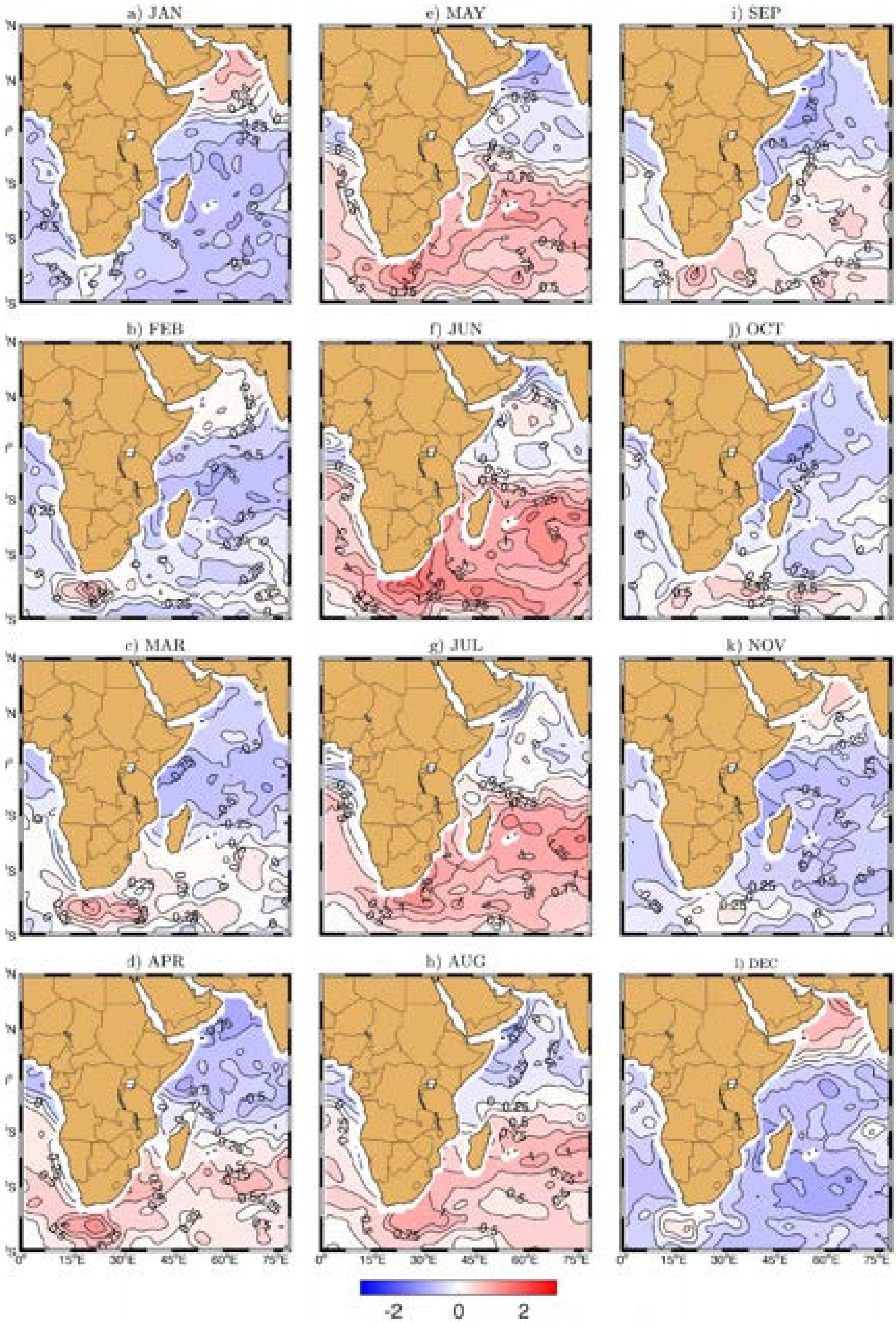


Figure 11: Monthly mean climatology (January to December) of air-sea density fluxes (with contours of 2.5×10^{-6} kg m^{-2} s^{-1}) estimated using heat fluxes and freshwater fluxes to and from the ocean. Negative (Positive) values are indicative of density loss (gain) respectively. Density loss (gain) are indicative of oceanic heating (cooling) respectively.
Courtesy of the dataset Howe, 2008.

The imprint of the ocean-atmosphere density fluxes is closely related to the local hydrology, notably on precipitation and evaporation rates (Howe, 2008). Therefore, it has a strong influence on the formation and characteristics of the dominant water masses locally generated. The description of the WIO's main water masses is presented in the section below.

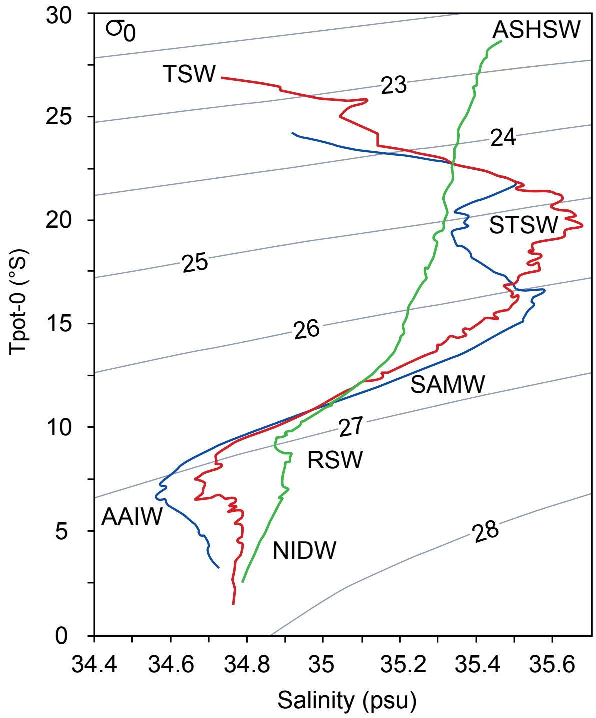


Figure 12: Temperature/Salinity (T/S) diagram illustrating predominant water masses of the WIO. Antarctic Intermediate Water (AAIW), Arabian Sea water (ASHSW), Northern Indian Deep Water (NIDW), Red Sea Water (RSW), SubAntarctic Mode Water (SAMW), Subtropical Surface Water (STSW), Tropical Surface Water (TSW).
Adapted from Halo et al. (2017).

Upper waters

At the surface and sub-surface layers (0–500 m depth), the south-west Indian Ocean is mainly occupied by tropical and subtropical water masses (Fig. 12). The tropical surface water (TSW) is originated close to the equatorial band, in the central Indian Basin. Its formation is related to both the excessive rates of precipitation over evaporation in the tropics (Wyrtki, 1971; Toole and Warren, 1993) and the influence of the low salinity waters of the Indonesian Throughflow, also referred as the Australasian Mediterranean Sea Waters (AAMW) (Tomczak and Godfrey, 1994). The TSW is characterized by salinity values lower than 35.5 and a neutral density less than 25.5 kg m^{-3} (Beal et al., 2006). Specific values of salinity and temperature range between 34.91 and 35.31, and between 24.7°C and 26.3°C, respectively (Donohue and Toole, 2003). The TSW enters in the subtropics via the branches of the South Equatorial Current (SEC) (Swallow et al., 1988; Schott et al., 1988), propagating along the east coast of Madagascar and along the east coast of Africa mainland (Fig. 5). The main route taken by the TSW during its southward spreading is made through the Mozambique Channel (Beal et al., 2006; Swallow et al., 1988).

At the sub-surface, or thermocline layers (200–500 m depth), the flow is mainly dominated by the subtropical surface water (STSW) and the AAMW. The STSW is formed within the subtropical gyre of the south Indian Ocean, to the east of 90°E (Wyrtki, 1971), and between latitudes 25°S and 35°S (Tomczak and Godfrey, 1994; DiMarco et al., 2002). Its formation is due to the excess of evaporation rates over precipitation. It is characterized by salinity greater than 35.5 (Fig. 12) and a neutral density range between 25.5 kg m^{-3} and 26.4 kg m^{-3} (Beal et al., 2006). STSW is transported westward by the SEC and enters into the greater Agulhas system by the flow of the SEMC (Beal et al., 2006). According to Donohue and Toole (2003), a distinction between TSW and STSW is made by a strong boundary between them, formed at about 28°S, where it generates a sharp gradient of temperature and salinity.

The AAMW is originated in the tropics, from the Pacific Central Waters, and enters in the Indian Ocean through the ITF (see Fig. 5), between Timor and the islands to the east of Bali, and forms one of the strongest thermocline fronts of the world's ocean (Tomczak and Godfrey, 1994). It is characterized by temperatures between 8°C and 23°C, and salinities between 34.4 and 35 (Emery, 2001). Another subsurface water mass in the region is the SubAntarctic Mode Water (SAMW) (Fig. 12), formed at the subtropical convergence front, mainly between 46°E and 62°E (Fine, 1993), due to the winter cooling and deep convection (Fig. 11) to the south of the front (McCartney, 1977). It subducts into the thermocline and propagates northward into the subtropical gyre (Toole and Warren, 1993). The core of the SAMW is at about 500 m depth, where it holds an oxygen maximum (DiMarco et al., 2002). To the north of about 28°S, the potential temperature and density of SAMW varies from 13°C and 26.65 kg m^{-3} respectively at about 28°S, to values close to 11°C and 26.8 kg m^{-3} at 20°S (Donohue and Toole, 2003).

The zonal gradient of temperature, salinity and density along the southern Indian Ocean Basin, and the entrainment of the SAMW in the subtropical gyre results in a subtropical distribution of this water, with highest values of oxygen found in the south-east Indian Ocean (Donohue and Toole, 2003). This extension is also termed South-East Indian SubAntarctic Mode Water (SEISAMW), characterized by a concentration of oxygen above 4.9 ml l^{-1}, and a neutral density of 26.8 kg m^{-3} (Wyrtki, 1971). This water mass enters into the WIO region through the westward branch of the subtropical gyre (Donohue and Toole, 2003). Similar to SAMW is the Indian Central Waters (ICW). The ICW is also originated in the subtropics and is characterized by temperatures ranging between 8°C and 25°C, and salinity between 34.6 and 35.8 (Emery, 2001). Beal et al. (2006), observed this water mass at a depth below the thermocline 300 m deep.

Intermediate waters

At the intermediate layers, or below the thermocline waters (500–1500 m depth), the WIO is mostly occupied by the Antarctic Intermediate Waters (AAIW), in the southern hemisphere, and the Red Sea Waters (RSW) and the Arabian Sea Low Oxygen Waters (ASLOW) (Beal et al., 2006), in the northern hemisphere (Wyrtki, 1971) (Fig. 12). The AAIW is thought to be formed in the south-eastern Pacific and enters into the Atlantic Ocean through the Drake Passage, and continues flowing eastward along the Subantarctic front (McCartney, 1977). Once in the Indian Ocean, at about 60°E, the AAIW flows northward into the subtropical gyre (Fine, 1993; Beal et al., 2006). However, it does not cross the 10°S latitude, because its propagation is blocked by the equatorial current systems (Tomczak and Godfrey, 1994). The fresher AAIW is characterized by a minimum in salinity ranging between 33.8 and 34.6, and temperature between 2°C and 10°C (Emery, 2001).

In contrast, the RSW is very saline (Fig. 12), being a water mass formed in the Red Sea Basin, from excessive evaporation over precipitation, which leads to a sinking of surface waters in the Gulf of Aden (Wyrtki, 1971; Tomczak and Godfrey, 1994). This process induces a local formation of maximum salinity, with reduced oxygen concentration (DiMarco et al., 2002). The RSW is characterized by a potential temperature of about 22°C, salinity of about 39, and a density of 27.25 kg m^{-3} (Tomczak and Godfrey, 1994). It flows southwards, concentrated along the African coast, below the Zanzibar Current (Wyrtki, 1971; Beal et al., 2000; Donohue and Toole, 2003), and passes through the Mozambique Channel, eventually reaching the Agulhas Current (Beal et al., 2000; Donohue and Toole, 2003).

The ASLOW originates in the Arabian Basin and has been observed at about 1200 m depth (Beal et al., 2006). It is characterized by high values of salinity, a relatively lighter neutral density of 25.5 kg m^{-3}, and low oxygen, less than 3.8 ml l^{-1}. Such minimum oxygen concentrations are due to the high consumption rates associated to the seasonal high productivity. The ASLOW propagates southwards, concentrated along the western boundary of the Indian Ocean. During its journey it also enters the Mozambique Channel. The ASLOW is a result of a mixing process between the Arabian Sea Water (ASW) and Bengal Bay Water (BBW). The former is characterized by values of temperature ranging between 24°C and 30°C, and salinity between 35.5 and 36.8; while the latter is characterized by temperature ranging between 25°C and 29°C, and salinity between 28 and 35 (Emery, 2001).

Deep and abyssal waters

The deep layer of the Indian Ocean is filled by the Indian Deep Water (IDW). To the north of the Equator this water mass is usually termed as the northern, or NIDW (Fig. 12), and in the south as the southern, or SIDW. To the north of 45°S this deep water mass exists between 1500 and 3800 m depth, while to the south of this latitude it shallows to about 500 m depth (Tomczak and Godfrey, 1994). The IDW is characterized by a salinity greater than 34.8 in the western side of the Indian Ocean, and by 34.75 in its eastern side (Tomczak and Godfrey, 1994). It is formed in the Atlantic Ocean as a remaining part of the North Atlantic Deep Water that did not convert into the intermediate waters within the Atlantic sector. The IDW is carried eastwards by the Antarctic Circumpolar Current (ACC). In the South Indian Ocean, it propagates northwards, concentrated along the western boundary. On reaching the northern Indian Ocean, at the Somali Basin, this water mass flows eastwards, and upwells in the Arabian Seas and in the Bay of Bengal (Wyrtki, 1971). The deep circulation is below the permanent thermocline and is influenced by the inflow of the RSW and Persian Gulf Waters (PGW).

To the bottom, below 3800 m, the Indian Ocean is dominated by the Antarctic Bottom Water (AABW), also called Circumpolar Deep Waters (CDW). This water mass is characterized by a range of potential temperature between 1°C and 2°C, and salinity between 34.62 and 34.73 (Emery, 2001). It is formed in the Southern Ocean and enters the South-West Indian Ocean via the Mozambique and Madagascar Basins (Fig. 1a), through the deep fractures of the South-West Indian Ocean Ridge, near 30°S, and 56°E–59°E (Tomczak and Godfrey, 1994). The flow in the Mozambique Basin is blocked within the Mozambique Channel by the Davie Ridge. In the Madagascar Basin (Fig. 1a), the water propagates further north and forms a western boundary current along the continental slope of the east coast of Madagascar.

In the south-east, this water mass enters the Indian Ocean through the South Australasian Bight, around 50°S and 124°E (Tomczak and Godfrey, 1994). To the south of Australia, it flows along the southern and western slope of Australia, and further north it escapes to the central Indian Ocean propagating along the eastern slope of the Ninetyeast Ridge (Tomczak and Godfrey, 1994). After crossing through the fractures of the Ridge, the water flows westwards and eventually reaches the north-east coast of Africa. Through the slope of the African continent, it gradually upwells to form the North Indian Deep Waters.

REFERENCES

Ansorge, I. and Lutjeharms, J. (2003). Eddies originating at the South-West Indian Ridge. *J. Mar. Sys*. 39, pp. 1–18. https://doi.org/10.1016/S0924-79639(02)00243-9

Avraham, Z.B. and Bunce, E.T. (1977). Geophysical study of the Chagos-Laccadive Ridge, Indian Ocean. *J. Geophys. Res*. 82 (8), pp. 1295–1305. https://doi.org/10.1029/JB082i008p01295

Baines, A.G., Cheadle., M.J., Dick., H.J.B., Scheirer, A.H., John, B.E., Kusznir, N.J. and Matsumoto, T. (2007). Evolution of the Southwest Indian Ridge from 55°45'E–62°E: Changes in plate-boundary geometry since 26 Ma. *Geochem. Geophys. Geosys*. 8(6), Q06022. https://doi.org/10.1029/2006GC001559

Barlow, R.G., Marsac, F.S., Ternon, J.F. and Roberts, M. (2014). The Mozambique Channel: Mesoscale Dynamics and Ecosystem responses. *Deep Sea Res. Part II*, 100, pp. 1–9. https://doi.org/ 10.1016/j.dsr2.2013.10.012

Bassias, Y. (1992). Petrological and geochemical investigation of rocks from the Davie fracture zone (Mozambique Chan- nel) and some tectonic implications. *J. Afr. Earth. Sci*. 15, pp. 321–339. https://doi.org/10.1016/0899-5362(92)90018-8

Beal, L.M., De Ruijter, W.P.M., Biastoch, A., Zahn, R., Cronin, M., Hermes, J., Lutjeharms, J., Quartly, G., Tozuka, T., Baker-Yeboah, S., Bornman, T., Cipollini, P., Dijkstra, H., Hall, I., Park, W., Peeters, F., Penven, P., Ridderinkhof, H. and Zinke, J. (2011). On the role of the Agulhas System in ocean circulation and climate. *Nature* 472, pp. 429–436. https://doi.org/10.1038/nature09983

Beal, M.L., Chereskin, T.K., Lenn, Y.D. and Elipot, S. (2006). The Sources and Mixing Characteristics of the Agulhas Current. J. *Phys. Oceanogr*. 36, pp. 2060–2074. https://doi.org/10.1175/JPO2964.1

Beal, M.L., Ffield, A. and Gordon, A.L. (2000). Spreading of the Red Sea Overflow waters in the Indian Ocean. *J. Geophys. Res*. 105, pp. 8549 – 8564. https://doi.org/10.1029/1999JC900306

Bernard, A., Munschy., M., Rotstein, Y. and Sauter, D. (2005). Refined spreading history at the Southwest Indian Ridge for the last 96 Ma, with the aid of satellite gravity data. *Geophys. J. Int*. 162, pp. 765–778. https://doi.org/10.1111/j.1365- 246X.2005.02672.x

Biastoch, A., Böning, C.W., Schwarzkopf, F.U. and Lutjeharms, J.R.E. (2009). Increase in Agulhas leakage due to poleward shift of Southern Hemisphere westerlies. *Nature* 462, pp. 495–498. https://doi.org/10.1038/nature08519

Biastoch, A., Riesen, C.J.C., Lutjeharms, J.R.E. and Boebel, O. (1999). The importance of flow in the Mozambique Channel to seasonality in the greater Agulhas Current System current. *Geophys. Res. Lett*. 26(21), pp. 3321–3324. https://doi.org/10.1029/1999GL002349

Bond, G. (1979). Evidence for some uplifts of large magnitude in continental platforms. *Tectonophysics* 61, pp. 28–305. https://doi.org/10.1016/0040-1951(79)90302-0

Bruce, J.G. and Beatty, W.H. (1985). Some observations of the coalescing of Somali eddies and a description of the Socotra eddy. *Oceanol. Acta* 8(2), pp. 207–219

Cabral, P., Gabriela, A., Akandea, A., Costa, A., Amade, N., Niquisse, S., Atumane, A., Cuna, A., Kazemi, K., Mlucasse, R. and Santha, R. (2017). Assessing Mozambique's exposure to coastal climate hazards and erosion. *Int. J. Disaster Risk Reduction* 23, pp. 45-52. https://doi.org/10.1016/j.ijdrr.2017.04.002

Cedras, R.B., Halo, I. and Gibbons, M.J. (2020). Biogeography of pelagic calanoid copepods in the Western Indian Ocean. *Deep Sea Research II* 179, 104740. https://doi.org/10.1016/j.dsr2.2020.104740

Chapman, P., DiMarco, S.F., Davis, R.E. and Coward, A.C. (2003). Flow at intermediate depths around Madagascar based on ALACE float trajectories. *Deep Sea Res. Part II* 50(12–13), pp. 1957–1986. https://doi.org/10.1016/S0967-0645(03)00040-7

Chevane, C.M., Penven, P., Nehama, F.P.J. and Reason, C.J.C. (2016). Modelling the tides and their im- pacts on the vertical stratification over the Sofala Bank, Mozambique. *Afr. J. Mar. Sci*. 38, pp. 465–479. https://10.2989/1814232X.2016.1236039

Cossa, O., Penven, S.P.P., Capet, X. and Reason, C. J. C. (2016). Modelling cyclonic eddies in the Delagoa Bight region. *Clim. Dyn*. 39, pp. 509–529. https://doi.org/ 10.1007/s00382-011-1170-6

De Ruijter, W.P.M., Ridderinkhof, H., Lutjeharms, J.R.E., Schouten, M.W. and Veth, C. (2002). Observations of the flow in the Mozambique Channel. *Geophys. Res. Lett*. 29, pp. 1401–1403. https://doi.org/10.1098/rsta.2004.1478

De Ruijter, W.P.M., Ridderinkhof, H. and Schouten, M.W. (2005). Variability of the southwest Indian Ocean. Philos. *Trans. R. Soc. London A* 363, pp. 63–76. https://doi.org/10.1098/rsta.2004.1478

De Ruijter, W.P.M., van Aken, H.M., Beier, E.J., Lutjeharms, J.R.E., Matano, R.P. and Schouten, M.W. (2004). Eddies and dipoles around South Madagascar: formation, pathways and large-scale impact. *Deep Sea Res. Part II* 51, pp. 383–400. https://doi.org/ 10.1016/j.dsr.2003.10.011

Defant, A. (1961). *A Physical Oceanography*. Vol. 1. Pergamon Press, New York, USA, 729 pp

DiMarco, S.F., Chapman, P., Nowlin, W.D., Hacker, P., Donohue, K., Luther, M., Johnson, G.C. and Toole, J. (2002). Volume transport and property distribution of the Mozambique Channel. *Deep Sea Res. Part II* 49, pp. 1481–1511. https://doi.org/10.1016/S0967-0645(01)00159-X

Donohue, K.A. and Toole, J.M. (2003). A near synoptic survey of the SouthWest Indian Ocean. *Deep Sea Res. Part II* 50, pp. 1893–1931. https://doi.org/10.1016/S0967-0645(03)00039-0

Emery, W.J. (2001). Water Types and Water Masses. *Encyclopedia of ocean sciences* 6, pp. 3179-3187. https://doi.org/10.1006/rwos.2001.0108

Fenta, A.A., Tsunekawa, A., Haregeweyn, N., Poesen, J., Tsubo, M., Borrelli, P., Panagos, P., Vanmaercke, M., Broeckx, J., Yasuda, H., Kawai, T. and Kurosaki, Y. (2020). Land susceptibility to water and wind erosion risks in the East Africa region. *Sci. Tot. Environ*. 703, p. 135016. https://doi.org/10.1016/j.scitotenv.2019.135016

Fine, R. (1993). Circulation of Antarctic Intermediate Water in the South Indian Ocean. *Deep Sea Res.* 40, pp. 2021–2042. https://doi.org/10.1016/0967-0637(93)90043-3

Gamoyo, M., Reason, C.J.C. and Collins, C. (2017). A numerical investigation of the Southern Gyre using ROMS. *J. Mar. Sys*. 169, pp. 11–24. https://doi.org/ 10.1016/j.jmarsys.2017.01.002

Goodman, S.M., Weyeneth, N., Ibrahim, Y., Saïd, Y. and Ruedi, M. (2010). A review of the bat fauna of the Comoro Archipelago. *Acta Chiropterologica* 12(1), pp. 117–141. https://doi.org/ 10.3161/150811010X504635

Goschen, W.S., Bornman, T.G., Deyzel, S.H.P. and Schumann, E.H. (2015). Coastal upwelling on the far eastern Agulhas Bank associated with large meanders in the Agulhas Current. *Cont. Shelf Res.* 101, pp. 34–46. https://doi.org/10.1016/jcsr.2015.04.004

Goslin, J., Segoufin, J., Schlich, R. and Fisher, R.L. (1980). Submarine topography and shallow structure of the Madagascar Ridge, western Indian Ocean. *Geol. Soc. Am. Bull*. 91(12), pp. 741–753. https://doi.org/10.1130/0016-7606(1980)91- 741:STASSO-2.0.CO2

Graham, D.W., Johnson, K.T.M., Priebe, L.D. and Lupton, J.E. (1999). Hotspot-ridge interaction along the Southeast Indian Ridge near Amsterdam and St. Paul islands: helium isotope evidence. *Earth Planet. Sci. Lett*. 167, pp. 297–310. https://doi.org/10.1016/S0012-821X(99)00030-8

Gustella, L. A. and Roberts, M.J. (2016). Dynamics and role of the Durban cyclonic eddy in the KwaZulu-Natal Bight ecosystem. ajms 38, S23–S42. https://doi.org/10.2989/1814232X.2016.1159982

Halo, I. (2012). The Mozambique Channel eddies: Characteristics and Mechanisms of formation. Ph.D. thesis, University of Cape Town

Halo, I., Backeberg, B., Penven, P., Ansorge, I., Reason, C. and Ullgren, J.E. (2014a). Eddy properties in the Mozambique Channel: A comparison between observations and two numerical ocean circulation models. *Deep Sea Res. Part II* 100, pp. 38–53. https://doi.org/ 10.1016/j.dsr2.2013.10.15

Halo, I., Penven, P., Backeberg, B., Ansorge, I., Shillington, F. and Roman, R. (2014b). Mesoscale eddy variability in the southern extension of the East Madagascar Current: Seasonal cycle, energy conversion terms, and eddy mean properties. *J. Geophys. Res*. 119, pp. 7324–7356. https://doi.org/10.1002/2014JC009820

Halo, I., Malauene, B. and Ostrowski, M. (2017). Physical oceanography. In *The RV Dr Fridtjof Nansen in the Western Indian Ocean: Voyages of marine research and capacity development* (eds. Groeneveld, J.C. and Koranteng, K.A.), pp. 37–53. FAO. Rome, Italy

Halo, I. and Raj, P.R. (2020). Comparative oceanographic variability during climate change in the Agulhas Current and Somali Coastal Current Large Marine Ecosystems. *Env. Dev.* 36, 100586. https://doi.org/10.1016/j.envdev.2020.100586

Harris, D.J. and Rocha, S. (2009). Comoros, pp. 177–180. In *Encyclopedia of Islands* (eds. R.G. Gillespie and D. A. Clague) xxxii + 1074 pp. University of California Press, Berkeley

Ho, C.-R., Zheng, Q. and Kuo, N.-J. (2004). SeaWiFs observations of upwelling south of Madagascar: long-term variability and interaction with East Madagascar Current. *Deep-Sea Res. II Top. Stud. Oceanogr*. 51(1-3), pp. 59–67, https://doi.org10.1016/j.dsr2.2003.05.001

Howe, N. (2008). The impact of air-sea fluxes on the thermohaline circulation. *Royal Met Soc*. 63(8), pp. 236–239. https://doi.org/10.1002/wea.277

Huggett, J. and Kyewalyanga, M. (2017). Ocean productivity. In *The RV Dr Fridtjof Nansen in the Western Indian Ocean: Voyages of marine research and capacity development* (eds. Groeneveld, J.C. and Koranteng, K.A.), pp. 55-80. FAO. Rome, ItalyJung, G., Prange, M. and Schultz, M. (2016). Influence of topography on tropical African vegetation coverage. *Clim. Dyn*. 46, pp. 2535–2549. https://doi.org/10.1007/s00382-015-2716-9

Köning, M. and Jokat, W. (2010). Advanced insights into magmatism and volcanism of the Mozambique Ridge and Mozambique Basin in the view of new potential field data. *Geophys. J. Int.* 180, pp. 158–180. https://doi.org/10.1111/j.1365- 246X.2009.04433.x

Kruger, G.P. (1983). Terrain Morphology Map of Southern Africa. Soil and Irrigation Research Institute, Department of Agriculture.

Lamont, T., Barlow, G., Morris, T. and van der Berg, M.A. (2014). Characterization of mesoscale features and phytoplankton variability in the Mozambique Channel. *Deep Sea Res. Part II* 100, pp. 94–105. https://doi.org/10.1016/j.dsr2.2013.10.019

Lamont, T., Roberts, M.J., Barlow, R.G., Morris, T. and van der Berg, M.A. (2010). Circulation patterns in the Delagoa Bight, Mozambique, and the influence of deep ocean eddies. *Afr. J. Mar. Sci.* 32, pp. 553–562. https://doi.org/10.2989/1814232X.2010.538147

Leslie, A.D. (1991). Agroforestry practices in Somalia. *For. Ecol. Manage* 45, pp. 293-308. https://doi.org/10.1016/0378-1127(91)90224-J

Lutjeharms, J.R.E. (2006). *The Agulhas Current*. Vol. 1. Springer-Verlag, Berlin

Lutjeharms, J.R.E. and Ansorge, I. (2001). The Agulhas Return Current. *J. Mar. Sys.* 30, pp. 115–138. https://doi.org/10.1016/S0924-7963(01)00041-0

Lutjeharms, J.R.E. and Machu, E. (2000). An upwelling cell inshore of the East Madagascar Current. *Deep-Sea Res.* I 47, pp. 2405-2411. https://doi.org/10.1016/S0967-0637(00)00026-1

Lutjeharms, J.R.E. and Cooper, J. (1996). Interbasin leakage through Agulhas Current filaments. Deep Sea Res., Part I 43 pp. 213–238. https://doi.org/10.1016/S0924-7963(01)00041-0

Lutjeharms, J.R.E. and da Silva, A.J. (1988). The Delagoa Bight eddy. *Deep Sea Res*. 35, pp. 619–634. https://doi.org/10.1016/0198-0149(88)90134-3

Lutjeharms, J.R.E. and Roberts, H.R. (1988). The Natal Pulse: an extreme transient on the Agulhas Current. *J. Geophys. Res*. 93, pp. 631–645. https://doi.org/10.1029/JC093iC01p00631

Machu, E., Lutjeharms, J.R.E., Webb, A.M. and Van Aken, H. M. (2002). First hydrographic evidence of the southeast Madagascar upwelling cell. *Geophys. Res. Lett.* 29(21). https://doi.org/10.1029/2002GL015381

Maia, M., Diament, M. and Recq, M. (1990). Isostatic response of the lithosphere beneath the Mozambique Ridge (SW Indian Ocean) and geodynamic implications. *Geophys. J. Int*. 100(3), pp. 337–348. https://doi.org/10.1111/j.1365-246X.1990.tb00689.x

Malauene, B.S., Moloney, C.L., Lett, C., Roberts, M.J., Marsac, F. and Penven, P. (2018). Impact of offshore eddies on shelf circulation and river plumes of the Sofala Bank, Mozambique Channel. *J. Mar. Sys*. 185, pp. 1–12. https://doi.org/10.1016/j.jmarsys.2018.05.001

Manyilizu, M., Dufois, F., Penven, P. and Reason, C.J.C. (2014). Interannual variability of sea surface temperature and circulation in the tropical western Indian Ocean. *Afr. J. Mar. Sci.* 36 (2), pp. 233–252. https://doi.org/10.2989/1814232X.2014.928651

Masson, D.G (1984). Evolution of the Mascarene basin, western Indian Ocean, and the Significance of the Amirante arc. *Mar. Geophys. Res.* 6, pp. 365-382.

Matano, R.P. and Simionato, C. (1999). Modeling the Wind-Driven Variability of the South Indian Ocean. *J. Phys. Oceanogr*. 29, pp. 217–230. https://doi.org/10.1175/1520-0485(1999)029<0217:MTWDVO>2.0.CO;2

Mayorga-Adame, C.G., Strub, P.T., Batchelder, H.P. and Spitz, Y.H. (2016). Characterizing the circulation off the Kenyan-Tanzanian coast using an ocean model. *J. Geophys. Res*. 121, pp. 1377–1399. https://doi.org/10.1002/2015JC010860

McCartney, M. (1977). Subantarctic Mode Water, in: A voyage of discovery. *Deep Sea Res*. 24, (Suppl.), pp. 103–119.

Moore, A., Blenkinsop, T. and Cotterill, F. (2009). Southern African topography and erosion history: plumes or plate tectonics? *Terra Nova* 21, pp. 310–315. https://doi.org/ 10.1111/j.1365-3121.2009.00887.x

Müller, C.O. (2017). The Central Mozambique continental margin - Its tectonic evolution as the centrepiece of the initial Gondwana break-up. Ph.D. thesis, Alfred Wegener Institute, Bremen, Germany

Nair, N., Anand, S.P. and Rajaram, M. (2013). Tectonic framework of Laccadive Ridge in Western Continental Margin of India. *Mar. Geolog*. 346, pp. 79–90. https://doi.org/10.1016/j.margeo.2013.08.009

NationMaster (2018). Countries Compared by Geography – Terrain. International Statistics at NationMaster.com, CIA World Factbooks 18. NationMaster.com, http://www.nationmaster.com/country-info/stats/Geography/Terrain

Nehama, F.P.J. and Reason, C.J.C. (2015). Modelling the Zambezi River Plume. *Afr. J. Mar. Sci.* 37, pp. 593–604. https://doi.org/doi.org/10.2989/1814232X.2015.1113202

New, A.L., Alderson, S.G., Smeed, D.A. and Stansfield, K.L. (2007). On the circulation of water masses across the Mascarene Plateau in the South Indian Ocean. *Deep Sea Res. Part I* 54, pp. 42–74. https://doi.org/10.1016/j.dsr.2006.08.016

New, A.L., Stansfield, K.L., Smythe-Wright, D., Smeed, D.A., Evans, A.J. and Alderson, S.G. (2005). Physical and Biological Aspects of the Flow across the Mascarene Plateau in the South Indian Ocean. *Philos. Trans. R. Soc.* 363(1826), pp. 151–168. https://doi.org/10.1098/rst.2004.1484

Newell, B.S. (1959). The hydrography of the British East African Coastal Waters. London, East African Marine Fisheries Research Organisation. Fishery Publications, 12, 23p.

Nyandwi, N. (2013). The effects of monsoons on the east African coastal current through the Zanzibar Channel, Tanzania. *J. Ocean Technol*. 8(No4), pp. 64–74

Oettli, P. and Camberlin, P. (2005). Influence of topography on monthly rainfall distribution over East Africa. *Climate Research* 28(3), pp. 199–212. https://doi.org/10.3354/cr028199

Palalane, J., Larson, M., Hanson, H. and Juízo, D. (2016). Coastal erosion in Mozambique: Governing processes and remedial measures. Coastal Erosion in Mozambique. *J. Coastal Res*. 32 (3), pp. 700–718. https://doi.org/10.2112/JCOASTRES-D-14-00020.1

Palastanga, V., van Leeuwen, P.J. and Ruijter, W.P.M.D. (2006). A link between low-frequency mesoscale eddy variability around Madagascar and the large-scale Indian Ocean variability. *J. Geophys. Res*. 111(C9). https://doi.org/10.1029/2005JC003081

Palastanga, V., van Leeuwen, P.J., Schouten, M.W. and Ruijter, W.P.M.D. (2007). Flow structure and variability in the subtropical Indian Ocean. *J. Geophys. Res.* 112, C01001. https://doi.org/10.1029/2005JC003395

Parson, L.M. and Evans, A.J. (2005). Seafloor topography and tectonic elements of the Western Indian Ocean. *Philos. Trans. R. Soc*. 363(1826), pp. 15–24. https://doi.org/10.1098/rsta.2004.1472

Partridge, T.C., Dollar, E.S.J., Moolman, J. and Dollar, L.H. (2010). The geomorphic provinces of South Africa, Lesotho and Swaziland: A physiographic subdivision for earth and environmental scientists. *Trans. Roy. Soc. S. Afr.* 65(1), pp. 1 – 47. https://doi.org/ 10.1080/00359191003652033

Patriat, P., Sauter, D., Munschy, M. and Parson, L. (1997). A Survey of the Southwest Indian Ridge Axis Between Atlantis II Fracture Zone and the Indian Ocean Triple Junction: Regional Setting and Large Scale Segmentation. *Mar. Geophys. Res.* 19, pp. 457–480. https://doi.org/ 10.1023/A:1004312623534

Penven, P., Lutjeharms, J.R.E. and Florenchie, P. (2006). Madagascar: a pacemaker for the Agulhas Current system? *Geophys. Res. Lett*. 33, L17609. https://doi.org/10.1029/2006GL026854

Pepper, J.F. and Everhart, G.M. (1963). The Indian Ocean: *The Geology of Its Bordering Lands and The Configuration of Its Floor.* Tech. rep., Department of the Interior United States Geological Survey

Pollard, R. and Read, J. (2017). Circulation, Stratification and seamounts in the Southwest Indian Ocean. *Deep Sea Res. Part II* 136, pp. 36–43. https://doi.org/ 10.1016/j.dsr2.2015.02.018

Quartly, G.D., Buck, J.J.H., Srokosz, M.A. and Coward, A.C. (2006). Eddies around Madagascar - The Retroflection re-considered. *J. Mar. Sys*. 63, pp. 115–129. https://doi.org/ 10.1016/j.jmarsys.2006.06.001

Quartly, G.D. and Srokosz, M.A. (2004). Eddies in the southern Mozambique Channel. *Deep Sea Res. Part II* 51, pp. 69–83. https://doi.org/ 10.1098/rsta.2002.1108

Raju, K.A.K., Samudrala, K., Drolia, R.K., Amarnath, D., Ramachandran, R. and Mudholkar, A. (2012). Segmentation and morphology of the Central Indian Ridge between 3°S and 11°S, Indian Ocean. *J. Tectonophys*. 554, pp. 114–126. https://doi.org/10.1016/j.tecto.2012.06.001

Ramanantsoa, J. D., Penven, P., Krug, M., Gula, J. and Rouault, M. (2018). Uncovering a new current: The Southwest Madagascar Coastal Current. *Geophys. Res. Lett*., 45, https://doi.org/10.1002/2017GL075900

Read, J. and Pollard, R. (2017). An introduction to the physical oceanography of six seamounts in the southwest Indian Ocean. *Deep Sea Res. Part II* 136, pp. 44–58. https://doi.org/ 10.1016/j.dsr2.2015.06.022

Ridderinkhof, H. and de Ruijter, W.P.M. (2003). Moored current observations in the Mozambique Channel. *Deep Sea Res., Part II* 50, pp. 1933–1955. https://doi.org/10.1016/S0967-0645(03)00041-9

Ridderinkhof, W., Le Bars, D., von der Heydt, A.S. and de Ruijter, W.P.M. (2013). Dipoles of the South-east Madagascar Current. *Geophys. Res. Lett*. 40(3), pp. 558-562. https://doi.org/10.1002/GRL.50157

Risien, C. and Chelton, D.B. (2008). A Global Climatology of Surface Wind and Wind Stress Fields from Eight Years of QuikSCAT Scatterometer Data. *J. Phys. Oceanogr.* 38, pp. 2379–2413. https://doi.org/10.1175/2008JPO3881.1

Roberts, M. and Nieuwenhuys, C. (2016). Observations and Mechanisms of upwelling in the northern Kwazulu-Natal Bight, South Africa. *Afr. J. Mar. Sci.* 38, pp. S43–S63. https://doi.org/ 10.2989/1814232X.2016.1194319

Robinson, A.R. (1983). *Eddies in Marine Science*. Springer-Verlag, 644 pp.

Royer, J. and Schlich, R. (1988). Southeast Indian Ridge Between the Rodriguez Triple Junction and the Amsterdam and Saint-Paul Islands: Detailed Kinematics for the Past 20 m.y. *J. Geophys. Res. Solid-Earth* 93(B11), pp. 13524–13550. https://doi.org/10.1029/JB093iB11p13524

Saji, H.N., Xie, S.P. and Yamagata, T. (2006). Tropical Indian Ocean Variability in the IPCC twentieth-century climate. *J. Clim*. 29(11), pp. 4397–4417. https://doi.org/10.1175/JCLI3847.1

Schott, F.A., Fieux, M., Kindle, J., Swallow, J. and Zantopp, R. (1988). The boundary currents east and north of Madagascar: 2 Direct measurements and model comparisons. *J. Geophys. Res.* 93, pp. 4963–4974. https://doi.org/10.1029/JC093iC05p04963

Schott, A.F., Xie, S.P. and McCreary Jr, J.P. (2009). Indian Ocean Circulation and Climate Variability. *Rev. Geophys*. 47, pp. 1–46. https://doi.org/10.1029/2007RG000245

Schott, F.A. and McCreary, J.P. (2001). The Monsoon circulation of the Indian Ocean. *Prog. Oceanog.* 51, pp. 1–123.

Schouten, M.W., De Ruijter, W.P.M. and van Leeuwen, P.J. (2002). Upstream control of Agulhas ring shedding. *J. Geophys. Res*. 107, C83109. https://doi.org/10.1029/2001JC000804

Schouten, M.W., De Ruijter, W.P.M., van Leeuwen, P.J. and Ridderinkhof, H. (2003). Eddies and variability in the Mozambique Channel. *Deep Sea Res. Part II* 50, pp. 1987–2003. https://doi.org/10.1016/S0967-0645(03)00042-0

Sepulchre, P., Ramstein, G., Fluteau, F., Schuster, M.S., Tiercelin, J.J. and Brunet, M. (2006). Tectonic uplift and esatern africa aridification. *Science* 313, pp. 1419–1423. https://doi.org/10.1126/science.1129158

Shigalla, B.M. and Shaghude, Y.W. (2014). Modelling the dynamics of the Tanzanian coastal waters. *J. Ocean. Marine Science* 5(1), pp. 1–7. https://doi.org/10.5897/joms2013.0100

Siedler, G., Rouault, M., Biastoch, A., Backeberg, B., Reason, C.J.C. and Lutjeharms, J.R.E. (2009). Modes of the southern extension of the East Madagascar Current. *J. Geophys. Res*. 114, C01005. https://doi.org/10.1029/2008JC004921

Siedler, G., Rouault, M. and Lutjeharms, J.R.E. (2006). Structures and origin of the subtropical South Indian Ocean. *Geo- phys. Res. Lett*. 33, L24609. https://doi.org/10.1029/2006GL027399

Small, C., Cochran, J.R., Sempere, J. and Christie, D. (1999). The structure and segmentation of the southeast indian ridge. *Mar. Geolog*. 161, pp. 1–12. https://doi.org/10.1016/S0025-3227(99)00051-1

Spencer, T., Laugton, A.S. and Flemming, N.C. (2005). Variability, interaction and change in the atmosphere-ocean-ecology system of the Western Indian Ocean. *Philos. Trans. R. Soc*. 363, pp. 3–13. https://doi.org/10.1098/rsta.2004.1495

Stramma, L. and Lutjeharms, J.R.E. (1997). The flow field of the subtropical gyre in the South Indian Ocean into the Southeast Atlantic Ocean: A case study. *J. Geophys. Res.* 99, pp. 14053–14070. https://doi.org/10.1029/96JC03455

Swallow, J., Fieux, M. and Schott, F.A. (1988). The boundary currents east and north of Madagascar 1: Geostrophic currents and transports. *J. Geophys. Res*. 93, pp. 4951–4962. https://doi.org/10.1029/JC093iC05p04951

Tomczak, M. and Godfrey, J.S. (1994). Regional Oceanography: An Introduction. *Bull. Amer. Meteor. Soc.* 2, p. 390

Toole, J.M. and Warren, B.A. (1993). A hydrographical section across the subtropical South Indian Ocean. *Deep Sea Res*. 40 (10), pp. 1973–2019. https://doi.org/10.1016/0967-0637(93)90042-2

UMLP (2016). *World Regional Geography: People, Places and Globalization*. Vol. 2, ISBN: 978-1-946135-27-8. Minnesota Library Publishing. https://doi.org/10.24926/8668.2701

Westerberg, L.O. and Christiansson, C. (1999). Highlands in East Africa: Unstable Slopes, Unstable Environments? *Royal Swedish Academy of Sci* 28, p. 5. In *Research for Mountain Area Development: Africa and Asia*, pp. 419–429

Wieners, C., Dijkstra, H.A. and de Ruijter, W.P.M. (2019). The interaction between the western Indian Ocean and ENSO in CESM. *Clim. Dynamic* 52, pp. 5153-5172. https://doi.org/10.1007/s00382-018-4438-2

Wiles, E.A. (2014). Tectonic history, microtopography and bottom water circulation of the Natal Valley and Mozambique Ridge, southwest Indian Ocean. Ph.D. thesis, University of Kwazulu Natal, Durban, South Africa.

Wyrtki, K. (1971). Oceanographic Atlas of the International Indian Ocean Expedition. National Science Foundation.

Wyrtki, K. (1973). An equatorial jet in the Indian Ocean. *Sci.* 181, pp. 262–264. https://doi.org/10.1126/science.181.4096.262

Yang, W., Seager, R. and Cane, M. (2015). The annual cycle of east African precipitation. *J. Clim*. 28, pp. 2385–2404. https://doi.org/10.1175/JCLI-D-14-00484.1

Yossi, M. (1988). The Tectonic Setting of the Seychelles, Mascarene and Amirante Plateaus in the Western Equatorial Indian Ocean. *Mar. Geo.* 79, pp. 261–274. https://doi.org/10.1016/0025-3227(88)90042-4

Zavala-Garay, J., Theiss, J., Moulton, M., Walsh, C., Woesik, R., Mayorga-Adame, C.G., García-Reys, M., Mukaka, D.S., Whilden, K. and Shaghude, Y.W. (2015). On the dynamics of the Zanzibar Channel. *J. Geophys. Res*. 120. https://doi.org/10.1002/2015JC010879

Zinke, J., Pfeiffer, M., Timm, O., Dullo, W., Kroon, D. and Thomassin, B. (2008). Mayotte coral reveals hydro-logical changes in the western Indian Ocean between 1881 and 1994. *Geophys. Res. Lett*. 35, L23707. https://doi.org/10.1029/2008/GL035634,2008

CRITICAL HABITATS

LAND-SEA CONNECTIVITY

Joseph M. Maina

Land-sea interface near Cape Town, South Africa. © UN Environment

BACKGROUND

The WIO coastal zone region includes major cities, harbours, industries and other social and economic infrastructures increasingly affecting the marine environment (Celliers and Ntombela, 2015). These coastal areas are experiencing an accelerated increase in population as people seek job opportunities and better living conditions (Neumann et al., 2015). The mainland states with large catchments adjoining the coast and major cities along the coast are the most prone to the ecological footprints of urbanization. These cities include Mombasa (Kenya), Dar es Salaam (Tanzania), Maputo (Mozambique) and Durban (South Africa), each of which supports populations of 2–4 million people (Diop et al., 2016). Among other pressures, urbanization and forest conversion for agriculture continue to alter hydrological processes and regimes within coastal catchments. These processes underpin land-sea connectivity and all ecological functions and water quality outcomes directly linked to the health of the adjacent marine environment.

Coastal marine ecosystems have developed on a coastline that has delivered low-moderate nutrient concentrations and suspended sediment (Fig. 1). Therefore, the ecosystems have adapted to a specific range of sediment and nutrient conditions influenced by the linked coastal watersheds. Human forest conversion on coastal watersheds to other land uses has altered the annual load of nutrients and suspended sediment exported from the coastal catchments flowing into the marine environment, with a detrimental impact on marine ecosystems (Fleitmann et al., 2007; Maina et al., 2013) (see Fig. 1). For example, reports from Kenya's Sabaki River have estimated an increase in sediment discharge into the Indian Ocean from 1900 to 1990's to be between five and six times (Fleitmann et al., 2007). Increased suspended sediment concentrations have been linked with land-use and soil erosion changes in the Sabaki basin. The actual increases strongly depend on the type of land-use change (Fleitmann et al., 2007).

Despite the ongoing changes occurring to coastal land and pollution impacts on marine ecosystems, implementing the Integrated Coastal Zone Management (ICZM) approach is complicated due to a lack of knowledge of historical trends and baselines of nutrient and sediment emanating land. With no understanding of the critical baselines at discharge locations in the WIO, ICZM is handicapped, owing to attribution challenges for ecosystem condition to changes on land, among other drivers of change. Considering that few catchments in the region are gauged for river flow, let alone sediment load, a key priority for the respective national ICZM and basin management agencies is to establish a sediment and river flow monitoring system. Furthermore, catchments should be subjected to appraisal and evaluation of the amount of nutrients and sediment flowing into the ocean, potential impacts on biodiversity, and how different activities on land, such as agriculture, mining and land clearing in general, and climate change may impact on the fluvial ecology and sediment budget.

Another challenge is to link land-based activities and sediment effluent to changes in marine ecosystems. For example, it is of paramount importance that knowledge of sediment and nutrient thresholds is established for the different linked ecosystems. Attribution of the changes in habitats to pollution can not only create awareness and provide scientific support and a basis necessary for the formulation and implementation of land-sea policy and management actions, but also it would lead to the establishment of critical targets for sediment reduction and measure of success of the management actions.

LAND-SEA CONNECTIVITY IN THE WIO

The concept of land-sea connections in the WIO is dependent on several factors but is primarily driven by hydrological connectivity between freshwater, estuarine and coastal ecosystems (Fig. 1). The interface between the coastal, estuarine and freshwater system is a very productive component of the food chain and a critical corridor for movement between ecosystems (Sheaves et al., 2015). Several marine and estuarine fish species use the freshwater systems for part of their life cycle. For example, mangroves and estuaries are vital habitats that support the life cycle of many shrimp species (eg Abreu et al., 2017), reflected in the detailed studies from Kenya (Munga et al., 2007; Fulanda et al., 2011) that demonstrated the significance of land-sea connectivity to shrimp species diversity and fisheries in Kenya's Tana and Sabaki Estuaries. Also, in Mozambique, there is evidence of the link between catchment discharges and shrimp commercial catches in several estuaries, such as the Zambezi delta (Gammelsrød, 1992) and the Maputo Bay catchments (Bacaimane and Paula e Silva, 2014; Nordez, 2014).

The freshwater flows from the various waterways also serve as the primary delivery mechanism for materials that runoff the catchment, including a range of pollutants. These physical exchanges underpin functional

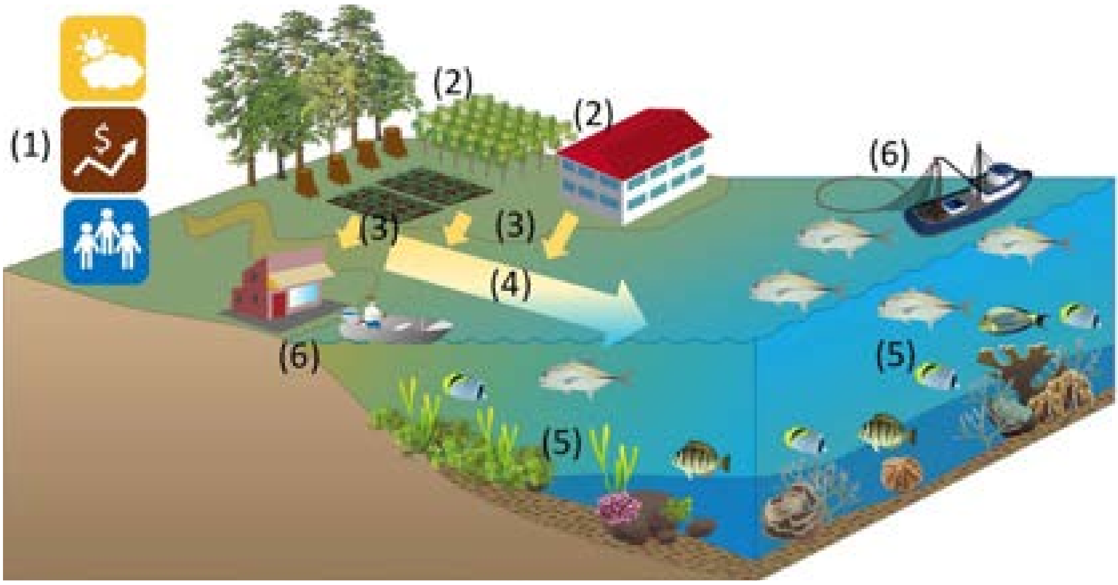


Figure 1: Land-sea connections (source: Brown et al., 2019).

(1) Climate, economic and societal drivers of land-use change.
(2) Human activities that change pollutant runoff, including forestry, agriculture and urbanisation.
(3) Sediment and nutrient runoff from activities on land enter streams and eventually the ocean.
(4) Resulting changes in water quality as pollutants are dispersed and transformed in the ocean.
(5) Changes in marine ecosystems and fished populations, including interactions between predators, prey and between fished species and their habitats.
(6) Impacts of ecological change on fisheries and social and economic responses to change in fisheries.

land-sea connectivity. Therefore, understanding the physical connectivity in terms of catchment dynamics, outlets, sediment and flow volumes, and sensitivity to climate and land use is critical to managing the functional connectivity. Depending on each catchment characteristics and type of land-sea (estuarine) interface, the mixing of waters of different origin occurs with the associated transformation of matter from land origin through diverse biogeochemical processes (eg Meybeck and Dürr, 2009; Dürr et al., 2011). The coastal interfaces are strongly affected by the whole catchment, and freshwater runoff, erosion and biogeochemical processes at basin scale modulate downstream characteristics and their variability at varying temporal scales, as seen for coastal resources within the Maputo Bay in Mozambique (eg Monteiro and Marchand, 2009). In addition, the inherent optical and chemical characteristics of the terrestrially sourced plume entering the ocean, such as nutrients, salinity and sediment load, and associated organisms, influence the adjacent marine environment through the transformation of dissolved and particulate materials (eg Frankignoulle et al., 1998; Dagg et al., 2004). Although most continental drainage effects on the coastal zone come from rivers and associated coastal interfaces, diffuse groundwater flows are still largely unknown.

River basins in the tropical world can broadly be classified based on climate, primarily rainfall, wet tropics basins (WTBs) and dry tropics basins (Latrubesse et al., 2005). The WTBs are characterized by wetter climates with average annual rainfall exceeding 3000 mm, intensive agricultural land uses and their associated fertilizer and pesticide loads. In the WIO, examples of WTBs include regions in the low-mid latitude, including northern Mozambique, Tanzania, Kenya and the southern parts of Somalia, that receive high-intensity rainfall during the wet season from March through May (Scheren et al., 2016). In addition, oceanic islands, such as those of Seychelles and other island states, receive high-intensity rainfall strongly influenced by the monsoon (FAO, 2005). Changes in rainfall patterns and human activities in wet and dry tropics have contributed to changes in sediment regimes along WIO coastlines. In contrast, catchments in arid tropical areas, for example, south-western Madagascar and drier parts of southern Kenya (Lower Tana basin), have average annual rainfall in the 500–750 mm range. Such areas tend to be dominated by original savannah/woodland rangeland.

While the flow volume determines river-borne sediment transport capacity, the rainfall intensity influences the erosion potential and dislodgement of soil particles. Thus, rivers draining high and intense rainfall watersheds exhibit higher flow and sediment discharges. Reports indicate that catchments in the wet tropics are very sensitive to deforestation, with minimal forest decline leading to large river runoff and sediment discharge (Maina et al., 2013). In dry catchments, however, sedimentation is less sensitive to forest decline, as is the case in southwestern Madagascar. A large degree of deforestation led to a marginal decline in sediment discharge (Maina et al., 2013). According to Scheren et al. (2016), in the northern parts of the WIO region (eg Somalia and Kenya), the estimated total annual river discharge is in the range 1.8–4.95 km^3/yr. River discharge volume along the WIO coastline (eg Tanzania, Mozambique and South Africa) is estimated to be in the range of 2.9–106 km^3 (Scheren et al., 2016). These large sediment loads discharged to the ocean have generated estuarine formations, particularly in the southern parts of the WIO region (Mozambique) with extensive mangrove forest development (Taylor et al., 2003; Scheren et al., 2016).

HYDROLOGICAL LANDSCAPE

Rivers discharge nodes are the main features that connect coastal catchments to marine ecosystems. These could be large perennial all-season rivers or seasonal rivers active during flash floods or the wet season. Similarly, the watersheds drained by these rivers could be large basins or smaller watersheds within the larger basin. Most national water management bodies are based on a basin-scale; for example, Tanzania's Rufiji Basin Water Office and Kenya's Tana Basin Corporation manage the water at the basin level. Consequently, hydrological reports and data are aggregated at the basin level, which in most cases can be too coarse for their consideration as land-sea connectivity units. On the other hand, small-medium catchments within larger basins are ideal for land-sea management, given the scale at which activities on land take place (eg small scale farming) and the significance of smaller sub-catchment in erosion and transportation of sediment.

Despite the ecological significance of smaller catchments, the management focus has been on larger basins and estuaries. In Chapter 11, 12 large river basins are described and presented as the main estuaries in the WIO. However, many estuaries that are not considered large but are of significance nevertheless are largely undocumented. Small estuaries are mainly fed by seasonal rivers and are distributed across many WIO countries. They are characterized by inactivity during the dry season, but they discharge a large runoff during the wet season. The freshwater flows from the various rivers profoundly affect coastal marine ecosystems in the region, driving multiple ecological processes and providing nutrients for many biota (Kairu and Nyandwi, 2000). Given the significance of land-sea connectivity, freely available topography data was interrogated to determine the spatial distribution of all estuaries, including those that are small and seasonal.

To characterize the catchment ecosystem in the region to a scale relevant to various features of physical and biological connections, spatial data on watersheds was downloaded from the Hydrosheds website (http://www.hydrosheds.org). This database was delineated from the Remote Sensing derived elevation data (Shuttle Radar Topographic Mission-SRTM) (Lehner et al., 2008). The data obtained was used to evaluate the number and size of catchments and main rivers draining the catchments. From these, a total of 83 river discharge points were delineated for continental WIO and Madagascar, draining a capacity of 72 (sub) watersheds (Fig. 2). For small islands in the WIO, catchment and rivers' data were unavailable in the SRTM watershed product.

LINKED MARINE ECOSYSTEMS

Coral reefs

Coral reefs are described in detail in Chapter 10. Natural land-ocean linkages through runoff and sedimentation have been altered due to increased sediments and pollutants deposited in the coastal waters to the detriment of coral reefs. Among the leading causes of the global coral decline, terrestrially sourced pollutants rank as the top causes (Gardner et al., 2003). Modification of terrestrial sediment fluxes can increase sedimentation and turbidity in receiving waters, with detrimental impacts on coral reef ecosystems. Preventing anthropogenic sediment from reaching coral reefs requires a better understanding of the specific characteristics, sources and processes generating the anthropogenic sediment so that effective watershed management strategies can be implemented. This information, however, is unavailable for most catchments. At the basic level, linkages between sedimentation and coral reef decline in the region need to be demonstrated empirically to establish thresholds.

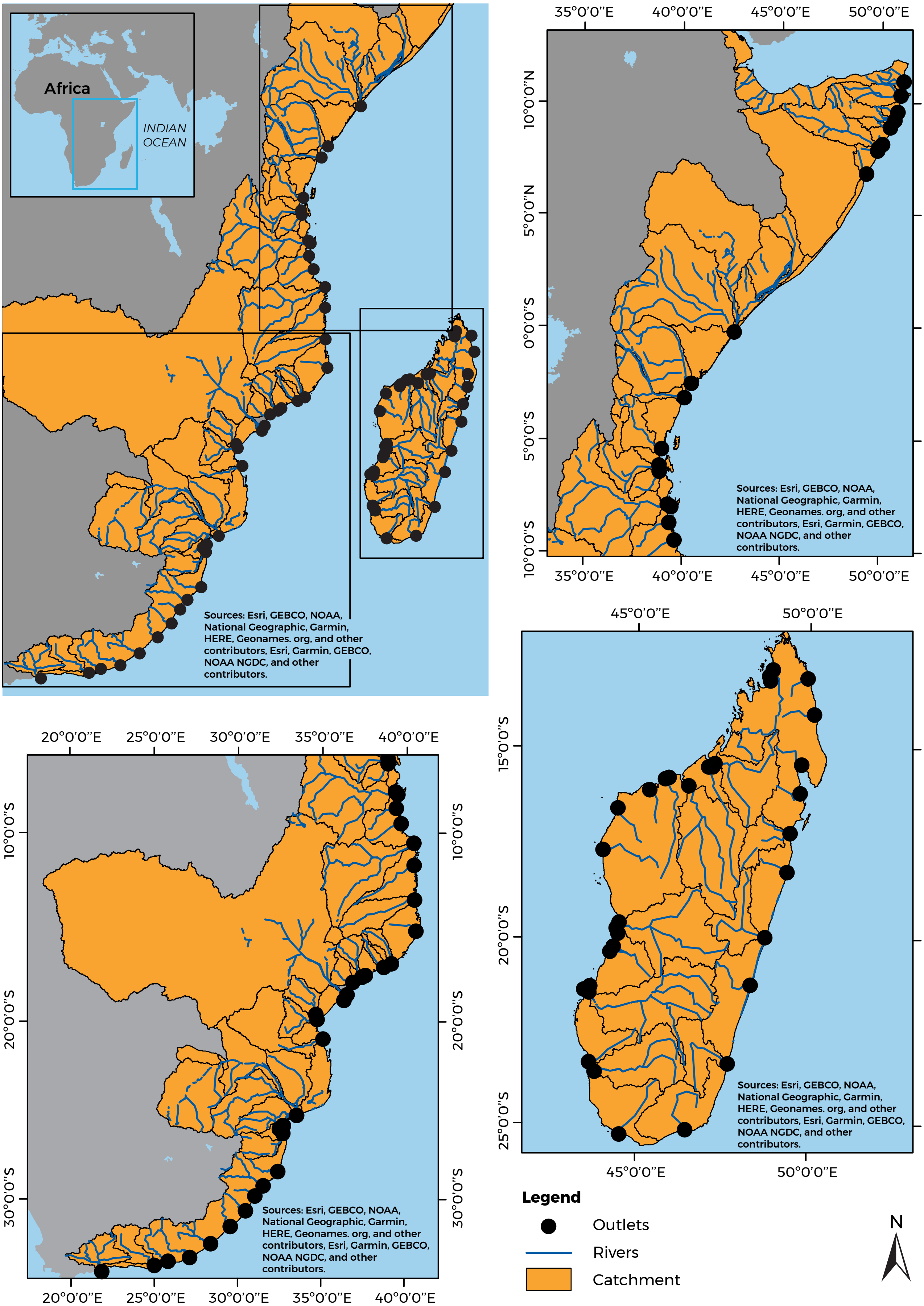


Figure 2: Distribution of 83 discharge points from all types of rivers in the continental WIO and Madagascar.

Establishing sediment concentration thresholds at which coral reefs begin to deteriorate is necessary for management intervention.

Assessments of sediment impacts on reefs and linkages between catchments and coral reefs have been undertaken for various locations in the region. For example, in a study in Kenya, coral diversity and evenness decreased, and the dominance index increased due to the selective survivorship of coral species resistant to elevated sediment (for example, *Millepora* sp.) (McClanahan and Obura, 1997). Another study revealed high sedimentation rates on the Malindi reef emanating from the Sabaki River, which did not impact coral recruitment and general coral health (Mwachireya et al., 2015). However, increased hydrodynamics and enhanced flushing rates have been credited for the nuanced observations of sediment impacts on reefs. A more recent threat, however, is the synergy between global and local stressors. The global coral decline has been attributed to interacting with multiple international and local disturbances. These include terrestrially sourced pollutants (Gardner et al., 2003; Wilkinson, 2004); overfishing and loss of herbivores (Mumby et al., 2006); and climate-related changes in sea surface temperature and acidity (Fabricius et al., 2011; Hoegh-Guldberg et al., 2007) among others. These compound drivers may interact with varying consequences in different coral reef systems. Furthermore, they operate at different spatial scales, where local factors may exacerbate the effects of global processes, and at different temporal scales, where longer-term trends may be obscured by short-term, inter-annual or seasonal variability (Chabanet et al., 2005; Habeeb et al., 2005). Simultaneous assessments of drivers of change may provide insight into the inter-linkages and relationships between physical and biological processes in coastal watersheds and the adjacent coral reefs.

Mangroves

Mangrove forests are described in detail in Chapter 7. The WIO mangroves, with an estimated coverage of 1 million hectares, represents 5 per cent of the global mangrove coverage (Bosire, 2016). The dense distribution of mangrove forests in the WIO occurs in deltas and estuaries (Spalding, 2010; Hamilton and Casey, 2016). The nature of mangrove distribution and easy accessibility has exposed them to unprecedented human pressure in recent years. Human activities such as reclamation for expansion of residential housing, tourist installations and agriculture; commercial or artisanal extraction of wood for timber, fuelwood and poles; and freshwater diversion are happening to the detriment of mangrove ecosystems. Deforestation of coastal watersheds throughout the WIO has altered water, nutrients, and sediments to mangrove estuaries and coastal oceans. This happens because of robust couplings linking land-use changes on upland watersheds to receiving aquatic ecosystems down the topographical gradient.

Eutrophication is one of the major causes of coastal ecosystem degradation. Eutrophication leads to an increase in the occurrence of algal blooms (Paerl, 1997), degradation of coral reefs (Lapointe, 1997) and reductions in seagrass cover (Van Katwijk et al., 2011). Persistent eutrophication can also adversely affect mangroves, which have the potential to assimilate nutrients in eutrophicated coastal environments (Robertson and Phillips, 1995). Lovelock et al. (2009) have suggested that nitrogen enrichment may reduce the resilience of mangroves to environmental stress, thereby increasing mortality. Nitrogen enrichment of terrestrial and coastal ecosystems produces similar effects. In a terrestrial system with little or no harvesting, for example, wooded semi-natural terrestrial systems, nitrogen may be taken up into the local nutrient cycles. Nitrogen enrichment of the terrestrial forest floor usually increases growth and nutrient (soil and foliar) levels (Lovelock et al., 2014). Similar effects have been found in mangrove ecosystems. Mangroves showed enhanced growth and increased foliar nitrogen (N) and phosphorus (P) concentrations under P enrichment conditions. The addition of N and P (each 300 g/tree) to mangrove soils resulted in increases of up to 30 per cent for foliar N and 40–100 per cent for foliar P levels. In a similar study, Boto and Wellington (1983) reported that the addition of N and P (each 100 kg/ha) increased mangrove foliar N and P levels by 22 and 7 per cent, respectively.

Estuarine wetlands

Estuaries are assessed in detail in Chapter 11. Estuarine habitats may include forests (mangroves), coastal saltmarshes (grass, sedge and herb swamps), salt flats and saltpans, mudflats and intertidal seagrass ecosystems. Beneath the water, estuarine habitats can include soft-bottom communities, hard-bottom communities, and ecosystems dominated by coral and seagrass. Estuaries are located at the terminus of coastal catchments and receive runoff and contained loads of sediment, nutrients and other contaminants from contributing catchment areas. Due to these biophysical linkages, the condition of an estuary is mediated by the state of its catchment to varying degrees. Therefore, estuaries are susceptible to catchment land use and development that alter freshwater flows or elevate loads of sediment, nutrient, and other contaminants exported downstream (see Chapter 11).

This affects their susceptibility to water quality impacts associated with contaminant loads in the runoff.

Estuarine ecosystems in the WIO are exposed to extreme environmental conditions, from large freshwater flows during the wet season leading to hyposalinity to hypersaline conditions caused by the cessation of flows and evaporation during the dry season. Estuarine ecosystems have adapted to these conditions but are dependent on connectivity and tidal exchange for ongoing health and resilience. Changes to river flow regimes and tidal connectivity between individual habitat components can cause phase shifts in estuarine communities. Recovery time from disturbance can be as long as 20 years. Saltmarsh communities are generally more susceptible to human disturbance than mangrove areas.

PLANNING FOR LAND-USE CHANGE

It is inevitable that with competing demand for land and ongoing climatic change, optimal land allocation will increasingly become complicated and will require decision support planning tools for prioritization. Like elsewhere in the world, marine management in the WIO will need to plan for conservation amid the ongoing global environmental changes, including climate change and conversion of forest land to agriculture and economic development in coastal catchments. Decision support tools, including the land-sea models, are indispensable tools for conservation planning (Brown et al., 2017). Given the multidisciplinary nature of the land-sea environment, different tools will need to be coupled to cover other sectors of the land-sea continuum. For example, hydrological modelling may need to be linked with coastal hydrodynamics to determine where the sediment comes from and where it disperses to post-discharge. Similarly, socio-ecological and economic models would be required to quantify the impacts of sediment on ecosystems and the socio-economic consequences. Incorporating this information into a quantitative planning framwork provides a transparent and repeatable approach to land-sea planning (Game et al., 2013).

A review of studies across the region reveals no precedence on coupled integrated land-sea spatial planning, exposing critical knowledge and management gaps. Elsewhere in the tropical world, land-sea planning has been applied to manage the Great Barrier Reef (GBR) catchments in Australia to reduce sedimentation from impacting the GBR (Dale et al., 2017). Table 1 provides examples of research studies along the land-sea continuum in the WIO. From this review, it is clear that most of the studies in the region are on establishing the impacts of sediment on ecosystems and quantifying the amount of river flow and/or sediment discharge from catchments. The table also highlights the gaps, especially in integrating all the datasets for spatial planning.

Table 1: Examples of quantitative studies along the land-sea continuum have linked the land-use change to coastal ecosystems and fisheries. Dark boxes indicate steps where a specific quantitative model was used; empty boxes indicate no quantitative model was used, though that step may have been considered conceptually (adapted and contextualized for the WIO after Brown et al., 2017).

EXAMPLE:	DRIVERS OF LAND-USE CHANGE	HUMAN ACTIVITIES THAT CAUSE LAND-USE CHANGE	FRESH WATER AND POLLUTANTS IN THE OCEAN	DISPERSION OF POLLUTANTS IN THE OCEAN	ECOSYSTEM RESPONSE	SOCIO-ECONOMIC RESPONSE	LAND-SEA SPATIAL PLANNING
Madagascar, coral reefs	■	■	■	■	■	■	
Rufiji Basin, mangroves						■	
Athi/Sabakibasin (Malindi), coral reefs	■					■	
Tana Basin, mangroves							
Pangani Basin (Tanga)		■	■				

References for example: Maina et al., 2012; Maina et al., 2013; Minu et al., 2020; McClanahan and Obura 1997; Fleitmann et al., 2007; Ndomba, 2010

CASE STUDY

Groundwater linkages to nearshore marine areas

While rivers form the most obvious land-sea connectivity pathway, another important pathway is submarine groundwater discharge (SGD) – a hydrogeological process by which groundwater enters the sea. The global sub-surface flux is approximately 10 per cent of the gross river discharge (Taniguchi et al., 2002). This process, increasingly recognized as a nutrient and pollutant pathway from land to sea, transports bioactive solutes, including nutrients (nitrogen, phosphorous, silica), gases (methane, carbon dioxide), and trace metals (iron, nickel, zinc) (Moosdorf and Oehler, 2017). Nutrient addition to nearshore marine ecosystems can be beneficial because their availability largely controls primary production (Duarte et al., 2010). Conversely, reduced salinity may also undermine productivity, as salinity is an important physiochemical attribute for nearshore biodiversity. Thus the distribution and abundance of marine life may also be directly affected by SGD (Krause-Jensen et al., 2008). Coastal aquifer salinization from seawater intrusion is a common phenomenon in many coastal cities. When groundwater is pumped from coastal aquifers, potential SGD is intercepted, disrupting the natural equilibrium and causing the freshwater-seawater interface to locally migrate landward and/or vertically upward (Manivannan and Elango, 2019). This Case Study focuses on both terrestrial groundwater discharges to the sea (terrestrial fraction or fresh SGD) and marine fractions (saline SGD), and the environmental and biological impacts based on WIO studies.

In the WIO region, few case studies of SGD exist, including one on Mauritius on the impacts of the micro-environmental conditions caused by SDG on reef fish (Povinec et al., 2012; Lilkendey et al., 2019). They found that physiologically favorable conditions created by SGD elevate the survival potential of marine fish (Lilkendey et al., 2019). Given the observed benefits, their study highlights the need for ground-water fluxes to be included in environmental management plans, with regards to addressing potential future challenges such as trade-offs between anthropogenic freshwater needs and coastal fisheries productivity. In neighboring Reunion, groundwater discharge onto coral reefs, in particular La-Saline reef, was discovered in early 1980s (Naim, 1993). Subsequent studies have reported coral cover decline and algal overgrowth (eg Chabanet et al., 2002; Chazottes et al., 2002), attributed to nutrients from SGD causing eutrophication on this reef. This can complicate the management of coral reefs, considering that nutrient reduction is one of the commonly recommended strategies for enhancing coral reef resilience to climate change. At a global scale, a recent high-resolution estimate of SGD flux indicated that 23 per cent of the global coastline is at risk of eutrophication by terrestrially derived groundwater (Luijendijk et al., 2020). Based on the report, some of the high-risk areas in the WIO include north-eastern Madagascar, central Mozambique and Dar as Salaam in Tanzania and parts around Durban in South Africa (Luijendijk et al., 2020).

As coastal cities in the WIO witness increasing population and industrial development, demand for water increases and, in most cases, surpasses the capacity of governments to provide. Consequently, WIO coastal cities experience high groundwater exploitation (Bakari et al., 2012). Intensive use of coastal aquifers often results in their salinization from seawater intrusion. A case study for Dar es Salaam simulated the different pathways of saltwater intrusion and found that intrusion depended on depth of wells and their distance from the coastline (Van Camp et al., 2014). The study demonstrated the importance of formulating and enforcing evidence-based recommendations when drilling new wells for a better monitoring of the salinization process (Van Camp et al., 2014). The overdependence on SGD, and its active use by coastal populations demonstrates its role for coastal societies (Luijendijk et al., 2020). In the WIO, fresh SGD is widely valued as a water resource for drinking, hygiene, agriculture and culture, among other uses. For example, in Quissico, Mozambique, locals use intertidal springs for bathing and laundry (Moosdorf and Oehler, 2017).

Despite the wide-ranging benefits, the region lacks adequate policies for safeguarding the integrity of the SGD. Furthermore, hydrogeological knowledge is fragmented, groundwater lacks a long-term monitoring infrastructure and information transfer to water users is limited. A logical step towards sustainable use of SGD is to incorporate it within the Integrated Coastal Zone system and as part of the environmental flow, such that its role in the system is clearly outlined and considered in the formulation of relevant policies.

CONCLUSION AND FUTURE DIRECTIONS

The strong linkages between land-based activities and nearshore marine ecosystems and associated socio-economics demand that marine resource management evolve to consider human activities on land. The complexity of processes linking basin land-use change to change in coastal ecosystems hinders effective integrated land-sea planning. Overcoming this complexity can be facilitated through efforts to integrate models from the drivers of land-use change to management responses for marine ecosystems.

Based on the critical knowledge gaps identified in this review, the following future research directions for connecting land and sea models and actions that could be taken that will assist integrated land-sea planning are proposed:

1. A thorough scientific assessment of the hydrological processes within the coastal catchments in the WIO to be carried out, including the seasonal to annual water balances, streamflow and hydrograph characteristics, the role of land-use in runoff processes, ground and surface water interactions from the hillslopes to the lower floodplains, overland flow extent and floodplain inundation frequencies, the role of in-stream storages (dams and weirs) in the catchment hydrology, and the dependence of event scale variability on various synoptic processes (for example, ENSO and dipole).

2. An analysis of climatic and streamflow trends within the coastal catchments of the WIO, including statistical tests for changes in sediment and river discharge, is needed.

3. An analysis of the sensitivity of streamflow to other changes in the catchment water and energy balances, especially precipitation and vegetation, and the feedback between them.

4. A detailed analysis of the overland flow (floodplain) transport pathways where nutrient addition is of primary concern, including their inundation, flow hydraulics, infiltration, changes in water quality, and the subsequent return flow to river channels or the coast. This would also consider the relation of these processes to the river channel hydrograph and the relative contribution of return flow from floodplains to flood plumes, ultimately reaching the nearshore marine ecosystems.

5. A comprehensive study on the coupling between nutrient kinetics and the hydrological transport processes in the landscape in selected watersheds. This would include critical biogeochemical kinetic factors (organic matter, dissolved oxygen, microbial processing), the role of event and seasonal hydrology in nutrient export, and how this links with the surface, hyporheic zone, and groundwater exchanges and flow paths.

6. Establish the impact of sediment and nutrient pollution (or lack of) on marine ecosystems and determine the socio-economics and livelihoods consequences of sediment pollution. This would facilitate the trade-off between land-based activities based on the potential impacts on ecosystems and livelihoods.

7. Finally, for long-term management effectiveness, it is essential to establish a robust monitoring network, preferably where streamflow records are already continuously monitored, that would determine surface, groundwater and hyporheic zone water exchanges, continuously monitor key kinetic determinants of nutrient concentrations (organic matter fluorescence, dissolved oxygen). Monitoring sites would also serve as locations for surface water and groundwater sampling.

Creek at the Ruvuma delta, Quionga, Mozambique.
© José Paula

These actions provide the critical information necessary to understand how catchment processes impact the marine nearshore environments. However, they also proceed in a logical, independent order: with actions 1–3 enabling areas of *critical hydrological function* within the WIO coastal catchments to be effectively identified, and actions 4–5 enabling areas of *critical biogeochemical function* to be identified. If these can be achieved in combination, it is possible to establish areas requiring further investigation as an *immediate priority for marine biodiversity protection*. Therefore, it would be the target of action 7. Finally, the complexity of comprehensive modelling of linked land-sea processes should not hold back the development of management plans.

A pragmatic way to proceed in the absence of planning tools that account for land-sea impacts is to devise strategies using expert input and then evaluate ecological and socio-economic outcomes post-hoc using existing modelling tools. Quantitative planning for the effects of land-use change on coastal fisheries requires linking models across many disciplines. Doing so can be a challenge for the small teams often tasked with developing land-sea plans. Addressing the research challenges outlined above should help those teams create plans that focus on outcomes, like fish yield, rather than more abstract objectives of reducing the threat. Outcome-driven planning is likely to be more effective for driving land-sea plans and evaluating competing trade-offs.

REFERENCES

Abreu, D.C., Paula, J. and Macia, A. (2017). Tropical seascapes as feeding grounds for juvenile penaeid shrimps in southern Mozambique revealed using stable isotopes. *Estuar. Coast. Shelf Sci.* 198, pp. 21−28. https://doi.org/10.1016/j.ecss.2017.08.040

Bacaimane, C. and Paula e Silva, R. (2014). Influence of the precipitation and river runoff on the semi-industrial shrimp catches in Maputo Bay, pp. 285−286. In *The Maputo Bay Ecosystem*. WIOMSA, Zanzibar Town, 427 pp.

Bakari, S.S., Aagaard, P., Vogt, R.D., Ruden, F., Johansen, I. and Vuai, S.A. (2012). Delineation of groundwater provenance in a coastal aquifer using statistical and isotopic methods, Southeast Tanzania. *Environ. Earth Sci.* (3), pp. 889-902

Bosire, J.O. (ed.) (2016). Mangroves of the Western Indian Ocean: Status and Management. WIOMSA, Zanzibar

Boto, K.G. and Wellington, J.T. (1983). Phosphorus and nitrogen nutritional status of a northern Australian mangrove forest. *Mar. Ecol. Prog. Ser.* 11(1), pp. 63-69

Brown, C.J., Jupiter, S.D., Lin, H.Y., Albert, S., Klein, C., Maina, J.M., Tulloch, V.J., Wenger, A.S. and Mumby, P.J. (2017). Habitat change mediates the response of coral reef fish populations to terrestrial runoff. *Mar. Ecol. Prog. Ser.* 576, pp.55−68

Brown, C.J., Jupiter, S.D., Albert, S., Anthony, K.R., Hamilton, R.J., Fredston-Hermann, A., Halpern, B.S., Lin, H.Y., Maina, J., Mangubhai, S. and Mumby, P.J. (2019). A guide to modelling priorities for managing land-based impacts on coastal ecosystems. *J. Appl. Ecol.* 56(5), pp. 1106−1116

Celliers, L. and Ntombela, C. (2015). Urbanisation, coastal development and vulnerability, and catchments, pp. 377−394. in *Regional State of the Coast Report: The Western Indian Ocean*. UNEP and WIOMSA, Nairobi

Chabanet, P., Bigot, L., Naim, O., Garnier, R., Tessier, E. and Moyne-Picard, M. (2002). Coral reef monitoring at Reunion island (Western Indian Ocean) using the GCRMN method. *In Proceedings of the Ninth International Coral Reef Symposium, Bali, 23−27 October 2000*. Vol. 2, pp. 873−878

Chabanet, P., Adjeroud, M., Andréfouët, S., Bozec, Y.M., Ferraris, J., Garcìa-Charton, J.A. and Schrimm, M., (2005). Human-induced physical disturbances and their indicators on coral reef habitats: A multi-scale approach. *Aquat. Living Resour.* 18(3), pp. 215−230

Chazottes, V., Le Campion-Alsumard, T., Peyrot-Clausade, M. and Cuet, P. (2002). The effects of eutrophication-related alterations to coral reef communities on agents and rates of bioerosion (Reunion Island, Indian Ocean). *Coral reefs* 21(4), pp. 375-390

Dagg, M., Benner, R., Lohrenz, S. and Lawrence, D. (2004). Transformation of dissolved and particulate materials on continental shelves influenced by large rivers: plume processes. *Cont. Shelf Res.* 24, pp. 833−858

Dale, A. P., Vella, K., Gooch, M., Potts, R., Pressey, R. L., Brodie, J. and Eberhard, R. (2017). Avoiding implementation failure in catchment landscapes: a case study in the governance of the Great Barrier Reef. *Environ. Manage.* pp. 1−12

Diop, S., Scheren, P. and Machiwa, J.F. (eds.) (2016). *Estuaries: A Lifeline of Ecosystem Services in the Western Indian Ocean.* Springer

Duarte, T.K., Pongkijvorasin, S., Roumasset, J., Amato, D. and Burnett, K. (2010). Optimal management of a Hawaiian Coastal aquifer with nearshore marine ecological interactions. *Water Resour. Res.* 46(11)

Dürr, H.H., Laruelle, G.G., van Kempen, C.M., Slomp, C.P., Meybeck, M. and Middelkoop, H. (2011). Worldwide typology of nearshore coastal systems: defining the estuarine filter of river inputs to the oceans. *Estuaries Coasts* 34, pp. 441−458

Fabricius, K.E., Langdon, C., Uthicke, S., Humphrey, C., Noonan, S., De'ath, G., Okazaki, R., Muehllehner, N., Glas, M.S. and Lough, J.M. (2011). Losers and winners in coral reefs acclimatized to elevated carbon dioxide concentrations. *Nat. Clim. Change* 1(3), pp. 165−169

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

Fleitmann, D., Dunbar, R.B., McCulloch, M., Mudelsee, M., Vuille, M., McClanahan, T.R., Cole, J.E. and Eggins, S. (2007). East African soil erosion recorded in a 300 year old coral colony from Kenya. *Geophys. Res. Lett.* 34(4), pp. 1737−1739

Frankignoulle, M., Abril, G., Borges, A., Bourge, I., Canon, C., Delille, B., Libert, E. and Theate, J. M. (1998). Carbon dioxide emission from European estuaries. *Science* 282, pp. 434−436

Fulanda, B., Ohtomi, J., Mueni, E. and Kimani, E. (2011). Fishery trends, resource-use and management system in the Ungwana Bay fishery Kenya. *Ocean Coast. Manag.* 54(5), pp. 401−414

Game, E.T., Kareiva, P. and Possingham, H.P. (2013). Six common mistakes in conservation priority setting. *Conserv. Biol.* 27(3), pp. 480−485

Gammelsrød, T. (1992) Variation in shrimp abundance on the Sofala bank, Mozambique, and the Zambezi river runoff. *Estuar. Coast. Shelf Sci.* 39, pp. 91−103

Gardner, T.A., Côté, I.M., Gill, J.A., Grant, A. and Watkinson, A.R. (2003). Long-term region-wide declines in Caribbean corals. *Science* 301(5635), pp. 958−960

Habeeb, R.L., Trebilco, J., Wotherspoon, S. and Johnson, C.R. (2005). Determining natural scales of ecological systems. *Ecological Monographs* 75(4), pp. 467−487

Hamilton, S.E. and Casey, D. (2016). Creation of a high spatio-temporal resolution global database of continuous mangrove forest cover for the 21st century (CGMFC-21). *Glob. Ecol. Biogeogr.* 25(6), pp. 729−738

Hoegh-Guldberg, O., Mumby, P.J., Hooten, A.J., Steneck, R.S., Greenfield, P., Gomez, E., Harvell, C.D., Sale, P.F., Edwards, A.J., Caldeira, K. and Knowlton, N. (2007). Coral reefs under rapid climate change and ocean acidification. *Science* 318(5857), pp. 1737−1742

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

Krause-Jensen, D., Sagert, S., Schubert, H. and Boström, C. (2008). Empirical relationships are linking distribution and abundance of marine vegetation to eutrophication. *Ecol. Indic.* 8(5), pp. 515−529

Lapointe, B.E. (1997). Nutrient thresholds for bottom-up control of macroalgal blooms on coral reefs in Jamaica and southeast Florida. *Limnol. Oceanogr.* 42(5part2), pp. 1119−1131

Latrubesse, E.M., Stevaux, J.C. and Sinha, R. (2005). Tropical rivers. *Geomorphology* 70 (3), pp. 187−206

Lehner, B., Verdin, K. and Jarvis, A. (2008). New global hydrography derived from spaceborne elevation data. *AGU Publications* 89(10), pp. 93−94

Lilkendey, J., Pisternick, T., Neumann, S.I., Dumur Neelayya, D., Bröhl, S., Neehaul, Y. and Moosdorf, N. (2019). Fresh submarine groundwater discharge augments growth in a reef fish. *Front. Mar. Sci.* 6, p. 613

Lovelock, C.E., Adame, M.F., Bennion, V., Hayes, M., O'Mara, J., Reef, R. and Santini, N.S. (2014). Contemporary rates of carbon sequestration through vertical accretion of sediments in mangrove forests and saltmarshes of South East Queensland, Australia. *Estuaries Coasts* 37(3), pp. 763−771

Lovelock, C.E., Ball, M.C., Martin, K.C. and Feller, I.C. (2009). Nutrient enrichment increases mortality of mangroves. *PLoS One* 4(5), p. e5600

Luijendijk, E., Gleeson, T. and Moosdorf, N. (2020). Fresh groundwater discharge insignificant for the world's oceans but important for coastal ecosystems. *Estuaries Coasts* 11(1), pp. 1−12

Maina, J., de Moel, H., Vermaat, J.E., Bruggemann, J.H., Guillaume, M.M., Grove, C.A., Madin, J.S., Mertz-Kraus, R. and Zinke, J. (2012). Linking coral river runoff proxies with climate variability, hydrology and land-use in Madagascar catchments. *Mar. Pollut. Bull.* 64(10), pp. 2047−2059

Maina, J., De Moel, H., Zinke, J., Madin, J., McClanahan, T. and Vermaat, J.E. (2013). Human deforestation outweighs future climate change impacts of sedimentation on coral reefs. *Nat. Commun.* 4, p. 1986

Manivannan, V. and Elango, L. (2019). Seawater intrusion and submarine groundwater discharge along the Indian coast. *Environ. Sci. Pollut. Res.* pp. 1−17

McClanahan, T.R. and Obura, D. (1997). Sedimentation effects on shallow coral communities in Kenya. *J. Exp. Mar. Biol. Ecol.* 209(1−2), pp. 103−122

Meybeck, M. and Dürr, H.H. (2009). Cascading filters of river material from headwaters to regional seas: the European example, pp. 115−139. In *Watersheds, Bays, and Bounded Seas – the science and management of semi-enclosed marine systems* (eds. Sundby, B., Malanotte-Rizzoli, P. and Melillo, J.M.) Island Press, Washington

Minu, A., Routh, J., Machiwa, J. F. and Pamba, S. (2020). Spatial variation of nutrients and primary productivity in the Rufiji Delta mangroves, Tanzania. *Afr. J. Mar. Sci.* 42(2), 221−232

Monteiro, P.M.S. and Marchand, M. (eds.) (2009). *Catchment2Coast: a systems approach to coupled river-coastal ecosystem science and management*. IOS Press, pp. 75. Amsterdam

Moosdorf, N. and Oehler, T. (2017). Societal use of fresh submarine groundwater discharge: An overlooked water resource. *Earth Sci. Rev.* 171, pp. 338−348

Mumby, P.J. (2006). The impact of exploiting grazers (Scaridae) on the dynamics of Caribbean coral reefs. *Ecol. Appl.* 16(2), pp. 747−769

Munga, D., Mwangi, S., Kamau, M.M., Nguli, P.O., Gwada. L.N., Daudi, H., Ong'anda, S.M., Mwaguni, H.S., Massa, M., Tole, J.M., Onyari, J., Makopa, A., Gachanja, G., Opello, A., Kheir, M. and 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, 56 pp. Nairobi, Kenya

Mwachireya, S.A., McClanahan, T.R., Hartwick, B.E., Cote, I.M. and Lesack, L. (2015). Growth and corallite characteristics of Kenyan scleractinian corals under the influence of sediment discharge. *Int. J. Biodivers. Conserv.* 7(8), pp. 364−377

Naim, O. (1993). Seasonal responses of a fringing reef community to eutrophication (Reunion Island, Western Indian Ocean). *Mar. Ecol. Prog. Ser.* 99, pp. 137−151

Ndomba, P.M. (2010). Modelling of sedimentation upstream of Nyumba ya Mungu reservoir in Pangani River Basin. *Nile Basin Water Science and Engineering* 3(2), pp. 25−38

Neumann, B., Vafeidis, A.T., Zimmermann, J. and Nicholls, R.J. (2015). Future coastal population growth and exposure to sea-level rise and coastal flooding− A global assessment. *PLoS One* 10 (3), pp. e0118571.

Nordez, S. (2014). Influence of estuarine flow rates on the artisanal shrimp catches in Maputo Bay. I*n The Maputo Bay Ecosystem*. WIOMSA, Zanzibar, pp. 287−288

Paerl, H.W. (1997). Coastal eutrophication and harmful algal blooms: Importance of atmospheric deposition and groundwater as "new" nitrogen and other nutrient sources. *Limnol. Oceanogr.* 42(5 part 2), pp. 1154−1165

Povinec, P.P., Burnett, W.C., Beck, A., Bokuniewicz, H., Charette, M., Gonneea, M.E., Ishitobi, T., Kontar, E., Kwong, L.L. and Marie, D.E.P. (2012). Isotopic, geophysical and biogeochemical investigation of submarine groundwater discharge: IAEA-UNESCO intercomparison exercise at Mauritius Island. *J. Environ. Radioact.* 104, pp. 24−45

Robertson, A.I. and Phillips, M.J. (1995). Mangroves as filters of shrimp pond effluent: predictions and biogeochemical research needs. *Hydrobiologia* 295(1−3), pp. 311−321

Scheren, P., Diop, S., Machiwa, J. and Ducrotoy, J.P. (2016). The Western Indian Ocean: a wealth of life-supporting ecosystem goods and services. In Estuaries: *A Lifeline of Ecosystem Services in the Western Indian Ocean*, pp. 1−23. Springer, Cham

Sheaves, M., Baker, R., Nagelkerken, I. and Connolly, R.M. (2015). True value of estuarine and coastal nurseries for fish: incorporating complexity and dynamics. *Estuaries Coasts* 38(2), pp. 401-414

Spalding, M., Kainuma, M. and Collins, L. (2010). *World Atlas of Mangroves*. Routledge. Washington DC, USA

Taniguchi, M., Burnett, W.C., Cable, J.E. and Turner, J.V. (2002). Investigation of submarine groundwater discharge. *Hydrol. Process.* 16(11), pp. 2115−2129

Taylor, M., Ravilious, C. and Green, E.P. (2003). Mangroves of East Africa. UNEP-WCMC Biodiversity Series 13. Cambridge

Van Camp, M., Mtoni, Y., Mjemah, I.C., Bakundukize, C. and Walraevens, K. (2014). Investigating seawater intrusion due to groundwater pumping with schematic model simulations: the example of the Dar es Salaam coastal aquifer in Tanzania. *J. Afr. Earth Sci*. 96, pp. 71–78

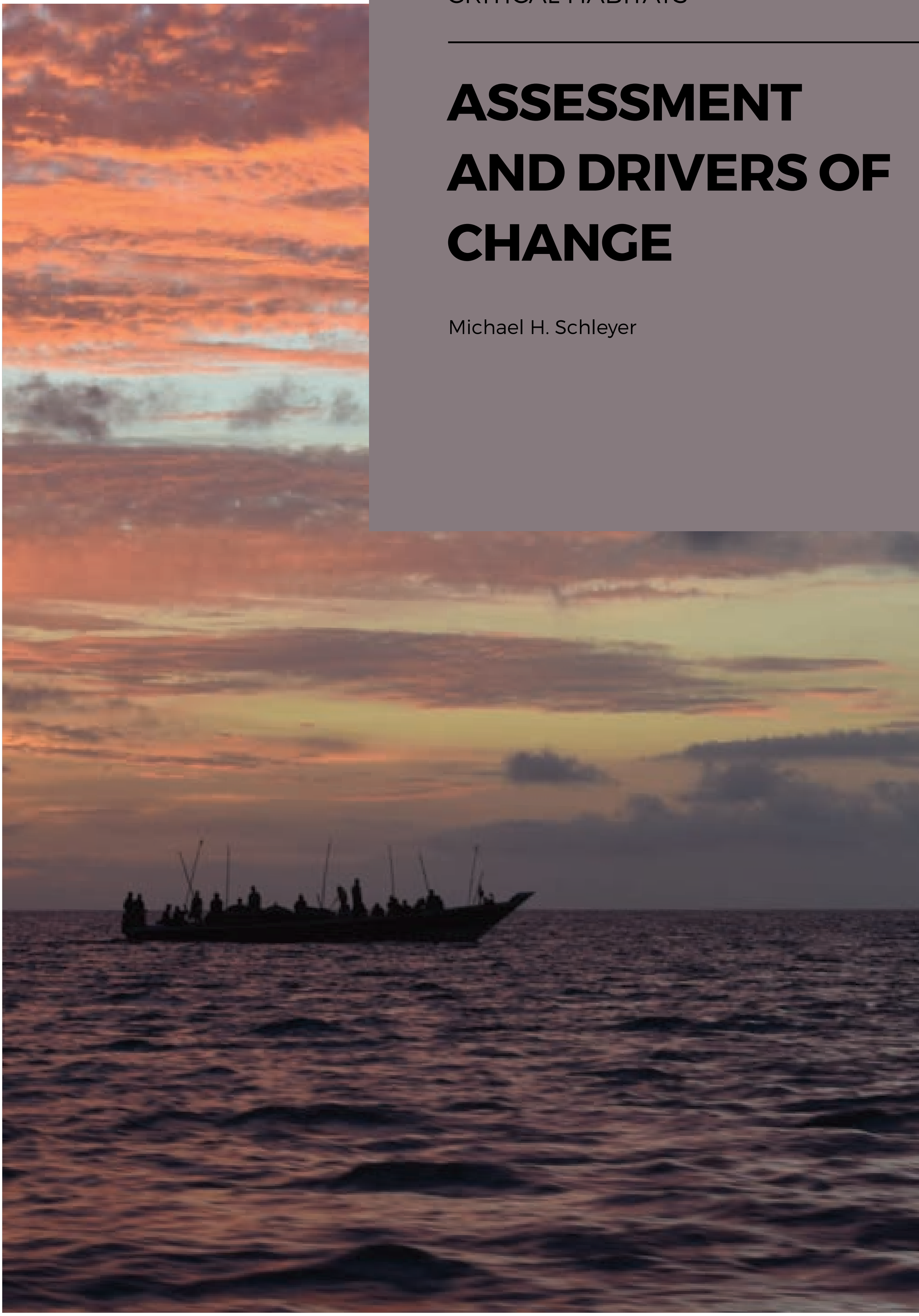
Van Katwijk, M.M., Van der Welle, M.E.W., Lucassen, E.C.H.E.T., Vonk, J.A., Christianen, M.J.A., Kiswara, W., al Hakim, I.I., Arifin, A., Bouma, T.J., Roelofs, J.G.M. and Lamers, L.P.M. (2011). Early warning indicators for river nutrient and sediment loads in tropical seagrass beds: a benchmark from a near-pristine archipelago in Indonesia. *Mar. Pollut. Bull.* 62(7), pp. 1512−1520

Wilkinson, C. (2004). *Status of coral reefs of the world: 2004*. Australian Institute of Marine Science. Townsville, Australia

CRITICAL HABITATS

ASSESSMENT AND DRIVERS OF CHANGE

Michael H. Schleyer



Artisanal fishers returning to port, Tanzania. © Jennifer O'Leary

CRITICAL HABITATS

The term Critical Habitat appears first to have been coined in the US Endangered Species Act (ESA), this being part of US enacting legislation for implementation of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES[1]). The term defines areas essential for the conservation of endangered or threatened species and, paradoxically, such areas need not be inhabited by the species in question when designated, but are needed for their recovery.

The US Fish and Wildlife Service take into consideration areas needed for individual and population growth of a species; its normal behaviour, shelter, and nutritional and physiological requirements; sites for breeding and rearing offspring; and habitats protected from disturbance or representative of its historical geographical and ecological distribution[2]. The focus is clearly species-orientated and overlooks other important attributes of habitats.

The International Finance Corporation of the World Bank thus devised a broader, more rigorous definition for Critical Habitats:

> *'Critical habitats are areas with high biodiversity value, including a) habitat of significant importance to Critically Endangered and/or Endangered species; b) habitat of significant importance to endemic and/or restricted-range species; c) habitat supporting globally significant concentrations of migratory species and/or congregatory species; d) highly threatened and/or unique ecosystems; and/or e) areas associated with key evolutionary processes.'*

These broader World Bank criteria have been employed in identifying and assessing the critical nature of habitats in the WIO. The focus has thus not been on species alone; threatened and unique ecosystems have been included.

It is important to note that biodiversity offsets are considered an option in the World Bank criteria, but only on a 'like-for-like' or 'better than' basis that will provide a net gain in habitat.

1. www.cites.org

2. https://www.fws.gov/endangered/what-we-do/critical-habitats-faq.html

CRITICAL HABITATS ASSESSMENT

Critical habitat assessment can be a difficult and intricate process. However, the Convention for Biodiversity (CBD) has developed an elegant process to identify Ecologically or Biologically Sensitive Areas (EBSAs; see Dunn et al., 2014), which helps promote sites for their protection. This is done within the framework of a marine spatial planning process.

In its development, the CBD basically adopted and adapted a Canadian process for environmental evaluation, employing the following criteria to evaluate marine habitats for their:

Uniqueness or rarity

Examples here would be the coelacanth *(Latimeria chalumnae)*, which appears limited in habitat to submarine canyons, or the dugong *(Dugong dugon)*, populations of which have been harvested (or caught as bycatch) to extinction in many WIO countries.

Special importance for life history stages of species

A good example here would be fish aggregation sites for breeding, such as Pinnacle Reef in southern Mozambique, which is annually visited for this purpose by the giant trevally, *Caranx ignobilis*.

Importance for threatened, endangered or declining species and/or habitats

Turtles are vulnerable when nesting and the nesting beaches of endangered species warrant protection.

Vulnerability, fragility, sensitivity or slow recovery

The most obvious case here would be that of coral reefs, which are globally under threat from human activities and climate change. In terms of species, an example would be the African penguin *(Spheniscus demersus)*, which is classified as endangered, and Bird Island in Algoa Bay provides its most important WIO breeding habitat.

Biological productivity

Probably the best example here would be the rich tuna fishing grounds in and around the EEZ of the Seychelles.

Biological diversity

Here coral reefs again provide a good example as they support some of the richest biological diversity in the world. Similarly, the islands of Mayotte and Europa have diverse habitats that are rich in biodiversity.

Naturalness

The relatively pristine habitats of the Bassas da India atoll and Europa Island provide good examples of truly natural habitats.

There has been some debate as to the efficacy of the EBSA process (Dunn et al., 2014). Clearly, the process is qualitative rather than quantitative and the assessments, unless undertaken by specialists in the relevant fields, may be subjective. Furthermore, it merely identifies priority areas in need of protection but cannot institute the process whereby this is accomplished.

The IUCN ecosystem status categories for ecosystems (Bland et al., 2017) have been applied to the habitats assessed in this document (Fig. 1). The categories employed are summarized below, details of which can be found in Bland et al. (2017):

Collapsed (CO)

A habitat has Collapsed when its defining biological or physical features are lost, and its characteristic natural fauna and flora are no longer maintained. This category has been used in the context of national or regional, not global, collapse. The IUCN (2016) makes allowance for this in what it terms 'sub-global' assessments that can be confined to political or ecological boundaries, such as a river catchment or ocean basin.

This is the most straightforward category with assessment based on the following criteria:

- **Reduction in distribution:** The extent to which a habitat has been reduced over the last 50 years, or since 1750, or is likely to be reduced in the next 50 years.
- **Environmental degradation:** The severity and extent to which a habitat has become environmentally degraded over the last 50 years, or since 1750, or is likely to become degraded in the next 50 years.
- **Disruption of biological processes:** The severity and extent to which the biological processes have been disrupted in a habitat over the last 50 years, or since 1750, or are likely to become degraded in the next 50 years.
- **Restricted distribution:** Habitats that are naturally restricted in distribution and are potentially threatened as they occur at very few localities or are very small in area.
- **Quantitative analysis:** The probability of collapse of a habitat based on modelling and risk assessment.

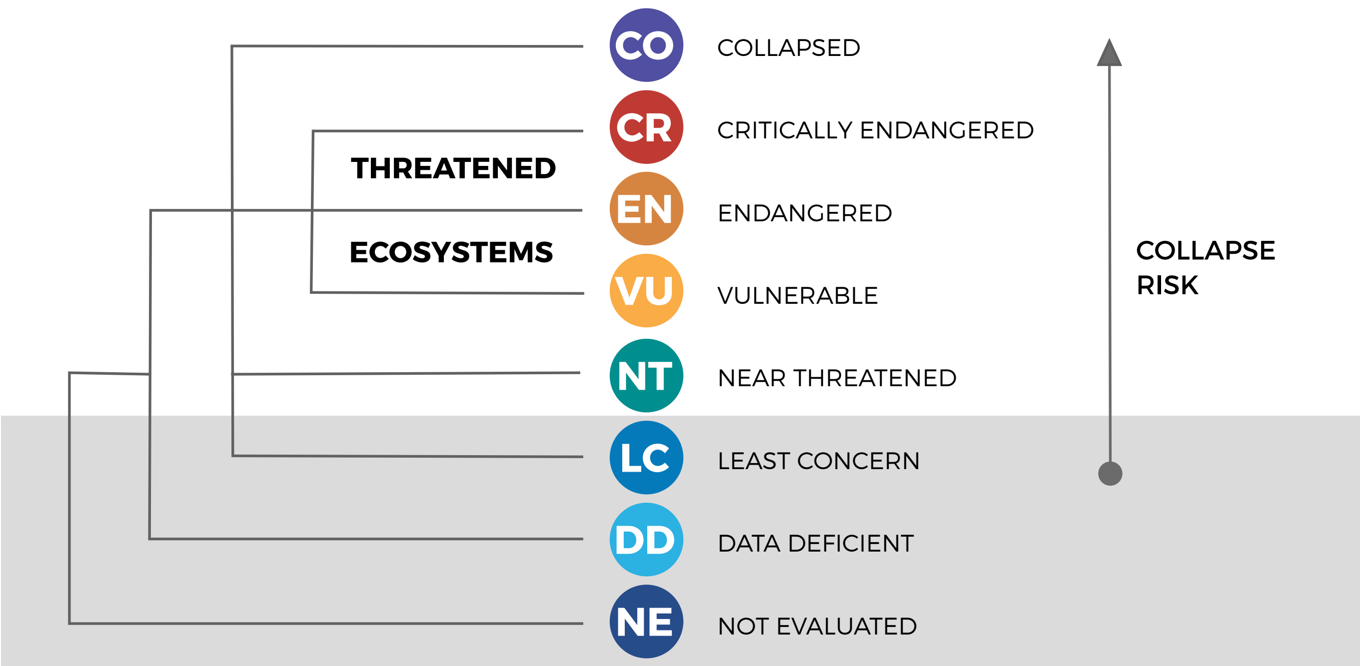


Figure 1. The IUCN ecosystem status categories (Bland et al., 2017). Shaded categories were not considered in this document. The category 'Collapsed (CO)' was used in the context of national or regional, not global, collapse.

Critically Endangered (CR)

A habitat is Critically Endangered when it has, for example, suffered a reduction in geographic distribution by 80 per cent over the past 50 years, or 90 per cent since 1750, or is likely to be reduced by 80 per cent in the next 50 years. The assessment of the level to which habitats are restricted in distribution is more subjective and is made by specialists. The probability of collapse of a Critically Endangered habitat is ≥50 per cent within 50 years.

Endangered (E)

A habitat is Endangered when it has, for example, suffered a reduction in geographic distribution by 50 per cent over the past 50 years, or 70 per cent since 1750, or is likely to be reduced by 50 per cent in the next 50 years. The assessment of the level to which habitats are restricted in distribution is more subjective and is made by specialists. The probability of collapse of an Endangered habitat is ≥20 per cent within 50 years.

Vulnerable (VU)

A habitat is Vulnerable when it has, for example, suffered a reduction in geographic distribution by 30 per cent over the past 50 years, or 50 per cent since 1750, or is likely to be reduced by 30 per cent in the next 50 years. The assessment of the level to which habitats are restricted in distribution is more subjective and is made by specialists. The probability of collapse of a Vulnerable habitat is ≥10 per cent within 100 years.

Near Threatened (NT)

A habitat is Near Threatened when, upon assessment, it does not qualify for a threatened category but is close to qualifying, or is likely to qualify, for a threatened category in the future.

Levels of protection of the habitats are provided using the South African National Biodiversity Act (NBA) codes (Sink et al., 2012):

Well protected

100 per cent or more of the SDG 14 target (10 per cent) for a habitat is protected within an MPA (or similar structure), with sufficient 'no-take'.

Moderately protected

50 to <100 per cent of the SDG target is in an MPA (or similar structure).

Poorly protected

5 to <50 per cent of the SDG target is in an MPA (or similar structure).

Hardly protected

1 to <5 per cent of the SDG target is in an MPA (or similar structure).

Zero protection

No formal protection.

The scale is based on the proportion of each habitat or ecosystem under protection but the actual degree of protection will vary – in 'paper parks' it might technically be zero. Where this is known, the information has been included in the assessment. However, where unknown and the assessments are based only on gazetted proclamations, they may be exaggerated.

CRITICAL HABITATS OF THE WIO

Critical habitats in the WIO comprise a wide suite of tropical, subtropical and temperate habitats, both coastal and marine; these are dealt with in separate chapters. In brief, they range from:

Coastal forests, lakes and lagoons

Expansive coastal forests are found along the shores of the WIO mainland states, Madagascar and some of the islands. While they are of value and are exploited, their biodiversity has not been fully explored. For example, a new endemic coastal forest tree species, *Incuria dunensis*, was recorded as recently as 2004 near Moma, central Mozambique (Albano, 2004). What are possibly the highest vegetated dunes in the world are also found along the north-east coast of South Africa, extending into southern Mozambique.

Coastal lakes and lagoons are found in this area up to Vilanculos and along the east coast of Madagascar, ranging in salinity from fresh to saline, and are fished by local communities.

Mangroves and estuaries with associated salt marshes and seagrass beds

While most estuaries along the South African coastline are unwooded, these become heavily forested with mangroves to the north, with WIO mangroves accounting for 5 per cent of global mangrove coverage. A narrow band of salt marsh plants is found at the landward edge of many mangroves but becomes more extensive in temperate South African estuaries and lagoons.

Expansive seagrass beds are found on shallow sedimentary banks in the tropics and are often associated with estuarine systems. These may themselves be small in comparison to the habitats they link. All are valuable for their resources and mangroves also play a significant role in shoreline protection.

Beaches and the nearshore habitat

Beaches are not considered a productive habitat yet are used both for fishing and recreational activities, and are vital for turtle nesting. The nearshore habitat is commercially important as it supports artisanal commercial fisheries for prawns, squid, finfish and sharks.

Rocky shores, reefs and coral reefs

- *Rocky shores and reefs:* Rocky shores and reefs are far more accessible to fishers than the nearshore environment and both are gleaned for a diversity of algal and invertebrate resources as well as finfish.

- *Coral reefs:* These are amongst the most biodiverse and productive of habitats, terrestrial and marine. They are valuable for their resources and shoreline protection, yet are globally threatened by human activities and climate change.

Offshore habitats

These are the least known and understood of habitats, yet are the most expansive. WIO countries have, within their EEZs, offshore banks, submarine canyons, sunken atolls, seamounts and deep trenches. Few have been explored but offer diverse resources, ranging from rich fish stocks to gas and mineral deposits.

The livelihood of coastal communities depends on these habitats and, to a greater or lesser degree all are used – and threatened – by coastal populations. Individual assessments of these critical habitats have been made at the national as well as the regional level and the evaluated information falls into three categories:

Habitats where single species or phyla are considered

For example, canyons where the coelacanth is found, pinnacles where fish breeding aggregations occur (eg the aforementioned giant trevally, *Caranx ignobilis* – accompanied by the predatory bull shark, *Carcharhinus leucas* – on Pinnacle Reef in southern Mozambique), or remote localities are used (nearly) exclusively by seabirds (eg Île du Lys in the Glorioso Archipelago and the aforementioned Bird Island in Algoa Bay).

Expansive habitats

Such as the mangroves and seagrass beds in Mozambique that are connected through estuaries which are small in comparison. These habitats may be many square kilometres in extent, comprising nearly mono-specific stands of the dominant plant species, but provide shelter, nursery areas and feeding grounds for diverse animal life.

Island habitats characterized by diverse communities that are in close proximity and intimately connected

For example, oceanic islands where coral reefs, seagrasses and mangroves are found, and the diversity of habitats are used by sea turtles for nesting, seabirds and a diversity of marine life. These incorporate all the WIO marine habitats in a microcosm, frequently include endemic species, and are vulnerable to disruption by introduced species such as rats or invasive marine life. They range from relatively pristine (for instance, the Aldabra Atoll and Europa Island) to heavily impacted (the biodiversity hotspot of Mayotte).

KEY DRIVERS OF CHANGE

Threats to the environment can be broadly categorized as those which are natural, for example episodic events (cyclones, tsunamis, floods) and climate change, and anthropogenic or human in cause, for example exploitation (direct and indirect), habitat destruction (land 'reclamation', urbanization, dredging, mining and oil/gas extraction), pollution (point and diffuse sources) and climate change (including ocean acidification and sea level rise).

The World Wildlife Foundation (WWF) broadly lists the following as threats to the environment[3]:

Effects of climate change

This is the first environmental threat that the average person thinks of in the present age because of the level to which it has been publicized. However, what most people do not immediately appreciate is that the only thing constant about climate is change, and some of the present changes in global climate are, in fact, natural. However, a problem arises when human activities increase the rate of climate change and this is occurring because of the unprecedented rate at which humans are generating greenhouse gases, principally atmospheric carbon dioxide (CO_2). This is causing the earth's climate to warm, resulting in changes in climate that include more violent and frequent storms, melting of the polar ice caps and expansion of the seas with a rise in sea level, and acidification of the seas as

[3] https://www.worldwildlife.org/threats

the CO_2 dissolves in seawater forming weak carbonic acid. Life within habitats has to adjust to these changes but, in many cases, cannot do so fast enough.

The exploitation of fossil fuels

Burning fossil fuels generates the greatest source of CO_2 (but it also comes from digestion in ruminants and the decomposition of waste material in eg municipal dumps). Increasing prospecting for and exploitation of fossil fuels is causing loss of habitat and poses major pollution risks.

Deforestation

Growing plants 'fix', or sequestrate, CO_2 and trees play the biggest role in this regard. Greenhouse gas production could thus be alleviated by afforestation. However, deforestation of natural forests is occurring at an increasing rate for timber and agriculture, especially in the tropics where decomposition of the forest floor after clear-felling adds further CO_2 to the atmosphere. The rich biodiversity in these forests is being lost as valuable forest habitats diminish.

Infrastructural development

Human populations are increasing, with most demographic growth occurring in cities and urban areas, particularly along the coast. Human needs for infrastructural development require space and resources, resulting in loss of habitat. This includes land and natural environment lost not only to cities but also developments such as dams, harbours, roads and airports.

Pollution

Urbanization, industrialization and inadequate waste disposal associated with demographic growth in the WIO are resulting in pollution on land, the introduction of solid waste to waterways, dissolved toxins to the sea, and atmospheric pollution. Waste material from other countries is also being introduced to the WIO by currents. Persistent particulate material and micro-plastics are a growing global concern.

Soil erosion and degradation

Poor land use and agricultural practices, particularly overgrazing, are at the root of this environmental concern. Soil, nutrients and agrochemicals are being washed into waterways and lost to the ocean, resulting in turbidity, sedimentation and nutrient enrichment.

Water scarcity

Of the ever-increasing demand for resources, the scarcity and quality of potable water is probably the greatest concern. Sources are being over-extracted or becoming contaminated, and water is the most limited commodity in many countries. This adversely affects particularly the aquatic habitats from which water is extracted but also the habitats into which this life-giving commodity is introduced.

Overfishing, illegal fishing and bycatch

The world's growing demand for food has led to overfishing of many living marine resources. Much fishing is unregulated and there is substantial wastage of what is known as bycatch, unwanted fish caught incidentally while targeting a desirable species. 'Ghost-fishing', the unnecessary and untargeted death of aquatic life in lost or disposed fishing gear is further reducing fish stocks.

The illegal trade in wildlife

Wild animals and plants are hunted for bush meat, traditional medicine or the curio and pet trade, and are being captured and harvested to the point that they many have become threatened or endangered. Despite the practice being illegal, the rewards are high, and poachers continue their activities unabated. Many species are thus endangered, examples within WIO critical habitats being the dugong *(Dugong dugon)*, which has been radically reduced in number for its meat, and the coelacanth *(Latimeria chalumnae)*, targeted in the past for scientific specimens and more recently as bycatch of the gillnet fishery for sharks.

References

Albano, G. (2004). *Coastal forests of Mozambique: Socio-economic aspects (A review).* Unpublished report, final version, 22 pp

Bland. L.M., Keith, D.A., Miller, R.M., Murray, N.J. and Rodríguez, J.P. (eds.) (2017). Guidelines for the application of IUCN Red List of Ecosystems Categories and Criteria, Version 1.1. IUCN, 99 pp. Gland, Switzerland

Dunn, D.C., Ardron, J., Bax, N., Bernal, P., Cleary, J., Cresswell, I., Donnelly, B., Dunstan, P., Gjerde, K., Johnson, D., Kaschner, K., Lascelles, B., Rice, J., von Nordheim, H., Wood, L. and Halpin, P.N. (2014). The Convention on Biological Diversity's Ecologically or Biologically Significant Areas: Origins, development, and current status. *Mar. Policy* 49, pp. 137–145. http://dx.doi.org/10.1016/j.marpol.2013.12.002i

IUCN (2016). An Introduction to the IUCN Red List of Ecosystems: The Categories and Criteria for Assessing Risks to Ecosystems. IUCN, 14 pp. Gland, Switzerland

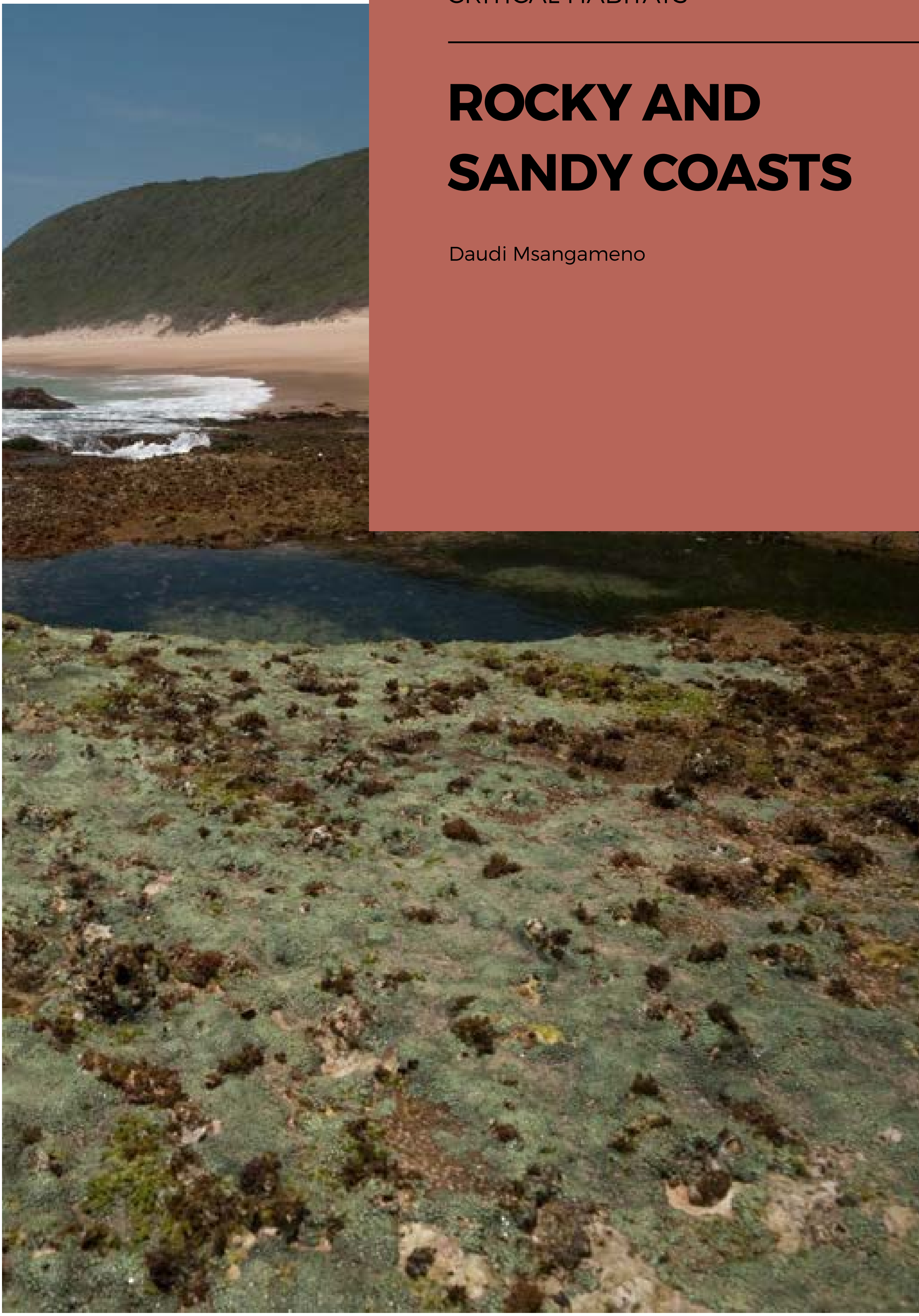
Sink, K., Holness, S., Harris, L., Majiedt, P., Atkinson, L., Robinson, T., Kirkman, S., Hutchings, L., Leslie, R., Lamberth, S., Kerwath, S., von der Heyden, S., Lombard, A., Attwood, C., Branch, G., Fairweather, T., Taljaard, S., Weerts, S., Cowley, P., Awad, A., Halpern, B., Grantham, H. and Wolf, T. (2012). *National Biodiversity Assessment 2011: Technical Report*. Volume 4: Marine and Coastal Component. South African National Biodiversity Institute, Pretoria. 325 pp

PART 3: CRITICAL HABITATS IN THE WESTERN INDIAN OCEAN

CRITICAL HABITATS

ROCKY AND SANDY COASTS

Daudi Msangameno



Rocky shore and coastal dunes at Cape Inhaca, Southern Mozambique. © José Paula

BACKGROUND

Regional coverage

Rocky shores and sandy beaches are some of the most common features along coastlines of the Western Indian Ocean (WIO) countries, though information on the spatial extent and total areal coverage of such habitats in the region is scant. However, estimates for the whole Indian Ocean coast are put at approximately 3000 km^2, excluding the western Australian coast (Wafar, 2011). This has been computed as a product of the total coastline lengths of all maritime states of the Indian Ocean (66 000 km) and an average intertidal width of 50 m (Qasim, 1998).

Substrate formation and characterization

Nearshore habitats are among the most bio-physically dynamic marine environments, being also characterized by their high vulnerability to anthropogenic and natural drivers. Worldwide, such habitats have been highly altered to meet the demands of growing populations, as well as for various subsistence and economic ends. The nearshore environment is generally defined as the area encompassing the transition from subtidal marine habitats to associated upland systems. It includes the subtidal zone, the intertidal, and the upland area of the shore. Depending on the type of substrate such habitats may either be rocky, sandy or muddy in nature, thus intertidal rocky shores, sandy beaches, muddy shores, mangroves, seagrass meadows and coral reefs tend to fall under this broad category (Maina, 2015). While coral reefs, seagrass meadows and mangroves are addressed in separate parts of the outlook, this chapter will focus on the subtidal and intertidal rocky and sandy shore habitats.

The major difference between rocky and sandy shore habitats is the nature of the substrate, leading to fundamental differences in the biophysical characteristics of these equally important marine biotopes. Rocky shores are mostly formed as a result of denudation of the over-burden and bedrock caused by a combination of sea level rise and wave action in areas of limited sedimentation (Ruwa, 1996). Rocky shore habitats can also be extended by the presence of artificial coastal structures such as seawalls, groynes, dykes and jetties (Moschella et al., 2005). The physical properties of a given rocky shore are chiefly determined by the mode of its geological formation (Yorath and Nasmith, 2001). Pleistocene limestone are the main geological formations in the WIO region, dominating the intertidal zone and the subtidal in Madagascar, northern Mozambique, Tanzania and Kenya, while aeolianite is common along the north-eastern coast of South Africa and southern coasts of Mozambique (Kalk, 1995; Ramsay, 1996). Some rocky shores in the WIO oceanic islands are granitic and basaltic in origin, with notable examples being the granitic reefs of Mahe, Seychelles (eg Hill and Currie, 2007). Basaltic reefs are common in Mauritius and the Comoros. Some islands in the region comprise atolls, formed from coral and have limestone cliffs of interglacial origin. Aldabra Atoll in Seychelles is one such island (Ruwa, 1996). Table 1 summarizes some of the major rocky shore geological formations in the WIO region.

However, as for most tropical marine regions in the WIO, most rocky shores are biogenic, being formed from raised fossil corals. Such reefs are therefore characterized by the presence of pits, cracks and crevices, creating extremely heterogeneous environments with numerous rock pools, overhangs, gullies and caves.

Physical attributes such as hardness and porosity vary significantly among different rock formations, with limestone and basaltic rocks being more porous than granitic ones. Such variation in physical properties invariably determines the nature of various biological processes such as larval settlement and recruitment, and thus the nature of the climax benthic biological communities found in such habitats (Raimondi, 1988).

While rocky shores often occur in areas of high wave energy, sandy shores are characteristic of areas of high depositional activity, resulting in wave-deposited sediment accumulations on or close to the shoreline. For such habitats to form there must be a basement (hard stratum) which is typically the bedrock, waves to shape them, sediment, and in most cases also rivers and/or tides to bring the sediments on the foreshore (Short, 2012). In most cases the accumulation of beach sediment extends from where waves begin to influence the seabed, extending across the nearshore zone through the intertidal area to the upper limit of wave swash.

A major source of beaches in the WIO region is carbonate-sourced sediments, derived from sediment resulting from either weathered dead shells or coralline algae (eg *Halimeda*) that has been transported to the shore from shallow marine environments (shelf sediments) or eroded from nearby shores and coral reefs and transported by long-shore currents (Fennessy and Green, 2015). However, in coastal areas drained by major rivers that discharge large amounts of sediments, the beach and nearshore would typically be dominated by sediment of terrigenous origin. Such sediments are geo-chemically

Table 1: Examples of rocky reef formations at selected locations in the WIO.

LOCATION	TYPE OF FORMATION
Dar es Salaam (Tanzania)	Limestone (Hartnoll, 1976)
Inhaca (Mozambique)	Sandstone (Kalk, 1995)
Maputaland (South Africa)	Sandstone (Ramsay, 1996)
Durban (South Africa)	Sandstone (Martin & Flemming, 1988)
Seychelles	Coral rock, granite (Ngusaru, 1997)
Mauritius	Basalt, limestone (Hartnoll, 1976)
Kenya	Limestone (Ngusaru, 1997)
Tulear (Madagascar)	Limestone (Hartnoll, 1976)
Comoros	Basalt (Ngusaru, 1997)
Northern Mombasa (Kenya)	Limestone (Ngusaru, 1997)

Source: UNEP/Nairobi Convention Secretariat (2009).

characterized by the presence of quartz and feldspar minerals, and thus siliciclastic in nature. However, where these riverine clastic sediment inputs are small, biogenic (bioclastic) sedimentation, produced by erosion of the skeletal carbonate remains of marine organisms, can dominate. The distribution patterns of the various sediment textures (mud, sand, gravel, etc) vary according to proximity to river mouths, depth, wave action and currents (Nichols, 2009), with the fine fractions (mud and fine sand) being the most easily dispersed. The type of sediment can thus change substantially along and across a shore depending on the relative contributions from carbonate and/or clastic sediment production (Fennessy and Green, 2015).

The amount and patterns and distribution of shelf sediments reaching the intertidal and nearshore is mainly determined by the bathymetric characteristics and hydrographic dynamics of the continental shelf. For instance, the location of the continental shelf break, which determines the width of the shelf, is a function of interaction between sedimentation processes, sea level changes and tectonics (sea floor emergence or submergence). In addition, reefs and submerged shorelines, for example, form barriers, allowing sediment to accumulate between them and the shore, partly helping to retain the sediments in the nearshore (eg Puga-Bernabéu et al., 2011). Since coral reef coverage in the WIO is minimal compared to the total estimated shelf area, it is obvious that the vast majority of the seabed in the WIO is comprized of unconsolidated sediments (Fennessy and Green, 2015), creating a reliable source for sediments essential for maintaining sandy shores.

Patterns of biological distributions

A common feature of most rocky shores is their ability to support diverse assemblages of benthic organisms which exhibit peculiar distribution patterns. Such patterns (or zones) are in response to a number of biophysical factors operating at different spatial and temporal scales (Menge and Sutherland, 1987). While variations at a bio-geographical level may be explained by large-scale factors notably ocean current systems and broad-scale seawater temperature regimes (Bustamante and Branch, 1996), local variations in species composition are invariably a result of factors operating at smaller spatial scales. These include physical attributes such as extent of wave exposure, insolation, temperature, aspect and substratum type. The combined effect of such environmental attributes is the creation of unique zones of species distribution on most intertidal rocky shores (Stephenson and Stephenson, 1972). According to Lewis (1964) the following broad zones can therefore be distinguished on a typical intertidal rocky shore: littoral fringe, eulittoral zone and sublittoral zone.

While patterns of species distribution on rocky shores are primarily determined by physico-chemical gradients along the shore height axis, the role of biological interactions is also important. For instance, processes such as competition, facilitation and predation are crucial in shaping the final assemblages of species in given biological communities (Steffani, 2000; Coleman et al., 2006). One of the most notable examples of the influence of biological interactions on biotic patterns on rocky shores is the role played by processes such as grazing and competition in setting species distributional limits in the lower parts of the shores (Boaventura et al., 2002). Several ecological models have included biotic interactions as important determinants of the structure of biological communities.

One of the classic examples of such models is by Menge and Sutherland (1976), predicting the comparative importance of predation in determining community composition in relatively benign environments, with competition being progressively more important as the environment becomes harsher. The model was however, modified to include the effects of recruitment variations (Menge and Sutherland, 1987), mainly downplaying the importance of predation and competition in areas with low recruitment.

Predicting the influence of gradients of physical and biological factors in determining patterns of biological distribution on rocky shores may be made more complex by the presence of rock/tide pools. These important features of most rocky shores significantly interrupt the otherwise simplified zonation by enhancing species abundance and richness (Firth et al., 2013). This, invariably extends the distributional upper limits of many species, making the biological zonation less pronounced (Steffani, 2000).

Unlike rocky shores where the substrate is mostly consolidated and stable, organisms on sandy shores need to be highly adapted to living on or within substrate that is unstable and constantly disturbed by swash, tides and wind (Janssen and Mulder, 2005). However, in contrast with rocky shores, atmospheric exposure and desiccation are not a major concern for sandy shore benthos, as they can retreat into the substratum or below the water table. Though tides disturb sandy shore benthos, most of the organisms present depend on the tides for feeding, as flooding tides bring in suspended food particles on which many filter-feeders depend. To cope with tidal movements many species of meiofauna use vertical tidal migrations through the sand (McLachlan, 1977; Steyaert et al., 2001), while other motile species move up and down the beach with the tides. The movement of the fauna along the shore axis is in response to various stimuli, which are both directional (such as light, slope of the beach and water currents) and non-directional (such as disturbance of the sand, changes in temperature and hydrostatic pressure). Dominant functional groups on many sandy shores are filter feeders and scavengers. As in other marine benthic environments biotic distributions and abundance of sediment infauna is mostly controlled by complex interactions between the physicochemical and biological properties of the sediments (Knox, 2001). These include grain size, water content, flushing rate of water through the sediment, oxidation-reduction levels, dissolved oxygen, temperature, light, organic content, food availability and feeding activity, reproductive effects on dispersal and settlement, behaviour that induces movement and aggregation, intraspecific competition, interspecific competition and competitive exclusion, and predation effects (see Rodriguez et al., 2001; Moreno et al., 2006).

Rock pools on the intertidal at Inhaca Island, Mozambique, © Mariana Cravo

Associated key species

The complexity and habitat heterogeneity found within rocky and sandy shores attract considerable diversity of benthic fauna, with almost all invertebrate phyla represented (Deepananda and Macusi, 2012). These habitats host a wide range of species either living permanently, spending part of their life cycles (Gibson and Yoshiyama, 1999) or simply using them as feeding grounds or refugia. Among important taxa on rocky shores are the macroalgae. These are habitat-forming organisms with which other organisms associate (Casu et al., 2006). They also contribute significantly to the pelagic energy-biomass budgets by acting as a source of pelagic carbon. There are also a wide range of benthic invertebrate assemblages associated with rocky shores in the WIO region. These include molluscs (bivalves, gastropods and cephalopods) and crustaceans (eg Postaire et al., 2014) and other invertebrate taxa such as cnidaria, echinoderms, bryozoans and marine worms. Besides, several commercially important species of finfish are associated with rocky reefs (Durville and Chabanet, 2009; Maina, 2015; Sindorf et al., 2015). They include a number of species belonging to families such as Moringidae, Muraenidae, Pseudochromidae, Kuhliidae, Lutjanidae, Chaetodontidae, Pomacentridae, Labridae, Mugilidae, Blenniidae, Gobiidae, Tripterygiidae, Acanthuridae, Bothidae, Sparidae and Carangidae (Durville and Chabanet, 2009; Maina, 2015). The majority of these fish use rocky shores as nurseries and temporary refugia (eg during stranding in rock pools).

Sandy beaches are well known for their recreational value. However, these habitats host considerable marine biodiversity, supporting diverse species of both invertebrates and vertebrates. Invertebrate infauna and important taxa associated with sandy shores in the WIO region

Table 2: Nesting status for species of sea turtles in the WIO countries

COUNTRY	*Chelonia mydas*	*Caretta caretta*	*Dermochelys coriacea*	*Eretmochelys imbricata*	*Lepidochelys olivacea*
Kenya	Nesting	Sighting	Sighting	Nesting	Rare nesting
Madagascar	Possible nesting	Possible nesting	Sighting	Possible nesting	Possible nesting
Mauritius	Possible nesting outer islands	–	–	Possible nesting outer island	–
Mozambique	Nesting	Nesting	Nesting	–	–
Reunion/Eparses Island/Mayotte	Nesting	Very rare sighting	Very rare sighting	Nesting	Very rare sighting
South Africa	In water sighting	Nesting	Nesting	In water sighting	Rare nesting
Tanzania	Nesting	In water sighting	In water sighting	Nesting	Rare nesting

Modified from Bourjea et al., 2008.

are dominated by interstitial meiofauna (eg Barnes et al., 2011). These form an important trophic compartment of the benthic food webs of sandy shores and adjacent habitats, by acting as a source of food for both benthic and pelagic consumers. Invertebrate macrofauna are also common on sandy shores, with various species of crustacean (eg crabs), molluscs (eg bivalves and gastropods) and worms such as annelids being abundant. One of the most common and visible taxa are ghost crabs, which are normally used as ecological indicators for ecosystem health, with respect to anthropogenic impacts on beaches. Apart from being rich in invertebrate fauna, sandy shores of the WIO region are ecologically important for certain vertebrate taxa. Sea turtles for example are reliant on sandy beaches for their life histories. Along several coastlines in the region sandy shores are important nesting areas for several species of marine turtle, to which periodically adult female turtles return to lay eggs.

Five species of sea turtles have been documented in the WIO (Marquez, 1990). Of these, the Green turtle (*Chelonia mydas*) and Hawksbill (*Eretmochelys imbricata*) are most widely distributed and most numerous. Loggerheads (*Caretta caretta*) and Leatherbacks (*Dermochelys coriacea*) used to be abundant along the South African coast, but less common in the rest of the region. Relatively little has been documented about the Olive Ridley (*Lepidochelys olivacea*) (Bourjea et al., 2008). All these species are protected under CITES. Besides, Hawksbill and Green turtle are listed as Critically Endangered and Endangered on the IUCN Red List, respectively. The rest are listed as Vulnerable[1]. Table 2 shows the nesting species of sea turtles as recorded per each WIO country.

1. www.iucnredlist.org.

ECOLOGICAL AND SOCIO-ECONOMIC IMPORTANCE

Intertidal and nearshore habitats offer considerable benefits, in terms of their ecological and socio-economic values. Both in isolation and as part of the wider seascape, such habitats contribute significantly to marine productivity, ecological integrity, as well as local livelihoods and economies.

Ecological importance

Being highly complex and topographically heterogeneous, rocky shores and sandy beaches offer a multitude of microhabitats and niches for diverse groups of benthic organisms (Kostylev et al., 2005). Although stable over ecological temporal scales, topographic heterogeneity is crucial in influencing spatial structure of environmental variables and in turn, biological processes. For instance, by creating desiccation stress gradients, topographic heterogeneity on rocky shores provides refuges against desiccation as well as competition and predation (Sebens, 1991). Besides, the increased substrate heterogeneity conferred by topographic features such as crevices, cracks and rock boulders protect many mobile animals against thermal stress by providing shade (Williams and Morritt, 1995). Although sandy beaches have more unstable substrate compared to rocky shores, organisms on such habitats are highly adapted to withstanding the impact of physical perturbation. Consequently, sandy shores are home to a variety of burrowing macrofauna such crabs and bivalves, and interstitial invertebrate infauna, with the most important taxa being the meiofauna. These represent an important part of the food web base for benthic and pelagic systems (Barnes et al., 2011).

Rocky and sandy shore habitats have high ecological connectivity with other intertidal and nearshore habitats, contributing to the productivity of the latter. For instance, one of the common features of rocky shores is the presence of micro and macroalgae. These abundant producers contribute significantly to marine productivity, supporting the wider marine food web (Branch et al., 2008). For instance, macroalgae are highly productive, and act as a major source of organic matter in the marine environment (Worm and Lotze, 2006). Together with seagrasses, macroalgae are estimated to account for up to 40 per cent of primary productivity in the coastal zone (Charpy-Roubaud and Sournia, 1990) and contribute significantly to the global marine plant biomass. Besides, such macrophytes fulfil crucial ecological functions in the marine environment, including carbon storage and nutrient cycling (Worm et al., 2000). The interaction between rocky shores and sandy beaches with other marine ecosystems also ensures a constant interchange of biomass and energy within the marine environment (Menge et al., 1997). For instance, the reproductive histories of most invertebrates associated with such habitats involve the production of large quantities of eggs and pelagic larvae. These create an important source of food for juvenile fish and other marine animals, thus enhancing overall fisheries production.

Rocky reefs and sandy beaches are also important areas for spawning, nursing, foraging and nesting for various marine species. For instance, the presence of tide pools on most intertidal rocky shores offers an opportunity for several marine species to withstand the extreme environmental conditions observed in intertidal habitats during low tide (Sindorf et al., 2015). While tide pools can save some stranded reef fish and other visiting pelagic fauna during low tide, such pools may also act as temporary residences for several species, using the habitat during specific seasons or life history stages (Gibson and Yoshiyama, 1999). These include larval or juvenile fish recruits which leave the pools once they reach a certain body size or maturity stage. This makes intertidal rocky shores one of the important nurseries for ecologically and commercially important deep-water species in several locations (Cunha et al., 2008), capable of replenishing surrounding reef populations and nearshore waters (Mahon and Mahon, 1994). A study by Sindorf et al. (2015) on one of the intertidal rocky shores of the WIO demonstrated the importance of such habitats as nursery areas for reef-associated and deeper water fish populations. For instance, in the study, it was indicated that over half of the fish observed in the tide pools were juveniles, confirming that such habitats are used as nurseries. In the same study several other species were found to be residents of the area, with ten such species being found in no other habitat in the surrounding area (Sindorf et al., 2015). Several other studies in the WIO have depicted similar patterns (see Durville et al., 2003; Durville and Chabanet, 2009).

In most areas in the WIO, rocky shores and sandy beaches serve as important feeding and foraging grounds for both terrestrial and marine animals. During low tide for instance, flocks of foraging seabirds are a common scene on most intertidal and nearshore areas while during high tide marine animals such as fish also feed on benthic invertebrates and plants (Worm et al., 2000).

Zoanthid soft corals at Inhaca rocky shore, Mozambique.
© José Paula

Socio-economic importance

Apart from their ecological importance, nearshore habitats such as rocky shores and sandy beaches contribute considerably to coastal livelihoods and economies. Sandy beaches are a major attraction on which coastal tourism is based. Several places across the WIO region are famous for their white sandy beaches and crystal-clear waters, attracting thousands of tourists every, in turn supporting local and national economies in the region. Rocky reefs form natural sea defences along several coastlines, in so doing protecting valuable investments such as residential and commercial properties and associated infrastructure. By contributing to nearshore and pelagic productivity (Raffaelli and Hawkins, 1996) and acting as refugia and nursery grounds for some reef and deep-water fish (Durville and Chabanet, 2009; Sindorf et al., 2015), nearshore habitats play a significant role in supporting artisanal and commercial fisheries in the inshore waters, thus contributing to dietary needs, food security and incomes for local communities and beyond.

Subsistence gleaning for invertebrate resources is also widespread along the coastlines in the region (Kyle et al., 1997). Various types of whelks, mussels, oysters, cockles, abalones, crabs, octopuses, barnacles, sponges and echinoderms are widely collected for food and income generation. Fishing activities are also common in many rock pools and shallow intertidal lagoons. Although practiced at limited scales, farming of several invertebrate species is also common in certain parts of the region. These include stalked barnacles, abalones, sponges and certain species of oysters (eg Troelli et al., 2006). Besides traditional subsistence gleaning activities on the intertidal and nearshore habitats, such habitats and associated species support emerging commercial activities in the region. These include marine based ornamental (curio) trade, as well as seaweed, sponge, finfish cage and half pearl farming (Gössling et al., 2004; Branch et al., 2008; Gibbons and Remaneva, 2011).

Shelfish collectors at Ruvuma delta near Mtwara, Tanzania.
© José Paula

THREATS TO ROCKY SHORES, SANDY BEACHES AND NEARSHORE HABITATS

Several phenomena and activities threaten the nearshore habitats such as rocky shores and sandy beaches in the WIO region, affecting their ecological productivity, integrity and by extension, livelihoods and economies. Although many of these habitats have the capacity to adapt to high levels of natural environmental stress, such ability is threatened by various human-related activities.

Resource over-exploitation and physical disturbances

Most intertidal and nearshore habitats provide highly accessible platforms from which natural resources can be harvested at low cost. During low tide, such habitats are typically visited by numerous people, gleaning for shellfish and other organisms, in places, placing enormous pressure on the resources, though detailed studies are few. Overharvesting may have profound impacts on the functioning and integrity of the many intertidal ecosystems, not only in terms of the direct effects on populations of the targeted species but also in terms of habitat destruction and recruitment failure. Modifications to benthic community composition through species overexploitation have been reported in the region. For instance, Siegfried (1994) reports on transformation of a filter feeder-dominated habitat into a community dominated by coralline algae, due to over-exploitation of intertidal invertebrates by subsistence collectors. The impact of over-exploitation of invertebrate resources becomes more pronounced if the targeted organisms are keystone species (Little and Kitching, 1996) as such organisms play important roles in supporting other species within the ecosystems (Lindberg et al., 1998). Apart from excessive harvesting of resources, human visitation on the intertidal zone can cause significant physical disturbance on the benthic biological communities, as human trampling may lead to changes in species composition (Casu et al., 2006). This can be either due to the direct physical impact, leading to dislodgement, mortality and general structural deformation of the organisms or indirectly, by causing loss in physiological efficiency, such as reproductive potential or competitive ability against other species (Denis, 2003).

Pollution

Being mostly intertidal or shallow-water habitats, rocky and sandy shores are affected by both sea-borne and land-based activities. Pollution has serious impact on biological communities on such shores. For instance,

Table 3: Major types of marine pollution and their impacts.

TYPE OF POLLUTION	SOURCES	IMPACTS
Eutrophication	– Natural sources eg animal droppings (Bosman and Hockey, 1988) – Sewage outfalls and agricultural run-off (Clark et al., 1997)	– Transformation of stable benthic communities, eg replacement of perennial macroalgae by ephemeral algae, diatoms and cyanobacteria (Schramm, 1996)
Siltation	– River discharge, shore erosion, sediment re-suspension, and atmospheric transport (Airoldi, 2003) – Industrial and domestic discharges (Kim et al., 1998) – Mining, construction and dredging (MacDonald et al., 1997) – Aquaculture (Holmer et al., 2001)	– Reduced species abundance (Saiz-Salinas and Urdangarin, 1994) – Transformation of certain biological assemblages (Branch et al., 1990) – Effects on larval settlement, recruitment, growth and survival (Airoldi, 2003)
Oil pollution	– Oil spills (Watt et al.,1993)	– Partial or complete loss of macrobenthic diversity (Jones et al., 1996)
Heavy metal pollution	– Denudation of ore-containing rocks and volcanism (Clark et al., 1997) – Domestic and industrial discharge and urban run-off (Anderlini, 1992; Clark et al., 1997)	– Reduced benthic growth eg in mussels and fucoid algae (Munda and Hudnik, 1986) – Effects on larval development (Fichet et al., 1998)
Plastic and microplastic pollution	– Marine and land-based sources (GESAMP, 1991)	– Possible ingestion by benthic organisms (Thompson et al., 2004; Rezania et al., 2018)

Modified from Msangameno (2015).

excessive input of nutrients, silt, pesticides, heavy metals and debris into the marine environment has been demonstrated to have adverse effects on biological ecosystems (Crowe et al., 2000). Table 3 summarizes the potential impact of some of the major types of pollution on rocky shores. This may also apply to other nearshore habitats such as sandy beaches.

Coastal development and urbanization

Coastal development and urbanization, if not well managed may have deleterious effect on shoreline habitats. Growth of urban areas along the coast and the associated activities leading to the use and conversion of coastal land have been having negative impact on marine biodiversity in the WIO (Celliers and Ntombela, 2015). Poorly planned and uncoordinated development in most coastal urban centres has resulted from inadequate management of the coastal zone (Fraschetti et al., 2011), with significant implications for marine ecosystems such as rocky shores and sandy beaches.

For instance, in many areas along the coast, construction and infrastructure development have led to increased coastal erosion and water turbidity, affecting certain sessile communities. Moreover, the construction of certain defensive structures for shoreline protection (eg seawalls) interferes with key hydrographic and biological processes essential to maintenance of the integrity of such ecosystems (Branch et al., 2008; Bertasi et al., 2009), by interrupting the supply of recruits, nutrients and food. Such improper beach armouring can exacerbate beach erosion and loss of turtle nesting habitats, block access to nesting turtles and fatally entrap turtles (Eckert et al., 1999). Likewise, another threat to sandy shore habitats is sand mining, with the persistent removal of beach sand disrupting stabilizing vegetation, also exacerbating beach erosion.

Climate change

It is no longer a debate whether global climate is changing. The global average temperature is now about 0.85°C above the pre-industrial range. Over the past 100 years, global sea level has risen by an average of 1–2 mm yr^{-1}, with current projections being a rise of between 0.29m and 1.1m by the end of this century (IPCC, 2019). Global climate change is seriously impacting the ecological systems in marine waters worldwide, with sea level rise and increases in sea surface temperature being regarded as some of the most important aspects of change along the coast (Tsyban et al., 1990).

Climate change is predicted to drive significant shifts in the structure of biological communities (Helmuth et al., 2006; Hawkins et al., 2009), with recent climatic events already affecting the survival, development, phenology, physiology and ecology of a wide range of species within marine ecosystems (Walther et al., 2002). Potential impacts of climate change on the intertidal habitats of the WIO are summarized in Table 4, being modelled around the potential environmental alterations to be caused by various climate-related stressors such as increased sea surface temperature, increased irradiation, sea level rise and changes in the patterns of atmospheric and oceanographic processes (Steffani, 2000).

CONSERVATION STATUS AND LEVEL OF THREAT

Conservation status

Despite their considerable socio-economic and ecological importance, rocky shores in the WIO have received relatively little attention. Among the many intertidal habitats in the region, rocky shores are generally understudied, and undermanaged, with poor or no monitoring (Nordlund et al., 2014; Maina, 2015). Therefore, apart from their known goods and services in support of livelihoods, little is known of the quantifiable status of the resources within such habitats, as well as their generalized conservation status. This can be attributed to the general lack of dedicated, comprehensive, multispecies region-wide assessments made of such habitats.

However, it must be appreciated that the conservation status of rocky and sandy shore habitats can be site-specific, varying across different geographical settings and jurisdictions. This could be mainly due to geographical variations in the levels of direct threats to such habitats, as specified in the 2012 IUCN-CMP Unified Classification Scheme (Version 3.2) and as in the detailed assessment made in the subsequent section of this chapter. Although constantly under higher anthropogenic pressure, nearshore habitats in the WIO can be generally considered 'vulnerable' at worst (refer IUCN Conservation status categories). As far as the IUCN Threat Impact Scoring System (IUCN, 2012) the threat to nearshore habitats such as rocky and sandy shores can putatively be judged to be continuing, with slow increase in severity, affecting the majority of the habitats in the region, thus the threats are of medium level impact.

Table 4: Potential impacts of global climate change on the rocky reef ecosystems

CLIMATE RELATED PHENOMENON	ECOSYSTEM IMPACT
Increased seawater temperature	– Changes in biological composition in favour of more heat-resistant organisms (Barry et al., 1995; Steffani, 2000) – Effects on trophic interactions within benthic biological communities (Sanford, 1999) – Polar-ward shifts in genetic range shifts for certain species (Ling et al., 2009) – Local extinction of certain species (Helmuth et al., 2002)
Sea level rise	– Submergence and loss of benthic biological assemblages, especially on reef flats and wave-cut platforms (Steffani, 2000) – Upward displacement of benthic organisms on gentle sloping shores (Jackson and McIlvenny, 2011)
Ocean acidification	– Reduced calcification in calcareous benthos eg certain species of crustaceans, molluscs, echinoderms and calcareous algae – Impairment of physiological and developmental processes in many benthic species, especially in the early life history stages (Gaylord et al., 2011) – Shifts in the community structure, dynamics and productivity
Changes in oceanic and nearshore circulations	– Changes in rates of settlement and recruitment of benthic organisms – Changes in patterns of biotic interactions such as predation, herbivory and competition (Menge and Sutherland, 1987) – Reduced productivity due to possible changes in patterns of nutrient and plankton supply (Menge et al., 1997)
Increased intensity and frequency of storms	– Reduced natural succession and/or recovery of benthic communities – Reduced habitat heterogeneity and species diversity (Sousa, 1985) – Community transformation eg reduced abundance of perennial species in favour of short-lived, fast-growing ephemeral species (Steffani, 2000)
Increased coastal erosion and changes in sediment dynamics	– Loss of sand-intolerant species; increased dominance by sand-tolerant species; thus, reduced diversity within benthic communities (Menge et al., 1983)

Source: Msangameno (2015).

Assessing current levels of threat

Although several observations in the region reveal continued support of rocky shores and sandy beaches for the sustainable exploitation of their resources (Msangameno, 2015), sustained pressures in the form of over-harvesting, habitat alterations, pollution and possibly climate change, may have serious impact on the ability of such ecosystems to offer these vital goods and services. A comprehensive prognosis for the future of rocky reef habitats was given by Branch et al. (2008), in which a wide range of impacts are predicted at all ecological levels (individual, populations and communities). Although many of these impacts are predicted to be localized, such as point pollution sources and local human resource exploitation, they are projected to affect entire coastlines if unchecked, leading to widespread changes in primary and secondary productivity, with consequences for both commercial and unexploited species, together with associated ecosystem goods and services (Branch et al., 2008).

However, it has to be appreciated that the ability to predict the consequences of changes in a single impact may vary from reasonable certainty to considerable uncertainty, for example in terms of ecosystem responses to changes in global climate or the introduction of non-native species (Thompson et al., 2002). As the ability to forecast the interactive effects of several environmental factors is at best fairly modest, unpleasant surprises can be expected in the future (Branch et al., 2008). This will happen where environmental change induces shifts between alternate states (Paine et al., 1998); an organism is particularly susceptible to a pollutant; or an exotic species has potentially a much more prominent role in an invaded community than in its home range (Branch et al., 2008).

EXISTING PROTECTION LEVELS

To safeguard the provision of ecosystem goods and services by marine ecosystems there need to be deliberate actions to protect such ecosystems against a number of anthropogenic stressors. Various forms of marine habitat protection exist in the WIO, with considerable success. There are seldom any exclusive measures for the protection of rocky reef habitats and sandy shores in the region of the same level as accorded to such 'keystone' ecosystems as coral reefs and mangroves. However, some nearshore habitats tend to fall within the broader realms of the existing area-based protection mechanisms such as Marine Protected Areas (MPAs) and Locally Managed Marine Areas (LMMAs).

Currently there are 143 MPAs (UNEP-Nairobi Convention and WIOMSA, 2021) in the WIO region. This is on top of 11 000 km^2 being protected under various LMMAs arrangements (Rocliffe et al., 2014). By assisting in regulating levels of resource harvesting, and improving rates of recruitments, growth and survivorships of various benthic species, these area-based measures have proved to be effective in maintaining or even improving biological abundances, diversity and endemism on rocky shores and similar habitats, as has been shown in certain parts of the WIO (see Postaire et al., 2014). Despite their usefulness, MPAs and LMMAs will only offer effective protection of such ecosystems and their resources if their spatial coverage becomes large enough to include the majority of such habitats. Currently, the protection level of rocky reef habitats in the region can at best be described as poor (< 20 per cent).

PRIORITY OPTIONS FOR CONSERVATION

Since only a fraction of the rocky reef habitats in the WIO are under any form of formal protection, there are considerable opportunities for improving their protection levels to 20 per cent of total areal extent. Although there is no definitive figure of the areal extent of rocky shores in the region, one of the best options for prioritizing and increasing such spatial coverage would be the incorporation of additional areas into the existing MPAs and LMMAs within each national jurisdiction.

To assist in the identification of such additional areas, assessments can be undertaken to measure the level of their ecological potential, based on measurable criteria such as those used to designate the Ecologically or Biologically Significant Areas (EBSA) (CBD Secretariat, 2008). These are areas meeting at least one of the following criteria: ecological uniqueness of rareness; special importance for life history stages of species; importance for threatened, endangered or declining species and/or habitats; vulnerability, fragility, sensitivity, or slow recovery; biological productivity; biological diversity; or naturalness (Dunn et al., 2014). There are 39 marine EBSAs identified in the WIO region, and these can be used as the basis for increasing areal coverage by placing rocky shores and reefs, sandy beaches and other near-shore habitats under formal protection.

REFERENCES

Airoldi, L. (2003). The effects of sedimentation on rocky coast assemblages. In *Oceanography and marine Biology: an annual review* (eds. R.N. Gibson and Atkinson, R.J.A.) vol. 41 pp. 161–236

Anderlini, V.C. (1992). The effect of sewage on trace metal concentrations and scope for growth in *Mytilus edulis aoteanus* and *Perna canaliculus* from Wellington Harbour, New Zealand. *Sci. Total Environ.* 125 pp. 263–288

Barnes, N., Bamber, R.N., Bennell, G., Cornelius, N., Glassom, D., Henderson, S.C., Jiddawi, N., Lee, C.N.W., Macia, A., Msangameno, D.J., Paula, J., Yahya, S. and Ferrero, T.J. (2011). Assessment of regional and local biodiversity in tropical and subtropical coastal habitats in the East African Marine Ecoregion. *Biodivers. Conserv*. 20(2075), p. 2109. https://doi.org/10.1007/s10531-011-0076-2

Barry, J.P., Baxter, C.H., Sagarin, R.D. and Gilman, S.E. (1995). Climate-related, long-term faunal changes in a California rocky intertidal community. *Science* 267, pp. 672–675

Bertasi, F., Colangelo, M.A., Colosio, F., Gregorio, G., Abbiati, M. and Ceccherelli, V.U. (2009). Comparing efficacy of different taxonomic resolutions and surrogates in detecting changes in soft bottom assemblages due to coastal defence structures. *Mar. Pollut. Bull*. 58, pp. 686–694

Boaventura, D., Alexander, M., Santina, P.D., Smith, N.D., Ré, P., da Fonseca, L.C. and Hawkins, S.J. (2002). The effect of grazing on the distribution and composition of low shore algal communities on the central coast of Portugal and on the southern coast of Britain. *J. Exp. Mar. Biol. Ecol*. 267, pp. 185–206

Bosman, A.L. and Hockey, P.A.R. (1988). The influence of seabird guano on the biological structure of rocky intertidal communities on islands off the west coast of Southern Africa. *Afr. J. Mar. Sci.* 7, pp. 61–68

Bourjea, J., Nel, R., Jiddawi, N.S., Koonjul, M.S. and Bianchi, G. (2008). Sea turtle bycatch in Indian ocean. *West. Indian Ocean J. Mar. Sci*. 7(2), pp. 137–150

Branch, G.M., Eekhout, S. and Bosman, A. L. (1990). Short-term effects of the 1988 Orange River floods on the intertidal rocky-shore communities of the open coast. *Trans. R. Soc. South Africa* 47, pp. 331–354

Branch, G., Thompson, R.C., Crowe, T.P., Castilla, J.C., Langmead, D. and Hawkins, S.J. (2008). Rocky intertidal shores: prognosis for the future. In *Aquatic Ecosystems Trends and Global Prospects* (ed. N.V.C Polunin). Cambridge University Press

Bustamante, R.H. and Branch, G.M. (1996). Large scale patterns and trophic structure of southern African rocky shores: the roles of geographic variation and wave action. *J. Biogeogr*. 23, pp. 339–35

Casu, D., Ceccherelli, G., Curini-galletti, M. and Castelli, A. (2006). Short-term effects of experimental trampling on polychaetes of a rocky intertidal substratum (Asinara Island MPA, NW Mediterranean. In *Scientific Advances in Polychaete Research* (eds. R. Sarda, G. San Martin, E. Lopez, and D. George), pp. 368. Institut de Ciències del Mar

CBD Secretariat (2008). Decision IX/20 Marine and Coastal Biodiversity. UNEP/CBD/COP/DEC/IX/20. http://www.cbd.int/doc/decisions/cop-09/cop-09-dec-20-en.pdf>-doi:10.1016/j.marpol.2013.12.002

Celliers, L. and Ntombela, C. (2015). Urbanisation, Coastal Development and Vulnerability, and Catchments. In *The Regional State of the Coast Report: Western Indian Ocean*. UNEP-Nairobi Convention and WIOMSA. Nairobi, Kenya, 546 pp.

Charpy-Roubaud, C. and Sournia, A. (1990). The comparative estimation of phytoplanktonic, microphytobenthic and macro-phytobenthic primary production in the oceans. *Mar. Microb. Food Webs* 4, pp. 31–57

Clark, R.B., Frid, C. and Attrill, M. (1997). *Marine Pollution* (4th ed.). Clarendon Press, Oxford

Coleman, R.A., Underwood, A.J., Benedetti-Cecchi, L., Åberg, P., Arenas, F., Arrontes, J., Castro, J., Hartnoll, R.G., Jenkins, S.R., Paula, J., Della-Santina, P. and Hawkins, S.J. (2006). A continental scale evaluation of the role of limpet grazing on rocky shores. *Oecologia* 147, pp. 556–564

Crowe, T.P., Thompson, R.C., Bray, S. and Hawkins, S.J. (2000). Impacts of anthropogenic stress on rocky intertidal communities. *J. Aquat. Ecosyst. Stress Recovery* 7, pp. 273–297

Cunha E.A., Carvalho, R.A.A., Monteiro-Neto, C., Moraes, L.E.S. and Araújo, M.E. (2008). Comparative analysis of tidepool fish species composition on tropical coastal rocky reefs at State of Ceará, Brazil. *Iheringia. Série Zoologia* 98(3), pp. 379–390

Deepananda, K.H.M.A and Macusi, E.D. (2012). Human disturbance on tropical rocky shore assemblages and the role of marine protected areas in reducing its impact. *Philipp. Agric. Sci*. 95 (1), pp. 87–98

Denis, T.G. (2003). Effects of human foot traffic on the standing stock, reproduction, and the size structure of populations of the intertidal rockweed *Silvetia compressa (O. Fucales)*. MSc. dissertation, California State University. California, USA

Dunn, D.C., Ardron, J.A., Bax, N.J., Bernal, P., Cleary, J., Cresswell, I., Donnelly, B., Dunstan, P.K., Gjerde, K.M., Johnson, D., Kaschner, K., Lascelles, B., Rice, J., von Nordheim, H., Wood, L. and Halpin, P.N. (2014). The Convention on Biological Diversity's Ecologically or Biologically Significant Areas: origins, development, and current status. *Mar. Policy* 49, pp. 137–145. https://doi.org/10.1016/j.marpol.2013.12.002

Durville, P. and Chabanet, P. (2009). Intertidal rock pool fishes in the natural reserve of Glorieuses Islands (Western Indian Ocean). *Western Indian Ocean Journal of Marine Sciences* 8(2), pp. 225–230

Durville, P., Chabanet, P. and Quod, J.P. (2003). Visual census of the reef fishes in the natural reserve of the Glorieuses Islands (Western Indian Ocean). *West. Indian Ocean J. Mar. Sc*i. 2(2), pp. 59–104

Eckert, K.L., Bjorndal, K.A. Abreu, F.A. and Donnelly, M. (eds.) (1999). Research and management techniques for the conservation of sea turtles. IUCN/SSC Marine Turtle Specialist Group Publication No. 4. Washington, D.C.

Fennessy, S. and Green, A. (2015). Shelf sediments and biodiversity. In *The Regional State of the Coast Report: Western Indian Ocean*. pp. 103–112. UNEP and WIOMSA. Nairobi, Kenya

Fichet, D., Radenac, G. and Miramand, P. (1998). Experimental studies of impacts of harbour sediments re-suspension to marine invertebrate larvae: Bioavailability of Cd, Cu, Pb and Zn and toxicity. *Mar. Pollut. Bull*. 36, pp. 7–12

Firth, L.B., Thompson, R.C., White, F.J., Schofield, M., Skov, M.W., G. Hoggart, S.P.G., Jackson, J., Knights, A.M. and Hawkins, S.J. (2013). The importance of water-retaining features for biodiversity on artificial intertidal coastal defence structures. *Divers. Distrib*. 19(10), pp. 1275–1283

Fraschetti, S., Terlizzi, A., Guarnieri, G., Pizzolante, F., D'Ambrosio, P., Maiorano, P., Beqiraj, S. and Boero, F. (2011). Effects of unplanned development on marine biodiversity: A lesson from Albania (Central Mediterranean Sea). *J. Coast. Res.* 58, pp. 106–115

Gaylord, B., Hill, T.M., Sanford, E., Lenz, E.A., Jacobs, L.A., Sato, K.N., Russell, A.D. and Hettinger, A. (2011). Functional impacts of ocean acidification in an ecologically critical foundation species. *J. Exp. Biol*. 214 (15), pp. 2586–2594

GESAMP (1991). *The State of Marine Environment*. Group of Experts on the Scientific Aspects of Marine Pollution. Blackwell Scientific Publications, London

Gibbons, E. and Remaneva, L. (2011). Curio trade: Southwest Madagascar. Reef Doctor, London, 41 pp

Gibson, R.N. and Yoshiyama, R.M. (1999). Intertidal fish communities. *In Intertidal fish: life in two worlds* (eds. M.H. Horn, K.L. Martin, M.A. Chotkowski), pp. 264–296. Academic Press, San Diego, CA

Gössling, S., Kunkel, T., Schumacher, K. and Zilger, M. (2004). Use of molluscs, fish and other marine taxa by tourism in Zanzibar, Tanzania. *Biodivers. Conserv*. 13(12), pp. 2623–2639

Hartnoll, R.G. (1976). The Ecology of some rocky shores in tropical East Africa. *Estuar. Coast. Shelf Sci*. 4(1), pp. 1–10

Hawkins, S.J., Sugden, H.E., Mieszkowska, N., Moore, P.J., Poloczanska, E., Leaper, R., Herbert, R.J.H., Genner, M.J., Moschella, P.S., Thompson, R.C., Jenkins, S.R., Southward, A.J. and Burrows, M.T. (2009). Consequences of climate-driven biodiversity changes for ecosystem functioning of North European rocky shores. *Mar. Ecol. Prog. Ser*. 396, pp. 245–259

Helmuth, B., Harley, C.D.G., Halpin, P.M., O'Donnell, M., Hofmann, G.E. and Blanchette C.A. (2002). Climate change and latitudinal patterns of intertidal thermal stress. *Science* 298, pp. 1015–1017

Helmuth, B., Mieszkowska, N., Moore, P. and Hawkins, S.J. (2006). Living on the edge of two changing worlds: forecasting the responses of rocky intertidal ecosystems to climate change. *Annu. Rev. Ecol. Evol. Syst*., pp. 373–404

Hill, M. and Currie, D. (2007). *Wildlife of Seychelles*. HarperCollins Publishers Ltd., London. 272 pp.

Holmer, M., Lassus, P., Stewart, J.E. and Wildish, D.J. (2001). ICES symposium on environmental effects of mariculture. Introduction. *ICES J. Mar. Sci*. 58, pp. 363–368

IPCC, (2019). *IPCC Special Report on the Ocean and Cryosphere in a Changing Climate.* (eds. Pörtner, H.-O., Roberts, D.C., Masson-Delmotte, V., Zhai, P., Tignor, M., Poloczanska, E., Mintenbeck, K., Alegría, A., Nicolai, M., Okem, A., Petzold, J., Rama, B., Weyer, N.M.)

Jackson, A.C. and McIlvenny, J. (2011). Coastal squeeze on rocky shores in northern Scotland and some possible ecological impacts. *J. Exp. Mar. Biol. Ecol.* 400, pp. 314–321

Janssen, G. and Mulder, S. (2005). Zonation of macrofauna across sandy beaches and surf zones along the Dutch coast. *Oceanologia* 47(2), pp. 265–282

Jones, D.A., Watt, I., Plaza, J. and Woodhouse, T.D. (1996). Natural recovery of the intertidal biota within the proposed marine habitat and wildlife sanctuary for the Gulf (Saudi Arabia) after the 1991 Gulf War oil spill. In *A Marine Wildlife Sanctuary for the Arabian Gulf. Environmental Research and Conservation Following the 1991 Gulf War Oil Spill* (eds. F. Krupp, H. Abuzinada and I.A. Nader), pp. 138–158. NCWCD, Riyadh and Senckenberg Research Institute. Frankfurt, Germany

Kalk, M. (1995). *A Natural History of Inhaca Island, Mozambique*. Witwatersrand University Press, South Africa, 395 pp.

Kim, K.Y., Choi, T.S., Huh, S.H. and Garbary, D.J. (1998). Seasonality and community structure of subtidal benthic algae from Daedo Island, Southern Korea. *Botanica Marina* 41, pp. 357–365

Knox, G.A. (2001). *The Ecology of Seashores*. p. 557. CRC Press

Kostylev, V.E., Erlandsson, J., Ming, M.Y. and Williams, G.A. (2005). The relative importance of habitat complexity and surface area in assessing biodiversity: Fractal application on rocky shores. *Ecol. Complex*. 2, pp. 272–286

Kyle, R., Pearson, B., Fielding, P.J., Robertson W.D. and Birnie, S.L. (1997). Subsistence shellfish harvesting in the Maputaland Marine Reserve in Northern KwaZulu-Natal, South Africa: Rocky Shore Organisms. *Biol. Conserv*. 82, pp. 183–192

Lewis, J.R. (1964). *The Ecology of Rocky Shores*. 323 pp. English Universities Press, London

Lindberg D.R., Estes, J.A. and Warheit, K.I. (1998). Human influences on trophic cascades along rocky shores. *Ecol. Appl*. 8, pp. 880–890

Ling, S.D., Johnson, C.R., Ridgway, K., Hobday, A.J. and Haddon, M. (2009). Climate-driven range extension of a sea urchin: inferring future trends by analysis of recent population dynamics. *Glob. Chang. Biol.* 15, pp. 719–731

Little, C. and Kitching, J.A. (1996). *The Biology of Rocky Shores*. 352 p. Oxford University Press

MacDonald, L.H., Anderson, D.M. and Dietrich, W.E. (1997). Paradise threatened: land use and erosion on St. John, US Virgin Islands. *Environ. Manage*. 21, pp. 851–863

Mahon, R. and Mahon, S. (1994). Structure and resilience of a tidepool fish assemblage at Barbados. *Devel. Enviro. Biol. Fish* 15, pp. 171–190

Maina, J. (2015). Beaches and the Nearshore. In *The Regional State of the Coast Report: Western Indian Ocean*. pp. 37–48 UNEP and WIOMSA. Nairobi, Kenya

Marquez, R.M. (1990). Sea turtles of the world, FAO Species catalogue (11) pp. 25–30. Rome, Italy

Martin, A.K. and Flemming, B.W. (1988). Physiography, structure and geological evolution of the Natal continental shelf. In *Coastal Ocean studies off Natal, South Africa* (ed. E.H. Schumann). pp. 11–46. Springer-Verlag, New York

McLachlan, A., Erasmus, T. and Furstenberg, J.P. (1977). Migrations of Sandy Beach Meiofauna. *Afr. Zool*. 12(2), pp. 257–277. https://doi.org/10.1080/00445096.1977.11447575

Menge, B.A., Daley, B.A., Wheeler, P.A., Dahlhoff, E., Sanford, E. and Strub, P.T. (1997). Benthic-pelagic links and rocky intertidal communities: Bottom-up effects on top-down control? *Proceedings of the National Academy of Sciences USA* 94, pp. 14530–14535

Menge, B.A., Ashkenas, L.A. and Matson, A. (1983). Use of artificial holes in studying community development in cryptic marine habitats in a tropical rocky intertidal region. *Mar. Biol*. 77, pp. 129–142

Menge, B.A. and Sutherland, J.P. (1976). Species diversity gradients: synthesis of the roles of predation, competition and temporal heterogeneity. *Am. Nat*. 110, pp. 351–369

Menge, B.A. and Sutherland, J.P. (1987). Community regulation: variation in disturbance, competition, and predation in relation to environmental stress and recruitment. *Am. Nat*. 130, pp. 730–757

Moreno, M., Ferrero, T.J., Granelli, V., Albertelli, G. and Fabiano, M. (2006). Across shore variability and trophodynamic features of meiofauna in a microtidal beach of the NW Medditerranean. *Estuar. Coast. Shelf Sci*. 66, pp. 357–367

Moschella, P.S., Abbiati, M., Åberg, P., Airoldi, L., Anderson, J.M., Bacchiocchi, F., Bulleri, F., Dinesen, G.E., Frost, M., Gacia, E., Granhag, L., Jonsson, P.R., Satta, M.P., Sundelöf, A., Thompson, R.C. and Hawkins, S.J. (2005). Low-crested coastal defence structures as artificial habitats for mamartrine life: Using ecological criteria in design. *Coast. Eng*. 52, pp. 1053–1071

Msangameno, D. (2015). Intertidal and Nearshore Rocky Reefs. In *The Regional State of the Coast Report: Western Indian Ocean*. pp 85–101. UNEP and WIOMSA. Nairobi, Kenya

Munda, I.M. and Hudnik, V. (1986). Growth response of *Fucus vesiculosus* to heavy metals, singly and in dual combinations, as related to accumulation. *Botanica Marina* 29, pp. 410–412

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

Nichols, G. (2009). *Sedimentology and stratigraphy*. 2nd edition. Wiley-Blackwell

Nordlund, L.M., Torre-Castro, M., Erlandsson, J., Conand, C., Muthiga, N., Jiddawi, N. and Gullström, M. (2014). Intertidal Zone Management in the Western Indian Ocean: Assessing Current Status and Future Possibilities Using Expert Opinions. *Ambio* 43(8). https://doi.org/10.1007/s13280-013-0465-8

Paine, T., Tegner, M.J. and Johnson, E.A. (1998). Compounded Perturbations Yield Ecological Surprises. *Ecosystems* 1, pp. 535–545

Postaire, B,, Bruggemann, J.H., Magalon, H. and Faure, B. (2014). Evolutionary Dynamics in the Southwest Indian Ocean Marine Biodiversity Hotspot: A Perspective from the Rocky Shore Gastropod Genus Nerita. *PLoS one* 9(4), e95040. https://doi.org/10.1371/journal.pone.0095040

Puga-Bernabéu, A., Webster, J.M., Beaman, R.J. and Guilbaud, V. (2011). Morphology and controls on the evolution of a mixed carbonate-siliciclastic submarine canyon system, Great Barrier Reef margin, north-eastern Australia. *Mar. Geol.* 289, pp. 100–116

Qasim, S.Z. (1998). *Glimpses of the Indian Ocean*. pp. 206 Universities Press (India) Ltd

Raffaelli, D. and Hawkins. S. (1996). *Intertidal Ecology*. pp. 356 Chapman and Hall, London

Raimondi, P.T. (1988). Rock type affects settlement, recruitment, and zonation of the barnacle *Chthamalus anisopoma Pilsbury. Journal of Experimental Marine Biology and Ecology* 123, pp. 253–267

Ramsay, P.J. (1996). Quaternary marine geology of the Sodwana Bay continental shelf, northern KwaZulu-Natal. *Bul. Geol. Survey South Africa* 117, pp. 1–86

Rezania, S., Park, J., Din, M.F.M., Taib, S.M., Talaiekhozani, A., Yadav, K.K. and Kamyeb, H. (2018). Microplastics pollution in different aquatic environments and biota: A review of recent studies. *Mar. Pollut. Bull*. 133, pp. 191–208

Rocliffe, S., Peabody, S., Samoilys, M. and Hawkins, J.P. (2014). Towards A Network of Locally Managed Marine Areas (LMMAs) in the Western Indian Ocean. *PLoS one* 9(7), p. e103000. https://doi.org/10.1371/journal.pone.0103000

Rodriguez, J.G., Lopez, J. and Jaramillo, E. (2001). Community structure of the intertidal meiofauna along a gradient of morphodynamic sandy beach types in southern Chile. *Rev. Chil. Hist. Nat*. 74, pp. 885–897

Ruwa, R.K. (1996). Intertidal wetlands. In *East African Ecosystems and Conservation* (eds. T.R. McClanahan and T.P. Young), pp. 101–127. Oxford University Press

Saiz-Salinas, J.I. and Urdangarin, I.I. (1994). Response of sublittoral hard substrate invertebrates to estuarine sedimentation in the outer harbour of Bilbao (N. Spain). *Mar. Ecol.* 15, pp. 105–131

Sanford, E. (1999). Regulation of keystone predation by small changes in ocean temperature. *Science* 283, pp. 2095–2097

Schramm, W. (1996). The Baltic Sea and its transition zones. In *Marine Benthic Vegetation: Recent Changes and the Effects of Eutrophication* (eds. W. Schramm and P. Nienhuis), pp. 131–164. Springer, Berlin

Sebens, K.P. (1991). Habitat structure and community dynamics in marine benthic systems. In *Habitat Structure: the Physical Arrangement of Objects in Space* (eds. S.S. Bell, E.D. McCoy and H.R. Mushinsky), pp. 211–234. Chapman and Hall, London

Short, A.D. (2012). Coastal Processes and Beaches. *Nat. Edu. Knowledge* 3(10), p. 15

Siegfried, W.R. (1994). *Rocky Shores-Exploitation in Chile and South Africa*. 177 p. Springer-Verlag

Sindorf, V., Cowburn, B. and Sluka, R. (2015). Rocky intertidal fish assemblage of the Watamu Marine National Park, Kenya (Western Indian Ocean). Environmental Biology of Fishes 98(7). DOI:10.1007/s10641-015-0397-1

Sousa, W.P. (1985). Disturbance and patch dynamics on rocky intertidal shores. In *The Ecology of Natural Disturbance and Patch Dynamics* (eds. S.T. Pickett and P.S. White), pp. 101–124. Academic Press, New York

Steffani, N.C. (2000). Possible effects of climate change on the biodiversity of Southern African rocky shores. In *The effects of climate change on Marine Biodiversity in South Africa* (eds. B.M. Clark, N.C. Steffani, S. Young, A.J. Richardson and A.T. Lombard). Report prepared for the Foundation for Research Development, South African country study on Climate Change, Vulnerability and Adaptation Assessment. Marine Biodiversity Section pp. 129

Stephenson, T.A. and Stephenson, A. (1972). *Life between Tidemarks on Rocky Shores.* 423 p. W.H. Freeman and Co., San Francisco.

Steyaert, M., Herman, P.M.J., Moens, T., Widdows, J. and Vincx, M. (2001). Tidal migration of nematodes on an estuarine tidal flat (the Molenplaat, Schelde Estuary, SW Netherlands). *Mar. Ecol. Prog. Ser.* 224, pp. 299–304

Thompson, R.C., Crowe, T.P. and Hawkins, S.J. (2002). Rocky intertidal communities: past environmental changes, present status and predictions for the next 25 years. *Environ. Conserv.* 29, pp. 168–191

Thompson, R.C., Olsen, Y., Mitchell, R.P., Davis, A., Rowland, S.J., John, A.W.G., McGonigle, D. and Russell, A.E. (2004). Lost at Sea: Where Is All the Plastic? *Science* 304(5672), pp. 838

Troelli, M., Robertson-Andersson, D., Anderson, R.J., Bolton, J.J. Maneveldt, G.C., Halling, C. and Probyn, T. (2006). Abalone farming in South Africa: An overview with perspectives on kelp resources, abalone feed, potential for on-farm seaweed production and socio-economic importance. *Aquaculture* 257(1–4), pp. 266–281

Tsyban, A.V., Everett, J.T. and Titus, J.G. (1990). World oceans and coastal zones. In *Climate Change: The IPCC Impacts Assessment. Contribution of Working Group II to the First Assessment Report of the Intergovernmental Panel on Climate Change* (eds. W.J.McG. Tegart, G.W. Sheldon and D.C. Griffiths), chapter 6, pp. 1–28. Australian Government Service, Canberra, Australia

UNEP/Nairobi Convention Secretariat (2009). *Transboundary Diagnostic Analysis of Land-based Sources and Activities Affecting the Western Indian Ocean Coastal and Marine.*

UNEP-Nairobi Convention and WIOMSA. (2021). *Western Indian Ocean Marine Protected Areas Outlook: Towards achievement of the Global Biodiversity Framework Targets.* UNEP and WIOMSA, Nairobi, Kenya, 298 pp.

Wafar, M., Venkataraman, K., Ingole, B., Khan, S.A. and LokaBharathi, P. (2011). State of knowledge of coastal and marine biodiversity of Indian Ocean countries. *PLoS one* 6, e14613

Walther, G.R., Post, E., Convey, P., Menzel, A., Parmesan, C., Beebee, T.J.C., Fromentin, M., Hoegh-Guldberg, O. and Bairlein, F. (2002). Ecological responses to recent climate change. *Nature* 416, pp. 389–395

Watt, I., Woodhouse, T. and Jones, D.A. (1993). Intertidal clean-up activities and natural regeneration on the Gulf Coast of Saudi Arabia from 1991 to 1992 after the 1991 Gulf oil spill. *Mar. Pollut. Bull*. 27, pp. 325–331

Williams, G.A. and Morritt, D. (1995). Habitat partitioning and thermal tolerance in a tropical limpet, *Cellana grata. Mar. Ecol. Prog. Ser.* 124, pp. 89–103

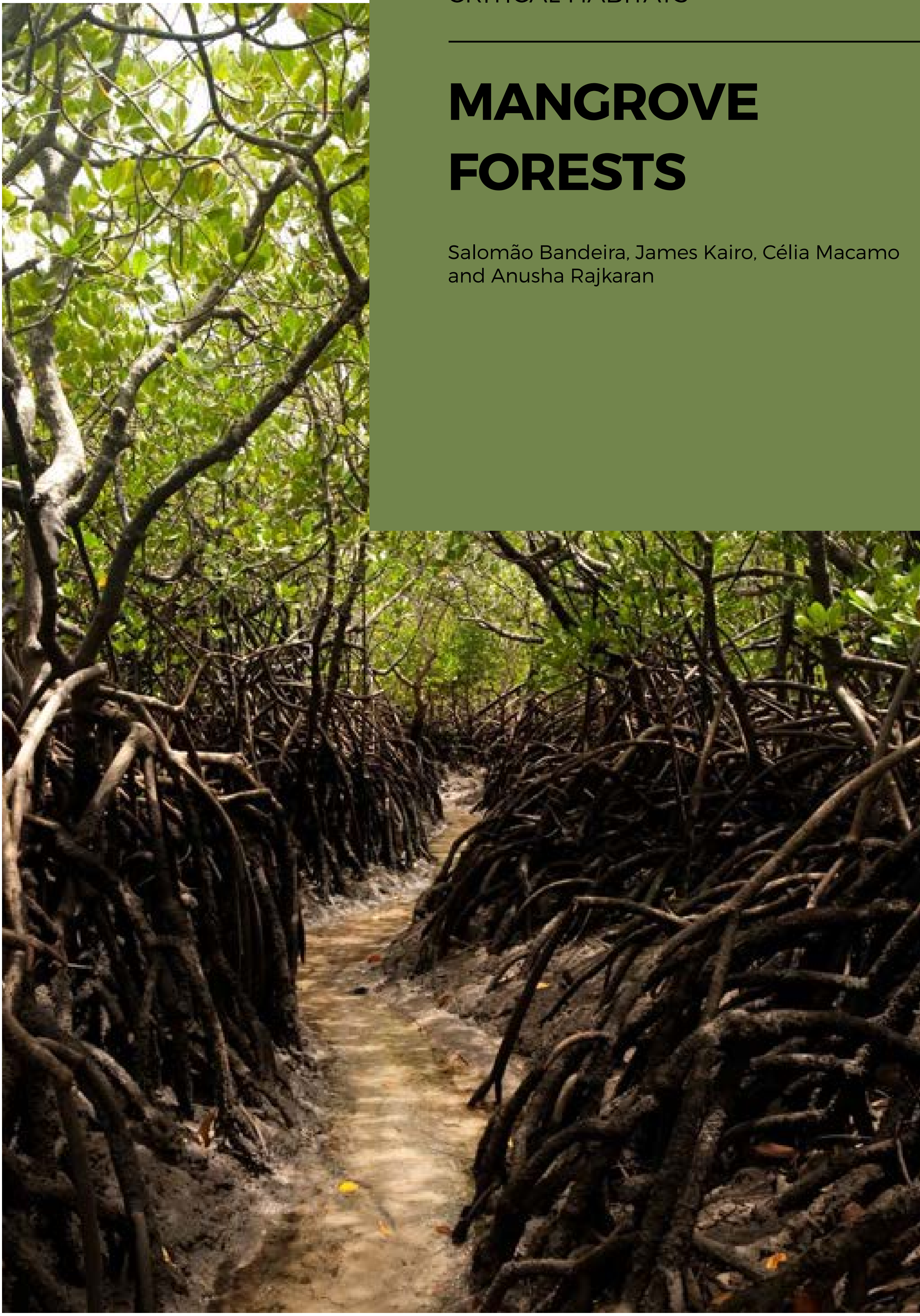
Worm, B. and Lotze, H.K. (2006). Effects of eutrophication, grazing and algal blooms on rocky shores. *Limnol. Oceanogr.* 51(1), pp. 569–579

Worm, B., Lotze, H.K. and Sommer, U. (2000). Coastal food web structure, carbon storage and nitrogen retention regulated by consumer pressure and nutrient loading. *Limnol. Oceanogr*. 45, pp. 339–349
Yorath, C.J. and Nasmith, H.W. (2001). *The Geology of Southern Vancouver Island; a field guide*. Orca Book Publishers

CRITICAL HABITATS

MANGROVE FORESTS

Salomão Bandeira, James Kairo, Célia Macamo and Anusha Rajkaran



Mangroves at the mouth of the Ruvuma Delta near Quionga in North Mozambique. © José Paula

BACKGROUND

Description and characterization of mangroves

Mangroves comprise forests and indeed ecosystems that surround estuaries and sheltered coastlines in the Western Indian Ocean (WIO) tropical and subtropical areas that are now extending into the warm temperate regions. Mangroves are adapted to live under conditions of: (i) changing water level that enables them to develop pneumatophores (aerial roots adapted to gas exchange); (ii) change in salinity prompting the development of mechanisms for salt excretion or avoidance of leaf evaporation and transpiration; and (iii) scarcity of oxygen, enabling the emergence of viviparous fruits common in several mangrove species. Finally, some mangrove species such as *Heritiera littoralis* and *Xylocarpus granatum* also possess floating, water drifting seeds and fruits, respectively.

Five forms of mangrove forest occur in the WIO: (i) fringing mangroves that are widespread and occur along the mostly sheltered coastlines; (ii) riverine mangroves, that are common along the banks of rivers; (iii) basin mangroves, that are widespread and occupy larger areas at the back of both fringing and riverine mangroves; (iv) dwarf or stunted mangroves, common in abnormal or equinoctial tidal reach, having tidal inundation of only a few days in a month; and (v) over-washed mangroves, that occur as isolated stands that are washed over by high tides, and include species such as *Sonneratia alba*, being common in coral limestone areas of northern Mozambique, Tanzania and Kenya. Figure 1 depicts the distribution of mangrove forests on the coasts of the WIO region and their coastal types according to Dürr et al. (2011).

Mangrove occurrence and distribution in the WIO region

Around 6200 km^2 (620 000 ha) of mangroves occur in the WIO, corresponding to about 25 per cent of the African mangrove area and 4.1 per cent of the world's mangrove area (Beentje and Bandeira, 2007; Fatoyinbo and Simard, 2013; Spalding et al., 2010). The largest expanse is found in Mozambique, followed by Madagascar, Tanzania, Kenya, Seychelles and South Africa. Small island states have the smallest areas, nonetheless significant.

Somalia

The most important mangrove formations of Somalia grow in the south and central parts of the country, in the estuary of the Juba River and near the Kenyan border (lower Juba Province). Notable formations also occur on the Bajuni Islands. North of Mogadishu, mangroves are disperse and tend to grow in monospecific stands of *Avicennia marina* that grow behind sand spits (eg in Zeylac, Berbera, Xabo and Caluula) (Carbone and Accordi, 2000; Mumuli et al., 2010; Spalding et al., 2010). There are also reports of past occurrence of mangroves in the north, in an area currently designated an Ecologically or Biologically Significant Marine Areas (EBSA) site (The Great Whirl and Gulf of Aden Upwelling Ecosystem EBSA) which covers mostly northern Somalia but also Gulf of Aden and Socotra Island. The area still needs further assessment regarding the historical occurrence of mangroves, however it was declared an EBSA site because of its high productivity and unique megafauna and birdlife, both resulting from patterns of currents and winds in the area. The areas of historical occurrence of mangrove might lead to their selection for mangroves restoration activities once they have been thoroughly studied.

Kenya

Kenya mangroves cover 61 272 ha (GoK, 2017) covering the Kilifi, Kwalie, Lamu, Mombasa and Tana Counties, the first three areas encompassing around 90 per cent of the country's mangrove forest area. Lamu County contributes 62 per cent of the national total area with 37 350 ha. These forests are dominated by mixed stands of *Rhizophora* but also *Avicennia*, particularly on the landward side as well as stands of *Ceriops* in the mid-zone of the forest. Lamu mangroves occur in the Northern Swamps, Pate Island Swamps, North Central Swamps, Southern Swamps, Mongoni and Dodori Creek Swamps. Twenty per cent (or 7628 ha) of the mangroves of Lamu County are protected in the Kiunga Marine National Reserve (KMNR), which extends mostly into the Northern Swamps and parts of the Northern Central Swamps.

Tana River County mangrove forests cover a total area of 3260 ha stretching from Ngomeni to Kipini. Mangroves in Kilifi County occur in small patches covering Mtwapa, Kilifi-Takaungu, Mida and Ngomeni, with an approximate total area of 8535 ha (Ngomeni accounting for almost 50 per cent). Watamu EBSA encompasses Watamu National Park as part of a complex of marine and tidal habitats along Kenya's north coast south of Malindi covering Malindi Marine National Reserve and Park (mostly with seagrasses) and Mida Creek which has important mangrove forests, with a high diversity of species, including *Ceriops tagal*, *Rhizophora mucronata*, *Bruguiera gymnorrhiza*, *Avicennia marina* and *Sonneratia alba*. It is a key spawning ground for many fish species. Watamu/Malindi Marine Parks and Reserve covers an area of 22 900 ha and is part of a United Nations Biosphere Reserve that also includes the Arabuko Sokoke Coastal Forest.

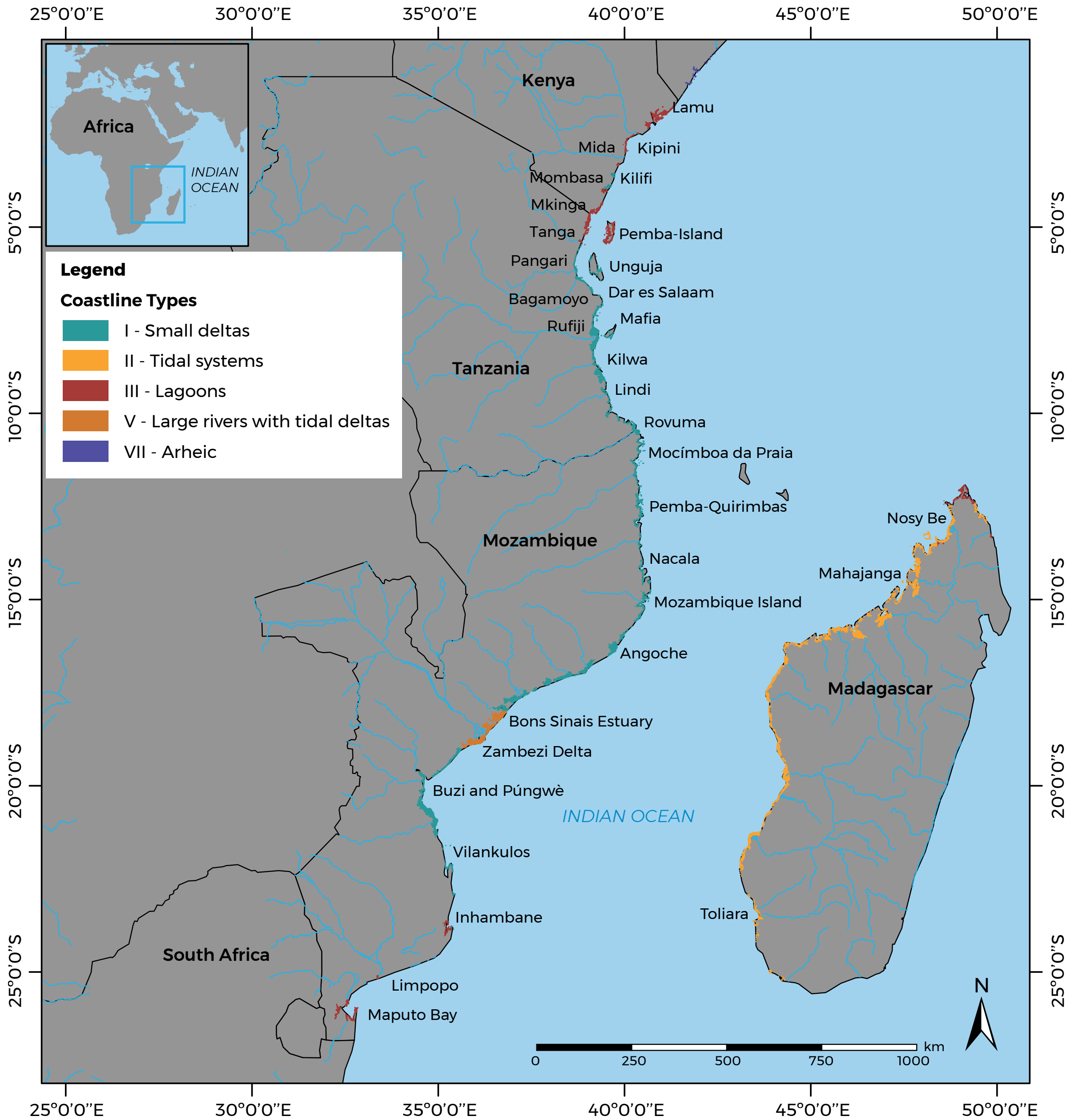


Figure 1: Mangrove nearshore coastal types and their distribution in Kenya, Tanzania, Mozambique, and Madagascar. Five nearshore coastal types are illustrated for the region, including small deltas (type I), tidal systems (II), lagoons (III). Large rivers (V) largely bypass the nearshore filter, while karstic (VI), and arheic coasts (VII) act as inactive filters (Dürr et al., 2011). Adapted from Maina et al. (2021).

Mombasa County, dominated by *Ceriops – Rhizophora* and mixed stands of *Rhizophora*, has 3771 ha of mangroves, distributed mostly along Port Reitz and Tudor Creeks. The forest is heavily degraded through illegal harvesting, land-encroachment and pollution. Reportedly 1850 ha of mangroves in Mombasa County are degraded and in urgent need of rehabilitation. Kwale County mangroves comprise Vanga-Funzi, Gazi Bay, and Ukunda areas, with an area of approximately 8354 ha, dominated by mixed stands of *Ceriops* and *Rhizophora*, as well as pure stands of *Avicennia*.

Pemba-Shimoni-Kisite EBSA lies between the border of Kenya (part of Kwale County) and Tanzania and includes Pemba Island. In Tanzania, the Pemba Island marine area has high diversity of mangroves species (notably *A. marina*, *C. tagal*, *Lumnitzera racemosa*, *Xylocarpus granatum*, *X. moluccensis*, *Heritiera littoralis*, *B. gymnorrhiza*, *R. mucronata* and *S. alba)* often associated with extensive seagrass and algal beds. With a coverage of 13 919 ha, the mangrove stands of Pemba Island appear to be of better condition than those of Unguja, with 5003 ha (Mangora et al., 2016).

Tanzania

Tanzania mangrove area was estimated at 108 300 ha for the mainland (Wang et al., 2003), however recent studies downgraded this to 80 900 ha (Fatoyinbo and Simard, 2013). Meanwhile the area for Zanzibar was estimated in 19 748 ha, with 13 919 ha on Pemba and the remaining on Unguja (Leskinen et al., 1997). For management purposes, the mangroves are divided into 12 blocks (in decreasing order of size): Rufiji, Kilwa, Pemba, Mkinga and Tanga, Mtwara, Unguja, Bagamoyo, Lindi, Mafia, Mkuranga, Pangani and Dar es Salaam.

The Rufiji Delta comprises about 50 per cent of the country's mangrove estate, whilst other important formations within the above referred management blocks include Tanga, Pangani, Wami and Ruvu, and Ruvuma estuary (the last one with approximately 9500 ha) (Ferreira et al., 2009; Mangora et al., 2016). Other formations include those of Zanzibar, Pemba, Mafia Islands (Mangora, 2011) and Unguja Island with the great formation at Chwaka Bay (Shunula and Whittik, 1999). Around Dar es Salaam city, mangroves tend to grow shrubby and up to 5–7 m high. Such are the cases of mangrove formations at Mbweni (located some 30 km north of Dar es Salaam at the mouth of Mpigi River), Msimbazi, Kunduchi, and Mtoni estuary (Mremi and Machiwa, 2003; Kruitwagen et al., 2008). These forests grow under great anthropogenic pressure, including urban expansion, port activities and other associated services (Kruitwagen et al., 2008).

Tanzania jurisdictional area is home to the following EBSAs: Tanga Coelacanth Marine Park with major mangroves within and north of Tanga Bay; Zanzibar (Unguja) – Saadani of which the Unguja island itself supports 5003 ha; Rufiji-Mafia-Kilwa with Rufiji concentrations representing approximately half of Tanzania mangroves.

Mozambique

The Mozambique coastline (of ca 2700 km) can be divided into three parts: (i) coralline type in the north covering the province of Cabo Delgado and Nampula; (ii) swampy coast covering the Sofala bank, massive mangroves occur quite continuously, the largest one being the Zambezi River delta mangrove; (iii) sandy dune in the south with major mangroves in Save River estuary, Inhambane Bay, Limpopo River estuary and Maputo Bay. Globally, Mozambique ranks 13th in mangrove coverage, representing the equivalent of approximately 2.3 per cent of the global mangrove forest area (Giri et al., 2011).

The country's mangrove area is about 305 400 ha (Fatoyinbo and Simard, 2013), with the highest expanses concentrated at Zambézia and Sofala provinces (Table 1). All nine species that occur in the WIO can be found in Mozambique. *Avicennia marina* is widely distributed from north to south, while *S. alba* only occurs from Inhambane province to the north. *Xylocarpus moluccensis* has only been recorded at the Zambezi River delta so far (Trettin et al., 2015).

Table 1: Mozambique mangrove area.

PROVINCE	AREA (ha)
Cabo Delgado	34 730.1
Gaza	465.7
Inhambane	26 055.9
Maputo	17 596.0
Nampula	50 015.1
Sofala	73 553.6
Zambézia	115 337.2
TOTAL	317 753.6

Source: H. Mabilana, H. Balidy, unpublished and ongoing mapping of mangroves for selected provinces.

The transboundary formation of the Rovuma River estuary has approximately 9500 ha and is the largest in Cabo Delgado province. Other important areas include Quissanga-Ibo-Quirimba Island (4300 ha), Macomia (4395 ha), Pemba (2700 ha), Palma (1010 ha) and Mocímboa da Praia (6536 ha). All these formations lie within the Mtwara-Pemba Bay EBSA that was created based on the combination of critical habitats but also charismatic and endangered species in this region including turtles, whales and dugongs.

In Nampula Province, the largest mangrove areas occur within Angoche (18 135 ha), Mussoril (7354 ha), Moma covering also Ligonha River mangroves (16 119 ha), Monjicual (5128 ha), Memba (2229 ha) and Mozambique Island district including mainland areas (339 ha). Nampula and Zambézia provinces are home to Baixo Pinda-Pembane (Primeiras and Segundas Islands) EBSA, covering in its southern regions one of the largest WIO MPA known as Area of Environmental Protection of the Islands Primeiras and Segundas (covering the districts of Angoche, Larde, Moma and Pebane). About 570 ha of mangroves are protected within this MPA.

The Zambezia Province holds the most important mangrove formation of the country, the Zambezi River delta, which is also one the largest single mangrove formations of the WIO region. The Zambezi River delta is also part of an EBSA site which extends between Quelimane city

(Bons Sinais River estuary) and Zuni River (wider Zambezi River delta), the latter in Sofala province. The southern areas of Zambezi River delta is also a Ramsar site. Around 33 per cent of Mozambique mangroves occur in the Zambezi River delta. The immediate seas of the delta give rise to the Sofala Bank, which extends from Save River (south of Beira Town) to the chain islands of Ilhas Primeiras and Segundas (north, in Nampula province), the largest and among the most productive fishing areas in Mozambique attaining close to 50 per cent of the entire industrial catches (some 50 000 t in 2002). Fish comprise most of the catch, followed by the shrimp species *Penaeus indicus* and *Metapenaeus monoceros* that use the mangroves as nursery areas.

Other important mangrove areas in Zambezia province include Inhassunge (21 112 ha), Pebane (35 772 ha), Nicoadala (4241 ha) and Namacurra (3998 ha). Sofala province has the second most extensive mangroves of the country, extending from north of the Save River up to the southern limit of the Zambezi River delta. Sofala Bay and the estuaries of Búzi and Púgue rivers are amongst the most important areas.

In southern Mozambique, there are four main mangrove areas where they occur abundantly: (1) the area between Save River and Bazaruto National Park, which is also an EBSA site (Save River to San Sebastian). At the Saver River mouth, mangroves were estimated to cover as much as 14 500 ha; (2) Inhambane and Morrumbene Bay, with around 4000 ha (Amone-Mabuto et al., 2017), being part of the Morrumbene to Zavora EBSAs; (3) the Limpopo River estuary, with some 928 ha (Bandeira and Balidy, 2016) and (4) Maputo Bay, with approximately 18 000 ha (Ferreira and Bandeira, 2014). Maputo Bay is the second most important fishing ground of the country, and it is known that mangroves play a particularly important role in supporting shrimp and fish fisheries. These mangroves are also part of an EBSA site (Incomáti River to Ponta do Ouro).

South Africa

Mangroves in South Africa are patchy, limited to the provinces of Kwazulu-Natal and the Eastern Cape, from Kosi Bay (near the border with Mozambique) down to the Nahoon Estuary and Tyolomnqa Estuary. The total area of mangrove forests is estimated at 1672 ha (Adams and Rajkaran, 2021), with the most important formations occurring in Kosi Bay, Santa Lucia and Mfolozi, Richards Bay, Mngazana River estuary (118 ha) and the Mhlathuze River estuary (the largest formation of the country with ~ 793 ha) (Adams and Rajkaran, 2021). Mangroves at Nahoon River estuary were initially planted in 1969 and more recently, mangroves were planted at Tyolomnqa River estuary which are now the most southern mangroves on the east coast of Africa. In terms of species distribution, six species of mangrove can be found in South African mangrove forests, which are: *A. marina*, widely distributed up to the southernmost limit at Nahoon and Tyolomnqa river estuaries; *B. gymnorhiza* and *R. mucronata*, which can be found at temperate latitudes; *C. tagal*, *L. racemosa* and *X. granatum* with more limited distribution, found only at Kosi Bay.

Madagascar

Madagascar has the second largest mangrove area in the WIO region which accounts for 2 per cent of global coverage, estimated at 205 900–210 000 ha (Fatoyinbo and Simard, 2013; Jones et al., 2015). The west coast supports more than 98 per cent of the national cover, while the remaining 2 per cent can be found on the north-east coast.

The major formations occur in the estuaries of major rivers such as the Mahanjamba forests (the largest of the country), the Ambaro-Ambaja Complex and Ambaro. Table 2 below details over 50 per cent of mangrove area in Madagascar.

Table 2: Major mangrove areas in Madagascar, covering over 50 per cent of the country's mangrove area.

LOCATION	ZONE	AREA (ha)
Ambaro	North	18 490
Tsiribihina	Central West	12 197
Ambaro-Ambanja	North West	25 902
Mahanjamba Bay	North West	26 678
Antsohihy	North West	13 838
Tamboharano	Central west	13 418

Source: Ratsimbazafy et al., 2016.

Seychelles

The Seychelles has an estimated mangrove area of 2500 ha (Appadoo et al., 2016) with mangroves occurring on 17 islands. The largest area is found in the Aldabra atoll, comprising over two thirds (2000 ha) of the total mangrove area of the country (Spalding at al., 2010). *Avicennia marina* is the most common species occurring in those areas, followed by *R. mucronata*. Other species are *C. tagal*, *B. gymnorhiza*, *L. racemosa*, *X. granatum and S. alba* (Appadoo et al., 2016). Silhouette, Praslin and La Digue islands have important areas of mangroves. Faunal diversity is also high, with several reported species of crab, fish and birds.

Comoros Archipelago

The Comoros Archipelago supports some 105 ha of mangroves, dispersed around Moheli, Grande Comoro and Anjouan Islands, with Moheli having the largest assemblages (Appadoo et al., 2016) (Table 3).

Table 3: Mangrove area and species in Comoro Archipelago.

ISLAND	AREA (ha)	LOCATIONS	SPECIES PER ISLAND
Mohéli / Mwali	91	South coast Damou Mapiachingo Bangoi Kouni Chindini	*A. marina* *B. gymnorhiza* *C. tagal* *H. littoralis* *L. racemosa* *R. mucronata* *S. alba*
Anjouan / Nzduani	8	Bimbini, Chissioini	*A. marina* *B. gymnorhiza* *R. mucronata* *S. alba*
Grande Comore / Ngazidja	6	Domoni Hahaya Ouroveni Iconi Moroni Voidjou	*A. marina* *R. mucronata*

Source: Appadoo et al., 2016.

Mauritius

Mauritius has 145 ha of mangroves (Government of Mauritius, 2009), distributed between Mauritius main island (mostly on the east coast) and Rodrigues, the latter with an estimated 24 ha of mangroves forests, thought to have been planted.

Bruguiera gymnorhiza and *R. mucronata* are reported to be the only mangrove species in Mauritius. Juan da Nova Island (France) in Mozambique Channel has 700 ha of mangroves as per the global assessment (Spalding et al., 2010).

Key species associated with the habitat, with identification of nature of association

The WIO region is ranked second area in the world, after South-East Asia, with the most diverse mangroves (Spalding et al., 2010). Nine mangrove species occur in WIO: *A. marina*, *B. gymnorhiza*, *C. tagal*, *H. littoralis*, *L. racemosa*, *R. mucronata*, *S. alba*, *X. granatum* and *X. moluccensis* (Bosire et al., 2016). All nine species can be found in Kenya, Tanzania and Mozambique.

Comoros and Seychelles have seven species, Somalia and South Africa have six, and Mauritius has the lowest diversity of true mangrove species with just two species (Spalding et al., 2010; Mumuli et al., 2010; Lugendo, 2015) (Table 4).

Mangrove associated species in the WIO include the fern *Achrosticum aureum*, palm *Phoenix eclinate* and other woody trees and shrubs such as *Brexia madagascariensis*, *Hibiscus tiliaceus*, *Terminalia catappa*, *Thespesia polpunea*, *Barringtonia racemosa* and *Pemphis acidula*. Among the herbs and succulents are included *Salicornia* sp., *Suaeda monoica*, *Sesuvium portulacastrum* and *Arthcrocnemon* sp.

Table 4: Mangrove area coverage and species assemblages in WIO.

COUNTRY	MANGROVE AREA (ha)	Am	Bg	Ct	Hl	Lr	Rm	Sa	Xg	Xm
Somalia	3000	X	X	X			X	X	X	
Kenya	45 560	X	X	X	X	X	X	X	X	X
Tanzania	80 900	X	X	X	X	X	X	X	X	X
Mozambique	305 400	X	X	X	X	X	X	X	X	X
South Africa	1672	X	X	X		X	X		X	
Madagascar	205 900	X	X	X	X	X	X	X	X	
Seychelles	2500	X	X	X		X	X	X	X	
Mauritius	145		X				X			
Comoros	91	X	X	X	X	X	X	X		

Species name abbreviations: Am *A. marina*, Bg *B. gymnorrhiza*, Ct *C. tagal*, Hl *Heritiera littoralis*, Lr *Lumnitzera racemosa*, Rm *R. mucronata*, Sa *S. alba*, Xg *Xylocarpus granatum* and Xm *X. moluccensis*.

Sources: Mumuli et al., 2010; Spalding et al., 2010; Bosire et al., 2016; GoK, 2017; Rajkaran and Adams, 2016; Adams and Rajkaran, 2021.

Mangrove forest in Malindi, Kenya. © Timothy K.

IMPORTANCE

Ecological importance

The ecological importance of WIO mangroves goes from coastal protection to biodiversity maintenance and mitigation and adaptation to climate changes (Bosire et al., 2016). Mangrove forests sustain extensive fisheries in addition to being directly used, mainly as building, material and firewood.

Mangrove degradation in areas such as Gazi Bay, Vanga (Kenya) and Nhangau (Mozambique) had led to coastal erosion and loss of temperature regulatory services, with severe impacts on the housing of local communities and loss of other infrastructure (Bosire et al., 2016). The value of shoreline protection in Kenya was estimated as USD 1300 ha/year (Kairo et al., 2009). In south-central Mozambique it was also demonstrated that a healthy mangrove can be an effective protection from cyclones and other climatic events, as the mangroves growing along the Save River mouth protected the Nova Mambone village during the Category 4 Eline cyclone in 2000 (Massuanganhe et al., 2015; Macamo et al., 2016a). Mangrove forests in the WIO buffer the coast against tidal and storm surges, and absorb storm water thus protecting the coastal zone. Beira Town in central Mozambique, known as being below sea level, is crossed by the small Chiveve River and, recently its surrounding mangrove riverbed was widened to improve protection of the town against storm water and extreme tides.

Mangroves can store up to 3–5 times the amount of carbon accumulated by other terrestrial vegetation systems, with African mangrove forest carbon storage calculated by Fatoyinbo and Simard (2013) (Table 5).

Table 5: Mangrove area and biomass in WIO countries.

COUNTRY	AREA (ha)	TOTAL BIOMASS (Mg)	MEAN BIOMASS (Mg ha^{-1})
Kenya	19 200	2 294 820	119
Madagascar	205 900	24 856 900	121
Mozambique	305 400	30 974 100	101
Somalia	3000	436 907	143
South Africa	1200	40 018	100
Tanzania	80 900	11 037 800	136

Source: Fatoyinbo and Simard, 2013.

Mangroves sustain tangible livelihoods, especially considerable ecotourism, mostly in Kenya and Tanzania, but also for some island states (Appadoo et al., 2016). Madagascar and Mozambique, with the largest stands of mangroves in the WIO, sustain some of the largest fisheries such as Sofala Bank (central Mozambique) with high fisheries production. Similarly in Maputo Bay, the 18 000 ha of mangroves, and around 4000 ha of seagrasses, contribute to this area being the second most important fishing ground in this country. Island mangroves also support biodiversity, provide shoreline protection and water quality control, among other ecosystem services.

In the WIO region there are around 25 EBSAs and over three-quarters of these have mangrove forests (Table 6). The EBSAs designation strictly follows seven criteria,

Table 6: EBSAs containing mangrove forests in the WIO region.

EBSAs	COUNTRY	MAIN AREAS/OBSERVATION
1. The Great Whirl and Gulf of Aden Upwelling Ecosystem	Somalia	With historical mangroves, now extinct
2. Lamu-Kiunga	Kenya	Northern and southern swamps
3. Watamu area	Kenya	Mida creak
4. Pemba–Shimoni–Kisite	Kenya and Tanzania	Pemba
5. Tanga Coelacanth Marine Park	Tanzania	Tanga
6. Zanzibar (Unguja)-Saadani	Tanzania	Unguja
7. Rufiji–Mafia–Kilwa	Tanzania	Rufiji
8. Northern Mozambique Channel	Tanzania–Mozambique–Comoro–Madagascar– Seychelles (Aldrabra region)	Rovuma, Ibo-Quissanga, Messalo, Pemba Bay
9. Pemba Bay–Mtwara	Tanzania–Mozambique	Ruvuma, Messalo, Ibo–Quissanga, Pemba Bay
10. Baixo Pinda–Pebane (Primeiras & Segundas Islands)	Mozambique	Memba, Nacala, Mussoril, Angoche, Ligonha and Pebane

namely (i) Uniqueness or rarity, (ii) Special importance for life-history stages of species, (iii) Importance for threatened, endangered or declining species and/or habitats, (iv) Vulnerability, fragility, sensitivity, or slow recovery, (v) Biological productivity, (vi) Biological diversity and (vii) Naturalness. Unique or rare mangrove could be small and limited in area, such as those in South Africa and small state Islands (Seychelles, Mauritius, Comoros). Somalian mangroves need more research attention as they are quite unknown, especially those of the northern and central regions, though they nevertheless are believed to sustain activities such as fisheries.

Social and economic importance

In addition to their potential ecotourism potential, mangroves are important sources of livelihood for coastal communities throughout the WIO region (Taylor et al., 2003; Bosire et al., 2016). They provide building material, food, animal fodder, tannins, etc, and are sites for the development of several activities that provide livelihood to the communities (fishing, honey production, salt production among others). For example, it was estimated that mangroves provide 70 per cent of the wood requirements for the communities in coastal areas in Kenya (Bosire et al., 2016). Mangrove pastoralism is quite common in countries like South Africa, also reported for Somalia, where it even represents a threat to forests conservation (Mumuli et al., 2010; Rajkaran and Adams, 2016). Mangroves are also intrinsically related to local community culture. For example, in Somalia the coastal communities from the South Centre obtain frankincense from mangroves while in Mozambique and Kenya mangrove ecosystems are cultural sites where traditional healers perform rituals (Taylor et al., 2003).

Mangroves also contribute in a great manner to local economies through the harvesting and commercialization of its products. For example, the communities from the Zambezi River delta in central Mozambique claim that the commercialization of mangrove products and mangrove related activities are their primary source of livelihood (Machava-António et al., 2020). At national levels, mangroves offer significant economic gains to the economies, by supporting different types of fisheries. In the Seychelles, mangroves also provide fishing grounds, shoreline protection, fishing bait and ecotourism, whilst the bark of *R. mucronata* is traditionally used to polish wooden floors.

The mangrove forests of South Africa provide important social and ecological services to the estuarine communities where they occur. The most common uses are wood extraction for timber, building material, fish traps and fuelwood. Other uses include recreational fishing, and tourism (bird-watching, boat trips, etc). Mangroves are also sites of high biodiversity of fauna including birds, hippos, crocodiles, and terrestrial visitors such as the blue duiker, bush pig and others. However, overexploitation, unsustainable use and coastal development have led to mangrove loss and degradation, particularly in those forests outside protected areas such as Durban Bay where harbour and industrial development caused the loss of 440 ha (Adams and Rajkaran, 2021). Impacts on mangroves include livestock browsing, agriculture, pollution, sand mining and also climate change (Rajkaran and Adams, 2016). Regardless of these losses, mangrove area seems to be increasing in some areas, particularly in remote areas and those within protected areas, such as in northern Kwazulu-Natal province (Spalding et al., 2010).

THREATS

Status

Kenya mangroves are reported as being impacted by natural and anthropogenic causes. Between 1985 and 2009, the country lost about 40 per cent of its mangrove cover, about 450 ha of mangrove area loss per year (Table 7). Mangrove loss tends to be more pronounced in urban settings compared to rural areas, as reported in other countries too, eg Mozambique. Kenya mangroves are also threatened by climatic issues such as heavy rains and associated sediment deposition as Kitheka et al. (2002) have reported, with widespread mangrove dieback. Loss and degradation of mangrove habitat have been identified as mainly being caused directly and indirectly by population pressure, overexploitation as well, poverty and inequality. KMFRI data indicate over 80 per cent loss of mangroves around Mombasa County.

The historical degradation of mangroves in Somalia is yet to be assessed, and the accuracy of mangrove mapping is needed to ascertain historical mangrove loss in other countries. For example, the 6 per cent, 4 per cent and 8 per cent loss reported for Tanzania, Madagascar and Comoros, respectively (Wang et al., 2003; Lugendo, 2015). Similar precision estimates may be necessary for some of the mangrove stands of Mozambique. Somalia is reported as having historical mangroves on its northern coast (Spalding et al., 2010). Fewer mangrove stands exists on the Somalia south coast, near the border with Kenya and may be already threatened. Therefore some form of protection and reinforcement might be needed.

Tanzania mangroves have been impacted due to harvesting and conversion of mangrove forests into rice farming in Rufiji River delta; construction of salt plans have impacted mangroves in Tanga, Bagamoyo and Mtwara. Climatic impacts such as flooding and related sedimentation appears to be also important issues driving mangrove degradation in Tanzania (Mangora et al., 2016).

Table 7: Kenya mangrove forest area and proportion of degradation.

COUNTY	MANGROVE AREA (ha)	DEGRADED MANGROVE (ha)	% DEGRADED AREA
Lamu	37 350	14 407	38.6
Tana River	3260	1180	36.2
Kilifi	8536	3422	40.0
Mombasa	3771	1850	49.1
Kwale	8354	3725	44.6
TOTAL (ha)	61 271	24 585	40.1

Anthropogenic impacts in Mozambique are related to deforestation for firewood, for shrimp aquaculture, historical salt pans production and to a small extent to a port development in Nacala Bay (Macamo et al., 2016b).

Extreme event impacts are those related with cyclones and floods. Within the WIO region, Madagascar and Mozambique experience yearly several cyclones, tropical depressions and resulting excessive rains, floods and sedimentation or erosion that impact both mangroves and seagrass beds. Recent studies in Save River estuary (central Mozambique) documented extensive mangrove die back due to mainly the 2000 Cyclone Eline that brought wind and sedimentation as well as a prolonged flooding and drowning of mangrove stands (Massuanganhe et al., 2015; Macamo et al., 2016a). Limpopo River in southern Mozambique is another site with mangrove destruction caused from prolonged flooding up to 45 days, accompanied by extensive sedimentation, impacting at least half of the original mangrove stand (Bandeira and Balidy, 2016).

In Mozambique, the condition or status of mangroves were analyzed for 11 sites in all regions. Table 8 presents the national status showing areas where mangrove stands have been impacted but also areas that are expanding. The results show area increase in several locations, but also areas where significant losses are notable. In the province of Cabo Delgado, for example, there is a general tendency for mangrove areas to increase (Ferreira et al., 2009), but there are specific places within this province, such as Olumbi, where there is a great reduction of area and a general degradation of the condition of mangrove stands (Macamo et al., 2018). In central Mozambique, remote areas such as the Zambezi River delta experienced an increase of about 10 per cent between 1994 and 2003 (Shapiro et al., 2015), while the urban area on the Chiveve River (Beira city) suffered a loss of about two-thirds of its

Table 8: Mangrove status, trend and impacts in Mozambique.

LOCATIONS	REGIONS OF MOZAMBIQUE	PERIOD OF THE STUDY	AREA (km^2)		% OF CHANGE	MAIN IMPACTS	SOURCE
			T_0	T			
Cabo Delgado	North	1995–2005	325	369	+13.5	Local uses such as fuel and construction	Ferreira et al., 2009
Olumbi	North	1991–2013	7.24	5.56	-25.4	Local uses such as fuel and for construction, shellfish harvest and footpaths	Macamo et al., 2018
Pemba Bay	North	1991–2013	21.43	31.30	+23.1	Aquaculture, salt pans, timber harvest for local use	Macamo et al., 2018
Quirimbas National Park	North	1991–2013	112.44	123.48	+9.8	Timber harvest for local use, natural sedimentation	Nicolau et al., 2017
Nacala Bay	North	2013–2016	0.365	0.276	-24.9	Clearance for construction of a new port	Salomão Bandeira, unpubl.
Zambezi Delta	Center	1994–2013	333.11	370.34	+10.1	Erosion, natural causes	Shapiro et al., 2015
Chiveve Canal	Center	2016–2017	0.23	0.1	-43.5	Restoration of canal	Uacane and Ombe, 2016; obs. Salomão Bandeira, unpubl.
Save mangroves	South	1999–2014	147.44	106.66	-27.7	Cyclone and floods during 2000	Macamo et al., 2016a
Limpopo Estuary	South	1999–2001	9.28	3.82	-58.8	Floods during 2000	Bandeira and Balidy, 2016
Incomati Estuary	South	1991–2003	42.31	44.51	+5.1	Local uses	Macamo et al., 2015
Maputo Bay	South			175.96		Urban expansion and local uses	Paula et al., 2014

area for rehabilitation of the canal to increase its flood regulation capacity (Machava-António et al., 2020). There are other areas throughout the country where there are reports of non-sustainable exploitation of mangrove wood. Some of these locations are the south of the city of Quelimane in the province of Zambézia and the south of the city of Beira in the province of Sofala.

In South Africa, overexploitation, unsustainable use and coastal development are leading to mangrove loss and degradation, particularly in those forests outside protected areas (Table 9). Examples can be seen in Durban Bay where harbour and industrial development caused the loss of 440 ha (Adams and Rajkaran, 2021); and in other minor estuaries where mangroves were completely lost due to overexploitation and estuarine closure (Rajkaran et al., 2009).

Mangroves in South Africa have been lost from ten temporarily closed estuaries, these represented small areas of mangroves, with losses due to mouth closure and increasing water levels. In KwaZulu-Natal agriculture (sugar cane), river mouth closure, and poorly placed transport infrastructure are the main drivers of mangrove loss in small systems. In the Eastern Cape, unsustainable harvesting, cattle browsing and changes in river mouth conditions are the main threats. Currently and into the future St Lucia/Mfolozi area may experience mangrove losses due to freshwater inflow and siltation (Adams and Rajkaran, 2021). Other major human induced impacts on mangroves in South Africa include livestock browsing, agriculture, pollution, sand mining and water abstraction, while climate change is the most prominent natural threat (Rajkaran and Adams, 2016).

Historical records indicated that Madagascar has lost more than 20 per cent of its mangrove forest between 1990 and 2010 (Jones et al., 2016), however some areas experienced a slight increase due to accretion (eg the Mangolovo region increased its area from 981 ha to 1172 ha between 2000 and 2010; and Sohany increased from 1984 ha to 2025 ha within the same period) (Table 10). The main causes of mangrove loss include urban expansion, agriculture, aquaculture, wood harvesting and siltation. These causes are a result of human population growth. Natural causes such as cyclones (in areas such

Table 9: Mangrove status and trend in South Africa's main forests (> 10 ha).

SITE	AREA (ha)	OVERALL CONDITION	TREND	MAIN THREATS
Durban Bay	13.4	–	Increasing	Pollution, coastal development
Mgeni	26.8	Good *	Decreasing	Pollution, sand mining Water abstraction, coastal development
Mlalazi	60.7	Good *	Increasing	River mouth closure
Mhlathuze	793	Good *	Increasing	Wood harvesting, sand mining, water abstraction, coastal development
Richards bay	171	Good *		Pollution, coastal development
Mfolozi	60.7			Agriculture, field fires
St lucia	287.7	Good *		Sand mining, water abstraction
Kosi bay	71	Good *		Wood harvesting
Nqabara	11.8			Trampling, browsing, wood harvesting, agriculture, bush encroachment, pollution, field fires
Xora	25			
Mtata	31			
Mtakatye	10			
Mngazana	118			
Mntafufu	12.4			

* Protected in conservation areas.
Sources: Adams and Rajkaran, 2021; Rajkaran and Adams, 2016; Rajkaran, 2011; Spalding et al., 2010.

as Tsiribihina and Mangoky) and decrease in precipitation (eg in the protected area of Ambodivahibe) have also been documented (Ratsimbazafy et al., 2016).

The status of mangroves in the small island states of the WIO is provided in Table 11. In general, mangrove areas are decreasing, except for those in protected areas where their coverage appears stable. Mangroves in the Seychelles are usually well conserved, and overall coverage is stable (Lugendo, 2015); however, reports indicate threats from land reclamation, pollution by solid waste, rubbish and chemicals, especially at Mahé Island (Appadoo et al., 2016). Meanwhile, in Mauritius, mangroves might be threatened by clearing for coastal development; past threats were sugar cane pollution via effluents. Comoros mangroves are threatened by deforestation and water abstraction of freshwaters upstream as well as sand deposition and wastewater pollution (Appadoo et al., 2016).

Gillnets at a creek of the Rovuma Delta, North Mozambique (Cabo Delgado province). © José Paula

Table 10: Mangrove trend comparison between 1990 and 2010 in Madagascar.

SITE	AREA (ha)	TREND	MAIN THREATS
Mahanjamba Bay	26 677	Stable	Conversion to agriculture, logging, conversion to aquaculture, urban development, cyclone impacts
Ambaro-Ambaja Bays	25 664	Decreasing	
Tsirihibina and Manambolo Deltas	20 242	Decreasing	
Antsohihy	13 838	Decreasing	
Tambohorano	13 418	Decreasing	
Sahamalaza	10 956	Decreasing[1]	Rice farming
Mahavavy su Sud	10 654	Stable	
Mahajanga	9574	Decreasing	
Mangoky	9431	Decreasing	Cyclone impacts
Morondava-Bosy	6213	Decreasing	Grazing and browsing, wood harvesting
Kamendriky-Tsilambana	5924	Decreasing	
Mahabo-Andramy	5905	Increasing	
Maintirano	5900	Decreasing	
Boeny	3867	Stable	
Baly-Soalala	3507	Decreasing[2]	
Besalampy	3287	Decreasing	
Rigny-Irody	3231	Stable	
Morombe	3035	Decreasing	
Mariarano	2330	Decreasing	
Narinda	2036	Decreasing	
Sohany	2025	Decreasing	
Belo sur Mer	1917	Decreasing[3]	
Vilamatsa	1847	Stable	
Kabatomena	1529	Decreasing	
Reharaka	1406	Decreasing	
Assassins	1360	Decreasing	
Manampatra	1327	Decreasing	
Morovasa	1199	Decreasing	
Mangolovo	1172	Decreasing	
Ambondrombe	1109	Decreasing	

1. Biosphere Reserve of Sahamalaza
2. National Park of Baly Bay
3. National Park of Kirindi Mitea

Table 11: Mangrove status and trend in Small Island States.

SITE		AREA (ha)	TREND	MAIN THREATS
Comoros	Moheli	91	Decreasing	Wood harvesting, water abstraction for agriculture, sand deposition, wastewater pollution, water abstraction, infrastructure development, urbanization, sand and coral extraction
	Anjouan	8		
	Grande Comoro	6		
Mauritius	Main Island	121	Increasing from 1980 till 2009, then stable or decreasing	Wetland destruction, coastal development, pollution
	Rodrigues	24 (planted)		
Seychelles	Mahé	100	Decreasing	Land reclamation, pollution, dredging, agriculture runoff, water abstraction (hotels, tourism), wastewater discharge
	Cousin	0.8		
	Aldraba group *	2000		
	Port Glaud and Port Launay *	20		
	Anse intendance	13		
	Anse a la Mouche	10		

* Protected in conservation areas.
Sources: Appadoo et al., 2016; Spalding et al., 2010.

Climatic influences

Central Mozambique was badly hit by Category 4 Cyclone Idai which made landfall in Beira port city on May 14, 2019 (Fig. 2), and was the deadliest tropical cyclone recorded in the South-West Indian Ocean and the second deadliest in the Southern Hemisphere. It affected 3 million people (at least 1300 dead) and infrastructure damages amount to more than USD 2 billion.

A preliminary aerial survey was conducted a month after the cyclone. Preliminary results of this assessment show that mangroves were severely impacted by the cyclone. Near Beira, mangroves grow in the south around Púnguè and Buzi river estuaries, but also extend further south to Buzi village and to Govuro near Bazaruto Archipelago; and north of Nhangau (north of Beira city) and beyond to near Zambezi River estuary region.

These mangroves suffered from storm surges, wind action and inundation, causing mangrove death, mainly from drowning. However, the impacts were distinct. South of Beira, a rather sparsely populated area, the mangroves grow extensively, with minimal human impact. Tree defoliation appears to be the main impact, although many trees also fell due to wave and wind action brought by the cyclone. Damage was extensive, giving the forest a greyish appearance, with the tallest trees more impacted.

North of Beira, in more populated areas, mangroves are intensively exploited for wood and the forests appeared mostly shrubby and generally suffered less from the cyclones, with exception of seaward mangroves and the few tall remaining mangrove stands. Shortly after the cyclone, mangrove logging intensified as people needed wood to rebuild their houses. Mangrove logging (including for charcoal production) was also seen as a means to increase family income in the post-cyclone recovery period.

In the south of Mozambique, the highest losses were recorded in the mangroves of Save River and Limpopo River estuary, associated with Cyclone Eline and 45 days of floods during 2000, respectively (Bandeira and Balidy, 2016; Macamo et al., 2016a). Extreme events such as floods are regularly reported in the coastal towns of Pemba, Nacala, Quelimane, Beira, Inhambane and Maxixe, and Maputo city.

It is important to note that area estimates alone may not indicate the ecological condition of the forest. For example, it was estimated that for the Save River mangroves Cyclone Eline caused the loss of about 28 per cent of the habitat, but the normalized difference vegetation index (NDVI) estimate shows that up to 50 per cent of the entire forest area was impacted though without total destruction (Macamo et al., 2016a). It is recommended that mapping studies be accompanied by assessments of forest structure and condition on the ground.

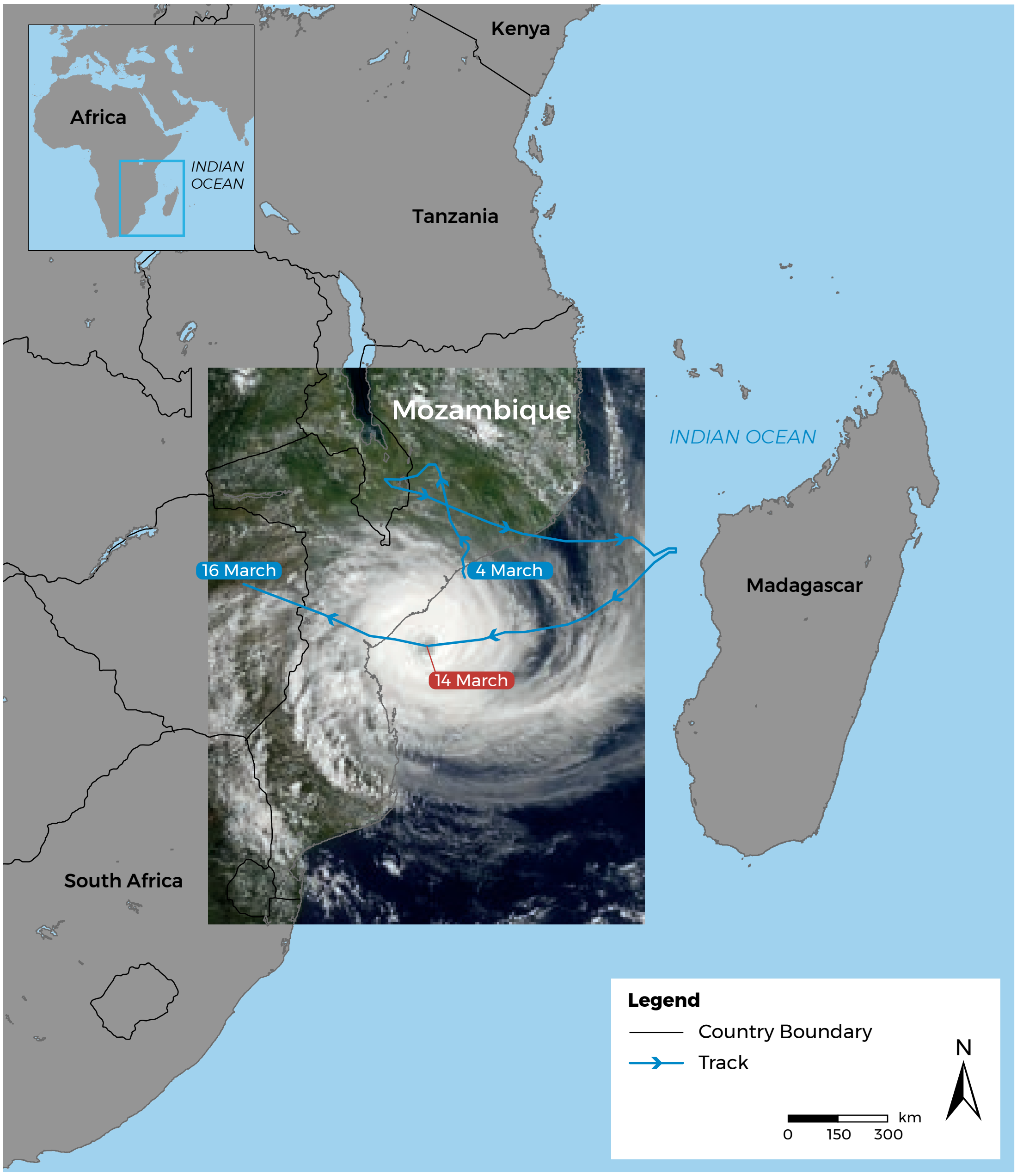


Figure 2: Intense Tropical Cyclone Idai approaching the Sofala province of Mozambique on 14 March 2019, shortly after reaching its peak intensity.

Adapted from https://pt.m.wikipedia.org/wiki/Ficheiro:Idai_2019-03-14_1135Z.jpg (MODIS image captured by NASA's Aqua satellite) and https://reliefweb.int/map/mozambique/tropical-cyclone-idai-impact-overview-emergency-response-coordination-centre-ercc-dg (trekking information).

Cyclone Kenneth resulted in significant damage to the Comoros, Mozambique and Tanzania. Fifty-two people died due to this cyclone. Kenneth Category 4 Cyclone was the most intense land falling tropical cyclone (on April 25, 2019) in the recorded history of Mozambique. Confirmed impacts on mangroves were reported around Ibo, northern Mozambique, however a detailed assessment on mangrove impacts of this cyclone in Comoros, Mozambique and Tanzania are yet to be undertaken. As sea level rises, mangroves will retreat or 'creep' inland depending on the topography and available space. Mangrove expansion into new inland areas will therefore depend on whether the inshore space is already occupied by human habitation or infrastructure farming or other land use. Thus, there is a need to evaluate the potential for landward mangrove creep for all major existing mangrove sites and to integrate the future space requirements into urban and municipal development plans.

EXISTING PROTECTION LEVEL

Mangrove protection level is non-existent in Somalia.

Kenya mangrove enjoys protection within Kiunga Marine National Reserve and Watamu Marine National Park, both UNESCO Man and the Biosphere (MAB) Programme sites, Mombasa National Park, Mombasa National Reserve and Gazi Mokoko Pamoja community area (Spalding et al., 2010).

In Tanzania (including Zanzibar), from north to south, mangroves area protected under national forestry legislation, as well as within 12 protected areas:

- Forest Reserve # 10
- Forest Reserve # 11
- Coelacanth Marine Park
- Pemba Channel Conservation Area
- Saadani National Park
- Chumbe Island Coral Park (CHICOP) Marine Sanctuary and Forest Reserve
- Chwaka Bay-Jozani Forest Conservation Area
- Menai Bay Conservation Area
- Fungu Yasini Marine Reserve;
- Mbudya Marine Reserve
- Dar es Salaam Marine Reserve
- Mafia Island Marine Park
- Mnazi Bay-Ruvuma Estuary Marine Park

Mozambique possesses the following protected areas with mangroves:

- Quirimbas National Park (also UNESCO MAB, declared in 2018)
- Area of Environmental Protection of the Archipelago of Primeiras and Segundas Islands
- Marromeu Reserve (Ramsar Site)
- Bazaruto Archipelago National Park
- Pomene Reserve
- Ponto do Douro Partial Marine Reserve

Details in terms of area coverage for each protected area, species, importance and threats are presented in Table 12.

South Africa mangrove protected areas:

- Kosi Bay (part of iSimangaliso Wetland Park)
- Great Saint Lucia Wetland Marine Park (World Heritage Site)
- Umlalazi Nature Reserve
- Beachwood Mangroves Nature Reserve
- Umtamvuma Nature Reserve
- Mkambati Wildlife Reserve
- Hluleka Marine Sanctuary
- Dwesa-Cwebe Nature Reserve

Madagascar conservation areas with mangroves forests:

- Baie de Baly National Park
- Kirindy Mitea National Park

Table 12: Mangroves and conservation areas in Mozambique.

SITE	AREA (ha)	SPECIES DIVERSITY	USES AND THREATS
Bazaruto Archipelago National Park	64.6	5	Sustainable local use is allowed, threats include illegal cutting, extreme events
Area of Environmental Protection of the Archipelago of Primeiras and Segundas Islands	56 994	6	Illegal cutting
Marromeu National Reserve	14 740	5	Mangroves appear intact, possibly associated with their remoteness
Pomene Reserve	157	7	Forests are quite intact; Cyclone Eline (in 2000) impacts now recovered; possible current human use threat
Ponto do Ouro Partial Marine Reserve (including Maputo Estuary Reserve)	4516	5	Mangroves at Inhaca and Matutuíne region are in good condition
Quirimbas National Park	12 348	6	Main threat is local use (for house and boat construction), however use is tolerated/ allowed in specific areas.
Area of Total Protection of San Sebastião	2550	5	Mangroves fairly pristine

Sources: Balidy et al., 2005; DNAC, 2011; ANAC, 2016a; 2016b; Nicolau et al., 2017.

Seychelles enjoys considerable mangrove protection, especially in the Aldabra region (1552 ha), however a degree of natural impacts is nevertheless experienced (Constance et al., 2021). Mahé island mangroves appears in good condition despite past report of some impacts. The planted mangrove forests in Rodrigues Island (Mauritius) appear protected (Perry and Berkeley 2009).

PRIORITY AREAS FOR CONSERVATION

In Somalia, the priority for intervention in mangrove conservation is to undertake detailed evaluation of mangrove coverage and status across the country. If possible, to include a historical evaluation of mangrove status. Depending on the findings, Somalia mangroves may require approaches for restoration and management especially given reports of past intensive deforestation.

In Kenya, a vibrant mangrove management action plan was recently approved and, having also a Mikoko Pamoja program and a research output portfolio for this region (Gazi Bay) has led to Kenya becoming recognized as an African leader on this activity, as well as a global model of mangrove conservation and restoration; setting the stage for replication of these successes elsewhere. Despite the progress, Kenya nevertheless lists as priority the need to sustain wider community participation and ecosystem-based management of mangroves.

Mangrove poles for construction at Gazi Bay, south Kenya.
© José Paula

Reviving a management action plan comes as a priority for Tanzania, as stated by Mangora et al. (2016). Such an action plan would need to update the country's mangrove area, strengthen reinforcement, involve a wide community's participation in mangrove best practices and develop and expand mangrove rehabilitation. Tanzania, as with Mozambique, has highlighted the need to establish permanent mangrove areas set aside for applied research.

Mozambique has recently (in 2020) approved a mangrove management action plan. And there are already a series of ongoing restoration activities targeted to address the Sustainable Development Goal (SDG) 14.2. Priority is to be given to appropriate techniques for restoration, such as in high tide amplitude abandoned shrimp ponds (such as around Zambezi River delta), integration of community-based approaches in mangrove management and restoration, and integration of blue carbon focused approaches in management initiatives as part of attempts to address both anthropogenic and natural or climate related impacts. To align with the nationally determined contributions (NDCs) for Mozambique, as integrated in a climate adaptation portfolio, issues of reinforcement and wider environmental awareness and education about mangroves are very relevant. Specifically, as listed in Table 13, a series of priority activities can be implemented in specific locations in Mozambique that also serve as examples for other countries in the WIO region.

In South Africa, mangrove forests at Kosi Bay, uMhlathuze River estuary and Mngazana River estuary should be prioritized for protection. Kosi Bay is the only site with all six mangrove species as well as mangrove associates. uMhlathuze represents the largest area in the country and mangroves are continuing to increase in that estuary. Mngazana River estuary supports the largest forest in the Eastern Cape and is one of the most important estuaries in that province.

An important priority for Madagascar is to establish a mangrove regulatory and management framework for mangrove management and conservation. Madagascar highlights the need to control current, but also future, anthropogenic and natural pressures on mangroves, as well as the need to implement community-based mangrove management, ecosystem-based management and development of technical capacities in mangrove research and management. Madagascar has also committed to incorporating more mangrove forest in the network of MPAs (Ratsimbazafy et al., 2016).

Table 13: Priority activity fort selected mangrove locations in Mozambique.

AREA	PROVINCE	SUGGESTED TYPE OF INTERVENTION
Palma (Quionga)	Cabo Delgado	Mapping of mangrove condition
Olumbi	Cabo Delgado	Restoration, sensitization and environmental education, and promotion of sustainable practices
Mecufi	Cabo Delgado	Restoration of abandoned salt pans (as initiated a few years ago)
Angoche	Nampula	Options for possible restoration, sensitization and environmental education, and promotion of sustainable practices
Moma	Nampula	Restoration, sensitization and environmental education, and promotion of sustainable practices
Quelimane	Zambezia	Mangrove mapping and studies of its condition; continuation of restoration activities, sensitization and environmental education
Inhassungue	Zambezia	Restoration of abandoned aquaculture ponds
Micaune	Zambezia	Mangrove condition and mapping studies
Nhangau	Sofala	Sensitization and promotion of good practices
Buzi and Pungue river estuaries		Mapping and evaluation of forest condition
Inhambane Bay	Inhambane	Sensitization and environmental education, and promotion of best practices
Xai Xai district (Limpopo River estuary)	Gaza	Continuous environmental sensitization and mangrove restoration
North of Maputo city	Maputo	Environmental sensitization
Matola	Maputo	Restoration of abandoned salt pans

RECOMMENDATIONS

National agendas on mangroves have to be re-visited so that they are mainstreamed with global platforms such as the main targets of the SDGs. Mangrove area mapping, improved understanding of their socio-economic importance and sustainable usage and research on impacts are also to be pursued for many countries of the WIO. WIO country assessments of knowledge gaps on management and the integration of the wider society, both at local and country level, will help improve in steering the discussion on tackling the wider mangrove management challenges. Regional networks such as the WIO Mangrove Network and other platforms with existing experience and approaches on mangrove conservation and management, such as "Save our mangroves now" form an appropriate avenue for wider discussion on mangrove management and conservation.

Given the wider deforestation history in many WIO countries, the WIO region needs better to strategize its implementation of mangrove restoration strategies together with appropriate social adaptation strategies. The region would need to strategize towards mangrove management following existing assessment and models of vulnerability related to climate events and sea level rise (eg Cinco-Castro and Herrera-Silveira, 2020; Charrua et al., 2020).

In Somalia, basic studies are needed on mangrove condition, occurrence and utilization. This can be linked with the design of mangrove management plans specific for regions in the country. For Tanzania, more is needed on updating information on mangrove distribution and to restart the previously approved mangrove management plan. The focus for Mozambique is to further document the impact of climate change on mangrove forests, especially associated to recent cyclones and to undertake implementation of a recently approved mangrove management plan. In South Africa, priority for protection is suggested for mangrove forests at Kosi Bay, uMhlathuze and Mngazana river estuaries. Madagascar might find itself prioritizing means to achieve a reduction of intense mangrove deforestation as reported (Ratsimbazafy et al., 2016). An update of mangrove mapping is needed in Mauritius to strengthen monitoring of the few mangrove stands in several sites on the Island as well as of the planted mangroves on Rodrigues Island.

REFERENCES

Adams, J. and Rajkaran, A. (2021). Changes in mangroves at their southernmost African distribution limit. *Estuar. Coast. Shelf Sci*. 247, p. 107158. https://doi.org/10.1016/j.ecss.2020.106862

Amone-Mabuto, M., Bandeira, S. and da Silva, A. (2017). Long term changes in seagrass coverage and potential links to climate-related factors: the case of Inhambane Bay, southern Mozambique. *West. Indian Ocean J. Mar. Sci.* 16(2), pp. 13–25

ANAC (2016a). Plano de maneio da Reserva Nacional do Pomene. Volume I. Plano de maneio. República de Moçambique, Ministério da terra, ambiente e desenvolvimento rural, República de Moçambique, Ministério da terra, ambiente e desenvolvimento rural, 150 pp.

ANAC (2016b). Plano de maneio da Reserva de Marromeu 2016–2025. 150 pp

Appadoo, C., Rabi, F.A., Antha, S., Bastienne, L.J., Hurbungs, M., Mougal, J., Rogoonaden, R., Soogun, N. and Vel, T. (2016). Mangroves of small Islands. In *Mangroves of the Western Indian Ocean: Status and Management* (eds. Bosire, J.O., Mangora, M.M., Bandeira, S., Rajkaran, A., Ratsimbazafy, R., Appadoo, C. and Kairo, J.G.). WIOMSA, Zanzibar Town, pp. 115–135

Balidy, H.J., Sitoe, A., Menomussanga, M. and Pires, P.L. (2005). Avaliação dos níveis de corte, composição específica e regeneração natural de mangal no Sul de Moçambique. CDS-ZC. Xai-xai. 20 pp

Bandeira, S. and Balidy, H. (2016). Limpopo estuary mangrove transformation, rehabilitation and management. In *Estuaries: a Lifeline of Ecosystem Services in the Western Indian Ocean* (eds. Scheren, P., Salif, D. and Machiwa, J.), chapter 14, pp. 227–238. Springer

Beentje, H. and Bandeira, S. (2007). *A Field Guide to the Mangrove Trees of Africa and Madagascar*. Royal Botanic Gardens, Kew. 91 p. ISBN 978-1-84246-135-8

Bosire, J.O., Mangora, M.M., Bandeira, S., Rajkaran, A., Ratsimbazafy, R., Appadoo, C. and Kairo, J.G. (eds.) (2016). *Mangroves of the Western Indian Ocean: Status and Management*. WIOMSA, Zanzibar Town, 156 pp

Carbone, F. and Accordi, G. (2000). The Indian Ocean Coast of Somalia. *Mar. Pollut. Bull.* 41, pp. 141–159

Charrua, A.B., Bandeira, S.O., Catarino, S., Cabral, P. and Romeiras, M.M. (2020). Assessment of the vulnerability of coastal mangrove ecosystems in Mozambique. *Ocean Coast. Manag*. 189, p. 105145. https://doi.org/10.1016/j.ocecoaman.2020.105145

Cinco-Castro, S. and Herrera-Silveira, J. (2020). Vulnerability of mangrove ecosystems to climate change effects: The case of the Yucatan Peninsula. *Ocean Coast. Manag*. 192(1), p. 105196. https://doi.org/10.1016/j.ocecoaman.2020.105196

Constance, A., Haverkamp, P.J., Bunbury, N. and Schaepman-Strub, G. (2021). Extent change of protected mangrove forest and its relation to wave power exposure on Aldabra Atoll. *Glob. Ecol. Conserv*. 27, e01564, https://doi.org/10.1016/j.gecco.2021.e01564

DNAC (2011). Plano de maneio da Reserva Marinha Parcial da Ponta do Ouro. Ministério do Turismo, Direcção Nacional das Áreas de Conservação (DNAC)

Dürr, H.H., Laruelle, G.G., van Kempen, C.M., Slomp, C.P., Meybeck, M. and Middelkoop, H. (2011). Worldwide typology of nearshore coastal systems: defining the estuarine filter of river inputs to the oceans. *Estuaries Coast* 34(3), pp. 441–458

Fatoyinbo, T. and Simard, M. (2013) Height and biomass of mangroves in Africa from ICESat/GLAS and SRTM. *Int. J. Remote Sens.* 34(2), pp. 668–681

Ferreira, M.A., Andrade, F., Bandeira, S.O., Cardoso, P., Nogueira-Mendes, R. and Paula, J. (2009). Analysis of cover change (1995-2005) of Tanzania/Mozambique trans-boundary mangroves using Landsat Imagery. *Aquat. Conserv. Mar. Freshw. Ecosyst.* 19, pp. S38–S45

Ferreira, M.A. and Bandeira, S. (2014). Maputo Bay's coastal habitats. In *The Maputo Bay Ecosystem* (eds. Bandeira, S. and Paula, J.). WIOMSA, Zanzibar Town, pp. 21–24

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

Gok (2017). National Mangrove Ecosystem Management Plan. Kenya Forest Service, Nairobi, Kenya. 102 pp

Government of Mauritius (2009). Environmentally sensitive areas report. Ministry of and NDU. Republic of Mauritius. 198 pp.

Jones, T.G., Glass, L., Gandhi, S., Ravaoarinorotsihoarana, L., Carro, A., Benson, L., Ratsimba, H.R., Giri, C., Randriamanatena, D. and Cripps, G. (2016). Madagascar's mangroves: quantifying nation-wide and ecosystem specific dynamics, and detailed contemporary mapping of distinct ecosystems. *Int. J. Remote Sens.* 8(2). https://doi.org/10.3390/rs8020106

Jones, T.G., Ratsimba, H.R., Ravaoarinorotsihoarana, L., Glass, L., Benson, L., Teoh, M., Carro, A., Cripps, G., Giri, C., Gandhi, S., Andriamahenina, Z., Rakotomanana, R. and Roy, P-F. (2015). The Dynamics, Ecological Variability and Estimated Carbon Stocks of Mangroves in Mahajamba Bay, Madagascar. *J. Mar. Sci. Eng*. 3(3), pp. 793–820. https://doi.org/10.3390/jmse3030793

Kairo. J., Wanjiru., C. and Ochiewo., J. (2009). Net Pay: Economic analysis of a Replanted Mangrove Plantation in Kenya. J. Sustain. For. 28(3), pp. 395–414

Kitheka, U.J., Ongwenyi, S.G. and Mavuti, M.K. (2002). Dynamics of suspended sediment exchange and transport in a degraded mangrove creek in Kenya. *Ambio* 31, pp. 580–587

Kruitwagen, G., Pratap, H.B., Covaci, A. and Wendelaar Bonga, S.E. (2008). Status of pollution in mangrove ecosystems along the coast of Tanzania. *Mar. Pollut. Bull.* 56, pp. 1022–1042

Leskinen, J., Pohjonen, V.M. and Mbarouk, S.A. (1997). Woody Biomass Inventory of Zanzibar Islands. Zanzibar Forestry Development Project Technical Paper No. 40. Commission for Natural Resources, Zanzibar and Forest and Park Services, Helsink. 103 pp

Lugendo, B. (2015). Mangroves, salt marshes and seagrass beds. In *The Regional State of the Coast report: western Indian Ocean*. UNEP and WIOMSA, Nairobi Convention, pp. 49–64. Nairobi, Kenya

Macamo, C., Bandeira, S., Muando, S., Abreu, D. and Mabilana, H. (2016b). *Mangroves of Mozambique. In Mangroves of the Western Indian Ocean: Status and Management* (eds. Bosire, J.O., Mangora, M.M., Bandeira, S., Rajkaran, A., Ratsimbazafy, R., Appadoo, C. and Kairo, J.G.). WIOMSA, Zanzibar Town, pp. 51–73

Macamo, C.C., Balidy, H., Bandeira, S.O. and Kairo, J.G. (2015). Mangrove transformation in the Incomati Estuary, Maputo Bay, Mozambique. WIO J. Mar. Sci. 14(1&2) pp. 11–22

Macamo, C.C.F., Adams, J.B., Bandeira, S.O., Mabilana, H.A. and Machava António, V. (2018). Spatial Dynamics and Structure of Human Disturbed Mangrove Forests in Contrasting Coastal Communities in Eastern Africa. *Wetlands* 38, pp. 509-523. https://doi.org/10.1007/s13157-018-0996-7

Macamo, C.C.F., Massuanganhe, E., Nicolau, D.K., Bandeira, S.O. and Adams, J.B. (2016a). Mangrove's response to cyclone Eline (2000): what's happening 14 years. *Aquat. Bot.* 134, pp. 10–17

Machava-António, V., Bandeira, S.O., Macamo, C. and Mahanzule, R. (2020). Value Chain Analysis of Mangrove Forests in Central Mozambique: Uses, Stakeholders and Income. *West. Indian Ocean J. Mar. Sci.* 19(1), pp. 1–17

Maina, J., Bosire, J.O., Kairo, J.G., Bandeira, S.O., Mangora, M.M., Macamo, C., Ralison, H. and Majambo, G. (2021). Identifying global and local drivers of change in mangrove cover and the implications for management. *Glob. Ecol. Biogeogr*. 30(10), pp. 2057–2069

Mangora, M. (2011). Poverty and Institutional Management stand-off: a restoration and conservation dilemma for mangrove forests of Tanzania. *Wetl. Ecol. Manag*. 19, pp. 533–543. https://doi.org/10.1007/s11273-011-9234-2

Mangora, M.M., Lugendo, B.R., Shalli, M.S. and Semesi, S. (2016). Mangroves of Tanzania. In *Mangroves of the Western Indian Ocean: Status and Management* (eds. Bosire, J.O., Mangora, M.M., Bandeira, S., Rajkaran, A., Ratsimbazafy, R., Appadoo, C. and Kairo, J.G.). WIOMSA, Zanzibar Town, pp. 31–49

Massuanganhe, E.A., Macamo, C., Westerberg, L-O., Bandeira, S., Mavume, A. and Ribeiro, E. (2015). Deltaic coasts under climate-relate catastrophic events - insights from the Save River delta, Mozambique. *Ocean Coast. Manag.* 116, pp. 331–340

Mremi, S.D. and Machiwa, J.F. (2003). Heavy metal contamination of mangrove sediments and the associated biota in Dar es Salaam, Tanzania. *Tanz. J. Sci.* 29(1), pp. 61–75

Mumuli, S.O., Alim, M. and Oduori, G. (2010). *Monitoring of Mangroves in Somalia (Puntland, Somaliland and South Central Somalia).* FAO-SWALIM, Project Report No. L-19. Nairobi, Kenya

Nicolau, D., Macamo, C.C., Mabilana, H.A., Taju, A. and Bandeira, S.O. (2017) Mangrove change detection, structure and condition in a protected area of eastern Africa: the case of Quirimbas National Park, Mozambique. *West. Indian Ocean J. Mar. Sci.* 16, pp. 47–60

Paula, J., Macamo, C. and Bandeira, S. (2014). Mangroves of Maputo Bay. In *The Maputo Bay Ecosystem* (eds. Bandeira, S. and Paula, J.). WIOMSA, Zanzibar Town, pp. 109–126

Perry, C.T. and Berkeley, A. (2009). Intertidal substrate modification as a result of mangrove planting: Impacts of introduced mangrove species on sediment microfacies characteristics. *Estuar. Coast. Shelf Sci*. 81(2), pp. 225–237. https://doi.org/10.1016/j.ecss.2008.10.019.

Rajkaran, A. (2011). A status assessment of mangrove forests in South Africa and the utilization of mangroves at Mngazana Estuary. PhD Thesis, Nelson Mandela Metropolitan University, 155 pp

Rajkaran, A. and Adams, J. (2016). *Mangroves in South Africa. In Mangroves of the Western Indian Ocean: Status and Management* (eds. Bosire, J.O., Mangora, M.M., Bandeira, S., Rajkaran, A., Ratsimbazafy, R., Appadoo, C. and Kairo, J.G.). WIOMSA, Zanzibar Town, pp. 115–135

Rajkaran, A., Adams, J. and Taylor, R. (2009). Historic and recent (2006) state of mangroves in small estuaries from Mlalazi to Mtamvuna in KwaZulu-Natal, South Africa. *Southern Forests* 71(4), pp. 287–296

Ratsimbazafy, R., Randriamanatena, D., Rakotondrazafy, J., Rakotomalala, H., Ramahery, V., Roger, E., Razakanirina, H., Rabarison, H., Lavitra, T., Mahafina, J., Ravaoarinorotsihoarana, L., Cripps, G., England, K., Carro, A., Gareth Jones, T., Glass, L., Taylor, B. and Danhaive, L. (2016). Mangroves in Madagascar. In *Mangroves of the Western Indian Ocean: Status and Management* (eds. Bosire, J.O., Mangora, M.M., Bandeira, S., Rajkaran, A., Ratsimbazafy, R., Appadoo, C. and Kairo, J.G.). WIOMSA, Zanzibar Town, pp. 95–112

Shapiro, A.C., Trettin, C., Küchly, H., Alavinapanah, S. and Bandeira, S. (2015). The Mangroves of the Zambezi delta: Increase in extent observed via satellite from 1994 to 2013. *Remote Sensing* 7, pp. 16504–16518

Shunula, J.P. and Whittik, A. (1999). Aspects of Litter Production in Mangroves from Unguja Island, Zanzibar, Tanzania. *Estuar. Coast. Shelf Sci.* 49, pp. 51–54

Spalding, M., Kainuma, M. and Collins, L. (2010). *World Atlas of Mangroves.* A collaborative project of ITTO, ISME, FAO, UNEP-WCMC, UNESCOMAB, UNU-INWEH and TNC. London (UK): Earthscan, London. 319 pp

Taylor, M., Ravilious, C. and Green, E.P. (2003). *Mangroves of East Africa.* UNEP-WCMC Biodiversity Series 13. Cambridge, 24 pp

Trettin, C.C., Stringer, C.E. and Zarnoch, S.J. (2015). Composition, biomass and structure of mangroves within the Zambezi River Delta. *Wetl. Ecol. Manag.* 24(2), pp. 173–186. https://doi.org/10.1007/s11273-015-9465-8

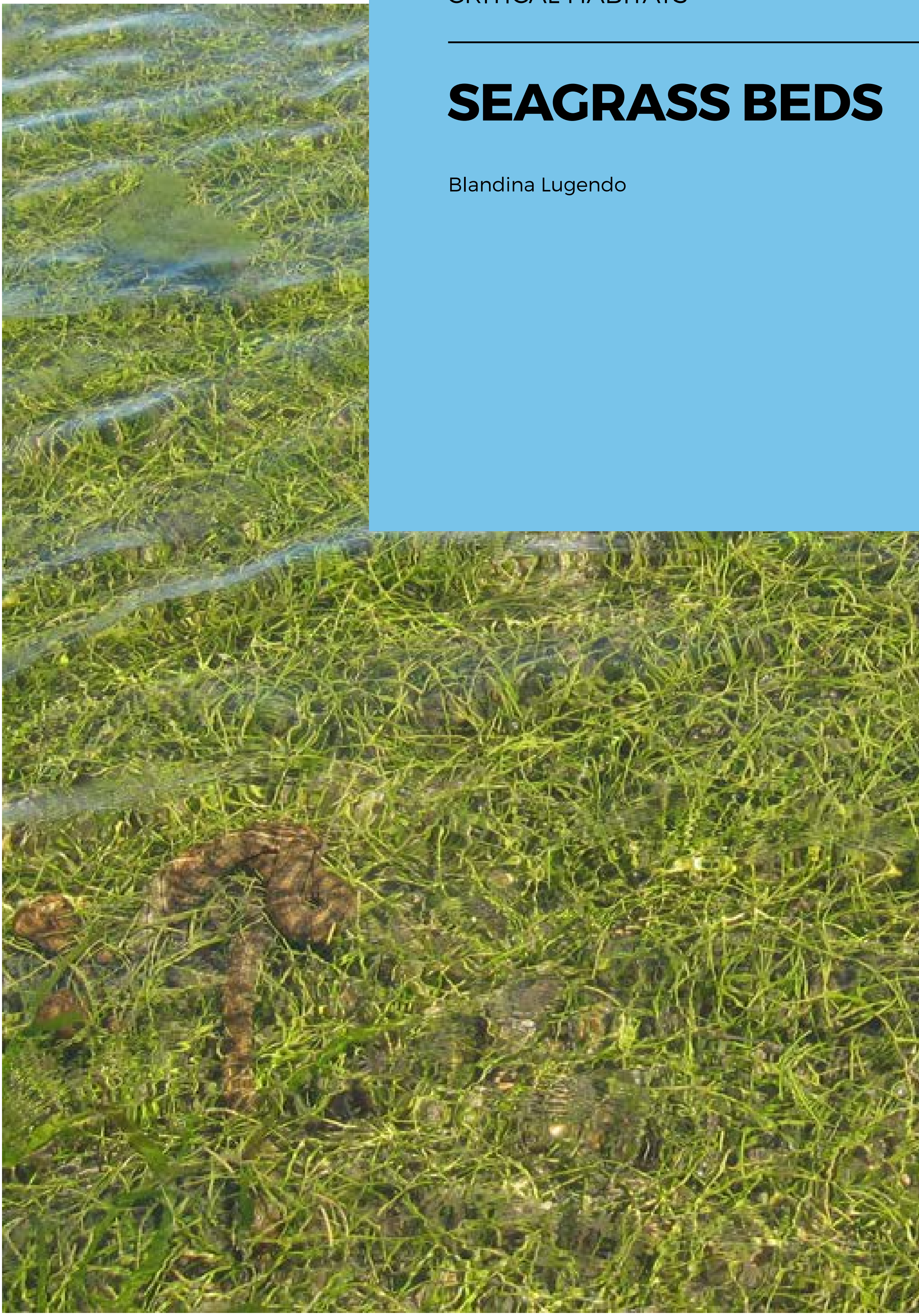
Uacane, M.S. and Ombe, Z.A. (2016). Modificação das áreas húmidas adjacentes ao Chiveve face às formas de ocupação do espaço na cidade da Beira. *Geoamazonia*, pp. 199–212. https://doi.org/10.17551/2358-1778/geoamazonia.v4n7p199-212

Wang, Y., Bonynge, G., Nugranad, J., Traber, M., Ngusaru, A., Tobey, J., Hale, L., Bowen, R. and Makota, V. (2003). Remote sensing of mangrove change along the Tanzania coast. *Mar. Geod*. 26, pp. 1–14

CRITICAL HABITATS

SEAGRASS BEDS

Blandina Lugendo



Thalassia hemprichii / *Syringodium isoetifolium* **mixed meadow at Inhaca Island, Mozambique.** © José Paula

BACKGROUND

Seagrasses are submerged flowering plants found in shallow marine and estuarine waters such as bays and lagoons in tropical and temperate areas. In the Western Indian Ocean (WIO) region seagrasses are distributed throughout the coastline of the mainland states (from the north coast of Somalia to the north-east coast of South Africa), and on coastlines of the Island States (Gullström et al., 2002). Being photosynthetic plants, seagrasses are commonly found in shallow depths where light levels are high. In the WIO region seagrasses are distributed from the intertidal zone down to about 40 m (UNEP, 1998; Gullström et al., 2002). In most countries of the region, seagrass beds often occur in close connection with coral reefs and mangroves, however, in the sub-tropical-temperate regions of South Africa, the seagrass *Zostera capensis* Setchell commonly occurs at the low water mark, with the salt marsh cordgrass *Spartina maritima* and other salt marsh species dominant at higher tidal levels (Adams and Bate 1995). Seagrasses form key components of marine ecosystems; however, they have received limited scientific attention compared to mangroves and coral reefs (Gullström et al., 2002).

Species composition

Fourteen species grouped into three families occur in the region (Table 1). Common species include *Thalassodendron ciliatum*, *Thalassia hemprichii*, *Cymodocea rotundata*, *C. serrulata*, *Enhalus acoroides*, *Syringodium isoetifolium* and *Halodule uninervis*. Another commonly reported species – *Halophila minor* is now considered a member of the *Halophila ovalis* complex. The occurrence of *Halodule wrightii* in the region is still debatable. According to Ochieng and Erftermeijer (2003), Waycott et al. (2004), and Bandeira (2011), this species does not occur in the region, however, it is still being reported in some countries, hence calling for further taxonomic research regarding its status. Two species, *Thalassodendron leptocaule* and *Halophila decipiens* shows very limited distribution, the former being reported in Mozambique and South Africa (Duarte et al., 2012; 2014; Browne et al., 2013), and the latter only in Seychelles. Exact location of one more species, *Halophila beccarrii* is subject to further research and observations in subtidal locations (Bandeira, 2011; Waycott et al., 2004). One other species, *Zostera capensis* is listed as Vulnerable in IUCN red list (Bandeira, 2014).

Table 1: Seagrass diversity for different countries within the WIO region.

FAMILY	SPECIES	COMOROS	FRENCH TERRITORIES	KENYA	MADAGASCAR	MAURITIUS	MOZAMBIQUE	SEYCHELLES	SOMALIA	SOUTH AFRICA	TANZANIA
Cymodoceaceae	*Cymodocea rotundata* Ehrenberg & Hemprich ex Ascherson	●	●	●	●		●	●	●	●	●
	Cymodocea serrulata (R. Brown) Ascherson	●	●	●	●	●	●	●	●	●	●
	Halodule uninervis (Forsskal) Ascherson	●	●	●	●	●	●	●	●		●
	Syringodium isoetifolium (Ascherson) Dand	●	●	●	●	●	●	●	●		●
	Thalassodendron ciliatum (Forsskal) den Hartog	●	●	●	●	●	●	●			●
	Thalassodendron leptocaule M.C. Duarte, Bandeira & Romeiras						●			●	
Hydrocharitacea	*Enhalus acoroides* (L.f) Royle	●		●			●	●			●
	Halophila ovalis (R. Brown) Hooker f.	●	●	●	●	●	●	●	●	●	●
	Halophila stipulacea (Forsskal) Ascherson	●		●	●	●	●	●	●		●
	Halophila decipiens Ostenfeld, 1902							●			
	Halophila beccarrii Ascherson, 1871										
	Thalassia hemprichii (Ehrenbergi) Ascherson	●	●	●	●		●	●	●	●	●
Zosteraceae	*Zostera capensis* Setchell	●		●			●			●	●
	Zostera muelleri subsp. *capricorni* (Aschers.)		●								
	Total number of species	10	8	10	8	6	11	10	7	6	10

Species richness of seagrasses varies among countries in the region. Mozambique with a total of eleven species supports the highest seagrass diversity, followed by Comoros, Kenya, Seychelles and Tanzania that harbour ten species each. Mauritius and South Africa with six species each harbour the lowest number of species (Table 1). Mixed seagrass beds are common in the region, in some areas up to eight or ten species can be found at the same locality, example in Mozambique (Bandeira, 2000) and Tanzania (Lugendo et al., 1999).

Seagrass coverage

Comprehensive mapping of seagrass beds has not been done for most countries in the region, and hence total seagrass coverage in the WIO region is not known. Total area estimates exist for some countries, for example Comoros (Poonian et al., 2016), Kenya (KWS, 2013), Reunion (Cuvillier et al., 2017) and South Africa (Adams, 2016). Some attempts to map seagrasses have however been undertaken at some localities within several WIO countries eg in French Territories, Mauritius, and Mozambique (Bandeira and Gell, 2003; Daby, 2003; Amone-Mabuto et al., 2017; Cuvillier et al., 2017).

The global dataset provides coverage data for some countries in the WIO, however, the provided data shows a high degree of discrepancy when compared with available country data (Table 2). For example, coverage of seagrasses in Kenya ranges between 31 710 – 33 600 ha, (KWS, 2013; Harcourt et al., 2018), but the provided figure in the global dataset ie 13 518.73 ha and 9 038.35 ha is far below country estimates. Discrepancies also exist for Kenya and Comoros (see Table 2). Although countrywide data does not exist for some countries, the provided data for Tanzania, for example, are too low, indicating the need to conduct comprehensive region-wide habitat mapping in order to establish baseline information for regional as well as countrywide purposes.

Country information

Although seagrasses are found in all countries of the WIO region, they are not distributed ubiquitously along the coastlines. Furthermore, due to limited research regarding seagrass distribution in the region, data is limited to areas where information is available. The following section provides detailed information on species composition, distribution and coverage of seagrasses in different countries of the WIO.

Comoros

In Comoros, seagrasses occur around all three islands, namely Grande Comoro, Anjouan and Mohéli. The largest seagrass bed occurs around Anjouan Island (along the Bimbini Peninsula) and covers an area of 1419 ha. In Grande Comoro seagrasses occur in large, continuous beds in the shallow waters at the northern and southern tips. Five separate seagrass beds occur in northern Grande Comoro and cover a combined area of 333 ha. Two separate beds occur in southern Grande Comoro Island covering a total area of 379 ha. The seagrass beds

Table 2: Global dataset and country data for seagrass coverage in WIO.

COUNTRY	GLOBAL DATASET (ha)		COUNTRY DATA (ha)
	UNEP – WCMC	RCMRD	
Comoros	5 663.73	No data	2 131
Reunion (France)	No data	No data	2.63
Kenya	13 518.89	9 038.35	31 710 – 33 600
Madagascar	87 668.60	22 129.55	No Data
Republic of Mauritius	No data	No data	No Data
Mozambique	80 662.0	48 073.75	No Data
Seychelles	No data	No data	No Data
Somalia	No data	No data	No Data
South Africa	No data	No data	1200
Tanzania	No data	70 509.85	No Data

Data source: UNEP World Conservation Monitoring Centre and Regional Centre for Mapping of Resources for Development (RCMRD).

around Mohéli Island are fragmented and generally occur in low densities, possibly due to sedimentation (Poonian et al., 2016). They occur around the whole island, but they are more prevalent in the southern part of the island, most likely because of the protection provided by Mohéli Marine Park. Also, the area receives less human pressure since the majority of Mohéli Island's population (75 per cent) lives in the north (UNEP, 2002). Ten seagrass species (including *H. wrightii*, a species thought not to occur in the region) are reported in Comoros, with *Thalassia hemprichii* being the most dominant (Poonian et al., 2016). Other species that occur in the Comoros are listed in Table 1.

French Territories

Three French territories occur in the WIO, namely, Reunion Island, Mayotte and Iles Eparses (Europa, Glorieuses Islands, Juan de Nova, Tromelin and Bassas da India). Seagrasses in Reunion Island are found in very shallow waters (<2 m). Their coverage is low (maximum of 2.63 ha), fluctuating with impacts of cyclonic events (summer) or high austral swells (winter). They are mainly located on the west coast, among the Hermitage/La Saline coral reef complex (8 km long, 500 m width maximum), and are part of the reef complexes (Cuvillier et al., 2017). A very small seagrass bed also occurs in the south-east coast (Sainte-Rose). Reunion Island harbours mono-specific seagrass meadows consisting of *Syringodium isoetifolium* (Cuvillier et al., 2017). On Mayotte, seagrasses cover 760 ha (Trégarot et al., 2017). Eight seagrass species, dominated by *Syringodium isoetifolium* (Ballorain et al., 2010) occur around Mayotte (Table 1). Other species include *Halophila ovalis*, *Cymodocea serrulata*, *C. rotundata*, *Thalassia hemprichii* and *Thalassodendron ciliatum*. Also observed in Mayotte is *Zostera muelleri* subsp. *capricorni*, (Aschers.) (Ballorain et al., 2010), but it is uncommon. Five seagrass species have been identified in Iles Eparses namely *Halodule uninervis*, *Halophila ovalis*, *C. rotundata*, *T. hemprichii* and *T. ciliatum* (Mattio et al., 2016).

Catshark in seagrass. © Shannon Moran

Kenya

Seagrass beds in Kenya are estimated to cover a surface area of between 31 710 and 33 600 ha (KWS, 2013; Harcourt et al., 2018), with the most extensive cover occurring in Lamu-Kiunga area, Malindi-Ungwana Bay, Watamu, Mombasa, Diani-Chale, Shimoni-Funzi Bay, Gazi Bay and Mida Creek (Dahdouh-Guebas et al., 1999; ASCLME, 2012a; KWS, 2013; Harcourt et al., 2018). Seagrass habitats are mostly lagoonal and associated with mangroves in some areas. Seagrasses can be seen in low water depths, in the nearshore areas exposed during low tide, in channels 5 m deep, and beyond 15 m deep in offshore waters. Comprehensive country map is not existent, however, a recent study by Maina et al. (2015) provides a map for about 300 km of the Kenyan Coast. Ten seagrass species occur in Kenya (Table 1), with *Thalasondendron ciliatum* being the most dominant (Ochieng and Erftemeijer, 2003).

Madagascar

Seagrass in Madagascar are known to occur off the north-west and north-east coast of the country. Off the north-west coast, seagrasses occur notably around Nosy-Be, Nosy Iranja, Sahamalaza, Analalava, Mahajamba and Mahajanga; and off Nosy Hara in the north region. In the north-east, they occur between Antsiranana and Vohémar (Tantely F. Tianarisoa, pers. comm. 2018). Further south, seagrasses are common within the islets of Barren Archipelago (Cripps, 2010) and in the Sahamalaza Marine Park in the mid-western Madagascar, and around Tulear. A total of eight species have been identified so far (Table 1). Common species *include Thalassodendron ciliatum* and *Thalassia hemprichii* (Bandeira and Gell, 2003) and are mainly found on stable substrates, such as within coastal lagoons, and within inner edge of coral reef flats (Obura et al., 2011). Monospecific meadows of *T. ciliatum* occur in Sahamalaza. All species except *T. ciliatum* are found on muddy shallow areas. A total area of 106 105 ha of seagrasses have been mapped using satellite images and classified according to their density (Lantoasinoro Ranivoarivelo pers. comm. 2018). Sparse seagrass beds dominate with a surface of 48 062 ha, followed by dense seagrass beds (40 988 ha), and finally moderately dense seagrass beds (17 055 ha).

Mauritius

Seagrass meadows in the Republic of Mauritius are located mainly in coastal lagoons and are most abundant in the south-west, south-east and north-east regions. Six species have been reported in the country (Table 1). Two species, *Halophila stipulacea* and *H. ovalis* occur in Rodrigues, mainly in Anse aux Anglais and Baladirou. Seagrass cover is estimated at 55 ha for Mauritius and 649 ha for Rodrigues (Turner and Klaus, 2005). Seagrass beds of Mauritius were mapped in a study commissioned by the Government in 2009 (ESA Policy Guidance Report, 2009), however, details could not be accessed.

Mozambique

In northern Mozambique, seagrasses occur abundantly mainly within Cabo Delgado Province (around some islands of Quirimbas Archipelago, Pemba Bay, Chuíba – Murrèbuè and Mecúfi), and Nampula Province (within Nacala-a-Velha, Mozambique Island and surroundings the Primeiras and Segundas Islands). Extensive seagrass meadows occur in Quirimbas Archipelago and are common in the back reef and lagoons but also in the subtidal areas. In southern Mozambique, seagrass occur mainly within Inhassoro, Vilanculos and Bazaruto archipelago region; Inhambane Bay and Maputo Bay. A total of eleven species occur in Mozambique. Dominant species include *Thalassodendron ciliatum*, *Thalassia hemprichii*, *Halodule uninervis*, *Syringodium ioetifolium*, *Cymodocea rotundata* and *Halophila ovalis* complex. Bazaruto Archipelago harbours rich seagrass meadows dominated by *T. ciliatum*, *T. hemprichii*, *H. uninervis* and *C. rotundata* (Bandeira et al., 2008). The tallest individuals of *T. ciliatum* (126 cm) were collected from this area (at Inhassoro) and are deposited at UEM Herbarium (Bandeira et al., 2014). Total seagrass cover is not known for Mozambique; however, some estimates exist for some areas (Table 3).

Table 3: Seagrass coverage in some localities within Mozambique.

LOCALITY	AREA (ha)	REFERENCE
Quirimbas archipelago, mainland regions and Pemba	17 000–22 800	Bandeira and Gell (2003)
Pemba- Mecufi	2800	Bandeira and Gell (2003)
Fernão Veloso	7500	Bandeira and Gell (2003)
Ilha de Moçambique and surroundings, Lumbo, Cunducia, Mussoril, Quissimajulo, relanzapo areas	80 000	Bandeira and Gell (2003)
Bazaruto Archipelago and adjacent mainland regions	8800	Bandeira et al., (2008)
Inhambane Bay	6200	Amone-Mabuto et al., (2017)
Maputo Bay	3875	Bandeira et al., (2014)

Seychelles

In Seychelles, seagrasses occur in almost all islands, and in Aldabra – one of the largest coral atolls in the world, where stands of *Thalassodendron ciliatum* are common (Jeanne Mortimer pers. comm. 2017). A total of ten seagrass species occurs in Seychelles (Table 1), with *T. ciliatum*, *Thalassia hemprichii*, *Siringondium isoetifolium*, *Cymodocea rotundata*, *C. serrulata*, and *Halodule uninervis* being common. Seychelles encompasses the deepest seagrass meadows in the WIO region with *T. ciliatum* recorded over 30 ms deep (Bandeira and Gell, 2003) and *Halophila decipienes*, in Daros Island (Amirantes Group), down to 26 m (Jeanne Mortimer pers. comm. 2017).

Somalia

In Somalia, seagrasses occur along the whole coastline from north-west of Sa'adiin Island and north-eastern Hafun-Somalia to south-east Kismaayo (Bajuuni Island) at the coastal border with Kenya. Abundant seagrass beds have been reported to occur from Adale to Ras Chiamboni and to a smaller extent along the north coast (ASCLME, 2012b). Seven species occur in Somalia (Table 1), with *Thalassodendron ciliatum* being abundant (UNEP - Nairobi convention and WIOMSA, 2015). Similar to many other countries within the WIO, no attempts have been made to map the seagrass beds of Somalia and hence comprehensive seagrass maps do not exist.

South Africa

Only 20 per cent of South African estuaries (being 72 estuaries) support submerged aquatic vegetation, as these plants are sensitive to changes in water level, turbidity, nutrients and salinity (Adams, 2016). The total submerged macrophyte habitat in South Africa covers 2564.78 ha (Adams, 2016). Six species occur in South Africa (Table 4) with *Zostera capensis* Setchell dominating. *Zostera capensis* occurs mainly in permanently open estuaries (POEs) from the Kosi River estuary on the east coast to the Olifants River estuary on the west coast (Table 1). *Zostera capensis* currently covers approximately 1200 ha (Adams, 2016). Area cover is dynamic and changes in response to estuary mouth conditions, water level fluctuations and river flooding. The largest area is found in the estuarine bay of Knysna, followed by the Berg River estuary on the west coast. Seagrasses are mostly absent from estuaries on the east coast due to development changes and high turbidity (Adams, 2016).

Table 4: Estuaries with the largest area of *Zostera capensis* (ha) in South African estuaries from east to west.

LOCALITY	AREA (ha)	PROTECTION STATUS	REFERENCE
Kosi	5.0	iSimangaliso Wetland Park Authority	DWS (2016)
Mhlathuze	28.5	Ezemvelo KwaZulu-Natal Wildlife	Cyrus et al. (2008)
Qora	8.5	None	Colloty (2000)
Keiskamma	12.0	None	Colloty et al. (2002)
Kariega	32.6	None	NBA (2012)
Bushmans	39.8	None	Jafta (2011)
Kromme	34.0	None	Schmidt (2013); Department of Water Affairs and Forestry (2005)
Swartkops	44.7	None	Bornman et al. (2016)
Keurbooms	64.0	CapeNature (partial protection)	Africa (2008)
Knysna	353.0	South African National Parks (small section)	Schmidt (2013)
Langebaan	85.6	South African National Parks	Van Der Linden (2014)
Berg	206.0	IBA (Important Bird Area), water and intertidal CapeNature, under consideration for RAMSAR	Boucher and Jones (2007)
Olifants	47.7	None	DWAF (2006); Forestry (2006); Taljaard et al. (2006)

Tanzania

Seagrasses are widely distributed along the coast of Tanzania. Extensive beds occur on the northern coast of Tanzania mainland (along the Tanga coast), in the deltas of the Ruvu, Wami and Rufiji rivers, around Mafia Island, the Songo Songo Archipelago and around Kilwa. Seagrass beds also form a dominant feature in Chwaka Bay, Zanzibar (UNEP-Nairobi Convention and WIOMSA, 2015; ASCLME, 2012c; UNEP/Nairobi Convention Secretariat, 2009). Extensive meadows also occur in Chwaka Bay Zanzibar (Ochieng and Eftermeijer, 2003). Ten species occur in Tanzania (Table 1), with *Thalassia hemprichii*, *Syringodium isoetifolium* and *Thalassodendron ciliatum* dominating (ASCLME, 2012c). Seagrass area coverage has not been established in Tanzania, and therefore maps are non existent. However, in one area (Chwaka Bay) seagrasses are estimated to cover 10 000 ha (UNEP/Nairobi Convention Secretariat, 2009).

IMPORTANCE

Seagrasses are one of the most productive aquatic ecosystems in the world (Duarte and Chiscano, 1999). They possess a complex habitat structure; as a result, seagrass meadows are inhabited by a myriad of other species. In South Africa for example, despite their small area the *Zostera capensis* beds serve as a substrate for epiphytes and periphyton, which are then used as a food source for other organisms (Adams, 2016). Seagrasses are used as shelter against predation, as foraging and nursery areas by many fish and invertebrates. A few animals (eg dugongs, green turtles, sea urchins and some herbivorous fishes) feed directly on seagrasses. However, the large proportion of seagrass biomass enter the marine food web through detritus, thereby supporting productivity through recycling of nutrients and carbon (Hemminga et al., 1991). Their loss has been shown to result in decreased primary productivity and loss of invertebrate and fish abundance (Froneman and Henninger, 2009; Sheppard et al., 2011).

Other ecological functions of seagrasses include bottom sediment stabilization, dampening of the wave energy and current velocity, thereby enhancing sediment settling, consequently reducing turbidity and coastal erosion (Green and Short, 2003). Further, seagrass beds trap large amounts of nutrients and organic matter in the bottom sediment (Green and Short, 2003).

Sometimes, seagrass meadows occur in synergistic relationships with mangroves and coral reefs (Björk et al., 2008). In such cases, seagrass meadows and mangroves stabilize sediments, dampen water movements, trap heavy metals and nutrients, filter freshwater discharges from land, maintaining water quality for coral reef growth. In turn coral reefs buffer seagrasses and mangroves from waves and ocean currents (Björk et al., 2008). Connectivity among these three habitats has also been observed in terms of reef fish migrations (Unsworth, et al. 2008), and often form intermingled mosaic habitats which are ecologically interdependent.

Seagrasses also provide many important ecosystem services to humans as well. This is accomplished through support to fisheries and tourism industries, as well as in coastal protection. Seagrass fisheries are conducted for income and subsistence, as well as for recreational purposes (Nordlund et al., 2017) and are reliant on the ability of healthy seagrass beds to support finfish, shellfish and other fishery related products. In Kenya, the seagrass ecosystem is vital to the fishing industry as it serves as an important habitat to approximately 70 per cent of fish species, for at least a part of their life cycle. Examples of commercially important fish associated with seagrass beds belong to the families Labridae, Lethrinidae, Lutjanidae, Monacanthidae, Scaridae, Siganidae and Teraponidae. Seagrass beds are also widely used for collection of invertebrates. In Maputo Bay for example, invertebrates, mostly bivalves (such as *Meretrix meretric*, *Eumarcia paupercula*, *Solen cylindaceus* and *Pinctada capensis*), snails (*Volema pyrum*), crustaceans (*Portunus pelagicus*) and sea urchins (*Tripneutis gratilla*) are collected from this habitat (see Vicente and Bandeira, 2014; Ferreira and Bandeira, 2014), however there are evidences of depletion of such resources within the region (Nordlund and Gullström, 2013).

Contribution of seagrasses to the tourism industry depends mainly upon sediment stabilization which supports existence of the beautiful sandy beaches and healthy coral reefs. In Mauritius for example, seagrasses grow close to beaches and their presence is associated with sediment stabilization along the coast. Beach erosion was observed to be prominent in areas where loss of seagrasses has occurred, and loss of seagrass meadows has been observed to impact the health of coral reefs (JICA, 2015). Seagrasses are also used as habitats for seaweed farming (Hedberg et al., 2018). A recent study by Gullström et al. (2017) gives insights to seagrass beds of the WIO as potential storage habitats for blue carbon.

Studies that have explored the monetary value of seagrasses are limited. Contanza et al. (1997) estimated the value of the water purification service produced by seagrass beds at USD 19 002/ha/year (or € 1 732 255/km^2/year). A study by Trégarot et al. (2017) estimated the monetary value of various ecosystem services provided by coastal ecosystems in Mayotte. The values reported for seagrasses are € 353 170/km^2/year for coastal protection, € 1 243 759/km^2/year for water purification, € 2154/km^2/year for fish biomass, and € 1911/km^2/year for carbon sequestration. These values may vary in different WIO countries depending on the size (coverage) and ecological status (eg pristine, moderately degraded or poor condition) of the seagrass meadows, however, they provide a very important rationalization to emphasize the economic value of seagrass conservation efforts.

Key species associated with seagrass habitats

Seagrasses provide nursery and foraging grounds for an array of animals. For some, their existence depends solely on the presence of seagrasses, such as the threatened Green sea turtle (*Chelonia mydas*) and the Dugong (*Dugong dugon*). Both are herbivores, with the Dugong feeding exclusively on seagrasses (Erftemeijer et al., 1993). While the Green turtle is considered Endangered on the IUCN Red List of Threatened Species, the Dugong is considered Vulnerable. Small populations of Dugong are reported in Kenya, Tanzania and Mozambique. On the other hand, Green turtles have been reported to occur in seagrass beds of Kenya, Tanzania, French Territories (Reunion and Mayotte), Seychelles and Mauritius. Another species of marine turtle, the Hawksbill (*Eretmochelys imbricate*) is also known to feed over seagrass beds as observed in Tanzania and Mauritius. In South Africa, the eelgrass (*Zostera capensis*) provides critical habitat for a number of species including the range-restricted Knysna seahorse *Hippocampus capensis* and Bot klipfish *Clinus spatulatus* (Lockyear et al., 2006). The critically endangered estuarine pipefish *Syngnathus watermeyeri* occurs in submerged macrophyte beds consisting of *Z. capensis* and *Ruppia* spp. (Whitfield and Bruton, 1996), while the spotted sea hare *Aplysia oculifera* and shaggy sea hare *Bursatella leachii* often occur in high densities, grazing in *Z. capensis* beds (Stephen J. Lamberth, pers. comm.; Adams, 2016). Many herbivorous fish species are found within seagrass beds, with marbled parrotfish *Leptoscarus vaigiensis* being the most dominant. In addition, many sea cucumbers (eg *Stichopus chloronotus*, *Synapta maculata* and *Holothuria leucospilota*), and sea urchins (eg *Toxopneustes pileolus*, *Tripneustes gratilla* and *Echinometra mathaei*) are abundant in seagrass beds within the region.

Impacts of seagrass loss and degradation

When seagrass beds are degraded, they lose their complexity and functionality, leading to severe impacts on biodiversity, other marine ecosystems as well as human wellbeing. Impacts may range from loss of their nursery role for example with consequent reduction in fisheries resources. Loss of seagrass cover results in the decline in biodiversity and the local extinction of some species, especially those exclusively dependent on this ecosystem. Other impacts include loss of shoreline stability leading to increased risk of shoreline erosion, and decreases in their function as a global carbon sink (Sifleet et al., 2011). Impacts on human well-being associated with the loss of these ecosystem goods and services include food insecurity and the loss of livelihoods for communities that depend on fishery resources. Also, seagrass degradation and or loss may lead to loss of revenues accrued from coastal tourism as a result of deterioration of coral reef habitats and sandy beaches.

THREATS

No country in the WIO region has established threat levels for seagrass ecosystems. This is probably due to lack of adequate information as well as the generally low profile of seagrasses in countries' conservation agenda. However, seagrass meadows are on a declining trend globally. According to Waycott et al. (2009), seagrass meadows disappeared at a rate of 7 per cent of their total global area per year between 1990 and 2006. Seagrass loss, degradation and fragmentation is still continuing in many areas.

Threats to seagrass habitats in the WIO region are more or less similar across the region (UNEP-Nairobi Convention and WIOMSA, 2015). Most of these threats are a result of human activities, though natural causes can also account for seagrass loss in the region. Habitat destruction resulting from fishing activities, particularly the use of beach seines and trawls by artisanal fishers over seagrass beds is a widespread threat (UNEP-Nairobi Convention and WIOMSA, 2015). In Tanzania, although beach seining is an illegal fishing method and hence prohibited, it is a common undertaking on most intertidal seagrass beds and shallow waters (Green and Short, 2003). The use of seine and drag nets by artisanal fishermen is also a daily activity in the shallow waters of Gazi Bay (Kenya) and is thought to cause significant damage to seagrasses in the area (Githaiga et al., 2017).

Another threat that is related to fishing activities is the collection of invertebrates (gleaning) in the intertidal area that often involves digging and revolving huge amounts of sediments as well as trampling over seagrasses. Sediment turnover due to digging and collection of bait caused habitat loss in Langebaan Lagoon, South Africa (Pillay et al., 2010), and in Maputo Bay (Vicente and Bandeira, 2014). *Zostera capensis* beds have disappeared in the main village of Inhaca in Mozambique due to heavy concentration of fishing, trampling and tourist activities (ASCLME/SWIOFP, 2012). Inappropriate fishing practices have also been reported in Primeiras and Segundas Environmental Protected Area (PSEPA) (WWF, 2017).

Another important threat to seagrasses is eutrophication as a result of excessive nutrient input into the coastal waters, with consequent proliferation of algae that obscures seagrasses from getting sufficient light, and ultimately resulting into their death. Loss of *Zostera capensis* habitat as a result of eutrophication and smothering with macroalgae *Ulva lactuca* mats has been reported in Knysna River estuary (South Africa) (Human et al., 2016). Loss of intertidal seagrass habitat due to eutrophication has also occurred in Ocean Road beach, Dar es Salaam (Tanzania), leading to proliferation of *Ulva* spp. that smothered the seagrasses (Blandina Lugendo pers. obs.). Faulty and overloaded wastewater treatment plants and agricultural return flows are increasing nutrient inputs to South African estuaries and represent a growing threat for the future fate of eelgrass (Adams, 2016). Pollution (mainly organic matter and nutrients) is also reported to affect seagrass beds in Reunion.

Seagrasses of the WIO region are also threatened by sedimentation – with sediments originating from various sources (eg deforestation of coastal vegetation and catchment areas, reclamation, dredging and agricultural activities). Sedimentation may smother seagrasses or increase turbidity leading to seagrass death. Sedimentation and nutrient runoff are among stressors for seagrasses occurring in PSEPA in Mozambique (WWF, 2017). Destruction of seagrasses due to sedimentation has also been reported in Mohéli Marine Park in Comoros (ASCLME, 2012d) and in Sabaki catchments in Kenya (Katwijk et al., 1993).

Seagrass meadows also suffer from physical destruction related to water-based leisure activities. Seagrass habitat degradation due to boating activities is reported in South Africa (Adams, 2016). In Mauritius, seagrasses are removed by hotels believing that they are unsightly and harbour organisms that may cause injury to swimmers (Daby, 2003). Another mechanical form of destruction of seagrass beds is associated with coastal development activities, such as port expansion, which results in its physical removal as well as increased turbidity. For instance, in South Africa, expansion of Richards Bay harbour and the disposal of dredge spoil threatened the existence of one of the important *Z. capensis* habitat on the east coast in Mhlathuze River estuary (Cyrus et al., 2008). Seagrasses found in estuaries suffer especially from freshwater abstraction, river mouth closure to the sea and freshening (Whitfield and Elliott, 2011), a situation particularly detrimental in South Africa where almost all seagrasses are found in estuaries.

Natural threats to seagrasses include diseases (den Hartog, 1987), storms (Gallegos et al., 1992) and overgrazing by sea urchins (McClanahan et al., 1994; Wallner-Hahn et al., 2015). Overfishing of sea urchin predators, particularly triggerfish, and consequent population explosion of sea urchins, can lead to degradation of seagrass beds (McClanahan, 2000). For instance, sea urchin (*Tripneustes gratilla*) herbivory was associated with a 50 per cent reduction of *Thalassodendron ciliatum* beds in Diani-Chale and Watamu in Kenya between 2001 and 2005 (Eklof et al., 2008).

CONDITION/STATUS

Information regarding the status of seagrass beds within the WIO is largely lacking, however, considering that the threats are continuing, then it is logical to generalize that seagrass beds in the WIO are following a declining trend. A recent review of the seagrasses of South Africa indicates decline in coverage for 13.6 per cent of the estuaries (Adams, 2016). Some *Zostera capensis* beds have also disappeared from some estuaries in South Africa (eg northern St. Lucia Estuary, Richards Bay harbour) as a result of the harbour development. Seagrass disappearance has also occurred in Ocean Road intertidal area in Dar es Salaam (Tanzania) as a result of eutrophication (Blandina Lugendo pers. obs.). About 20 per cent of seagrasses were lost in Nacala-a-Velha (Mozambique) in 2013–2016 due to construction of a new coal terminal (Salomao Bandeira pers. comm).

Significant increase in seagrass coverage has also occurred in the region, for example in Reunion (France) where over 82 per cent increase has been recorded between 2008 and 2014 (Cuvillier et al., 2017). Increase in seagrass coverage has also been recorded for 3 per cent of the South African estuaries (Adams, 2016). Overall, no country in the WIO region has established threat levels for seagrass ecosystems.

PROTECTION LEVEL

Although seagrass meadows are threatened, there exists some degree of protection for this important ecosystem in the region. Seagrass beds often form important parts of Marine Protected Areas (MPAs), which due to control of activities within their boundaries offer a certain level of protection. Unlike in many other areas, almost all seagrass beds in Reunion (France) are under protection (Cuvillier et al., 2017) while none in Somalia receives any formal protection.

Another form of protection for seagrass beds is through National Parks and related wildlife conservation areas. Examples include Saadani National Park in Tanzania which extends into the marine environment, Ezemvelo KwaZulu Natal Wildlife which offers some protection to seagrass beds on the east coast of Mhlathuze River Estuary, West Coast National Park and Garden Route National Park that offer protection to large areas of *Zostera capensis* in the Langebaan and Knysna systems, respectively (Adams, 2016). Regardless of the form of protection, however, the area of seagrasses under protection is generally small compared to that outside protection (Green and Short, 2003).

Currently, no legislation to protect seagrass beds as such exists in the WIO. Instead seagrasses are inclusively protected by other regulation aimed at protecting fisheries resources and or the general environment. In 2013, Kenya established a coral reef and seagrass ecosystems conservation strategy 2014–2018 (KWS, 2013). The strategy forms an important framework for the management of these unique ecosystems and provides a robust framework for building the necessary partnerships to guide conservation and restoration of these critical habitats. In order for Kenya to realize the impact of this strategy, adequate implementation of the strategy is required.

Other initiatives to protect seagrasses include education and awareness raising programmes on the value of seagrasses, their threats and need for conservation, and community participation in seagrass related research. Example of this include the ongoing activities by Sea Sense (an NGO in Tanzania) to educate the community on the importance of seagrasses in supporting the existence of marine turtles and dugongs (Sea Sense, 2012). There are also some initiatives for restoring seagrass beds in Tanzania (Blandina Lugendo pers. obs.), Kenya (Mutisia, 2009; Dago, 2020; Jacqueline Uku pers. comm.) and in Mozambique (Bandeira et al., 2020). The new Guidelines on Seagrass Ecosystem Restoration for the Western Indian Ocean Region (UNEP-Nairobi Convention/WIOMSA, 2020) offer the necessary technical guidance in support of successful restoration of the degraded seagrass habitats in the WIO region.

No country in the WIO has established protection level solely for seagrass beds. For seagrass beds found within MPAs their protection level automatically follows that of the respective MPA, which ranges from no-take to multiple-use areas. In South Africa, where seagrass habitats are mostly found in estuaries, their protection status follows that of a particular estuary. For example, approximately 30 per cent of all estuaries with *Zostera capensis* receives some formal protection status (poorly protected 0–30 per cent) and approximately 46 per cent fall under estuary management plans (Adams, 2016), which is a requirement of the National Environmental Management: Integrated Coastal Management Act (Act 24 of 2008).

Another opportunity that exists for the protection of seagrasses is the Dugong MoU (Memorandum of Understanding on the Conservation and Management of Dugongs and their Habitats throughout their Range), under the Convention on the Conservation of Migratory Species of Wild Animals (CMS) that was adopted by the first Signatories in 2007. Some countries in the WIO (eg Comoros, Kenya, Madagascar, Mozambique, Seychelles, Somalia, and Tanzania) are signatories. The implementation of this MoU could result in significant increase in seagrass protection.

Also, there are increased efforts towards establishments of networks of scientists that work on seagrasses both regionally and internationally. Among other objectives, these networks aim at raising the profile of seagrasses in the different countries' conservation agenda. Such networks include the WIO Seagrass Monitoring Network and Indo Pacific Seagrass Network which have already been established, and the International Network of Seagrass Experts which is in the process of being established.

PRIORITY AREAS FOR CONSERVATION

There is inadequate protection of seagrass habitats in the WIO region, and hence there is a need to identify priority areas for conservation as well as opportunities that can be used to enhance seagrass protection.

Monitoring efforts require baseline information, and therefore at the beginning and in the absence of new studies, efforts should be directed on areas which contain baseline information. Such areas would include those that support endangered species, particularly marine turtles and Dugong. There are a number of research as well as community-based conservation initiatives currently ongoing in these areas, and therefore, regional or country efforts can build on the existing initiatives. Also, areas where research on the role of seagrasses as carbon sinks and potential sustainable financing options through blue carbon initiatives, offers opportunity for further research on seagrass meadows. Priority should also be given to areas where seagrass beds have been proven to play important roles in supporting adjacent ecosystems like coral reefs and mangroves (Björk et al., 2008). Furthermore, areas that are managed by communities such as Community Managed Areas (CMA) and Locally Managed Marine Areas (LMMAs) (Rocliffe et al., 2014, Samoilys et al., 2017) are potential candidates for prioritization as they serve to increase the coverage of protected areas along the coastlines of the WIO.

Priority should also be given to important seagrass areas outside MPAs as these receive relatively little or no protection. For example, in South Africa a population of *Zostera capensis* that would serve as an important nursery habitat was recorded at the Rail Balloon area in Richards Bay harbour (Cyrus and Vivier, 2014), and this would require protection. On the east coast, Mhlathuze River estuary is protected by Ezemvelo KwaZulu Natal Wildlife but not from the threats of nearby dredging and the associated increase in turbidity (Cyrus et al., 2008). As a result, *Z. capensis* has been lost from the northern St Lucia Estuary and recovery chances are almost impossible due to the freshwater and silty conditions. Therefore, monitoring and management actions should be directed to Mhlathuze River estuary to ensure protection of the limited *Z. capensis* beds there. Priority should also be given to the large *Z. capensis* habitats at the Keiskamma River estuary (12 ha) followed from north to south by Kariega, Bushmans, Swartkops and Kromme (30–40 ha in each estuary), none of which have protection status (see Table 4).

In Mauritius, priority could be given to Albion (in the west) and Pointe aux Canonniers (in the north) due to available data collected by the Albion Fisheries Research Centre (Nabeehah B. Roomaldawo pers. comm.). Also, some seagrass areas in Mauritius are classified as Environmentally Sensitive Areas (ESA) and are a priority for conservation in this country (ESA Policy Guidance Report, 2009). In Somalia, where knowledge on seagrasses and conservation initiatives are very limited, priority should be given to larger seagrass meadows located from Adale to north Mogadishu, and Kismaayo south to the Bajuuni Islands off the coast of south Somalia (Mohamud. H. Ali pers. comm.).

RECOMMENDATIONS

Seagrasses receive low priority in conservation mainly due to their lack of visibility and poorly recognized ecological and economic roles, at local, national as well as regional levels. At regional level, efforts should be made to ensure that the profile of seagrasses is raised high up on countries' conservation agendas. Mechanisms should be put in place at the regional level to ensure regional collaborations and joint actions for the conservation of seagrass ecosystems. These may include:

1. Mainstreaming seagrass conservation needs into policy frameworks in the WIO countries.

2. Habitat mapping and modelling to generate basic information useful for sustainable management of seagrass ecosystems in the WIO.

3. Establishment of Regional Seagrass Task Force to spearhead seagrass conservation in the region, and this should be echoed at country levels.

4. Education and awareness raising regarding the importance, threats and need for protecting seagrasses.

5. Establishment of robust monitoring methodologies so that changes in seagrass abundance and distribution in these sensitive coastal environments can be understood.

6. Increase of research and monitoring endeavours to establish the condition/status of seagrass habitats and inform (science-based) decision-making.

7. Seagrass restoration.

8. Community involvement in seagrass conservation efforts.

9. Capacity-building at institutional level to equip technical staff with knowledge regarding seagrasses.

10. Establishment of comprehensive strategies that will protect seagrass ecosystems together with coral reefs and mangrove ecosystems as a continuum.

11. Economic valuation of seagrass habitats – that helps to raise the profile of seagrasses and attract attention of decision makers.

12. Increase protection through the establishment of new MPAs as well as encouraging establishment of community and locally managed areas.

REFERENCES

Adams, J.B. (2016). Distribution and status of *Zostera capensis* in South African estuaries – A review. *S. Afr. J. Bot.* 107, pp. 63–73

Adams, J.B. and Bate, G.C. (1995). Ecological implications of tolerance of salinity and inundation by *Spartina maritima. Aquat. Bot.* 52, pp. 183–191

Africa, B.M.S. (2008). *Keurbooms estuary rapid reserve determination study (version 2) — draft technical report.* Prepared by L. Van Niekerk of the CSIR for Ninham Shand Consulting Engineers. Report No. CSIR/NRE/ECO/ER/2008/0086/C

Amone-Mabuto, M., Bandeira, S. and da Silva, A. (2017). Changes in seagrass coverage after 21 years of climate impact – case of Inhambane Bay (southern Mozambique). *West. Indian Ocean J. Mar. Sci.* 16(2), pp. 13–25

ASCLME (2012a). National Marine Ecosystem Diagnostic Analysis. Kenya. Contribution to the Agulhas and Somali Current Large Marine Ecosystems Project (supported by UNDP with GEF grant financing)

ASCLME (2012b). National Marine Ecosystem Diagnostic Analysis. Somalia. Contribution to the Agulhas and Somali Current Large Marine Ecosystems Project (supported by UNDP with GEF grant financing)

ASCLME (2012c). National Marine Ecosystem Diagnostic Analysis. Tanzania. Contribution to the Agulhas and Somali Current Large Marine Ecosystems Project (supported by UNDP with GEF grant financing)

ASCLME (2012d). National Marine Ecosystem Diagnostic Analysis. Comoros. Contribution to the Agulhas and Somali Current Large Marine Ecosystems Project (supported by UNDP with GEF grant financing)

ASCLME/SWIOFP (2012). Transboundary Diagnostic Analysis for the western Indian Ocean. Volume 1: Baseline. ISBN: 978-0-620-57042-8

Ballorain, K., Ciccione, S., Bourjea, J., Grizel, H., Enstipp, M. and Georges, J–Y. (2010). Habitat use of a multispecific seagrass meadow by green turtles *Chelonia mydas* at Mayotte Island. *Mar Biol* 157, pp. 2581-2590. https://doi.org/10.1007/s00227-010-1520-7

Bandeira, S.O. (2000). Diversity and Ecology of Seagrasses in Mozambique: Emphasis on *Thalassodendron ciliatum* Structure, Dynamics, Nutrients and Genetic Variability. PhD Thesis, Göteborg University, Sweden

Bandeira, S.O. (2011). Seagrasses. In *A Field Guide to the Seashores of Eastern Africa and the Western Indian Ocean Island.* (ed. Richmond, M.D.), 3rd Edition, pp. 74–77, Sida/WIOMSA

Bandeira, S.O. (2014). *Zostera capensis*, a vulnerable seagrass species. In *The Maputo Bay Ecosystem.* (eds. Bandeira, S. and Paula, J.), pp. 171–173. WIOMSA, Zanzibar Town

Bandeira, S. and Gell, F. (2003). The Seagrasses of Mozambique and Southeastern Africa. In *Seagrass Atlas of the World.* (eds. Green, E. and Short F.), pp. 93−100. World Conservation Monitoring Centre. University of California press. ISBN 0-520-24047-2

Bandeira, S., Gullström, M., Balidy, H., Samussone, D. and Cossa, D. (2014). Seagrass meadows in Maputo Bay. In *The Maputo Bay Ecosystem.* (eds. Bandeira, S. and Paula, J.), pp. 147−169. WIOMSA, Zanzibar Town

Bandeira, S., Muiocha, D. and Schleyer, M.H. (2008). Seagrasses beds. In *A Natural History of the Bazaruto Archipelago, Mozambique* (eds. Everett, B.I., van der Elst, R.P. and Schleyer, M.H.), pp. 65-69. Special Publication No 8

Bandeira, S., Amone-Mabuto, M., Cuambe, M., Cossa, D. and Guissamulo, A. (2020). Seagrass restoration in Mozambique − How it began. *The WIOMSA Magazine*, Issue No. 12, December 2020

Björk, M., Short, F., Mcleod, E. and Beer, S. (2008). Managing Seagrasses for Resilience to Climate Change. IUCN, Gland, Switzerland. 56 pp

Bornman, T.G., Schmidt, J., Adams, J.B., Mfikili, A.N., Farre, R.E. and Smit, A.J. (2016). Relative sea-level rise and the potential for subsidence of the Swartkops Estuary intertidal salt marshes, South Africa. *S. Afr. J. Bot*. 107. https://doi.org/10.1016/j.sajb.2016.05.003

Boucher, C. and Jones, L. (2007). Chapter 6: Estuarine and floodplain vegetation. In Berg River Baseline Monitoring Programme Final Report (ed. Clark, B.M.), Volume 2. Berg River Estuary, September 2004. Unpublished report to the Department of Water Affairs and Forestry, 194 pp

Browne, C.M., Maneveldt, G.W., Bolton, J.J. and Anderson, R.J. (2013). Abundance and species composition of non-geniculate coralline red algae epiphytic on the South African populations of the rocky shore seagrass *Thalassodendron leptocaule* M.C. Duarte, Bandeira & Romeiras. *S. Afr. J. Bot.* 86, pp. 101−110. https://doi.org/10.1016/j.sajb.2013.02.173

Colloty, B.M. (2000). The botanical importance rating of the estuaries in the former Ciskei/Transkei, Department of Botany. University of Port Elizabeth, p. 202

Colloty, B.M., Adams, J.B. and Bate, G.C. (2002). Classification of estuaries in the Ciskei and Transkei regions based on physical and botanical characteristics. *S. Afr. J. Bot.* 68, pp. 312−321

Costanza, R., dArge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., Oneill, R.V., Paruelo, J., Raskin, R.G., Sutton, P. and van den Belt, M., (1997). The value of the world's ecosystem services and natural capital. *Nature* 387, pp. 253−260

Cripps, G. (2010). Feasibility study on the protection and management of the Barren Isles ecosystem, Madagascar. Blue Ventures Conservation Report (2009), for WWF and the "Réseau interdisciplinaire pour une gestion durable de la biodiversité marine: diagnostic environnemental et social autour des tortues marines dans le sud-ouest de l'Océan Indien". 272 pp

Cuvillier, A., Villeneuve, N., Cordier, E., Kolasinski, J., Maurel, L., Farnier, N. and Frouin, P. (2017). Causes of seasonal and decadal variability in a tropical seagrass seascape (Reunion Island, south western Indian Ocean). *Estuar. Coast. Shelf Sci.* 184, pp. 90−101

Cyrus, D.P. and Vivier, L. (2014). *Overall findings and assessment: specific abiotic and biotic components associated with priority habitats in Transnet Capital Projects Richards Bay Port Expansion Project.* CRUZ Environmental Report No. 18, 35 pp.

Cyrus, D.P., MacKay, C.F. and Weerts, S.P. (2008). Intrusion of beach-disposed dredger spoil into the Mhlathuze Estuary, South Africa, and its impact on *Zostera capensis. Afr. J. Aquat. Sci.* 33, pp. 223−231

Daby, D. (2003). Some quantitative aspects of seagrass ecology in a coastal lagoon of Mauritius. *Mar. Biol.* 142(1), pp. 193−203

Dago, T.M. (2020). Passive seagrass restoration in Mombasa Marine Protected Area. The WIOMSA Magazine. Issue No. 12, December 2020

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

den Hartog, C. (1987). "Wasting disease" and other dynamic phenomena in *Zostera* beds. *Aquatic Botany* 27, pp. 3−14

Department of Water Affairs and Forestry (DWAF) (2006). *Olifants/Doring catchment ecological water requirements study.* RDM Report on Estuarine Component. Project Number: 2002−377. CSIR Report No. ENV-S-C 2005-096 A

Department of Water Affairs and Forestry (DWAF). (2005). *Kromme/Seekoei catchments reserve determination study — technical component. Main Report. Prepared by Coastal and Environmental Services.* DWAF Report No. RDM/K90/00/CON/1205, 108 pp

Duarte, C.M. and Chiscano, C.L. (1999). Seagrass biomass and production: A reassessment. *Aquat. Bot.* 65, pp. 159−174

Duarte, M.C., Bandeira, S. and Romeiras, M. (2014). *Thalassodendorn leptocaule* a new species of seagrass from rocky habitats. In *The Maputo Bay Ecosystem* (eds. Bandeira, S., and Paula, J.) pp. 175−179. WIOMSA, Zanzibar Town

Duarte, M.C., Bandeira, S. and Romeiras, M.M. (2012). Systematics and Ecology of a New Species of Seagrass (*Thalassodendron*, Cymodoceaceae) from Southeast African Coasts. *Novon* 22, pp. 16−24

DWS, Sanitation, D.o.W.a. (2016). *Resource Directed Measures: Reserve determination study of selected surface water and groundwater resources in the Usutu/Mhlathuze Water Management Area.* Kosi Estuary Rapid Environmental Water Requirements Determination. Report produced by CSIR. Report no: RDM/WMA6/CON/COMP/141

Eklof, J.S., de la Torre-Castro, M., Gullström M., Uku, J., Muthiga, N., Lyimo, T. and Bandeira, S.O. (2008). Sea urchin overgrazing of seagrasses: A review of current knowledge on causes, consequences, and management. *Estuar. Coast. Shelf Sci.* 79, pp. 569–580

Erftemeijer, P.L.A., Djunarlin, L.A. and Moka, W. (1993). Stomach Content Analysis of a Dugong (*Dugong dugon*) from South Sulawesi. *Aust. J. Mar. Freshwater Res.* 44, pp. 229–233

ESA Policy Guidance Report (2009). Policy Guidance for Environmentally Sensitive Areas (ESA) in Mauritius and Rodrigues Report 2009

Ferreira, M.A. and Bandeira, S. (2014). Maputo bay´s coastal habitats. In *The Maputo Bay Ecosystem* (eds. Bandeira, S. and Paula, J.), pp 21–24. WIOMSA, Zanzibar Town

Forestry, D.o.W.A.a. (2006). *Olifants/Doring catchment ecological water requirements study.* RDM Report on Estuarine Component. Project Number: 2002–377. CSIR Report No. ENV-S-C 2005-096 A

Froneman, P.W. and Henninger, T.O. (2009). The influence of prolonged mouth closure on selected components of the hyperbenthos in the littoral zone of the temporarily open/closed Kasouga Estuary, South Africa. *Estuar. Coast. Shelf Sci.* 83, pp. 326–332

Gallegos, M.E., Marba, N., Merino, M. and Duarte, C.M. (1992). Flowering of *Thalassia testudinum* in the Mexican Caribbean: age-dependence and inter-annual variability. *Aquat. Bot.* 43, pp. 249–55

Githaiga. M.N., Kairo, J.G., Gilpin, L. and Huxham, M. (2017). Carbon storage in the seagrass meadows of Gazi Bay, Kenya. *PLoS ONE* 12(5): e0177001. https://doi.org/10.1371/journal.pone.0177001

Green, E.P. and Short, F.T. (2003). *World atlas of seagrasses*. Berkeley: University of California Press

Gullström, M., de la Torre Castro, M., Bandeira, S.O., Björk, M., Dahlberg, M., Kautsky, N., Rönnbäck, P. and Öhman, M. C. (2002). Seagrass Ecosystems in the Western Indian Ocean. *Ambio* 31(7–8), pp. 588–596. https://doi.org/10.1579/0044-7447-31.7.588

Gullström, M., Lyimo, L.D., Dahl, M., Samuelsson, G.S., Eggertsen, M., Anderberg, E., Rasmusson, L.M., Linderholm, H.W., Knudby, A., Bandeira, S., Nordlund, L.M. and Björk, M. (2017). Blue Carbon Storage in Tropical Seagrass Meadows Relates to Carbonate Stock Dynamics, Plant–Sediment Processe`\s, and Landscape Context: Insights from the Western Indian Ocean. *Ecosystems*. https://doi.org/10.1007/s10021-017-0170-8

Harcourt, W.D., Briers, R.A. and Huxham, M. (2018). The thin(ning) green line? Investigating changes in Kenya's seagrass coverage. *Biol. Lett.* 14, p. 20180227. http://dx.doi.org/10.1098/rsbl.2018.0227

Hedberg, N., von Schreeb, K., Charisiadou, S., Jiddawi, N.S., Tedengren, M. and Nordlund, L.M. (2018). Habitat preference for seaweed farming – A case study from Zanzibar, Tanzania. *Ocean Coast. Manage.* 154, pp. 186–195

Hemminga, M.A., Harrison, P.G. and Lent, F. (1991). The balance of nutrient losses and gains in seagrass meadows. *Mar. Ecol. Prog. Ser.* 71, pp. 85–96

Human, L.R.D., Adams, J.B. and Allanson, B.R. (2016). Insights into the cause of an *Ulva lactuca* Linnaeus bloom in the Knysna Estuary. *S. Afr. J. Bot*. 107, pp. 55–62

Jafta, N. (2011). The botanical importance and health of the Bushmans Estuary, Eastern Cape, South Africa. Department of Botany, Nelson Mandela Metropolitan University, Port Elizabeth, South Africa

JICA (2015). The Project for Capacity Development on Coastal Protection and Rehabilitation in the Republic of Mauritius. Final Report. Coastal Conservation Plan. (Volume 2). 286 pp

Katwijk, M.M., Meier, N.F., van Loon, R., van Hove, E.M., Giesen, W.B.J.T, van der Velde, G. and 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, pp. 675–683

KWS (2013). Seagrass and coral reef ecosystems conservation strategy 2014–2018. Kenya Wildlife Service (KWS), Nairobi, 58 pp

Lockyear, J.F., Hecht, T., Kaiser, H. and Teske, P.R. (2006). The distribution and abundance of the endangered Knysna seahorse *Hippocampus capensis* (Pisces: Syngnathidae) in South African estuaries. *Afr. J. Aquat. Sci.* 31, pp. 275–283

Lugendo B.R., Mgaya, Y.D. and Semesi A.K. (1999). The seagrass and associated macroalgae at selected beaches along Dar es Salaam coast. In *Marine Science Development in Tanzania and Eastern Africa*. (eds. Richmond, M.D. and Francis, J.), pp. 359–373. Proceedings of the 20th Anniversary Conference on Advances inn Marine Science in Tanzania. 28 June–1 July 1999, Zanzibar, Tanzania

Maina, J.M., Jones, K.R., Hicks, C.C., McClanahan, T.R., Watson, J.E.M., Tuda, A.O. and Andréfouët, S. (2015). Designing Climate-Resilient Marine Protected Area Networks by Combining Remotely Sensed Coral Reef Habitat with Coastal Multi-Use Maps. *Remote Sens.* 7, pp. 16571–16587. https://doi.org/10.3390/rs71215849

Mattio, L., Zubia, M., Maneveldt, G.W., Anderson, R.J., Bolton, J.J., de Gaillande, C., De Clerck, O. and Payri, C.E. (2016). Marine flora of the Iles Eparses (Scattered Islands): A longitudinal transect through the Mozambique Channel. *Acta Oecologica* 72, pp. 33–40

McClanahan, T.R. (2000). Recovery of a coral reef keystone predator, *Balistapus undulates*, in East African marine parks. *Biol. Conserv.* 94, pp. 191–198

McClanahan, T.R., Nugues, M. and Mwachireya, S. (1994). Fish and sea-urchin herbivory and competition in Kenyan coral-reef lagoons – the role of reef management. *J. Exp. Mar. Biol. Ecol.* 184, pp. 237–254

Mutisia, L.N.D. (2009). Restoration of Kenyan seagrass beds: a functional study of the associated fauna and flora. MSc. Thesis. Vrije Universiteit Brussel (VUB), Universiteit Antwerpen (UA) 94 pp

NBA (2012). An assessment of South Africa's biodiversity and ecosystems. National Biodiversity Assessment (NBA)

Nordlund, L.M. and Gullström, M. (2013). Biodiversity loss in seagrass meadows due to local invertebrate fisheries and harbour activities. *Estuar. Coast. Shelf Sci.* 135, pp. 231–240

Nordlund, L.M., Unsworth, R.K.F., Gullström, M. and Cullen-Unsworth, L.C. (2017). Global significance of seagrass fishery activity. *Fish and Fisheries*, pp. 1–14. https://doi.org/10.1111/faf.12259

Obura, D.O., Di Carlo, G., Rabearisoa, A. and Oliver, T. (eds.) (2011). *A Rapid Marine Biodiversity Assessment of the coral reefs of northeast Madagascar.* RAP Bull Biol Assessment 61 Conservation International Arlington, VA. 110 pp

Ochieng, C.A. and Erftemeijer, P.L.A. (2003). Seagrasses of Kenya and Tanzania. In *World Atlas of Seagrasses* (eds. Green, E.P. and Short, F.T.), pp. 82–92. University of California press, Berkeley, Los Angeles, London

Pillay, D., Branch, G.M., Griffiths, C.L., Williams, C. and Prinsloo, A. (2010). Ecosystem change in a South African marine reserve (1960–2009): role of seagrass loss and anthropogenic disturbance. *Mar. Ecol. Prog. Ser.* 415, pp. 35–48

Poonian, C.N.S., Tuharska, K. and Hauzer, M.D. (2016). Diversity and distribution of seagrasses in the Union of the Comoros, *Afr. J. Mar. Sci.* 38(2), pp. 263–268, https://doi.org/10.2989/1814232X.2016.1181103

Rocliffe, S., Peabody, S., Samoilys, M. and Hawkins, J. P. (2014). Towards a network of Locally Managed Marine Areas (LMMAs) in the Western Indian Ocean. *PLoS ONE* 9(7)

Samoilys, M., Osuka, K., Muthiga, N. and Harris, A. (2017). Locally managed fisheries in the Western Indian Ocean: a review of past and present initiatives, iv + 40p. WIOMSA. WIOMSA Book Series 17

Schmidt, J. (2013). Influence of predicted sea-level rise on the salt marsh of the Swartkops, Kromme and Knysna Estuaries, Department of Botany. Nelson Mandela Metropolitan University, Port Elizabeth, South Africa

Sea Sense (2012). *Community Based Conservation of Threatened Marine Species and Habitats in Tanzania.* Annual Report. 40 pp

Sheppard, J.N., James, N.C., Whitfield, A.K. and Cowley, P.D. (2011). What role do beds of submerged macrophytes play in structuring estuarine fish assemblages? Lessons from a warm-temperate South African estuary. *Estuar. Coast. Shelf Sci.* 95, pp. 145–155

Sifleet, S., Pendleton, L. and Murray, B.C. (2011). State of the science on coastal blue carbon: A Summary for policy makers. Nicholas Institute for Environmental Policy Solutions. School of the Environment, Duke Univ.

Taljaard, S. and Key Specialists (2006). RDM report on estuarine component: Olifants/Doring Catchment Ecological Water Requirements Study. Prepared by CSIR for Department of Water Affairs and Forestry

Trégarot, E., Failler, P. and Maréchal, J-Pe. (2017). Evaluation of coastal and marine ecosystem services of Mayotte: Indirect use values of coral reefs and associated ecosystems. *Int. J. Biodivers. Sci. Ecosyst. Serv. Manag.* 13(3), pp. 19–34, https://doi.org/10.1080/21513732.2017.1407361

Turner, J.R. and Klaus, R. (2005). Coral Reefs of the Mascarenes, Western Indian Ocean. *Philos. Trans. Royal Soc.* 363, pp. 1–22

UNEP (1998). *Eastern Africa Atlas of Coastal Resources:* Kenya, 119 pp

UNEP (2002). L'Afrique Orientale Atlas des Ressources Côtières: République Fédérale Islamique des Comores. Nairobi, UNEP

UNEP/Nairobi Convention Secretariat (2009). Transboundary Diagnostic Analysis of Land-based Sources and Activities Affecting the Western Indian Ocean Coastal and Marine Environment. Nairobi, Kenya

UNEP-Nairobi Convention and WIOMSA (2015). The Regional State of the Coast Report: Western Indian Ocean. UNEP and WIOMSA, 546 pp. ISBN: 978-9987-9559-7-8. Nairobi, Kenya

UNEP-Nairobi Convention/WIOMSA (2020). Guidelines for Seagrass Ecosystem Restoration in the Western Indian Ocean Region. UNEP, Nairobi, 63 pp

Unsworth, R.K.F., De León, P.S., Garrard, S.L., Jompa, J., Smith, D.J. and Bell, J.J. (2008). High connectivity of Indo-Pacific seagrass fish assemblages with mangrove and coral reef habitats. *Mar. Ecol. Prog. Ser.* 353, pp. 213–224

Van Der Linden, S.C. (2014). Salt marsh distribution and structure at Langebaan Lagoon, Department of Botany. Nelson Mandela Metropolitan University, Port Elizabeth, South Africa

Vicente, I.E. and Bandeira, S. (2014). Socio-economic aspects of gastropod and bivalve harvest from seagrass beds – comparison between urban (disturbed) and rural (undisturbed) areas. In *The Maputo Bay Ecosystem* (eds. Bandeira, S. and Paula, J.), pp. 329–334. WIOMSA, Zanzibar Town

Wallner-Hahn, S., de la Torre-Castro, M., Eklof, J.S., Gullström, M., Muthiga, N.A. and Uku, J. (2015). Cascade effects and sea-urchin overgrazing: An analysis of drivers behind the exploitation of sea urchin predators for management improvement. *Ocean Coast. Manage.* 107, pp. 16–27

Waycott, M., Duarte, C.M., Carruthers, T.J.B., Orth, R.J., Dennison, W.C., Olyarnik, S., Calladine, A., Fourqurean, J.W., Heck, K.L., Hughes, R., Kendrick, G., Kenworthy, W.J., Short, F.T. and Williams S.L. (2009). Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proc. Natl. Acad. Sci. U. S. A.* 106, pp. 12377–12381 https://doi.org/10.1073/pnas.0905620106

Waycott, M., McMahon, K., Mellors, J., Calladine, A. and Kleine, D. (2004). A guide to tropical seagrasses of the Indo-West Pacific. Townsville, James Cook University. 72 pp

Whitfield, A. and Elliott, M. (2011). Ecosystem and Biotic Classifications of Estuaries and Coasts. In *Treatise on Estuarine and Coastal Science* (eds. Wolanski, E. and McLusky, D.S.) Volume 1, pp. 99–124. Waltham: Academic Press

Whitfield, A.K. and Bruton, M.N. (1996). Extinction of the river pipefish *Syngnathus watermeyeri* in the Eastern Cape Province. South Africa. *S. Afr. J. Zool.* 92, pp. 59–61

WWF (2017). Conservation status assessment of the sea-grass meadows of Primeiras and Segundas Environmental Protected area Nampula & Zambézia provinces, North of Mozambique. 80 pp

CRITICAL HABITATS

SALT MARSHES

Janine Adams and Salomão Bandeira

Salt marsh vegetation along a tidal gradient, Langebaan Lagoon, South Africa. © Janine Adams

BACKGROUND

Salt marshes are defined as "areas, vegetated by herbs, grasses or low shrubs, bordering saline water bodies. They are subjected to periodic flooding as a result of fluctuations (tidal or non-tidal) in the level of the adjacent water body" (Adam, 1990). Salt marshes occur in arctic and temperate regions, as well as in the subtropics and tropics where they occur in areas where mangrove development is precluded, or as a component of a salt marsh-mangrove ecotone (Adam, 1990). Macnae (1969) stated that in drier regions where rainfall is seasonal, the lower tidal range is occupied by mangroves and the higher shoreline by salt marsh. This is supported by the distribution of salt marshes in the tropics and northern Africa where this habitat replaces mangroves where the sediment is too dry or saline. In the subtropical estuaries of South Africa mangroves occupy the lower tidal zone and salt marshes occur in the higher, drier areas (Adams et al., 2016). Sometimes this is a very narrow band and therefore difficult to map as a separate habitat.

In addition, according to Adam (1990), on arid or strongly seasonal tropical / subtropical coasts salt marshes may not extend upwards to the highest tide level but may be fringed on their landward side by extensive hypersaline flats, known in the Middle East as *sabkha*. These areas exist in the Western Indian Ocean (WIO) region but have not been described in detail. In Africa salt marshes are described for the Mediterranean, north-west, south and south-west coasts.

Mcowen et al. (2017) defined salt marshes as "tidal communities that comprise the upper, vegetated portion of intertidal mudflats, lying approximately between mean high water neap tides and mean high water spring tides". Other global definitions of salt marshes also only consider tidal systems (eg Weis and Butler, 2009). However, in South Africa the seldom flooded supratidal habitat that has halophytic species has been included as salt marsh and is considered a component of the estuary. The 5 m topographical contour is used to demarcate the estuarine functional zone (EFZ) and the lateral boundaries.

Supratidal salt marsh occurs at > 1.5 m above mean sea level (amsl) and an ecotone with terrestrial species can occur from > 2.5 m amsl (Veldkornet et al., 2015). The supratidal salt marsh may only be flooded twice a year during exceptional spring tide events. Other studies refer to this as the high marsh (Boorman, 2003). Thus, a description of the distribution of salt marsh is influenced by its definition. This has not been well described for the WIO region and information on this important habitat is lacking, posing a threat to conservation efforts.

Salt marsh plants are typically succulents and grasses with common species including *Bassia diffusa*, *Sarcocornia mossambicensis*, *Sarcocornia natalensis*, *Suaeda* spp., *Sporobolus virginicus*., *Salicornia* spp., *Sesuvium portulacastrum* and *Juncus kraussii*. Some of these species (eg *Sarcocornia natalensis*) extend from South Africa to the lower regions of Mozambique and Madagascar. Species common in Mozambique salt marshes are *Arthrocnemon* sp., *Pemphis acidula*, *Portulaca oleracea*, *Salicornia* sp., *Sporobolus virginicus*, *Suaeda monoica* and *Suriana maritima*. Some authors consider *Pemphis* as a member of the mangrove community.

In Somalia, as well as in North West Africa and South East Asia, species such as the halophytic grass *Urochondra setulosa* occur in salt marshes (Khan et al., 2009). *Salicornia virginica* is also common (Carbone and Accordi, 2000). In Tanzania and Madagascar sea purslane *Sesuvium partulacastrum* and *Suaeda monoica* occur on soils too saline for mangrove species. The annual *Salicornia pachystachya* occurs from Kenya to South Africa, Madagascar and other Indian Ocean islands in salt marsh associated with mangrove swamps. It is clear that salt marsh species occur widely in the WIO although the overall habitat is poorly described.

Mangroves bordered by salt marsh at the Nxaxo River estuary, South Africa. © Janine Adams

DISTRIBUTION

Distribution is discussed in this section for only some of the WIO countries, based on available information.

Mozambique

Salt marshes occur in most areas adjacent to mangroves and estuaries, but some occur further inland associated with the brackish lakes of southern Mozambique. Salt marshes are extensive in lowland areas and in areas with a wide tidal range with limited freshwater drainage or seepage. Large salt marshes are common in Maputo Bay, the Limpopo Estuary and areas around the estuaries of Cabo Delgado and Nampula provinces (Table 1). They also occur on the Changane River, a tributary of the Limpopo, where salt tolerant species such as *Suaeda* sp. and *Salicornia* sp. are found at Chibuto village near Xai-Xai town.

In Maputo Bay salt marsh occurs in extensive areas between mangroves and terrestrial vegetation; generally in degraded mangroves and around coastal lakes (UNEP and WIOMSA, 2015). Maputo Bay is surrounded by mangroves, but extensive areas, especially in the northern Maputo city regions of Bairro dos Pescadores, Mapulene and Muntanhana, are covered by salt marshes. Unfortunately, these areas have been heavily impacted by housing expansion.

The Limpopo River Estuary supports extensive areas with grassy *(Sporobolus virginicus)* salt marshes. River flooding in 2000 halved the original mangrove areas, thus enabling salt marsh colonization (Bandeira and Balidy, 2016).

Lowland areas between the sea and mangroves near Inhambane Town are colonized by salt marsh. The small Chiveve River, that runs through Beira City in central Mozambique, supports extensive salt marshes, especially around the Beira Golf Club, and dominant species include *Phragmites australis*, *Sporobolus virginicus*, *Cynodon dactylon*, *Panicum maximum*, *Pennisetum* sp., *Cyperus compressus* and *Urochloa mosambicensis*. Quelimane (on the northernmost arm of the Zambezi Delta) has extensive grassland areas adjacent to mangroves, especially around the Chuabo Dembe area. Some salt marsh species occur in the ecotone between the mangrove species and the grasses with species such as *Sporobolus* sp., *Sesuvium portulacastrum*, *Arthrocnemum* sp. and *Salicornia* sp. present. Furthermore, it has been reported that new areas are being covered by sea water, thus creating more habitat for colonization by mangroves and salt marshes in the outer areas. There are also examples of salt marshes occurring in abandoned salt pans in areas near Quelimane, with adjacent areas such as Mirazane being used for rice farming.

Table 1: Known salt marshes in Mozambique and estimates of area, habitat trend (increasing, decreasing, stable) and protection status.

LOCATION	REGION /PROVINCE	ESTIMATE OF SIZE (ha)	HABITAT TREND	PRESSURES	PROTECTION STATUS
Montepuéz River Estuary	Cabo Delgado	Unknown	S	Agriculture	Quirimbas National Park
Paquitequete, Pemba Town	Cabo Delgado	< 10	D	Development	None
Mecúfi	Cabo Delgado	> 50	D	Salt plans, road upgrade	None
Memba northern side of Mecuburi Estuary	Nampula	Unknown	S	Salt pans	None
Nacala Bay	Nampula	Unknown	D	Port development, town expansion	None
Quelimane	Zambezia	> 80	I	Development, rice production	None
Cabo São Sebastião	Inhambane	Unknown	S		Area of total protection (privately managed)
Limpopo Estuary	Gaza	> 300	S	Flooding	Community managed wider mangrove area
Maputo City	Maputo	> 500	D	Urban development	None

Figure 1: The Montepuez river delta leading to Montepuez Channel, Mozambique. The surrounding outer mangrove channels depicts salt marshes. Source: GoogleEarth.

In northern Mozambique salt marshes occur in several places, but information is scarce. Salt marshes have been observed in Nacala Bay (Nampula), in areas around the Memba mangroves (particularly near the northern side of the Mecuburi Estuary), between Pemba Bay and Mecufi, and in Pemba Town, especially in the lowland area of Paquitequete. Some of the dominant species here include *Suaeda monoica*, *Suriana maritima*, *Pemphis acidula* and *Sporobolus* sp. Freshwater reeds and sedges also occur around some of these areas. The delta of the Montepuez River in northern Mozambique (Fig. 1) has extensive salt marshes. Fieldwork is needed to verify the diversity and extent of these habitats within the Montepuez Channel, facing the southern Quirimbas Archipelago.

Saline lagoons and lakes, situated within a short distance from the coastline, are a common feature in southern Mozambique (south of Vilanculos region). From a geomorphological perspective these are mostly Pleistocene formations and are brackish or near freshwater. The vegetation in and around these lakes is dominated by *Hibiscus tiliaceus*, *Phoenix reclinata*, *Phragmites australis*, *Ruppia maritima* and *Typha latifolia*. The dominant succulent is *Portulaca oleracea*, though not all areas surrounding these lakes qualify as salt marshes.

Some of these areas lie adjacent to the Ecologically or Bio-logically Significant Area (EBSA) of Save-São Sebastian region (encompassing Bazaruto Archipelago and some inland swamps). Salt marsh habitat has also been identified in the Incomati-Ponta de Ouro EBSA that encompasses Maputo Bay, Lingamo (near Maputo and Port) and Inhaca Island, and in some areas near Saco da Inhaca.

Madagascar

The western and northern parts of Madagascar support large mangrove areas. Intertidal areas are generally wide and salt marsh occurs on the landward side of mangrove stands. Salt marshes occur near Tsangajoly, Toliara Province in south-western Madagascar, while salt pans and associated marsh habitat also exist at Morondava on the central west coast. Mudflats of up to 1 km wide occur together with mangroves, lakes and salt marshes in the Ambavanankarana wetlands along the north-western coast (Marnewick et al., 2015). Similarly, in the Mahavavy delta in the Mahajanga region, mangroves occupy an area of 16 000 ha, and mudflats 5200 ha.

The bay has 7500 ha of mangrove inlets, mudflats and marshes. Given the extent of mangroves and associated lowland areas in Madagascar, it is expected that salt marshes exist in adjacent areas. However, there is a need for detailed mapping and distribution studies to verify the full extent of salt marshes in the country.

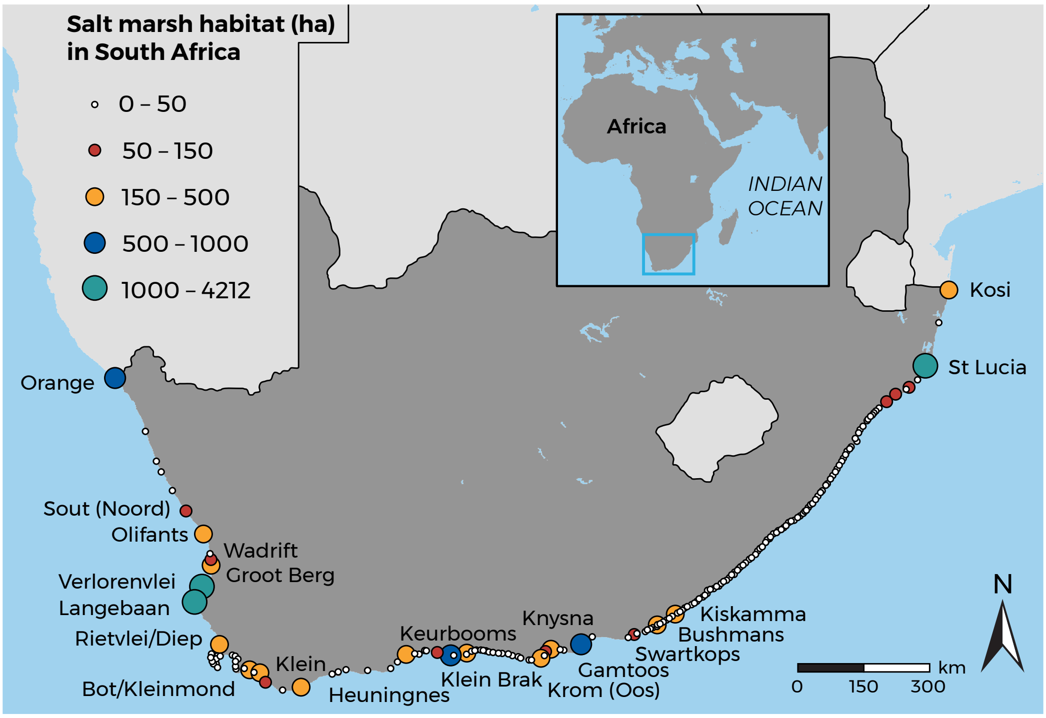


Figure 2: Distribution of salt marsh habitat in South African estuaries.

South Africa

In South Africa intertidal salt marsh occurs below the mean spring high water mark, with supra-tidal salt marsh above this. Salt marsh plants are distributed according to inundation and salinity gradients, often resulting in distinct zonation patterns particularly in permanently open estuaries with large tidal ranges (Adams et al., 2016).

Salt marsh occurs in the sheltered estuaries across all four geographic zones (cool temperate, warm temperate, subtropical and tropical) with a total area of 14 437.2 ha (Fig. 2, Table 2). The Berg Estuary on the west coast supports the largest salt marsh area in the country (4212 ha).

Table 2: Area of salt marsh habitat types (ha) across bioclimatic regions in South Africa.

	COOL TEMPERATE	WARM TEMPERATE	SUB-TROPICAL	TOTAL AREA
Intertidal salt marsh	2476.5	1418.9	217.4	4112.8
Supratidal salt marsh	6611.7	2570.3	1131.1	10313.1
Total area (ha)	9088.2	3989.2	1348.5	14425.9

IMPORTANCE OF SALT MARSHES

Salt marshes are productive ecosystems important for carbon storage, water purification, flood control, refugia, and habitat for other organisms such as juvenile fish (Paterson and Whitfield, 2003; Barbier et al., 2011; Tabot and Adams, 2013). They also serve as critical habitat for migratory fish and birds. Juvenile fish make use of the intertidal salt marsh habitat as refugia, with a high density of fish larvae and juveniles found on the marsh-creek edge (Whitfield, 2017). Marsh hydroperiod and vegetation stem density is important in determining fish access and invertebrate prey abundance (Rozas, 1995). The type of vegetation is also important as *Spartina* (grass) and *Sarcocornia* (succulent) species offer different food sources and structural protection (Hettler, 1989, cited in Whitfield, 2017). Recent studies that have described the importance of South African estuaries as a nursery habitat are Becker et al. (2011), Whitfield and Pattrick (2015), and Edworthy and Strydom (2016). Earlier studies such as Booth (2007) found that juvenile estuarine dependant fish made up 83 per cent of all fish sampled in the intertidal salt marsh under flooded conditions (Kariega Estuary, South Africa).

Salt marshes serve as important habitats for birds in terms of breeding, roosting and feeding. Birds such as herons, gulls, waders, terns and cormorants (and summer migrant waders that make up 90 per cent of the salt marsh bird community) prey on prawns, marsh crabs and other invertebrates, pencil bait and fish. Salt marshes also provide important high tide and overnight roosting areas and secondary feeding habitat (Saintilan et al., 2018). In Southern Africa salt marshes, more than 100 migratory bird species have been recorded. At least 20 of the 112 IBAs of South Africa support salt marsh habitat. The Berg River estuary on the west coast supports in the region of 8000 migrant waders. The peak number of birds is dependent on the productivity of the marshes, with the highest density recorded over the summer period.

The urbanized Swartkops Estuary and surrounding salt pans on the south-east coast of South Africa support high bird numbers with an average of 14 500 birds per year (Birdlife South Africa, 2015a). Of these birds, at least 4000 are Palearctic migrants, present mainly in summer. Threatened species frequenting the intertidal mudflats and salt marshes of the Swartkops Estuary include the Damara Tern *Sterna balaenarum*, African Black Oystercatcher, Cape Cormorant *Phalacrocorax capensis*, Greater and Lesser Flamingo, Martial Eagle *Polemaetus bellicosus*, and Chestnut-banded Plover *Charadrius pallidus*.

The rich mud of salt marshes supports dense populations of mollusks and crustaceans. In South Africa, small gastropods called marsh snails such as *Assiminea ovata*, *A. globulus* and *A. bifasciata* are common detritus feeders in salt marshes, occurring in very high numbers around the high-water mark (Wooldridge, 1998). A few crab species (*Paratylodiplax edwardsii*, *P. algoense* and *Sesarma catenata*) are also abundant in salt marshes but are generally confined to *Spartina maritima* stands.

Various reptiles forage in the salt marsh and fringing terrestrial habitat including snakes, lizards and tortoises, as well as frogs. Small and larger mammals also make use of this habitat, however not exclusively. Some endemic chameleons occur in the fringing vegetation of salt marshes such as the vulnerable Cape dwarf chameleon *Bradypodion pumilum* found at Rietvlei Estuary in Cape Town, South Africa, and the southern dwarf chameleon *Bradypodion ventrale* endemic to the Swartkops Estuary in Port Elizabeth, South Africa (Birdlife South Africa, 2015a; Birdlife South Africa, 2015b). Gronovi's dwarf burrowing skink *Scelotes gronovii*, Kasner's dwarf burrowing skink *S. kasneri*, and the large-scaled girdled lizard *Cordylus macropholis* are endemic to South Africa's west coast and occur in the xeric salt marsh of Langebaan Lagoon (Bird Life South Africa, 2015c). Peringuey's Leaf-toed Gecko (*Cryptactities peringueyi*) is the only gecko in the world that lives in salt marshes. It is only known from the Kromme River estuary and a few sites near Port Elizabeth, South Africa (Adams, 2020).

Besides grazing by livestock there is little direct use of salt marsh as a food source (see Box opposite). In Mozambique, the leaves of the succulent herb *Sesuvium portulacastrum* are added to soups and salads and used for a traditional plant dish only cooked for Christmas (also known as *mpfixiri*) (Célia Macamo pers. comm.). Seablight *Suaeda monoica* grows amongst mangroves, and its young leaves can be picked and eaten in salads or cooked in curries.

(Left) Samphire (*Salicornia*) sold as a vegetable at Woolworths South Africa. © Woolworths. **(Top) Sea purslane *Sesuvium portulacastrum* occurs worldwide and is harvested from the wild to be eaten as a vegetable. It also has ornamental and medicinal value and is used as a ground cover to prevent erosion in dune vegetation.** © Forest & Kim Starr

CASE STUDY

Economic potential of salt marsh plants

Taryn Riddin

Approximately 25 per cent of the world's soils are too saline for cultivation. As this value increases annually, alternate agricultural practices are being investigated. With the growing scarcity of freshwater, halophytes provide a viable solution for the production of food, fibre and fodder.

Salicornia brachiata **plants irrigated with seawater in Bhavnagar, India.** © Muppala P. Reddy

The salt marsh genera *Salicornia* and *Sarcocornia* have extreme salt tolerances (up to 500 mM) and have had a long record of use in the wild, both for food and medicinal uses. They are sought after for their salty nature, palatability, digestibility and high nutritional value. Recent studies have shown that halophytes can yield as much as conventional crops on a commercial scale, even when irrigated with seawater. They can be cultivated under nets or in greenhouses. Halophytes harvested in the wild include species such as Sea Spinach (*Aster tripolium*), a Northern European plant found in salt marshes and estuaries. During times of famine in the Netherlands it was eaten, these days it is considered a delicacy. *Salicornia europaea*, known by a variety of names from Sea Beans, Sea Asparagus to Samphire, is eaten in salads, steamed, boiled or sautéed. In South Africa it is now being sold in shops as gourmet food. Another South African species *Tetragonia decumbens* is harvested for use in traditional cooking as a spinach alternative.

Halophytes have also been harvested for the production of soda ash for glass and soap making, including *Suaeda*, *Salicornia*, *Salsola*, and *Haloxylon*. They are also effective biofilters, removing up to 90 % of nutrients from saline aquaculture waste. Genera used include *Suaeda*, *Salicornia*, *Sarcocornia*, *Atriplex*, *Tamarisk* and *Portulaca*. The added benefit of this practice is the production of seed oil for biofuel, for example from *Salicornia bigelovii*, as well as for human and animal consumption. Whereas the plant accumulates salt, the seeds do not. *Salicornia bigelovii* yields up to 2 t/ha of seed which contain 28 per cent oil and 31 per cent protein.

In Eritrea *Salicornia bigelovii* was grown and irrigated with seawater as part of an integrated aquaculture project; an Initiative by Seawater Foundation for Greening Eritrea and the brainchild of scientist Carl Hodges. The nutrient rich outflow from a shrimp farm passed through a fish farm and then through 250 acres of *Salicornia* plantations. Finally, water soaked through a mangrove forest into the sea. The project was unfortunately shutdown due to various socio-political reasons. A similar project in Adu Dhabi investigated intercropping of *Salicornia* and mangroves for the production of renewable jet fuel.

Recent studies in Spain have shown the potential of growing *Salicornia biglovii* with Seabass in marine aquaponics, the combination of hydroponics and aquaculture. This eliminates the leaching of salts and nutrients into freshwater aquifers. Plants are either grown in sand or in floating rafts on top of the fish tanks. Grown in this manner, the plant was nutritionally better than the wild harvested plants. There was the additional benefit of the farmed fish as a food protein source. No fertilizer is used and there is a complete re-use of water compared to the example in Eritrea. Production values of 5 kg/ha has been estimated.

All these applications need investigation and implementation in the WIO region.

The grass *Sporobolus virginicus* can be used for stabilization of wind-eroded shorelines, stream banks and road slopes. Traditionally, it has been used to relieve urinary tract and throat irritation (Fern, 2014). In some countries Samphire *Sarcocornia* and *Salicornia* spp. as well as *Aster tripolium* are sold at comparatively high prices as sea vegetables and salad crops. The South American seed crop quinoa (Chenopodium quinoa) has gained popularity among the health conscious for its highly nutritional seeds (Ventura et al., 2014). The common ice plant *Mesembryanthemum crystallinum* is a salt tolerant succulent native to southern and eastern Africa. It is consumed as a vegetable crop in India, California, Australia, New Zealand and in some countries of Europe (Herppich et al., 2008). It is also used medicinally for its demulcent, diuretic, and antiseptic effects (Bouftira et al., 2012; Ksouri et al., 2008), and as protection from the sun (Bouftira et al., 2008).

Table 3: Total salt marsh area in South Africa and changes in area for specific estuaries.

HABITAT	PAST (ha)	PRESENT (ha)	CHANGE (ha) & %
Intertidal salt marsh	5373.8	4170.8	1203 (22 % loss)
Supratidal salt marsh	15 078.6	10 542	4536.6 (30 % loss)
ESTUARY	PAST (ha)	PRESENT (ha)	LOSS (ha)
Berg	4891	3488	1403
Gamtoos	779	177.1	601.9
Orange	1465	771	694

There is potential to utilize salt marsh species for the production of bioenergy. Many of these species produce high quantities of oil producing seeds and lignocellulosic biomass suitable for biofuel production. Promising genera include *Salicornia* (glasswort), *Suaeda* (sea-blite), *Atriplex* (saltbush), arid salt grass *Distichlis* and the succulent-leaved ground cover *Batis*. Halophyte-based agriculture is beneficial in that marginalized land and saline water from the sea are used and limited inputs are required. Halophyte agriculture has the potential to reclaim salinized habitats in the Sahara desert, Western Australia and Southern Africa (Sharma et al., 2016). In terms of recreation and tourism, scenic views and vistas are created by salt marshes and this is desirable in the coastal residential property market. For example, properties on Thesen Island on the Knynsa River estuary, South Africa, are marketed for their salt marsh views.[1]

THREATS AND STATUS

In South Africa, salt marshes are threatened by sea level rise at the sea interface, and development at the land interface (Adams, 2020). Other threats are salinization due to upstream water abstraction as well as changes in rainfall patterns. There has been an estimated loss of ~6000 ha of salt marsh in South Africa due to construction of bridges, causeways and jetties in the intertidal salt marsh habitat, and agriculture and development in the supratidal area (Table 3).

Agriculture within the floodplain of the Berg and Gamtoos river estuaries has resulted in the loss of the largest supratidal salt marsh habitat in the country. Roughly half of the natural salt marsh habitat from the Berg and nearly 90 per cent from the Gamtoos Estuary has been lost due to vegetable cultivation and cattle grazing. The Orange River estuary, a Ramsar wetland of special concern, is the boundary between South Africa and Namibia. It was placed on the Montreux Record in 1995 because 300 ha of salt marsh had become desertified. This loss was attributed to leakage of diamond mine water, the effect of windblown dried slimes dam sediment on the marsh vegetation, construction of flood protection structures and a beach access road, and the elimination of tidal exchange into the wetland due to a causeway constructed at the river mouth (Shaw et al., 2008). Due to the low rainfall on the west coast and highly salinized nature of the desertified marsh area there has been little change in the salt marsh status over the last 10 years.

Other stressors that have been described but not quantified in terms of loss of salt marsh area include salinization and desiccation due to upstream freshwater abstraction (Bornman et al., 2002; Bornman et al., 2004). Reduced freshwater inflow causes extended mouth closure of temporarily open/closed estuaries, and inundation and flooding of the salt marsh (Riddin and Adams, 2010; Riddin and Adams, 2012). In urbanized estuaries, salt marsh loss is related to a restriction of tidal exchange, freshening, and invasion by alien invasive plants (O'Callaghan, 1990). Eutrophication, macroalgal blooms and smothering of salt marsh is a growing concern in South African estuaries (Nunes and Adams, 2014; Human et al., 2016). In the rural areas, livestock browsing and trampling of the salt marsh is extensive but unquantified.

Mozambique's salt marshes are threatened by transformation for human settlement through the expansion of

[1] http://www.thesenislandsliving.co.za/

the urban areas of Maputo, Chiveve and Paquitequete in Pemba. The Maputo population, especially within the Costa de Sol and Muntanhana region, has been expanding resulting in the conversion of the nearby salt marshes and adjacent dwarf mangroves. Urbanization and an increase in freshwater run-off also threaten the salt marshes. In the past, and to some extent today, salt pans pose a threat to salt marshes. Small scale rice production impacts salt marshes near Quelimane. Beira, the second largest town in Mozambique, is built in a former swamp and salt marsh area. This city has the highest density of man-made channels in eastern Africa. The remaining salt marsh areas are threatened by urban expansion and port development.

In Madagascar the western wetlands are affected by many different pressures, mainly drainage for agriculture and aquaculture. Pond farming often occurs behind the mangrove areas in the salt flats. This results in removal of this habitat, followed by erosion and siltation. The chemical buffering function of this salt marsh / salt flat area cannot be underestimated. In the Mahavavy delta wetlands, mangroves and marshes have been converted to rice fields, birds have been hunted, and mangroves exploited for firewood (Marnewick et al., 2015). The declining population of the Madagascar Marsh Harrier has been attributed to this loss of habitat. Where precipitation is low and salinity is high mangroves degrade and salt marsh develops in its place. This process is known as "tannification" which results in highly saline areas with low herbaceous vegetation (Bosire et al., 2016). This can evolve into completely denuded areas and cover extensive areas of several hundred hectares as observed in the marine protected area of Ambodivahibe.

Salt marshes are susceptible to fluctuations in abiotic conditions due to their unique placement at the interface of the sea and land. Salt marsh plants exhibit unique tolerances to abiotic conditions and thus occur in distinct zonation patterns determined by inundation and salinity gradients (Tabot and Adams, 2013). Thus, the different species will respond to changing climatic conditions in different ways and need to be managed accordingly. Climate change is expected to manifest in salt marshes as increased submergence, changes in salinity and drought. Salt marshes respond to increasing sea level through landward migration, should suitable slopes and potential habitat be available. However, this migration is hindered by "coastal squeeze" – development and hard structures in the floodplain of estuaries preventing landward movement.

In South Africa, reduced freshwater inflow results in the mouth of temporarily closed estuaries remaining closed to the sea for longer periods. An increase in water level and salt marsh submergence causes dieback of salt marsh vegetation, particularly supratidal species that are not adapted to flooding (Riddin and Adams, 2008).

Loss of these species decreases habitat heterogeneity, creating a more homogenous landscape with less niches. Prolonged submergence affects the ability of salt marsh plants to regenerate from seed banks and vegetative propagules. The life cycles of the plants are not completed and over time the seed banks decrease, ultimately resulting in the loss of this habitat.

EXISTING PROTECTION

Table 4 indicates that there is some protection for estuaries with large salt marsh areas in South Africa. In addition, estuary management plans are a requirement of the National Environmental Management: Integrated Coastal Management Act (Act 24 of 2008). These plans can be effective in protecting sensitive habitats such as salt marshes through zonation of activities such as boating that leads to disturbance and erosion.

In Mozambique, swamps, including salt marshes and mangroves, are legally protected under the following legislation:

- The Environmental Impact Assessment Process (Law 54 of 2015) regulates transformation promoted by enterprises.
- The Land Law (Law 19 of 1997) defines areas of total and partial protection. Mangroves and adjacent marshes should be exempt from the allocation of land use deeds.
- The Environment Law (Law 20 of 1997), Art 12 mentions threatened species and Art 13 the need for environmental protection. Chapter 5 refers to environmental licensing and auditing for areas such as mangroves and adjacent salt marshes.

São Sebastião in southern Mozambique is managed as a conservation area, while the Montepuez River estuary and river occur in the Quirimbas National Park. In 2015 the protected area in Chiveve River estuary in Mozambique was increased to assist with flood protection in Beira City, as it occurs in a lowland area. The wetlands are protected as their role in flood protection is recognized. In Madagascar some mangroves are found within existing marine parks such as the Mananara Biosphere Reserve. Here, salt marsh may occur amongst the mangroves and thus receive some protection.

Table 4: Estuaries in South Africa with the largest salt marsh areas showing habitat trends – stable (S), increasing (I), decreasing (D), pressures and protection status.

ESTUARY	INTERTIDAL SALT MARSH	SUPRATIDAL SALT MARSH	HABITAT	PRESSURES	PROTECTION STATUS
Orange	144	627	D	Salinisation	Ramsar site
Olifants	96.6	879	D	Salinisation	None
Berg	1310	2178	D	Agriculture	Partial CapeNature
Langebaan	806	1132	I	Grazing pressure removed	South African National Parks
Heuningnes	16.2	259	D	Agriculture	CapeNature
Knysna	295	221	S	Development	Partial SANParks
Swartkops	192.6	359	D	Development and industry	None
Kariega	36.5	378.7	S	Agriculture and development	None
Keiskamma	189.8	180.5	D	Agriculture and grazing	None
Kosi	58	229	D	Grazing, trampling, fires	ISimangaliso Wetland Park, World Heritage site

PRIORITY OPTIONS FOR CONSERVATION

In South Africa there is a need for formal protection status for the Berg River estuary. The estuary is currently designated as an Important Bird Area (IBA) where the water and intertidal habitat is managed by CapeNature and the local municipality. Restoration of the salt marsh at Orange River mouth is also needed as well as greater protection for the large intertidal salt marshes of Knysna River estuary. In the South African National Estuary Biodiversity Plan habitat targets were set as 20 per cent of the total area of each estuarine habitat type, however this has not been implemented or addressed in any way (Turpie et al., 2012).

Overall, there is little protection of salt marshes in Mozambique and on a local scale management plans are needed, especially in populated areas. The Zambezi River delta is recognized as an IBA and the salt marshes are therefore not protected but candidates for protection status.

RECOMMENDATIONS

Salt marshes in the WIO region form an important buffer between land and intertidal mangrove/estuarine habitats. However, in most countries this habitat has not received priority attention and future research is needed to describe the distribution and status of salt marshes in the region. It should be emphasized that once lost due to salinization and development, there is no buffer to replace salt marshes, and these areas become barren unproductive salt flats.

Remote sensing approaches, supported by field studies, are recommended for identifying and quantifying habitat distribution. In addition, citizen science could be harnessed using cell phones to upload geotagged photos of salt marsh to verify distribution.

Salt marshes represent an opportunity for the study of biodiversity and ecological processes in general, and restoration in particular. These studies need to be linked with wider management initiatives that may include adjacent swamps, mangroves and dune forest. Formal protection status is needed for several estuaries and associated salt marshes in South Africa, for example, while in some cases active restoration needs to take place. Ideally, buffer areas need to be identified to allow for the landward expansion of salt marshes in response to sea level rise, including the removal of hard structures where necessary.

Overall it is apparent that a great deal of work needs to be done on salt marshes to fully understand their distribution and importance at a WIO regional level. Experience from some countries such as South Africa, where significant effort has been placed in mapping and researching salt marshes, needs to be harnessed for the benefit of the other countries of the region.

REFERENCES

Adam, P. (1990). *Salt Marsh Ecology*. Cambridge University Press, Cambridge, UK, 461 pp

Adams, J.B. (2020). Salt marsh at the tip of Africa: Patterns, processes and changes in response to climate change. *Estuar. Coast. Shelf Sci.* 237: 106650. https://doi.org/10.1016/j.ecss.2020.106650.

Adams, J.B., Veldkornet, D. and Tabot, P. (2016). Distribution of macrophyte species and habitats in South African estuaries. *S. Afr. J. Bot.* 107, pp. 5–11

Bandeira, S. and Balidy, H. (2016). Limpopo estuary mangrove transformation, rehabilitation and management. In *Estuaries: a Lifeline of Ecosystem Services in the Western Indian Ocean* (eds. Scheren, P., Salif, D. and Machiwa, J.). Springer, Chapter 14, pp. 227–238

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.M., Stier, A.C. and Silliman, B.R. (2011). The value of estuarine and coastal ecosystem services. *Ecol. Monogr.* 81, pp. 169–193

Becker, A., Cowley, P.D., Whitfield, A.K., Järnegren, J. and Næsje, T.F. (2011). Diel fish movements in the littoral zone of a temporarily closed South African estuary. *J. Exp. Mar. Biol. Ecol.* 406, pp. 63–70

BirdLife South Africa (2015a). Rietvlei Wetland: Table Bay Nature Reserve. https://www.birdlife.org.za/conservation/important-bird-areas/iba-directory/item/253-sa111-rietvlei-wetland-tablebay-nature-reserve. Accessed 19 July 2018

BirdLife South Africa (2015b). Swartkops Estuary-Redhouse and Chatty Salt Pans. https://www.birdlife.org.za/conservation/important-bird-areas/iba-directory/item/238-sa096-swartkops-estuary-redhouse-chatty-saltpans. Accessed 19 July 2018

BirdLife South Africa (2015c). West Coast National Park and Saldanha Bay Islands. https://www.birdlife.org.za/conservation/important-bird-areas/iba-directory/item/247-sa105-west-coast-national-park-and-saldanha-bay-islands. Accessed 19 July 2018

Booth, T.L. (2007). The ichthyofauna associated with Taylor's salt marsh, Kariega Estuary (Eastern Cape), South Africa. PhD Thesis, Rhodes University, p. 117

Boorman, L.A. (2003). Saltmarsh Review. An Overview of Coastal Saltmarshes. Their Dynamic and Sensitivity Characteristics for Conservation and Management. Joint Nature Conservation Committee. JNCC Report No. 334. Peterborough.

Bornman, T.G., Adams, J.B. and Bate, G.C. (2002). Freshwater requirements of a semi-arid supratidal and floodplain salt marsh. *Estuaries* 25(6B), pp. 1394-1405

Bornman, T.G., Adams, J.B. and Bezuidenhout, C. (2004). Adaptations of salt marsh to semi-arid environments and management implications for the Orange river mouth: anthropogenic effects on arid systems. *Trans. R. Soc. South Africa*, 59, pp. 125–131

Bosire, J.O., Mangora, M., Bandeira, S., Rajkaran, A., Ratsimbazafy, R., Appadoo, C. and Kairo, J.G. (2016). *Mangroves of the Western Indian Ocean: Status and Management*. WIOMSA, 156 pp

Bouftira, I., Abdelly, C. and Sfar, S. (2008). Characterization of cosmetic cream with *Mesembryanthemum crystallinum* plant extract: influence of formulation composition on physical stability and anti-oxidant activity. *International Journal of Cosmetic Science* 30, pp. 443–452

Bouftira, I., Hela, H., Amor, M., Abdelly, C. and Souad, S. (2012). Effect of *Mesembryanthemum crystallinum* extract against DMH-induced colon carcinogenesis in experimental animals. *Int. J. Cosmet. Sci.* 3, pp. 1038–1043

Carbone, F. and Accordi, G. (2000). The Indian Ocean Coast of Somalia. *Mar. Pollut. Bull.* 41(1–6), pp. 141–159

Edworthy, C. and Strydom, N. (2016). Habitat partitioning by juvenile fishes in a temperate estuarine nursery, South Africa. *Sci. Mar.* 80(2), pp. 151–161

Fern, K. (2014). Useful Tropical Plants. [Online] Tropical. theferns.info. Available at: http://tropical.theferns.info/viewtropical. Accessed 19 July 2018

Herppich, W.B., Huyskens-Keil, S. and Schreiner, M. (2008). Effects of saline irrigation on growth, physiology and quality of *Mesembryanthemum crystallinum* L., a rare vegetable crop. *J. Appl. Bot. Food Qual.* 82, pp. 47–54

Human, L.D.R., Adams, J.B. and Allanson, B.R. (2016). Insights into the cause of an *Ulva lactuca* Linnaeus bloom in the Knysna Estuary. *S. Afr. J. Bot.* 107, pp. 55–62

Khan, M.A., Ansari, R., Gul, B. and Li, W. (2009). Dormancy and germination responses of halophyte seeds to the application of ethylene. *C. R. Biol.* 332, pp. 806–815

Ksouri, R., Megdiche, W., Falleh, H., Trabelsi, N., Boulaaba, M., Smaoui, A. and Abdelly, C. (2008). Influence of biological, environmental and technical factors on phenolic content and antioxidant activities of Tunisian halophytes. *C. R. Biol.* 331, pp. 865–873

Macnae, W. (1969). A General Account of the Fauna and Flora of Mangrove Swamps and Forests. In *The Indo-West-Pacific Region* (eds. Russell, F.S. and Yonge, M.). Advances in Marine Biology. Academic Press, pp. 73–270

Marnewick, M.D., Retief, E.F., Theron, N.T., Wright, D.R. and Anderson, T.A. (2015). *Important Bird and Biodiversity Areas of South Africa*. BirdLife South Africa, Johannesburg

Mcowen, C.J., Weatherdon, L.V., Bochove, J.V., Sullivan, E., Blyth, S., Zockler, C., Stanwell-Smith, D., Kingston, N., Martin, C.S., Spalding, M. and Fletcher, S. (2017). A global map of saltmarshes. *Biodivers. Data J.* 5, pp. 1–13

Nunes, M. and Adams, J.B. (2014). Responses of primary producers to mouth closure in the temporarily open/closed Great Brak Estuary in the warm temperate region of South Africa. *Afr. J. Aquat. Sci.* 39, pp. 387–394

O'Callaghan, M. (1990). The ecology of the False Bay estuarine environments, Cape, South Africa. 2. Changes during the last fifty years. *Bothalia* 20, pp. 113–122

Paterson, A.W. and Whitfield, A.K. (2003). The fishes associated with three intertidal salt marsh creeks in a temperate southern African estuary. *Wetl. Ecol. Manag.* 11, pp. 305–315

Riddin, T. and Adams, J.B. (2008). Influence of mouth status and water level on the macrophytes in a small temporarily open/closed estuary. *Estuar. Coast. Shelf Sci.* 79, pp. 86–92

Riddin, T. and Adams, J.B. (2012). Predicting macrophyte states in a small temporarily open/closed estuary. *Mar. Freshw. Res.* 63, pp. 616–623

Riddin, T.R. and Adams, J.B. (2010). The effect of a storm surge event on the macrophytes of a temporarily open / closed estuary, South Africa. *Estuar. Coast. Shelf Sci.* 89, pp. 119–123

Rozas, L.P. (1995). Hydroperiod and its influence on nekton use of the salt marsh: A pulsing ecosystem. *Estuaries* 18, pp. 579–590

Saintilan, N., Rogers, K., Kelleway, J., Ens, E. and Sloane, D. (2018). Climate Change Impacts on the Coastal Wetlands of Australia. *Wetlands*. https://doi.org/10.1007/s13157-018-1016-7

Sharma, R., Wungrampha, S., Singh, V., Pareek, A. and Sharma, M.K. (2016). Halophytes As Bioenergy Crops 7, pp. 1–8

Shaw, G.A., Adams, J.B. and Bornman, T.G. (2008). Sediment characteristics and vegetation dynamics as indicators for the potential rehabilitation of an estuary salt marsh on the arid west coast of South Africa. *J. Arid Environ.* 72, pp. 1097–1109

Tabot, P.T. and Adams, J.B. (2013). Ecophysiology of salt marsh plants and predicted responses to climate change in South Africa. *Ocean Coast. Manag.* 80, pp. 89–99

Turpie, J.K., Wilson, G. and Van Niekerk, L. (2012). National Biodiversity Assessment 2011: *National Estuary Biodiversity Plan for South Africa.* Anchor Environmental Consulting, Cape Town. Report produced for the Council for Scientific and Industrial Research and the South African National Biodiversity Institute

UNEP and WIOMSA (2015). Mangroves, salt marshes and seagrass beds by B. Lugendo In: *The Regional State of the Coast Report: Western Indian Ocean.* UNEP – Nairobi Convention and WIOMSA, Nairobi, Kenya, 546 pp

Veldkornet, D.A., Adams, J.B. and Potts, A.J. (2015). Where do you draw the line? Determining the transition thresholds between estuarine salt marshes and terrestrial vegetation. *S. Afr. J. Bot.* 101, pp. 153–159

Ventura, Y., Eshel, A., Pasternak, D. and Sagi, M. (2014). The development of halophyte-based agriculture: past and present. *Ann. Bot.* 115(3), pp. 529–540

Weiss, J.S. and Butler, C.A. (2009). Salt Marshes: A natural and UnNatural history. Rutgers University Press, New Jersey.

Whitfield, A.K (2017). The role of seagrass meadows, mangrove forests, salt marshes and reed beds as nursery areas and food sources for fishes in estuaries. *Rev. Fish Biol. Fish.* 27, pp. 75–110

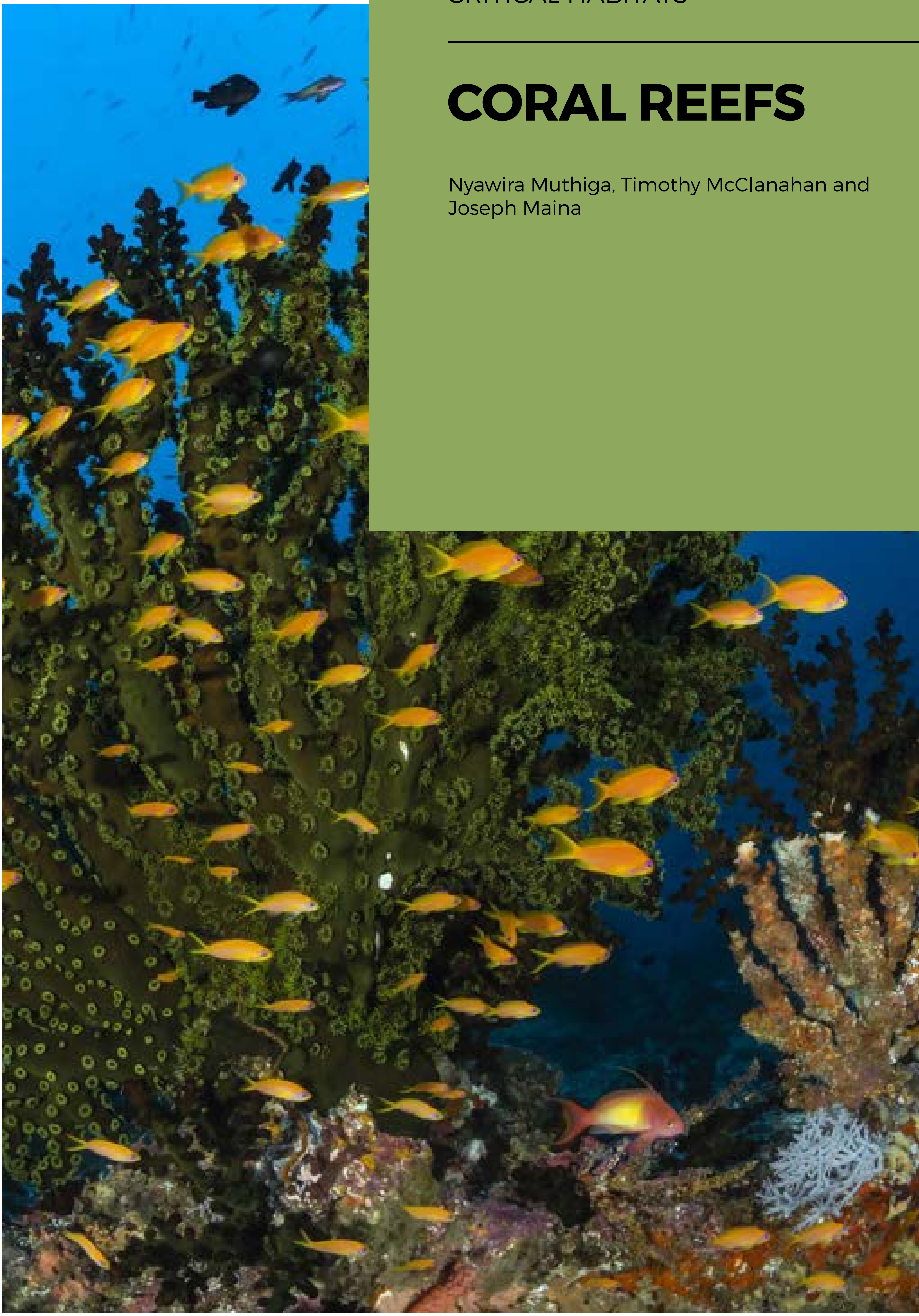
Whitfield, A.K. and Pattrick, P. (2015). Habitat type and nursery function for coastal marine fish species, with emphasis on the Eastern Cape region, South Africa. *Estuar. Coast. Shelf Sci.* 160, pp. 49–59

Wooldridge, T.H. (1998). Estuarine Invertebrates. In: *Field Guide to the Eastern and Southern Cape Coasts* (eds. Lubke, R. and de Moor, I.). University of Cape Town Press, pp. 205–215

CRITICAL HABITATS

CORAL REEFS

Nyawira Muthiga, Timothy McClanahan and Joseph Maina



Coral reef environment at the Indian Ocean. © Fabrice Dudenhofer / Ocean Image Bank.

BACKGROUND

Coral reefs fringe the vast majority of shorelines in the Western Indian Ocean (WIO). Reefs extend as shallow-water structures visible from space from the ancient reef island in northern Kenya (3°S) to Mozambique Island in north-central Mozambique (15°S). South of central Mozambique, there are diverse reef communities composed of hard and soft corals that occupy shallow hard bottom areas including in Bazaruto and Inhaca Islands, but reefs, as large emergent visual carbonate structures, largely disappear south of Mozambique. These reefs are isolated from the more connected northern region (Maina et al., 2020) and do not build reefs due to insufficient calcification relative to reef erosion and dissolution (Schleyer, 2000; Schleyer et al., 2018). Offshore from the African continent, there are also emergent reefs associated with oceanic islands as far east as Rodrigues Island of Mauritius (20.0°S, 63.0°E) and Madagascar.

There are also locations without carbonate deposited reefs with scattered coral colonies such as north-eastern Reunion Island, south-east Madagascar and colonizing volcanic substrates, such as the island of the Comoros. These coral structures are associated with recent volcanism or environmental conditions of light and temperature that are not suitable for net reef growth. Thus, with few exceptions, the reef structure and ecosystem are extensive (Fig. 1) and are critical for shoreline protection, nearshore fisheries production, and tourism throughout the region.

Reef areas have been calculated according to habitat typology (Andréfouët et al., 2006) and estimates of coral and fish species diversity, and coral cover and fish biomass (McClanahan et al., 2014; Obura, 2012; Ateweberhan and McClanahan, 2016). Formal estimates of nationally protected areas have been compiled and recently updated in an *MPA Outlook* report (Richmond and Sisitka, 2021). In addition, the emergence of community-level management, fisheries management feasibility, restrictions, and impacts on fish have been studied (eg Rocliffe et al., 2014; McClanahan et al., 2016; Jones et al., 2018; Samoilys et al., 2017). Moreover, the environmental forces affecting reefs, such as light, temperature, currents, and plankton have been evaluated and mapped on a large-scale using satellite proxies (Maina et al., 2011; Crochelet et al., 2016).

Thus, the information on coral reefs in the region has expanded these last decades since the 1998 massive bleaching event and these syntheses provide a basis for better policies, planning, and management.

BIOGEOGRAPHY

Biogeographically, the WIO is a province within the western part of the West Indo Pacific realm, which contains nine ecoregions (Spalding et al., 2007). The taxonomy of the region has not been adequately sampled, leading to changing ideas about the origins and locations of diversity and their boundaries (Obura, 2012, 2016; Ateweberhan and McClanahan, 2016). The lack of geographically resolved data resulted in underestimates of diversity based on its distance from a centre of diversity in the Coral Triangle of Austral-Asia reefs (Connolly et al., 2003; Tittensor et al., 2010). More recent evaluations suggest this distance-from-biodiversity centre effect is small relative to other factors, such as historical climate refuge, coast length, reef area, connectivity, and variability in the temperature distributions (Pellissier et al., 2014; Ateweberhan et al., 2018).

The WIO is also one of the more sensitive provinces using the functional-loss criteria for tropical fish (Parravacini et al., 2013; D'agata et al., 2016). Since number of species has often been a basis for establishing protected areas, some habitats and taxa are underrepresented in the existing protected area networks (McClanahan, 2020a). Additionally, while species in this province are often pooled with those from other regions, there is increasing evidence that they are often genetically distinct enough to be a concern for the genetic loss of local adaptation potential (Ridgway and Sampayo, 2005; Hoareau et al., 2013; van der Ven et al., 2020). There is also growing evidence that portions of this region have unique environmental conditions that may make them important locations for climate refuge including reefs in the Kenya-Tanzania, Tanzania-Mozambique borders as well as north-western Madagascar (eg Maina et al., 2011; Beyer et al., 2018; McClanahan, 2020b). Finally, larval connectivity studies indicate that most legally gazetted protected areas are not well connected (Maina et al., 2020). Moreover, north-western Madagascar is a notably important location for larval connectivity to many other locations (Crochelet et al., 2016; Maina et al., 2020).

At the provincial scale, several diversity patterns emerge among the better studied taxa of fish and corals. For example, a large field study of the generic coral richness found that community diversity peaked at 10°S and was associated with a moderate temperature standard deviation of 1.4°C (Fig. 2). The high diversity coral areas overlap with the climate refugia areas, and each of these areas has both high local and between-reef diversity (Ateweberhan and McClanahan, 2016).

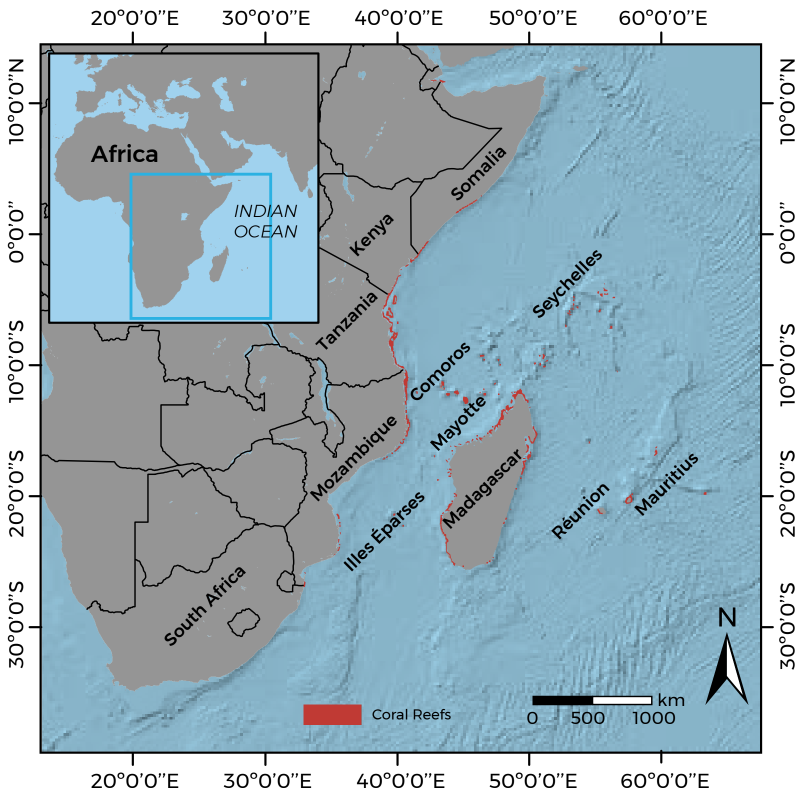


Figure 1: Distribution of coral reefs in the WIO region.

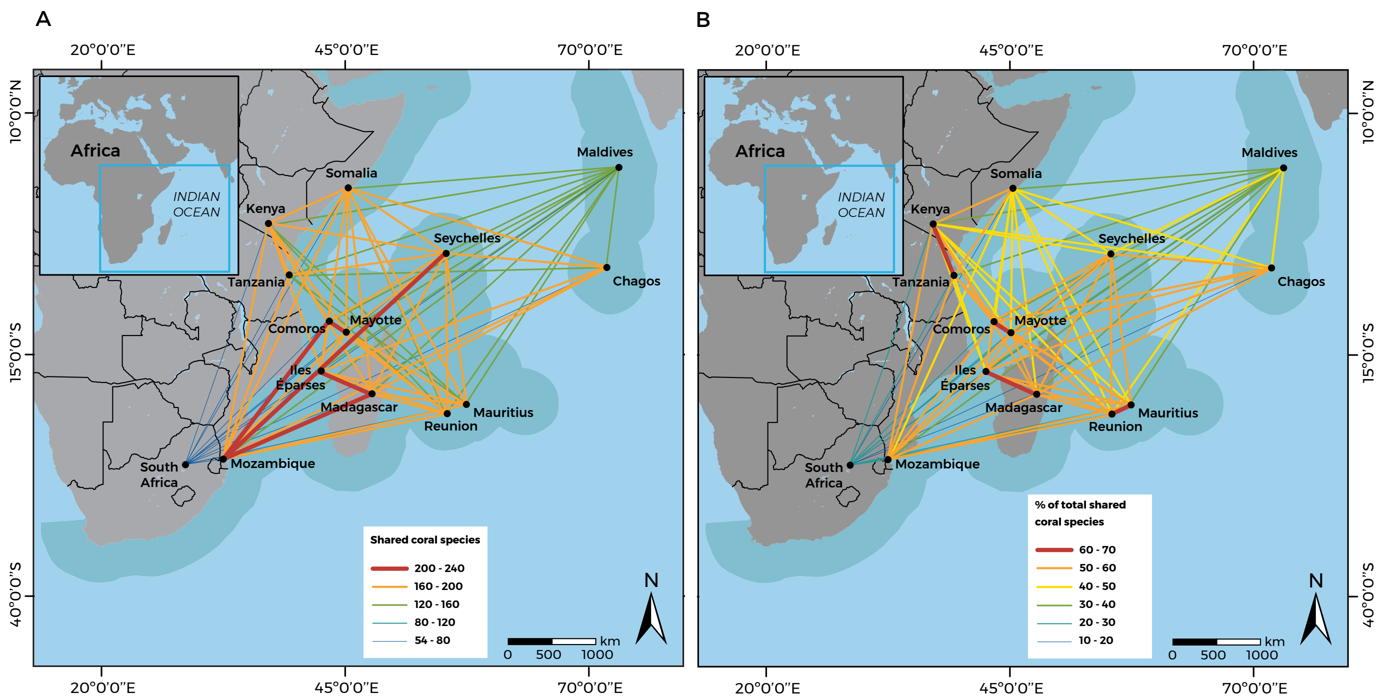


Figure 2: Shared coral species between WIO countries, in absolute numbers (A) and in percentages (B), based on species concurrence among WIO countries (existence of the same species in pairs of countries). Data on the extent of occupancy of coral species was compiled from the IUCN Red List of Threatened Species (2017), comprising 433 coral species. The map was originally published in Levin et al. (2018).

Studies of fish diversity have shown similar geographic patterns and found that diversity is primarily driven by the biomass of fish and some of the critical habitat characteristics, such as the extent of shallow water and the hard and soft coral cover (McClanahan, 2019). Interestingly, some reef fish diversity patterns appear to differ between the African continent and Madagascar. Species accumulate and level faster along the African coastline than Madagascar due to fewer small-bodied taxa in Madagascar (McClanahan and Jadot, 2017), a body size-diversity pattern that has been observed broadly (Kulbicki et al., 2015).

IMPORTANCE

Coral reefs support a range of goods and ecosystem services that collectively provide seafood, habitats and other genetic resources that are important for the livelihoods and economies of coastal nations. Coral reefs also provide regulatory services such as climate change regulation, beach replenishment and coastal protection that support the foundations for tourism, fisheries and trade and reefs are key cultural assets supporting recreation and education. Moreover, coral reefs are connected to and interact with adjacent coastal and marine ecosystems such as mangroves and seagrass beds that contribute to the ecological functioning and hence the overall value of these ecosystems. Globally, coral reefs provide ecosystem services valued at an estimated USD 9.9 trillion (Costanza et al., 2014). In the WIO, the coastal economy is valued at USD 20.8 billion (Obura et al., 2017), which includes fisheries, coastal tourism and other economic sectors that coral reefs contribute to (Table 1). Coral reefs are particularly important in the WIO, where a high population of the coastal community are reef dependent either for livelihood or for food (eg Hicks, 2011) and a significant number are living in poverty (Whittingham et al., 2003).

Coverage and distribution

In the WIO, reefs cover an area of about 11 980 km^2 (Table 1) which is about 35 per cent of the coastal habitats of East African mainland states where more than 87 per cent of the region's population resides (Whittingham et al., 2003). Coral reefs are distributed in clear oceanic waters to stressful and turbid environments next to populated coastlines, although they are limited where major rivers input into the Indian Ocean (Sheppard, 2000).

Four main types of reef structures occur: fringing, patch, barrier reefs, and atolls, but reef habitats often occur as a continuum of reef types (Andréfouët et al., 2006). Non-reefal habitats include limestone ridges and banks. Fringing reefs are predominantly found along with the Somalia, Kenya, Tanzania and Mozambique coastline and the eastern coastlines of islands including Pemba and Unguja, islands of Zanzibar (Fig. 1). Patch reefs often occur where there is a wider continental shelf, for example along the Tanzania mainland coastline. Barrier reefs occur off Mayotte island, Mauritius and Tulear in Madagascar, and atolls around oceanic islands like Aldabra, Farquhar, Cosmoledo and Alphonse Islands (Seychelles) and Bassas da India in the Mozambique channel (Sheppard, 2000; Arthurton, 2003). Reefs are also found in various offshore banks, including Malindi Bank off the Kenyan coastline, Leven and Castor in NW Madagascar, Africa Bank in Mozambique and the Mascarene Plateau and North Seychelles Banks (Sheppard, 2000; Spalding et al., 2001). Reefs in South Africa occur at the southern limit of coral reefs in the WIO and exist as coral communities with higher soft coral than hard coral cover and are not true accretive reefs (Schleyer, 2000).

Biodiversity

Reefs support more species per unit area than any other marine and coastal ecosystems, although they cover only 0.1 per cent of the world's oceans (WCMC, 1992). Globally, 93 000 species of reef inhabiting animals and plants have been described, but this is thought to be a fraction of the total (Spalding et al., 2001; Paulay, 1997). Regional diversity estimates are inadequate, although some taxa are better understood than others. The Indo-West Pacific, which includes the WIO region, has approximately 700 species of hard coral, more than 690 species of soft corals (Veron, 1995; Paulay, 1997), 1350 species of echinoderms (Clark and Rowe, 1971), approximately 4000 species of reef fish that is almost a third of the worlds marine fish species (Lieske and Myers, 1994). Hard corals are one of the more studied species, and the pattern of coral species diversity in the WIO region reinforces the concept that this is a biogeographic region (Sheppard, 2000; Obura, 2012). The diversity of hard corals in the Indian Ocean includes 87 genera and 491 species (Sheppard, 2000) and species diversity is better studied in some subregions and countries (Rosen, 1971; Riegl, 1996; Schleyer and Celliers, 2003) than in others.

Fishing

Small scale fishers account for 90 per cent of the world's fishers (Teh and Sumaila, 2013), and the WIO region contributes ~5 per cent of global fish catch. In small island

Table 1: General geographic, demographic, trade and conservation statistics for each of the WIO countries. Area and number of MPAs includes only designated MPAs within the World database of protected areas (WDPA) as of 2017. Population and GDP data is from 2012; temporal change is assumed to be proportional across countries.

INDIAN OCEAN	AREA (1,000 km²)	POP 2012 (10^6)	GDP 2012 ($10^9$$)	GDP 2012 (PER CAPITA)	IMPORTS ($10^9$$)	EXPORTS ($10^9$$)	IMPORTS/EXPORTS RATIO (%)	IMPORTS FROM INDIAN OCEAN (% OF TOTAL)	EXPORTS TO INDIAN OCEAN (% OF TOTAL)	EXPORTS FISH PRODUCTS (% OF TOTAL)	FOREIGN AID (% OF GDP)	NO. OF AGREEMENTS	EEZ AREA (1,000 km²)	NO. OF MPAs	AREA OF MPAs (1,000 km²)	AREA OF MPAs (% OF EEZ)	CORAL AREAS (km²)
British Indian Ocean Territory	0.1	0.0	N/A	N/A	0.1	0.0	322	0	1	11.0	N/A	40	636	7	634.8	100	1,923
Comoros	1.7	0.7	0.6	831	0.2	0.1	355	14	3	2.1	9.4	27	164	0	0.0	0.0	221
French Southern Territories (Iles Esparses)	0.05	–	N/A	N/A	-	-	N/A	0	0		N/A	47	623	4	0.05	0.0	752
Kenya	593.3	43.2	40.7	943	15.1	5.2	291	7	13	1.7	4.9	41	111	7	0.6	0.5	506
Madagascar	590.3	22.3	10.0	447	2.7	1.2	217	10	8	6.0	5.2	36	1,191	34	8.3	0.7	3,773
Maldives	0.3	0.3	2.2	6,567	1.6	0.2	962	0	0	0.0	2.7	23	915	0	0.0	0	2,696
Mauritius	2.0	1.3	10.5	8,120	5.8	2.3	256	9	22	3.4	1.4	41	1,270	6	0.0	0.003	716
Mayotte (Fr)	0.4	0.2	1.0	4,484	0.4	0.0	1281	12	22	63.2	56.8	47	63	1	62.7	100	295
Mozambique	786.0	25.2	14.2	565	6.2	3.5	178	32	20	0.7	14.2	33	566	3	11.6	2.1	2,090
Reunion (Fr)	2.5	0.8	18.8	22,355	2.3	0.3	843	9	25	32.6	N/A	47	315	1	0.0	0.01	12
Seychelles	0.5	0.1	1.1	12,783	0.7	0.5	141	12	6	25.1	2.6	34	1,329	10	0.2	0.02	1,572
Somalia	632.7	10.2	5.9	578	1.3	0.2	777	0	0	0.5	13.6	18	665	0	0.0	0	248
South Africa	1,220	52.3	384.3	7,352	101.6	86.7	117	2	5	0.5	0.3	45	317	8	1.2	0.4	3
Tanzania	939.8	47.8	28.2	609	11.7	5.5	211	13	25	3.0	9.6	35	240	9	5.4	2.2	2,413

nations such as the Seychelles, Mayotte and Reunion, fishing is a key economic sector contributing 25 to 60 per cent of GDP (Table 1). However, most of these are offshore and not reef fisheries. Reef fisheries are crucial for the rest of the WIO, supporting food security and livelihoods of ~250 000 people, excluding South Africa (Teh and Sumaila, 2013).

An estimated reef area of ~50 000 km^2 with a maximum yield of ~6 t/km^2 would produce ~300 000 t of reef fish per year in the region (Samoilys et al., 2017; McClanahan and Azali, 2020). However, ~40 per cent of the reefs have biomasses below sustainable yields due to overfishing. Countries where reefs are close to shore, and people depend on reef resources, such as Kenya, Tanzania, Mozambique, Madagascar, and Mauritius, are particularly vulnerable. Reefs with fish biomass above levels of maximum production typically occur in sparsely populated countries, where reefs are either far from markets, people do not eat reef fish, or reefs are contained within high compliance closures, such as Seychelles, Maldives, South Africa and Iles Eparses (Southern French Territories) (McClanahan et al., 2021).

Tourism

Coral reefs are the foundation of the coastal tourism sector providing ecosystems services such as sandy beaches, shoreline protection and fish for food and recreation. The WIO coastal tourism sector has been growing at a rapid rate. It includes hotels, restaurants, diving and recreational fishing, all providing thousands of jobs and driving the economies of many nations of the WIO. Coastal tourism is reported to contribute ~ USD 11 billion and accounts for 40 per cent of the value of marine and coastal resources and is the largest contributor to the GDP of WIO countries (ASCLME/SWIOFP, 2012). The contribution of coral reefs is substantial given the value of the numerous ecosystem services that coral reefs generate for the WIO.

Shoreline protection

Coral reefs also protect coastal areas from erosion and buffer them from cyclones and wave surges, protecting beaches and coastal habitats that are important for fisheries and tourism. This function is especially important for the coastal communities living along low-lying coastlines, where reefs protect their infrastructure, crops and homes. Since approximately 58 million people live within the coastal areas of the WIO, the provision and value of coastal protection is high. In addition to shoreline protection, coral islands are also maintained by the continual generation of sand allowing habitation by people and habitats for nesting birds and sea turtles.

Connectivity

Coral reefs exist within a wider coastal and ocean space and are connected by the exchange of currents, nutrients, sediments and dispersing organisms in the larval or adult stages. Most reef organisms have sessile lifestyles but produce pelagic larval stages that are dispersed to adjacent reefs or other habitats such as mangroves and seagrass beds that are inhabited by numerous other species. This larval dispersion has implications for conservation planning (Crochelet et al., 2016). Reefs also provide feeding and breeding grounds for numerous pelagic and coastal species that disperse over large distances, such as turtles and marine mammals.

THREATS

Coral reefs in the WIO are being threatened by multiple factors, of which the three main forces are climate associated disturbances, fishing, and the interrelated factors of nutrient pollution and sedimentation caused by human influences on land. The spatial scale of these disturbances reduces along this continuum, but the intensity is patchy in space and time. Coral reefs can therefore experience one, all three, or all possible combinations of these degrading forces. Thus, a reef's ecological state often reflects the combined forces and hence the impacts that will need to be managed (Fig. 3). Other threats include human induced pressures from tourism and coastal development, invasive species and coral diseases, some of which are driven by and exacerbated by human pressures, including fishing and nutrient pollution (Darling et al., 2010; Maina et al., 2013). The status of these threats on WIO reefs is detailed in the following sub-sections.

Climate-associated disturbances

According to the Intergovernmental Panel on Climate Change (IPCC), coral reefs are one of the Earth's ecosystems experiencing drastic climate-induced change (IPCC, 2014). Climate associated disturbances include thermal stress, ocean acidification (OA) and increased extreme weather events and storms (Ban et al., 2014). Thermal stress impacts in the region have been severe and the impacts may be expanding and accelerating (eg Hughes et al., 2018). Some bleaching events probably occurred

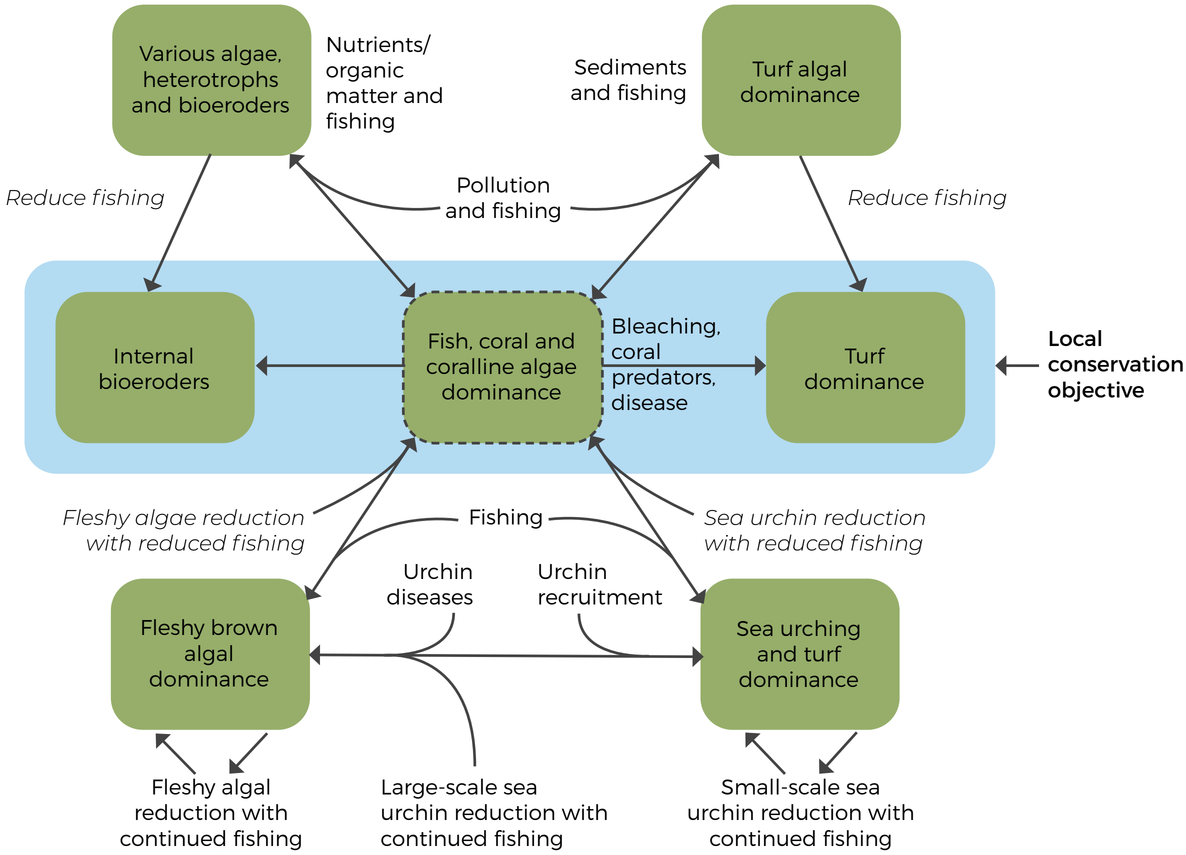


Figure 3: A conceptual model of coral reefs.

in the early 1980s but were seldom observed or quantified (Faure et al., 1984). The first large-scale bleaching observations in 1998 reported ~50 per cent mortality but with considerable local variability (eg Goreau et al., 2000; Ateweberhan et al., 2011). Overall, northern reefs in the WIO, including Kenya and Seychelles were badly affected with poor and patchy recovery (Darling et al., 2013; Graham et al., 2017). Reefs in the southern WIO were more patchily affected by subsequent warm water disturbances in 2000, 2005, 2008, 2010, 2013, and 2016 but mortality and recovery rates have been poorly quantified (eg Schleyer et al., 2008; Obura et al., 2018; Bigot et al., 2019; McClanahan and Muthiga, 2021).

Some studies have suggested considerable stability in reef structures in Reunion Island (Scopélitis et al., 2009) but not SW Madagascar (Andréfouët et al., 2013). Surveys have consistently found more thermal sensitive hard corals in the southern than northern WIO region indicating either less stress or more resilience to stress (McClanahan et al., 2014). The general picture is that the northern Indian Ocean reefs were severely disturbed prior to 2004 and showed some patchy recovery thereafter (Ateweberhan et al., 2011). After 2004, the southern Indian Ocean experienced faster declines in coral cover than the northern Indian Ocean as thermal stress expanded to the south, especially in *Acropora* species. However, some locations within the southern Indian Ocean reefs have only been patchily disturbed and able to maintain high diversity and, are therefore of regional conservation priority (McClanahan and Muthiga, 2017).

Climate change is also causing disruptions in ocean chemistry, and as oceans absorb more and more anthropogenic carbon dioxide (CO_2), seawater pH is lowering, leading to OA. The world ocean's pH is projected to decrease to 0.2–0.4 pH by 2104 (IPCC, 2014), which threatens reef organisms that need $CaCo_2$ for their skeletons, for example, hard corals, mollusks, coralline and calcareous algae that are the main reef builders, as well as other invertebrates key to ecological functioning, such as sea urchins (Moulin et al., 2014). OA also impacts processes such as recruitment and food webs hence threatening reefs more broadly. The impacts are expected to vary across ocean basins (van Hooidonk et al., 2014) and are likely to be exacerbated by other anthropogenic stressors such as increased sea water temperatures, pollution and fishing that could cause functional ecosystem collapse and the loss of reef biodiversity (Hoegh-Guldberg, 2011). The state of OA is known from a range of studies across the oceans on the physiological, molecular and community response to acidification and the association with seagrass beds and mangroves (Camp et al., 2016). Although OA occurs globally, the impacts are local and vary across ecosystems and populations. Therefore, to mitigate or adapt locally, monitoring and experimental information are required at the local level yet, little is known about OA and its effects in the WIO (WIOMSA, 2017).

Thermal stress and OA associated with climate change also affect the health of coral reef organisms (Hoegh-Guldberg et al., 2007; Harvell et al., 2009). Coral diseases have been reported globally, mainly affecting a few species, although localized outbreaks have caused high mortalities of hard corals and other organisms, including sea urchins, reef fish, sponges and algae (Peters, 1993; Harvell et al., 1999). Forty diseases have been classified worldwide, mostly occurring in the Caribbean, with only 14 per cent of observations from the Indo-Pacific (Weil et al., 2006). In the WIO, bacteria induced bleaching was reported in Zanzibar (Ben-Haim and Rosenberg, 2002) and isolated incidences of Black, White and Yellow Band diseases were reported in Kenya (McClanahan et al., 2004), while Black Band disease and a yellowing disease were reported in South Africa (Jordan and Samways, 2001) and a mass die-off of *Montipora* and *Astreopora* species caused by a white syndrome was reported in Kenya (McClanahan et al., 2004). The most recent studies of coral diseases in Reunion, Mayotte and South Africa reported six main diseases with the highest disease prevalence in Reunion, followed by South Africa and Mayotte, the common genera *Acropora* and *Porites* were the most affected (eg Séré et al. 2015, 2016).

Disease outbreaks and increasing coral mass mortality may increase as the oceans continue to heat up since thermally stressed coral are more susceptible (eg Harvell et al., 2009; Séré et al., 2016). Corals have also been reported to bleach due to bacterial infection (Ben-Haim and Rosenberg, 2002) and bleaching was also associated with increased incidences of Black Band disease and cyanobacterial films in South Africa (Celliers and Schleyer, 2002). Water quality, including turbidity, nutrient load, sediments and pollutants, also projected to increase due to climate change, can also reduce coral resilience and increase disease outbreaks (Burge et al., 2013). Climate change associated factors also impact coral reef organisms at the reproductive and embryonic stages (Negri et al., 2007; Albright et al., 2010), potentially affecting recruitment and the ability of coral reefs to recover after bleaching and mass mortality events (Szmant and Gassman, 1990; Weil et al., 2009).

Fishing

Fishing disturbances are pervasive in the region and associated with patterns of demand and management systems (McClanahan et al., 2015; Cinner et al., 2016). Some remote island reefs in Seychelles and the Mozambique Channel are not disturbed by fishing (eg Friedlander et al., 2014; Chabanet et al., 2016). Nevertheless, the African coastline and the occupied larger islands are considerably impacted by fishing in a way that suggests that high reef fish biomass is largely restricted to high compliance fisheries closures (McClanahan et al., 2016; McClanahan, 2019). These high compliance closures can be used to set a benchmark for the status of reefs in terms of fishing impacts (McClanahan, 2018).

A regional evaluation of the status of fished seascapes around reefs in the WIO shows that some countries like Tanzania and Madagascar have less fish biomass with the proposed sustainable yield levels because of the lack of managed areas (McClanahan et al., 2016, 2021). Other countries like Kenya have a similar pattern but with some reef areas maintaining high biomass in the better managed reefs. Overall, the model predicts that ~40 per cent of the reefs in the region have biomasses of <600 kg/ha, which is considered a key cut-off for sustaining and reducing the degradation of coral reefs (Graham et al., 2017). About the same percentage of reefs have higher biomasses suggestive of fish communities with minor human disturbances.

The rates of recovery of fish stocks have also been studied on a regional level allowing calculations of both the time to recovery to proposed sustainable levels as well as yield potentials. Mean recovery times for all reefs in the region is ~3 years of no fishing to achieve the 600 kg/ha biomass level (McClanahan et al., 2016). Recovery times vary, for example, with Reunion, Mauritius, Madagascar, Mozambique, Tanzania and Kenya requiring ~ 4 to 7 years while the remote islands of Seychelles and Maldives requiring ~1 year. Model estimates suggest that the maximum yields are ~6 t/km^2/y at a biomass of 50 t/km^2, which fit well with empirical studies for the region and particularly for Kenya where long-term trends have been measured (Samoilys et al., 2017; McClanahan, 2018). Most Kenyan reef fisheries are overexploited and have biomasses of ~25 t/km^2, below maximum yields, and slowly declining at around 2-3 per cent per year (McClanahan and Azali, 2020).

Whether or not this model applies to other areas in the region remains to be determined. Still, these fisheries yields and modelling studies imply that nearly half the region is slowly losing catch potential. Areas not fished unsustainably are remote or in reefs with more fishing restrictions, such as limited gear use (McClanahan, 2019).

Pollution and sedimentation

Nutrient pollution and sedimentation are emerging as major threats to coral reefs compounding the already intensifying climate change and fishing threats (Fig. 4; Maina et al., 2013; Ban et al., 2014). While most of the coral reefs in the WIO, especially on the islands, experience low levels of pollution, sediment and nutrient pollution from land discharged through the main rivers in the region has been increasing.

The main rivers in proximity to coral reefs include Rivers Sabaki and Tana (Kenya), Pangani, The Great Ruaha/ Rufiji and Ruvuma (Tanzania), Zambezi (Mozambique), Mangoky, Onilahy, Betsiboka and Sofia (Madagascar), and The Grand River South East (GRSE) and Citron rivers in Mauritius. These rivers deposit >100 t of sediment annually into the Indian Ocean that disperses to near-shore coral reefs with variable consequences (Maina et al., 2011). For example, studies have shown sediment discharges from the Onihaly river correlate with a coral core derived sediment proxy (barium/calcium, luminescence) and the condition of reefs in SW Madagascar (Grove et al., 2012; Maina et al., 2012). The degradation of the SW Madagascar Great Barrier Reef is therefore partially attributed to heavy sedimentation from the Onihally and Mangoky rivers.

Increased soil erosion due to local land-use change, deforestation and intensifying tropical cyclones also caused significant changes in the sedimentary budget and biodiversity shift in NW Madagascar (Fontanier et al., 2018). Although Fleitmann et al. (2007) demonstrated a five to tenfold increase in sediment flux after the 1900s relative to natural levels in East Africa, this increase showed no significant impact on diversity and ecological health of the sediment influenced reefs of Malindi in the mid 1990s (McClanahan and Obura, 1997). Nonetheless, the negative impacts of sedimentation are expected to intensify given climate change and rise in temperature acting in synergy with sediment and nutrient pollution (eg Ban et al., 2014).

It is unlikely that there will be a reduction in sediment and nutrient pollution without a significant improvement in socio-economic conditions and adoption of green growth strategies, especially given the large agricultural and infrastructural projects (ports and oil pipelines) currently underway or planned in the WIO. Examples include the Southern Agricultural Growth Corridor of Tanzania (SAGCOT[1]), the 1.7 million-acre Galana-Kulalu irrigation scheme project in Kenya[2]. Extensive land conversion of ~3 million ha in coastal catchments will negatively impact freshwater, coastal habitats, fisheries and biodiversity resources through land-based runoff. The agricultural expansion initiatives may also inadvertently endanger peoples' future food security and fisheries resources, which are already threatened by subsistence fishing and climate change. Thus, development plans for marine and terrestrial natural resources must consider and mitigate impacts on communities and livelihoods.

Coastal development

A broad range of anthropogenic activities on the coast, including the construction of roads, tourism facilities and homes, seawalls and dredging and land conversion for ports, airports and oil facilities, can negatively impact coral reefs and associated ecosystems (Celliers and Ntombela, 2015). In particular, large infrastructure projects and ports in Kenya (Lamu Port and Lamu-Southern Sudan-Ethiopia Transport Corridor), in Tanzania (Bagamoyo Port) and in Mozambique (oil and gas licenses and exploration activities) that are under consideration or underway have the potential to cause negative impacts (ASCLME/ SWIOFP, 2012). Construction, dredging and disposal of dredge materials, clearing or modification of beaches and other coastal habitats such as mangroves, and through sedimentation and oil spills could negatively impact coral reefs.

The WIO coastal cities are also undergoing rapid urbanization, and management of solid and liquid waste is a major challenge (Celliers and Ntombela, 2015). Construction of soakage pits close to shore and discharge of sewage at sea is a common problem as most WIO nations have few land-based sewage facilities. Seepage and discharge in areas near coral reefs could encourage algal blooms and macroalgae growth that outcompete corals. Coastal development also has an indirect threat through pressure from population density and population growth. Burke et al. (2011) estimated that 35 per cent of reefs in the Indian Ocean have a medium to high risk from local pressures, including coastal development. Climate impacts, including sea level rise, also can negatively affect coastal developments through accelerated erosion and flooding (Kebede et al., 2012), producing discharges and pollutants that in the long term could have adverse effects on reefs and associated ecosystems.

1. http://www.sagcot.com
2. http://www.nib.or.ke/research-centre/84-nib/127-galana-kulalu-ranchirrigation-project-green-revolution.html

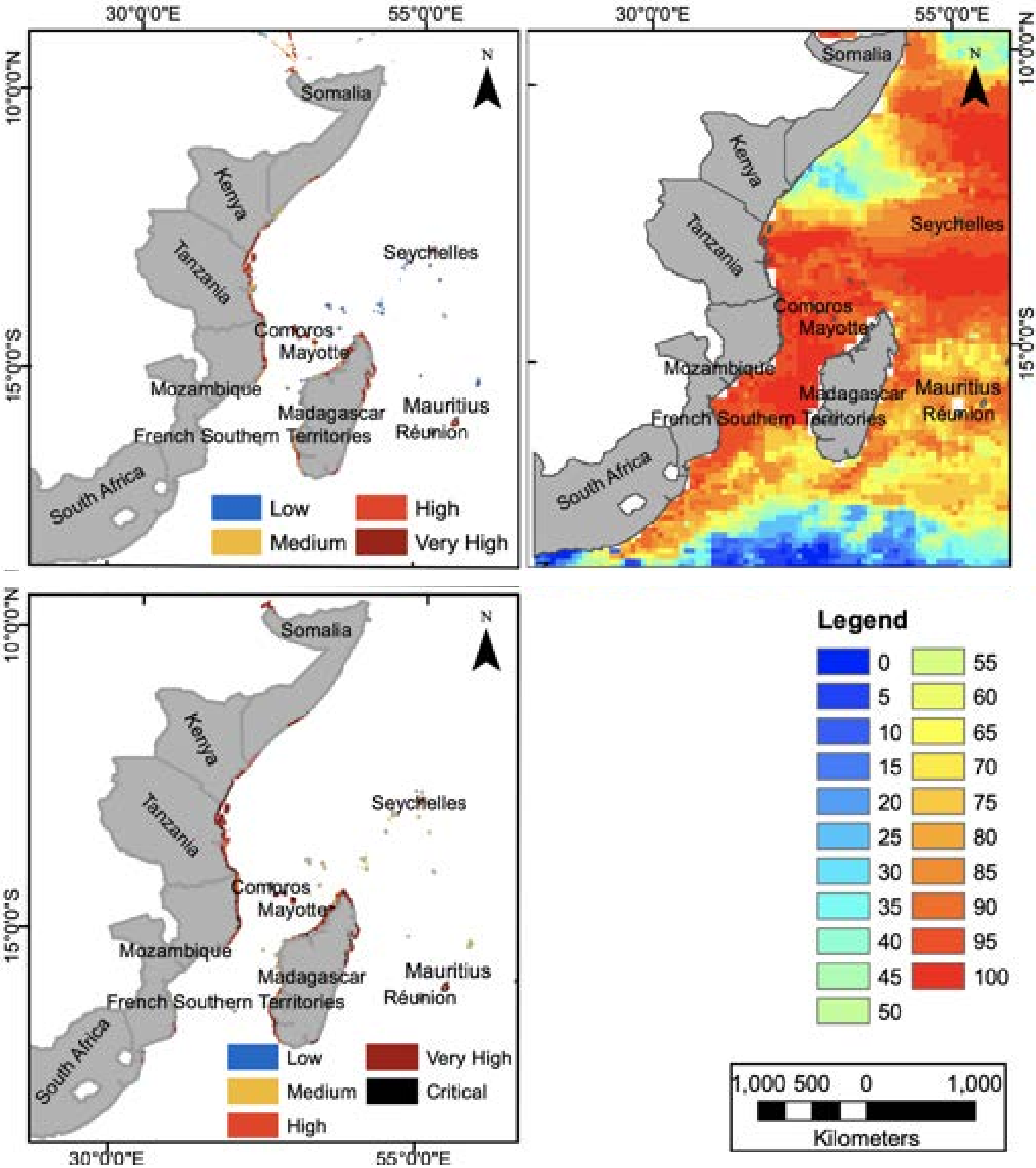


Figure 4: Map of the distribution of (A) local threats (B) climate threats in the form thermal stress by 2050, and (C) integrated local and global threats. Data used here was derived from Reef at Risk (Burke et al., 2011).

Marine litter has also become an emerging global ocean health issue that is exacerbated by coastal development. The first regional assessment of marine litter (Lane et al., 2007) showed that little data exists in the WIO except in South Africa. Most marine litter was plastics, and it accumulated on beaches, coral reefs, estuaries and other coastal habitats across the WIO. Marine litter is generated locally and can come from as far away as South East Asia, as reported on the remote Alphonse Islands, Seychelles (Duhec et al., 2015). Marine litter has also been shown to increase coral disease risk from contamination amongst other negative effects (Lamb et al., 2018). Although few studies have been undertaken on the effect of plastics in the WIO, several countries have implemented plastic bag bans (Kenya, the Republic of Tanzania and Madagascar), partial bans (Somalia) and taxes (Mozambique and South Africa).

Existing protection

The WIO nations have experienced an evolution in marine resource management that has an impact on coral reefs. Over time, nations of the region have instituted a variety of modern conservation and management practices. These include small to medium scale area-based management approaches such as national Marine Protected Areas (MPAs), co-management and community managed areas, locally managed marine areas (LMMAs), and community fisheries closures, for example, "Tengefu" in Kenya. Protected areas managed by non-governmental organizations (NGOs) include Cousin Island, managed by Nature Seychelles, and by private companies, include Chumbe Island Coral Park in Zanzibar (eg Muthiga et al., 2000; Rocliffe et al., 2014; Kawaka et al., 2017). Larger scale area management, including integrated coastal zone management (ICZM), national marine spatial plans (MSP) and transboundary conservation areas (TBCAs), such as the proposed TBCA between Kenya and Tanzania, have been instituted or are under discussion (MPRU/KWS, 2015). National, regional and global laws anchor these approaches, initiatives and agreements such as the Convention on Biological diversity's Aichi marine target, the International Coral Reef Initiative's Call to Action, the UNEP/Nairobi Convention Action plan and work programme and the Coral Reef Task Force's EA Coral Reef Action Plan amongst others (ASCLME/SWIOFP, 2012; MPRU/KWS, 2015; WCS, 2019; Coral Reef Task Force, 2008).

Marine Protected Areas (MPAs) are the most commonly implemented area-based tools in the WIO. In many cases however, the design of MPAs did not take into account marine zoning considerations such as representativeness (ecological and biodiversity), adequacy (size), and irreplaceability (Margules and Pressey, 2000). A recent regional assessment of MPAs indicates there are 143 MPAs in the WIO covering ~555 437 km^2 or ~7 per cent of the combined EEZs (Richmond et al., 2021). Of these, coral reefs are estimated to cover 17 720 km^2 which is 7.5 per cent of total coral reef area (Maina et al., 2020). These estimates are based on area calculations of the MPA and coral reefs spatial features, which will differ from the national reported areas since the areas reported are not based on corresponding MPA habitats for most countries. Further, the area of coral reefs is dependent on the resolution of the gridded data. The calculations used a 1 km^2 grid size for coral reef locations. Hence, few nations in the region have met the CBD Aichi target 11 (Table 1) and numerous challenges related to the management effectiveness of existing MPAs remain (Chadwick, 2021).

The second most common area-based approach is the co-management approach, a decentralized management model focusing on fisheries. The co-management legislation built on already exiting traditional systems such as the 'dina' in Madagascar or the national social Ujamaa system in Tanzania and there is now a proliferation of co-managed areas across the region (eg Cinner et al., 2012; Rocliffe et al., 2014; Kawaka et al., 2017). Co-management effectiveness evaluated in Kenya, Madagascar and Tanzania showed that it sustained fisheries but that compliance and institutional strength were key factors influencing the outcomes (Cinner et al., 2012).

Other management approaches include restricting use of destructive gears (dynamite and beach seines) and reducing gears that have high bycatch through trap modifications (eg Mbaru and McClanahan, 2013), temporal fisheries closures, such as those in Madagascar and elsewhere for octopus (eg Benbow et al., 2014; Gardner et al., 2020) and management and protection of spawning aggregation sites, for example, the rabbitfish (*Siganus sutor*) in Kenya and Seychelles (Robinson et al., 2011; Samoilys et al., 2017).

PRIORITY OPTIONS FOR CONSERVATION OF CORAL REEFS

Countries of the WIO are already undertaking many actions to manage coral reefs. These include regional and national strategies for climate change, coral reef action plans, ICZM and marine spatial planning as well as establishment of MPAs and other area-based and fisheries-based approaches and interventions. However, many actions recommended in the regional Coral Reef Task

Force Action plan (Coral Reef Task Force, 2008), the previous Regional State of the Coast Report (Obura, 2015) and national coral reef strategies and commitments to CBD and Sustainable Development Goals (SDGs) amongst others are yet to be implemented. The following recommendations draw on these reports and knowledge from new scientific studies undertaken in the WIO focusing on the most urgent actions.

1. Prioritization of MPAs based on climate change impacts

Climate change impacts are presently the major threat to coral reefs in the WIO, but studies have shown that not all areas experience the same impacts (eg McClanahan et al., 2020). Areas that are buffered from stress or are less sensitive to climate change have characteristics that allow the ecological and socio-economic values to persist overtime. Such areas termed climate refugia have been identified at the border regions of Kenya/Tanzania, Tanzania/Mozambique, and, north-west Madagascar.

Currently, there are efforts/initiatives to increase protection of the Kenya/Tanzania border as a proposed TBCA and proposed projects under the Northern Mozambique Channel initiative (Obura et al., 2015). There is a need to identify and map these areas and undertake socioecological studies to better understand the threats and develop mechanisms to ensure their protection. As these areas include several countries, joint governance strategies will also have to be developed. Identifying and mapping in more detail other potential climate refugia and other areas of conservation concern throughout the WIO, and providing this information to countries for inclusion in their strategies for national MSP, Blue Economy planning and global MPA commitments (Aichi Target 11) is needed. Setting aside large wilderness areas, TBCAs, and "Other Effective Area-based Conservation Measures" (OECMs) (see Jonas and Sandwith, 2019) are potential pathways to meeting this commitment. An evaluation of potential financing options such as debt refinancing used by the Seychelles to protect 30 per cent of its EEZ, Blue Natural Capital and other innovative financing mechanisms can provide support for increasing MPA coverage.

2. Improving effectiveness of fisheries management

Reefs that are well managed are able to resist and recover from disturbances to a degree. A range of actions can be undertaken to control fishing to sustainable levels. In many WIO countries, the appropriate regulations exist. Still, the main limitations are weak governance and lack of adequate scientific information for sustainable fisheries, weak application of co-management, and lack of or poor integration with other sectors.

Management should be ecosystem focused, based on the best available scientific knowledge, use precautionary principles and engage the key stakeholders if it is to be effective. Future subsidies should focus on the recovery of stocks rather than reducing access costs through gear, fuel and vessel subsidies. Co-managed areas can also be improved by sharing the costs of governance, strengthening monitoring and enforcement capacity, and improving intra and inter-community engagement.

3. Evaluating and promoting alternative and sustainable livelihoods

Coastal communities in the WIO are highly dependent on coral reefs and associated ecosystems. Different types of livelihood projects have been initiated in the WIO, and the most common are mariculture projects (seaweeds, sea cucumbers, mud crabs), octopus closures, modified fishing traps, curios (pearl oysters jewellery) and ecotourism (provision of boats for tourism, mangrove restoration and board walks, and community closures) amongst others. However, many of these projects remain as trials or small-scale interventions and are often not adequately coordinated, capacitated and scaled up to ensure harmonized and broad scale impact.

Conflicts between national and local programs and poor coordination with and amongst NGOs and community-based organizations (CBOs) is a challenge. Overarching national livelihood strategies that provide a framework for coordinated development of livelihoods will reduce the current ad hoc and poorly coordinated efforts and can build on some of the successful livelihood initiatives that have been trialled across the region. To be sustainable in the long term, livelihood interventions should ideally include providing educational and technical support, access to credit markets, skills training, and be sustained for the length of time communities need support.

In addition, continued research and monitoring are required to ensure adaptive management, especially as social and ecological outcomes are often dependent on social, political and other conditions at the project site (see Eklöf et al., 2006; Fröcklin et al., 2012). Such strategies also need to incorporate climate change vulnerability assessments and forecast impacts of future local and national developments as these may affect coastal communities and their livelihoods.

4. Improving management effectiveness of marine protected areas

The WIO countries have invested in establishing 143 MPAs and this is one of the core strategies for protecting coral reefs. These MPAs have the potential to provide ecological, social and economic benefits. However, declaration of MPAs does not automatically lead to actions that result in adequate protection. Many MPAs lack the basic resources for management, effectively functioning as "paper parks", and fail to achieve conservation goals that exceed well-managed fisheries.

There is a need to undertake a comprehensive regional Management Effectiveness Assessment (MEA) to augment the results of the WIO *MPA Outlook* report (Chadwick, 2021). This can also build on and incorporate previous capacity building efforts (Wells et al., 2003) such as the MPA management course and MPA manager certification (COMPAS) programs of WIOMSA, the MEA WIO training workshop (Wells, 2004), support for the national coral reef task forces and monitoring programs and the Network for MPA managers.

In addition, a cost benefit assessment showing the cost of ineffective management could serve to influence decision-makers in allocating adequate funds for MPAs. As financing is one of the leading limitations (Chadwick, 2021), a review of sustainable financing option is also needed.

5. Formalizing and operationalizing co-management of small-scale fisheries

One of the limitations to expanding MPAs and coral reef conservation across the WIO is the resistance by local communities who were not adequately engaged in the establishment of MPAs and fisheries regulations and weak fisheries management inside and outside protected areas. Across the WIO, the predominantly top-down fisheries management approach has changed to co-management, that puts stakeholders at the centre of management. Co-management has not only been shown to improve biodiversity protection and livelihoods, but it also has the potential to increase spatial management in countries where resistance to area-based management is high. Although the legal instruments for co-management are in place in many countries, the implementation, especially the operationalization of co-management, has not been adequately supported.

Studies of co-management have shown that the most effective communities had local rules, strong leadership and regular monitoring of their resources. However, co-management is effectively still in the teething stage in the WIO, often plagued by lack of understanding of roles and responsibilities of the community, poor outreach by the management institutions, lack of skills in adaptive management and corruption and elite capture. Given the high reliance of local communities and their impact on coral reefs, improving their co-management ability will greatly benefit reefs in the WIO. Learning networks (national and regional) such as the LMMA network in Madagascar and the Fishers' forum in Kenya can enhance co-management capacity.

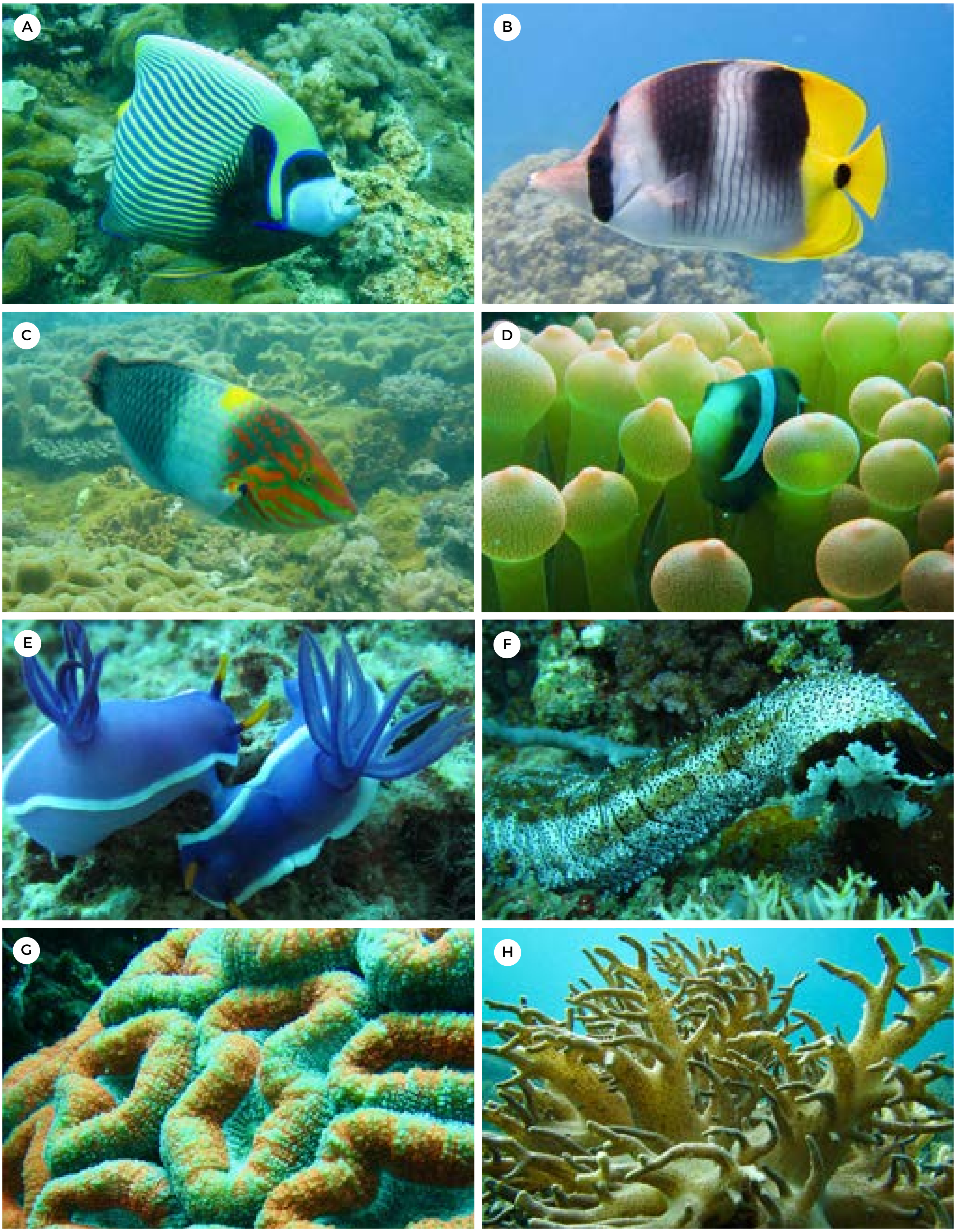
Supporting leaders to adapt traditional practices that benefit the environment (McClanahan and Rankin, 2016) can also increase the effectiveness of co-managed areas hence improving the ability of nations to meet Aichi Target 11 and other SDGs goals.

6. Improving overall governance of the coastal zone

Coral reefs exist within a much larger spatial and social context. They are therefore impacted by other anthropogenic stressors such as coastal development, tourism, land-based runoff and threats from mining and other extractive uses and marine litter. These impacts can be managed and mitigated through integrated approaches such as ICZM and MSP that facilitate coordination across multiple jurisdictions and sectors within governments and with stakeholders. Countries in the WIO have made some progress towards implementing ICZM through the development of policies, but this needs to be accelerated to the level of implementation (ASCLME/SWIOFP, 2012).

Recognition of the Blue Economy has also motivated some countries in the WIO to undertake national MSP. The regional MSP strategy being developed for the WIO will serve as a useful guide, especially where shared stocks and resources are concerned. Developments for the Blue Economy often include large infrastructure projects such as ports, mining and industrial fishing, all with potentially harmful environmental impacts.

The Mitigation Hierarchy approach that includes actions that avoid, minimize, remediate and offset impacts is a strategy that can help anticipate and avoid impacts on coral reefs and the ecosystem services they provide. This approach is sometimes mandated by financial lending institutions such as the World Bank and International Finance Corporation and can be incorporated in national ICZM and MSP strategies.



The variety of organisms inhabiting coral reefs: (A) Emperor angelfish (B) butterflyfish; (C) Checkerboard wrasse; (D) clownfish on its host anemone; (E) mating sea slugs; (F) sea cucumber *Pearsonothuria graeffei*; (G) hard coral of the genus *Lobophyllia;* (H) soft coral. © T.R. McClanahan

7. Enhancing research and monitoring

The need for research and monitoring for adaptive management is well understood in the WIO, and there are currently national and regional programs that undertake research and monitoring of coral reefs. Nonetheless, limitations include inadequate funds and low capacity and skills leading to a lack of reliable and long-term monitoring data in many WIO countries. Multidisciplinary research approaches combining natural sciences and social sciences should be further developed.

There is also the challenge of low capacity in some crucial areas such as economic valuation, ocean acidification, marine litter and social change. A regional evaluation building on past research and capacity building assessments (Francis et al., 2015) would help guide management courses, higher education training and post graduate studies and provide information on data gaps to better inform management. Knowledge transfer can be improved by using approaches such as joint research design, knowledge co-production and collaboration with academic institutions, NGOs, civil groups and communities.

In addition, packaging research findings so that information is more accessible to targeted groups will enhance knowledge uptake and use of this information for management. Information from research and monitoring is a powerful tool to help increase understanding of the value of reefs that is essential for generating the political and community will to improve sustainable use and conservation of coral reefs.

8. Enhancing outreach and stakeholder engagement

Human impacts on coral reefs can be minimized if stakeholders are aware and engaged in reducing harmful practices. This can be achieved by promoting the growth of an informed and engaged public. The importance of marine and coastal ecosystems and their value for national and local economies is broadly understood in the WIO. However, there is less understanding on the linkages between ecosystem health, food security, economic development, climate change, and effective management. Because of the projected severe consequences of climate change on coral reefs, there is an urgent need for more targeted programs to raise awareness, experiential exchanges, and knowledge forums specific to coral reefs. A harmonized and targeted program is needed that can build on long term learning and information exchange programs such as the Fishers' forum in Kenya, and the *MItantana HArena an-dRanomasina avy eny Ifotony* (MIHARI) in Madagascar, and networks such as the BMU network in Kenya, and the national coral reef task forces.

Awareness programs are standard activities with NGOs and regional programs that support linkages, and partnerships between these programs and stakeholders from fisher/reef user groups to management/policymakers can improve the effectiveness of information exchange. Other opportunities for raising awareness include programs that engage communities in monitoring (citizen science) to increase understanding of the value of coral reefs.

CONCLUSIONS AND RECOMMENDATIONS

Climate change and overfishing disturbances are the greatest threats to WIO coral reefs. These threats are exacerbated by increased resource use driven by population growth, expansion of economic activities in the coastal zone and other pressures mainly from anthropogenic sources. Coral reefs generate many benefits for local and national economies and the livelihoods of WIO communities. It is important therefore, that management and conservation efforts are provided with the needed resources for adequately reducing these pressures.

Although countries of the WIO have invested in many programs and initiatives to protect and manage coral reefs, more concerted effort is urgently needed because coral reefs are in imminent danger due to climate change disturbances, fishing and the drive for coastal development and the Blue Economy. The suggestions detailed above, therefore, focus on practical solutions that can accelerate coral reef conservation and restoration. The science to implement better management has increased considerably in the past few decades, and now it is time to act on these recommendations.

CASE STUDY

Models of fish biomass recovery

Kendall Raward Jones

Broad-scale overharvesting of fish is one of the major drivers of marine biodiversity loss and poverty in the WIO, particularly in countries with large populations and high dependence on coral reefs for food and income. Given that fishing effort and management success vary spatially across the WIO, and the general scarcity of management resources, it is necessary to identify broad-scale locations for promoting successful fisheries management and conservation. There are many objectives and strategies used to prioritize locations for fisheries management and conservation, so here we explored numerous strategies for doing so. We utilized models of fish biomass recovery across the WIO, combined with data on fishing effort and management feasibility to assess how fisheries management and conservation priorities in the WIO would change if the objectives were to (1) minimize lost fishing opportunity, (2) minimize the time for fish biomass to recover, (3) avoid locations of low management feasibility based on historical management outcomes, and (4) incorporate international collaboration to optimize the rate for achieving goals. All objectives aimed to include 50% of coral reef in sustainable fishing zones, and 20% of coral reef in conservation zones. The general priority areas remained similar across most objectives, with conservation areas located in remote, high-biomass areas (e.g. Seychelles, Chagos), and sustainable fishing areas located along the East African

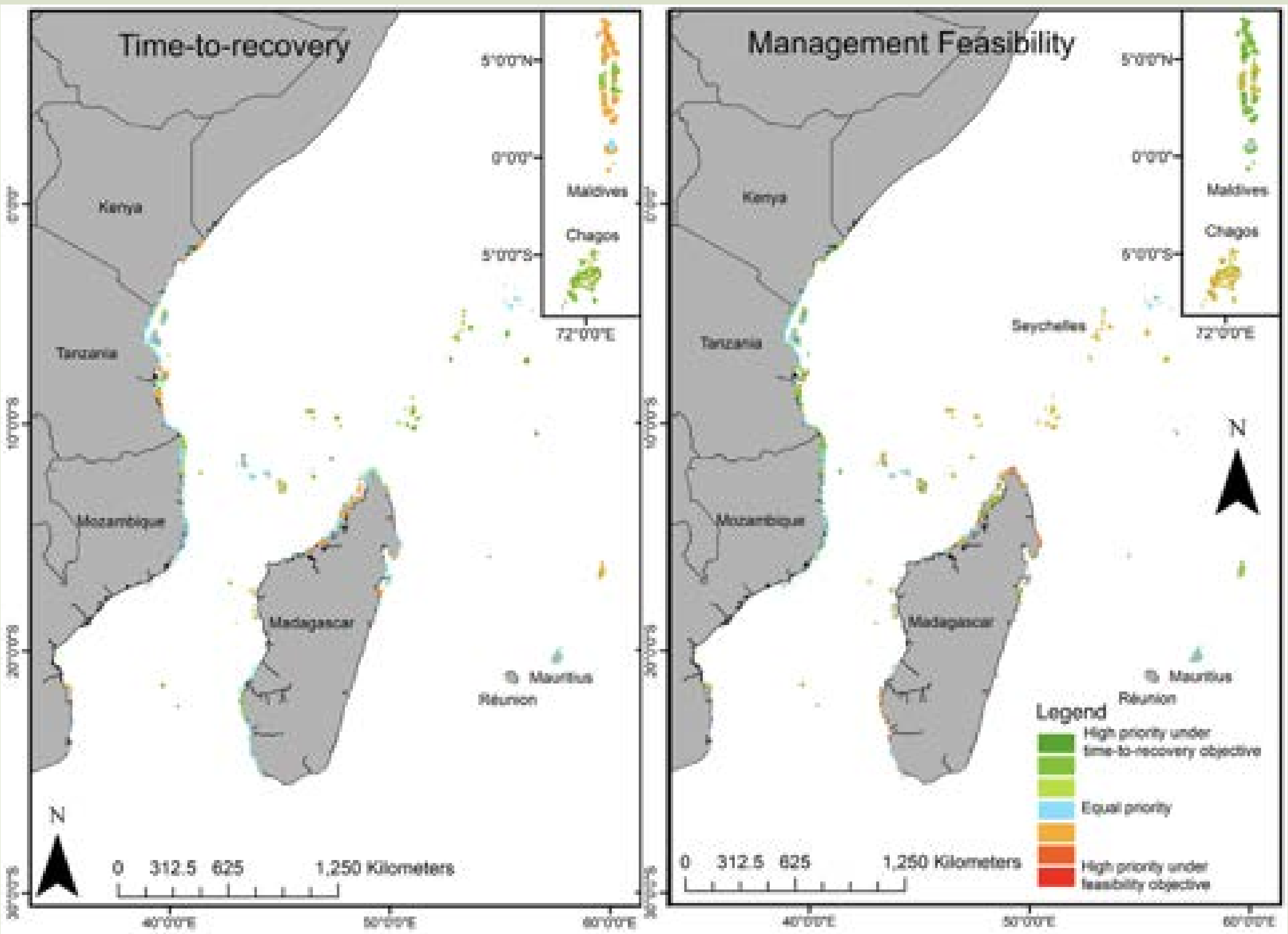


Figure 5: Difference in planning unit selection frequency for (a) conservation zones, and (b) sustainable fishing zones when aiming to minimize the time required for fish biomass recovery (green areas) and when aiming to avoid locations of low management feasibility (red areas). Planning units are blue if they had equal selection frequencies under both objectives.

coast and Madagascar. When aiming to minimize lost fishing opportunities, the total area of conservation zones was much larger than when aiming to minimize fish biomass recovery time or avoid locations of low management feasibility. Average time-to-recovery increased six-fold for conservation zones when considering management feasibility compared to simply aiming to minimize fish biomass recovery, but sustainable fishing zones had similar fish biomass recovery times.

When allowing sustainable fishing and conservation zones to be spread across the entire WIO, rather than placed in each nation individually, the total area of management zones and the time required for fish biomass recovery were substantially reduced. Even when collaboration only occurred between member nations of the Nairobi Convention and the Indian Ocean Commission, the total area required for management zones was almost halved.

Our results show that incorporating management feasibility into spatial prioritizations can help avoid spending resources where effective management seems unlikely, and that collaboration between nations can increase the efficiency of management plans (ie decrease total area and cost of management zones). Furthermore, using fish biomass recovery models to provide information on the length of time required for management to meet demonstrable ecological targets should increase knowledge and gain support from stakeholders. However, these prioritization strategies result in an uneven distribution of management priorities and may further burden people in poorer countries where effective fishery management is badly needed to promote food security. It is clear that for spatial prioritization analyses to be useful and incorporated into decision making, many possible values, incentives, scenarios, and metrics must be considered.

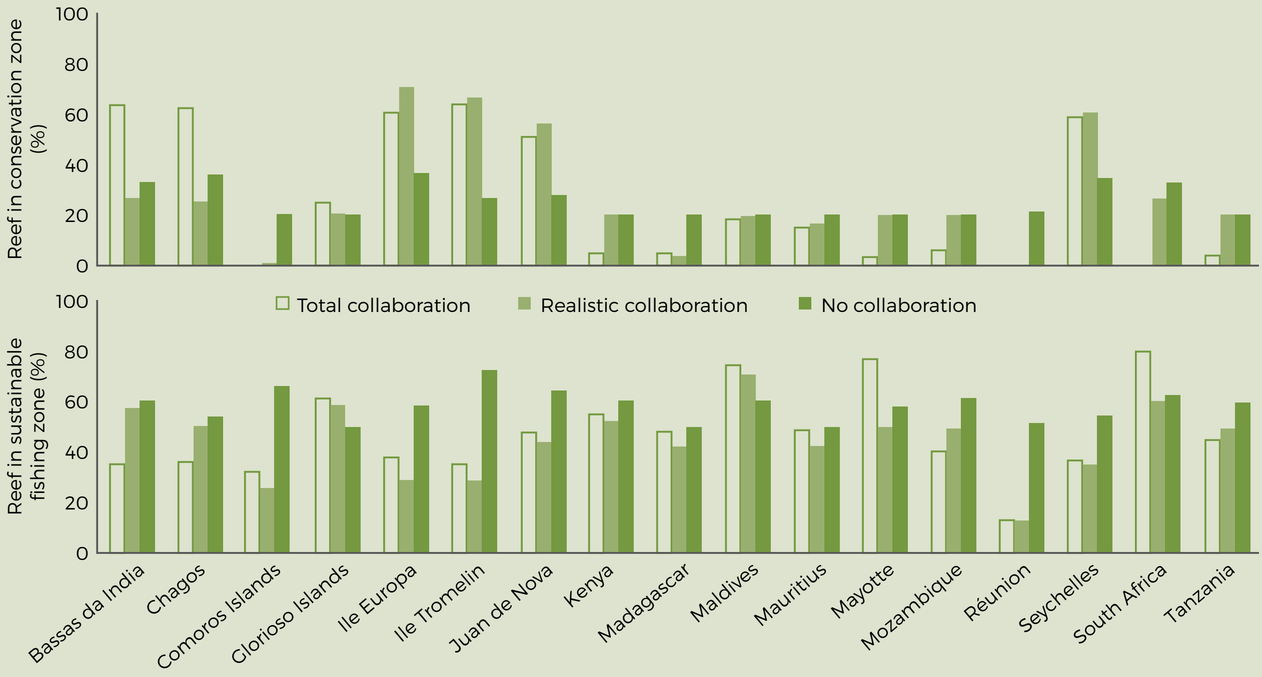


Figure 6: Percentage of reef of each country contained in priority areas identified under three international collaboration scenarios when aiming to minimize the time required for fish biomass recovery. All scenarios aim to include 50% of reef area in sustainable fishing zones and 20% of reef area in conservation zones. In the total collaboration scenario these targets can be met across the entire WIO, whereas in the no collaboration scenario the targets must be met for each country individually. The partial collaboration scenario allows collaboration between Kenya and Tanzania (members of the Nairobi convention) and between Comoros, Madagascar, Mauritius, Seychelles and Réunion (members of the Indian Ocean Commission).

REFERENCES

Albright, R., Mason, B., Miller, M. and Langdon, C. (2010). Ocean acidification compromises recruitment success of the threatened Caribbean coral *Acropora palmata*. *PNAS* 107(47), pp. 20400–20404

Andréfouët, S., Muller-Karger, F.E., Robinson, J.A., Kranenburg, C.J., Torres-Pulliza, D., Spraggins, S.A. and Murch, B. (2006). Global assessment of modern coral reef extent and diversity for regional science and management applications: a view from space. Proceedings of 10th International Coral Reef Symposium, pp. 1732–1745

Andréfouët, S., Guillaume, M. M., Delval, A., Rasoamanendrika, F. M. A., Blanchot, J. and Bruggemann, J. H. (2013). Fifty years of changes in reef flat habitats of the Grand Récif of Toliara (SW Madagascar) and the impact of gleaning. *Coral Reefs* 32(3), pp. 757–768.

Arthurton, R. (2003). The fringing reef coasts of eastern Africa-present processes in their long-term context. *West. Indian Ocean J. Mar. Sci.* 2(1), pp. 1–14

ASCLME/SWIOFP (2012). Transboundary diagnostic analysis for the western Indian Ocean Volume 1Baseline. pp 122

Ateweberhan, M. and McClanahan, T.R. (2016). Partitioning scleractinian coral diversity across reef sites and regions in the Western Indian Ocean. *Ecosphere* 7(5), e01243

Ateweberhan, M., McClanahan, T.R., Graham, N.A.J. and Sheppard, C.R.C. (2011). Episodic heterogeneous decline and recovery of coral cover in the Indian Ocean. *Coral Reefs* 30(3), pp. 739–752

Ateweberhan, M., McClanahan, T.R., Maina, J. and Sheppard, C. (2018). Thermal energy and stress properties as the main drivers of regional distribution of coral species richness in the Indian Ocean. *J. Biogeogr.* 45(6), pp. 1355–1366

Ban, S.S., Graham, N.A. and Connolly, S.R. (2014). Evidence for multiple stressor interactions and effects on coral reefs. *Glob. Change Biol.* 20(3), pp. 681–697

Benbow, S., Humber, F., Oliver, T.A., Oleson, K.L.L., Raberinary, D., Nadon, M., Ratsimbazafy, H. and Harris, A. (2014). Lessons learnt from experimental temporary octopus fishing closures in southwest Madagascar: benefits of concurrent closures. *Afri. J. Mar. Sci* 36 (1), pp. 31–37

Ben-Haim, Y. and Rosenberg, E. (2002). A novel Vibrio sp. pathogen of the coral *Pocillopora damicornis. Mar. Biol* 141(1), pp. 47–55

Beyer, H.L., Kennedy, E.V., Beger, M., Chen, C.A., Cinner, J.E., Darling, E.S., Eakin, R., Gates, D., Heron, S.F., Knowlton, N. and Obura, D.O. (2018). Risk-sensitive planning for conserving coral reefs under rapid climate change. *Conserv. Lett.* 11(6), e12587

Bigot, L., Chabanet, P., Cuet, P., Cauvin, B., Durville, P., Mulochau, T., Naim, O., Nicet, J.B., Tessier, E., Thomassin, B. and Wickel, J. (2019). World seas: An environmental evaluation Volume 2: The Indian ocean to the pacific – French Territories in the Western Indian Ocean, *Elsevier Academic press*, pp. 279–302

Burge, C.A., Kim, C.J., Lyles, J.M. and Harvell, C.D. (2013). Special issue oceans and humans health: The ecology of marine opportunists. *Microb. Ecol.* 65(4), pp. 869–879

Burke, L., Reytar, K., Spalding, M. and Perry, A. (2011). Reefs at risk revisited (eds. E. Cooper, B. Kushner, E. Selig, B. Starkhouse, K. Teleki, R. Waite, C. Wilkinson and T. Young.). World Resources Institute. Washington, D.C.

Camp, E.F., Suggett, D.J., Gendron, G., Jompa, J., Manfrino, C. and Smith, D.J. (2016). Mangrove and seagrass beds provide different biogeochemical services for corals threatened by climate change. *Front. Mar. Sci.* 3(52). https://doi.org/ 10.3389/fmars.2016.00052

Celliers, L. and Ntombela, C. (2015). Urbanization, coastal development and vulnerability, and catchments. In *Regional State of the Coast Report: Western Indian Ocean* Nairobi, Kenya (ed. J. Paula) pp. 286–404. UNEP-Nairobi Convention and WIOMSA

Celliers, L. and Schleyer, M.H. (2002). Coral bleaching on high-latitude marginal reefs at Sodwana Bay, South Africa. *Mar. Pollut. Bull.* 44(12), pp. 1380–1387

Chabanet, P., Bigot, L., Nicet, J.B., Durville, P., Massé, L., Mulochau, T., Russo, C., Tessier, E. and Obura, D. (2016). Coral reef monitoring in the Iles Eparses, Mozambique Channel (2011–2013). *Acta Oecol.(Montrouge)* 72, pp. 62–71

Chadwick, P. (2021). Part IV: Management Effectiveness. In *Western Indian Ocean Marine Protected Areas Outlook: Towards Achievement of the Global Biodiversity Framework Targets* (eds. L. Sisitka and M.D. Richmond) pp. 229–256. UNEP-Nairobi Convention and WIOMSA. Nairobi, Kenya

Cinner, J.E., Huchery, C., MacNeil, M.A., Graham, N.A., McClanahan, T.R., Maina, J., Maire, E., Kittinger, J.N., Hicks, C.C., Mora, C., Allison, E.H., D'Agata, S., Hoey, A., Feary, D.A., Crowder, L., Williams, I.D., Kulbicki, M., Vigliola, L., Wantiez, L., Edgar, G., Stuart-smith, R.D., Sandin, S.A., Green, A.L., Hardt, M.J., Beger, M., Friedlander, A., Campbell, S.J., Holmes, K.E., Wilson, S.K., Brokovish, E., Brooks, A.J., Curz-Motta, J.J., Booth, D.J., Chabanet, P., Gough, C., Tupper, M., Ferse, S.C.A., Sumaila, R.U. and Mouillot D. (2016). Bright spots among the world's coral reefs. *Nature* 535(7612), pp. 416–419

Cinner, J.E., Daw, T.M., McClanahan, T.R., Muthiga, N., Abunge, C., Hamed, S., Mwaka, B., Rabearisoa, A., Wamukota, A., Fisher, E. and Jiddawi, N. (2012). Transitions towards co-management: the process of marine resource management devolution in three east African countries. *Glob. Environ. Change* 22 (3), pp. 651–658

Clark, A.M. and Rowe, F.W.E. (1971). *Monograph of the Shallow-Water Indo-West Pacific Echinoderms* (Vol. 690). London: Trustees of the British Museum (Natural History)

Connolly, S.R., Bellwood, D.R. and Hughes, T.P. (2003). Indo-Pacific biodiversity of coral reefs: Deviations from a mid-domain model. *Ecology* 84(8), pp. 2178–2190

Coral Reef Task Force (2008). Strengthening capacity for the sustainable management of coral reefs in the western Indian Ocean. Coral Reef Task Force. pp. 36

Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S.J., Kubiszewski, I., Farber, S. and Turner, R.K. (2014). Changes in the global value of ecosystem services. *Glob. Environ. Change* 26, pp. 152–158

Crochelet, E., Roberts, J., Lagabrielle, E., Obura, D., Petit, M. and Chabanet, P. (2016). A model-based assessment of reef larvae dispersal in the western Indian Ocean reveals regional connectivity patterns - Potential implications for conservation policies. *Reg. Stud. Mar. Sci. 7*, pp. 159–167

D'agata, S., Mouillot, D., Wantiez, L., Friedlander, A.M., Kulbicki, M. and Vigilola, M. (2016). Marine reserves lag behind wilderness in the conservation of key functional roles. *Nat. Commun.* 7, 12000. https://doi.org/10.1038/ncomms12000

Darling, E.S., McClanahan, T.R. and Côté, I.M. (2010). Combined effects of two stressors on Kenyan coral reefs are additive or antagonistic, not synergistic. *Conserv. Letters* 3(2), pp. 122–130

Darling, E.S., McClanahan, T.R. and Côté, I.M. (2013). Life histories predict coral community disassembly under multiple stressors. *Glob. Change. Biol.* 19(6), pp. 1930–1940

Duhec, AV., Jeanne, RF., Maximenko, N. and Hafner, J. (2015). Composition and potential origin of marine debris stranded in the Western Indian Ocean on remote Alphonse Island, Seychelles. *Mar. Pollut. Bull.* 96(1–2), pp. 78–96

Eklöf, J.S., Henriksson, R. and Kautsky, N. (2006). Effects of tropical open-water seaweed farming on seagrass ecosystem structure and function. *Mar. Ecol. Prog. Ser.* 325, pp. 73–84

Faure, G., Guillaume, M., Payri, C., Thomassin, B.A., Vanpraet, M. and Vasseur, P. (1984). Massive bleaching and death of corals in the Mayotte Reef ecosystem (SW Indian-Ocean). *Comptes Rendus Academie Des Sciences Serie III -Sciences De La Vie-Life Sciences*, 299(15), pp. 637–642

Fleitmann, D., Dunbar, R.B., McCulloch, M., Mudelsee, M., Vuille, M., McClanahan, T.R., Cole, J.E. and Eggins, S. (2007). East Africa soil erosion recorded in a 300 year old coral colony from Kenya. *Geophys. Res. Lett.* 34(4), L04401

Fontanier, C., Mamo, B., Toucanne, S., Bayon, G., Schmidt, S., Deflandre, B., Dennielou, B., Jouet, G., Sakai, S. and Lamas, R.M. (2018). Are deep-sea ecosystems surrounding Madagascar threatened by land-use or climate change? *Deep Sea Res. Part I: Oceanogr. Res. Pap.* 131, pp. 93–100

Francis, J., Celliers, L. and Rosendo, S. (2015). Coastal and marine research and capacity building. In *Regional State of the Coast Report: Western Indian Ocean* (ed. J. Paula), pp. 473–494. UNEP-Nairobi Convention and WIOMSA. Nairobi, Kenya

Friedlander, A.M., Obura, D., Aumeeruddy, R., Ballesteros, E., Church, J., Cebrian, E. and Sala, E. (2014). Coexistence of low coral cover and high fish biomass at Farquhar Atoll, Seychelles. *PLoS One*, 9(1), e87359. https://doi.org/10.1371/journal.pone.0087359

Fröcklin, S., de la Torre-Castro, M., Lindström, L., Jiddawi, N.S. and Msuya, F.E. (2012). Seaweed mariculture as a development project in Zanzibar, East Africa: a price too high to pay? *Aquaculture* 356–357, pp. 30–39

Gardner, C.J., Cripps, G., Day, L.P., Dewar, K., Gough, C., Peabody, S. and Harris, A. (2020). A decade and a half of learning from Madagascar's first locally managed marine area. *Conserv. Sci. Pract*. 2(12), e298

Goreau, T., McClanahan, T., Hayes, R. and Strong, A. (2000). Conservation of coral reefs after the 1998 global bleaching event. *Conserv. Biol.* 14(1), pp. 5–15

Graham, N.A.J., Cinner, J.E., Holmes, T.H., Huchery, C., MacNeil, M.A., McClanahan, T.R. and Wilson, S.K. (2017). Human disruption of coral reef trophic structure. *Curr. Biol.* 27(2), pp. 231–236

Grove, C.A., Zinke, J., Scheufen, T., Maina, J., Epping, E., Boer, W. and Brummer, G.J. (2012). Spatial linkages between coral proxies of terrestrial runoff across a large embayment in Madagascar. *Biogeosciences* 9(8), pp. 3063–3081

Harvell, C.D., Kim, K., Burkholder, J.M., Colwell, R.R., Epstein, P.R., Grimes, D.J., Hoffmann, E.E., Lipp, E.K., Osterhaus, A.D.M.E., Overstreet, R.M., Porter, J.W. and Vasta G.R. (1999). Emerging marine diseases-climate links and anthropogenic factors. *Science* 285(5433), pp. 1505–1510

Harvell, D., Altizer, S., Cattadori, I.M., Harrington, L. and Weil, E. (2009). Climate change and wildlife diseases: when does the host matter the most? *Ecology* 90(4), pp. 912–920

Hicks, C.C. (2011). How do we value our reefs? Risks and tradeoffs across scales in "biomass-based" economies. *Coast. Manage*. 39(4), pp. 358–376

Hoareau, T.B., Boissin, E., Paulay, G. and Bruggemann, J.H. (2013). The Southwestern Indian Ocean as a potential marine evolutionary hotspot: Perspectives from comparative phylogeography of reef brittle-stars. *J. Biogeogr.* 40(11), pp. 2167–2179

Hoegh-Guldberg, O. (2011). Coral reef ecosystems and anthropogenic climate change. *Reg. Environ. Change* 11(1), pp. 215–227

Hoegh-Guldberg, O., Mumby, P.J., Hooten, A.J., Steneck, R.S., Greenfield, P., Gomez, E. and Hatziolos, M.E. (2007). Coral reefs under rapid climate change and ocean acidification. *Science* 318(5857), pp. 1737–1742

Hughes, T.P., Anderson, K.D., Connolly, S.R., Heron, S.F., Kerry, J.T., Lough, J.M, Baird, A.H., Baum, J.K., Berumen, M.L., Bridge, T.C. and Claar, D.C. (2018). Spatial and temporal patterns of mass bleaching of corals in the Anthropocene. *Science* 359(6371), pp. 80–83

IPCC (2014). Climate change (2014). Synthesis Report. Contribution of working groups I, II, and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change (eds. Core writing team, R.K. Pachauri and L.A. Meyer). IPCC, Geneva, Switzerland, 151 pp.

IUCN (2017). The IUCN Red List of Threatened Species. Version 2017. IUCN: Gland, Switzerland. https://www.iucnredlist.org

Jonas, H. and Sandwith, T. (eds.) (2019). Towards Recognising, Reporting and Supporting OECMs: Reportof the Fourth Expert Meeting of the IUCN-WCPA Task Force on Other Effective Area-based Conservation Measures. IUCN: Gland, Switzerland

Jones, K.R., Maina, J.M., Kark, S., McClanahan, T.R., Klein, C.J. and Beger, M. (2018). Incorporating feasibility and collaboration into large-scale planning for regional recovery of coral reef fisheries. *Mar. Ecol. Prog. Ser.* 604, pp. 211–222

Jordan, I.E. and Samways, M.J. (2001). Recent changes in coral assemblages of a South African coral reef, with recommendations for long-term monitoring. *Biodivers. Conserv.* 10(7), pp. 1027–1037

Kawaka, J.A., Samoilys, M.A., Murunga, M., Church, J., Abunge, C. and Maina, GW. (2017). Developing locally managed marine areas: Lessons learnt from Kenya. *Ocean Coast. Manag.* 135, pp. 1–10

Kebede, A.S., Brown, S. and Nicholls, R.J. (2012). *Synthesis Report: The Implications of climate change and sea-level rise in Tanzania – The Coastal Zones.* Report submitted to the Global Climate Adaptation Partnership (GCAP). Oxford, UK.

Kulbicki, M., Parravicini, V. and Mouillot, D. (2015). Patterns and processes in reef fish body size. In *Ecology of Fishes on Coral Reefs* (ed. C. Mora), pp. 104–115. Cambridge University Press, Cambridge, UK

Lamb, J.B., Willis, B.L., Fiorenza, E.A., Couch, C.S., Howard, R., Rader, D.N., True, J.D., Kelly, L.A., Ahmad, A., Jompa, J. and Harvell, D.C. (2018). Plastic waste associated with disease on coral reefs. *Science* 359(6374), pp. 460–462

Lane, S.B., Ahamada, S., Gonzalves, C., Lukambuzi, L., Ochiewo, J., Pereira, M. and Seewoobaduth, J. (2007). *Regional overview and assessment of marine litter related activities in the West Indian Ocean region*. Report to the United Nations Program, pp. 1–91

Levin, N., Beger, M., Maina, J., McClanahan, T. and Kark, S. (2018). Evaluating the potential for transboundary management of marine biodiversity in the Western Indian Ocean. *Australas. J. Environ. Manag.* 25(1), pp. 62–85

Lieske, E. and Myers, R.M. (1994). *Collins' Hand Guide to the Coral Reef Fishes of the World*. London: Harper-Collins.

Maina, J.M., Gamoyo, M., Adams, V.M., D'agata, S., Bosire, J., Francis, J. and Waruinge, D. (2020). Aligning marine spatial conservation priorities with functional connectivity across maritime jurisdictions. *Conserv. Sci. Pract.* 2(2), e156

Maina, J., de Moel, H., Vermaat, J.E., Bruggemann, J.H., Guillaume, M.M., Grove, C.A. and Zinke, J. (2012). Linking coral river runoff proxies with climate variability, hydrology and land-use in Madagascar catchments. *Mar. Pollut. Bull.* 64(10), pp. 2047–2059

Maina, J., de Moel, H., Zinke, J., Madin, J., McClanahan, T.R. and Vermaat, J.E. (2013). Human deforestation outweighs future climate change impacts of sedimentation on coral reefs. *Nature Communications* 4(1), pp. 1–7. https://doi.org/10.1038/ncomms2986

Maina, J., McClanahan, T.R., Venus, V., Ateweberhan, M. and Madin, J. (2011). Global gradients of coral exposure to environmental stresses and implications for local management. *PLoS One* 6(8), e23064. https://doi.org/10.1271/journal.pone.0023064

Margules, C.R. and Pressey, R.L. (2000). Systematic conservation planning. *Nature* 405(6783), pp. 243–253

Mbaru, E.K. and McClanahan, T.R. (2013). Escape gaps in African basket traps reduce bycatch while increasing body sizes and incomes in a heavily fished reef lagoon. *Fish. Res*. 148, pp. 90–99

McClanahan, T.R. (2018). Multicriteria estimate of coral reef fishery sustainability. *Fish. Fish*. 19(5), pp. 807–820

McClanahan, T.R. (2019). Functional communities, diversity, and fisheries status of coral reefs fish in East Africa. *Mar. Ecol. Prog. Ser*. 632, pp. 175–191

McClanahan, T.R. (2020a). Decadal turnover of thermally stressed coral taxa support a risk-spreading approach to marine reserve design. *Coral Reefs* 39, pp. 1549–1563

McClanahan, T.R. (2020b). Coral community life histories and population dynamics driven by seascape bathymetry and temperature variability. In *Advances in Marine Biology: Population Dynamics of The Reef Crisis* (eds. B. Reigl and P.W. Glynn), vol. 87, pp. 291–230. Academic Press, London, UK

McClanahan, T.R. and Azali, M.K. (2020). Improving sustainable yield estimates for tropical reef fisheries. *Fish. Fish.* 21(4), pp. 683–699

McClanahan, T.R. and Jadot, C. (2017). Managing coral reef fish community biomass is a priority for biodiversity conservation in Madagascar. *Mar. Ecol. Prog. Ser.* 580, pp. 169–190

McClanahan, T.R. and Muthiga, N.A. (2017). Environmental variability indicates a climate-adaptive center under threat in northern Mozambique coral reefs. *Ecosphere* 8(5), e01812

McClanahan, T.R. and Muthiga, N.A. (2021). Oceanic patterns of thermal stress and coral community degradation on the island of Mauritius. *Coral Reefs* 40, pp. 53–74

McClanahan, T.R. and Obura, D. (1997). Sedimentation effects on shallow coral communities in Kenya. *J. Exp. Mar. Biol. Ecol.* 209(1), pp. 103–122

McClanahan, T.R. and Rankin, P.S. (2016). Geography of conservation spending, biodiversity, and culture. *Conserv. Biol.* 30(5), pp. 1089–1101

McClanahan, T.R., Ateweberhan, M., Darling, E.S., Graham, N.A.J., and Muthiga, N.A. (2014). Biogeography and change among regional coral communities across the Western Indian Ocean. *PLoS One* 9(4), e93385

McClanahan, T.R., Baird, A.H., Marshall, P.A. and Toscano, M.A. (2004). Comparing bleaching and mortality responses of hard corals between southern Kenya and the Great Barrier Reef, Australia. *Mar. Pollut. Bull.* 48(3), pp. 327–335

McClanahan, T.R., Friedlander, A.M., Graham, N.A., Chabanet, P. and Bruggemann, J.H. (2021). Variability in coral reef fish baseline and benchmark biomass in the central and western Indian Ocean provinces. *Aquat. Conser: Mar. Freshw. Ecosyst*. 31(1), pp. 28–42. https://doi.org/10.1002/aqc.3448

McClanahan, T.R., Graham, N.A.J., MacNeil, M.A. and Cinner, J.E. (2015). Biomass-based targets and the management of multispecies coral reef fisheries. *Conserv. Biol.* 29(2), pp. 409–417

McClanahan, T.R., Maina, J.M., Darling, E.S., Guillaume, M.M.M., Muthiga, N.A., D'agata, S., Leblond, J., Arthur, R., Jupiter, S.D., Wilson, S.K., Mangubhai, S., Ussi, A.M., Humphries, A.T., Patankar, V., Shedrawi, G., Pagu, J., Ndagala, J. and Grimsditch, G. (2020). Large geographic variability in the resistance of corals to thermal stress. *Glob. Ecol. Biogeogr.* 29(12), pp. 2229–2247

McClanahan, T.R., Maina, J.M., Graham, N.A. and Jones, K.R. (2016). Modeling reef fish biomass, recovery potential, and management priorities in the Western Indian Ocean. *PLoS One* 11(5), e0154585

Moulin, L., Grosjean, P., Leblud, J., Batigny, A. and Dubois, P. (2014). Impact of elevated pCO2 on acid–base regulation of the sea urchin *Echinometra mathaei* and its relation to resistance to ocean acidification: A study in mesocosms. *J. Exp. Mar. Biol. Ecol*. 457, pp. 97–104

MPRU/KWS (2015). A Proposed Marine Transboundary Conservation Area Between Kenya and Tanzania. Joint Technical Paper. Marine Parks Reserves Unit, United Republic of Tanzania and Kenya Wildlife Service, Kenya. pp. 72

Muthiga, N.A., Riedmiller, S., Carter, E., van der Elst, R., Mann-Lang, J., Horrill, C. and McClanahan, T.R. (2000). Management status and case studies. In *Coral Reefs of the Indian Ocean: Their Ecology and Conservation* (ed. T.R. McClanahan, C.S. Sheppard, and D. Obura) pp. 473–505. Oxford University Press, New York

Negri, A.P., Marshall, P.A. and Heyward, A.J. (2007). Differing effects of thermal stress on coral fertilization and early embryogenesis in four Indo Pacific species. *Coral Reefs*, 26(4), pp. 759–763

Obura, D. (2012). The diversity and biogeography of Western Indian Ocean reef-building corals. *PloS One* 7(9), e45013

Obura, D. (2015). Coral and biogenic reef habitats. In *Regional State of the Coast Report: Western Indian Ocean* (ed. J. Paula). UNEP-Nairobi Convention and WIOMSA, Nairobi, Kenya

Obura, D., Bigot, L. and Benzoni, F. (2018). Coral responses to a repeat bleaching event in Mayotte in 2010. *Peer. J.* 6, e5305

Obura, D., Smits, M., Chaudhry, T., McPhillips, J., Beal, D. and Astier, C. (2017). Reviving the Western Indian Ocean economy: actions for a sustainable future." *World Wide Fund for Nature (Formerly World Wildlife Fund), Gland, Switzerland*. 63 pp.

Obura, D.O. (2016). An Indian Ocean centre of origin revisited: Palaeogene and Neogene influences defining a biogeographic realm. *J. Biogeogr.* 43(2), pp. 229–242

Obura, D.O., Burgener, V., Nicoll, M.E., Ralison, H.O. and Scheren, P. (2015). The northern Mozambique Channel. Setting the foundations for a regional approach to marine governance. A background document. WWF International and CORDIO East Africa

Parravicini, V., Kulbicki, M., Bellwood, D.R., Friedlander, A. M., Arias-Gonzalez, J.E., Chabanet, P., Floeter, S.R., Myers, R., Vigliola, L., D'Agata, S. and Mouillot, D. (2013). Global patterns and predictors of tropical reef fish species richness. *Ecography* 36(12), pp. 1254–1262.

Paulay, G. (1997). Diversity and distribution of reef organisms. In *Life and Death of Coral Reefs* (ed. C.E. Birkeland) pp. 298–353. Chapman and Hall, New York

Pellissier, L., Leprieur, F., Parravicini, V., Cowman, P.F., Kulbicki, M., Litsios, G., Olsen, S.M., Wisz, M.A., Bellwood, D.R. and Mouillot, D. (2014). Quaternary coral reef refugia preserved fish diversity. *Science* 344(6187), pp. 1016–1019

Peters, E.C. (1993). Diseases of other invertebrate phyla: Porifera, Cnidaria, Ctenophora, Annelida, Echinodermata. In *Pathobiology of Marine and Estuarine Organisms* (eds. J.A. Couch and J.W. Fourine), pp. 393–449. CRRC Press, Boca Raton, Florida

Richmond, M.D. and Sisitka, L. (2021). Part III : Summary of MPAs : Classification, characterization and main achievements in relation to conservation targets. In *Western Indian Ocean Marine Protected Areas Outlook: Towards Achievement of the Global Biodiversity Framework Targets*, (eds. L. Sisitka and M.D. Richmond) pp. 215–228. UNEP-Nairobi Convention and WIOMSA. Nairobi, Kenya

Richmond, M.D., Horigue, V., Maina, J. and Tuda, A.O. (2021). Part V: Meeting the global goals and marine biodiversity conservation targets. In *Western Indian Ocean Marine Protected Areas Outlook: Towards Achievement of the Global Biodiversity Framework Targets*, pp. 279–298. UNEP-Nairobi Convention and WIOMSA. Nairobi, Kenya

Ridgway, T. and Sampayo, E.M. (2005). Population genetic status of the Western Indian Ocean: What do we know? *West. Indian Ocean J. Mar. Sci.* 4(1), pp. 1–10

Riegl, B. (1996). Hermatypic coral fauna of subtropical southeast Africa: A checklist. *Pac. Sci.* 50(4), pp, 404–414

Robinson, J., Samoilys, M., Grandcourt, E.M., Danny, J., Cedras, M. and Calvin, G. (2011). The importance of targeted spawning aggregation fishing to the management of Seychelles' trap fishery. *Fish. Res.* 112, pp. 96–103

Rocliffe, S., Peabody, S., Samoilys, M. and Hawkins, J.P. (2014). Towards a network of locally managed marine areas (LMMAs) in the Western Indian Ocean. *PLoS One* 9(7), e103000

Rosen, B.R. (1971). The distribution of reef coral genera in the Indian Ocean. In *Regional Variation in Indian Ocean Coral Reefs* (eds. D.R. Stoddart and M. Yonge), pp. 263–299, Symp. Zool. Soc. Lond. Academic Press, London

Samoilys, M., Osuka, K., Harris, A. and Muthiga, N. (2017). Locally managed fisheries in the Western Indian Ocean: A review of past and present initiatives. WIOMSA Book Series 17, iv+40pp.

Schleyer, M.H. (2000). South Africa coral communities. In *Coral reefs of the Indian Ocean: Their Ecology and Conservation* (eds. T.R. McClanahan, C.R.C. Sheppard, and D.O. Obura), pp. 83–105, Oxford University Press, New York

Schleyer, M.H. and Celliers, L. (2003). Biodiversity on the marginal coral reefs of South Africa: What does the future hold? *Zoologische Verhandelingen*. 345, pp. 387–400

Schleyer, M.H., Floros, C., Laing, S.C., Macdonald, A.H., Montoya-Maya, P.H., Morris, T., Porter, S.N. and Seré, M.G. (2018). What can South African reefs tell us about the future of high-latitude coral systems? *Mar. Pollut. Bull*. 136, pp. 491–507

Schleyer, M.H., Kruger, A. and Celliers, L. (2008). Long-term community changes on a high-latitude coral reef in the Greater St Lucia Wetland Park, South Africa. *Mar. Pollut. Bull.* 56(3), pp. 493–502

Scopélitis, J., Andréfouët, S., Phinn, S., Chabanet, P., Naim, O., Tourrand, C. and Done, T. (2009). Changes of coral communities over 35 years: Integrating in situ and remote-sensing data on Saint-Leu Reef (la Réunion, Indian Ocean). *Estuar. Coast. Shelf Sci.* 20(84), pp. 342–352

Séré, M.G., Chabanet, P., Turquet, J., Quod, J-P. and Schleyer M.H. (2015). Identification and prevalence of coral diseases on three Western Indian Ocean coral reefs. *Dis. Aquat. Organ*. 114(3), pp. 249–261

Séré, M.G., Wilkinson, D.A., Schleyer, M.H., Chabanet, P., Quod, J-P. and Tortosa, P. (2016). Characterization of an atypical manifestation of black band disease on *Porites lutea* in the Western Indian Ocean. *Peer J*, 4, e2073

Sheppard, C.R.C. (2000). Coral reefs of the Western Indian Ocean. In *Coral Reefs of the Indian Ocean, Their Ecology and Conservation* (eds. T.R. McClanahan, C.R.C. Sheppard, and D.O. Obura) pp. 3–38. Oxford University Press, New York

Spalding, M.D., Fox, H.E., Allen, G.R., Davidson, N., Ferdana, Z.A., Finlayson, M.A.X., Halpern, B.S., Jorge, M.A., Lombana, A.L., Lourie, S.A. and Martin, K.D. (2007). Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. *BioScience*, 57(7), pp. 573–583

Spalding, M.D., Ravilious, C. and Green, E.P. (2001). *World Atlas of Coral Reefs*. University of California Press. Berkeley, USA

Szmant, A.M. and Gassman, N.J. (1990). The effects of prolonged "bleaching" on the tissue biomass and reproduction of the reef coral *Montastrea annularis*. *Coral Reefs* 8(4), pp. 217–224

Teh, L.C.L. and Sumaila, U.R. (2013). Contribution of marine fisheries to worldwide employment. *Fish Fish*. 14(1), pp. 77–88

Tittensor, D.P., Mora, C., Jetz, W., Lotze, H.K., Ricard, D., Berghe, E.V. and Worm, B. (2010). Global patterns and predictors of marine biodiversity across taxa. *Nature* 466(7310), pp. 1098–1101

van der Ven, R.M., Flot, J-F., Buitrago-López, C. and Kochzius, M. (2020). Population genetics of the brooding coral *Seriatopora hystrix* reveals patterns of strong genetic differentiation in the Western Indian Ocean. *Heredity* 126(2), pp. 351–365

van Hooidonk, R., Maynard, J.A., Manzello, D. and Planes, S. (2014). Opposite latitudinal gradients in projected ocean acidification and bleaching impacts on coral reefs. *Global Change Biol*. 20(1), pp. 103–112

Veron, J.E.N. (1995). *Corals in space and time*. University of New South Wales Press. Sydney

WCMC (1992). Global biodiversity: Status of the earth's living resources. Chapman and Hall, London

WCS (2019). Implementing preparatory activities to initiate the Transboundary Conservation Area (TBCA) between Kenya and Tanzania. Final report for the EU/IOC Biodiversity Project. 79 pp.

Weil, E., Cróquer, A. and Urreiztieta, I. (2009). Yellow band disease compromises the reproductive output of the Caribbean reef-building coral *Montastraea faveolata* (Anthozoa, Scleractinia). *Dis. Aquat. Organ*. 87(1–2), pp. 45–55

Weil, E., Smith, G. and Gil-Agudelo, D.L. (2006). Status and progress in coral reef disease research. *Dis. Aquat. Organ*. 69(1), pp. 1–7

Wells, S. (2004). Assessment of management effectiveness in selected marine protected areas in the Western Indian Ocean. IUCN, Eastern and Southern Africa Programme. Nairobi

Wells, S., Francis, J. and Muthiga, N. (2003). Marine protected area management in the Western Indian Ocean. In *Capacity needs to manage protected areas: Africa* (eds. J. Carabias, J. De la Maza, and R. Cadena), pp. 97–103. The Nature Conservancy, Virginia USA

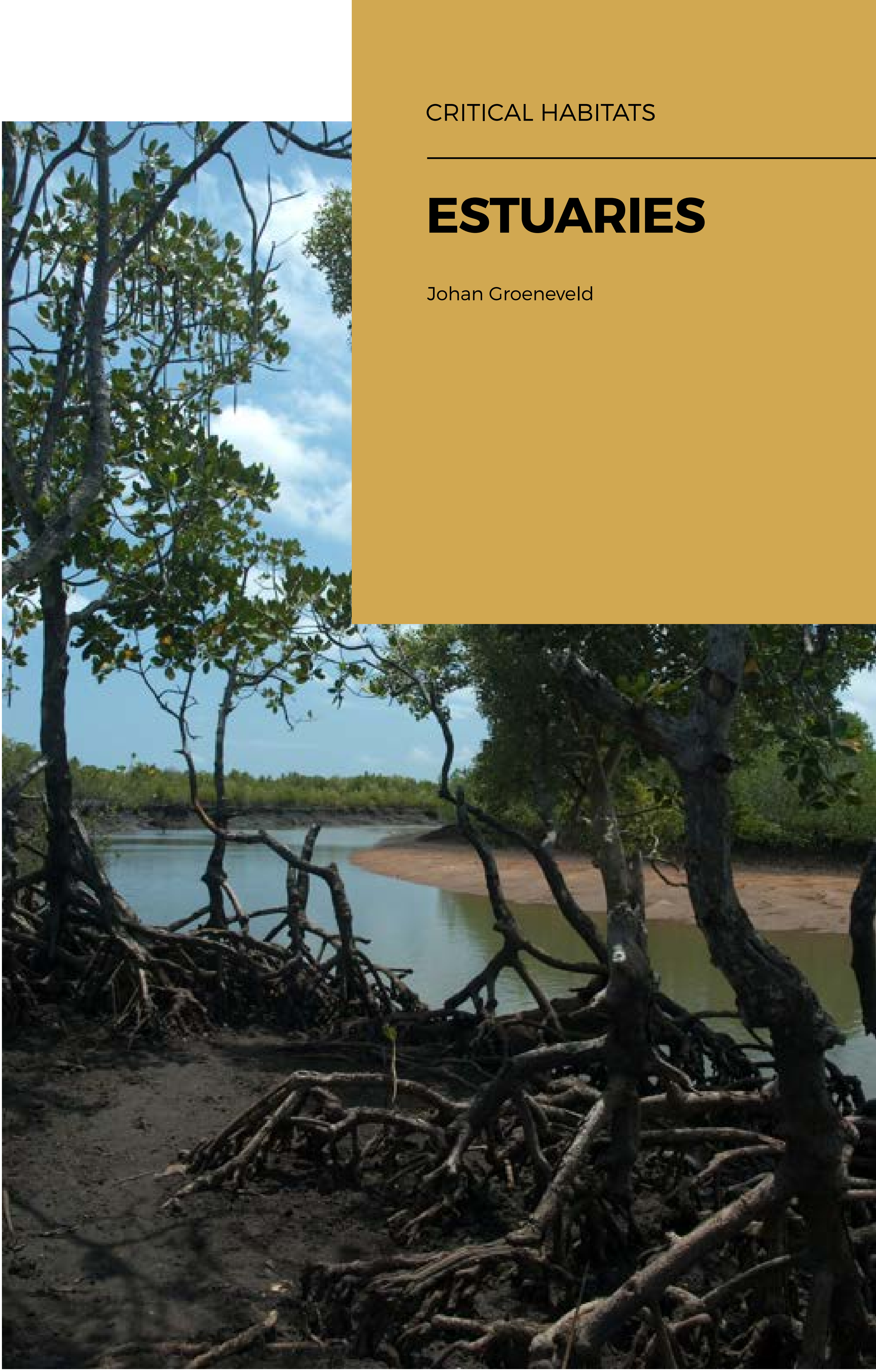
Whittingham, E., Campbell, J. and Townsley, P. (2003). *Poverty and reefs* (Vol. 1). Exeter, UK: IMM Ltd.

WIOMSA (2017). Ocean acidification workshop – Developing regional capacity for ocean observations in support of SDG target 14.3. October 2017. Workshop report. www.wiomsa.org

CRITICAL HABITATS

ESTUARIES

Johan Groeneveld



Creek at Ruvuma delta, Tanzania. © José Paula

BACKGROUND

Estuaries are partially enclosed coastal water bodies where rivers reach the sea. They are dynamic brackish water ecosystems that form the transition between freshwater and marine environments. The mixing of sea and fresh water in estuaries provides high levels of nutrients in the water column and sediments and their shallow depth ensures that biological processes take place near the surface, making them among the most productive natural habitats in the world (Elliott and McLusky, 2002).

Estuaries are home to unique plant and animal communities that have adapted to brackish water, and they export sediments, nutrients and organic matter to nearshore habitats on the continental shelf, enriching marine ecosystems. Estuaries have been focal points of human settlement and resource use throughout history (Lotze et al., 2006). Over time, complex socio-ecological systems have evolved around estuaries, in which the 'human system' (eg, communities, society, economy) interacts with the 'natural system' (eg, coastal and marine ecosystems and renewable resources) in an adaptive and resilient manner.

Estuaries are not usually considered to be critical habitats; rather, they are ecosystems where complexes of organisms and their associated physical environments interact within a specified area (Tansley, 1935). Estuaries enclose, connect and support the functioning of many different habitats, often unique and critical for the maintenance of surrounding coastal and marine ecosystems. Critical habitats associated with Western Indian Ocean (WIO) estuaries are mangroves (Chapter 7), seagrasses (Chapter 8), salt marshes (Chapter 9), coral and biogenic reefs (Chapter 10) and nearshore environments that receive terrigenous sediments, nutrients and biological propagules from estuaries (Chapter 6). Mesohaline pelagic- or benthic habitats that support brackish water species, coastal lagoons, freshwater wetlands and coastal forests within the estuarine functional zone are common in the WIO region and are critical from ecological and socio-economic perspectives.

Estuaries are one of the coastal areas most at risk from human activities (Diop et al., 2016; Santos et al., 2021). Estuarine processes rely on riverine (fluvial) runoff from catchment basins and are vulnerable to changes in the frequency and volume of flooding. Disruption in fluvial regimes may originate far inland, when land-use patterns and vegetation cover are changed to increase agricultural production, water is abstracted for irrigation, or dams are built for hydroelectrical power (HP) generation (Duvail et al., 2017). All these activities have a high priority for food production and economic growth in developing countries, and take place far away from estuaries – nevertheless, their downstream impacts on estuarine processes and functioning can be severe from ecological and socio-economic perspectives. Estuaries are also influenced by sea level rise and storm surges, which are set to increase under the present climate change predictions. Coastal erosion and salt-intrusion, driven by increasing tidal reach, can alter the physical state and hydrobiology of estuaries, affecting key processes and productivity.

Rapid population growth over the past decades, especially in the developing world, has increased the anthropogenic pressure on natural resources. Human population growth in the coastal areas of the WIO has likewise been rapid, and several cities have developed along the banks of estuaries (eg, Maputo, Beira, Dar es Salaam and Mombasa), where they are strategically placed as ports for ocean-going vessels and transport of goods, and to benefit from ecosystem goods, such as fish, fuelwood and mangrove wood for construction materials (Groeneveld et al., 2021a).

Within the context of this *Critical Habitats Outlook*, 'WIO estuaries' was defined as the unit of assessment. Twelve estuaries, for which reports and published information could be found, were assessed for this chapter but reference is made to several other estuaries where it could assist in understanding. The scale of the assessment was regional (ie, sub-global), dealing with estuaries located in tropical / subtropical climates with strongly seasonal rainfall patterns. Ecologically meaningful similarities among the biota, abiotic environments and their interactions were therefore assumed to occur in estuaries across the WIO region.

DESCRIPTION OF WIO ESTUARIES

Steep headwater gradients and deltaic systems

Most of the rivers that drain into the WIO have steep headwater gradients along the high-elevation ridge stretching southwards from the Red Sea coast, through the Ethiopian relief and along the eastern edge of the Great Rift Valley down to southern Lake Malawi (Duvail et al., 2017) (Fig. 1). South of that, several rivers that discharge in southern Mozambique and eastern South Africa originate in the highlands of Swaziland and Lesotho. In Madagascar, rivers that drain into the Mozambique

Figure 1: Representative estuaries selected for this study, showing their locations and catchment basins.

Channel originate on the central plateau. Steep headwater gradients combined with erodible volcanic rock in source areas often result in high sediment loads (Vanmaercke et al., 2014). Deltaic estuarine systems, common in the WIO, then form when sediments are deposited on coastal floodplains, and are reshaped over time by sediment compaction, dewatering and renewed sedimentation by successive flood deposits (Duvail et al., 2017). Major fan-shaped deltas in the region have formed where the Zambezi, Rufiji and Tana rivers meet the ocean, and many other smaller estuaries also form deltas that discharge into Maputo Bay (eg, Incomati, Maputo), and further to the north (eg, Ruvuma, Ruvu, Athi-Sabaki).

Seasonal rainfall

Rainfall and flow characteristics of WIO estuaries are highly seasonal. Heavy extended rainfall occurs in March to May in northern Mozambique, Tanzania, Kenya and southern Somalia, before the SE monsoon, and short rains occur in the same region in October to December during the NE monsoon (Kitheka et al., 2004). The alternating dry and wet months result in high seasonal variability in runoff and sediment transport, with profound effects on estuarine functioning and exports of sediments and nutrients to nearby marine ecosystems. Peak rainfall in southern Mozambique and eastern South Africa occurs during summer, between November and February. The greater frequency and severity of floods and droughts resulting from climate change and their influences on river discharge and sediment transport further affects estuarine conditions (Mwaguni et al., 2016), adding to natural variability brought by the seasonal wet / dry cycle. Traditional use systems in the deltas and floodplains of the WIO are adapted to seasonal floods and dry periods, with fishing taking place during the floods, planting of rice and other crops during flood recession and grazing by livestock afterwards (Duvail et al., 2017).

Latitudinal rainfall trend

By latitude, annual rainfall decreases northwards from Mozambique (530–1140 mm per year) to Somalia (250–375 mm), and as a result larger estuaries are more prevalent in the southern part of the WIO, particularly in Mozambique (Taylor et al., 2003). The estimated total annual discharge in the northern part of the WIO (Kenya and Somalia) is in the range of 1.8–4.95 km^3/y, but it is substantially greater at 2.9–106 km^3/y in the central and southern WIO region (Tanzania, Mozambique and South Africa) (Hatziolos et al., 1996; Hirji et al., 1996; UNEP, 2001).

Tidal influence

The WIO deltas are mostly river dominated during periods of high rainfall, but tidal and wave processes are more important during droughts (Hoguane et al., 2021). The tidal range is 2–4 m (mesotidal), and tidal currents can be strong (> 2 m/s) in estuaries, and influence headwaters far upstream, especially where dams and upstream water abstraction reduces runoff, or during droughts. For example, salt water intrusion occurs up to 80 km from the mouth of the Zambezi delta and 55 km in the Thukela River (Scheren et al., 2016). Seasonal freshwater wetlands are common in low-lying areas of WIO deltas. Stratification of the estuary water-column, when dense seawater enters estuaries as a wedge below the freshwater layer at incoming tides (Hoguane and Antonio, 2016) facilitates irrigation of rice and other crops that are cultivated in wetlands, often in areas cleared of mangroves.

Key habitats of WIO estuaries

Mangrove forests (see Chapter 9) are the most ubiquitous habitat in WIO estuaries, covering an estimated 1 million ha, mostly in Mozambique, Tanzania, Kenya and Madagascar (Bosire et al., 2016). Nine mangrove species occur in the region, with zonation from seaward to landward edges of estuaries depending on factors such as tolerance to salinity, varied tidal regimes and substrate types (Bosire et al., 2016). The most expansive mangrove forests occur in deltas, but small estuaries and non-estuarine habitats harbouring mangroves are also important ecosystems (Kimirei et al., 2016). Mangroves extend upstream along the banks of estuaries up to the furthest extent of seawater intrusion, whereafter pioneering species such as *Avicennia marina* make way for vegetation that are salt-intolerant.

Seagrasses are distributed throughout the WIO region, from the intertidal zone to about 40 m deep, and often occur in close connection with mangroves and coral reefs (Lugendo, 2015). Extensive seagrass beds occur in WIO deltas (eg, Rufiji, Tana, Ruvu and Wami), and they are sometimes limited to the sheltered waters of estuaries. Seagrasses are one of the most productive aquatic ecosystems in the world, and they serve as critical habitats (as nurseries and foraging grounds) for numerous fish and invertebrate species (see Chapter 10).

Mesohaline pelagic- or benthic habitats that support brackish water species are other key habitats of WIO estuaries. Mesohaline conditions are important as nursery or breeding areas for marine fish and crustaceans with estuary-dependent life-history phases, and loss of this habitat might have implications for nearshore fisheries. A stark example was the collapse of a prawn trawl fishery on the Thukela Bank in eastern South Africa, when the mouth of the St Lucia Estuary closed as a result of sedimentation. In that case, juvenile prawns in the St Lucia Estuary could not recruit to the nearshore marine mudbanks, leading to recruitment failure and the collapse of the fishery (Ayers et al., 2013).

Urbanization around estuaries

Human population densities in deltas between Somalia and central Mozambique (incl. Zambezi delta) range between 25 and 249 inhabitants/km^2, well below the world average of around 500 inhabitants/km^2 (Overeem and Syvitski, 2009). Densities are much higher around deltas in southern Mozambique and Madagascar, ranging from 250 to 999 inhabitants/km^2.

Urbanization along the banks of estuaries takes advantage of the plentiful resources that estuaries provide, including products from mangrove forests, fish resources, and their proximity to navigable waters and ports, to transport goods. Cities and large towns on the banks of estuaries in the WIO continue to expand rapidly (eg, Maputo, Beira, Quelimane, Dar es Salaam, Mombasa).

Selection of representative estuaries

For this chapter, 12 representative estuaries were selected between the equator (northern Kenya) and 30°S (eastern South Africa), including one in north-west Madagascar (Fig. 1). These estuaries represent the major rivers that discharge into the WIO (eg, Zambezi, Rufiji), as well as rivers with medium-sized (eg, Tana, Athi-Sabaki, Ruvuma, Incomati, Maputo and Thukela) and small basins (Umgeni). Basin sizes ranged from 4500 km^2 for the Umgeni River to 1.3 million km^2 for the Zambezi, and estuaries likewise ranged from small and shallow (Umgeni, Thukela, Athi-Sabaki) to the extensive Zambezi delta which stretches > 100 km along the coast.

The estuaries and their catchments covered a range of average flow rates, lengths, source altitudes and slopes, and have transboundary catchment basins in at least 12 sub-Saharan countries, with estuaries in five of them (Table 1). Anthropogenic impacts such as urbanization, increasing agricultural cultivation, deforestation, water abstraction and infrastructure building differed in scale and relative importance between the chosen estuaries.

Information was obtained from unpublished reports and the peer-reviewed literature to review the threats (drivers and pressures), present state, anthropogenic and climate-related impacts at ecosystem and socio-economic levels, and resulting policy and management responses regarding WIO estuaries. The importance of WIO estuaries as providers of ecosystem goods and services is highlighted, including their role in maintaining crucial supporting and regulating ecological processes. A key question is what the impacts of reduced (or absent) provisioning, supporting and regulating services provided by WIO estuaries would be on social well-being and economic growth. Policy and management options are recommended.

ECOLOGICAL AND SOCIO-ECONOMIC IMPORTANCE OF WIO ESTUARIES

Estuaries provide unique goods and services on which the social and economic prosperity of coastal communities rely. Estuarine goods and services are important at spatially different scales – locally for food security, coastline stabilization and economic activity; regionally for providing nutrients to marine habitats and nurseries for dispersive marine species; and globally as a carbon sink in dense mangrove forests (Bosire et al., 2016) and seagrass beds.

Table 1: Selected rivers and estuaries of the WIO.

ESTUARY AND CATCHMENTS BY COUNTRY	CATCHMENT km^2/1000	LENGTH km	AVG FLOW m^3/s	ALTITUDE m	AVG SLOPE m/km	REGULATION 2014 Index
Tana (**Ken**)	127	1100	99	5199	5.2	0.32
Athi-Sabaki (**Ken**)	69	650		5000	7.7	0
Rufiji (**Tan**)	177	900	800	2400	2.6	0.01
Ruvuma (Tan-**Moz**)	155	800	450	1560	2.0	0
Zambezi (Zam-Ang-Nam-Bot-Zim-**Moz**)	1300	2600	3424	1524	0.6	2.24
Pungwe (Zim-**Moz**)	31	400	120	2592	6.5	0
Limpopo (Bot-Saf-Zim-**Moz**)	416	1750	170	2300	1.3	1.31
Incomati (Swa-Saf-**Moz**)	47	480	111	1800	3.8	0.82
Maputo (Saf-Swa-**Moz**)	29	380		2391	6.3	
Thukela (Les-**Saf**)	30	400		3000	7.5	
Umngeni (**Saf**)	4.4	232		1825	7.8	
Betsiboka (**Mad**)	49	525	271	1755	3.3	0.02

Adapted from Duvail et al., 2017, and Scheren et al., 2016.
Rivers discharging into the WIO from Africa (north to south) and Madagascar showing catchment area, length of river, average flow, source altitude, average slope and regulation index (=total storage capacity of the dam reservoirs / mean annual discharge).
Country codes: Ang – Angola; Bot – Botswana; Ken – Kenya; Les – Lesotho; Mad – Madagascar; Moz – Mozambique; Saf – South Africa; Swa – Swaziland; Zam – Zambia; Zim – Zimbabwe. Estuary location in **bold**.

CASE STUDY

Coastal lagoons and lakes

Fiona Mackay

Coastal lagoon in south-east Madagascar.
© Fiona Mackay

Coastal lagoons are shallow, coastal bodies of water, separated from the ocean by a barrier, formed by a coral reef, barrier islands, or sand bars (Kjerfve, 1994). Most lagoons formed during rises in sea level, particularly during the early Holocene (12 000–8 000 years ago) and they are often considered to be ephemeral, continually changing over time (Hill, 1975). Lagoons are found along low-lying coasts all over the world, and they are extensive along the coasts of Africa, where they cover nearly 18 per cent of the continent's coastline (Miththapala, 2013). The terms 'coastal lagoon' and 'estuary' are often used interchangeably, but there are characteristic differences between them. Lagoons are generally shallow, without major rivers discharging into them, and the flow of water is sluggish and slow. They can be classified into three geomorphic types (choked-, restricted-, and leaky lagoons), based on how they exchange water with the ocean. Other classification systems are according to their mean salinity, or to tidal regime.

Lake St Lucia in eastern South Africa is a typical choked lagoon, with a narrow inlet (often closed) which prevents exchange of water with the ocean. High evaporation rates and reduced tidal inflow combine to make it hypersaline for extended periods (Cyrus et al., 2011), with major consequences for the biota and ecological functioning of the lake and the adjacent nearshore marine environment (Cyrus et al., 2010). Extensive barrier lakes, swamps and temporarily rain-filled pans occur along the coast of southern Mozambique, between Ponta do Ouro and Bazaruto. They occur on a low elevation plain, are shallower than 5 m, and are separated from the sea by a well-developed longshore dune system. Most of the barrier lakes do not have links with the sea and typical freshwater or brackish fish species occur in them. Unlike the others, Lake Bilene is linked to the sea via a channel, which occasionally closes through formation of a sand bar – hence a choked lagoon. Relative sea level changes and sediment supply generated the barrier system along the southern Mozambique coast, and present climatic conditions contribute to a transgressive SE–NW dune migration rate of >22 m/y which can rapidly modify or bury lake and lagoon systems (Miguel et al., 2017). Their transitional nature makes coastal lagoons naturally stressed systems that experience frequent environmental disturbances and fluctuation.

Much of Kenyan and Tanzanian coastlines are fronted by fringing reefs, with lagoonal platforms between the coast and the reef. The reefs and platforms were formed during a Quaternary accretion and erosion of biogenic reef and backreef sediments, composed of calcium carbonate, in response to sea level variation (Arthurton, 2003). These are typical 'leaky lagoons', with an unhindered exchange of water, filling up with seawater during high tide and emptying during low tide. Fringing reefs and their associated lagoons are absent around the outflows of major rivers.

Similar to estuaries, lagoons provide coastal communities with important ecosystems goods and services. Flushing rates of lagoons are low and nutrients brought into them can be cycled many times before they are flushed out to the ocean, resulting in high primary productivity (Kennish and Paerl, 2010). Fish resources have long been exploited by local inhabitants for subsistence and commercial purposes, and lagoons provide breeding and nursery habitats for juvenile fish and crustaceans. Lakes that are not connected to the sea provide freshwater reservoirs for domestic use and agriculture. As critical habitats, coastal lagoons and lakes have high socio-ecological importance, but are vulnerable to anthropogenic interference and the effects of climate change.

Ecosystem goods and services are typically grouped into four broad categories (Millennium Ecosystem Assessment, 2005; Agbenyega et al., 2009), that are also applicable to WIO estuaries:

1. Regulating, or the capacity to regulate essential ecological processes and life support systems, such as climatic, water, soil, nutrient, ecological and genetic conditions.

2. Supporting, or the provision of a place for plants and animals, and thus helping with the conservation of genetic, species and ecosystem diversity.

3. Cultural, or the capacity to contribute to human well-being through knowledge and experience and sense of relationship with context eg spiritual experiences, aesthetic pleasure, cognition and recreation.

4. Provisioning, defined as the capacity to create biomass and thereby produce goods such as food, raw materials, and energy resources.

McNally et al. (2016) further subdivided the ecosystem goods and services provided by a typical WIO estuary (Wami, Tanzania) into roughly 30 specific ecosystem services, ranked according to stakeholder perceptions (Table 2).

The relative scale and importance of goods and services differ between individual estuaries.

Ecological importance

Key ecological functions, common to all WIO estuaries, are the provision, maintenance and connectivity of critical habitats for unique brackish water plant and animal species. Estuaries function as spawning, nursery and feeding grounds for marine fish and crustaceans that later migrate to nearshore marine environments to complete their life cycles. Palearctic birds use WIO estuaries as migratory stopovers. Extensive mangrove forests and seagrass beds trap and recycle nutrients and sediments, contributing to high biological productivity that is common to estuarine ecosystems. Some estuaries support large floodplains and wetlands, including coastal forests, lakes, and surrounding habitats for many wildlife species, including legendary herds of large mammals, which have been severely depleted outside of protected areas (Beilfuss, 1999).

Nutrient-rich sediments support a productive soft sediment ecosystem dominated by benthic-feeding penaeid shrimps, both in the lower reaches of estuaries and on offshore mudbanks, maintained by exported terrigenous sediments and riverine organic matter (Forbes et al., 2002;

Table 2: Typical ecosystem services and goods provided by WIO estuaries.

CATEGORY	ECOSYSTEM SERVICE
Regulating	Water filtration (clean water) Flood mitigation (water retention capacity) Minimizing drought effects (storage capacity) Prevention of salt-water intrusion Delivery of water and sediments for nursery areas Nutrient cycling and export to marine systems Carbon sequestration in mangroves
Supporting	Habitats for wetland plant and animal species Nursery habitats (eg prawns with an estuarine-dependent juvenile phase) Shelter and protection of animals against predators Conservation areas for animals and plants Erosion control/stabilization by vegetation Coastal protection of beach and coastline from storm surges, waves, floods
Cultural	Aesthetic and intrinsic values Recreation, tourism and spiritual function
Provisioning	Water for domestic use Fish / shrimp for food and commerce Mangrove wood for fuel and construction Fertile land for flood-recession agriculture, fruit and vegetables and grazing Transport of people and goods Traditional medicinal plants Inorganic raw materials (gravel, sand for building) Employment

Adapted from McNally et al., 2016.

de Lecea and Cooper, 2016). Nearshore marine mudbanks that are ecologically associated with WIO estuaries are the Thukela Bank (Thukela River in eastern South Africa), Maputo Bay (Maputo, Espirito Santo and Incomati estuaries in southern Mozambique), Sofala Bank (Zambezi delta in central Mozambique), shallow banks off the Rufiji delta (Tanzania) and off the Tana and Athi-Sabaki estuaries in Ungwana Bay (Kenya). The marine ecosystems of these nearshore shelf areas rely heavily on the catchment / coastal sea interactions, for sediments, organic matter, nutrients and recruitment of diadromous prawns and fishes. As an example, Hoguane and Armando (2015) emphasized the importance of Zambezi River runoff on fisheries production and artisanal catches on the Sofala Bank, a major fisheries zone in Mozambique.

Also of high ecological importance, is the natural buffer that estuaries form against biophysical stressors, predicted to intensify with climate change. Estuarine vegetation, especially mangroves, play a major role in erosion control and stabilization of sediments, thus protecting the coastline and low-lying areas from saltwater intrusion, storm surges and floods. The water retention capacity of estuarine wetlands mitigates the effects of droughts, by storing water for domestic use, livestock and agriculture. Stakeholders around the Wami estuary in Tanzania placed a high value on provision of domestic water, habitats for wild animals and plants and erosion control (McNally et al., 2016), but a lower value on saltwater intrusion, perhaps because the Wami estuary is located in a national park and is in an ecologically good condition. Upstream water abstraction appears to be comparatively moderate in the Wami catchment, but Kiwango et al. (2015) stressed the need to maintain minimum environmental flow requirements to preserve estuarine habitats and functioning.

Socio-economic importance

The capacity to create biomass and thereby produce goods such as food, raw materials, and energy resources make estuarine ecosystems valuable socio-economic assets, reflected in the growth of agriculture, towns and centres of commerce around them. In the WIO region, Maputo (pop. 1.2 million, 2018) is located on the shores of Maputo Bay – and surrounded by the Maputo, Incomati and Espirito Santo estuaries (Bandeira and Paula, 2014); Beira (pop. 0.53 million, 2017) is on the bank of the Pungwe; Quelimane (pop. 0.35 million, 2017) on the upper reaches of the Bons Sinais; Pangani (pop. 54 000, 2012) sits at the mouth of the Pangani estuary in Tanzania; and Kipini is located at the junction of the Tana delta with Ungwana Bay in Kenya. Both Dar es Salaam and Mombasa are located adjacent to estuaries and use them as ports for shipping. Increasing urbanization on the banks of WIO estuaries can be a mixed blessing, however, with the use of estuarine goods and services for socio-economic growth potentially bringing about their eventual depletion or degradation. For example, the use of peri-urban mangrove stands in Mombasa contributed to a 70 per cent loss between 1985 and 2009 (Bosire et al., 2016).

Ecosystem goods and services typical of WIO estuaries (Table 2) include the provisioning of water for domestic use; fisheries for prawns by small-scale and industrial fishers, and for finfish, bivalves, gastropods and crabs for food security and local markets; cutting mangrove poles for construction; using wood (mainly mangrove cuttings) to make and sell charcoal for fuel; harvest of traditional medicinal plants; and mining of inorganic raw materials such as sand, or gravel for construction purposes. The WIO estuaries provide fertile land for flood recession agriculture and grazing, and for vegetable and fruit production. Commercial salt pans for making salt or ponds for mariculture are also common uses. Tourism is a growing activity, with several estuaries located within national parks, such as the transboundary Ruvuma estuary, shared by the Mnazi Bay-Ruvuma Estuary Marine Park in Tanzania and the Quirimbas National Park in northern Mozambique (Scheren et al., 2016). The Wami estuary in Tanzania is in the Saadani National Park, which was formally established in 2005 on land acquired over three decades (Anderson et al., 2007).

THREATS TO WIO ESTUARINE ECOSYSTEMS: MULTIPLE SCALES

Global scale

Climate change and human population growth, in terms of increasing demands for resources and living space, and the building of infrastructure for energy, transport and industry, are the most persistent pressures on the 21st century WIO environment (UNEP-Nairobi Convention and WIOMSA, 2015; Diop et al., 2016; McNally et al., 2016; Duvail et al., 2017) (Table 3). Estuaries are under increasing pressure from natural processes and human activity. On a global scale, climate change effects are clearly noticeable as changes in weather patterns, and as sea level rise. More severe floods and / or droughts are predicted to influence freshwater runoff. Predicted changes in the seasonality of rainfall – and hence seasonal flood-regimes – will further affect estuarine processes, and their ecological functioning. Rising sea-level and greater tidal ingress and saltwater intrusions are expected, thus altering the chemical properties of productive brackish-water habitats. Critical habitats of many plant and animal species adapted to brackish-water environments for food, shelter or reproduction, may be degraded, displaced or will disappear (Furaca et al., 2021).

Regional scale

At a regional (catchment basin) scale, the pressure on natural resources emanates from rapid economic and infrastructure development, driven by the fundamental needs (food security, energy, clean water) of rapidly growing human populations (Table 3).

Major threats to estuarine processes in catchment basins are the abstraction of water for irrigation and damming of rivers for hydro-electric power or drinking water for cities. Building of dams have often disregarded environmental flows required to support downstream habitats or ecosystems, including runoff volumes and seasonality of flooding (Duvail et al., 2017). Changes in land-use and vegetation cover in catchment basins, through conversion of natural habitats to cultivated croplands, also affect freshwater runoff and increase sediment loads in runoff water, resulting in downstream accretion and changes to estuarine, beach and nearshore morphology (Kitheka and Mavuti, 2016). The seasonal (or even permanent) closure of river mouths due to sedimentation and the formation of sandbanks is common in smaller estuaries in eastern South Africa, with major implications for their ecological functioning (Whitfield et al., 2012). At a regional scale, disrupted connectivity between estuarine and marine environments results from upstream water abstraction, higher sediment loads and droughts.

Local scale

At a local scale (within and around individual estuaries) pressures are a reduced freshwater discharge, seasonally altered flood-cycles from dam releases and increasing coastal populations (Table 3). Economic growth and building of urban infrastructure are gradually replacing traditional (or subsistence) livelihood systems which are based on seasonal flood cycles (Francisco et al., 2021). WIO estuaries have supported subsistence livelihoods over many centuries, with interactions relying on timely societal adaptations and natural biophysical resilience (Leauthaud et al., 2013). Fishing for food, use of mangroves and other wood for fuel and construction, flood-recession planting of rice and other crops, and seasonal grazing of livestock all contributed to livelihoods in flexible traditional-use systems (Hamerlynck et al., 2010; 2020).

Table 3: Pressures and threats affecting WIO estuaries on multiple scales.

SCALE	PRESSURE	THREAT
Global	Climate change Global population growth	Sea level rise Erratic weather condition or seasonality More frequent and intense storms Natural resource demand outstrips production
Regional (WIO)	Economic & infrastructure development Regional population growth	Upstream water abstraction for irrigation / industry Dams for hydro-electric power Accelerated sedimentation or erosion Eutrophication and chemical / organic pollution Closure of river mouths and disrupted nutrient exchange Recruitment failure of estuarine dependent marine fish/prawns
Local (Estuary)	Reduced freshwater discharge Altered flood-cycles from dam releases Increasing coastal populations Change in traditional lifestyle	Loss of critical habitats for estuarine animal and plant species Loss of core estuarine processes and functions Loss of water quality (domestic use) Less fish and prawns (food security, local economy) Loss of fuelwood and construction materials from mangroves Reduction of space for flood-recession agriculture Increasing upstream salinity and salt intrusions Disrupted livelihood systems based on seasonal floods Loss of cultural and aesthetic value Vegetation change Loss of biodiversity Increased vulnerability to floods and storms Increased socio-economic stresses

Adapted from UNEP-Nairobi Convention and WIOMSA 2015; Diop et al. 2016; McNally et al. 2016; Duvail et al. 2017; Furaca et al., 2021; Groeneveld et al., 2021b; Mugabe et al., 2021; Mwamlavya et al., 2021.

Coastal livelihoods around estuaries are now threatened by human population growth and a greater demand for biological resources than that which can be obtained from estuarine ecosystems (UNEP-Nairobi Convention and WIOMSA, 2015). Unplanned urban and industrial development around estuaries is accompanied by overharvesting of nearby natural resources (fish and mangroves), pollution, and altering estuarine water flow through canalization or infilling.

Threats from human activities include loss of critical habitats, loss of core estuarine processes and functions, reduced water quality and lowered food security through destructive fishing (Table 3). Increasing upstream salinity and salt intrusions affect wetlands and vegetation, with implications for biodiversity, agriculture and traditional livelihood systems. Loss of traditional livelihoods is expected to accelerate urbanization, followed by increased unemployment and socio-economic stress.

The loss of the estuarine protection function against climatic events (ie, mangrove barriers and stabilized banks of estuaries) further increases the vulnerability of local inhabitants and infrastructure to storms and floods.

STATE OF THE WIO ESTUARIES

The WIO estuaries were all data deficient relative to the key aspects of distribution of critical habitats, ecological processes, and historical trends in freshwater and marine influences. A formal assessment against the IUCN Ecosystems categories and criteria (IUCN, 2015) could therefore not be undertaken. In its place, a simple qualitative index was developed to assess the present status of 12 WIO estuaries for placement into one of five categories (Table 4). None of the 12 estuaries qualified for the 'excellent' (least concern) category because all of them are increasingly affected by human activities in catchments or in the estuarine functional zone.

Estuary status relative to their condition 50 years ago (default assumption of pristine, except where contrary evidence exists) was assessed against six indicators, selected to represent the condition of critical habitats (core and support), functioning of estuarine processes, water quality, biodiversity and to what extent freshwater discharge into estuaries is affected by upstream dams (Table 5).

Assessment by estuary

Tana

Estuary state = Poor (Endangered)

The Tana delta is comprised of four estuaries of which the northern-most one has been channelled to form the main river mouth into Ungwana Bay (Scheren et al., 2016). Construction of hydroelectric power (HP) plants and dams in the Upper Tana basin has reduced runoff and affected the seasonal flood cycle. Water abstraction is increasingly affecting run-off to the estuary. Impacts of land-use change, damming and climate variability are high turbidity, heavy sedimentation, changes in beach morphology, and degradation of mangrove forests and marine ecosystems (Kitheka and Mavuti, 2016). The surface area and longevity of flood-supported riverine forests, wetlands and mangroves have been reduced. The estuarine fishery remains a relatively intact socio-ecological system (Manyenze et al., 2021).

Athi-Sabaki

Estuary state = Poor (Endangered)

There are no upstream dams, but urbanization in the headwaters have led to water abstraction and reduced infiltration of rainfall, manifesting as a diminished base flow and more rapid and short-lived flooding events (Scheren et al., 2016). Higher sedimentation loads because of land use change to agriculture have negatively impacted nearshore corals, but have led to an increased area colonized by mangroves (Kitheka and Mavuti, 2016). The estuary remains important in terms of biodiversity, providing habitats and nursery grounds for prawns and feeding grounds for birds. It plays an important role in sustaining the productivity of Ungwana Bay, but is threatened by heavy accretion.

Rufiji

Estuary state = Poor (Endangered)

The Rufiji is the largest river in Tanzania, and its delta extends 65 km across and 23 km in length, with a surface area of 1200 km^2. Sediment carried by the river has caused accretion with a seaward shift of the shore-line over millennia. It supports the largest mangrove area in Tanzania (approx. 50 000 ha), which, together with nearby seagrass beds, coral reefs and small islands form an interacting seascape which provides invaluable ecological services to the WIO (Wagner and Sallema-Mtui, 2016). Erosion and loss of mangroves at the seaward edge of the Rufiji delta have been attributed to sea-level rise; the loss at the seaward edge is counter-balanced by inland migration of mangroves at the landward edge (Wagner and Sallema-Mtui, 2016). Livelihoods are traditionally based on agriculture and fishing. Agriculture, particularly rice farming, has increased significantly over the past

Table 4: A simple qualitative index for assessment of WIO estuaries (adapted from Forbes and Demetriades, 2008), based on an evaluation of key indicators (see Table 5). No estuaries qualified for the Least concern (Excellent) category because all of them are increasingly affected by human activities in catchments or in the estuarine functional zone.

DESCRIPTION OF THE FIVE CATEGORIES USED	
Least concern (EXCELLENT)	Estuaries with high level of habitat integrity, good water quality, high diversity and high provision of goods and services
Near threatened (GOOD)	Estuaries with most core estuarine habitats and support habitats still present, good water quality, high diversity of species, estuarine processes in place
Vulnerable (FAIR)	Estuaries with core estuarine habitat intact, some estuarine support habitats, impacted water quality, some loss of diversity, key estuarine processes in place
Endangered (POOR)	Substantially reduced or no estuarine support habitats, polluted water, substantial loss of diversity and/or abundance and key estuarine processes impaired
Critically endangered (DEGRADED)	Estuaries which have major impacts on core estuarine habitats through infilling, canalization and pollution, substantially reduced or no estuarine support habitats and major loss of key estuarine processes

Table 5: Key indicators used to evaluate the status of individual estuaries, for placement into categories defined in Table 4.

INDICATORS USED FOR ASSESSMENT OF ESTUARIES	
Core estuarine habitats	Mesohaline pelagic- or benthic habitats that support brackish water species; mangrove forests
Estuarine support habitats	Intertidal and freshwater wetland habitats
Water quality	Levels of organic pollution, eutrophication, salt intrusion
Species diversity	Diversity and abundance of brackish water plants; birds; mammals; fishes; crustaceans
Estuarine processes	Seasonal and inter-annual flood cycle; high productivity, nutrient cycling, erosion/accretion, water storage processes retained
Freshwater supply from catchment basins	Regulation index (total storage of dam reservoirs/mean annual discharge for 2014) (Table 1)

Table 6: Estuary status inferred from qualitative scoring of key indicators, based on information obtained from available literature.

	Core habitat	Support habitat	Water quality	Species diversity	Estuarine processes	Catchment dams	OVERALL STATUS
Tana							POOR
Athi-Sabaki							POOR
Rufiji							POOR
Ruvuma							GOOD
Zambezi							POOR
Pungue							POOR
Limpopo							POOR
Incomati							DEGRADED
Maputo							FAIR
Thukela							POOR
uMngeni							DEGRADED
Betsiboka							DEGRADED

Colour code: The present (2018) status of the indicator is similar / worse / much worse relative to the status 50 years ago.		
SIMILAR	WORSE	MUCH WORSE

Flamingos foraging over shallow intertidal areas in southern Mozambique. © Fiona Mackay

two decades, and is associated with clearing of mangrove areas (Tumbo et al., 2015). The invasive giant freshwater prawn *Macrobrachium rosenbergii* has established a population in the delta and is occasionally fished and sold on markets (Kuguru et al., 2019). There are no major dams in the catchment basin, but large-scale conversion to agricultural land and increasing water abstraction for farming and hydroelectric power generation are increasingly contributing to water stress in the delta (Shaghude, 2016). Human population density in the delta is low, although projected to increase steeply based on census data from the past 30 years (Tumbo et al., 2015). Improved road access has attracted external traders, potentially increasing the extraction of natural products (fish, timber) for markets in Dar es Salaam. Whereas the Rufiji delta retains its core and support estuarine habitats intact, with most key estuarine processes in place, there has been loss of habitats and water quality, which is set to accelerate if a planned upstream dam in the main river at Stiegler's Gorge is built (Tumbo et al., 2015).

Ruvuma

Estuary state = Good (Near threatened)

The Ruvuma is a transboundary estuarine system, protected by the Mnazi-Bay Ruvuma Estuary Marine Park in Tanzania and the Quirimbas National Park in Mozambique. There are no major dams in its catchment basin and the human population density in the parks is low. The estuary is well-known for its beaches, mangroves and tropical coastal marine resources (Scheren et al., 2016). A recent bird survey showed high diversity, including palearctic migrants, species restricted to the East African biome and a globally threatened heron species (Borghesio et al., 2009). Large land mammals occur, and turtles nest on beaches. Locals rely mainly on farming and fishing, but also harvest mangroves for tannins, fuel wood, medicine, boat-building and carpentry. Some activities are not allowed within the park boundaries (Mangora et al., 2014). Based on Landsat imagery, the area of mangrove cover remained similar between 1995 and 2005 (Ferreira et al., 2009). The Ruvuma retains its core estuarine and support habitats, estuarine processes are in place, water quality is good and species diversity high – and protected. Although well preserved, the delta is threatened by expanding agriculture, hunting, planned oil and gas drilling and illegal timber extraction.

Zambezi

Estuary state = Poor (Endangered)

The Zambezi is the largest delta along the East Africa coast, and is 100 km long and 120 km wide at the coast, covering about 15 000 km^2 (Chenje, 2000). Its ecological functions include the support and maintenance of the Sofala Bank habitats and their rich nearshore fisheries, through the discharge of vital nutrients and organic matter into the sea (Hoguane and Armando, 2015). It provides spawning and nursery areas for penaeid prawns (Malauene et al., 2021), the target species of nearshore trawl and artisanal fishers, and is a productive feeding area for many fish species, which are also harvested. The delta supports large mangrove forests, which appear to be in a good conservation status, showing increased coverage between 1994 and 2013 (Macamo et al., 2016). The human population density in the delta is relatively low, and local livelihoods rely on farming, fishing and harvesting mangroves for fuel and construction. The construction of the Kariba (1959) and Cahora Bassa dams (1974) for hydro-electric power generation did not take downstream impacts of a change in flood regime into account, and has led to "reduced artisanal fisheries and shrimp industry productivity, reduced silt deposition and nutrient availability, severe coastal erosion, soil salinization, salt water intrusion, replacement of wetland vegetation by invasive upland species, reduction in coastal mangroves, failure of vegetation to recover from grazing, and disrupted or mistimed reproductive patterns for wildlife species" (Beilfuss, 1999). Biodiversity and the abundance of wild animals have declined, following loss of habitats.

From a socio-economic perspective, the dams reduced the land available for flood-recession agriculture and grazing practices. Based on the literature, there is little doubt that the Zambezi core and supporting habitats have been substantially reduced and altered by upstream dams, and that there has been a loss of estuarine processes. Biodiversity and abundance of terrestrial and marine species have been reduced – as reflected by reduced distribution and numbers of large mammals, and by lower fish and prawn catches by the Sofala Bank fisheries.

Pungue

Estuary state = Poor (Endangered)

The Pungue catchment basin is characterized by a low degree of development, few abstractions, diversions or regulation. There are no large dams, but a water pipeline transfers approximately 22 million m^3 per year to supply Mutare in Zimbabwe (Van der Zaag, 2000). It also supplies freshwater to Beira (pop. 500 000 in 2017), located on the north bank of the estuary where it meets the ocean. Water shortages in the Pungue basin are uncommon (Droogers and Terink, 2014), but during low flow conditions, the upstream freshwater intake for Beira is affected by salt intrusion. Large amounts of sediments discharged by the river minimize the effects of coastal erosion. The rate of mangrove deforestation in Beira is high (Barbosa et al., 2001).

Prawn aquaculture takes place in the estuary. The Port of Beira is situated on the north bank of the Pungue near the river mouth and is an important centre for shipping and logistics in the central Mozambican and Central African regions. The river mouth is dredged. Rural livelihoods rely on small-scale agriculture in flood plains, and some fishing. Since 2013, an invasive prawn, called 'rainbow prawns', make up a large portion of artisanal catches in the estuary – it is unclear whether they escaped from prawn farms in the estuary, or were brought by ballast water. The Pungue displays impacted core and estuarine support habitats, impairment of key estuarine processes, reduced water quality as a result of the port and city, and loss of biodiversity, reflected by the presence of invasive species. Interestingly, its endangered status results from a port and urbanization near the estuary mouth, as opposed to damming and water abstraction in catchment basins, which affects several other deltas in the region, for example the Zambezi and Tana.

Limpopo

Estuary state = Poor (Endangered)

Although not dammed, the river drains catchments in three countries and flows through the economic hub of southern Africa – resulting in high abstraction of water for domestic use and agriculture, and a high pollution load (Earle et al., 2006). The river experiences high streamflow variability, including very low conditions during droughts, and devastating floods that occur every few years after torrential rainfall in catchments. A severe flood in 2000 temporally increased the width of the estuary from 200 m to several kilometres, causing sediment transformation, mangrove forest degradation, uprooting and dieback, and affecting around 2 million people. Two mangrove species, *Xylocarpus granatum* and *Ceriops tagal* disappeared from the estuary after the 2000 flood but have since been replanted (Bandeira and Balidy, 2016). Severe flooding events over the past 70 years occurred in 1955, 1967, 1972, 1975, 1977, 1981, 2000 and 2013. Climate is therefore the major driver of the Limpopo estuary status (Bandeira and Balidy, 2016). The estuary also provides a nursery ground for fish and prawns, and its flood plains are under extensive agricultural cultivation (Louw and Gichuki, 2003). Core and support estuarine habitats have been substantially reduced by farming and recurrent floods, and water quality is low. Mangrove rehabilitation through replanting seedlings is underway to improve estuarine health.

Incomati

Estuary state = Degraded (Critically Endangered)

The Incomati River is shared by Mozambique, Swaziland and South Africa, and is intensively used for irrigation in South Africa (Hoguane and Antonio, 2016). It is of high ecological importance for the maintenance of the Maputo Bay ecosystem and fisheries. The Incomati estuary is located near a major urban centre (Maputo) and therefore suffers high anthropogenic pressure. The estuary is about 40–50 km long and meanders within the coastal plain separated from the ocean by a narrow sand dune. It has reduced freshwater inputs and is shallow, with islands and sandbars. Mangrove deforestation (Le Marie, et al., 2006) and severe degradation (Paula et al., 2014) have been attributed to overharvesting for fuelwood and construction, and modifications to river flow resulting from damming. The estuary remains a sanctuary for breeding colonies of aquatic birds, and a nursery ground for fish and crustacean species (Sengo et al., 2005). The estuary contributes approximately 20 per cent of the overall prawn catch in Maputo Bay (Anon., 2001). The reduction of estuarine core habitats (mangroves), impairment of key estuarine processes because of reduced freshwater input and polluted water endanger the ecological role of the estuary.

Maputo

Estuary state = Fair (Vulnerable)

The Maputo estuary is small with most of its catchment in Swaziland and South Africa, where its tributary (Pongola River) is dammed. The estuary is of high ecological importance for the maintenance of the Maputo Bay ecosystem and fisheries. It is located further from Maputo city than the Incomati and suffers less from anthropogenic pressures – with parts of the estuary located in the Maputo Special Reserve. Mangroves are in a good, though not pristine condition (Paula et al., 2014). Most core estuarine and support habitats are intact, and key estuarine processes are in place, with biodiversity receiving protection in the reserve.

Thukela

Estuary state = Poor (Endangered)

The Thukela River rises in the Drakensberg Mountain Range and has a steep gradient. It is highly impounded, with >600 smaller dams in its tributaries, seven major dams, and several inter-basin transfers that supply water to South Africa's economic hubs. Freshwater inflow is highly seasonal, with summer rains and dry months in winter – and variable with occasional major floods. The estuary has large mud flats, and although open to the sea, sand bars have occasionally closed the mouth in recent years. The Thukela estuary is small, without mangroves, but has high ecological importance as a source of organic material, nutrients and sediments to the nearshore Thukela Bank (Turpie and Lamberth, 2010; de Lecea and Cooper, 2016), which forms the southernmost prawn trawl grounds in the WIO. It is also an important habitat for resident and non-resident birds, fish and crustaceans. Water quality is affected by upstream industries and pesticides used in extensive sugarcane monoculture.

Umgeni

Estuary state = Degraded (Critically Endangered)

The Umgeni River flows through a dense urban metropolis (Durban) and the mainstream is impounded by a major dam within 30 km of its mouth. The estuary is severely affected by loss of habitat, sedimentation, freshwater deprivation, chemical and organic pollution, and modifications to its mouth (Forbes and Demetriades, 2008). The estuary has a dense mangrove stand on its north bank, and a diverse fish community, possibly because of its permanently open mouth. It also has an abundant and diverse bird community, because of intertidal sand / mud banks.

Betsiboka

Estuary state = Degraded (Critically Endangered)

The Betsiboka is Madagascar's largest river stretching 600 km from the high central plateau to the north-west coast, where it discharges into Bombetoka Bay (Scheren et al., 2016). The city of Mahajanga (pop. 220 000 in 2013) with a port is located on its northern bank. The Betsiboka transports lateritic soils and sediments derived from the highlands of central Madagascar to the sea, colouring the river a blood-red hue. The evolution of the bay, coastline, delta, and change detection results derived from Landsat satellite images recorded in 1973, 1989, 1999, 2000 and 2003, show that sedimentary transport and suspension in Bombetoka Bay has increased dramatically over the past 30 years, attributed to increased erosion following large-scale deforestation, bush fires, and overgrazing in the river basin (Raharimahefa and Kusky, 2010). These changes have adversely affected core estuary habitats, estuarine processes and water quality – as reflected in negative changes in agriculture (rice paddies and shrimp pens), fisheries and transportation.

Other estuaries

Estuary state = Not assessed

Numerous other rivers and estuaries discharge into the WIO from eastern Africa and Madagascar, and it is unlikely that many of them have escaped the dual influences brought by human activities and climate change effects. For example, the 98 km coastline of the Ethekwini municipality around Durban incorporates 16 estuaries, of which 13 are temporarily open / closed estuaries, two are permanently open and one is an estuarine bay (Forbes and Demetriades, 2008). Apart from the estuarine bay (Durban Bay), they are small estuaries (<10 to 230 ha), and their current health status range from highly degraded to good (Forbes and Demetriades, 2008).

EXISTING PROTECTION

International frameworks

Rivers that cross borders between countries are shared resources subject to equitable utilization by riparian states, and they are therefore fundamentally part of international and national watercourses governance and laws (Birnie et al., 2009). As a part of this, common management models include the creation of international

Kosi Bay, the northernmost of four interlinked lakes in northern KwaZulu-Natal, South Africa, showing traditional fish fence traps. © Fiona Mackay

institutions in which all riparian states cooperate in formulating and implementing policies for the development and use of a water course. In the past, these institutions focussed mainly on the access and allocation of water between upstream and downstream states, but in recent years, more attention has been given to broader ecological implications, within a legal framework more attuned to sustainability and water shortage (Momanyi, 2016). For example, Article 23 (protection and conservation of the marine environment) of the 1997 Watercourses Convention makes specific mention of the preservation of estuaries, 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".*

Several deltas and estuaries in the WIO have Ramsar status, as part of an international agreement for the protection and wise use of wetlands (1971 Wetlands Convention), which imposes conservation and management duties and responsibilities on states. Three examples of estuarine Ramsar sites in the WIO are the Tana delta (since 2012), the Marromeu complex in the Zambezi delta (2003) and the Kosi Bay complex (1991) in eastern South Africa, which is composed of four interconnected tidally influenced lakes (Momanyi, 2016).

WIO regional frameworks and Nairobi Convention

www.unep.org/nairobiconvention/

The estuarine ecosystems of the WIO have a strong regional legal framework for protection and sustainable use, grounded in various river basin organizations throughout the region, and in the provisions of the 1985 Nairobi Convention and the 2010 Amended Nairobi Convention (Momanyi, 2016). River basin organizations, such as the Incomati Basin Water Authority (South Africa, Mozambique, Swaziland) and the Zambezi River Authority (Mozambique, Zambia and Zimbabwe) operate according to clearly defined mandates for a specific purpose, such as shared dam construction or operation, hydropower generation or irrigation, but do not engage in interstate negotiations or policy formulation.

The 2010 Amended Nairobi Convention defines 'Convention Area' to include WIO estuarine ecosystems, as well as watersheds, and has provisions on pollution from various sources, including from ships (Article 5), by dumping (Article 6) and from land-based sources and activities (Article 7). In particular, Article 7 exhorts contracting parties 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 Land Based Sources and Activities (LBSA) Protocol to the Nairobi Convention has detailed provisions regarding measures to control, reduce and prevent downstream pollution, physical alterations and habitat destruction emanating from land-based sources.

Overall, there appears to be an elaborate framework of regional, national and local legal, policy and institutional arrangements in WIO riparian countries, but they treat estuaries as a part of international watercourses or river basin systems. Because the framework does not specify or isolate estuaries, they generally fall short of effective protection of estuarine ecosystems (Momanyi, 2016). Main challenges regarding the conservation of estuarine ecosystems *per se* are therefore policy and legislative inadequacies, limited institutional capacities, inadequate awareness, inadequate financial resources and mechanisms, and poor knowledge management (UNEP/Nairobi Convention Secretariat and WIOMSA, 2009).

Integrated Coastal Management Act (South Africa)

The Integrated Coastal Management Act (ICM; Act No. 24 of 2008) of South Africa has made some progress towards providing a more specific and integrated environmental management framework for estuaries (de Villiers, 2016). As in other WIO countries, South Africa has a mass of legislation that applies to estuaries at all levels of government (local, district, provincial, national and international) and because of multiple-use of land and water resources – from the catchment to the coast – the responsibility for their management and development falls under various government departments and acts, often with conflicting objectives. To deal with the complexity, the ICM Act includes a section that deals specifically with estuaries and estuary management.

The ICM Act prescribes that estuary management plans are developed for individual estuaries, by independent service providers, and with multiple stakeholder engagements, where government departments are represented to outline / discuss official management mandates. A Generic Framework for Estuary Management Plans (van Niekerk and Taljaard, 2007) provides broad guidelines to follow when developing individual estuary management plans, including structure and content, and ensures that all aspects are taken into account, including biodiversity value, social and economic values, goods and services, and environmental flow requirements. After stakeholder approval, the plan is submitted to a management authority for adoption. The management authority is a government department, mandated according to the National Estuary Management Protocol (Gov. Gazette No. 36432, 10 May 2013). Each estuary management plan includes an implementation plan, which reports to the management authority, where it can be audited. The formation of multiple stakeholder fora for individual estuaries improves the efficiency of estuarine management plans. It is envisaged that sufficient data will be collected to develop a 'State of the Estuaries' report that can be used to assess South African estuaries each year. The 2011 National Biodiversity Assessment (Driver et al., 2012) was the first such assessment that linked the states of estuaries and catchments.

PRIORITY OPTIONS FOR CONSERVATION

The over-riding importance of changes in catchment areas

The conservation of estuaries is a complex matter because their health and functioning depend not only on activities within the estuarine functional zone, but also on upstream land-use activities and water abstraction in catchment areas, which may span more than one country. Energy (dams for hydroelectric power generation), food security (water abstraction for irrigation), and clean water for urban and industrial growth are fundamental raw materials of economic development. Throughout the WIO region, upstream water abstraction to satisfy these demands has overshadowed initiatives to conserve downstream estuarine ecosystems. A first conservation priority is to increase awareness of the importance of changes in catchment basins on downstream ecosystems – especially at a political level.

Flood-pulses are a part of natural cycles and cannot be ignored

Damming and water abstraction in catchment basins affect estuaries by reducing freshwater runoff into estuaries, and by disrupting seasonal flood regimes. Reduced freshwater runoff changes the dynamics of tidal and fluvial influences, allowing for erosion, salt intrusion, and changes in habitat distribution. The disruption of seasonal flood-pulses causes radical changes in the floodplain ecology, leading to multiple environmental problems, as well as loss of biological productivity. Several studies (Drijver and Marchand, 1985; Junk et al., 1989; Opperman et al., 2013) show a direct relationship between flood extent and ecosystem production. Managed flood releases from hydropower dams to restore, maintain and improve estuarine ecosystem service delivery or 'environmental flows' is a key conservation priority (Duvail et al., 2017), supported by World Bank guidelines, best scientific practices, and a broad base in civil society (Acreman et al., 2010). The timing and volume of flood-releases – to fit with seasonal cycles and interannual variability – is critical.

Coastal communities, traditional livelihoods and socio-economic consequences

Socio-economic consequences of changes to estuarine and coastal ecosystems are fundamentally important in the WIO region (Leauthaud et al., 2013; Duvail et al., 2017), where flood-recession agriculture and fishing are important livelihood activities (Groeneveld et al., 2021a, Santos et al., 2021). Direct impacts of a loss of ecosystems goods and services on coastal communities are an increased vulnerability to climate change and other natural causes, including erosion, reduced fish catches, declining water quality for drinking and domestic use, and the loss of traditional livelihoods, farming systems and food security. Loss of estuarine ecosystems and their associated unique habitats and high biodiversity has also occurred, with long-term impacts on livelihoods, and on the economic growth potential of the region, through tourism development. Conservation of estuarine ecosystems, with a focus on their specific supporting, regulating, provisioning and cultural goods and services, is fundamental in maintaining traditional livelihoods over a longer term, to facilitate local socio-economic stability.

Mangroves are critical habitats in the WIO

Mangroves are a key component of WIO estuarine ecosystem productivity with influence well beyond their physical limits. They recycle nutrients which can be exported to marine ecosystems, provide habitats for many brackish water species, and nursery areas for marine fish and prawns, which support artisanal and commercial fisheries. They trap sediments and form barriers against storms and floods, stabilize mud- and sand banks against erosion, and provide fuel and construction wood to coastal communities. The conservation of mangroves and the hydrological systems that support them is therefore a high priority (see Chapter 9).

MPAs are only a partial solution

Marine Protected Areas (MPAs) in several WIO estuarine systems (Wami, Ruvuma) may protect critical habitats and biodiversity in and around estuaries from human activities in the estuaries, but these MPAs remain exposed to changes in land and water use in catchment basins. The habitats and ecosystems that the MPAs are meant to protect are increasingly vulnerable to saltwater intrusions, especially when freshwater flow has been reduced. The situation is exacerbated by rising sea level and more frequent storms resulting from climate change. Water stress originating from outside MPA boundaries is therefore likely to disrupt ecological processes and degrade critical habitats within their boundaries. Ensuring the maintenance of 'environmental flow', originating in catchment basins far removed from the MPA jurisdiction, is therefore also a key conservation priority.

Further, MPAs often face challenges in translating the accrued resource protection benefits into enhanced livelihoods of local communities in and around areas of their jurisdiction (Mangora et al., 2014). In the Mnazi Bay-Ruvuma Estuary Marine Park, traditional access and user rights have been reduced by MPA operations, thus affecting livelihoods of local communities, without providing commensurate alternative livelihood strategies (Mangora et al., 2014). Finding a way to redress the cost of denied access to livelihood resources, by provision of alternative livelihood means accepted by stakeholders, is therefore a priority when establishing MPAs.

Integration of ecological considerations into development politics, planning and design

From the above, the conservation of WIO estuaries and multiple ecosystems goods and services that they provide can only be approached in a true cooperative manner, with stakeholder participation. Stakeholders should include government departments with relevant mandates (agriculture, fisheries, water provisioning, environmental affairs, industrial development, energy, forestry, rural development and coastal protection), local jurisdictions and municipalities, NGOs, the private sector and local communities. The catchment basin, estuary and nearshore marine environment all need to be considered in planning and design – which will need to take ecological considerations, such as maintaining seasonal environmental flows, into account.

Treat estuaries individually – with specifically designed estuary management plans

Individual estuaries differ substantially in size, flow regime, anthropogenic impacts, geomorphology, habitats and ecosystem services that they provide. Managing them successfully requires individual estuary management plans, designed according to specific circumstances and characteristics.

The South African Integrated Coastal Management Act (Act No. 24 of 2008) prescribes that estuary management plans are developed for individual estuaries, by independent service providers, and with multiple stakeholder engagements. The design and implementation of individual estuary management plans in South Africa is transformative, and the strategy can easily be adapted for use in other WIO countries (Momanyi, 2016).

RECOMMENDATIONS

1. Increase awareness of the importance of changes in catchment basins to the health and functioning of water-dependent downstream ecosystems, especially estuaries. Increase the awareness of the ecosystems goods and services provided by estuaries, and how important they are to the socio-economies of coastal communities. An awareness campaign should target multiple levels (political, executive, middle management, field officers and affected communities).

2. Treat estuaries individually, within a specific estuaries framework that takes a broad spectrum of environmental, ecological, socio-economic, and economic development indicators into account. An example of an operational framework is provided by the South African Integrated Coastal Management Act (Act No. 24 of 2008).

3. Individual estuary management plans and implementation plans must be built on wide stakeholder participation, by identifying common ground, areas of mutual interest and shared concerns among stakeholders, while also recognizing potential tensions among them. A final plan must be acceptable to all affected parties, where possible.

4. 'Environmental flow', or the volume and seasonality of freshwater discharge into estuarine systems, to restore or maintain critical habitats, ground-water reserves and ecosystem functioning, needs to be determined for individual estuaries. Managed flood releases from upstream hydropower dams must be used to maintain natural environmental flow conditions through estuaries.

5. Mangrove forests are a key habitat of the WIO estuaries with multiple ecological functions which affect marine ecosystems and commercial fisheries. They are also a key source of fuel and construction material for coastal communities. The coverage and condition of mangroves should be monitored, potentially with a time series of remote sensing images backed up by ground-truthing, and where changes exceed a threshold, restorative actions should be taken, for example replanting (Bandeira and Balidy, 2016).

6. Human encroachment through building or flood plain agriculture also needs to be monitored, potentially with comparative remote sensing images backed up by ground-truthing. They can be managed to remain within specific estuary management plans.

7. Estuarine MPAs are a useful tool for managing exploitation of natural resources and reducing degradation of habitats within a protected area, thus conserving biodiversity and natural habitats. Nevertheless, they fall short when freshwater flow and natural flood-cycles are disrupted. The design of MPAs should account for activities in catchment basins and nearshore marine influences, as part of individual estuary management plans.

REFERENCES

Acreman, M., Farquharson, F.A.K., McCartney, M.P., Sullivan, C., Campbell, K., Hodgson, N., Morton, J., Smith, D., Birley, M., Knott, D. and Lazenby, J. (2000). Managed flood releases from reservoirs: issues and guidance. *Report to DFID and the World Commission on Dams*. Centre for Ecology and Hydrology, 2000, p. 86. Wallingford, UK

Agbenyega, O., Burgess, P.J., Cook, M. and Morris, J. (2009). Application of an ecosystem function framework to perceptions of community woodlands. *Land Use Policy* 26(3), pp. 551–557

Anderson, E.P., McNally, C., Kalangahe, B., Ramadhani, H. and Mhitu, H. (2007). *A Rapid Ecological Assessment of the Wami River Estuary, Tanzania*. Global Water for Sustainability Program, Florida International University

Anon. (2001). T*ransboundary water management as an international public good – development financing 2000*. A report prepared for the Ministry of Foreign Affairs of Sweden. Department for International Development Cooperation, Ministry for Foreign Affairs/Norstedts, Stockholm. ISBN 91-7496-250-7

Arthurton, R. (2003). The fringing reef coasts of eastern Africa—present processes in their long-term context. *West. Indian Ocean J. Mar. Sci.* 2(1), pp.1–14

Ayers, M.J., Scharler, U.M. and Fennessy, S.T. (2013). Modelling ecosystem effects of reduced prawn recruitment on the Thukela Bank trawling grounds, South Africa, following nursery loss. *Mar. Ecol. Prog. Ser.* 479, pp. 143–161

Bandeira, S. and Balidy, H. (2016). Limpopo Estuary mangrove transformation, rehabilitation and management. In Estuaries: *A Lifeline of Ecosystem Services in the Western Indian Ocean*, pp. 227–237. Springer, Cham

Bandeira, S. and Paula, J. (eds.) (2014). *The Maputo Bay Ecosystem*. WIOMSA, Zanzibar Town, 427 pp.

Barbosa, F.M.A., Cuambe, C.C. and Bandeira, S.O. (2001). Status and distribution of mangroves in Mozambique. *South Afr. J. Bot.* 67(3), pp. 393–398

Beilfuss, R. (1999). Can this river be saved? Rethinking Cahora Bassa could make a difference for dam-battered Zambezi. *World Rivers Rev.* 14(1), pp. 8–11

Birnie, P.W., Boyle, A. and Redgwell, C. (2009). *International Law and the Environment*. Third Edition, Oxford University Press, 579 pp.

Borghesio, L., Armakobe, B., Bakari, S., Balidy, H., Biasol, D. and Menomussanga, M. (2009). A bird survey of the Ruvuma Delta, northern Mozambique. *Bull. Afr. Bird Club* 16(2), pp. 197–203

Bosire, J., Mangora, M., Bandeira, S., Rajkaran, A., Ratsimbazafy, R., Apadoo, C. and Kairo, J.G. (eds.) (2016). *Mangroves of the Western Indian Ocean: status and management*. WIOMSA, Zanzibar Town, 161 pp.

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

Cyrus, D., Jerling, H., MacKay, F. and Vivier, L. (2011). Lake St Lucia, Africa's largest estuarine lake in crisis: combined effects of mouth closure, low levels and hypersalinity. *South Afr. J. Sci.* 107(3–4), pp. 1–13

Cyrus, D.P., Vivier, L. and Jerling, H.L. (2010). Effect of hypersaline and low lake conditions on ecological functioning of St Lucia estuarine system, South Africa: An overview 2002–2008. *Estuar. Coast. Shelf Sci.* 86(4), pp. 535–542

de Lecea, A.M. and Cooper, R. (2016). 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. In *Estuaries: a lifeline of ecosystem services in the Western Indian Ocean*, pp. 53–65. Springer, Cham

de Villiers, P. (2016). Estuary management in South Africa – An overview of the challenges and progress made to date. In *Estuaries: A Lifeline of Ecosystem Services in the Western Indian Ocean*, pp. 301–311. Springer, Cham

Diop, S., Scheren, P. and Machiwa, J.F. (eds.) (2016). *Estuaries: A Lifeline of Ecosystem Services in the Western Indian Ocean*. Springer International Publishing, 322 pp.

Drijver, C.A. and Marchand, M. (1985). Taming the floods: Environmental aspects of floodplain development in Africa. agris.fao.org

Driver, A., Sink, K.J., Nel, J.L., Holness, S., van Niekerk, L., Daniels, F., Jonas, Z., Majiedt, P.A., Harris, L. and Maze, K. (2012). *National Biodiversity Assessment 2011: An assessment of South Africa's biodiversity and ecosystems*

Droogers, P. and Terink, W. (2014). Water Allocation Planning in Pungwe Basin Mozambique. *Future Water Report*, 129

Duvail, S., Hamerlynck, O., Paron, P., Hervé, D., Nyingi, W.D. and Leone, M. (2017). The changing hydro-ecological dynamics of rivers and deltas of the Western Indian Ocean: Anthropogenic and environmental drivers, local adaptation and policy response. *C. R. Geosci.* 349(6), pp. 269–279

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

Elliott, M. and McLusky, D.S. (2002). The need for definitions in understanding estuaries. *Estuar. Coast. Shelf Sci.* 55(6), pp. 815–827

Ferreira, M.A., Andrade, F., Bandeira, S.O., Cardoso, P., Mendes, R.N. and Paula, J. (2009). Analysis of cover change (1995–2005) of Tanzania/Mozambique trans-boundary mangroves using Landsat imagery. *Aquat. Conserv.* 19(S1), pp. S38–S45

Forbes, A.T. and Demetriades, N.T. (2008). *Estuaries of Durban*. Report for the Environmental Management Department, eThekwini Municipality

Forbes, A.T., Demetriades, N.T., Bosman, C., Leuci, R., Sinclair, D., Smith, A.M., Fennessy, S., Robertson, W.D., Sink, K., Kruger, A. and Everett, B.I. (2002). Impacts of proposed dams on the Thukela estuary and inshore marine environment, KwaZulu-Natal, South Africa. *African basins: LOICZ global change assessment and synthesis of river catchment-coastal sea interaction and human dimensions, LOICZ reports and studies* (25), pp. 168–173

Francisco, R.P., Hoguane, A.M., Simbine, R.L. and Mabota, H.S. (2021). Household dependence on fish-based farming systems in the Bons Sinais Estuary in Mozambique. *West. Indian Ocean J. Mar. Sci*. Special Issue 1/2021, pp. 29–41

Furaca, N.B., Hoguane, A.M., MacKay, F., Willemse, M. and Langa, A.A. (2021). Exploring urbanization and critical habitat loss through land cover change around the Bons Sinais Estuary, Mozambique. *West. Indian Ocean J. Mar. Sci.* Special Issue 1/2021, pp. 43–58

Groeneveld, J.C., Hoguane, A.M., Kuguru, B., MacKay, F., Munga, C. and Santos, J. (2021a). Estuarize-WIO: A socio-ecological assessment of small-scale fisheries in estuaries of the Western Indian Ocean. *West. Indian Ocean J. Mar. Sci.* Special Issue 1/2021, pp. 1–15

Groeneveld, J.C., MacKay, F., Kuguru, B. and Mchomvu, B. (2021b). Socio-ecological change in the Ruvu Estuary in Tanzania, inferred from land-use and land-cover (LULC) analysis and estuarine fisheries. *West. Indian Ocean J. Mar. Sci.* Special Issue 1/2021, pp. 75-92

Hamerlynck, O., Duvail, S., Hoag, H., Yanda, P. and Paul, J.L. (2010). The large-scale irrigation potential of the lower Rufiji floodplain: Reality or persistent myth. *Shared waters, shared opportunities: Hydropolitics in East Africa*, IFRA, pp. 219–234. Nairobi

Hamerlynck, O., Nyingi, D.W., Paul, J.L. and Duvail, S. (2020). Fish-based farming systems: maintaining ecosystem health and flexible livelihood portfolios (chapitre 11). In *Farming systems and food security in Africa: Priorities for science and policy under global change*, pp. 354–392. Routledge

Hatziolos, M., Gustaf, C. and Alm, A. (1996). Africa: A framework for integrated coastal zone management. *The World Bank*, pp. 150. Washington DC

Hill, B.J. (1975). The origin of Southern African coastal lakes. *Trans. R. Soc. South Africa* 41(3), pp. 225–240

Hirji, R., Partorni, F.M. and Rubin, D. (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* 227 pp.

Hoguane, A.M. and Antonio, M.H.P. (2016). The hydrodynamics of the Incomati Estuary–An alternative approach to estimate the Minimum Environmental Flow. In *Estuaries: A Lifeline of Ecosystem Services in the Western Indian Ocean*, pp. 289–300. Springer, Cham

Hoguane, A.M., Gammelsrod, T., Furaca, N.B., Cafermane, A.C. and António, M.H. (2021). The residual circulation profile of the Bons Sinais Estuary in central Mozambique-potential implications for larval dispersal and fisheries. *West. Indian Ocean J. Mar. Sci.* Special Issue 1/2021, pp. 17–27

Hoguane, A.M. and Armando, E.V. (2015). The influence of the river runoff in the artisanal fisheries catches in tropical coastal waters – The case of the Zambezi River and the fisheries catches in the northern Sofala Bank, Mozambique. *J. Integr. Coast. Zone Manag.* 15(4), pp. 443–451

IUCN (2015). *Guidelines for the application of IUCN Red List of Ecosystems categories and criteria, version* 1.0 (eds. L.M., Keith, D.A., Miller, R.M., Murray, N.J. and Rodríguez, J.P.). Bland, 93 pp. Gland, Switzerland

Junk, W.J., Bayley, P.B. and Sparks, R.E. (1989). The flood pulse concept in river-floodplain systems. *Canadian Special Fish. Aquatic Sci.* 106(1), pp. 110–127

Kennish, M.J. and Paerl H.W. (2010). *Coastal lagoons: critical habitats of environmental change.* In Coastal lagoons: critical habitats of environmental change (eds. M.J. Kennish and H.W. Paerl). USA: Marine science series, CRC Press, pp. 1–16

Kimirei, I.A., Igulu, M.M., Semba, M. and Lugendo, B.R. (2016). Small estuarine and non-estuarine mangrove ecosystems of Tanzania: overlooked coastal habitats? In *Estuaries: A Lifeline of Ecosystem Services in the Western Indian Ocean*, pp. 209–226. Springer, Cham

Kitheka, J.U. and Mavuti, K.M. (2016). Tana Delta and Sabaki Estuaries of Kenya: Freshwater and sediment input, upstream threats and management challenges. In *Estuaries: A Lifeline of Ecosystem Services in the Western Indian Ocean*, pp. 89–109. Springer, Cham

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

Kiwango, H., Njau, K.N. and Wolanski, E. (2015). The need to enforce minimum environmental flow requirements in Tanzania to preserve estuaries: case study of mangrove-fringed Wami River estuary. *Ecohydrol. Hydrobiol.* 15(4), pp. 171–181

Kjerfve, B. (1994). Coastal Lagoons. In *Coastal Lagoon Processes* (ed. Kjerfve, B.). Elsevier Oceanography Series 60, pp. 1–8

Kuguru, B., Groeneveld, J., Singh, S. and Mchomvu, B. (2019). First record of giant freshwater prawn *Macrobrachium rosenbergii* (de Man, 1879) from small-scale fisheries in East Africa, confirmed with DNA barcoding. *Bioinvasions Rec.* 8(2), pp. 379–391

Leauthaud, C., Duvail, S., Hamerlynck, O., Paul, J.L., Cochet, H., Nyunja, J., Albergel, J. and Grünberger, O. (2013). Floods and livelihoods: The impact of changing water resources on wetland agro-ecological production systems in the Tana River Delta, Kenya. *Glob. Environ. Change* 23(1), pp. 252–263

LeMarie, M., Van der Zaag, P., Menting, G., Baquete, E. and Schotanus, D. (2006). The use of remote sensing for monitoring environmental indicators: the case of the Incomati estuary, Mozambique. *Physics and Chemistry of the Earth, Parts* A/B/C 31(15–16), pp. 857–863

Lotze, H.K., Lenihan, H.S., Bourque, B.J., Bradbury, R.H., Cooke, R.G., Kay, M.C., Kidwell, S.M., Kirby, M.X., Peterson, C.H. and Jackson, J.B. (2006). Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science* 312(5781), pp. 1806–1809

Louw, A. and Gichuki, F. (2003). *Limpopo Basin Profile: Strategic research for enhancing agricultural water productivity.* Baseline reports. Colombo, Sri Lanka, CGIAR Challenge Programme on Water and Food

Lugendo, B. (2015). Mangroves, salt marshes and seagrass beds. In *The Regional State of the Coast Report: Western Indian Ocean* (ed. Paula, J.) pp. 49–64. UNEP-Nairobi Convention and WIOMSA, Nairobi, Kenya

Macamo, C.C.F., Massuanganhe, E., Nicolau, D.K., Bandeira, S.O. and Adams, J.B. (2016). Mangrove's response to cyclone Eline (2000): What is happening 14 years later. *Aquat. Bot.* 134, pp. 10–17

Malauene, B.S., Lett, C., Marsac, F., Roberts, M.J., Brito, A., Abdula, S. and Moloney, C.L. (2021). Spawning areas of two shallow-water penaeid shrimps *(Penaeus indicus and Metapenaeus monoceros)* on the Sofala Bank, Mozambique. *Estuar. Coast. Shelf Sci.* 253, p. 107268

Mangora, M.M., Shalli, M.S. and Msangameno, D.J. (2014). Livelihoods of coastal communities in Mnazi Bay-Ruvuma Estuary Marine Park, Tanzania. In *Vulnerability of Agriculture, Water and Fisheries to Climate Change*, pp. 271–287. Springer, Dordrecht

Manyenze, F., Munga, C.N., Mwatete, C., Mwamlavya, H. and Groeneveld, J.C. (2021). Small-scale fisheries of the Tana Estuary in Kenya. *West. Indian Ocean J. of Mar. Sci*. Special Issue 1/2021, p. 93–114

McNally, C.G., Gold, A.J., Pollnac, R.B. and Kiwango, H.R. (2016). Stakeholder perceptions of ecosystem services of the Wami River and Estuary. *Ecol. Soc.*, 21(3)

Miguel, L.L.A.J., Nehama, F.P.J. and Castro, J.W.A. (2017). Lagoon-barrier system response to recent climate conditions and sea level rise, Mozambique, Africa. *Estuar. Coast. Shelf Sci.* 216, pp. 71–86

Millennium Ecosystem Assessment (2005). *Ecosystem and human well-being: biodiversity synthesis. World Resources Institute.* Washington, DC

Miththapala, S. (2013). Lagoons and Estuaries. Coastal Ecosystems Series, Vol. 4, vi + 73 pp. IUCN Sri Lanka Country Office, Colombo

Momanyi, A. (2016). Re-thinking estuarine ecosystem governance in the WIO region. In *Estuaries: A Lifeline of Ecosystem Services in the Western Indian Ocean*, pp. 241–257. Springer, Cham

Mugabe, E.D., Madeira, A.N., Mabota, H.S., Nataniel, A.N., Santos, J. and Groeneveld, J.C. (2021). Small-scale fisheries of the Bons Sinais Estuary in Mozambique with emphasis on utilization of unselective gear. *West. Indian Ocean J. Mar. Sci.* Special Issue 1/2021, pp. 59-74

Mwaguni, S., Ruwa, R.K., Odhiambo-Ochiewo, J. and Osore, M. (2016). 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. In *Estuaries: A Lifeline of Ecosystem Services in the Western Indian Ocean*, pp. 111–129. Springer, Cham

Mwamlavya, H.M., Munga, C.N., Fulanda, B.M., Omukoto, J.O., Thoya, P.Z., MacKay, F., Manyenze, F.H. and Groeneveld, J.C. (2021). Natural resource-use in the Lower Tana River Delta based on household surveys and remote sensing of land cover and land use patterns. *West. Indian Ocean J. Mar. Sci.* Special Issue 1/2021, pp. 115-130

Opperman, J., Galloway, G.E. and Duvail, S. (2013). The multiple benefits of river-floodplain connectivity for people and biodiversity. In: *Encyclopedia of biodiversity* (ed. Levin, S.A.) vol. 7. Waltham: Academic Press, pp. 144–160. ISBN 978-0-12-384720-1

Overeem, I. and Syvitski, J.P.M. (2009). Dynamics and vulnerability of delta systems. GKSS Research Center, LOICZ International Project Office, Institute for Coastal Research. Geesthacht, Germany

Paula, J., Macamo, C. and Bandeira, S. (2014). Mangroves of Maputo Bay. In *The Maputo Bay ecosystem* (eds. Bandeira, S. and Paula J.). WIOMSA, Zanzibar Town, pp. 101–146

Raharimahefa, T. and Kusky, T.M. (2010). Environmental monitoring of Bombetoka bay and the Betsiboka estuary, Madagascar, using multi-temporal satellite data. *J. Earth Sci.* 21, pp. 210–226. https://doi.org/10.1007/s12583-010-0019-y

Santos, J., Groeneveld, J.C., MacKay, F. and Munga, C.N. (2021). A regional assessment of seasonal-to-decadal changes in socio-ecological systems in the Western Indian Ocean. West. *Indian Ocean J. Mar. Sci.* Special Issue 1/2021, pp. 131-162

Scheren, P., Diop, S., Machiwa, J. and Ducrotoy, J.P. (2016). The Western Indian Ocean: a wealth of life-supporting ecosystem goods and services. In *Estuaries: A Lifeline of Ecosystem Services in the Western Indian Ocean*, pp. 1–23. Springer, Cham

Sengo, D.J., Kachapila, A., Van der Zaag, P., Mul, M. and Nkomo, S. (2005). Valuing environmental water pulses into the Incomati estuary: key to achieving equitable and sustainable utilisation of transboundary waters. *Physics and Chemistry of the Earth, Parts A/B/C* 30(11-16), pp. 648–657

Shaghude, Y.W. (2016). Estuarine environmental and socio-economic impacts associated with upland agricultural irrigation and hydropower developments: the Case of Rufiji and Pangani Estuaries, Tanzania. In *Estuaries: A Lifeline of Ecosystem Services in the Western Indian Ocean*, pp. 169–182. Springer, Cham

Tansley, A.G. (1935). The use and abuse of vegetational concepts and terms. *Ecology* 16(3), pp. 284–307

Taylor, M., Ravilious, C. and Green, E.P. (2003). *Mangroves of East Africa.* UNEP-WCMC Biodiversity Series 13. Cambridge

Tumbo, M., Mangora, M.M., Pauline, N.M. and Kuguru, B. (2015). Review of literature for a climate vulnerability assessment in the RUMAKI Seascape, Tanzania

Turpie, J.K. and Lamberth, S.J. (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(3), pp. 613–624

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

UNEP / Nairobi Convention Secretariat and WIOMSA (2009). *Regional synthesis report on the review of the policy, legal and institutional frameworks in the Western Indian Ocean (WIO) region.* UNEP. Nairobi, Kenya

UNEP-Nairobi Convention and WIOMSA (2015). *The Regional State of the Coast Report: Western Indian Ocean.* UNEP and WIOMSA, pp. 546. Nairobi, Kenya

Van der Zaag, P. (2000). *The Pungwe River Basin.* SADC Water Sector. ICP collaboration Portal. http://www.icpconfluence-sadc.org/riverbasin/102

van Niekerk, L. and Taljaard, S. (2007). Proposed generic framework for estuarine management plans. *CAPE Generic Estuary Management Plan.* Version 1, pp. 114

Vanmaercke, M., Poesen, J., Broeckx, J. and Nyssen, J. (2014). Sediment yield in Africa. *Earth Sci. Rev.* 136, pp. 350–368

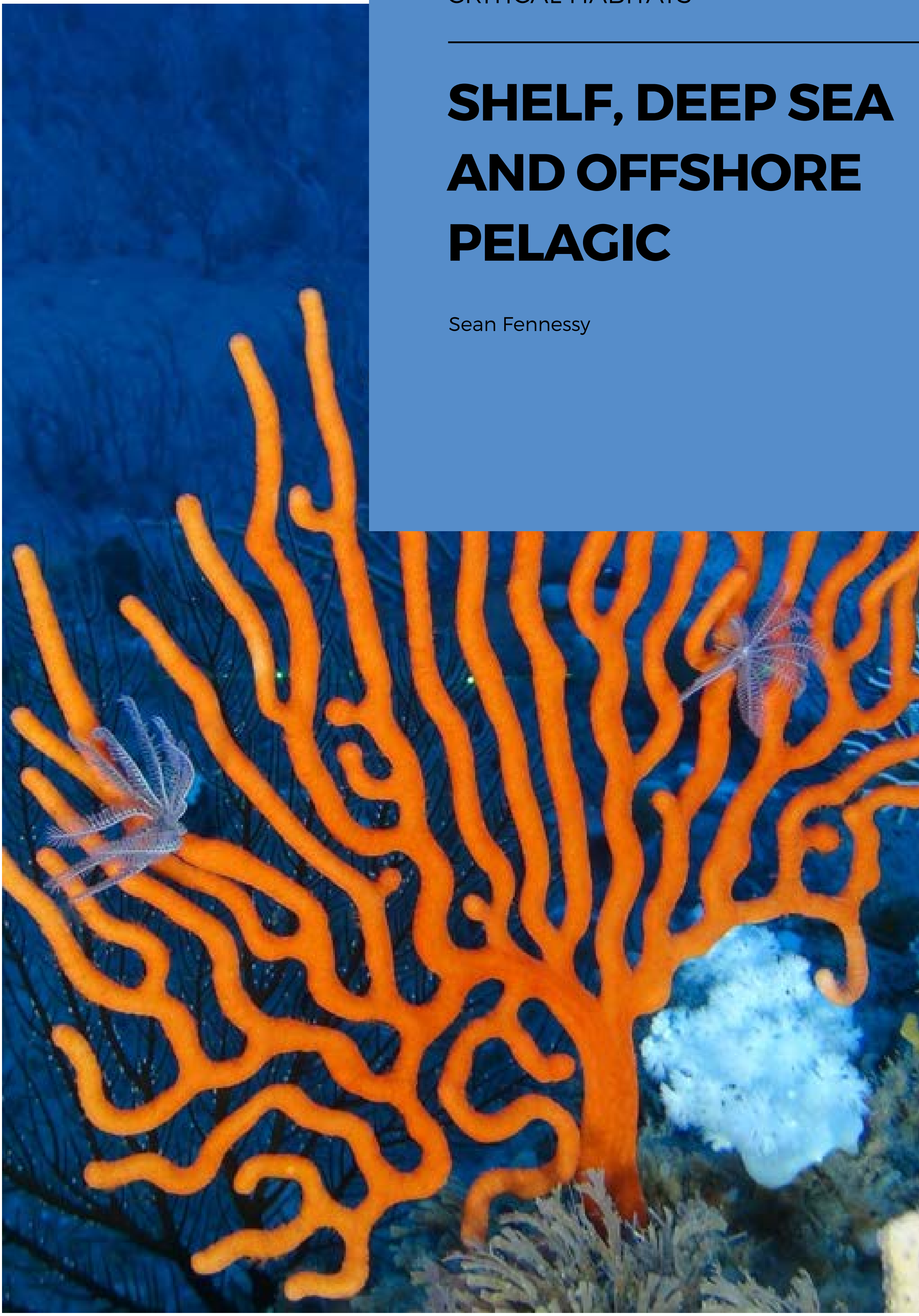
Wagner, G.M. and Sallema-Mtui, R. (2016). The Rufiji Estuary: climate change, anthropogenic pressures, vulnerability assessment and adaptive management strategies. In *Estuaries: A Lifeline of Ecosystem Services in the Western Indian Ocean*, pp. 183–207. Springer, Cham

Whitfield, A.K., Bate, G.C., Adams, J.B., Cowley, P.D., Froneman, P.W., Gama, P.T., Strydom, N.A., Taljaard, S., Theron, A.K., Turpie, J.K. and van Niekerk, L. (2012) A review of the ecology and management of temporarily open/closed estuaries in South Africa, with particular emphasis on river flow and mouth state as primary drivers of these systems. *Afr. J. Mar. Sci.* 34(2), pp. 163–180

CRITICAL HABITATS

SHELF, DEEP SEA AND OFFSHORE PELAGIC

Sean Fennessy



Mesophotic reefs in the WIO – a poorly-known offshore habitat. © DSI/NRF/ACEP Deep Forests project, SA

BACKGROUND

Shelf sediments, the offshore pelagic and deep-sea habitats are spatially vast areas, encompassing the seabed and the water above it; as with seamounts and ridges which are dealt with separately (Part III, Chapter 17), they differ from the other critical habitats covered in this book in that they are not coastal. Yet the proximity of shelf sediments to critical coastal habitats means there is spatial overlap, and physical and biological processes taking place in shelf sediments, offshore pelagic and deep-sea habitats have profound effects on critical coastal habitats too. These areas are also affected by coastal processes and land runoff, through sedimentary fluxes, and chemical and biological interlinkages. For the sake of simplicity, though, the habitats described herein are termed "offshore habitats", because that is what they largely cover.

Their spatial scale means they include a diverse variety of habitats within their realms. But, as described in the relevant chapters of the *Regional State of the Coast Report for the Western Indian Ocean* (Fennessy and Green, 2015; Obura, 2015a), offshore habitats are poorly known for the region, particularly with respect to the seabed. The approach taken here is to use the considerable regional knowledge on habitats and biodiversity in the WIO which formed the basis of the identification and prioritization of regional Ecologically or Biologically Significant Areas (EBSAs) (Dunn et al., 2014). The rationale for this is further elaborated in Part II, Chapter 4. Notwithstanding shortcomings (Johnson et al., 2018), the EBSA process has aggregated arguably the best available knowledge on biodiversity and habitats within and beyond state jurisdiction, and continues to evolve (CBD, 2018). This process is being co-ordinated by the Secretariat of the Convention on Biological Diversity (SCBD, 2013; see Existing Protection below), to which Convention all Western Indian Ocean (WIO) states have consented to be bound. From a jurisdiction perspective, the Offshore Habitat EBSAs described here include those in the exclusive economic zones (EEZs) of WIO coastal states, as well as in Areas Beyond National Jurisdiction (ABNJ). Further, since the Nairobi Convention applies to signatory coastal states, also included are Offshore Habitat EBSAs in the South African EEZ to the west of Cape Agulhas – in other words, in the south-east Atlantic (SCBD, 2014), albeit that these technically fall outside of the WIO biogeographic region (Part II, Chapter 4). Reference to coastal critical habitats occurring within these EBSAs has been minimized herein, as these are covered in other chapters. The Prince Edward and Crozet Islands EBSA is also not included here, as it falls in the Southern Ocean. The 18 Offshore habitat EBSAs dealt with in this chapter are listed in Table 1. It being impractical to reproduce the extensive bibliographies supporting the rationale for these EBSAs here, readers are referred to the specific EBSA reports (SCBD, 2013; 2014), and references therein, as well as the website[1], for more detail. Where additional supporting literature has been consulted, citations are provided.

Supply vessel hookup in offshore oil activities, Tanzania.

The WIO region covers a very large ocean area of around 25 million km^2, of which WIO EBSAs make up around 6.4 million km^2 (Table 2, Fig. 1). About 39 per cent of the area of these EBSAs falls in state EEZs, the remainder is in ABNJ. Of the WIO EBSAs in EEZs, only 22 per cent of their area is on the continental shelf (< 200 m depth), reflecting that they are mainly offshore. The seven EBSAs falling within the South African EEZ, off its west coast, comprise an area of around 193 000 km^2, 44 per cent of which is on the continental shelf. Around 14 per cent (~300 000 km^2) of the area of the WIO EBSAs falls into existing Marine Protected Areas (MPAs) within EEZs; overall, only 5 per cent of the area of Offshore habitat EBSAs in the WIO is in MPAs.

The WIO is home to an extraordinarily diverse suite of species (Griffiths, 2005; Richmond, 1997), but those known are mainly from coastal shelf waters and many more remain to be discovered or described from deeper

[1] www.cbd.int/ebsa/

Table 1: Summarized characteristics of 18 EBSAs (www.cbd.int/ebsa/) predominantly comprising offshore habitats. The seven scientific criteria developed by the Convention on Biological Diversity for scoring are provided below the table. H = High, M = Medium, L = Low.

EBSA	CBD scientific criteria scores*							Key Offshore habitat features	Other critical habitat features	Country	Threats
	1	2	3	4	5	6	7				
WIO EBSAs											
Agulhas Front	H	H	H	M	H	M	L	Extremely high pelagic productivity, high biodiversity	Birds, cetaceans, bluefin tuna	South Africa, France, ABNJ	–
Agulhas Bank	H	H	H	M	M	M	M	Unique and/ or rare sand/ mud/ gravel, deep corals, nursery areas, oceanography, productivity, threatened benthic habitats, threatened endemic fishes, spawning aggregations	–	South Africa	Fisheries, oil/ gas
Agulhas slope	M	H	M	H	H	H	H	Highly diverse pelagic and benthic habitats, threatened benthic habitats, endemic cold-water corals, high productivity, fish spawning/ recruitment area, bird foraging	Seamounts, turtles, birds, sharks	South Africa	–
Offshore of Port Elizabeth	M	H	H	M	H	H	L	Vulnerable canyons, rare and threatened mixed sediments and gravels, deep reef corals, unique pelagic features, high productivity, spawning/ recruitment area, bird foraging	Turtles, birds	South Africa	–
Protea Banks and sardine route	H	H	M	M	M	M	L	High benthic and pelagic complexity, unique deep reefs, canyons, endemic benthos, threatened endemic fishes, spawning aggregations of threatened fishes, migration pathway	Birds, sharks, cetaceans	South Africa	Fishing
Natal Bight	M	H	H	M	H	L	L	Unique and threatened sediments and gravels, strong terrestrial-marine connection, locally high productivity, unique and endemic benthos and fishes, nursery area, threatened fishes, deep reefs	Estuary, elasmobranchs, turtles	South Africa	Fishing, oil/ gas, mining, pollution
Delagoa shelf edge	M	H	M	M	M	H	H	Diverse benthic and pelagic habitats, ecoregion transition zone, high species diversity, vulnerable canyons, deep reefs/ corals, threatened habitats	Corals, sharks, coelacanths, turtles	South Africa, Mozambique	Mining, oil/ gas,
Quelimane to Zuni River	H	H	M	L	H	–	M	High benthic productivity, extensive mud habitat	Estuary, mangroves, mammals	Mozambique	Fishing
Mozambique Channel	H	H	H	H	H	H	M	Globally unique eddy dynamics influenced by complex seabed geology, both influential in cross-channel connectivity and pelagic productivity; high levels of biodiversity	Corals, sharks, turtles, birds, mammals	South Africa, Mozambique, Tanzania, Comoros, Madagascar, France	Oil/gas
Southern Madagascar	H	H	H	M	H	H	H	Transition zone between tropical and temperate waters, high wave energy, high pelagic productivity, high biodiversity and endemicity	Seamounts, turtles, birds, cetaceans	Madagascar, ABNJ	–

EBSA	CBD scientific criteria scores*							Key Offshore habitat features	Other critical habitat features	Country	Threats
	1	2	3	4	5	6	7				
Northern Mozambique Channel	H	H	H	H	H	H	L	Eddy dynamics leading to high interconnectedness in a region of high biodiversity	Mangroves, seagrass, corals, turtles, elasmobranchs, birds, coelacanths, mammals	Mozambique, Tanzania, Seychelles, Comoros, Madagascar, France	Oil/gas
EBSAs in the South African EEZ to the west of Cape Agulhas (ie in the adjacent south-east Atlantic)											
Subtropical Convergence Zone	M	H	H	M	M	M	L	High pelagic productivity and biodiversity	Birds, bluefin tuna, cetaceans	South Africa, ABNJ	–
Benguela Upwelling System **	H	H	H	M	H	M	M	Oceanographically unique, high biological productivity, fish spawning and nursery areas, endemic biodiversity	Birds, cetaceans	South Africa (Namibia, Angola)	Oil/gas
Browns Bank	H	H	H	M	M	L	M	Unique, endangered gravel habitat, high benthic biodiversity, deep corals, fish spawning and nursery areas, high pelagic productivity	Birds	South Africa	–
Cape Canyon and Surrounds	M	H	H	H	H	M	M	Rare, endangered and unique benthic habitat types, deep corals, high pelagic productivity	Islands, birds, cetaceans	South Africa	Oil/gas, mining
Childs Bank	H	L	M	H	L	M	H	Unique benthic habitat, including cold-water corals, high fish biodiversity	Sharks	South Africa	Fishing
Orange Shelf Edge	L	M	H	M	M	H	H	High demersal fish biodiversity	Estuary	South Africa (Namibia)	–
Orange Cone **	H	H	M	M	M	M	M	Unique area, high productivity, fish recruitment	Estuary, salt marsh, birds	South Africa (Namibia)	Mining

* CBD scientific criteria scores:
1. Uniqueness or Rarity
2. Special importance for life history stages of species
3. Importance for threatened, endangered or declining species and/or habitats
4. Vulnerability, Fragility, Sensitivity, or Slow recovery
5. Biological Productivity
6. Biological Diversity
7. Naturalness

** Only areas falling within South Africa's EEZ considered.

Table 2: Indicative calculated areas (all km^2), rounded off for convenience. Note that these figures refer to all WIO EBSAs to the west of 70° E and to the east of 20° E (ie. South-East Atlantic EBSAs are not included) including those which contain islands and seamounts, although areas of EBSAs designated specifically for these features (eg Tromelin, Walters Shoals) were excluded as they are dealt with in other chapters; areas of overlapping EBSAs were not counted twice in calculations.

WIO EBSA area 6 400 000			Total WIO area 24 500 000		
In EEZs		In ABNJ	In EEZs		In ABNJ
2 500 000		3 900 000	9 390 000		15 270 000
On shelf	Beyond shelf		On shelf	Beyond shelf	
267 000	2 214 000	–	590 000	8 800 000	–
MPA area			MPA area		
59 000	300 000	0	81 000	357 000	0

Source: J. Maina, University of Queensland, unpubl. data. MPA figures are as per the *MPA Outlook*.

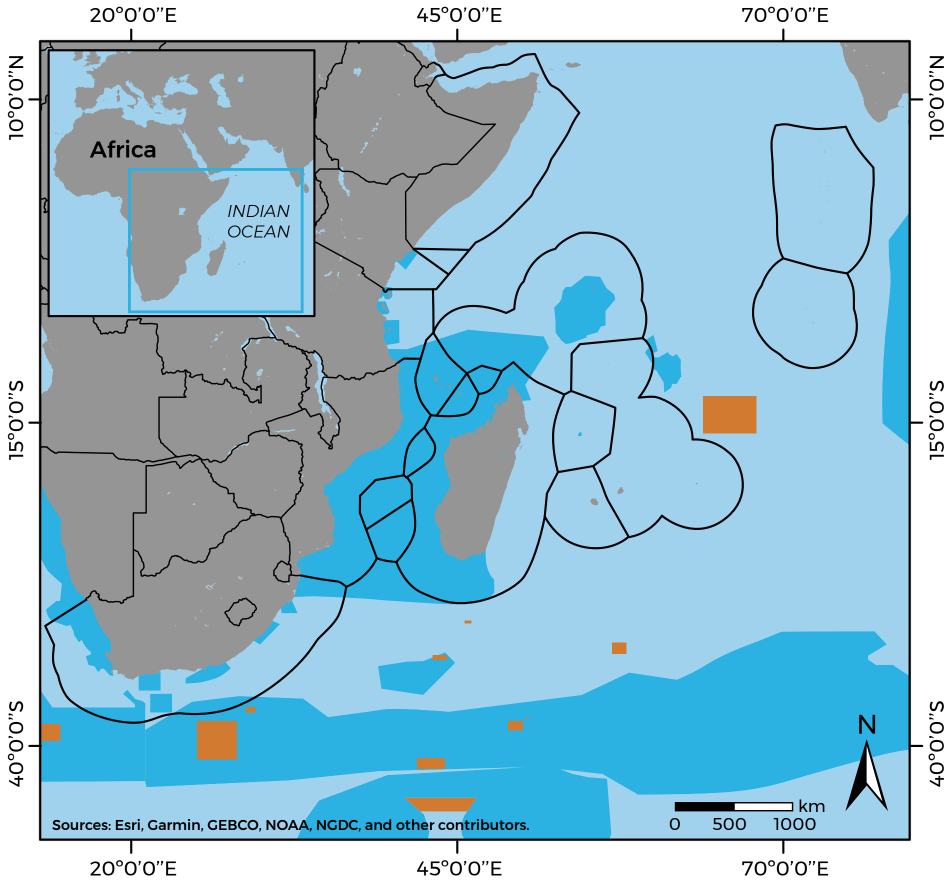


Figure 1: Western Indian Ocean area showing 18 Offshore habitat EBSAs (green) and Benthic Protected Areas (purple). Note: Readers should consult the South Indian Ocean Fisheries Agreement website for confirmation of BPA status.
Source: J. Maina, University of Queensland, unpubl. data. EEZs denoted by black lines.

waters. It may be expected, given their vast spatial extent, that Offshore habitats will add considerably to species counts as explorations expand into these areas. This particularly applies to benthic fauna from shelf sediments and deeper seabed habitats which have been consistently under-sampled (Griffiths, 2005). This is the case even in South Africa, where most ecological seabed research in the region has taken place – with vastly more in shallower (< 100 m) west coast waters, notwithstanding the enormously greater area of seabed > 100 m deep (Griffiths et al., 2010). Broadly, in shelf waters, biodiversity increases from the cool west coast ecoregion, through the warm-temperate Agulhas region off the south of the continent, into the subtropical KwaZulu-Natal province and ultimately the extensive tropical WIO commencing off southern Mozambique (Spalding, 2007). The biota of the western and southern regions are quite different and more variable compared to those of the eastern, which are more uniform (Griffiths, 2005). However, deep water habitats are known to be more stable and usually their biota distributes throughout larger areas when compared to that of shallow water (Longhurst, 2007). At depths of 800–3000 m, the WIO forms part of the proposed Indian Ocean lower bathyal province, but based on physico-chemical proxies rather than species, while at depths > 3000 m, the south-west Indian Ocean region is proposed to be distinct from the remainder of the Indian Ocean abyssal province, based on sea temperature (UNESCO, 2009).

Most offshore EBSAs characteristically identify the more readily observable and/or charismatic and endangered species, such as seabirds and marine mammals, together with threatened fishes, as part of the rationale for their ecological or biological significance. These faunal elements will be addressed more comprehensively in the chapters on Marine Birds (see Chapter 16), and Threatened Species (see Chapter 15), but the following key species in offshore habitats are particularly noteworthy. Commencing in EBSAs off the South African west and southern coasts, some encompass key foraging areas for Southern right (*Eubalaena australis*) and Humpback whales (*Megaptera novaeangliae*) and have some of the highest known densities of several endemic seabirds. African penguin (*Spheniscus demersus*), Cape gannet (*Morus capensis*), Cory's shearwater (*Calonectris borealis*) and the Atlantic

Yellow-nosed and Tristan albatrosses (*Thalassarche chlororhynchos*, *Diomedea dabbenena*), amongst several other seabird species, are heavily reliant on these areas for foraging and breeding. Keystone small pelagic fish species, notably sardine (*Sardinops sagax*), anchovy (*Engraulis encrasicolus*) and Horse mackerel (*Trachurus delogae*), are reliant on these areas for spawning and as nurseries, as are the keystone demersal Cape hakes (*Merluccius* spp.). Endangered Southern Bluefin tuna (*Thunnus maccoyii*) make extensive use of productive offshore pelagic areas for foraging. Knowledge of deeper (> 200 m) pelagic biodiversity is limited (UNESCO, 2009), although studies have been made by oil prospecting companies, however with very restricted dissemination. Critical aggregating areas for several threatened endemic deep reef fish species are found off the southern Cape, such as Red steenbras (*Petrus rupestris*), as well as aggregation of their counterparts from shelf sediment habitats eg Silver kob (*Argyrosomus inodorus*). Vulnerable cold-water corals, such as *Goniocorella dumosa* and *Solenosmilia variabilis*, as well as hydrocorals, gorgonians and glass sponges are found on the shelf edge as well as on deep reefs and in canyons in several of these offshore habitats.

Moving north-eastwards into the offshore habitats of the WIO proper, the wide-ranging whales and Southern Bluefin tuna persist, and there are high abundances in one of the most diverse seabird communities known. Some members of this are endangered and reliant on this region as their most important feeding area, such as Barau's petrel (*Pterodroma baraui*) and Amsterdam albatross (*Diomedea amsterdamensis*). Further north, tropical species such as frigatebirds (*Fregatta* spp) and Red-tailed tropicbirds (*Phaethon rubricauda*) are heavily reliant on waters there for foraging. On the shelf, aggregations of over-exploited endemic deep reef fishes such as Seventy-four (*Polysteganus undulosus*) and Slinger (*Chrysoblephus puniceus*) are found. Apart from their reliance on inshore waters, vulnerable and/or threatened turtles (all five WIO species) make extensive use of offshore waters here, as do migrating Humpback whales. Several threatened elasmobranch species form critical aggregations for nursery, feeding or mating purposes, either associated with shelf sediments or deep reefs; these include Ragged tooth shark (*Carcharias taurus*), Scalloped hammerhead (*Sphyrna lewini*), Whale shark (*Rhincodon typus*) and manta rays (*Manta* spp.). There are benthic communities of invertebrates and fishes specifically adapted to muddy habitats on the shelf and in deep water, the former closely associated with outflows from large rivers. The critically endangered Coelacanth (*Latimeria chalumnae*) is found in certain shelf-edge habitats, and vulnerable reef-building cold-water coral sites are known in deep water (> 900 m) in some areas.

IMPORTANCE

For some of these offshore habitats, particularly those which are spatially extensive, physical oceanographic processes are extremely influential (see also Part II, Chapter 5). In some areas, such as on the west coast shelf of South Africa, and around oceanic fronts and convergence zones, very high levels of pelagic productivity are found, as a consequence of the interaction of currents and wind. These produce strong gradients of salinity and temperature, with vertical stratification of the water column allowing nutrients to be concentrated in the upper euphotic layers, resulting in plankton blooms and associated energy transfer higher up the food web. This accounts for the reliance of seabirds, mammals and pelagic fishes on these areas for feeding (Boersch-Supan et al., 2017). In shelf areas, the pelagic energy is transferred to benthic habitats too, permitting high levels of biomass over shelf sediments on the South African west coast. Mobile and semi-permanent oceanic mesoscale eddies typify the WIO region, also elevating nutrient levels (but not to the same extent as on the west coast), either by upwelling at their cores, or by advecting and retaining nutrients from shelf regions; these features, too, are associated with enhanced biological production (reviewed in Ternon et al., 2014). The eddies are predominantly propagated to the south/south-west, and are heavily influenced by seabed morphology, continental land masses, islands and bathymetric ridges; eddy pathways suggest inter-basin transfers of upwelled products (Chapter 5). The productivity of several WIO upwelling sites, some of which spatially coincide with offshore habitats, are being investigated in the ongoing Western Indian Ocean Upwelling Research Initiative (Roberts, 2015) ending in 2025. Over shelf sediments, large rivers are recognized as important providers of nutrients (Huggett and Kyewalyanga, 2017), also evidenced from the presence of nearby industrial fisheries. However, the importance of land-based nutrient sources relative to oceanic upwelled sources in shelf environments is not fully understood, notwithstanding preliminary findings on the east coast of South Africa, reviewed in Fennessy et al. (2016). Much of the WIO, though, is naturally low in productivity, especially in surface waters (Kyewalyanga, 2015; Obura, 2015a; Huggett and Kyewalyanga, 2017), and this is reflected in the low biomasses of fishes in contrast to those on the Atlantic coast (Fennessy et al. 2017; Krakstad et al., 2017). Further, the pathways and extent of bentho-pelagic coupling facilitating energy transfer to deep seabed habitats in the region are not well understood.

Apart from the productivity features of offshore habitats, other physical processes were found to be critical for sustaining the organisms which occur there. Migration corridors facilitate essential seasonal movements of adult organisms to and from areas for feeding and reproduction, such as those undertaken by Humpback whales from the Antarctic to the central WIO (Best et al., 1998), or for example by marine turtle migrations (Lambardi et al., 2008). Some species, such as spiny lobsters (*Palinurus* spp.), produce pelagic larval recruits which can be transported vast distances and over long (six months) periods before settling in shelf nursery areas, but other species utilize physical oceanographic features to ensure their offspring are retained close to their origin soon after hatching. However, even with a relatively short pelagic duration (one month), the larvae of some species can be transported across the Mozambique Channel and remain viable (Ockhuis et al., 2017). Similarly, Maina et al. (2020) recently demonstrated in the WIO the potential for larvae generated in the high-seas and from MPAs to be advected to coastal waters. Recruitment can be mediated by major current systems, such as the South Equatorial Current and the Agulhas Current, or by more localized, smaller features such as eddies and gyres, or coastal currents (see Part II, Chapter 5). Recruitment processes of less mobile organisms which occur in, or close to, shelf and deep-sea sediments are not well-known in most cases. As already indicated, the biota in these habitats are poorly known throughout much of the region – with the exception of parts of the South African west and east shelves and upper slopes, and also if they are the target of fisheries (discussed below).

The significance of these offshore habitats for biodiversity conservation has been elaborated in EBSA workshops with the participation of local and regional experts (SCBD, 2013; 2014). Criteria for inclusion as an EBSA include high levels of productivity, vulnerability and biodiversity (reviewed in Part II, Chapter 4), and the scoring of these EBSAs, together with the offshore habitat features which qualify them for conservation, are summarized in Table 1.

The value of the WIO marine economy was recently reviewed by Obura et al. (2017). While tourism is the major contributor, offshore habitats offer limited scope for such activities, owing to their largely inaccessible nature. Nevertheless, they provide ecosystem services in support of coastal habitats which do support tourism, albeit that the value of this support has not been calculated. Similarly, quantification of the value of offshore habitats, particularly pelagic waters and deep-sea sediments, towards another major economic contributor, carbon sequestration, has also not been determined for the WIO region and is not as yet included in state economies. Fisheries are traditionally identified as having more obvious direct economic benefit to states, particularly those in their EEZs. Notwithstanding the socio-economic importance of the region's coastal small-scale fisheries (van der Elst et al., 2009), fisheries in offshore habitats are economically important to WIO nations, particularly as a source of foreign currency. These are generally of an industrial nature, owing to the infrastructure required to access and process offshore resources, and are regionally epitomized by the fisheries for large pelagic fishes (long-line and purse-seine for tunas), crustaceans such as prawns and langoustines trawled over shelf and/or deep-sea sediments, and demersal fishes on seamounts in ABNJ eg trawling for Orange roughy (*Hoplostethus atlanticus*) and Alfonsino (*Beryx* spp).

The offshore fisheries and their target species in the WIO region are described in van der Elst and Everett (2015). For many states, the investment and expertise to harvest offshore resources is not available, and the fishing rights to their EEZs are sold to interests outside the region. The most valuable fishery is that for tuna, and most catches are made in the high seas. Annual WIO tuna catches are around 850 000 tonnes, and are valued at over USD1.3 billion (Barnes and Mfodwo, 2012), although these figures are under-estimates. The most economically important South African west and south coast industrial fisheries in offshore habitats are atypical of the WIO region in terms of the cold-water species targeted and the types of fishing gear. They take the form of demersal trawling for hakes (*Merluccius* spp.) and purse-seining for small pelagic anchovies (*Engraulis encrasicolus*) and sardines (*Sardinops sagax*) (Cochrane et al., 1997), with a combined value in the region of USD 0.5 billion. These fisheries are heavily dependent on the elevated nutrient productivity generated by upwelling, and current-mediated recruitment.

Unsurprisingly, formal literature sources on the economic value of non-renewable marine resources such as oil and gas, and polymetallic nodules, sulphides and crusts, are difficult to obtain. Revenues can be very large – the annual value of diamonds mined in shelf sediments off the west coast of South Africa and (mainly) Namibia in 2012 was around USD 3.5 billion (reported in Baker et al., 2016). The predicted potential for polymetallic nodules in the WIO is not as high as in other oceans (Petersen et al., 2016), although some reports indicate otherwise (eg Rona, 2008); nor are mining activities for these already occurring in the region, although further exploration is likely. Owing to the depth at which these features occur, sites are often beyond EEZs (Petersen et al. 2016). The WIO has several oil/gas fields, onshore, nearshore and offshore; with some of the latter currently being developed for production (Richmond, 2015), most recently initiated in northern Mozambique, and there are considerable estimated reserves in the region, albeit not necessarily economically viable, and much of the area remains under-explored (U.S. Geological Survey, 2012). Phosphate and diamond mining interests are currently restricted to the shelf off the west coast of South Africa, while heavy minerals such as titanium ores are often found in shelf sediments off large river mouths in the WIO (Rona, 2008), although no offshore mining of these has commenced.

THREATS

These can be broadly grouped into three categories – extraction of resources (renewable and non-renewable), contamination and pollution (some of which is directly associated with resource extraction), and climate change. All of these threats are, to a greater or lesser degree, anthropogenic. The ASCLME/SWIOFP Transboundary Diagnostic Analysis (2012) for the region identified several drivers which exacerbate threats to habitats, including: unsuitable governance, economic factors, insufficient financial resources, a lack of knowledge, and population growth. At the regional scale, cumulative human impacts in the WIO based on 2004-2006 data were less intense than in other regions, but with elevated levels in the north-east and south-west of the region (Halpern et al., 2008). However, a follow-up review shows that regional impacts, particularly those linked to climate change, had intensified considerably by 2013, particularly in the Mozambique Channel (Tanzania) and to the east of Madagascar, around Comoros, Reunion and Seychelles (Halpern et al., 2015). Threats frequently imply declines in habitat status, and threats to offshore habitats identified during the EBSA process are therefore included in the summary table of habitat status in the following section (Table 3).

Harvesting of renewable resources, largely in the form of fishing, is widely recognized as being a threat via habitat modification and/or unsustainable removal of large amounts of biota, either as bycatch or targets, causing disruptions to ecosystem functioning. While over-exploitation of some species in offshore habitats in the WIO region is known, for example for large pelagic tunas (Pillai and Satheeshkumar, 2012), evidence of changes to ecosystem functioning is limited, largely because of an absence of suitable, long-term datasets. Increasing sea temperatures and altered upwelling patterns are predicted to result in distributional shifts and changing abundance of tunas; incidences of this have already been seen in the WIO, with low primary production and major changes in tuna distribution in the late 1990s causing an eastward shift in fishing fleet operations (Robinson et al., 2010). There is some evidence of altered composition of fish families from shelf sediment habitats in Mozambique, potentially attributable to coastal over-fishing (Fennessy et al., 2017; Krakstad et al., 2017). While demersal trawling is generally recognized as having negative physical impacts on the seabed, in the WIO region this activity is at a relatively low level within EEZs, being mainly concentrated off central Mozambique and the west coast of Madagascar (van der Elst and Everett, 2015) and there is limited scope for its increase, at least at depths from 200-600 m (Everett et al., 2015). The potential for demersal trawling at depths greater than this, within EEZs in the region, is largely unknown. There are indications that industrial trawling effort in depths < 100 m is declining owing to reduced viability (Fennessy and Everett, 2015), and the smaller island states have all banned demersal trawling. The situation is somewhat different off the south and west coasts of South Africa, where there is considerably greater demersal trawling effort (as well as purse-seining for small pelagic fishes), and where there is stronger (although not always unequivocal) evidence of alteration of offshore habitats, and composition and distribution of species, due to fishing (Atkinson et al., 2011; Coetzee et al., 2008; Sink et al., 2012). Trawl-associated deep-water communities from ABNJs, frequently associated with seamounts and ridges (see Part III Chapter 17), are poorly documented in the formal literature, although sharp changes in effort and catch indicate over-exploitation of the highly vulnerable target species (eg Orange roughy), and damage to habitats with vulnerable epifauna such as deep-water corals is known (reviewed in Clark et al., 2016). A threat still unquantified for WIO offshore habitats is bioprospecting for marine natural products, although several states have been involved in this activity in coastal habitats (Wynberg, 2015).

Exploration for and extraction of non-renewable resources both pose threats, and there is increasing interest in identifying and utilizing marine sources as terrestrial sources diminish. Methods for identifying mineral resources initially rely on remote sensing to identify promising indicative geological features – such methods frequently involve use of seismic and sonar equipment. Depending on the frequencies and intensities of the sounds generated, negative impacts on a wide range of organisms, from benthic infauna to cetaceans, are possible, including disrupted communication, hearing and orientation, although there is considerable lack of knowledge of effects for many taxa (Hawkins et al., 2015). Mining of minerals generally results in disruption of sediments, leading to increased turbidity and modification or loss of habitats, and contamination and destruction of biota (Ahnert and Borowski, 2000; Levin et al., 2016). Even excluding catastrophic failure of infrastructure leading to widespread oil spillage, drilling for and extraction of petroleum products results in contamination of sediments and surrounding water, with extirpation or modification of benthic and pelagic biological communities. Effects of these extractive activities can be localized or dispersed over thousands of kilometres (Smith et al., 2008), depending on current regimes and the extent of contamination, and can persist for many years in deep-sea habitats, though except for accidental events, impacts are in general localized, restricted to within 2 km from the well or deep-sea installation (Cordes et al., 2016).

The most well-known extraction of non-renewable marine resources in the WIO region is of oil and gas within state EEZs, with licenses being granted for large prospecting areas by several states. In South Africa, for example, 98 per cent of the EEZ has been assigned for this activity, and there are large prospecting blocks in southern Tanzania and northern Mozambique. While impact assessments have been undertaken for prospecting and extraction (Richmond, 2015), and monitoring is underway at localities where extraction has already commenced, details of impacts are not available in the formal literature.

Extraction of metallic ores has not commenced in offshore habitats in the WIO region, though hydrothermal vent areas of potential interest have been identified, and, as interest in these resources increases, exploration is expanding. The International Seabed Authority (ISA)[2], which regulates deep-sea mining in ABNJs, is granting increasing numbers of licenses to contractors for deep-sea exploration for polymetallic nodules, massive sulphides and cobalt-ferro-manganese crusts (Boetius and Haeckel, 2018). While most licenses are in other oceans, there are some on the Central Indian Ridge and the South West Indian Ridge (Levin et al., 2016). Indications are that hydrothermal vent communities are intolerant of disturbance, but, more concerningly, elements released from the vents have a critical biogeochemical role in the wider ocean, for example via mediation of micronutrient productivity associated with phytoplankton blooms (eg German et al., 2016). There are concerns that the ISA's governance processes are not sufficiently transparent and that it has limited means to enforce conditions of exploration contracts (Johnson et al., 2016). Deposits of titanium-based minerals in shelf sediments are known for several areas, notably in areas adjacent to where coastal mining is already occurring; locations of phosphate accumulations in shelf sediments are similarly known (Rona, 2008). In South Africa, prospecting rights in these habitats have been granted for both of these minerals. Probability of commencement of extraction of non-renewables depends on the availability of the minerals from terrestrial sources, their prices, and on technological capabilities – these are all changeable, so the imminence of the threats posed is difficult to assess.

Shipping traffic in the region is also related to the regional economy and extraction of resources (both renewable and non-renewable), and the Indian Ocean has demonstrated very rapid growth in shipping subsequent to 2002; although considerable traffic passes through the Mozambique Channel, the major route is between southern Africa, passing to the south of Madagascar, to and from Asia, and the relative densities of ships are considerably lower than in the Northern Indian Ocean (Tournadre, 2014). If, as anticipated, oil and gas activities in the WIO region continue to grow apace, greater shipping traffic can be expected in the region, with associated increased pollution, ship strikes on cetaceans, and invasive species from ballast water and fouling.

By far the most marine contamination and pollution originates from the land (Hassan, 2017), and the proximity of coastal and shelf habitats means they are the main recipients, while impacts in offshore habitats tend to be less noticed owing to dispersion and their being out of the public eye. However, plastic, the most pervasive type of marine litter, has been found in sediments even in remote habitats several thousand meters deep (Woodall et al., 2015). Plastics can entangle organisms, smother habitats and alter community structure (Gregory, 2009), and when ingested can reduce stomach capacity, affect growth, cause internal injury, and block intestines (Plot and Georges, 2010). The WIO region is less threatened by pollution than other oceans, although this is changing (Obura, 2015b); for example, the Indian Ocean had higher numbers and weights of plastic particles compared to other southern hemisphere oceans (Eriksen et al., 2014), and around 50 per cent of Loggerhead turtles (*Caretta caretta*) examined between Reunion and Madagascar from 2007-2013 had ingested plastic, with ingested amounts higher than in turtles from other oceans (Hoarau et al., 2014). Notwithstanding the expansion of oil and gas activities in the WIO region, there are no formal publications on impacts of associated pollutants on offshore habitats, although monitoring has commenced in some areas, notably northern Mozambique by the *RV Dr Fridtjof Nansen* in 2018. Similarly, assessments of the threats posed by other pollutants, such as metals and organic compounds from terrestrial sources, is rare (see UNEP/Nairobi Convention Secretariat, CSIR and WIOMSA, 2009; Fennessy and Green, 2015).

In contrast to the threats posed by land-derived pollutants and contaminants, some WIO offshore habitats and

[2] www.isa.org

communities, including some in deep water, rely on sediments and nutrients provided by rivers (Gammelsrod, 1992; Fennessy et al., 2016; Scharler et al., 2016). Reduced flow, because of impoundments or climate change, compromises this delivery (Lamberth et al., 2009), as well as reducing recruitment of estuarine-dependent organisms to offshore habitats (Scharler et al., 2016), thereby threatening ecosystem functioning,

The deep-sea plays a major role in reducing anthropogenic impacts on climate – its capacity is substantially larger than the atmosphere and land, and it has absorbed between 25-40 per cent of human-generated atmospheric carbon dioxide (CO_2) (Khatiawala et al., 2013; McKinley et al., 2016). Oceanic absorption of atmospheric CO_2 involves chemical, physical, and biological processes, which are all sensitive to temperature; and yet absorption of CO_2 is making the sea warmer and more acidic, thus reducing its ability to hold oxygen (Cao and Zhang, 2017). The Indian Ocean sea temperature is known to be increasing faster than other oceans (Hoegh-Guldberg et al., 2014; Roxy et al., 2014), of concern since increased temperatures have effects on marine communities. A meta-analysis of over 600 publications by Nagelkerken and Connell (2015) revealed that primary production by non-calcifying plankton in temperate waters increases with elevated temperature and CO_2 levels, whereas productivity of tropical plankton decreases because of acidification. Temperature increases metabolic rates in herbivores (and hence their consumption), but does not result in greater secondary production; instead, there are decreases in both calcifying and non-calcifying species. In carnivores, metabolic and foraging costs increase with increasing temperature. Species diversity and abundance decline with acidification in both tropical and temperate species, with a trend towards communities dominated by non-calcifying organisms. The CO_2 concentration affects the aragonite saturation state (ASS) of the ocean, and as ASS levels drop, the ability of calcifying organisms such as corals and shelled invertebrates to create calcium carbonate skeleton structures is reduced (Halpern et al., 2015).

A recently identified threat in offshore habitats, even less quantified for the WIO, is posed by methyl hydrates, and an overview is given in Bollmann et al. (2010). From sediments at depths greater than 350 m, and with water temperature of < 4°C, natural methane gas production in sediments can be stabilized into hydrates on the seabed, but with warming, the hydrates can break down, releasing methane. The hydrates are concentrated on continental slopes because that is where suitable conditions (depth, temperature and sufficient organic matter) are found to facilitate their production. Vast amounts of methane hydrate are buried in sediments on the slopes – containing far more carbon than released by fossil fuels. Micro-organisms oxidize the resulting methane gas to form the greenhouse gas CO_2 which will not only contribute to further global warming, it will also lead to increased acidification of oceans. There is also interest in mining of seabed hydrates from offshore habitats, which would accelerate release of methane. It is likely that there have been large-scale natural releases of methane over geological time which could have resulted in mass extinctions of deep-sea organisms – further investigations are needed to assess the scale at which climate change will accelerate due to changing temperatures at depth causing methane gas release at the sea floor.

Likely effects of climate change on global ocean hydrodynamics and circulation are still being debated. Global circulation includes transport of warm, less-saline water from the Pacific to the Indian Ocean and then into the Atlantic. Datasets are not available to assess long-term change in this thermohaline circulation, but there are recent indications that in the last ten years, the Indian Ocean has increasingly been taking up warmer water from the Pacific (Lee et al., 2015). Accelerated warming, together with intensifying winds, is reported to be accounting for the widening of the Agulhas Current in the south-west Indian Ocean (Beal and Elipot, 2016), which transports water to the Atlantic Ocean. Broadly, climate change effects are intensifying in the WIO region, particularly in areas that were previously less impacted, but available data sets for the region are limited, or are based on proxies rather than direct evidence (Halpern et al., 2015; Mahongo, 2015). Regional surveys, for example as part of the Second International Indian Ocean Expedition, and by the *RV Dr Fridtjof Nansen*, will improve predictive models.

STATUS

Given the vast spatial extent of the EBSAs, the multiplicity of habitats within these areas, and the lack of information on offshore habitats in the WIO, it is not practical to use the standard ecosystem indicators or IUCN ecosystem categories to assess status. The only WIO country which has made some progress towards this is South Africa, which has categorized 62 offshore (deeper than 30 m) benthic habitats that were defined on the basis of substrate, depth, slope, geology, grain size and biogeography, and 16 offshore pelagic habitats, that were defined based on sea surface temperature, productivity, chlorophyll, depth, eddies and fronts (Sink et al., 2012; see Case Study). Therefore, indicators of the status of offshore habitats herein is based on the “Naturalness” category determined during the EBSA process: "Area with a comparatively higher degree of naturalness as a result of the lack of or low level of human-induced disturbance or degradation". The Naturalness category/status was determined during the EBSA workshops by expert assessment of habitats within each EBSA, if information was available (SCBD, 2014; 2013). The rationale for assigning a particular Naturalness category (low, medium and high) to each EBSA is summarized in Table 3. For some of the very large EBSAs, coastal habitats are included, ie, the assigned category is not exclusively representative of the status of offshore habitats.

Of the 18 EBSAs, five scored high for Naturalness, seven scored medium and six scored low. Offshore habitats in a poor state are invariably affected by extraction of renewable and/or non-renewable resources. Very broadly, however, it may be assumed that offshore habitats are in a better state than coastal habitats, owing to remoteness from human populations, and pelagic habitats are in a better state than benthic habitats, owing to an absence of vulnerable static features which support communities in the former. To some extent the global meta-analysis by Halpern et al. (2008) supports this contention, with low cumulative human impact scores in deep-water ecosystems (although long-standing fishery effects there were underestimated); surprisingly, continental shelf sediments were considered to be as heavily impacted as hard shelf and rocky reef ecosystems, owing to influences from both land and ocean.

Table 3: Status of offshore habitats expressed in terms of 18 Offshore EBSAs and their Naturalness categories.

EBSA	Naturalness	Rationale
WIO EBSAs		
Agulhas Front	Low	Long-line fisheries operate in the area, and their bycatch of seabirds, particularly albatrosses, has caused considerable declines; there are whale entanglements in fishing gear and ships strikes on whales; however, this EBSA is still highly productive and large numbers of seabirds still feed there, indicating it retains some functionality. Areas around the islands are protected and managed, and have a high degree of Naturalness.
Agulhas Bank	Medium	Several pelagic and demersal fishery types operate in this area - they have caused damage to some reefs, and declines in several endemic fishes. Petroleum-related activities are expanding. Consequently, several species and habitats (sediment and reef) are categorized as Threatened (IUCN categories ranging from Critically Endangered to Vulnerable). There is only one pelagic habitat in this EBSA, which is in a good state, while the state of the various benthic habitats ranges from poor to good depending on exposure to fishing and petroleum activities.
Agulhas slope	High	Several pelagic and demersal fishery types operate in this area, and there are threatened species (turtles and seabirds) and threatened habitats (pelagic, sediment and reef). However, threat levels in this EBSA are lower than in other slope areas, partly because oceanographic and seabed features limit the potential for disturbance.
Offshore of Port Elizabeth	Low	There are a variety of pressures in offshore habitats in this EBSA, including a variety of fishery types. The overall state is declining, with fair to poor conditions in most habitats. There are a variety of species (turtles and seabirds) and multiple habitat types (including muds, canyons, sandy shelf) categorized as Threatened (from Critically Endangered to Vulnerable). However there are many areas which are in a good state.
Protea Banks and sardine route	Low	There are threatened habitats (particularly reefs), and threatened demersal fish species (due to fishing); the pelagic habitat state is good, with benthic habitats ranging from poor to good. Overall the state is categorized as fair to poor.
Natal Bight	Low	Threats to offshore habitats in this EBSA include demersal fisheries on shelf and slope sediments and reefs, developing petroleum and mining interests, and further reductions in nutrient and sediment supply from riverine runoff. Fisheries and dams have already resulted in Endangered states of some rare habitats, and Threatened species (turtles and fishes) occur here. The overall state is fair to poor, but parts of some habitats (reef, mud, gravel) are in a good state.
Delagoa shelf edge	High	There are limited current threats here, with existing protection and usage management zones in MPAs covering habitats to the shelf edge, and consequently most of the offshore habitats are in a good state (largely undisturbed), particularly documented for South African shelf habitats, which have had a longer period of more managed protection. Potential threats are petroleum exploration and proposed port development; pelagic longline fishing is not permitted within 20 nm of the coast.

EBSA	Naturalness	Rationale
Quelimane to Zuni River	Medium	Much of the shelf sediment habitat to ~100 m depth has been trawled extensively for many years, with both targeted prawns and some bycatch fish species being overexploited. This habitat is heavily reliant on nutrients and sediments from the Zambezi River which has been affected by existing dams; others are planned, which are likely to compromise the currently relatively pristine mangrove habitats which serve as nursery areas for the communities in offshore habitats. Reef habitats have also been heavily fished.
Mozambique Channel	Medium	This huge EBSA has very high levels of biodiversity; some Offshore habitats in the EEZs of several bordering countries have MPA protection at varying spatial scales and management levels, suggesting healthy status for these. There are numerous species (cetaceans, birds, fishes) categorized in different threat levels (Critically Endangered to Vulnerable) which occur here. Some offshore habitats are remote from human populations and are consequently less impacted, but, at the regional scale, vulnerability is high.
Southern Madagascar	High	There are low coastal population levels and limited pressures on offshore habitats here, but these threats may develop as other fishing areas become depleted. There are numerous species of cetaceans and birds categorized in different threat levels (Critically Endangered to Vulnerable) which occur here. Status can generally be categorized as good.
Northern Mozambique Channel	Low	Levels of human impacts differ in various locations in this large EBSA, but there are some areas still in a good undisturbed state. Numerous species (cetaceans, birds, fishes), categorized in different threat levels (Critically Endangered to Vulnerable), occur here. Overall status is poor (low Naturalness category), but this is due to the disturbed state of coastal habitats within the EBSA which have high population pressure; offshore habitats are likely to be in a better state.
EBSAs in the South African EEZ to the west of Cape Agulhas (ie, in the adjacent south-east Atlantic)		
Subtropical Convergence Zone	Low	Harvesting of whales took place here for many years, although population levels are recovering. There are likely to be fishing effects, but the area is still naturally highly productive, supporting bird and fish communities which feed here. This may be affected by climate change, but other human pressures on the area are not expected in the near future.
Benguela Upwelling System	Medium	Historical over-fishing, mining and petroleum exploration and production have had impacts on offshore habits in this EBSA, and there are additional pressures such as pollution, invasive species and altered freshwater outflows. The southern part of the EBSA (off South Africa) appears to have been more stable, assisted by conservative fisheries management, but eastward shifts in distribution of several key species have had negative effects on seabird populations. However, many habitats are in good condition, and overall the area can be considered to be in a moderately natural state.
Browns Bank	Medium	There is considerable trawl fishing pressure in offshore habitats here, with most outer shelf sediments in a poor state; one habitat is Critically Endangered with a very limited spatial extent, while some shelf-edge reefs are in a good state as they have not been trawled. The pelagic habitat is considered Vulnerable and is the most threatened of the pelagic habitats in the area. Of the bird species occurring here, the most threatened is categorized as Critically Endangered.
Cape Canyon and Surrounds	Medium	Several fisheries operate here, and the state of offshore habitats ranges from good to poor. Pressures in the form of petroleum exploration and prospecting for seabed mining are increasing. There are some habitats in a good state, particularly around the canyons and on reefs where trawling is limited.
Childs Bank	High	Much of this offshore habitat is in a good state, but with parts that are fair or poor, with fishing impacts on biodiversity or ecological process. Fishing effect has been declining, but damage to sessile benthic organisms on reef slope areas is continuing. Other anthropogenic pressures are low.
Orange Shelf Edge	High	In this EBSA, while the shelf edge and shelf sediment offshore habitats are in IUCN threatened categories of either Critically Endangered or Vulnerable, with varying degrees of habitat degradation and loss of ecosystem function, there are still parts which are in a good state, particularly in South African waters, because there are reduced threats in the form of fishing, mining or pollution.
Orange Cone	Medium	Several demersal and pelagic communities from offshore habitats are reliant on Orange River flow, and changes have been recorded as flows have altered. Coastal mining impacts are considerable, albeit confined to depths of 30 m; the inner shelf area is considered to be largely in a good state, but there have been long-term declines in fish catches, suggesting changing communities.

CASE STUDY

Assessing marine habitats in SA

Kerry Sink

Woman collecting ascidians on the rocky shore.
© José Paula

The South African National Biodiversity Institute (SANBI) uses a consistent approach to assess Ecosystem Threat Status and Protection Levels in the marine, terrestrial and inland aquatic realms. A practical, science-based method is used to assess the state of marine and other ecosystems and identify national priorities (SANBI and UNEP-WCMC, 2016). The 2018 and 2011 National Biodiversity Assessment classified, mapped and assessed 150 marine ecosystem types (Sink et al., 2019) and 136 marine habitat types (Sink et al., 2012). respectively. A systematic spatial approach is used to determine extent of threats to ecosystems ie, how much of each type is in a natural or near natural state, or alternatively, are losing key aspects of their structure, function and composition (SANBI and UNEP-WCMC, 2016), with the 2018 assessment aligned to the IUCN'S Red List of Ecosystems (RLE). Protection level is also determined as an indicator of the extent to which ecosystem types are represented in the protected area network.

The key requirements for such a national assessment include four key datasets. Primary inputs include a marine ecosystem classification and map, a map of ecological condition, a map of marine protected areas (MPAs), and biodiversity targets or thresholds that set a minimum proportion of each ecosystem type that should remain in good condition. More detail on the five specific steps for assessing thereat status and protection level are explained in the guidelines for mapping biodiversity priorities (SANBI and UNEP-WCMC, 2016) with a summary of key datasets and results presented in this case study. This approach can also be applied at a regional level as was undertaken for the Benguela Current Ecosystems (Holness et al., 2014).

South Africa's marine and coastal habitat and ecosystem classifications incorporate several key drivers of marine biodiversity pattern: terrestrial and benthic-pelagic connectivity, substrate, depth and slope, geology, grain size, wave exposure and biogeography. The 2018 ecosystem classification identified and mapped a total of 150 ecosystem types in six ecoregions (Sink et al., 2019). In 2011, a separate classification was undertaken to define 16 different offshore pelagic habitat types based on differences in sea surface temperature, depth productivity, chlorophyll and frequency of eddies, temperature fronts and chlorophyll fronts (Roberson et al., 2017). In 2018, key advances in the map of marine ecosystems included very fine-scale shore mapping with alignment between marine, terrestrial and estuarine realms in the coastal zone; the inclusion of kelp forests, bays, fluvial fans and stromatolite shores as distinct ecosystem types and the introduction of finer depth strata across shelves and the slope. This was as a result of major efforts to collate or increase relevant historic and current data sets to support improved ecosystem classification and mapping. Analyses of patterns in benthic fauna from grab, trawl and remotely operated vehicle surveys and other types of visual seabed survey including submersible, tow camera and baited underwater video, were used to inform the ecosystem map. More than 2000 visual surveys of the seabed were used to help classify seabed type and to provide information about biological assemblages. Kelp forests and the surf zone were mapped by contemporary remote sensing. Ecosystems were grouped into 15 broad groups to assess and report on patterns of threat and protection level.

Marine ecosystems and species face pressures from an increasing range and intensity of human activities. Pressure data is the second key input into ecosystem assessments including the new IUCN RLE. To assess marine ecosystem condition, a cumulative, pressure-mapping approach based on Halpern et al. (2008; 2009; 2015), was used in both 2011 and 2018. Maps reflecting the relative intensity of 31 pressures or drivers of ecosystem change were produced to determine ecosystem threat status.

These include 20 fisheries sectors, petroleum activities, mining, shipping, ports and harbours, coastal development, mariculture, freshwater flow reduction and pollution. Emerging pressures include plastic pollution, increased underwater noise and desalination. Pressures were considered individually and cumulatively as a surrogate for ecosystem degradation. Both the extent and intensity of pressures were considered in addition to an impact score per broad ecosystem type. An ecosystem condition map was produced with degradation assessed in four categories aligned to the IUCN RLE approach. IUCN RLE criterion C3, which focuses on ecosystem degradation, was the primary assessment criterion but ecosystem distribution was also considered in line with criterion B. The 2018 results are not directly comparable to the 2011 NBA results because of changes to the ecosystem maps and pressure data, and also differences in assessment methods. Key differences in 2018 include six additional pressures, finer-scale assessment at pixel level, application of four rather than three categories of ecosystem condition, and differences in thresholds for threat status categories.

The third key input is the need for quantitative biodiversity targets representing the minimum proportion of an ecosystem type that needs to be kept in a near natural or natural state. This is still a developing science and in 2011 South Africa used a pragmatic approach with a standard 20 per cent target for all marine ecosystem types. This also aligns with the IUCN RLE, which assigns Critically Endangered status to ecosystems that have lost more than 80 per cent of their geographic distribution over 50 years (Bland et al., 2017). Critically Endangered (CR) ecosystem types have ≤ 20 per cent of their original extent in good/natural ecological condition and are considered likely to have lost important components of biodiversity pattern, community structure and functioning.

In 2018, South Africa aligned more closely with the IUCN thresholds and further changes to this recent assessment are anticipated in the near future, as more work is needed to interrogate the criteria, thresholds and input data. Currently, half of South Africa's 150 marine ecosystem types are threatened. By area this equates to only 5 per cent of the ocean space around South Africa with more inshore and shelf ecosystem types threatened than those in the slope and abyss. Only two ecosystem types (1 per cent of types) are considered Critically Endangered; the Agulhas Muddy Mid Shelf and Browns Bank Rocky Shelf Edge. A further 22 types are considered to be Endangered (15 per cent) and 51 types are considered Vulnerable (34 per cent). The most threatened broad ecosystem groups include bays, islands, muddy ecosystem types and rocky ecosystems on the shelf and shelf edge. The cold temperate Southern Benguela ecoregion has more threatened ecosystem types than the warm temperate Agulhas ecoregion, while the subtropical Natal and Delagoa ecoregions have the least threatened ecosystems.

Ecosystem Protection Level is a major biodiversity indicator calculated to assess the extent to which ecosystem types are represented in South Africa's protected area network. To calculate this indicator, the distribution of ecosystems in relation to the distribution of protected areas was assessed using a standard 20 per cent biodiversity target, taking into account ecosystem condition. Protection level was determined on a scale that ranged from Not Protected (< 0.2 per cent of an ecosystem type in the protected area network) to Well Protected (≥ 20 per cent of an ecosystem type in good condition in the protected area network). The year 2018 was a significant year for Marine Protected Area (MPA) expansion with 20 new MPAs approved for declaration by the South African cabinet. These MPAs were proclaimed in 2019, increasing the number of South African MPAs from 26 to 42, inclusive of the Prince Edward Islands MPA in the Southern Ocean, and noting that some existing MPAs were expanded, or merged and expanded, in the new MPA network. Protection of the marine environment around mainland South Africa increased from < 0.5 per cent (approximately 4 900 km^2) in 2018 to 5.4 per cent (57 900 km^2) in 2019. Protection level was calculated for each of South Africa's 150 marine ecosystem types, post-declaration of the expanded MPA network (2019) and for comparison, before declaration (2018). In 2018, when there were 25 mostly coastal MPAs in existence, only 20 per cent of marine ecosystem types were Well Protected while 47 per cent were Not Protected.

Of the 70 ecosystem types that were Not Protected in 2018, 51 received their first protection with the declaration of the new MPAs in 2019. Thus, the introduction of the 20 new MPAs reduced the number of ecosystem types within the Not Protected category from 47 to 13 per cent. Most Not Protected ecosystem types are located in the deeper offshore environment, particularly on the slope, with most of the slope and abyssal ecosystem types still Poorly Protected. Thirteen previously Not Protected and four Moderately Protected ecosystem types advanced to Well Protected, an improvement from 20 per cent to 31 per cent of ecosystem types in this category. A total of 87 per cent of the 150 marine ecosystem types now have at least some representation in the MPA network.

The systematic spatial assessment of marine ecosystems in South Africa demonstrates the value of a spatial approach that can objectively inform spatial planning and prioritization where management resources are scarce. The case study on South Africa's Phakisa Blue Ocean Economy (Fielding and Sink, 2021) shows how these same spatial layers were used to develop a proposed network of offshore MPAs.

EXISTING PROTECTION

A large part of the WIO falls within the EEZs of the bordering states and territories (see Table 2), all of which, with the exception of Somalia, have mechanisms for the inclusion of offshore habitats in declarations of formal MPAs. Further, most of these states have some capacity for at least sector-based marine spatial planning which can offer protection eg spatial fisheries closures, potentially considered under the category of Other Effective Area-based Conservation Measures (OECMs). As will be seen below, there are a variety of regional, transnational, national and sub-national institutions and actors in the WIO whose mandates and/or mechanisms provide for protection of offshore habitats. These will be considered separately in terms of jurisdiction – for states, this means mandates and mechanisms applicable to their EEZs, with their equivalents in ABNJs being considered separately.

There are overlaps though – for example, pelagic fishes such as tuna within a state's EEZ are under the jurisdiction of the regional fisheries body (see below), while benthic organisms and mineral resources are under the jurisdiction of the state. The *MPA Outlook* report (UNEP-Nairobi Convention, GEF and WIOMSA, 2021) describes in detail the mandates, governance and management of protected areas in EEZs of WIO states and should be consulted in this regard – only a brief summary thereof is provided here. Locally Managed Marine Areas (LMMAs) are not considered here as they do not currently offer protection to offshore habitats (UNEP-Nairobi Convention, GEF and WIOMSA, 2021). Therefore the focus here is on formal MPAs for habitat protection, although OECMs in the form of fishery reserves form part of MPAs in some instances. In this regard, it is worth noting that most of the WIO island states (excluding Madagascar) do not permit demersal trawling in their EEZs. While there is a WIO Regional Fisheries Body in the form of the South West Indian Ocean Fisheries Commission, under the aegis of FAO, its advisory role is confined to state EEZs and it has no mandate for declaration of protected areas.

In the three islands comprising the Union of the Comoros, there is only one formal protected marine area, the Marine Park of Moheli, declared by national decree of the head of state, and administered by the Ministry of Environment, with a park management committee incorporating local communities. There is no continental shelf as such, a consequence of the volcanic origins of the islands; protected habitats are thus essentially coastal, with complete protection limited to smaller areas within the Park. A national network of Protected Areas is planned, which will be declared by a superseding decree, and which will be co-managed by a protected areas agency and village communities, legislated under the Comoros Protected Areas Act and its laws which are still being considered by the state government, and which will be under the jurisdiction of the proposed Ministry of Protected Areas. Various National Parks boards and committees will also play a management role. Offshore habitats which could receive formal protection under these mechanisms are deep volcanic slopes and pelagic marine areas.

There are a variety of types amongst the five MPAs in the islands comprising the Indian Ocean French Territories. Similarly to the Comoros, continental shelf areas are negligible; the motivation for the protection is mostly vulnerable, coastal, shallow-water habitats, while inhabitants of epipelagic habitats (eg fishes, mammals and birds) also effectively receive protection depending on levels of compliance with the zonation of protection levels; deep (> 1000 m) sea-bed habitats falling within the MPA boundaries also benefit. Mandate for declaration mainly stems from the French Ministry for Ecological and Inclusive Transition, and management responsibility often rests with the French Biodiversity Agency, which convenes a forum of MPA managers. The MPAs are proclaimed by decree of the relevant local authority (island prefecture). There is also management input from a range of advisory panels and committees specific to each MPA, as well as a variety of other interest groups.

Kenya has six MPAs containing zoned no-take/no disturbance areas, which were proclaimed for protection of coastal habitats. These are under the authority of the Wildlife (Conservation and Management) Act of the Ministry of Environment and Natural Resources. The parastatal Kenya Wildlife Service is responsible for management, with participation by a range of government agencies, NGOs, local communities and the private sector. Offshore habitats thus do not currently feature in MPAs; there are plans for a transboundary marine conservation area between Kenya and Tanzania, the seaward boundary of which corresponds with the 200 m depth contour (approximately five nautical miles offshore), and which will afford protection to shelf sediment habitats.

The 22 MPAs in Madagascar are decreed by the Ministry of Environment, Ecology and Forests, and are under their guardianship as well as under the Ministry of Marine Resources and Fisheries. They are managed by the parastatal Madagascar National Parks agency, collaborating with NGOs, local communities and the private sector, either individually or in combination; most MPAs are co-managed. Offshore habitats receive limited protection, other than some areas of the continental shelf, as the MPAs are essentially designated for coastal habitat protection.

In Mauritius, while the Maritime Zones Act provides for some elements of protection of the marine environment, the Environment Protection Act provides the legal framework for such protection and management thereof, while the Fisheries and Marine Resources Act provides for proclamation and management of MPAs. Altogether there are 18 MPAs in Mauritius and Rodriguez, in the form of marine parks, fishing reserves and marine reserves, but these are all essentially coastal, with the parks having the furthest seaward extent, out to only 1 km beyond the fringing reefs. In Mauritius they are managed by the Ministry of Ocean Economy, Marine Resources, Fisheries and Shipping (Fisheries Division), with activities in fishing reserves being controlled by the Fisheries Protection Service and local coast guard stations. In Rodrigues, management is mainly via the Commission for Agriculture, Forestry, Fisheries and Marine Parks.

In Mozambique, the Law for the Protection, Conservation and Sustainable Use of Biological Diversity provides the framework for protection of habitats, while the Fisheries Law deals specifically with fisheries conservation areas and closed seasons. Proclamation of individual protected areas is via specific decrees sanctioned by the National Council of Ministers. There are seven MPAs in Mozambique (two national parks, one marine and two national reserves, one total protection zone, and one environmental protection area). Of these, four include offshore habitats in the form of offshore pelagic habitats, and deep-sea benthic habitats including canyons, seamounts and ridges. The legally mandated management institution is the Ministry for Land, Environment and Rural Development, through its National Administration for Conservation Areas, with various advisory management committees. These include representatives of local government, local communities, NGOs and the private sector. Interestingly, agreements have been signed with the Ministry in some MPAs, permitting non-profit conservation organizations to manage the areas for fixed periods of up to 50 years.

In Seychelles, the 16 formal MPAs for habitat protection have been designated either under The Environment Protection Act or (mostly) the National Parks and Nature Conservancy Act, and provide for protection of habitats in four categories: Area(s) of Outstanding Natural Beauty, Strict Natural Reserves, Special Reserves and National Parks. Most only protect shallow-water coastal habitats, with only the two most extensive including deep-sea and offshore pelagic habitats. The MPAs are variously managed by foundations, societies and authorities, frequently with directors appointed by the President of Seychelles. A Protected Area Policy was recently developed as a framework for establishment and management of protected areas (including MPAs). Several new extensive MPAs have recently been finalized, and include offshore habitats for protection, supported by a new institutional framework for their management.

South Africa has 42 formal MPAs, with protected habitats in sanctuaries, and restricted, controlled and no-take zones. This total includes the Prince Edward Islands MPA, within its own EEZ. Of the remaining 41 sites, many are coastal MPAs, and only 15 could be considered to protect offshore habitats in the EEZ of mainland South Africa (as distinct from the islands of the Southern Ocean). The primary legal instrument for the establishment and protection of MPAs is the National Environmental Management: Protected Areas Act, promulgated by the Department of Environment Affairs (DEA), which is the nationally-mandated management authority for all MPAs, and which has contracted a range of (mainly) local government conservation authorities to manage them. DEA's National Protected Area Expansion Strategy developed conservation targets to particularly address protection

Plumes from major rivers in the WIO extending far offshore. © Fiona MacKay

of offshore habitats, and 22 new/extended MPAs were proposed in 2016 for promulgation in this regard, largely coinciding with EBSAs. On 26th October 2018, 20 of these proposed MPAs were approved by Parliament and are included in the current total of 41 MPAs within the mainland EEZ.

In the United Republic of Tanzania, the 18 Mainland MPAs fall under the Marine Parks and Reserves Act under the auspices of the Ministry of Livestock and Fisheries; the Marine Parks and Reserves Unit which manages MPAs was also constituted by this Act, with oversight by the Board of Trustees for Marine Parks and Reserves, although there is co-management with community members, advisory committees and other stakeholders. The MPAs are mostly coastal, with minor inclusion of deep sea/epipelagic areas for protection. Proposed new MPas are also coastal, and the proposed trans-frontier marine conservation area between Kenya and Tanzania will include part of the Pemba Channel which will afford some protection to deeper shelf habitats. In Zanzibar, the Environmental Management Act allows for areas to be declared for protection, with management by the Marine Conservation Unit now the Department of Marine Conservation, under the new Ministry of Blue Economy and Fisheries. The nine MPAs of Zanzibar are partially protected areas having a focus on fisheries management, with extensive involvement of communities; none can be considered to protect offshore habitats, apart from one which borders the deep ~1000 m Pemba Channel.

Regarding offshore habitats beyond EEZs, notwithstanding that the Nairobi Convention has no specific mandate in ABNJ, member states agreed in 2015 to co-operate in improving governance beyond their EEZs, and the Convention secretariat assumed a co-ordinating and advisory role; it became a partner in activities dealing with ABNJ governance and mechanisms for habitat protection, and has facilitated several projects in this regard – of particular significance for offshore habitats being the EBSA process. Readers should consult Wright and Rochette (2017) and UNEP-WCMC (2017) for more comprehensive reviews of governance of ABNJ in the WIO. There are several international organizations and/or legal instruments which have mandates that incorporate mechanisms for protection of habitats in ABNJs, and there are Regional Seas Conventions establishing MPAs in ABNJ in the Atlantic, Pacific and Southern oceans (Rochette et al., 2014). Albeit that there are benefits to be gained by the Nairobi Convention assuming a more prominent role in ABNJ governance, it is concerning that, given non-regional states' interest in the WIO, Regional Seas MPAs are only binding on parties to the Regional Seas Programme. International legal instruments are promoted by the Division for Ocean Affairs and the Law of the Sea (DOALOS) of the United Nations General Assembly (UNGA), which is the only global platform at which ABNJ habitat protection can be discussed. The organizations and instruments with mandates for habitat protection, can be broadly categorized into four sectors, dealing with fisheries, shipping, mining and environmental protection.

For WIO fisheries, the regional bodies with ABNJ mandates are the Indian Ocean Tuna Commission (IOTC) and the South Indian Ocean Fisheries Agreement (SIOFA), with their counterparts of the South African west coast – the International Commission for the Conservation of Atlantic Tunas (ICCAT) and the South East Atlantic Fisheries Organization (SEAFO). The Commission for the Conservation of Southern Bluefin Tuna (CCSBT) can be included here too. Various international instruments (eg United Nations Convention on the Law of the Sea – UNCLOS) and UNGA resolutions require these Regional Fisheries Management Organizations (RFMOs) to take conservation actions. The tuna RFMOs have numerous binding and non-binding measures to reduce bycatch of cetaceans, seabirds, turtles and sharks, and their websites should be consulted for these. The IOTC currently has no designated areas for habitat protection (a small area with a time/area closure existed from 2010-2015), and in 2006, SIOFA, through its members, the South Indian Ocean Deepwater Fisheries Association (SIODFA), declared self-enforced Benthic Protected Areas (BPAs) in its area of competence, which exclude trawling by members of SIODFA, and added to these areas in 2013 (Fig. 1). There are ongoing initiatives by Contracting Parties to SIOFA to reduce the amount and spatial scale of trawling effort to existing fished areas, and to formally record instances of encounters with vulnerable habitats for potential protection (Shotton, 2018). This is in response to the UNGA 2006 resolution that measures be implemented by fisheries organizations to protect habitats, which included: impact assessments to prevent significant adverse impacts on vulnerable marine ecosystems (VMEs) in ABNJ; catch threshold protocols which require vessels to move away when they encounter a VME; and closure to demersal fishing in areas where VMEs occur or are likely to occur, until conservation measures have been established. Currently there are no formally designated VME areas in the WIO ABNJ, and there has been resistance from some SIOFA parties to the conversion of the SIODFA BPAs into VMEs (Guduff et al., 2018). For the purposes of this report the Africana Seamount VME could be considered to be within the WIO, being well to the east of Cape Agulhas, but falls with the mandate of SEAFO. It is of significance that the WIO BPAs only apply to SIODFA members – so non-members are not bound by them; additionally, it is of concern that benthic fishing effort in the SIOFA competence area will expand (Guduff et al., 2018).

For shipping-related habitat protection mechanisms, the International Maritime Organization (IMO) is a UN agency with responsibility for safety and security of shipping, and ship-derived pollution. Member states can designate Special Areas under the International Convention for the Prevention of Pollution from Ships (MARPOL) and Particularly Sensitive Sea Areas (PSSAs), in both EEZs and ABNJ, through the IMO's Marine Environment Protection Committee, to protect habitats from environmental impacts due to shipping. From a habitat protection perspective, the relevant associated protective measures (APMs) include pollution control measures, and navigational measures such as areas to be avoided and preferential routeing. However, few PSSAs exist, there are currently none designated in ABNJ, and states are not legally bound to adhere to designating resolutions; experience to date with PSSAs suggests the process is challenging (Wright and Rochette, 2017).

Regarding mining, as constituted under UNCLOS, the International Seabed Authority (ISA) is responsible for activities associated with exploration for, and exploitation of, non-renewable mineral resources (solid, liquid or gaseous mineral resource) on the seabed in ABNJ (see section on Threats above). The ISA can designate Areas of Particular Environmental Interest (APEI) to exclude mining, and its regulations can prevent prospecting if there is considerable evidence that serious harm to habitats can be incurred; to date, although exploration contracts in the WIO have been awarded, no APEIs have yet been contemplated. Contractors who receive permits can independently designate Impact and Preservation reference zones to assist with assessing impacts.

Several international environmental conventions provide mechanisms for protection of habitats or species. The CBD of the United Nations Environment Programme (UNEP) facilitated the development of scientific criteria to identify and justify EBSAs for protection, and, partly facilitated by the Nairobi Convention, several such areas have been listed for the WIO region (Table 1). The World Heritage Convention (WHC) of the United Nations Educational, Scientific and Cultural Organization (UNESCO) has designated two marine world heritage sites in the region for protection, based on their natural significance, but only one (iSimangaliso on the east coast of South Africa) can be considered to protect offshore habitats by virtue of its seaward extent. The Convention on Migratory Species (CMS), under the aegis of UNEP, has several instruments for the protection of habitats of endangered and vulnerable species, the most prominent of these in the WIO being dugongs and turtles. Albeit not dealing with habitat protection as such, the International Whaling Commission designated the Indian Ocean Sanctuary in 1979, which prohibits commercial whaling in the whole of the Indian Ocean, effectively including the WIO.

In summary, although there are a variety of mechanisms for protection of offshore habitats, both within EEZs and in ABNJs, they are largely uncoordinated, mainly being predicated on the interests of the state concerned (in EEZs), and on the requirements of a specific sector or sectors (eg fisheries, mining). The following section discusses this further in the context of the necessity for coordinated conservation mechanisms in the face of the need for additional protected offshore habitats.

PRIORITY OPTIONS FOR CONSERVATION

It is apparent from the previous section, and previously confirmed by Chevallier (2017), that WIO protected areas have many different governmental processes involved in their designation and management, and that the legal, institutional and policy frameworks of WIO states are not coordinated or integrated. The complexity of the governance partly stems from the diversity of legal regimes governing marine and coastal zones which include internal waters, territorial seas, contiguous zones, continental shelves, slopes and rises, EEZs, and the high seas. These zones each have a rationale for their designation, predicated on their value and importance to the adjacent terrestrial state, largely in terms of the exploitation or use which takes place there (or may take place in the future). The situation has heightened complexity because some WIO states have made submissions to the Commission on the Limits of the Continental Shelf to extend the outer limits of their continental shelves; others have disputes around the extent of their EEZs. Beyond EEZs, the international community has become increasingly aware of growing interest in resources in ABNJs and the threats this poses (Wright and Rochette, 2017). Offshore habitats, particularly those in deep waters, are extremely vulnerable to disturbance, owing to the long-lived, slow-growing attributes of the faunal inhabitants (Clark et al., 2016). Notwithstanding that the seabed in ABNJs is considered to be the common heritage of humankind and is subject to the provisions of UNCLOS, there has been slow progress in ensuring this (see below).

Much of the protection, or at least potential for protection, currently afforded to habitats is designated in terms of a specific sector, such as fisheries (particularly), as well as shipping or mining. Protection based on fisheries mechanisms does not necessarily protect habitats from other exploitation threats, and frequently only addresses one type of fishing, permitting other types; it also tends to focus on harvestable organisms rather than the habitat itself. So the downside of a sector-based approach to governance and regulations for habitat protection is that there are often spatial and legal gaps in management – without an overarching mechanism, which is currently lacking, some offshore habitats which require protection may not be afforded it (Gjerde, 2008; Gjerde et al., 2013).

It is also apparent from the previous section, and from Table 2, that offshore habitats in EEZs and ABNJ have little formal protection in the region, as most MPAs have a coastal focus. There are no MPAs in ABNJ in the WIO region, and only 5 per cent of the total EEZ area falls within MPAs, most of which is continental shelf area, while only South Africa and Seychelles include protection for beyond-shelf areas (UNEP-Nairobi Convention, GEF and WIOMSA, 2021). A similar conclusion was recently drawn by Fischer et al. (2019), in a meta-analysis of MPA coverage of 19 offshore geomorphic seabed features – globally, none of these features receive more than 7 per cent protection, in contrast to coral reefs, mangroves and seagrasses which receive protection of between 18-41 per cent of their area. Demersal trawling is potentially feasible throughout the region, apart from in the SIODFA Benthic Protected Areas (which only apply to the nine contracting states), and in the EEZs of small island states; effectively, though, extreme depth (> 2000 m) precludes this type of fishing, and much of the demersal trawling which occurs is focussed on seamounts because of their fish aggregations (see Chapter 17). Fishing for medium-sized and large pelagic species, via small-scale gillnets and/or industrial longline and purse seine, occurs over most of the region, in EEZs and ABNJs. There are no PSSAs or APEIs to exclude shipping or mining activities from vulnerable offshore habitats in the WIO. A recently published study shows that the Indian Ocean has no areas that are not exposed to anthropogenic stressors (Jones et al., 2018). At the same time, there is a paucity of information to help prioritize such habitats for protection, particularly beyond the continental shelf. Lack of knowledge should not be a deterrent, however. Indeed, it is the lack of knowledge itself which should encourage caution.

EBSAs are the most suitable regional approach to elaborate the need for additional protection for offshore habitats, by applying internationally-agreed scientific criteria. Identification of EBSAs is intended to alert states, and regional and global intergovernmental agencies, about the significance of habitats and to motivate for their protection. The broadening of EBSAs to include areas within the jurisdiction of states means that they "...can use the EBSA process to (1) support CBD National Biodiversity Strategies and Action Plans, (2) promote the status of previously identified national protected areas, and (3) potentially increase access to international funding for area-based planning, resource management and conservation efforts" (Dunn et al., 2014). Albeit that these authors make the point that there is no obligation for a state to convert an EBSA into a protected area or to manage it accordingly, the EBSA process can contribute to planning and designation for protection. The size of some EBSAs (eg the Mozambique Channel) makes it impractical for them to be either designated or managed as fully protected areas, and smaller individual EBSAs should be considered within these (SCBD, 2013).

The EBSA process continues to evolve, and the evidence in support of existing EBSAs, as well as that for identifying new EBSAs, is being updated and strengthened to fill gaps, especially in ABNJ. Recent modifications include discussions around categorization of EBSAs into four site categories: fixed, transient (mobile fronts), scattered or grouped, and ephemeral (seasonal). Globally, several states have used EBSAs to inform their national processes for habitat protection and management, or to motivate for research funding to support gathering of additional evidence. In the WIO, South Africa's Blue Economy initiative (Operation Phakisa) has used EBSAs in combination with other marine spatial planning products to propose expanded protection in its offshore habitats (see Case Study), and the Northern Mozambique Channel spatial planning initiative (WWF and CORDIO, 2018) has built on the EBSA process's identification of the importance of this region for protection.

EBSAs will be an important component in the development of an international legally-binding instrument to enhance protection in ABNJ (see Recommendations). As early as 2004, the UNGA created a working group on biodiversity beyond national jurisdiction (BBNJ Working Group) to discuss conservation and sustainable use of marine biological diversity in ABNJs. Following its concluding meeting in 2015, states recommended to the UNGA that it open negotiations for a legally binding instrument under UNCLOS. The recommendation was endorsed. Subsequently, four meetings of a Preparatory Committee (PrepCom) took place in 2016 and 2017, which culminated in a resolution at the end of 2017 to convene an intergovernmental conference, comprising four ten-day meetings over three years, commencing in September 2018. The conference considered the PrepComm final report recommendations, and elaborated the text of an international legally binding instrument which addresses, amongst others, MPAs in ABNJs. Regrettably, the COVID pandemic meant that the fourth and final negotiating session was postponed, and agreement with the terms of the instrument was finally reached in March 2023.

The PrepComm final report (Morgera et al., 2017) reflected difficulties in achieving consensus amongst participants, particularly around terms defining geographic jurisdiction, potential prejudice of existing legal instruments and frameworks, wording expressing the trade-off between conservation and sustainable use, and about the need for area-based management tools including MPAs in ABNJs. Kraska (2018) also expresses doubts about states' willingness to accede to constraints on navigation and to give up their potential preferential access to adjacent offshore resources, and suggests that they would likely only support small changes to existing instruments. In contrast, Elferink (2018) provides some suggestions for the negotiations, emphasizing that due regard for the rights of coastal states' is paramount, while De Santo (2018) makes recommendations on improving the evidence to support MPAs in ABNJ, as well as on compliance mechanisms and stakeholder engagement. There is still lack of agreement between States on many of the draft provisions (Humphries and Harden-Davies, 2020), and the implementing agreement will have profound implications for governance of ABNJs and protection of Offshore habitats in the WIO. In order to inform the discussions, the ABNJ Deep Seas Project (UNEP-WCMC, 2018), jointly implemented by FAO and UNEP, aims to improve understanding, cooperation and capacity among the various stakeholders about the use of area-based planning methodologies, including EBSAs in ABNJ. Other initiatives are also working towards this goal, such as the STRONG High Seas project (Gjerde et al., 2018).

Weak governance in the face of increasing human pressures is a major impediment to protection of marine and coastal environments in the WIO, and Momanyi (2015) elaborates on the issues at some length, drawing on the work by UNEP/Nairobi Convention Secretariat, CSIR and WIOMSA (2009). Guerreiro et al. (2011) also highlight weaknesses, notably that an international treaty is not binding on states unless they ratify it, and that not all WIO states are party to these instruments, regardless of their merits. These authors, and more recently Sorby (2018), make the point that bilateral or sub-regional mechanisms can be simpler and more effective to achieve habitat protection; to some extent this is the case in the Lubombo transboundary MPA between South Africa and Mozambique, and steps are being taken towards a similar arrangement between Kenya and Tanzania. Wright and Rochette (2017) and UNEP-WCMC (2017) concur that the WIO is not as advanced as some regions in terms of governance of ABNJ, but that there are some positive signals.

Critically, few WIO states have resources to manage coastal protected areas under their jurisdiction, let alone offshore habitats, to ensure that the protection is effective (Obura, 2015a), and the *MPA Outlook* (UNEP-Nairobi Convention, GEF and WIOMSA, 2021) considers this in detail.

RECOMMENDATIONS

The main findings of this chapter can be summarized as follows: there is a need for protection for offshore habitats in the WIO as they are currently poorly protected; because of their vastness, there is a need to prioritize areas within these habitats, but the majority remain underexplored, and information is lacking; the EBSA process has prioritized some areas in terms of their vulnerability and environmental importance; the EBSA process continues to evolve as additional work is done and as new information becomes available; there is increasing interest in further exploitation of renewable and non-renewable resources in these habitats, with associated threats; there are mechanisms in place for declaration of protected areas within state EEZs and now the new agreement has been adopted and ratified, there is a mechanism to declare international MPAs; and there is need for effective management of existing protected areas in offshore habitats in the WIO.

From this follows recommendations for the main stakeholders; they are not in order of priority, and are not necessarily mutually exclusive. They are partly derived from several recent sources which recognize the urgency for improving offshore protection in the face of increasing threats, including: Johnson et al. (2016), Wright and Rochette (2017), UNEP-WCMC (2017), Wynberg (2015), Chevallier (2017), Guduff et al. (2018).

The Nairobi Convention and its structures should:
- negotiate with Parties to extend the Convention's mandate to include ABNJ;
- foster political awareness of issues relating to ABNJ;
- facilitate the production of an inventory (atlas) of existing and planned anthropogenic activities in WIO offshore habitats;
- continue to provide a platform to solicit global funding agencies for support with capacity building in marine spatial planning, offshore habitat research, and management of MPAs;
- facilitate the development of a standardized approach to assessing management effectiveness for offshore habitat MPAs in the WIO region;
- continue to facilitate initiatives to support and mentor MPA stakeholders in the WIO region;
- encourage regional cross-sectoral area-based planning to avoid the gaps in protection caused by a purely sectoral approach; and
- facilitate access by regional research institutions to scientific information from surveys of offshore habitats by national and multinational agencies and companies.

States (including non-WIO states in some instances), should:
- recognize that the vast majority of offshore habitats are under-explored, frequently vulnerable to anthropogenic impact, and that ecosystem services are not always able to be assigned a financial value; but should recognize that this does not imply these habitats can be exploited without caution;
- facilitate their scientists' participation in the EBSA process to identify/justify particularly vulnerable offshore habitat areas for potential protection;
- strive towards building WIO regional capacity for marine spatial planning, incorporating the prioritization of areas in offshore habitats for use zonation and protection;
- avoid declaration of protected areas if they do not have the will or resources to manage them;
- improve efforts towards ensuring that their citizens and state utilities, as well as companies utilizing state-owned resources, adhere to protected area and other pertinent environmental regulations;
- strive towards bilateral/sub-regional agreements on transboundary MPAs for offshore habitats;
- actively participate in the UNGA process which, amongst others, aims to facilitate declaration and governance of MPAs in ABNJ;
- encourage and make available resources to facilitate research on offshore habitats, particularly those that are under-explored; and
- urgently facilitate the process of final designation of their EEZs and extended continental shelf claims.

Regional fisheries bodies should:
- urgently prioritize converting the SIODFA's BPAs into formal VME closures;
- actively promote the identification and designation of additional BPAs, and VMEs; and
- promote the identification of and rationale for closed fishing areas for large pelagic fish species.

Shipping stakeholders, through the IMO, should:
- be more receptive to the designation of PSSAs and adoption of Associated Protective Measures in offshore habitats in the WIO.

Entities which prospect for or extract renewable or non-renewable resources should:
- ensure their operations comply with state regulations and laws of mandated regional and international bodies;
- through the ISA, be more proactive in regard to the need for and declaration of APEIs, Impact reference zones and Preservation reference zones in ABNJ; and
- on request, make environmental data in their prospecting and exploitation areas accessible to research institutions.

REFERENCES

Ahnert, A. and Borowski, C. (2000). Environmental risk assessment of anthropogenic activity in the deep-sea. *J. Aquat. Ecosyst. Stress Recovery* 7(4), pp. 299–315

ASCLME/SWIOFP (2012). Transboundary Diagnostic Analysis for the western Indian Ocean. Volume 1: Baseline. ISBN: 978-0-620-57042-8

Atkinson, L.J., Field, J.G. and Hutchings, L. (2011). Effects of demersal trawling along the west coast of southern Africa: multivariate analysis of benthic assemblages. *Mar. Ecol. Prog. Ser*. 430, pp. 241–255

Baker, E., Gaill, F., Karageorgis, A.P., Lamarche, G., Narayanaswamy, B., Parr, J., Raharimananirina, C., Santos, R., Sharma, R. and Tuhumwire, J. (2016). Chapter 23; Offshore Mining Industries. *First Global Integrated Marine Assessment* (First World Ocean Assessment). United Nations. 34 pp.

Barnes, C. and Mfodwo, K. (2012). *A market price valuation of tuna resources in the Western Indian Ocean – an indicative regional and country/EEZ perspective*. WWF unpublished report. 100 pp.

Beal, L.M. and Elipot, S. (2016). Broadening not strengthening of the Agulhas Current since the early 1990s. *Nature* 540, pp. 570–573

Best, P.B., Findlay, K.P., Sekiguchi, K., Peddemors, V.M., Rakotonirina, B., Rossouw, A. and Gove, D. (1998). Winter distribution and possible migration routes of humpback whales Megaptera novaeangliae in the southwest Indian Ocean. *Mar. Ecol. Prog. Ser*. 162, pp. 287–299

Bland, L.M., Keith, D.A., Miller, R.M., Murray, N.J. and Rodríguez, J.P. (eds.) (2017). Guidelines for the application of IUCN Red List of Ecosystems Categories and Criteria, Version 1.1. International Union for the Conservation of Nature. Gland, Switzerland

Boersch-Supan, P., Rogers, A. and Brierley, A. (2017). The distribution of pelagic sound scattering layers across the southwest Indian Ocean. *Deep Sea Res. Part II Top. Stud. Oceanogr*. 136, pp. 108–121

Boetius, A. and Haeckel, M. (2018). Mind the seafloor. *Science* 359(6371), pp. 34–36

Bollmann, M. (2010). World ocean review: living with the oceans. Maribus and Kiel Marine Sciences. 236 p.

Cao, L. and Zhang, H. (2017). The role of biological rates in the simulated warming effect on oceanic CO_2 uptake. *J. Geophys. Res. Oceans* 122(5), pp. 1098–1106

CBD (2018). *Report of the expert workshop to develop options for modifying the description of areas meeting the criteria for ecologically or biologically significant marine areas (EBSAs), for describing new areas, and for strengthening the scientific credibility and transparency of the EBSA process*. 5–8 December 2017. CBD/EBSA/EM/2017/1/3, 36 p. Berlin, Germany

Chevallier, R. (2017). Integrated marine and coastal management in the Western Indian Ocean: towards a sustainable oceans economy. South Africa Institute of International Affairs. Occasional Paper 258, 44 p.

Clark, M.R., Althaus, F., Schlacher, T.A., Williams, A., Bowden, D.A. and Rowden, A.A. (2016). The impacts of deep-sea fisheries on benthic communities: a review. *ICES J. Mar. Sci*. 73(suppl_1), pp. i51–i69

Cochrane, K.L., Butterworth, D.S.W. and Payne, A.I.L. (1997). South Africa's offshore living marine resources: the scientific basis for management of the fisheries. *Trans. R. Soc. South Africa* 52(1), pp. 149–176

Coetzee, J.C., Van der Lingen, C.D., Hutchings, L. and Fairweather, T.P. (2008). Has the fishery contributed to a major shift in the distribution of South African sardine? *ICES J. Mar. Sci*. 65(9), pp. 1676–1688

Cordes, E.E., Jones, D.O., Schlacher, T.A., Amon, D.J., Bernardino, A.F., Brooke, S., Carney, R., DeLeo, D.M., Dunlop, K.M., Escobar-Briones, E.G., Gates, A.R., Genio, L., Gobin, J., Henry, L-A., Herrera, S., Hoyt, S., Joye, M., Kark, S., Mestre, N.C., Metaxas, A., Pfeifer, S., Sink, K., Sweetman, A.K. and Witte, U. (2016). Environmental impacts of the deep-water oil and gas industry: a review to guide management strategies. *Front. Environ. Sci*. 4, pp. 1–25

De Santo, E.M. (2018). Implementation challenges of area-based management tools (ABMTs) for biodiversity beyond national jurisdiction (BBNJ). *Mar. Policy* 97, pp. 34–43

Dunn, D.C., Ardron, J., Bax, N., Bernal, P., Cleary, J., Cresswell, I., Donnelly, B., Dunstan, P., Gjerde, K., Johnson, D. and Kaschner, K. (2014). The convention on biological diversity's ecologically or biologically significant areas: origins, development, and current status. *Mar. Policy* 49, pp. 137–145

Elferink, A.G.O. (2018). Coastal states and MPAs in ABNJ: Ensuring consistency with the LOSC. *Int. J. Mar. Coast. Law* 33(3), pp. 437–466

Eriksen, M., Lebreton, L.C., Carson, H.S., Thiel, M., Moore, C.J., Borerro, J.C., Galgani, F., Ryan, P.G. and Reisser, J. (2014). Plastic pollution in the world's oceans: more than 5 trillion plastic pieces weighing over 250,000 tons afloat at sea. *PloS one* 9(12), e111913

Everett, B.I, Groeneveld, J.C, Fennessy, S.T., Dias, N., Filipe, O., Zacarias, L., Igulu, M., Kuguru, B., Kimani, E., Munga, C.M., Rabarison, G.A., Razafindrakoto, H. and Yemane, D. (2015). Composition and abundance of deep-water crustaceans in the Southwest Indian Ocean: Enough to support trawl fisheries? *Ocean Coast. Manag*. 111, pp. 50–61

Fennessy, S.T. and Everett, B.I. (2015). Crustacean shallow-water trawl fisheries, a retrospective analysis of their status in the Southwest Indian Ocean, Chapter 2, pp. 19−65. In *Offshore fisheries of the Southwest Indian Ocean: their status and the impact on vulnerable species* (eds. van der Elst, R.P. and Everett, B.I.). Oceanographic Research Institute, Special Publication, 10. 448 pp.

Fennessy, S.T. and Green, A. (2015). Shelf sediments and biodiversity, Chapter 8. In *The Regional State of the Coast Report: Western Indian Ocean* (ed. Paula, J.). UNEP and WIOMSA, pp. 103−114. Nairobi, Kenya

Fennessy, S.T., Krakstad, J., Groeneveld, J.C., Bianchi, G. and Everett, B.I. (2017). Demersal resources based on bottom trawl and other sampling methods. In *The RV Dr Fridtjof Nansen in the Western Indian Ocean: Voyages of marine research and capacity development* (eds. Groeneveld, J.C. and Koranteng, K.A), pp. 101−124. FAO, Rome, Italy

Fennessy, S.T., Roberts, M.J. and Barlow, R.G. (2016). An editorial synthesis of the ACEP project: Ecosystem Processes in the KwaZulu-Natal Bight. *Afr. J. Mar. Sci*. 38(sup1), pp. S217−S223

Fielding, P. and Sink, K.J. (2021). Phakisa Blue Ocean Economy. In UNEP-Nairobi Convention, GEF and WIOMSA (2021). *Western Indian Ocean Marine Protected Areas Outlook: Towards achievement of the Global Biodiversity Framework Targets*. UNEP and WIOMSA, Nairobi, Kenya. pp. 135−137

Fischer, A., Bhakta, D., Macmillan-Lawler, M. and Harris, P. (2019). Existing global marine protected area network is not representative or comprehensive measured against seafloor geomorphic features and benthic habitats. *Ocean Coast. Manag*. 167, pp. 176−187

Gammelsrød, T. (1992). Variation in shrimp abundance on the Sofala Bank, Mozambique, and its relation to the Zambezi river runoff. *Estuar. Coast. Shelf Sci*. 35, pp. 91−103

German, C.R., Casciotti, K.A., Dutay, J.C., Heimbürger, L.E., Jenkins, W.J., Mills, R.A., Obata, H., Schlitzer, R., Tagliabue, A., Turner, D.R. and Whitby, H. (2016). Hydrothermal impacts on trace element and isotope ocean biogeochemistry. *Phil. Trans. R. Soc. A*. 374(2081), p. 20160035

Gjerde, K., Boteler, B., Durussel, C., Rochette, J., Unger, S. and Wright, G. (2018). Conservation and sustainable use of marine biodiversity in Areas Beyond National Jurisdiction: Options for underpinning a strong global BBNJ agreement through regional and sectoral governance. STRONG High Seas Project. 24 p. https://doi.org/10.2312/iass.2018.015

Gjerde, K.M. (2008). *Regulatory and Governance Gaps in the International Regime for the Conservation and Sustainable Use of Marine Biodiversity in Areas beyond National Jurisdiction*. IUCN, Gland, Switzerland. x + 70 pp.

Gjerde, K.M., Currie, D., Wowk, K. and Sack, K. (2013). Ocean in peril: reforming the management of global ocean living resources in areas beyond national jurisdiction. *Mar. Pollut. Bull*. 74(2), pp. 540−551

Gregory, M.R. (2009). Environmental implications of plastic debris in marine settings: entanglement, ingestion, smothering, hangers-on, hitch-hiking and alien invasions. *Philos. Trans. R. Soc. Lond. B. Biol. Sci* 364, pp. 2013−2025

Griffiths, C.L. (2005). Coastal marine biodiversity in East Africa. *Indian J. Mar. Sci*. 34(1), pp. 35−41

Griffiths, C.L., Robinson, T.B, Lange, L. and Mead, A. (2010) Marine Biodiversity in South Africa: An Evaluation of Current States of Knowledge. *PLoS ONE* 5(8), e12008

Guduff, S., Rochette, J., Simard, F., Spadone, A. and Wright, G. (2018). *Laying the foundations for management of a seamount beyond national jurisdiction: A case study of the Walters Shoal in the South West Indian Ocean*. Unpublished report. International Union for Conservation of Nature (IUCN) and Institute for Sustainable Development and International Relations (IDDRI), 40 pp.

Guerreiro, J., Chircop, A., Dzidzornu, D., Grilo, C., Ribeiro, R., van der Elst, R. and Viras, A. (2011). The role of international environmental instruments in enhancing transboundary marine protected areas: An approach in East Africa. *Mar. Policy* 35, pp. 95−104

Halpern, B.S, Kappel, C.V., Selkoe, K.A., Micheli, F., Ebert, C.M., Kontgis, C., Crain, C.M., Martone, R., Shearer, C. and Teck, S.J. (2009). Mapping cumulative human impacts to California Current marine ecosystems. *Conserv. Lett*. 2, pp. 13−148

Halpern, B.S., Frazier, M., Potapenko, J., Casey, K.S., Koenig, K., Longo, C., Lowndes, J.S., Rockwood, R.C., Selig, E.R., Selkoe, K.A. and Walbridge, S. (2015). Spatial and temporal changes in cumulative human impacts on the world's ocean. *Nat. Commun*. 6, pp. 1−7, p. 7615

Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C., Fox, H.E., Fujita, R., Heinemann, D., Lenihan, H.S., Madin, E.M.P., Perry, M.T., Selig, E.R., Spalding, M., Steneck, R. and Watson, R. (2008). A global map of human impact on marine ecosystems. *Science* 319, pp. 948−952

Hassan, D. (2017). Protecting the marine environment from land-based sources of pollution: towards effective international cooperation. Routledge. 254 p.

Hawkins, A.D., Pembroke, A.E. and Popper, A.N. (2015). Information gaps in understanding the effects of noise on fishes and invertebrates. *Rev. Fish Biol. Fish*. 25(1), pp. 39−64

Hoarau, L., Ainley, L., Jean, C. and Ciccione, S. (2014). Ingestion and defecation of marine debris by loggerhead sea turtles, *Caretta caretta*, from by-catches in the South-West Indian Ocean. *Mar. Pollut. Bull*. 84(1−2), pp. 90−96

Hoegh-Guldberg, O.R., Cai, E.S., Poloczanska, P.G., Brewer, S. and Sundby, K. (2014). The ocean. In *Climate Change 2014: Impacts, Adaptation, and Vulnerability, Part B: Regional Aspects* (eds. Barros, V.R., Field, C.B., Dokken, D.J., Mastrandrea, M.D., Mach, K.J., Bilir, T.E., Chatterjee, M., Ebi, K.L., Estrada, Y.O., Genova, R.C., Girma, B. Kissel, E.S., Levy, A.N., MacCracken, S., Mastrandrea, P.R., and White L.L.). Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel of Climate Change. Cambridge University Press, Cambridge, New York

Holness, S., Kirkman, S., Samaai, T., Wolf, T., Sink, K., Majiedt, P., Nsiangango, S., Kainge, P., Kilongo, K., Kathena, J., Harris, L., Lagabrielle, E., Kirchner, C., Chalmers, R. and Lombard, M. (2014). *Spatial Biodiversity Assessment and Spatial Management, including Marine Protected Areas*. Final report for the Benguela Current Commission project BEH, 09–01

Huggett, J. and Kyewalyanga, M. (2017). Ocean Productivity. In *The RV* Dr Fridtjof Nansen *in the Western Indian Ocean: Voyages of marine research and capacity development* (eds. J.C. Groeneveld and K.A. Koranten), pp. 55–80. FAO, Rome, Italy

Humphries, F. and Harden-Davies, H. (2020). Practical policy solutions for the final stage of BBNJ treaty negotiations. *Mar. Policy* 122, p. 10421

Johnson, D., Weaver, P., Gunn, V., Spicer, W., Mahaney, S., Tladi, D., Perez, A.A. and Tawake, A. (2016). *Periodic review of the International Seabed Authority pursuant to UNCLOS article 154: interim report*. Seascape Consultants, Southampton, UK. 74 p.

Johnson, D.E., Froján, C.B., Turner, P.J., Weaver, P., Gunn, V., Dunn, D.C., Halpin, P., Bax, N.J. and Dunstan, P.K. (2018). Reviewing the EBSA process: Improving on success. *Mar. Policy* 88, pp. 75–85

Jones, K.R., Klein, C.J., Halpern, B.S., Venter, O., Grantham, H., Kuempel, C.D., Shumway, N., Friedlander, A.M., Possingham, H.P. and Watson, J.E. (2018). The location and protection status of Earth's diminishing marine wilderness. *Curr. Biol*. 28(15), pp. 2506–2512

Khatiwala, S., Tanhua, T., Mikaloff Fletcher, M., Gerber Doney, S.C., Graven, H.D., Gruber, N., McKinley, G.A., Murata, A., Rios, A.F. and Sabine, C.L. (2013). Global ocean storage of anthropogenic carbon. *Biogeosciences* 10, pp. 2169–2191

Krakstad, J., Munga, C.N. and Stromme, T. (2017). Pelagic resources based on acoustic recordings and trawl sampling. In *The RV* Dr Fridtjof Nansen *in the Western Indian Ocean: Voyages of marine research and capacity development* (eds. J.C. Groeneveld and K.A. Koranteng), pp. 81–99. FAO, Rome, Italy

Kraska, J. (2018). National security considerations for a binding instrument on managing biodiversity beyond national jurisdiction. *AJIL Unbound*, 112, pp. 150–154. https://doi.org/10.1017/aju.2018.43

Kyewalyanga, M. (2015). Phytoplankton primary production. In *The Regional State of the Coast Report: Western Indian Ocean* (ed. J. Paula), pp. 213–230. UNEP and WIOMSA, Nairobi, Kenya

Lambardi, P., Lutjeharms, J.R.E., Mencacci, R., Hays, G.C. and Luschi P. (2008). Influence of ocean currents on long-distance movement of leatherback sea turtles in the Southwest Indian Ocean. *Mar. Ecol. Prog. Ser.* 353, pp. 289–301

Lamberth, S.J., Drapeau, L. and Branch, G.,M.(2009). The effects of altered freshwater inflows on catch rates of non-estuarine-dependent fish in a multispecies nearshore line fishery. *Estuar. Coast. Shelf Sci*. 84, pp. 527–538

Lee, S.K., Park, W., Baringer, M.O., Gordon, A.L., Huber, B. and Liu, Y. (2015). Pacific origin of the abrupt increase in Indian Ocean heat content during the warming hiatus. *Nat. Geosci*. 8(6), p. 445

Levin, L.A., Mengerink, K., Gjerde, K.M., Rowden, A.A., Van Dover, C.L., Clark, M.R., Ramirez-Llodra, E., Currie, B., Smith, C.R., Sato, K.N. and Gallo, N. (2016). Defining "serious harm" to the marine environment in the context of deep-seabed mining. *Mar. Policy* 74, pp. 245–259

Longhurst, A. (2007). Ecological Geography of the Sea. Academic Press, 560 p.

Mahongo, S. (2015). Sea/Air Interaction, Chapter 17. In *The Regional State of the Coast Report: Western Indian Ocean* (ed. J. Paula), pp. 198-210. UNEP and WIOMSA, Nairobi, Kenya

McKinley, G.A., Pilcher, D.J., Fay, A.R., Lindsay, K., Long, M.C. and Lovenduski, N.S. (2016). Timescales for detection of trends in the ocean carbon sink. *Nature* 530(7591), p. 469

Momanyi, A. (2015). Chapter 33: Governance: Legal and institutional frameworks. In *The Regional State of the Coast Report: Western Indian Ocean* (ed. J. Paula), pp. 445–458. UNEP and WIOMSA, Nairobi, Kenya

Morgera, E., Appleton, A., Kantai, T. and Tsioumanis, A. (2017). Summary of the Fourth Session of the Preparatory Committee on Marine Biodiversity beyond areas of national jurisdiction: 10-21 July 2017. *Earth Negot. Bull*. 25, pp. 1–22

Nagelkerken, I. and Connell, S.D. (2015). Global alteration of ocean ecosystem functioning due to increasing human CO_2 emissions. *Proc. Natl. Acad. Sci*. 112(43), pp. 13272–13277

Obura, D. (2015a). Deep sea and offshore/pelagic habitats. In *The Regional State of the Coast Report: Western Indian Ocean* (ed. J. Paula), pp. 115–126. UNEP and WIOMSA, Nairobi, Kenya

Obura, D. (2015b). Coral and biogenic reef habitats. In *The Regional State of the Coast Report: Western Indian Ocean* (ed. J. Paula), pp. 71–82. UNEP and WIOMSA, Nairobi, Kenya

Obura, D., Burgener, V., Owen, S. and Gonzales, A. (2017). *Reviving the Western Indian Ocean Economy: Actions for a Sustainable Future*. WWF International, Gland, Switzerland. 64 p.

Ockhuis, S., Huggett, J., A., Gouws, G. and Sparks, C. (2017). The 'suitcase hypothesis': Can entrainment of meroplankton by eddies provide a pathway for gene flow between Madagascar and KwaZulu-Natal, South Africa? *Afr. J. Mar. Sci*. 39(4), pp. 435−451

Petersen, S., Krätschell, A., Augustin, N., Jamieson, J., Hein, J. and Hannington, M.D. (2016): News from the seabed – Geological characteristics and resource potential of deep-sea mineral resources. *Mar. Policy* 70, pp. 175−187

Pillai, N.G. and Satheeshkumar, P. (2012). Biology, fishery, conservation and management of Indian Ocean tuna fisheries. *Ocean Sci. J*. 47(4), pp. 411−433

Plot, V. and Georges, J.Y. (2010). Plastic debris in a nesting leatherback turtle in French Guiana. *Chelonian Conserv. Biol*. 9(2), pp. 267−270

Richmond, M.D. (1997). A *guide to the seashores of Eastern Africa and the Western Indian Ocean islands*. 1st edition. SIDA/Department for Research Cooperation, SAREC. 448 pp.

Richmond, M.D. (2015). Oil, gas and renewable energy. In *The Regional State of the Coast Report: Western Indian Ocean* (ed. Paula, J.). UNEP and WIOMSA, Nairobi, Kenya. Western Indian Ocean. pp. 342−359

Roberson, L.A., Lagabrielle, E., Lombard, A.T., Sink, K., Livingstone, T., Grantham, H. and Harris, J.K. (2017). Pelagic bioregionalisation using open-access data for better planning of marine protected area network. *Ocean Coast. Manag*. 148, pp. 214−230

Roberts, M.J. (2015). The Western Indian Ocean Upwelling Research Initiative (WIOURI): A Flagship IIOE2 Project. *CLIVAR Exchanges* 68, 19(3), pp. 26−30

Robinson, J., Guillotreau, P., Jiménez-Toribio, R., Lantz, F., Nadzon, L., Dorizo, J., Gerry, C. and Marsac, F. (2010). Impacts of climate variability on the tuna economy of Seychelles. *Clim. Res*. 43(3), pp. 149−162

Rochette, J., Unger, S., Herr, D., Johnson, D., Nakamura, T., Packeiser, T., Proelss, A., Visbeck, M., Wright, A. and Cebrian, D. (2014). The regional approach to the conservation and sustainable use of marine biodiversity in areas beyond national jurisdiction. *Mar. Policy* 49, pp. 109−117

Rona, P.A. (2008). The changing vision of marine minerals. *Ore Geol*. Rev. 33(3−4), pp. 618−666

Roxy, M.K., Ritika, K., Terray, P. and Masson, S. (2014). The curious case of Indian Ocean warming. *J. Clim*. 27(22), pp. 8501−8509

SANBI and UNEP-WCMC (2016). Mapping biodiversity priorities: A practical, science-based approach to national biodiversity assessment and prioritisation to inform strategy and action planning. UNEP-WCMC, Cambridge, UK

SCBD (2013). *Report of the Southern Indian regional workshop to facilitate the description of ecologically or biologically*. UNEP/CBD/RW/EBSA/SIO/1/4, 314 pp. Secretariat of the Convention on Biological Diversity

SCBD (2014). *Report of the South-eastern Atlantic regional workshop to facilitate the description of ecologically or biologically significant marine areas*. UNEP/CBD/RW/EBSA/SEA/1/4, 327 pp. Secretariat of the Convention on Biological Diversity

Scharler, U.M., van Ballegooyen, R.C. and Ayers, M.J. (2016). A system-level modelling perspective of the KwaZulu-Natal Bight ecosystem, eastern South Africa. In *Ecosystem processes in the KwaZulu-Natal Bight* (eds. Roberts, M.J., Fennessy, S.T., Barlow, R.G.). Afr. J. Mar. Sci. 38(Supplement), pp. S205−S216

Shotton, R. (2018). Considerations on the Purpose of Benthic Protected Areas in the Southern Indian Ocean Fisheries Agreement Area. Southern Indian Ocean Deepsea Fishers Association. Working Paper. SC-03-INFO-04. Presented at the 3rd Meeting of the Southern Indian Ocean Fisheries Agreement (SIOFA) Scientific Committee 20−24 March 2017. Saint Denis, La Reunion

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. and Wolf, T. (2012). *National Biodiversity Assessment 2011: Technical Report*. Volume 4: Marine and Coastal Component. South African National Biodiversity Institute, Pretoria. 325 pp.

Sink, K.J., van der Bank, M.G., Majiedt, P.A., Harris, L.R., Atkinson, L.J., Kirkman, S.P., and Karenyi, N. (eds.) (2019). *South African National Biodiversity Assessment 2018 Technical Report* Volume 4: Marine Realm. South African National Biodiversity Institute, Pretoria. http://hdl.handle.net/20.500.12143/6372

Smith, C.R., Levin, L.A., Koslow, A., Tyler, P.A. and Glover, A.G. (2008). The near future of the deep seafloor ecosystems. In *Aquatic ecosystems: trends and global prospects* (ed. N. Polunin). Cambridge University Press, pp. 334−349

Sorby, S. (2018). From universal to local law: prospects for the protection of whales in the western Indian Ocean through the Whale Route project. *West. Indian Ocean J. Mar. Sci.* 1(2018), pp. 75−83

Spalding, M.D., Fox, H.E., Allen, G.R., Davidson, N., Ferdaña, Z,A., Finlayson, M., Halpern, B.S., Jorge, M.A., Lombana, A. Lourie, S.A., Martin, K.D., McManus, E. Molnar, J., Recchia, C.A. and Robertson, J. (2007). Marine Ecoregions of the World: a bioregionalization of coast and shelf areas. *BioScience* 57, pp. 573−583

Ternon, J.F., Bach, P. Barlow, R. Huggett, J., Jaquemet, S., Marsac, F., Ménard, F., Penven, P., Potier, M. and Roberts, M.J. (2014). The Mozambique Channel: from physics to upper trophic levels. *Deep Sea Res*. 100, pp. 1−10

Tournadre, J. (2014). Anthropogenic pressure on the open ocean: The growth of ship traffic revealed by altimeter data analysis. *Geophys. Res. Lett*. 41, pp. 7924−7932

U.S. Geological Survey (2012). Assessment of undiscovered oil and gas resources of four East Africa geologic provinces. Fact Sheet 2012–3039. U.S. Department of the Interior. 4 pp.

UNEP/Nairobi Convention Secretariat, CSIR and WIOMSA (2009). *Regional synthesis report on the status of pollution in the western Indian Ocean region*. UNEP, Nairobi, Kenya

UNEP-WCMC (2017). Governance of areas beyond national jurisdiction for biodiversity conservation and sustainable use: Institutional arrangements and cross-sectoral cooperation in the Western Indian Ocean and the South East Pacific. UN Environment World Conservation Monitoring Centre, Cambridge, UK. 120 pp.

UNEP-WCMC (2018). Regional cross-sectoral planning in Areas Beyond National Jurisdiction. UN Environment World Conservation Monitoring Centre. 10 pp.

UNESCO (2009). Global Open Oceans and Deep Seabed (GOODS) – biogeographic classification. IOC Technical Series, 84. Paris: UNESCO-IOC. United Nations Educational, Scientific and Cultural Organization. 332 pp.

UNEP-Nairobi Convention, GEF and WIOMSA (2021). Western Indian Ocean Marine Protected Areas Outlook: Towards achievement of the Global Biodiversity Framework Targets. UNEP and WIOMSA, Nairobi, Kenya, 298 pp.

van der Elst, R.P. and Everett, B.I. (eds.) (2015). Offshore fisheries of the Southwest Indian Ocean: their status and the impact on vulnerable species. *Spec. publ. – Oceanogr. Res. Inst.* 10, pp. 448

van der Elst, R.P., Groeneveld, J.C., Baloi, A.P., Marsac, F., Katonda, K.I., Ruwa, R. and Lane, W.L. (2009). Nine nations, one ocean: A benchmark appraisal of the South Western Indian Ocean Fisheries Project (2008–2012). *Ocean Coast. Manag*. 52, pp. 258–267

Woodall, L.C., Robinson, L.F., Rogers, A.D., Narayanaswamy, B.E. and Paterson, G.L. (2015). Deep-sea litter: a comparison of seamounts, banks and a ridge in the Atlantic and Indian Oceans reveals both environmental and anthropogenic factors impact accumulation and composition. *Front. Mar. Sci*. 2, p. 3

Wright, G. and Rochette, J. (2017). Regional Management of Areas beyond National Jurisdiction in the Western Indian Ocean: State of Play and Possible Ways Forward. *Int. J. Mar. Coast. Law* 32(4), pp. 765–796

WWF and CORDIO (2018). White Paper for the Science-Policy Workshop of the 9th Conference of Parties to the Nairobi Convention, 9–11 July 2018. 4 pp.

Wynberg, R. (2015). Marine genetic resources and bioprospecting in the Western Indian Ocean. In *The Regional State of the Coast Report: Western Indian Ocean* (ed. J. Paula), pp. 407–416. UNEP and WIOMSA, Nairobi, Kenya

CRITICAL HABITATS

THREATENED SPECIES

Nyawira Muthiga, Remy Oddenyo, Rhett Bennett and Dave van Beuningen

Southern Right Whale *Eubalaena australis* **mother and calf.** © Lewis Burnett / Ocean Image Bank

INTRODUCTION

Nature provides invaluable benefits including biodiversity and ecosystem services that are of enormous value to humanity (Cardinale et al., 2012). However, globally, anthropogenic activities have caused and continue to cause loss of biodiversity. The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES, 2019) reported that "nature is declining globally at rates unprecedented in human history", extinction rates are estimated to be two to three orders of magnitude higher than background levels (Pimm et al., 2014). Other global assessments of hard corals, mangroves, seagrasses and sharks and rays (Carpenter et al., 2008; Polidoro et al., 2009; Polidoro et al., 2010; Short et al., 2011; Dulvy et al., 2014) have also sounded the alarm on the rapid loss of species. Biodiversity loss is particularly a challenge for tropical countries including those in the Western Indian Ocean (WIO) that have highly biodiverse ecosystems and where coastal people are highly dependent on nature for food and livelihoods. Species losses could weaken the resilience of ecosystems and reduce the ecosystem services they provide to the expanding human population in the WIO.

There are approximately 11 000 to 20 000 marine species in the WIO (Richmond, 2015) and most are inadequately documented. The identification of species that are at risk of extinction in highly diverse ecosystems is a challenge due to many factors including, the large number of species, their complex life histories, enormous geographical ranges, lack of taxonomic expertise and inadequate monitoring of populations. The International Union for Conservation of Nature (IUCN) Red List is the global standard for determining conservation status of species through a set of criteria that categorize the level of extinction risk of a species (Mace et al., 2008). This information can be used to assess threats and conservation status of single species or for multi-taxa analysis at the national, subregional and regional levels (Carpenter et al., 2008). This chapter therefore focuses on the threatened species, ie species considered to be facing a high to extremely high risk of extinction in the wild categorized as Critically Endangered (CR), Endangered (EN), and Vulnerable (VU), of nine taxa groups in the WIO based on the IUCN Red List and the changes that have occurred in the threat level since the last Regional State of the Coast Report (UNEP-Nairobi Convention and WIOMSA, 2015); hereafter referred to as RSCR-2015.

Over the last seven years, the number of marine species listed in the IUCN Red list that occur in the WIO increased from 161 to 231 (Table 1). Of these, 138 species showed no change, 14 species increased and nine species decreased in their threat status since the RSCR-2015 and 17 species were absent from the RSCR-2015 compilation (Table 2). Reasons for the increase mainly

Table 1: The number of threatened species by taxonomic grouping, their Red List categories and conservation measures in the Western Indian Ocean region.

	IUCN RED LIST CATEGORY (SPECIES COUNTS)				CONSERVATION MEASURES SPECIES COUNTS)		
TAXA	CR	EN	VU	TOTAL THREATENED	CMS	CITES	IOTC
Seagrasses	0	0	1	1	NA	NA	NA
Corals	0	8	77	85	NA	85	NA
Sea cucumbers	0	4	6	10	NA	2	NA
Gastropods	0	3	3	6	NA	NA	NA
Bony Fish	5	9	23	37	NA	NA	2
Batoids	9	12	14	35	10	14	7
Sharks	5	18	30	53	15	13	5
Turtles	2	1	2	5	5	5	5
Mammals	0	3	3	6	4	4	6
Total	21	58	159	238	34	123	25

CR=Critically Endangered, EN=Endangered, VU=Vulnerable.

Sources: IUCN Red List Version 2021-1. https://www.iucnredlist.org; CMS https://www.cms.int/en/species/appendix-i-ii-cms; IOTC https://iotc.org/cmms

Table 1: Changes in IUCN Red List threat status by taxa since the Regional State of the Coast Report, 2015 (RSCR-2015). Comparisons made with IUCN Red List of Threatened Species Version 2021-1 https://www.iucnredlist.org

MAJOR TAXA	LESS THREAT	NO CHANGE	VU	NOT INCLUDED IN RSCR-2015 *
Seagrasses	–	1	–	–
Hard corals	–	84	–	1
Sea cucumbers	–	10	–	–
Gastropods	–	2	–	4
Bony fish	3	9	–	28
Batoids	2	6	4	25
Sharks	1	9	14	29
Turtles	1	4	–	–
Mammals	1	4	–	1
Total	8	129	18	88

* Species included in this column were categorized as threatened in the IUCN Red List Version 2021-1 but not included in the RSCR-2015 evaluation.

stem from species more recently added to the endangered status as well as the slight increase in the area covered, with the current assessment including colder waters around South Africa that were not considered in Richmond (2015). Sharks and batoids showed the greatest change with 14 species showing an increase and nine species (two sharks, two rays, three bony fish, one turtle and one mammal) showing a decrease in their threat status. The following sections summarize the importance of the nine taxonomic groups, their taxonomy, distribution, threats and conservation measures. The changes in the extinction risk (IUCN, 2021) and changes in other global protection measures including the Convention on Migratory Species (CMS), Convention on International Trade in Endangered Species (CITES) are also discussed. Attention is also given to species reported from studies in the WIO as highly threatened or near extinct in the WIO but not globally. The individual taxa or taxa group sections end with recommendations, management and policy implications.

SEAGRASSES

Background

Seagrasses are marine flowering plants that are widely distributed throughout the WIO. Most seagrass habitats are composed of mono-or multi-species meadows that are highly productive and provide many ecosystem services that are important for the livelihoods, food security and well-being of coastal communities (Gullström et al., 2002; Nordlund et al., 2018; Unsworth et al., 2019). Seagrasses create habitats that harbor thousands of different species of organisms (Spalding et al., 2003) including invertebrates, fishes, sharks and rays, and endangered species such as turtles (*Chelonia mydas*) and dugong (*Dugong dugon*), many of which are also integral to the health of seagrass beds. Numerous other organisms use seagrass beds as nursery and feeding grounds and during different phases of their life cycle. Seagrasses also recycle nutrients from coastal runoff and by stabilizing sediments improve water quality and provide coastal protection. In the WIO, seagrass beds are often connected to mangrove ecosystems and together disproportionately sequester more "blue carbon" than other ecosystems (Mcleod et al., 2011). Seagrasses are also often connected to coral reefs and mangroves through nutrient and larval flows and species move between these ecosystems. Coastal communities in the WIO use seagrass beds primarily for fishing and gleaning, for mariculture of seaweeds and sea cucumbers, as traditional medicines and fertilizer and other ecosystem services including spiritual and religious (de la Torre-Castro and Rönnbäck, 2004; Muthiga and Conand, 2014).

Key species detailing taxonomy and threat levels

The diversity of seagrasses is relatively low compared to other marine plants such as macroalgae, represented by nine genera and 13 species in the families Cymodoceaceae, Hydrocharitaceae and Zosteraceae have been confirmed in the WIO (Ochieng and Erftemeijer, 2003). A low diversity of seagrasses is not uncommon in the tropics, as the countries with the highest diversity globally are those that span tropical and temperate regions (Spalding et al., 2003). Newly described species include *Halophila decipiens* and *H. beccarii* (Waycott et al., 2004; Bandeira, 2011) and *Thalassodendron leptocaule* (Duarte et al., 2012). Some taxonomic uncertainties remain and although morphological studies have suggested that *Halodule wrightii* is likely *H. uninervis* and *H. minor* belongs to the *H. ovalis* complex, however, *H. wrightii* continues to be reported in the literature (Ochieng and Erftemeijer, 2003). Only one species *Zostera capensis* is listed in the IUCN Red List (version 2020-1) as Vulnerable and its threat status has not changed since the RSCR-2015 (Table 1, Table 2). The species is reported in Kenya, Madagascar, Mozambique, South Africa and Republic of Tanzania (Appendix).

Distribution

Seagrasses are widely distributed in shallow intertidal to subtidal sandy and soft bottom substrates, lagoons, coral reefs, mangroves as well as in deeper habitats from southern Somalia to South Africa and the islands of the WIO (Appendix). Seagrasses predominate in different habitats within these areas, for example they occur in fringing and back reef lagoons in Kenya and Tanzania, in sandy and limestone areas in Mozambique, in offshore banks in the Seychelles and within estuaries in South Africa (Green and Short, 2003). Despite their wide distribution, few studies have been undertaken on long-term changes in seagrasses in the WIO. Harcourt et al. (2018) showed a loss of 2.7 Tg (teragram) of carbon from seagrasses since 1986 in Kenya and suggested fishing and poor land-use practices as the major causes of this loss. Cuvillier et al. (2017) showed long-term spatial and temporal variability in Reunion seagrass beds of *Syringodium isoetifolium*. The major causes of these fluctuations included physical factors such as swell events and cyclones and anthropogenic factors including nutrient inputs.

The distribution of the threatened species *Z. capensis* is relatively well documented in South Africa. A recent review (Adams, 2016) reported the species in 62 estuaries from Olifants on the west coast to Kosi Bay on the east coast. In Mozambique *Z. capensis* occurs in monospecies stands in the south and mixed species assemblages in the north (Bandeira and Gell, 2003). The species has also been reported in southern Kenya, north-west Madagascar and Tanzania. A newly described species *Thalassodendron leptocaule* has a very narrow distribution from Richard's Bay in South Africa to Inhambane in Mozambique (Duarte et al., 2012). The highest biomass of seagrasses has been recorded in Inhaca Island (Gullström et al., 2002) which also has the highest diversity of seagrasses recorded in the WIO (Bandeira and Gell, 2003).

Threats

Threats to seagrass habitats are increasing at an estimated 110 km^2/yr between 1980 and 2006 (Waycott et al., 2009). In the WIO, the threats vary in different regions but increasing anthropogenic pressures leading to habitat degradation, pollution and nutrient loading are suggested as the major causes of the declines (Duarte, 2002; Green and Short, 2003; Appendix). Overfishing and over exploitation of invertebrates not only cause physical damage but could disrupt the food web leading to changes in seagrass community structure (Heck and Valentine, 2006; Myers et al., 2007; Eklöf et al., 2008; Bandeira, 2011). Climate change is also projected to impact seagrasses through increased storms and floods, sea level rise, UV radiation, sediment hypoxia and anoxia and changing tidal cycles (Björk et al., 2008). Overgrazing by the sea urchin *Tripneustes gratilla* in Kenya has also been reported (Alcoverro and Mariani, 2002). Together these threats to seagrasses are projected to have severe consequences on biodiversity, the health of adjacent ecosystems and the livelihood and economies of the WIO and globally (Green and Short, 2003; Costanza et al., 2014). The main threat to the threatened species *Z. capensis* is habitat degradation (Appendix). For example, digging for bivalves in Bairros dos Pescadores, Mozambique caused a reduction from 60 per cent to 10 per cent cover. Also, flooding was reported in the east coast estuaries in South Africa and disappearance was recorded in the northern St Lucia estuaries (Bandeira and Gell, 2003). *Zostrea capensis's* threat level VU did not change since RSCR-2015 (Table 1).

Conservation measures

Seagrasses are protected through a number of measures (Appendix) such as MPAs, many of which are not directly targeted for seagrasses but are for spatial and/or threatened species protection. In Reunion, seagrass habitats are managed directly to protect seagrasses under protection level 1 while in South Africa estuaries where *Z. capensis* occurs have formal protection. Seagrasses occur in most of the MPAs in the WIO (Spalding et al., 2003) hence they receive some protection due to restrictions on fishing, anchoring or construction activities within these MPAs. Seagrasses are also cited along with coral reefs and mangrove as justifications for establishing MPAs and are mentioned in most management plans although no specific management measures for seagrasses are prescribed. Outside of the MPAs, other management measures such as fisheries restrictions, bans on destructive forms of fishing and dredging also confer some level of protection. However, because these fisheries measures are often poorly applied in many WIO countries, their effectiveness in reducing threats to seagrasses is likely low.

Recommendations, management and policy implications

Although seagrasses have high productivity and are important for fishing, they have not received as much attention as adjacent ecosystems such as coral reefs and mangroves in the WIO. This has led to a poor understanding of the ecology and the extent and underlying causes of the decline of seagrasses in the WIO. Hence one of the first steps that is needed is a regional initiative to map and evaluate threats and conduct valuation of seagrass habitats across the WIO. Understanding where the WIO seagrass biodiversity hotspots are will allow more targeted action in the region where resources and skills for management are limited. Specific knowledge of the value of seagrasses especially the Blue Carbon value will also assist in raising awareness of decision-makers and communities about the importance of seagrasses and generate interest in Blue Carbon financial instruments for protecting seagrasses.

Since threats to seagrasses are mainly anthropogenic including fishing, sedimentation, coastal development and climate change, an integrated system of management of human activities that affect seagrasses and other adjacent ecosystems is needed (Spalding et al., 2003; Nordlund et al., 2014). The Nairobi Convention initiated a process of national integrated coastal zone management (ICZM) policy development in the WIO and some countries have made progress in implementing ICZM. Other initiatives to reduce the threats to seagrasses include improving management of MPAs since many MPAs in the region encompass extensive seagrass beds. Specific measures targeted at seagrasses such as reduction in types of fishing that cause physical damage and interventions that reduce pollution and sedimentation can be trialled and then incorporated into MPA management plans. Despite the large extent of seagrasses in the WIO, few seagrass-based MPAs or Locally Managed Marine Areas (LMMAs) have been established. Blue Carbon initiatives could meet the dual requirements of conservation of seagrasses and community livelihoods and help sustain such interventions, a major impediment to their long-term sustainability (Mcleod et al., 2011).

Flagship species such as sea turtles and Dugong as well as other endangered species that occur in seagrass habitats such as the seahorses *Hippocampus capensis* and *Clinus spatulatus*, the pipefish, *Syngnathus watermeyeri*, Spotted sea-hare *Aplysia oculifera* and Shaggy sea-hare *Bursatella leachii* can also serve to draw attention for the protection of particular seagrass areas that are important for foraging or breeding of these species.

HARD CORALS

Background

Hard corals are invertebrates that build coral reef ecosystems that harbour the highest concentration of marine biodiversity in the world's oceans. Coral reef ecosystems occur from shallow intertidal hard substrate areas to depths of 40 m across the WIO (Sheppard, 2000). The calcareous framework that corals create provide topographically complex niches for thousands of species including fishes, sharks and rays, molluscs, and echinoderms that are important for ecological function and biodiversity, and are a foundation for livelihoods, food security and well-being of coastal communities (Moberg and Folke, 1999; McClanahan et al., 2000; Whittingham et al., 2003; Hicks, 2011).

Numerous marine species also utilize reefs during different phases of their life cycles including marine mammals and threatened species such as Hawksbill turtles (*Eretmochelys imbricata*). In addition, reefs are often linked to seagrass beds and mangroves and together provide ecosystem services such as coastal protection, nutrient recycling, and creation and provision of habitats that are the foundation for coastal economies including fishing, tourism and coastal development.

Key species detailing taxonomy and threat levels

Reef-building hard corals consist of the scleractinian corals that are widespread and diverse and the hydrozoan corals that are less diverse and more sparsely distributed. There are more than 300 species of hard corals reported in the WIO (Spalding et al., 2001; Schleyer, 2011; Obura, 2012) depending on the methods, sources of information and taxonomic detail of the studies. Obura (2012) compiled species richness information from studies in different countries and geographies and reported a total of 369 identified species and 44 unnamed species, indicating the need for more taxonomic studies especially given the large number of unknown species. The highest species richness of 380 species was reported for Madagascar (Veron and Turak, 2005) and the lowest 91 species was reported in South Africa where reef communities are limited due to suboptimal conditions for coral reef growth (Schleyer et al., 2018).

The Red List reports 398 species for the WIO of which 77 species are Vulnerable and eight species are Endangered (Table 1, Table 2, Appendix). Most of the species listed as vulnerable are members of the Acroporidae (30 species) and Poritidae (10 species) families (Table 3). This coincides with their important role in reef-building and the need for effective protection.

Richmond (2015) in the RSCR-2015 provided a comprehensive list of the threatened hard corals, but omitted *Goniopora burgosi* which is wide ranging, occurring from the Red Sea and across the Indian and Pacific oceans. In the WIO, the species has been reported only from Madagascar, and is listed due to reduction of its habitat and fishing for the aquarium trade (Sheppard et al., 2008).

Table 3: The number of threatened hard coral species and their Red List category in each hard coral family in the Western Indian Ocean region.

CORAL FAMILY	EN	VU	TOTAL
Acroporidae	3	30	33
Agariciidae	–	7	7
Dendrophylliidae	–	4	4
Euphyllidae	–	3	3
Faviidae	1	7	8
Fungiidae	–	2	2
Helioporidae	–	1	1
Meandrinidae	1	–	1
Milleporidae	1	–	1
Mussidae	–	4	4
Oculinidae	–	1	1
Pectiniidae	–	2	2
Pocilloporidae	2	3	5
Poritidae	–	10	10
Siderastreidae	–	3	3
Total	8	77	85

EN=Endangered, VU=Vulnerable.

Sources: I: IUCN Red List Version 2021-1. https://www.iucnredlist.org

Distribution

Corals occur in a wide variety of habitats from shallow intertidal hard substrate areas to deeper areas of up to 40 m depth across the WIO (Sheppard, 2000). Most species occur in fringing reefs along the East African coast, barrier reefs that occur off Tulear in Madagascar and atoll reefs such as the Aldabra atoll in the Seychelles. Coral communities can also be found on non-reefal habitats including limestone ridges and banks and Pleistocene sandstone substrate at the southern limits of coral growth along the east coast of South Africa (Sheppard, 2000; Schleyer et al., 2018).

Information on the broad distribution of coral reefs across the WIO is well documented, but information on the distribution of individual threatened species is scarce. Most of the threatened species of the WIO have broad distributions although some species have narrow ranges including *Pocillopora fungiformis*, *Stylophora madagascarensis* and *Millepora tuberosa* that occur in Madagascar and *Acropora rudis*, *Anacropora spinosa*, *Ctenella chagius* and *Parasimplastrea sheppardi* occur mostly in the South Western Indian Ocean (SWIO) (Appendix).

Threats

Hard corals are some of the most threatened species worldwide, they have been reported to have a higher extinction risk than terrestrial species and the rate of extinction has increased dramatically over time (Carpenter et al., 2008). The main drivers of hard coral decline (Appendix) include physical damage from fishing, changes in ecological function due to overexploitation of key species, climate change impacts including bleaching events and ocean acidification. While bleaching kills corals and also makes them susceptible to diseases that have also been increasing over time, ocean acidification inhibits the coral from building skeletons that are the main framework of reefs (Gattuso et al., 1999; Celliers and Schleyer, 2002; Hoegh-Gulberg, 2011).

Local stressors such as fishing, increased sedimentation, nutrients and eutrophication due to coastal development and poor watershed management and agricultural activities are an increasing problem for corals (Burke et al., 2011). Diseases are not as common in the WIO as in the Caribbean, nonetheless the reported incidences show severe local level impacts including total die-off (McClanahan et al., 2004). Black band and yellowing disease (Jordan and Samways, 2001) and six other diseases have been reported in the SWIO (Séré et al., 2013). Increasing incidences of infestations of the coral eating starfish *Acanthaster plancii* (crown of thorns) have also been reported in the region (Celliers and Schleyer, 2006). Collection for the aquarium trade and for ornamental purposes also pose a threat to some species such as *G. burgosi* and the Blue coral *Heliopora coerulea* that is listed as Vulnerable in the Red List and CITES Appendix II. Species that have limited geographic and depth ranges which in the WIO mainly occur in the SWIO islands such as *Ctenella chagius*, *Parasimplastrea sheppardi*, *Poccillopora fungiformis* and *Stylophora madagascarensis* are particularly threatened and are listed as Endangered (IUCN Red List Version 2021-1).

Conservation measures

Several management measures are utilized that can protect hard corals in the WIO including area management (MPAs and community managed areas), fisheries management such as requirements for licensing numbers of users and therefore controlling effort; controls on harvesting keystone species; gear restrictions, and regulation of collection and trade of ornamental species. In addition, bans and restrictions on destructive gears, such as dynamite or beach seines, that cause physical damage to corals are employed. These however are poorly applied or not enforced in many countries hence the level of protection is low.

All 85 hard coral species listed in the IUCN Red List are also listed under Appendix II of the Convention on International Trade in Endangered Species (CITES). Most other species mainly receive protection through spatial management since most MPAs in the WIO encompass coral reefs (Appendix). Regulations that minimize the effects of coastal development, pollution and sedimentation that are in place in some countries although not specifically targeted at corals can mitigate impacts of these threats. Few countries list hard coral species in their conservation and management legislation except South Africa that includes all hard coral species in the legislation. Some countries have national action plans like Kenya's Coral Reef, Seagrass and Mangrove Action Plan. Regional conservation action is also coordinated through the Nairobi Convention Regional Coral Reef Task Force (CRTF).

Recommendations, management and policy implications

Hard corals are considered to be the taxa with the greatest and increasing risk of extinction. There is therefore an urgent need to prioritize efforts to protect hard corals.

There are many actions that are currently being undertaken including regional and national strategies for climate change, coral reef action plans, ICZM and marine spatial planning as well as establishment of MPAs and other area-based entities such as co-managed and community managed areas, gear management and other fisheries interventions. A regional evaluation of the effectiveness of these measures especially of MPAs, one of the main management interventions widely utilized in the WIO, is urgently needed as well as strengthening regional and national actions through the Nairobi CRTF and national coral reef committees. Although coral reefs are one of the most studied ecosystems in the WIO, efforts to protect individual species are limited by lack of appropriate information on population abundances, ecology and biogeography of threatened hard coral species.

Countries of the WIO that have ratified the Convention on Biological Diversity agreed to Aichi target 11 protecting 10 per cent of their marine waters by 2020, yet few have met this target (Wells et al., 2007; Chadwick et al., 2021). The CBD post 2020 framework that includes a target to increase coverage of MPAs to 30 per cent, ie "30 x 30" (UNEP-WCMC et al., 2021) is generating renewed interest. Establishing MPAs in wilderness areas (McClanahan, 2020) and climate refugia, many of which are transboundary areas, would meet several needs including increased area coverage, conservation of large areas, protection of

A shoal of Endangered spinetail devil rays, also known as giant devil rays (*Mobula mobular*) off North West Madagascar.
© Stella Diamant

migratory routes, and climate mitigation amongst others. Area management can also be expanded by increasing the coverage of co-managed and community managed areas or "Other effective area-based conservation measures" (OECMs), that can be included as part of countries marine targets and are crucial for ensuring local community input in marine conservation (IUCN-WCPA Task Force on OECMs, 2019). Studies have identified areas that are potential climate refugia including reefs in the border regions of Kenya/Tanzania, Tanzania/Mozambique, and north-west Madagascar (eg McClanahan and Muthiga, 2016; Ateweberhan et al., 2018b) and these need protection. Green listing of protected areas could encourage a higher level of protection of coral reefs in MPAs (Wells et al., 2016). Taking a watershed management approach and ICZM will also reduce the impacts of land-based sources of pollution that will likely increase with climate change and coastal development.

SEA CUCUMBERS

Background

Sea cucumbers are widely distributed and ecologically important members of marine benthic communities. Their ecological roles include deposit feeding, which reduces organic load and redistributes sediments; excretion of inorganic nitrogen and phosphorous, that enhances productivity in coral reef and other oligotrophic ecosystems; their feeding and excretion increases seawater alkalinity potentially buffering ocean acidification; they host more than 200 species in commensal, symbiotic and parasitic relationships; and, are prey to numerous marine species (Wolkenhauer et al., 2010; MacTavish et al., 2012; Purcell et al., 2016). Sea cucumbers are commercially exploited and provide a livelihood for millions of small-scale fishers worldwide including in the WIO, and nutrition for more than 1 billion consumers (Purcell et al., 2016). Sea cucumbers have traditionally been used as medicines in Asia and are under investigation for their anti-inflammatory and anti-tumor properties (Fan, 1979; Alfonso et al., 2007; Janakiram et al., 2015).

Key species detailing taxonomy and threat levels

Sea cucumbers are in a large group of marine species, the echinoderms, in the family Holothuroidea and most species occur in the order Aspidochirotida that is characterized by a relatively tough body wall and oral tentacles. They occur in a range of marine ecosystems, including coral reefs, seagrass beds, mangroves and estuaries, and in a range of habitats such as exposed reef flats, lagoons and slopes, shallow hard substrate banks and deep soft bottom areas. Rowe and Richmond (2011) reported 140 species across the WIO while Muthiga and Conand (2014) reported 44 species for Kenya, 37 species for La Reunion, 125 species for Madagascar, 125 species for the Seychelles and 26 species for Tanzania.

New species include *Holothuria arenacava* sp. nov. found only in Kenya to date (Samyn et al., 2001), *Bohadschia atra* reported in several countries in the region (Massin et al., 1999), *Labidodemas quadripartitum* sp. nov. reported in South Africa (Massin et al., 2004) and *Actinopyga capillata* that was first reported in La Reunion and Rodrigues (Rowe and Massin, 2006) and in Madagascar and Mayotte (Ducarme, 2016). There is still some taxonomic uncertainty for some species such as *Holoturia (Microthele)* sp. Pentard (Aumeeruddy and Conand, 2008) that is an important fishery in the Seychelles and queries continue about the large number of species recorded by Cherbonnier (1988) in Madagascar (Thandar and Samyn, 2004). However, taxonomic capacity for sea cucumber identification is very low in the WIO.

A total of ten species are threatened (Table 1, Table 2, Appendix), four are Endangered *(H. scabra*, *H. nobilis*, *H. lessoni* and *Thelenota ananas*, and six are Vulnerable (*Actinopyga echinites*, *A. mauritiana*, *A. miliaris*, *H. arenacava H. fuscogilva*, *Stichopus herrmanni*). Because of the pressure of the international trade in sea cucumbers, discussions also progressed to list some species on CITES Appendix II. Three species were evaluated, of these, *H. fuscogilva* and *H. nobilis* that occur in the region were proposed for appendix listing with entry into effect in August 2020 (Source CITES Appendices I, II, and III valid for 26 November 2019). *Holothuria arenacava* is currently only reported in Kenya and is listed as Vulnerable under Criterion D2 of the Red List because of its limited range.

Threats

Sea cucumbers are threatened mainly by overfishing, destructive fishing practices, habitat destruction (Appendix), and global warming (Toral-Granda et al., 2008). Because of the ease of capture, processing, storage and trade driven by a strong Asian market, the fishery is overexploited and often illegal, unregulated and unreported (IUU) in most tropical countries (Toral-Granda et al., 2008; Anderson et al., 2011; Eriksson and Clarke 2015). Many stocks in the WIO have exhibited boom and bust cycles with increasing catches when new species are identified and rapid depletion and collapse due to overfishing (Muthiga and Conand, 2014). As shallow habitats have become depleted, fishing has expanded spatially to deeper waters but also exhibits a roving dynamic, crossing borders and fishing grounds (Eriksson et al., 2012; Muthiga and Conand, 2014). Most stocks are overexploited except in La Reunion where exploitation has never been sanctioned and in the Seychelles that has a relatively well managed fishery.

More than 20 species of sea cucumbers are harvested in the WIO, including the high to medium commercial value species *Holothuria fuscogilva*, *H. nobilis*, *H. fuscopunctata*, *H. scabra*, *H. lessoni*, *Thelenota ananas* and *Actinopyga mauritiana*. Species that are only commercially exploited in one country include *H. (Microthele)* sp. Pentard in the Seychelles, and *H. notabilis* and *Stichopus horrens* in Madagascar. The stocks of high to medium value species have been depleted over time in most countries and the fisheries are increasingly based on low value species. A global evaluation of the extinction risk of all 377 species known to be exploited (Purcell et al., 2014) showed that high value increased extinction risk and was exacerbated by accessibility and market familiarity.

The impacts of climate change on sea cucumbers has not received as much attention as overfishing but also has the potential to negatively impact these animals. For example, increases in severe weather patterns such as typhoons, storm surges and coral bleaching events could cause physical disturbances and changes to intertidal zones, increase sedimentation and erosion that could severely impact these organisms. Increasing seawater temperatures, and changes in weather patterns could also cause disruptions in key biological processes such as reproduction and recruitment (Morita et al., 2010; Huo et al., 2019), feeding and burrowing, but also increase disease outbreaks. Sea cucumbers could also mitigate ocean acidification through increasing seawater pH and reducing the impacts of acidification (Schneider et al., 2011).

Conservation measures

A range of management and conservation measures are deployed in the WIO region (Appendix). The primary management measures used in most countries are fisheries regulations requiring the licensing, harvesting, trading or exporting of sea cucumbers and their products. These measures are poorly applied in most countries leading to overexploitation of this fishery. Other measures include gear restrictions, such as bans on the use of scuba (Kenya), size and catch limits (Seychelles) or even total bans on collection and trade (Mainland Tanzania, Mayotte, Comoros), and CITES Appendix II listing. Conservation measures such as MPAs indirectly protect sea cucumbers as shown in Kenya and Zanzibar where there are more and larger sea cucumbers in MPAs compared to adjacent fished waters (Orwa et al., 2009; Eriksson et al., 2010).

Recommendations, management and policy implications

Sea cucumbers are primarily threatened by fishing pressure and countries differ in the size of their stocks, the level of exploitation and the capacity for management, therefore different approaches are needed. For countries with relatively healthy stocks, management can be strengthened by developing management plans, building capacity for monitoring the stocks and trade, and improved control and surveillance. Since in most countries there are large data gaps especially on stocks, socio-economic and ecological information, the collection of this data needs to be institutionalized. Trade data is collected more systematically but it too has challenges due to the lack of taxonomic capacity. This can be addressed through training exercises such as the FAO workshop on ecosystem approaches to management of sea cucumbers (FAO, 2013; Purcell et al., 2012; Eriksson et al., 2015).

Since the fishery is mainly artisanal involving poor communities, who are highly dependent on marine resources, livelihood diversification programs could help reduce fishing pressure and dependence on marine resources. There is a growing interest in livelihoods diversification in coastal communities in the WIO (Rönnbäck et al., 2002; Ireland et al., 2004) and sea cucumber farming of the high value species *H. scabra* is successfully undertaken by communities in Madagascar (Ateweberhan et al., 2014). Mariculture programs of sea cucumbers are under consideration in Kenya, Zanzibar, Mozambique and Mainland Tanzania (Muthiga and Conand, 2014; Ateweberhan et al., 2018a). Reproduction studies have been conducted on three of the four high value species including *H. scabra*, *H. fuscogilva and H. nobilis* (Rasolofonirina, 2005; Muthiga and Kawaka, 2009; Muthiga et al., 2009) that have provided crucial information for mariculture, including reproductive cycles and size at sexual maturity for these species, however, much more research is needed to tailor mariculture to the local conditions (Eriksson et al., 2012). There is also limited stakeholder engagement that could enhance compliance and the quality of voluntary reporting. Engagement is also required at the regional and international levels to improve monitoring of trade that shows a diverse pattern of different routes to the global market, as well as connections to IUU trade (Eriksson et al., 2012; 2015).

GASTROPODS

Background

Gastropods are a group of molluscs that include species of snails and slugs that exhibit great diversity in size and shape and occur in saltwater, freshwater and on land. In the WIO gastropods occur in all the major coastal and marine ecosystems, including coral reefs, mangroves, seagrass beds and estuaries and in different habitats from sandy beaches to intertidal and subtidal to deeper oceans. Marine gastropods play an important role in the food wed for example serving as prey for many species, including fish, rays, dolphins, and wading birds and feeding on many other species including corals that are preyed upon by *Drupella cornus* and *Coralliophila violacea* (McClanahan, 1994) and sea urchins that are eaten by *Cypraecaassis rufa* (McClanahan and Muthiga, 1989). Gastropods are also important herbivores grazing in seagrass beds and mangroves and recycling nutrients in these ecosystems (Fratini et al., 2004) as well as suspension feeders especially in estuaries.

Coastal communities have been fishing and gleaning for gastropods an important source of protein for millennia (Crawford et al., 2010; Nordlund et al., 2010; 2018) and the shells also provide a source of livelihoods in the curio and shell collecting trade (Newton et al., 1993; Wells, 1997; Peters et al., 2013). Although gastropods are ubiquitous and provide important ecosystem functions in the WIO, relatively little information is available on the taxonomy and ecology of these species.

Key species detailing taxonomy and threat levels

Gastropods belong to the class Gastropoda and phylum Mollusca. There are 2500 species of gastropods described in the WIO, comprising more than three quarters of the molluscs in the region (Richmond and Rabesandratana, 2011). This is likely an underestimate as few comprehensive taxonomic and biogeographic studies have been undertaken, with studies limited in geographical range and ecosystems focusing mainly in shallow intertidal, coral reefs and mangroves or individual families of gastropods.

A study in coral reefs reported 135 species of gastropods in 25 families with little endemism or faunal affinities (McClanahan, 1990) and 291 species of molluscs dominated by gastropods and bivalves (Borri et al., 2005) in a mangrove creek in Kenya. In a review of museum collections of opisthobranchs found in the WIO, Yonow (2012) reported 70 species of which ten were new species for the WIO.

The cone shells *C. jeanmartini* and *C. julii* reported in Richmond (2015) are Vulnerable and their status has not changed (Table 1, Table 2, Appendix). Four additional species are absent in Richmond (2015) because they were considered outside of the geographical range and/or were added to the Red List more recently. These include, *Conus immelmani* a Vulnerable species that occurs on the southeast coast of South Africa (southern Kwazulu-Natal and northern Eastern Cape provinces) and whose population has declined due to collection (Peters et al., 2013), and the Endangered species *Chrysomallon squamiferum* that inhabits deep sea vents in the SWIO ridge, along with *Alviniconcha marisindica*, and *Desbruyeresia marisindica* that occur in deep sea vents in Mauritius's EEZ. The latter three species were added to the Red List in 2019. Other mollusc species that are at risk include the giant clams *Tridacna* that are collected for the curio trade and also impacted by habitat degradation and are listed in CITES Appendix II.

Distribution

Gastropods live throughout the world's coastal and marine waters with steep latitudinal diversity gradients from the tropics to temperate regions (Kohn and Perron, 1994). Distribution varies widely with some species occurring across the entire tropical Indo-Pacific but others restricted to a single bay or seamount (Rockel et al., 1995). The depth ranges of individual species varies considerably with some species living on rocky intertidal shores at sea level and some deep-water species occurring at 500 m or more (Rockel et al., 1995).

Gastropods are widely distributed throughout the WIO and occur in all the coastal and marine ecosystems including coral reefs, mangroves and seagrass beds. The distribution patterns of coastal species are influenced by their food preferences and their ability to withstand desiccation. For example, intertidal and mangrove gastropods move up and down the shores depending on the tides.

Most gastropods live in microhabitats that vary by species, some burrow in sand or mud, some attach to intertidal limestone benches with sand or algal turf, subtidal reef platforms within living or dead corals, or boulders with sandy layers (Kohn, 1968). The threatened species have limited ranges, occurring in coral reefs in Mauritius, Reunion and South Africa or deep hydrothermal vents off Mauritius (Appendix). High endemism has been reported for the genus *Nerita* in the SWIO (Postaire et al., 2014) and *Conus*, namely *C. jeanmartini* and *C. julii* that occur in Mauritius and Reunion (Raybaudi-Massilia, 2013).

Threats

The primary threats to gastropods are loss or alteration of habitat from coastal development and climate change, fishing and gleaning for food and collection for the shell trade. Alteration and loss of habitats is mainly caused by fishing, pollution and sedimentation that occurs across the WIO and climate change impacts such as coral bleaching and storm damage. Gastropods are mainly collected for food by women and children gleaning on shallow intertidal areas in most WIO countries (Crawford et al., 2010; Nordlund et al., 2018) and though an importance source of protein, the impacts of the collected species across the region are unknown.

The shells of gastropods have been harvested and traded in the WIO for millennia, yet information that is needed for management of this resource is scarce. Studies on the ecology and distribution of gastropods in Kenya showed low endemism and densities, and high diversity, with MPAs having higher diversities than fished reefs (McClanahan, 2002). In Zanzibar, cowries *Cypraea* were reported to be 18 times less abundant in exploited tourist areas (Newton et al., 1993) and in the Mascarenes two rare *Conus* species are threatened by collectors, namely, *C. immelmani* and *C. julii* (Peters et al., 2013). Rapid coastal development also provides a large market for shells and species that are already facing pressures from pollution may be pushed further towards extinction. Yet, warning indicators such as sudden price inflation on the shell market may not trigger inclusion in the Red List (Peters et al., 2013). In South Africa, Mauritius and Reunion, *C. immelmani* and *C. julii,* have both declined in numbers likely from over-collection, and *C. jeanmartini*, also from Reunion, is impacted by intensive trawling in its deep-water habitat (Peters et al., 2013). Throughout the Indo-Pacific, the giant triton (*Charonia tritonis*), has been extensively fished and in many areas has been extirpated and its status in the WIO is unknown. This shell is a major predator in coral reefs and its removal impacts the ecology of reefs.

Conservation measures

Several protection measures that are currently utilized in the WIO (Appendix) such as area management (MPAs and community managed areas), fisheries management such as requirements for licensing collectors, gear restrictions that limit destructive gears, and numbers of users, all indirectly protect gastropods. In addition, legislation to control shell collection and bans on collection of Red Listed species assist in the conservation of gastropods.

The species of cowries that are included on the Red List are threatened by over collection and are protected through bag limits while the species in the deep vents are threatened by mining. Protection of gastropods however is limited by ineffective implementation of these measure in many WIO countries.

Recommendations, management and policy implications

Gastropods are small and generally inconspicuous species yet have a disproportionate contribution to the productivity of marine and coastal ecosystems. Unfortunately, their sedentary nature and life histories limit their ability to cope with changes due to anthropogenic pressure. In order to improve protection and management of gastropods a better understanding of the distribution, abundance and life histories of the key species is needed. This information can be used to produce a regional status report and map the key species into management units that can be tracked to monitor change over time. This information can then be incorporated into national action plans for example as coral reef, mangrove and seagrass action plans that some countries in the WIO already have in place. National action plans can serve to harmonize the activities of the various stakeholders as they identify joint actions, foster partnerships and raise awareness for conservation. Improving the management measures already in place such as the effectiveness of fisheries and MPA management and working with key stakeholders to incorporate gastropods in awareness programs is also needed. In addition, as gastropods have received little attention, research and training in taxonomy, studies on population trends and the role of keystone species, anthropogenic and climate change impacts, and a valuation of the gastropods as a protein source will provide the knowledge needed to manage this resource in the long term.

BONY FISHES

Background

Bony fishes are a large class of species that include ray-finned and lobe-fined fishes that live throughout the world's oceans. They are adapted to survive in many different habitats from shallow intertidal to pelagic, to the deep sea. Some species are sedentary living in one place for their entire life, others move between coral reefs and mangroves, while others are migratory, travelling long distances to feed and breed including into freshwater bodies. Bony fishes therefore occupy many niches and are integral to the ecological functioning of critical habitats of the WIO such as coral reefs, seagrasses and mangroves.

Bony fishes also provide the main source of protein and micronutrients for many coastal communities and are a vital source of income and other benefit flows for poor communities in the WIO (eg Moberg and Folke, 1999; Hicks, 2011). The bony fishes support artisanal and industrial fisheries that are important for livelihoods and national economies. Despite the ecological and social importance of bony fishes, and although they receive more attention than most species in the region, there is still a scarcity of information that is needed for their conservation and management especially of threatened species.

Key species detailing taxonomy and threat levels

Bony fishes belong to the Osteichthyes that predominantly have bony skeletons. Osteichthyes is divided into the ray-fined fishes Actinopterygii a very diverse and abundant group of fishes, and the lobe-fined fishes the Sarcopterygii. The WIO region has one of the most speciose fauna of marine fishes, estimates range from 1900 and 2200 species of bony fishes (Smith and Heemstra, 1986; Nelson, 2006; Essen and Richmond, 2011) predominately represented by the ray-fined fishes. This number is likely an underestimate as a 20-year collaboration of fish taxonomists estimated 3600 species of coastal of species occurring above 200 m in the WIO (Heemstra and Heemstra 2019). The WIO region not only has approximately 20 per cent of reported marine fish fauna in the world, approximately 1 per cent (36 species) are classified as threatened including iconic and critically endangered species such as the Coelacanth (*Latimeria chalumnae*) and the Bluefin tuna (*Thunnus maccoyii*).

Thirty-seven species of bony fish are currently Red listed for the WIO (Table 1, Table 2). In 2014, during the preparation of the RSCR-2015, there were only 13 listed (Richmond, 2015) with additional species published during 2014, after the preparation of the RSCR-2015 chapter, highlighting the challenge of keeping up with changes in the endangered status of species and especially how such changes impact on the timing of publication of scientific literature. Currently, five species are critically endangered (Table 1, Appendix) including the Dageraad seabream (*Chrysoblephus cristiceps*), the Sev-entyfour seabream (*Polysteganus undulosus*) and the estuarine pipefish (*Syngnathus watermeyeri*). Nine species are Endangered, including the South Africa endemics, the Blaasop beauty (*Chelonodon pleurospilus*), the Red Steenbras (*Petrus rupestris*) and the White Steenbras (*Lithognathus lithognathus*), and the blenny (*Springeratus polyporatus*). While 23 species are Vulnerable (Table 1) including the seahorse (*Hippocampus kuda*) and the Ocean sunfish (*Mola mola*). In addition, the threat level for three species in the Vulnerable category was downgraded for the Giant grouper (*Epinephelus lanceolatus*) to Data Deficient, the Blacksaddled coral grouper (*Plectropomus laevis*) to least concern and the African weakfish (*Atractoscion aequidens*) from Vulnerable to Near Threatened.

Distribution

Bony fishes occur in coastal and marine waters and in many habitats across the WIO, however, most of the threatened species occur in coastal waters which reflects to an extent the research effort that has been undertaken and proximity to human habitation. Many bony fish species in the WIO are distributed throughout the Indo-Pacific although studies specially of coral reefs provide indications of biogeographic and subregional characteristics (Sheppard, 2000; Obura, 2012; McClanahan, 2015). There is also a high level of endemism especially in the SWIO islands and in the Mozambique to South Africa area (McAllister et al., 1994; Elst et al., 2005).

The Red listed threatened bony fishes display wide to very narrow distribution patterns (Table 4, Appendix). The Ocean sunfish and the Bigeye tuna occur throughout the tropical and temperate oceans. Some species are distributed all along the eastern Africa coast from the Red Sea to the entire WIO region such as the grouper *Polysteganus praeorbitalis*, the Sky Emperor (*Lethrinus mahsena*) and the Brown marbled grouper (*Epinephelus fuscoguttatus*). Several threatened species are restricted from Mozambique to South Africa such as the River seabream (*Acanthopagrus vagus*) and seabream (*Polysteganus praeorbitalis*). Further, some threatened species are restricted to the islands including the Blenny (*Mimoblennius lineathorax*) that only occurs in Mauritius and the Squaretail grouper (*Plectropomus areolatus*) reported in the Comoros. The species with the narrowest ranges included the mud blenny *Parablennius lodosus* that is known only from a 20 km^2 area in Delagoa Bay, Mozambique and the Blaasop beauty (*Chelonodon pleurospilus*) that is reported between the mouth of the Xora River to Durban South Africa.

Table 4: The list of threatened bony fishes in the Western Indian Ocean, their Red List category, distribution and threats up to 2021.

SPECIES	RED LIST CATEGORY	DISTRIBUTION	THREATS	RED LIST YEAR PUBLISHED
Chrysoblephus cristiceps	CR	Very rare found in South Africa from Cape Point to Durban in KwaZulu-Natal	e, f, g, h	2014
Polysteganus undulosus	CR	Very rare, occur from Cape Point, Western Cape to the mouth of the Limpopo River in southern Mozambique	e, f, h	2014
Syngnathus watermeyeri	CR	Very rare, Eastern Cape Province South Africa	a, f, h	2017
Chelonodon pleurospilus	EN	Very rare, from Xora mouth to Durban (South Africa)	a, f, h	2014
Petrus rupestris	EN	Very rare, from Table Bay, south-western Cape to St Lucia, KwaZulu-Natal (South Africa)	e, f, g, h	2014
Springeratus polyporatus	EN	Very rare, found in Mauritius and Reunion	a, f, h	2014
Argyrosomus japonicus	EN	Found only in Mozambique in WIO but also in other countries in the IWP	a, e, f	2020
Argyrosomus thorpei	EN	Occurs on the south-east African coast from Port Elizabeth, South Africa to Xai Xai, Mozambique	a, e, f, i	2020
Lethrinus mahsena	EN	Found throughout WIO	e	2019
Lithognathus lithognathus	EN	Endemic found from the mouth of the Orange River to KwaZulu-Natal, South Africa	a, e, f	2014
Upeneus saiab	EN	Very rare, recorded only in Mozambique	a, e,f, h	2020
Acanthopagrus vagus	VU	Rare, endemic to southern Africa, from Knysna, Western Cape to southern Mozambique	a, e, f	2014
Awaous commersoni	VU	Rare, fount in Mauritius and Reunion	a	2020
Polysteganus praeorbitalis	VU	Occurs from Algoa Bay in the Eastern Cape, South Africa; to Beira, Mozambique	e, f, g	2014
Cymatoceps nasutus	VU	Very rare, found from Cape Agulhas, Western Cape to St. Lucia in KwaZulu-Natal	e, f, g, h	2014
Arothron inconditus	VU	Very rare, found in Durban South Africa	a, f, h	2014
Parablennius lodosus	VU	Very rare, known only from Delagoa Bay, Mozambique in an area of less than 20 km²	a, c, f, h	2014
Mimoblennius lineathorax	VU	Very rare, known only from Reunion Island	a, f	2014
Mola mola	VU	Found globally	e, i	2015
Plectropomus areolatus	VU	Found in the IWP	a, e	2018
Epinephelus marginatus	VU	From western South Africa to southern Mozambique, south-eastern Madagascar and Reunion Island	e, g	2018
Epinephelus fuscoguttatus	VU	Found in the IWP excluding South Africa	a, e	2018
Epinephelus polyphekadion	VU	Found in the IP excluding South Africa	a, e	2018
Pomatomus saltatrix	VU	Found in the WIO	e	2015
Oxymonacanthus longirostris	VU	Found in the Indo excluding South Africa	a, e	2018
Hippocampus kuda	VU	Found in Mozambique, South Africa and Tanzania	a, i	2014
Trachurus indicus	VU	Found in Kenya, Mauritius and Somalia	e	2018

CR=Critically Endangered, EN=Endangered, VU=Vulnerable. Indo-West Pacific region (IWP); Indo-Pacific region (IP); Western Indian Ocean (WIO).

Notes on threat codes: a) overall species habitat degradation, used as a proxy for population decline; c) narrow depth range; e) overharvested (for food); f) restricted geographic range; g) low fecundity; h) rare; i) bycatch. Source IUCN Red List version 2021-1, https://www.iucnredlist.org.

Threats

The major threats to bony fishes globally include overfishing and habitat destruction (Worm et al., 2009; Neubauer et al., 2013), these are also the main risks for the threatened species in the WIO (Table 4, Appendix). A third of threatened species are negatively affected by over harvesting and 8 per cent by incidental capture. Reliance on fishing by most coastal communities for food and livelihoods and rapid development of coastal areas will continue to stress the already overexploited fisheries of the WIO (Zeller et al., 2020) and consequently negatively affect threatened species. This is especially a challenge for species inhabiting coastal areas that are targeted disproportionately in the artisanal and commercial fisheries and in critical habitats such as coral reefs, mangroves and seagrasses. Fishing also causes reduction in the biomass of species that are caught through IUU fishing especially the tuna and tuna-like species.

Habitat degradation is the second highest threat affecting 23 per cent of threatened species in the WIO. Degradation can be caused directly by physical damage from destructive gears (dragnets, trawls, dynamite), coastal development activities (ports, tourism, mining), and sedimentation and pollution that reduce the quality of the habitats species depend upon. Climate change can also exacerbate these impacts due to widespread damage from increased frequency and severity of storms. Increased seawater temperatures due to climate change can also destroy habitats for example through bleaching events in coral reefs that kill hard corals and also cause geographical shifts in species ranges (McClanahan et al., 2004; Feary et al., 2014; Pinsky et al., 2020).

Bony fish distribution, larval development and behavior can also be affected by ocean acidification (Nilsson et al., 2016; Poloczanska et al., 2013; Nagelkerken and Munday, 2016) but the long-term effects of this on ecological function are not well studied (Ferrari et al., 2015). All these factors are especially a challenge for species that are rare, that have restricted geographical and depth ranges or that have low fecundity (13 per cent, 20 per cent, 6 per cent of threatened species respectively).

Conservation measures

A suit of measures have been implemented by WIO countries that directly or indirectly protect threatened species (Appendix). These include fisheries management measures (bans on destructive gears, gear management, species harvest restrictions), MPAs and other area-based measures, community managed areas and co-management of fisheries resources. Additional measures to protect individual species that are harvested include size and effort limits and protection of spawning aggregations. As the concepts of ecosystem-based management, ICZM, Marine Spatial Planning (MSP) and fisheries co-management have become more accepted and utilized, there has been a shift away from thinking of conservation and fisheries management as separate tools towards more collaboration across sectors.

Some WIO countries also have national plans of action for some species as well as critical habitat plans such as coral reef and mangrove action plans that indirectly protect threatened species. In addition, species may benefit indirectly from general management measures that are recommended through regional agreements such as the Nairobi Convention, Indian Ocean Tuna Commission (IOTC), South West Indian Ocean Fisheries Commission (SWIOFC) and global agreements (CMS, United Nations Convention on the Law of the Sea-UNCLOS). Specific measures intended to improve fisheries such as the IOTC resolutions requiring tuna statistics for landings of Bigeye and Southern Bluefin tuna also confer some level of protection. However, because these fisheries measures are often poorly applied in many WIO countries, their effectiveness in reducing threats to threatened bony fishes is low.

Recommendations, management and policy implications

In the last few decades with the increasing reliance on marine resources for economic, social and well-being of their people, countries of the WIO have come to the appreciation of the importance of bony fishes that comprise much of their fisheries stocks and biodiversity value. The priority actions required to reverse the trend in the loss of threatened species of bony fish include addressing issues of governance, research and, monitoring, training, conservation and livelihoods detailed below.

Governance needs

Despite relatively robust legal frameworks, management of natural resources remains weak in most WIO countries and overfishing, destructive fishing and habitat degradation continue. Improving governance of fisheries, MPAs and the coastal zones are therefore priorities. This will include building capacity (finances and staff) for ecosystem-based management, surveillance and enforcement, ecological monitoring and collection of species population, harvest and trade trends in order to improve the effectiveness of current management measures. The capacity to reduce fisheries effort, control local and foreign fleets and reduce IUU is also needed. In addition, for species that are highly threatened, there is a need for stricter national level protection and specific species recovery plans. In countries where the co-management model is in place, guidelines and training is required for communities to enhance their effective engagement in fisheries management of their co-management areas. Collaboration across sectors through ICZM can improve management of the coastal zones and mitigate harmful impacts.

Research, monitoring and training needs

There is generally limited capacity in skills for fisheries and ecosystem management in the WIO. Improved stock assessment, fish identification, collection and evaluation of fisheries catch and trade trends skills are needed. Specific information for management of threatened species such as size at sexual maturity, maximum sustainable yield estimates and other metrics needed to set limits and manage fisheries are often lacking. Knowledge on the impacts of local and foreign fleet on fisheries, ecosystems and economies are also lacking in most cases. Although regional courses on stock assessment, fisheries and MPA management have been supported by the Western Indian Ocean Marine Science Association (WIOMSA) and under various projects (SWIOFISH etc), more managers require such training through programs that are designed as learning while doing modules so that the skills and knowledge imparted are put to practical use.

Conservation needs

Bony fishes also occur in MPAs, community managed and co-managed areas hence, improving their management, especially the capacity for monitoring and enforcement is needed. Species recovery plans that include conservation actions and research and monitoring are needed, especially for the critically endangered species. There is a lack of adequate information for assessments for most threatened species, such as distribution and population estimates, and size compositions that can be tracked for assessing declines in area and extent or quality of habitat, sexual maturity and population growth. Lastly, as there are several areas that have been identified as important for biodiversity such as climate refugia, ecologically or biologically significant marine areas (EBSAs), Key Biodiversity Areas (KEBs) and Biosphere reserves, a regional MSP process can assist in prioritization and management of these areas.

Livelihood needs

Many coastal communities in the WIO depend on bony fish for their food and livelihood and overfishing continues to erode the resource base that these communities depend on. Apart from improving the overall management of fisheries to sustain communities in the long term, there is also the need to promote sustainable fishing practices and to develop alternative livelihoods. There are several alternative livelihood projects currently undertaken in the WIO such as seaweed, mud crab, blacklip pearls, milkfish and sea cucumber culture.

A comprehensive review of the social, ecological and economic outcomes of these projects is needed to understand the long-term outcomes, scalability and financial sustainability of these interventions. In addition, there is a need to trial innovative and practical interventions such as retrofitting traps with escape gaps and different types of hooks to reduce bycatch and ecological impacts, cold storage and credit schemes to reduce waste and add value. Lastly, interest in community Blue Carbon projects is increasing and projects such as the Mikoko Pamoja initiative spearheaded by Kenya Marine & Fisheries Research Institute (KMFRI) can provide practical local expertise.

Regional and international collaboration needs

Most countries of the WIO are members of several regional and international agreements. However, participation is often ineffective and resolutions are poorly implemented nationally. In addition, there is often a sectoral division with different ministries involved in different agreements, yet these may overlap institutionally and spatially. A mechanism to coordinate all the resolutions from national, regional and global agreements is needed. There is also a need to develop capacity for management of transboundary areas, refugia areas and other large wilderness areas.

Awareness needs

Anthropogenic impacts can be minimized if stakeholders are aware and engaged in reducing harmful practices. In general, there is low awareness of bony fishes except for iconic species such as the coelacanth in the WIO. Awareness programs that are targeted to address lack of knowledge, experiential exchanges and knowledge fora for specific species and the habitats they inhabit are needed. These programs can be coordinated by the Threatened Species Task Force in collaboration with long-term learning and information exchange programs such the Fishers' Forum in Kenya, MIHARI in Madagascar, the Beach Management Units (BMUs) networks in WIO countries.

Improved linkages and coordination with community-based organizations (CBOs) and non-governmental organizations (NGOs) that conduct regular awareness programs and coordinate citizen monitoring can provide the widespread dissemination and effectiveness of information.

The Endangered whale shark (*Rhincodon typus*), the largest fish in the sea, rises to the surface. © Stella Diamant, Madagascar Whale Shark Project

BATOIDS AND SHARKS

Background

Chondrichthyan species, including sharks, batoids (wedgefishes, sawfishes, guitarfishes, skates, rays) and chimaeras, form important components of marine ecosystems, acting as apex and meso predators, as well as prey for larger species (Prugh et al., 2009). The WIO is considered a global hotspot for chondrichthyan diversity (Dulvy et al., 2014), with 227 species recorded to date (c. 20 per cent of all known chondrichthyan species). This rich diversity includes at least 135 shark species, 83 batoid species and 9 chimaera species (eg Ebert and van Hees, 2015; Fricke et al., 2018). However, despite the diversity of species and global importance of the WIO region for this group of fishes, there is limited biological and ecological information on most chondrichthyan species, and limited information on how they are impacted by fisheries.

Key species detailing taxonomy and threat levels

The WIO region includes several large charismatic species, such as whale sharks (*Rhincodon typus*), great white sharks (*Carcharodon carcharias*), tiger sharks (*Galeocerdo cuvier*), hammerhead sharks (Sphyrnidae), thresher sharks (Alopiidae) and manta and mobula rays (Mobulidae). Historically, the WIO was also home to two critically endangered sawfish species (Pristidae); however, the Green sawfish (*Pristis zijsron*) is thought to have been extirpated completely from the region, and the Largetooth sawfish (*P. pristis*) is now extremely rare in the WIO and possibly locally extinct in some previous range states, such as South Africa and Seychelles (Kyne et al., 2013; Leeney, 2017).

There is a considerable level of endemicity among chondrichthyan species in the WIO, which, along with the high species richness, gives the WIO chondrichthyans a high irreplaceability index (Stein et al., 2018). Approximately 51 WIO chondrichthyan species are endemic to the region, and some to a single WIO country; for example, the Madagascar pygmy skate (*Fenestraja maceachrani*) and Madagascar numbfish (*Narcine insolita*) have only been recorded thus far in Madagascar (Last et al., 2016).

The number of recorded species has increased exponentially over the past few decades (Ebert and van Hees, 2015), through new species descriptions, taxonomic assessments and improved field data collection resulting in new species distribution records. However, there remains taxonomic uncertainty within several orders and families, particularly within the batoids. The whiptail stingrays (Dasyatidae), represented by at least 12 genera and 16 species in the WIO (Ebert and van Hees, 2015; Last et al., 2016), require taxonomic confirmation and verification of geographic distributions for certain genera. Similarly, the Rhinopristiformes (sawfishes, wedgefishes, guitarfishes and giant guitarfishes) require confirmation of their distributions and taxonomy, particularly the guitarfish genera *Acroteriobatus and Rhinobatos*.

The Rajiformes (skates) and Torpediniformes (electric rays) are represented in the WIO by 27 species (13 genera) and nine species (six genera), respectively (Ebert and van Hees, 2015; Last et al., 2016), although more than one quarter and one half, respectively, of the species in these orders remain data deficient on the IUCN Red List of Threatened Species.[1] There is also considerable taxonomic uncertainty within the Squalidae family (dogfish sharks), thought to be represented in the WIO by at least six species.

Distribution

Chondrichthyans occur in the waters of all WIO countries and occupy a diversity of coastal and marine habitats (Appendix). Estuaries, mangroves and atolls provide important nursery habitats for species such as bull sharks (*Carcharhinus leucas*), sawfishes (Pristidae) and many batoid species. Coral reefs are critical habitats for species such as blacktip reef sharks (*Carcharhinus melanopterus*) and whitetip reef sharks (*Triaenodon obesus*), while inshore soft sediment habitats are important for many batoid species, such as whitespotted wedgefish (*Rhynchobatus djiddensis*) and whiptail stingrays (Dasyatidae). Several large charismatic species, such as whale sharks (*R. typus*), occupy the pelagic zone, while many poorly known species, such as lantern sharks (Etmopteridae) and dogfish sharks (Squalidae), occupy deep-water habitats.

The coastal waters of South Africa, Mozambique and Madagascar have the highest chondrichthyan species richness of the WIO countries (Dulvy et al., 2014), with at least 152 species in South Africa (from Port Elizabeth eastwards and northwards to Mozambique), 129 in Mozambique and 114 in Madagascar (Ebert and van Hees, 2015; Fricke et al., 2018; Ebert et al., 2021). Species richness decreases northwards to Kenya and Somalia, and among the WIO island states.

There are 44 shark and 23 batoid species in the WIO that are classified as migratory, or possibly migratory, based on movement behaviour and known or potential movements across jurisdictional boundaries (Fowler, 2014). Migratory behaviour and broad geographic distributions complicate management, as such species are vulnerable to fisheries in the waters of multiple countries (Barkley et al., 2019).

[1] IUCN, 2021, www.iucnredlist.org

Threats

Chondrichthyans are generally slow growing, with late maturity and low reproductive capacity, making them highly susceptible to population disturbances, such as fishing (Worm et al., 2013). In the WIO, there is intense fishing pressure on chondrichthyan species (Appendix), from a range of fishing gears used in domestic artisanal, commercial and industrial fisheries, as well as from foreign fleets through fishing rights agreements, and through illegal, unregulated and unreported (IUU) fishing (Kiszka and van der Elst, 2015).

While sharks have been targeted in several WIO countries for more than a century (Marshall and Barnett, 1997), fishing pressure and demand for chondrichthyan products have grown rapidly over the past few decades. Overfishing (including directed fishing and bycatch) occurs in most sectors and is the primary threat to chondrichthyan species in the WIO.

While chondrichthyans are taken as bycatch in certain WIO fisheries, they constitute important targets in most. There is legal and illegal trade in chondrichthyan products, with a high demand for shark meat for local consumption and export, and as coastal human populations grow this demand for shark meat will also grow. There is also a huge demand for shark and batoid fins for the global shark fin trade, particularly for certain species with high value fins such as the critically endangered wedgefishes (genus *Rhynchobatus*) and scalloped hammerhead sharks (*Sphyrna lewini*) and, more recently, a burgeoning demand for the gill rakers of mobulid rays for the Asian market – a relatively new threat in the WIO. There is also demand for other commercial products, such as shark livers for oil, particularly for pharmaceutical products (Samoilys et al., 2015).

Overexploitation of chondrichthyan species can have direct negative impacts on their populations and indirect impacts through cascading effects on the ecosystems and trophic webs. As millions of people living in coastal communities within the WIO countries are dependent on fishes, chondrichthyans and other marine resources for their income and livelihoods, as well as cultural and traditional uses, sustainable utilization of these resources is paramount, and as much a social issue as it is an ecological issue.

Human populations, and consequently the demand for marine resources (including chondrichthyan products), are increasing throughout the WIO, and there is evidence of human migrations towards and among coastal areas in search of improved food security and livelihoods (Barnes-Mauthe et al., 2013). Consequently, the impacts on chondrichthyan populations are likely increasing. However, there are currently limited data on the catches of chondrichthyans (particularly at species level), most fisheries (particularly the artisanal, small-scale and IUU fisheries) are poorly monitored and total catches remain unknown (Worm et al., 2013). Furthermore, there is limited legislation in most WIO countries for the protection of threatened chondrichthyan species and many of the WIO fisheries that catch chondrichthyans remain poorly regulated.

Habitat degradation is also a major threat to chondrichthyan species in the WIO, with many important or sensitive habitats, such as mangroves and coral reefs that are used by these species for crucial life stages, such as pupping grounds and nurseries, under threat from habitat alteration and coastal development (*inter alia* land reclamation, building of ports and pollution).

Conservation status

Owing to overfishing and other human impacts, the stocks of numerous WIO chondrichthyan species have declined dramatically (Dulvy et al., 2014; Kiszka and van der Elst, 2015), with the WIO now considered a global "darkspot" in terms of the number of imperiled chondrichthyan species (Davidson and Dulvy, 2017). A global assessment of shark abundance on coral reefs (MacNeil et al., 2020) indicated major population declines in many reef-associated shark species, particularly in parts of the WIO.

Following the recent global revision of chondrichthyan IUCN Red List statuses (Dulvy et al., 2021; IUCN, 2021), 90 (40 per cent) of the 227 chondrichthyan species in the WIO are now considered threatened. These include 14 (~6 per cent) critically endangered, 32 (~14 per cent) endangered and 44 (~19 per cent) vulnerable species (Table 1). This represents an increase from 49 threatened species (22 per cent of the WIO chondrichthyan species), over the past five to ten years. The number of critically endangered chondrichthyan species has increased threefold over this period from just four to 14, with nine batoid species (from four families) and five shark species (from four families) now categorized as critically endangered.

In addition, 39 species (17 per cent) are classified as data deficient by the IUCN, ie, there is inadequate information to make a direct or indirect assessment of the species' risk of extinction, and six species (~3 per cent) have not yet been evaluated (IUCN, 2021). Furthermore, the assessments of three (1 per cent) of the 221 species that have been categorized on the IUCN Red List are more than ten years old, rendering them out of date according to the IUCN categories and criteria (IUCN, 2001).

Considering the extensive fishing pressure in the region and that at least one of three species with an outdated Red List assessment was already endangered in 2006, many of these data deficient species may in fact be threatened, while the three species with outdated assessments may meet the criteria of a higher (more threatened) threat category. There is thus a critical need for improved knowledge, corrective management and improved conservation of the chondrichthyan species in the WIO, particularly those that are threatened or likely to become threatened.

The WIO is also home to 27 (57 per cent) of the 47 chondrichthyan species that are listed on Appendix I or Appendix II[2] of CITES – species that are considered to be in need of trade controls due to the impacts that harvesting and international trade have on their populations. The region also has 25 (68 per cent) of the 37 chondrichthyan species listed on Appendices I and II[3] of CMS – migratory species considered to be threatened throughout their ranges or at greater risk due to movement patterns that span multiple countries and jurisdictions. These listings of such species are a good reflection of their poor conservation status, and the high levels of threat that they face.

The chondrichthyan families most at risk in the WIO include Pristidae (sawfishes), Rhinidae (wedgefishes), Myliobatidae (eagle rays), Sphyrnidae (hammerhead sharks), Mobulidae (manta and mobula rays), Lamnidae (great white and mako sharks) and Alopiidae (thresher sharks). The most threatened species are generally either impacted by both inshore (mainly artisanal) and offshore (mainly industrial) fisheries (Dulvy et al., 2014), such as hammerhead sharks, or are targeted for specific body parts, such as the wedge fishes for their highly valued fins and manta and mobula rays for their gill plates. All of these species have low resilience to overexploitation, and should be prioritized for WIO regional conservation efforts.

2. https://www.cites.org/eng/app/appendices.php
3. https://www.cms.int/en/page/appendix-i-ii-cms

Recommendations, management and policy implications

1. A high proportion (40 per cent) of WIO chondrichthyan species are classified as threatened on the IUCN Red List, an increase of 18 per cent (near doubling) in the past five to ten years. There is thus a critical need for a reduction of fishing pressure, fishery-related mortality and incidental bycatch of these species, and for the implementation of appropriate species recovery plans for overexploited species.

2. Despite the high proportion of threatened chondrichthyan species, few are protected under national legislation. There is a need for protective legislation for threatened species, and stricter management, regulations and enforcement.

3. Considering the intense fishing pressure and the high level of dependence of WIO communities on coastal (specifically fishery) resources, there is a need to promote sustainable fishing practices and alternative livelihoods.

4. The WIO countries fall within the geographic coverage of the *Nairobi Convention for Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region*, a Regional Seas Program of the United Nations Environment Program (UNEP). While no chondrichthyans are yet listed under the Nairobi Convention annexes (which list species requiring protection or restricted use), the Convention provides a framework for the management and sustainable utilization of the marine and coastal resources in the WIO, and could provide a suitable legal platform for the protection of threatened chondrichthyans at the regional level.

5. There is poor control of the trade in chondrichthyan products both within and out of the WIO region, and considerable discrepancies in statistics between export volumes (and taxa) from the WIO countries and import volumes into other countries, indicating inaccurate reporting, non-reporting and illegal trade in certain instances. There is thus a need for stricter trade controls and improved monitoring, reporting and enforcement.

6. There is generally poor information on the catches of chondrichthyans in most fisheries in the WIO, which leads to underestimating the true scale of the current impacts of the different sectors on chondrichthyan populations. There is, therefore, a need for the establishment or improvement of programs that monitor chondrichthyan catches and for socio-economic profiling of fisheries that impact on chondrichthyan species, in order to identify i) species under the greatest threat, ii) trends in the status of the resources and resource use, and iii) mechanisms for improving the sustainability of these fisheries.

7. Most WIO countries are signatory to numerous international conventions and agreements, such as CITES, CMS and the IOTC, which list species (including many chondrichthyan species found in the WIO, Appendix) that require protection or limitations on harvesting, more stringent trade controls and/or that are under threat by virtue of their migratory ecology. However, the full benefits of such conventions will only be witnessed with improved and effective enforcement of the associated regulations.

8. There is limited awareness of the poor conservation status of chondrichthyan species in the WIO and generally a poor understanding of their important ecological roles in their relevant ecosystems and trophic webs, and thus a lack of cognizance of the impacts (direct and indirect) of overfishing these species. There is a need to raise awareness of these issues, among fishers, governments and other stakeholders.

TURTLES

Background

Sea turtles are marine reptiles that are widely distributed in tropical and subtropical oceans. They have a complex life history and use different habitats to forage and breed. Sea turtles therefore play an important role in the ecology of marine ecosystems, including nutrient recycling, habitat maintenance and stabilizing marine food webs through their roles as consumers, prey and competitors (Bjorndal, 1985). Sea turtles also provide traditional and cultural services and livelihoods for coastal communities (Troëng and Drews, 2004; Humber et al., 2011. However, because they are slow growing, late maturing and have low reproductive rates, they are biologically more susceptible. In addition, their life history exposes them to many threats from anthropogenic activities that have decimated their populations over centuries resulting in most species being listed as endangered (IUCN, 2021). There have been long-term and intensive studies of sea turtles in the SWIO but there remains major data gaps including reliable population estimates that are crucial for management (Bourjea, 2015).

Key species detailing taxonomy and threat levels

Sea turtles are reptiles that belong to the order Testudines characterized by a soft bony shell. There are only seven species of sea turtles existing worldwide and five of these occur in the WIO namely, the Olive ridley (*Lepidochelys olivacea*), the Green (*Chelonia mydas*), the Hawksbill *Eretmochelys imbricata*, the Leatherback (*Dermochelys coriacea*) and the Loggerhead (*Caretta caretta*) turtles. Of these species, the Green and Hawksbill turtles are the most common in the region (Bourjea, 2015).

All five species are categorized as threatened with the Hawksbill and Leatherback categorized as Critically Endangered, the Green as Endangered, while the Olive ridley and Loggerhead are categorized as Vulnerable (Table 1, Table 2). Global assessments rank *C. caretta*, *D. coriacea* and *L. olivacea* as high-risk species in the WIO with *L. olivacea* and *C. mydas* considered to face the greatest levels of threat from bycatch (Wallace et al., 2013).

Distribution

Sea turtles have a pan-tropical distribution (Nel et al., 2013a). They use a variety of habitats to complete their life cycle, ranging from coastal-terrestrial breeding beaches to neritic and pelagic feeding habitats (Godley et al., 2008), coral reefs and seagrass beds (Appendix). Satellite tracking, tagging and other studies show that migration routes may be oceanic or coastal, characterized by fixed or seasonal movements to local or pelagic foraging grounds.

Green and Hawksbill turtles are the most abundant and widely distributed in the WIO (Bourjea, 2015; Appendix). Green turtles feed predominantly on seagrass beds that extend along coastal east Africa to the western coast of Madagascar (Bjorndal, 1997; Gullström et al., 2002). Gravid green turtle females have a more diverse diet that includes 58 per cent seagrasses, 15 per cent algae and 7 per cent sponges (Stokes et al., 2019). Green turtles also nest throughout the region (Bourjea et al., 2008) and migrate to nesting beaches along coastal east Africa and the northern Mozambique Channel (Dalleau et al., 2019). The WIO region contains some of the world's most important rookeries for Green turtles (Dalleau et al., 2012).

Population genetics studies showed that there are two main genetic stocks of Green turtles, the southern Mozambique Channel and the northern SWIO stocks that can be considered the management units for these populations (Bourjea et al., 2017. A recent study modelling migration patterns of the Green turtle indicate the likelihood of a northern migration corridor between the north of Madagascar and the northern coast of Mozambique (Dalleau et al., 2019). The WIO also has globally important nesting beaches for Hawksbills especially along the coast and islands of the Mozambique Channel with Madagascar and Seychelles leading in number of nests (Mortimer and Donnelly, 2007).

Studies on loggerheads on the other hand suggest a trans-equatorial life cycle utilizing the Indian Ocean basin (Dalleau et al., 2014) and nesting in southern Mozambique, South Africa, mainland Tanzania and Zanzibar. Leatherbacks though sighted in most countries are less common and nest in South Africa. Olive ridleys are reported across the region but are rare and confirmed nests have only been reported in Kenya, Madagascar, South Africa and Tanzania. Little is known on the migration behavior of this species.

Threats

In coastal (neritic) waters sea turtles are exposed to artisanal (eg gillnets) as well as commercial fishing activities. On the high seas, sea turtles interact with industrial fisheries such as longlines or purse seines. Overlap with fishing activities results in drowned sea turtles caught in fishing gear (active or discarded), cuts or other injuries due to boat strikes or pollution (Nel et al., 2013a). The complex life cycle of turtles, incidental and targeted fishing, as well as the large spatial overlap between sea turtles and human activities, has resulted in most sea turtles being listed as Endangered in the IUCN Red List (Appendix).

In the WIO there are three major gear types that catch large numbers of turtles namely gillnets, prawn trawls and longlines (Bourjea et al., 2008; FAO, 2010). Small-scale fisheries also have incidental and targeted catches, with gillnet and hand line gears posing the greatest threat (Temple et al., 2018; Poonian et al., 2008; Kiszka et al., 2010). For instance, 10 000–16 000 Green turtles were captured and sold for consumption in local markets each year in south-west coast of Madagascar (Humber et al., 2011). Targeted and incidental catch has also been reported in Mozambique, Tanzania and Kenya (Okemwa et al., 2004; Moore et al., 2010; Williams et al., 2017).

Coastal development also poses a threat to sea turtles through, alteration of nesting beaches such as beach walls and other barriers, changes in the quality of nesting beaches caused by erosion, plastic, noise and light pollution (Nicholas, 2001). Climate change impacts including increased storms that damage nesting beaches, sea level rise that inundates nesting beaches and increases in nest temperature that control the sex of hatchings will all have a negative effect on turtles (Butler, 2019). Coral bleaching could also result in reduced habitat for turtles that forage in coral reefs (Fuentes et al., 2011).

Conservation measures

Sea turtles have received the most conservation attention of the threatened species in the WIO. There are more than 140 sites focused on turtle conservation activities (Bourjea et al., 2008). Nest monitoring and protection, MPAs, education/awareness interventions occur at 20–25 per cent of these sites; vehicle/access control at ~17 per cent, egg relocation/hatcheries, beach cleanups and predator controls, and other activities ~10 per cent, while building/design/light regulations and fishing gear modifications at 6–8 per cent. This investment in conservation and management activities especially actions targeted at waters off nesting beaches, protection of females when nesting, protection of nests and hatchlings and MPAs has resulted in maintenance of populations and population increases of some species such as the Green turtles in Aldabra atoll (Mortimer, 1985) and Grande Glorieuse and Europa islands (Lauret-Stepler et al., 2007), Hawksbills from Aldabra atoll and Cousin Island Seychelles (Wood, 1986) and Loggerhead and Leatherbacks in South Africa (Nel et al., 2013b). Many of these efforts are primarily supported by national, international and regional NGOs and community based interventions across the WIO (Okemwa et al., 2005).

Efforts to conserve turtles on a large scale benefit from national legislation, policies and action plans in the WIO and turtle species are often specifically mentioned in national species conservation schedules. For example, all five threatened species are listed in the species schedules of the Wildlife Conservation and Management Act of Kenya and Leatherbacks and Hawksbills in the National Environment Management Biodiversity Act of South Africa. Many of these policies and strategies meet the obligations of regional and global conservation actions that most WIO countries are signatory to such as the Indian Ocean South East Asian MOU for the conservation and management of turtles, the IOTC, the CMS, and the Western Indian Ocean Marine Turtle Task Force (WIO-MTTF). However, the level of compliance of the resolutions passed in these regional and global instruments is often low due to inadequate monitoring, lack of capacity and insufficient attention to turtle conservation in some countries.

Recommendations, management and policy implications

The strategies laid out in the "Marine conservation strategy and action plan for the Western Indian Ocean" (IUCN, 1996) are still valid today. These include research and monitoring, integrated management of turtle populations, community participation, building research, conservation and management capacity, public awareness and education programs, regional and international cooperation and sustainable financing. Some of these strategies are better applied than others.

There are some long-term studies for example of Loggerhead and Leatherbacks in South Africa (Nel et al., 2013b) and Green and Hawksbill turtles in the Seychelles (Mortimer, 1985). However, there are still major data gaps on species abundance and nesting data that are crucial for management. Because turtles have a highly mobile and complex life cycles it is difficult to collect reliable population level data for most species (Bourjea, 2015). Satellite tagging could be used to monitor turtle movement patterns and identify possible nesting sites but a structured and scientifically robust system is needed (Bourjea et al., 2008; Webster, 2013). Some genetic studies on stock structures and population dynamics of turtles have been undertaken (Dalleau et al., 2012), but life history characteristic such as hatching success, sex ratios, and natural mortality for the species are needed (Bourjea et al., 2008).

Another area of research that needs attention is experimentation with mitigation measures, including TEDs and circle hooks that are recommended by IOTC and other global conventions. TEDs were first experimented in Kenya (Wamukoya and Salm, 1997), and the work continues

Hawksbill turtle (*Eretmochelys imbricata*) in the Mombasa Marine National Park. © T.R. McClanahan.

through the Bycatch Assessment and Mitigation in the WIO fisheries project (BYCAM) supported by WIOMSA. Information is also needed to better understand the increasing threat of climate change (Butler, 2019).

To address the general lack of information an integrated system that incorporates data from artisanal and industrial fisheries, and that involves local communities, fisheries and conservation agencies and scientists is needed (Bourjea et al., 2008). This can be driven by the national turtle committees that are in place in some countries. The WIO-Marine Turtle Task Force ideally integrates national actions and enhances cross-border collaboration and involvement with the Regional Fisheries Management Organizations. In addition, adherence to national on-board observer programmes which have been implemented in five WIO countries with varying levels of success need to be strengthened (Migraine and Hykle, 2014).

Turtles are popular for community engagement and public awareness campaigns and there are many community-based turtle conservation groups in the region. Some of these programs are an alternative source of livelihoods for local communities either as employees or provide direct revenue from tourism activities. The programs have varying capacities and have not been evaluated for conservation efficacy. There is a need to monitor, strengthen and harmonize these programs to enhance learning and exchange (Migraine and Hykle, 2014).

MARINE MAMMALS

Background

Marine mammals play key roles in marine ecosystem structure and function (Katona and Whitehead, 1988). Because these species are mostly large bodied, they affect prey populations and influence their life history traits and community structure. They also play a role in the storage and redistribution of nutrients as they feed at depth and defecate in the shallower waters and when large species such as Blue whales die, they sink to the ocean bottom removing carbon from circulation hence potentially helping to mitigate the impacts of climate change (Pershing et al., 2010).

Feeding activities by Dugongs and Grey whales have been reported to restructure seagrass meadows and seabeds (Nerini, 1984; Preen, 1995). Marine mammals are also hunted by local communities (Cerchio et al., 2015) and contribute to livelihoods through whale and dolphin watching in the WIO (Berggren et al., 2007; Kiszka et al., 2009). Populations of many marine mammal species have declined worldwide and the conservation of these species faces numerous challenges globally (Kaschner et al., 2012).

Key species detailing taxonomy and threat levels

Marine mammals fall into the orders Cetacea and Sirenia. Cetaceans are composed of two suborders; the Odontoceti (toothed whales) including dolphins and beaked whales, and the Mysticeti (baleen whales) including the Blue, Gray, and Humpback whales. Sirenians include the Dugong and Manatees but only the Dugong (*Dugong dugon*) occurs in the WIO. About 37 species of marine mammals have been recorded in the WIO, but there are several taxonomic uncertainties (Kiszka, 2015). Six marine mammal species are currently listed as Threatened (Table 1; Appendix), the Blue whale (*Balaenoptera musculus*) and the Sei whale (*Balaenoptera borealis*) are categorized as Endangered as well as the recent addition Indian Ocean Humpback dolphin (*Sousa plumbea*) newly listed as Endangered due to the small size of the populations and threats over a large portion of its range (Braulik et al., 2015). The Sperm whale (*Physeter microcephalus*), the Fin whale (*Balaenoptera physalus*) and the Dugong (*Dugong dugon*) are listed as Vulnerable.

Distribution

Marine mammal distribution and abundances are influenced by a number of factors including depth, habitat, productivity and prey availability. Some species are predominantly coastal and restricted to nearshore waters while others, especially the large and beaked whales, are mainly oceanic and occur in deeper waters (Appendix). Kiszka's (2015) review of information on the distribution of marine mammals in the WIO, and more recently surveys along the Tanzanian coastline (Braulik et al., 2018) have increased knowledge of WIO cetacean species distributions.

The most common dolphins occurring in most countries of the WIO include the Indo Pacific bottlenose (*Tursiops andicus*), Indian Ocean Humpback (*S. plumbea*), Spinner (*Stenella longirostris*), Spotted (*Stenella attenuata*) and Common bottlenose (*Tursiops truncatus*) dolphins (Kiszka, 2015). The Indian Ocean Humpback dolphins have been reported in the widest range of habitats including mangroves, rocky reefs, coastal lagoons and shallow protected bays (Braulik et al., 2015).

Of the whale species, the most common are the Humpback whales (*Megaptera novaeangliae*) that are widely distributed, breed and have nursery grounds in the WIO (Kiszka, 2015). The Southern Right whale (*Eubalaena australis*) also breeds and calves between the south-west coast of South Africa to Maputo (Best, 1990). Other widely distributed whales include the Melon-headed (*Peponocephala electra*), Short finned pilot (*Globicephala macrorhynchus*), Blainville's beaked (*Mesoplodon densirostris*), False killer (*Pseudorca crassidens*) and Blue (*Balaenoptera musculus*) whales reported in several WIO countries (Kiszka, 2015).

Dugongs occur in most WIO countries (Appendix) but their populations have decreased markedly and the only viable population left occurs in Bazaruto Archipelago, Mozambique (Muir and Kiszka, 2012). Dugongs have also been reported in the Rufiji delta, Tanzania, in Lamu archipelago and southern Kenya, in the Comoros, Mayotte, Madagascar, and Aldabra atoll, Seychelles (Wamukoya et al., 1996; Hamylton et al., 2012; Muir and Kiszka, 2012; Pusineri et al., 2013). The species is the most endangered marine mammal in the WIO and is considered extinct in Mauritius, near-extinct in Kenya and Tanzania and barely surviving in Mozambique (Muir and Kiszka, 2012).

There is limited information on the abundances, population structures and migratory routes of cetaceans in the WIO and what data is available is limited to a few species in restricted areas. Populations of the Indo-Pacific bottlenose and Indian Ocean Humpback dolphins were estimated off Zanzibar and the Kisite-Mpunguti MPA, Kenya (Stensland et al., 2006, Meyler et al., 2012). In addition, populations of the Indo-Pacific bottlenose dolphins were estimated off the islands of Mauritius (Cadinouche et al., 2010, Webster et al., 2014), Mayotte (Kiszka et al., 2010, Pusineri et al., 2014) and La Réunion (Dulau et al., 2017), and the Spinner dolphins off Mauritius (Cadinouche et al., 2010). Humpback whale populations have been estimated off KwaZulu-Natal (Findlay and Best, 2006), Madagascar (Best et al., 2003) and La Reunion (Dulau-Drouot et al., 2012).

Threats

Marine mammals face a number of threats in the WIO (Appendix) including, but not limited to, incidental and targeted catch in small-scale and commercial fisheries, habitat destruction and fragmentation, coastal pollution, noise and physical threat from dynamite fishing (Temple et al., 2018). Although information on threats to specific species is scarce, targeted and incidental catch of marine mammals was the largest threat documented in small scale fisheries in the Comoros, Mayotte, Tanzania, Kenya, Madagascar, Mozambique and La Réunion (Amir, 2010; Cerchio et al., 2015; Braulik et al., 2018; Temple et al., 2018). Bycatch was highest in drift and set coastal gillnets and lowest in the longline fisheries (Temple et al., 2018). Blast fishing in Tanzania is also a potential threat (Braulik et al., 2018).

Coastal developments that cause habitat destruction, fragmentation and loss may pose threats to nearshore species (Condet and Dulau-Drouot, 2016). Nearshore species are also particularly vulnerable to pollution discharged by rivers and surface runoff especially as coastal human populations increase (Dirtu et al., 2016; Lane et al., 2014). The potential threat from ship and boat collisions and noise disturbances from shipping and seismic surveys will increase with the projected increase in mining and port developments.

Conservation measures

The establishment of the Indian Ocean Whale sanctuary in 1979 stopped whaling in the WIO and since then several other global conservation strategies have been adopted. Whales and dolphins are included in the highly migratory species list of UNCLOS (1982) ratified by all countries in the WIO and trade in these species and their products is restricted in countries that have ratified CITES (Table 1, Appendix). A global assessment of marine mammals identified 20 key conservation sites that included the southern coast of South Africa (Pompa et al., 2011). In addition, the Indo-Pacific Humpback dolphin and the Killer whale (*Orcinus orca*) are included in Appendix II of CMS. More recently, 20 Important Marine Mammals Areas (IMMA) were approved for the WIO by the marine mammal protected areas task force (IUCN, 2019).

Within individual countries management straegies include MPAs, conservation and fisheries legislation, and research and monitoring, however, the continued decrease of some species of marine mammals indicates that these measured need to be strengthened. Experiments to reduce bycatch included using acoustic deterrents, or making the nets more acoustically conspicuous and replacing shark nets with baited drumlines, which do not catch dolphins (Cliff and Dudley, 2011; Atkins et al., 2013). More recently BYCAM project experimented with recycled bottles and C hooks on longlines and suggested larger scale experiments (Berggren et al., 2019). Community-based ecotourism of dolphins has replaced hunting (Berggren et al., 2007; Cerchio et al., 2015) resulting in reductions in directed-take from historical levels in Madagascar and Zanzibar.

Recommendations, management and policy implication

Information on distribution, long-term population trends, and behavior of cetaceans is scarce in the WIO, making it difficult to manage these species. National research and monitoring programs are needed especially for species that are currently classified as threatened, are hunted or that are targets for tourism. Research findings can inform and strengthen national legislation and recovery plans for these species. Monitoring programs could incorporate citizen science initiatives, such as the Kenya Marine Mammal Network, the Zanzibar monitoring program and community ecotourism initiatives in Madagascar, that not only collect data but are effective networking and awareness tools.

It is also recommended that countries continue to experiment and develop fishing technologies that reduce bycatch. Ocean noise from shipping, marine construction and other causes impacts marine mammals (Erbe et al., 2019), but few studies have been undertaken in the WIO. Assessments of the threats of ocean noise are especially urgent given the projected increase of shipping in the WIO (Halpern et al., 2015) and the CBD (2016) decision on addressing impacts of marine debris and anthropogenic underwater noise on marine and coastal biodiversity.

With the rapid growth of coastal areas and the growing interest in developing the Blue Economy in the WIO, the pressure from coastal development and oil and gas mining will greatly increase. National marine spatial planning should include marine mammals to ensure their feeding, breeding, nursery areas and migratory routes are protected. Major threats from coastal development can be mitigated by implementing national IZCM and Environmental Impact Assessment policies in those countries that have not initiated them and strengthening management and conservation actions for marine mammals.

Due to the highly migratory nature of marine mammals, discussions to create Transboundary Conservation Areas, especially incorporating some of the recently approved IMMAs such as the area on the Kenya/Tanzania border and Pemba Channel and the Mozambique Channel, need to be supported. Regional bodies such as the IOTC, SWIOF also assist in coordinating conservation action in areas beyond national jurisdictions. Lastly, awareness, and other conservation measures including community-based ecotourism, local protection and enforcement as well as social outreach campaigns can complement national conservation measures and strengthen co-management of marine resources.

REFERENCES

Adams, J.B. (2016). Distribution and status of *Zostera capensis* in South African estuaries. *S. Afr. J. Bot*. 107, pp. 63–73

Alcoverro, T. and Mariani, S. (2002). Effects of sea urchin grazing on seagrass (*Thalassodendron ciliatum*) beds of a Kenyan lagoon. *MEPS* 226, pp. 255–263

Alfonso, I., Tacoronte, J.E. and Mesa, J.A. (2007). Isostichotoxin isolated from *Isostichopus badionotus* (Selenta, 1867) sea cucumber processing byproducts. *SPC. Bech-de-Mer Inf. Bull*. 25, pp. 29–31

Amir, O.A. (2010). *Biology, ecology and anthropogenic threats of Indo-Pacific bottlenose dolphins in east Africa*. PhD thesis, Stockholm University, Sweden

Anderson, S.C., Mills Flemming, J., Watson, R. and Lotze, H.K (2011). Rapid global expansion of invertebrate fisheries: Trends, drivers, and ecosystem effects. *PloS One* 6, p. e14735

Ateweberhan, M., Hudson, J., Rougier, A., Harris, A., Jiddawi, N. and Msuya, F.E. (2014). Community based aquaculture in the Western Indian Ocean: Challenges faced and lessons learned. Zanzibar, December 9–11, 2013. Blue Ventures Conservation, London

Ateweberhan, M., Hudson, J., Rougier, A., Jiddawi, N.S., Msuya, F., Stead, S.M. and Harris, A. (2018a). Community based aquaculture in the western Indian Ocean: challenges and opportunities for developing sustainable coastal livelihoods. *Ecol. Soc*. 23(4), p. 17

Ateweberhan, M., McClanahan, T.R., Maina, J. and Sheppard, C. (2018b). Thermal energy and stress properties as the main drivers of regional distribution of coral species richness in the Indian Ocean. *J. Biogeogr*. 45(6), pp. 1355–1366. https://doi.org/10.1111/jbi.13224

Atkins, S., Cliff, G. and Pillay, N. (2013). Humpback dolphin bycatch in the shark nets in KwaZulu-Natal, South Africa. *Biol. Conserv*. 159, pp. 442–449

Aumeeruddy, R. and Conand, C. (2008). Seychelles: a hotspot of sea cucumber fisheries in Africa and the Indian Ocean region. In *Sea cucumbers: a global review on fisheries and trade*. (eds. V. Toral-Granda, A. Lovatelli and M. Vasconcellos). FAO Fisheries and Aquaculture Technical Paper 516, pp. 195–209

Bandeira, S.O. (2011). Seagrasses. In *A Field Guide to the Sea-shores of Eastern Africa and the Western Indian Ocean Islands* (ed. M.D. Richmond) 3rd Edition, pp. 74-77. SIDA/WIOMSA

Bandeira, S.O. and Gell, F. (2003). The seagrasses of Mozambique and Southeastern Africa. In *World Atlas of Seagrasses* (eds. E.P. Green and F.T. Short), pp. 93–100. University of California press

Barkley, A.N., Gollock, M., Samoilys, M., Llewellyn, F., Shivji, M., Wetherbee, B. and Hussey, N.E. (2019). Complex transboundary movements of marine megafauna in the Western Indian Ocean. *Anim. Conserv*. 22(5), pp. 420–431. https://doi.org/10.1111/acv.12493

Barnes-Mauthe, M., Oleson, K.L.L. and Zafindrasilivonona, B. (2013). The total economic value of small-scale fisheries with a characterization of post-landing trends: An application in Madagascar with global relevance. *Fish. Res*. 147, pp. 175–185

Berggren, P., Amir, O.A., Guissamulo, A., Jiddawi, N.S., Ngazy, Z., Stensland, E., Sarnblad, A. and Cockroft, V.G. (2007). *Sustainable Dolphin Tourism in East Africa: MASMA Technical Report*. WIOMSA Book Series, No. 7, pp. ix+72

Berggren, P., Sharpe, M., Temple, A.J., Yang, L., Amir, O.A, Jiddawi, N.S. and Neashan, J. (2019). Recycled bottles offer potential low-cost solution to marine mammal bycatch. Abstract WIOMSA Scientific Symposium. 1–5 July 2019

Best, P.B. (1990). Trends in the inshore right whale population off South Africa 1969–1987. *Mar. Mamm. Sci*. 6(2), pp. 93–108

Best, P.B., Rademeyer, R.A., Burton, C., Ljungblad, D., Sekiguchi, K., Shimada, H., Thiele, D., Reeb, D. and Butterworth, D.S. (2003). The abundance of blue whales on the Madagascar Plateau, December 1996. *J. Cetacean Res. Manag*. 5(3), pp. 253–260

Björk M., Short F., Mcleod, E. and Beer, S. (2008). *Managing Seagrasses for Resilience to Climate Change*. IUCN, Gland, Switzerland. 56 pp.

Bjorndal, K. A. (1985). Nutritional ecology of sea turtles. *Copeia*, pp. 736–751

Bjorndal, K.A. (1997). Foraging ecology and nutrition of sea turtles. In *The biology of sea turtles* (eds. P.L. Lutz and A. Musick), pp. 199–231. CRC Press. Boca Raton, Florida

Borri M., Cianfanelli S., Martignono R. and Volpi, C. (2005) Taxonomic and ecological remarks on the marine molluscs of the Mida Creek area (Kenya, Western Indian Ocean). *Atti della Societa Italiana di scienze naturali e del museo civico di storia naturale di Milano* 146, pp. 79–94

Bourjea, J. (2015). Sea Turtles: a review of status, distribution and interaction with fisheries in the Southwest Indian Ocean. In *Offshore Fisheries of the Southwest Indian Ocean* (eds. R. van der Elst and B. Everett). Special Publication 10, pp. 325–350. Oceanographic Research Institute. Durban, South Africa

Bourjea, J., Lapegue, S., Gagnevin, L., Broderick, D., Mortimer, J. A., Ciccione, S., Roos, D., Taquet, C. and Grizel, H. (2017). Phylogeography of the green turtle, *Chelonia mydas*, in the Southwest Indian Ocean. *Mol. Ecol.* 16(1), pp. 175–186

Bourjea, J., Nel, R., Jiddawi, N.S., Koonjul, M.S. and Bianchi, G. (2008). Sea turtle bycatch in the West Indian Ocean: review, recommendations and research priorities. *West. Indian Ocean J. Mar. Sci*. 7(2), pp. 137–150

Braulik, G.T., Findlay, K., Cerchio, S. and Baldwin, R. (2015). Assessment of the conservation status of the Indian Ocean humpback dolphin (*Sousa plumbea*) using the IUCN Red List Criteria. In *Advances in Marine Biology* (eds. T.A. Jefferson and B.E. Curry), Vol. 72, pp. 119–141. Academic Press

Braulik, G.T., Kasuga, M., Wittich, A., Kiszka, J.J., MacCaulay, J., Gillespie, D., Gordon, J., Said, S.S., Hammond, P.S. and Hammond, P.S. (2018). Cetacean rapid assessment: An approach to fill knowledge gaps and target conservation across large data deficient areas. *Aquat. Conserv. Mar. Freshw. Ecosyst*. 28(1), pp. 216−230

Burke, L., Reytar, K., Spalding, M. and Perry, A. (2011). *Reefs at risk revisited*. World Resources Institute. Washington, D.C.

Butler, C.J. (2019). A review of the effects of climate change on Chelonians. *Diversity*, pp. 1−22

Cadinouche, A., Webster, I. and Cockcroft, V.G. (2010). *Sustainable management of marine resources, cetaceans and reefs of Mauritius: Dolphin study*. Final report of the Mauritius Marine Conservation Society

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., A.Wardle, D., Kinzig, A.P., Daily, G.C., Loreau, M., Grace, J.B., Larigauderie, A., Srivastava, D.S. and Naeem, S. (2012). Biodiversity loss and its impact on humanity. *Nature* 489, pp. 326−326

Carpenter, K. E., Abrar, M., Aeby, G., Aronson, R. B., Banks, S., Bruckner, A., Chiriboga, A., Cortés, J., Charles Delbeek, J. C., DeVantier, L., Edgar, G. J., Edwards, A. J. Fenner, D., Guzmán, H. M., Hoeksema, B. W. Hodgson, G., Johan, O., Licuanan, W. Y., Livingstone, S. R., Lovell, E. R., Moore, J. A., Obura, D. O., Ochavillo, D., Polidoro, B. A., Precht, W. F., Quibilan, M. C., Reboton, C., Richards, Z. T., Rogers, A. D., Sanciangco, J., Sheppard, A., Sheppard, C., Smith, J., Stuart, S., Turak, E., Veron, J. E. N., Wallace, C., Weil, E. and Wood, E. (2008). One-third of reef-building corals face elevated extinction risk from climate change and local impacts. *Science* 321(5888), pp. 560−563

Celliers, L. and Schleyer, M. (2006). Observations on the behaviour and character of an *Acanthaster planci* (L.) aggregation in a high-latitude coral community in South Africa. *West. Indian Ocean J. Mar. Sci*. 5, pp. 105−113

Celliers, L. and Schleyer, M.H. (2002). Coral bleaching on high-latitude marginal reefs at Sodwana Bay, South Africa. *Mar. Pollut. Bull*. 44(12), pp. 1380−1387

Cerchio, S., Andrianarivelo, N. and Andrianantenaina, B. (2015). Ecology and conservation status of Indian Ocean humpback dolphins (*Sousa plumbea*) in Madagascar. *Adv. Mar. Biol*. 72, pp. 163−199

Chadwick, P., Maina, J. and Tuda, A.O. (2021). Part IV: Management Effectiveness. p. 229-256. In: UNEP-Nairobi Convention and WIOMSA. *Western Indian Ocean Marine Protected Areas Outlook: Towards achievement of the Global Biodiversity Framework Targets*. UNEP and WIOMSA, Nairobi, Kenya, 276 pp.

Cherbonnier, G. (1988). *Echinodermes: Holothurides*. Faune de Madagascar. ORSTOM, Vol. 70, 292 pp.

Cliff, G. and Dudley, S.F.J. (2011). Reducing the environmental impact of shark-control programs: a case study from KwaZulu-Natal, South Africa. *Mar. Freshw. Res*, 62(6), pp. 700−709

Condet, M. and Dulau-Drouot, V. (2016). Habitat selection of two island-associated dolphin species from the south-west Indian Ocean. *Cont. Shelf Res*. 125, pp. 18−27

CBD (2016). Decision adopted by the conference of the parties to the Convention on Biological Diversity. XIII/10. Addressing impacts of marine debris and anthropogenic underwater noise on marine and coastal biodiversity. CBD – Convention on Biological Diversity. CBD/COP/DEC/XIII/10.

Costanza, R., de Groot, R., Sutton, P., Van der Ploeg, S., Anderson, S. J., Kubiszewski, I., Farber, S. and Turner, R.K. (2014). Changes in the global value of ecosystem services. *Glob. Environ. Chang*. 26, pp. 152−158

Crawford, B., Herrera, M.D., Hernandez, N., Leclair, C.R., Jiddawi, N., Masumbuko, S. and Haws, M. (2010). Small scale fisheries management: Lessons from cockle harvesters in Nicaragua and Tanzania. *Coast. Manage*. 38(3), pp. 195−215

Cuvillier, A., Villeneuve, N., Cordier, E., Kolasinski, J., Maurel, L., Farnier, N., and Frouin, P. (2017) Causes of seasonal and decadal variability in a tropical seagrass seascape (Reunion Island, south western Indian Ocean). *Estuar. Coast. Shelf Sci*. 184, pp. 90−101

Dalleau, M., Ciccione, S., Mortimer, J.A., Garnier, J., Benhamou, S. and Bourjea, J. (2012). Nesting phenology of marine turtles: Insights from a regional comparative analysis on green turtle (*Chelonia mydas*). *PLoS One*, 7, e46920.

Dalleau, M., Benhamou, S., Sudre, J., Ciccione, S. and Bourjea, J. (2014). The spatial ecology of juvenile loggerhead turtles (*Caretta caretta*) in the Indian Ocean sheds light on the "lost years" mystery. *Mar. Biol*. 161(8), pp. 1835−1849

Dalleau, M., Kramer-Schadt, S., Gangat, Y., Bourjea, J., Lajoie, G. and Grimm, V. (2019). Modeling the emergence of migratory corridors and foraging hot spots of the green sea turtle. *Ecol. Evol*. 9(18), pp. 10317−10342

Davidson, L. and Dulvy, N.K. (2017). Global marine protected areas to prevent extinctions. *Nat. Ecol. Evol*. 1(2), pp. 1-6. https://doi.org/10.1038/s41559-016-0040

de la Torre-Castro, M. and Rönnbäck, P. (2004). Links between humans and sea grasses − an example from tropical East Africa. *Ocean Coast. Manag*. 47, pp. 361−387

Dirtu, A.C., Malarvannan, G., Das, K., Dulau-Drouot, V., Kiszka, J., Lepoint, G., Mongin, P. and Covaci, A. (2016). Contrasted accumulation patterns of persistent organic pollutants and mercury in sympatric tropical dolphins from the South-Western Indian Ocean. *Environ. Res*. 146, pp. 263−273

Duarte, C.M. (2002). The future of seagrass meadows. *Environ. Conserv*. 29, pp. 192−206

Duarte, M.C., Bandeira, S. and Romeiras, M.M. (2012). Systematics and ecology of a new species of seagrass (*Thlassodendron*: Cymodoceacea) from Southeast African coasts. *Novon* 22, pp. 16–24

Ducarme, F. (2016). Revision of the geographical range of *Actinopyga capillata* Rowe and Massin 2006 (Echinodermata: Holothuridea). *SPC Beche-de-mer Info. Bull*. 37, pp. 92–94

Dulau-Drouot, V., Fayan, J., Mouysset, L. and Boucaud, V. (2012). Occurrence and residency patterns of humpback whales of Réunion Island during 2004 -10. *J. Cetacean Res. Manag*. 12(2), pp. 255–263

Dulau, V., Estrade, V. and Fayan, J. (2017). Identifying key demographic parameters of a small island-associated population of Indo-Pacific bottlenose dolphins (Reunion, Indian Ocean). *PLoS One* 12(6), e0179780

Dulvy, N.K., Fowler, S.L., Musick, J.A., Cavanagh, R.D., Kyne, P.M., Harrison, L.R., Carlson, J.K., Davidson, L.N.K., Fordham, S.V, Francis, M.P., Pollock, C.M., Simpfendorfer, C.A., Burgess, G.H., Carpenter, K.E., Compagno, L.J. V, Ebert, D.A., Gibson, C., Heupel, M.R., Livingstone, S.R., Sanciangco, J.C., Stevens, J.D., Valenti, S. and White, W.T. (2014). Extinction risk and conservation of the world's sharks and rays. *Elife* 3, pp. 1–35. https://doi.org/10.7554/eLife.00590

Dulvy, N.K., Pacaureau, N., Rigby, C.L., Pollom, R.A., Jabado, R.W., Ebert, D.A., Finucci, B., Pollock, C.M., Cheok, J., Derrick, D.H., Herman, K.B., Sherman, C.S., VanderWright, W.J., Lawson, J.M., Walls, R.H.L., Carlson, J.K., Charvet, P., Nineesh, K.K., Fernando, D., Ralph, G.M., Matsushib, J.H., Hilton-Taylor, C., Fordham, S.J. and Simpfendorfer, C.A. (2021). Overfishing drives over one-third of all sharks and rays toward a global extinction crisis. *Current Biology* 31, pp. 1–15, e1-e8. https://doi.org/10.1016/j.cub.2021.08.062

Ebert, D.A. and van Hees, K.E. (2015). Beyond Jaws: rediscovering the 'lost sharks' of southern Africa. *Afri. J. Mar. Sci*. 37(2), pp. 141–156

Ebert, D.A., Fowler, S. and Compagno, L.J.V. (2013). *Sharks of the world: a fully illustrated guide*. Wild Nature Press. Plymouth, UK

Ebert, D.A., Wintner, S.P. and Kyne, P.M. (2021). An annotated checklist of the chondrichthyans of South Africa. *Zootaxa* 4947(1), pp. 1–127

Eklöf, J.S., de la Torre-Castro, M., Gullström, M., Uku, J., Muthiga, N., Lyimo, T. and Bandeira, S.O. (2008). Sea urchin overgrazing of seagrasses: A review of current knowledge on causes, consequences, and management. *Estuar. Coast. Shelf Sci*. 79(4), pp. 569–580

Elst, R., Bernadine, E., Jiddawi, N., Mwatha, G., Paula, A. and David, B. (2005). Fish, fishers and fisheries of the Western Indian Ocean: Their diversity and status. A preliminary assessment. *Philos. Trans. R. Soc. A*. 363, pp. 263–284

Erbe, C., Marley, S.A., Schoeman, R.P., Smith, J.N., Trigg, L.E. and Embling, C.B. (2019). The effects of ship noise on marine mammals – A review. *Front. Mar. Sci*. 6, p. 606

Eriksson, H. and Clarke, S. (2015). Chinese market responses to overexploitation of sharks and sea cucumbers. *Biol. Conserv*. 184, pp. 163–173

Eriksson, H., Conand, C., Lovatelli, A., Muthiga, N.A. and Purcell, S. W. (2015). Governance structures and sustainability in Indian Ocean sea cucumber fisheries. *Mar. Policy*. 56, pp. 16–22

Eriksson, H., de La Torre-castro, M. and Olsson, P. (2012). Mobility, expansion and management of a multi-species scuba diving fishery in East Africa. *PLoS One* 7(4), e35504

Eriksson, H.B., de la Torre-Castro, M., Eklöf, J. and Jiddawi, N. (2010). Resource degradation of the sea cucumber fishery in Zanzibar, Tanzania: a need for management reform. *Aquat. Living Resour*. 23, pp. 387–398

Essen, M. and Richmond, M.D. (2011). Superclass Pisces – Fish. *In A Field Guide to the Seashores of Eastern Africa and the western Indian Ocean Islands* (ed. M.D. Richmond) 3rd edition, pp. 340–379. Sida/WIOMSA

Fan, H.Z. (1979). Study on the physical, chemical, toxin and anti-tumor characteristics of mucopolysaccharide. *Communication on Ocean Medicine* 1, pp. 61–67

FAO (2010). *Guidelines to reduce sea turtle mortality in fishing operations*. FAO Fisheries and Aquaculture Department. Rome, Italy

FAO (2013). *Report on the FAO workshop on sea cucumber fisheries: An ecosystem approach to management in the Indian Ocean (SCEAM Indian Ocean), Mazizini, Zanzibar, the United Republic of Tanzania, 12-16 November 2012*. FAO Fisheries and Aquaculture Report, n°1038, 92 pp. Rome, Italy

Feary, D.A., Pratchett, M.S., Emslie, M., Fowler, A.M., Figueira, W.F., Luiz, O.J., Nakamura, Y. and Booth, D.J. (2014). Latitudinal shifts in coral reef fishes: why some species do and others do not shift. *Fish. Fish*. 15(4), pp. 593–615

Ferrari, M.C.O., Munday, P.L., Rummer, J.L., McCormick, M.I., Corkill, K., Watson, S-A., Allan, B.J.M., Meekan, M.G. and Chivers, D.P. (2015). Interactive effects of ocean acidification and rising sea temperatures alter predation rate and predator selectivity in reef fish communities. *Glob. Chang. Biol*. 21(5), pp. 1848–1855

Findlay, K. and Best, P. (2006). *The migrations of humpback whales past Cape Vidal South Africa and a preliminary estimate of the population increase rate*. Report SC/A06/HW16 to the Scientific Committee of the International Whaling Commission

Fowler, S. (2014). *The Conservation Status of Migratory Sharks*. UNEP/CMS Secretariat, 30 pp. Bonn, Germany

Fratini, S., Vigiani, V., Vannini, M. and Cannicci, S. (2004). *Terebralia palustris* (Gastropoda; Potamididae) in a Kenyan mangal: size structure, distribution and impact on the consumption of leaf litter. *Mar. Biol*. 144, pp. 1173–1182

Fricke, R., Mahafina. J., Behivoke, F., Jaonalison, H., Léopold, M. and Ponton, D. (2018). *Annotated checklist of the fishes of Madagascar, southwestern Indian Ocean, with 158 new records. J. Fish. Taxonomy*. 3(1), pp. 66−376

Fuentes, M., Limpus, C. and Hamann, M. (2011). Vulnerability of sea turtle nesting grounds to climate change. *Glob. Change. Biol*. 17(1), pp. 140−153

Gattuso, J., Allemand, D. and Frankignoulle, M. (1999). Photosynthesis and calcification at cellular, organismal and community levels in coral reefs: A review on interactions and control by carbonate chemistry. *Amer. Zool*. 39(1), pp. 16−183

Godley, B.J., Blumenthal, J.M., Broderick, A.C., Coyne, M.S., Godfrey, M.H., Hawkes, L.A. and Witt, M.J. (2008). Satellite tracking of sea turtles: Where have we been and where do we go next? *Endanger. Species Res*. 4, pp. 3−22

Green, E.P. and Short, F.T. (2003). *World Atlas of Seagrasses*. Prepared by the UNEP World Conservation Monitoring Centre. University of California Press. Berkeley, USA

Gullström, M., de la Torre Castro, M., Bandeira, S.O., Björk, M., Dahlberg, M., Kautsky, N., Rönnbäck, P. and Öhman, M.C. (2002). Seagrass ecosystems in the western Indian Ocean. *Ambio* 31(7), pp. 588−596

Halpern, B.S., Frazier, M., Potapenko, J., Casey, K.S., Koenig, K., Longo, C., Lowndes, J.S., Rockwood, R.C., Selig, E.R., Selkoe, K.A. and Walbridge, S. (2015). Spatial and temporal changes in cumulative human impacts on the world's ocean. *Nat. Commun*. 6, p. 7615

Hamylton, S. M., Hagan, A. B. and Doak, N. (2012). Observations of dugongs at Aldabra Atoll, western Indian Ocean: Lagoon habitat mapping and spatial analysis of sighting records. *Int. J. Geogr. Inf. Sci*. 26(5), 839−853

Harcourt, W. D., Briers, R. A. and Huxham, M. (2018). The thin (ning) green line? Investigating changes in Kenya's seagrass coverage. *Biol. Lett.* 14(11), 20180227. https://doi.org/10.1098/rsbl.2018.0227

Heck Jr., K.L. and Valentine, J.F. (2006). Plant-herbivore interactions in seagrass meadows. *J. Exp. Mar. Biol. Ecol*. 330, pp. 420−436

Heemstra, P. and Heemstra, E (2019). Coastal fishes of the Western Indian Ocean. https://www.saiab.ac.za/uploads/files/saiab_pre_publication_brochure_2019_web_72dpi_new.pdf

Hicks, C.C. (2011). How do we value our reefs? Risks and tradeoffs across scales in "biomass-based" economies. *Coast. Manage*. 39(4), pp. 358−376

Hoegh-Guldberg, O. (2011). Coral reef ecosystems and anthropogenic climate change. *Reg. Environ. Change* 11(1), pp. 215−227

Humber, F., Godley, B.J., Ramahery, V. and Broderick, A.C. (2011). Using community members to assess artisanal fisheries: The marine turtle fishery in Madagascar. *Anim. Conserv*. 14(2), pp. 175−185

Huo, D., Sun, L., Zhang, L., Ru, X., Liu, S. and Yang, H. (2019). Metabolome responses of the sea cucumber *Apostichopus japonicus* to multiple environmental stresses: Heat and hypoxia. *Mar. Pollut. Bull*. 138, pp. 407−420

IPBES (2019). *Global assessment report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*. (eds. S. Díaz, J. Settele, E. Brondízio and N.T. Ngo) Bonn, Germany, IPBES Secretariat: 1753. https://doi.org/10.5281/zenodo.3831674

Ireland, C., Malleret-King, D. and Baker, L. (2004). *Alternative sustainable livelihoods for coastal communities- a review of experience and guide to best practice*. IUCN Eastern Africa Programme

IUCN (1996). A Marine Turtle Conservation Strategy and Action Plan for the Western Indian Ocean. IUCN East Africa Regional Office and IUCN/SSC Marine Turtle Specialist Group

IUCN (2001). *IUCN Red List Categories and Criteria: Version 3.1*. IUCN Species Survival Commission. IUCN, Gland, Switzerland and Cambridge, United Kingdom: 30 pp

IUCN (2019). *The IUCN Red List of Threatened Species. Version 2019-2*. Gland, Switzerland: IUCN http://www.iucnredlist.org.

IUCN (2021). *The IUCN Red List of Threatened Species. Version 2021−1*. https://www.iucnredlist.org

Janakiram, N.B., Mohammed, A. and Rao, C.V. (2015). Sea cucumbers metabolites as potent anti-cancer agents. *Mar. Drugs* 13(5), pp. 2909−2923

Jordan, I.E. and Samways, M.J. (2001). Recent changes in coral assemblages of a South African coral reef, with recommendations for long-term monitoring. *Biodivers. Conserv*. 10(7), pp. 1027−1037

Kaschner, K., Quick, N., Jewell, R., Williams, R. and Harris, C.M. (2012). Global coverage of cetacean line-transect surveys: Status quo, gaps and future challenges. *PLoS One* 7(9), e44075

Katona, S. and Whitehead, H. (1988). Are cetacea ecologically important? *Oceanogr. Mar. Biol*. 26, pp. 553−568

Kiszka, J. (2015). Marine mammals: A review of status, distribution and interaction with fisheries in the Southwest Indian Ocean. In *Offshore fisheries of the southwest Indian Ocean*, (eds. van der Elst, R. and Everett, B.) Special Publication 10, pp. 303−324. Oceanographic Research Institute. Durban, South Africa

Kiszka, J., Bein, A., Bach, P., Jamon, A., Layssac, K., Labart, S. and Wickel, J. (2010). *Catch and bycatch in the pelagic longline fishery around Mayotte (NE Mozambique Channel), July 2009−September 2010*. IOTC WPEB-19

Kiszka, J., Berggren, P., Rosenbaum, H.C., Cerchio, S., Rowat, D., Drouot-Dulau, V., Razafindrakoto, Y., Vely, M. and Guissamulo, A. (2009). *Cetaceans in the southwest Indian Ocean: A review of diversity, distribution and conservation issues*. Report SC/61/ O18 to the Scientific Committee of the International Whaling Commission

Kiszka, J. and van der Elst, R.P. (2015). Elasmobranchs (sharks and rays). In *Offshore fisheries of the Southwest Indian Ocean: their status and the impact on vulnerable species* (eds. van der Elst, R.P. and Everett, B.I.). Oceanographic Research Institute Special Publication 10, 448 pp. Durban, South Africa

Kohn, A.J. (1968). Microhabitats, abundance and food of *Conus* on Atoll Reefs in the Maldive and Chagos Islands. *Ecology* 49, pp. 1046–1062

Kohn, A.J. and Perron, F.E. (1994). *Life History and Biogeography: Patterns in Conus*. Oxford Science Publications

Kyne, P.M., Carlson, J. and Smith, K. (2013). *Pristis pristis* (errata version published in 2019). The IUCN Red List of Threatened Species 2013: e.T18584848A141788242

Lane, E.P., De Wet, M., Thompson, P., Siebert, U., Wohlsein, P. and Plön, S. (2014). A systematic health assessment of Indian ocean bottlenose (*Tursiops aduncus*) and Indo-Pacific humpback (*Sousa plumbea*) dolphins incidentally caught in shark nets off the KwaZulu-Natal coast, South Africa. *PloS One* 9(9), e107038

Last, P.R., White, W.T., de Carvalho, M.R., Séret, B., Stehmann, M.F.W. and Naylor, G.J.P. (2016). *Rays of the world*. CSIRO Publishing, Clayton South. 790 pp.

Lauret-Stepler, M., Bourjea, J., Roos, D., Pelletier, D., Ryan, P.G., Ciccione, S. and Grizel, H. (2007). Reproductive seasonality and trend of *Chelonia mydas* in the SW Indian Ocean: a 20 yr study based on track counts. *Endanger. Species Res*. 3, pp. 217–227

Leeney, R. (2017). Are sawfishes still present in Mozambique? A baseline ecological study. *PeerJ* 5, p. e2950

Mace, G.M., Collar, N.J., Gaston, K.J., Hilton-Taylor, C., Akçakaya, H.R., Leader-Williams, N., Milner-Gulland, E.J. and Stuart, S.N. (2008). Quantification of extinction risk: IUCN's system for classifying threatened species. *Conserv. Biol*. 22, pp. 1424–1442

MacNeil, M. A., Chapman, D. D., Heupel, M., Simpfendorfer, C. A., Heithaus, M., Meekan, M., Harvey, E., ... Cinner, J. E. (2020). Global status and conservation potential of reef sharks. *Nature, 583*(7818), pp. 801–806

MacTavish, T., Stenton-Dozey, J., Vopel, K. and Savage, C. (2012). Deposit-feeding sea cucumbers enhance mineralization and nutrient cycling in organically-enriched coastal sediments. *PLoS One* 7(11), e50031

Marshall, N. T. and Barnett, R. (eds.) (1997). *The Trade in Sharks and Shark Products in the Western Indian and Southeast Atlantic Oceans*. Traffic, East-Southern Africa. iii + 130 pp.

Massin, C., Rasolofonirina, R., Conand, C. and Samyn, Y. (1999). A new species of *Bohadschia* (Echinodermata, Holothuroidea) from the Western Indian Ocean with a redescription of *Bohadschia subrubra* (Quoy & Gaimard, 1833). *Bull. Inst. R. Sci. Nat. Belg.* 69, pp. 151–160

Massin, C., Samyn, Y. and Thandar, A.S. (2004). The genus *Labidodemas* (Holothuroidea: Aspidochirotida) revisited, with description of three new species and with re-positioning of *Holothuria (Irenothuria) maccullochi Deichmann*, 1958. *J. Nat. Hist*. 38 (14), pp. 1811–1847

McAllister, D.E., Schueler, F.W., Roberts, C.M. and Hawkins, J.P. (1994). Mapping and GIS of global distribution of coral reef fishes on an equal area grid. In *Mapping the biodiversity of nature* (ed. Miller, R.I.), pp. 153–173. Chapman and Hall, London

McClanahan, T.R. (1990). Kenyan coral reef-associated gastropod assemblages: distribution and diversity patterns. *Coral Reefs* 9, pp. 63–74

McClanahan, T.R. (1994). Coral-eating snail *Drupella cornus* population increases in Kenyan coral reef lagoons. *MEPS* 115, pp. 131–137

McClanahan, T.R. (2002). The effects of time, habitat and fisheries management on Kenyan coral reef associated gastropods. *Ecol. Appl*. 12(5), pp. 1485–1495

McClanahan, T.R. (2015). Biogeography versus resource management: How do they compare when prioritizing the management of coral reef fish in the south-western Indian Ocean? *J. Biogeogr*. 42, pp. 2414–2426

McClanahan, T.R. (2020). Wilderness and conservation policies needed to avoid a coral reef fisheries crisis. *Marine Policy 119*, 104022. https://doi.org/10.1016/j.marpol.2020.104022

McClanahan, T.R. and Muthiga, N.A. (1989). Patterns of predation on a sea urchin, *Echinometra mathaei* (de Blainville) on Kenyan coral reefs. *J. Exp. Mar. Biol. Ecol*. 126(1), pp. 77–94

McClanahan, T.R. and Muthiga, N.A. (2016). Similar impacts of fishing and environmental stress on calcifying organisms in Indian Ocean coral reefs. *MEPS*. 560, pp. 87–103

McClanahan, T.R., Sheppard, C.R.C. and Obura, D.O. (2000). *Coral Reefs of the Indian Ocean: Their Ecology and Conservation*. Oxford University Press

McClanahan, T.R., McLaughlin, S.M., Davy, J.E., Wilson, W.H., Peters, C.E., Price, K.L. and Maina, J. (2004). Observations of a new source of coral mortality along the Kenyan coast. *Hydrobiologia* 530(1), pp. 469–479

Mcleod, E., Chmura, G.L., Bouillon, S., Salm, R., Björk, M., Duarte, C.M., Lovelock, C.E., Schlesinger, W.H. and Silliman, B.R. (2011). A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO_2. *Front. Ecol. Environ*. 9, pp. 552–560

Meyler S.V, Felix, H. and Crouthers, R. (2012). Abundance and distribution of Indo-Pacific humpback dolphins (*Sousa chinensis*) in the Shimoni Archipelago, Kenya. *West. Indian Ocean J. Mar. Sci.* 10(2), pp. 201–209

Migraine, P. and Hykle, D. (2014). Socio-economic and cultural implications of marine turtle use and conservation: A review of the literature from the IOSEA region. *Memorandum of Understanding on the Conservation and Management of Marine Turtles and their Habitats of the Indian Ocean and South-East Asia*. MT IOSEA/SS.7/Doc. 10.3. Seventh Meeting of the Signatory States. Bonn, Germany

Moberg, F. and Folke, C. (1999). Ecological goods and services of coral reef ecosystems. *Ecol. Econ*. 29, pp. 215−233

Moore, J.E., Cox, T.M., Lewison, R.L., Read, A.J., Bjorkland, R., McDonald, S.L., Crowder, L.B, Aruna, E., Ayissi, I., Espeut, P., Joynson-Hicks, C., Pilcher, N., Poonian, C.N.S., Solarin, B. and Kiszka, J. (2010). An interview-based approach to assess marine mammal and sea turtle captures in artisanal fisheries. *Biol. Conserv*. 143(3), pp. 795−805

Morita, M., Suwa, R., Iguchi, A., Nakamura, M., Shimada, K., Sakai, K. and Suzuki, A. (2010). Ocean acidification reduces sperm flagellar motility in broadcast spawning reef invertebrates. *Zygote* 18(2), pp. 103−107

Mortimer, J.A. (1985). Recovery of green turtles on Aldabra. *Oryx* 19, pp. 146-150

Mortimer, J.A. and Donnelly, M. (2007). Marine turtle specialist group 2007 Red List status assessment: Hawksbill turtle (*Eretmochelys imbricata*). pp. 1−121

Muir, C.E. and Kiszka, J. (2012). Eastern African dugongs. In *Sirenian Conservation: Issues and strategies in developing countries* (eds. E. Hines, J. Reynolds III, L.V.A.A. Mignucci-Giannoni and M. Marmontel), pp 84−90. USA, University Press of Florida

Muthiga, N.A. and Kawaka, J. (2009). The breeding pattern and variations in timing and reproductive output of the commercial sea cucumber *Holothuria fuscogilva* in Kenya. West. *Indian Ocean J. Mar. Sci*. 8(2), pp. 183−192

Muthiga, N.A. and Conand, C. (eds.) (2014). *Sea cucumbers, a Poorly Understood but Important Coastal Resource: A Regional Analysis to improve management*. WIOMSA Book Series 14. viii+74 pp.

Muthiga, N.A., Kawaka, J.A. and Ndirangu, S. (2009). The timing and reproductive output of the commercial sea cucumber *Holothuria scabra* on the Kenyan coast. *Estuar. Coast. Shelf Sci*. 84, pp. 353−360

Myers, R.A., Baum, J.K., Shepherd, T.D., Powers, S.P. and Peterson, C.H. (2007). Cascading effects of the loss of apex predatory sharks from a coastal ocean. *Science* 315, p. 1846

Nagelkerken, I., and Munday, P.L. (2016). Animal behaviour shapes the ecological effects of ocean acidification and warming: moving from individual to community-level responses. *Glob. Chang. Bio*l. 22, pp. 974−989

Nel, R., Wanless, R.M., Angel, A., Meklet, B. and Harris, L. (2013a). *Ecological Risk Assessment and Productivity - Susceptibility Analysis of sea turtles overlapping with fisheries in the IOTC region.* Unpublished Report to IOTC and IOSEA Marine Turtle MoU

Nel, R., Punt, A.E. and Hughes, G.R. (2013b). Are coastal protected areas always effective in achieving population recovery for nesting sea turtles. *PLoS One* 8(5), e63525. https://doi.org/10.1371/journal.pone.0063525

Nelson, J.S. (2006). *Fishes of the world, 4th edition.* John Wiley and Sons, Inc. New Jersey

Nerini, M. (1984). A review of gray whale feeding ecology. In *The Gray Whale, Eschrichtius Robustus* (eds. Jones, M.L., Swartz, S.L. and Leatherwood, S.), pp 243−450. Academic Press

Neubauer, P., Jensen, O.P., Hutchings, J.A. and Baum, J.K. (2013). Resilience and recovery of overexploited marine populations. *Science* 340(6130), pp. 347−349

Newton, L.C., Parkes, E.V.H. and Thompson, R.C. (1993). The effects of shell collecting on the abundance of gastropods on Tanzanian Shores. *Biol. Conserv*. 63, pp. 241−245

Nicholas, M. (2001). Light pollution and marine turtle hatchlings: The straw that breaks the camel's back? *The George Wright Forum* 18(4), pp. 77−82

Nilsson, G.E. and Lefevre, S. (2016). Physiological challenges to fishes in a warmer and acidified future. *Physiology* 31, pp. 409−417

Nordlund, L., Erlandsson, J., de la Torre-Castro, M. and Jiddawi, N. (2010). Changes in an East African social-ecological seagrass system: Invertebrate harvesting affecting species composition and local livelihood. *Aquat. Living Resour*. 23(4), pp. 399−416

Nordlund, L.M., de la Torre-Castro, M., Erlandsson, J., Conand, C., Muthiga, N., Jiddawi, N. and Gullström, M. (2014). Intertidal zone management in the western Indian Ocean: Assessing current status and future possibilities using expert opinions. *AMBIO* 43(8), pp. 1006−1019

Nordlund, L.M., Unsworth, R.K.F., Gullström, M. and Cullen-Unsworth, L.C. (2018). Global significance of seagrass fishery activity. *Fish Fish.* 19, pp. 399−412

Obura, D. (2012). The diversity and biogeography of Western Indian Ocean reef-building corals. *PloS One* 7(9), p. e45013

Ochieng, C.A. and Erftemeijer, P.L.A. (2003). Seagrasses of Kenya and Tanzania. In *World Atlas of seagrasses* (eds. Green, E.P. and Short, F.T.), pp. 82−92. Prepared by the UNEP World Conservation Monitoring Centre, University of California Press. Berkeley, USA

Okemwa, G. M., Nzuki, S. and Mueni, E.M. (2004). The status and conservation of sea turtles in Kenya. *Mar. Turtle Newsl.* 105, pp. 1−6

Okemwa, G.M., Muthiga, N.A. and Mueni, E.M. (2005). Proceedings of the Western Indian Ocean Marine Turtle Conservation Workshop. Mombasa, Kenya

Orwa, P.O., Ntiba, M.J., Muthiga, N.A. and Kawaka, J. (2009). Resource assessment of the holothurian populations in Mombasa marine national park and nearby unprotected reefs. In *People, Processes and Ecosystems* (eds. Hoorweg, J. and Muthiga, N.A.). African Studies Centre Collection vol. 20, pp. 162−177

Pershing, A., Christensen, L., Record, N., Sherwood, G. and Stetson, P. (2010). The impact of whaling on the ocean carbon cycle: Why bigger was better. *PLoS One 5* (8), e12444

Peters, H., O'Leary, B.C., Hawkins, J.P., Carpenter, K.E. and Roberts, C.M. (2013). Conus: First comprehensive conservation Red List assessment of a marine gastropod mollusc genus. *PLoS One 8*(12), p. e83353

Pimm, S.L., Jenkins, C.N., Abell, R., Brooks, T.M., Gittleman, J.L., Joppa, L.N., Raven, P.H., Roberts, C.M. and Sexton, J.O. (2014). The biodiversity of species and their rates of extinction, distribution, and protection. *Science* 344(6187), p. 1246752

Pinsky, M.L., Selden, R.L. and Kitchel, Z.J. (2020). Climate-driven shifts in marine species ranges: Scaling from organisms to communities. *Ann. Rev. Mar. Sci.* 12(1), pp. 153−179

Polidoro, B.A., Livingstone, S.R., Carpenter, K.E., Hutchinson, B., Mast, R.B., Pilcher, N., Sadovy de Mitcheson, Y. and Valenti, S. (2009). Status of the world's marine species. In *Wildlife in a Changing World* (eds. J.-C. Vié., Hilton-Taylor, C. and Stuart, S.N.), Island Press, IUCN. Gland, Switzerland

Polidoro, B.A., Carpenter, K.E., Collins, L., Duke, N.C., Ellison, A.M., Ellison, J.C., Farnsworth, E.J., Fernando, E.S., Kathiresan, K., Koedam, N.E., Livingstone, S.R., Miyagi, T., Moore, G.E., Nam, V.N., Ong, J.E., Primavera, J.H., Salmo, S.G., Sanciangco, J.C., Sukardjo, S., Wang, Y. and Yong, J.W.H. (2010). The loss of species: Mangrove extinction risk and geographic areas of global concern. *PLoS One* 5(4), p. e10095. https://doi.org/10.1371/journal.pone.0010095

Poloczanska, E.S., Brown, C.J., Sydeman, W.J., Kiessling, W., Schoeman, D.S., Moore, P.J., Brander, K., Bruno, J.F., Buckley, L.B., Burrows, M.T., Duarte, C.M., Halpern, B.S., Holding, J., Kappel, C.V., O'Connor, I.M., Pandolfi, J.M., Parmesan, C., Schwing, F., Thompson, S.A. and Richardson, A.J. (2013). Global imprint of climate change on marine life. *Nat. Clim Change* 3(10), pp. 919−925

Pompa, S., Ehrlich, P.R. and Ceballos, G. (2011). Global distribution and conservation of marine mammals. *PNAS* 108 (33), pp. 13600−13605

Poonian, C.N.S., Hauzer, M.D., Allaoui, A.B., Cox, T.M., Moore, J.E., Read, A.J., Lewison, R.L. and Crowder, L.B. (2008). Rapid assessment of sea turtle and marine mammal bycatch in the Union of the Comoros. *West. Indian Ocean J. Mar. Sci.* 7(2), pp. 207−215

Postaire, B., Bruggemann, J.H., Magalon, H. and Faure, B. (2014). Evolutionary dynamics in the Southwest Indian Ocean Marine Biodiversity Hotspot: A perspective from the rocky shore Gastropod Genus *Nerita. PLoS One* 9, p. e95040

Preen, A. (1995). Impacts of dugong foraging on seagrass habitats: observational and experimental evidence for cultivation grazing. *MEPS* 124(1), pp. 201−213

Prugh. L.R., Stoner, C.J., Epps, C.W., Bean, W.T., Ripple, W.J., Laliberte, A.S. and Brashares, J.S. (2009). The rise of the mesopredator. *Bioscience* 59, pp. 779−791

Purcell, S.W., Samyn, S. and Conand, C. (2012). Commercially important sea cucumbers of the world. FAO Species catalogue for fishery purposes. No. 6., FAO. 150 pp. 30 colour plates. Rome, Italy

Purcell, S.W., Conand, C., Uthicke, S. and Byrne, M. (2016). Ecological roles of exploited sea cucumbers. *Oceanogr. Mar. Biol.* 54, pp. 367−386

Purcell, S.W., Polidoro, B.A., Hamel, J.F., Gamboa, R.U. and Mercier, A. (2014). The cost of being valuable: Predictors of extinction risk in marine invertebrates exploited as luxury seafood. *Proc. R. Soc. Lond.* 281(1781), p. 20133296

Pusineri, C., Kiszka, J., Quillard, M. and Caceres, S. (2013). The endangered status of dugongs *Dugong dugon* around Mayotte (East Africa, Mozambique Channel) assessed through interview surveys. *Afri. J. Mar. Sci* 35(1), pp. 111−116

Rasolofonirina, R. (2005). Reproduction et développement de l'holothurie *Holothuria scabra à* Madagascar (Holothuroidea: Echinodermata). PhD, ULB, Bruxelles

Raybaudi-Massilia, G. (2013). *Conus julii*. The IUCN Red List of Threatened Species. Version 2014.2

Richmond, D. (2015). Threatened Marine Species. In The *Regional State of the Coast Report: Western Indian Ocean* (ed. J. Paula) pp. 129−152, UNEP-Nairobi Convention and WIOMSA. Nairobi, Kenya

Richmond, M.D. and Rabesandratana, H.D. (2011). Phylum Mollusca – Molluscs. In *A Field Guide to the Seashores of Eastern Africa and the western Indian Ocean Islands* (ed. M.D. Richmond) 3rd edition, pp. 248−257 and 284−297. Sida/WIOMSA

Rockel, D., Korn, W. and Kohn, A.J. (1995). Manual of the Living Conidae, Vol 1. Verlag Christa Hemmen

Rönnbäck, P., Bryceson, I. and Kautsky, N. (2002). Coastal aquaculture development in eastern Africa and the western Indian Ocean: Prospects and problems for food security and local economies. *Ambio* 31, pp. 537−542

Rowe, F.W.E. and Massin, C. (2006). On a new species of *Actinopyga* Bronn, 1860 (Echinodermata, Holothuroidea) from the Indo-West Pacific. *Zoosystema* 28(4), pp. 955−961

Rowe, F.W.E. and Richmond, M.D. (2011). Phylum Echinodermata-Echinoderms. In *A Field Guide to the Seashores of Eastern Africa and the western Indian Ocean Islands* (ed. Richmond, M.D.) 3rd Edition, pp. 300−329. Sida/WIOMSA

Samoilys, M., Pabari, M., Andrew, T., Maina, G.W., Church, J., Momanyi, A., Mibei, B., Monjane, M., Shah, A., Menomussanga, M. and Mutta, D. (2015). *Resilience of coastal systems and their human partners in the Western Indian Ocean*. IUCN ESARO, WIOMSA, CORDIO and UNEP Nairobi Convention. x + 74pp. Nairobi, Kenya

Samyn, Y., Massin, C. and Muthiga, N.A. (2001). A new species of *Holothuria* (Aspidochirotida, Holothuriidae) from Kenya. *Annals. Sci. Zool*. 285, pp. 101–110

Schleyer, M. (2011). Order Scleractinia (Madreporaria) - Hard Corals. In *A Field Guide to the Seashores of Eastern Africa and the western Indian Ocean Islands* (ed. M.D. Richmond) 3rd edition, pp. 150–163. Sida/WIOMSA

Schleyer, M.H., Floros, C., Laing, S.C., Macdonald, A.H., Montoya-Maya, P.H., Morris, T., Porter, S.N. and Seré, M.G. (2018). What can South African reefs tell us about the future of high-latitude coral systems? *Mar. Pollut. Bull*. 136, pp. 491–507

Schneider, K., Silverman, J., Woolsey, E., Eriksson, H., Byrne, M. and Caldeira, K. (2011). Potential influence of sea cucumbers on coral reef $CaCO_3$ budget: A case study at One Tree Reef. *J. Geophys. Res*. 116(G4). https://doi.org//10.1029/2011JG001755

Séré, M.G., Tortosa, P., Chabanet, P., Turquet, J., Quod, J.-P. and Schleyer, M.H. (2013). Bacterial communities associated with *Porites* white patch syndrome (PWPS) on three Western Indian Ocean (WIO) coral reefs. *PLoS One* 8(12), p. e83746

Sheppard, C.R.C. (2000). Coral reefs of the western Indian Ocean an overview. In *Coral reefs of the Indian Ocean* (eds. T.R. McClanahan, C.R.C. Sheppard and D.O. Obura), pp. 3–38. Oxford University Press

Sheppard, A., Fenner, D., Edwards, A., Abrar, M. and Ochavillo, D. (2008). *Goniopora burgosi. The IUCN Red List of Threatened Species* 2008, e.T133391A3721140

Short, F.T., Polidoro, B., Livingstone, S.R., Carpenter, K.E., Bandeira, S., Bujang, J.S., Calumpong, H.P., Carruthers, T.J.B., Coles, R.G., Dennison, W.C., Erftemeijer, P.L.A., Fortes, M.D., Freeman, A.S., Jagtap, T.G., Kamal, A.H.M., Kendrick, G.A., Judson Kenworthy, W., La Nafie, Y.A., Nasution, I.M., Orth, R.J., Prathep, A., Sanciangco, J.C., Tussenbroek, B. Van, Vergara, S.G., Waycott, M. and Zieman, J.C. (2011). Extinction risk assessment of the world's seagrass species. *Biol. Conserv.* 144, pp. 1961–1971

Smith, M.M. and Heemstra, P.C.H. (1986). *Smiths' Sea Fishes*. McMillan, Johannesburg

Spalding, M., Ravilious, D.C. and Green, E.P. (2001). *World Atlas of Coral Reefs*. Berkeley, USA, University of California Press

Spalding, M., Taylor M., Rayilious C., Short, F. and Green, E. (2003). The distribution and status of seagrasses. In *World Atlas of seagrasses* (eds. E.P. Green and F.T. Short), pp. 82-92. University of California press, Berkeley, USA

Stein, R.W., Mull, C.G., Kuhn, T.S., Aschliman, N.C., Davidson, L.N.K., Boy, J.G., Smith, G.J., Dulvy, N.K. and Mooers, A.O. (2018). Global priorities for conserving the evolutionary history of sharks, rays and chimaeras. *Nat. Ecol. Evol*. 2(2), pp. 288-298. https://doi.org/10.1038/s41559-017-0448-4

Stensland, E., Carlen, I., Sarnblad, A., Bignert, A. and Berggren, P. (2006). Population size, distribution, and behavior of Indo-Pacific bottlenose (*Tursiops aduncus*) and humpback (*Sousa chinensis*) dolphins off the south coast of Zanzibar. *Mar. Mamm. Sci*. 22(3), pp. 667–682

Stokes, H.J., Mortimer, J.A., Hays, G.C., Unsworth, R.K.F., Laloë, J-O. and Esteban, N. (2019). Green turtle diet is dominated by seagrass in the Western Indian Ocean except amongst gravid females. *Mar. Biol*. 166, p. 135

Temple, A.J., Kiszka, J.J., Stead, S.M., Wambiji, N., Brito, A., Poonian, C.N., Amir, O.A., Jiddawi, N., Fennessy, S.T., Pérez-Jorge, S. and Berggren, P. (2018). Marine megafauna interactions with small-scale fisheries in the southwestern Indian Ocean: A review of status and challenges for research and management. *Rev. Fish Biol. Fish* 28(1), pp. 89–115

Thandar, A.S. and Samyn, Y. (2004). Shallow-water holothuroid (Echinodermata: Holothuroidea) biodiversity and biogeography of the subtropical east coast of South Africa. Proceedings of the 11th International Echinoderm Conference, October 2003. Munich, Germany

Toral-Granda, V., Lovatelli, A. and Vasconcellos, M. (2008). *Sea cucumbers. A global review of fisheries and trade*. FAO Fisheries and Aquaculture Technical Paper, nº516, 317 p. Rome, FAO

Troëng, S. and Drews, C. (2004). *Money Talks: Economic Aspects of Marine Turtle Use and Conservation*, WWF-International. Gland, Switzerland

UNEP-WCMC, IUCN and NGS (2021). *Protected planet live report 2021*. UNEP-WCMC., IUCN and NGS

UNEP-Nairobi Convention and WIOMSA (2015). *The Regional State of the Coast Report: Western Indian Ocean*. UNEP and WIOMSA, Nairobi, Kenya, 546 pp.

Unsworth, R.K.F., Nordlund, L.M. and Cullen-Unsworth, L.C. (2019). Seagrass meadows support global fisheries production. *Conserv. Lett.* 12(1), p. e12566. https://doi.org/10.1111/conl.12566

Veron, J.E.N. and Turak, E. (2005). Zooxanthellate scleractinia of Madagascar. In *A rapid marine biodiversity assessment of the coral reefs of northwest Madagascar* (eds. S.A. McKenna and G.R. Allen). RAP Bulletin of the Rapid Assessment Program 31. Conservation International. Washington DC

Wallace, B.P., Kot, C.Y., DiMatteo, A.D., Lee, T., Crowder, L.B. and Lewison, R.L (2013). Impacts of fisheries bycatch on marine turtle populations worldwide: Toward conservation and research priorities. *Ecosphere* 4(3), pp. 1–49

Wamukoya, G.M., Mirangi, J.M. and Ottichillo, W.K. (1996). *Report on the marine aerial survey of the marine mammals, turtles, sharks and rays.* Kenya Wildlife Service Technical Report Series N°1. Mombasa, Kenya

Wamukoya, G.M. and Salm, R.V. (eds.) (1997). *Proceedings of the Western Indian Ocean turtle excluder device (TED) training workshop*, Mombasa, Kenya, 27–31 January 1997, IUCN. 30 pp.

Waycott, M., McMahon, K., Mellors, J., Calladine, A. and Kleine, D. (2004). *A guide to tropical seagrasses of the Indo-West Pacific*. James Cook University Townsville Australia

Waycott, M., Duarte, C.M., Carruthers, T.J.B., Orth, R.J., Dennison, W.C., Olyarnik, S., Calladine, A., Fourqurean, J.W., Heck Jr., K.L., Hughes, A.R., Kendrick, G.A., Kenworthy, W.J., Short, F.T. and Williams, S.L. (2009). Accelerating loss of seagrasses across the globe threatens coastal ecosystems. PNAS *106*(30), pp. 12377-12381

Webster, I. (2013). Observations of green and hawksbill turtles on the Southwest Coast of Mauritius. *Marine Turtle Newsletter*, n°138, pp. 15–17

Webster, I., Cockcroft, V.G. and Cadinouche, A. (2014). Abundance of the Indo-Pacific bottlenose dolphin *Tursiops aduncus* off south-west Mauritius. *Afri. J. Mar. Sci.* 36(3), pp. 293–301

Wells, S. (1997). Giant clams: Status, trade and mariculture, and the role of CITES in management. IUCN-the World Conservation Union. Gland, Switzerland

Wells, S., Addison, P.F.E., Bueno, P.A., Costantini, M., Fontaine, A., Germain, L., Lefebvre, T., Morgan, L., Staub, F., Wang, B., White, A. and Zorrilla, M.X. (2016). Using the IUCN Green List of Protected and Conserved Areas to promote conservation impact through marine protected areas. *Aquatic Conserv: Mar. Freshw. Ecosyst*. 26, pp. 24–44

Wells, S., Burgess, N. and Ngusaru, A. (2007). Towards the 2012 marine protected area targets in Eastern Africa. *Ocean Coast. Manag*. 50(1), pp. 67–83

Whittingham, E., Campbell, J. and Townsley, P. (2003). *Poverty and reefs*. DFID-IMM-IOC/UNESCO, 260 pp.

Williams, J.L., Pierce, S.J., Rohner, C.A., Fuentes, M.M.P.B. and Hamann, M. (2017). Spatial distribution and residency of green and loggerhead sea turtles using coastal reef habitats in southern Mozambique. *Front. Mar. Sci.* 3, p. 288. https://doi.org/10.3389/ fmars.2016.00288

Wolkenhauer, S.M., Uthicke, S., Burridge, C., Skewes, T. and Pitcher, R. (2010). The ecological role of *Holothuria scabra* (Echinodermata: Holothuroidea) within subtropical seagrass beds. *J. Mar. Biol. Assoc*. 90, pp. 215–223

Wood, V.E. (1986). Breeding success of hawksbill turtles *Eretmochelys imbricata* at Cousin Island, Seychelles and the implications for their conservation. *Biol. Conserv*. 37, pp. 321–332

Worm, B., Davis, B., Kettemer, L., Ward-Paige, C.A., Chapman, D., Heithaus, M.R., Kessel, S.T. and Gruber, S.H. (2013). Global catches, exploitation rates, and rebuilding options for sharks. *Mar. Policy* 40, pp. 194–204

Worm, B., Hilborn, R., Baum, J.K., Branch, T.A., Collie, J.S., Costello, C., Fogarty, M.J., Fulton, E.A., Hutchings, J.A., Jennings, S. and Jensen, O.P. (2009). Rebuilding global fisheries. *Science* 325(5940), pp. 578–585

Yonow, N. (2012). Opisthobranchs from the western Indian Ocean, with descriptions of two new species and ten new records (Mollusca, Gastropoda). *ZooKeys* (197), pp. 1–130

Zeller, D., Hood, L., Palomares, M.L.D., Sumaila, U.R., Khalfallah, M., Belhabib, D., Woroniak, J. and Pauly, D. (2020). Comparative fishery yields of African Large Marine Ecosystems. *Environ. Dev*. 36. https://doi.org/10.1016/j.envdev.2020.100543

APPENDIX

Threatened species in the Western Indian Ocean region, their Red List categories, protection measures, threats, habitats and national distribution.

MAJOR TAXA/ SCIENTIFIC NAME	COMMON NAME	RED LIST CATEGORY	PROTECTION MEASURES	THREATS	HABITAT	NATIONAL DISTRIBUTION
Seagrasses						
Zostera capensis	Eelgrass	VU	MPAs	a	Estuaries, lagoons and flats in muddy substrates	KEN; MDG; MOZ; ZAF; TZA
Hard corals						
Acropora roseni	Scleractinian hard coral	EN	MPAs; CITES II	a, b, d	Shallow coral reefs on upper reef slopes exposed to strong waves	COM; MDG; MYT; SYC
Acropora rudis	Scleractinian hard coral	EN	MPAs; CITES II	a, b, d	Shallow fringing coral reef	MUS; SYC
Anacropora spinosa	Scleractinian hard coral	EN	MPAs; CITES II	a, b	Shallow coral reefs on soft substrate lower reef slopes	MUS
Ctenella chagius	Scleractinian hard coral	EN	MPAs; CITES II	a, b	Coral reef slopes and lagoons to 45 m	MUS; REU
Millepora tuberosa	Hydrozoan fire coral	EN	MPAs; CITES II	a, b, e	Inshore coral reefs in turbid waters	MUS
Parasimplastrea sheppardi	Scleractinian hard coral	EN	MPAs; CITES II	a, b	Subtidal rocks and rocky reefs and back and fore slopes and lagoons of coral reef	MUS; REU; SOM
Pocillopora fungiformis	Scleractinian hard coral	EN	MPAs; CITES II	a, b	Shallow coral reefs exposed to strong waves	MDG
Stylophora madagascarensis	Scleractinian hard coral	EN	MPAs; CITES II	a, b	Shallow coral reefs and sheltered lagoons	MDG
Sea cucumbers						
Holothuria lessoni	Golden sandfish	EN	Banned in MYT ; SCUBA ban in KEN and MDG	e	Shallow lagoons and reef flat inner slopes and sea-grass beds	COM; KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; TZA
Holothuria nobilis	Black teatfish	EN	Banned in MYT ; SCUBA ban in KEN and MDG; CITES II proposed	e	Shallow coral reef, on flats and outer slopes and sea-grass beds	COM; KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; TZA; ZAF
Holothuria scabra	Sandfish	EN	Banned in MYT ; Inhambane MOZ; SCUBA ban KEN and MDG	e	Intertidal seagrass beds, inner coral reef flats and lagoons and muddy habitats	COM; KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; TZA; ZAF
Thelenota ananas	Prickly redfish	EN	Banned in MYT ; SCUBA ban in KEN and MDG	e	Sandy to hard bottoms on coral reef flats, slopes and passes	COM; KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; TZA; ZAF
Actinopyga echinites	Brownfish	VU	Banned in MYT; SCUBA ban in KEN and MDG	e	Shallow littoral zones in rubble and reef flats, islet-la-goon reefs, seagrass beds	KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; TZA

MAJOR TAXA/ SCIENTIFIC NAME	COMMON NAME	RED LIST CATEGORY	PROTECTION MEASURES	THREATS	HABITAT	NATIONAL DISTRIBUTION
Actinopyga mauritiana	Surf redfish	VU	Banned in MYT; SCUBA ban in KEN and MDG	e	Outer reef flat with surf	KEN; MDG; MYT; MOZ; REU; SYC; SOM; TZA
Actinopyga miliaris	Hairy blackfish	VU	Banned in MYT; SCUBA ban in KEN and MDG	e	Reef flats of fringing coral reef, lagoons and seagrass beds	KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; TZA
Holothuria arenacava		VU	Banned in MYT; SCUBA ban in KEN and MDG	e	Fine sand in shallow lagoons	KEN
Holothuria fuscogilva	White teatfish	VU	Banned in MYT; SCUBA ban in KEN and MDG; CITES II proposal	e	Sandy bottoms on reef slopes and lagoons	KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; TZA
Stichopus herrmanni	Herrmann's sea cucumber	VU	Banned in MYT; SCUBA ban in KEN and MDG	e	Lagoons, seagrass beds, rubble over sandy muddy substrate	KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; TZA
Gastropods						
Chrysomallon squamiferum	Scaly-foot snail	EN		a	Hydrothermal vents	MUS
Alviniconcha marisindica		EN		a	Hydrothermal vents	MUS
Desbruyeresia marisindica		EN		a	Hydrothermal vents	MUS
Conus jeanmartini		VU		a, e	Depths of 700 meters	REU
Conus julii	Snowflake cone	VU	MPAs; control of collection	e	Coral reef rubble and under rocks	MUS; REU
Conus immelmani	Amadis cone	VU	MPAs	e	Coral reefs 35 to 50 m depth	ZAF
Bony fish						
Chrysoblephus cristiceps	Daggerhead seabream	CR	MPAs	e, f, g, h, i	Coral reef	ZAF
Latimeria chalumnae	Coelacanth	CR	CMS I; CITES I	a, e	Continental, oceanic shelfs	COM
Polysteganus undulosus	Seventyfour seabream	CR	MPAs	e, f, h, i	Offshore	ZAF
Syngnathus watermeyeri	River pipefish	CR	MPAs	a, e, f, h, i	Estuaries	ZAF
Thunnus maccoyii	Southern Bluefin tuna	CR	MPAs	i	Continental, oceanic shelfs	MDG; ZAF
Argyrosomus japonicus	Dusky Meagre	EN	MPAs; Harvest management	a, e, f	Subtidal rock and rocky reefs, sandy-mud, muddy; lagoon reef; estuaries	Pantropical in IWP; ZAF to Beira MOZ
Argyrosomus thorpei	Squaretail Kob	EN	MPAs; Harvest management	a, e, f, i	Subtidal rock and rocky reefs, sandy-mud	ZAF; MOZ; west coast MDG
Cheilinus undulatus	Squaretail Kob	EN	MPAs; CITES II	e, i	Coral reef	COM; KEN; MDG; MUS; MYT; MOZ; SYC; SOM; TZA

MAJOR TAXA/ SCIENTIFIC NAME	COMMON NAME	RED LIST CATEGORY	PROTECTION MEASURES	THREATS	HABITAT	NATIONAL DISTRIBUTION
Chelonodon pleurospilus	Blaasop beauty	EN	MPAs	a, f, h	Subtidal rock, rocky reefs, sandy mud, muddy	Mouth of Xora R. to Durban ZAF
Lethrinus mahsena	Sky Emperor	EN	MPAs	e	Subtidal sandy, reef, seagrass	KEN; MDG; MUS; MYT; MOZ; REU; ZAF; TZA,
Lithognathus lithognathus	White Steenbras	EN	MPAs	a, e, f	Subtidal loose rock, pebble, gravel, sandy, estuary	Endemic to ZAF
Petrus rupestris	Red Steenbras	EN	MPAs	e, f, g, h	Subtidal rock and rocky reefs	Endemic to ZAF, South Western Cape to St Lucia
Springeratus polyporatus		EN	MPAs	a, f, h	Tidepools	MUS; REU
Upeneus saiab	SAIAB Goatfish	EN	NA	a, e, f, h	Marine neritic, subtidal sandy	Endemic to MOZ
Acanthopagrus vagus	Riverbream	VU	MPAs; Harvest management	a, e, f, h	Subtidal loose rock, pebble, gravel, sandy, estuary, man-grove submerged roots, coastal freshwater lakes	Endemic to ZAF
Albula glossodonta	Shortjaw Bonefish	VU	MPAs; Harvest management	a, e	Subtidal rock and rocky reefs, pebble, gravel, sandy, mangrove submerged roots, seagrass	MDG; SYC; SOM
Arothron inconditus	Bellystriped Blaasop	VU	MPAs	a, f, h	Subtidal sandy, sandy-mud, muddy, intertidal sandy shore, tidepools	ZAF
Awaous commersoni	Commerson's Freshwater Goby	VU	MPA	a	Wetlands, loose rock, sub-tidal sandy, estuaries	MUS; REU
Bolbometopon muricatum	Green Humphead Parrotfish	VU	MPAs	a, e	Coral reef	KEN; MDG; MUS; SYC; SOM
Cymatoceps nasutus	Black Musselcracker	VU	MPAs; Harvest management	e, f, g, h	Subtidal rock and rocky reefs, subtidal loose rock, pebble, gravel	ZAF
Epinephelus albomarginatus	White-edged Rockcod	VU	Harvest management	e, h	Subtidal rock and rocky reefs, coral reef	MOZ; ZAF
Epinephelus fuscoguttatus	Brown-marbled Grouper	VU	MPAs; Harvest management	a, e	Coral reef, seagrass, estuaries	COM; KEN; MDG; MUS; MYT ; MOZ; REU; SYC; TZA
Epinephelus marginatus	Dusky Grouper	VU	MPAs	e, g	Subtidal rock and rocky reefs	MDG; MOZ; REU; ZAF
Epinephelus polyphekadion	Camouflage Grouper	VU	MPAs; Harvest management	a, e	Coral reef	COM; KEN; MDG; MUS; MYT ; MOZ; REU; SYC; SOM; TZA
Hippocampus histrix	Thorny Seahorse	VU	MPAs	a, e, i	Macroalgal, kelp, coral reef, seagrass	MDG, MUS, MOZ, REU, SYC, ZAF, TZA
Hippocampus kelloggi	Great Seahorse	VU	MPAs	e, i, h	Subtidal muddy, Coral reef	TZA, KEN
Hippocampus kuda	Spotted Seahorse	VU		a, i	Pelagic, subtidal muddy, seagrass, estuaries, man-grove submerged roots	MOZ; ZAF; TZA

MAJOR TAXA/ SCIENTIFIC NAME	COMMON NAME	RED LIST CATEGORY	PROTECTION MEASURES	THREATS	HABITAT	NATIONAL DISTRIBUTION
Makaira nigricans	Blue Marlin	VU		e, i	Pelagic epipelagic	MDG, MUS, REU, SYC, ZAF, TZA
Mimoblennius lineathorax		VU	MPAs	a, f	Subtidal rock and rocky reefs	REU
Mola mola	Sunfish	VU	MPAs	e, i	Epipelagic, mesopelagic	Global
Oxymonacanthus longirostris	Harlequin Filefish	VU	MPAs	a, e	Subtidal rock and rocky reefs, loose rock, pebble, gravel, coral reef, estuaries	COM; KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; TZA
Parablennius lodosus	Mud Blenny	VU	MPAs	a, c, f, h	Subtidal rock and rocky reefs, rocky shoreline, tidepools	MOZ
Plectropomus areolatus	Squaretail coral grouper	VU	MPA; Harvest management	a, e	Coral reef	REU
Polysteganus praeorbitalis	Scotsman seabream	VU	MPAs	e, f, g, h	Subtidal rock and rocky reefs, subtidal loose rock, pebble, gravel	ZAF; MOZ
Pomatomus saltatrix	Bluefish	VU	MPAs; Harvest management	e,	Pelagic, estuaries	MDG; MYT; SOM; ZAF, TZA
Thunnus obesus	Bigeye Tuna	VU		e	Pelagic, epipelagic, meso-pelagic, bathypelagic	KEN; MUS; MOZ; SYC; SOM; ZAF; TZA
Trachurus indicus	Arabian Scad	VU	Harvest management	e	Pelagic, subtidal sandy	KEN; MUS; SOM
Batoids						
Aetomylaeus bovinus	Duckbill ray	CR	MPAs	a, e, i	Estuarine, coastal, man-groves, soft substrates	MOZ; ZAF; TZA
Glaucostegus halavi	Halavi guitarfish	CR	CITES II; MPAs	a, e, f, h i	Coastal, inshore, coral reefs, sand	KEN
Myliobatis aquila	Eagle ray	CR	MPAs	a, e, i	Estuaries, coastal, inshore, offshore, soft substrates	KEN; MUS; MOZ; REU; ZAF; TZA
Pristis pristis	Largetooth sawfish	CR	CITES I; CMS I, II; MPAs; MLRA (ZAF) b; REPMARj; TFRi	a, c, e, i	Estuarine, mangroves, inshore, offshore	KEN; MDG; MUS; MOZ; REU; SYC; SOM; ZAF; TZA
Pristis zijsron	Green sawfish	CR	CITES I; CMS I, II; MLRA (ZAF)b; MPAs; REPMARj; TFRi	a, c, e, i	Estuarine, mangroves, inshore, offshore	KEN; MDG; MUS; MOZ; REU; SOM; ZAF; TZA
Rhina ancylostoma	Bowmouth guitar-fish or shark ray	CR	CITES I; CMS II; KWCMAg; MPAs	a, c, e, i	Coastal, mangroves, coral reefs, sand	KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; ZAF; TZA
Rhynchobatus australiae	Bottlenose wedgefish	CR	CITES II; CMS II; MPAs	a, c, e, i	Coastal and continental shelf, sand, coral reef	KEN; MDG; MOZ; SYC; SOM; TZA
Rhynchobatus djiddensis	Whitespotted wedgefish	CR	CITES II; CMS II KWCMAg; MPAs; TOPS (ZAF)a	a, c, e, f, i	Continental shelf, estuaries, soft substrates	MOZ; ZAF (pres-ence uncertain in KEN, TZA, SOM)

MAJOR TAXA/ SCIENTIFIC NAME	COMMON NAME	RED LIST CATEGORY	PROTECTION MEASURES	THREATS	HABITAT	NATIONAL DISTRIBUTION
Rhynchobatus laevis	Smoothnose wedgefish	CR	CITES II	a, c, e, i	Coastal, estuaries, mangroves, inshore, soft substrates	Distribution uncertain within WIO
Aetomylaeus vespertilio	Ornate eagle ray	EN	MPAs; TFRi	e, i	Continental, oceanic shelfs, sandy substrate, muddy banks and coral reefs	KEN; MOZ; ZAF; TZA
Acroteriobatus leucospilus	Greyspot guitarfish	EN	MPAs	a, e, i	Coastal and continental shelf, reef and soft substrates	MDG; MOZ; ZAF
Himantura uarnak	Reticulate whipray	EN	MPAs	a, c, e, i	Coastal, sand, sandy areas of coral reefs, shallow estuaries, mangroves	KEN; MDG; MUS; MYT; MOZ; SYC; SOM; ZAF; TZA
Mobula birostris	Giant manta ray	EN	CITES II; CMS I, II; IOTCc; MPAs; REPMARj	a, e, g, i	Pelagic-neritic; inshore and offshore; oceanic islands and seamounts	COM; KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; ZAF; TZA
Mobula eregoodoo	Longhorned pygmy devil ray	EN	CITES II; CMS I, II; IOTCc; MPAs; REPMARj	a, e, g, i	Inshore and offshore on continental shelf; coral reef	ZAF; TZA
Mobula kuhlii	Shortfin devil ray	EN	CITES II; CMS I, II; IOTCc; MPAs; REPMARj	a, e, g, i	Inshore; primarily pelagic on continental shelf	KEN; MDG; MOZ; ; SYC; SOM; ZAF; TZA
Mobula mobular	Giant devil ray	EN	CITES II; CMS I, II; IOTCc; MPAs; REPMARj	a, e, g, i	Pelagic-neritic; inshore and offshore	KEN; MOZ; MYT; SOM; ZAF;
Mobula tarapacana	Sicklefin devil ray	EN	CITES II; CMS I, II; IOTCc; MPAs; REPMARj	a, e, g, i	Pelagic, coastal waters Seasonal oceanic island groups, and near offshore pinnacles and seamounts	MUS; REU; ZAF;
Mobula thurstoni	Bentfin devil ray	EN	CITES II; CMS I, II; IOTCc; MPAs; REPMARj	a, e, g, i	Coastal, inshore, pelagic	MDG; ZAF; TZA
Raja ocellifera	Twineyed skate	EN		a, e, i	Continental shelf, soft substrates	ZAF
Rhinoptera jayakari	Javanese cownose ray	EN	MPAs	a, c, e, g, i	Coastal, mud, sand, sandy areas of coral reefs, shallow estuaries, lagoons	KEN; MDG; MOZ; SOM; ZAF; TZA
Rostroraja alba	Spearnose skate	EN	TFRi	e, i	Coastal to upper slope, sand and detrital bottoms, deep water	KEN; MDG; MOZ; SYC; ZAF
Acroteriobatus annulatus	Lesser guitarfish	VU	MPAs	a, c, e, i	Coastal, inshore, estuaries	ZAF
Aetobatus ocellatus	Spotted eagle ray	VU	MPAs	a, e, i	Coastal and continental shelf, pelagic, coral reef, lagoons, estuaries and mangroves	COM; KEN; MDG; MUS; MYT; MOZ; SYC; SOM; ZAF; TZA
Bathytoshia lata	Brown stingray	VU	MPAs	e, g, i	Coastal, and continental shelf and slop, soft sediments and reefs	COM; MOZ; REU; TZA; ZAF
Dipturus crosnieri	MDG skate	VU		f, i	Continental slope, deep water, soft substrates	MDG

MAJOR TAXA/ SCIENTIFIC NAME	COMMON NAME	RED LIST CATEGORY	PROTECTION MEASURES	THREATS	HABITAT	NATIONAL DISTRIBUTION
Gymnura poecilura	Longtail butterfly ray	VU	MPAs	a, c, e, i	Coastal, inshore, mangroves, soft substrates	KEN; SOM (unconfirmed in MDG, MOZ and TZA)
Himantura leoparda	Leopard whipray	VU	MPAs	a, c, e, i	Coastal, inshore, sand, sandy areas of coral reefs, mangroves	KEN; MOZ; SOM; ZAF; TZA
Leucoraja wallacei	Yellowspot skate	VU		e, i	Demersal on outer continental shelf and upper slope on soft substrates, deep-water	MOZ, ZAF
Mobula alfredi	Reef manta ray	VU	CITES II; CMS I, II; IOTCc; MPAs; REPMARj; TOPS (ZAF)a	a, e, g, i	Epipelagic and mesopelagic Inshore but also offshore coral reefs, rocky reefs and seamounts	COM; MDG; MYT; MOZ; SYC; ZAF
Pastinachus ater	Broad cowtail ray	VU		a, e, g, i	Coastal, sand, sandy areas of coral reefs, lagoons, mangroves	KEN; MDG; MYT; MOZ; SYC; SOM; ZAF; TZA
Pateobatis fai	Pink whipray	VU	MPAs	a, e, i	Inshore, soft substrates	MYT; MOZ; ZAF
Pateobatis jenkinsii	Jenkins whipray	VU	MPAs	a, c, e, i	Inshore, soft substrates	KEN; MDG; MOZ; SOM; ZAF; TZA
Taeniurops meyeni	Blotched fantail ray	VU	KWCMAg; MPAs	a, e, i	Continental shelf, coral reefs, sand	KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; ZAF; TZA
Urogymnus asperrimus	Porcupine ray	VU	KWCMAg; MPAs	a, c, e, i	Inshore, coral reef, sand, mud, mangroves	KEN; MDG;MYT; SYC; SOM; ZAF; TZA
Urogymnus granulatus	Mangrove whipray	VU	MPAs	a, c, e, i	Inshore, sand, mangroves, estuaries	MYT; SYC
Sharks						
Carcharhinus longimanus	Oceanic whitetip shark	CR	CITES II; CMS I; IOTCd; KWCMAg; REPMARj	e, i	Continental, oceanic shelfs	COM; KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; TZA; ZAF
Carcharias taurus	Grey nurse shark/ ragged-tooth shark	CR	KWCMAg; MPAs; TOPS (ZAF)a	e, i	Continental, oceanic shelfs	MOZ; SOM; ZAF; TZA
Pseudoginglymostoma brevicaudatum	Shorttail nurse shark	CR	KWCMAg; MPAs	a, c, e, i	Inshore, continental and insular shelves, coral reefs	KEN; TZA; MDG, MOZ
Sphyrna lewini	Scalloped hammerhead shark	CR	CITES II; CMS II; MLRA (SA)b; TOPS (ZAF)a	e, i	Coastal and semi-oceanic pelagic, continental and insular shelves, estuaries	COM; KEN; MDG; MYT; MOZ; MUS; SYC; SOM; ZAF; TZA
Sphyrna mokarran	Great hammerhead shark	CR	CITES II; CMS II; MLRA (SA)b; TOPS (ZAF)a; TFRi	a, e, i	Continental, oceanic, estuaries, mangroves	COM; KEN; MDG; MUS; MYT ; MOZ; REU; SYC; SOM; ZAF; TZA
Alopias pelagicus	Pelagic thresher Shark	EN	CITES II; CMS II; IOTCf; REPMARj	e, i	Epipelagic and mesopelagic	COM; KEN; MDG; MYT ; MOZ; SYC; SOM; ZAF; TZA

MAJOR TAXA/ SCIENTIFIC NAME	COMMON NAME	RED LIST CATEGORY	PROTECTION MEASURES	THREATS	HABITAT	NATIONAL DISTRIBUTION
Carcharhinus amblyrhynchos	Grey reef shark	EN	MPAs	a, e, i	Coastal, coral reefs	COM; KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; ZAF; TZA
Carcharhinus obscurus	Dusky shark	EN	CMS II; MPAs	e, i	Continental, oceanic shelfs	MDG; MOZ; SOM; ZAF
Carcharhinus plumbeus	Sandbar shark	EN	MPAs	e, i	Continental, oceanic shelfs, estuaries, soft substrates, reefs	COM; KEN; MDG; MYT; MOZ; REU; SYC; SOM; ZAF; TZA
Centrophorus granulosus	Gulper shark	EN		e, i	Continental and insular shelves and slopes, deep water	COM; MDG; MYT ; MOZ; SYC; SOM; ZAF
Centrophorus squamosus	Leafscale gulper shark	EN		e, i	Continental and insular shelves and slopes, abyss, deep water	MOZ, SYC, ZAF
Cetorhinus maximus	Basking shark	EN	CITES II; CMS I, II; MLRA (ZAF)b; TFRi; TOPS (ZAF)a	i	Coastal, epi- and meso-pelagic, deep water	ZAF
Echinorhinus brucus	Bramble shark	EN		e, i	Continental and insular shelves and slopes, deep water	MOZ, SOM, ZAF
Holohalaelurus favus	Honeycomb catshark	EN	MPAs	a, i	Pelagic-oceanic Found on the upper to mid slope	MOZ; ZAF
Holohalaelurus punctatus	African spotted catshark	EN	MPAs	a, e	Continental, oceanic shelfs	MDG; MOZ; ZAF
Isurus oxyrinchus	Shortfin mako shark	EN	CITES II; CMS II	e, i	Coastal, offshore, epipelagic	COM; KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; ZAF; TZA
Isurus paucus	Longfin mako shark	EN	CITES II; CMS II	e, i	Epi-, meso- and bathypelagic	COM; MDG; MYT; SYC; TZA
Mustelus manazo	Starspotted smoothhound	EN	MPAs	e, i	Continental shelf, inshore, soft substrates	KEN; MDG; SYC; TZA (unconfirmed in MOZ)
Mustelus mustelus	Smoothhound shark	EN	MPAs	e, i	Inshore and offshore on continental shelf, upper slope, sand and muddy substrates	ZAF
Negaprion acutidens	Sicklefin lemon shark	EN	MPAs	a, c, e, i	Shallow inshore, offshore, coral reefs and sandy plateaus near coral, mangroves	COM; KEN; MDG; MUS; MYT; MOZ; SYC; SOM; ZAF; TZA
Oxynotus centrina	Angular rough shark	EN		e, i	Continental shelf and upper slope, deep water.	MDG (unconfirmed in MOZ)
Rhincodon typus	Whale shark	EN	CITES II; CMS I, II; IOTCe; KWCMAg; MLRA (ZAF) b; REPMARj; SWABPAh; TFRi; TOPS (ZAF)a	i	Coastal, pelagic	COM; KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; ZAF; TZA

MAJOR TAXA/ SCIENTIFIC NAME	COMMON NAME	RED LIST CATEGORY	PROTECTION MEASURES	THREATS	HABITAT	NATIONAL DISTRIBUTION
Stegostoma tigrinum (fasciatum)	Zebra shark	EN	MPAs	a, e, i	Shallow inshore and offshore waters, coral and rocky reefs and sandy plateaus near coral	KEN; MUS; MDG; MYT; MOZ; SYC; SOM; ZAF; TZA
Alopias superciliosus	Bigeye thresher shark	VU	CITES II; CMS II; IOTCf; REPMARj	e, i	Coastal, continental shelf, epipelagic	COM; KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; ZAF; TZA
Alopias vulpinus	Common thresher shark	VU	CITES II; CMS II; IOTCf; REPMARj	e, i	Coastal, epipelagic	KEN; ZAF
Carcharhinus albimarginatus	Silvertip shark	VU		e, i	Continental shelf, offshore islands, coral reefs, offshore banks	COM; KEN; MDG; MYT; MOZ; REU; SYC; SOM; ZAF; TZA
Carcharhinus amblyrhynchoides	Graceful shark	VU		a, e, i	Coastal, mud, sand, estuaries, pelagic	SOM
Carcharhinus amboinensis	Pigeye shark	VU		a, e, g, i	Coastal, continental shelf, soft sediments, shallow bays, estuaries, mangroves	KEN; MDG; MUS; MOZ; SYC; SOM; ZAF, TZA
Carcharhinus brachyurus	Copper shark	VU		e, i	Continental shelf, inshore and offshore, estuaries	MDG; SYC; ZAF (unconfirmed in MOZ)
Carcharhinus brevipinna	Spinner shark	VU		e, i	Coastal, pelagic, continental shelves, estuaries	MDG; MUS; MOZ; REU; SYC; SOM; ZAF
Carcharhinus falciformis	Silky shark	VU	CITES II; CMS II	e, i	Continental, oceanic shelfs	COM; KEN; MDG; MYT; MOZ; REU; SYC; SOM; ZAF; TZA
Carcharhinus leucas	Bull shark	VU	MPAs	a, e, g, i	Coastal, pelagic, sand, mud, reef, estuaries, rivers	COM; KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; ZAF; TZA
Carcharhinus limbatus	Blacktip shark	VU	MPAs	a, e, g, i	Continental shelf, demersal, pelagic, reefs, bays, mangroves, lagoons	COM; KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; ZAF; TZA
Carcharhinus melanopterus	Blacktip reef shark	VU	MPAs	c, e, i	Coastal, inshore, coral reefs, estuaries, mangroves	COM; KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; ZAF; TZA
Carcharodon carcharias	Great white shark	VU	CITES II; CMS I, II; KWCMAg; MLRA (ZAF)b; REPMARj; TOPS (ZAF)a	e, i	Coastal, pelagic	COM; KEN; MDG; MYT; MOZ; REU; SYC; SOM; ZAF; TZA
Centrophorus moluccensis	Smallfin gulper shark	VU		e, g, i	Continental shelf and upper slope, deep water	MDG; MYT; MOZ; SYC; ZAF; TZA
Centroscymnus owstoni	Roughskin dogfish	VU		e, i	Continental slopes, oceanic ridges, and seamounts	MDG; ; REU; SYC

MAJOR TAXA/ SCIENTIFIC NAME	COMMON NAME	RED LIST CATEGORY	PROTECTION MEASURES	THREATS	HABITAT	NATIONAL DISTRIBUTION
Dalatias licha	Kitefin shark	VU		e, h, i	Continental and insular shelf, mesopelagic	MDG; MOZ; ZAF
Deania quadrispinosa	Longsnout dogfish	VU		e, i	Outer continental shelves, slopes and seamounts	MDG; MOZ; ZAF
Halaelurus boesemani	Speckled catshark	VU	MPAs	c, f, i	Continental shelf, benthic	KEN; SOM
Halaelurus natalensis	Tiger catshark	VU	MPAs	a, i	Inshore and offshore on continental shelf, benthic	ZAF
Haploblepharus fuscus	Brown shyshark	VU	MPAs	a, c, e, f, i	Inshore, rocky reefs	ZAF
Haploblepharus kistnasamyi	Natal shyshark	VU	MPAs; TOPS (ZAF)a	c, i	Inshore, rocky and coral reefs, sand	ZAF
Hemigaleus microstoma	Sicklefin weasel shark	VU	MPAs	e, g, i	Inshore and offshore on continental and insular shelves	TZA
Hemipristis elongata	Snaggletooth shark	VU	MPAs	a, e, i	Inshore and offshore on continental and insular shelves	KEN; MDG; MOZ; SYC; SOM; ZAF; TZA
Nebrius ferrugineus	Tawny nurse shark	VU	MPAs	a, c, e, g, i	Coastal, lagoons, coral and rocky reefs, seagrass, sandy areas near reef	COM; KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; ZAF; TZA
Notorynchus cepedianus	Broadnose seven-gill shark	VU	MPAs	e, i	Coastal, inshore and offshore	ZAF
Odontaspis ferox	Smalltooth sand tiger shark	VU	MPAs	i	Islands, seamounts, continental outer shelves and upper slopes	COM; MDG; SYC; ZAF; TZA
Paragaleus leucolomatus	Whitetip weasel shark	VU	MPAs	a, c, g, h, i	Shallow coastal waters 1-20 m deep	MDG; MOZ; ZAF
Rhizoprionodon acutus	Milk shark	VU	MPAs	e, i	Continental shelf, inshore, offshore	COM; KEN; MDG; MUS; MOZ; SYC; SOM; ZAF; TZA
Scylliogaleus quecketti	Flapnose houndshark	VU	TOPS (ZAF)a	a, e, f, h, i	Inshore	ZAF
Sphyrna zygaena	Smooth hammer-head shark	VU	MLRA (ZAF)b; CITES II; CMS II	e, i	Coastal-pelagic and semi-oceanic, continental shelf	COM; MDG; MUS; MOZ; SYC; SOM; ZAF
Triaenodon obesus	Whitetip reef shark	VU	MPAs	e, i	Coastal, inshore	COM; KEN; MDG; MUS; MYT; MOZ; REU; SYC; SOM; ZAF; TZA
Turtles						
Dermochelys coriacea	Leatherback turtle (sub-pop)	CR	MPAs; CMS I, II; CITES I	a	Continental, oceanic shelfs	COM; KEN; MDG; MUS; MYT; MOZ; SYC; TZA; ZAF
Eretmochelys imbricata	Hawksbill turtle	CR	MPAs; CMS I, II; CITES I	a	Continental, oceanic shelfs	COM; KEN; MDG; MUS; MYT; MOZ; SYC; TZA

MAJOR TAXA/ SCIENTIFIC NAME	COMMON NAME	RED LIST CATEGORY	PROTECTION MEASURES	THREATS	HABITAT	NATIONAL DISTRIBUTION
Chelonia mydas	Green turtle	EN	MPAs; CMS I, II; CITES I	i	Continental, oceanic shelfs	COM; KEN; MDG; MUS; MYT; MOZ; SYC; TZA; ZAF
Caretta caretta	Loggerhead turtle	VU	MPAs; CMS I, II; CITES I	a	Continental, oceanic shelfs	COM; KEN; MDG; MUS; MOZ; SYC; TZA; ZAF
Lepidochelys olivacea	Olive ridley	VU	MPAs; CMS I, II; CITES I	a	Continental, oceanic shelfs	KEN; MDG; MOZ; SOM; TZA
Mammals						
Balaenoptera borealis	Sei whale	EN	CMS I, II; CITES I protected from commercial whaling by the IWC	e	Continental, oceanic shelfs	MDG; REU; SYC; ZAF
Balaenoptera musculus	Blue whale	EN	CMS I, II; CITES I protected from commercial whaling by the IWC	a	Continental, oceanic shelfs	COM; KEN; MDG; MUS; MOZ; REU; SYC; SOM; TZA; ZAF
Sousa plumbea	Indo-Pacific humpback dolphin	EN	CMS I, II; CITES I	i	Continental, oceanic shelfs	COM; KEN; MDG; MYT; MOZ; SOM; TZA
Balaenoptera physalus	Fin whale	VU	CMS I, II; CITES I protected from commercial whaling by IWC	e	Continental, oceanic shelfs	MDG; MUS; MOZ; REU; SYC; TZA; ZAF
Dugong dugon	Sea cow	VU	CMS Appendix II	a	Continental, oceanic shelfs	COM; KEN; MDG; MUS; MYT; MOZ; SYC; SOM; TZA
Physeter macrocephalus	Sperm whale	VU	CMS I, II; CITES I protected from commercial whaling by the IWC	a	Continental, oceanic shelfs	COM; KEN; MDG; MUS; MOZ; SYC; SOM; TZA

Notes on abbreviations/acronyms:
CR=Critically Endangered, EN=Endangered, VU=Vulnerable.

Notes on threat codes:
a) overall species habitat degradation, used as a proxy for population decline; b) susceptible to coral bleaching and disease; c) narrow depth range; d) Crown-of-thorns (COT) predation; e) overharvested (for food); f) restricted geographic range; g) low fecundity; h) rare; i) bycatch.

Notes on regulations;

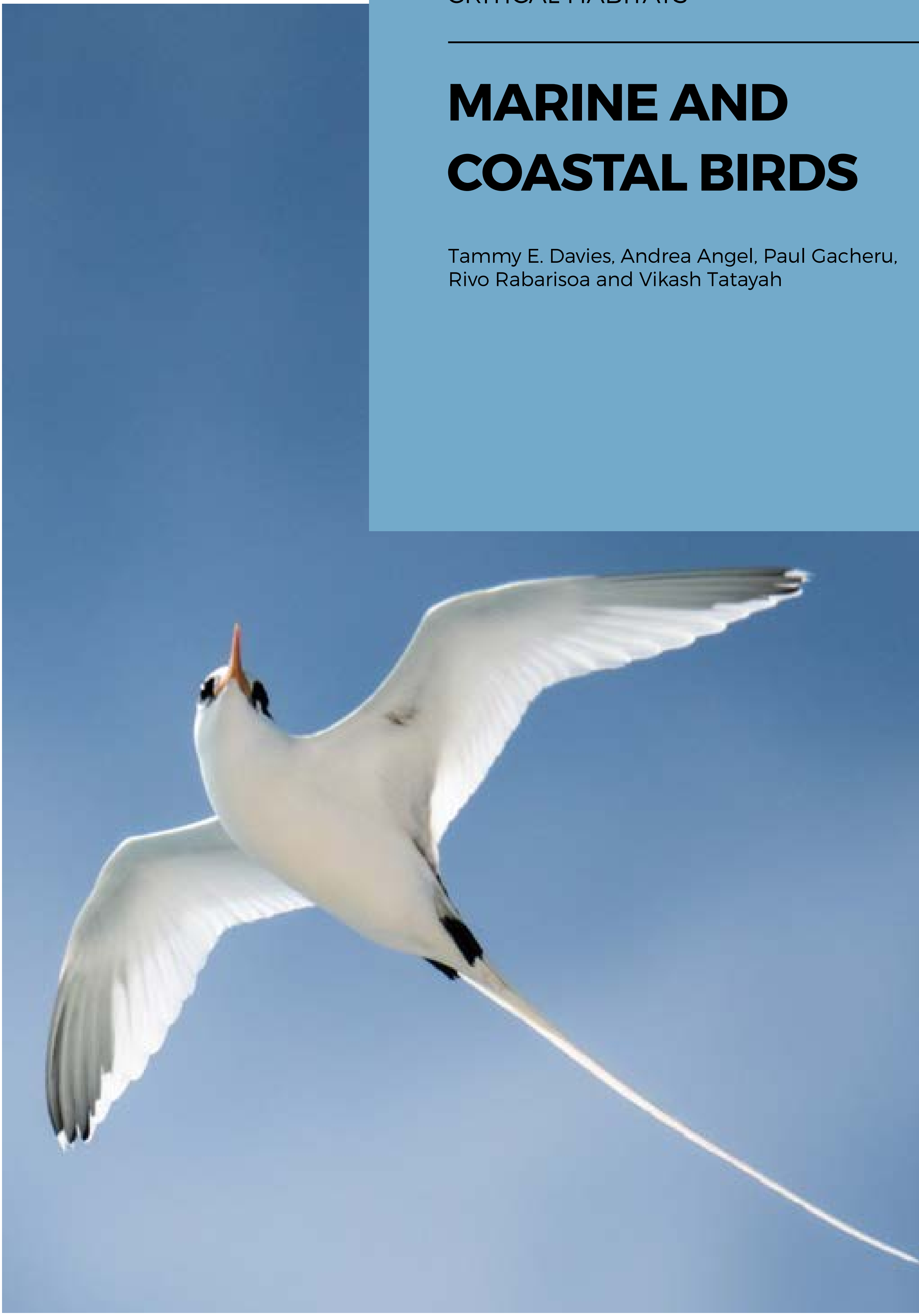
a. Threatened or Protected Species (TOPS) Regulations, Government Gazette 40876 (2017) https://cer.org.za/wp-content/uploads/2004/09/Marine-TOPS-Regulations.pdf;
b. Marine Living Resources Act 1998 (Act no. 18 of 1998). https://www.environment.gov.za/sites/default/files/legislations/marine_livingresources_act18_0.pdf
c. IOTC Resolution 19/03(https://iotc.org/cmm/resolution-1903-conservation-mobulid-rays-caught-iin-association-fisheries-iotc-area-competence)
d. IOTC Resolution 13/06 (http://www.iotc.org/cmm/resolution-1306-scientific-and-management-framework-conservation-sharks-species-caught)
e. IOTC Resolution 13/05 (http://www.iotc.org/cmm/resolution-1305-conservation-whale-sharks-rhincodon-typus)
f. IOTC Resolution 12/09 (http://www.iotc.org/cmm/resolution-1209-conservation-thresher-sharks-family-alopiidae-caught-association-fisheries-iotc)
g. KEN Wildlife Conservation and Management Act, 2013. KEN Gazette Supplement No. 181 (Acts No. 47), 27th December 2013. "47.
h. SYC Wild Animals and Bird Protection Act 1961 – Wild Animals (Whales Shark) Protection Regulation, 2003. "2.
i. TZA Fisheries Regulations 2009 "67. (1)
j. MOZ Sea Fisheries Regulation/Regulamento da Pesca Marítima (REPMAR) Decree n.º 89/2020. Annex XIII. "List of Prohibited Species to Capture (Regarding paragraph 2 of article 146)".

Source:
The IUCN Red List of Threatened Species. Version 2021-1. https://www.iucnredlist.org. MPA: Marine Protected Area. CITES: Convention on International Trade in Endangered Species of Wild Fauna and Flora; Country names from ISO 3166, Comoros: COM, Kenya: KEN, Madagascar: MDG, Mauritius: MUS, Mayotte: MYT, Mozambique: MOZ, Seychelles: SYC, Somalia: SOM, South Africa: ZAF, Tanzania: TZA. IWC: International Whaling Commission, IWP Indo-West Pacific

CRITICAL HABITATS

MARINE AND COASTAL BIRDS

Tammy E. Davies, Andrea Angel, Paul Gacheru, Rivo Rabarisoa and Vikash Tatayah



White tailed tropicbird. © Sharif Uddin, Macauly Library

BACKGROUND

The Western Indian Ocean (WIO) has a high diversity of seabird and coastal birds that can be found in all habitats, including several endemic and near-endemic species. It is an important region for 20 globally threatened seabirds, and migratory shorebirds from Europe and Asia. Seabirds are the most threatened group of birds (Croxall et al., 2012) and are a conservation priority. Compared with other taxa in the marine realm, seabirds are exceptionally well-studied (Schreiber and Burger, 2001). Consequently, knowledge of their conservation status is more comprehensive and reliable than for any comparable group of marine organisms (Vié et al., 2009) and this understanding can have many benefits for conservation, including identifying sites for conservation, acting as ecosystem indicators, and understanding threats.

Seabirds are useful indicators for identifying priority sites for conservation and their distributions can provide surrogates for biodiversity hotspots in marine spatial planning (Zacharias and Roff, 2001; Lascelles et al., 2012; Aslan et al., 2015). This is because their distributions often overlap with other taxa (eg, cetaceans, Hebshi et al., 2008), they occupy a high level on the food chain, and their food webs encompass a wide diversity of marine taxa (eg, Forero et al., 2004). Seabirds are also established ecosystem indicators because changes in seabird populations can be used to infer changes in the marine environment (Parsons et al., 2008) and for example, pollutant concentrations found in seabirds may be an indication of the pollutant load in other economic and environmental keystone marine predators (van der Schyff et al., 2021). Seabird research is increasingly providing information on the health of fish stocks, ecosystem health, ecosystem change, and the impacts of climate change (Einoder, 2009). For example, breeding failure or low recruitment of sensitive seabird species is linked to widespread declines in fish stocks (Piatt et al., 2007).

The Brown noddy (*Anous stolidus*). © José Pedro Granadeiro

Seabirds can be defined as species for which a large proportion of the total population relies on the marine environment for at least part of the year (Croxall et al., 2012). Seabirds derive all or most of their food from the marine environment and spend a considerable part of their time at sea, except when breeding, which occurs on land. Seabirds tend to be grouped together by habitat and morphological characteristics rather than relatedness. Following the classifications of seabirds followed by BirdLife International (2021), the seabird families occurring in the WIO are considered to be penguins (Spheniscidae), albatrosses (Diomedeidae), petrels and shearwaters (Procellariidae), both southern and northern storm-petrels (Oceanitidae and Hydrobatidae), tropicbirds (Phaethontidae), gannets and boobies (Sulidae), frigatebirds (Fregatidae), skuas (Stercorariidae), cormorants (Phalacrocoracidae), gulls and terns (Laridae) and pelicans (Pelecanidae). Shorebirds derive varying amounts of energy from the marine environment but may spend much of their time in terrestrial, freshwater and estuarine environments. Generally shorebirds are considered to be groups such as herons and egrets (Ardeidae), ibises (Threskiornithidae), plovers (Charadriidae), and sandpipers (Scolopacidae).

Outputs reflect data available in November 2018.

DATA ON SEABIRDS

Many seabirds spend much of their lives at sea, often far from land, and our knowledge of their distributions and behaviours during this time can be extremely poor. However, because seabirds breed on land they are relatively more accessible to study compared to many other migratory marine taxa. Additionally, technological advances over recent years, including miniaturization of tracking devices, battery engineering and software development have changed the way the study seabirds is undertaken, providing insights into locomotion, foraging behaviour, migration and exposure to anthropogenic risks at sea (Hussey et al., 2015). There are now a range of devices that are able to document where birds go, such as satellite trackers, geolocators, and global positioning system (GPS) loggers (Burger and Shaffer, 2008). Some devices need to be recollected whereas others transmit

Table 1: Overview of the different tags used to track seabirds.

DEVICE TYPE	DESCRIPTION	PROS	CONS	SUITED TO
Geolocator (GLS)	Uses changes in ambient light levels to estimate sunrise/sunset and hence longitude and latitude	Very lightweight (down to 0.3 g); long battery life; slightly larger tags can record location data for 2–10 years	Coarse resolution; accuracy decreases near the equator and close to equinox; tags must be retrieved to download data	Tracking long distance movements over an annual cycle (migration, long-distance foraging trips)
Argos Platform Transmitter Terminals (PTT)	A network of satellites devoted to environmental studies	Location data are transmitted and not stored so tag recovery is not mandatory	Minimum size of approx. 30 g, so only suitable for larger seabirds (eg, gannets)	Initial dispersion and juveniles, where tag recovery is not possible
Global Positioning System (GPS)	Uses global positioning of satellites to determine price locations	Fine spatial resolution; light-weight device (down to 1 g)	Tags must be retrieved to download data	Ground speed, micro-movements, and area-restricted behaviour

data to satellites, and these devices enable researchers to plot the latitude and longitude of the birds' locations and better understand their distributions. As technology has improved, smaller, cheaper, and more reliable devices have been developed that are allowing researchers to collect tracking data for a broad range of seabird species around the world (Table 1).

The resultant data have provided insights into feeding areas, distribution of seabirds at-sea and overlap with human activities – in particular fisheries and offshore industries. Such information can indicate not only the distribution of tracked birds, but also their behaviour and the way they utilize their surrounding environment. These data are central to seabird conservation because they facilitate unprecedented understanding of the distributions of species throughout their various life history stages (Carneiro et al., 2020).

In recognition of the importance of these data, BirdLife International collaborated with seabird researchers to establish the Seabird Tracking Database with the aim to foster collaboration for seabird research and conservation. This database now holds data for more than 13 million location points from more than 115 seabird species, contributed by more than 180 researchers. Within the Nairobi Convention area there are almost 2000 tracks from 33 species. Bringing together many datasets from various populations and life histories through this collaborative approach has enabled the development of a more complete understanding of how seabirds behave, move and migrate across oceans. This has been integral in developing our understanding of their ecology, how they interact with marine systems, and in promoting their protection from the various anthropogenic pressures that threaten many species with extinction.

IMPORTANT BIRD AND BIODIVERSITY AREAS

Important Bird and Biodiversity Areas (IBAs) are areas of global significance for birds that are based on a set of standardized and objective criteria (Donald et al., 2018). To qualify as an IBA, a site must hold a regular presence of a threshold number of birds (eg, ≥ 1 per cent of a global or biogeographic population). For globally threatened species with very small populations (ie, Critically Endangered or Endangered) regular presence alone may be enough to warrant designation, but for other species, abundance thresholds must also be met. IBA criteria are readily applicable to most threatened and congregatory species during different life history stages and can be used to identify areas such as breeding colonies, foraging areas around colonies, non-breeding congregations, migratory bottlenecks and pelagic feeding aggregations (Osieck, 2004). See Appendix 1 for full IBA criteria.

IBAs provide a focus for conservation action, planning and advocacy, and are recognized by the Convention on Biological Diversity (CBD) as the basis of a worldwide network of priority sites for conservation (Waliczky et al., 2018). IBAs are also a subset of Key Biodiversity Areas (KBAs), which are "sites that contribute significantly to the global persistence of biodiversity." KBAs can have a broader focus than IBAs (ie, any taxa) and present a unifying framework and an overarching set of criteria that can be applied across taxonomies and ecosystems. The IBA framework remains operational, and the majority of marine IBAs also qualify as KBAs.

While it is relatively easy to assess seabird breeding colonies against IBA criteria, it is more difficult to locate

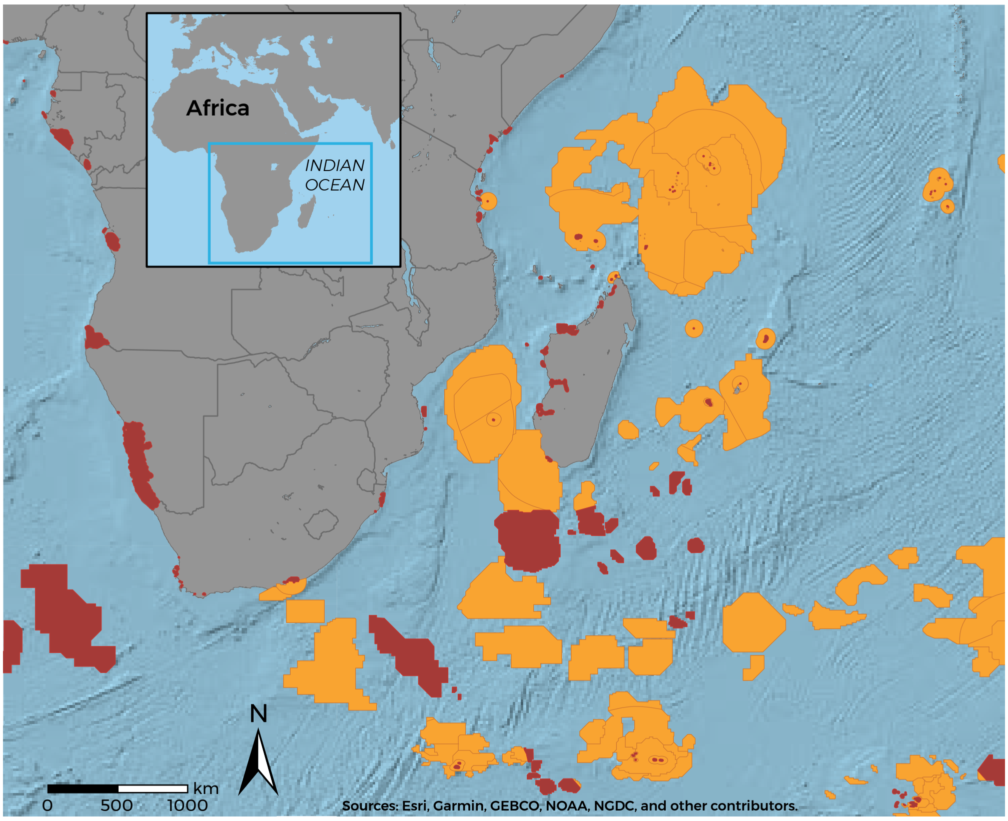


Figure 1:Map of marine Important Bird and Biodiversity Areas in the West Indian Ocean. Red polygons are confirmed marine IBAs, orange polygons are candidate marine IBAs (meaning they are waiting additional data or action to be confirmed). Outputs reflect data available in November 2018. The latest sites can be viewed online at: https://maps.birdlife.org/marineibas

areas of aggregation at sea and determine if they warrant designation. Vessel-based observations cannot always adequately describe the at-sea distributions of seabirds, and therefore tracking data have filled this key data gap, providing an understanding of where important areas occur and when these are being used by different species and life-history stages (Lascelles et al., 2016b). A range of IBAs have been identified within the WIO (Fig. 1), including seabird IBAs (terrestrial colony locations) and marine IBAs (those entirely within the marine environment), with the latter identified via seabird extensions to colony locations (foraging radius approach; Soanes et al., 2016) or from analyzing seabird tracking data (Beal et al., 2021).

THREATS TO SEABIRDS

Seabirds face threats both when breeding on land – including predation by invasive species such as rats and cats, harvesting and human disturbance; and also when at sea – including from fisheries (Fig. 2) both through direct competition for food sources, and mortality on fishing gear (Dias et al., 2019). Seabird populations in the WIO are thought to be a fraction of the historical estimates (Danckwerts et al., 2014). Many colonies have become extinct and those that still exist are greatly reduced in size (eg, Barau's petrel *Pterodroma baraui*, Mascarene petrel *Pseudobulweria aterrima*, Tropical shearwater *Puffinus bailloni* etc; Feare et al., 2007).

A recent assessment (2019) by BirdLife International provides new insights into the major threats affecting seabirds in the WIO:

Invasive alien species (IAS)

These animals (or plants) are a major threat to seabirds globally (in terms of species affected), and also in the WIO (Russell et al., 2016). IAS are organisms that are introduced intentionally or accidentally into a new habitat out of their natural range, without the natural checks on their populations, which allows them to spread and have detrimental impacts on the ecology of the new environment (Russell and Blackburn, 2017). Rats *Rattus* spp. and cats *Felis catus* are the invasive alien species impacting the highest number of seabird species (Dias et al., 2019). Introduced mammals are present on every island group in the WIO (Russell et al., 2016). Islands are particularly vulnerable

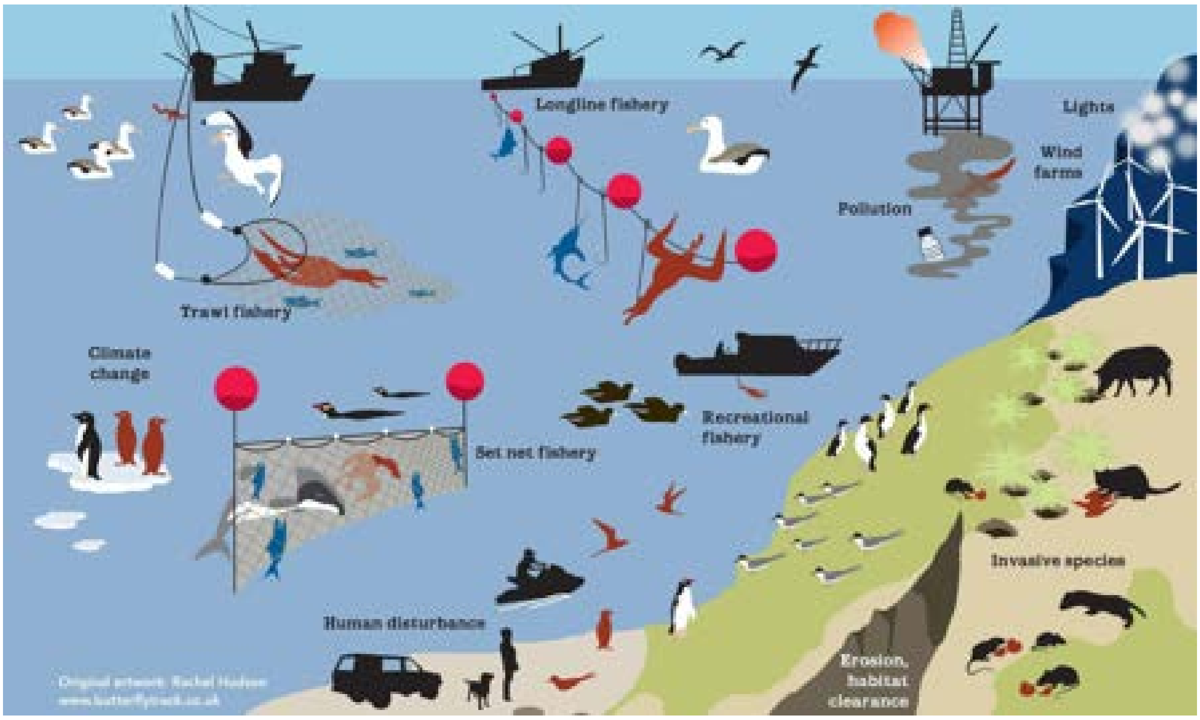


Figure 2: An illustration of some key threats to seabirds on land and at sea.

to IAS because island ecosystems tend to be less complex than mainland ones and many animal species have evolved in the absence of predators. It is estimated that 90 per cent of historical bird extinctions have occurred on islands (Whittaker and Fernandez-Palacios, 2007).

Invasive species have the biggest effect on seabirds at breeding colonies, where they can cause devastating impacts through direct predation and habitat disturbance (Croxall et al., 2012). The mainland states in the WIO are affected by IAS, including the Indian house crow (*Corvus splendens*) in Tanzania and the coastal regions of Kenya, and this species may also be impacting the recolonization of birds on Ile aux Aigrettes, Mauritius (V. Tatayah, pers. com.). However, the effects of IAS on the islands of the WIO are much greater, and multi-island states, such as those in the WIO, are even more at risk of invasions because of a lack of inter-island quarantine procedures for islands within the same state (Mauremootoo, 2003). Ten invasive species present on the WIO islands are listed by the IUCN Invasive Species Specialist Group in the top 100 of the world's worst invaders (Lowe et al., 2000). These ten species include two plants (Strawberry guava *Psidium cattleianum* and the prickly lantana *Lantana camara*), one bird (Indian myna *Acridotheres tristis*) and seven mammals (Domestic cat *Felis domesticus*, House mouse *Mus musculus*, Rat *Rattus rattus*, Pig *Sus scrofa*, Rabbit *Oryctolagus cuniculus*, Long-tailed crab-eating macaque *Macaca fascicularis*, and the Small Indian mongoose).

Bycatch in fisheries

Incidental mortality (bycatch) in fisheries is a major threat to seabirds, particularly albatross, large petrels and shearwaters, and penguins, with large-scale fisheries driving declines in twice as many species as small-scale fisheries (Dias et al., 2019). In the tropical WIO there is thought to be a low level of seabird bycatch as the species present are not generally attracted to fishing vessels (Anderson et al., 2011). But south of 25°S, bycatch has caused rapid population declines of albatross and petrels (Petersen et al., 2008; Watkins et al., 2008) and a hotspot of fisheries overlap has been identified in the south-west Indian Ocean (pelagic longline) for albatross and petrel populations (Clay et al., 2019).

Overall, there is limited information on seabird bycatch for the Indian Ocean and the data that is available is skewed towards tuna fisheries, and hampered by the aggregation of all seabirds (Pott and Wiedenfeld, 2017).

Overfishing

This is the depletion of food resources for seabirds as a consequence of human extraction (fisheries). Overfishing is the main cause of decline of 24 species, including WIO species such as the African penguin *Spheniscus demersus*, Cape Gannet *Morus capensis* and Cape Cormorant *Phalacrocorax capensis* (Dias et al., 2019).

In the WIO, annual catches of tuna have increased massively over the last 50 years, and tuna now makes up 17 per cent of the total catch throughout the WIO (FAO, 2011; Le Corre et al., 2012; BirdLife International, 2015). The tuna fisheries in the tropical WIO do not compete directly with seabirds for fish, because seabirds target epipelagic fish while the fisheries target predatory tuna and billfish (Le Corre et al., 2012). However, most tropical seabirds feed in association with predatory fish, which force their prey species to the surface (Ashmole and Ashmole, 1967), and some seabird species are 'near-obligate commensals' of suface-dwelling tunas (Au and Pitman, 1986).

If the abundance of the large tunas and billfish is decreased through overfishing, seabirds might find it hard to locate and catch their prey (Le Corre et al., 2012). For example, competition with commercial fisheries poses the greatest threat to the Wedge-tailed shearwater *Ardenna pacifica* because over-exploitation of tuna fisheries is greatly reducing prey availability (Brooke, 2004) as the shearwaters rely on tuna to drive shoals of small fish to the surface where they become available for surface-feeding (Ratcliffe, 1999; BirdLife International, 2021).

Hunting/trapping

Harvesting of seabird adults, chicks, eggs, and feathers have been important activities for some coastal communities for many centuries, but have also driven seabird declines. Hunting/trapping at colonies is the second major terrestrial threat in terms of number of species affected, and the top threat to coastal globally threatened species (Dias et al., 2019). Harvest methods have changed over time to include more efficient tools, meaning seabirds are more exposed to excessive harvesting. This is a complex issue because the meat and eggs may be important sources of protein for many communities, but a number of species declines have been attributed to unregulated harvesting and over-exploitation.

Harvesting was a major problem on Round Island, Mauritius, for 150 years with subsequent population declines in Red- and White-tailed tropic birds, and to some extent Round Island petrels (Tatayah, 2010). In Madagascar and Mozambique, harvesting remains a problem (Le Corre et al., 2022) and is likely to be unsustainable, especially the take of adults and collection of eggs. In some locations, harvesting quotas have been implemented (eg, 20 per cent of Sooty tern eggs each year; Lascelles et al., 2016a), and some colonies (eg, in Seychelles and Madagascar) are sustainably managed but this can only be achieved through rigorous monitoring of seabird breeding success and harvest levels (Feare and Doherty Jr, 2004; Le Corre et al., 2022).

Disturbance

Human intrusion into seabird colonies is increasing as recreational (including tourism), scientific and developmental demands increase. The impacts of disturbance are species-specific, with some species considerably more sensitive to human intrusion than others. For example, Round Island petrels have become skittish due to an increased frequency of intervention at nests for monitoring (V. Tatayah, pers. com.). Disturbance of seabirds at their colonies can lead to reduced breeding success or even permanent abandonment of the site. Globally threatened coastal species are particularly affected (eg, penguins, cormorants, pelicans; see Dias et al., 2019).

Unsustainable coastal development (including agricultural expansion to infrastructure development) is a major concern to conservation of birds and their habitats in the marine and coastal environment of the WIO (BirdLife International, 2015). At-sea, shipping is a cause of widespread disturbance to seabirds with moving vessels causing temporary habitat loss, collisions, and pollution (see more details in 'pollution') for foraging and resting seabirds.

Pollution

The threat from pollution is largely related to oil spills. Oil pollution is also a major threat to biodiversity in the marine and coastal zone of the WIO. A large proportion (30 to 40 per cent) of the world's oil is produced in the Middle East and most of this is exported across the Indian Ocean (BirdLife International, 2015). This increases the risk of low-level chronic oil pollution and catastrophic spills (Vethamony et al., 2007). There is a high overlap of maritime traffic and the movement patterns of birds in the southern Mozambique Channel, the south of Madagascar and the Mascarene Archipelago, putting them at a high risk of oil pollution (Le Corre et al., 2012).

Pollution in various forms is a widespread problem adversely affecting many seabirds. Oil spills, from both offshore facilities and shipping tankers, can cause mortalities that lead to population-level impacts, particularly when they occur within the most sensitive sites. Single spills have been recorded as killing up to a quarter of a million birds (García et al., 2003) and causing the loss of 7 per cent of regional populations of certain species (Piatt and Ford, 1996). In the south-west Indian Ocean, after *Kapodistrias* ran aground off Cape Recife, in South Africa's Eastern Cape province in 1985, at least 137 penguins died from oiling and 1043 oiled Jackass penguins were rescued for rehabilitation (Randall and Randall, 1986). Such events remain a risk in the region. For example, in 2020 the cargo ship MV *Wakashio* ran aground on the south-east coast of Mauritius, leaking diesel and fuel oil covering 15 km stretch of coastline (Lewis, 2020; Pasnin et al., 2020) and affecting a protected island – Ile aux Aigrettes and several other islets that are home to seabirds.

Light pollution

The widespread and ever-growing use of artificial light at night is an increasing threat to seabirds, impacting mostly gadfly petrels, large petrels/shearwaters and storm-petrels (Rodríguez et al., 2017). One of the most severe ecological consequences of light pollution is light-induced mass fatality events, for example, petrel fledglings are attracted to and disoriented by artificial lights when they leave their nests for the first time and fly towards the sea at night (Rodríguez et al., 2014; Rodríguez et al., 2017). Grounded birds are not able to take off again and they are exposed to several sources of mortality (collisions with infrastructures or vehicles, predation, dehydration or starvation).

On Réunion Island, fledgling mortality has been recorded for all nine species of breeding petrel (Le Corre et al., 2002), including the endangered Barau's Petrel *Pterodoma baraui*, the Critically Endangered Mascarene Petrel *Pseudobulweria aterrima*, and Tropical shearwaters *Puffinus bailloni* (Gineste et al., 2017). The only record of a Mascarene petrel on Mauritius is of a bird found dead under an isolated street light just outside the Black River Gorges National Park (Tatayah et al., 2011).

Climate change/severe weather

The resulting impacts from meteorological events on seabirds are mostly due to habitat shifting and alteration, and temperature extremes (Dias et al., 2019). The populations and breeding success of marine and coastal birds are influenced by various climate-related factors, which make them good indicators of ecosystem change. The impacts of climate change on these species are likely to be direct (eg, extreme weather, flooding of low-lying colonies from sea level rise); and/or indirect (eg, sea-surface temperature affecting plankton biomass and prey stock distributions). Cyclones are projected to become more frequent and more intense (Knutson et al., 2010), and such changes in cyclones could negatively affect breeding seabirds (Nicoll et al., 2017). However, because marine systems are so variable, it is difficult to reliably predict future climate change impacts on marine ecosystems (with a few exceptions: sea level rises, storm surge impacts). Given the knowledge gaps and lack of feasible solutions for projected impacts, the standard approach remains to monitor and evaluate, and establish baseline information in order to be able to detect any changes. Such monitoring is a priority because it may provide early warnings of related changes to other components of the marine system.

Seabirds are generally long-lived and can usually survive adverse short-term environmental events. For example, the large Indian Ocean tsunami of 26 December 2004 reshaped the beach of Latham Island, Tanzania, but it did not appear to have a major impact on seabird populations breeding at the island, which likely avoided excessive mortality by taking flight (Crawford et al., 2006a). However, small populations tied to restricted habitat, such as the African penguin in South Africa, may be threatened by long-term climate warming that alters the distribution patterns of their prey (Crawford et al., 2015). The behavioural, social and life history traits of seabirds may make them particularly susceptible to climate change (Grémillet and Boulinier, 2009).

Many seabirds have specialized diets, which make them vulnerable to any change in the distribution, abundance, or predictability of their prey. Many species are also highly philopatric, returning to the same site to breed even if conditions become unfavourable (Grémillet et al., 2008). Several groups of seabirds have been identified as being particularly vulnerable to climate change: including the Diomedeidae (albatrosses), Spheniscidae (penguins), Procellariidae, and Hydrobatidae (petrels and shearwaters) families (Foden et al., 2008). By contrast, the Ardeidae family (herons and egrets) have a low susceptibility to climate change (Foden et al., 2008). Although a study on the projected impacts of climate change on non-breeding albatross in the Indian Ocean found little projected change in distribution by the end of the century, this could be because of the longevity of albatross and the comparatively short-time frame of the study (Somveille et al., 2020).

Climate change could also affect the long-range migrants that frequent the East African coast because other regions have been found to be affected; for example, in Australia, arrival dates of migrants have advanced by 3.5 days per decade since 1960 (Beaumont et al., 2006). However, to date there have not been any studies in the WIO.

SEABIRDS DIVERSITY IN THE WESTERN INDIAN OCEAN

Seabird diversity is high in the WIO, but endemism is low. There are four seabirds that are endemic to the Nairobi Convention area and seven near endemic species (Table 2; Wanless, 2015).

The WIO has been divided into three broad zones based on the diversity of the seabird species they contain (BirdLife International, 2015): 1) Tropical Waters; 2) Coastal waters of Southern Africa; 3) Temperate and sub-Antarctic (Fig. 3).

1. Tropical waters
(north of ~25°S)

This zone is dominated in numbers by tropicbirds (2 spp), boobies (3 spp), frigatebirds (2 spp) and terns (>10 spp; Wanless, 2015). There are 7.4 million pairs of breeding seabirds of 31 different species in this zone (Le Corre et al., 2012; BirdLife International 2015). Most of the breeding colonies are on remote oceanic islands in the Seychelles area (3.4 million pairs), the Mozambique Channel (3 million pairs) and the Mascarene Archipelago (0.7 million pairs) (Le Corre et al., 2012; Evans et al., 2016). There are relatively few seabird colonies on the coasts of Madagascar and East Africa, with just 13 seabird species breeding on islets off the west coast of Madagascar, and two known sites on the Mozambique coast (Le Corre and Jaquemet, 2005). Important foraging areas for seabirds that breed in the WIO are thought to be south of Madagascar (the continental shelf and the Walters Shoals) and the central Indian Ocean (Le Corre and Jaquemet, 2005). The Mozambique Channel is a region of exceptionally abundant and diversified seabird communities, with the southern part of the channel an important over-wintering area for many sub-Antarctic species, including endangered albatrosses and petrels (more details in section 3: Temperate and sub-Antarctic).

Tropical waters are more homogeneous and less productive than other oceanic areas, with patches of prey available for seabirds rare and unpredictably distributed (Balance et al., 1997). In this poor environment, the presence of large schools of surface-dwelling predatory fishes or marine mammals plays a major role in the availability of prey (Jaquemet et al., 2004). To increase foraging efficiency, these birds have evolved an association with feeding subsurface predators, mainly schooling tuna and

Table 2: Endemic and near-endemic seabird species in the Western Indian Ocean.

	SPECIES	IUCN RED LIST STATUS	BREEDING LOCATIONS
ENDEMIC	Barau's petrel *Pterodroma baraui*	EN	Réunion
	Jouanin's petrel *Bulweria fallax*	NT	Socotra archipelago and islands off Oman
	Mascarene petrel *Pseudobulweria aterrima*	CR	Réunion
	Socotra cormorant *Phalacrocorax nirgrogularis*	VU	Islands in Persian Gulf and Arabian Sea
NEAR ENDEMIC	African penguin *Spheniscus demersus*	EN	Islands off coast of South Africa and Namibia
	Cape gannet *Morus capensis*	VU	Islands off coast of South Africa and Namibia
	Cape cormorant *Phalacrocorax capensis*	LC	Islands off coast of South Africa and Namibia
	Bank cormorant *Phalacrocorax neglectus*	EN	Islands off coast of South Africa and Namibia
	Crowned cormorant *Phalacrocorax coronattus*	NT	Islands off coast of South Africa and Namibia
	Hartlaub's gull *Larus hartlaubi*	LC	Islands off coast of South Africa and Namibia
	Damara tern *Sterna baleanarum*	NT	Islands off coast of South Africa and Namibia

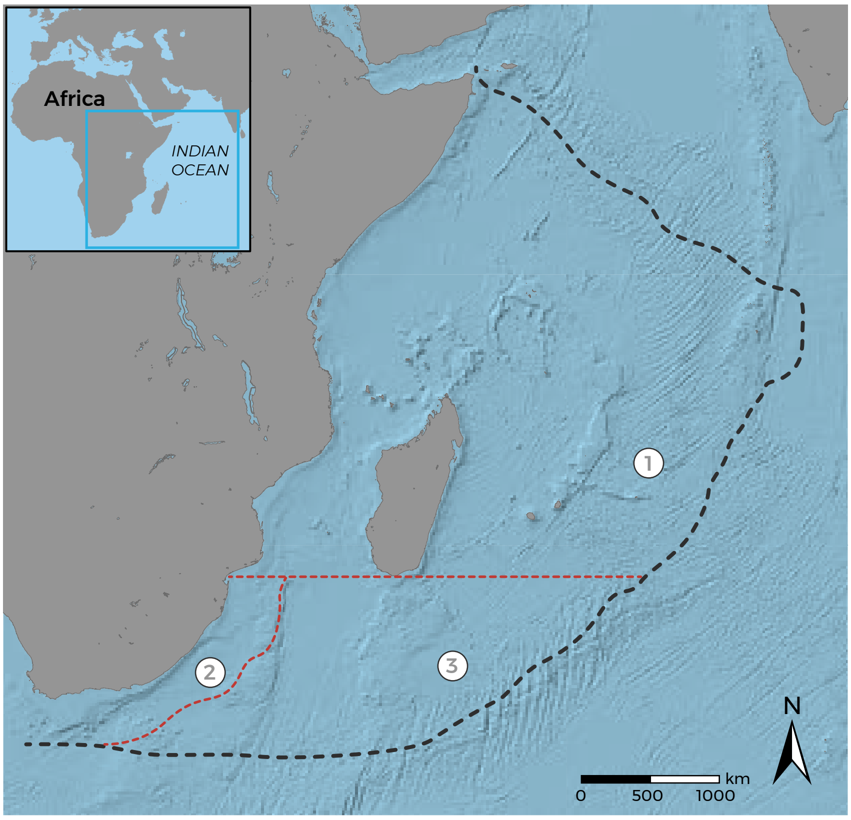


Figure 3: Schematic map of broad zones of seabird species: 1) Tropical waters; 2) Coastal waters of Southern Africa; 3) Temperate and sub-Antarctic. Grey dashed line represents area considered in this study. Note, boundaries are for illustration purposes only.

dolphins. When feeding, these predators drive forage fish to the surface making them available to seabirds feeding from above (eg, Le Corre and Jaquemet, 2005). Some seabird species are even regarded as near obligate commensals of tuna with more than 70 per cent of all feeding activity occurring in association with schools of large predatory fish (eg, Jaquemet et al., 2004). In the WIO, it has been shown that almost all feeding by terns, noddies and shearwaters occurs when surface-dwelling tunas or mammals are present (Jaquemet et al., 2004). Because of this strong association, seabirds are frequently used and have a long history of being used to detect schools of surface tuna, from artisanal fishers using binoculars to follow seabirds, to purse-seine vessels that use radar to detect seabirds to discover tunas. Three species from these waters are:

Barau's petrel

Pterodroma baraui (EN) An endemic, medium-sized, grey-and-white gadfly petrel – that nests on cliff-ledges on Réunion (one nest has been discovered on Rodriguez). The population is estimated to be 15 000 to 20 000 breeding pairs (Danckwerts et al., 2016). But as it nests in fewer than five locations, and the population is thought to be declining, it is thus classified as Endangered.

They are surface feeders, mainly feeding on fish and squids, foraging alone or in small flocks, often associating with Sooty terns *Onychoprion fuscata* and Tropical shearwaters *Puffinus bailloni*. Tracking research has shown a broad at-sea distribution when breeding, encompassing a wide area from Réunion to 1000 km south of Madagascar and up to the South African coast (Pinet et al., 2011).

They are synchronous annual breeders. Birds (both sexes of breeders and non-breeders) return to colony in September coinciding with the full moon (Pinet et al., 2009). Birds lay synchronously in November each year with adults leaving the colony in late March, and fledglings leaving throughout April. During chick rearing, birds adopt a dual foraging strategy with a clear alternation of short trips (two to three days) around Réunion Island and long trips (10 to 14 days) as far as the continental plateau of South Africa and the Walters Shoal area (a collection of seamounts), 1000 km south of Madagascar.

After breeding, the species migrates eastward to a wide area located on both sides of the Ninety East Ridge (Pinet et al., 2011) to regions characterized by warm sea surface temperatures and low productivity. There is considerable consistency in wintering areas (Pinet et al., 2011).

Sooty tern

Onychoprion fuscata (LC)

The most abundant marine seabird in all tropical waters and is widely distributed in the tropical Indian Ocean (Schreiber et al., 2002). It is a dispersive and migratory surface feeder that associates with schools of surface tunas for its food (Jaquemet et al., 2005). Within the Indian Ocean it breeds at numerous islands from 11 to 26°S. It is especially abundant in the Mozambique Channel (Le Corre and Jaquemet, 2005) and on the Seychelles Archipelago (Feare et al., 2007), where it represents 99 per cent (>3 050 000 pairs) and 83 per cent (>3 420 000 pairs) of the total number of seabird breeding pairs, respectively (Le Corre and Jaquemet, 2005). The species nests in large synchronized dense colonies. Juan de Nova in the northern Mozambique Channel is one of the largest tropical seabird colonies in the world, hosting approximately two million breeding Sooty tern pairs (Feare et al., 2007). However, of the 46 known Sooty tern colonies in the WIO, 14 have been extirpated and eight have declined in number over the last century (Feare et al., 2007). Sooty tern breeding phenology varies considerably within the south-west Indian Ocean from annual and seasonal at Europa Island (winter) and Juan de Nova Island (summer) to non-seasonal at Lys Island (Le Corre and Jaquemet, 2005). These differences in breeding phenology seem to be driven by large oceanic patterns related to the climate in the Western Indian Ocean that influence regional oceanic production and prey availability (Jaquemet et al., 2007). Tracking of Sooty terns revealed they undertake extensive post-breeding migrations, lasting on average about seven months, with average distances exceeding 50 000 km per individual (Jaeger et al., 2017), meaning individual birds fly on average about 240 km/day.

Red-footed booby

Sula sula (LC)

This is the smallest and most pelagic of the boobies and found in the tropical and subtropical regions of the Atlantic, Pacific and Indian Oceans. The species can range out to ~250 km from the colony, and parents return to feed their chicks once a day (Harrington, 1977). Red-footed boobies often hunt in large groups and feed mainly on flying-fish and squid. Prey are caught by plunge-diving, but flying fish are also taken in flight, especially when feeding in association with underwater predators. These birds mostly forage in low dynamic divergence zones, where prey are diluted near the surface, but are accessible below the sea surface by diving (Jaquemet et al., 2014). It is mostly diurnal, but also feeds on squid at night.

Tracking of Red-footed boobies from Europa found they undertook relatively short and exclusively diurnal foraging trips (Mendez et al., 2016). It often rests on boats using them as vantage points. Breeding is not seasonal in most of its range. Individuals form large colonies, nesting and roosting mainly in trees or on islets with abundant vegetation. They are highly social birds with ritualized behaviours. Red-footed boobies are sympatric with frigatebirds over most of their range (Le Corre and Jouventin, 1997). The global population is suspected to be in decline owing to habitat loss, predation by invasive species and unsustainable levels of exploitation (BirdLife International, 2021).

2. Coastal waters of Southern Africa (30°–38°S)

This zone is home to numerous near endemic seabirds: the African penguin (*Spheniscus demersus*), Cape gannet (*Morus capensis*), three cormorant species, Hartlaub's gull (*Larus hartlaubii*) and the Damara tern (*Sterna baleanarum*; Crawford et al., 2006b; BirdLife International, 2015). These waters are also important for non-endemic coastal species such as Kelp gulls (*Larus dominicanus*) and several tern species as well as northern hemisphere larids and other migratory seabirds, eg, Cory's shearwater *Calonectris borealis* that migrate here in the austral summer (Wanless, 2015). Three species from these waters, the African penguin, Cape gannet and Cory's shearwater are described in more detail below:

African penguin

Spheniscus demersus (EN)

An endemic species to the region that breeds in colonies from Hollam's Bird Island in Namibia to Bird Island (Algoa Bay, Eastern Cape) in South Africa (Crawford et al., 2011). Outside of the breeding season, the species ranges coastally between 18°S in Namibia and 29°S in KwaZulu-Natal, South Africa. There have been sightings as far north as Sette Cama in Gabon and the Limpopo River mouth in Mozambique (Hagen, 2015). The African penguin forages for small pelagic schooling fish (mainly sardines and anchovy) in the inshore waters, usually within 15 km of the coast. During breeding they can travel up to 50 km from colonies but during non-breeding they travel much further, although adults generally remain within 400 km of their breeding locality, but have been recorded up to 900 km away, and can move further offshore, especially when foraging on the Agulhas Bank off the south coast of South Africa (Petersen et al., 2006; Crawford et al., 2011). The species breeds and moults on coastal islands and protected mainland sites (Crawford et al., 2011).

The foraging strategy and breeding requirements of the species make them particularly susceptible to over-fishing and oil spills. Population declines have been attributed to

Cape gannet – Lambert's Way, South Africa. © Paul F. Donald

food shortages, resulting from shifts in the distributions of prey species, competition with commercial purse-seine fisheries and environmental fluctuations (Crawford et al., 2011; Crawford et al., 2018). There is increasing evidence that fish availability during different times of the year affects breeding success and adult survival (Crawford et al., 2011; Sherley et al., 2014). Past mortality from oil spills has been serious (Wolfaardt et al., 2009) and may increase if proposed development of harbours close to colonies proceeds, particularly because most of the population is confined to areas that are near existing or planned major shipping ports (Nel and Whittington, 2003). Potential for catastrophic large-scale oil spills is likely to increase, given developments planned along the coast of South Africa and further north. Chronic oiling from leaking wrecks, washing of ship's tanks and other sources are also a threat (Wolfaardt et al., 2009). Predation by Cape fur seals *Arctocephalus pusillus* is causing mortalities at some colonies, and seals have also displaced penguins from some breeding sites (Hagen, 2015).

Cape gannet

Morus capensis (EN)

This gannet species breeds at just six islands, with three in South Africa (Bird at Lambert's Bay, Malgas and Bird at Algoa Bay) and three in Namibia (Mercury, Ichaboe and Possession). Birds range east to KwaZulu-Natal and Mozambique and as vagrants to Tanzania (Klages, 1994). The species usually remains over the continental shelf, often within 100 km of land (Hockey et al., 2005). Outside the breeding season, adults may disperse up to ~1000 km from colonies, whereas juvenile birds may move more than 2000 km (Klages, 1994). It feeds mainly on shoaling pelagic fish, such as anchovy, sardine and saury, and individuals can travel as far as 450 km in a day in search of food.

The Cape gannet has experienced significant population declines. These have been attributed to food shortage following the collapse of the Namibian sardine fishery and an eastward displacement of epipelagic fish off South Africa (Crawford, 2007; Crawford et al., 2014; Crawford et al., 2015). Competition with purse-seine fisheries targeting small pelagic fish has been demonstrated in South Africa, with a negative impact on foraging success, adult body condition and chick growth rates (Cohen et al., 2014; Grémillet et al., 2016). When there is a lack of their natural prey, Cape gannets exploit fishery waste from trawlers, but this low-quality resource leads to reduced adult body condition and reproductive performance (Grémillet et al., 2016). Oil spills are also a serious threat, with approximately 5000 individuals oiled during an incident in 1983 (Altwegg et al., 2008). Oiling by fish oil from vessels processing fish or on-shore factories poses a chronic threat in Namibia (Du Toit and Bartlett, 2001).

Cory's shearwater
Calonectris borealis (LC)
An abundant seabird species in the Atlantic Ocean, visiting the WIO region mostly in the austral summer, during its non-breeding season. Cory's shearwaters are trans-equatorial migrants from the North Atlantic that travel more than 12 000 km every year to visit the waters of Agulhas Current and Mozambique Channel during their non-breeding season (Dias et al., 2011; Dias et al., 2012a). Cory's shearwaters are remarkably flexible in their migratory behaviour, with some individuals only visiting the area occasionally, while others return every year (Dias et al., 2011). The reason for this variability is mostly unknown, but studies show that the breeding success of the birds can play an important role in the decision to travel far or stay closer to the colony (Catry et al., 2013). It is also known that females are more likely to visit the distant wintering area (Pérez et al., 2013). Cory's shearwaters eat mostly fish and squid (Alonso et al., 2014), although their diet during the non-breeding period is still poorly known. Tracking data have shown that Cory's shearwaters are mostly diurnal foragers when foraging in the Agulhas Current, contrasting with their mostly nocturnal behaviour when wintering in the deep waters of the central Atlantic (Dias et al., 2012b). This suggests that they might take advantage of foraging in aggregation with underwater predators, as do other shearwaters occurring in the WIO (Catry et al., 2009).

3. Temperate and Sub-Antarctic (south of ~25°S)

This zone covers the pelagic waters, Sub-Antarctic and cool-temperate islands, and highly productive South African continental shelf, which are dominated by procellariform seabirds (albatrosses, petrels and allies, storm-petrels and diving-petrels) (BirdLife International, 2015). Many of these birds do not breed in the region but spend their non-breeding season here, travelling thousands of kilometres to forage in these productive waters. Three emblematic species from these waters are the Indian yellow-nosed albatross, the White-chinned petrel and the Wandering albatross, described in more detail below, with more information on these species (and others) available on the BirdLife International DataZone[1].

Indian yellow-nosed albatross
Thalassarche carteri (EN)
This is the smallest black and white albatross, which breeds on Prince Edward Island (South Africa), and on Amsterdam, Crozet Islands, Kerguelen Islands, and St Paul Islands (French Southern Territories). It is listed as Endangered based on a very rapid and ongoing decline, with the overall decline on Amsterdam Island documented at over 30 per cent between 1982-2006 (Rolland et al., 2009). It is an annual breeder, and feeds mainly on fish and squid, and less frequently on crustaceans. Colonies may partition feeding grounds during breeding (Makhado et al., 2018). Satellite-tracking showed that breeding birds from Amsterdam Island foraged up to 1500 km from the colony (Pinaud and Weimerskirch, 2007) whereas those from Prince Edward Island having older chicks fed along the Agulhas Bank (Makhado et al., 2018). Outside the breeding season, the species disperses throughout the southern Indian Ocean between 30 to 50°S, and birds are frequently observed off southern Africa and south-western Australia, extending as far as north-eastern New Zealand (BirdLife International, 2021). The population is experiencing rapid declines over its range as a result of adult mortality and poor recruitment owing to interactions with fisheries, and disease (Weimerskirch et al., 2018a). Indian yellow-nosed albatross are known to interact with fisheries across their range, including tuna longliners in subtropical waters (Weimerskirch and Jouventin, 1998) and demersal longline fisheries off the west coast off New-Zealand (Delord et al., 2014).

White-chinned petrel
Procellaria aequinoctialis (VU)
A large burrow-nesting, black petrel with a pale bill, that breeds on South Georgia (UK), Prince Edward Islands (South Africa), Crozet Islands, Kerguelen Islands (French Southern Territories), Auckland, Campbell and Antipodes Islands (New Zealand), and in small numbers in the Falkland Islands. The species forages from equatorial waters to Antarctica and is frequently found in convergence zones, where strong currents meet, and in areas of upwelling, where cold nutrient rich waters from the ocean depths rise to the surface (eg, Phillips et al., 2006).

Birds that breed on islands from South Africa and the French Southern Territories spend the non-breeding season off the coasts of South Africa and Namibia, including over the Benguela Current. However, birds that breed on the Antipodes Islands winter off Peru and Chile. The White-chinned petrel feeds on cephalopods, crustaceans and fish, and also fisheries processing waste or discarded longline baits (eg, Delord et al., 2010). Birds range widely to search for food, travelling up to 8000 km on feeding forays during the breeding season (Phillips et al., 2006). Individuals breeding at the Crozet and Kerguelen islands display a bimodal foraging strategy, conducting either short trips to the surrounding shelf or long trips ranging from subtropical waters in the north

[1] BirdLife International (2021) IUCN RedList for birds. http://data zone.birdlife.org/species/search

to Antarctic waters in the south (Catard et al., 2000). This species is particularly vulnerable to incidental mortality in longline fisheries because it forages at night (Mackley et al., 2011) and their deep-diving capabilities mean they can retrieve baited hooks at greater depths (Jiménez et al., 2012). The White-chinned petrel are the most commonly caught seabird species in the Southern Ocean, and comprise 10 per cent of pelagic and 55 per cent of demersal longline fisheries bycatch in South Africa (Petersen et al., 2007). On land, the species is also threatened by invasive species at breeding sites, including rats on Crozet, and cats on Kerguelen and Cochons Island, and formerly Marion Island. This species has a low reproductive rate, and suspected rapid population declines means this species is listed as Vulnerable (BirdLife International, 2021).

Wandering albatross
Diomedea exulans (VU)
A large albatross, with ~44 per cent of the population breeding on Prince Edward Islands (South Africa), and ~38 per cent breeding on Crozet and Kerguelen Islands (French Southern Territories). This species has a circumpolar distribution, and both breeding and non-breeding birds have very large foraging ranges; during non-breeding some individuals remain in the WIO (3000-4000 km around the colony) whereas others complete triple circumpolar navigation (Weimerskirch, 2018). It is mostly a biennial breeding species that feeds by surface-seizing. Adults feed mainly on cephalopods and fish, and often follow ships, feeding on offal and galley refuse. Non-breeding and juvenile birds remain north of 50°S between sub-Antarctic and subtropical waters with a significant proportion crossing the Indian Ocean to wintering grounds around the southern and eastern coast of Australia (Weimerskirch et al., 2014).

Satellite tracking shows that juvenile birds tend to forage further north than adults, and that females tend to forage further north than males (Weimerskirch et al., 2006). This means juveniles and female birds have a greater overlap with longline tuna fisheries and are at greater risk of incidental capture. Longline fishing is the main cause of decline in this species, reducing adult survival and juvenile recruitment, and this threat is ongoing. Fisheries were responsible for a 54 per cent decrease in numbers on the Crozet Islands between 1970 and 1986 (Weimerskirch et al., 1997).

GPS tracking indicates that nearly 80 per cent of individuals from Crozet encounter boats during their foraging trips (Weimerskirch et al., 2018b). The Crozet and Prince Edward Island populations are most vulnerable to pelagic longline fishing in the Indian Ocean and Australian region.

MECHANISMS FOR SEABIRD PROTECTION

Given their imperilled conservation status, many seabirds have been highlighted for special conservation status and action under a range of international, regional and national agreements and mechanisms. These include:

Nairobi Convention
This is a partnership between governments, civil society and the private sector. The Convention provides a regional legal framework and coordinates the efforts of the member states to plan and develop programmes that strengthen their capacity to protect, manage and develop their coastal and marine environment of the WIO. Annex II of the Convention lists species of wild flora and fauna requiring special protection, which are those contained in the IUCN Red List of Threatened Species (ie, globally threatened species).

Agreement on the Conservation of African-Eurasian Migratory Waterbirds (AEWA)
An intergovernmental treaty dedicated to the conservation of migratory waterbirds and their habitats across Africa, Europe, the Middle East, Central Asia, Greenland and the Canadian Archipelago. AEWA covers 255 species of birds ecologically dependent on wetlands for at least part of their annual cycle, including many species of divers, grebes, pelicans, gannets, cormorants, herons, storks, rails, ibises, spoonbills, flamingos, ducks, swans, geese, cranes, waders, gulls, terns, tropicbirds, auks, frigatebirds and the African penguin.

Convention on the Conservation of Migratory Species of Wild Animals (CMS)
A framework convention that provides a global platform for the conservation and sustainable use of migratory animals and their habitats. CMS brings together the States through which migratory animals pass, the Range States, and lays the legal foundation for internationally coordinated conservation measures throughout a migratory range.

Agreement on the Conservation of Albatrosses and Petrels (ACAP)
A multilateral agreement which seeks to conserve albatrosses and petrels by coordinating international activity to mitigate known threats to their populations. ACAP came into force in February 2004 and currently has 13 member countries and covers 31 species of albatrosses, petrels and shearwaters.

The Ramsar Convention (the Convention on Wetlands)
An intergovernmental treaty that provides the framework

for national action and international cooperation for the conservation and wise use of wetlands and their resources. Wetlands are important areas for many birds throughout the Nairobi Convention area, especially as feeding grounds for migratory birds.

Regional Fisheries Management Organisations (RFMOs)
These are multilateral agreements to regulate fishing on the high seas and stocks that straddle international borders. The Indian Ocean Tuna Commission (IOTC) and South West Indian Ocean Fisheries Commission (SWIOFC) operate in the WIO. RFMOs are directly or indirectly mandated to manage relevant fishing impacts on all affected species, including bycatch, not just the target species.

PRIORITY OPTIONS FOR SEABIRD CONSERVATION

Conservation actions for seabirds will depend on the context, including primary threats and species susceptibility. However, general conservation actions that are required include:

1. Site-based conservation

The IBA inventory has made a crucial contribution to identifying places in need of international protection. IBAs provide a focus for conservation action, planning and advocacy, and are recognized by the CBD as the basis of a worldwide network of priority sites for conservation (Donald et al., 2018). IBAs can be used to inform marine spatial planning and area-based conservation measures, including protected areas (both terrestrial and marine) and integrated coastal zone management (Waliczky et al., 2018). Some key sites for seabirds have been protected in the WIO, including colonies in Madagascar (Le Corre et al., 2022), a network of sites in South Africa, and the islands of the French Southern Territories, but overall the WIO has a low coverage of protection.

For site protection to be effective, it should ensure that areas are large enough to capture critical behaviour (such as key breeding sites, the marine areas around them used for maintenance and more distant feeding and aggregation sites), consider temporal and spatial variations, and have adequate regulation to minimize effects of any threats.

Management of a marine IBA does not necessarily imply advocating for bans on extractive or recreational activities and may involve promoting best practices for management threat mitigation (eg, promotion of fisheries bycatch mitigation measures, limits on visitation during breeding). Often, a case-specific approach that takes into account the species at the site and their sensitivity to various threats or activities that may occur in the site is needed, rather than a 'one-size fits all'. Conserving a network of sites (IBAs) across the WIO that are important for birds will also help protect many other species and the wider ecosystem (Brooks et al., 2001).

2. Removal or control of invasive species

The removal of predatory, alien species from areas used for seabird breeding, feeding and/or aggregation, is a crucial part of habitat and species recovery initiatives. Where eradications and/or controls of IAS have been undertaken, recoveries of seabird populations have often been rapid and dramatic (Jones et al., 2016). Mammal eradications have been attempted on 45 islands in the WIO, the majority in the Seychelles and Mauritius (See Case Study opposite), and where successful have resulted in spectacular recovery of species and ecosystems (Russell et al., 2016).

A number of larger islands are now being tackled, including plans to eradicate mice from Marion Island in 2024 by the South African government and BirdLife South Africa. Eradicating IAS can have ecosystem wide benefits, for example, eradication of rats on Chagos Archipelago benefitted the terrestrial ecosystem and also enhanced coral reef productivity and functioning by restoring seabird-derived nutrient subsidies from large areas of the ocean (Graham et al., 2018). Once IAS have been eradicated, there needs to be diligent biosecurity to prevent reinvasion (Russell et al., 2016). Native species may also need to be reintroduced to recover populations that were extirpated and restore ecosystem functioning (this is currently in progress on Ile aux Aigrettes, Mauritius). Translocations of some species to new locations have also proved an effective conservation strategy for several species (eg, Carlile et al., 2003). Additionally, the islands in the WIO would benefit from better cooperation on biosecurity. The enforcement of quarantine procedures at ports of arrival and of departure will reduce significantly the spread of invasive species (UNEP, 2012).

3. Control of unsustainable seabird harvest

A better understanding of the threat seabird harvesting poses to populations across the WIO is needed to be able to clarify conservation priorities, alongside understanding

CASE STUDY

Impact of IAS removal on seabird populations

Invasive mammals have devastated endemic island communities throughout the world, and seabirds have proven particularly vulnerable with many species extinctions. Invasive mammals have the biggest impact through direct predation, but also have indirect effects that involve unpredictable interactions (Russell and Le Corre, 2009). The eradication of invasive mammals can benefit seabird populations on islands where they nest through improving breeding success, including enhancing hatching success and/or fledging success, and even improving adult survival. Additionally, islands abandoned by certain species can be recolonized – although this is not guaranteed. Seabird recovery following the eradication of predators is influenced by complex and interacting environmental and demographic factors.

Tromelin island (15°53'S, 54°31'E) was discovered in 1722, and remained relatively undisturbed and free from introduced mammals until 1761, with the Norway rat (*Rattus norvegicus*) introduced sometime between 1857 and 1953 (Russell and Le Corre, 2009). In 2005, an island wide eradication of rats was undertaken, and the island was then surveyed eight years after eradication (detailed in Le Corre et al., 2015).

Following rat eradication the vegetation recovered from being the staple food source of rats. Red-footed and Masked booby populations increased rapidly (22–23 per cent per year), due to no predation of their chicks (Le Corre et al., 2015). Furthermore, three seabird species returned to Tromelin or bred for the first time on the island nine years after rats were eradicated: the White tern, Brown booby, and Brown noddy. The White tern had not bred on the island since 1856, and the Brown booby had never bred on the island.

The study by Le Corre et al., (2015) concluded that long-term monitoring is important for understanding seabird and ecosystem recovery after invasive species removal, and they suggested that building the cost of long-term monitoring (ten years) into conservation project budgets would help scientists understand how invasive species removal benefits seabirds, and inform selection of future restoration sites.

the subsistence needs and contributions to food security. Madagascar has seen increases in seabird populations where harvesting has been controlled either in protected areas (eg, Nosy Hara) or community protections (eg, Ambodivahie, and Red-tailed tropicbirds *P. rubricauda* at Nosy Ve; Bemanaja, 2009; Rabarisoa, 2018).

4. Integrated Coastal Zone Management (ICZM)

Birds and biodiversity should be mainstreamed in ICZM. Conservation and economic development are often presented as conflicting activities. Efforts should be made towards integrating environmental and sustainability considerations into economic development, especially in the context of marine spatial planning. This could be achieved by creating regional plans, mapping hotspots and critical habitats, and making those products available to environmental impact assessors. Also, support for mechanisms to ensure adequate enforcement of existing environmental legislation is needed. Research is needed on the economic value of marine ecosystem services in order to compare the cost of potential damage to ecosystem services with the costs and benefits of planned developments.

Residential, commercial and agricultural development along the coast is a major cause of the loss and degradation of habitats such as mangroves and coastal dunes. A set of coastal zone management guidelines would be beneficial to regulate coastal development and assist countries in managing developments so they have minimal impacts on sensitive habitats. Additionally, there is a need for a set of guidelines for environmental impact assessments for oil and gas exploration, deep-sea mining, and other developments in the coastal and marine environment. Sensitive areas should be identified (ideally as part of national and regional marine spatial planning initiatives), and attempts made to mitigate against any actual and potential detrimental activities occurring in these areas, including development of contingency plans for spills, regular training and inspection to ensure disaster readiness and early warning systems. If necessary, the

CASE STUDY

Reducing seabird bycatch

Globally, one of the biggest threats to seabirds is incidental capture in fishing gears, known as bycatch. Birds are caught and drowned on baited longline hooks, in nets, or killed by collisions with trawl cables. An estimated 300 000 seabirds are killed every year by longline and trawl vessels (Anderson et al., 2011), and a further 400 000 seabirds are estimated to die in gillnets each year (Žydelis et al., 2013). The main fishing gears responsible for seabird bycatch are: trawls, longlines, purse seines, and gillnets. There are well established and effective bycatch mitigation measures for some gear types (longline and trawl), including night-setting, bird-scaring lines, and/or line weighting. Where these have been implemented effectively, bycatch rates have been reduced by at least 80 per cent, demonstrating the scale of potential success (Maree et al., 2014; Da Rocha et al., 2021). However, for other gear types, such as gill nets, mitigation methods are still being trialled (eg, Rouxel et al., 2021).

99 per cent reduction in albatrosses killed in South African hake trawl fishery

Hake is a key species for the South African fishing industry, accounting for half of the country's annual catch. Forty-six vessels land up to 160 000 tonnes every year. In 2004, the fishery was certified by the Marine Stewardship Council (MSC) and part of the certification criteria was that the fishery must assess and reduce seabird bycatch. In 2004–5, the South African Deep Sea Trawl Industry Association (SADSTIA) investigated the scale of bycatch using onboard observations and video cameras and estimated that 9300 seabirds were killed by cable strikes each year, and that 7200 of these were globally threatened albatrosses. In 2006, the South African government introduced bycatch regulations, and the trawl vessels began working with BirdLife's Albatross Task Force (ATF) to mitigate seabird bycatch. Bird-scaring lines were tested in the fishery, and by 2010 seabird deaths had been reduced by 73–95 per cent. Seabirds are at highest risk of collisions with trawl cables when the net is being set, so in 2011 SADSTIA decided to deploy bird-scaring lines during net setting to mitigate this risk, and a year later this was made a condition of the fishing permits. After a multi-year study, the ATF demonstrated that albatross deaths had been reduced by an astounding 99 per cent across the demersal trawl fishery. This amazing improvement was all down to a simple mitigation method being adopted by all vessels in the fleet.

For further information see Maree et al., 2014; BirdLife International Marine Programme 2017.

Nairobi Convention "Protocol Concerning Cooperation in Combating Marine Pollution in Cases of Emergency in the Eastern African Region" could be amended to include oil and gas exploration, drilling and mineral extraction.

5. Reduction of bycatch

This is needed, at least to levels that do not pose a threat of species decline. For many species that occur in very low numbers and/or have low reproductive rates, this likely can only be achieved when bycatch is reduced to near zero. Other, more generic actions, such as education and awareness-raising and accompanying stakeholder involvement, are also high priorities. Where simple seabird bycatch mitigation measures have been implemented (such as night-setting, line weighting, or bird-scarring lines (see Case Study above), there is evidence that this can substantially reduce bycatch (eg, Anderson et al., 2011), including more than 95 per cent reductions in some fisheries (eg, Maree et al., 2014; Da Rocha et al., 2021).

The main tuna RFMOs now have voluntary or binding regulations in place that require the use of combinations of mitigation techniques, though their effectiveness may be hampered by a lack of monitoring and/or enforcement. Sustaining reductions through effective compliance mechanisms, ideally driven by governments, should be a priority. Electronic monitoring provides an opportunity for enforcing compliance, and minimum data standards for Electronic Monitoring Systems have been proposed for the Indian Ocean region (Murua et al., 2020; Brown et al., 2021).

SEABIRD CONSERVATION RECOMMENDATIONS

Recommendations for actions at the regional policy level

It will be important to establish clear links between the Nairobi Convention and complementary Multilateral Environmental Agreements such as the AEWA, the CMS, CBD, the Ramsar Convention, and regional fisheries bodies. Under the Nairobi Convention, the Contracting Parties agreed to the Protocol Concerning Protected Areas and Wild Fauna and Flora in the Eastern African Region. The Protocol recognizes the importance of protecting and improving the state of the wild fauna and flora of the Eastern African region as natural resources that constitute important cultural, scientific, recreational, educational and economic heritage. The Protocol urges the Contracting Parties to take the appropriate measures to "maintain essential ecological processes and life support systems, to preserve genetic diversity, and to ensure the sustainable utilization of harvested natural resources under their jurisdiction". It also lists the species of wild flora and fauna requiring special protection (Annex II), which are those contained in the IUCN Red List of Threatened Species of Wild Fauna in the WIO; and Migratory Species (Annex IV), although no seabirds are listed on Annex IV, despite many migratory seabirds within the WIO. A revised bird list for Annex II (that includes seabirds) was published in "Status of Birds in the Marine and Coastal Environment of the Nairobi Convention Area: Regional Synthesis Report" (BirdLife International, 2015). AEWA lists 68 seabirds among its 255 species of bird dependent on wetlands[2] and the CMS also lists seabird species that are protected under this Convention, mostly related to albatross and petrel species[3].

Many seabirds of the WIO cross national boundaries and spend long periods at sea, and they require regional cooperation for their effective conservation. Regional conservation actions and policies that encompass Areas Beyond National Jurisdiction (ABNJ). A new International legally binding instrument under the United Nations Convention on the Law of the Sea (UNCLOS) on the conservation and sustainable use of marine biological diversity of ABNJ are important (General Assembly resolution 72/249) is currently under negotiation, and once agreed will pave the way for conservation action in ABNJ[4].

2. (https://www.unep-aewa.org/en/species)
3. (https://www.cms.int/en/species)
4. (https://www.un.org/bbnj/)

Recommendations for actions at the national and site level

At a site level, protection and restoration of seabird colonies are a priority, and IBAs can guide conservation action (see Appendix 2, detailing all confirmed marine and coastal IBAs for each WIO country, and current level of protection).

Long-term monitoring is often overlooked, but it is essential to understand the outcomes of conservation actions, such as invasive species eradications. In addition, long-term monitoring is important to understand ecosystem change and to enable predictions of where such changes may occur so conservation actions can be developed accordingly. Seabird population dynamics and behaviour are expected to change with the projected changes in climate, and in order to be able to understand such ecosystem changes long-term data of seabird colonies, and population dynamics and spatial ecology are needed.

Tracking data can provide critical insights in seabird ecology. The more tracking data there are (species from the same colonies, different colonies, and across multiple years), the more robust will be the outcomes of the analyses to identify important sites. In the WIO region, there are tracking data gaps for the majority of colonies in East Africa and Madagascar, and also of juvenile birds. Seabirds that occur in the WIO, but do not yet have any tracking data stored in the Seabird Tracking Database are shown in Table 3.

Table 3: Species that occur in the WIO but do not yet have tracking data stored in the Seabird Tracking Database (www.seabirdtracking.org).

COMMON NAME	SCIENTIFIC NAME
Matsudaira's storm-petrel	*Hydrobates matsudairae*
Cape gannet	*Morus capensis*
Cape cormorant	*Phalacrocorax capensis*
Mascarene petrel	*Pseudobulweria aterrima*
Damara tern	*Sternula balaenarum*
Southern royal albatross *	*Diomedea epomophora*
Southern rockhopper penguin **	*Eudyptes chrysocome*
Salvin's albatross *	*Thalassarche salvini*

* No data within the Indian Ocean

** No data from the French Southern Territories

REFERENCES

Alonso, H., Granadeiro, J.P., Waap, S., Xavier, J., Symondson, W.O., Ramos, J.A. and Catry, P. (2014). An holistic ecological analysis of the diet of Cory's shearwaters using prey morphological characters and DNA barcoding. *Mol. Ecol.* 23, pp. 3719–3733

Altwegg, R., Crawford, R.J.M., Underhill, L.G. and Williams, A.J. (2008). Long-term survival of de-oiled Cape gannets Morus capensis after the Castillo de Bellver oil spill of 1983. *Biol. Conserv.* 141, pp. 1924–1929

Anderson, O.R., Small, C.J., Croxall, J.P., Dunn, E.K., Sullivan, B.J., Yates, O. and Black, A. (2011). Global seabird bycatch in longline fisheries. *Endanger. Species Res.* 14, pp. 91–106

Ashmole, N.P. and Ashmole, M.J. (1967). *Comparative feeding ecology of sea birds of a tropical oceanic island.* Peabody Museum of Natural History, Yale University New Haven, Connecticut.

Aslan, C., Holmes, N., Tershy, B., Spatz, D. and Croll, D.A. (2015). Benefits to poorly studied taxa of conservation of bird and mammal diversity on islands. *Conserv. Biol.* 29, pp. 133–142

Au, D.W. and Pitman, R.L. (1986). Seabird interactions with dolphins and tuna in the eastern tropical Pacific. *The Condor* 88, pp. 304–317

Ballance, L.T., Pitman, R.L. and Reilly, S.B. (1997). Seabird community structure along a productivity gradient: importance of competition and energetic constraint. *Ecology* 78, pp. 1502–1518

Beal, M., Oppel, S., Handley, J., Pearmain, E.J., Morera-Pujol, V., Carneiro, A.P.B., Davies, T.E., Phillips, R.A., Taylor, P.R., Miller, M.G.R., Franco, A.M.A., Catry, I., Patrício, A.R., Regalla, A., Staniland, I., Boyd, C., Catry, P. and Dias, M.P. (2021). track2KBA: An R package for identifying important sites for biodiversity from tracking data. *Methods Ecol. Evol.* 12(12), pp. 2372-2378. https://doi.org/10.1111/2041-210X.13713

Beaumont, L.J., McAllan, I.A. and Hughes, L. (2006). A matter of timing: changes in the first date of arrival and last date of departure of Australian migratory birds. *Glob. Chang. Biol.* 12, pp. 1339–1354

Bemanaja, E. (2009). Les colonies d'oiseaux marins des futures aires protégées marines et côtières du sud-ouest, de l'ouest et du nord de Madagascar: diagnostic écologique et socio-économique pour une exploitation durable de la ressource. Thèse de Doctorat en océanologie appliquée, IH-SM, Université de Toliara

BirdLife International (2015). Status of Birds in the Marine and Coastal Environment of the Nairobi Convention Area: Regional Synthesis Report (eds. C. Moseley, A. Ajagbe and J. Bosire), pp. 124. United Nations Environment Programme/Nairobi Convention Secretariat, Nairobi, Kenya.

BirdLife International (2021). IUCN RedList for birds. Downloaded from http://www.birdlife.org.

BirdLife International Marine Programme (2017). Towards seabird-safe fisheries: global efforts and solutions. pp. 21. BirdLife International Cambridge UK

Brooke, M.D.L. (2004). *Albatrosses and Petrels Across the World.* Oxford University Press, Oxford, UK

Brooks, T., Balmford, A., Burgess, N., Hansen, L.A., Moore, J., Rahbek, C., Williams, P., Bennun, L., Byaruhanga, A. and Kasoma, P. (2001). Conservation priorities for birds and biodiversity: do East African Important Bird Areas represent species diversity in other terrestrial vertebrate groups. *Ostrich* 15, pp. 3–12

Brown, C.J., Desbiens, A., Campbell, M.D., Game, E.T., Gilman, E., Hamilton, R.J., Heberer, C., Itano, D. and Pollock, K. (2021). Electronic monitoring for improved accountability in western Pacific tuna longline fisheries. *Mar. Policy* 132, p. 104664

Burger, A.E. and Shaffer, S.A. (2008). Application of tracking and data-logging technology in research and conservation of seabirds. *The Auk* 125, pp. 253–264

Carlile, N., Priddel, D., Zino, F., Natividad, C. and Wingate, D.B. (2003). A review of four successful recovery programmes for threatened sub-tropical petrels. *Mar. Ornithol.* 31, pp. 185–192

Carneiro, A.P.B., Pearmain, E.J., Oppel, S., Clay, T.A., Phillips, R.A., Bonnet-Lebrun, A.-S., Wanless, R.M., Abraham, E., Richard, Y., Rice, J., Handley, J., Davies, T.E., Dilley, B.J., Ryan, P.G., Small, C., Arata, J., Arnould, J.P.Y., Bell, E., Bugoni, L., Campioni, L., Catry, P., Cleeland, J., Deppe, L., Elliott, G., Freeman, A., González-Solís, J., Granadeiro, J.P., Grémillet, D., Landers, T.J., Makhado, A., Nel, D., Nicholls, D.G., Rexer-Huber, K., Robertson, C.J.R., Sagar, P.M., Scofield, P., Stahl, J.-C., Stanworth, A., Stevens, K.L., Trathan, P.N., Thompson, D.R., Torres, L., Walker, K., Waugh, S.M., Weimerskirch, H. and Dias, M.P. (2020). A framework for mapping the distribution of seabirds by integrating tracking, demography and phenology. *J. Appl. Ecol.* 57, pp. 514–525

Catard, A., Weimerskirch, H. and Cherel, Y. (2000). Exploitation of distant Antarctic waters and close shelf-break waters by white-chinned petrels rearing chicks. *Mar. Ecol. Prog. Ser.* 194, pp. 249–261

Catry, P., Dias, M.P., Phillips, R.A. and Granadeiro, J.P. (2013). Carry-over effects from breeding modulate the annual cycle of a long-distance migrant: an experimental demonstration. *Ecology* 94, pp. 1230–1235

Catry, T., Ramos, J.A., Le Corre, M. and Phillips, R.A. (2009). Movements, at-sea distribution and behaviour of a tropical pelagic seabird: the wedge-tailed shearwater in the western Indian Ocean. *Mar. Ecol. Prog. Ser.* 391, pp. 231–242

Clay, T.A., Small, C., Tuck, G.N., Pardo, D., Carneiro, A.P.B., Wood, A.G., Croxall, J.P., Crossin, G.T. and Phillips, R.A. (2019). A comprehensive large-scale assessment of fisheries bycatch risk to threatened seabird populations. *J. Appl. Ecol.* 56, pp. 1882–1893

Cohen, L.A., Pichegru, L., Grémillet, D., Coetzee, J., Upfold, L. and Ryan, P.G. (2014). Changes in prey availability impact the foraging behaviour and fitness of Cape gannets over a decade. *Mar. Ecol. Prog. Ser.* 505, pp. 281–293

Crawford, R., Altwegg, R., Barham, B., Barham, P., Durant, J., Dyer, B., Geldenhuys, D., Makhado, A., Pichegru, L. and Ryan, P. (2011). Collapse of South Africa's penguins in the early 21st century. *Afr. J. Mar. Sci.* 33, pp. 139–156

Crawford, R., Asseid, B., Dyer, B., Hija, A., Mwinyi, A., Shinula, P. and Upfold, L. (2006a). The status of seabirds at Latham Island, Tanzania. *Afr. J. Mar. Sci.* 28, pp. 99–108

Crawford, R., Goya, E., Roux, J. and Zavalaga, C. (2006b). Comparison of assemblages and some life-history traits of seabirds in the Humboldt and Benguela systems. *Afr. J. Mar. Sci.* 28, pp. 553–560

Crawford, R.J. (2007). Food, fishing and seabirds in the Benguela upwelling system. *J. Ornithol.* 148, pp. 253–260

Crawford, R.J., Makhado, A.B. and Oosthuizen, W.H. (2018). Bottom-up and top-down control of the Benguela ecosystem's seabirds. *J. Mar. Syst.* 188, pp. 133–141

Crawford, R.J., Makhado, A.B., Waller, L.J. and Whittington, P.A. (2014). Winners and losers–responses to recent environmental change by South African seabirds that compete with purse-seine fisheries for food. *Ostrich* 85, pp. 111–117

Crawford, R.J., Makhado, A.B., Whittington, P.A., Randall, R.M., Oosthuizen, W.H. and Waller, L.J. (2015). A changing distribution of seabirds in South Africa—the possible impact of climate and its consequences. *Front. Ecol. Evol.* 3, p. 10

Croxall, J.P., Butchart, S.H., Lascelles, B., Stattersfield, A.J., Sullivan, B., Symes, A. and Taylor, P. (2012). Seabird conservation status, threats and priority actions: a global assessment. *Bird Conserv. Int.* 22, pp. 1–34

Da Rocha, N., Oppel, S., Prince, S., Matjila, S., Shaanika, T.M., Naomab, C., Yates, O., Paterson, J.R., Shimooshili, K. and Frans, E. (2021). Reduction in seabird mortality in Namibian fisheries following the introduction of bycatch regulation. *Biol. Conserv.* 253, p. 108915

Danckwerts, D., McQuaid, C., Connan, M., Smale, M., Le Corre, M., Humeau, L., Kaehler, S., Juhasz, C., Orlowski, S. and Tourmetz, J. (2016). Intra-annual variation in the foraging ecology of the endangered endemic Barau's Petrel (Pterodroma baraui) from Réunion Island, south-western Indian Ocean: insights from a multifaceted approach. *Mar. Biol.* 163, p. 18

Danckwerts, D.K., McQuaid, C.D., Jaeger, A., McGregor, G.K., Dwight, R., Le Corre, M. and Jaquemet, S. (2014). Biomass consumption by breeding seabirds in the western Indian Ocean: indirect interactions with fisheries and implications for management. *ICES J. Mar. Sci.* 71, pp. 2589–2598

Delord, K., Barbraud, C., Bost, C., Cherel, Y., Guinet, C. and Weimerskirch, H. (2014). Atlas of top predators from french southern territories in the Southern Indian Ocean. CNRS.

Delord, K., Cotté, C., Péron, C., Marteau, C., Pruvost, P., Gasco, N., Duhamel, G., Cherel, Y. and Weimerskirch, H. (2010). At-sea distribution and diet of an endangered top predator: relationship between white-chinned petrels and commercial longline fisheries. *Endanger. Species Res.* 13, pp. 1–16

Dias, M., Granadeiro, J. and Catry, P. (2012a). Do Seabirds Differ from Other Migrants in Their Travel Arrangements? On Route Strategies of Cory's Shearwater during Its Trans-Equatorial Journey. *PLOS ONE 7*, e49376

Dias, M.P., Granadeiro, J.P. and Catry, P. (2012b). Working the day or the night shift? Foraging schedules of Cory's shearwaters vary according to marine habitat. *Mar. Ecol. Prog. Ser.* 467, pp. 245–252

Dias, M.P., Granadeiro, J.P., Phillips, R.A., Alonso, H. and Catry, P. (2011). Breaking the routine: individual Cory's shearwaters shift winter destinations between hemispheres and across ocean basins. *Proc. R. Soc. B: Biol. Sci.* 278, pp. 1786–1793

Dias, M.P., Martin, R., Pearmain, E.J., Burfield, I.J., Small, C., Phillips, R.A., Yates, O., Lascelles, B., Borboroglu, P.G. and Croxall, J.P. (2019). Threats to seabirds: a global assessment. *Biol. Conserv.* 237, pp. 525–537

Donald, P.F., Fishpool, L.D.C., Ajagbe, A., Bennun, L.A., Bunting, G., Burfield, I.J., Butchart, S.H.M., Capellan, S., Crosby, M.J. and Dias, M.P. (2018). Important Bird and Biodiversity Areas (IBAs): the development and characteristics of a global inventory of key sites for biodiversity. *Bird Conservation International*, pp. 1–22

Du Toit, M. and Bartlett, P.A. (2001). 'Soaked' Cape Gannets at Ichaboe Island, Namibia. *Bird Numbers*, 10, pp. 8–9

Einoder, L.D. (2009). A review of the use of seabirds as indicators in fisheries and ecosystem management. *Fish. Res.* 95, pp. 6–13

Evans, S.W., Cole, N., Kylin, H., Choong Kwet Yive, N.S., Tatayah, V., Merven, J. and Bouwman, H. (2016). Protection of marine birds and turtles at St Brandon's Rock, Indian Ocean, requires conservation of the entire atoll. *Afr. J. Mar. Sci.* 38, pp. 317–327

FAO (2011). Review of the state of world marine fishery resources. *FAO Fisheries and Aquaculture Technical Paper No. 569*, pp. 334. Food and Agriculture Organisation (FAO), Rome

Feare, C.J. and Doherty Jr, P.F. (2004). Survival estimates of adult sooty terns Sterna fuscata from Bird Island, *Seychelles. Ibis* 146, pp. 475–480

Feare, C.J., Jaquemet, S. and Le Corre, M. (2007). An inventory of Sooty Terns (Sterna fuscata) in the western Indian Ocean with special reference to threats and trends. *Ostrich-J. of Afr. Ornithology* 78, pp. 423–434

Foden, W., Mace, G., Vié, J.-C., Angulo, A., Butchart, S., DeVantier, L., Dublin, H., Gutsche, A., Stuart, S. and Turak, E. (2008). Species susceptibility to climate change impacts. *Wildlife in a changing world: an analysis of the 2008 IUCN Red List of threatened species* (eds. J.-C. Vié, C. Hilton-Taylor and S.N. Stuart), pp. 77–87. IUCN, Gland, Switzerland

Forero, M.G., Bortolotti, G.R., Hobson, K.A., Donazar, J.A., Bertelloti, M. and Blanco, G. (2004). High trophic overlap within the seabird community of Argentinean Patagonia: a multiscale approach. *J. Anim. Ecol.* 73, pp. 789–801

García, L., Viada, C., Moreno-Opo, R., Carboneras, C., Alcade, A. and Gonzalez, F. (2003). Impacto de la marea negra del "Prestige" sobre las aves marinas. SEO/BirdLife, Madrid, Spain

Gineste, B., Souquet, M., Couzi, F.-X., Giloux, Y., Philippe, J.-S., Hoarau, C., Tourmetz, J., Potin, G. and Le Corre, M. (2017). Tropical Shearwater population stability at Reunion Island, despite light pollution. *J. Ornithol.* 158, pp. 385–394

Graham, N.A.J., Wilson, S.K., Carr, P., Hoey, A.S., Jennings, S. and MacNeil, M.A. (2018). Seabirds enhance coral reef productivity and functioning in the absence of invasive rats. *Nature* 559, pp. 250–253

Grémillet, D. and Boulinier, T. (2009). Spatial ecology and conservation of seabirds facing global climate change: a review. *Mar. Ecol. Prog. Ser.* 391, pp. 121–137

Grémillet, D., Péron, C., Kato, A., Amélineau, F., Ropert-Coudert, Y., Ryan, P.G. and Pichegru, L. (2016). Starving seabirds: unprofitable foraging and its fitness consequences in Cape gannets competing with fisheries in the Benguela upwelling ecosystem. *Mar. Biol.* 163, p. 35

Grémillet, D., Pichegru, L., Kuntz, G., Woakes, A.G., Wilkinson, S., Crawford, R.J. and Ryan, P.G. (2008). A junk-food hypothesis for gannets feeding on fishery waste. *Proc. R. Soc. B: Biol. Sci.* 275, pp. 1149–1156

Hagen, C. (2015). African Penguin Spheniscus demersus. In T*he Eskom Red Data Book of Birds of South Africa, Lesotho and Swaziland* (eds. M.R. Taylor, F. Peacock and R.W. Wanless), pp. 146–148. BirdLife South Africa, Johannesburg, South Africa

Harrington, B.A. (1977). Winter distribution of juvenile and older Red-footed Boobies from the Hawaiian Islands. *The Condor* 79, pp. 87–90

Hebshi, A.J., Duffy, D.C. and Hyrenbach, K.D. (2008). Associations between seabirds and subsurface predators around Oahu, Hawaii. *Aquat. Biol.* 4, pp. 89–98

Hockey, P.A.R., Dean, W.R.J. and Ryan, P.G. (2005). *Roberts birds of southern Africa.* John Voelcker Bird Book Fund, Cape Town

Hussey, N.E., Kessel, S.T., Aarestrup, K., Cooke, S.J., Cowley, P.D., Fisk, A.T., Harcourt, R.G., Holland, K.N., Iverson, S.J. and Kocik, J.F. (2015). Aquatic animal telemetry: a panoramic window into the underwater world. *Science* 348, 1255642

Jaeger, A., Feare, C.J., Summers, R.W., Lebarbenchon, C., Larose, C.S. and Le Corre, M. (2017). Geolocation Reveals Year-Round at-Sea Distribution and Activity of a Superabundant Tropical Seabird, the Sooty Tern Onychoprion fuscatus. *Front. Mar. Sci.* 4

Jaquemet, S., Le Corre, M., Marsac, F., Potier, M. and Weimerskirch, H. (2005). Foraging habitats of the seabird community of Europa Island (Mozambique Channel). *Mar. Biol.* 147, pp. 573–582

Jaquemet, S., Le Corre, M. and Quartly, G. (2007). Ocean control of the breeding regime of the sooty tern in the southwest Indian Ocean. *Deep-Sea Res. I: Oceanogr. Res. Pap.* 54, pp. 130–142

Jaquemet, S., Le Corre, M. and Weimerskirch, H. (2004). Seabird community structure in a coastal tropical environment: importance of natural factors and fish aggregating devices (FADs). *Mar. Ecol. Prog. Ser.* 268, pp. 281–292

Jaquemet, S., Ternon, J.F., Kaehler, S., Thiebot, J.B., Dyer, B., Bemanaja, E., Marteau, C. and Le Corre, M. (2014). Contrasted structuring effects of mesoscale features on the seabird community in the Mozambique Channel. *Deep Sea Res. Part II Top. Stud. Oceanogr.* 100, pp. 200–211

Jiménez, S., Domingo, A., Abreu, M. and Brazeiro, A. (2012). Bycatch susceptibility in pelagic longline fisheries: are albatrosses affected by the diving behaviour of medium-sized petrels? *Aquat. Conserv.* 22, pp. 436–445

Jones, H.P., Holmes, N.D., Butchart, S.H., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D.P., Bonnaud, E. and Burbidge, A.A. (2016). Invasive mammal eradication on islands results in substantial conservation gains. *Proc. Natl. Acad. Sci.* 113, pp. 4033–4038

Klages, N.T. (1994). Dispersal and site fidelity of Cape gannets Morus capensis. *Ostrich* 65, pp. 218–224

Knutson, T.R., McBride, J.L., Chan, J., Emanuel, K., Holland, G., Landsea, C., Held, I., Kossin, J.P., Srivastava, A.K. and Sugi, M. (2010). Tropical cyclones and climate change. *Nature Geoscience* 3, pp. 157–163

Lascelles, B., Rice, J., Sato, M., Tarzia, M. and Wanless, R. (2016a). *The First Global Integrated Marine Assessment – World Ocean Assessment I – Chapter 38. Seabirds*

Lascelles, B.G., Langham, G.M., Ronconi, R.A. and Reid, J.B. (2012). From hotspots to site protection: Identifying Marine Protected Areas for seabirds around the globe. *Biol. Conserv.* 156, pp. 5–14

Lascelles, B.G., Taylor, P., Miller, M., Dias, M., Oppel, S., Torres, L., Hedd, A., Le Corre, M., Phillips, R. and Shaffer, S.A. (2016b). Applying global criteria to tracking data to define important areas for marine conservation. *Divers. Distrib.* 22, pp. 422–431

Le Corre, M., Bemanaja, E., MBelomanana, A., Jaeger, A., Rabarisoa, R., Andriatahina, L. and Davies, T.E. (2022). The seabirds of Madagascar: population status, threats and conservation. *The new natural history of Madagascar* (ed. S. Goodman). Princeton University Press

Le Corre, M., Danckwerts, D., Ringler, D., Bastien, M., Orlowski, S., Rubio, C.M., Pinaud, D. and Micol, T. (2015). Seabird recovery and vegetation dynamics after Norway rat eradication at Tromelin Island, western Indian Ocean. *Biol. Conserv.* 185, pp. 85–94

Le Corre, M., Jaeger, A., Pinet, P., Kappes, M.A., Weimerskirch, H., Catry, T., Ramos, J.A., Russell, J.C., Shah, N. and Jaquemet, S. (2012). Tracking seabirds to identify potential Marine Protected Areas in the tropical western Indian Ocean. *Biol. Conserv*. 156, pp. 83–93

Le Corre, M. and Jaquemet, S. (2005). Assessment of the seabird community of the Mozambique Channel and its potential use as an indicator of tuna abundance. *Estuar. Coast. Shelf Sci.* 63, pp. 421–428.

Le Corre, M. and Jouventin, P. (1997). Kleptoparasitism in tropical seabirds: vulnerability and avoidance responses of a host species, the red-footed booby. *The Condor 99*, pp. 162–168

Le Corre, M., Ollivier, A., Ribes, S. and Jouventin, P. (2002). Light-induced mortality of petrels: a 4-year study from Réunion Island (Indian Ocean). *Biol. Conserv.* 105, pp. 93–102

Lewis, D. (2020). How Mauritius is cleaning up after major oil spill in biodiversity hotspot. *Nature* 585, pp. 172–173

Lowe, S., Browne, M., Boudjelas, S. and De Poorter, M. (2000). *100 of the world's worst invasive alien species: a selection from the global invasive species database*. Invasive Species Specialist Group Auckland

Mackley, E.K., Phillips, R.A., Silk, J.R., Wakefield, E.D., Afanasyev, V. and Furness, R.W. (2011). At-sea activity patterns of breeding and nonbreeding white-chinned petrels Procellaria aequinoctialis from South Georgia. *Mar. Biol.* 158, pp. 429–438

Makhado, A., Crawford, R., Dias, M., Dyer, B., Lamont, T., Pistorius, P., Ryan, P., Upfold, L., Weimerskirch, H. and Reisinger, R. (2018). Foraging behaviour and habitat use by Indian Yellow-nosed Albatrosses (Thalassarche carteri) breeding at Prince Edward Island. *Emu-Austral Ornit.* 118, pp. 353–362

Maree, B.A., Wanless, R.M., Fairweather, T., Sullivan, B. and Yates, O. (2014). Significant reductions in mortality of threatened seabirds in a South African trawl fishery. *Animal Conserv.* 17, pp. 520–529

Mauremootoo, J.R. (2003). Regional Workshop on Invasive Alien Species and Terrestrial Ecosystem Rehabilitation for Western Indian Ocean Island states – Sharing Experience, Identifying Priorities and Defining Joint Action. Indian Ocean Commission Regional Environment Programme, Mauritius

Mendez, L., Cotté, C., Prudor, A. and Weimerskirch, H. (2016). Variability in foraging behaviour of red-footed boobies nesting on Europa Island. *Acta Oecol.* 72, pp. 87–97

Murua, H., Fiorellato, F., Ruiz, J., Chassot, E. and Restrepo, V. (2020). Minimum standards for designing and implementing Electronic Monitoring systems in Indian Ocean tuna fisheries. *IOTC – 23rd Meeting of the Scientific Committee.* Indian Ocean Tuna Commission, online

Nel, D.C. and Whittington, P. (2003). *Rehabilitation of oiled African Penguins: a conservation success story*. BirdLife South Africa

Nicoll, M.A., Nevoux, M., Jones, C.G., Ratcliffe, N., Ruhomaun, K., Tatayah, V. and Norris, K. (2017). Contrasting effects of tropical cyclones on the annual survival of a pelagic seabird in the Indian Ocean. *Glob. Chang. Biol.* 23, pp. 550–565

Osieck, E.R. (2004). Towards the identification of marine IBAs in the EU: an exploration by the Birds and Habitat Directives Task Force. BirdLife International, Cambridge, UK

Parsons, M., Mitchell, I., Butler, A., Ratcliffe, N., Frederiksen, M., Foster, S. and Reid, J.B. (2008). Seabirds as indicators of the marine environment. *ICES J. Mar. Sci.* 65, p. 1520–1526

Pasnin, O., Sunassee, S., Tatayah, V., Turner, A. and Ward, C. (2020). What will be the environmental impact of the Mauritius oil spill? *African Arguments*. https://africanarguments.org/2020/09

Pérez, C., Granadeiro, J.P., Dias, M.P., Alonso, H. and Catry, P. (2013). When males are more inclined to stay at home: insights into the partial migration of a pelagic seabird provided by geolocators and isotopes. *Behav. Ecol.* 25, pp. 313–319

Petersen, S., Nel, D. and Omardien, A. (2007). Towards an Ecosystem Approach to Longline Fisheries in the Benguela: an assessment of impacts on seabirds, sea turtles and sharks. FAO, Rome, Italy

Petersen, S.L., Honig, M.B., Ryan, P.G. and Underhill, L.G. (2008). Seabird bycatch in the pelagic longline fishery off southern Africa. In *Understanding and Mitigating Vulnerable Bycatch in southern African Trawl and Longline Fisheries* (eds. S.L. Petersen, D.C. Nel, P.G. Ryan and L.G. Underhill). WWF South Africa

Petersen, S.L., Ryan, P.G. and Gremillet, D. (2006). Is food availability limiting African Penguins Spheniscus demersus at Boulders? A comparison of foraging effort at mainland and island colonies. *Ibis* 148, pp. 14–26

Phillips, R.A., Silk, J.R., Croxall, J.P. and Afanasyev, V. (2006). Year-round distribution of white-chinned petrels from South Georgia: relationships with oceanography and fisheries. *Biol. Conserv.* 129, pp. 336–347

Piatt, J.F. and Ford, R.G. (1996). How Many Seabirds Were Killed by the Erron Valdez Oil Spill? *American Fisheries Society Symposium*, pp. 712–719

Piatt, J.F., Sydeman, W.J. and Wiese, F. (2007). Introduction: seabirds as indicators of marine ecosystems. *Mar. Ecol. Prog. Ser*. 352, pp. 199–204

Pinaud, D. and Weimerskirch, H. (2007). At-sea distribution and scale-dependent foraging behaviour of petrels and albatrosses: a comparative study. *J. Anim. Ecol.* 76, pp. 9–19

Pinet, P., Jaquemet, S., Pinaud, D., Weimerskirch, H., Phillips, R.A. and Le Corre, M. (2011). Migration, wintering distribution and habitat use of an endangered tropical seabird, Barau's petrel Pterodroma baraui. *Mar. Ecol. Prog. Ser*. 423, pp. 291–302

Pinet, P., Salamolard, M., Probst, J.-M., Russell, J.C., Jaquemet, S. and Le Corre, M. (2009). Barau's petrel (Pterodroma baraui): history, biology and conservation of an endangered endemic petrel. *Mar. Ornithol.* 37, pp. 107–113

Pott, C. and Wiedenfeld, D.A. (2017). Information gaps limit our understanding of seabird bycatch in global fisheries. *Biol. Conserv.* 210, pp. 192–204

Rabarisoa, R. (2018). New observation on the breeding of the crested tern (Thalasseus bergii) in Madagascar. Indian Ocean Seabird Group Newsletter

Randall, R. and Randall, B. (1986). The Kapodistrias affair-another oiling incident affecting seabirds. *Bokmakierie* 38, pp. 37–40

Ratcliffe, N. (1999) Seabirds on Ascension Island. *World Birdwatch* 21, pp. 16–18

Rodríguez, A., Burgan, G., Dann, P., Jessop, R., Negro, J.J. and Chiaradia, A. (2014). Fatal attraction of short-tailed shearwaters to artificial lights. *PLOS ONE 9*, e110114

Rodríguez, A., Holmes, N.D., Ryan, P.G., Wilson, K.J., Faulquier, L., Murillo, Y., Raine, A.F., Penniman, J.F., Neves, V. and Rodríguez, B. (2017). Seabird mortality induced by land-based artificial lights. *Conserv. Biol.* 31, pp. 986–1001

Rolland, V., Barbraud, C. and Weimerskirch, H. (2009). Assessing the impact of fisheries, climate and disease on the dynamics of the Indian yellow-nosed albatross. *Biol. Conserv*. 142, pp. 1084–1095

Rouxel, Y., Crawford, R., Cleasby, I.R., Kibel, P., Owen, E., Volke, V., Schnell, A.K. and Oppel, S. (2021). Buoys with looming eyes deter seaducks and could potentially reduce seabird bycatch in gillnets. *R. Soc. Open Sci*. 8, 210225

Russell, J.C. and Blackburn, T.M. (2017). The rise of invasive species denialism. *Trends Ecol. Evol*. 32, pp. 3–6

Russell, J.C., Cole, N.C., Zuël, N. and Rocamora, G. (2016). Introduced mammals on western Indian Ocean islands. *Glob. Ecol. Conserv.* 6, pp. 132–144

Russell, J.C. and Le Corre, M. (2009). Introduced mammal impacts on seabirds in the Îles Éparses, Western Indian Ocean. *Mar. Ornithol.* 37, pp. 121–128

Schreiber, E.A. and Burger, J. (2001). *Biology of marine birds*. CRC Press

Schreiber, E.A., Feare, C.J., Harrington, B.A., Murray, B.G., Robertson, W., Robertson, M.J. and Woolfenden, G.E. (2002). *Sooty tern (Onychoprion fuscatus).* Cornell Lab of Ornithology, Ithaca, NY, USA

Sherley, R.B., Abadi, F., Ludynia, K., Barham, B.J., Clark, A.E. and Altwegg, R. (2014). Age-specific survival and movement among major African Penguin Spheniscus demersus colonies. *Ibis* 156, pp. 716–728

Soanes, L.M., Bright, J.A., Angel, L.P., Arnould, J.P.Y., Bolton, M., Berlincourt, M., Lascelles, B., Owen, E., Simon-Bouhet, B. and Green, J.A. (2016). Defining marine important bird areas: Testing the foraging radius approach. *Biol. Conserv.* 196, pp. 69–79

Somveille, M., Dias, M.P., Weimerskirch, H. and Davies, T.E. (2020). Projected migrations of southern Indian Ocean albatrosses as a response to climate change. *Ecography* 43, pp. 1683–1691

Tatayah, R. (2010). The breeding biology of the Round Island Petrel (Pterodroma arminjoniana) and factors determining breeding success. Unpublished PhD Thesis, University of Mauritius, Moka, Mauritius

Tatayah, R., Jones, C., Birch, D. and Salamolard, M. (2011). first record of Réunion Black Petrel Pseudobulweria aterrima on Mauritius. *Bull. Br. Ornithol. Club* 131, pp. 64–66

UNEP (2012). Status of Birds in the Marine and Coastal Environment of the Nairobi Convention Area: Regional Synthesis Report. pp. 128. Nairobi, Kenya

van der Schyff, V., Yive, N.S.C.K., Polder, A., Cole, N.C., Tatayah, V., Kylin, H. and Bouwman, H. (2021). Persistent organic pollutants in sea bird eggs from the Indian Ocean's Mascarene Basin. *Sci. Total Environ*. 771, p. 145348

Vethamony, P., Sudheesh, K., Babu, M., Jayakumar, S., Manimurali, R., Saran, A., Sharma, L., Rajan, B. and Srivastava, M. (2007). Trajectory of an oil spill off Goa, eastern Arabian Sea: Field observations and simulations. *Environ. Pollut.* 148, pp. 438–444

Vié, J.-C., Hilton-Taylor, C. and Stuart, S.N. (2009). *Wildlife in a changing world: an analysis of the 2008 IUCN Red List of threatened species*. IUCN

Waliczky, Z., Fishpool, L.D.C., Butchart, S.H.M., Thomas, D., Heath, M.F., Hazin, C., Donald, P.F., Kowalska, A., Dias, M.P. and Allinson, T.S.M. (2018) Important Bird and Biodiversity Areas (IBAs): their impact on conservation policy, advocacy and action. *Bird Conserv. Int*. 29, pp. 199–215

Wanless, R.M. (2015). Seabirds; a review of status, distribution and interaction with fisheries in the Southwest Indian Ocean. In *Offshore fisheries of the Southwest Indian Ocean: their status and the impact on vulnerable species* (eds. R.P. Van der Elst and B.I. Everett), pp. 351–364. Oceanographic Research Institute

Watkins, B., Petersen, S. and Ryan, P. (2008). Interactions between seabirds and deep-water hake trawl gear: an assessment of impacts in South African waters. *Anim. Conserv*. 11, pp. 247–254

Weimerskirch, H. (2018). Linking demographic processes and foraging ecology in wandering albatross—Conservation implications. *J. Anim. Ecol*. 87, pp. 945–955

Weimerskirch, H., Åkesson, S. and Pinaud, D. (2006). Postnatal dispersal of wandering albatrosses Diomedea exulans: implications for the conservation of the species. *J. Avian Biol.* 37, pp. 23–28

Weimerskirch, H., Brothers, N. and Jouventin, P. (1997). Population dynamics of wandering albatross Diomedea exulans and Amsterdam albatross D. amsterdamensis in the Indian Ocean and their relationships with long-line fisheries: conservation implications. *Biol. Conserv*. 79, pp. 257–270

Weimerskirch, H., Cherel, Y., Delord, K., Jaeger, A., Patrick, S.C. and Riotte-Lambert, L. (2014). Lifetime foraging patterns of the wandering albatross: life on the move! *J. Exp. Mar. Biol. Ecol*. 450, pp. 68–78

Weimerskirch, H., Delord, K., Barbraud, C., Le Bouard, F., Ryan, P.G., Fretwell, P. and Marteau, C. (2018a). Status and trends of albatrosses in the French southern territories, western Indian Ocean. *Polar Biol.* 41, pp. 1963–1972

Weimerskirch, H., Filippi, D., Collet, J., Waugh, S.M. and Patrick, S. (2018b). Use of radar detectors to track attendance of albatrosses at fishing vessels. *Conserv. Biol*. 32, pp. 240–245

Weimerskirch, H. and Jouventin, P. (1998). Changes in population sizes and demographic parameters of six albatross species breeding on the French sub-Antarctic islands. *Albatross Biol. Conserv.* pp. 84–91

Whittaker, R.J. and Fernandez-Palacios, J.M. (2007). *Island biogeography: Ecology, Evolution and Conservation.* Oxford University Press, Oxford, UK

Wolfaardt, A., Williams, A., Underhill, L., RJM, C. and Whittington, P. (2009). Review of the rescue, rehabilitation and restoration of oiled seabirds in South Africa, especially African penguins Spheniscus demersus and Cape gannets Morus capensis, 1983–2005. *Afr. J. Mar. Sci.* 31, pp. 31–54

Zacharias, M.A. and Roff, J.C. (2001). Use of focal species in marine conservation and management: a review and critique. *Aquat. Conserv*. 11, pp. 59–76

Žydelis, R., Small, C. and French, G. (2013). The incidental catch of seabirds in gillnet fisheries: A global review. *Biol. Conserv.* 162, pp. 76–88

APPENDIX 1

Categories and criteria used to select IBAs at the global level. Sites may qualify for multiple categories and criteria. Marine IBAs typically qualify under the A1 and/or A4 category (http://datazone.birdlife.org/site/ibacritglob).

CATEGORY	CRITERION	NOTES	EXAMPLE
A1. Globally threatened species	The site is known or thought regularly to hold significant numbers of a globally threatened species.	The site qualifies if it is known, estimated or thought to hold a population of a species categorized by the IUCN Red List as Critically Endangered, Endangered or Vulnerable. Specific thresholds are set for species within each of the threat categories that need to be exceeded at a particular IBA. The list of globally threatened species is maintained and updated annually for IUCN by BirdLife International (www.birdlife.org/datazone/species).	Algoa Bay Islands: Addo Elephant National Park (South Africa ZA074). This IBA consists of two groups of three islands each within the large arc of Algoa Bay. The IBA is important for two Endangered species: the Cape gannet and the African penguin, and holds 43% of the global African penguin population.
A2. Restricted-range species	The site is known or thought to hold a significant population of at least two range-restricted species.	Restricted-range bird species are those having a global range size less than or equal to 50 000 km^2. 'Significant population': it is recommended that site-level populations of at least two restricted-range species should be equal to or exceed 1% of their global population. This criterion can be applied to species both within their breeding and non-breeding ranges. This criterion has not typically been applied to seabirds.	
A3. Biome-restricted species	The site is known or thought to hold a significant component of the group of species whose distributions are largely or wholly confined to one biome.	Bioregion-restricted assemblages are groups of species with largely shared distributions which occur (breed) mostly or entirely within all or part of a particular bioregion. Many biome-realms hold large numbers of species restricted to them, often across a variety of different habitat types; networks of sites must be chosen to ensure, as far as possible, adequate representation of all relevant species. In data-poor areas, knowledge of the quality and representativeness of the habitat types within sites alongside incomplete knowledge of the presence of bioregion-restricted species can be used to inform site selection. Many biome-realms cross political boundaries; where this is so, national networks of sites are selected to ensure that all relevant species in each country are adequately represented in IBAs. Thus biome-realms require that the networks of sites take account of both the geographical spread of the biome-realm and the political boundaries that cross them, as appropriate. Under 'significant component' it is recommended to use 30% of the number of bioregion-restricted species within a biome-realm within a country or five bioregion-restricted species, whichever is greatest. This criterion has not typically been applied to seabirds.	
A4. Congregations	The site is known or thought to hold congregations of ≥ 1% of the global population of one or more species on a regular or predictable basis.	This criterion can be applied to seasonal (breeding, wintering or migratory) congregations of any waterbird, seabird or terrestrial bird species. Sites can qualify whether thresholds are exceeded simultaneously or cumulatively, within a limited period. In this way, the criterion covers situations where a rapid turnover of birds takes place (including, for example, for migratory landbirds).	Cargados Carajos shoals (Mauritius, MU016) – a collection of low islets, coral reefs and sandbanks lying 350 km north-north-east of Mauritius, often called Saint Brandon. The site supports a congregation of Least Concern species, estimated to total more than 1 million birds, including Sooty terns (LC), Lesser noddy (LC), Brown noddy (LC), Common white tern (LC), Roseate tern (LC) (Evans et al., 2016).

APPENDIX 2

Summary details of marine and coastal IBAs in each WIO country (as of November 2018). Details include IBA name, species information (scientific name, common name, IUCN Red list category, population estimate, the unit of the population estimate, and the corresponding year of the population estimate, the season the bird is present in the IBA), the criteria used to designate the IBA, and if the IBA is protected. For updated information please see: http://datazone.birdlife.org/site/search

COUNTRY	IBA NAME	*SCIENTIFIC NAME*	COMMON NAME	IUCN	POP YEAR	POP ESTIMATE	UNIT	SEASON	IBA CRITERIA	PROTECTED
Comoros	Mont Mlédjélé (Hauts de Mwali)	*Puffinus persicus*	Persian Shearwater	LC		300	breeding pairs	breeding	A4ii	N
Kenya	Kisite island	*Sterna dougallii*	Roseate Tern	LC		1000	breeding pairs	breeding	A4i	Y
	Kisite island - Marine	*Sterna dougallii*	Roseate Tern	LC		2400	individuals	breeding	A4i	N
	Kiunga Marine National Reserve	*Dromas ardeola*	Crab-plover	LC	1996	800	individuals	non-breeding	A4i	Y
		Sterna dougallii	Roseate Tern	LC	1970	6200	breeding pairs	breeding	A4i	Y
	Mida Creek, Whale Island and the Malindi - Watamu coast	*Charadrius leschenaultii*	Greater Sandplover	LC		1250	individuals	winter	A4i	Y
		Charadrius mongolus	Lesser Sandplover	LC		1500	individuals	winter	A4i	Y
		Dromas ardeola	Crab-plover	LC		800	individuals	non-breeding	A4i	Y
		Sterna dougallii	Roseate Tern	LC		1500	breeding pairs	breeding	A4i	Y
		Sternula saundersi	Saunders's Tern	LC		5700	individuals	non-breeding	A4i	Y
	Sabaki River Mouth	*Glareola ocularis*	Madagascar Pratincole	VU	1978	2500–10 000	individuals	non-breeding	A4i	N
		Larus hemprichii	Sooty Gull	LC	1995	410	individuals	non-breeding	A4i	N
		Sternula saundersi	Saunders's Tern	LC	1995	900	individuals	non-breeding	A4i	N
		Thalasseus bengalensis	Lesser Crested Tern	LC	1995	270	individuals	non-breeding	A4i	N
	Tana River Delta	*Anastomus lamelligerus*	African Openbill	LC	1993	3530	individuals	non-breeding	A4i	N
		Ardea alba	Great White Egret	LC	1993	2560	individuals	non-breeding	A4i	N
		Ardea brachyrhyncha	Yellow-billed Egret	LC	1993	2000	individuals	non-breeding	A4i	N
		Bubulcus ibis	Cattle Egret	LC	1993	11 270	individuals	non-breeding	A4i	N
		Calidris ferruginea	Curlew Sandpiper	NT	1993	12 960	individuals	winter	A4i	N
		Calidris minuta	Little Stint	LC	1993	15 310	individuals	winter	A4i	N
		Gelochelidon macrotarsa	Australian Gull-billed Tern	LC	1993	1450	individuals	winter	A4i	N
		Gelochelidon nilotica	Common Gull-billed Tern	LC	1993	1450	individuals	winter	A4i	N
		Charadrius marginatus	White-fronted Plover	LC	1993	1070	individuals	non-breeding	A4i	N
		Charadrius mongolus	Lesser Sandplover	LC	1993	2340	individuals	winter	A4i	N
		Chlidonias hybrida	Whiskered Tern	LC	1993	1450	individuals	non-breeding	A4i	N
		Hydroprogne caspia	Caspian Tern	LC	1993	1340	individuals	winter	A4i	N
		Larus genei	Slender-billed Gull	LC	1993	490	individuals	winter	A4i	N

COUNTRY	IBA NAME	SCIENTIFIC NAME	COMMON NAME	IUCN	POP YEAR	POP ESTIMATE	UNIT	SEASON	IBA CRITERIA	PROTECTED
Kenya	Tana River Delta	*Larus hemprichii*	Sooty Gull	LC	1993	830	individuals	non-breeding	A4i	N
		Mycteria ibis	Yellow-billed Stork	LC	1993	970	individuals	non-breeding	A4i	N
		Pelecanus onocrotalus	Great White Pelican	LC	1993	2070	individuals	non-breeding	A4i	N
		Pelecanus rufescens	Pink-backed Pelican	LC	1993	2500	individuals	non-breeding	A4i	N
		Phoenicopterus roseus	Greater Flamingo	LC	1993	2240	individuals	non-breeding	A4i	N
		Platalea alba	African Spoonbill	LC	1993	3680	individuals	non-breeding	A4i	N
		Plectropterus gambensis	Spur-winged Goose	LC	1993	5400	individuals	non-breeding	A4i	N
		Sternula saundersi	Saunders's Tern	LC	1993	3610	individuals	non-breeding	A4i	N
		Thalasseus bengalensis	Lesser Crested Tern	LC	1993	1670	individuals	non-breeding	A4i	N
		Tringa stagnatilis	Marsh Sandpiper	LC	1993	1690	individuals	winter	A4i	N
Madagascar	Zone humide d'Ambava-nankarana	*Anas bernieri*	Madagascar Teal	EN	1999	unknown	–	resident	A1, A2, A3	N
		Ardea humbloti	Madagascar Heron	EN	1999	unknown	–	resident	A1, A2, A3	N
		Dromas ardeola	Crab-plover	LC	1999	900	individuals	non-breeding	A4i	N
		Sternula saundersi	Saunders's Tern	LC	1999	236	individuals	non-breeding	A4i	N
		Thalasseus bengalensis	Lesser Crested Tern	LC	1999	350	individuals	non-breeding	A4i	N
	Reserve Speciale d'Analamerana	*Ardea humbloti*	Madagascar Heron	EN	1997	unknown	–	resident	A1, A2, A3	N
		Ardeola idae	Madagascar Pond-heron	EN	1986	unknown	–	breeding	A1	Y
		Lophotibis cristata	Madagascar Crested Ibis	NT	1986	unknown	–	resident	A1	Y
	Zone humide d'Ankobohobo	*Ardea humbloti*	Madagascar Heron	EN	1997	unknown	–	resident	A1, A3	N
		Actophilornis albinucha	Madagascar Jacana	NT	1997	unknown	–	resident	A2	Some
		Anas bernieri	Madagascar Teal	EN	1997	unknown	–	resident	A1, A2, A3	Some
		Ardea humbloti	Madagascar Heron	EN	1997	unknown	–	resident	A1, A2, A3	Some
		Ardeola idae	Madagascar Pond-heron	EN	1997	unknown	–	breeding	A1	Some
		Charadrius thoracicus	Black-banded Plover	VU	1997	19	individuals	resident	A1, A2, A3	Some
		Lophotibis cristata	Madagascar Crested Ibis	NT	1997	unknown	–	resident	A1	Some
		Tachybaptus pelzelnii	Madagascar Grebe	VU	1997	unknown	–	resident	A1	Some
		Thalasseus bengalensis	Lesser Crested Tern	LC	1997	5000	individuals	non-breeding	A4i	Some
	AMP des Iles Barren	*Ardea humbloti*	Madagascar Heron	EN	1998	unknown	–	resident	A1, A3	N
		Sterna dougallii	Roseate Tern	LC	1998	1480	individuals	breeding	A4i	N

COUNTRY	IBA NAME	*SCIENTIFIC NAME*	COMMON NAME	IUCN	POP YEAR	POP ESTIMATE	UNIT	SEASON	IBA CRITERIA	PROTECTED
Madagascar	Forêt et zones humides de Cap Saint André	*Actophilornis albinucha*	Madagascar Jacana	NT	1998	unknown	–	resident	A2	N
		Lophotibis cristata	Madagascar Crested Ibis	NT	1998	unknown	–	resident	A1	N
		Tachybaptus pelzelnii	Madagascar Grebe	VU	1998	unknown	–	resident	A1	N
	NAP Archipel Cap Anorontany	*Sterna dougallii*	Roseate Tern	LC	1997	500	individuals	breeding	A4i	N
		Thalasseus bengalensis	Lesser Crested Tern	LC	1997	350	individuals	breeding	A4i	N
		Thalasseus bergii	Greater Crested Tern	LC	1997	3200	individuals	breeding	A4i	N
	Parc National de Kirindy Mite et extension	*Anas bernieri*	Madagascar Teal	EN	1999	unknown	–	resident	A1, A2, A3	Y
		Lophotibis cristata	Madagascar Crested Ibis	NT	1999	unknown	–	resident	A1	Y
	Parc National de Kirindy Mite et extension	*Actophilornis albinucha*	Madagascar Jacana	NT	1997	unknown	–	resident	A2	Y
		Ardea humbloti	Madagascar Heron	EN	1997	unknown	–	resident	A1, A2, A3	Y
		Ardeola idae	Madagascar Pond-heron	EN	1997	unknown	–	breeding	A1	Y
		Charadrius thoracicus	Black-banded Plover	VU	1997	50	individuals	resident	A1, A2, A3, A4i	Y
		Chlidonias hybrida	Whiskered Tern	LC	1997	503	individuals	non-breeding	A4i	Y
		Dromas ardeola	Crab-plover	LC	1997	500	individuals	non-breeding	A4i	Y
		Lophotibis cristata	Madagascar Crested Ibis	NT	1997	unknown	–	resident	A1	Y
		Phoeniconaias minor	Lesser Flamingo	NT	1997	unknown	–	non-breeding	A1	Y
		Tachybaptus pelzelnii	Madagascar Grebe	VU	1997	unknown	–	resident	A1	Y
		Tachybaptus ruficollis	Little Grebe	LC	1997	620	individuals	non-breeding	A4i	Y
		Zapornia olivieri	Sakalava Rail	EN	1962	unknown	–	resident	A1, A2, A3	Y
	Lacs Anony et Erombo	*Ardea humbloti*	Madagascar Heron	EN	1997	unknown	–	resident	A1, A2, A3	N
		Phoeniconaias minor	Lesser Flamingo	NT	1997	unknown	–	non-breeding	A1	N
		Phoenicopterus roseus	Greater Flamingo	LC	1997	3000	individuals	non-breeding	A4i	N
		Tachybaptus pelzelnii	Madagascar Grebe	VU	1997	unknown	–	resident	A1	N
	Complexe de la Baie de Mahajamba – Anjavavy	*Ardea humbloti*	Madagascar Heron	EN	1997	unknown	–	resident	A1, A2, A3	N
		Egretta gularis	Western Reef-egret	LC	1997	447	individuals	non-breeding	A4i	N
	NAP Zone Humide de Mahavavy – Kinkony	*Anas bernieri*	Madagascar Teal	EN	1997	10	individuals	resident	A1, A2, A3, A4i	N
		Anas melleri	Meller's Duck	EN	1997	unknown	–	resident		N
		Ardea humbloti	Madagascar Heron	EN	1997	unknown	–	resident	A1, A2, A3	N
		Ardeola idae	Madagascar Pond-heron	EN	1997	unknown	–	breeding	A1	N

COUNTRY	IBA NAME	SCIENTIFIC NAME	COMMON NAME	IUCN	POP YEAR	POP ESTIMATE	UNIT	SEASON	IBA CRITERIA	PROTECTED
Madagascar	NAP Zone Humide de Mahavavy – Kinkony	*Phoeniconaias minor*	Lesser Flamingo	NT	1997	unknown	–	non-breeding	A1	N
		Phoenicopterus roseus	Greater Flamingo	LC	1997	4200	individuals	non-breeding	A4i	N
		Thalasseus bengalensis	Lesser Crested Tern	LC	1997	2523	individuals	non-breeding	A4i	N
	Parc National de Mananara-Nord	*Ardeola idae*	Madagascar Pond-heron	EN	1997	unknown	–	breeding	A1	Y
		Lophotibis cristata	Madagascar Crested Ibis	NT	1997	unknown	–	resident	A1	Y
		Mentocrex kioloides	Madagascar Wood-rail	LC	1997	unknown	–	resident	A3	Y
		Sarothrura insularis	Madagascar Flufftail	LC	1997	unknown	–	resident	A3	Y
		Tachybaptus pelzelnii	Madagascar Grebe	VU	1997	unknown	–	resident	A1	Y
	Parc National de Masoala	*Ardeola idae*	Madagascar Pond-heron	EN	1994	unknown	–	breeding	A1	Y
		Lophotibis cristata	Madagascar Crested Ibis	NT	1994	unknown	–	resident	A1	Y
		Mentocrex kioloides	Madagascar Wood-rail	LC	1994	unknown	–	resident	A3	Y
		Sarothrura insularis	Madagascar Flufftail	LC	1994	unknown	–	resident	A3	Y
	Complexe de la Forêt du Menabe	*Actophilornis albinucha*	Madagascar Jacana	NT	1993	unknown	–	resident	A2	Some
		Ardeola idae	Madagascar Pond-heron	EN	1993	unknown	–	breeding	A1	Some
		Lophotibis cristata	Madagascar Crested Ibis	NT	1993	unknown	–	resident	A1	Some
		Phoeniconaias minor	Lesser Flamingo	NT	1993	unknown	–	non-breeding	A1	Some
		Tachybaptus pelzelnii	Madagascar Grebe	VU	1993	unknown	–	resident	A1	Some
	Aire Protégée de Mikea	*Actophilornis albinucha*	Madagascar Jacana	NT	1998	unknown	–	resident	A2	N
		Lophotibis cristata	Madagascar Crested Ibis	NT	1998	unknown	–	resident	A1	N
		Tachybaptus pelzelnii	Madagascar Grebe	VU	1998	unknown	–	resident	A1	N
	Pangalane Nord	*Anas melleri*	Meller's Duck	EN	1997	unknown	–	resident	A1, A3	N
		Glareola ocularis	Madagascar Pratincole	VU	1997	121	individuals	non-breeding	A4i	N
	Nosy Manitse Future SAPM Marine et zones humides ajacentes	*Ardea humbloti*	Madagascar Heron	EN	1998	unknown	–	resident	A1, A2, A3	N
		Charadrius thoracicus	Black-banded Plover	VU	1998	25	individuals	resident	A1, A2, A3, A4i	N
		Sterna dougallii	Roseate Tern	LC	1998	9000	individuals	non-breeding	A4i	N
		Thalasseus bengalensis	Lesser Crested Tern	LC	1998	1300	individuals	non-breeding	A4i	N
		Thalasseus bergii	Greater Crested Tern	LC	1998	800	individuals	non-breeding	A4i	N

COUNTRY	IBA NAME	SCIENTIFIC NAME	COMMON NAME	IUCN	POP YEAR	POP ESTIMATE	UNIT	SEASON	IBA CRITERIA	PROTECTED
Madagascar	Parc National Marin Sahamalaza - Iles Radama	*Ardea humbloti*	Madagascar Heron	EN	1997	unknown	–	resident	A1, A3	N
		Ardeola idae	Madagascar Pond-heron	EN	1997	unknown	–	breeding	A1	N
		Lophotibis cristata	Madagascar Crested Ibis	NT	1997	unknown	–	resident	A1	N
		Thalasseus bengalensis	Lesser Crested Tern	LC	1997	575	individuals	non-breeding	A4i	N
	NAP Zone Humide de Tambohorano	*Actophilornis albinucha*	Madagascar Jacana	NT	1998	unknown	–	resident	A2	N
		Anas bernieri	Madagascar Teal	EN	1998	67	individuals	resident	A1, A2, A3, A4i	N
		Ardea humbloti	Madagascar Heron	EN	1998	unknown	–	resident	A1, A2, A3	N
		Charadrius thoracicus	Black-banded Plover	VU	1998	61	individuals	resident	A1, A2, A3, A4i	N
		Sterna dougallii	Roseate Tern	LC	1998	883	individuals	non-breeding	A4i	N
		Thalasseus bengalensis	Lesser Crested Tern	LC	1998	3200	individuals	non-breeding	A4i	N
	Zones humides du Delta et de la Haute Tsiribihina	*Ardea humbloti*	Madagascar Heron	EN	1998	50	individuals	resident	A1, A2, A3, A4i	N
		Ardeola idae	Madagascar Pond-heron	EN	1998	unknown	–	breeding	A1	N
		Charadrius thoracicus	Black-banded Plover	VU	1998	47	individuals	resident	A1, A2, A3, A4i	N
		Glareola ocularis	Madagascar Pratincole	VU	1998	250	individuals	non-breeding	A4i	N
		Thalasseus bengalensis	Lesser Crested Tern	LC	1998	3300	individuals	non-breeding	A4i	N
		Xenus cinereus	Terek Sandpiper	LC	1998	642	individuals	non-breeding	A4i	N
		Anas bernieri	Madagascar Teal	EN	1998	40	individuals	resident	A1, A2, A3, A4i	N
Mauritius	Cargados Carajos Shoals	*Anous stolidus*	Brown Noddy	LC		4500	breeding pairs	breeding	A4i	N
		Anous tenuirostris	Lesser Noddy	LC		15 000	breeding pairs	breeding	A4i	N
		Gygis alba	Common White Tern	LC		5000	breeding pairs	breeding	A4i	N
		Onychoprion fuscatus	Sooty Tern	LC		20 000	breeding pairs	breeding	A4i	N
		Sterna dougallii	Roseate Tern	LC		400	breeding pairs	breeding	A4i	N
	Relict Forests of the Central Plateau	*Anas melleri*	Meller's Duck	EN		unknown	–	non-breeding	A1	N
	Rodrigues Islets	*Onychoprion fuscata*	Sooty tern	LC	2019	5000–6500	breeding pairs	breeding	A4iii	Some
		Anous stolidus	Brown noddy	LC	2019	5000–10 000	breeding pairs	breeding	A4iii	Some
		Anous tenuirostris	Lesser noddy	LC	2019	10 000–20 000	breeding pairs	breeding	A4iii	Some
	Round Island	*Ardenna pacifica*	Wedge-tailed Shearwater	LC		50 000	breeding pairs	breeding	A4ii	Y
		Phaethon lepturus	White-tailed Tropicbird	LC		1000	breeding pairs	breeding	A4ii	Y
		Phaethon rubricauda	Red-tailed Tropicbird	LC		700	breeding pairs	breeding	A4ii	Y
		Pterodroma arminjoniana	Trindade Petrel	VU		400	breeding pairs	breeding	A1, A4ii	Y

COUNTRY	IBA NAME	SCIENTIFIC NAME	COMMON NAME	IUCN	POP YEAR	POP ESTIMATE	UNIT	SEASON	IBA CRITERIA	PROTECTED
Mauritius	Serpent Island	*Anous stolidus*	Brown Noddy	LC		100 000	breeding pairs	breeding	A4i	Y
		Anous tenuirostris	Lesser Noddy	LC		100 000	breeding pairs	breeding	A4i	Y
		Onychoprion fuscatus	Sooty Tern	LC		500 000	breeding pairs	breeding	A4i	Y
Mozambique	Bazaruto Archipelago	*Calidris alba*	Sanderling	LC		2273	individuals	winter	A4i	Y
		Charadrius mongolus	Lesser Sandplover	LC		476	individuals	winter	A4i	Y
		Pluvialis squatarola	Grey Plover	LC		2029	individuals	winter	A4i	Y
		Sterna hirundo	Common Tern	LC		20 000	individuals	winter	A4i	Y
		Sternula albifrons	Little Tern	LC		1883	individuals	winter	A4i	Y
		Thalasseus bengalensis	Lesser Crested Tern	LC		5895	individuals	non-breeding	A4i	Y
	Moebase region Pomene	*Ardeola idae*	Madagascar Pond-heron	EN		unknown	–	non-breeding	A1	N
		Morus capensis	Cape Gannet	EN		unknown	–	non-breeding	A1	Some
	Zambezi River Delta	*Anastomus lamelligerus*	African Openbill	LC		1000	individuals	non-breeding	A4i	Unknown
		Bugeranus carunculatus	Wattled Crane	VU		70	breeding pairs	resident	A1, A4i	Unknown
Seychelles	Bancs Africains	*Anous stolidus*	Brown Noddy	LC		5900	breeding pairs	breeding	A4i	N
		Onychoprion fuscatus	Sooty Tern	LC		10 000	breeding pairs	breeding	A4i	N
		Sterna sumatrana	Black-naped Tern	LC		10	breeding pairs	breeding	A4i	N
	Réserve Spéciale d'Aldabra	*Ardeola idae*	Madagascar Pond-heron	EN		50	breeding pairs	breeding	A1, A4i	Y
		Arenaria interpres	Ruddy Turnstone	LC		1000	individuals	winter	A4i	Y
		Egretta gularis	Western Reef-egret	LC		3000	breeding pairs	breeding	A4i	Y
		Dromas ardeola	Crab-plover	LC		2800	individuals	non-breeding	A4i	Y
		Fregata ariel	Lesser Frigatebird	LC		6000	breeding pairs	breeding	A4ii	Y
		Fregata minor	Great Frigatebird	LC		4000	breeding pairs	breeding	A4ii	Y
		Phaethon lepturus	White-tailed Tropicbird	LC		2500	breeding pairs	breeding	A4ii	Y
		Phaethon rubricauda	Red-tailed Tropicbird	LC		1900	breeding pairs	breeding	A4ii	Y
		Sterna sumatrana	Black-naped Tern	LC		150	breeding pairs	breeding	A4i	Y
		Sula sula	Red-footed Booby	LC		7000	breeding pairs	breeding	A4ii	Y
	Reserve Spéciale de l'Ile Aride	*Anous stolidus*	Brown Noddy	LC		11 600	breeding pairs	breeding	A4i	Some
		Anous tenuirostris	Lesser Noddy	LC		197 000	breeding pairs	breeding	A4i	Some
		Ardenna pacifica	Wedge-tailed Shearwater	LC		28 400	breeding pairs	breeding	A4ii	Some
		Gygis alba	Common White Tern	LC		5900	breeding pairs	breeding	A4i	Some
		Onychoprion fuscatus	Sooty Tern	LC		366 000	breeding pairs	breeding	A4i	Some
		Phaethon lepturus	White-tailed Tropicbird	LC		972	breeding pairs	breeding	A4ii	Some
		Puffinus bailloni	Tropical Shearwater	LC		72 000	breeding pairs	breeding	A4ii	Some
		Sterna dougallii	Roseate Tern	LC		1300	breeding pairs	breeding	A4i	Some

COUNTRY	IBA NAME	SCIENTIFIC NAME	COMMON NAME	IUCN	POP YEAR	POP ESTIMATE	UNIT	SEASON	IBA CRITERIA	PROTECTED
Seychelles	Ile aux Vaches (Bird Island)	*Anous stolidus*	Brown Noddy	LC		10 000	breeding pairs	breeding	A4i	Y
		Onychoprion fuscatus	Sooty Tern	LC		600 000	breeding pairs	breeding	A4i	Y
	Boudeuse Island	*Sula dactylatra*	Masked Booby	LC		5000	breeding pairs	breeding	A4ii	Y
	Cosmolédo	*Arenaria interpres*	Ruddy Turnstone	LC		400	individuals	winter	A4i	N
		Dromas ardeola	Crab-plover	LC		2000	individuals	non-breeding	A4i	N
		Onychoprion fuscatus	Sooty Tern	LC		1 100 000	breeding pairs	breeding	A4i	N
		Phaethon rubricauda	Red-tailed Tropicbird	LC		200	breeding pairs	breeding	A4ii	N
		Sterna sumatrana	Black-naped Tern	LC		100	breeding pairs	breeding	A4i	N
		Sula dactylatra	Masked Booby	LC		6000	breeding pairs	breeding	A4ii	N
		Sula sula	Red-footed Booby	LC		15 000	breeding pairs	breeding	A4ii	N
		Thalasseus bergii	Greater Crested Tern	LC		500	breeding pairs	breeding	A4i	N
	Réserve Spéciale de l'Ile Cousin	*Anous tenuirostris*	Lesser Noddy	LC	1999	90 000	breeding pairs	breeding	A4i	N
		Ardenna pacifica	Wedge-tailed Shearwater	LC		16 900	breeding pairs	breeding	A4ii	N
		Gygis alba	Common White Tern	LC	1997	4080	breeding pairs	breeding	A4i	N
		Phaethon lepturus	White-tailed Tropicbird	LC	1999	1540	breeding pairs	breeding	A4ii	N
	Ile Cousine	*Anous tenuirostris*	Lesser Noddy	LC		71 200	breeding pairs	breeding	A4i	Y
		Ardenna pacifica	Wedge-tailed Shearwater	LC		31 000	breeding pairs	breeding	A4ii	Y
		Gygis alba	Common White Tern	LC		5000	breeding pairs	breeding	A4i	Y
		Phaethon lepturus	White-tailed Tropicbird	LC	1999	850	breeding pairs	breeding	A4ii	Y
	Ile Desnoeufs	*Onychoprion fuscatus*	Sooty Tern	LC		500 000	breeding pairs	breeding	A4i	N
	Etoile island	*Sterna dougallii*	Roseate Tern	LC		150	breeding pairs	breeding	A4i	Y
	Farquhar – Ile du sud et Ãlots	*Onychoprion fuscatus*	Sooty Tern	LC		200 000–400 000	breeding pairs	breeding	A4i	N
		Sterna sumatrana	Black-naped Tern	LC		20	breeding pairs	breeding	A4i	N
	Ile Frégate	*Anous tenuirostris*	Lesser Noddy	LC		7250	breeding pairs	breeding	A4i	N
		Gygis alba	Common White Tern	LC		3010	breeding pairs	breeding	A4i	N
	Ile Marie-Louise	*Anous stolidus*	Brown Noddy	LC		2000	breeding pairs	breeding	A4i	N
		Anous tenuirostris	Lesser Noddy	LC		3500	breeding pairs	breeding	A4i	N
		Gygis alba	Common White Tern	LC		3000	breeding pairs	breeding	A4i	N
Somalia	Ceel Munye – Ceel Torre	*Dromas ardeola*	Crab-plover	LC		300	individuals	non-breeding	A4i	N
	Jasiira Ceebaad and Jasiira Sacaada Diin	*Dromas ardeola*	Crab-plover	LC		1000	breeding pairs	breeding	A4i	N
		Larus leuco–phthalmus	White-eyed Gull	NT		200	breeding pairs	breeding	A1, A4i	N
		Onychoprion anaethetus	Bridled Tern	LC		100 000	breeding pairs	breeding	A4i	N
	Jasiira lagoon and Muqdisho islets	*Sterna dougallii*	Roseate Tern	LC	1980	2000	breeding pairs	breeding	A4i	N
	Jasiira Maydh	*Anous stolidus*	Brown Noddy	LC	1979	20 000	breeding pairs	breeding	A4i	N

COUNTRY	IBA NAME	*SCIENTIFIC NAME*	COMMON NAME	IUCN	POP YEAR	POP ESTIMATE	UNIT	SEASON	IBA CRITERIA	PROTECTED
South Africa	Agulhas Plain – Heuningnes Estuary	*Larus dominicanus*	Kelp Gull	LC		375	breeding pairs	resident	A4i	Some
		Sternula balaenarum	Damara Tern	VU		unknown	–	breeding	A1	Some
		Haematopus moquini	African Oystercatcher	LC		20	breeding pairs	resident	A1, A4i	Y
		Larus dominicanus	Kelp Gull	LC		700	breeding pairs	resident	A4i	Y
		Morus capensis	Cape Gannet	EN		59 000	breeding pairs	resident	A1, A4ii	Y
		Spheniscus demersus	Jackass Penguin	EN		60 000	individuals	non-breeding	A1, A4ii	Y
		Spheniscus demersus	Jackass Penguin	EN		21 200	breeding pairs	resident	A1, A4ii	Y
		Sterna dougallii	Roseate Tern	LC		180	breeding pairs	breeding	A4i	Y
		Sterna dougallii	Roseate Tern	LC		400	individuals	non-breeding	A4i	Y
		Sterna vittata	Antarctic Tern	LC		5000	individuals	non-breeding	A4i	Y
	Berg River Estuary	*Calidris ferruginea*	Curlew Sandpiper	NT		8281–16 881	individuals	winter	A4i	Y
		Charadrius pecuarius	Kittlitz's Plover	LC		300	breeding pairs	resident	A4i	Y
		Chlidonias leucopterus	White-winged Tern	LC		394–2623	individuals	winter	A4i	Y
		Hydroprogne caspia	Caspian Tern	LC		150	individuals	non-breeding	A4i	Y
		Larus dominicanus	Kelp Gull	LC		375	breeding pairs	resident	A4i	Y
		Larus hartlaubii	Hartlaub's Gull	LC		585	breeding pairs	resident	A4i	Y
		Phoenicopterus roseus	Greater Flamingo	LC		2748	individuals	non-breeding	A4i	Y
		Platalea alba	African Spoonbill	LC		35	breeding pairs	resident	A4i	Y
		Recurvirostra avosetta	Pied Avocet	LC		236–2273	individuals	non-breeding	A4i	Y
		Spatula smithii	Cape Shoveler	LC		360	individuals	non-breeding	A4i	Y
		Thalasseus bergii	Greater Crested Tern	LC		400	individuals	non-breeding	A4i	Y
		Thalasseus sandvicensis	Sandwich Tern	LC		695–1555	individuals	winter	A4i	Y
	Bird Island	*Microcarbo coronatus*	Crowned Cormorant	NT		7	breeding pairs	resident	A1, A4i	Y
		Morus capensis	Cape Gannet	EN		5000	breeding pairs	resident	A1, A4ii	Y
		Spheniscus demersus	African Penguin	EN		5	breeding pairs	resident	A1	Y
	Boland Mountains	*Anthropoides paradiseus*	Blue Crane	VU		unknown	–	resident	A1	Most
	Boulders Beach	*Spheniscus demersus*	African Penguin	EN	1997	700	breeding pairs	resident	A1	Y
	Cape Whale Coast	*Fulica cristata*	Red-knobbed Coot	LC		18 283–36 000	individuals	resident	A4i	N
		Netta erythro-phthalma	Southern Pochard	LC		326–1132	individuals	resident	A4i	N

COUNTRY	IBA NAME	SCIENTIFIC NAME	COMMON NAME	IUCN	POP YEAR	POP ESTIMATE	UNIT	SEASON	IBA CRITERIA	PROTECTED
South Africa		*Podiceps cristatus*	Great Crested Grebe	LC		62–152	individuals	non-breeding	A4i	N
		Podiceps nigricollis	Black-necked Grebe	LC		68–1100	individuals	non-breeding	A4i	N
		Spatula smithii	Cape Shoveler	LC		172–404	individuals	non-breeding	A4i	N
		Tadorna cana	South African Shelduck	LC		229–787	individuals	non-breeding	A4i	N
	Dassen Island	*Arenaria interpres*	Ruddy Turnstone	LC		416	individuals	winter	A4i	Y
		Haematopus moquini	African Oystercatcher	LC		100	breeding pairs	resident	A1, A4i	Y
		Larus dominicanus	Kelp Gull	LC		6250	breeding pairs	resident	A4i	Y
		Larus hartlaubii	Hartlaub's Gull	LC		65	breeding pairs	resident	A4i	Y
		Microcarbo coronatus	Crowned Cormorant	NT		130	breeding pairs	resident	A1, A4i	Y
		Phalacrocorax capensis	Cape Cormorant	EN		13 767–48 000	breeding pairs	resident	A1, A4i	Y
		Phalacrocorax neglectus	Bank Cormorant	EN		40	breeding pairs	resident	A1	Y
		Spheniscus demersus	African Penguin	EN		8500	breeding pairs	resident	A1, A4ii	Y
		Thalasseus bergii	Greater Crested Tern	LC		3038	breeding pairs	resident	A4i	Y
	De Hoop Nature Reserve	*Anas undulata*	Yellow-billed Duck	LC		319–4626	individuals	resident	A4i	Y
		Anthropoides paradiseus	Blue Crane	VU		unknown		resident	A1	Y
		Fulica cristata	Red-knobbed Coot	LC		2886–24 400	individuals	resident	A4i	Y
		Haematopus moquini	African Oystercatcher	LC		60	individuals	resident	A1, A4i	Y
		Phoenicopterus roseus	Greater Flamingo	LC		1473	individuals	resident	A4i	Y
		Podiceps cristatus	Great Crested Grebe	LC		140	individuals	non-breeding	A4i	Y
		Spatula smithii	Cape Shoveler	LC	0	604–3004	individuals	non-breeding	A4i	Y
	Dwesa-Cwebe Nature Reserve	*Haematopus moquini*	African Oystercatcher	LC		unknown	–	non-breeding	A1	Y
	Dyer Island Nature Reserve	*Haematopus moquini*	African Oystercatcher	LC	2000	47	breeding pairs	resident	A1, A4i	Y
		Larus dominicanus	Kelp Gull	LC		110	breeding pairs	resident	A4i	Y
		Larus hartlaubii	Hartlaub's Gull	LC		110	breeding pairs	resident	A4i	Y
		Microcarbo coronatus	Crowned Cormorant	NT		60–238	breeding pairs	resident	A1, A4i	Y
		Phalacrocorax capensis	Cape Cormorant	EN		35 580	breeding pairs	resident	A1, A4i	Y
		Phalacrocorax neglectus	Bank Cormorant	EN		unknown	–	breeding	A1	Y
		Spheniscus demersus	African Penguin	EN		3050	breeding pairs	resident	A1, A4ii	Y
		Thalasseus bergii	Greater Crested Tern	LC		300	breeding pairs	resident	A4i	Y

COUNTRY	IBA NAME	*SCIENTIFIC NAME*	COMMON NAME	IUCN	POP YEAR	POP ESTIMATE	UNIT	SEASON	IBA CRITERIA	PROTECTED
South Africa	False Bay Nature Reserve	*Chlidonias leucopterus*	White-winged Tern	LC		1025–6832	individuals	winter	A4i	Y
		Larus dominicanus	Kelp Gull	LC		996–3685	individuals	non-breeding	A4i	Y
		Larus hartlaubii	Hartlaub's Gull	LC		1156–3506	individuals	non-breeding	A4i	Y
		Netta erythro-phthalma	Southern Pochard	LC		346–1332	individuals	non-breeding	A4i	Y
		Phalacrocorax capensis	Cape Cormorant	EN		1000–15 000	breeding pairs	resident	A1, A4i	Y
		Phoeniconaias minor	Lesser Flamingo	NT		unknown	–	non-breeding	A1	Y
		Phoenicopterus roseus	Greater Flamingo	LC		187	individuals	non-breeding	A4i	Y
		Podiceps cristatus	Great Crested Grebe	LC		38	individuals	non-breeding	A4i	Y
		Podiceps nigricollis	Black-necked Grebe	LC		328–1380	individuals	non-breeding	A4i	Y
		Recurvirostra avosetta	Pied Avocet	LC		467–942	individuals	non-breeding	A4i	Y
		Spatula smithii	Cape Shoveler	LC		603–1418	individuals	non-breeding	A4i	Y
		Tachybaptus ruficollis	Little Grebe	LC		403–628	individuals	non-breeding	A4i	Y
		Tadorna cana	South African Shelduck	LC		87–477	individuals	non-breeding	A4i	Y
		Thalasseus bergii	Greater Crested Tern	LC		753	individuals	non-breeding	A4i	Y
		Thalasseus sandvicensis	Sandwich Tern	LC		3027	individuals	winter	A4i	Y
	iSimangaliso Wetland Park	*Anas undulata*	Yellow-billed Duck	LC		503–1706	individuals	non-breeding	A4i	Y
		Chlidonias hybrida	Whiskered Tern	LC		64–179	individuals	non-breeding	A4i	Y
		Geronticus calvus	Southern Bald Ibis	VU	1998	unknown	–	resident	A1	Y
		Hydroprogne caspia	Caspian Tern	LC		240	breeding pairs	breeding	A4i	Y
		Pelecanus onocrotalus	Great White Pelican	LC		1000	breeding pairs	resident	A4i	Y
		Phoeniconaias minor	Lesser Flamingo	NT		unknown	–	non-breeding	A1	Y
		Phoenicopterus roseus	Greater Flamingo	LC		6000	breeding pairs	resident	A4i	Y
		Platalea alba	African Spoonbill	LC		350	breeding pairs	resident	A4i	Y
		Recurvirostra avosetta	Pied Avocet	LC		1265–3460	individuals	non-breeding	A4i	Y
		Spatula smithii	Cape Shoveler	LC		110–512	individuals	non-breeding	A4i	Y
	Maitland - Gamtoos coast	*Haematopus moquini*	African Oystercatcher	LC		65	breeding pairs	resident	A1, A4i	N
	Olifants river estuary	*Calidris ferruginea*	Curlew Sandpiper	NT		2131–5363	individuals	winter	A4i	N
	Overberg Wheatbelt	*Anthropoides paradiseus*	Blue Crane	VU		2914–3484	individuals	non-breeding	A1, A4i	N

COUNTRY	IBA NAME	SCIENTIFIC NAME	COMMON NAME	IUCN	POP YEAR	POP ESTIMATE	UNIT	SEASON	IBA CRITERIA	PROTECTED
South Africa	Prince Edward Islands Special Nature Reserve	*Aphrodroma brevirostris*	Kerguelen Petrel	LC		20 000	breeding pairs	resident	A4ii	Y
		Aptenodytes patagonicus	King Penguin	LC		220 000	breeding pairs	resident	A4ii	Y
		Diomedea exulans	Wandering Albatross	VU		3000	breeding pairs	resident	A1, A4ii	Y
		Eudyptes chrysolophus	Macaroni Penguin	VU		417 000	breeding pairs	resident	A1, A4ii	Y
		Fregetta tropica	Black-bellied Storm-petrel	LC		2000	breeding pairs	resident	A4ii	Y
		Phalacrocorax atriceps	Imperial Shag	LC		463–961	breeding pairs	resident	A4i	Y
		Halobaena caerulea	Blue Petrel	LC		200 000	breeding pairs	resident	A4ii	Y
		Larus dominicanus	Kelp Gull	LC		230	breeding pairs	resident	A4i	Y
		Macronectes giganteus	Southern Giant Petrel	LC		3310	breeding pairs	resident	A1, A4ii	Y
		Macronectes halli	Northern Giant Petrel	LC		590	breeding pairs	resident	A1, A4ii	Y
		Pachyptila salvini	Salvin's Prion	LC		200 000	breeding pairs	resident	A4ii	Y
		Pterodroma gouldi	Grey-faced Petrel	LC		20 000	breeding pairs	resident	A4ii	Y
		Pterodroma macroptera	Great-winged Petrel	LC		20 000	breeding pairs	resident	A4ii	Y
		Phoebetria fusca	Sooty Albatross	EN		2400	breeding pairs	resident	A1, A4ii	Y
		Phoebetria palpebrata	Light-mantled Albatross	NT		290	breeding pairs	resident		Y
		Procellaria aequinoctialis	White-chinned Petrel	VU		20 000	breeding pairs	resident	A1, A4ii	Y
		Procellaria cinerea	Grey Petrel	NT		2000	breeding pairs	resident	A1, A4ii	Y
		Pterodroma mollis	Soft-plumaged Petrel	LC		2000	individuals	non-breeding	A4ii	Y
		Pterodroma mollis	Soft-plumaged Petrel	LC		2000	breeding pairs	breeding	A4ii	Y
		Eudyptes chrysocome	Southern Rockhopper Penguin	VU		208 000	breeding pairs	resident	A1, A4ii	Y
		Eudyptes moseleyi	Northern Rockhopper Penguin	EN		208 000	breeding pairs	resident	A1, A4ii	Y
		Catharacta antarctica	Brown Skua	LC		960	breeding pairs	resident	A4ii	Y
		Sterna virgata	Kerguelen Tern	NT		35	breeding pairs	resident	A1, A4i	Y
		Sterna vittata	Antarctic Tern	LC		50	breeding pairs	resident	A4i	Y
		Thalassarche carteri	Indian Yellow-nosed Albatross	EN		7000	breeding pairs	breeding	A1, A4ii	Y
		Thalassarche carteri	Indian Yellow-nosed Albatross	EN		20 000	individuals	non-breeding	A1, A4ii	Y
		Thalassarche chrysostoma	Grey-headed Albatross	EN		8100	breeding pairs	resident	A1, A4ii	Y
	Rietvlei Wetland: Table Bay Nature Reserve	*Haematopus moquini*	African Oystercatcher	LC		unknown	–	non-breeding	A1	Y
		Larus dominicanus	Kelp Gull	LC		665	individuals	non-breeding	A4i	Y

COUNTRY	IBA NAME	SCIENTIFIC NAME	COMMON NAME	IUCN	POP YEAR	POP ESTIMATE	UNIT	SEASON	IBA CRITERIA	PROTECTED
South Africa		*Larus hartlaubii*	Hartlaub's Gull	LC		262	breeding pairs	resident	A4i	Y
		Podiceps cristatus	Great Crested Grebe	LC		54	individuals	non-breeding	A4i	Y
		Recurvirostra avosetta	Pied Avocet	LC		263–669	individuals	non-breeding	A4i	Y
		Spatula smithii	Cape Shoveler	LC		337–506	individuals	non-breeding	A4i	Y
	Robben Island	*Haematopus moquini*	African Oystercatcher	LC		30	breeding pairs	resident	A1, A4i	N
		Larus hartlaubii	Hartlaub's Gull	LC		1000–4309	breeding pairs	resident	A4i	N
		Microcarbo coronatus	Crowned Cormorant	NT		40–108	breeding pairs	resident	A1, A4i	N
		Phalacrocorax capensis	Cape Cormorant	EN		2000	breeding pairs	resident	A1, A4i	N
		Phalacrocorax neglectus	Bank Cormorant	EN		57–106	breeding pairs	resident	A1, A4i	N
		Spheniscus demersus	African Penguin	EN		4500	breeding pairs	resident	A1, A4ii	N
		Thalasseus bergii	Greater Crested Tern	LC		2300	breeding pairs	resident	A4i	N
	Swartkops Estuary – Redhouse and Chatty Saltpans	*Arenaria interpres*	Ruddy Turnstone	LC		363	individuals	winter	A4i	Some
		Haematopus moquini	African Oystercatcher	LC		unknown	–	non-breeding	A1	Some
		Larus dominicanus	Kelp Gull	LC		430	breeding pairs	resident	A4i	Some
		Recurvirostra avosetta	Pied Avocet	LC		490	individuals	non-breeding	A4i	Some
	Umlalazi Nature Reserve	*Crex crex*	Corncrake	LC		unknown	–	winter	A1	Y
		Geronticus calvus	Southern Bald Ibis	VU	1998	unknown	–	resident	A1	Y
	Verlorenvlei Estuary	*Larus hartlaubii*	Hartlaub's Gull	LC		293	individuals	non-breeding	A4i	N
		Podiceps cristatus	Great Crested Grebe	LC		12	breeding pairs	resident	A4i	N
		Recurvirostra avosetta	Pied Avocet	LC		78–452	individuals	non-breeding	A4i	N
		Spatula smithii	Cape Shoveler	LC		103–600	individuals	non-breeding	A4i	N
		Tadorna cana	South African Shelduck	LC		380	individuals	non-breeding	A4i	N
	West Coast National Park and Saldanha Bay islands	*Arenaria interpres*	Ruddy Turnstone	LC		1963–4587	individuals	winter	A4i	Y
		Calidris alba	Sanderling	LC		1229–2643	individuals	winter	A4i	Y
		Calidris canutus	Red Knot	NT		2504–6219	individuals	winter	A4i	Y
		Calidris ferruginea	Curlew Sandpiper	NT		17 940–25 347	individuals	winter	A4i	Y
		Charadrius marginatus	White-fronted Plover	LC		3000	individuals	non-breeding	A4i	Y
		Charadrius pecuarius	Kittlitz's Plover	LC		1500	individuals	non-breeding	A4i	Y
		Haematopus moquini	African Oystercatcher	LC		160	breeding pairs	resident	A1, A4i	Y
		Larus dominicanus	Kelp Gull	LC		500–3347	breeding pairs	resident	A4i	Y
		Larus hartlaubii	Hartlaub's Gull	LC		2500	breeding pairs	resident	A4i	Y

COUNTRY	IBA NAME	SCIENTIFIC NAME	COMMON NAME	IUCN	POP YEAR	POP ESTIMATE	UNIT	SEASON	IBA CRITERIA	PROTECTED
South Africa		*Microcarbo coronatus*	Crowned Cormorant	NT		550	breeding pairs	resident	A1, A4i	Y
		Morus capensis	Cape Gannet	EN		20200	breeding pairs	resident	A1, A4ii	Y
		Phalacrocorax capensis	Cape Cormorant	EN		4000–10 000	breeding pairs	resident	A1, A4i	Y
		Phalacrocorax neglectus	Bank Cormorant	EN		220–300	breeding pairs	resident	A1, A4i	Y
		Phoenicopterus roseus	Greater Flamingo	LC		3791–8724	individuals	non-breeding	A4i	Y
		Pluvialis squatarola	Grey Plover	LC		3643–8228	individuals	winter	A4i	Y
		Recurvirostra avosetta	Pied Avocet	LC		217	individuals	non-breeding	A4i	Y
		Spheniscus demersus	Jackass Penguin	EN	0	unknown	–	resident	A1	Y
		Sterna hirundo	Common Tern	LC	0	1299–9658	individuals	winter	A4i	Y
		Thalasseus bergii	Greater Crested Tern	LC	0	50–4070	breeding pairs	breeding	A4i	Y
		Thalasseus bergii	Greater Crested Tern	LC	0	203–9000	individuals	non-breeding	A4i	Y
	Wilderness – Sedgefield Lakes Complex	*Fulica cristata*	Red-knobbed Coot	LC	0	5280–18 698	individuals	resident	A4i	Some
		Netta erythrophthalma	Southern Pochard	LC	0	825–2795	individuals	non-breeding	A4i	Some
		Podiceps cristatus	Great Crested Grebe	LC	0	212	individuals	resident	A4i	Some
		Podiceps nigricollis	Black-necked Grebe	LC	0	382–1738	individuals	non-breeding	A4i	Some
		Spatula smithii	Cape Shoveler	LC	0	767–2700	individuals	resident	A4i	Some
	Woody Cape Section: Addo Elephant National Park	*Haematopus moquini*	African Oystercatcher	LC	0	50–249	individuals	breeding	A1	Some
		Sternula balaenarum	Damara Tern	VU	0	unknown	–	breeding	A1	Some
Tanzania	Dar es Salaam coast	*Egretta gularis*	Western Reef-egret	LC	1995	400	individuals	non-breeding	A4i	N
		Dromas ardeola	Crab-plover	LC	1995	700	individuals	non-breeding	A4i	N
		Glareola ocularis	Madagascar Pratincole	VU	1982	2000	individuals	non-breeding	A4i	N
		Larus hemprichii	Sooty Gull	LC	1995	400	individuals	non-breeding	A4i	N
		Sterna dougallii	Roseate Tern	LC	1995	3000	individuals	non-breeding	A4i	N
		Sternula saundersi	Saunders's Tern	LC	1995	1000	individuals	non-breeding	A4i	N
		Thalasseus bengalensis	Lesser Crested Tern	LC	1995	500	individuals	non-breeding	A4i	N
	Latham Island	*Anous stolidus*	Brown Noddy	LC	1989	10 000	individuals	breeding	A4i	N
		Onychoprion fuscatus	Sooty Tern	LC	1989	35 000	individuals	breeding	A4i	N
		Sula dactylatra	Masked Booby	LC	1989	1500	adults	breeding	A4ii	N
		Thalasseus bergii	Greater Crested Tern	LC	1971	1000	individuals	breeding	A4i	N

COUNTRY	IBA NAME	SCIENTIFIC NAME	COMMON NAME	IUCN	POP YEAR	POP ESTIMATE	UNIT	SEASON	IBA CRITERIA	PROTECTED
Tanzania	Mafia Island	*Egretta gularis*	Western Reef-egret	LC	1988	461	individuals	non-breeding	A4i	Most
		Dromas ardeola	Crab-plover	LC	1988	1887	individuals	non-breeding	A4i	Most
		Xenus cinereus	Terek Sandpiper	LC	1988	489	individuals	winter	A4i	Most
	Mnazi Bay	*Charadrius leschenaultii*	Greater Sandplover	LC	1995	1823	individuals	winter	A4i	N
		Dromas ardeola	Crab-plover	LC	1995	750	individuals	non-breeding	A4i	N
	Rufiji Delta	*Calidris ferruginea*	Curlew Sandpiper	NT	2000	16 043	individuals	winter	A4i	Unknown
		Gelochelidon macrotarsa	Australian Gull-billed Tern	LC	2000	3427	individuals	winter	A4i	Unknown
		Dromas ardeola	Crab-plover	LC	2000	3402	individuals	non-breeding	A4i	Unknown
		Gelochelidon nilotica	Common Gull-billed Tern	LC	2000	3427	individuals	winter	A4i	Unknown
		Sternula saundersi	Saunders's Tern	LC	2000	203	individuals	non-breeding	A4i	Unknown
		Thalasseus bengalensis	Lesser Crested Tern	LC	2000	1939	individuals	non-breeding	A4i	Unknown
		Xenus cinereus	Terek Sandpiper	LC	1988	708	individuals	winter	A4i	Unknown
	Tanga North – Kibo saltpans	*Charadrius leschenaultii*	Greater Sandplover	LC	1995	2200	individuals	winter	A4i	N
	Tanga South	*Charadrius leschenaultii*	Greater Sandplover	LC	1995	1823	individuals	winter	A4i	Unknown
		Dromas ardeola	Crab-plover	LC	1995	750	individuals	non-breeding	A4i	Unknown
	Zanzibar Island-East Coast	*Charadrius leschenaultii*	Greater Sandplover	LC	1998	1805	individuals	winter	A4i	N
		Dromas ardeola	Crab-plover	LC	1998	1633	individuals	non-breeding	A4i	N
		Sternula saundersi	Saunders's Tern	LC	1989	3050	individuals	non-breeding	A4i	N
		Dromas ardeola	Crab-plover	LC	1998	712	individuals	non-breeding	A4i	N
		Sterna dougallii	Roseate Tern	LC	1994	750	breeding pairs	breeding	A4i	N
		Xenus cinereus	Terek Sandpiper	LC	1998	1083	individuals	winter	A4i	N

CRITICAL HABITATS

SEAMOUNTS AND RIDGES

Aurélie Spadone and Sabrina Guduff

Aspects of sea surface. © José Paula

BACKGROUND

Oceans cover nearly 71 per cent of the Earth's surface. With an average depth of almost 4000 m, the oceans provide more than 90 per cent of the habitable area for life on Earth. Beyond the continental shelves, 88 per cent of the oceans are deeper than 1 km and 76 per cent have depths of 3000–6000 m (UNEP, 2006). The seafloor is reached at a depth of about 4000 m and extends over the ocean basins at depths of 5000 m on average. This is called the abyssal plain. The zone between the continental shelf and the abyssal plain is the bathyal zone. In some places, the seafloor drops again into elongated trenches with depths of 10–11 km. This region is the hadal zone. The ocean floor is interrupted by a mountain chain known as the mid-oceanic ridge system. Other features on the ocean floor are seamounts and hydrothermal vents (Kaiser, 2005).

Seamounts occur from the Equator to the Poles and are morphologically distinct elevations beneath the surface of the sea, rising relatively steeply from the seabed, but they do not emerge above the surface (Santos et al., 2009; Rogers, 2012). They are present throughout the world's ocean basins across a wide range of latitudes and depths (Fig. 1) and form distinctive habitats in areas that would otherwise be dominated by sedimentary plains (Clark et al., 2010). Most seamounts are of volcanic origin, although some, such as the Atlantis Bank in the South-West Indian Ocean (SWIO), are formed by tectonic uplift or even from serpentine mud (Fryer, 1992). They are commonly conical in shape, with a circular, elliptical or more elongated base (Consalvey et al., 2010).

Geologists have traditionally defined seamounts as topographic features with an elevation exceeding 1000 m above the seabed. In most current definitions of seamounts, however, the restriction to a minimum height of 1000 m seems to be based primarily on practical criteria since elevations of less than 1000 m on the seafloor may enclose morphologic structures of diverse origins such as fault blocks or blocks within debris avalanche deposits (Menard, 1964; Schmidt and Schmincke, 2000). Smaller submarine knolls (with an elevation of 500–1000 m) and hills (elevation of less than 500 m) also share many of the environmental characteristics of larger features and, given that the size distribution of such elevations are continuous, the term 'seamount' is used interchangeably for most features of more than 100 m in elevation (Wessel, 2007; Staudigel and Clague, 2010).

Because seamounts do not break the sea's surface, knowledge of their distribution comes primarily from remote sensing. The abundance and distribution of seamounts at a global scale have been predicted many times, mostly based on satellite altimetry and ship-based sounding extrapolations (Costello et al., 2010; Wessel et al., 2010; Yesson et al., 2011). At present, these approaches are unable to adequately detect small and deep peaks, and thus estimates of the global abundance of seamounts are still uncertain (Morato et al., 2013).

Recent estimates (Wessel et al., 2010; Kim and Wessel, 2011; Yesson et al., 2011) of the number of seamounts in the world's underwater topography range approximately from 25 000 to 140 000 large features and potentially from 125 000 to 25 million small seamounts or knolls greater than 100 m in height. Despite this uncertainty and a general perception that seamounts are small, isolated spots scattered in remote areas, this habitat is one of the most extensive of all oceanic environments, encompassing an estimated area of about 28.8 million square kilometres (Etnoyer et al., 2010). The largest contiguous area of seamounts is found in the central portion of the Pacific Plate, where most studies have been conducted (Gubbay, 2003), with lower numbers in the Indian, Atlantic, Arctic and Southern Oceans (Wessel, 2007). The Indian Ocean has a surface area of 70.5 million km^2 and is characterized by a system of three active spreading mid-oceanic ridges (MOR): the Central Indian Ridge (CIR), the South-West Indian Ridge (SWIR) and the South-East Indian Ridge (SEIR) (Das et al., 2005). More details on the active spreading plate boundaries and associated ridges in the Western Indian Ocean (WIO) are provided in Chapter 5 (this volume).

The SWIO region corresponds to the western Indian marine ecoregion which includes an island, Madagascar, and several archipelagos such as Comoros, Mascarenes and Seychelles, each with different origins and ages (Spalding et al., 2007). The continental land mass of Africa, the micro-continent Madagascar and the North Seychelles Bank are fragments of the supercontinent Gondwana, dating from Precambrian times, more than 650 million years ago (mya) and which started to break up 180 mya (Peng and Mahoney, 1995). The SWIR is a slowly spreading ridge system separating the African, Australian and Antarctic tectonic plates with seamounts strongly marking the limits between the African and Antartic plates (Fig. 1). It extends from north-east to south-west in the west of the Indian Ocean basin, extending over 1800 km and varies from 300 to 450 km in width (Romanov, 2003).

Compared with the East Pacific Rise and Mid-Atlantic Ridge, the region of the SWIO has been less studied. Recently, the SWIR's ultra-slow and oblique spreading

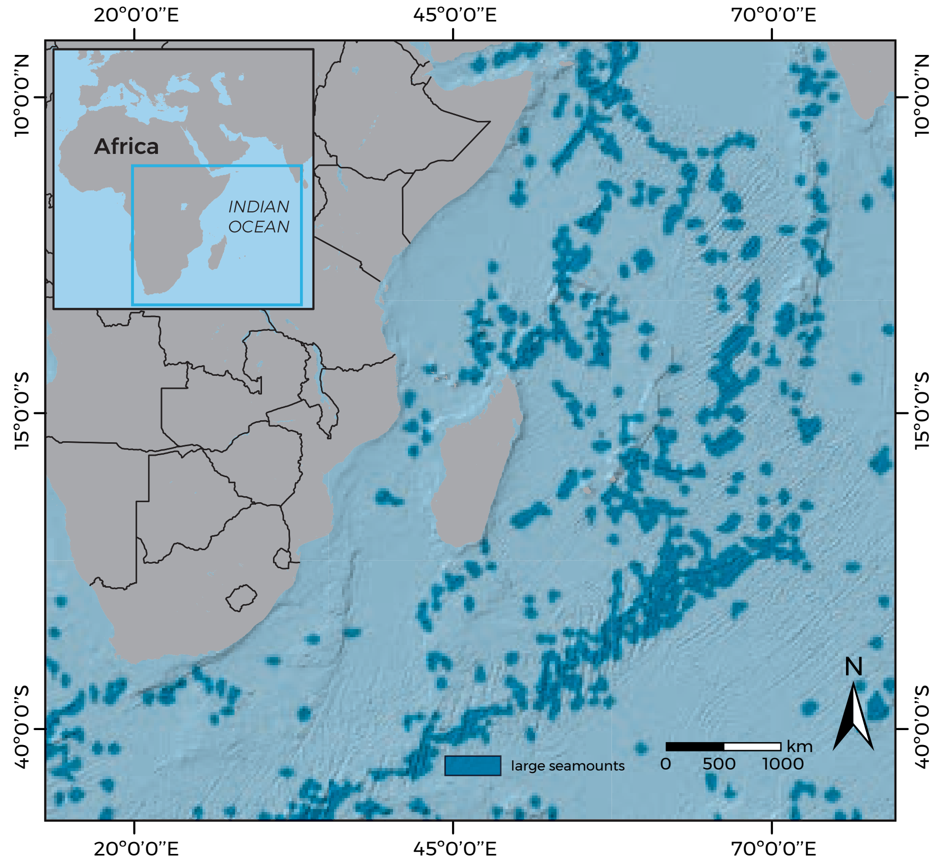


Figure 1: The distribution of seamounts predicted by Kitchingman and Lai (2004).

characteristics have attracted increasing international attention (Dick et al., 2003) and revealed that, rather than being formed of volcanic rock, parts of the ridge comprise large areas where mantle has been extruded onto the seafloor (Rogers and Taylor, 2012). Oceanographically, the SWIR is influenced by several fronts with the combined effect of the retroflection of the Agulhas Current (Lutjeharms, 2007) and the Subantarctic Front creating one of the most productive areas in the ocean (Read et al., 2000). It is also known that the SWIO area is characterized by substantial sea surface temperature (SST) variations (Annamalai and Murtugudde, 2004).

The Madagascar Ridge consists of a massive elevation of the seafloor, extending between the micro-continent of Madagascar and the SWIR for a distance of almost 1130 km. The ridge crest is wide and has depths ranging from 1000 to 2500 m (at the positions of seamounts up to 567 m). The minimum depth falls on the Walters Shoal to less than 20 m. The shoal was discovered in 1963 by the South African Hydrographic Frigate *SAS Natal* and named after its captain.

To date, more studies have been undertaken on the Walters Shoal than other seamounts, probably because it is closer to land than other areas and because of commercial fisheries interests in the region. The shoal was sampled during the 1964 Indian Ocean expedition by the research vessel *Anton Bruun* and subsequently by the *Vityaz* (Rogers, 2012). This chapter refers to Rogers (2012) and Rogers and Taylor (2012) for a complete list of Walters Shoal endemic species. Additionally, the research article published by Vereshchaka (1995) lists a large number of taxa as occurring on the Walters Shoal and summarized several investigations on the macroplankton occurring on slopes and seamounts in the Indian Ocean.

Walters Shoal is a group of seamounts located near the southern end of the Madagascar Ridge and consists of a large number of knolls, seamounts and ridges (Fig. 2). It is distinctive because the shallow areas of the seamount reach 18 m below the surface and it is characterized by high biodiversity.

IMPORTANCE

Seamounts, underwater mountains of volcanic and tectonic origin, are recognized as significant habitats for a wide diversity of species (Clark et al., 2012) and

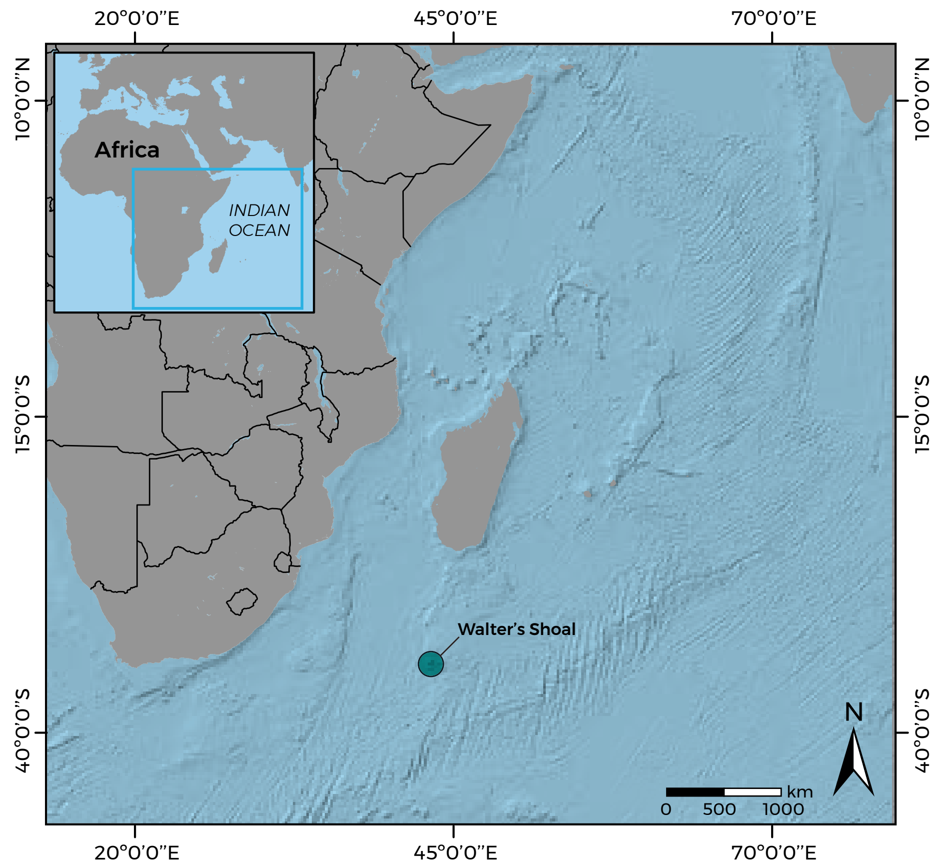


Figure 2: The Walters Shoal is located on the Madagascar Ridge, 833 km south of Madagascar and 1296 km east of South Africa.

considered hotspots of biodiversity (Postaire et al., 2014), attracting a range of oceanic predators, including sea-birds, whales and sharks. They also attract deep-water fisheries, as they host many species of commercial interest, and are subject to human exploitation (Rowden et al., 2010). Most of the deep-water species are very vulnerable to over-exploitation. Despite an increase in research on the ecology and biogeography of seamounts and oceanic islands, many basic aspects of their biodiversity are still unknown.

As described by Rogers (1994, 2004, 2012) and Rogers et al., (2007), the dominant large fauna of hard substrate on many deep-sea seamounts are attached, sessile organisms that feed on particles of food suspended in the water (Fig. 3). Also, pelagic species of fishes, sharks, squid and whales tend to aggregate over shallow seamounts because of the Taylor columns that form over them. Taylor columns are gently rotating water eddies that can aggregate food resources (small fishes, larvae and plankton), due to down-welling currents around the seamounts. The predominant seamount's phylum is Cnidaria, which includes black, stony and gorgonian corals, sea pens and anemones, and hydroids (Consalvey et al., 2010).

As biodiversity hotspots, seamounts have high endemism relative to other habitats (Richer de Forges et al. 2000; Morato and Clark 2007; Rogers, 2004, 2012). Understanding of global seamount biodiversity, however, is still poor, as fewer than 300 seamounts have been properly studied (Con-salvey et al., 2010), which is not enough to allow a reliable description of the benthic community. Furthermore, sampling has been biased towards larger fauna such as fishes, crustaceans and corals (Stocks, 2009).

Limited biological surveys of seamounts are a problem for assessing accurate levels of species richness and endemism (Stocks and Hart, 2007) and therefore conservation measures. Also, the hypothesis of high endemism has been questioned in recent years (Rowden et al., 2010). As a result, seamount data are very sparse and the 'oasis hypothesis' (Samadi et al., 2006), related to biomass, remains quantitatively untested (Rowden et al., 2010).

At the macro-ecological scale, the fauna of individual seamounts has been found to reflect the species groups present on neighbouring seamounts and continental margins (Samadi et al., 2006; Stocks and Hart, 2007; McClain et al., 2009; Brewin et al., 2009; Clark et al., 2010). Even where evidence suggests that the broad assemblage composition may resemble the surrounding deep-sea environments, community structure may differ between these habitats (Consalvey et al., 2010).

Rogers (2012) described in detail seamount habitats and associated communities, and the different hypotheses by which seamounts are locations of enhanced trophic input and determinants of community composition. However, to date, understanding of seamount ecosystems is hindered by significant gaps in global sampling, diverse analytical and scientific approaches, and sampling methods, as well as a lack of large-scale data synthesis and sharing.

Overall, the seamount ecosystem can host abundant and diverse benthic and pelagic communities. As previously stated, however, several studies have demonstrated that in many instances community composition might be similar to that of adjacent habitats including continental slopes. In general, acquisition of knowledge about seamount ecosystems and their associated resources is still ongoing.

THREATS

Since the second half of the 20th century, seamounts have faced two emerging threats: the exploitation of fishery resources and the potential for seabed mining (FFEM, 2013).

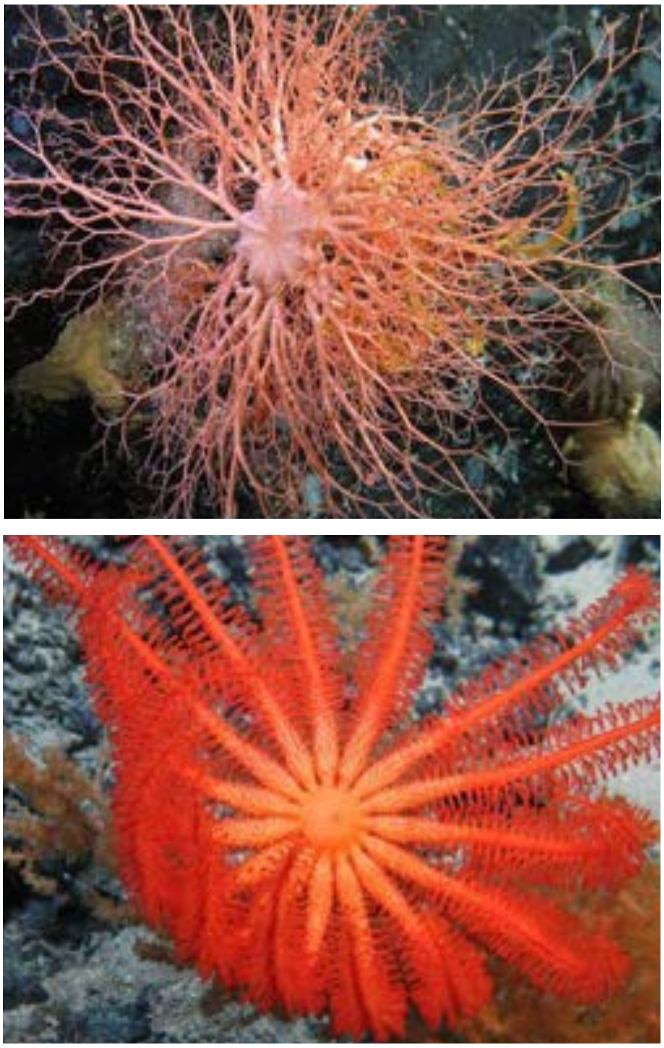


Figure 3: Examples of sessile fauna living on seamounts of the South-West Indian Ridge: (top) Basket star (*Gorgonocephalus* sp, Echinodermata); (bottom) Brisingid sea star (Order Brisingida, Echinodermata). © NERC/IUCN

Fishing

The depletion of biological resources is one of the major risks associated with the fishing trade that the targeted ecosystems are facing. In only a short time, these areas can be strongly impacted by the pressure of fisheries activity. The target species are often of low global abundance and their aggregation on seamounts at certain stages of their life (eg reproduction) makes them particularly vulnerable. The isolation of seamounts also makes the evolutionary and ecological mechanisms of these ecosystems substantially different from those in the surrounding waters. Due to limited exchanges with communities of other seamounts or coastal communities, it would take decades to rebuild numbers in the event of weakening stocks (Simard and Spadone, 2012).

Habitat degradation and its effects on associated communities, through the mechanical impact on ecosystem structure, is another of the bottom fisheries related threats. The resuspension of sediments is also an indirect consequence of this type of fishing (bottom trawling), combined with the lack of selectivity of catches. Trawl bycatch can include a broad range of benthic invertebrates, fish and seabirds, including sensitive or vulnerable species. The repercussions on these ecosystems could be observed particularly in terms of predator-prey relationships. The threat with ghost fishing gear, which continues to "fish" once lost or discarded, is thought to be low on seamounts, but is also a potential concern (Simard and Spadone, 2012).

There are an estimated 268 seamounts in this part of the Indian Ocean at 'fishing depth', ie summit areas shallower than 2000 m. FAO reported in 2009 that the SWIO was experiencing a significant increase in catches, however, fishing statistics in the regional are underdeveloped, with limited accessibility (Kimani et al., 2009). Fishery research programmes and fishing companies have provided the most detailed biological data and bathymetric maps of the region (FAO, 2002; Romanov, 2003; Shotton, 2006). Only syntheses of such data are publicly available and there is no compilation on species distribution. Data obtained from research on longline and commercial fisheries are generally not published (Tracey et al., 2011).

Nevertheless, almost 40 years of fishing mark the history of SWIO seamounts (Zucchi et al., 2018). Industry and research for Soviet fishery resources began experimental fishing in the 1970s on the Southwest Indian Ridge (SWIR), Mozambique Ridge and Madagascar Ridge, while bottom trawling started in 1980s (Romanov, 2003; Clark et al., 2007). The French fleet also conducted experimental trawl fisheries over the same period, on the Madagascar Ridge and SWIR, and in particular on the Walters Shoal and Sapmer Bank (Collette and Paring, 1991).

As previously described by Rogers et al. (2009), fisheries activities in the SWIO targeted Redbait (*Emmelichthys nitidus*) and Rubyfish (*Plagiogeneion rubiginosus*) with catches peaking in about 1980 and then decreasing to the mid-1980s (Clark et al., 2007). Later, fishing switched to Alfonsino (*Beryx splendens*) in the 1990s as new seamounts were exploited and the longline fleet was developing on the SWIR. While in the late 1990s, a new fishery developed on SWIR with trawlers targeting deep-water species such as Orange roughy (*Hoplostethus atlanticus*), Black cardinal fish (*Epigonus telescopus*), Southern boarfish (*Pseudopentaceros richardsoni*), Oreo (*Oreosomatidae*) and Alfonsino (Clark et al., 2007). More recently, longliners from Réunion have developed the tuna fishery in southern Madagascar, with a major effort devoted to this type of fishing in the SWIO region (Zucchi et al., 2018).

Species mainly targeted by these fisheries have a low reproductive rate and gather at seamounts during breeding season. They are therefore particularly exposed and vulnerable (of low resilience) to overexploitation. Target species include the above-mentioned species as well as the pelagic Armourhead, pelagic armourhead (*Pseudopentaceros richardsoni*) (Clark et al., 2007).

The Walters Shoal, an area beyond national jurisdiction (ABNJ), is considered in particular to be a productive fishing ground (Zucchi et al., 2018). It is a known fishing ground for demersal species (Romanov, 2003; Bach et al., 2011), and it has also been targeted for deep-sea lobster fishing, including the famous *Palinurus barbarae* (Rogers and Gianni, 2011; Bensch et al., 2008), and recreational fishing. The potential productivity of green prawns (*Palinurus delagoa*) in this area was estimated at 1000 t per year (Andrianaivojaona et al., 1992; Gopal et al., 2006). Exploitation of these stocks, as well as new targets such as the spiny lobster (*Palinurus barbarae*) recently discovered on the Walters Shoal (Groeneveld et al., 2006), continues (Bensch et al., 2008).

Mining

Mining exploration activities have been conducted since the 1970s–1980s (mainly in the Clarion-Clipperton zone, in the Pacific Ocean) (Cuyvers et al., 2018). The number of metals exploited worldwide has tripled since the 1970s to meet industrial needs with resources on land becoming scarce, thus there is increasing interest in exploiting the deep seabed.

The concentration of metals in the marine environment is found in three forms: polymetallic nodules on the abyssal plains; crusts on seamounts; and hydrothermal sulphides along the ridges. Currently, engineering for the extraction of polymetallic crusts located on seamounts is the least developed. Despite the economic interest and the shallowness of the crusts (above 2500 m), extraction processes are still technically complex for this resource (Hein et al., 2009, in Cuyvers et al., 2018). However, extraction processes will likely cause destruction of habitat and associated fauna. They may also generate fine particles rich in toxic metals, which can be transported by bottom currents to the pelagic and suspension feeder fauna (FFEM, 2013).

Potential threats from mining also include the following: noise pollution from extraction techniques (air guns, sonar, machines, drilling); pollution from sludge and drilling piles that may be contaminated by oil, chemicals and drilling fluids; and oil and gas leaks and spills (Simard and Spadone, 2012).

To date, the International Seabed Authority (ISA) has granted 29 contracts for exploration of seabed minerals in ABNJ, representing more than 1.2 million km^2 of seabed. Five contracts, for the exploration of two types of mineral, have been awarded in the Indian Ocean:

Polymetallic nodules
1. Location: Central Indian Ocean Basin – Contractor: Government of India.

Polymetallic sulphides
2. Location: Central Indian Ocean (Mid-Indian Ridge and SWIR) – Contractor: Government of India.
3. Location: Central Indian Ocean (Mid-Indian Ridge) – Contractor: Federal Institute for Geosciences and Natural Resources, Federal Republic of Germany.

4. Location: Mid-Indian Ridge – Contractor: Government of the Republic of Korea.
5. Location: SWIR – Contractor: China Ocean Mineral Resources Research and Development Association (COMRA).

While the number of exploration contracts granted has been increasing in recent years, exploitation is yet to begin.

In addition to these deep-sea mining and fisheries-related threats, seamounts are subject to direct or indirect impacts from other human activities, such as:

- accidental and/or deliberate (operational) discharges from vessels;
- anchoring;
- collisions (ship strikes) with, for example marine mammals, sharks and turtles;
- grounding and shipwreck;
- invasive alien species (IAS); and
- noise.

Seamount ecosystems could also be impacted by activities for which the ship serves primarily as a platform, such as:

- archaeology;
- artificial islands and fixed/floating installations;
- bioprospecting;
- dumping;
- marine mining for oil and gas;
- marine scientific research;
- military activities;
- ocean-based climate change mitigation;
- piracy/criminal activities;
- recreation;
- salvage; and
- undersea cable- and pipeline-laying.

Finally, there are threats from activities not involving ships, such as:

- anthropogenic climate change;
- land-based activities;
- marine debris or litter;
- overflight; and
- radionuclides.

Seamount ecosystems are particularly fragile and vulnerable to anthropogenic threats and hence their ecosystem structure is likely to have or be vulnerable to tipping points. Any additional or new activity, or the intensification of an ongoing activity, could trigger a tipping points, leading to the collapse of a seamount ecosystem.

STATUS / LEVEL OF THREAT

The SWIO region hosts an extraordinary proportion of endemic species and is highly threatened by human activities, hence its classification as a marine biodiversity hotspot (Roberts et al., 2002; Bellard et al., 2013; Gopal et al., 2006). It is known that the reproduction rate for these species is generally low and they form breeding aggregations on seamounts, making them particularly susceptible to overexploitation (Koslow et al., 2000).

In particular, the Orange roughy is described as having a low resilience and high vulnerability to fishing (Branch, 2001). In the late 1980s, an estimated annual catch of more than 10 000 t led to the subsequent rapid collapse of the population. In 2006, some participants in the fishing industry (bottom trawlers) voluntarily closed a small portion of the Walters Shoal for conservation purposes (Coyle et al., 2007).

Seamount and hydrothermal vent ecosystems display common features. Both ecosystem types:

- are considered 'hotspots' of species biodiversity;
- are already under potential threat from intensive commercial exploitation (such as mining, fishing, pharmaceutical research) (UNEP, 2006;
- could be proposed as Marine Protected Areas (MPAs), Ecologically or Biologically Significant Marine Areas (EBSAs) or Areas of Particular Environmental Interest (APEIs); and
- need a higher and targeted level of protection in particular for vulnerable and unique associated species.

In this respect, considerably more exploration and investigation, that follow responsible research practices for new sites at key locations (see the six recommendations promoted in Devey et al. (2007)) are essential to fill important gaps in the understanding of biogeographical, ecological, geological, evolutionary and genetic enigmas associated with hydrothermal vents and seamounts. Only then will it be possible to advise the public and policy-makers on how best to preserve these ecosystems and their outstanding beauty and uniqueness for future generations.

EXISTING PROTECTION

Three Regional Fisheries Management Organizations (RFMOs) operate in the Western Indian Ocean (WIO) region, each with different mandates and competences:

- The *Indian Ocean Tuna Commission (IOTC)* promotes cooperation with the aim of ensuring management, conservation, and optimum utilisation of stocks of tuna and tuna-like species in the Indian Ocean. The IOTC covers both national waters and ABNJ of the Indian Ocean.

- The *South Indian Ocean Fisheries Agreement (SIOFA)* aims to ensure the long-term conservation and sustainable use of fishery resources in ABNJ of the Indian Ocean through cooperation among the Contracting Parties. SIOFA only covers waters beyond national jurisdiction.

- The *Southwest Indian Ocean Fisheries Commission (SWIOFC)* is an advisory fisheries body that promotes sustainable utilisation of the living marine resources of the SWIO region. SWIOFC only covers waters under national jurisdiction.

In addition to these RFMOs, it is also worth noting that two additional management bodies have mandates covering adjacent waters (Fig. 4). The Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) aims to conserve Antarctic marine life and takes an ecosystem-based approach to managing the area. The South East Atlantic Fisheries Organisation (SEAFO) aims to ensure the long-term conservation and sustainable use of living marine resources and safeguard the environment and marine ecosystems in the South East Atlantic Ocean.

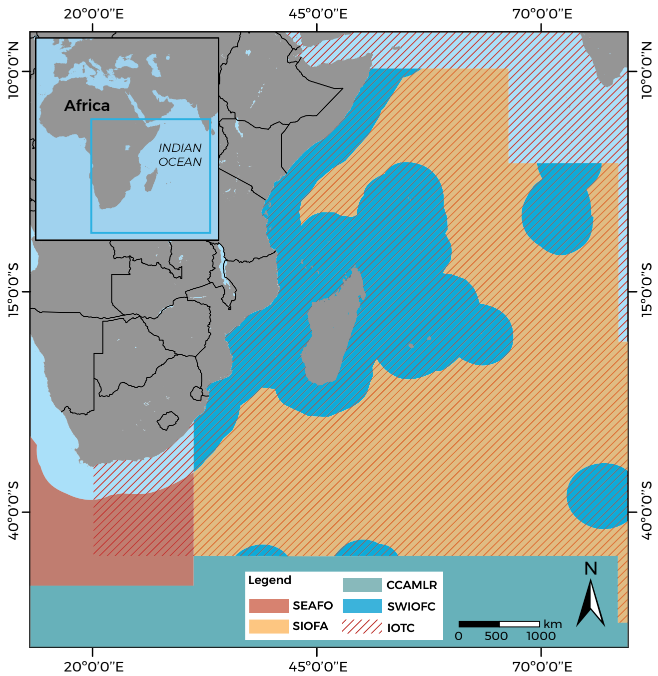


Figure 4: Areas of competence of RFMOs in the SWIO and adjacent waters.

There may be value in increasing cooperation and information exchange between these bodies in order to better understand connectivity and provide further support for the development of appropriate management actions.

Complementary to these RFMOs, the operators of the vessels conducting deep-sea fishing in the region established the Southern Indian Ocean Deep Sea Fishers Association (SIODFA) in 2006. This industry association aims to promote responsible management of the deep-water fishery while conserving biodiversity, especially the deep-water benthos.

Progress has been made in the southern Indian Ocean towards better protection of biodiversity in the high seas. In Phuket, Thailand, on the week of 25–29 June 2018, the Southern Indian Ocean Fisheries Agreement (SIOFA) declared five new Protected Areas in the high seas at its 5th Meeting of the Parties (MoP5). These closures, defined as benthic protected areas (BPAs) apply only to bottom trawling and do not cover other fishing gear such as bottom long lining and trap fisheries which, nevertheless, will have the obligation to have observers on board 100 per cent of the time, if fishing in the designated areas. The protected sites are Atlantis Bank, Coral, Fool's Flat, Middle of What and Walters Shoal, all of them being important features of the ocean floor for biodiversity – such as banks or seamounts – and covering an area of over 25 000 km^2.

PRIORITY OPTIONS FOR CONSERVATION

Possible options for the conservation and management of the Walters Shoal are given here as an example of what could be the foundations for management of a seamount in the WIO. In addition to its existing BPA status, several options are possible to better conserve and manage the seamount, from the adoption of sectoral measures aimed at limiting impacts from certain maritime activities to the establishment of an MPA. This section studies and assesses the opportunities and feasibility of such measures.

Limiting impacts from maritime activities

Fishing

IOTC fisheries closures

There are currently few operational examples of fisheries closures for highly migratory pelagic species, though in recent years interest has been growing in understanding and developing such measures (Game et al., 2009; Harley and Suter, 2007; Hyrenbach et al., 2000; Kaplan et al., 2010; Kaplan et al., 2014; Maxwell and Morgan 2012; Torres-Irineo et al., 2011; Young et al., 2015). Pelagic ecosystems are generally characterized by high levels of species mobility, large spatial scales, and limited scientific knowledge, such that existing practice in relation to fisheries closures and MPAs cannot necessarily be applied directly to this context. Some have called for development of pelagic MPAs (Game et al., 2009; Robison, 2009; Young et al., 2015), noting that "recent advances across conservation, oceanography and fisheries science provide the evidence, tools and information to address these criticisms and confirm MPAs as defensible and feasible instruments for pelagic conservation" (Game et al., 2009). However, few scientific studies have so far accurately determined if such measures are effective (Kaplan et al., 2014) and no consensus exists as yet on effectiveness and good practice. Some commentators have tentatively noted the success of certain measures (Kaplan et al., 2014; Torres-Irineo et al., 2011), but others have argued that the benefits of closures and area-based measures decrease significantly for mobile species (Grüss et al., 2011; Le Quesne and Codling, 2008; Moffitt et al., 2009).

In any case, scientists currently consider tuna fisheries to have little to no impact on the Walters Shoal ecosystems. As illustrated by Fig. 5, longline fisheries are distant from the Walters Shoal and there are no purse seine tuna fisheries south from 15 S, except in the Mozambique channel, consequently all purse seine fisheries are well outside the Walters Shoal area. Against this background, it does not seem appropriate to propose an IOTC fisheries closure in the Walters Shoal area.

SIOFA Fisheries closures

In contrast to pelagic ecosystems, benthic ecosystems are well suited to area-based management tools (ABMTs), including fisheries closures. Bottom fishing has been reported in the Walters Shoal area (FAO, 2010), thus it would be relevant to consider whether the area contains vulnerable marine ecosystems (VMEs) that should be closed to fishing or whether other management measures might be appropriate.

Although the BPAs currently in place will remain in force for the members of SIODFA, it is clear that Parties to SIOFA are also obliged to take certain measures: the UNFSA makes it clear that RFMOs are the primary vehicle for collaboration on fisheries management and United National General Assembly (UNGA) resolutions require closures and other measures for the protection of VMEs.

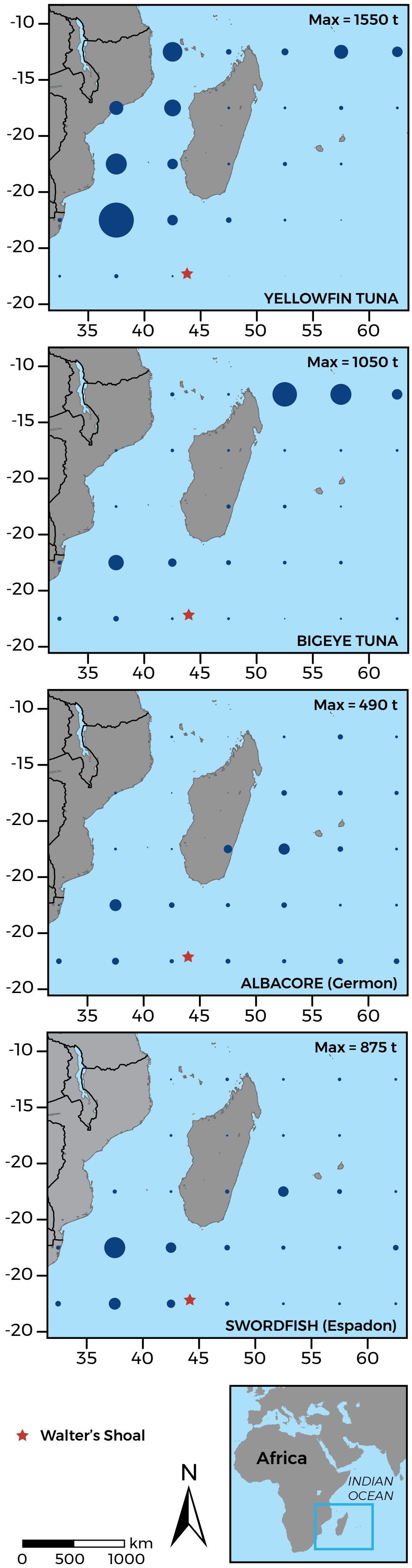


Figure 5: Main areas of longlines fisheries in SWIO.

As highlighted above, no fisheries closures have been adopted by SIOFA so far. Pressure on SIOFA to take such measures as soon as possible is however mounting. At the 2nd SIOFA meeting, SIODFA submitted an "Expression of Concern" over the failure to adopt measures, and as reported by FAO (2002), the Deep Sea Conservation Coalition (DSCC) highlighted that:

> *The draft measure CMM 14.02 for the protection of VMEs circulated last year falls far short of the commitments to protect VMEs that States Parties to SIOFA have repeatedly made through the UNGA resolutions over the past 11 years. A new measure or measures for the protection of VMEs should be drafted, adopted and implemented on an urgent basis.*

One relatively simple route for the adoption of VME closures within the SIOFA framework would be to study the feasibility of converting the SIODFA's BPAs – which include the Walters Shoal – into formal VME closures. Such a proposal was tabled at the 3rd (La Réunion, France, 3-8 July 2016) and 4th (Mauritius, 26-30 June 2017) meetings of the SIOFA.

This proposal was supported by the majority of parties and civil society but was ultimately not passed due to the objections of France and South Korea, which highlighted the lack of scientific data reviewed by the SIOFA Scientific Committee. France, representing its Territories in the region, also argued that the closure should apply to bottom trawling but not to other fishing gears, such as bottom longlining. This position is supported by a French legal provision that aims to expand the fishing fleet in the SIOFA area, including in several areas currently covered by the SIODFA BPAs (Decree of 6 February 2017 transposing the recommendation CMM 2016/01 of the Southern Indian Ocean Fisheries Agreement NOR: DEVM1625024A). In turn, states that practice bottom trawling have rejected this counter-proposal.

There is also ongoing debate amongst the SIOFA member states regarding the procedure for defining fisheries footprints. If the transformation of the whole set of BPAs into formal RFMO fisheries closures is not politically viable, an alternative option could be to discuss proposals for each area separately.

Unilateral national initiatives

Flag states retain the right to regulate their vessels even where the relevant RFMO has not adopted measures, and nothing prevents one or several states from unilaterally declaring that they will prohibit or restrict fishing in the Walters Shoal area by vessels flying their flag. There is some precedent for a unilateral national initiative to prohibit or restrict fishing in ABNJ.

In the south-west Atlantic, Spain, the only state known to conduct significant bottom fishing activities, published a list of authorized vessels and, in the absence of a RFMO for the region, unilaterally declared nine areas closed to bottom fishing by its vessels in July 2011 (pursuant to the European Union (EU) Council Regulation (EC) No 734/2008 of 15 July 2008). Between 2007-2009, Spain's Oceanographic Institute (Instituto Español de Oceanografía; IEO) conducted a series of 11 multidisciplinary research cruises with the aim of identifying VMEs in the region and making a preliminary assessment of how fishing activity was affecting these areas (Portela et al., 2010).

The research found that, overall, the particular fisheries in question only had a small adverse impact on VMEs in the region, but nonetheless identified nine areas that should be closed to bottom trawling to prevent significant adverse impacts. Beginning in July 2011, these areas were closed for bottom fishing for a period of six months (Gianni et al., 2011).

In New Zealand, the Government worked in consultation with industry, environmental NGOs and government departments to implement closures in its footprint area in advance of measures being formally taken by the competent South Pacific RFMO – SPRFMO (New Zealand Government, 2012). Lightly trawled areas were closed to bottom fishing, moderately trawled areas were opened subject to application of a move-on rule, and heavily trawled blocks generally remained open to bottom fishing. Although these closures no doubt represent an improvement on a business-as-usual scenario, Penney and Guinotte (2013) conducted a detailed analysis of the New Zealand closures, concluding that the existing sites are "sub-optimal for protecting likely coral VMEs" (Penney and Guinotte, 2013) and Penney et al. (2009) concluded that "effective protection of benthic VMEs in the Pacific Ocean high seas will probably require the establishment of a series of international spatial closures designed to protect adequate and representative areas of habitats and ecosystems."

Shipping

The designation of a Particularly Sensitive Sea Area (PSSA) is made by a non-legally binding resolution from the International Maritime Organization (IMO) Marine Environment Protection Committee (MEPC). This resolution is then given effect by the adoption of "associated protective measures" (APMs). It seems that there is no specific threat to the Walters Shoal system. The IMO may also pursue the development and adoption of other measures, provided they have an identified legal basis.

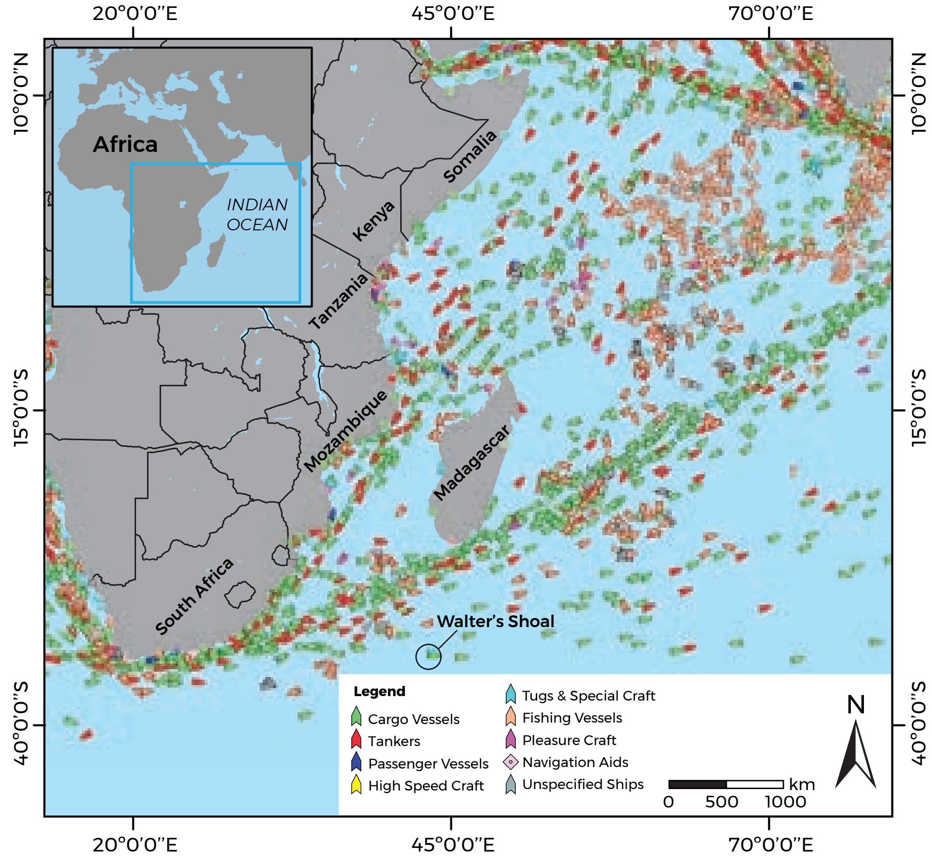


Figure 6: Shipping traffic in the Western Indian Ocean.

As illustrated in Fig. 6, major shipping routes do not pass through the Walters Shoal area, therefore the establishment of a PSSA and APMs do not seem particularly relevant.

Mining

In 2012, as part of its Environmental Management Plan for the Clarion-Clipperton Zone (ISA, 2011), the ISA designated nine APEIs in the Area where no mining is permitted. These designations were made in advance of contractor-designated "impact reference zones" (areas to be used for assessing the effect of each contractor's activities in the Area on the marine environment and which are representative of the environmental characteristics of the area) and "preservation reference zones" (areas in which no mining shall occur to ensure representative and stable biota of the seabed in order to assess any changes in the flora and fauna of the marine environment).

At the same time, the ISA Regulations on prospecting and exploration for polymetallic nodules, polymetallic sulphides, and ferromanganese crusts in the Area provide that "prospecting shall not be undertaken if substantial evidence indicates the risk of serious harm to the marine environment" (Regulation 2(2)).

Exploration for mineral resources is ongoing in the Indian Ocean, including in its western part. The ISA is yet to define any APEIs in this region, while no assessment has so far been conducted regarding their need and feasibility. This is therefore a step WIO states, and the international community more generally, may be interested in taking in conjunction with the ISA.

Establishment of Marine Protected Area

MPAs are widely acknowledged as an important tool for biodiversity conservation, and ecologically connected networks of MPAs are crucial for sustaining high seas ecosystems (Sumaila et al., 2007). The international community has committed, in numerous global fora, to establish a network of MPAs covering a significant percentage of the oceans (Rochette et al., 2014a). Therefore, interest in the establishment of multi-purpose MPAs in ABNJ is strong, yet currently no global mechanism exists to make this possible.

Nonetheless, some efforts have been made to develop specific initiatives to conserve marine biodiversity in ABNJ through the creation of MPAs. Against this background, several options exist to establish an MPA in the Walters Shoal area.

Establishing a marine protected area through the Nairobi Convention

Some regional initiatives and organizations have progressively extended their activities to ABNJ, including through the establishment of MPAs (Rochette et al., 2014b). Four areas are currently covered by a Regional Sea with a specific mandate in ABNJ: the Mediterranean through the Barcelona Convention (Convention for the Protection of the Marine Environment and the Coastal Region of the Mediterranean from 1995), the Southern Ocean through the Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR) (Convention on the Conservation of Antarctic Marine Living Resources from 1980), the North-East Atlantic through the OSPAR Convention (Convention for the Protection of the Marine Environment of the North-East Atlantic from 1992) and the South Pacific through the Nouméa Convention (Convention for the Protection of the Natural Resources and Environment of the South Pacific Region from 1986).

Three Regional Seas have already developed specific actions in ABNJ through the creation of MPAs:

Mediterranean
The Pelagos Sanctuary for Mediterranean Marine Mammals was created in 1999 by France, Italy and Monaco. The Sanctuary was recognized as a Specially Protected Area of Mediterranean Importance (SPAMI) in 2001 (Scovazzi, 2011) and incorporates the territorial waters of these three states, but also ABNJ.

Southern Ocean
In 2009, CCAMLR endorsed a roadmap established by its Scientific Committee in order to fulfil the international requirements to establish a coherent and representative network of MPAs by 2012. The same year, CCAMLR adopted its first MPA on the South Orkney Islands continental shelf (CM 91-03 from 2009), and in 2016 the Ross Sea was also designated as an MPA.

North-East Atlantic
Contracting Parties to the OSPAR Convention established a network of six MPAs in ABNJ in 2010 (O'Leary et al., 2012), and agreed an additional MPA in 2012 (Freestone et al., 2014).

As previously noted, the Nairobi Convention geographical coverage is limited to areas within national jurisdiction. The designation of the Walters Shoal as an MPA is therefore not currently possible.

However, the opportunity of extending the geographical coverage of the framework convention into ABNJ could be considered. Indeed, the United Nations Environment Assembly (UNEA) of UNEP adopted a resolution in 2016 that "encourages the contracting parties to existing regional seas conventions to consider the possibility of increasing the regional coverage of those instruments in accordance with international law" (Christiansen, 2010). The parties to the Convention could therefore continue their discussions on the extension of the Nairobi Convention mandate, with a view to instituting a process to develop MPAs in ABNJ.

Expansion of the mandate of the Nairobi Convention would in theory allow for such action to be taken in the WIO region. However, some important limitations are to be noted. First, such MPAs are binding only on the parties to the Regional Seas Programme and not on third parties. This means that even if the Nairobi Convention were to take this step, any future MPA or management measures would not be applicable to non-parties. Second, the management of such MPAs would also require coordination and cooperation with other bodies. As the Nairobi Convention's mandate is limited, it would need to cooperate with other bodies to ensure that complementary protective measures were taken, by, for example SIOFA on fisheries and the ISA on deep-sea mining. Without cooperation between these organizations, any MPA declared under a Regional Seas Programme would be little more than "lines on a map".

A coalition-based approach

An alternative to the Regional Sea approach would be the use of a coalition-based approach (described above). Inspiration could be taken from the Pelagos Sanctuary, a small-scale, state-led effort focussing on cetacean conservation, and the efforts of the Sargasso Sea Alliance (SSA) (now the Sargasso Sea Commission), a broad and cooperative initiative launched and led by civil society and a champion territory.

The Pelagos Sanctuary incorporated both the territorial waters of these three states and areas that were, at that time ABNJ, which was subsequently recognized as a SPAMI by the Parties to the Protocol concerning Specially Protected Areas and Biological Diversity in the Mediterranean; consequently, all contracting parties to this Protocol must abide by the regulations adopted for the Sanctuary. A joint management plan was approved in 2004 and steps have been taken to respect the MPA (Mangos and André, 2012; Mayol et al, 2013). The founding states have also committed to seeking recognition as a PSSA by the IMO (Freestone et al., 2014).

In comparison to other regional marine areas, the institutional landscape in the Sargasso Sea is underdeveloped. No Regional Seas Programme or broad-based RFMO covers the region. The only land in this area is Bermuda, a British overseas island territory. The SSA, a partnership between the Government of Bermuda, NGOs, scientists and private donors, was launched in 2011 with the aim of establishing a management regime using existing sectoral bodies and measures, and to act as a case study of what can, and cannot, be achieved within existing institutions covering ABNJ (Kaplan et al., 2014). Bermuda, with the support of the SSA, has already submitted information regarding the Sargasso Sea for its potential designation as an EBSA, and a range of additional actions for advancing the conservation of this region are currently being considered.

The Pelagos and Sargasso Sea examples demonstrate that a limited number of states can advance conservation and sustainable use of ABNJ, but with considerable limitations. Learning from this approach, some WIO states could champion a process towards a better conservation of ABNJ ecosystems, including by jointly declaring the Walters Shoal as an MPA and committing to conserving its biodiversity. This process could also be a first step to ultimately recognizing the area as an MPA through an extended Nairobi Convention.

Inscription as a World Heritage Site

Nominating the Walters Shoal for inscription on the World Heritage List appears, at present, to be unfeasible. Parties to the World Heritage Committee (WHC) would first have to decide to allow for this possibility under the WHC. Assuming that the WHC is ultimately extended to ABNJ, the Walters Shoal would then have to be nominated in accordance with the agreed procedures for recognition of its "outstanding universal value". Nonetheless, states in the SWIO may wish to keep in mind the possibility for such recognition as they further develop scientific knowledge of the region and the Walters Shoal.

Dissociated management between the water column and the seabed

Should Madagascar's submission on the extent of its continental shelf be accepted by the United Nation's Commission on the Limits of the Continental Shelf (CLCS), this would have significant ramifications for the potential options available for the protection of the Walters Shoal. In particular, such a ruling would give Madagascar exclusive rights to explore and exploit the resources of the seabed around the Shoal (the status of the superjacent waters would, however, remain unchanged). This would mean that the ISA and RFMOs would have no mandate to implement management measures for the resources of the seabed in the area. In such a case, the establishment of a comprehensive MPA or other ABMT in the area would require action by Madagascar to implement measures concerning the continental shelf, along with complementary action by sectoral bodies concerning the superjacent waters that would remain part of the high seas.

RECOMMENDATIONS

As already emphasized by Rogers (2012), our knowledge of seamount and hydrothermal vent distribution and associated communities remains poor; in particular sampling on seamounts at equatorial latitudes is lacking. Previous surveys mainly focused on a few geographic areas (such as the North Atlantic and South-West Pacific), while little data exist for seamounts in other regions such as the Indian and the Southern Oceans. Consequently, the biological communities of tropical seamounts are poorly documented for large parts of the world. Most biological surveys on seamounts have been relatively shallow (for example, mostly less than 1500 m) and thus the great majority of deeper seamounts remain largely unexplored. As a result, the seafloor of the oceans is not mapped to a sufficient resolution to determine the position, size and shape of the majority of the seamounts, particularly those of less than 1000 m in elevation.

In spite of a series of intensive efforts in the 1960s (Zeitzschel, 1973), the basin-scale ecology and the fauna inhabiting seamounts of the Indian Ocean and the SWIR remain poorly known, in part because of the ocean's remoteness to nations with large-scale historical oceanographic research programmes. However, there is now an urgent need to explore these ecosystems to complete the picture of the biodiversity and productivity associated with the Indian Ocean (Demopoulos et al., 2003). Deep-sea studies on the SWIR are limited to a series of geological surveys of the Atlantis Bank (Dick, 1998) and to the hydrothermal vents in the vicinity of Melville Banks (Tao et al., 2007).

Studies of seamount and hydrothermal vent geology and physical oceanography are as a consequence limited. In addition, available biological data mainly originate from the deep-sea fishing industry or from national fisheries research programmes prospecting for exploitable fish stocks (FAO, 2002; Romanov, 2003). Until recently, the most detailed bathymetric charts of seamounts in the Indian Ocean and SWIO were those generated by fishing companies (Shotton, 2006). Thus, the two major international scientific databases of seamount information held predicted bathymetries for only three seamounts in this region and few biological records (Seamounts Catalog[1]).

Seamounts have an impact on circulation of the water masses (White et al., 2007) and their correct position is also necessary to forecast tsunami propagation accurately (Mofjeld et al., 2001). In this respect, a detailed list of seamounts, with their position and summit depth, can be invaluable for fisheries management (Fonteneau, 1991; Rogers, 1994), of particular interest for conservation, ideal candidates for offshore and high-seas MPAs (Roberts et al., 2002; Alder and Wood, 2004; Schmidt and Christiansen, 2004; Davies et al., 2007) and to implement the tsunami hazard mitigation programme. An accurate inventory of seamounts is necessary at both national and regional scales.

The growth of the research effort beyond national programmes, together with the ability to plan and carry out research at broader geographic scales, has considerably improved understanding over the last few decades of how seamounts and hydrothermal vents are structured, how they function as ecosystems and to what extent human activity has impacted them (Woodall et al., 2015, Serpetti et al., 2016). This scientific progress is evident in different fields, such as oceanography, geology, biology, ecology, taxonomy, conservation and fisheries.

The lack of knowledge about the location of seamounts and hydrothermal vents is, however, affecting a series of functional aspects, such as understanding of habitat and community heterogeneity and complexity (for example, species composition, distribution and growth rates), connectivity and faunal dispersal, the impact of human activities (long-term biomonitoring, species recovery, assessment of trawling impacts, etc), as well as conservation and management strategies and the development of marine protected areas. In particular, and as Rogers (2012) has to a certain extent already stated, scientists, conservation actors and managers should focus on the following aspects to further our understanding of seamounts and hydrothermal vents:

- Food-chain architecture (such as seamount associated fish and prey populations, bentho-pelagic coupling).
- Factors influencing the seamount-scale distribution of benthic organisms.
- Role of upwelling, vertical mixing, retention and resuspension on primary production.
- Life histories of seamount species (use of genetic studies).
- Long-term implications of climate change and threats (for example, fisheries, pollution, seabed mining, ocean acidification and presence of alien species) to seamount and hydrothermal vent communities (introduction of database for habitat loss and degradation).
- Seamount microbial communities (substantially underestimated at present).
- Linkages of the bottom fauna with the water column.
- Comparative studies, in order to compare fauna of seamounts and plumes with that of other bathyal bottoms at equivalent depths.
- Measurable conservation objectives that are relevant to current policies and sensitive to meaningful thresholds in order to establish meaningful indicators and monitoring protocols (Failing and Gregory 2003).
- Creation of EBSAs and MPAs.
- Identification of potential and new stressors (debris, noise, traffic vessels, tourism, etc).
- Creation of a list of endangered species (for both types of ecosystem).
- Improving access to data from seabed mining and high-seas fisheries activities, which is dramatically affecting scientific understanding and potential conservation measures.
- Identification of meaningful indicators, monitoring protocols and strategies to assess whether an MPA is achieving the established conservation and management objectives is a key component of overall management planning and implementation.

Overall knowledge of high-seas ecosystems remains limited due to insufficient funding for exploring and studying seamounts and hydrothermal vents. To meet these challenges, funding for field programmes is required. However, to ensure compatible sharing of result, standardized sampling methods and taxonomic resolution (inter-calibration assessment studies) should be introduced as different collecting instruments have different performances and data comparison may be biased to a certain degree. In the near future it will be particularly important to enhance collaboration among scientific communities of numerous countries and multiple disciplines. In addition, a minimum set of standardized seamount sampling protocols should be embraced as widely as possible by countries endorsing seamount and hydrothermal vent sampling programmes.

Additionally, to strengthen conservation and management of ABNJ, such as seamounts and hydrothermal

[1] www.earthref.org/databases/SC/main.htm

vents, marine resources and ecosystems, molecular tools need to be introduced and applied in all field programmes in order to:

- reveal evolutionary histories of marine species;
- discriminate between cryptic species (increasing information concerning existing biodiversity and associated distribution patterns);
- track effects of climate change (von der Heyden et al., 2010);
- identify marine invasive alien species (Darling and Tepolt, 2008); and
- identify potentially suspiciously-labelled seafood (von der Heyden et al., 2010).

Furthermore, genetic studies might demonstrate whether fragile and unique biota, such as that of seamount and vent ecosystems, are at an appropriate scale for protection, or whether they should be carefully protected (UNEP, 2006). Finally, as mentioned in the UNEP report (2006), availability of data regarding seamounts represents a problem. For many seamount studies, only summary data are publicly available, with analysis of species distribution patterns and studies on assemblage composition across different seamounts and regions not aggregated and often contained in the 'grey literature', such as unpublished fisheries research, trawler and commercial catch records (Tracey et al., 2011), thus not always readily accessible.

The conservation and management of marine biodiversity based on precautionary and ecosystem approaches are consequently hampered by the lack of fundamental scientific knowledge and understanding of these areas and their relationship with benthic and pelagic fish species of commercial interest. Furthermore, many seamounts are located in international waters, so the control of human activities that might adversely impact oceanic features (fishing, seabed mining activities, etc) is a major challenge. To address these issues, appropriate mechanisms that bridge science and policy-making must be established.

The knowledge gaps mentioned above need to be addressed and discussed internationally in order to create solid scientific evidence that might enable institutions, local communities and, in particular, scientists, to interpret the causes and impacts of present and future environmental changes and threats and consequently to integrate seamount and hydrothermal vent ecosystems into conservation strategies.

REFERENCES

Alder, J. and Wood, L. (2004). Managing and protecting seamounts ecosystems. *Fish. Cent. Res. Rep*. 12(5), pp. 67–73

Andrianaivojaona, C., Kasprzyk, Z. and Dasylva, D. (1992). *Malagasy Fisheries and Aquaculture, a diagnostic assessment*. Project report FAO, Rome, Italy

Annamalai, H. and Murtugudde, R. (2004). *Role of the Indian Ocean in regional climate variability. Earth's climate: the ocean–atmosphere interaction.* Geophysical Monograph Series 47. American Geophysical Union, Washington DC, USA

Bach, P., Romanov, E., Rabearisoa, N., Akbaraly, A. and Sharp, A. (2011). *Report for 2010 exhaustive data collected by observers on board largest pelagic long-liners based in La Reunion*. IOTC–2011–WPEB07–INF29, pp. 24–27. North Malé, Maldives

Bellard, C., Leclerc, C. and Courchamp, F. (2013). Impact of sea level rise on the 10 insular biodiversity hotspots. *Glob. Ecol. Biogeogr.* 23, pp. 203–212. https://doi.org/10.1111/geb.12093

Bensch, A., Gianni, M., Grébroval, D., Sanders, J.S. and Hjort, A. (2008). *Worldwide review of bottom fisheries in the high seas.* FAO Fisheries and Aquaculture Technical Paper No. 522. Rome, Italy

Branch, T.A. (2001). A review of orange roughy *Hoplostethus atlanticus* fisheries, estimation methods, biology and stock structure. In *A Decade of Namibian Fisheries Science* (eds. A.I.L. Payne, S.C. Pillar and R.J.M. Crawford). *South Afr. J. Mar. Sci*. 23, pp. 181–203

Brewin, P.E., Stocks, K.I., Haidvogel, D.B. (2009). Effects of oceanographic retention on decapod and gastropod community diversity on seamounts. *Mar. Ecol. Prog. Ser.* 383, pp. 225–237. https://doi.org/10.3354/meps07987

Christiansen, S. (2010). *Background document for the High Seas MPAs: Regional Approaches and Experiences side event at the 12th UNEP Global Meeting of the Regional Seas Conventions and Action Plans, 20th of September 2010*. UNEP. Bergen, Norway

Clark, M.R., Rowden, A.A. and Schlacher, T. (2010). The ecology of seamounts: Structure, function and human impacts. *Annu. Rev. Mar. Sci.* 2, pp. 253–278. https://doi.org/10.1890/09-0548.1; https://doi.org/10.1890/09-1268.1

Clark, M.R., Schlacher, T., Rowden, A.A., Stocks, K.I. and Consalvey, M. (2012). Science Priorities for Seamounts: Research Links to Conservation and Management. *PLoS one* 7, e29232. https://doi.org/10.1371/journal.pone.0029232 Accessed at 01 May 2018

Clark, M.R., Vinnichenko, V.I., Gordon, J.D.M., Beck-Bulat, G.Z., Kukharev, N.N. and Kakora, A.F. (2007). Large-scale distant-water trawl fisheries on seamounts. In *Seamounts: ecology, fisheries and conservation* (eds. T.J. Pitcher, T. Morato, P.J.B. Hart, M.R. Clark, N. Haggan and R.S. Santos), pp. 361–399. Oxford, UK: Blackwell Publishing. https://doi.org/10.1002/9780470691953.ch17

Collette, B.B. and Paring, N.V. (1991). Shallow-water fishes of Walters Shoals, Madagascar Ridge. *Bul. Mar. Sci.* 48, pp. 1–22

Consalvey, M., Clark, M.R., Rowden, A.A. and Stocks, K.I. (2010). *Life on Seamounts. In Life in the World's Oceans: diversity, distribution, and abundance* (ed. A.D. McIntyre), pp. 123-138. Oxford, UK: Wiley Blackwell. https://doi.org/10.1002/9781444325508.ch7

Costello, M.J., Cheung, A. and De Hauwere, N. (2010). The surface area, and the seabed area, volume, depth, slope, and topographic variation for the world's seas, oceans and countries. *Environ. Sci. Technol*. 44, pp. 8821–8828. https://doi.org/10.1021/es1012752

Coyle, K., Konar, B., Blanchard, A., Highsmith, R., Carroll, J., Carroll, M., Denisenko, S. and Sirenko, B. (2007). Potential effects of temperature on the benthic in faunal community on the southeastern Bering Sea shelf: Possible impacts of climate change. *Deep-Sea Res.* II 54, pp. 2885–2905. https://doi.org/10.1016/j.dsr2.2007.08.025

Cuyvers, L., Berry, W., Gjerde, K., Thiele, T. and Wilhem, C. (2018). *Deep seabed mining: a rising environmental challenge.* Gland, Switzerland: IUCN and Gallifrey Foundation. x + 74 pp.

Darling, J.A. and Tepolt, C.K. (2008). Highly sensitive detection of invasive shore crab (*Carcinus maenas* and *Carcinus aestuarii*) larvae in mixed plankton samples using polymerase chain reaction and restriction fragment length polymorphisms (PCR-RFLP). *Aquat. Invasions* 3(2), pp. 141–152. https://doi.org/10.3391/ai.2008.3.2.4

Das, P., Iyer, S.D., Kodagali, V.N. and Krishna, K.S. (2005). A new insight into the distribution and origin of seamounts in the Central Indian Ocean Basin. *Mar. Geod.* 28, pp. 259-269. https://doi.org/10.1080/01490410500204637

Davies, A.J., Roberts, J.M. and Hall-Spencer, J. (2007). Preserving deep-sea natural heritage: emerging issues in offshore conservation and management. *Biol. Conserv.* 138(3-4), pp. 299–312. https://doi.org/10.1016/j.biocon.2007.05.011

Demopoulos, A.W.J., Smith C.R. and Tyler, P.A. (2003). Ecology of the deep Indian Ocean floor. In *Ecosystems of the World Volume 28: Ecosystems of the Deep Ocean* (ed. P.A. Tyler). Elsevier, Amsterdam, Netherlands

Devey, C.W., Fisher, C.R. and Scott, S. (2007). Responsible science at hydrothermal vents. *Oceanography* 20, pp. 162–171. https://doi.org/10.5670/oceanog.2007.90

Dick, H. (1998). Indian Ocean's Atlantic Bank yields deep-Earth insight. *Oceanus* 41

Dick, H.J.B., Lin, J. and Schouten, H. (2003). Ultra-Slow Spreading—A New Class of Ocean Ridge. *Nature* 426, pp. 405–412. https://doi.org/10.1038/nature02128

Etnoyer, P., Wood, J. and Shirley, T. (2010). How large is the seamount biome? *Oceanography* 23, pp. 206–209. https://doi.org/10.5670/oceanog.2010.96

Failing, L. and Gregory, R. (2003). Ten common mistakes in designing biodiversity indicators for forest policy. J. *Environ. Manage.* 68(2), pp. 121–132. https://doi.org/10.1016/S0301-4797(03)00014-8

FAO (2002). *Report of the Second Ad Hoc Meeting on Management of Deepwater Fisheries Resources of the Southern Indian Ocean*. FAO Fisheries Report. No. 677, p. 106 Fremantle. FAO, Rome, Italy

FAO (2010). *Management of demersal fisheries resources of the Southern Indian Ocean*. FAO, Rome

FFEM (2013). Note d'Engagement de Projet (NEP). Conservation et exploitation durable des écosystèmes de monts sous-marins et sources hydrothermales du Sud-Ouest de l'Océan Indien au-delà des zones de juridiction nationale. 67 pp.

Fonteneau, A. (1991). Monts sous-marins et thons dans l'Atlantique tropical est. *Aquat. Living Resour.* 4(1), pp. 13–25. https://doi.org/10.1051/alr:1991001

Freestone, D., Johnson, D., Ardron, J., Morrison, K.K. and Unger, S. (2014). Can existing institutions protect biodiversity in areas beyond national jurisdiction? Experiences from two on-going processes. *Mar. Policy* 49, pp. 167–175. https://doi.org/10.1016/j.marpol.2013.12.007

Fryer, P. (1992). Mud volcanoes of the Marianas. *Scientific American* 266, pp. 46–52. https://doi.org/10.1038/scientificamerican0292-46

Game, E.T., Grantham, H.S., Hobday, A.J., Pressey, R.L., Lombard, A.T., Beckley, L.E., Gjerde, K., Bustamante, R. Possingham, H.P. and Richardson, A.J. (2009). Pelagic protected areas: the missing dimension in ocean conservation. *Trends Ecol. Evol*. 24(7), pp. 360–369. https://doi.org/10.1016/j.tree.2009.01.011

Gianni, M., Currie, D., Fuller, S., Speer, L., Ardron, J., Weeber, B., Gibson, M., Roberts, G. Sack, K., Owen, S. and Kavanagh, A. (2011). *Unfinished business: a review of the implementation of the provisions of United Nations General Assembly resolutions 61/105 and 64/72, related to the management of bottom fisheries in areas beyond national jurisdiction.* Deep Sea Conservation Coalition, 53 pp. http://www.savethehighseas.org/

Gopal, K., Tolley, K.A., Groeneveld, J.C. and Matthee, C.A. (2006). Mitochondrial DNA variation in spiny lobster *Palinurus delagoae* suggests genetically structured populations in the southwestern Indian Ocean. *Mar. Ecol. Prog. Ser*. 319, pp. 191–198. https://doi.org/10.3354/meps319191

Groeneveld, J.C., Griffiths, C.L. and Van Dalsen, A.P. (2006). A new species of spiny lobster, *Panulirus barbarae* (Decapoda, Palinuridae) from Walters Shoals on the Madagascar Ridge. *Crustaceana* 79(7), pp. 821–833. https://doi.org/10.1163/156854006778008177

Grüss, A., Kaplan, D. M., Hart, D. R (2011). Relative Impacts of Adult Movement, Larval Dispersal and Harvester Movement on the Effectiveness of Reserve Networks. *PLoS one* 6(5), e19960. https://doi.org/10.1371/journal.pone.0019960

Gubbay, S. (2003). *Seamounts of the North-East Atlantic.* Hamburg and Frankfurt am Main, Germany: OASIS and World Wildlife Fund (WWF). Germany

Harley, S.J. and Suter, J.M. (2007). The potential use of time-area closures to reduce catches of bigeye tuna (*Thunnus obesus*) in the purse-seine fishery of the eastern Pacific Ocean. *Fish. Bull.* 105(1), pp. 49–62

Hyrenbach, K.D., Forney, K.A. and Dayton, P.K. (2000). Marine protected areas and ocean basin management. *Aquat. Conserv.* 10, pp. 437–458

ISA (2011). Draft environmental management plan for the Clarion-Clipperton Zone, 28 January 2011, adopted 22 July 2012, ISBA/18/C/22. International Seabed Authority (ISA)

Kaiser, M. (2005). *Marine ecology: Processes, systems and impacts*. Oxford University Press. Oxford, UK

Kaplan, D.M., Chassot, E., Amande, J.M., Dueri, S., Dagorn, L. and Fonteneau, A. (2014). Spatial management of Indian Ocean tropical tuna fisheries: potential and perspectives. *ICES J. Mar. Sci*. 71(7), pp. 1728–1749

Kaplan, D., Chassot, E., Gruss, A. and Fonteneau, A. (2010). Pelagic MPAs: The devil is in the details. *Trends Ecol. Evol.* 25(2), pp. 62–63. https://doi.org/10.1016/j.tree.2009.09.003

Kim, S.S. and Wessel, P. (2011). New global seamount census from altimetry-derived gravity data. *Geophys. J. Int*. 186, pp. 615–631. https://doi.org/10.1111/j.1365-246X.2011.05076.x

Kimani, E.N., Okemwa G.M and Kazungu, J.M (2009). Fisheries in the Southwest Indian Ocean: Trends and Governance Challenges. In *The Indian Ocean; Resource and Governance Challenges* (eds. Laipson, E. and A. Pandya), The Henry L. Stimson Centre, pp. 3–90. Washington DC, USA

Kitchingman, A. and Lai, S. (2004). Inferences of potential seamount locations from mid-resolution bathymetric data. In *Seamounts: Biodiversity and Fisheries* (eds. T. Morato and D. Pauly), pp.7–12. Fisheries Centre Research Report 12(5)

Koslow, J.A., Boehlert, G., Gordon, J., Lorance, P. and Parin, N. (2000). Continental slope and deep-sea fisheries: implications for a fragile ecosystem. *ICES J. Mar. Sci.* pp. 548–557. https://doi.org/10.1006/jmsc.2000.0722

Le Quesne, W.J.F. and Codling, E.A. (2008). Managing mobile species with MPAs: the effects of mobility, larval dispersal, and fishing mortality on closure size. *ICES J. Mar. Sci.* 66(1), pp. 122–131. https://doi.org/10.1093/icesjms/fsn202

Lutjeharms, J.R.E. (2007). Three decades of research on the greater Agulhas Current. *Ocean Sci.* 3, pp. 129–147. https://doi.org/10.5194/os-3-129-2007

Mangos, A. and André, S. (2012). A*nalysis of Mediterranean marine environment protection: the case of the Pelagos Sanctuary.* Blue Plan Notes No. 20, Plan Bleu, UNEP/MAP Regional Activity Centre, 4 pp

Maxwell, S. M. and Morgan, L. E. (2012). *Examination of Pelagic Marine Protected Area Management with Recommendations for the Pacific Remote Islands Marine National Monument*. Seattle: Marine Conservation Institute.

Mayol, P., Labach, H., Couvat, J., Ody, D. and Robert, P. (2013). Particularly Sensitive Sea Area (PSSA): An IMO status as an efficient management tool of Pelagos. *IMPAC* 3, Oct. 2013, Marseille, pp. 21-27

McClain, C.R., Lundsten, L., Ream, M., Barry J. and DeVogelaere, A. (2009). Endemicity, biogeography, composition and community structure on a northeast Pacific seamount. *PLoS one* 4(1), e4141. https://doi.org/10.1371/journal.pone.0004141

Menard, H.W. (1964). *Marine Geology of the Pacific*. McGraw-Hill, New York, USA, 271 pp.

Moffitt, E.A., Botsford, L.W., Kaplan, D.M. and O'Farrell, M.R. (2009). Marine reserve networks for species that move within a home range. *Ecol. Appl.* 19(7), pp. 1835–1847. https://doi.org/10.1890/08-1101.1 38

Mofjeld, H.O., Titov, V.V., Gonzalez, F.I. and Newman, J.C. (2001). Tsunami scattering provinces in the Pacific Ocean. *Geophys. Res. Lett.* 28(2), pp. 335–337. https://doi.org/10.1029/2000GL011710

Morato, T. and Clark, M.R. (2007). Seamount fishes: ecology and life histories. In *Seamounts: ecology, fisheries and conservation* (eds. T.J. Pitcher, T. Morato, P.J.B. Hart, M.R. Clark, N. Haggan and R.S. Santos), pp.170–188. Oxford, UK: Blackwell Publishing. https://doi.org/10.1002/9780470691953.ch9

Morato, T., Kvile, K.Ø., Taranto, G.H., Tempera, F., Narayanaswamy, B.E., Hebbeln, D., Menezes, G.M., Wienberg, C., Santos, R.S. and Pitcher, T.J. (2013). Seamount physiography and biology in the north-east Atlantic and Mediterranean Sea. *Biogeosciences* 10, pp. 3039–3054. https://doi.org/10.5194/bg-10-3039-2013

New Zealand Government (2012). Report on New Zealand's Implementation of Operative Paragraphs 80 and 83-90 of Resolution 61/105. ISBN No: 978-0-478-38833-6. Available at http://www.un.org/depts/los/general_assembly/contributions_fisheries/new_zealand.pdf. Accessed 25 February 2017

O'Leary, B.C., Brown, R.L., Johnson, D.E., von Nordheim, H., Ardron, J., Packeiser, T. and Roberts, C.M. (2012). The first network of marine protected areas (MPAs) in the high seas: The process, the challenges and where next. *Mar. Policy*, 36(3), 598–605. https://doi.org/10.1016/j.marpol.2011.11.003

Peng, Z.X. and Mahoney, J.J. (1995). Drillhole lavas from the northwestern Deccan Traps, and the evolution of Reunion hotspot mantle. *Earth Planet. Sci. Lett*. 134(1-2), pp. 169–185

Penney, A.J. and Guinotte, J.M. (2013). Evaluation of New Zealand's high-seas bottom trawl closures using predictive habitat models and quantitative risk assessment. *PloS One*, 8(12). https://doi.org/10.1371/journal.pone.0082273

Penney, A., Parker, S. and Brown, J. (2009). Protection measures implemented by New Zealand for vulnerable marine ecosystems in the South Pacific Ocean. *Mar. Ecol. Prog. Ser.* 397 (Unga 2006), pp. 341–354. https://doi.org/10.3354/meps08300

Portela, J.M., Pierce, G.J., del Río, J.L., Sacau, M., Patrocinio, T. and Vilela, R. (2010). Preliminary description of the overlap between squid fisheries and VMEs on the high seas of the Patagonian Shelf. *Fish. Res.* 106(2), pp. 229–238. https://doi.org/10.1016/j.fishres.2010.06.009

Postaire, B., Bruggemann, J.H., Magalon, H. and Faure, B. (2014). Evolutionary Dynamics in the Southwest Indian Ocean Marine Biodiversity Hotspot: A Perspective from the Rocky Shore Gastropod Genus *Nerita. PLoS one* 9(4), e95040. https://doi.org/10.1371/journal.pone.0095040 accessed 1 May 2018

Read, J.F., Lucas, M.I., Holley, S.E. and Pollard R.T. (2000). Phytoplankton, nutrients and hydrography in the frontal zone between the Southwest Indian Subtropical gyre and the Southern Ocean. *Deep Sea Res. Part I* 47(12), pp. 2341–2368. https://doi.org/10.1016/S0967-0637(00)00021-2

Richer De Forges, B., Koslow, J.A. and Poore, G.C.B. (2000). Diversity and endemism of the benthic seamount fauna in the south-west Pacific. *Nature* 405, pp. 944–947. https://doi.org/10.1038/35016066

Roberts, C.M., McClean, C.J., Veron, J.E.N., Hawkins, J.P. and Allen, G.R. (2002). Marine biodiversity hotspots and conservation priorities for tropical reefs. *Science* 295, pp. 1280–1284. https://doi.org/10.1126/science.1067728

Robison, B.H. (2009). Conservation of deep pelagic biodiversity. *Conserv. Biol.* 23(4), pp. 847–858. https://doi.org/10.1111/j.1523-1739.2009.01219.x

Rochette, J., Gjerde, K., Druel, E., Ardron, J. a., Craw, A., Halpin, P., Pendleton, L., Teleki, K. and Cleary, J. (2014a). Delivering the Aichi target 11: challenges and opportunities for marine areas beyond national jurisdiction. *Aquat. Conserv.* 24(S2), pp. 31–43. https://doi.org/10.1002/aqc.2507

Rochette, J., Unger, S., Herr, D., Johnson, D., Nakamura, T., Packeiser, T., Proelss, A., Visbeck, M., Wright, A. and Cebrian, D. (2014b). The regional approach to the conservation and sustainable use of marine biodiversity in areas beyond national jurisdiction. *Mar. Policy* 49, pp. 109–117. https://doi.org/10.1016/j.marpol.2014.02.005

Rogers, A., Alvheim, O., Bemanaja, E., Benivary, D., Boersch-Supan, P., Bornman, T., Cedras, R., Plessis, N.D., Gotheil, S., Høines, Å., Kemp, K., Kristiansen, J., Letessier, T., Mangar, V., Mazungula, N., Mørk, T., Pinet, P., Read, J. and Sonnekus, T. (2009). *Preliminary Cruise Report "Dr. Fritjof Nansen"*, Southern Indian Ocean Seamounts. IUCN/UNDP/ASCLME/NERC/EAF Nansen Project 2009 Cruise 410, 12 November–19 December

Rogers, A.D. (2004). *The biology, ecology and vulnerability of seamount communities.* Report for the World Conservation Union for the 7th Convention of Parties, Convention for Biodiversity, Kuala Lumpur, pp. 8–19 February

Rogers, A.D. (2012). *An ecosystem approach to management of seamounts in the Southern Indian Ocean: volume 1: overview of seamount ecosystems and biodiversity.* IUCN. Gland, Switzerland

Rogers, A.D. (1994). The biology of seamounts. *Adv. Mar. Biol.* 13, pp. 305–350

Rogers, A.D. and Gianni, M. (2011). *The Implementation of UNGA Resolutions 61/105 and 64/72 in the Management of Deep-Sea Fisheries on the High Seas.* Report prepared for the Deep-Sea Conservation Coalition. International Programme on State of the Ocean, London, United Kingdom, 97 pp.

Rogers, A.D. and Taylor, M.L. (2012). Benthic biodiversity of seamounts in the southwest Indian Ocean Cruise report. R/V James Cook 066 Southwest Indian Ocean Seamounts expedition, 7 November–21 December 2011

Rogers, A.D., Baco, A. and Griffiths, H. (2007). Corals on seamounts. In *Seamounts: ecology, fisheries and conservation* (eds. T.J. Pitcher, T. Morato, P.J.B. Hart, M.R. Clark, N. Haggan and R.S. Santos), pp. 141–169, Blackwell Publishing. Oxford, UK. https://doi.org/10.1002/9780470691953.ch8

Romanov, E.V. (2003). *Summary and review of Soviet and Ukrainian scientific and commercial fishing operations on the deepwater ridges of the southern Indian Ocean.* FAO Fisheries Circular 991. Rome, Italy

Rowden, A.A., Dower, J.F., Schlacher, T.A., Consalvey, M. and Clark, M.R. (2010). Paradigms in seamount ecology: fact, fiction, and future. *Mar Ecol.* 31 (suppl 1), pp. 226–239

Samadi, S., Bottan, L. and Macpherson, E. (2006). Seamount endemism questioned by the geographical distribution and population genetic structure of marine invertebrates. *Mar. Biol.* 149, pp. 1463–1475. https://doi.org/10.1007/s00227-006-0306-4

Santos, R.S., Christiansen, S., Christiansen, B. and Gubbay, S. (2009). Toward the conservation and management of Sedlo Seamount: A case study. *Deep-Sea Research Part II* 56, pp. 2720–2730. https://doi.org/10.1016/j.dsr2.2008.12.031

Schmidt, R. and Schmincke, H.-U. (2000). Seamounts and Island Building. In *Encyclopedia of Volcanoes* (ed. H. Sigurdsson), pp. 383–402. Academic Press, San Diego, USA

Schmidt, S. and Christiansen, S. (2004). *The Offshore MPA Toolbox: Implementing Marine Protected Areas in the North-east Atlantic Offshore: Seamounts—A Case Study*. Hamburg and Frankfurt am Main, Germany: OASIS and WWF, Germany

Scovazzi, T. (2011). *Note on the establishment of marine protected areas beyond national jurisdiction or in areas where the limits of national sovereignty or jurisdiction have not yet been defined in the Mediterranean Sea.* Tunis RAC/SPA

Serpetti, N., Taylor, M.L., Brennan, D., Green, D.H., Rogers, A.D., Paterson, G.L.J. and Narayanaswamy, B.E. (2016). Ecological adaptations and commensal evolution of the Polynoidae (Polychaeta) in the Southwest Indian Ocean Ridge: A phylogenetic approach. *Deep Sea Res. Part II* 137, pp. 273-281. https://doi.org/10.1016/j.dsr2.2016.06.004

Shotton, R. (2006). *Management of demersal fisheries resources of the Southern Indian Ocean*. FAO, Rome, Italy

Simard F. and Spadone, A. (eds.) (2012). *An Ecosystem Approach to Management of Seamounts in the Southern Indian Ocean.* Volume 2 – Anthropogenic Threats to Seamount Ecosystems and Biodiversity, iv+64 pp. IUCN, Gland, Switzerland

Spalding, M.F., Fox, H.E., Allen, G.R., Davidson, N., Ferdaña, Z.A., Finlayson, M., Halpern, B.S., Jorge, M.A., Lombana, A., Lourie, S.A., Martin, K.D., McManus, E., Molnar, J., Recchia, C.A. and Robertson, J. (2007). Marine ecoregions of the world: a bioregionalization of coastal and shelf areas. *Bioscience* 57(7), pp. 573–583. https://doi.org/10.1641/B570707

Staudigel, H. and Clague, D.A. (2010). The geological history of deep-sea volcanoes: Biosphere, hydrosphere, and lithosphere interactions. *Oceanography* 23(1), pp. 58–71. https://doi.org/10.5670/oceanog.2010.62

Stocks, K.I. (2009). SeamountsOnline: an online information system for seamount biology. Version 2009-1. Available at: http://seamounts.sdsc.edu accessed 1 May 2018

Stocks, K.I. and Hart, P.J.B. (2007). Biogeography and biodiversity of seamounts. In *Seamounts: ecology, fisheries and conservation* (eds. T.J. Pitcher, T. Morato, P.J.B. Hart, M.R. Clark, N. Haggan and R.S. Santos), pp. 252–281. Blackwell Publishing, Oxford, UK. https://doi.org/10.1002/9780470691953.ch13

Sumaila, U., Zeller, D., Watson, R., Alder, J. and Pauly, D. (2007). Potential costs and benefits of marine reserves in the high seas. *Mar. Ecol. Prog. Ser.* 345, pp. 305–310. https://doi.org/10.3354/meps07065

Tao, C., Lin, J., Guo, S., Chen, Y.J., Han, G.W.X., German, C.R., Yoerger, D.R., Zhu, J., Zhou, N., Su, X., Baker, E.T. and DY115-19 Science Party. (2007). First discovery and investigation of a high-temperature hydrothermal vent field on the ultra-slow spreading Southwest Indian Ridge. *EOS Trans AGU*, Fall Meet Suppl: Abstract T52B-07

Torres-Irineo, E., Gaertner, D., Delgado De Molina, A. and Ariz, J. (2011). Effects of time-area closure on tropical tuna purse-seine fleet dynamics through some fishery indicators. *Aquat. Living Resour*. 24, pp. 337–350. https://doi.org/10.1051/alr/2011143

Tracey, D., Rowden, A., Mackay, K. and Compton, T. (2011). Habitat-forming coldwater corals show affinity for seamounts in the New Zealand region. *Mar. Ecol. Prog. Ser.* 430, pp. 1–22. https://doi.org/10.3354/meps09164

UNEP (2006). *Ecosystems and Biodiversity in Deep Waters and High Seas.* UNEP Regional Seas Reports and Studies No. 178. UNEP and IUCN, Nairobi, Kenya and Gland, Switzerland

Vereshchaka, A.L. (1995). Macroplankton in the near-bottom layer of continental slopes and seamounts. *Deep Sea Res. Part I* 42, pp. 1639–1668. https://doi.org/10.1016/0967-0637(95)00065-E

von der Heyden, S., Lipinski, M.R. and Matthee, C.A. (2010). Remarkably low mtDNA control region diversity in an abundant demersal fish. *Mol. Phylogenet. Evol.* 55, pp. 1183–1188. https://doi.org/10.1016/j.ympev.2009.09.018

Wessel, P. (2007). *Seamount characteristics. In Seamounts: ecology, fisheries and conservation* (eds. T.J. Pitcher, T. Morato, P.J.B. Hart, M.R. Clark, N. Haggan and R.S. Santos), pp. 3–40. Blackwell Publishing, Oxford, UK https://doi.org/10.1002/9780470691953.ch1

Wessel, P., Sandwell, D.T. and Kim, S.-S. (2010). The global seamount census. *Oceanography* 23, pp. 24–33. https://doi.org/10.5670/oceanog.2010.60

White, M., Bashmachnikov, I., Aristegui, J. and Martins, A. (2007). Physical processes and seamount productivity. In *Seamounts: ecology, fisheries and conservation* (eds. T.J. Pitcher, T. Morato, P.J.B. Hart, M.R. Clark, N. Haggan and R.S. Santos), pp. 65–84. Blackwell Publishing, Oxford, UK. https://doi.org/10.1002/9780470691953.ch4

Woodall, L.C., Robinson, L.F., Rogers, A.D., Narayanaswamy, B.E. and Paterson, G.L.J. (2015). Deep-sea litter: a comparison of seamounts, banks and a ridge in the Atlantic and Indian Oceans reveals both environmental and anthropogenic factors impact accumulation and composition. *Front. Mar. Sci.* 2. https://doi.org/10.3389/fmars.2015.00003

Yesson, C., Clark, M.R., Taylor, M. and Rogers, A.D. (2011). The global distribution of seamounts based on 30-second bathymetry data. *Deep Sea Res. Part I* 58, pp. 442–453. https://doi.org/10.1016/j.dsr.2011.02.004

Young, H.S., Maxwell, S.M., Conners, M.G. and Shaffer, S.A. (2015). Pelagic marine protected areas protect foraging habitat for multiple breeding seabirds in the central Pacific. *Biol. Conserv.* 181, pp. 226–235. https://doi.org/10.1016/j.biocon.2014.10.027

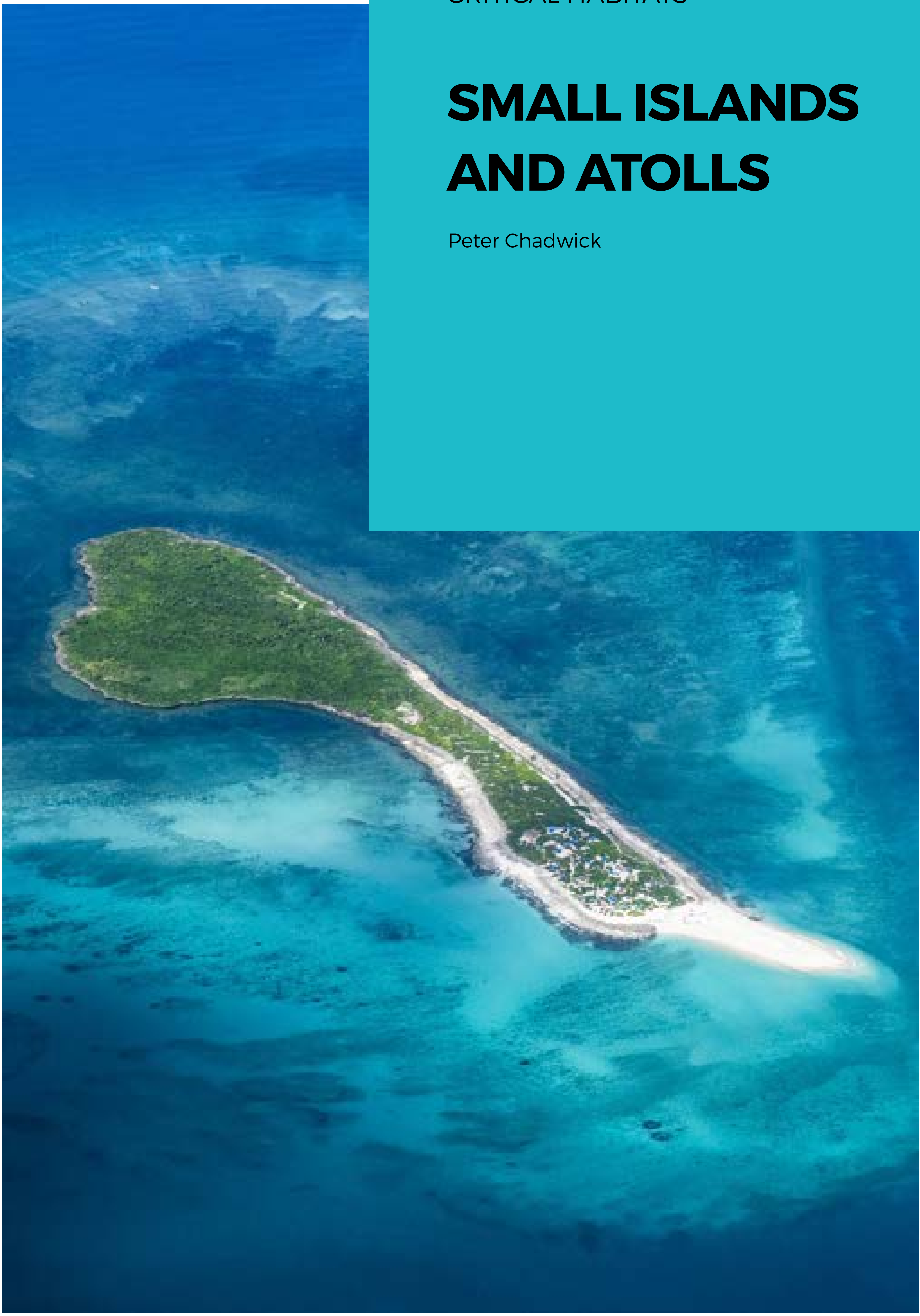
Zeitzschel, B. (ed.) (1973). *The biology of the Indian Ocean.* Ecological studies 3. Springer-Verlag, New York, Heidelberg and Berlin. https://doi.org/10.1007/978-3-642-65468-8

Zucchi, S., Ternon, J.-F., Demarcq, H., Ménard, F., Guduff, S. and Spadone, A. (2018). Oasis for marine life. State of knowledge on seamounts and hydrothermal vents, vi + 50pp. IUCN, Gland, Switzerland https://doi.org/10.2305/IUCN.CH.2018.14.en-fr

CRITICAL HABITATS

SMALL ISLANDS AND ATOLLS

Peter Chadwick



A small island surrounded by tropical reef in the northern Quirimbas, Mozambique. © Peter Chadwick

DESCRIPTION AND CHARACTERISTICS

Globally, islands only make up about 5 per cent of the Earth's landmass and yet they are home to over 20 per cent of the world's biodiversity. Given their high levels of endemism, 41 per cent of the world's critically endangered and endangered species are found on islands (Spatz et al., 2017). They are important roosting and nesting sites for seabirds and migrant birds and are essential nesting sites for turtles. Their surrounding waters are vital nursery areas for many fish species and particularly sharks and rays. In tropical waters, almost all have well-established coral reef systems with a high diversity of associated species. Around the more isolated of these islands and atolls, some corals have also shown resilience against bleaching from warming events thus making them critically important as seeding areas to allow re-establishment.

Because of their small size and low height above sea level, islands and atolls are amongst the first systems that will suffer the consequences of accelerating sea level rise as a result of climate change. Despite their often remoteness, they are being plagued by an increasing accumulation of waste pollution and they have been invaded with alien invasive species with hugely destructive impacts to their fragile biodiversity.

As the focus is on critical habitats and associated species, this chapter provides descriptions and analysis of small islands (in terms of area, which should in most cases not exceed 100 km^2) that are essentially oceanic and isolated. A few exceptions may exist for those small islands that are distinct even though they might be considered coastal islands, not too distant from major land masses. These small islands are hotspots for biodiversity and essential for the reproduction and migratory routes of many charismatic fauna.

The small islands of the Western Indian Ocean (WIO) fall under a full suite of country designations and vary in size from relatively large landmasses to small isolated coral atolls that are widely scattered across the ocean. Together they have been identified as one of the world's biodiversity hotspots (Myers et al., 2000). Formal protection has already been afforded to some of the islands and two sites have been listed for UNESCO World Heritage status. However, far more conservation effort is needed to ensure the preservation of these biodiversity hotspots through additional proclamation of marine protected areas (MPAs) and through ensuring that those currently under formal protection are effectively managed.

The islands of the WIO can be distinguished into three groups, namely the granitic islands, the low coralline islands and the raised coralline islands. The granitic islands are mostly built of ancient continental granite that is at least 650 million years old, but others are more recent, comprising volcanic syenite rock that is about 60 million years old. The low coralline islands and sand cays are all formed relatively recently from marine coral and shell sand sediments. Most of these are less than 3 m above sea level, last emerging around 5000 years ago and are prone to periodic inundation. Reef-building corals that were then uplifted also formed the raised coralline islands, such as Aldabra.

These islands have been submerged and have emerged again above sea level several times with the most recent emergence being about 125 000 years ago (Hill and Curry, 2007). There are more than 40 main island and atoll groups within the WIO with brief descriptions of each listed in Table 1.

In addition, there are many smaller rocky outcrops that occur, but these are too small to be of major conservation significance and have therefore not been listed.

Table 1: Main island and atoll groups within the Western Indian Ocean.

AFRICAN BANKS	
JURISDICTION	Seychelles
AREA	8.27 km^2
GEOGRAPHY	Pseudo atoll with two islands.
DESCRIPTION	Comprises two small sandstone islands. The underwater bank extends 4 km north/south and 3 km east/west. It has a shallow coral ring around it that extends to 20–36 m depth.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Charter yachts occasionally visit North island.

BIODIVERSITY	It is a significant nesting site for terns, the colonies of which are subject to frequent exploitation and disturbance by poachers. It has important populations of black-naped and sooty terns and brown noddies. Green and Hawksbill turtles nest there. The surrounding waters are important for large pelagic fish and sharks.
CONSERVATION STATUS	Listed as a protected area and also forms a 750 ha Important Bird Area (IBA) (Birdlife International, 2018a).
AGALÉGA ISLANDS	
JURISDICTION	Mauritius
AREA	70 km^2 (North island: 2.25 km^2; South Island: 31.5 km^2)
GEOGRAPHY	Coralline islands separated by a sand bank that can be crossed at low tide.
DESCRIPTION	Comprises of two islands with an airstrip being present on North Island.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	The island is leased to the Indian Military for development of strategic assets. There are two villages on North Island and one small village on South Island. The economy is based on coconut oil with plantations covering a vast portion of the islands.
BIODIVERSITY	The endemic Agalega day gecko (*Phelsuma borbonica agalegae*) occurs on the islands. Green and Hawksbill turtles regularly breed on the islands but evidence of poaching still exists (Webster et al., 2016).
CONSERVATION STATUS	Managed by the Outer Island Development Corporation with no formal conservation status.
ALDABRA ISLAND GROUP	
JURISDICTION	Seychelles
AREA	210 000 km^2
GEOGRAPHY	Atolls and raised reef.
DESCRIPTION	Comprises of Aldabra Atoll (four main atolls and 40 smaller islets), Assumption Island (raised reef), Cosmoledo (two main atolls and 18 smaller islets) and Astove Island (a raised atoll with one island).
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Three of the islands are inhabited and one is uninhabited. Assumption Island is the main settlement where a military base is in the process of being constructed. Aldabra has a well-established research station and Astove has a population of only a few people.
BIODIVERSITY	Harbours some of the least impacted coral reefs in the region. Green and Hawksbill turtles breed on the islands. Aldabra is the second largest atoll in the world, supporting a wide range of species. It is an important breeding and roosting area for many seabirds and has high levels of endemism associated with it (Friedlander et al., 2015).
CONSERVATION STATUS	Special nature reserve, World Heritage Site (WHS), Ramsar wetland and IBA.
ALPHONSE ISLAND GROUP	
JURISDICTION	Seychelles
AREA	1.71 km^2
GEOGRAPHY	Atoll
DESCRIPTION	Comprises Alphonse Island, Bijoutier and St Francois.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Small resident population on the island with luxury accommodation and game fishing tourism.
BIODIVERSITY	Various seabirds breed on the islands despite the presence of rats and cats. Coral reefs have been negatively impacted by sea temperature warming events. Green and Hawksbill turtles nest on the islands.
CONSERVATION STATUS	Managed by the Island Conservation Society with the Alphonse Foundation to fund conservation efforts. No formal conservation status.
ARIDE ISLAND	
JURISDICTION	Seychelles
AREA	0.683 km^2
GEOGRAPHY	Granitic island.
DESCRIPTION	The site also includes 105 ha of coastal marine habitat.

SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Human activities are limited to research and day visitor tourism that stay on designated paths.
BIODIVERSITY	Former coconut plantations are being progressively eliminated. Approximately one million breeding seabirds of ten species make this one of the most important seabird colonies in the Indian Ocean. Green and Hawksbill turtles nest on the beaches.
CONSERVATION STATUS	Aride Island Special Nature Reserve and IBA (Birdlife International 2018b).
BAJUNI ISLANDS	
JURISDICTION	Somalia
AREA	40 km^2
GEOGRAPHY	Low coral formations covered by scrub and a few trees. Forms a barrier reef protecting the mainland.
DESCRIPTION	Nine low-lying islands with only one significant village on Chula. There are six main islands, Chandra (2.95 km^2), Chovaye (5.46 km^2), Chula (1.99 km^2), Koyama (6.38 km^2), Darakasi (1.99 km^2) and Ngumi (2.56 km^2).
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Artisanal fishing within the reef. The growing population is causing widespread degradation and pollution.
BIODIVERSITY	Coral reef systems.
CONSERVATION STATUS	Falls within a larger EBSA submission.
BANC DU GEYSER	
JURISDICTION	French Territories in the Western Indian Ocean, contested by Madagascar and Comoros
AREA	40 km^2
GEOGRAPHY	Mostly submerged reef that is exposed at low tide.
DESCRIPTION	Some rock formations remain exposed at the southern end of the reef and there are some sandy cays covered with grass and small bushes.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	There is a possibility of oil fields being present.
BIODIVERSITY	There is an abundance of seabirds that cover the cays in tonnes of guano.
CONSERVATION STATUS	France included the reef as a marine protected area in 2012.
BASSAS DA INDIA	
JURISDICTION	French Territories in the Western Indian Ocean, contested by Madagascar
AREA	80 km^2
GEOGRAPHY	Atoll consisting of ten barren rocky islets.
DESCRIPTION	12 km in diameter with a shallow sandy lagoon surrounded by a reef slope that drops quickly to 3000 m.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	It is a site of numerous shipwrecks.
BIODIVERSITY	Aggregations of juvenile sharks (*Carcharhinus galapagensis*).
CONSERVATION STATUS	Declared a nature reserve in 1975. Part of the Iles Eparses potential WHS (Obura et al., 2012).
BAZARUTO ARCHIPELAGO	
JURISDICTION	Mozambique
AREA	1.583 km^2
GEOGRAPHY	Sandy Archipelago, apart from Santa Carolina, which is a rock island.
DESCRIPTION	Six islands (Bazaruto, Benguerra, Magarugue, Banque, Santa Carolina and Shell).
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	The area is a popular tourism destination with opportunities for diving, fishing and surfing. Seventy per cent of the resident population of about 4000 people are directly dependant on fishing as a primary livelihood.
BIODIVERSITY	Home to the largest population of dugongs along the eastern coastline of Africa, south of the Red Sea. The population comprises an estimated 250 individuals. The protected area is also home to coral reefs, whale sharks, manta rays and cetaceans. Five species of turtles breed on the beaches (IUCN, 2015).

CONSERVATION STATUS	The archipelago became a National Park in 1971. African Parks took over the conservation management of the National Park in December 2017.
BIRD ISLAND – SEYCHELLES	
JURISDICTION	Seychelles
AREA	0.94 km^2
GEOGRAPHY	Coralline island.
DESCRIPTION	Vegetated interior (including coconut plantations) with 5 km of sandy beaches.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Small resident population and high-end luxury tourism facilities.
BIODIVERSITY	Known for its large breeding colonies of Sooty terns and Common noddies. Hawksbill and Green turtles nest on the beaches.
CONSERVATION STATUS	Private conservation measures include the eradication of rats and rabbits. No formal conservation status.
BOOBY ISLAND	
JURISDICTION	Seychelles
AREA	0.023 km^2
GEOGRAPHY	Granitic island.
DESCRIPTION	Granitic island topped with tropical vegetation.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Uninhabited but visited for snorkelling and diving.
BIODIVERSITY	No significance.
CONSERVATION STATUS	None.
BOUDEUSE ISLAND	
JURISDICTION	Seychelles
AREA	0.03 km^2
GEOGRAPHY	Sandstone platform.
DESCRIPTION	4.6 m above sea level with a small sandy beach. Heavy swells make landing difficult
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Uninhabited.
BIODIVERSITY	No introduced vegetation, making it one of the most pristine cays in the area. Supports a population of 4000 Masked boobies and a small colony of Brown boobies. Green and Hawksbill turtles nest on the beaches.
CONSERVATION STATUS	IBA (Birdlife International 2018c).
SAINT BRANDON (also known as the Cargados Carajos Shoals)	
JURISDICTION	Mauritius
AREA	250 km^2 with a 190 km^2 surrounding reef.
GEOGRAPHY	Coral ridges and sand flats on an extended reef system.
DESCRIPTION	A group of 50 islands. Considered as part of the Mascarene Island group.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	There is a small transient population of Creole fishermen. Tourism operators offer fishing and diving trips.
BIODIVERSITY	Green and Hawksbill turtles breed on the beaches and seabird breeding colonies of various species are present. Introduced rats are considered one of the most significant causes of species loss on the islands. Mice, chickens, geckos and rabbits also infest the islands.
CONSERVATION STATUS	Recognized as an IBA with an estimate of 100 000–500 000 birds occurring there (Evans et al., 2016).

CHAGOS ARCHIPELAGO	
JURISDICTION	British Indian Ocean Territory (BIOT), contested by Mauritius
AREA	Land area is 56.13 km^2. Total area including lagoons and atolls is 15 000 km^2.
GEOGRAPHY	Coralline rock structures that top a submarine ridge.
DESCRIPTION	Seven atolls comprising 60 individual islands and nine reefs and banks.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Chagos is constituted as a BIOT and has the USA Diego Garcia military base on it.
BIODIVERSITY	The Great Chagos Bank is the largest acknowledged atoll structure in the world and it supports half the total of good quality reefs in the Indian Ocean. Ecosystems have thus far proven resilient to climate change. Seventy-six species that occur within the Chagos are listed in the IUCN Red List of Threatened Species.
CONSERVATION STATUS	In 2010, the UK government designated the area around the Chagos as the world's largest no-take reserve covering 544 000 km^2 (Marine Protection Atlas, 2017).
CHUMBE ISLAND	
JURISDICTION	Zanzibar, United Republic of Tanzania
AREA	The coral reef sanctuary is 0.55 km^2 and a forest reserve of 0.17 km^2.
GEOGRAPHY	Coral reef and coral rag island.
DESCRIPTION	Located off west coast of Unguja Island (Zanzibar Archipelago's main island). Comprises of a coral rag island approximately 1.1 kms long and 300m at its widest point. There is a shallow fringing reef running north-west to south-west.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	The island is managed by a not-for-profit company (CHICOP ltd) that reports to the Zanzibar Investment Promotion Agency and MANRLF.
BIODIVERSITY	Over 500 species of fish and 59 genera of reef-building corals have been identified. There are populations of coconut crab, Aders' duiker and the rare Roseate tern (*Sterna dougalli*) that breed on the island.
CONSERVATION STATUS	Designated as a no-take area.
COËTIVY ISLAND	
JURISDICTION	Seychelles
AREA	9.33 km^2
GEOGRAPHY	Coralline island.
DESCRIPTION	Low lying and heavily wooded.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Coëvity Island is used as an active prison and recently the government of the Seychelles and the government of the People's Republic of China have come to an agreement regarding a Chinese military base placed at the Northern end of Coëtivy Island (Global Powers, 2017).
BIODIVERSITY	No information.
CONSERVATION STATUS	None.
COUSIN ISLAND	
JURISDICTION	Seychelles
AREA	0.34 km^2 with a 1.05 km^2 coastal marine habitat.
GEOGRAPHY	Granitic island.
DESCRIPTION	A plateau covered with woodlands and with a granitic central hill 69 m above sea level.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Research station that allows day visitors that stay on designated trails during weekdays.
BIODIVERSITY	Small areas of mangroves occur with three freshwater swamps. Important for endemic land birds as well as extensive seabird breeding populations of seven species. Green and Hawksbill turtles nest on the beaches. Coral restoration project underway. An extensive restoration project is underway that is removing the relict coconut plantations.
CONSERVATION STATUS	Managed by Nature Seychelles. Cousin Island Special Reserve and identified as one of the most important IBAs in the Seychelles (Birdlife International, 2018d).

COUSINE ISLAND	
JURISDICTION	Seychelles
AREA	0.3 km^2
GEOGRAPHY	Granitic island.
DESCRIPTION	An ancient granitic ridge overlooking a small coastal plateau and fringed by a long sandy beach on the eastern side.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	A small exclusive resort exists on the island.
BIODIVERSITY	Considered one of the most important nesting sites for Hawksbill and Green turtles in the WIO. Important populations of endemic land birds and substantial populations of several seabird species occur on the island. A vegetation rehabilitation plan is underway to remove coconut trees.
CONSERVATION STATUS	Cousine Special Reserve and recognized as an IBA (Birdlife International, 2018e).
CURIEUSE ISLAND	
JURISDICTION	Seychelles
AREA	2.93 km^2
GEOGRAPHY	Granitic island.
DESCRIPTION	Bare red earth and coco-de-mer palms.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	An old leper colony that now hosts a satellite camp for conservation volunteers who undertake monitoring projects.
BIODIVERSITY	Endemic land bird, coco-de-mer palms and nesting Hawksbill turtles.
CONSERVATION STATUS	Curieuse Marine National Park.
D'ARROS ISLAND AND ST JOSEPH ATOLL	
JURISDICTION	Seychelles
AREA	D'Arros is 1.71 km^2 and St Josephs Atoll is 1.63 km^2.
GEOGRAPHY	Coralline island and atoll.
DESCRIPTION	A low lying vegetated coralline island that is separated from St Josephs Atoll by a deep-water channel.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Privately owned with a small resident population. Save Our Seas foundation runs a flourishing research centre on the island. Save Our Seas are keen to see the area fully designated as an MPA and various negotiations have been made with the Seychelles government to implement this.
BIODIVERSITY	The atoll is an important nursery area for several shark and ray species and an important feeding ground for turtles. Several seabird species roost and nest on the island and atoll. Extensive intact coral reef systems occur.
CONSERVATION STATUS	None. Managed by Save Our Seas Foundation.
DENIS ISLAND	
JURISDICTION	Seychelles
AREA	1.4 km^2
GEOGRAPHY	Coralline island.
DESCRIPTION	The second northernmost island in the Seychelles.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	There is a small resident population with a small resort on site that offers diving, snorkelling and fishing.
BIODIVERSITY	There is an abundance of coconut palms and other alien invasive tree species, but restoration projects are underway. There are several endemic land bird species and a fledging seabird colony. Green and Hawksbill turtles nest on the island.
CONSERVATION STATUS	The Green Islands Foundation, an NGO with a decade-long track record of conservation success in Seychelles, has a permanent outpost on the island and oversees a variety of programmes. No formal conservation status.

DESNOEUFS ISLAND	
JURISDICTION	Seychelles
AREA	0.457 km^2
GEOGRAPHY	Exposed sandstone island.
DESCRIPTION	Circular island with a high rim surrounding a central depression. The island has a fringing reef.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	The island is a base for the commercial exploitation of seabirds, especially the eggs of Sooty tern (*Sterna fuscata*). There is no permanent human population, but buildings have been constructed. Poaching of turtles and their eggs may be taking place.
BIODIVERSITY	Hawksbill and Green turtles nest on the island. Breeding seabird populations have declined.
CONSERVATION STATUS	IBA (Birdlife International, 2018f).
DESROCHES ISLAND	
JURISDICTION	Seychelles
AREA	4027 km^2
GEOGRAPHY	Coralline and fringed by reef of atoll character.
DESCRIPTION	5.5 km long with a 13 km beach circumference.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	There is a permanent population of around 100 people and a hotel is also found on the island.
BIODIVERSITY	Green and Hawksbill turtles nest on the beaches. Small numbers of Wedge-tailed shearwaters breed despite the presence of rats.
CONSERVATION STATUS	Managed by the Island Conservation Society. No formal conservation status.
ÉTOILE CAY	
JURISDICTION	Seychelles
AREA	0.05 km^2
GEOGRAPHY	Coral cay.
DESCRIPTION	Circular cay lying in the Amirantes outer islands of the Seychelles.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Uninhabited.
BIODIVERSITY	One of only three known nesting locations in the Seychelles for Roseate terns. Sooty terns and Common noddies also nest. Green and Hawksbill turtles nest on the beaches.
CONSERVATION STATUS	IBA.
EUROPA ISLAND	
JURISDICTION	French Territories in the Western Indian Ocean
AREA	28 km^2
GEOGRAPHY	Low sand cay.
DESCRIPTION	It is surrounded by coral beaches and a fringing reef and encloses a shallow mangrove lagoon of around 9 km^2 and open to the sea on one side.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	There is a small French military garrison on the island.
BIODIVERSITY	One of the worlds' largest nesting sites for Green turtles (8000 – 15 000 females). It supports a large and diverse population of breeding seabirds. It is the only known breeding site outside Aldabra and Madagascar for Malagasy pond herons. Seabirds include the second largest colony in the WIO of Great frigate birds and Audubon's shearwaters. It has the most diverse seabird fauna of the scattered islands in the WIO. Blacktip reef sharks (*Carcharinus melanopterus*), Lemon sharks (*Negarprion acutidens*) and hammerhead sharks occur in healthy numbers.
CONSERVATION STATUS	Listed as an IBA. Part of the Iles Eparses potential WHS (Obura et al., 2012).

FARQUHAR GROUP	
JURISDICTION	Seychelles
AREA	13.567 km² with 370 km² of atolls.
GEOGRAPHY	Atolls and submerged reefs
DESCRIPTION	Lying in the outer islands of the Seychelles. The island group comprises Farquhar Atoll, Providence Atoll, St Pierre Island, Wizard Reef, Umzinto bank (submerged), Bulldog Bank (submerged) and McLeod Bank (submerged).
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	There is only one settlement on Ile du Nord of Faquhar Atoll. The atolls are a favourite fly-fishing and game fishing destination and SCUBA diving and snorkelling also takes place.
BIODIVERSITY	Fossil coral cliffs on St Pierre Atoll. The site is an important nursery area for juvenile sharks. The island hosts significant numbers of breeding seabird species.
CONSERVATION STATUS	IBA (Birdlife International, 2018g).
FREGATE ISLAND	
JURISDICTION	Seychelles
AREA	2.07 km²
GEOGRAPHY	Granitic island.
DESCRIPTION	The easternmost of the granitic islands in the Seychelles, it comprises two hills with low-lying coastal plateaus.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	A privately owned island with a luxury resort.
BIODIVERSITY	Hawksbill turtles nest on the island. The island historically held large seabird colonies, but these have mostly become extinct, with now only small numbers of terns still breeding.
CONSERVATION STATUS	IBA for its endemic land birds (Birdlife International, 2018h).
GLORIEUSES OR GLORIOSO ISLANDS	
JURISDICTION	French Territories in the Western Indian Ocean
AREA	165 km²
GEOGRAPHY	Sandy cay and coral bank.
DESCRIPTION	17 km long with two main islands, Grand Clorieuse (7 km²) and Lys Island (600 m long).
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Grande Glorieuse has an airstrip. It is thickly vegetated with the remnants of a coconut plantation and casuarina trees.
BIODIVERSITY	The island has the second largest population of breeding Sooty terns in the Indian Ocean with 760 000 pairs. Turtles nest on the sandy beaches.
CONSERVATION STATUS	Declared a nature reserve in 1975.
INHACA ARCHIPELAGO	
JURISDICTION	Mozambique
AREA	52km²
GEOGRAPHY	Sandy archipelago.
DESCRIPTION	The Inhaca Archipelago comprises of the Inhaca and Portuguese Islands and separates the Maputo Bay from the Indian Ocean. It lies approximately 32 km east from the city of Maputo.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	There is a population of about 6000 people living on the Archipelago that subsist on fishing and agriculture and the area is also a popular destination for South African tourists.
BIODIVERSITY	Several species of corals are recorded in the surrounding waters and the islands provide nesting sites for four endangered sea turtle species. Mangroves and seagrass are also found on and around the islands.
CONSERVATION STATUS	The Archipelago is part of a conservation area since 1965, and has recently been incorporated into the Ponta do Ouro Partial Marine Reserve.

JUAN DE NOVA ISLAND	
JURISDICTION	French Territories in the Western Indian Ocean
AREA	The island is 4.4 km^2 and the coral reef platform is 250 km^2.
GEOGRAPHY	Beachrock and sand dunes surrounded by a coral reef platform.
DESCRIPTION	The coral structures extend 12 km north and 2 km south of the island. The tilting structure of the island results in differing reef morphologies that vary between 3 m and 20 m before dropping to 2000 m.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	There is a French garrison of troops on the island with an airstrip.
BIODIVERSITY	Over two million breeding pairs of Sooty terns nest on the island make it the most important nesting site for this species in the Indian Ocean. It is an important nesting site for Hawksbill turtles and the surrounding waters are an important nursery area for Grey reef sharks.
CONSERVATION STATUS	Designated as a nature reserve and also an IBA. It forms part of the Iles Eparses potential UNESCO World Heritage Site (Obura et al., 2012).
MARIANNE ISLAND	
JURISDICTION	Seychelles
AREA	0.96 km^2
GEOGRAPHY	Granitic island.
DESCRIPTION	A long beach occurs on the western side of the island.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Uninhabited but visited by tourists and boaters for its diving locations.
BIODIVERSITY	No information.
CONSERVATION STATUS	None.
MARIE LOUISE ISLAND	
JURISDICTION	Seychelles
AREA	0.556 km^2
GEOGRAPHY	Coralline island.
DESCRIPTION	The island is located at the southern end of the Amirantes. A low-lying coral sandy cay, oval in shape.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	In 2012, the Seychelles government established a new prison on the island.
BIODIVERSITY	Coconut palms and casuarina trees dominate the vegetation. It is used as a support base for the harvesting of Sooty tern eggs. It has breeding colonies of various tern species and Hawksbill turtles nest on the beaches.
CONSERVATION STATUS	None.
MISALI ISLAND	
JURISDICTION	Zanzibar, United Republic of Tanzania
AREA	0.9 km^2
GEOGRAPHY	Coralline Island.
DESCRIPTION	The coralline island is covered with thick vegetated scrub.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	The waters around the island support key fishing grounds.
BIODIVERSITY	There is a high diversity of over 350 reef fish species and 40 coral species around the island. Dugong have also been reported on occasion and Hawksbill turtles have bred on the island.
CONSERVATION STATUS	Listed under the Misali Forest Order as a protected forest and multiple use zone now integrated in the Pemba Channel Conservation Area (PECCA).

MNEMBA	
JURISDICTION	Zanzibar, United Republic of Tanzania
AREA	28 km^2
GEOGRAPHY	Coralline island
DESCRIPTION	Mnemba Island is a small single island located 2 km off the north-east corner of Unguja Island (Zanzibar Archipelago's main island). It is surrounded by an oval coral reef system.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Mnemba is a private island that has a luxury lodge on it that is managed by &Beyond. There is a 200 m exclusion zone around the island which may only be used by guests visiting the lodge.
BIODIVERSITY	Green turtles nest on the beaches and the island is a refuge for the introduced Aders' duiker.
CONSERVATION STATUS	It has been designated as the Mnemba Island Marine Conservation Area (MIMCA).
NORTH ISLAND – SEYCHELLES	
JURISDICTION	Seychelles
AREA	2.01 km^2
GEOGRAPHY	Granitic island.
DESCRIPTION	One of the Seychelles inner islands situated 27 km north of Mahe.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Privately owned with a private resort.
BIODIVERSITY	Endemic land birds. Vegetation restoration projects are being undertaken. Hawksbill and Green turtles nest on the beaches. There are small numbers of nesting seabirds.
CONSERVATION STATUS	None.
ILE PLATTE	
JURISDICTION	Seychelles
AREA	0.578 km^2
GEOGRAPHY	Sandy cay.
DESCRIPTION	The island is a low and wooded sandy cay about 1300 m long and 250 m wide. Barrier reefs extend 5 km north and about 2 km south of the island. There is also a submerged coral reef rim that extends 12 km west and 18 km south of the island.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	There is a small settlement on the western shore with a few guest cottages on the north-west corner.
BIODIVERSITY	A small population of Hawksbill turtles nest on the island and White-tailed tropicbirds and tern species breed on the island. The island is known for its rich fish life.
CONSERVATION STATUS	Managed by the Island Development Company. No formal conservation status.
POIVRE ATOLL	
JURISDICTION	Seychelles
AREA	20.24 km^2
GEOGRAPHY	Atoll.
DESCRIPTION	Poivre Atoll is on the eastern edge of the Amirantes Bank and comprises of the atoll and four islands. Poivre North and Poivre South are joined by a 750 m causeway crossing the reef flats. There is a large, elongated lagoon between the four islands that dries out during low tide.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	There is a 30-room hotel on the Florentine Island.
BIODIVERSITY	The Island is known for its rich fish life. Hawksbill turtles nest on the beaches and there are a few species of breeding seabird on the island.
CONSERVATION STATUS	No formal conservation status.

PRIMEIRAS AND SEGUNDAS ARCHIPELAGOS	
JURISDICTION	Mozambique
AREA	The area extends for more than 10 000 km² and over 205 km of coastline.
GEOGRAPHY	Archipelago.
DESCRIPTION	The Primeiras and Segundas Archipelago is a chain of ten sparsely inhabited barrier islands and two coral reef complexes off the coast of central Mozambique, near the coastal town of Angoche. The islands lie in two groups along the western side of the Mozambique Channel. The five Segundas islands are in the north and separated by a stretch of open water and reefs from the five islands of the Primeiras chain to the south.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Due to the lack of reliable sources of freshwater, habitation on the islands is sparse and is mainly in support of fishing.
BIODIVERSITY	The eastern sides of the islands are fringed with coral reefs, comprised mainly of soft corals with hard corals at the southern edge. Beds of seagrass are situated between the islands and mainland which are important habitat for sea turtles and dugongs. The southern islands support Mozambique's largest nesting grounds for Green and Hawksbill sea turtles. The archipelago also hosts an important dugong population.
CONSERVATION STATUS	No formal conservation status though it has been identified as an area of high priority.
QUIRIMBAS ISLANDS	
JURISDICTION	Mozambique
AREA	31 islands stretching across approximately 350 km.
GEOGRAPHY	Archipelago.
DESCRIPTION	Low lying islands covered with patches of dense woodland through to grasslands. The main islands of the Quirimbas include Ibo, Quirimba, Matemo, Vamizi, Quilalia, das Rolas and Medjumbi. While the southern Quirimbas are under formal protection, there is a need to prioritise protection for the northern islands of Metundo Vamizi, Rongi and Tecomaji.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Many of the islands are now inhabited with historical settlements also occurring on a number of the islands. The area is popular as a diving destination.
BIODIVERSITY	The Quirimbas are important in terms of its input of larvae into the south equatorial current. There has been limited evidence of bleaching mortality, which indicates some resilience to bleaching events. Mangroves occur along the more sheltered shorelines. The coral reef systems within the northern Quirimbas are among the world's most biologically diverse, having the highest recorded diversity of corals outside the Coral Triangle. Three species of turtle breed on the islands but are threatened by poaching for meat and eggs. The Quirimbas Islands would provide suitable habitat for dugongs if proper conservation management effort was put in place. (Hill et al., 2010).
CONSERVATION STATUS	Linked to the terrestrial Quirimbas National Park in northern Mozambique (1185 km² are marine and island habitats). Submitted for World Heritage Status in 2008.
REMIRE ISLAND	
JURISDICTION	Seychelles
AREA	0.3 km²
GEOGRAPHY	Coralline island surrounded by coral reef.
DESCRIPTION	It is located 2.5 km south of the southern extremity of Remire Reef.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	There is a small settlement on the western shores of the island.
BIODIVERSITY	The Island is known for its rich fish life. Hawksbill turtles nest on the beaches and there are a few species of breeding seabird on the island but numbers are declining.
CONSERVATION STATUS	Managed by the Island Development Company. No formal conservation status.

SAINTE ANNE ARCHIPELAGO	
JURISDICTION	Seychelles
AREA	3.87 km^2 with a marine national park of 14.43 km^2.
GEOGRAPHY	Granitic islands.
DESCRIPTION	Comprise eight islands.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Ste. Anne, Round and Long Islands have luxury resorts on them, while Cerf Island has a small resident population of around 100 persons. The islands are a popular snorkelling and diving spot.
BIODIVERSITY	Cachee Island is a nesting site for seabirds. The area contains one of the largest seagrass meadows in the granitic islands of the Seychelles and is therefore an important feeding ground for turtles.
CONSERVATION STATUS	Sainte Anne Marine National Park.
SILHOUETTE ISLAND	
JURISDICTION	Seychelles.
AREA	20.1 km^2.
GEOGRAPHY	It is the third largest granitic island in the Seychelles.
DESCRIPTION	The Island is mountainous with five peaks over 500 m above sea level.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	The population of around 200 persons live in three villages on the west coast of the island. There is a luxury hotel on the island.
BIODIVERSITY	Numerous endemic land-based species occur on the island. Coastal reef flats surround the island.
CONSERVATION STATUS	Silhouette National Park and IBA. The Island Conservation Society manages it. Not significant for seabird or turtle nesting.
SONGO SONGO ARCHIPELAGO	
JURISDICTION	Tanzania
AREA	40 km^2
GEOGRAPHY	Coralline islands.
DESCRIPTION	The Songo Songo Archipelago is made up of a collection of five tiny islands surrounded by reefs that are located to the south of Mafia Island. The islands stretch for approximately 40 km.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	Fanjove Private Island contains a small eco-friendly lodge that can accommodate 12 persons, and Songo Songo Island support a population of some 3500 people, while the other smaller islands house temporary fishing camps
BIODIVERSITY	A band of fringing reef with high coral diversity protects the islands from the open sea. The islands are important feeding and nesting grounds for Hawksbill turtles. They are also important for coconut crabs and nesting seabirds.
CONSERVATION STATUS	No formal conservation status, but included in the Mafia-Rufiji-Kilwa Ramsar Site.
TROMELIN ISLAND	
JURISDICTION	French Territories in the Western Indian Ocean, contested by Mauritius
AREA	1.77 km^2
GEOGRAPHY	Atoll.
DESCRIPTION	A low flat island not reaching more than 7 m above sea level and fringed by coral reef.
SOCIO-ECONOMIC/ GEOPOLITICAL STATUS	There is a short airstrip on the island.
BIODIVERSITY	Masked and Red-footed boobies nest on the island. It is a key site for nesting Green turtles (Derville et al., 2015).
CONSERVATION STATUS	It has been identified as an IBA due to its significance as a seabird-breeding site.

A Noddy Tern perches on top of a granite outcrop on Cousin Island, Seychelles. © Peter Chadwick

KEY SPECIES

Seabirds

The WIO islands are globally important roosting and breeding grounds for several seabird species (see Table 2). For example, Juan de Nova hosts over two million breeding pairs of Sooty terns and the Glorieuses has the second largest colony of Sooty terns with 760 000 pairs. Europa Island has the most diverse seabird fauna of the scattered islands of the WIO and also has the second largest breeding colonies of Frigate birds and Audubon's shearwaters. Aldabra with its high endemism, hosts the world's second largest colony of nesting Frigate birds and Aride Island has over one million breeding seabirds of ten different species. Masked and Red-footed boobies both have limited breeding populations with colonies occurring on Tromelin and Bordeuse islands, while St Joseph's Atoll is an important roosting site for Red-footed boobies and hosts over 1000 Greater *(Fregata minor)* and Lesser frigate birds. Sooty tern eggs are still being legally harvested in large numbers on Desnoeufs Island with Marie Louise Island acting as a secondary base for this harvesting and it has been noted that the populations of these terns are declining on the islands.

Unchecked development on small islands can have major negative consequences for seabird populations and rising sea levels will also result in the flooding of low-lying areas necessary for breeding terns, tropicbird and shearwater species. Invasive rats have historically been one of the leading causes of seabird extinctions and they continue to threaten island bird species by predating on eggs, chicks and adult birds. According to Graham et al. (2018), seabird droppings that are rich in nutrients, leach into and benefit surrounding reef systems and the fish on the reefs adjacent to islands with seabirds were larger for their age than fish on the reefs next to rat-infested islands. Increasing levels of tourism are adding to their vulnerability through disturbance, trampling and habitat destruction and over-utilization of resources. In addition to seabird colonies, islands within the WIO are important stop over, resting and feeding points for numerous migrating birds.

Turtles

Five species of turtle occur within the WIO with two of these, the Hawksbill *(Eretmochelys imbricata)* and Green *(Chelonia mydas)*, breeding regularly on many of the islands (see Table 2). They have traditionally been exploited for their meat, shells and eggs and are particularly vulnerable when they return to land to lay their eggs. As a result of past exploitation they are now listed as threatened. Most WIO countries have agreed upon and have implemented conservation measures to protect turtles with extensive monitoring and research being undertaken on an ongoing basis. The islands are important as feeding, nursery and nesting grounds. Europa Island is considered as one of the most important nesting sites for Green turtles in the world with between 8000 and 15 000 females nesting on the island each year. Cousin Island Special Reserve is considered as the most important nesting site for Hawksbill turtles in the WIO and has one of the longest running monitoring programs for this species in the world (Allen et al., 2010). The shallow reef systems surrounding islands and the lagoons within atoll systems, where seagrass beds occur, are critically important feeding grounds for the turtles.

Sharks

With 230 shark and ray species having been identified in the WIO, it is one of only a handful of global shark and ray hotspots. At the same time, with over 70 million people living within 100 km of the coastline, there is tremendous pressure on marine resources and shark populations have faced massive declines. Sharks are slow growing, late to mature and produce few young and therefore battle to recover from over-exploitation. Additionally, destruction of habitats, such as mangroves (Save our Sharks, 2017) and coral reefs that are important nursery areas for sharks, are also impacting negatively on their populations.

Several of the islands in the WIO are important nursery areas for shark species (see Table 2), with Bassas da India holding aggregations of juvenile Galapagos sharks *(Carcharhinus galapagensis)* (Hammerschalg, 2005). D'Arros and St Joseph's Atoll are important nursery areas for several shark and ray species, particularly juvenile Lemon sharks *(Negaprion brevirostris)* (Filmalter et al., 2013) and Europa Island is important for Blacktip reef sharks *(Carcharinus melanopterus)*, Lemon sharks *(Negarprion acutidens)* and Greater hammerhead sharks *(Sphyrna mokarran)*. Further research is required to determine the importance of other island systems for shark species.

Dugongs

Dugong populations have declined rapidly across the WIO region largely due to deliberate and accidental capture in fishing nets. Dugongs are now listed as threatened, though several hundred may still occur along the eastern African coast, with the Bazaruto Archipelago being one of the most important of these (see Table 2). According to Findlay et al. (2011), Bazaruto holds a population of between 247–359 dugongs.

Table 2: Importance of key animal groups in small islands of the Western Indian Ocean.

ISLAND	SEABIRDS	TURTLES	SHARKS	OTHER
African Banks	X	X	X	
Agaléga Islands		X		
Aldabra Island Group	X	X	X	
Alphonse Island Group	X	X		
Aride Island	X	X		
Bajuni Islands				
Banc du Geyser	X			
Bassas da India			X	
Bazaruto Archipelago		X	X	X (Dugong)
Bird Island – Seychelles	X	X		
Booby Island				
Boudeuse Island	X	X		
Cargados Carajos	X	X		
Chagos Archipelago				
Coetivy Island				

ISLAND	SEABIRDS	TURTLES	SHARKS	OTHER
Cousin Island	X	X		
Cousine Island	X	X		
Curieuse Island		X		
D'Arros Island and St Joseph Atoll	X	X	X	
Denis Island		X		
Desnoeufs Island	X	X		
Desroches Island		X		
Étoile Cay	X	X		
Europa Island	X	X	X	
Farquhar Group	X		X	
Fregate Island	X	X		
Glorieuses	X	X		
Inhaca Archipelago		X		
Juan de Nova Island	X	X		
Marianne Island				
Marie Louise Island	X			
Mnemba		X		
North Island - Seychelles		X		
Ile Platte	X	X		
Poivre Atoll	X	X		
Primeiras and Segundas Archipelagos	X	X		X (Dugong)
Quirimbas Islands		X		
Remire Island		X		
Sainte Anne Marine National Park	X			
Silhouette Island				
Songo Songo Archipelago		X		
Tromelin Island	X	X		

Booby Island in the Seychelles – a granitic island that is covered in tropical vegetation. © Peter Chadwick

SOCIO-ECONOMIC AND GEOPOLITICAL IMPORTANCE

Throughout history, the islands of the WIO have played both an important socio-economic role and a maritime strategic role. These islands span the ocean and their strategic importance is highlighted by their location along key sea lines of communication (Baruah, 2018). In addition, given the enormous energy and natural resources of the region, islands will become increasingly important for the economy of emerging countries to grow.

These islands are vital to facilitating the ability of naval forces to continue to protect the key shipping lanes, with military bases located on a number of these islands. Diego Garcia is the biggest island of the Chagos Archipelago and the US military use this as a base for all of their Indian Ocean operations, while Juan de Nova and Europa Islands have garrisons of French troops stationed there. Agaléga Island is leased to the Indian Military for development of strategic assets and in 2015, the Seychelles and India signed an agreement for constructing and operating a joint military facility on Assumption Island. However, the Seychelles opposition party recently nullified this agreement (Eurasian Times, 2018). India and China have been making recent efforts to garner power and influence across the region and this military base would have given a strong strategic advantage. The Chinese have built a military base on the northern end of Coetivy Islands in the Seychelles (Global Powers, 2017).

Given the strategic importance of these islands, it is understandable that countries are contesting their claim of ownership. Banc du Geyser is contested between France, Madagascar and the Comoros, Bassas da India's jurisdiction is contested by France and Madagascar, the Chagos are contested between the UK and Mauritius, and Tromelin island is contested between France and Mauritius.

With the decline of agricultural activities in the 1970s and 1980s, tourism activities have become the focus for many of the small islands, particularly those in the Seychelles, and tourism now makes a substantial contribution to the gross domestic product (GDP) and foreign exchange of island states. Facilities vary from small fishing guesthouses to luxury high-end lodges. These venues then become the base point for numerous water-based activities that include snorkelling, diving, boating, fly-fishing and deep-sea game fishing (UNDP, 2013). These activities all rely on maintaining the healthy functioning of marine and island ecosystems and development needs to be carefully planned and impacts must be mitigated. Coral reefs and mangroves are deteriorating from the impacts of local use and important fish stocks are declining due to overfishing and mismanagement (Obura et al., 2017). With island states recognizing the importance of healthy island ecosystems, most of the islands now have small but highly efficient research and monitoring stations based on them. These stations are undertaking long-term projects with a particular focus on turtles, seabirds, coral reef systems and climate change.

Oil and gas development is becoming a potentially significant economic activity with numerous companies now conducting exploratory drilling. This activity will need careful monitoring in terms of its alignment with conservation priorities.

THREAT LEVEL

As countries within the WIO intensify their efforts for a sustainable oceans economy, this places an increasing burden on the diverse ecosystems and biodiversity of the region's islands and atolls. Mounting resource utilization, habitat degradation, tourism and development, alien invasive species, pollution and climate change all impact negatively on these already fragile systems. For the blue economy to reach its full potential, the region's governments will need to ensure that risk mitigation is maximized and that careful management and conservation of the islands and atolls is ensured to safeguard sustainability and the ongoing delivery of ecosystem services (Chevallier, 2017).

Climate change

Given the limited land area, low level above the sea and high exposure to unpredictable marine weather, it can be expected that islands and atolls will have a high vulnerability to climate change. Although sea level rise is considered as one of the most widely recognized threats of climate change to small islands and atolls, tropical cyclones, increasing air and sea surface temperatures and changing rainfall patterns are additional negative impacts that can be expected (Nurse et al., 2014). Low lying areas can expect storm surges and swell waves to increase, as sea levels rise and this will increase the rates of erosion and also impact on fresh groundwater resources as overwash of seawater occurs.

Increases in sea temperature are already resulting in coral bleaching and reef degradation that in turn will reduce their benefits of providing coastal protection from storms and will negatively impact on island community livelihoods as tourism and subsistence fishery opportunities are reduced.

To clearly understand the impacts of climate change, the difference between observed and projected impacts of these changes into the future will need to be carefully recorded through increased baseline monitoring that in turn can heighten confidence in prediction models. Increased assistance from the international community will also be required to assist with adaptation and mitigation measures.

Pollution

In spite of international policies and conventions, countries are still dumping millions of tonnes of waste per year into the ocean (Galgani et al., 2010). Despite small islands and atolls being isolated, they are not immune from this debris. Morishige et al. (2007) and Bouwman et al. (2016) both refer to the potential of isolated islands acting as traps or sinks for marine rubbish due to the nature of currents and gyres around these islands. The spatial distribution of this waste is also not homogenous and factors such as the size, shape, density and distance from source all play a role in determining debris deposition. In the case of the WIO islands, most waste has been found to emanate from the mainland of south-east Asia and Africa, as a result of inadequate waste management practices (Duhec et al., 2015).

Plastics in particular have become problematic and given their low density, they float on the sea surface and can be transported over large distances by wind and currents. It is also well known that these plastic debris can accumulate chemical pollutants with a resultant increased concentration of harmful chemicals and heavy metals being found along island shorelines. Seabirds that roost and breed on small islands and atolls have also recently been found to be a source of accumulating microplastics and debris (Provencher et al., 2018). The seabirds take in the plastic particles when they are mistaken for food items and are then later excreted or vomited out back at their colonies. This can then create concentrated areas of pollution and chemical pollutants such as DDT and PCB's that can have a negative impact on the wider ecosystem.

Invasive alien species

Islands are prone to invasion by alien species because of the lack of natural competitors and predators that control populations in their native ecosystems. In addition, islands often have ecological niches that have not been filled because of the distance from colonizing populations, also increasing the probability of successful invasions. These invasions pose a severe risk to small island developing states by threatening the ecosystems, livelihoods and local economies.

Mammals have invaded all island groups within the WIO and cats and rats in particular have had an extremely negative impact on seabird populations (Russel et al., 2016). These mammals have been introduced since at least the second half of the last millennium and new introductions still continue unintentionally with, for example, shrews arriving on the island of Rodrigues in 1998. The impacts of these introductions are clear with huge losses of land birds, seabirds and reptiles from predators. Habitat degradation is also occurring from herbivores such as deer, goats and pigs. (Russell and Le Corre, 2009). The granitic Seychelles and the Mascarene Islands hold globally important species of plants with high levels of endemism that are currently threatened by past habitat destruction and the current impact of alien invasive species such as the coconut and the creeper *Canavalia cathartica*. Mammal eradications have been attempted on 45 islands in the WIO region and where they have been successful, they have resulted in a spectacular recovery of species and ecosystems. Overall, the task of removing these invasive species will require government commitment with the provision of financial and human resources.

Tourism and development

According to Kumar (2002), marine tourism within the island states of the WIO has the risk of causing irreversible degradation from excess development, over-utilization of water and energy resources, trampling of sensitive systems, overfishing and mangrove clearing. In addition, increased trade and tourism are significant vectors for the introduction of invasive alien species as listed above. These invasive species arrive via the ships' ballast, cargo and unprocessed commodities. Given that this tourism is often the largest source of foreign exchange for many of the small island states it is therefore imperative to ensure that the pressures of this industry are carefully monitored, mitigated and managed.

Illegal extractive use and poaching

The poaching of seabird eggs, turtle eggs, turtles and numerous fish species and sharks is occurring across WIO islands but the full impacts of this are not quantifiable or clearly understood. Destruction of mangroves, is similarly having detrimental impacts on healthy island functioning, increasing erosion from storm surges, reducing nursery areas for sharks and fish species and reducing available nesting habitat for many seabirds.

PROTECTION LEVEL

Currently only 17 of the listed small islands have some form of legally protected area status (see Table 3). Of these, only 14 of the sites have marine areas also included under legal protection. While some of the other islands and atolls have indeed been recognized for their biodiversity and ecological value, IBA and ecologically or biologically significant marine areas (EBSAs) carry no formal protection status. In 2014, it was announced that D'Aross Island was to be made a protected nature reserve under the Seychelles Nature Park and Conservancy Act but to date this has not been formally designated (Nature Seychelles Blue Economy, 2016).

From these results, it is explicitly clear that much work still needs to be done to ensure better legal protection status for these critical biodiversity areas and particular focus must be given to ensuring protection status for the marine areas surrounding these islands.

Table 3: Small islands of the Western Indian Ocean protection status. Note: XX indicates marine area also protected.

ISLAND	PROTECTED AREA STATUS	IMPORTANT BIRD AREA	EBSA	OTHER	NO PROTECTION STATUS
African Banks	XX	X			
Agaléga Islands					X
Aldabra Island Group	XX	X		Ramsar wetland & UNESCO World Heritage Status	
Alphonse Island Group			X		X
Aride Island	X	X			
Bajuni Islands			X		
Banc du Geyser	XX				
Bassas da India	XX			UNESCO WHS application	
Bazaruto Archipelago	XX				
Bird Island - Seychelles					X
Booby Island					X
Boudeuse Island		X			
Cargados Carajos		X			
Chagos Archipelago	XX				
Coetivy Island					X
Cousin Island	XX	X			
Cousine Island	X	X			
Curieuse Island	XX				
D'Arros Island and St Joseph Atoll					X
Denis Island					X
Desnoeufs Island		X			
Desroches Island					X
Étoile Cay		X			

ISLAND	PROTECTED AREA STATUS	IMPORTANT BIRD AREA	EBSA	OTHER	NO PROTECTION STATUS
Europa Island		X		Ramsar Site, and UNESCO WHS application	
Farquhar Group		X			
Fregate Island		X			
Glorieuses	XX				
Inhaca Archipelago				Inclusion into the Ponto do Ouro Partial Marine Reserve	X
Juan de Nova Island	X	X		UNESCO WHS application	X
Marianne Island					
Marie Louise Island					X
Mnemba				Marine Conservation Area	X
North Island - Seychelles					X
Ile Platte					X
Poivre Atoll					
Primeiras and Segundas Archipelagos					X
Quirimbas Islands	X X (partial area)			UNESCO WHS application	X
Remire Island					X
Sainte Anne Marine National Park	XX				
Silhouette Island				Ramsar	X
Songo Songo Archipelago		X	X		X
Tromelin Island					

An aerial view of St Joseph Atoll and lagoon that lies adjacent to D'Arros Island in the outer Amirantees of the Seychelles.
© Peter Chadwick

PRIORITY AREAS FOR CONSERVATION

For island states to fully maximize the socio-economic opportunities of the blue economy, it will be imperative to ensure adequate protection of the biodiversity and ecosystems of the region's islands and their surrounding waters. Focused effort will be required to expand the area under legal conservation status and this will need both local and global support and funding.

Adequate understanding of the biodiversity and ecology of many of the islands is still deficient and this lack of information makes it difficult to ensure appropriate decision-making to protect these systems. Those areas already under formal proclamation will need to be effectively managed with regular review of management effectiveness taking place. The threats of alien invasive species, over-exploitation of resources, pollution and irresponsible development will also need to be addressed.

Opportunities for increasing protection status lie in concentrating on the islands where the terrestrial component is already proclaimed and on those islands that have already been identified as important bird areas, falling within EBSAs or those areas being identified for UNESCO World Heritage Status. Islands that already have surrounding waters proclaimed need to be assessed further to determine if protection of the marine components is adequate. The delineation of marine priorities for conservation may further be refined with information on seabird, turtle and marine mammal foraging ranges (Ronconi et al., 2012) and current information on coral reef systems.

Formalized conservation planning processes will need to be initiated to determine the full extent of new protected area boundaries but immediate opportunities should be focused on the following:

- The Northern Quirimbas Archipelago is one of the most diverse, productive and intact ecosystems in the WIO due to the unique blend of environmental, social and historical drivers. The coral reef systems within the northern Quirimbas are among the world's most biologically diverse, having the highest recorded diversity of corals outside the Coral Triangle. The Archipelago comprises Rongui, Vamizi and Metendo Islands.

- The Primeiras and Segundas Archipelagos host Mozambique's largest nesting grounds for Green and Hawksbill sea turtles and the most important dugong populations in the WIO. Extensive beds of seagrass are situated between the islands and mainland which are important habitats for sea turtles and dugongs.

- Juan de Nova Island already has protection status for the terrestrial area, has been identified as an IBA and is included in a UNESCO WHS application. It is important to both seabirds and turtles.

- Europa Island has been identified as an IBA and Ramsar Site and is included in a UNESCO WHS application. It is important to seabirds, turtles and sharks.

- Aride and Cousine Islands already have the terrestrial areas under protection status and now require marine area expansion. They are important to both seabirds and turtles.

- Tromelin Island is listed as an IBA and falls within an EBSA. It is important to both seabirds and turtles.

- Boudeuse Island, Cargados Carajos, Desnoeufs Island, Étoile Cay, the Farquhar Group and Fregate Island are all listed as IBAs and have the potential for formal proclamation.

REFERENCES

Allen, Z.C., Shah, N.J., Grant, A., Derand, G-D. and Bell, D. (2010). Hawksbill turtle monitoring in Cousin Island Special Reserve, Seychelles: an eight-fold increase in annual nesting numbers. *Endang. Species Res.* 11, pp. 195

Baruah, D. (2018). I*slands as game changers in the Indian Ocean.* Asia Maritime Transparency Initiative. https://amti.csis.org/ISLANDS-GAME-CHANGERS-INDIAN-OCEAN/ Accessed at 20 April 2015

BirdLife International (2018a). *Important Bird Areas factsheet: African Banks.* BirdLife International, Cambridge, UK

BirdLife International (2018b). Important Bird Areas factsheet: Aride Island Special Reserve. BirdLife International, Cambridge, UK

BirdLife International (2018c). *Important Bird Areas factsheet: Boudeuse Island*. BirdLife International, Cambridge, UK

BirdLife International (2018d). *Important Bird Areas factsheet: Cousin Island Special Reserve*. BirdLife International, Cambridge, UK

BirdLife International (2018e). *Important Bird Areas factsheet: Cousine island*. BirdLife International, Cambridge, UK

BirdLife International (2018f). *Important Bird Areas factsheet: Desnoeufs island.* BirdLife International, Cambridge, UK

BirdLife International (2018g). *Important Bird Areas factsheet: Farquhar – South Island and islets*. BirdLife International, Cambridge, UK

BirdLife International (2018h). *Important Bird Areas factsheet: Frégate.* BirdLife International, Cambridge, UK

Bouwman, H., Evans, S.W., Cole, N., Yive, N. and Kylin, H. (2016). The flip-or-flop boutique: Marine debris on the shores of St Brandon's rock, an isolated tropical atoll in the Indian Ocean. *Mar. Environ. Res.* 114, pp. 58–64

Chevallier, R. (2017). Integrated marine and coastal management in the Western Indian Ocean: Towards a sustainable ocean economy. *South African Institute of International Affairs.* SAIIA OCCASIONAL PAPER 258

Derville, S., Jean, C., Dalleau, M., Le Gall, J., Ciccione, S. and Bourjea, J. (2015). Long term monitoring of Green Turtles nesting on Tromelin Island demonstrates stable reproduction and population parameters. *Chelonian Conserv. Biol.* 14(1), pp. 11–20

Duhec, A.V., Jeanne, R., Maximenko, N. and Hafner, J. (2015). Composition and potential origin of marine debris stranded in the Western Indian Ocean on remote Alphonse Island, Seychelles. *Mar. Pollut. Bull.* 96(1–2), pp. 76–86

Eurasian Times (2018). Is China against the expansion of Indian military base in Seychelles? *The Eurasian Times.* https://eurasiantimes.com/indian-military-base-seychelles/ Accessed at 7 August 2018

Evans, S.W., Cole, N.C., Kylin, H., Yive, R., Tatayah, V., Merven, J. and Bouwman, H. (2016). The protection of the marine birds and sea turtles of St Brandon's Rock, Indian Ocean, requires the conservation of the entire atoll as an ecosystem. *Afr. J. Mar. Sci.* 38(3), pp. 317–327

Filmalter, J.D., Daghorn, L. and Cowley, P.D. (2013). Spatial behaviour and site fidelity of the sicklefin lemon shark *Negaprion acutidens* in a remote Indian Ocean atoll. *Mar. Biol.* 160, pp. 2425-2436

Findlay, K.P., Cockcroft, V.G., and Guissamulo, A.T. (2011). Dugong abundance and distribution in the Bazaruto Archipelago, Mozambique. *Afr. J. Mar. Sci.* 33, pp. 441–452. https://doi.org/10.2989/1814232X.2011.637347

Friedlander, A.M., Ballesteros, E., Beets, J., Brown, E.K., Fay, J.M., Haup, P. Henning, B., Rose, P. and Sala, E. (2015). Biodiversity and ecosystem health of the Aldabra Group, southern Seychelles: Scientific report to the government of Seychelles. *National Geographic Pristine Seas*. https://doi.org/10.13140/RG.2.2.14095.69282

Galgani, F., Fleet, D., Van Franeker, J., Katsanevakis, S., Maes, T., Mouat, J., Oosterbaan, L., Poitou, I., Hanke, G., Thompson, R., Amato, E., Birkun, A. and Janssen, C. (2010). Marine Strategy Framework Directive, Task Group 10 Report: Marine Litter. In: JRC Scientific and Technical Reports (ed. Zampoukas, N.). European Commission Joint Research Centre, Ispra

Global Powers (2017). *Chinese military base built on the northern end of Coetivy Island*. Global Powers. https://www.reddit.com/r/GlobalPowers/comments/4o01z0/diplomacy_chinese_military_base_built_on_northern/

Graham, N.A.J., Wilson, S.K., Carr, P., Hoey, A.S., Jennings, S. and MacNeil, M.A. (2018). Seabirds enhance coral reef productivity and functioning in the absence of invasive rats. *Nature* 559, pp. 250–253. https://doi.org/10.1038/s41586-018-0202-3

Hammerschalg, N. (2005). Galapagos sharks (*Carcharhinus galapagensis*) at the Bassas da India atoll: first record from the Mozambique Channel and possible significance as a nursery area: research letter. *South Afr. J. Sci.* 101, pp. 7–8

Hill, M. and Curry, D. (2007). Wildlife of the Seychelles. Collins. Traveller's Guide.

Hill, N., Davidson, J., Marques da Silva, I., Mucave, S., Muaves, L., Guissamulo, A., Debney, A. and Garnier, J. (2010). Coral and Reef Fish in the Northern Quirimbas Archipelago, Mozambique – A First Assessment. *West. Indian Ocean J. Mar. Sci*. 8(1). https://doi.org/10.10.4314/wiojms.v8i1.56680

IUCN (2015). Bazaruto Archipelago: an area of globally outstanding conservation. https://www.iucn.org/content/bazaruto-archipelago-area-globally-outstanding-conservation

Kumar, S.S. (2002). Importance of Marine Tourism and Environmental Protection: in Some Selected Indian Ocean Islands. *Journal of Applied Sciences*. https://doi.org/10.3923/jas.2002.739.741

Marine Protection Atlas (2017). British Indian Ocean Territory Marine Protected Area (Chagos). *Marine Protection Atlas*. https://mpatlas.org/zones/7707499/

Morishige, C., Donohue, M.J., Flint, E., Swenson, C. and Woolaway, C. (2007). Factors affecting marine debris deposition at French Frigate Shoals, Northwestern Hawaiian Islands Marine National Monument, 1990-2006. *Mar. Pollut. Bull.* 54(8), pp. 1162–1169

Myers, N., Mittermeier, R.A., Mittemeier, C.G., da Fonseca, G.A.B. and Kent, J. (2000). Biodiversity hotspots for conservation priorities. *Nature* 403, pp. 853–858

Nature Seychelles Blue Economy (2016). D'Arros Island; Seychellois demand answers. *Nature Seychelles Blue Economy News*. http://blueeconomyseychelles.org/item/60-d-arros-island-seychellois-demand-answers Accessed at 16 October 2016

Nurse, L.A, McLean, R.F., Agard, J., Briguglio, L.P., Duvat-Magnan, V., Pelesikoti, N., Tompkins E. and A. Webb (2014). Small islands. In *Climate Change 2014: Impacts, Adaptation, and Vulnerability, Part B: Regional Aspects*. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Barros, V.R., C.B. Field, D.J. Dokken, M.D. Mastrandrea, K.J. Mach, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, and L.L. White (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 1613–1654.

Obura, D., Church, J. and Gabrié, C. (2012). *Assessing Marine World Heritage from an Ecosystem Perspective: The Western Indian Ocean.* World Heritage Centre, United Nations Education, Science and Cultural Organization (UNESCO), pp. 124

Obura, D. Burgener, V. Gamblin, P. Gonzales, A. and Owen, S. (2017). *Reviving the Western Indian Ocean Economy: Actions for a Sustainable Future.* WWF International, Gland, Switzerland, pp. 64

Provencher, J.F., Vermaire, J.C., Avery-Gomm, S., Braune, B.M. and Mallory, M.L. (2018). Garbage in guano? Microplastic debris found in faecal precursors of seabirds known to ingest plastics. *Sci. Total Environ.* 644, pp. 1477–1484. https://doi.org/10.1016/j.scitotenv.2018.07.101

Ronconi, R.A., Lascelles, B., Langham, G.M., Reid, J.B. and Oro, D. (2012). The role of seabirds in Marine Protected Area identification, delineation, and monitoring: Introduction and synthesis. *Biol. Conserv.* 156, pp. 1–4

Russel, J.C, Cole, N.C., Zuel, N. and Rocamora, G. (2016). Introduced mammals on Western Indian Ocean Islands. *Global Ecology and Conservation* 6(2016) pp. 132–144

Russell, J.C. and Le Corre, M. (2009). Introduced mammal impacts on seabirds in the Iles Eparses, Western Indian Ocean. *Mar. Ornithol*. 37(2), pp. 121–128

Save our Sharks (2017). *The value of mangroves to sharks.* Save our sharks. http://saveoursharks.nl/en/the-value-of-mangroves/

Spatz, D.R., Zilliacus, K.M., Holmes, N.D., Butchart, S.H.M., Genovesi, P., Ceballos, G., Tershy, B.R. and Croll, D.A. (2017). Globally threatened vertebrates on islands with invasive species. *Sci. Adv.* 3(10), e1603080

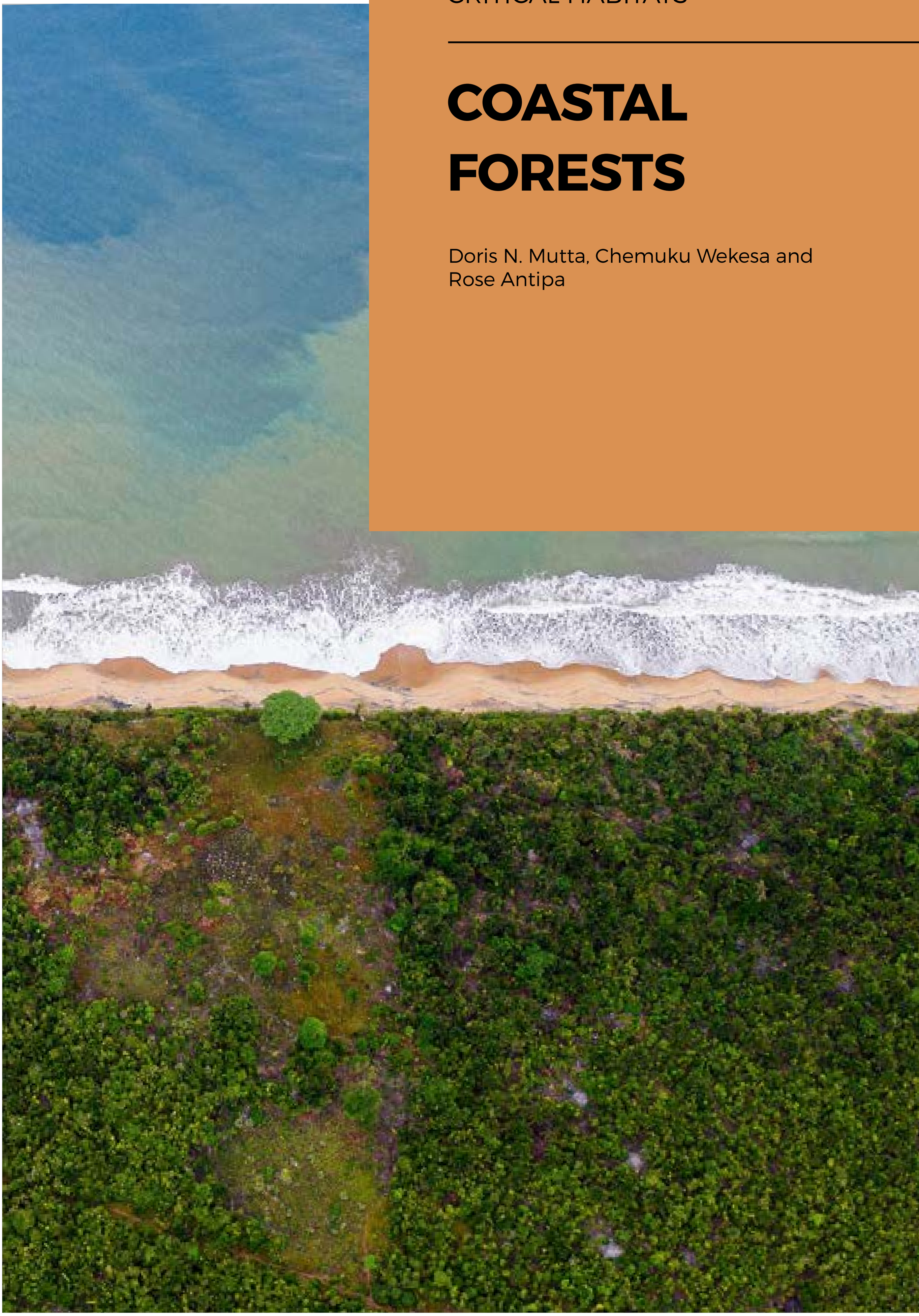
UNDP (2013). *Expansion and strengthening of the protected area subsystem of the outer islands of Seychelles and its integration into the broader land and seascape*. Project document, Government of the Seychelles and UNDP

Webster, I., Cadinouche, A. and Huggins, A. (2016). Evidence of on-going turtle poaching on Agalega, Mauritius. *Marine Turtle Newsletter* 151, pp. 21–23

CRITICAL HABITATS

COASTAL FORESTS

Doris N. Mutta, Chemuku Wekesa and Rose Antipa



Coastal forest of Tampolo, District of Fenerive-Est, Region Analanjirofo. © Patrick Laby

BACKGROUND

Typology of coastal forests

The term 'coastal forest' refers to a mosaic of forest types adapted to coastal habitats, and transitional zones. These include the typical semi-evergreen and evergreen dry forests, scrub forest and *Brachystegia* forest and transitional vegetation formations, such as riverine, swamp and Afromontane forests (Azeria et al., 2005). Coastal forests are found at elevations of 0–500 m above sea level (m.a.s.l.) up to a maximum of 1100 m.a.s.l. depending on ecological conditions (Burgess and Clarke, 2000). In Eastern Africa, the limits to the coastal forest area are also set by the amount of rainfall, and seasonality. Coastal forests include inland coastal forests and woodlands that are typically terrestrial and inter-phase with the coastal habitat.

Forest types adapted to the coastal habitat

Forest types adapted to the coastal habitat include beach forests, dune vegetation and sand forests which typically occur above the high-tide mark on sandy soil and may merge into agricultural land or higher elevation or upland forests (Burgess et al., 1998; Burgess and Clarke, 2000). They are mostly scrub-like with a high presence of stunted tree growth due to edaphic or climatic extremes including strong winds, salt spray and lack or excess of humidity. Beach forests are found above the high-tide mark on sandy soil and may merge into upland forests and are important in soil stabilization. They hold together the sandy soil, prevent coastal erosion, and act as barriers against storm waves from the sea. Besides, they maintain moisture in the sandy soil, which is crucial to the survival of numerous living organisms along the coastal areas (Burgess et al., 1998).

The coastal dune vegetation occupies the small but highly dynamic zone at the intersection of ocean and land and provides important ecosystem functions: they act as buffers against storms, wave impact, and erosion of the hinterland, as well as provide a unique habitat for flora and fauna (Sigren et al., 2014). Sand forests are relics of coastal dune forests which separated from the ocean when the shoreline and water levels slowly shifted. Sand forests have distinct boundaries and exhibit a narrow zone of nearly bare soil directly bordering it (Mathews, 2007). The vegetation that grows in the sand forests is very specialized hence they have unique biodiversity and high levels of animal and plant endemism (Van Wyk and Smith, 2001; Botes et al., 2006). These forest types play a critical role in land stabilization hence prevent the silting up of coastal lagoons and rivers and protecting human settlements further inland from where they occur (Burgess and Clarke, 2000). Given the sensitivity of these ecosystems to modifications, they are important ecological indicators of the groundwater level (IUCN, 1986; Burgess and Clarke, 2000). Most of these forests have been destroyed and the areas developed for tourism to the extent that the original state of the forests has been obliterated, eg Diani Beach in Kenya. Local communities have also planted crops and exotic species such as coconut, ornamental palms and *Casuarina equisetifolia* in place of the indigenous species altering the forest ecosystems. These forest types have not been comprehensively studied.

Inland coastal forests and woodlands

Inland coastal forests and woodlands are those adapted to the terrestrial habitat but interphased with coastal habitats. These include: freshwater swamp forests, riparian forests, savannah woodlands, scrub forests, dry forests and low land rain forests. Freshwater swamp forests contain a diverse assemblage of vegetation that occur next to the mangroves and are inundated by freshwater from rivers, creeks and lagoons hence could be permanent or periodically flooded. The floodplain forests are periodically flooded and occur in areas where the water table is high or where drainage is poor, eg Tana floodplain forest in Kenya (Younge et al., 2002). Some species in floodplain forests include *Cyperus papyrus* in Madagascar that was introduced from Central Africa.

Riparian forests, also called riverine or gallery forests, are found adjacent to or near rivers, where the water table is high (White, 1983). In the tropics, riparian forests are extremely dense and productive, and have large numbers of climbers including lianas. Besides their aesthetic and recreational values, riparian forests are important in preserving water quality, controlling erosion, siltation, and are wildlife refugia especially for amphibians and reptiles, otters and hippopotomi, birds and monkeys and other tree-dwelling mammals (IUCN, 1986; Burgess and Clarke, 2000). Examples include forests that occur along the Shebelle and Jubba Rivers at Bu'ale in Somalia; Tana and Galana River in Kenya; and Rufiji and Rovuma (or Ruvuma) rivers in Tanzania.

The Savannah woodlands are transition forests dominated by either *Brachystegia spiciformis* or *Brachystegia microphylla*, and usually occur on degraded/poor soils. The tree crowns do not interlock (Burgess et al., 1998) and lianas are usually scarce. Grasses are scarce or absent in these woodlands and hence fire does not normally penetrate this vegetation type. Examples of woodlands include Chiniziua Forest in Mozambique; parts of Arabuko-Sokoke Forest in Kenya and parts of Tong'omba forest in Tanzania.

Scrub forests are intermediate in structure between forests (canopy height > 10 m) and bushlands or thickets (canopy height < 10 m). Along Eastern Africa coasts, scrub forests have a lower canopy (> 4 m) than the lower 7 m limit as indicated by White (1983) but retain other forest features such as overlapping tree crowns, abundant lianas, a leaf-litter layer and emergent trees which often exceed 10 m in height (Younge et al., 2002). Herbs are scarce to absent in scrub forests.

Other inland coastal forests include dry and lowland rain forests. Dry forests are semi-evergreen or evergreen undifferentiated forests occurring in areas where atmospheric humidity is high throughout the dry season and have a canopy of up to 7 m, lower than the minimum limit of 10 m as adopted in White (1983). Examples include '*Cynometra webberi-Manilkara sulcata*' community of the Arabuko-Sokoke forest in Kenya and Inhansato and Inhamitanga forests, in Cheringoma in Mozambique. On the other hand, lowland rain forests are found in lowland areas where the amount of rainfall is high (Younge et al., 2002). In areas with well-drained areas such as on ridgetops, the lowland rain forests are replaced by dry forests. Lowland rain forests in the region include forests on the summit of the Shimba Hills in Kenya, and Tongwe Hill and East Usambara in Tanzania.

COASTAL FOREST FORMATIONS IN THE WESTERN INDIAN OCEAN REGION

Coastal forests in the Western Indian Ocean (WIO) region occur as regional forest mosaics traversing the Eastern Africa coastal states from Somalia to South Africa, and the island state of Madagascar and Small Island States of Comoros, Mauritius and Seychelles (Fig. 1).

Various regional forest mosaics have been described on the coastal plain of mainland Eastern Africa. The forest mosaics lie on a coastal plain sloping gently upwards away from the sea, interrupted in places by low hills and plateaus with widths at the northern limit (southern Somalia and northern Kenya), and southern limit (southern Mozambique) extending inland over 200 km, and on the southern Kenyan, northern Tanzanian coasts, and along the northern Mozambique coast, at less than 30 km (Kent, 1972). In most cases, Eastern Africa coastal forests comprise of vegetation recognized as distinct from most vegetation types further inland and those at increasing altitude (Burgess and Clarke, 2000; Younge et al., 2002; White, 1983; Joordens et al., 2019), and are also distinct from the forests of the Eastern Arc Mountains in terms of climate, elevation and dominant plant species.

The forests are highly fragmented extending along the coastal plain of the region between 1° N to 25° S and 34° to 41° E. Comprising a set of tiny forests found on the coastal belt up to 500 m.a.s.l., although in Tanzania they occur up to 1030 m.a.s.l. on the Handeni Hill, they are often embedded within larger habitat mosaic of farmland, savannah-woodland, lowland forests and thickets covering 3172 km^2. There are over 400 forest patches of closed canopy varying in size and degree of isolation; most of them less than 500 ha in size (Burgess and Clarke, 2000). The minimum area under forest and closely related habitats adds up to over 6200 km^2 or about 2 per cent of the total area of the Eastern African coastal Ecoregion. Owing to the high species endemism in these forests, the White (1983) classification of forests has defined them as the Zanzibar-Inhambane regional mosaic and floristic region. It is also classified as a regional center of endemism and has been recognized as one of the earth's biologically richest places and designated the Eastern Africa Biodiversity Hotspot (Mittermeier et al., 1999, Myers et al., 2000). Within this mosaic is the Northern Zanzibar-Inhambane coastal forest mosaic extending from southern Somalia through coastal Kenya to southern Tanzania, including the islands of Zanzibar and Pemba; and the Southern Zanzibar-Inhambane coastal forest mosaic extending from southern Tanzania along the Mozambique coast to the mouth of river Limpopo. Northern and Southern Zanzibar-Inhambane mosaics make up the Eastern Africa coastal forests.

Zanzibar-Inhambane regional mosaic

This mosaic comprises a forest expanse along the coastal plain extending from Somalia, through Kenya and Tanzania up to the southern part of Mozambique. The landscape is covered with sand dunes and plateaus in the southern part and mountains in the northern part, whilst tropical dry forests are found within farmlands, savannah grasslands, savannah woodlands, and wetlands habitats (Burgess and Clarke, 2000; Timberlake et al., 2011).

These forests are home to about 4050 plant species, the African violet (*Saintpaulia teitensis*) being one of the most popular plants. The mosaic is recognized as one

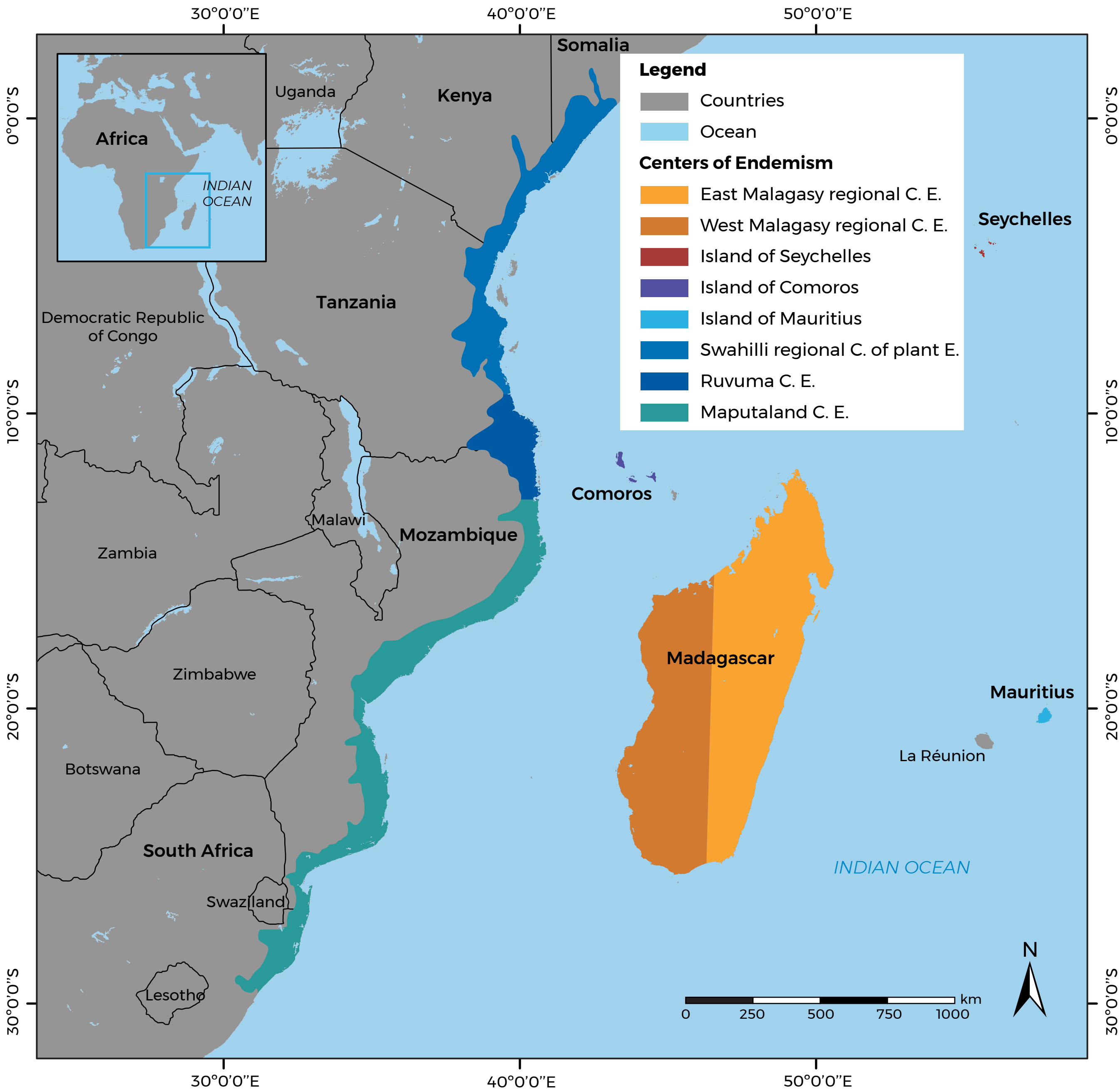


Figure 1: Coastal forest mosaics in the Western Indian Ocean region.

of Africa's most important area of species endemism and diversity (Younge et al., 2002) – approximately 200 mammals, 250 reptiles, and 85 amphibians live in the habitat. There are about 220 freshwater fish species with 30 being endemic including fish adapted to temporary swamps and floodplains of the coast, such as lungfishes (*Protopterus amphibious* and *P. annectens*) that can come up for air and hibernate for over a year in dried mud. Five primate species are endemic in the forests. These are Rondo galago (*Galagoides rondoensis*, endangered), Tana River red colobus (*Procolobus rufomitratus*, critically endangered), Zanzibar red colobus (*Procolobus kirkii*, endangered), Kenya coastal galago (*Galagoides cocos*) and the Tana River Mangabey (*Cercocebus galeritus*). More than 633 bird species can also be found, of which 11 are endemic to the forests.

Northern Zanzibar-Inhambane coastal forest mosaic

Northern Zanzibar-Inhambane coastal forest mosaic stretches from southern Somalia through Kenya to southern Tanzania. The 69 000 km^2 expanse includes Zanzibar and Pemba Islands. The region is generally dry, and the mosaic made up of tropical and subtropical moist broadleaf forests (Clarke, 2000; WWF-US, 2003). The forest mosaic has 154 forest patches covering approximately 3107 km^2 that were not included in Burgess and Clarke

(2000). Approximate number of species is 3000 with trees carrying the bulk of this population (Saleem and Hussein, 2016). High levels of species endemism characterize this region placing it among the highest in the world. *Berlinia orientalis*, *Dialium holtzi*, and *Millettia stuhlmannii* are some of the dominant tree species. Pemba sunbird (*Nectarina pemba*), Clarke's weaver (*Ploceus golandi*), and the Sokoke scops owl (*Otus ireneae*) are some of the endemic bird species found in the region. Mammals are represented by about 158 species including Black rhino (*Diceros bicornis*), the Zanzibar red colobus (*Piliocolobus kirkii*), Pemba Island flying fox (*Pteropus comorensis*), Savannah elephant (*Loxodonta africana*), and the Zanj elephant shrew (*Rhynchocyon petersi*). Three of the five endemic primate species described in the Zanzibar-Inhambane regional mosaic are also found in this forest mosaic (IUCN, 1986).

Swahili regional centre of plant endemism is found within this mosaic and hosts 4000 vascular plants of which about 1200 are endemic representing 25 genera. A further 287 species and eight genera are near endemic extending into the neighbouring Swahili-Maputaland regional transition zone, a few of which intrude into northern Natal.

Southern Zanzibar-Inhambane coastal forest mosaic

Southern Zanzibar-Inhambane coastal forest mosaic stretches from southern Tanzania at the Lukuledi River, along the Mozambique coast to the mouth of the Limpopo River. It extends approximately 2000 km along the coast and lies within 50 km of the Indian Ocean. Characteristics of this mosaic are similar to its northern counterpart. However, compared to the north, less rainfall is received. Savannah forests, and swamp forests are part of this mosaic. The mosaic hosts the Rovuma Centre of Endemism (RCoE) located in north-eastern Mozambique.

The RCoE is composed of coastal forests and woodlands of Cabo Delgado Province in north-east Mozambique (Burrows and Timberlake, 2011; Darbyshire et al., 2019) and is shared with south-east Tanzania. It is named after the Rovuma River which forms much of the border between the two countries. Of over 3000 collections made between 2003 and 2009, 738 plant taxa from 105 families (the largest of which is Rubiaceae with 83 taxa) were documented. A total of 68 new country records for Mozambique were noted, including scarce RCoE endemics such as *Drypetes sclerophylla* Mildbr. In addition, 36 taxa were identified as either entirely new to science or known previously from insufficient material to have been formally described (Timberlake et al., 2010, 2011). The family Rubiaceae was also represented by the largest number of new species (13) and a total of 55 endemic species. A recent report by Burrows et al. (2018) has provided additional woody flora records for this region, including additional rare RCoE endemics such as *Grewia filipes* Burret. The mosaic also hosts the Swahili-Maputaland regional transition zone, which contains 3300 species of which about 100 are endemic. A further 40 species intrude into Maputaland-Pondoland regional mosaic that extends to the south of KwaZulu-Natal in northern South Africa.

Maputaland-Pondoland Regional mosaic

The Maputaland-Pondoland Regional Mosaic covers mainly the Maputo province in southern Mozambique and extends southward across the Limpopo River into northern South Africa (as far as Port Elisabeth), and from the coastline inland to the Libombo Mountains (covering also Swaziland). The forest types are woody grasslands, sand forest, bushveld and subtropical thicket. The Maputaland center of endemism is found within this mosaic, stretching from the southern tip of Mozambique to South Africa and Swaziland, and is shared with northern KwaZulu-Natal. It covers approximately 17 000 km^2 and houses 230 endemic species (Van Wyk and Smith, 2001). The woody grassland and sand forest have the largest number of endemic species (Mucina et al., 2006).

Within this mosaic is the designated Maputaland Pondoland Albany Biodiversity Hotspot which lies on the east coast of southern Africa, housing about 600 tree species and representing the highest tree richness of any known temperate forest. It comprises 80 per cent of South Africa's remaining forests (Silander, 2001). The biodiversity hotspot also covers the KwaZulu Cape coastal forest mosaic that is also renowned for high levels of endemism (Burgess et al., 2004).

Coastal forests of the island states of the region

Island ecosystems are vastly different from those of the mainland due to their isolation and increased speciation. Consequently, forest resources of island states are of global importance in terms of their role in conservation of biological diversity, in particular of endemic species and genetic variability (White, 1983). While the forest cover of island states is relatively insignificant regionally in terms of area, the high ratio of coastline to land area and the short distances between the uplands and the coastal areas mean that coastal forests perform major ecosystem functions that sustain island livelihoods and the economies of these island states. The WIO island states are well endowed with forests, and the extent of forest cover varies between 18.6 and 73.2 per cent of land cover (FAO, 2016). The forests comprise lowland forests, montane rain forests, dry deciduous and moist evergreen forests. The forest cover is variously distributed in the island states as follows: Madagascar 124 830 km^2 at 21.3 per cent; Comoros 346.7 km^2 at 18.6 per cent; Mauritius 382.9 km^2 at 18.8 per cent; and Seychelles 337.0 km^2 at 73.2 per cent (FAO, 2016). These are however only a fraction of the original forest cover before human exploitation of the islands, and the associated deforestation due mainly to agricultural expansion. Of particular importance is that these remaining forest patches contain species that are unique to the island concerned.

Madagascar

The Republic of Madagascar is the largest island in Africa and fourth largest in the world. The coastal forests comprise dry deciduous forest, tropical dry and spiny forests. The massive island is a biodiversity hotspot with high species endemism (Myers et al., 2000). About 89 per cent of the plants, 95 per cent of reptiles, and 92 per cent of mammals are endemic to the island, with some also endangered (IUCN, 1986). Several populations of endangered endemic species include birds, reptiles and mammals. The Madagascar fish eagle and Madagascar plover are such endemic birds found only on the west and south coast. More than 70 endemic chameleon species occur on the island (Raselimanana and Rakotomalala, 2003; Glaw and Vences, 2007) and at least two thirds of the world's chameleon species occur in Madagascar (Sayer et al., 1992). Lemur species include indri, black lemur, and hairy-eared dwarf lemur. The island hosts two centres of endemism: East Malagasy Regional Centre of Endemism and West Malagasy Regional Centre of Endemism.

East Malagasy Regional Centre of Endemism occupies eastern Madagascar and descends from the central highlands with mountains up to above 2000 m.a.s.l., which run almost the entire length of the island, to sea level in the east and to approximately 800 m on the west side of the mountain ridge. The Sambirano Domain forms a small exclave on the north-west coast. To the east, the central highlands end abruptly in steep escarpments overlooking the narrow coastal plain. Extensive marshes and lagoons are found in the coastal plains. The forests comprise moist and drought resistant montane forest and lowland rain forest. Bamboo thickets and grasslands occur as secondary vegetation caused by deforestation. There are about 6100 plant species in this forest type, with 4800 of them (about 80 per cent) being endemic. Of the 1000 genera, 160 (16 per cent) are endemic (White, 1983). Though there still are extensive areas of forest, deforestation continues at pace due to agricultural expansion for rice production.

West Malagasy Regional Centre of Endemism occupies the western side of the island up to about the 800 m contour. Towards the east where the land rises to meet the central highlands, there are outcrops of crystalline Precambrian rocks, but the greater part of the region is underlain by sediments of Triassic, Jurassic, Cretaceous and Tertiary age. The flat plains of the west coast are wider than those along the east coast and cover 322 000 km^2. The vegetation varies from dry deciduous forests to deciduous thicket and grasslands. The grasslands are extensive, covering about 80 per cent of the area, and are secondary in origin, having been caused by deforestation and being maintained by regular fires. There are about plant 2400 species of which 1900 (79.2 per cent) are endemic and about 700 genera of which 140 (20 per cent) are endemic (White, 1983).

Comoros

There are four main Comoros islands, forming an archipelago between Madagascar and East Africa. The largest and youngest island is Grande Comoro (Ngazidja). Others are Anjouani (Ndzuani), Mohéli (Mwali), and Mayotte that is the oldest in terms of formation. Mt. Katharla is an active volcano that provides the highest point at 2361 m.a.s.l. The vegetation types include lowland and montane rain forests and to a lesser extent mangrove forest (Sayer et al., 1992). High endemism is evident (Sayer et al., 1992). Among the avian community, of which there are fewer than 100 species, many are critically endangered. The Anjouan sunbird (*Nectarinia comorensis*) and the Anjouan Brush-warbler (*Nesillas longicaudata*) are strictly endemic to Anjouan, while the Mayotte drongo (*Dicrurus waldenii*, endangered) and the Anjouan scops owl (*Otus capnodes*, critically endangered) are almost extinct. *Pteropus livingstonii* and *Rousettus obliviosus* are two endemic fruit bats. Nine species of geckos, chameleons, and shining-skink are strictly endemic to Comoros.

Mauritius

The Republic of Mauritius consists of two main islands, Mauritius and Rodrigues. St. Brandon, Agalega and other smaller islands are also part of the Republic of Mauritius. The islands' forest cover acts as wind buffers during frequent strong cyclones. The flora and fauna are unique due to the islands' location, age, isolation and varied topography and are characterized by high species endemism. There are approximately 685 indigenous plant species of which 267 (39 per cent) are endemic to the island of Mauritius, with 150 being endemic to the Mascarene Archipelago (which includes the French territory of La Réunion). Two endemic bird species are the Mauritius fody and the Olive white eye. There are rare seabirds that breed in the Mascarene Archipelago that are also believed to be endemic. Round Island is specifically regarded as a special nature reserve due to the presence of high number of rare reptiles.

Mauritius also has one of the most threatened island floras in the world. About 89 per cent of the endemic flora is threatened. Sixty-one of the indigenous species are already classified as extinct while 141 of the Mascarene endemic flowering plant species are classified as critically endangered, 55 species are endangered and 98 are classified as vulnerable (IUCN, 1986). Other taxa under threat are the giant tortoise, other reptiles and butterfly species. Two endemic giant tortoise species have become extinct while reptiles have reduced from 17 to 12 native species and five of the 39 native butterfly species are threatened (Republic of Mauritius, 2010).

Seychelles

The Seychelles islands are made up of over 115 granitic and coral islands, 40 of which are granitic islands while the rest are of coral limestone origin. Four of the granitic islands (Mahe, Prasin, Silhouette, and La Digue) are inhabited (Government of Seychelles, 2011). The coastal forest is a narrow strip measuring between 10 to 30 m wide comprised of moist evergreen tropical rain forest (Senterre and Wagner, 2014). The forest provides habitat to unique flora and fauna of biological importance. They include thousands of plants, birds, amphibians, mammals, reptiles and fish, some of which are endemic. For plants, there is one endemic family, 12 endemic genera, and 72 endemic species from a flora of about 233 native plants (White, 1983; Procter, 1984; Robertson and Luke, 1993). The endemic palm species include 'Coco de mer' (*Lodoicea maldivica*) which is endangered; Millionaire's salad (*Deckenia nobilis*) which is vulnerable; Seychelles stilt palm (*Verschaffeltia splendida*) and Latanier palm (*Roscheria melanochaetes*) which are near threatened; and Thief palm (*Phoenicophorium borsigianum*) and Latanier millepattes palm (*Nephrosperma vanhoutteanum*) that are the more dominant. There are also 12 endemic bird species, five endemic frog species, five endemic bat species, and two endemic freshwater fish species, among others (Government of Seychelles, 2011).

IMPORTANCE OF COASTAL FORESTS

Biological significance of coastal forests

The WIO coastal forest landscape is an area of remarkable biological diversity for both flora and fauna. Comprising small and fragmented patches, the forest mosaics are host to high biological diversity of global significance. The most distinctive biological attribute of forests in the region is the exceptionally high levels of species endemism found within the closed canopy forest patches (see Tables 1 and 2). Numerous studies have revealed a large number of endemic species, and a high concentration of rare and threatened taxa (Hawthorne, 1993; Burgess and Clarke, 2000; Myers et al., 2000; Brooks et al., 2001, 2002; Burgess et al., 2004).

Biological studies on the Eastern Africa coastal forests (Lovett and Wasser, 1993; Burgess et al., 1998; Burgess and Clarke, 2000; Myers et al., 2000) have documented more than 4500 plant species and 1050 plant genera, with around 3000 species and 750 genera occurring in closed forests (Burgess et al., 2003). Out of these more than 1750 plant species are strictly endemic with 554 of them occurring in the lowland forests. The concentration of species endemism is at the rate of 9.6 endemics per 100 km^2 of the habitat. Among the best-known plants in the hotspot are the species of African violets (*Saintpaulia* spp.) as well as 11 species of wild coffee, eight of which are endemic and none having been commercially exploited.

In terms of faunal species richness there are at least 158 species of mammals (17 per cent of all Afro-tropical species), 94 reptiles and 1200 molluscs. Faunal endemism rates are highest in the invertebrate groups such as millipedes (80 per cent of all the forest species), molluscs (68 per cent) and forest butterflies (19 per cent). Amongst the vertebrates, 7 per cent of mammals, 10 per cent of birds, 57 per cent of reptiles and 36 per cent of amphibians found in the area are endemic. Of a total of at least 80 endemic vertebrate species, 52 occur in the lowland forest habitat. Endemic animals include two ancient African mammal groups, the elephant shrews (one endemic species) and bush babies (two endemic species). There are also three endemic monkey species, all confined to tiny patches of remaining forest habitat (WWF, 2006).

The coastal forests of the Swahili regional centre of plant endemism, that extends along much of the mainland Africa coast, are also rich in bird life with more than 633 species found, 11 of which are endemic. Among them are the Clarke's weaver (*Ploceus golandi*), Sokoke scops owl (*Otus ireneae*), Pemba sunbird (*Nectarina pemba*), Fischer's tauraco (*Tauraco fishceri*) and the Tana River cisticola (*Cisticola restrictus*).

Most endemic species are concentrated in the forests of the Tana River, between Malindi in Kenya to Tanga in northern Tanzania, and in southern Tanzania. Forests with highest numbers of endemics are: lower Tana River, Arabuko-Sokoke and Shimba Hills in Kenya; lowland East

Usambara, Pugu Hills, Matumbi Hills, Rondo and Litipo and other plateaux near Lindi in Tanzania; the Tanzanian offshore island of Pemba; Bazaruto archipelago and tiny forest remnants in Mozambique.

Most coastal forest endemics have a narrow distributional range, often exhibiting single-site endemism or with scattered or disjunct distribution patterns. Such narrowly restricted endemics are often among the species most sensitive to environmental change and disturbance, and so at highest risk of extinction (Borokini, 2014; Abdelaal et al., 2018). These species are therefore important in identifying and conserving biodiversity priorities, such as Important Plant Areas (Darbyshire et al., 2017), and Key Biodiversity Areas (IUCN, 2016). Due to the intense threats to the survival of forest habitats in these areas, the centres of endemism have been declared priority areas for biodiversity conservation in Africa, and globally. Furthermore, a recent botanical expedition in the early 2000s to the northern Mozambican coast identified plant species that had never been formally described (Barratt, 2017). These biological values thus offer a powerful motivation for the protection and conservation of the coastal forests and for bioprospecting.

Table 1: Species endemism in coastal forests in WIO region.

COASTAL FOREST AREA	COUNTRIES COVERED	EXTENT (km^2)	NUMBER OF SPECIES	NUMBER OF ENDEMIC SPECIES
Swahili regional centre of plant endemism	Kenya, Somalia, Malawi, Mozambique, Tanzania, Zimbabwe	3167	4000 plant species 633 bird species 3 monkey species	1200 plant species 25 plant genera 11 bird species 3 highly threatened monkey species
Rovuma centre of endemism	Mozambique, Tanzania	1182	738 plant taxa (species and sub species) from 105 families Rubiaceae is the largest family 36 plant taxa newly identified	55 endemic species including *Drypetes sclerophylla, Pavetta Lindina, Grewia filipes*
Maputaland center of endemism	Mozambique, South Africa, Swaziland	17 000	8100 plant species 631 bird species 200 mammal species 200 reptile species 72 amphibian species	1900 plant species 14 bird species 4 mammal species 30 reptile species 11 amphibian species
East Malagasy regional centre of endemism	Madagascar	272 000	6100 plant species – 1000 genera, 25 species of reptiles 144 species of amphibian 25 mammal species 106 bird species	800 plant species – 160 genera 22 mammal species 83 bird species 232 reptile species 141 amphibian species
West Malagasy regional centre of endemism	Madagascar	322 000	2400 plant species 700 genera 106 bird species	1900 species (79.2%) 140 (20%) plant genera 12 bird species
Sand Forest	Mozambique, South Africa	5000	33 plant species 3 bird species	20 plant species 3 bird species
Beach Dunes	Somalia – South Africa	16 254	1000 plant species	6 plant species 2 reptile species 2 bird species 2 mammal species
Island of Comoros	Comoros	370	935 plant species 21 species of birds 9 species of reptiles	136 plant species
Island of Mauritius	Mauritius	390	685 plant species 16 bird species 13 reptile species	267 plant species 10 reptile species 13 bird species
Island of Seychelles	Seychelles	410	233 plant species	72 plant species – 12 plant genera – 1 plant family 12 endemic bird species 5 endemic frog species 5 endemic bat species 2 endemic freshwater species


(Left) Degraded sacred forest Kaya Lunguma in Kwale county, Kenya, due to illegal harvesting of indigenous trees for fuelwood and charcoal. (Right) Kilibasi Hills in Kwale County, Kenya. © Chemuku Wekesa

Table 2: Dominant woody species in the coastal forests of the WIO region.

COASTAL FOREST AREA	COUNTRIES COVERED	EXTENT (km^2)	DOMINANT TREE SPECIES
Swahili regional centre of plant endemism	Kenya, Somalia, Tanzania	3167	*Afzelia quanzensis, Scorodophloeus fischeri, Dialium holtzii, Hymenaea verrucosa, Millettia struhlmanni, Berlinia orientalis, Cynometra spp., and Xylia africana.*
Rovuma centre of endemism	Mozambique, Tanzania	1182	*Cryptosepalum exfoliatum, Brachystegia floribunda, Brachystegia spiciformis, Julbernardia globiflora, Burkea africana, Terminalia sericea, Milicia excelsa, Baikiaea plurijuga, Colophospermum mopane*
Maputaland center of endemism	Mozambique, South Africa, Swaziland	17 000	*Cleistanthus schlechteri, Newtonia hildebrandtii, Hymenocardia ulmoides, Psydrax fragrantissima, Croton pseudopulchellus, Drypetes arguta, Vitex ferruginea subsp. amboniensis, Hyperacanthus microphyllus, Combretum mkuzense*
East Malagasy regional centre of endemism	Madagascar	272 000	*Angraeceum sesquipedale, Typhonodorum lindleyanum, Ravenala madagascariensis*
West Malagasy regional centre of endemism	Madagascar	322 000	*Delonix regia, Pachypodium spp., Adansonia madagascariensis, Pachypodium decaryi, Adenia neohumbertii*
Beach Dunes	Somalia – South Africa	16 254	*Halopyrum mucronatum, Azima tetracantha, Phyllanthus reticulatus, Buxus hildebrantii, Maytenus undata, Vepris eugeniifolia, Dirachma somalensis*
Island of Comoros	Comoros	2155	*Philippia comorensis, Nuxia pseudodentata, Typhonodorum lindleyanum, Wielandia fadenii, Calophyllum inophyllum, Cordia subcordata*
Island of Mauritius	Mauritius	1900	*Diospyros egrettarum, Syzygium contractum, Zanthoxylum heterophylum, Eugenia hastilis, Sideroxylon cinereum, Souriana maritima, Olax psittacorum*
Island of Seychelles	Seychelles	260	*Cocos nucifera, Lodoicea maldivica, Mimusops sechellarum, Vateriopsis seychellarum, Intsia bijuga, Northea hornei, Dillenia ferruginea, Pisonia grandis*

Socio-ecological and economic value of coastal forests

The coastal forest mosaic provides a diversity of ecosystem services that are directly and indirectly linked to livelihoods of the coastal communities, both rural and urban, that are of significant environmental and socio-economic importance critical for the long-term survival of the region's economy. Directly linked ecosystem services are provisioning, regulating and cultural services while supporting services are indirect. Table 3 lists the various use values of coastal forests. The sections below provide more details on the four forms of ecosystem services derived from the coastal forests of the WIO region.

Provisioning services

The forested habitats harbour a wealth of species and genetic diversity that provide raw materials for livelihood support systems including food such as fruits, nuts, honey and fodder, as well as biomass for energy, such as wood fuel and charcoal for cooking. In Tanzania, rural communities depend on wood and charcoal for cooking, accounting for at least 92 per cent of the country's energy consumption and around 95 per cent of wood products. The forests also provide construction materials such as timber and poles, and for thatching, similarly offering commercially-valuable products including timber and other forest products whose direct beneficiaries are small and large-scale forest industries including sawmills, pulp and paper mills, furniture makers, and commercial handcraft producers spread along the coast. Other direct benefits are non-timber forest products such as health care products including herbal medicines, ornamental products, and as sources of freshwater supply for towns and villages. African violets (*Saintpaulia* spp.) whose 40 000 varieties are cultivated to form the basis of a US$ 100 million/year global houseplant trade, are all derived from a handful of species found in the coastal forests in Tanzania and Kenya. The forests directly provide habitat, subsistence and livelihood to forest dwellers, such as the Mikea of Madagascar and the Wasanya in Malindi, Kenya, among others, thereby supplying the means to hold these communities together. Butterflies are collected from Arabuko Sokoke Forest in Kenya, reared and exported to generate income for local communities living around the forest.

Table 3: Goods and services provided by coastal forests.

FOREST ECOSYSTEM SERVICE	
Provisioning of goods (Direct use)	Timber, poles Non-timber forest products (herbal medicine) Food (mushrooms, wild fruits, honey) Genetic resources Biomass energy (fuelwood)
Regulating services (Direct use)	Coastal, shoreline protection Air and water pollution and siltation reduction Microclimate function Carbon sequestration Maintaining water catchments
Cultural services (Direct use)	Cultural heritage Human habitat Recreation Nature tourism (ecotourism) Forests as objects of intrinsic value, or as a responsibility (stewardship) Cherished landscapes Sites for traditional rituals and ceremonies
Supporting services (Indirect use)	Endangered and charismatic species Threatened or rare habitats/ecosystems Biodiversity habitat Nutrient cycling (including detritus for aquatic food web), soil formation Watershed protection Wildlife habitat (including birds and aquatic species) Habitat for potential new future drugs, genes for plant breeding, new technologies development

Adapted from Pearce (1991).

Regulating services

The principle regulating functions include protecting shorelines from erosion and storms. Shoreline forests are a buffer against the actions of wind, waves and water currents. For example, beach forests hold together sandy soil and also act as barriers against storm waves from the sea. Vegetated dunes that are more resilient to wave-induced erosion act as a stabilizing agent in dune systems to protect shorelines through reduced erosion. Where land and sea and their processes are intermixed – where rivers link terrestrial habitats – to marine habitats riverine forests reduce soil erosion and siltation from upstream areas, and nutrients into the Indian Ocean that would have led to the degradation of the marine habitat and impact on marine life. For instance, Sigi River in East Usambara Mountain forests that drain into the Indian Ocean at Tanga. Riverine forests also filter degrading pollutants from inland freshwater systems thus maintaining water quality, as for example in the lower Tana River forest in Kenya. The forests also maintain ecological cycles and microclimate and sequester carbon. In addition, beach forests maintain moisture in the sandy soil, which is crucial to the survival of numerous living organisms along the coastal beaches.

Cultural services

Some of the coastal forests have an important cultural value to rural people, such that forests and people are historically and spiritually bound. They host indigenous forest dwellers and provide cultural services such as recreation, tourism, and sacred spaces, eg Pugu Hills Forest

Reserve in Tanzania and Kaya forests in Kenya provide groves for worship, ceremonies, burial grounds and meeting places for special occasions. A variety of wildlife species, many of them endangered and charismatic, that are found in the coastal forests have high aesthetic values and are valuable for ecotourism, tourism and recreation. Arabuko Sokoke in Kenya is the second most important bird conservation area and is an important tourist attraction for bird watchers. Quirimbas National Park, Bazaruto and Gorongosa National Parks and the Inhaca and Maputo Special Reserves in Mozambique are also important ecotourism areas.

The presence of lemurs that are endemic to Madagascar in the Ranomafana, Masoala and Andasibe-Mantadia national parks have made these parks important tourist attractions. Also, the presence of the 'Coco de mer' endemic palm to Seychelles in the islands of Pralin and Curieuse are important tourist attractions. Vegetated dunes have also high aesthetic value in addition to yielding substantial economic benefits by reducing damage to infrastructure during severe storms.

Supporting services

Coastal forests provide supporting services first as habitats for biological diversity that include valuable plants and animals of economic and biological importance, some of which in addition may have the potential to house bioactive compounds with biotechnology and medicinal applications. Consequently, coastal forests serve as areas for biodiversity conservation for wildlife. For example, Pugu Forest Reserve is the southernmost point in the range of an endemic East African coastal bird, the Sokoke pipit (*Anthus sokokensis*), and a number of bird species with unusual or restricted distributions are found in the forest. *Anthus sokokensis* is also found in Arabuko Sokoke forest in Kenya. It is endangered because it has very small habitat range which is becoming more fragmented. Rondo Forest Reserve provides habitat for the endemic and critically endangered primate, *Galagoides rondoensis* as well as serving as an important breeding site for bird species such as the endangered *Sheppardia gunning* (Samoilys et al., 2015).

Coastal forests also provide life supporting systems for forest dwellers, such as the Mikea of Madagascar and the Wasanya in Malindi Kenya, among others, through provision of habitat, subsistence and livelihood opportunities. They support productivity of the habitats through soil formation, storing and cycling nutrients. For example, riverine forests support the biological productivity through leaf litter and detrital matter which is exported to lagoons and the nearshore coastal environment, where it enters the marine food web.

Status and threats to coastal forests

Trends in coastal forest cover show a general decline characterized by fragmentation and shrinking of the forest patches. Large areas of the forest have been converted to farmland over many years of human habitation resulting in considerable loss of habitat and habitat continuity between the natural fragments. As a result, the forest's production potential and service provision to local livelihoods decline and the associated unique biodiversity of the region is under severe threat.

Of the original 291 250 km^2 of coastal forests in the Eastern Africa Hotspot, only 10 per cent (29 125 km^2) of the natural vegetation remains (WWF, 2006). In the coastal lowland forests of Cabo Delgado, the 6087 km^2 cover historically declined to only 1182 km^2, a loss of about 80 per cent (Timberlake et al., 2011). The remaining cover is distributed across more than 400 forest fragments of closed canopy Eastern Africa coastal lowland forest most of which are less than 20 km^2 in size. Remaining pristine forest covers a total of 6259 km^2 with the majority (4778 km^2) located in Mozambique. It is estimated that the remaining closed canopy forest extends over 4 km^2 in Somalia, approximately 1050 km^2 in Kenya, at least 970 km^2 in Tanzania (Burgess et al., 2003).

The coastal forests have experienced great pressure from several threats that have impacted negatively on the forest ecosystems and surrounding habitats and consequently threatening the survival of valuable biodiversity. Since forest resources form an integral part of the livelihood strategies of forest proximate coastal communities, anthropogenic factors attributable to increased human population and associated developments are the greatest threats. Poverty has been identified as the main root cause of the deforestation and forest degradation, particularly when forest dependent communities with limited alternative livelihood options resort to forests as a safety net, thus increasing the pressure on the forest resources. The impacts increase due to growing demand for agricultural land as well as for forest goods. Unsustainable use of forest resources, inappropriate land use practices, such as shifting cultivation frequent and uncontrolled bush fires, also threaten the forests. Economic development, comprising the conversion of forest land to other non-forest uses such as mining, hydropower, and urban centres is a major threat to coastal forests and is a consequence of a lack of awareness of the value of coastal forests and implications of their loss. In summary, the main threats have been: expanding agriculture; uncontrolled charcoal production and fuelwood collection; unsustainable and illegal logging; uncontrolled fires; unplanned human settlement and urbanization; destructive mining practices;

and infrastructural development. These are described in more detail in the sections that follow.

Expanding agriculture

Agricultural expansion is the biggest threat facing the coastal forests of the region. With human population growth at 2.5–3.5 per cent annually, and the need to feed the increasing population, the demand for farmland to support subsistence agriculture has led to more agricultural land to be carved out of the forests for crop cultivation and animal husbandry. The challenge of further expansion is accelerated by the fact that the soils are not fertile, and the farmers opt to practice shifting cultivation prospecting for higher yields. The growing population pressures also tend to decrease the length of fallow periods. Ultimately this leaves the remaining more fertile lowland forests vulnerable as subsistence as well as commercial farming continue to consume more and more of the region's forest habitat. Commercial agriculture for coconut, sisal, spices, fruit trees, rice, sugar and cashew nut plantations now occupy considerable areas of coastal land, having replaced lowland coastal forests and other natural habitats (WWF, 2006). Most of these excisions have been illegal but are favoured by weak administration of forests in respective jurisdictions. Unsustainable agricultural practices such as uncontrolled burning to clear for farmland, to drive animals for hunting and to reduce Tsetse flies further threaten the coastal forest and thicket patches, often replacing rare endemic coastal forest species with more common wide-ranging fire-adapted species (WWF-EARPO, 2002). The consequence of deforestation from conversion to agricultural land include increasing flooding and sedimentation of rivers and streams leading to increasing sediment loads that degrade marine habitats and threaten their associated biodiversity.

Degraded sacred coastal forest due to agricultural expansion, Kenya. © Chemuku Wekesa

Uncontrolled charcoal production and fuelwood collection

Charcoal production is a major threat to the coastal forests, largely because it is a socio-economic activity that does not require a lot of capital to engage, coupled with the readily available market for charcoal. Charcoal production therefore serves as a safety net for the rural population living adjacent to the forests with limited alternative livelihood options. The huge amount of charcoal produced from Eastern Africa coastal forests is causing major habitat loss near coastal towns and alongside main roads. In Tanzania for instance, in the last decade, coastal forests near Dar es Salaam (Pande Game Reserve, Pugu, Kazimzumbwi and Ruvu South Forest Reserves) have lost significant areas due to charcoal burning. Such areas remain a challenge to conservation particularly as they are also being encroached by expanding urbanization from Dar es Salaam. Further away from towns and main roads, fuel wood collection threatens the forests (Younge et al., 2002). This is because the majority of the rural population uses firewood as the main source of energy, and whose demand continues to increase due to rapid population growth.

Unsustainable and illegal logging for timber

Unsustainable logging is one of the direct causes of forest degradation. The commercial extraction of valuable timber species using licenses obtained from relevant authorities dates to the colonial period, but reforestation programmes after harvesting have been inadequate or lacking. Consequently, many of the slow growing timber species have been over-harvested until their populations collapsed. Further, even where licensed logging is undertaken, incidences of compromising forest administrations are prevalent leading to illegal logging in the region. The scale of illegal logging and timber trade is difficult to assess with accuracy, given the clandestine ways through which the related operations are conducted and these activities remain a problem in the region (WWF, 2006).

The main tree species targeted include *Pterocarpus angolensis*, *Milletia stuhlmannii*, *Afzelia quanzensis*, *Swartzia madagascariensis*, *Dalbergia melanoxylon* and *Milicia excelsa*. A number of logging concessions in Cabo Delgado Province of Mozambique are not well controlled leading to illegal activities linked to Tanzania and Chinese companies that decimate the tree populations of *P. angolensis*, *A. quanzensis* and *M. stuhlmannii* in the woodland forests on the slopes of the Messalo valley around Chitunda and Chai, now only discernible from a few large remnant trees (Mackenzie, 2006; Chilalo, 2008; Mackenzie and Ribeiro, 2009). In Madagascar, illegal logging of Rosewood and precious ebony-like timber from the genera *Dalbergia* and *Diospyros*, respectively, to meet increasing international demand has been a major cause of land degradation and deforestation (Ratsimbazafy et al., 2016). Wood carving industry has also contributed to illegal logging of trees. The most traded wood species are *Dalbergia melanoxylon* (African blackwood – often incorrectly referred to as ebony), *Brachylaena huillensis* (silver oak) and *Afzelia quanzensis* (pod mahogany). The slow growing *D. melanoxylon* is facing extinction in the region due to demand for its wood for making high quality musical instruments such as clarinets, oboes and piano keys, and for making woodcarvings for tourists. Unsustainable harvesting of *D. melanoxylon* and *B. huillensis* is common in southern Tanzania, as well as around Arabuko-Sokoke in Kenya.

Uncontrolled fires

Forest fires are very destructive, leading to loss of ecosystem services and economic opportunities that affect the livelihoods of forest dependent communities. Forest recovery from impacts of the destruction usually takes on average two decades. Uncontrolled fires are very extensive across coastal forests and the majority are associated with slash and burn of newly cleared fields, but there are also deliberate fires to drive animals for hunting, to facilitate collection of honey, and to remove Tsetse flies from an area. During the dry season, fire can invade lowland coastal forest patches and thicket vegetation destroying the vegetation that is less fire-adapted. Over time and with frequent and intense burning, fires suppress woody vegetation and create more fire-adapted wooded grasslands similar to Miombo woodlands (dominated by *Brachystegia* and *Julbernadia species*). This results in a loss of the narrowly endemic coastal forest specialist species and their replacement by wide-ranging species typical of Miombo (WWF, 2006).

Unplanned human settlements and urbanization

Encroachment by expanding local settlements is one of the major threats to coastal forests. Local populations have been moving away from the coastal margins into the wooded and forested interior of the coastal plateau in search of settlement areas and for cultivation. In much of the region, the expansion of settlements is mostly poorly planned and is strongly influenced by infrastructural development such as the presence of roads and passable tracks, including seismic survey cut-lines through forests that were used for oil and gas exploration many decades ago. Often the unplanned settlement is in forest areas that are poorly administered and protected, resulting in extensive forest and woodland clearance.

Strongly influenced by infrastructural development is rapid urbanization that also has a big impact on neighbouring forest resources. Poor planning has been a characteristic of establishment of urban centres in areas adjacent to coastal forests, ultimately encroaching on the forest reserves. Urban centres rely on natural assets such as forests for the well-being of their populations, especially for water and energy. The current rapid urbanization in the region means that urban planning should be undertaken in a way that can address the growing urban population and provide them with resources like water and energy in a sustainable manner. However, the increasing rural-urban migration to centres where people have inadequate services has resulted in the degradation of coastal forests. For example, in Dar es Salaam, expansion is starting to encroach into areas of natural habitat, some including lowland coastal forests with high biological values.

In some of the protected forests of Kenya and Tanzania (both Forest Reserves and traditionally protected forest areas), settlements have been established within the boundaries of the reserve. When this happens, farming activities also start and there can be much damage to the habitats (eg Kazimzumbwi and Vikindu Forest Reserves close to Dar es Salaam). In Zanzibar, the expansion of

Illegal logging in Marenje forest, Kwale, Kenya. © Chemuku Wekesa tourism into some coastal forests is also an issue for concern (WWF, 2006). Presently, most beach forests have been destroyed and developed for tourism, so much so that the original state of these forests has been obliterated. Additionally, people have planted alien species such as coconut and ornamental palm trees in place of the indigenous species for aesthetic purposes.

Destructive mining practices

Mining of underground minerals is also a major threat to coastal forests in the region. Countries in the region are endowed with a wealth of mineral resources and prospecting is still ongoing. These include gas, gemstones, iron ore, titanium ore, manganese ore, gold, limestone, uranium and kaolin. Mining of, for example, limestone, gold, gemstones, silica, iron ore, coral rag and manganese has destroyed large areas of forests as the value of minerals is assessed as more important for economic development than conserving the forests. The impacts of mining and prospecting on the forests are very severe to the forest ecosystem. The severity of the mining activities becomes more pronounced when the miners refuse to undertake any rehabilitation of the mined areas. This serves to destroy the forest ecosystem further, although in some instances rehabilitation is possible, as in the case of Bamburi cement rehabilitation initiative in Mombasa Kenya (WWF, 2006).

Infrastructural development

Infrastructural development in the region significantly contributes to forest habitat degradation and fragmentation leading to opening of forest resources to over-exploitation and invasion of exotic species. Such projects include road infrastructure development, industrial mining projects and development corridors. The oil and gas exploration work on land in Mozambique and Tanzania and the network of seismic survey cut-lines to allow for vehicle access (several decades ago in many cases) have served to facilitate movement and creation of new settlements in forested areas (Timberlake et al., 2010). The Exxon Mobil-led Rovuma LNG and Total-led Mozambique LNG projects – which will result in major onshore infrastructure and transport networks between Palma and Mocímboa da Praia, as well as the recent opening of the Mkapa Bridge over the Rufiji River and the Unity Bridge across the Rovuma River will probably accelerate commercial logging in the newly accessible forested areas.

Invasive species

Invasive alien plant species threaten biodiversity across the coastal forests, and are considered an increasing problem affecting many protected forests along the Eastern Africa coastal strip (Chenje and Mohamed-Katerere 2006; Wise et al., 2007). The alien invasive species is a common problem for many of the island states eg Rodrigues, and many parts of the mainland that needs close monitoring and control to minimize negative impacts on biodiversity.

MANAGEMENT AND GOVERNANCE OF COASTAL FORESTS

Management

Coastal forests fall under multiple management regimes at national and local authority levels. For instance, in Kenya, the protected area network at national level consists of national parks, national reserves, forest reserves, nature reserves and national monuments. National monuments which are part of coastal forests are sacred forests such as Kaya forests in Kenya. At a lower level, some of the forests are located on trust lands and fall under the control of local authorities. A few forests have no legal protection and fall within private land. Consequently, national parks, reserves, and monuments as well as forest reserves are managed by national governments while sacred forests in most cases are managed by the local elders. Forests within private land are at the management of individual landowners and hence classified as unprotected and highly vulnerable.

In the case of Tanzania, the protected area network at national level consists of national parks, game reserves, government catchment forests, game-controlled areas, forest reserves and nature reserves. Below the national level, a large number of forests, particularly in the coastal forest belt, fall under local authorities, owned and managed by the villagers. The management regimes in Kenya and Tanzania mirror those of other countries, namely Somalia, Comoros, Mozambique, Madagascar, Seychelles, Mauritius, and South Africa. In these management regimes no exploitation is allowed in national parks and protection levels are generally high.

Policy and legislation relating to the coastal forest areas include national policies for coastal zone management, policies and strategies for forest and wildlife, tourism strategies and the physical land planning laws. However, legislation on the management of the environment and natural resources are overlapping and confusing in most of these countries, posing serious challenges to sustainable management of the coastal forests and often exacerbating illegal exploitation of the forest resources. The legislation is therefore being rationalized through the enactment of new polices, amendments to existing legislation and new laws to remove the conflicting clauses and hence enhance coordinated management of natural resources including the coastal forests.

Institutional framework

The institutional frameworks that promote the interactions between people and the forests are largely an inheritance from the colonial governments. The countries have a civil service structure that includes ministries, permanent secretaries and national institutions dealing with different sectors of society and the economy. The countries have respective government ministries that oversee the management and conservation of forests either through departments of forestry or mandated parastatals. For instance, in Kenya, the Kenya Forest Service is responsible for the development and sustainable management, including conservation and rational utilization of all forest resources for socio-economic development. The Ministry of Natural Resources and Tourism is responsible for the protection of forests and the productive use of forest lands to meet demands for wood products in Tanzania, while in Mozambique, the Ministry of Land, Environment and Rural Development is responsible for sustainable management and conservation of forests in the country. Co-management of the forests through institutions such as community forest associations (CFAs) is also encouraged through the newly enacted forest conservation and management legislations/laws in most of the countries. This has allowed local communities to actively participate in forest management and conservation activities including decision-making. However, the institutional arrangements experience various challenges that constrain the effective management of coastal forests resources. The main constraints include weak coordination amongst involved sector agencies, lack of integrated and holistic land use plans, limited technical capacity to monitor and control coastal resource management and limited financial resources. Furthermore, institutional frameworks to address cross-border conservation and policy issues are inadequate.

Several NGOs are involved in forestry-related activities to support countries in the region to conserve coastal forests, key among them being WWF. Their interventions complement the government conservation and development initiatives and have greatly assisted the relevant ministries and government institutions responsible for conservation of forests to develop ecosystem specific conservation strategies, and enabling policies that will, if effectively implemented, improve the conservation status of the coastal forests.

CONCLUSIONS AND RECOMMENDATIONS

Conclusions

Coastal forests are critical to the survival of biodiversity, health of marine systems and in maintaining life support systems for coastal communities. Rich biodiversity and high levels of species endemism characterize the various forest mosaics in the WIO region placing them among some of the highest in the world, that include a wide variety of globally threatened fauna and flora. The forests are host to three out of 34 of the world's biodiversity hotspots – namely the Eastern Africa Coastal Forests Biodiversity Hotspot, the Madagascar Biodiversity Hotspot and the Maputaland–Pondoland–Albany Biodiversity Hotspot. Consequently, coastal forests are of global importance for their role in conservation of biological diversity. These biological values have contributed to these forests being at the centre of the tourism industry in most of the countries, a leading foreign exchange earner for their national economies.

These coastal forest ecosystems influence the marine systems. The forests reduce soil erosion in upstream areas and the resulting siltation and nutrient discharge into the Indian Ocean that could lead to the degradation of the marine habitats and impacts on marine life. Clearing of vegetation in coastal terrestrial ecosystems that neighbour marine ecosystems therefore has far-reaching impacts on the health of the marine ecosystem.

These forests provide the basis for a number of economic activities that are directly and indirectly linked to livelihoods of the coastal communities. They harbour a wealth of species and genetic diversity that provides raw materials for livelihood support systems including food such as fruits, nuts, honey, fodder, biomass for energy such as firewood and charcoal for cooking; construction materials such as timber and poles; and non-timber products such as herbal medicines. Some of the coastal forests have an important cultural value to rural people. They host indigenous forest dwellers and provide cultural services such as recreation, tourism, and sacred sites for rituals and ceremonies.

For a long time, coastal forests have been subject to considerable pressure including anthropogenic and natural threats leading to fragmentation and forest cover loss. The forests are threatened largely by anthropogenic activities and associated development pressures. The forests have declined due to expanding agriculture, over-exploitation for products, illegal logging and conversion to settlements, urbanization, infrastructural development and mining. Consequently, many species including endemic and rare species are facing tremendous pressure and hence threatened with extinction.

Poverty and increasing human population growth that have resulted in increased demand for farmland to support subsistence agriculture are also responsible for the degradation of the forests and pose the greatest threat to the survival of coastal forests. Other threats are the lack of alternative livelihood options for communities living adjacent to the forests, inadequate law enforcement, and low awareness of the value of coastal forests and consequences of the loss.

The coastal forest resources are managed by governments as national parks, nature reserves and/or national monuments through well-developed structures of forest management agencies. They are governed through forestry policies and legislation. However, there are shortcomings in their effectiveness to protect the forests. In most of the countries the agencies are often understaffed with inadequate funding to carry out their operations. Further, there are multiple sectoral policies that have conflicting objectives and hence impact negatively on the protection of the forests.

Recommendations

1. The governments in the region should promote the conservation of the coastal forests in the context of a framework that involves promoting a balance between the environment, society, and development and conservation strategies that meet the current needs without compromising those of the future generations.

2. Both *in situ* and *ex situ* conservation should be promoted to maintain the forest ecosystem and the natural habitats, and conserve biodiversity including critically endangered species that are endemic to these forests. This can be achieved through multi-stakeholder engagement bringing together governments, NGOs, CBOs, local communities and other interested players in conservation to map and develop strategies that can enhance sustainable management of the forests.

3. Integrated landscape management approaches that encourage communities to actively participate in forest management should be encouraged. Participatory forest management is such an example that gives community user rights and responsibilities that instil a sense of ownership and provide incentives to protect the forests. Also needed is proper land use planning to mitigate effects of urbanization and infrastructural development. Integrating local people's knowledge, values and cultural practices with science could lead to an improvement of conservation policies and implementation in terms of both conservation effectiveness and socio-economic equity.

4. Besides river basin and catchment management, invasive species control and removal and integrated coastal zone marine spatial planning should be supported to mitigate the threats to biodiversity attributed to invasion by alien species, and poor land use practices and planning.

5. National governance intersectoral platforms likely to affect the status of coastal forest landscapes should be institutionalised and strengthened to ensure adequate coordination and to minimize negative impacts on the forests. In this regard government ministries/departments with potentially conflicting objectives to biodiversity conservation and sustainable forest management should be sensitized.

6. Research that can generate robust evidence to guide formulation of policies is required to improve management and protection of the forests and hence achieve sustainable forest management. Research provides the foundation to monitoring changes in the forest ecosystems and responses by developing management interventions that can arrest degradation and forest cover loss.

7. Governments sharing the regional forest mosaics should promote and facilitate the development of partnerships between their forestry agencies to enhance exchange of knowledge and experiences and best management practices to enhance connectivity among the forest ecosystems, including those transboundary forest systems.

REFERENCES

Abdelaal, M., Fois, M., Fenu, G. and Bacchetta, G. (2018). Critical checklist of the endemic vascular plants of Egypt. *Phytotaxa* 360(1), pp. 19–34

Azeria, T., Sanmartin, I., Stefan, Âs., A Carlson and Burgess, N. (2005). Biogeographic patterns of the East African coastal forest vertebrate fauna. *Biodivers. Conserv*. 16(4), pp. 883–912. Springer Science Business Media B.V.

Barratt, C.D. (2017). Biodiversity patterns and conservation of the coastal forests of Eastern Africa PhD Thesis, Basel University, Basel, Switzerland

Borokini, T.I. (2014). A systematic compilation of endemic flora in Nigeria for conservation management. *J. Threat. Taxa* 6(11), pp. 6406–6426

Botes, A., McGeoch, M.A. and Van Rensburg, B.J. (2006). Elephant- and human-induced changes to dung beetle (Coleoptera: Scarabaeidae) assemblages in the Maputaland Centre of Endemism. *Biol. Conserv*. 130, pp. 573–582

Brooks, T., Balmford, A., Burgess, N., Fjeldsa, J., Hansen, L.A., Moore. J., Rahbek, C. and Williams, P. (2001). Toward a blueprint for conservation in Africa. *BioScience* 51, pp. 613–624

Brooks, T.M., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B., Rylands, A.B., Konstant, W.R., Flick, P., Pilgrim, J., Oldfield, S., Magin, G. and Hilton-Taylor, C. (2002). Habitat loss and extinction in the hotspots of biodiversity. *Biol. Conserv*. 16, pp. 909–923

Burgess, N.B., Negussie, G., Bechtel, P., Moises, N.O. and Doggart, N. (2003). Coastal forests in Northern Mozambique. *Arc*. 15, pp. 9–12

Burgess, N.D. and Clarke, G.P. (eds.) (2000). The coastal forests of Eastern Africa. IUCN, Gland, Switzerland and Cambridge, UK

Burgess, N.D., Clarke, G.P. and Rodgers, W.A. (1998). Coastal forests of Eastern Africa: status, endemism patterns and their potential causes. *Biol. J. Linn. Soc*. 64, pp. 337–367

Burgess, N.D., D'Amico Hales, J., Underwood, E., Dinerstein, E., Olson, D.M., Itoua, I., Schipper, J., Ricketts, T.H. and Newman, K. (2004). Terrestrial ecoregions of Africa and Madagascar: A conservation Assessment. 550 pp. Island Press, Washington DC

Burrows, J.E. and Timberlake, J.R. (2011). Mozambique's centres of endemism, with special reference to the Rovuma Centre of Endemism of NE Mozambique and SE Tanzania. In *Conference Abstracts: South African Association of Botanists – Annual Meeting 2011*. Abstracts of papers and posters presented at the 37th Annual Congress of the South African Association of Botanists held at Rhodes University, Grahamstown, 17–19 January 2011. *S. Afr. J. Bot*. 77(2), p. 518

Burrows, J.E., Burrows, S., Lötter, M. and Schmidt, E. (2018). Trees and Shrubs [of] Mozambique. Noordhoek, Cape Town, Publishing Print Matters

Chenje, M. and Mohamed-Katerere, J. (2006). Invasive Alien Species. In *Africa Environment Outlook 2, Our Environment, Our wealth*. pp. 331–349. UNEP.

Chilalo, M. (2008). Timber trade status in Mozambique: Effects on forest conserva¬tion, management and economic development. 80 pp. World Wildlife Fund (WWF). Washington DC

Clarke, G.P. (2000). Climate and climatic history. In *The Coastal Forests of Eastern Africa* (eds. N.D. Burgess and G.P. Clarke), pp. 47–67. IUCN: Cambridge and Gland

Darbyshire I., Timberlake J., Osborne J., Rokni S., Matimele H., Langa C., Datizua C., de Sousa C., Alves T., Massingue A., Hadj-Hammou J., Dhanda S., Shah T. and Wursten B. (2019). The endemic plants of Mozambique: diversity and conservation status. *Phytokeys* 136, pp. 45–96

Darbyshire, I., Anderson, S., Asatryan, A., Byfield, A., Cheek, M., Clubbe, C., Ghrabi, Z., Harris, T., Heatubun, C.D., Kalema, J., Magassouba, S., McCarthy, B., Milliken, W., de Montmollin, B., Nic Lughadha, E., Onana, J.M., Saïdou, D., Sârbu, A., Shrestha, K. and Radford, E.A. (2017). Important Plant Areas: Revised selection criteria for a global approach to plant conservation. *Biodivers. Conserv*. 26(8), pp. 1767–1800

FAO (2016). Country Profiles. FAO. Available at http://www.fao.org/countryprofiles/index/en/?iso3

Glaw, F. and Vences, M. (2007). *A Fieldguide to the Amphibians and Reptiles of Madagascar*. Third Edition. Vences and Glaw-Verlag, Cologne, Germany

Government of Seychelles (2011). *Fourth national report to the United Nations convention on biological diversity*. Environment Department, Government of Seychelles. Victoria, Republic of Seychelles

Hawthorne, W.D. (1993). East African coastal forest botany. *In Biogeography and ecology of the rain forests of Eastern Africa* (eds. J.C. Lovett and S.K. Wasser), pp. 57–59. Cambridge University Press. Great Britain, Cambridge

IUCN (1986). Review of the Protected Areas System in the Afrotropical Realm. pp. 275. Available at https://agris.fao.org/agris-search/search.do?recordID=XF2015044871

IUCN (2016). A global standard for the identification of Key Biodiversity Areas, version 1.0, 1st edition. IUCN, Gland. Available at https://portals.iucn.org/library/sites/library/files/documents/2016-048.pdf Accessed July 2021

Joordens, J., Feibel, C.S., Vonhof, H., Schulp, A.S. and Kroon, D. (2019). Relevance of the eastern African coastal forest for early hominin biogeography. *J. Hum. Evol*. 131(2019), pp. 176–202

Kent, P.E. (1972). Mesozoic history of the east coast of Africa. *Nature* 238, pp. 147–148

Lovett, J.C, and Wasser, S.K. (1993). Biogeography and Ecology of the Rain Forests of Eastern Africa Cambridge: Cambridge University Press.

Mackenzie, C. (2006). *Forest Governance in Zambézia, Mozambique: Chinese Take-away!* Final report for Fongza, Mozambique. Available at www.illegal-logging.info/uploads/Mozambique_China.pdf

Mackenzie, C. and Ribeiro, D. (2009). Tristezas tropicais: more sad stories from the forests of Zambezia. Amigos da Floresta and Justica, Ambiental. Maputo, Mozambique

Mathews, W.S. (2007). Contributions to the ecology of Maputaland, southern Africa, with emphasis on Sand Forest. PhD Thesis, University of Pretoria

Mittermeier, R.A., Myers, N., Mittermeier, C.G. and Robles, G.P. (1999). Hotspots: earth's biologically richest and most endangered terrestrial ecoregions. CEMEX, SA, Agrupación Sierra Madre, SC. Mexico, DF.

Mucina, L., Adams, J.B., Knevel, I.C., Rutherford, M.C., Powrie, L.W., Bolton, J.J., van der Merwe, J.H., Anderson, R.J., Bornman, T.G., le Roux, A. and Janssen, J.A.M. (2006). Coastal vegetation of South Africa. In *The vegetation of South Africa, Lesotho and Swaziland* (eds. L. Mucina and M.C. Rutherford), pp. 658–697. South African National Biodiversity Institute

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B. and Kents, J. (2000). Biodiversity hotspots for conservation priorities. *Nature* 403, pp. 853–858

Pearce, D.W. (1991). An economic approach to saving the tropical forests. In *An economic policy towards the environment* (ed. D. Helm). Oxford, UK, Blackwell Scientific Publications

Procter, J., (1984). Tropical forest litterfall. II. The data set. In Tropical Rain Forest: Ecology and Management (eds. Chadwick, A.C., Sutton, S.L.), Supplementary Volume Blackwell Scientific Publications, Oxford, pp. 83–113

Raselimanana, A.P. and Rakotomalala, D. (2003). Chamaeleonidae, Chameleons. In *The Natural History of Madagascar* (eds. S.M. Goodman and J.P. Benstead), pp. 961–969. The University of Chicago Press, Chicago, USA

Ratsimbazafy, C., Newton, D. and Ringuet, S. (2016). *Timber Island: The Rosewood and Ebony Trade of Madagascar*. TRAFFIC. Cambridge, UK

Republic of Mauritius (2010). *Fourth National Report on the Convention on Biological Diversity*. Ministry of Environment and Sustainable Development, Mauritius

Robertson, S.A. and Luke, W.R.Q. (1993). Kenya Coastal Forests. The report of the NMK/WWF Coast Forest Survey. WWF Project3256: Kenya Coast Forest Status, Conservation and Management, Nairobi, Kenya

Saleem, U. and Hussein, G. (2016). National Biodiversity Strategy and Action Plan of Somalia. FAO, Somalia

Samoilys, M., Pabari, M., Andrew, T., Maina, G.W., Church, J., Momanyi, A., Mibei, B., Monjane, M., Shah, A., Menomussanga, M. and Mutta, D. (2015). Resilience of Coastal Systems and Their Human Partners in the Western Indian Ocean. IUCN ESARO, WIOMSA, CORDIO, UNEP Nairobi Convention. Nairobi, Kenya

Sayer, J.A., Harcourt, C.S. and Collins, N.M. (eds.) (1992). *The conservation Atlas of Tropical Forests: Africa (1992)*.

Senterre, B. and Wagner, M. (2014). *Mapping Seychelles habitat-types on Mahé, Praslin, Silhouette, La Digue and Curieuse*. Consultancy Report by United Nations Development Programme. Victoria, Seychelles

Sigren, J.M., Figlus, J. and Armitage, A.R. (2014). Coastal sand dunes and dune vegetation: Restoration, erosion, and storm protection. *Shore Beach* 82(4), pp. 5–12

Silander, J.A.Jr. (2001). Temperate rainforests. In *Encyclopaedia of Biodiversity*, Vol. 5., pp. 607–626. Academic Press, San Diego, USA

Timberlake, J., Goyder, D., Crawford F., Burrows J., Clarke, G.P., Luke, Q., Matimele, H., Müller, T., Pascal, O., de Sousa, C. and Alves, T. (2011). Coastal dry forests in northern Mozambique. *Plant. Ecol. Evol.* 144(2), pp. 126–137

Timberlake, J.R., Goyder, D.J., Crawford, F. and Pascal, O. (2010). *Coastal dry forests in Cabo Delgado Province, northern Mozambique – botany and vegetation*. Report for "Our Planet Reviewed" a joint initiative Pro-Natura international and the French Museum of Natural History, 92 pp. Royal Botanic Gardens, Kew, London

Van Wyk, A.E. and Smith, G.F. (2001). Regions of floristic endemism in southern Africa. A review with emphasis on succulents. Umdaus Press, Pretoria

White, F. (1983). The vegetation map of Africa. UNESCO, Paris

Wise, R.M., van Wilgen, B.W., Hill, M.P., Schulthess, F., Tweddle, D., Chabi-Olay, A. and Zimmermann, H.G. (2007). The Economic Impact and Appropriate Management of Selected Invasive Alien Species on the African Continent. Final Report Prepared for Global Invasive Species Programme (GISP)

WWF (2006). The Eastern Africa Coastal Forest Ecoregion. Strategic Framework for Conservation, World Wide Fund for Nature (WWF). Eastern Africa Regional, Nairobi

WWF-EARPO (2002). *Coastal Forests of Eastern Africa: Draft Action Plan*. Nairobi: WWF-EARPO.

WWF-US (2003). Ecoregional reports: Northern Zanzibar-Inhambane coastal Forest Mosaic. Eastern and Southern Africa Bioregions. World Wide Fund for Nature (WWF)

Younge, A., Negussie, G. and Burgess, N. (2002). *Eastern Africa Coastal Forest Programme*. Regional workshop report February 4–7, 2002. World Wide Fund for Nature (WWF), Eastern African Programme Office, Nairobi

CRITICAL HABITATS

MARINE AND COASTAL CONNECTIVITY

Joseph M. Maina, Majambo Gamoyo, Vanessa Adams, Stephanie D'agata, Jared Bosire, Julius Francis and Dixon Waruinge

Hermit crab *Dardanus megistos* on seagrasses at Inhaca Island. © José Paula

INTRODUCTION

The health of global marine ecosystems is in severe decline from multiple pressures, including overfishing, pollution, invasive species, coastal development, and climate change, that compromise the ability of ocean and coastal ecosystems to support and sustain the essential goods and services for human persistence (Myers and Worm, 2003). Unregulated expansion of existing uses of the ocean, and the addition of emerging uses, such as renewable energy, large-scale aquaculture and mining, along with a rapidly growing coastal human population, are likely to exacerbate further the decline of marine ecosystem health (Cinner et al., 2018; Kroodsma et al., 2018; McCauley et al., 2015). As human populations continue to grow and technologies continue to advance, a significant challenge is to counteract ecosystems and biodiversity degradation across the oceans, particularly in high seas Areas Beyond National Jurisdiction (ABNJ) (Murawski, 2010).

The high seas make up two-thirds of oceans and are largely unclaimed and ungoverned. In effect, there are no legal mechanisms for governments to create marine reserves in these largely ungoverned, ecologically important areas. High seas include rare and fragile ecosystems and are critical migration routes that help sustain species, supporting ecosystems and livelihoods worldwide (Scovazzi, 2004). This notwithstanding, several human activities occur within ABNJ, including commercial shipping and fishing (Heffernan, 2018). Globally ABNJ accounts for up to USD 16 billion a year in fisheries catch (Sala et al., 2018) and is also a prime territory for discovering valuable mineral deposits, potent pharmaceuticals, and oil and gas reserves (Heffernan, 2018). At the same time, reciprocal legal obligations to protect the ABNJ are primarily overlooked (Ardron et al., 2008). Yet, as destructive activities continue to unfold on the high seas, management actions mainly focus on coastal and inshore regions, where our understanding of marine ecosystems is best (Heffernan, 2018). Improving our knowledge of marine ecosystems within marine protected areas (MPAs) and in the high seas and the foundational ecological process that functionally connects them is key to broadening conservation focus beyond territorial boundaries (Ardron et al., 2008).

Marine functional connectivity transcends maritime boundaries to support the most fundamental ecological function of connecting ecosystems, including the highly migratory species such as tuna, some sharks and long-lived species that move between the high seas and exclusive economic zones (EEZs) (Calich et al., 2018). Due to this highly migratory nature, these species tend to be intensely fished and overexploited (Campana, 2016). Oceanic sharks, of which 44 per cent are threatened (Dulvy et al., 2014), spend a lot of time in the high seas, where shark fishing is largely unregulated and unmonitored. As a platform to aid governments to conserve the high seas, where non-spatial monitoring is complex, and where data gaps obstruct conventional management approaches (Ardron et al., 2008), area-based planning across maritime boundaries, including marine spatial planning (MSP) and Ecologically or Biologically Significant Marine Areas (EBSAs) is a practical way forward.

Marine reserves, advocated as one tool to preserve and maintain biodiversity and to mitigate adverse effects of anthropogenic activities, have been implemented to a variable degree of success, including in the high seas where currently 12 marine protected areas (MPAs) exist (Smith and Jabour, 2017; Roberts, 2012). However, MPA design and implementation on the high seas is complicated because (i) little is known about the intricate ocean ecosystems far offshore, and (ii) the complex, slow and challenging process of planning and negotiations involved (Smith and Jabour, 2017). An evidence-based approach to protecting the high seas will require massive amounts of research. For example, to get a better sense of the scale of the looming ocean crisis, scientists need to map deep-seabed habitats (eg Harris et al., 2014) and understand key processes such as physical and functional connectivity. In the meantime, identification of suitable biodiversity surrogates, adoption of the precautionary principle (Lauck et al., 1998), and functional connectivity could be used as the main focus of the conservation goals guiding the identification of areas suitable for inclusion in the high seas MPAs (Álvarez-Romero et al., 2018).

Functional connectivity, or the exchange of individuals among marine populations, is fundamental for ecological processes such as population dynamics, evolution, and community responses to climate change (Cowen et al., 2007). Connectivity facilitates recovery processes after disturbance, through spillover of mobile juveniles and adults from MPAs into adjacent unprotected habitats and seeding unprotected sites with larvae spawned within MPAs (Roberts et al., 2017). Recovery through resettlement depends mainly on maintaining the supply of larvae, underpinning the need for functionally connected networks of marine reserves. Consequently, the long-term persistence of marine ecosystems and ecosystem services they provide hinges on identifying mesoscale connectivity patterns to link marine reserves within networks across the maritime jurisdiction.

In this chapter, marine connectivity is evaluated and used as one of the main focus of the conservation goals,

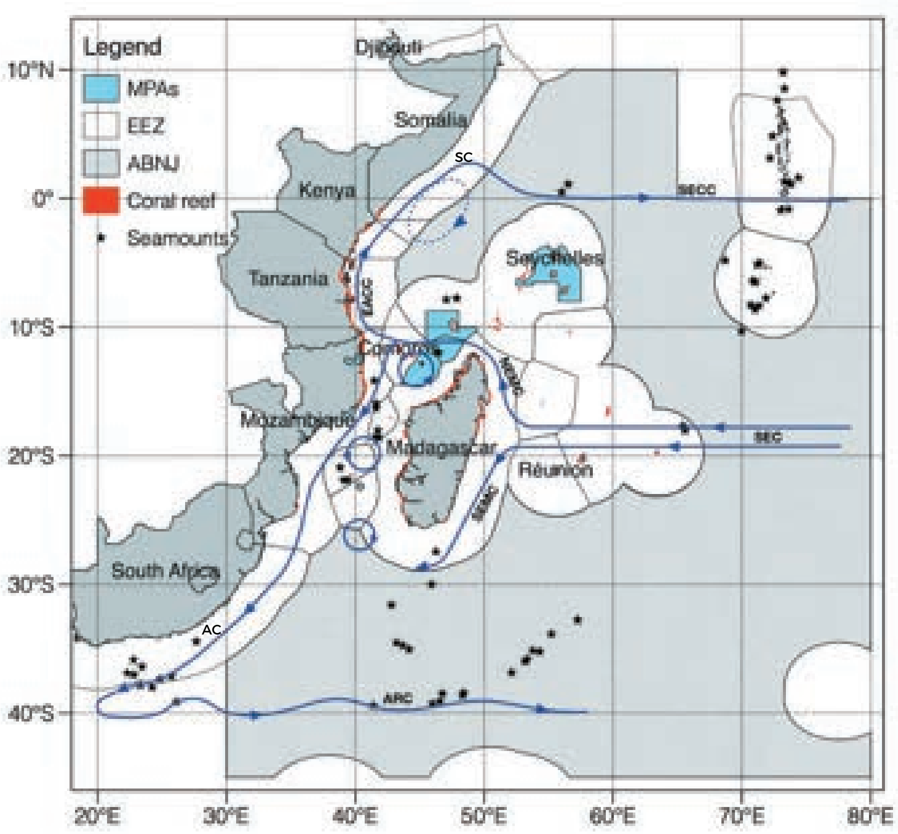


Figure 1: Map of the region showing the ABNJ, EEZ, MPA, geomorphic habitats and the main oceanographic circulation in summer (Northern Hemisphere) adapted from Schott and McCreary (2001). The major currents illustrated include; the South Equatorial Current (SEC), the North East Madagascar Current (NEMC) and the South East Madagascar Current (NEMC), the East African Coastal Current (EACC), Somalia Current (SC), the South Equatorial Counter Current (SECC). Further south is the Agulhas Current (AC) and the Agulhas Return Current (ARC).

guiding identifying areas suitable for inclusion in the high seas MPA. Connectivity patterns among existing MPAs, coral reefs and seamounts are assessed at large spatial scales to identify the gaps and opportunities for maintaining functional connectivity. Finally, regional scale prioritization across maritime zones of exclusive economic zones (EEZ) and ABNJ can be applied using area-based tools.

Three goals to maximize conservation outcomes guided the identification of areas suitable for inclusion in the MPA network. These goals apply nationally, and they guide the identification of representative marine reserves in all the marine regions. In the absence of comprehensive knowledge on high seas biodiversity, the planning goals were (i) to represent geomorphic seafloor habitats by protecting 10 per cent of their current distribution; (ii) to promote the long-term population viability of focal species by maintaining natural connections and connectivity corridors within marine reserve network mediated by larval dispersal, and (iii) to minimize human pressure on ecosystems in the EEZs, while promoting consensus by selecting less fished areas in the high seas.

METHODS

Study area

The Western Indian Ocean (WIO) covers 30 million km^2 of ocean off the coasts of eastern and southern African countries, equivalent to 8.1 per cent of the global ocean surface (Fig. 1). It comprises ten countries – Comoros, France (overseas territories), Kenya, Madagascar, Mauritius, Mozambique, Seychelles, Somalia, South Africa and Tanzania. Of these, five are mainland continental states on the western boundary of the WIO, four are small island states, and Madagascar is a large island, with a combined EEZ covering over 6 million km^2 and a combined coastline of over 15 000 km (UNEP/Nairobi Convention Secretariat, 2009).

The WIO is one of the Regional Seas identified by the United Nations Environment Programme (UNEP). The eastern limit of the WIO is not explicitly defined. For this study, the WIO ABNJ region as an intersection of FAO fishing zone 51 and the Regional Fisheries Management

Organisation (RFMO) defined Southern Indian Ocean Fisheries agreement areas (SIOFA) was adopted (Fig. 1). Consequently, the eastern and the southernmost boundaries were set to 75°E and 44°S, enclosing an ABNJ region of ~ 15.5 million km² (Fig. 1). Eight locations, covering 27 per cent of the WIO ABNJ, are designated as EBSAs.

Regional fisheries

WIO ABNJ experiences a high intensity of fishing, with an estimated cumulative effort of 265 000 hours by 19 countries, with net revenue of USD 537 million (Sala et al., 2018). Of the 19 countries that fished in FAO zone 51 in 2016, only four countries (Tanzania, Seychelles, Comoros and Maldives) were from the WIO region and earned ~USD 5 million (Sala et al., 2018). According to the data from SeaAroundUs (USA, 2007), the average fish landing within the EEZ from 2009–2014 was 682 265 t/year, with Tanzania, Mozambique, Madagascar and Somalia landing the highest amount. Industrial fishing, by comparison, was relatively low (21 per cent) compared to artisanal (61 per cent). The low industrial landing (primarily from the high seas) does not reflect the importance of the ABNJ to the WIO countries. The increased functional connectivity demonstrated in this and other studies suggest a high interconnection between the EEZ and high seas.

The fact that WIO countries don't have control over the exploitation of the adjacent high seas, an area with significant influence on fish stocks and fisheries of the ten WIO counties – and by extension, socio-economic settings raises important issues of justice, fairness and equity in the extraction of natural resources.

Dispersal modeling

The Mercator Ocean's Global ocean physical reanalysis GLORYS2V1 (Ferry et al., 2012) was employed to simulate larval dispersal, covering the WIO region extent (11°N to 40°S and 20°E to 75°E). The model's spatial resolution is 1/4°, and the temporal scope was daily from 1 January 2000 to 31 December 2010. Larval dispersal simulations for coral reefs, MPAs, ABNJ and seamounts were performed using Ichthyop (Lett et al., 2008) and run offline using the daily (24 h) velocity fields of the hydrodynamic model.

Advection of the virtual larvae was simulated using a 4th order *Runge-Kutta* integration scheme, and a random walk was applied using a dissipation rate of 1 x 10−9 m²/s³ for individual virtual larvae to account for turbulent motion not captured at the resolution of the oceanographic data (Peliz et al., 2017).

Connectivity among MPA's, coral reefs, and seamounts

Spatial data for MPAs for the WIO were obtained from a recently constructed WIO MPA comprehensive database containing 120 MPA records (unpublished data). Coral reef data were obtained from the Millennium Coral Reef Mapping Project archived at UNEP–WCMC as shapefile at 1 km resolution. Because the Mercator ocean data has a spatial resolution of ~25 km, the coral reef layer was re-sampled to 25 km² grids. Seamount data were obtained from the global seafloor habitat database (Harris et al., 2014). A subset of seamounts intersecting the study area at a depth range of 2–1000 m was defined for the analysis (Fig. 1). Centroids from MPA, coral reefs and seamounts (N=120, 242, and 67 respectively) were set as the release and settlement locations of virtual larvae.

One thousand virtual larvae were released from each centroid from January to December for 11 years (2000–2010) and tracked over 30 days, the average Pelagic Larval Duration (PLD) of fishes with a time step iteration of six hours (ie ~14 million virtual larvae released across all centroids) (Luiz et al., 2013; Andrello et al., 2017). The primary output of each simulation represented an estimate of the total amount of larvae transported between each of the 429 locations, including local retention.

Connectivity between ABNJ and EEZ and territorial waters

The EEZ is an area beyond and adjacent to a coastal state's territorial sea to a limit of 200 nautical miles from the baseline prescribed by the United Nations Convention on the Law of the Sea (1982). To estimate connectivity between EEZ and the high seas, at every grid in both areas, the release and tracking of particles was undertaken. ABNJ consisted of 16 515 grids, where larvae were released every 6 hours over ten years from January to December between 2000 and 2010 and tracked for 30 days (in total ~19 million virtual larvae). The EEZ dataset containing 21 EEZ features for the region was obtained from the UNEP-WCMC website[1] (Fig. 1). One thousand virtual larvae were released and tracked from within each EEZ from January to December between 2000–2010.

1. https://www.unep-wcmc.org

Connectivity indicators

Using the connectivity matrix as the input, the connectivity matrix *C* was defined as the matrix formed by the connection probabilities *C(i,j)* (Andrello et al., 2017). A suite of connectivity metrics was generated among the four habitats (ie MPAs, coral reefs, seamounts, and ABNJ-EEZ). *Connection probability C(i,j)* was the fraction of larvae originating in release point of interest *i* that ended up in destination point of interest *j* (Andrello et al., 2017). *Connectance* was defined as the fraction of connections with non-zero probability out of the total number of connections (ie the number of non-zero elements of *C* divided by the squared size of *C*). The *Betweenness Centrality* was calculated by determining the number of times a particular node, in this context, a reef, MPA or a seamount, served as a stepping-stone in the shortest paths between all other pairs of nodes in the network. *Betweenness Centrality* measure can be used to identify important stepping-stones that facilitate connectivity in a network. Also computed were the *degree* metrics: *in-degree* indicates the number of connections coming into each planning unit, and *out degree*, which shows the number of connections originating from each planning unit (Minor and Urban, 2008).

Designing a network of MPA's across maritime jurisdiction

The Marxan objective is to minimize the total cost of implementing the reserve network plan while ensuring the set conservation objectives are met. As part of the regional wide prioritization process, defining spatially consistent information on the habitat distributions across the planning domain was the first step. Given that prioritizing areas within EEZ and the high seas was required, Marxan was used with zones to differentiate between MPAs within EEZ and the ABNJ. This was done for two reasons: 1) the types of governance arrangements needed to designate and enforce MPAs are different between these two areas; therefore, zoning for them separately allows policymakers useful detail, 2) the types of human uses (and related cost measures) are different for these two areas and therefore to minimize the costs, the use of Marxan with zones allowed us to differentiate these costs.

For conservation features, seafloor morphology habitat maps were used as they are found in varying proportions within and outside EEZ. Three broad conservation goals were defined as follows: (i) to represent geomorphic seafloor habitats by protecting 10 per cent of their current distribution; (ii) to promote the long-term population viability of focal species by maintaining natural connections and connectivity corridors within marine reserves network mediated by larval dispersal, and (iii) to minimize human pressure on ecosystems in the EEZs, while promoting consensus by selecting less fished areas in the high seas. The connectivity metrics *Betweenness Centrality* and *Degree* were used to inform the selection of important areas for connectivity. A 100 per cent target for the connectivity measures was set to ensure the design of a connected reserve system that would be self-sustaining.

For the EEZ zone, the cost as the gravity of markets was set, which is a proxy for human pressure on marine ecosystems (Cinner et al., 2016; 2018). The gravity of markets represents the intensity of human impacts in the surrounding seascape. It is measured as a function of human population size and accessibility to marine resources ("gravity") (Cinner et al. 2018). For the ABNJ zone, the conservation cost was set as the fishing effort using Global Fishing Watch data based on an automatic vessel identification system for 2016 (Kroodsma et al., 2018). An optimal Boundary Length Modifier (BLM) value (0.007) was selected using Stewart and Possingham (2005) calibration method, which minimizes the trade-off in reduced boundaries and increased costs. All existing MPAs were locked into the analysis, as Watts et al. (2009) described.

Women walking along a beach in Zanzibar. © Rahim Saggaf

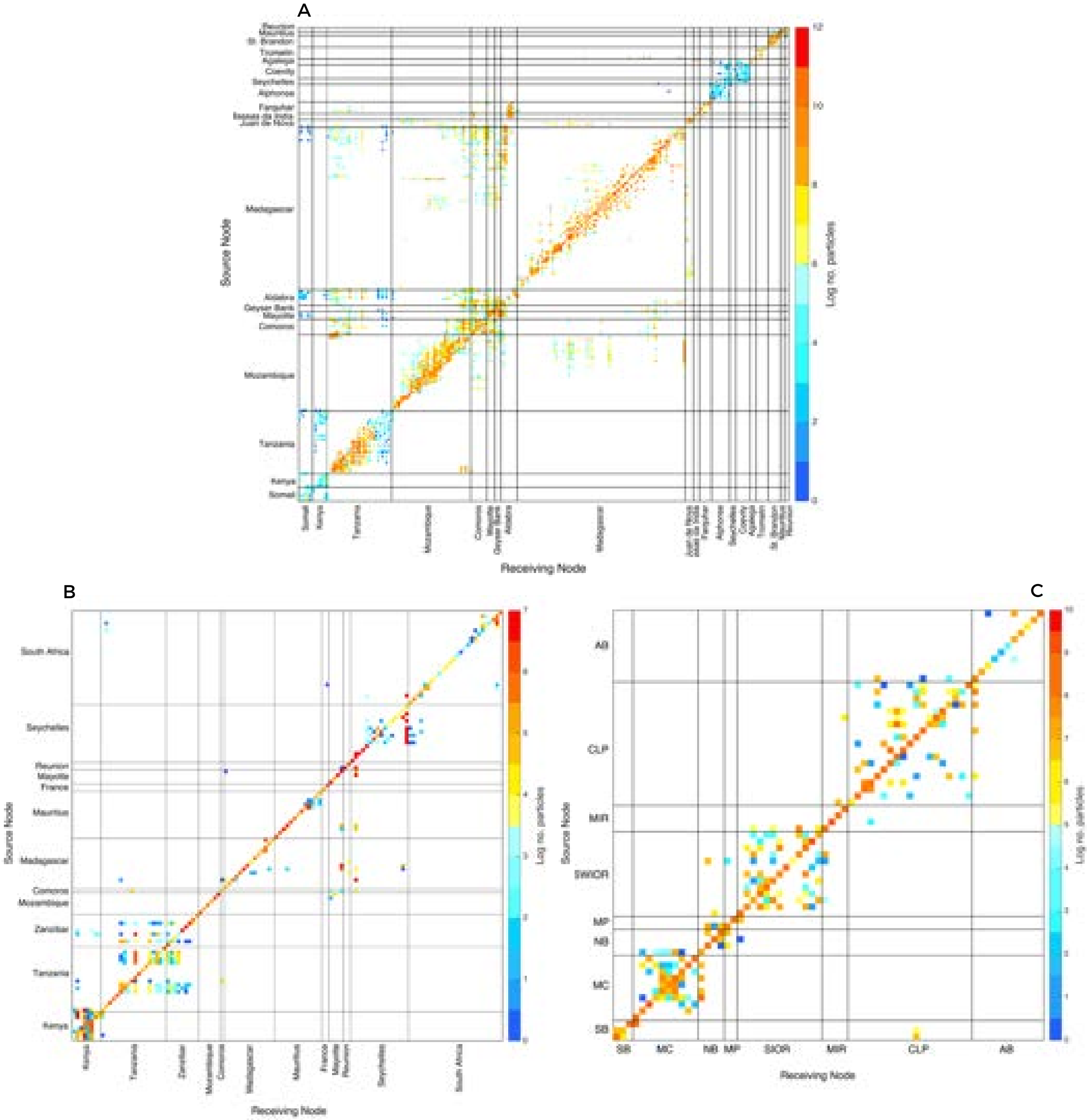


Figure 2: Connectivity matrices indicating the exchange of virtual larvae originating from a location *k* to recruit in a settlement location *l* after completion of a 30-day Pelagic Larval Duration (A) illustrates coral reefs, (B) MPAs, and (C) seamounts. Self-seeding (recruits that settled into their origin habitats) follows the diagonal. The connectivity matrices are made of 243 120 and 67 features of coral reefs, MPAs and seamounts, respectively, in the Western Indian Ocean.

The scale shows the log number of particles. Seamounts are grouped by Ocean Basins: SB = Somali Basin; MC = Mozambique Channel; MP = Madagascar Plate; NB = Natal Basin; SIOR = Southwest Indian Ocean Ridge; MIR=Mid Indian Ocean Ridge; CLR = Chagos-Lacadive plateau; AG = Agulhas Bank.

These are based on larval abundance at the end of a dispersal period. Consequently, the maps should be interpreted as potential larval export if larval production was constant across release locations and absent outside the release locations.

RESULTS

How connected are WIO MPAs?

Out of 14 280 possible paired connections, 248 connections were found (ie a connectance of 0.02 of the possible 1). When MPAs were connected, the connection probability was always low to moderate (median 0.07, interquartile range 0.29) (Fig. 2B). The connectivity of MPAs along the East African coast was the strongest (0.5–1) amidst the overall weak MPA connectivity in the region. Based on the degree metric of the total number of incoming and outgoing connections (Minor and Urban, 2008), MPAs in Tanzania (Mnazi Bay, Tanga, and Zanzibar) had the highest number of connections while Madagascar had the lowest. Half of the MPAs in the region are isolated, where 55 MPAs (46 per cent) are not seeded by any other MPA (zero incoming connections), and 62 do not seed any other MPA (50 per cent) (Fig. 2B). Overall, 38 MPAs (28 per cent) are completely isolated (zero incoming and outgoing connections).

Closeness centrality (how close a particular node is to the other nodes in the network) was overall very low (mean 0.00), reinforcing the finding that WIO MPAs are poorly connected. *Betweenness centrality* (identifies which MPAs act as gateways to larvae and gene transfer) was highest for the MPAs on the East Africa coast, with Menai Bay in Zanzibar, Mombasa, Mnazi Bay-Ruvuma Estuary, Tanga Coelacanth, and Malindi-Watamu among the highest larval corridor. Density maps of the larval flow indicate high density in Tanzania and Kenya, while in Zanzibar, in addition to self-seeding, MPA's tended to seed Tanzania mainland coast and Kenya (Fig. 2B).

Coral reef connectivity

Overall, WIO reefs are well connected, with a connectance of 0.05 (2868 connections out of possible 57 840). However, most connected coral reefs did not intersect with MPAs, as most connections were outside MPAs. The WIO coral reef network consists of densely connected clusters, sparsely interconnected (Fig. 2A). For example, along the East African coast, the dominant connectivity pattern is south to north, with Tanzania supplying coral larvae to Kenya and Kenya supplying to Somalia along the northward-flowing East African Coastal Current (EACC) (Fig. 2a). A north-south connection is also evident where reefs in Somalia seed the northern bank of Kenya during the reversal of the Somali Current. Islands in the Comoros Basin (Comoros, Mayotte, Geyser Bank and Aldabra) and Madagascar act as corridors for potential recruits *en route* to continental East Africa in Mozambique, Tanzania, Kenya, and Somalia. Self-recruitment (particles settling within their release location) dominated, as illustrated along the diagonal line. Madagascar appears to have the most connected reefs and primarily seeds Somalia, Kenya, Tanzania, Mozambique, Comoros, Mayotte and Aldabra to the north. At the same time, Madagascar receives less from other reefs except for Mozambican reefs.

Reefs in the south-eastern WIO (Agalega, Tromelin, St. Brandon, Mauritius and Reunion) are completely isolated from the western part of the domain except for rare westward dispersal from Agalega and Tromelin to Alphonse, Bassas da India and into Madagascar. There are two breaks/barriers to dispersal, as illustrated in the connectivity matrix, where the first barrier is located north of the Mozambique Channel. None or few particles cross into or out of the Mozambique Channel, effectively cutting the Channel off from the north (Fig. 1).

The central barrier separates the Seychelles archipelago from the southern (Mauritius, Reunion) and western reefs (Madagascar, continental East Africa); therefore, the isolated reefs depend entirely on recruits from local sources (ie self-recruitment). The central barrier may be from the South Equatorial Current (SEC), which forks northwards to create a barrier between the Seychelles and Madagascar/continental East Africa, and southwards to create a barrier between Madagascar and SE WIO reefs. The northern portion of the Mozambique Channel is a vital dispersal corridor for corals as it comprises dense coral reef networks (Fig. 1).

How connected are the seamounts?

Possible preferential routes for larvae exchanges among seamounts were explored to provide a comprehensive analysis of potential connectivity. Although less is known about patterns of connectivity of seamounts, model results show that overall WIO seamounts are moderately well connected, with a connectance of 0.05 (237 connections out of possible 4489). In pairwise comparisons, seamounts within the Mozambique Channel (MC), the South Indian Ocean Ridge (SIOR) and Chagos-Lacadive plateau (CLP) were connected (Fig. 2C). Long distance connection was also evident where seamounts within Chagos-Lacadive plateau were connected to those in the Mid-Indian Ridge (Fig. 2C).

Similar to shallow populations along coastlines, stepping-stone may be appropriate for many deep-sea species, particularly those arranged linearly along the mid-oceanic ridge or linear array of seamounts.

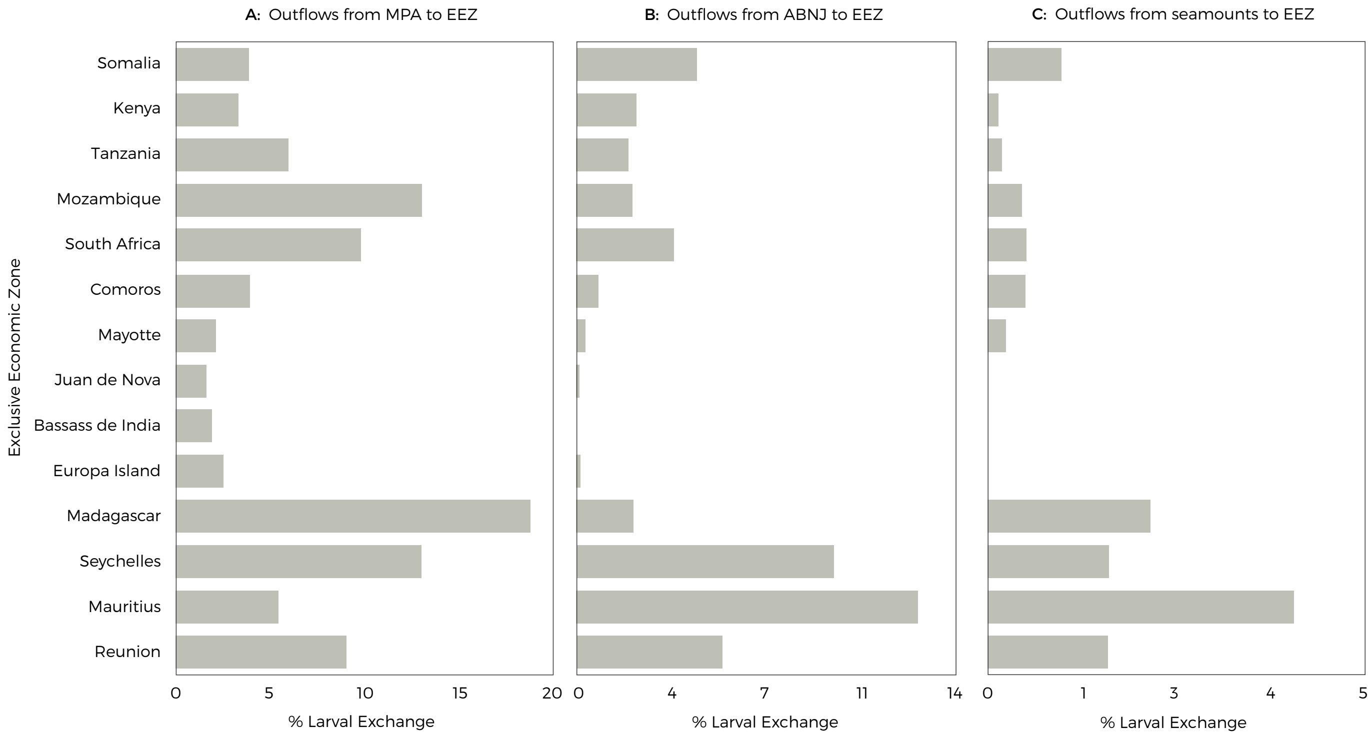


Figure 3: Bar graph indicating (A) proportion of larvae from MPAs into EEZs (by country); (B) proportion of larvae from ABNJ into EEZs; and (C) proportion of larvae from seamounts into EEZs.

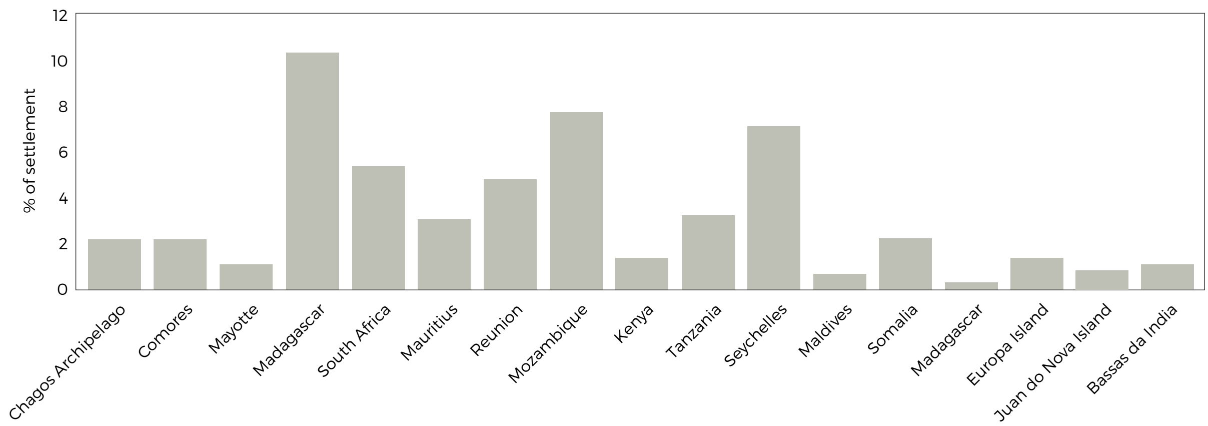


Figure 4: Proportion of larval released in ABNJ that settled within EEZ by country.

In contrast, open ocean spaces that separate a linear array of seamounts create an effective barrier to dispersal and connectivity decreases creating regionally isolated populations.

This scenario is evident in Fig. 2C, 15 seamounts were isolated. They didn't receive larvae from other seamounts, and 12 were non-seeding, while seven, located off the South African coast along the path of the Agulhas current, were completely isolated (Fig. 2C).

Connectivity between MPA, EEZ, and ABNJ

Madagascar, Mozambique and Seychelles receive most of the larvae generated within MPAs (Fig. 5), respectively 19, 14 and 15 per cent (Fig. 3A), while relatively fewer larvae settled in Kenya and Tanzania. Somalia, which has no MPAs, received larvae (5 per cent) from MPAs from other countries' EEZs. Most of the larvae released from ABNJ settled in Mauritius, Seychelles and Madagascar EEZs, while Somalia and Mozambique received a relatively high proportion compared to other continental countries (Fig. 3B).

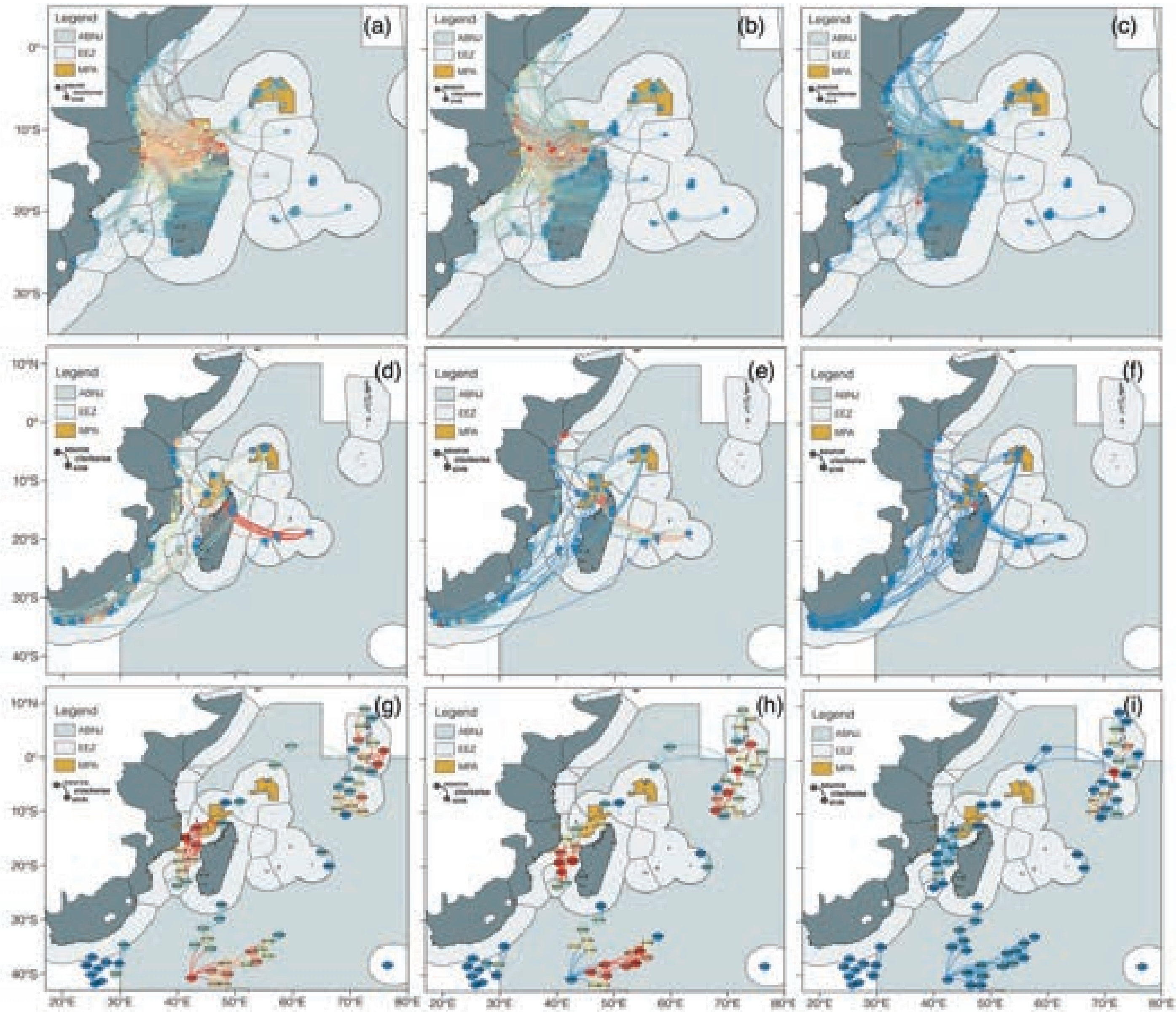


Figure 5: An illustration of larvae sources (panels a,d,g in the left column), sinks (panels b,e,h in the middle column), and corridors (panels c,f,i in the right column) within coral reefs (top row, panels a,b,c), MPAs (middle row, panels d,e,f) and seamounts (bottom row, panels g,h,i).

Similarly, larvae release from the seamounts in ABNJ settled in Mauritius, Seychelles and Somalia EEZ (Fig. 3C). Overall, 55 per cent of larvae released from ABNJ settled within the EEZ, with the greatest portion (10 per cent) settling in Madagascar, 7.3 per cent in Mozambique, 7.20 per cent in Seychelles, 5.45 per cent in South Africa and 4.86 per cent in Reunion (Fig. 4).

Priority area selections

The Marxan scenario sought to protect 10 per cent of seafloor geomorphic habitats while maintaining connections between and among coral reefs, seamounts and the existing MPAs (100 per cent target). Within the EEZ, a mix of off-shore and coastal areas selected include regions around the existing MPAs of *Amirantes to Fortune Bank in Seychelles* (Fig. 6). New areas were also selected in Comoros and Gloriosso Islands, in Somali EEZ, off-shore eastern Madagascar, Europa, Bassas da India, Mauritius and Reunion.

The ABNJ selected were off the Mauritius EEZ to the east and south. The northern part of the WIO ABNJ was not selected due to the high fishing effort in this zone; given that fishing effort was used at the cost in the prioritization analyses these areas were the least priority for Marxan (see methods) (Fig. 6).

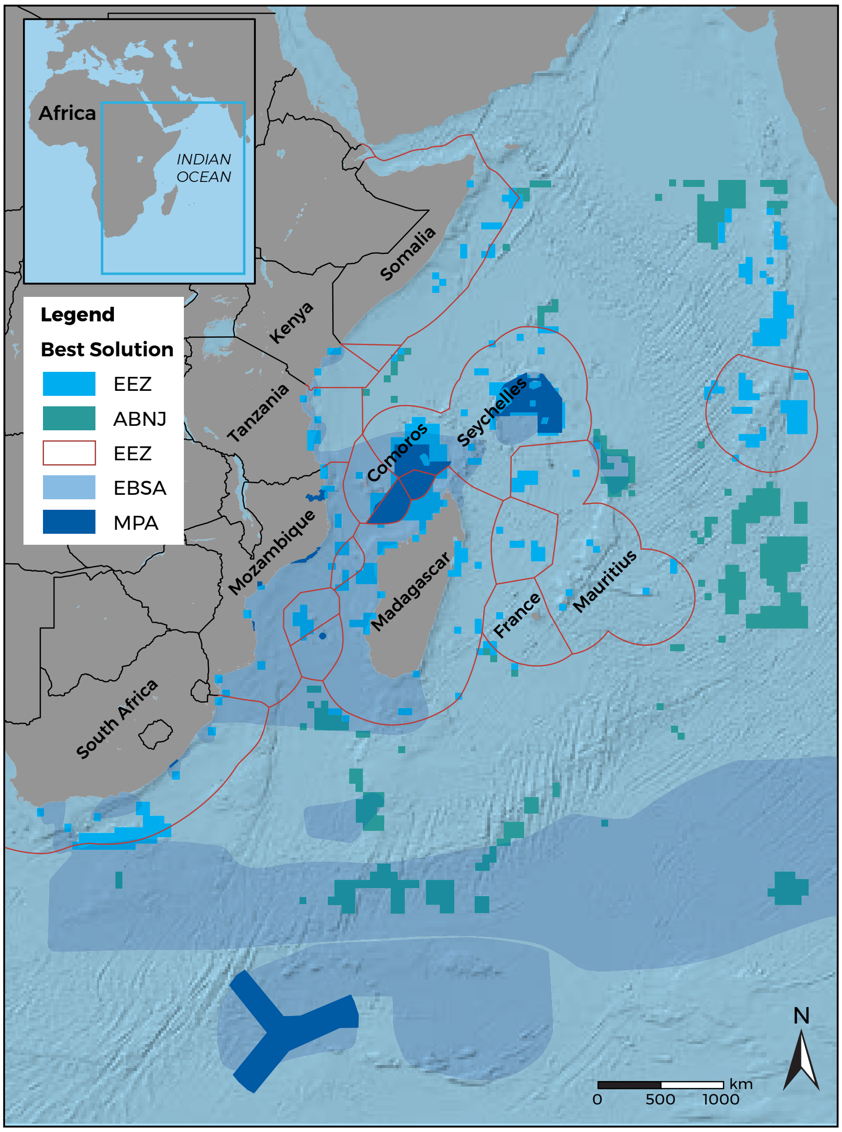


Figure 6: The best solution of priority area selection under the Marxan scenario.

Approximately 9.5 per cent of the total area was selected within the EEZ, while 1.8 per cent of the total area was from the ABNJ (Fig. 6), significantly lower than overall EEZ selection, but relatively higher than the individual country EEZ selections except Seychelles. Of the EEZ selections, rather large areas were selected from within the Seychelles EEZ (3.2 per cent) (Fig. 7). All other EEZs were <1 per cent of the total area.

DISCUSSION

Ocean connectivity is critical for the persistence of marine life and the vast benefits that accrue from them. Understanding broad-scale connectivity is crucial for managing the oceans, both within and outside areas of national jurisdictions. In this study, regional-scale connectivity among key habitats and maritime zones and MPAs in the WIO region was analyzed. As countries negotiate on the global platform between mechanisms for managing the high seas, a case study from the WIO on applying functional connectivity to a regional spatial prioritization process across maritime boundaries is presented. Three goals to maximize conservation outcomes guided the identification of areas suitable for inclusion in the MPA network: (i) representative area (10 per cent) of seafloor geomorphic habitats, (ii) protect coral reef and seamounts that enhance and maintain connectivity across maritime jurisdiction, and (iii) reduce human pressure on ecosystems. Objective setting using seafloor geomorphic habitats, which are distributed in both EEZ and the high seas, and using marine functional connectivity, provides an opportunity to prioritize areas of ecological and economic significance for conservation across maritime jurisdictions.

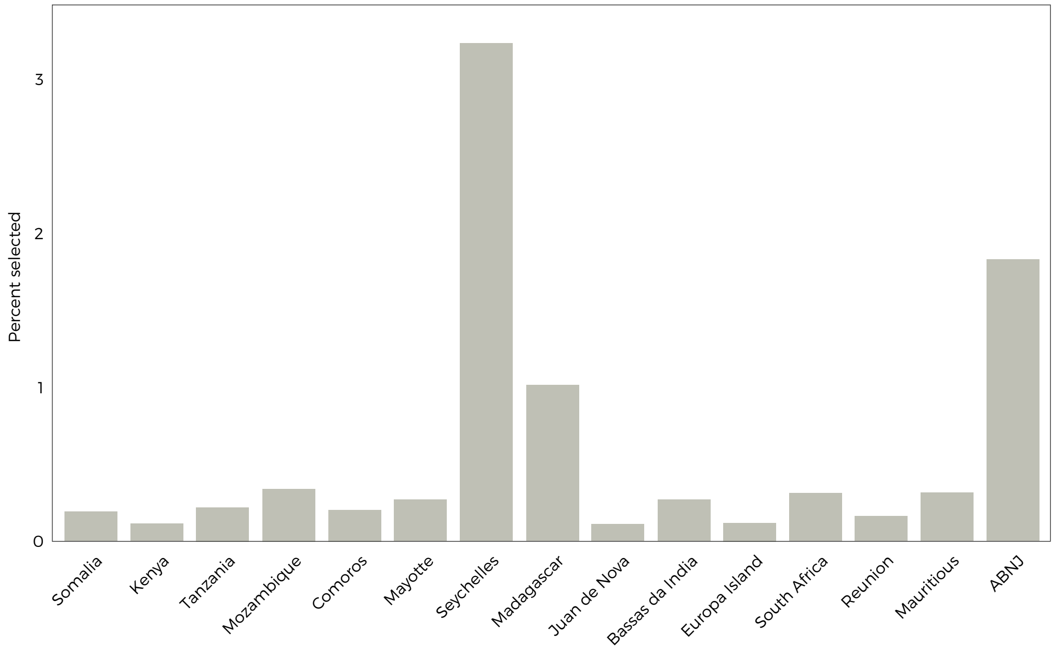


Figure 7: The proportion of area selected for protection within EEZ and ABNJ based on the Marxan scenario.

Connectivity between ABNJ and EEZ

The maritime boundaries between the EEZ and the ABNJ does not preclude a strong relationship between the High Seas and coastal states in practice. Many of these relationships have large economic value (Sala et al., 2018). Between 1970 and 2000, industrial marine fisheries catch in ABNJ increased by 10 per cent (Pauly et al., 2002). In spatial terms, the greatest expansion of fishing effort took place primarily beyond the continental shelf limits and in ABNJ. With evidence of pelagic species and larvae moving across the ocean from ABNJ into EEZ (Fig. 3B), the destruction of habitats in this area impacts the adjoining EEZs. The dispersal model suggests that island states of the WIO are more connected to the ABNJ than continental countries, with EEZs of Mauritius, Seychelles, Reunion and Madagascar being the destination for most of the larvae emanating from ABNJ. Among the continental countries, more larvae settle within Somalia EEZ than in any other continental country.

Aligning conservation areas to regional connectivity patterns

The current arrangement of 120 MPAs, the majority of which are on the western boundary of the WIO, means these sites are moderately connected, with connectance high along the east-west direction and following the major ocean currents. Most of these MPAs were established to protect biodiversity on the biodiversity hotspots in the region, which was underpinned by the high connectivity. Opportunities exist for looking at other highly diverse areas and could serve as biodiversity hotspots in the future. Of the 243 reef locations, 103 are located within MPAs and do not include the most connected reefs. In effect, highly connected reefs, which could serve as stepping-stones or supporting seeding of other coral reefs, are not protected.

The opportunity cost data greatly influenced priority selections. This is evident in the ABNJ, where fewer areas were selected. One of the Marxan objectives was to select locations that are least fished within ABNJ (ie to minimize costs) while meeting connectivity and seafloor habitat targets. While the objective to reduce cost associated with loss of fishing ground may not select the most productive or frequented areas, this scenario is realistic as it promotes consensus by preventing loss of fishing ground which is one of the issues that complicates country negotiations (Smith and Jabour, 2017). However, this may need to be balanced with ecological interests, where for instance, thresholds of effort are set such that the algorithm prioritizes both extremely fished and least fished areas.

Influence of oceanography on connectivity across ecosystems and maritime boundaries

These results elaborate how oceanic processes play an important role in larval dispersal and connectivity among populations. The westward flowing SEC carries waters from the Indonesian region across the Indian Ocean between 10–2°S (Schott et al., 2009) (Fig. 1). This zonal flow creates a physical and functional connectivity barrier to dispersal between Seychelles and Mascarene islands. On the east coast of Madagascar, the SEC accelerates past the tip of Madagascar as the NEMC while facilitating larval dispersal from the north-east tip of Madagascar into Comoros and further along the East African coast. These eddies have important implications for connectivity as they entrap larvae released within the Comoros Basin. The NEMC splits into the northward-flowing EACC and southwards as eddies in the Mozambique Channel on reaching the East Africa mainland coast. The NEMC creates a barrier between the reefs north and further south in the Mozambique Channel.

Along the East African coast (Tanzania, Kenya and Somalia), the dominant connectivity pattern is south to north connectivity for coral reefs. This is due to the constant northward flow of the EACC. It is also worth noting north to south (Somalia to Kenya) connections primarily for reefs found in the northern banks of Kenya because the north region is seasonally influenced by the reversal of the Somali Current (from northward-flowing current in south-west monsoon to southward flowing during north-east monsoon). Therefore, the strength of north to south connections depends on the strength of the reversing Somali Current. Further south, the north-west coast of Madagascar and Mozambique coast show a high level of connectivity to the high seas, and spatially explicit considerations for maintaining or restoring habitat diversity and connectivity across maritime jurisdictions.

Management and policy recommendations

While this work is a preliminary exploration of regional scale connectivity patterns in the WIO, the findings demonstrate the potential of using oceanographic modelling to estimate functional connectivity among zones of maritime jurisdictions. These assessments indicate well-connected marine areas and habitats, potentially impacting livelihoods, ecosystems, and economies. Maintaining functional connectivity in the WIO, and the well-being of ocean ecosystems across all maritime jurisdictions, including the high seas, as well as their ability to provide ecological functions and essential ecosystem services for human populations, is a challenge because of the current assortment of complex and unco ordinated regulations govering the use of coastal and the high seas (Dunn et al., 2014; Houghton, 2014). A sustainable future of marine areas in the WIO hinges on the formultion and implementation of a comprehensive governance framework that moves away from a within the country, sector-by-sector management approach to one that (i) incorporates appropriate ecological, socio-economic and geo-political perspectives across national and maritime boundaries; and (ii) supports management that is coordinated at the scale of ecosystems as well as political and maritime jurisdictions (Haque, 2015). These goals demand increased efforts to facilitate governing the high seas and spatially explicit considerations for maintaining or restoring habitat diversity and connectivity across maritime jurisdictions. Consequently, regional institutions should explore options for ocean governance and conservation of marine biodiversity in adjacent ABNJ.

The use of area-based tools, including marine spatial planning as demonstrated here, is a practical approach for contributing to protecting the high seas, where non-spatial monitoring is complex and where data gaps obstruct conventional management approaches. In adopting an evidence-based approach to protecting the high seas, research on migratory patterns of critical species and biological processes in the high seas should be promoted. A connectivity study focusing on coastal areas and spatially explicit linkages with ABNJ may help formulate possible decisions on offsetting mechanisms where activities in the high seas are linked to impacts on coastal areas.

Furthermore, studies on the feasibility, options, and scenarios for establishing MPAs in ABNJ, in consultation with the countries involved, are necessary. This may involve partnerships with the International Maritime Organization and UNCLOS to facilitate identifying and designating as particularly sensitive sea areas (PSSAs) which are of significance in terms of ecological, social, economic or scientific criteria and are vulnerable to damage by international shipping activities. Implementation of governance on the high seas may have to rely on effective satellite surveillance of fisheries activities on the open ocean. The International Maritime Organization (and Interpol) is already using vessel-monitoring technology to track ship movements and suspicious activity.

REFERENCES

Álvarez-Romero, J.G., Munguía-Vega, A., Beger, M., del Mar Mancha-Cisneros, M., Suárez-Castillo, A.N., Gurney, G.G., Pressey, R.L., Gerber, L.R., Morzaria-Luna, H.N., Reyes-Bonilla, H. and Adams, V.M. (2018). Designing connected marine reserves in the face of global warming. *Glob. Chang. Biol.* 24(2), pp. e671–e691

Andrello, M., Guilhaumon, F., Albouy, C., Parravicini, V., Scholtens, J., Verley, P., Barange, M., Sumaila, U.R., Manel, S. and Mouillot, D. (2017). Global mismatch between fishing dependency and larval supply from marine reserves. *Nat. Commun*. 8, p. 16039

Ardron, J., Gjerde, K., Pullen, S. and Tilot, V. (2008). Marine spatial planning in the high seas. *Mar. Policy* 32(5), pp. 832–839

Calich, H., Estevanez, M. and Hammerschlag, N. (2018). Overlap between highly suitable habitats and longline gear management areas reveals vulnerable and protected regions for highly migratory sharks. *Mar. Ecol. Prog. Ser.* 602, pp. 183–195

Campana, S.E. (2016). Transboundary movements, unmonitored fishing mortality, and ineffective international fisheries management pose risks for pelagic sharks in the Northwest Atlantic. *Can. J. Fish. Aquat. Sci.* 73(10), pp. 1599–1607

Cinner, J.E., Huchery, C., MacNeil, M.A., Graham, N.A., McClanahan, T.R., Maina, J., Maire, E., Kittinger, J.N., Hicks, C.C., Mora, C. and Allison, E.H. (2016). Bright spots among the world's coral reefs. *Nature* 535(7612), p. 416

Cinner, J.E., Maire, E., Huchery, C., MacNeil, M.A., Graham, N.A., Mora, C., McClanahan, T.R., Barnes, M.L., Kittinger, J.N., Hicks, C.C. and D'Agata, S. (2018). Gravity of human impacts mediates coral reef conservation gains. *Proc. Natl. Acad. Sci.* 115(27), pp. e6116–e6125

Cowen, R.K., Gawarkiewicz, G., Pineda, J., Thorrold, S.R. and Werner, F.E. (2007). Population connectivity in marine systems an overview. *Oceanography* 20(3), pp. 14–21

Dulvy, N.K., Fowler, S.L., Musick, J.A., Cavanagh, R.D., Kyne, P.M., Harrison, L.R., Carlson, J.K., Davidson, L.N.K., Fordham, S.V., Francis, M.P., Pollock, C.M., Simpfendorfer, C.A., Burgess, G.H., Carpenter, K.E., Compagno, L.J.V., Ebert, D.A., Gibson, C., Heupel, M.R., Livingstone, S.R., Sanciangco, J.C., Stevens, J.D., Valenti, S. and White, W.T. (2014). Extinction risk and conservation of the world's sharks and rays. *eLife* 3, p. e00590

Dunn, D.C., Ardron, J., Bax, N., Bernal, P., Cleary, J., Cresswell, I., Donnelly, B., Dunstan, P., Gjerde, K., Johnson, D., Kaschner, K., Lascelles, B., Rice, J., von Nordheim, Wood, L. and Halpin, P.N. (2014). The convention on biological diversity's ecologically or biologically significant areas: origins, development, and current status. *Mar. Policy* 49, pp. 137–145

Ferry, N., Parent, L., Garric, G., Bricaud, C., Testut, C.E., Le Galloudec, O., Lellouche, J.M., Drevillon, M., Greiner, E., Barnier, B. and Molines, J.M. (2012). GLORYS2V1 global ocean reanalysis of the altimetric era (1992–2009) at meso scale. *Mercator Ocean – Quaterly Newsletter* 44

Haque, A.E. (2015). Is maritime spatial planning a necessity. *Environ. Policy Law* 45(5), p. 212

Harris, P.T., Macmillan-Lawler, M., Rupp, J. and Baker, E.K. (2014). Geomorphology of the oceans. *Mar. Geol.* 352, pp. 4–24

Heffernan, O. (2018). How to save the High Seas Nature. *Nature* 557, pp. 154–156. https://doi.org/10.1038/d41586-018-05079-z

Houghton, K. (2014). Identifying new pathways for ocean governance: The role of legal principles in areas beyond national jurisdiction. *Mar. Policy* 49, pp. 118–126

Kroodsma, D.A., Mayorga, J., Hochberg, T., Miller, A.N., Boerder, K., Ferretti, F., Wilson, A., Bergman, B., White, T.D., Block, B.A., Woods, P., Sullivan, B., Costello, C. and Worm, B. (2018). Tracking the global footprint of fisheries. *Science* 359(6378), pp. 904–908

Lauck, T., Clark, C.W., Mangel, M. and Munro, G.R. (1998). Implementing the precautionary principle in fisheries management through marine reserves. *Ecol. Appl.* 8(sp1), pp. S72–S78

Lett, C., Verley, P., Mullon, C., Parada, C., Brochier, T., Penven, P. and Blanke, B. (2008). A Lagrangian tool for modelling ichthyoplankton dynamics. *Environ. Model. Softw*. 23(9), pp. 1210–1214

Luiz, O.J., Allen, A.P., Robertson, D.R., Floeter, S.R., Kulbicki, M., Vigliola, L., Becheler, R. and Madin, J.S. (2013). Adult and larval traits as determinants of geographic range size among tropical reef fishes. *Proc. Natl. Acad. Sci.* 110(41), pp. 16498–16502

McCauley, D.J., Pinsky, M.L., Palumbi, S.R., Estes, J.A., Joyce, F.H. and Warner, R.R. (2015). Marine defaunation: animal loss in the global ocean. *Science* 347(6219), p. 1255641

Minor, E.S. and Urban, D.L. (2008). A graph-theory framework for evaluating landscape connectivity and conservation planning. *Conserv. biol.* 22(2), pp. 297–307

Murawski, S.A. (2010). Rebuilding depleted fish stocks: the good, the bad, and, mostly, the ugly. *ICES J. Mar. Sci.* 67(9), pp. 1830–1840

Myers, R. A. and Worm, B. (2003). Rapid worldwide depletion of predatory fish communities. *Nature* 423, pp. 280–283

Pauly, D., Christensen, V., Guénette, S., Pitcher, T.J., Sumaila, U.R., Walters, C.J., Watson, R. and Zeller, D. (2002). Towards sustainability in world fisheries. *Nature* 418, pp. 689–695

Peliz, A., Marchesiello, P., Dubert, J., Marta-Almeida, M., Roy, C. and Queiroga, H. (2007). A study of crab larvae dispersal on the Western Iberian Shelf: Physical processes. *Journal of Marine Systems* 68(1-2), pp. 215–236

Roberts, C. (2012). Marine ecology: reserves do have a key role in fisheries. *Curr. Biol.* 22(11), pp. R444–R446

Roberts, C. M., O'Leary, B. C., McCauley, D. J., Cury, P. M., Duarte, C. M., Lubchenco, J., Pauly, D., Sáenz-Arroyo, A., Sumaila, R., Wilson, R. and Worm, B. (2017). Marine reserves can mitigate and promote adaptation to climate change. *Proc. Natl. Acad. Sci.* 114(24), pp. 6167–6775

Sala, E., Mayorga, J., Costello, C., Kroodsma, D., Palomares, M.L., Pauly, D., Sumaila, U.R. and Zeller, D. (2018). The economics of fishing the high seas. *Sci. Adv.* 4(6), p. eaat2504

Schott, F.A., and McCreary Jr, J.P. (2001). The monsoon circulation of the Indian Ocean. *Prog. Oceanogr*. 51(1), pp. 1–123

Schott, F.A., Xie, S.P. and McCreary, J.P. (2009). Indian Ocean circulation and climate variability. *Rev. Geophys*. 47(1), RG1002

Scovazzi, T. (2004). Marine protected areas on the high seas: some legal and policy considerations. *Int. J. Mar. Coast. Law* 19(1), pp. 1–17

Smith, D. and Jabour, J. (2017). MPAs in ABNJ: lessons from two high seas regimes. *ICES J. Mar. Sci.* 75(1), pp. 417–425

Stewart, R.R. and Possingham, H.P. (2005). Efficiency, costs and trade-offs in marine reserve system design. *Environ. Model. Assess.* 10(3), pp. 203–213

UNEP/Nairobi Convention Secretariat, WIOMSA (2009). Strategic Action Programme for the Protection of the Coastal and Marine Environment of the Western Indian Ocean from Land-Based Sources and Activities, 156 p. UNEP, Nairobi, Kenya

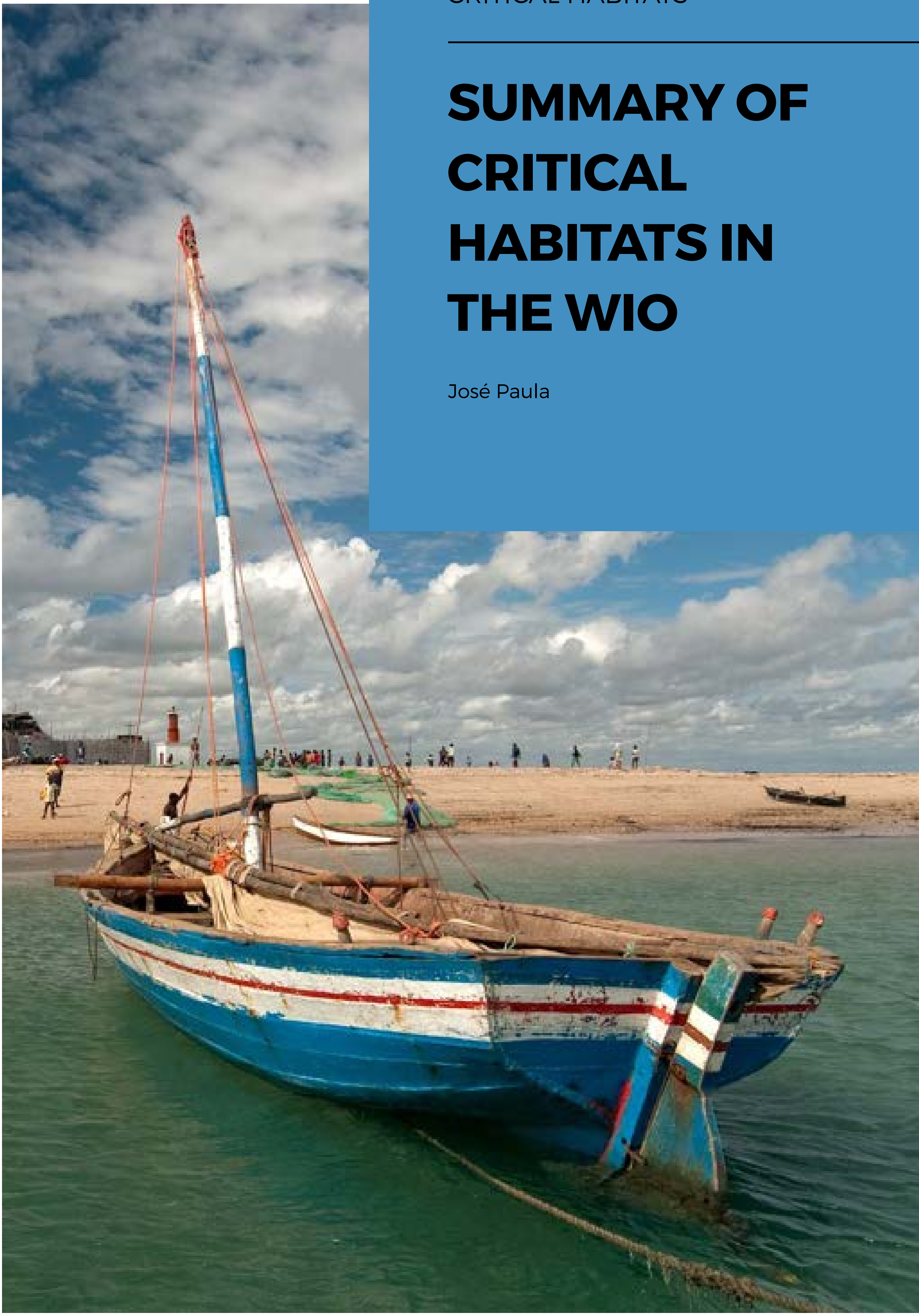
USA (2007). A Global Database on Marine Fisheries and Ecosystems. Fisheries Centre, University British Columbia. Vancouver, Canada

Watts, M.E., Ball, I.R., Stewart, R.S., Klein, C.J., Wilson, K., Steinback, C., Lourival, R., Kircher, L. and Possingham, H.P. (2009). Marxan with Zones: software for optimal conservation based land-and sea-use zoning. *Environ. Model. Softw.* 24(12), pp. 1513–1521

CRITICAL HABITATS

SUMMARY OF CRITICAL HABITATS IN THE WIO

José Paula



Fishing boat on Pemba Bay shore, North Mozambique. © José Paula

INTRODUCTION

The Western Indian Ocean (WIO) region encompasses tropical and subtropical regions of diverse nature, rich stretches of coast along the mainland countries of Somalia, Kenya, Tanzania, Mozambique and South Africa, and vast oceanic areas surrounding the island states of Madagascar, Seychelles, Comoros, Mauritius and the French Territories. The complexity and wide geographical span of the WIO region creates environmental gradients and contrasts, and these provide the basis for compartmentalization and regionalization based on different criteria and classification schemes (see Chapter 2).

The north of the WIO is strongly influenced by the monsoon regime of the Arabian Sea which pulses seasonally and triggers coastal upwelling and associated biological productivity. In the central WIO, the main equatorial current meets the African continent and splits into two major currents along the continental coastal waters to the north and south. In the Mozambican Channel the current moves southward through complex systems of gyres that meet the Agulhas Current and transports energy to higher latitudes in the southern hemisphere. The vastness of the WIO and its complex oceanographic dynamic (see Chapter 3) create a biophysical mosaic of coastal and offshore environments that spread from temperate to tropical habitats of diverse nature.

Threats to the environment in the WIO can be broadly categorized as those which are natural, for example episodic events (cyclones, tsunamis, floods) and anthropogenic or human in cause, for example exploitation (direct and indirect), habitat destruction (land 'reclamation', urbanization, dredging, mining and oil/gas extraction), pollution (point and diffuse sources) and climate change (including ocean acidification and sea level rise).

This chapter summarizes the information in the previous chapters of Part III, which describe and analyze habitats and special taxa of the WIO. A general overview of threats and general recommendations for protection is made.

CRITICAL HABITATS AND SENSITIVE TAXA

Rocky and sandy coasts

Status and importance

Rocky outcrops (both intertidal rocky shores and subtidal rocky reefs) and sedimentary formations create a diversity of coastal configurations along the coastlines of the WIO countries (see Chapter 6). Despite their spatial extent and area coverage, scientific information is scant. Sedimentary formations such as mud flats, sand shores and dunes are dynamic habitats and also highly vulnerable to both natural and anthropogenic drivers. Ecologically, these highly variable habitats are important areas on the coast-sea interface by providing a multitude of microhabitats and niches for organisms, including breeding and nursery areas for many species. They also serve as important feeding and foraging grounds for both terrestrial and marine animals. Due to their accessibility, rocky outcrops and sedimentary coastal resources are intensively used as a source for coastal livelihoods in the WIO, providing a major income for artisanal subsistence and food security in the region.

Threats

Several phenomena and activities threaten nearshore habitats in the WIO region, affecting their ecological productivity, integrity and by extension, livelihoods and economies. Although many of these habitats have the capacity to adapt to high levels of natural environmental stress, such ability is threatened by various human-related activities. Among the threats are the over-exploitation of resources and disturbance to habitats, pollution, coastal development and urbanization, as well as global phenomena related to climate induced changes.

Recommendations

There are very limited exclusive protection measures for subtidal rocky reef and sandy shore habitats in the region. However, some intertidal and nearshore habitats tend to fall within the coverage of the existing area-based protection mechanisms such as Marine Protected Areas (MPAs) and Locally Managed Marine Areas (LMMAs). Estimation of cover area of these nearshore habitats is necessary, as well as increasing their protection by incorporating additional areas into the existing MPAs and LMMAs within each national jurisdiction.

Mangrove forests

Status and importance

Mangrove forests are widespread in the WIO, with particular importance in the mainland countries and Madagascar. The WIO is ranked second in the world for mangrove areas after South-East Asia (see Chapter 7). The ecological importance of WIO mangroves extends from coastal protection to biodiversity maintenance, from mitigation of, to adaptation to climate-induced changes.Mangrove forests sustain extensive fisheries in addition to being directly used, mainly as building material and firewood.

Mangroves can store between 3–5 times the amount of carbon accumulated by other terrestrial vegetation systems, and sustain tangible livelihoods, including eco-tourism, while supporting some of the largest fisheries in the region such as Sofala Bank (central Mozambique) with high fisheries production related to the coastal mangrove forests nursery function. Island mangroves also support biodiversity, provide shoreline protection and water quality control, among other ecosystem services.

Threats

Anthropogenic threats to mangroves include habitat destruction for land reclamation and over-exploitation of their resources, in particular wood. Global phenomena also impact mangrove forests and contribute to their degradation, such as sea-level rise and extreme events like storm surges and floods.

Recommendations

National agendas on mangroves should be revisited so that they are mainstreamed with global platforms such as the main targets of the SDGs. Some information gaps still need to be addressed, such as mapping forests and vulnerable areas, assessing threats at local scales including strengthening land-use to ensure mangroves have space to develop inland as response to sea level rise. Integrating the wider society, both at local and country level, will help improve in steering the discussion on tackling the wider mangrove management challenges in the WIO. In view of local degradation and deforestation rates the WIO countries must also strategize the implementation of mangrove restoration programs involving local communities.

Seagrass beds

Status and importance

Seagrass meadows are distributed along the coastlines of the WIO mainland and the Island States. In most countries of the region, seagrass beds often occur in close connection with coral reefs and mangroves. Seagrasses form key components of marine ecosystems; however, they have received limited scientific attention compared to mangroves and coral reefs. Comprehensive mapping of seagrass beds has not been achieved yet for most countries in the region, and hence total seagrass coverage in the WIO region is not fully understood (see Chapter 8).

Seagrasses are one of the most productive aquatic ecosystems in the world. They possess a complex habitat structure and are used by a myriad of organisms as shelter against predation, foraging and nursery areas. Although a few animals can feed directly on seagrasses (eg many fish species, Dugongs or green turtles), the significant fraction of seagrass biomass enters the marine food web through detritus, thereby nourishing adjacent habitats and supporting productivity through recycling of nutrients and carbon. Seagrasses also stabilize sediment, thereby reducing coastal erosion and strengthening coastal protection. Seagrasses also provide many important ecosystem services through support to fisheries and tourism industries, reliant on the ability of healthy seagrass beds to support finfish, shellfish and other fishery related products.

Threats

Most threats to seagrasses are a result of human activities, though natural causes can also account for seagrass loss in the region. Habitat destruction resulting from fishing activities, particularly the use of beach seines and trawls by artisanal fishers over seagrass beds is a widespread threat. Another threat that is related to fishing activities is collection of invertebrates (gleaning) in the intertidal area that often involves digging and revolving large amounts of sediments as well as trampling over seagrasses. Other important threats to seagrasses are eutrophication as a result of excessive nutrient input into coastal waters, sedimentation originating from various sources, and physical destruction related to water-based leisure activities.

Recommendations

Information regarding the status of seagrass beds within the WIO is largely lacking, however, considering that the threats are continuing, then it is logical to generalize that seagrass beds in the WIO are following a declining trend. Although seagrass meadows are threatened, some degree of protection for this important ecosystem exists in the region, as for rocky shores and rocky reefs, mainly through their inclusion in marine protected areas (MPAs) and locally managed marine areas (LMMAs). Despite their importance, there is however inadequate protection of seagrass habitats in the WIO region, and hence there is a need to identify priority areas for conservation as well as opportunities that can be used to enhance seagrass protection. Mechanisms should be put in place at the regional level to ensure regional collaborations and joint actions for the conservation of seagrass ecosystems, including restoration programs.

Salt marshes

Status and importance

Salt marshes are typically temperate coastal habitats, and in the WIO occur mainly on temperate South African shores. However, the same type of habitat exists throughout the region on the upper edge of mangrove forests near hypersaline flats close to the terrestrial vegetation (see Chapter 9). Salt marshes are productive ecosystems important for carbon storage, water purification, flood control, refugia, and habitat for other organisms such as juvenile fish, and also serve as critical habitats for migratory fish and birds. Salt marsh plants are also increasingly used for human consumption.

Threats
Threats to salt marshes include sea level rise at the seaward interface, and development at the land interface. The latter include land reclamation for agriculture, seawater evaporation ponds for salt production, shellfish or fish farming ponds or livestock production, that restrict tidal exchange and promote the establishment of invasive species.

Recommendations
There is a degree of protection in South Africa of the larger salt marshes, and some degree in the legislation of other WIO countries, but overall, there is a need for better attention and research to fill gaps of knowledge regarding the distribution and condition of salt marshes in WIO countries.

Coral reefs

Status and importance
Coral reefs fringe most shorelines in the WIO (see Chapter 10), supporting a wide range of goods and ecosystem services, and generating many benefits for local and national economies. These include the provision of seafood and other resources that are important for the livelihoods of coastal communities. Coral reefs also provide regulatory services such as climate change regulation, beach replenishment and coastal protection. Coral reefs further support important revenues in tourism, fisheries and trade. Coral reefs are connected to and interact with adjacent coastal and marine ecosystems such as mangroves and seagrass beds that contribute to the integrated seascape ecological functioning. The information on coral reefs in the region has expanded since the 1998 massive bleaching event and the generated knowledge provides a basis for the development of conservation policies and integrated management.

Threats
WIO coral reefs are being threatened by multiple factors, of which the three main forces are climate associated disturbances, fishing, and the interrelated factors of nutrient pollution and sedimentation caused by human influences on land. The threat intensity is patchy in space and time. Coral reefs can therefore experience one, all three, or all possible combinations of these degrading forces.

Recommendations
MPAs are the most implemented area-based tools in the WIO for coral reefs. In many cases however, the design of MPAs did not consider marine zoning considerations such as representativeness (ecological and biodiversity), adequacy (size), and irreplaceability. The second most common area-based approach is the co-management approach, a decentralized management model focusing on fisheries. Although countries of the WIO have invested in many programs and initiatives to protect and manage coral reefs, more concerted effort is urgently needed because coral reefs are in imminent danger due to climate change disturbances, fishing and the drive for coastal development and the Blue Economy.

Estuaries

Status and importance
Estuaries are the transitional aquatic systems between the freshwater and marine environments, and are among the most productive natural systems of the world. These systems export sediments, nutrients and organic matter to the continental shelf enhancing coastal productivity. They often form complex ecosystems that include critical habitats such as mangroves, seagrass beds, salt marshes and extensive tidal flats. Due to their characteristics, estuaries have historically attracted the settlement of human communities, creating socio-ecological systems that have developed into most of the world's largest coastal cities.

Threats
Multiple stressors threaten the natural balance of WIO estuaries (see Chapter 11). Sea-level rise impacts low-lying estuarine land, and floods from extreme events induce erosion and mangrove destruction. Further human induced alterations at catchment scales, such as damming and water abstraction as well as intensive agriculture and alterations of vegetation cover, put pressure on the natural ecological balance. Widespread pollution and habitat destruction through land reclamation contribute to the degradation of estuaries and the natural habitats and resources they contain.

Recommendations
Protection for WIO estuaries is provided by international agreements on shared watersheds, and further promoted by wetland conventions. Conservation of estuaries is complex, because they not only include the activities with the estuarine system itself, but also the upstream land-use activities. Thus, there is a need to integrate management of the catchment. Another issue is that estuaries are very diverse in terms of hydrological and ecological regimes, further impacted by diverse anthro-

pogenic stressors, leading to the need for individually based management and action plans.

Shelf, deep sea and offshore pelagic

Status and importance

The vast majority of the sea in the WIO is offshore areas and the deep-sea. The offshore habitats, and in particular the deep seabed, are largely unknown (see Chapter 12). It may be expected, given their vast spatial extent, that offshore habitats will add considerably to species counts as explorations expand into these areas. This particularly applies to benthic fauna from shelf sediments and deeper seabed habitats which have been consistently under-sampled. Deep water habitats are known to be more stable and usually their biota distributes throughout larger areas when compared to that of shallow water.

Threats

The threats to the vast offshore areas and the deep sea can be broadly grouped into three categories – extraction of resources (renewable and non-renewable), contamination and pollution (some of which are directly associated with resource extraction), and climate change. Furthermore, several drivers exacerbate the threats, including: unsuitable governance, economic factors, insufficient financial resources, a lack of knowledge and population growth.

Harvesting of renewable resources such as deep sea fisheries resources modifies habitats and causes disruptions to ecosystem functioning. Exploration for and extraction of non-renewable resources both pose threats, including sound generated by seismic and sonar equipment, disruption of sediments leading to increased turbidity and modification or loss of habitats, and contamination and destruction of biota. Shipping traffic in the region is also related to the regional economy and extraction of resources, with associated increased pollution, ship strikes on cetaceans, and invasive species from ballast water and fouling.

Recommendations

There is a need for protection for offshore habitats in the WIO as they are currently poorly protected. Due to their vastness, there is a need to prioritize areas within these habitats, but the majority remain underexplored, and information is lacking. There are mechanisms in place for declaration of protected areas within state EEZs, but there is a need for a process to declare international MPAs, and there is also need for effective management of existing protected areas in offshore habitats in the WIO.

Threatened species

Status and importance

Over the last seven years, the number of marine species listed in the IUCN Red List that occur in the WIO increased from 161 to 231 (see Chapter 13). Conservation of threatened species implies conservation of their primary habitats, and among these seagrass meadows and coral reefs are of major importance. Among threatened species there are a variety of taxonomic groups from sea cucumbers, gastropod molluscs, and fish (including iconic species such as the Coelacanth), to sea turtles and marine mammals.

Threats

Threats to specific taxa depend on the species and its biology, ecology and distribution, but most ecosystems and species are prone to the impacts of global threats, derived from climate change, pollution and widespread environmental degradation. Other threats include overfishing for consumption and the ornamental species trade.

Recommendations

Appropriate management through integrated coastal zone management (ICZM) provides the best framework to protect vulnerable key threatened critical habitats, such as seagrasses and corals, and through these protect many other species that depend on the habitats, such as endangered fish, marine turtles and mammals. Other specific measures have to be adapted to individual taxa/ species and their conservation requirements, and MPAs and community managed areas are among the protection measures currently utilized in the WIO.

Marine and coastal birds

Status and importance

The WIO region supports a high diversity of seabird and coastal birds, found in all habitats, including several endemic and near-endemic species (see Chapter 14). Seabirds are useful indicators for identifying priority sites for conservation and their distributions can provide surrogates for biodiversity hotspots in marine spatial planning. Important Bird and Biodiversity Areas (IBAs) are areas of global significance for birds - a range of IBAs have been identified within the WIO, including seabird IBAs (terrestrial colony locations) and marine IBAs (those entirely within the marine environment), with the latter identified via feeding extensions to seabird colony locations. Seabird populations in the WIO are thought to be a fraction of the historical estimates. Many colonies have become extinct and those that still exist are greatly reduced in size.

Threats
Seabirds face threats when breeding (nesting) on land, including predation by invasive species such as rats and cats, harvesting and human disturbance, and when feeding at sea, threatened by fisheries activities, both through depletion of food sources, and mortality from fishing gear where are often captured as bycatch. In the WIO this is particularly the case south of 25°S. The threat from pollution is largely related to oil spills.

Recommendations
Conservation actions for seabirds will depend on the context, including primary threats and species susceptibility. However, general conservation actions that are required include: conserving a network of sites (IBAs) across the WIO that are important for birds, the removal of predatory, alien species from areas used for seabird breeding, feeding and/or aggregation, control of unsustainable seabird harvest, integrating bird conservation into integrated coastal zone management (ICZM), reduction of bycatch, and maintenance of long-term monitoring.

Seamounts and ridges

Status and importance
Seamounts and ridges are recognized as significant habitats for a wide diversity of species and considered hotspots of biodiversity, attracting a range of oceanic predators, including seabirds, whales and sharks (see Chapter 15). Many basic aspects of their biodiversity are still unknown, in particular in the south-Western Indian Ocean when compared to other regions. As biodiversity hotspots, seamounts have high endemism relative to other habitats. Overall, the seamount ecosystem can host abundant and diverse benthic and pelagic communities which are the target of significant fisheries, but also have non-renewable resources that can be exploited via deep-sea mining.

Threats
Seamounts and ridges are potentially impacted mainly by non-sustainable fisheries and seabed mining, especially considering that many of these habitats are located in international waters. On the other hand, the generalized lack of information regarding these systems creates an enormous difficulty in assessing threats and specific protection measures.

Recommendations
The knowledge of seamount, ridges and hydrothermal vent distribution and associated communities remains poor. Previous surveys mainly focused on a few geographic areas and little data exist for seamounts in regions such as the WIO. Thus, there is an urgent need to explore and survey these ecosystems to complete the picture of the biodiversity and productivity associated with the Indian Ocean. On the other hand, efforts should be made to extend the geographical coverage of regional areas beyond national jurisdiction (ABNJs) and MPAs beyond the regional national jurisdictions. Where relevant, promoting the establishment of Ecologically or Biologically Significant Marine Areas (EBSAs) may contribute to develop the conservation momentum for such sites.

Small islands and atolls

Status and importance
The small islands of the WIO are captured under a full suite of country designations and vary in size from relatively large landmasses to small, isolated coral atolls widely scattered across large ocean spaces (see Chapter 16). Together they have been identified as one of the world's biodiversity hotspots.

Threats
As countries within the WIO intensify their efforts to achieve sustainable ocean economies, this places an increasing burden on the diverse ecosystems and biodiversity of the region's islands and atolls. Mounting resource utilization, habitat degradation, tourism and development, alien invasive species, pollution and climate change all impact negatively on these already fragile systems.

Recommendations
Formal protection has already been afforded to some of the islands and two sites have been listed under UNESCO World Heritage status. However, far more conservation effort is needed to ensure the preservation of these biodiversity hotspots through additional proclamation of MPAs and through ensuring that those currently under formal protection are effectively managed.

Coastal forests

Status and importance
The WIO coastal forests comprise small and fragmented patches, which are host to high biological diversity of global significance (see Chapter 17). They provide a diversity of ecosystem services that are directly and indirectly linked to livelihoods of the coastal communities, both rural and urban, that are of significant environmental and socio-economic importance and critical for the long-term survival of the region's economy. The forests reduce soil erosion in upstream areas and mitigate potential siltation and nutrient discharge into the Indian Ocean that could

lead to the degradation of adjacent marine habitats and impacts on marine life.

Threats
Trends in coastal forest cover show a general decline characterized by fragmentation. Large areas of the forest have been converted to farmland over many years of human habitation resulting in considerable loss of habitat and habitat continuity between the natural fragments. As a result, the forest's production potential and service provision to local livelihoods decline and the associated unique biodiversity of the region is under severe threat.

Recommendations
Promotion of the conservation of the coastal forests should be contextualized under a framework that involves a balance between the environment, society, and development and conservation strategies. Additionally, forest conservation should be integrated in river basin and catchment management and invasive species control. Integrated coastal zone marine spatial planning should be supported to mitigate the threats to biodiversity attributed to invasion by alien species, and poor land use practices and planning.

Marine and coastal connectivity

Status and importance
The high seas comprise ecosystems that support ecologically important functions and livelihoods, and are critical migration routes that maintain biodiversity globally. Ocean connectivity is critical for the persistence of marine life and the vast benefits that derive from it. Regional scale connectivity patterns in the WIO demonstrate the potential of using oceanographic modelling to estimate functional connectivity among zones of maritime jurisdictions (see Chapter 18).

Threats
Threats to connectivity are of global nature and include unsustainable fisheries and uncontrolled shipping.

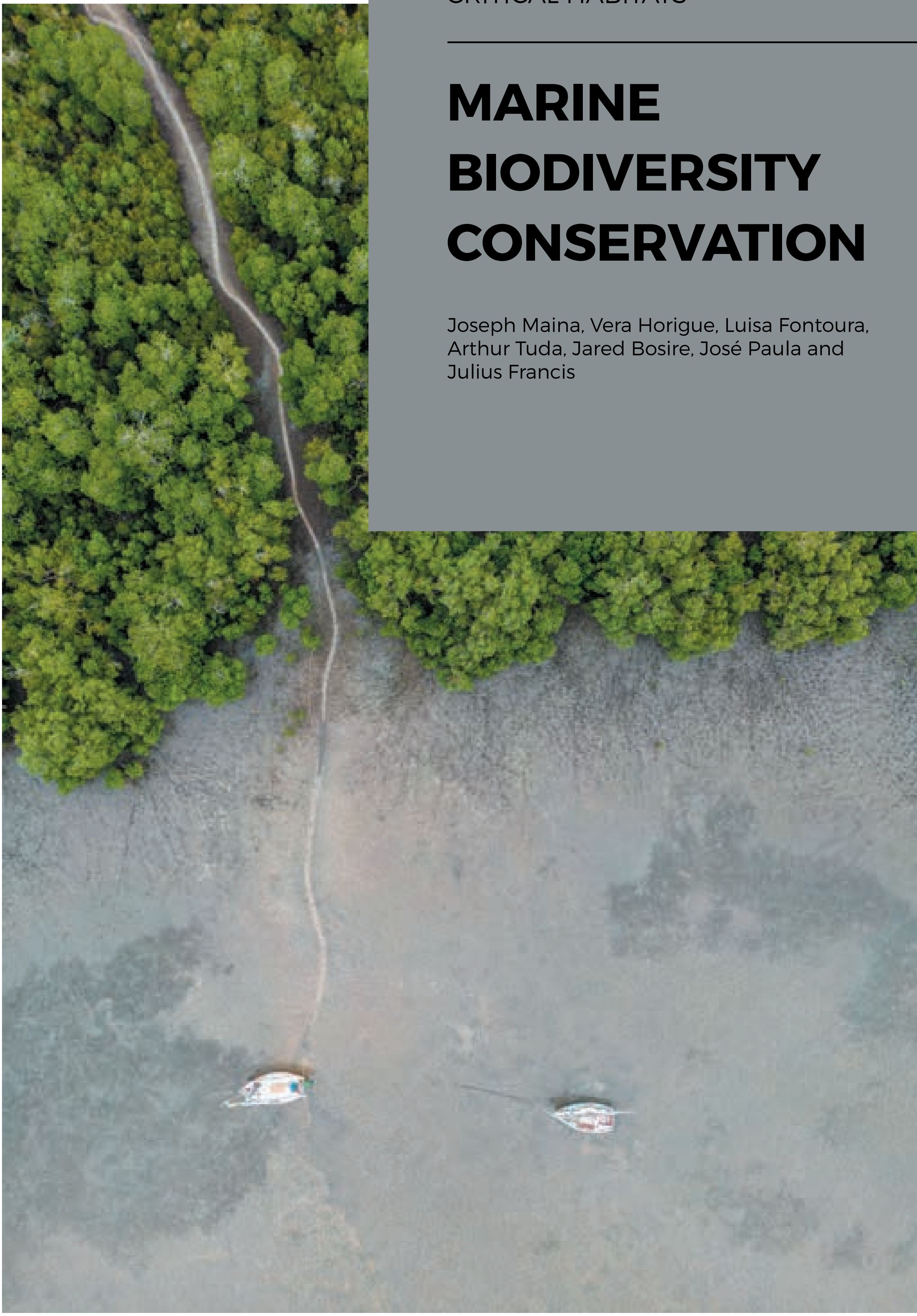
Recommendations
Knowledge of large-scale connectivity patterns is essential for managing the oceans, both within and outside areas of national jurisdictions. Furthermore, studies on the feasibility, options, and scenarios for establishing MPAs in ABNJ are necessary. This may involve partnerships with global organizations, such as the International Maritime Organization (IMO) and the United Nations Convention on the Law of the Sea (UNCLOS) to facilitate identifying and designating particularly sensitive sea areas (PSSAs).

PART 4: PROTECTING CRITICAL HABITATS IN THE WESTERN INDIAN OCEAN

CRITICAL HABITATS

MARINE BIODIVERSITY CONSERVATION

Joseph Maina, Vera Horigue, Luisa Fontoura, Arthur Tuda, Jared Bosire, José Paula and Julius Francis



Fishing boats on the edge of Saco mangrove, Inhaca Island (Mozambique). © David Mateus & José Paula

INTRODUCTION

Marine protected areas (MPAs) occupy ~7.92 per cent of the global ocean and are arguably one of the most important tools for biodiversity conservation and maintaining ecosystem goods and services (UNEP-WCMC, 2021). According to the governance style and the restrictions applied to marine management, effective area-based conservation measures are especially needed in regions where local communities heavily depend on natural resources and where a balance between conservation and exploitation is required most (Maina et al., 2020; Maire et al., 2021). Despite 10 years of conservation policy implementation (ie Aichi Targets), MPA networks still have significant quality gaps that limit their effectiveness on the social and ecological scales (Jantke et al., 2018). Upon the expiration of the previous Convention on Biological Diversity's (CBD) conservation policy framework (CBD, 2010), an assessment of the sustainability and status of existing conservation area networks can be useful for developing ecologically efficient MPA designs and sustainable management to ensure that MPAs achieve conservation and sustainable development objectives (Claudet et al., 2021; Reimer et al., 2021). To address conservation challenges from the previous decade and reverse the decline in biodiversity, an aspiration of the CBD's Kunming Montreal Global Biodiversity Framework (CBD, 2022) (GBF) and the Sustainable Development Goals (SDGs), it is critical to evaluate the gains made, and review of lessons learned from the contemporary eras (Petersson and Stoett, 2022; Reyers and Selig, 2020; Xu et al., 2021).

As part of the Strategic Plan for Biodiversity 2011–2020, countries committed to achieving specific conservation targets (the Aichi Targets), which outlined how biodiversity conservation should be accomplished (Campbell et al., 2014). Conservation goals for 2020 were intended to address the causes of the failed 2010 targets, including ensuring that they are SMART (specific, measurable, achievable, realistic, and time-bound) (Campbell et al., 2014). However, despite these efforts, most targets had no standardized indicators or associated data (Bhatt et al., 2020; Han et al., 2017). Under Target 13, for example, genetic diversity indicators focused on species of economic importance rather than genetic diversity across all species, thus undermining their relevance for global conservation goals (Hoban et al., 2020; Laikre et al., 2020). The same applies to Target 11, where there are multiple approaches to defining what constitutes an 'ecologically representative' protected area system and a 'well-connected' conservation strategy (Watson et al., 2016; Butchart et al., 2016; Hill et al., 2016). Furthermore, most targets lack quantitative definitions of 'success' for 2020, complicating progress measurements in terms of distance from a defined endpoint (Titensor et al., 2014). These have contributed to the paucity of comprehensive and holistic assessments of Aichi Targets, with most gap analyses tending to concentrate on the percentage area aspect because it is the easiest to measure (Hill et al., 2016; Adams et al., 2021). Furthermore, indicators may also vary spatially at regional and country scales due to diverse spatial, cultural, socio-economics and political contexts (Lengyel et al., 2008; Teixeira et al., 2016). Therefore, quantifying the relative efforts on preserving species and evaluating biodiversity trends, focusing on ecosystems and processes at a regional scale, may provide a regional-level understanding of the progress towards achieving the targets and inform the future management focus (Adams et al., 2021; Claudet et al., 2021).

There have been few retrospective evaluations of area-based measures from which lessons can be learned for implementing the GBF (Pressey et al., 2021). Previous regional and global evaluations have shown limited progress on most of the Aichi Biodiversity Targets despite ongoing conservation efforts (CBD, 2020; Xu et al., 2021). The global goal of preventing extinctions and halting species declines (Target 12) was not reached (Mace et al., 2018; Mair et al., 2021). Many assessments focusing on Target 11 have reported shortfalls in key components of the conservation policy (Hill et al., 2016). For example, connectivity conservation was found to be inadequate, with recent studies showing that 70 per cent of functionally important reefs are not protected (Fontoura et al., 2022). Similarly, representation was found to fall short, with 90.5 per cent of the 17 348 marine species evaluated shown to have less than 5 per cent of their range covered by strictly protected areas (Jetz et al., 2022; Klein et al., 2015). In addition, the rate of loss of natural habitats (Target 5) was not halved by 2020 (Mace et al., 2018), and all fish stocks were not sustainably harvested (Target 6) (Britten et al., 2021).

The GBF encompasses four goals and 23 targets (CBD, 2022). Generally, the goals envision reducing the rate and risk of species extinction, sustainable management of biodiversity as underpinning sustainable development and ecosystem services, equity and protection of genetic resources and the traditional knowledge associated with it, and providing financing mechanisms and capacity for implementing the framework. In addition, these goals are emboldened using 23 targets. Arguably, Target 3, which envisages the "*effective conservation of at least 30 per cent of degraded terrestrial, inland water and coastal/marine ecosystems by 2030 (i.e., 30 x 30), recognizing indigenous and traditional territories, where applicable*" is fundamental for

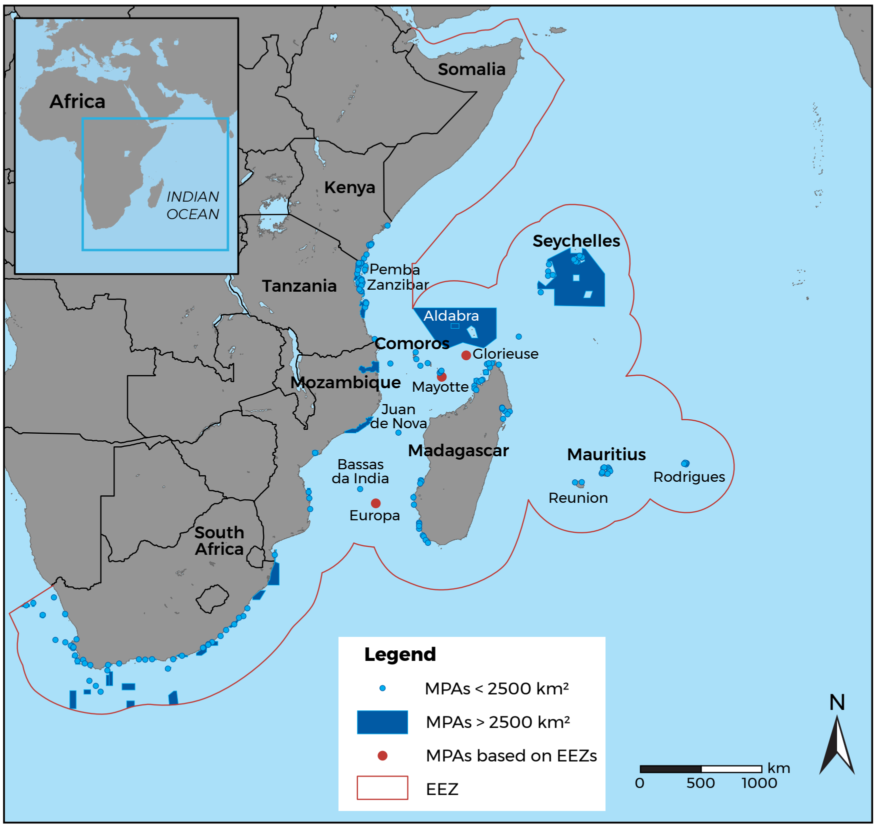


Figure 1: Distribution of marine protected areas (MPAs) in the Western Indian Ocean region.

achieving the other targets. As the target envisions, the protected area estate will need to be expanded at least three-fold by 2030. Given the countries' capacity and development priorities, it is critical that the global biodiversity finance gaps are addressed for countries in the developing world to implement this framework (Kedward et al., 2023). Progress towards the 30 by 30 must be accompanied by effective management to ensure the spatially expanded protected areas achieve the intended objectives.

In considering area-based management tools (ABMT) targets, to some extent, Aichi Target 11 aligns with GBF Target 3, at least in substance. Regarding ABMT, developing nations lay far behind developed countries in MPA implementation, with the high level of dependence on marine resource extraction being one of the impediments (Marinesque et al., 2012). The high coupling of social and ecological systems in developing regions has also led to the development of local marine management areas (LMMAs), a compromise between full closure and avoiding the opportunity cost with collaborative governance systems between local communities and government at different levels (Laffoley et al., 2017; Reimer et al., 2021). While most of these do not yet provide the levels of protection afforded by the more established, formal and effectively managed MPAs, these other effective area-based conservation measures (OECMs) have great potential to increase the coastal areas under conservation and contribute to both GBF and the ocean-related UN SDG-14 targets (Laffoley et al., 2017; Agung et al., 2022; Reyers and Selig, 2020), while noting that there are critical policy and governance limitations, which will need to be urgently addressed to ensure effective LMMA establishment and management, especially under the GBF policy.

The Western Indian Ocean (WIO) region includes ten countries (Comoros, France, Kenya, Madagascar, Republic of Mauritius, Mozambique, Seychelles, Somalia, South Africa and the United Republic of Tanzania), five of which are island states. WIO countries recently reported on the progress made under Target 11, particularly the area-based percentage on management effectiveness (UNEP-Nairobi Convention and WIOMSA, 2021).

Overall, there are 143 MPAs (or equivalent, see Figure 1), covering a total of 555 436.68 km^2, ~7 per cent of the total combined exclusive economic zone (EEZ) of nine (excluding Somalia) regional analyzed countries (UNEP-Nairobi Convention and WIOMSA, 2021). Since 2010, 46 MPAs have been established across these nine countries. Most MPAs predominantly protect critical coastal habitats, including mangroves, seagrass beds and coral reefs. Most existing MPAs across the region are not managed effectively due primarily to a lack of capacity and poor enforcement and compliance (UNEP-Nairobi Convention and WIOMSA, 2021). National-level assessments suggest a disparity in implementation efforts of area-based management tools (ABMTs), with most countries indicating shortfalls. Despite the regional approach to conservation policy implementation under regional mechanisms, eg the UNEP-Nairobi Convention, international commitments require implementation at the national level and are reported as such. However, the continuous nature of biodiversity and the socio-ecological interdependence require regionwide transboundary cooperation for biodiversity conservation to address representativeness, connectivity, and socio-economic benefits at the regional level (Mammides et al., 2021).

Furthermore, the push for quantity over quality undermines sustainability and social justice, especially in countries with high social and ecological coupling, further reinforcing a need for transboundary cooperation among countries (Mammides et al., 2021). Thus, evaluating conservation policy outcomes needs to focus on the quality of conservation efforts from the perspective of socio-economic benefits, threats and the condition of biodiversity and associated habitats (Islam, 2021).

As an extension of the recent comprehensive assessment of the progress of the world's regions on the area-based targets, we evaluate the WIO region's status in achieving Aichi Targets, which serves as a baseline for addressing the corresponding GBF targets. Specifically, we assess country and regional level progress regarding Target 5 (rate of loss of all-natural habitats); ten (anthropogenic pressures); 11 (10 per cent area coverage, connectivity, and representativeness); 13 (genetic diversity); and 14 (restoration and maintenance of ecosystem services). Considering the possible linkages between the five Aichi Targets under three strategic goals (B-reducing pressure on biodiversity; C-ensuring ecosystems, species, and genetic diversity; and D-ensuring ecosystem services), actions under these targets may contribute to achieving other targets (Laffoley et al., 2017).

To evaluate conservation efforts based on the indicators and targets, we use a variety of indicators within the limitations of the data available. We discuss the management performance of WIO MPAs and management and governance challenges that must be addressed to ensure their success and sustainability. Moreover, we provide recommendations for the regional pursuit of successful conservation outcomes under the GBF.

DATA AND METHODS

Based on regionwide data availability on indicators, we chose five targets from three strategic Aichi goals to evaluate the corresponding conservation policy outcomes in the WIO countries (Table 1). Next, for each of the five targets, we compiled spatial datasets for the corresponding indicators from freely available global and regional datasets, including critical habitats, climate exposure variables, and areas of high importance for species connectivity and population persistence (Table 1).

As part of the overall goal within Aichi targets of reducing direct pressure on biodiversity and promoting sustainable use, the aim of Target 5 was that by 2020, biodiversity loss in all-natural habitats would be reduced. This is analogous to Target 1 in the GBF, which aims to bring the loss to areas of high biodiversity importance to close to zero by 2030. Thus, given the availability of their spatial and temporal coverage, we use mangrove cover as the case habitat. To analyze whether habitat loss and fragmentation of a key coastal ecosystem were lower in protected areas than in unprotected areas, we used data from two time periods (2010 and 2016) (Aichi Target 5, see full definition in Table 1). We calculated and compared total mangrove coverage within protected areas established after 2010 and outside all protected areas by overlaying mangrove distribution data with protected area boundaries. Since mangroves are found in land-sea transitional areas, we used coastal protected area boundaries covering the transitional zone from the World Database on Protected Areas (WDPA) (IUCN and UNEP-WCMC, 2017). We only considered national protected areas that fall into established IUCN categories (I–V).

As part of the same goal, Aichi Target 10 aimed to minimize the impact of anthropogenic pressure on vulnerable ecosystems by 2015 (CBD, 2010). Similarly, Target 4 in the GBF calls for ensuring urgent management actions to stop the human-induced extinction of species and restore genetic diversity, while Target 8 aims to minimize the impact of climate change and ocean acidification on biodiversity. Hence, to assess the protection of vulnerable ecosystems from climate change, we evaluate the relative proportion of the most and least vulnerable coral reefs currently under protection. A coral reef map was obtained from Allen Atlas (Lyons et al., 2020) and spatially overlaid with predictions of annual severe coral bleaching onset (UNEP, 2020). Annual severe bleaching (ASB) maps estimate the year a reef location will suffer annual severe bleaching from climate change. ABS on coral reefs were characterized based on shared socio-economic pathway (SSP) scenarios: the mid-range

Table 1: Aichi Targets addressed in the study and their associated strategic goal, definitions and indicator(s). Strategic goals and definitions are available at https://www.cbd.int.

STRATEGIC GOAL	TARGET	DEFINITION/OBJECTIVES	INDICATOR(S) AND RESPECTIVE DATASET(S)
B: To reduce the direct pressure on biodiversity and promote sustainable use	5	By 2020, the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced.	Absolute and relative changes in mangrove cover inside and outside MPAs. Mangrove cover at two time points (2010 and 2016).
B: To reduce the direct pressure on biodiversity and promote sustainable use	10	By 2015, the multiple anthropogenic pressures on coral reefs and other vulnerable ecosystems impacted by climate change or ocean acidification are minimized so as to maintain their integrity and functioning.	The percentage of highly climate-vulnerable mangroves and coral reefs protected. Frequency of annual bleaching across coral reefs based on two climate change scenarios (IPCC-S585 and S245). Human gravity on coral reefs used as proxy of anthropogenic pressure on coral reefs. Mangrove vulnerability to climate and human threats (Maina et al 2021).
C: To improve the status of biodiversity by safeguarding ecosystems, species and genetic diversity	11	By 2020, at least 17 % of terrestrial and inland water, and 10 % of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes.	Coral reef, seagrass and mangrove area and seamount locations. Key biodiversity areas (KBAs). Reef-fish connectivity based on larval dispersal movement across the seascape.
C: To improve the status of biodiversity by safeguarding ecosystems, species and genetic diversity	13	By 2020, the genetic diversity of cultivated plants and farmed and domesticated animals and of wild relatives, including socio-economically as well as culturally valuable species, is maintained, and strategies have been developed and implemented for minimizing genetic erosion and safeguarding their genetic diversity.	Distance between coral reef and seagrass patches; Turtle nest locations, Important Bird areas (IBAs). Reef-fish connectivity based on larval dispersal movement across the seascape. Coral reef geomorphic data to identify outer reef and reef slope; both considered potential locations for spawning aggregation of fish species targeted (eg grouper).
D: Enhance the benefits to all from biodiversity and ecosystem services	14	By 2020, ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and well-being, are restored and safeguarded, taking into account the needs of women, indigenous and local communities, and the poor and vulnerable.	OECMs

fossil-fueled development (SSP5-4.5) and high-emissions global warming (SSP5-8.5).

Goal C under Aichi targets aspired to protect species, ecosystems, and genetic diversity for increased biodiversity (CBD, 2010). As part of this goal, Target 11 envisaged the protection of at least 10 per cent of coastal and marine ecosystem areas by 2020, while Target 13 aims to preserve the genetic diversity of socio-economically and culturally important species. Under the same overarching goal, Target 14 aimed to restore and protect ecosystems that provide essential services (CBD, 2010). The corresponding GBF Targets are 3, 4 and 2, respectively, where under Target 3, the proportional area was increased to 30 per cent, while Target 4 addresses the need to preserve genetic diversity, and Target 2 envisages effective restoration of at least 30 per cent of degraded terrestrial, inland water, and coastal/marine ecosystems by 2030. Moreover, GBF Target 9 envisions sustainable management and use of wild species, thereby supporting especially those individuals and communities most dependent on biodiversity.

We used ten indicators to measure the WIO's progress in achieving Aichi Target 11 and the baseline for corresponding GBF targets (Table A1). Using the most recent MPA database for the WIO (UNEP-Nairobi Convention and WIOMSA, 2021), for each indicator, we calculated the total cumulative percentage area for the entire region and for each country that was protected in MPAs. These indicators were: (i) Exclusive Economic Zone;(ii) Key Biodiversity Areas (KBAs) (Kullberg et al., 2019), (iii) most Critical Habitats (ie, mangroves, seagrass, seamounts, coral reefs); and (iv) functionally important coral reefs, which we define as reefs that both serve as larval sinks and sources.

To evaluate connectivity conservation, we used a database of connectivity matrices based on biophysical models (Fontoura et al., 2022). The database is based on larval dispersal simulations on coral reef habitats gridded at 8 x 8 km^2. One thousand fish larvae were released on each reef and tracked based on their pelagic larval duration and sensory zone. Using the connectivity matrix, we used the igraph package in R to estimate the connectivity characteristics of sources, sinks, and corridors (see Fontoura et al., 2022, for detailed methodology). Larval sinks are reefs that import more larval subsidies than they can export, while larval sources are reefs that export larval subsidies. Even though larval sources can replenish fish populations on downstream reefs (Harrison et al., 2020), the low likelihood of larval import could indicate greater vulnerability to overfishing. Due to their ability to receive larvae from other sources and to benefit from protected larval sources, larval sinks are an important location to consider for the growth of local, sustainable fisheries. Hence, all reefs exporting larvae into MPAs were considered larval sources. To quantify functionally important connectivity among coral reefs in the current MPA network arrangement, we calculated the proportion of potential fish larval sources within MPAs. Inter-habitat connectivity was also used as an indicator for connectivity within the complex ecological interdependence on mosaic seascapes. According to Berkstrom et al. (2020), seagrass patches within 8 km of coral reefs can potentially act as important nurseries for reef fish, thereby increasing coral reef fish biomass and promoting genetic diversity (Aichi Target 13; GBF Target 4). Thus, we used the 8 km distance threshold to identify seagrass patches that may serve as coral reef nurseries for MPAs and estimate their proportional representation within MPAs. The potential dependence of coral reef fish biomass and diversity on neighbouring seagrass patches is an example (and here assumed as a proxy) of the multiple functional linkages between habitats, that also include mangroves and other systems, as well as a variety of other taxa and organic and inorganic matter exchanges.

In the face of environmental disturbances, genetic diversity promotes species resilience and can prevent populations from abrupt decline and extinction (Hoban et al., 2020). Consequently, progress toward Target 13 could be assessed effectively with a comprehensive marine gene bank, which does not exist. On the other hand, halting or slowing the decline of species' populations might help prevent the loss of genetic diversity and erosion, as envisaged in Aichi Target 13/GBF Target 4 (Hoban et al., 2020).

To measure progress toward this goal within the WIO region, we examined how the existing MPAs support conservation strategies promoting species population persistence. To begin with, we evaluated the protection status of areas that are important for species reproduction (ie, important bird areas (IBAs), spawning sites of transient reef fish species); offspring survival (turtle nests, potential fish nurseries); and gene flow. The latter included coral reefs that serve as stepping stones for reef fish populations, herein after dispersal corridors (Palumbi, 2003). As a subset of KBAs, IBAs consist of only those parts of KBAs that have been confirmed as IBAs. Using the geomorphic spatial data based on the Allen Coral Reef Atlas (Lyons et al., 2020), the outer reef and reef slopes were compiled across the WIO region. Protecting turtle nesting sites was also used as an indicator for Aichi Target 13/GBF Target 4. Turtle data was downloaded from the Mozalink project website (Lagabrielle et al., 2018). To define potential seagrass nurseries, we used the same definition used for Target 11, which defined every seagrass patch located within 8 km of a coral reef patch as a reef fish nursery (Berkstrom et al., 2020). Using coral reef connectivity data described under Target 11, we identified important regional dispersal corridors based on fish larval dispersal across the seascape. The dispersal corridors we defined are reefs that export larval subsidies (outflow > 0) and are highly interconnected to other reefs (top 15 per cent of reefs with higher indegree – number of incoming connections).

Goal D within Aichi targets envisaged enhanced biodiversity and ecosystem services benefits (CBD, 2010), which is similar in substance to goal B of the GBF. In line with this goal, Target 14 aspired to restore and safeguard ecosystems that provide ecosystem services (Reyers and Selig, 2020). Targets 2 and 9 of the GBF have similar aspirations for restoration of ecosystems and their sustainable use and management.

LMMAs are important ABMT that support successful socio-economic outcomes and sustainable fisheries (Reimer et al., 2021; Gurney et al., 2021). As an indicator for Aichi Target 14 (and GBF Targets 2 and 9), we assessed the spatial distribution of the LMMAs relative to the exposure to climate change and the location of larval sources and sinks. When strategically located in source areas and relatively less climate-exposed areas, LMMAs can more effectively support sustainable fisheries outcomes (Fontoura et al., 2022). Thus, we characterized each LMMAs and MPA along with the sink-source gradient and climate exposure based on the two SSP scenarios considered in Target 10. We calculated the proportion of OECMs located within reef fish larval sinks and sources as well as their climate change exposure.

RESULTS

Reducing the rate of loss of all-natural habitats

(Aichi Target 5/GBF Target 1)

Using the change in mangrove cover as an indicator for Aichi Target 5/GBF Target 1, the average loss of mangrove cover between 2010 and 2016 was approximately 318 hectares, which represents well less than 1 per cent of the region's total mangrove area (estimated at around 1 million hectares: Bosire et al., 2016). The extent of mangrove cover varied within and outside protected areas post-2010 (Figure 2). In Kenya, Madagascar, Mozambique, and Tanzania, the countries with the highest mangrove areas (see Chapter 9), estimated mangrove losses ranged from 415 to 1368.8 hectares, representing 0.7 per cent of total mangrove areas across these countries (Figure 2).

In contrast, mangrove cover increased in countries with relatively fewer mangrove areas over the same period, with 4.4 per cent increase in Seychelles (~4 hectares), 1.1 per cent in Somalia and 0.7 per cent in Mauritius (Figure 2). There was no change in mangrove cover in Comoros.

Changes in mangrove cover differed according to conservation status (Figure 2). In Kenya, it increased within protected areas while it declined outside. Mangroves in Mauritius showed positive changes despite their unprotected status. In Mozambique and the French territory, mangroves decreased in protected areas while they increased outside. Seychelles showed no change inside protected areas, while mangroves increased outside protected areas. In Madagascar, the decline in mangrove cover was more pronounced inside protected areas than outside. Overall, mangroves expanded by 21.6 hectares (~0.003 per cent) throughout the region, mostly in Kenya, but decreased by 934 hectares (~1.5 per cent) within protected areas.

Evaluating the representation of mangroves in MPAs may be problematic. In many WIO countries, mangroves are forest reserves under the jurisdiction of Forestry Departments, even if small patches might be inside MPAs. In Tanzania, for example, mangroves are protected everywhere, whether in or outside MPAs. Therefore, the argument and any evidence presented on whether MPAs protect mangroves is questionable (UNEP-Nairobi Convention and WIOMSA, 2021).

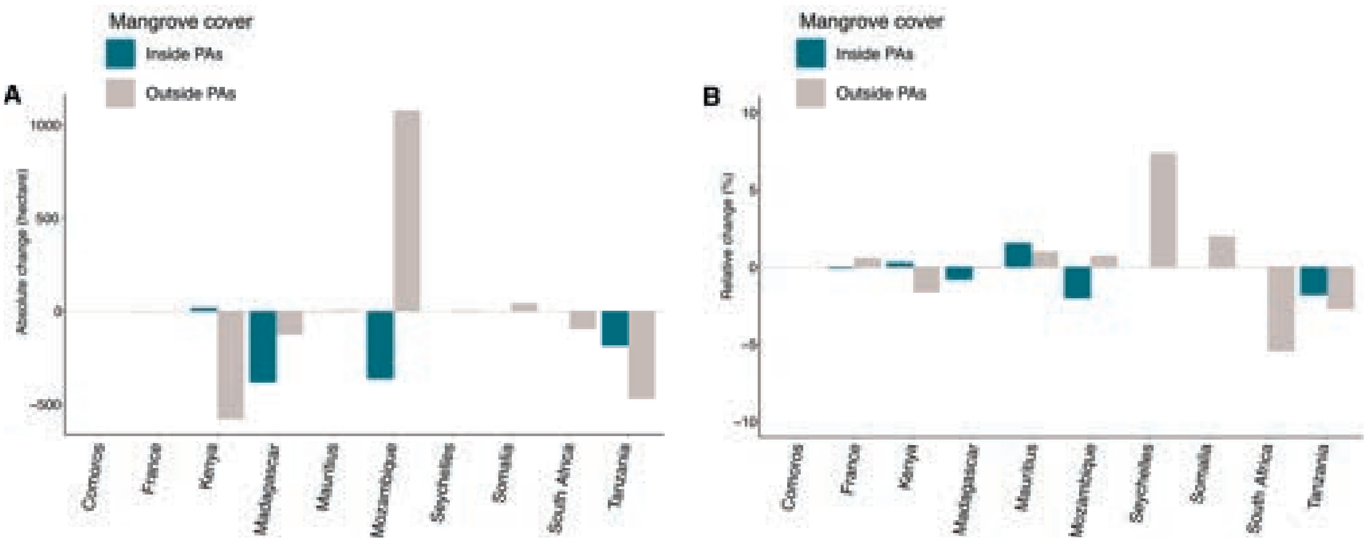


Figure 2: Absolute (A) and relative (B) temporal changes in mangrove cover across WIO countries inside (blue) and outside (grey) protected areas. Temporal changes were based on two time periods (2010 and 2016). Negative values indicate loss of mangrove cover measured in hectares (A) and relative to total mangrove coverage area recorded in 2010 within each national country (B). Positive values indicate expansion of mangrove cover measured in hectares (A) and relative to the total mangrove coverage area recorded in 2010 within each country (B).

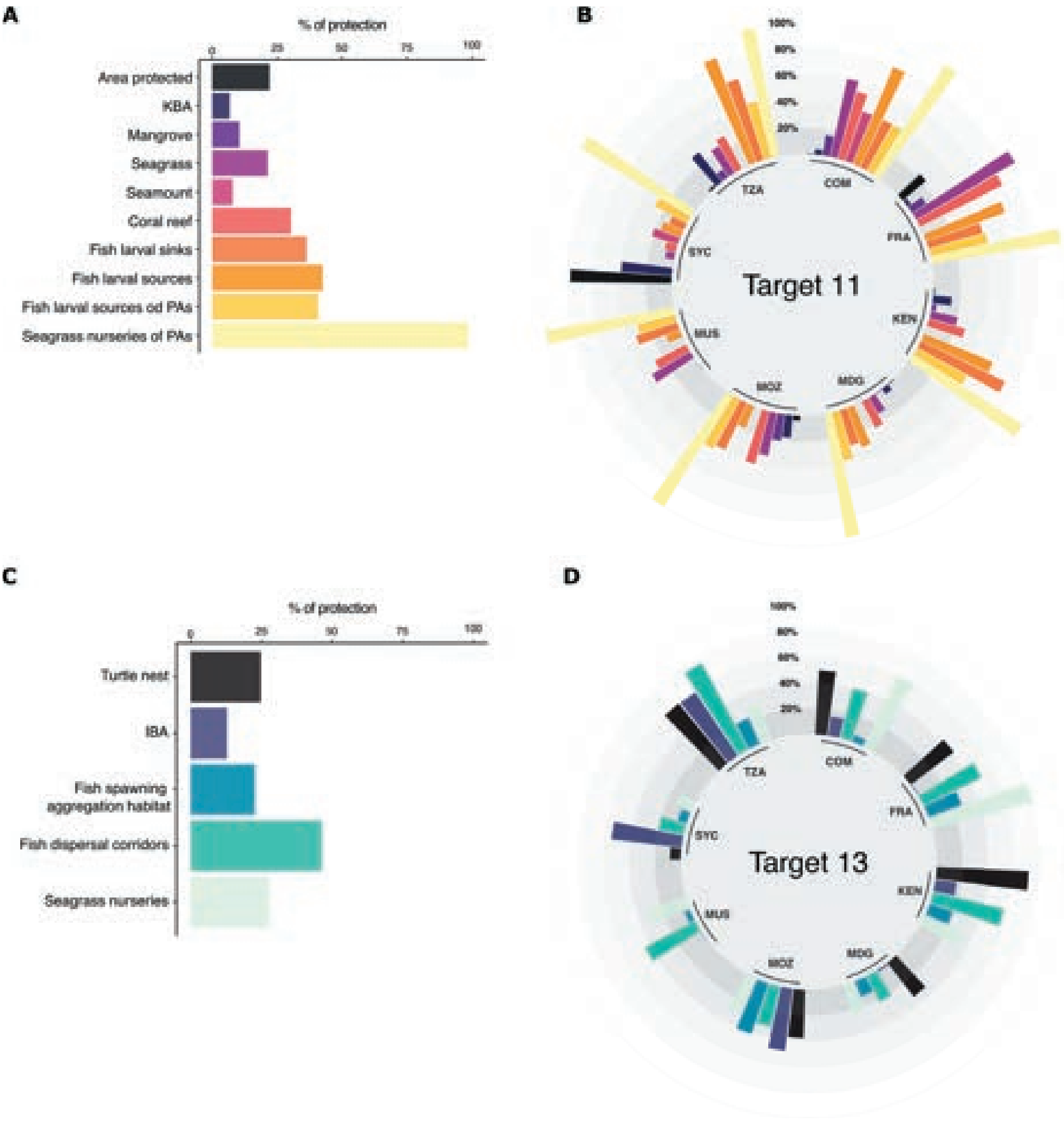


Figure 3: Marine protected area coverage by region and country and the representation of the critical habitats, key biodiversity areas, and ecological processes of functional connectivity (indicators of Aichi Targets 11 and 13). Bar and circular graphs represent the percentage of protected targets' Aichi Target indicators for the region and country-level, respectively. The percentage of protection was estimated based on the proportion of each indicator observed inside MPAs within the Western Indian Ocean region. KEN–Kenya, MDG–Madagascar, MOZ–Mozambique, MUS–Mauritius, SYC–Seychelles, TZA–Tanzania, COM–Comoros, FRA–France, excluding Somalia and South Africa.

Protecting areas of particular importance for biodiversity and ecosystem services
(Aichi Target 11/GBF Target 3)

The UNEP-Nairobi Convention and WIOMSA (2021) found that the existing MPAs cover nearly 7 per cent of the exclusive economic zones of WIO nations. However, there were marked differences in the representation of critical habitats, KBAs, and important connectivity attributes (Figure 3). Approximately 7.8 per cent of seamounts and 6.6 per cent of KBAs within the WIO region are protected, the lowest proportion of all indicators. The proportion of protected mangrove areas was lower than that of seagrass beds and coral reefs (10 per cent for mangroves and 20 and 30 per cent, respectively).

Regionally, the protection of important connectivity components for supporting ecosystem services provided by coral reefs varied from 36 per cent to 98 per cent. Among the critical larval sinks and sources identified in the region, 36 and 42 per cent are in protected areas, respectively. Similarly, 40 per cent of coral reef larval sources are located within marine protected areas. Approximately 98 per cent of seagrass beds that function as reef fish nurseries for MPAs are protected when considering the 8 km distance criteria (Berkstrom et al., 2020).

The percentage of protection for most indicators differed considerably between countries. A country's percentage of protection was calculated relative to the total area/occurrence of an indicator within its EEZ. Seychelles, South Africa, and French territories had the highest percentage of EEZ protection (77, 22, and 22 per cent, respectively). However, in Seychelles, less than 10 per cent of coral reefs, seagrass beds and mangroves are protected. Despite a relatively lower percentage of protection of the total EEZ area, other countries exhibited high levels of protection of other key indicators. MPAs cover less than 1 per cent of the EEZ of Mauritius, but despite this lower coverage, more than 20 per cent of KBAs, seagrass beds, and coral reefs lie within these MPAs. Similarly, in Kenya and Comoros, Mozambique, Tanzania and Madagascar, nearly all protection indicators were higher than 15 per cent, despite the total area protected by MPAs being below 1 per cent.

Maintaining genetic diversity

(Aichi Target 13/GBF Target 4)

Regionally, the percentage of indicators inside MPAs ranged from 12.8 per cent to 46.2 per cent, with important bird areas (IBAs) showing the lowest percentage and key dispersal corridors for reef fish the highest (Figure 3). More than 20 per cent of the spawning habitats of commercially important fish species, seagrass beds considered reef fish nurseries, and turtle nests were found in MPAs.

On a country scale, Tanzania and Kenya had the highest proportion of their dispersal corridors protected, exceeding 50 per cent. In addition, more than 48 per cent of IBAs in Seychelles, Mozambique, and Tanzania are located within protected areas, while less than 2 per cent are found within MPAs in Mauritius, Madagascar, and French territories. In Comoros, Mauritius, and Seychelles, the percentage of protected spawning sites of transient fish species was less than 10 per cent. With 72 and 60 per cent of their total turtle nests located inside protected areas, Kenya and Tanzania had more MPAs supporting turtle nest protection. Protection of seagrass beds that are potential fish nurseries was highest in Comoros and French territories (61 and 80 per cent, respectively) and lowest in Seychelles and Madagascar (12.8 per cent and 23.5 per cent, respectively).

Minimizing anthropogenic threats

(Aichi Target 10/GBF Targets 4 and 8)

We estimated annual severe bleaching (ABS) along with market gravity, a proxy for human pressure, for 184 267 coral reef patches in the WIO region. Based on a scenario of mid-level greenhouse emissions (SSP2-4.5), coral reefs classified as 'Climate-vulnerable' and 'Highly vulnerable' together represent 38 per cent of the total coral reefs assessed and are protected on average by 27 per cent of MPAs (Figure 4C).

According to the same SSP scenario, coral reefs less impacted by climate change are less vulnerable to human pressure and are 31 per cent protected (Figure 4C). Around 33 per cent of the total reef area consists of coral reefs with a higher human pressure but a relatively low climate vulnerability (ie, 'human pressure vulnerable'), 22 per cent of which are in MPAs.

When considering a high emissions global warming scenario (SSP5-8.5), the percentage of coral reefs vulnerable to climate change increased from 37 per cent to 47 per cent. An additional 48 km^2 of coral reef area was exposed to climate change. In this scenario, 33 per cent of coral reefs are less vulnerable to climate change, 28 per cent of coral reefs are more vulnerable to climate change, and 25 per cent of coral reefs with higher human pressure but relatively low vulnerability to climate change are protected by MPAs (Figure 4D).

Safeguarding ecosystems that provide essential services

(Aichi Target 14/GBF Targets 2 and 9)

In Mozambique, Madagascar, Kenya, and Tanzania, 23 per cent of coral reefs associated with OECMs are in fish larvae sinks, and 41 per cent are in fish larvae source areas (Figure 5). Furthermore, 52 per cent of coral reefs with a high probability of exporting larvae to OECMs are unprotected against unrestricted fishing, limiting their potential to support sustainable fisheries. Under fossil-fueled development and high emissions global warming (SSP5-8.5), 35 per cent of coral reefs in OECMs will experience coral bleaching that is more pronounced than that observed in MPAs (Figure 5).

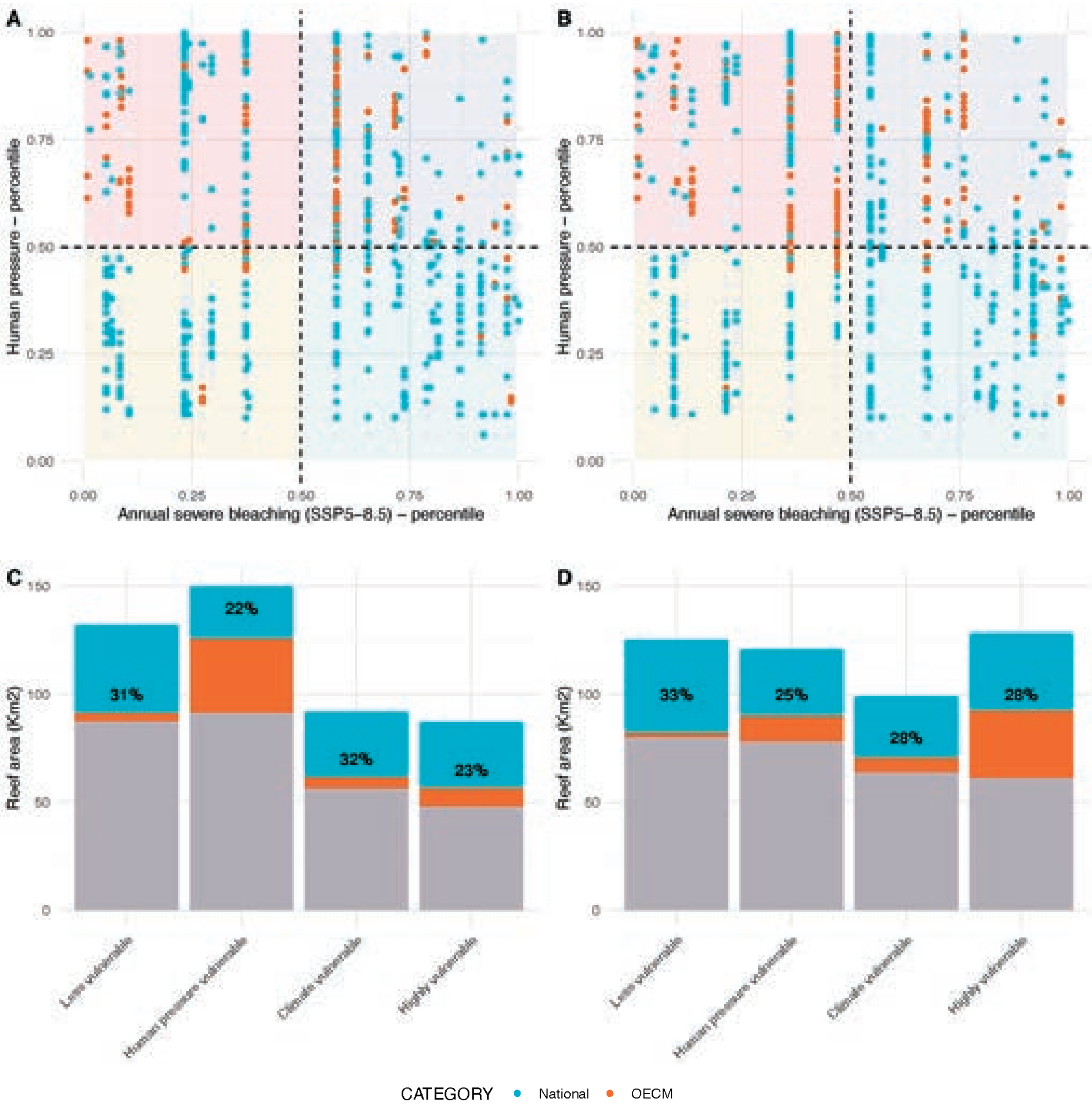


Figure 4: Exposure of coral reefs in the WIO based on human pressure and annual severe bleaching under mid (SSP2-4.5) to highest (SSP5-8.5) fossil-fuelled development and emissions scenarios. In (A) and (B), coral reefs are represented by points and coloured according to management status. The x-axis and y-axis represent human pressure and annual severe bleaching (ASB) percentiles for each coral reef. In (C) and (D), coral reefs with values lower than the 50th percentile of human pressure and higher than the 50th percentile of (ASB) were classified as "less vulnerable" (blue quadrat), and coral reefs with values lower than the 50th percentile of human pressure and lower than the 50th percentile of (ASB) were classified as "climate-vulnerable" (yellow quadrat), coral reefs with values higher than the 50th percentile of human pressure and higher than the 50th percentile of (ASB) were classified as "human pressure vulnerable" (purple quadrat) and coral reefs with values higher than the 50th percentile of human pressure and lower than the 50th percentile of (ASB) were classified as "highly vulnerable" (red quadrat). Bars represent the total area of coral reefs classified according to the vulnerability categories described above. Colours represent coral reefs patches inside MPAs (blue), OECMs (orange) and unprotected (grey). The percentage of coral reefs inside MPAs is represented by numbers on each bar by category.

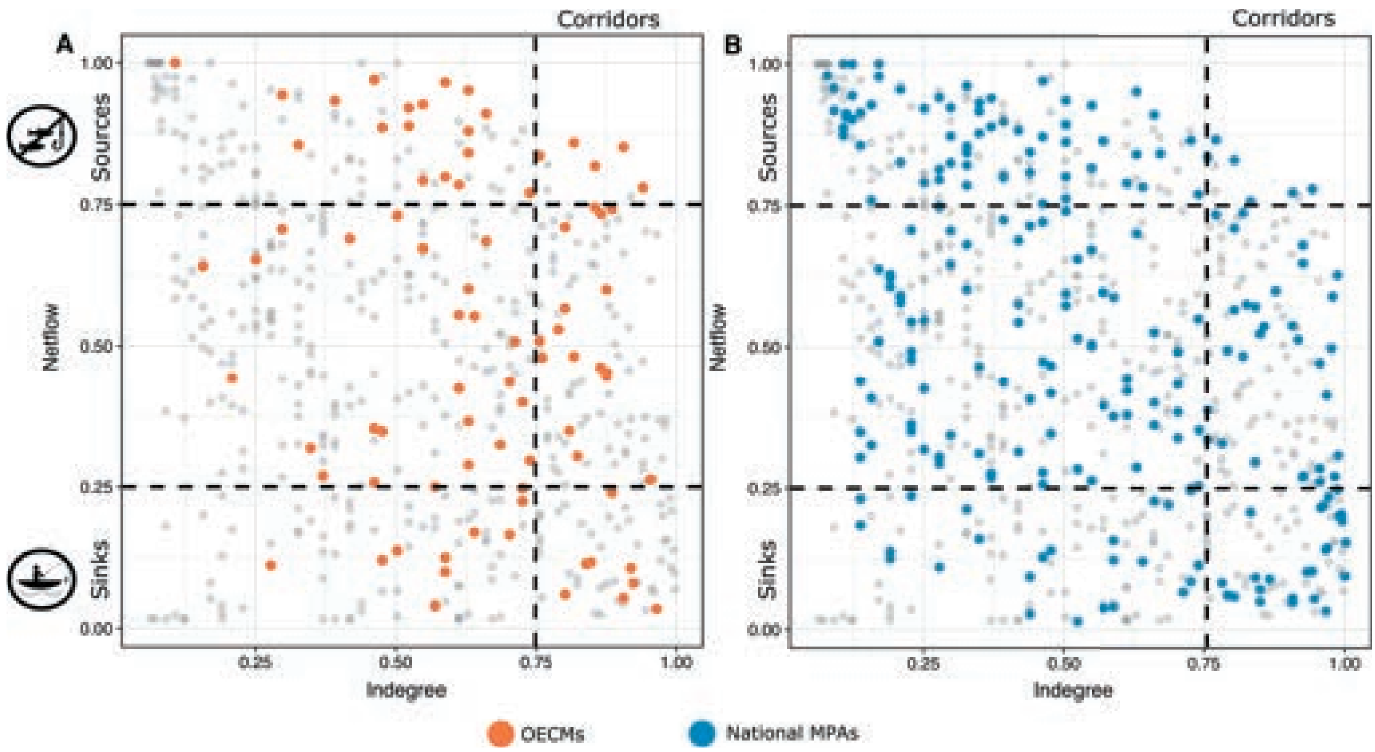


Figure 5: Representation of sinks, sources and corridors in MPAs and LMMAs.

DISCUSSION

Analysis of the progress in biodiversity conservation based on indicators of coverage of protected areas, representation of critical habitats, changes in habitat, connectivity conservation, genetic diversity and threats reveals significant efforts at the regional level, in addition to addressing the identification of knowledge gaps. The analyses also highlight the significant effort required and the obstacles that need to be overcome to reach GBF targets.

In assessing progress to 2020, a marginal net change in mangrove cover was observed in the period 2010–2016, with the Seychelles, Mauritius and Somalia reporting relative increases in mangrove cover. Although mangrove cover grew outside MPAs, it declined inside MPAs. Critical habitats (coral reefs, seagrass, and mangroves) are well represented in MPAs, with proportional protection exceeding 10 per cent of total regional coverage. In MPAs, offshore habitats such as seamounts, KBAs and IBAs had poor representation, with countries with a low MPA coverage (eg, Comoros) having a better representation than areas with a large MPA area (eg, Seychelles). Regarding connectivity, only 36 and 43 per cent of functionally important reefs (ie, sinks and sources) are adequately protected. The percentage of dispersal corridors protected by Tanzania and Kenya, each exceeding 50 per cent, was the highest of the nine countries in WIO. The protection of genetic diversity ranges from 12 to 46 per cent across the WIO. Under a fossil-fueled development scenario and high global warming (SSP5-8.5), 47 per cent of coral reefs in the region are at risk. Overall, OECMs do not appear to be strategically located, with 50 per cent of their identified source reefs (ie, with a high probability of exporting fish larvae) not currently in unrestricted fishing areas. About 35 per cent of coral reefs within OECMs will be increasingly vulnerable to coral bleaching under fossil-fueled development and high emissions in the global warming scenario (SSP5-8.5).

When a policy target date has passed, conservation gap analysis provides an opportunity to identify conservation gaps in the existing protected area network as well as set the stage for expanding the network by selecting high-priority areas for conservation (Knowlton, 2021; Klein et al., 2015). At the same time, this serves as a baseline for the future conservation policy, which establishes a pathway to address the gaps of the previous policy and, at the same time, achieve its targets. Indicators and target selection affect the results of systematic conservation planning and gap assessments (Vimal et al., 2011; Adams et al., 2021). While some indicators may vary by region, based on the context, they should be logical, described adequately, and evaluated for the impact of target selection (Hoban et al., 2020). Indicators and criteria used for evaluation need precise definitions to ascertain whether they are suitable for acting as adequate proxies for monitoring global trends. This is particularly important in functional processes such as connectivity patterns and ecological corridors, where multiple and complex linkages are established. In fact, under the GBF, marine connectivity indicators are highlighted as a gap that needs to be addressed. As in this analysis, emphasis was placed on the role of seagrass patches in maintaining biomass and diversity of nearby reef fish communities, but other habitat-based dependencies and other taxonomic/ecological entities may prove crucial at other temporal or spatial scales. There is still much work to be done to consolidate the use of solid functional indicators for general use throughout critical habitats at regional scales (Hu et al., 2022).

Our analysis of various indicators has shown that the WIO region has made modest efforts to preserve its biodiversity. While country-level statistics illustrate differences in the percentage of national EEZs covered by protected areas, they also show significant differences in the representation of critical habitats, key biodiversity areas, and conservation of connectivity and ecological fluxes. MPA coverage did not necessarily result in a better representation of critical habitats and ecological functions within protected areas. For example, Comoros had the lowest proportion of protected areas in the region but had over a 20 per cent representation of key indicators, including genetic diversity. Conversely, Seychelles had the highest percentage of marine areas conserved, yet their representation of the key indicators varied. A quality reserve is critically important, and places that offer little protection to species and ecosystems most at risk, known as residual reserves, should not be included in the overall protection inventory (Devillers et al., 2015). These residual MPAs typically occur in remote or unproductive sites, thus following the trend of terrestrial protected areas in being 'residual' to commercial activities.

With the implementation of the GBF and the unveiling of new targets, developing a roadmap for achieving socio-economic objectives and biodiversity targets in the draft framework is essential. To leverage initiatives that also benefit biodiversity, the roadmap should include complementary policies for climate change adaptation and sustainable development goals (Pahle et al., 2021), thus providing great policy convergence during this decade 2021–2030 under the different Multilateral Environmental Agreements, including Agenda 2030 United Nations Framework Convention on Climate Change (UNFCCC) climate agenda, UN Decade of Ecosystem Restoration and UN Decade of Ocean Science. It might be necessary to develop the regional governance framework to integrate national initiatives, goals, and objectives into broader regional and global goals.

The WIO Regional Ocean Governance Strategy currently being developed under the auspices of the African Ministerial Conference on the Environment (AMCEN) Decision of 2015 provides a timely opportunity. In addition to facilitating equitable protection efforts among countries, this would also facilitate transboundary conservation initiatives between them, as is the case for Kenya and Tanzania, which are both exploring establishing a transboundary marine conservation area. In addition, cooperation within the region can ensure a better representation of biodiversity and ecosystem processes throughout the region, thereby preventing the creation (or maintenance) of residual reserves for no other reason than to tick off a box. Developing and achieving a roadmap will require countries within the region to harmonize their national and regional goals within CBD's post-2020 GBF, UNFCCC, and SDGs.

Based on the findings of our study and conclusion from the UNEP, Nairobi Convention and WIOMSA (2021) *MPA Outlook*, the region needs to shift the focus away from percentage targets and towards quality and sustainable development targets since they are better aligned with national initiatives such as the strengthening of a Blue Economy supported by marine spatial planning (UNEP, 2021; Ferreira et al., 2022).

Moreover, common indicators are needed for conservation, sustainable development, and climate change adaptation. A lack of data and information may be one of the challenges, but creating capacity and taking precautionary measures may be the best strategies for achieving biodiversity conservation and sustainable development in the region. Restoration of degraded habitats has been promoted in the region for the past few years, including for mangroves and seagrass beds (eg UNEP-Nairobi Convention and WIOMSA, 2020), which can further contribute to maintaining biodiversity and ecosystem services, but also actively provide opportunities for engaging local communities in the conservation process.

The quality of the conservation status assessment depends on the indicators and data used (Adams et al., 2021). Our assessment was conducted using a set of specific indicators and within the limitations of availability and data quality for their description. Geospatial data on critical habitats and socio-ecological systems are essential to obtaining credible estimates as well as monitoring and evaluating the marine environment (Hoban et al., 2020). Here, we used the best available ecological and socio-economic data. However, the data may be incomplete and contain errors due to aggregating layers with a spatial mismatch.

Further, some of the official boundaries of the MPAs in the region may differ from the actual area under protection. The boundary problem extends to the LMMAs, which have yet to be mapped for the region. The actual measured area represented by LMMAs would be more informative than point geometry; we used LMMA data as points. There is a need to create a repository of data and indicators that can be used to measure conservation status and progress towards conservation goals for there to be consistency in assessments by various researchers. Territorial disputes and sensitivities on boundaries are other issues that may need to be addressed. Although regional evaluations somewhat circumvent the issue of border disputes between countries in the WIO, it is difficult to conduct country-level assessments supported by various governments when territorial problems exist. We have observed that, despite the limitations relating to the data used and the indicators used for assessment, a gap analysis has been performed in the region that goes beyond quantity but also addresses the quality of the conservation areas based on the Aichi Targets as a basis for developing a roadmap for implementing the GBF (Leadley et al., 2022).

The regional assessment of MPAs we conducted revealed overall considerable efforts in protected area coverage, but in terms of habitat quality and representation, climate change exposure, and the placement of protected areas relative to functionally connected areas, significant effort is still required, both at the country and regional level. OECMs, for instance, do not appear to be strategically located, with 50 per cent of identified source reefs (ie, with a high probability of exporting fish larvae) not being protected from unrestricted fishing. Understanding the source-sink dynamics of coral reef connectivity (and of other critical habitats) and applying it to the placement of LMMAs (Fontoura et al., 2022) might contribute to sustainable fisheries in the region. Towards implementing the post-2020 GBF, regional and national goals must be discussed and aligned with the GBF as part of developing the WIO roadmap for marine conservation and sustainable development.

An outcome of this roadmap could be a Biodiversity Framework for the WIO that can provide strategies for regional implementation of the GBF. The WIO *MPA Outlook* and its sister *Critical Habitats Outlook* could provide important foundational references to inform this regional framework.

REFERENCES

Adams, V.M., Visconti, P., Graham, V. and Possingham, H.P. (2021). Indicators keep progress honest: A call to track both the quantity and quality of protected areas. *One Earth* 4(7), pp. 901–906

Agung, M.F., Adhuri, D.S., Ferse, S.C., Sualia, I., Andradi-Brown, D.A., Campbell, S. J., ... and Ahmadia, G.N. (2022). Marine conservation beyond MPAs: Towards the recognition of other effective area-based conservation measures (OECMs) in Indonesia. *Mar. Policy* 137, 104939

Berkström, C., Eggertsen, L., Goodell, W., Cordeiro, C.A.M.M., Lucena, M.B., Gustafsson, R. ... and Ferreira, C.E.L. (2020). Thresholds in seascape connectivity: the spatial arrangement of nursery habitats structure fish communities on nearby reefs. *Ecography* 43(6), pp. 882–896

Bhatt, R., Gill, M.J., Hamilton, H., Han, X., Linden, H.M. and Young, B.E. (2020). Uneven use of biodiversity indicators in 5th National Reports to the Convention on Biological Diversity. *Environ. Conserv.* 47(1), pp. 15–21

Bosire, J.O., Mangora, M.M., Bandeira, S., Rajkaran, A., Ratzimbazafy, R., Appadoo, C. and Kairo, J.G (eds.) (2016). Mangroves of the Western Indian Ocean: status and management. WIOMSA, Zanzibar Town, 161 pp.

Britten, G.L., Duarte, C.M. and Worm, B. (2021). Recovery of assessed global fish stocks remains uncertain. *Proc. Natl. Acad. Sci.* 118(31), e2108532118

Butchart, S.H., Di Marco, M. and Watson, J.E. (2016). Formulating smart commitments on biodiversity: lessons from the Aichi Targets. *Conserv. Lett.* 9(6), pp. 457–468

Campbell, L.M., Hagerman, S. and Gray, N.J. (2014). Producing targets for conservation: science and politics at the Tenth Conference of the Parties to the Convention on Biological Diversity. *Glob. Environ. Polit.* 14(3), pp. 41–63

CBD (2010). Convention on biological diversity – Aichi biodiversity targets. http://www.cbd.int/sp/targets/.

CBD (2020). Global Assessment Report on Biodiversity and Ecosystem Services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. In *Global Biodiversity Outlook 5*. Secretariat of the Convention on Biological Diversity (CBD), Montreal. ISBN-9789292256883

CBD (2022). POST-2020 GLOBAL BIODIVERSITY FRAMEWORK,CBD/WG2020/5/L.2 5 December 2022, CBD secretariat, Montreal, Canada. https://www.cbd.int/doc/c/409e/19ae/369752b245f05e88f760aeb3/wg2020-05-l-02-en.pdf

Claudet, J., Loiseau, C. and Pebayle, A. (2021). Critical gaps in the protection of the second-largest exclusive economic zone in the world. *Mar. Policy* 124, 104379. https://doi.org/10.1016/j.marpol.2020.104379

Devillers, R., Pressey, R.L., Grech, A., Kittinger, J.N., Edgar, G.J., Ward, T. and Watson, R. (2015). Reinventing residual reserves in the sea: are we favouring ease of establishment over the need for protection? *Aquat. Conserv*. 25(4), pp. 480–504

Ferreira, M.A., Froján, C.B., Gunn, V. and Johnson, D.E. (2022). A role for UNEP's Regional Seas Programme under the post-2020 global biodiversity framework. *Mar. Policy* 136, 104930

Fontoura, L., D'agata, S., Gamoyo, M., Barneche, D.R., Luiz, O.J., Madin, E.M. ... and Maina, J.M. (2022). Protecting connectivity promotes successful biodiversity and fisheries conservation. *Science* 375(6578), pp. 336–340

Gurney, G.G., Darling, E.S., Ahmadia, G.N., Agostini, V.N., Ban, N.C., Blythe, J., ... and Jupiter, S.D. (2021). Biodiversity needs every tool in the box: use OECMs. *Nature* 595, pp. 646–694

Han, X., Josse, C., Young, B.E., Smyth, R.L., Hamilton, H.H. and Bowles-Newark, N. (2017). Monitoring national conservation progress with indicators derived from global and national datasets. *Biol. Conserv*. 213, pp. 325–334

Harrison, H.B., Bode, M., Williamson, D.H., Berumen, M.L. and Jones, G.P. (2020). A connectivity portfolio effect stabilises marine reserve performance. *Proc. Natl. Acad. Sci.* 117(41), pp. 25595–25600

Hill, L.S., Johnson, J.A. and Adamowski, J. (2016). Meeting Aichi Target 11: Equity considerations in marine protected areas design. *Ocean Coast. Manag.* 134, pp. 112–119

Hoban, S., Bruford, M., Jackson, J.D.U., Lopes-Fernandes, M., Heuertz, M., Hohenlohe, P.A., ... and Laikre, L. (2020). Genetic diversity targets and indicators in the CBD post-2020 Global Biodiversity Framework must be improved. *Biol. Conserv.* 248, 108654

Hu, Y., Wang, M., Ma, T., Huang, M., Huang, G., Zhou, W., ... & Wei, F. (2022). Integrated index-based assessment reveals long-term conservation progress in implementation of Convention on Biological Diversity. *Science advances*, 8 (1), eabj8093

Islam, M.M. (2021). Social dimensions in designing and managing marine protected areas in Bangladesh. *Hum. Ecol*. 49(2), pp. 171–185

IUCN and UNEP-WCMC (2017). The World Database on Protected Areas (WDPA). UNEP-WCMC. Cambridge, UK. www.protectedplanet.net

Jantke, K., Jones, K.R., Allan, J.R., Chauvenet, A.L., Watson, J.E. and Possingham, H.P. (2018). Poor ecological representation by an expensive reserve system: evaluating 35 years of marine protected area expansion. *Conserv. Lett.* 11(6), e12584

Jetz, W., McGowan, J., Rinnan, D.S., Possingham, H.P., O'Donnell, P., O'Donnell, B. and Londoño-Murcia, M.C. (2022). Include biodiversity representation indicators in area-based conservation targets. Nat. Ecol. Evol. 6(2), pp. 123–126

Kedward, K., Zu Ermgassen, S., Ryan-Collins, J. and Wunder, S. (2023). Heavy reliance on private finance alone will not deliver conservation goals. *Nat. Ecol. Evol*., pp. 1-4

Klein, C.J., Brown, C.J., Halpern, B.S., Segan, D.B., McGowan, J., Beger, M. and Watson, J.E. (2015). Shortfalls in the global protected area network at representing marine biodiversity. *Sci. Rep.* 5(1), pp. 1–7

Knowlton, N. (2021). Ocean optimism: moving beyond the obituaries in marine conservation. *Annu. Rev. Mar. Sci.* 13, pp. 479–499

Kullberg, P., Di Minin, E. and Moilanen, A. (2019). Using key biodiversity areas to guide effective expansion of the global protected area network. *Glob. Ecol. Conserv.* 20, e00768

Laffoley, D., Dudley, N., Jonas, H., MacKinnon, D., MacKinnon, K., Hockings, M. and Woodley, S. (2017). An introduction to 'other effective area-based conservation measures' under Aichi Target 11 of the Convention on Biological Diversity: Origin, interpretation and emerging ocean issues. *Aquat. Conserv.* 27, pp. 130–137

Lagabrielle, E., Sabinot, C., Obura, D., Nyigulila, R., Mwaipopo, P.C., Bandeira, S., ... and Mbugua, J. (2018). Linking marine science, traditional knowledge and cultural perceptions of the sea in the Mozambique Channel to build tomorrow's marine management using spatial simulation tools and educational game. Final report developed by the Research Institute for Development (IRD, France) with the contributions from CORDIO (Kenya), University Eduardo Mondlane (Mozambique). https://gred.ird.fr/programmes-de-recherche/autres-projets/mozalink

Laikre, L., Hoban, S., Bruford, M.W., Segelbacher, G., Allendorf, F.W., Gajardo, G. ... and Vernesi, C. (2020). Post-2020 goals overlook genetic diversity. *Science* 367(6482), pp. 1083–1085

Leadley, P., Gonzalez, A., Obura, D., Krug, C. B., Londoño-Murcia, M. C., Millette, K. L., ... & Xu, J. (2022). Achieving global biodiversity goals by 2050 requires urgent and integrated actions. *One earth*, 5(6), 597–603.

Lengyel, S., Kobler, A., Kutnar, L., Framstad, E., Henry, P.Y., Babij, V. ... and Henle, K. (2008). A review and a framework for the integration of biodiversity monitoring at the habitat level. Biodivers. Conserv. 17(14), pp. 3341–3356

Lyons, M.B., Roelfsema, C.M., Kennedy, E.V., Kovacs, E.M., Borrego-Acevedo, R., Markey, K. ... and Murray, N. (2020). Mapping the world's coral reefs using a global multiscale earth observation framework. *Remote. Sens. Ecol.* 6(4), pp. 557–568

Mace, G.M., Barrett, M., Burgess, N.D., Cornell, S.E., Freeman, R., Grooten, M. and Purvis, A. (2018). Aiming higher to bend the curve of biodiversity loss. *Nature* 1(9), pp. 448–451

Maina, J.M., Gamoyo, M., Adams, V.M., D'agata, S., Bosire, J., Francis, J. and Waruinge, D. (2020). Aligning marine spatial conservation priorities with functional connectivity across maritime jurisdictions. *Conserv. Sci. Pract.*2(2), e156

Mair, L., Byers, O., Lees, C.M., Nguyen, D., Rodriguez, J.P., Smart, J. and McGowan, P.J. (2021). Achieving International Species Conservation Targets. *Conserv. Soc.* 19(1), pp. 25–33

Maire, E., Graham, N.A., MacNeil, M.A., Lam, V.W., Robinson, J.P., Cheung, W.W. and Hicks, C.C. (2021). Micronutrient supply from global marine fisheries under climate change and overfishing. *Curr. Biol.* 31(18), pp. 4132–4138

Mammides, C., Goodale, E., Elleason, M. and Corlett, R.T. (2021). Designing an ecologically representative global network of protected areas requires coordination between countries. *Environ. Res. Lett.* 16(12), 121001

Marinesque, S., Kaplan, D.M. and Rodwell, L.D. (2012). Global implementation of marine protected areas: is the developing world being left behind? *Mar. Policy* 36(3), pp. 727–737

Pahle, M., Schaeffer, R., Pachauri, S., Eom, J., Awasthy, A., Chen, W. ... and Verdolini, E. (2021). The crucial role of complementarity, transparency and adaptability for designing energy policies for sustainable development. *Energy Policy* 159, 112662

Palumbi, S.R. (2003). Population genetics, demographic connectivity, and the design of marine reserves. *Ecol. Appl.* 13(sp1), pp. 146–158

Petersson, M. and Stoett, P. (2022). Lessons learnt in global biodiversity governance. *Int. Environ. Agreem.: Politics Law Econ.* 22(2), pp. 333–352

Pressey, R.L., Visconti, P., McKinnon, M.C., Gurney, G.G., Barnes, M.D., Glew, L. and Maron, M. (2021). The mismeasure of conservation. *Trends Ecol. Evol.* 36(9), pp. 808–821

Reimer, J.M., Devillers, R. and Claudet, J. (2021). Benefits and gaps in area-based management tools for the ocean Sustainable Development Goal. *Nature* 4(4), pp. 349–357

Reyers, B. and Selig, E.R. (2020). Global targets that reveal the social-ecological interdependencies of sustainable development. *Nat. Ecol. Evol.* 4(8), pp. 1011–1019

Teixeira, H., Berg, T., Uusitalo, L., Fürhaupter, K., Heiskanen, A.S., Mazik, K., ... and Borja, À. (2016). A catalogue of marine biodiversity indicators. *Front. Mar. Sci.* 3, 207

Tittensor, D.P., Walpole, M., Hill, S.L., Boyce, D.G., Britten, G.L., Burgess, N.D. ... and Ye, Y. (2014). A mid-term analysis of progress toward international biodiversity targets. *Science* 346(6206), pp. 241–244

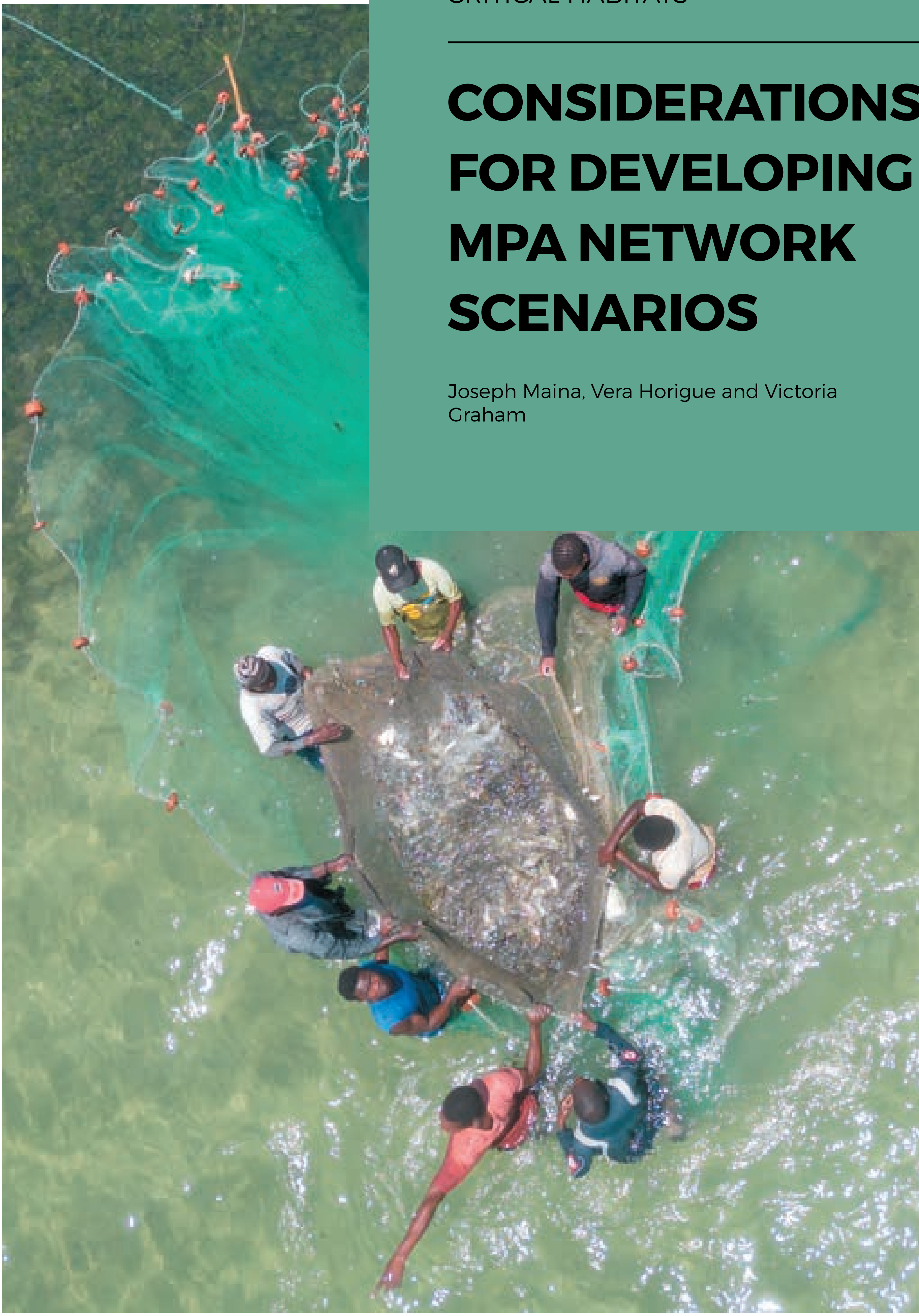
UNEP (2020). *Projections of Future Coral Bleaching Conditions using IPCC CMIP6 Models: Climate Policy Implications, Management Applications, and Regional Seas Summaries.* United Nations Environment Programme, Nairobi, Kenya. http://hdl.handle.net/20.500.11822/34219

UNEP (2021). United Nations Environment Programme, Regional Seas Biodiversity under the post-2020 Global Biodiversity Framework, Authors: D.E. Johnson, M.A. Ferreira, C. Barrio Froján, UNEP Regional Seas Working Paper, Nairobi. https://wedocs.unep.org/handle/20.500.11822/35102. (Accessed 15 April 2022).

UNEP-Nairobi Convention and WIOMSA (2020). Guidelines for Seagrass Ecosystem Restoration in the Western Indian Ocean Region. UNEP, Nairobi, 63 pp.
UNEP-Nairobi Convention and WIOMSA (2021). Western Indian Ocean Marine Protected Areas Outlook: Towards achievement of the Global Biodiversity Framework Targets. UNEP and WIOMSA, Nairobi, Kenya, 298 pp.
UNEP-WCMC (2021). Creating a Nature-Positive Future: The contribution of protected areas and other effective area-based conservation measures. UNDP, New York, USAVimal, R., Rodrigues, A.S., Mathevet, R. and Thompson, J.D. (2011). The sensitivity of gap analysis to conservation targets. Biodivers. Conserv. 20(3), pp. 531–543
Vimal, R., Rodrigues, A.S.L., Mathevet, R. and Thompson, J.D. (2011). The sensitivity of gap analysis to conservation targets. Biodivers. Conserv. 20, pp. 531–543
Watson, J.E., Darling, E.S., Venter, O., Maron, M., Walston, J., Possingham, H.P. ... and Brooks, T.M. (2016). Bolder science needed now for protected areas. Conserv. Biol. 30(2), pp. 243–248
Xu, H., Cao, Y., Yu, D., Cao, M., He, Y., Gill, M. and Pereira, H.M. (2021). Ensuring effective implementation of the post-2020 global biodiversity targets. Nat. Ecol. Evol. 5(4), pp. 411–418

CONSIDERATIONS FOR DEVELOPING MPA NETWORK SCENARIOS

Joseph Maina, Vera Horigue and Victoria Graham



Fishing with beach seine on a seagrass meadow at Maputo Bay, Mozambique. © David Mateus & José Paula

INTRODUCTION

As part of the global efforts to halt and reverse biodiversity loss and address the commitments to the Kunming Montreal Global Biodiversity Framework (hereafter GBF), there is a need to scale up conservation efforts. Towards the achievement of the goal and overall aspiration of harmony between humans and nature by 2050, it is essential that all regional, national, and local levels act in alignment on research and practice of conservation and development in order to overcome the past failures of conservation policies (Harrison et al., 2012). The emphasis of GBF Target 3, where, among others, the '30x30 target' was allocated, extends beyond the quantity of marine protected areas (MPAs) and stresses the need for quality in terms of effective management of protected areas as well as other conservation areas outside of protected areas. This is one of the main conclusions presented in the sister *MPA Outlook* to mitigate the risk of perpetuating the curse of 'paper parks'.

Driven by a broad 2050 vision of humans 'living in harmony with nature', the GBF aims to realize a world where biodiversity is valued, conserved, restored, and wisely used, maintaining ecosystem services, sustaining a healthy planet, and delivering benefits that are essential for all people. GBF's goals and targets encompass three key priorities: biodiversity conservation, sustainable use, and the equitable sharing of benefits arising from biodiversity, with a strong focus on developing indicators that meet the SMART (specific, measurable, ambitious, realistic, and time-bound) criteria (Adams et al., 2021; Green et al., 2019).

To inform these efforts, global research and development on ecological marine protected area networks (MPANs) predominantly occur at local to sub-national scales, such as lower government levels or finer scale ecological units like bays and gulfs, due to government jurisdictional limitations and pragmatic management and policy concerns (Harrison et al., 2012; Horigue et al., 2012; Abesamis et al., 2017). There are initiatives that have been able to do it on a national scale, such as the MPA estate expansion in Seychelles and South Africa, which took more than ten years to accomplish (Standing, 2023; Sink et al., 2023). Recent developments in conservation science show that increased protection of ecological processes in MPA designs (ie, size, spacing, and location) can ensure the persistence of biodiversity and support fisheries sustainability (Green et al., 2015; Magris et al., 2014). However, adequate representation of ecological processes within MPAs can be challenging because these processes often span larger (ie, >1000s of km) and multiple spatial scales (ie, local to global), transcending national boundaries (Mills et al., 2010; Fidelman et al., 2012). To address this, commonly referred recommendations include government cooperation, collaboration, and coordination to establish MPANs that can transcend jurisdictions and be nested within the different levels of the government organization (ie, local government, national government, regional associations) (Horigue et al., 2012; Levin et al., 2018; Chua, 2006).

A regional strategy is required to effectively implement GBF goals and targets, particularly for countries with high levels of social and ecological connectivity and localized ecosystems and species distribution. For the WIO countries, creating a unified implementation plan that aligns national strategies with the GBF would further enhance regional cooperation, streamline efforts, and facilitate the integration of social and ecological research findings into broader biodiversity conservation initiatives. By working together and integrating their respective national strategies, the WIO countries can contribute significantly to the broader vision of humans 'living in harmony with nature', valuing, conserving, restoring, and wisely using biodiversity to sustain a healthy planet and deliver benefits for all. MPANs are key to sustaining marine biodiversity and fisheries and ensuring the persistence of biodiversity in the face of climate change (IUCN-WCPA, 2008; Klein et al., 2015; Cabral et al., 2020).

Although the expansion through the establishment of more localized MPANs is a good start, it is necessary to scale up and align efforts, from local to national scales and eventually regional seas, to increase the protection of shared resources and oceans, manage boundary disputes, and increase the effectiveness of conservation by considering land-based and maritime activities (Levin et al., 2018; Abesamis et al., 2017; Horigue et al., 2012; Chua, 2006; Maina et al., 2020). The regional network of MPANs can also help strengthen the case for establishing region-specific biodiversity targets based on socio-economic needs and threats, among others. The establishment of regional MPANs could also facilitate the use and implementation of other spatial management tools to improve the management of shared seas and oceans and provide better safeguards against the increasing threat of climate change (Levin et al., 2018).

Establishing MPAs requires significant resources, technical expertise, and social capital among stakeholders, especially government institutions. Therefore, scaling up to form a regional MPAN would require countries to formulate concrete plans to develop national MPANs. Moreover, national governments will need to coordinate with neighbouring states to create synergies, address boundary disputes, and align development priorities with increasing the effectiveness of the regional network (Walton et al., 2014).

One of the UNEP-Nairobi Convention mandates is to support nations in the Western Indian Ocean (WIO) on biodiversity conservation and sustainable marine resource use in the region. This process includes tracking the progress of Contracting Parties towards achieving sustainable goals. In 2019, regional governments, through the UNEP-Nairobi Convention process, initiated a review of the progress made towards the achievement of the Convention of Biological Diversity (CBD) Aichi Target 11 and United Nations Sustainable Development Goals (ie, SDG 14). Under this process, a regional database of conservation areas in the WIO was established. It included an evaluation of MPA management performance using the Management Effectiveness Tracking Tool (Hockings et al., 2000; Stolton et al., 2019).

To establish a regional MPAN in the WIO, a systematic framework is necessary to organize and coordinate efforts. This systematic framework could be described as two major work streams. The first work stream sets targets for the individual WIO states to accomplish within their EEZ. In contrast, the second work stream requires concerted efforts by the WIO states to create a functional regional network. These work streams could help relevant government representatives within each country encourage other government officials to support and increase country-level efforts and encourage neighbouring states to increase government accountability and collaboration.

Despite the different capacities among WIO states, the general policy and technical recommendations revolve around: (i) improving the management and governance of existing MPAs; and (ii) increasing the area and quality of MPAs. Addressing these recommendations might involve establishing national MPANs that could organize relevant leaders, MPA managers, and technical resource persons to share information, coordinate efforts, and design an ecological network guided by a regional planning process. MPANs are nested structures with conservation properties at higher levels of organization (eg, species range extensions) that require substantial knowledge, planning and monitoring to become effective (Roff, 2014).

Establishing a functional ecological MPAN in each WIO country will require considerations of ecological design principles such as representativeness, connectivity, replication, and redundancy to ensure that individual MPAs within the network can act synergistically towards nature's wider-scale conservation goals and build resilience to current and anticipated anthropogenic impacts, including those of climate change (Fernandes et al., 2009; Grorud-Colvert et al., 2014).

THE IMPORTANCE OF CO-MANAGEMENT

Functional MPANs depend on effectively managed individual MPAs. Providing sufficient financial and human resources to sustain MPAs is a common problem for developing nations. Since government priorities often lean towards economic growth, financial support is usually allocated for infrastructure and other social development needs. Hence, as a way to address gaps in financial and human resource requirements, some developing countries also use participatory and co-management schemes between governments, communities and other stakeholders to support MPA management, including local marine management areas (LMMAs).

Integrating LMMAs more firmly in the process to improve management effectiveness at the local scale and build toward a national and eventual regional MPAN can be done by recognizing such community-led conservation initiatives and developing enabling legislation and policies to provide financial and technical support. These community initiatives could also be integrated within local and national-level MPA management plans. Involving communities and other stakeholders in MPA management will increase their awareness and understanding of the value of sustaining ecosystems and ecological processes, encourage stewardship, and increase compliance.

STRENGTHENING OF GOVERNMENT COMMITMENT AND COORDINATION

Identifying and addressing gaps in MPA planning in the WIO to develop functional MPAs requires political will, multidisciplinary information, coordinated action and time. The Nairobi Convention has recently developed the institutional structure and arrangements for developing and establishing regional MPAs. The regular regional fora and capacity-building initiatives organized by the UNEP, Nairobi Convention Secretariat and WIOMSA have also contributed to increasing social capital among WIO nations and government representatives. Moreover, some countries within the WIO have already engaged in bilateral agreements to share information and resources to support different management efforts. This provides opportunities to formalize further and align MPA initiatives across the region.

Regional MPAs will need to have different levels of organization and coordination. Apart from a functional ecological network, different governance and social networks will be required to coordinate efforts. These include forming networks of: (i) MPA managers, government officials, and scientists to exchange information and experiences; (ii) enforcers to conduct joint patrols; and (iii) MPA agency leaders and advocates from the governments to interface with other sectors and institutions to integrate the MPAN within broader planning and spatial management frameworks such as marine spatial planning (MSP).

Creating and formalizing these institutional structures and arrangements could lead to the development and implementation of ecological MPAs within each WIO country to properly represent and ensure the persistence of the region's biodiversity. In some cases, this may require retrofitting existing MPAs or changing them altogether. Following such national level improvements, the regional network could further develop other relevant initiatives and strategies to help sustain the national MPAs. This could include developing and implementing a systematic monitoring and evaluation framework to assess MPA management effectiveness in the region. Stressing the need for and importance of regular monitoring effectiveness of existing MPAs was one of the important recommendations of the *MPA Outlook* (see UNEP/Nairobi Convention and WIOMSA, 2021) and the recommendations from Tuda et al. (2022) from the WIO Science to Policy Platform Series.

The development of an effectively managed national MPAN that is overseen and implemented by institutions within existing governance structures from the regional to local levels will require the adoption of a regional and systematic MPA planning and implementation framework. Applying a systematic conservation planning approach will facilitate the development of the MPAN designs (ie, a spatial plan that identifies priority conservation areas) that adhere to ecological design principles and targets of the GBF and be attuned to social, economic, and political contexts and needs of stakeholders in the region.

The benefits of using the systematic conservation planning approach have been documented in the 10-fold increase of protected area coverage in South Africa (Sink et al., 2023). Learning from the experiences from South Africa and other parts of the world, developing systematic MPAN designs as part of mainstreaming GBF goals and targets, often takes time and requires the creation of new institutional arrangements or enhancing the alignment of existing ones to ensure that the resulting priority areas for conservation are established and implemented in a timely manner across the region to minimize further biodiversity loss. Identifying priority conservation areas that would be meaningful at the regional, national, and local scales, would require determining different MPAN scenarios that could result from the different governance and institutional arrangements. Using the scenarios as a guide, it could be used in stakeholder processes and discussions to facilitate the harmonization of regional and national priorities and potential interdependencies among countries for successful conservation outcomes.

As the first step in conceptualizing potential MPAN expansion scenarios to protect 30 per cent of critical habitats and other ecological features set by the GBF by 2030, we identified conservation planning objectives underpinned by an extensive review of the scientific literature and the GBF goals and targets (Table 1).

These objectives form part of the spatial prioritization framework to guide planning efforts of the UNEP-Nairobi Convention Contracting Parties, focusing on MPA expansion in the WIO region that supports progress towards the GBF goals and targets. The design approach is based on key ecological principles and best practices for MSP and MPAN plan development. These approaches are based on the precautionary principle and the need to achieve conservation and sustainable development goals. The objectives were separated into three categories: (1) Implementation and operations; (2) Ecological; and (3) Socio-economic. The process commences with writing clear statements of overarching goals that align with the national, regional or international conservation targets, followed by the development of specific, measurable objectives with associated indicators to use in the prioritization exercise. This list of generic considerations is not a definitive list of priorities, but a starting point with a recommendation to governments to recognize the need for a more comprehensive process of setting objectives with stakeholder involvement with representation from national and regional levels.

IMPLEMENTATION AND OPERATIONS

Under this category, promoting transboundary conservation is key for achieving ecological representativeness and promoting equity in conservation among WIO countries. Where necessary, conservation areas should transcend national boundaries.

Table 1: Setting the conservation priorities scenario.

CONSERVATION GOAL	INDICATOR(S) AND APPROACH
Implementation and operations	
Support transboundary (international) cooperation in the expansion of MPAs (Sala et al., 2021).	Distribute MPA and other marine conserved areas across all countries within the WIO EEZ
Protect ≥ 30 % of each country's marine area, with ≥ 15 % under strict protection (CBD-SBSTTA, 2020).	Capture ≥ 30 % of the EEZ of each country under some level of protection and ≥ 15 % under strict protection (no-take areas)
Ecological (Green et al., 2014)	
Habitat representation.	Represent ≥ 30 % of each marine ecoregion (eg, coral reefs, seagrass beds, mangroves) and seafloor habitat in management areas.
Replication to reduce risk.	Replicate protection of each major habitat within at least three reserves.
Protect special or unique habitats for key life stages and ensure connectivity between them (eg, spawning migrations).	Protect ≥ 30 % of larval sources and corridors; coral reef fish nurseries and coral reefs within 8 km of nurseries.
Protecting critical areas for rare and threatened species.	Protect 60 % of key biodiversity areas (KBAs) for flagship, endemic and endangered species (eg, turtles, dugongs, migratory birds) in no-take management zones.
Incorporating habitat connectivity.	Apply minimum and variable sizes to marine reserves, depending on focal species for protection, how far they move, and if other effective management is in place outside reserves.
Consider the ability of the network to support climate adaptation.	Protect a portfolio of areas predicted to experience varying levels of climatic risk (low and high) in marine reserves.
Manage key threats.	Minimize land and ocean-based threats within MPAs across land and seascapes (eg avoid placing reserves in areas that are heavily impacted by development threats).
Socio-economic	
Prioritize areas that are least impacted by current and projected coastal development (Green et al., 2014).	Separate reserves from coastal development, resource use conflicts (fishing, oil and gas, tourism).
Minimize human pressure on vulnerable habitats and/or ecosystems (Cinner et al., 2018; McClanahan et al., 2016)	Protect at least 50 % of areas that are more than 10 km from the shore and face lower relative human pressure using gravity of markets as a proxy.
Promote sustainable/ecotourism.	Maintain areas suitable for ecotourism (eg 10 %) with no-take restrictions.
Maintain long-term sustainability of deep-sea fish stocks (Green et al., 2014; Maina et al., 2020).	Set minimum distance (km) from aggregate fishing sites and sites with high catch rates.
Prioritize the protection of culturally significant areas (eg traditional fishing sites, coastal forests).	Prioritize the protection of culturally significant areas (eg traditional fishing sites, coastal forests).

A governance structure that allows transboundary arrangements should be in place to facilitate these. In terms of management, protecting 30 per cent of biodiversity features is one of the key targets in the GBF, which the WIO region needs to use as the goal moving forward to establish new conservation areas or retrofit existing ones where appropriate, all the while insisting on effective MPA management and/or community co-management depending on the case. In deciding on the targets to address, other country, and region-specific considerations, including transboundary opportunities, need to be considered. Furthermore, other effective area-based conservation measures (OECMS) in the region, which have evolved as LMMAs, will have a significant role in managing conservation areas in the region and globally.

A multi-objective approach that achieves the GBF targets while considering ecological and socio-economic benefits, among other uses, is necessary for considering this planning scenario.

ECOLOGICAL

Maintaining biodiversity for the future requires that all habitats are adequately protected and represented in the conservation area network. Their placement in space should be strategic to avoid or minimize a myriad of threats affecting critical habitats and ecosystems, including climate change and direct human threats. For example, using spatial attributes of functional connectivity, protected areas can be designed to take into account source and sink dynamics and species migratory corridors (Fontoura et al., 2022).

Moreover, recent studies have proposed that assessments of potential benefits or avoided biodiversity loss be the prerequisite for optimizing conservation areas, as opposed to the common strategy of retrospective evaluation of MPA effectiveness in conserving biodiversity. In addition, replicating the protection of each critical habitat within at least three reserves – or MPAs closed off to fishing and other activities may help distribute the risks. Optimizing conservation area selection for ecosystem persistence would also require protecting ecological processes such as larval connectivity and seascape connectivity across critical habitats.

Evidence-based and ecosystem-based should be some of the guiding principles for ecological goals. Other considerations are protecting threatened species, genetic diversity and climate resilience.

SOCIO-ECONOMIC

Socio-economic considerations are an integral part of selecting conservation areas in the region, both from resource use and minimizing threats perspectives. Conservation areas should be spatially configured to avoid present and future coastal developments to minimize conflict and promote compliance and effectiveness.

An important aspect of socio-economic considerations is minimizing human pressure on vulnerable ecosystems. This can be achieved by protecting marine areas with lower relative human pressure. On the other hand, protecting areas of high human pressure might be an excellent strategy to distribute effort evenly in marine areas (Maina et al., 2015).

Another key aspect is integrating socio-economic considerations. Our scenario also includes socio-economic considerations to identify locations that can provide maximum socio-economic benefits from small-scale fisheries. Equitable sharing of benefits arising from biodiversity is one of the three GBF priorities. In certain instances, this can be achieved by locating MPAs and LMMAs in larval sink areas and protecting the larval source and connectivity corridors, among other considerations (Fontoura et al., 2022).

Other socio-economic considerations include promoting sustainable/eco-tourism, maintaining long-term sustainability of deep-sea fish stocks, and prioritizing the protection of culturally significant areas (eg, traditional fishing sites and coastal forests).

REFERENCES

Abesamis, R.A., Saenz-Agudelo, P., Berumen, M.L., Bode, M., Jadloc, C.R.L., Solera, L.A., ... and Russ, G.R. (2017). Reef-fish larval dispersal patterns validate no-take marine reserve network connectivity that links human communities. *Coral Reefs* 36(3), pp. 791–801

Adams, V.M., Visconti, P., Graham, V. and Possingham, H.P. (2021). Indicators keep progress honest: A call to track both the quantity and quality of protected areas. *One Earth* 4(7), pp. 901–906

Cabral, R.B., Bradley, D., Mayorga, J., Goodell, W., Friedlander, A.M., Sala, E., ... and Gaines, S.D. (2020). A global network of marine protected areas for food. *Proc. Natl. Acad. Sci.* 117(45), pp. 28134–28139

CBD-SBSTTA (2020). *Zero draft of the post 2020 global biodiversity framework*. Convention on Biological Diversity, Subsidiary Body on Scientific, Technical and Technological Advice. Kunming, China

Chua, T. (2006). *The Dynamics of Integrated Coastal Management: Practical Applications in the Sustainable Coastal Development in East Asia*. Global Environmental Facility/ United Nations Development Programme/ International Maritime Organization Regional Programme on Building Partnerships in Environmental Management for the Seas of East Asia (PEMSEA)

Cinner, J.E., Maire, E., Huchery, C., MacNeil, M.A., Graham, N.A., Mora, C., ... and Mouillot, D. (2018). Gravity of human impacts mediates coral reef conservation gains. *Proc. Natl. Acad. Sci.* 115(27), pp. E6116–E6125

Fernandes, L., Day, J., Kerrigan, B., Breen, D., De'ath, G., Mapstone, B., ... and Kenchington, R. (2009). A process to design a network of marine no-take areas: Lessons from the Great Barrier Reef. *Ocean Coast. Manag.* 52(8), pp. 439–447

Fidelman, P., Evans, L., Fabinyi, M., Foale, S., Cinner, J. and Rosen, F. (2012). Governing large-scale marine commons: contextual challenges in the Coral Triangle. *Mar. Policy* 36(1), pp. 42–53

Fontoura, L., D'agata, S., Gamoyo, M., Barneche, D.R., Luiz, O.J., Madin, E.M., Eggertsen, L. and Maina, J.M. (2022). Protecting connectivity promotes successful biodiversity and fisheries conservation. *Science* 375(6578), pp. 336–340

Green, A.L., Fernandes, L., Almany, G., Abesamis, R., McLeod, E., Aliño, P.M., ... and Pressey, R.L. (2014). Designing marine reserves for fisheries management, biodiversity conservation, and climate change adaptation. *Coast. Manag.* 42(2), pp. 143–159

Green, A.L., Maypa, A.P., Almany, G.R., Rhodes, K.L., Weeks, R., Abesamis, R.A., ... and White, A.T. (2015). Larval dispersal and movement patterns of coral reef fishes, and implications for marine reserve network design. *Biol. Rev.* 90(4), pp. 1215–1247

Green, E.J., Buchanan, G.M., Butchart, S.H., Chandler, G.M., Burgess, N.D., Hill, S.L. and Gregory, R.D. (2019). Relating characteristics of global biodiversity targets to reported progress. *Conserv. Biol.* 33(6), pp. 1360–1369

Grorud-Colvert, K., Claudet, J., Tissot, B.N., Caselle, J.E., Carr, M.H., Day, J.C., ... and Walsh, W.J. (2014). Marine protected area networks: assessing whether the whole is greater than the sum of its parts. *PloS one* 9(8), e102298

Harrison, H.B., Williamson, D.H., Evans, R.D., Almany, G.R., Thorrold, S.R., Russ, G.R., ... and Jones, G.P. (2012). Larval export from marine reserves and the recruitment benefit for fish and fisheries. *Curr. Biol.* 22(11), pp. 1023–1028

Hockings, M., Stolton, S. and Dudley, N. (2000). *Evaluating Effectiveness: A Framework for Assessing the Management of Protected Areas*. IUCN

Horigue, V., Aliño, P.M., White, A.T. and Pressey, R.L. (2012). Marine protected area networks in the Philippines: Trends and challenges for establishment and governance. *Ocean Coast. Manag.* 64, pp. 15–26

IUCN-WCPA (2008). *Establishing Marine Protected Area Networks – Making it Happen.* IUCN-WCPA, National Oceanic and Atmospheric Administration and The Nature Conservancy. 118 p. Washington, D.C.

Klein, C.J., Brown, C.J., Halpern, B.S., Segan, D.B., McGowan, J., Beger, M. and Watson, J.E. (2015). Shortfalls in the global protected area network at representing marine biodiversity. *Sci. Rep.* 5(1), pp. 1–7

Levin, N., Beger, M., Maina, J., McClanahan, T. and Kark, S. (2018). Evaluating the potential for transboundary management of marine biodiversity in the Western Indian Ocean. *Australas. J. Environ. Manag.* 25(1), pp. 62–85

Magris, R.A., Pressey, R.L., Weeks, R. and Ban, N.C. (2014). Integrating connectivity and climate change into marine conservation planning. *Biol. Conserv.* 170, pp. 207–221

Maina, J.M., Jones, K.R., Hicks, C.C., McClanahan, T.R., Watson, J.E., Tuda, A.O. and Andréfouët, S. (2015). Designing climate-resilient marine protected area networks by combining remotely sensed coral reef habitat with coastal multi-use maps. *Remote Sensing* 7(12), pp. 16571–16587

Maina, J.M., Gamoyo, M., Adams, V.M., D'agata, S., Bosire, J., Francis, J. and Waruinge, D. (2020). Aligning marine spatial conservation priorities with functional connectivity across maritime jurisdictions. *Conserv. Sci. Pract.* 2(2), e156

McClanahan, T.R., Maina, J.M., Graham, N.A. and Jones, K.R. (2016). Modeling reef fish biomass, recovery potential, and management priorities in the Western Indian Ocean. *PLoS one* 11(5), e0154585

Mills, M., Pressey, R.L., Weeks, R., Foale, S. and Ban, N.C. (2010). A mismatch of scales: challenges in planning for implementation of marine protected areas in the Coral Triangle. *Conserv. Lett.* 3(5), pp. 291–303

Roff, J.C. (2014). Networks of marine protected areas–the demonstrability dilemma. *Aquat. Conserv.* 24(1), pp. 1–4

Sala, E., Mayorga, J., Bradley, D., Cabral, R.B., Atwood, T.B., Auber, A., ... and Lubchenco, J. (2021). Protecting the global ocean for biodiversity, food and climate. *Nature* 592(7854), pp. 397–402

Sink, K.J., Lombard, A.T., Attwood, C.G., Livingstone, T.-C., Grantham, H. and Holness, S.D. (2023). Integrated systematic planning and adaptive stakeholder process support a 10-fold increase in South Africa's Marine Protected Area estate. *Conserv. Lett.* n/a, e12954. https://doi.org/10.1111/conl.12954

Standing, A. (2023). The Financialization of Marine Conservation: The Case of Debt-for-Ocean Swaps. *Development*, pp. 1–12

Stolton, S., Dudley, N., Belokurov, A., Deguignet, M., Burgess, N.D., Hockings, M., ... and Young, L. (2019). Lessons learned from 18 years of implementing the management effectiveness tracking tool (METT): A perspective from the METT developers and implementers. *Parks* 25(2), pp. 79–92

Tuda, A., Horigue, V., Francis, J., Bosire, J. and Maina, J. (2022). Strengthening the WIO MPA System (WIOMPAS): institutionalising performance tracking and expansion of conservation areas through multi-level cooperation. *WIO Science to Policy Series* 1(1), pp. 131–138

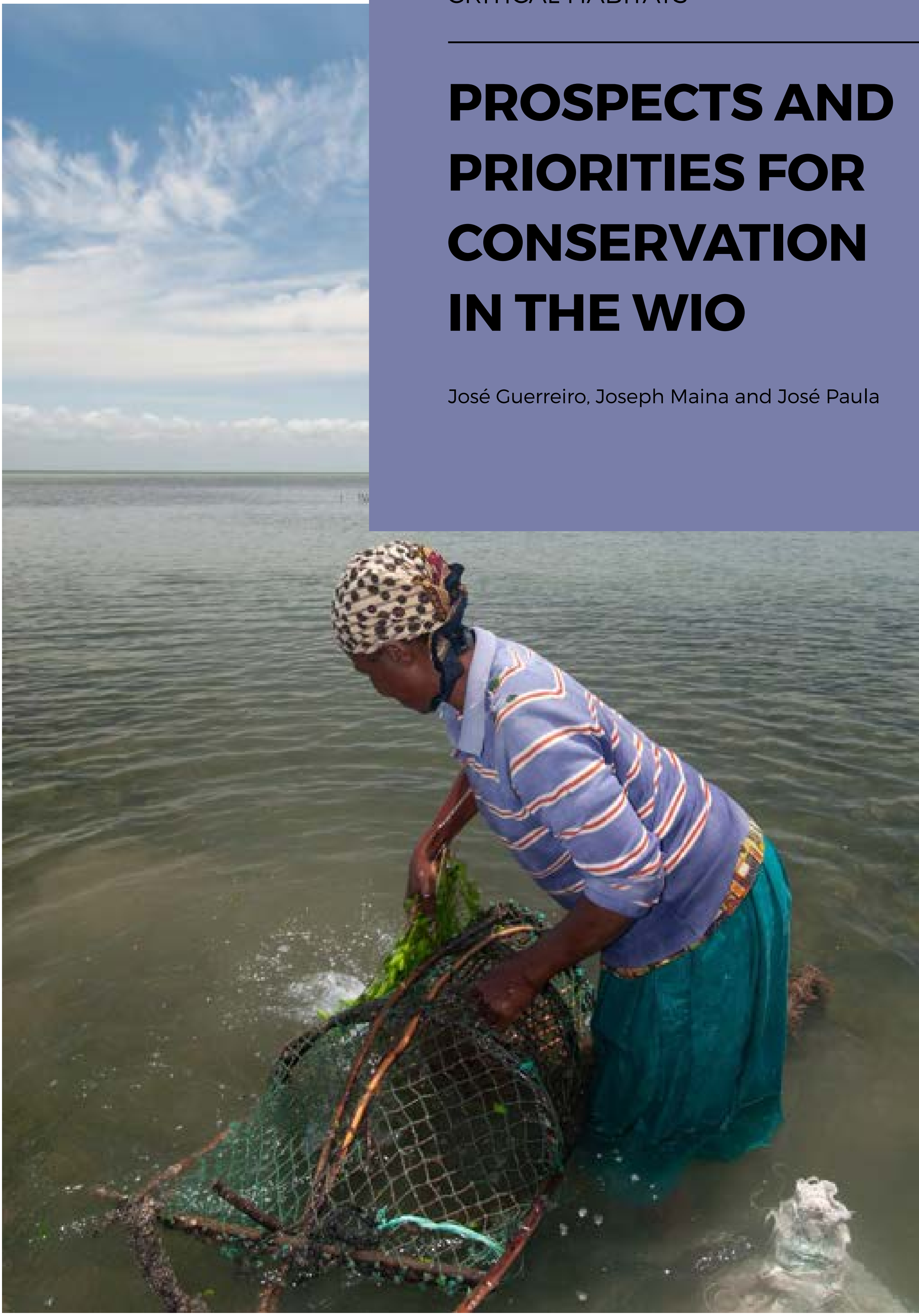
UNEP/Nairobi Convention and WIOMSA (2021). *Western Indian Ocean Marine Protected Areas Outlook: Towards achievement of the Global Biodiversity Framework Targets*. UNEP and WIOMSA, Nairobi, Kenya, 298 pp.

Walton, A., White, A.T., Tighe, S., Aliño, P.M., Laroya, L., Dermawan, A., ... and Green, A.L. (2014). Establishing a functional region-wide coral triangle marine protected area system. *Coast. Manag.* 42(2), pp. 107–127

CRITICAL HABITATS

PROSPECTS AND PRIORITIES FOR CONSERVATION IN THE WIO

José Guerreiro, Joseph Maina and José Paula



Fisherwoman with traps for fish and crabs at Inhaca Island, Mozambique. © José Paula

THE GLOBAL CONTEXT

At the start of the new millennium, the United Nations Organization (UN), through the General Assembly resolutions 57/141 and 58/240 (UN, 2003, 2004), established a regular process for global reporting and assessment of the state of the marine environment, including socio-economic aspects. This reporting, prepared by a broad group of international experts, is a landmark assessment of the global oceans' environmental status, contributing to documenting the achievement of the SDG 14 Targets. The reports (UN, 2017a; UN, 2021) state that the main threats to marine ecosystems, habitats and species come from human activities, such as fishing, aquaculture, shipping, sand and mineral extraction, oil and gas exploitation, building of renewable energy infrastructure, coastal infrastructure development and pollution, including the release of greenhouse gases.

Notwithstanding some increases in marine protected areas (MPAs) and the expansion of Ramsar Sites, mangroves (except in the Red Sea) and seagrass meadows, particularly in South-East Asia, continue to decline, with 19 per cent of mangrove and 21 per cent of seagrass species identified as Near-Threatened. The combined effects of ocean warming and human activities increasingly affect tropical and subtropical coral reefs and kelp forests globally. In recent years, coral reefs have undergone mass bleaching on an annual basis, while kelp forests have been affected by marine heat waves, resulting in rapid losses. Overall, about 6 per cent of known fish species and nearly 30 per cent of elasmobranch species are listed as Near-Threatened or Vulnerable. Globally, the status of marine mammals varies, with 75 per cent of species in some groups (sirenians, freshwater dolphins, polar bears and otters) being classified as Vulnerable, Endangered or Critically Endangered. Many large whale species are now recovering from past harvesting and benefiting from national recovery plans. The conservation status of marine reptiles varies greatly from region to region, according to the effectiveness of protection and conservation measures. The global conservation status of seabirds has worsened, with over 30 per cent of species now listed as Vulnerable, Endangered or Critically Endangered. Finally, communities associated with deep-sea features such as seamounts, pinnacles, ridges, trenches, hydrothermal vents and cold seeps remain under threat from fishing, offshore oil and gas exploration and development drilling, deep-sea mining and pollution, including plastic waste, and, to a lesser extent, climate change.

Responding to the need for increased marine conservation by 2020, MPAs cover 18 per cent of the ocean within national jurisdictions, representing approximately 8 per cent of the world's oceans. However, of marine areas beyond national jurisdiction, only about 1 per cent had been protected.

Understanding the distribution and status of species and habitats, how these are being affected by anthropogenic pressures, as well as addressing the gaps of knowledge, is critical for identifying conservation priorities and actions to achieve SDG's goals and Global Biodiversity Framework (GBF) targets. Although these instruments are reported on a national level, regional strategies and coordination are essential for successful biodiversity conservation and socioeconomic development. Such is the purpose of this volume, which focuses on one of the most important marine biodiversity regions of the world – the Western Indian Ocean (WIO). In cross-referencing the WIO MPA Outlook, it is necessary to develop a WIO regional Biodiversity Framework to serve as a roadmap for implementing the GBF commitments at the regional and national levels.

The Blue Economy and a new Ocean Governance Strategy

Recent developments and innovations in technology, particularly in the 21st century, have introduced several new demands on the maritime space, such as deep-sea mining, offshore renewable energies (particularly wind farms), offshore aquaculture and biotechnology, putting pressure on the ocean space, as well as increased potential conflicts between multiple interests. The concepts of 'Blue Growth' and 'Blue Economy' have grown worldwide, particularly following the Rio+20 Conference, where ocean governance and the blue economy were formally discussed and the subject of several side events.

These topics are now of the highest relevance to the management of the oceans (Campbell et al., 2013). Even during the preparatory meetings, the issue of the blue economy was formally debated, namely at the 2nd preparatory meeting in March 2011 (Intergovernmental Oceanographic Commission of United Nations Educational Scientific and Cultural Organization IOC–UNESCO, 2011). Moreover, at that same meeting, Pacific Small Island Developing States (SIDS), suggested the adoption of the blue economy as the approach that would best defend their development interests instead of the 'Green Economy' (which was central to the Rio+20 Conference). This thesis would make its way and be consolidated at the third SIDS conference in Apia, Samoa, on September 3, 2014, concluding that *"(…) Sustainable fisheries and aquaculture, coastal tourism, the possible use of seabed resources* *and renewable energy are among the main sectors of a sustainable ocean economy in small island developing states"* (UN, 2014). Within a few years, blue growth became a concept and policy approach all over the world, from Africa to Asia, reflecting a reinvention of maritime and marine governance and a redesign of legal and institutional frameworks, paving the way for integrated maritime policies, national ocean strategies, blue growth strategies and new instruments for maritime spatial planning (MSP), whilst also helping to reshape institutional and government frameworks (Guerreiro, 2021).

Over the last decade, new challenges for marine conservation have arisen, because blue growth strategies and integrated maritime policies can also be instruments to promote the conservation and restoration of habitats. It is widely recognized that MSP is commonly misconceived as being more of a technical tool focused on space allocation (zoning) and not on good governance (Flannery et al., 2018), lacking a real ecosystem approach by favouring economic priorities, within a concept of 'soft sustainability' (Frazão-Santos et al., 2014a, 2014b). Thus, the present global challenge is how to reconcile and use blue growth strategies and the maritime economy while reconciling the growing demands for increased and improved marine conservation, including in areas beyond national jurisdictions (ABNJ), where economic pressures are increasingly growing and international management instruments are slow to respond. The GBF's aspiration of 'harmony between human and nature' is strongly aligned with the ideals of blue growth and can provide guidance for the development of a sustainable ocean-based economy that safeguards natural integrity.

Responding to the global challenge – The UN Decade of Ocean Science for Sustainable Development and the Global Biodiversity Framework

Aware of this global evolution and the contrasting and conflicting demands on marine resources, the UN declared the decade 2021–2030 the Decade of Ocean Science for Sustainable Development (UN, 2017b), aiming to produce "*the science we need for the ocean we want*," by facilitating the generation of data, information and knowledge needed to catalyze transformative ocean science solutions for sustainable development, matching the SDGs and particularly SDG 14: Life below water. According to the Implementation Plan (UNESCO-IOC, 2021), from the seven outcomes foreseen, we must highlight, in the context of this volume, *"A healthy and resilient ocean where marine ecosystems are understood, protected, restored and managed."*

Furthermore, from the ten challenges identified, representing the highest level of the Ocean Decade Action Framework, Challenge 2 aims to "*Understand the effects of multiple stressors on ocean ecosystems, and develop solutions to monitor, protect, manage and restore ecosystems and their biodiversity under changing environmental, social and climate conditions.*"

Accordingly, the three objectives are: (i) Identify required knowledge for sustainable development and increase the capacity of the ocean; (ii) Build capacity and generate comprehensive knowledge and understanding of the ocean, including human interactions and interactions with the atmosphere, cryosphere and the land-sea interface; (iii) Increase ocean knowledge and understanding, and develop capacities to contribute to sustainable development solutions.

Complimentary to the UN Ocean Decade goals, the Convention on Biological Diversity (CBD) within COP 15[1] held in 2022, decided that the Kunming-Montreal Global Biodiversity Framework should be used as a strategic plan for the implementation of the Convention and its Protocols, over the period 2022–2030 reinforcing ocean conservation through Target 3 stating that *"... by 2030 at least 30 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem functions and services, are effectively conserved and managed through ecologically representative, well-connected and equitably governed systems of protected areas and other effective area-based conservation measures ... while ensuring that any sustainable use, where appropriate in such areas, is fully consistent with conservation outcomes, recognizing and respecting the rights of indigenous peoples and local communities ...".*

Furthermore, the UN also declared 2021–2030 as the Decade for Ecosystem Restoration (UN, 2019), aiming to prevent, halt and reverse the degradation of ecosystems on every continent and in every ocean, with a particular focus on oceans and coasts, with several programmes on-going. For the first time in the history of global conservation policy, GBF explicitly included restoration among its targets, calling on parties to implement effective restoration of at least 30 per cent of degraded terrestrial, inland water, and coastal/marine ecosystems by 2030 (Target 3).

In addition, the need for transboundary and cross-maritime conservation is highlighted in various conservation policies, including SDG 6 and GBF Goal A on connectivity.

1. Available at https://www.cbd.int/doc/c/e6d3/cd1d/daf663719a03902a9b116c34/cop-15-l-25-en.pdf (Accessed 29.1.23)

These complimentary initiatives that aim at ocean protection recognize the United Nations Convention on the Law of the Sea (UNCLOS) as the main umbrella instrument and its critical role in the most complex problem to be solved in this decade, which is the protection of the marine environment and biodiversity beyond national jurisdictions.

The 2022 UN Ocean Conference stressed this challenge by recognizing the work being undertaken by the intergovernmental conference on an international legally binding instrument, under UNCLOS, on the conservation and sustainable use of marine biological diversity of ABNJ and called on participating delegations to reach an ambitious agreement without delay.[2] This is the major challenge, to achieve the 30 per cent national target and significantly increase the previous 1 per cent of MPAs in ABNJ.

THE AFRICAN CONTEXT

Africa, with 38 coastal states and seven SIDS and 47 African States being Parties of UNCLOS, encompasses vast, highly productive oceanic and marine environments from the Atlantic to the Indian Ocean, the Red Sea and the Mediterranean, extending around approximately 30 000 km of coastline and over 10.5 million km^2 of Exclusive Economic Zones (EEZs). The diverse aquatic ecosystems and marine resources represent, on one hand, abundant opportunities for the new blue economy and, on the other hand, an increased challenge to tackle ecosystem degradation and habitat loss. Ensuring sustainable ocean and coastal development needs to be a non-negotiable condition for the economic development scenario. The main African umbrellas for marine conservation are the 1969 (revised in 2017) African Convention on the Conservation of Nature and Natural Resources[3] and three regional seas conventions (Abidjan, Nairobi and Jeddah) that provide a regional, legal and coordinated framework for strengthening country capacity to protect, manage and develop their coastal and marine environments. It is also important to refer to the Lomé Charter[4] which, besides addressing security and safety at sea, also aims to protect the maritime environment in coastal and island states and strengthen cooperation.

[2.] See https://sdgs.un.org/sites/default/files/2022-06/UNOC_political_declaration_final.pdf (Accessed 31st January 2023)

[3.] See https://au.int/en/treaties/african-convention-conservation-nature-and-natural-resources

[4.] See https://au.int/en/treaties/african-charter-maritime-security-and-safety-and-development-africa-lome-charter

Not surprisingly, Africa was one of the first regions in the world to embrace the concept of Integrated Maritime Policy and Blue Growth under the auspices of the African Union (AU). Adopted in 2014, the Africa Integrated Maritime Strategy 2050 (AIMS 2050[5]) introduced an innovative concept, the Combined Exclusive Maritime Zone of Africa (CEMZA) (Egede, 2023), aiming to foster increased wealth creation by developing a sustainable, thriving blue economy in a secure and environmentally sustainable manner. This vision was also incorporated in the Agenda 2063 (African Union Commission, 2015) within Aspiration 1: *"A prosperous Africa based on inclusive growth and sustainable development, considering that "Africa's Blue/ocean economy, …shall be a major contributor to continental transformation and growth, through knowledge on marine and aquatic biotechnology, the growth of an Africa-wide shipping industry, the development of sea, river and lake transport and fishing; and exploitation and beneficiation of deep-sea mineral and other resources".*

Accordingly, the Africa Blue Economy Strategy (ABES) approved in 2019 (African Union Commission, 2019) prioritizes five critical blue economy sectors:
1. Fisheries, aquaculture and ecosystems conservation.
2. Shipping, transportation and trade.
3. Sustainable energy, extractive minerals, gas, innovative industries.
4. Environmental sustainability, climate change and coastal infrastructure.
5. Governance.

ABES appeals to Member States to (i) value blue ecosystem services, (ii) consider institutionalizing Large Marine Ecosystem (LME) and Watershed Approach (WSA) for the assessment of the changing state of the ecosystem, and (iii) institutionalize MSP to allocate blue ecosystems' conservation. Several states are already developing MSP, particularly in the WIO, such as Mozambique and South Africa, both of which are also developing transboundary projects that utilize MSP for marine conservation (Guerreiro, 2022).

Finally, the roadmap for Africa Ocean Decade (UNESCO-IOC, 2022) included the Regional Consultation for Africa and the Adjacent Island States (the Nairobi Consultation), held in Nairobi, Kenya, in early 2020, the preparation of a Regional Gap Analysis in 2021, and the organization of a series of online multi-stakeholder workshops in early 2022, culminating in the identification of nine priority future Decade Actions. Pursuing the challenge to protect and restore ecosystems, the gap analysis identified:

[5.] Available at https://au.int/en/documents/20130225/2050-aim-strategy (Accessed in 31.1.23)

(i) insufficient fundamental knowledge/research on species diversity and taxonomy; (ii) insufficient understanding of ecosystem functions and services supported by different ecosystems at the scale required by relevant management, and (iii) limited mapping of marine and coastal ecosystems (eg for MPAs). Several programmes are already addressing these gaps, such as the *Challenger 150 – A Decade to Study Deep-Sea Life*[6] involving active members of the Indian Ocean regional group, namely Comoros, Kenya, Mauritius, France, Seychelles, South Africa and Tanzania.

Although the focus of these regional research initiatives seems to be centred on blue economy and growth, it is clear that ecosystem conservation is also a political priority, particularly with respect to mangrove and coral conservation and restoration, and the management of fisheries (including combating illegal, unreported and unregulated [IUU] fisheries), with several projects ongoing in the WIO region.

THE WIO REGION – OPTIONS AND CHALLENGES

The WIO region is a wide territory with major and complex geographical and environmental gradients. It encompasses diverse tropical and subtropical regions, rich stretches of coast along the mainland countries of Somalia, Kenya, Tanzania, Mozambique and South Africa, and vast oceanic areas surrounding the island states of Madagascar, Seychelles, Comoros, Mauritius and the French Territories. The complexity and wide geographical span of the WIO region create environmental gradients and contrasts, providing the basis for compartmentalization and regionalization based on different criteria and classification schemes. The vastness of the WIO and its complex oceanographic dynamic creates a biophysical mosaic of coastal and offshore environments and habitats that overlap temperate and tropical climatic zones.

Major marine coastal and oceanic environments exist in the WIO, and critical habitats such as mangrove forests, seagrass beds and coral reefs cover most of the region's coasts, enabling high levels of endemic biodiversity and supporting the livelihoods of millions. The vastness of the offshore ABNJ and largely unexplored deep-sea sediments provide potentially lucrative grounds for intensive industrial fisheries and deep-sea extraction of non-living resources.

6. See https://oceandecade.org/actions/challenger-150-a-decade-to-study-deep-sea-life/

Priorities, weaknesses and challenges for marine conservation

The highly diverse marine biodiversity of the WIO has provided a rich source of food security and livelihoods and has been a source of natural wonder for the people in the region. However, growing threats, from overexploitation of natural resources to climate change impacts and pollution, heavily impact marine ecosystems. Furthermore, many of the countries in the WIO are characterized by high population growth rates, high coastal population density, and substantial rural to urban migrations, increasing threats to the sustainable utilization of coastal resources (UNEP-Nairobi Convention and WIOMSA, 2015).

According to the IUCN 2021 regional assessment of the conservation status of marine biodiversity in the wider WIO (Bullock et al., 2021), 473 species were identified as Threatened (T) or Near Threatened (NT) and between 7–24 per cent of all species were estimated as being currently at risk of extinction, with a best estimate of 8 per cent. Major threats were found to be the over-use of biological resources, largely in the form of targeted fisheries and bycatch, as well as IUU fisheries. Overexploitation was flagged as a driver of population decline for all Threatened and Near Threatened cartilaginous fishes, mammals and sea turtles. The 237 Threatened and Near Threatened reef-building corals are impacted by the same suite of fishing threats, including fisheries-related habitat degradation. In general, habitat degradation and destruction through pollution, coastal development and other habitat modifications emerged as major threats.

Overall, the conservation status of marine species in the WIO region is moderately high relative to the status of the same taxonomic suite of species assessed in other regions, with a best estimate of 8 per cent threatened species. There is, however, a high level of uncertainty with respect to species status (16.9 per cent Data Deficient) compared to other tropical regions (11.0–15.8 per cent). The IUCN 2021 regional assessment report points out the need to identify Key Biodiversity Areas (KBAs) and the subsequent establishment of MPA networks that allow for high degrees of connectivity between sites, together with community-based management strategies establishing and maintaining small, no-take MPAs. Raising public awareness among the different stakeholders on the value of biodiversity was also (and still is) considered critical for the success of conservation measures.

In this regional context, the present *Critical Habitats Outlook* brings new light on the priorities, weaknesses and challenges for habitat and species conservation in the WIO. The findings highlight that mangroves and coral reefs are critical habitats, not only because of the high levels of biodiversity and ecosystems goods and services they provide but also for their socioeconomic relevance to countries, particularly for local communities. Both ecosystems are present along most shorelines of the WIO region, with mangroves covering 6200 km^2 (620 000 ha), about 25 per cent of the African mangrove area and 4.1 per cent of the world's mangrove area, while coral reefs cover about 11 980 km^2, equivalent to 35 per cent of the inshore coastal habitats of East African mainland states.

Mangroves are threatened by habitat destruction, land reclamation and overexploitation of natural resources, namely wood. Mapping forests and vulnerable areas, assessing threats at local scales, including strengthening land use to ensure mangroves have space to develop inland in response to sea level rise, and implementing mangrove restoration programs involving local communities, are critical measures to be implemented. Although coral reefs are the target of most MPAs, there is a need to improve management effectiveness and zoning considering representativeness, adequacy, and irreplaceability, even more so when blue growth exerts growing pressure on these areas, and particularly with climate change deeply impacting corals.

To address the threats and shortcomings, a combination of the above actions and the following is recommended: (i) improving the effectiveness of fisheries; (ii) evaluating and promoting alternative and sustainable livelihoods, and (iii) formalizing and operationalizing co-management of small-scale fisheries. These actions and measures demand a new approach to coastal governance, using MSP and integrated coastal zone management (ICZM), while also enhancing outreach and stakeholder engagement, research, and monitoring.

Seagrass beds, estuaries, salt marshes, rocky and sandy coasts and coastal forests are also of concern. Seagrass beds, although constituting a key component of inshore ecosystems, have comparatively received less scientific attention, and their comprehensive mapping is far from being completed (see Chapter 8). Major threats were identified as resulting from human activity, namely: (i) fishing activities, particularly the use of beach seines and trawls by artisanal fishers over seagrass beds; (ii) collection of invertebrates involving digging and revolving large amounts of sediments; (iii) eutrophication because of excessive nutrient input into coastal waters; and (iv) physical destruction related to water-based leisure activities. Although some of these areas are under MPA regimes, there is a need to identify priority areas for conservation and specific measures to enhance seagrass protection and restoration. The establishment of a Regional Seagrass Task Force would facilitate mainstreaming seagrass conservation within WIO countries, boost public awareness, and increase research and monitoring.

Estuaries are among the most productive natural systems of the world, often being part of complex ecosystems that include critical habitats such as mangroves, as well as seagrass beds, salt marshes and extensive tidal flats. Historically, river estuaries have attracted the settlement of human communities, creating socio-ecological systems that have developed into most of the world's largest coastal cities, with the inherent consequences, namely pollution resulting from untreated wastewater, dumping, industrial and agriculture pollution, often resulting in eutrophication and ecosystem degradation. The construction of dams in catchment basins and the alteration of water flows have contributed to the environmental degradation of many estuaries in the WIO region. Dealing with the threats of such complex socio-ecological systems demands integrated approaches at the level of all river basins and, mainly, the integration of environmental considerations into development politics, planning and design.

Salt marshes occur mainly in temperate South African shores or at the landward end of mangroves. They are very productive ecosystems serving as refugees and habitats for juvenile and migratory fishes and birds. The main threats are linked with land reclamation for agriculture, urbanization, evaporation salt plans, aquaculture, and livestock grazing. Recommendations for salt marsh conservation involve formal protection status for several estuaries and associated salt marshes, namely in South Africa, as well as active restoration actions. Buffer areas should also be considered to allow for the landward expansion of salt marshes in response to sea level rise, including removing hard structures where necessary.

Rocky and sandy shores are among the most biophysically dynamic marine environments in the transition coast-sea zone, and although providing a myriad of microhabitats and niches for organisms and breeding and nursery areas for many species, they suffer a wide range of threats, mainly related to human activities in coastal areas. Those threats include over-exploitation of resources, habitat disturbance, pollution, coastal development, and urbanization. Furthermore, they are among the habitats most vulnerable to the rise of sea level. The main recommendations include increasing the coverage of MPAs and locally managed marine areas (LMMAs) within each national jurisdiction, accompanied by improved management of coastal activities and river basins.

Coastal forests are usually linked to sandy soils, typically occurring above the high-tide mark and play a critical role in reducing soil and coastal erosion and by mitigating potential siltation and nutrient discharge. Trends in these important coastal ecosystems show a huge area decrease and fragmentation due to agricultural demand, uncontrolled charcoal production and fuelwood collection; unsustainable and illegal logging; uncontrolled fires; unplanned human settlement and urbanization; and destructive mining practices. Integrated management, both coastal and at the river basin scale, should be a priority, preferably through multi-stakeholder engagement, bringing together governments, non-governmental organizations (NGOs), community-based organizations (CBOs), local communities and other partners in conservation to map and develop strategies that can enhance sustainable management of coastal forests.

Offshore and deep-sea areas constitute most of the WIO region, and most of the habitats and species present, particularly in the deep-sea, remain unknown, most notably those associated with benthic environments. Similarly, seamounts and ridges, widely recognized as hotspots of biodiversity, also show high endemism, yet several basic bio-geological features remain unknown. These areas also attract a wide range of oceanic predators. Human extractive activities, particularly fisheries and deep-sea exploitation of mineral resources, are increasingly focused on such areas.

These ecosystems, present mainly in international waters, where maritime surveillance is weak, are under numerous pressures and threats, notably overfishing, IUU fisheries, seabed mining, pollution from illegal discharges (including ballast waters) and ocean dumping. Many of the recommendations shared in the *Critical Habitats Outlook* relate to the identification and prioritization of Ecologically or Biologically Significant Marine Areas (EBSAs) within these biomes in the first instance and MPAs at ABNJ scales as subsequent focus areas. There is also a need for effective management to fill gaps in information and to reinforce international cooperation for maritime surveillance beyond EEZs.

The WIO is characterized by hundreds of islands and atolls of various geophysical characteristics, scattered throughout the region. In combination, these are recognized as a global hotspot of marine biodiversity, housing several charismatic and endemic species. The human populations of the islands are highly dependent on natural resources and ecosystems goods and services and are on the frontline of consequences of global warming. The fragile ecosystems on which they depend are highly vulnerable to sea level rise and coral bleaching, while being at the same time under pressure from poorly regulated tourism, urbanization and logging, as well as illegal extractive uses and poaching. Not surprisingly, several WIO SIDS are leading the sustainable blue economy drive as a path to improve people's well-being while using international instruments to increase the level of protection and garner international support for the conservation of critical habitats (eg World Heritage, Ramsar) and endangered species from Dugongs to marine turtles.

The Nairobi Convention and the Regional Ocean Governance Strategy

The WIO region is aligned with the main global efforts within the Ocean's Decade conservation targets, particularly through the umbrella of the Nairobi Convention and its many initiatives, particularly the Regional Ocean Governance Strategy for WIO under project Sapphire.[7]

Several other initiatives are on-going, such as: (i) WIO-LME SAPPHIRE: The Western Indian Ocean Large Marine Ecosystems Strategic Action Programme Policy Harmonisation and Institutional Reforms; (ii) WIOSAP: Implementation of the Strategic Action Programme for the protection of the Western Indian Ocean from land-based sources and activities; (iii) NoCaMo: Nairobi Convention Clearinghouse and Information Sharing System; and (iv) WIOGI: Western Indian Ocean Governance Initiative. Furthermore, several other initiatives involving different institutional organizations, stakeholders and NGOs, are also underway, targeting biodiversity conservation. An example is the Great Blue Wall,[8] aiming at conserving and restoring marine and coastal biodiversity while unlocking the development of a regenerative sustainable blue economy. Additional initiatives and programmes address MSP/ICZM, transboundary MPAs and LME, and conservation and management of biodiversity in ABJN, contributing to SDG goals, conservation targets and a sustainable blue economy. These initiatives, programmes and action plans rely on an integrated approach to ocean governance in the region.

Within this conservation and governance setting, WIO governments, under the Nairobi Convention, agreed that regional ocean governance is the basis for the regional response and application of the UNCLOS commitments and those required to achieve SDG 14. This approach enables countries to discharge their obligations under

7. https://www.nairobiconvention.org/regional-ocean-governance-strategy/
8. https://www.iucn.org/news/secretariat/202111/global-launch-great-blue-wall

UNCLOS through cooperation with neighbouring countries – namely, the need to manage shared fish stocks, prevent transboundary pollution, conserve ecosystems, and facilitate marine transport. Although regional ocean governance faces several institutional challenges, such as decision-making; compliance, and changes to management structures and reform; it also relies on consensus between several partners to address thematic challenges, particularly regarding coastal and marine management, biodiversity conservation, marine pollution, and resource management and conservation in ABJN (UNEP-Nairobi Convention, 2020).

ICZM and MSP are two of the most relevant instruments to assist with the sustainable use of the marine environment and maritime space, respectively. The Nairobi Convention ICZM Protocol (adopted in September 2023, after 13 years of negotiations) provides a legal framework to promote regional and national ICZM and enhance cooperation for sustainable development. Moreover, most of the countries are developing MSP schemes and the corresponding legal framework as well as institutional arrangements (eg South Africa was the first African country to develop the legal basis for MSP), to respond to challenges associated with blue economy strategies, but also to address sustainable use of marine resources and their conservation (Guerreiro, 2022).

Considering that most MPAs are in the coastal zone, these instruments may enhance transboundary approaches for MPAs. This broader management approach also highlights the role of areas declared under international regimes such as Ramsar and World Heritage Conventions. Other complementary actions are in progress, such as those under SIOFA (Southern Indian Ocean Fisheries Agreement), the declaration of Vulnerable Marine Areas (VMAs), and associated 'no fishing' measures. Furthermore, the International Maritime Organization (IMO) instrument to declare Particularly Sensitive Sea Areas (PSSAs) can also be used for these purposes.

Finally, the creation of MPAs in the ABNJ is another recognized challenge, but one that could potentially be overcome through declarations accompanied by compliance measures that are progressively developed through regional consultations, including with the RFMOs and International Seabed Authority (ISA), both of whom have the mandate to develop legally-binding resolutions related to offshore areas within ABNJ.

The Nairobi Convention also has a protocol that provides for the protection of threatened and endangered species of flora and fauna, and important natural habitats in the region, which provides a platform for countries to protect rare or fragile ecosystems as well as rare, depleted, threatened or endangered species and their habitats in the region. A provisional list of species that are in need of protection was prepared, and numerous local and international NGOs are involved in conservation activities and the development of participatory approaches with relevant stakeholders. The Nairobi Convention has a particular focus on migratory species such as sharks, turtles and seabirds and considers the protected areas approach essential to preserve breeding sites, feeding grounds, nurseries, spawning grounds, and migratory routes and also draws attention to special requirements during spawning periods.

In the global context, efforts to combat marine sources of pollution and land-based sources of pollution (LBSP) have a key role in the conservation of habitats and species. Countries' engagement with MARPOL and London Conventions is central to these efforts. The Global Environment Facility-funded Strategic Action Programme for the protection of the WIO from land-based sources and activities (WIOSAP) – a project being executed by the Nairobi Convention – is an important regional initiative that assists WIO countries to combat LBSP and addresses many of the sources of pollution which present a major challenge in the WIO region. Of special focus is adequate sewage treatment and waste management, which continue to be responsible for the degradation of the environmental and health quality of coastal waters and shorelines in many parts of the region.

One of the most recent initiatives is the ABNJ/Biodiversity Beyond National Jurisdiction (BBNJ) negotiations to fulfil related gaps in the UNCLOS architecture, which was recently advanced (UN, 2022), and now focuses on four thematic areas: (i) Marine genetic resources (including benefit sharing); (ii) Area-based management tools (including MPAs); (iii) Environmental impact assessments; and (iv) Capacity building and technology transfer. The revised ABNJ draft text includes principles on ecosystem resilience, prevention of 'indirect' pollution through the transfer or transformation of pollution, internalization of environmental cost accountability, and non-regression and adaptive management.

Thus, the Nairobi Convention and its related programmes and initiatives, together with its direct relation with other global and regional conventions and strategies, play a central role in the development of regional ocean governance, responding to the major challenges of the Ocean Decade and SDG Agenda 2030.

PROSPECTS FOR MARINE CONSERVATION IN THE WIO REGION

To be effective, marine conservation needs to cover different themes that are features of the WIO region. The four principal ones are:

1. Ocean governance.
2. Scientific knowledge.
3. Implementation of effective conservation measures, management, and financial capacities.
4. Socioeconomic considerations.

Ocean governance

A common understanding of ocean governance is critical for a regional approach. This is supported by three main pillars: (i) Ratification of the main international instruments and their application under regional agreements and in domestic law (from UNCLOS to MARPOL or CBD); (ii) Policies, strategies, plans and norms endorsed at the international, regional or national level by States (from Integrated Maritime Policies to MSP and ICZM); and (iii) 'Soft' law instruments, such as codes of conduct (eg, the Code of Conduct for Responsible Fisheries), principles (eg, the precautionary principle) and international guidelines or recommendations (eg, IMO Guidelines for the Inventory of Hazardous Materials).

In the WIO region, countries are member parties of the main international conventions regarding ocean governance, biodiversity conservation and pollution prevention. Marine biodiversity conservation relies not only on global conventions from the CBD to Ramsar but also on the firm regional support provided by the Nairobi Convention. One of the challenges for regional marine pollution control is the fact that some countries in the region are still not full parties to the London or MARPOL conventions, and instruments such as the Declaration of Particular Sea Sensitive Areas (PSSAs) lack opportunities for their usage. Moreover, the regional standards of sewage treatment and waste management (both urban and industrial) are still very low. Hence, the degradation of coastal ecosystems and impacts on species abundance continues. Reversing these obstacles is probably one of the main challenges to overcome in the coming decade.

Most countries in the WIO region are now beginning to or are already fully embracing integrated maritime policies and blue growth, following AU encouragement and, accordingly, MSP and ICZM are being developed and implemented. There is, however, still a need to reinforce transboundary cooperation to obtain an adequate ecosystem-based approach. Regarding soft laws, most countries follow Food and Agriculture (FAO) guidelines and focus more on implementation and monitoring capacity, calling for more regional cooperation and data exchange. Finally, the ultimate challenge is the protection of biodiversity and sustainable management of natural resources in the ABNJ; although the recent evolution under UNCLOS provides a common ground, it is critical that the states of the region can implement those new legal measures, which greatly depends on their respective monitoring and surveillance capacity and resources.

Scientific knowledge

The conservation of habitats and species depends on sound scientific knowledge of ecological systems, biodiversity, population dynamics, species life cycles, larval dispersion, population genetics, and connectivity among critical habitats. These are requirements to guarantee that all habitats are adequately protected and represented in the conservation area network. The location of protected sites should avoid or minimize the complex threats affecting critical habitats and ecosystems, including climate change and anthropogenic pressures.

There are still major gaps of knowledge that need attention, particularly related to the high level of uncertainty in species status and the need to identify KBAs and EBSAs. Again, besides national efforts, collaborative transnational efforts that create dedicated thematic regional science networks can be a useful tool, even under international conventions such as Ramsar (eg focused on seagrass beds or coastal forests). The Science to Policy Platform under the Nairobi Convention, which seeks to strengthen the science-policy interface, should contribute to the much-needed information and data for evidence-based decision-making.

Conservation measures, management and financial capacities

It is widely agreed among the WIO states that MPAs are the best tool to protect marine habitats and species, supplemented with specific mechanisms to protect charismatic species (eg, turtles, dugongs, migratory birds). However, there is still a long way to go in order to reach the targets of ≥ 30 per cent of the EEZ of each country under some level of protection and ≥ 15 per cent under strict protection (no-take areas), including each of the principal marine habitats (eg, coral reefs, seagrass beds, mangroves), larval sources and corridors, coral reef fish nurseries and 60 per cent of KBAs for flagship, endemic and endangered species in no-take management zones.

For these targets to be reached, several issues must be addressed, namely: (i) ecosystem and transboundary approach, including LMEs, in order to constitute effective networks of MPAs guaranteeing representativity, connectivity and replicability; (ii) effectiveness of management plans, including a wide range of management levels, from transnational to local managed marine areas, also involving stakeholders and NGOs; and, (iii) financial resources, necessary both to assure effective measures of conservation and restoration of ecosystems, and to guarantee the availability of the necessary human resources. On the latter issue, IUCN recommended that, besides effective management plans, protected areas should also develop business plans to ensure sustainable financing (Emerton et al., 2006).

Finally, it is vital to monitor the success of ongoing conservation measures and the adequacy of their management plans by assessing management effectiveness. An assessment of potential benefits (or avoided biodiversity loss) could be the prerequisite for optimizing conservation areas. Adaptive management is now standard throughout any process that involves changing access to resources, changing resource user behaviour and practices and setting aside no-take areas.

Socio-economic considerations

The socio-economic component of nature conservation has received little attention for decades, failing to address conflicts with local communities and perpetuating inefficiencies in managing protected areas. Fortunately, this began to change dramatically after the Rio Conference of 1992, which stressed the need for the involvement of local communities in marine resource and biodiversity conservation, and this trend was consolidated during the Rio+10 conference in Durban in 2002.

Although the mindset has changed, the fundamental questions remain: how to compensate local communities and stakeholders for their losses in incomes that resource restrictions inevitably impose? How to involve local communities in management and conservation projects, avoiding a strict central, top-down governance model? How to value natural capital ecosystems goods and services? How to engage society in the cause of biodiversity protection? These reflect the major challenges that need to be overcome.

Some of the concrete suggested measures to consider include:

1. Separate reserves from coastal development to minimize resource-use conflicts (fishing, oil and gas, tourism).
2. Protect at least 50 per cent of areas that are more than 10 km from the shore and facing lower relative human pressure using the proximity to markets as a proxy.
3. Evaluate and promote alternative and sustainable livelihoods, such as areas suitable for eco-tourism with no-take restrictions.
4. Formalize and operationalize co-management of small-scale fisheries.
5. Set minimum distance (km) from aggregate fishing sites and sites with high catch rates.
6. Prioritize the protection of culturally significant areas (eg traditional fishing sites, coastal forests).
7. Promote people’s literacy on the goods and benefits of biodiversity conservation.

Together, these measures should contribute to a shift in the design and siting of future MPAs (and/or adapting existing ones) and marine biodiversity conservation strategies, while incorporating the complex socio-ecological setting and being guided by the adaptive management approach.

- **Goal A**
  supports SDG 14 and 15, as it contributes to the conservation of water and terrestrial ecosystems, protects against species loss and resilience of genetic diversity.

- **Goal B**
  also contributes to the advancement of ecosystem conservation, as indicated in SDG 14 and 15.

- **Goal C**
  advances SDGs 5, 10 and 11, as it emphasizes the importance of equitable and intergenerational sharing of benefits among all people. Goal C also supports the sharing of benefits which genetic resources provide to people, including food security, nutrition and sustainable agriculture (SDG 2), healthy lifestyles (SDG 3), access to water resources (SDG6), sustainable energy (SDG 7). Goal C supports SDG 16, as it promotes equitable and fair sharing on benefits, contributing to inclusive societies, as well as supports equitable governance systems.

- **Goal D**
  directly supports SDG 17's aim to strengthen the means of implementation and partnerships for sustainable development, as it promotes the means of implementation, including financial resources, mainstreaming, capacity building, awareness and communication, technology transfer, scientific cooperation and access to technology, be available to achieve the goals and targets of the GBF. Goal D additionally supports the achievement of SDG4, where equitable education on sustainable development, and biodiversity will be a crucial element for implementation to achieve the goals.

Figure 1: The relation between GBF 2050 goals and SDG targets.

REFERENCES

African Union Commission (2015). Africa Agenda 2063: the Africa We Want. African Union Commission, 20 pp. Addis Ababa, Ethiopia. https://www.un.org/en/ africa/osaa/pdf/au/agenda2063.pdf

African Union Commission (2019). Africa Blue Economy Strategy. African Union – Inter-African Bureau for Animal Resources. 33 pp. Nairobi, Kenya

Bullock, R., Ralph, G., Stump, E., Al Abdali, F., Al Asfoor, J., Al Buwaiqi, B., Al Kindi, A., … and Carpenter, K. (2021). The conservation status of marine biodiversity of the Western Indian Ocean. IUCN, vii + 32 pp. Gland, Switzerland

Campbell, L., Gray, N.J., Fairbanks, L., Silver, J. and Gruby, R. (2013). Oceans at Rio+20. *Conserv. Lett*. 6, pp. 439-447. https://doi.org/10.1111/conl.12035

Egede, E. (2023). UNCLOS 82: Africa's contributions to the development of modern law of the sea 40 years later. *Mar. Policy*, 148. https://doi.org/10.1016/j.marpol.2022.105463

Emerton, L., Bishop, J. and Lee, T. (2006). Sustainable Financing of Protected Areas. A global review of challenges and options. World Commission on Protected Areas Best Practice Protected Area Guidelines Series No. 13. The World Conservation Union (IUCN). Gland, Switzerland

Flannery, W., Healyb, N. and Luna, M. (2018). Exclusion and non-participation in Marine Spatial Planning. *Mar. Policy* 88, pp. 32–40. ttps://doi.org/10.1016/j.marpol.2017.11.001

Frazão-Santos, C., Domingos, T., Ferreira, M.A., Orbach, M. and Andrade, F. (2014a). How Sustainable Is Sustainable Marine Spatial Planning? Part I – Linking the Concepts. *Mar. Policy* 49, pp. 59–65. https://doi.org/10.1016/j.marpol.2014.04.004

Frazão-Santos, C., Domingos, T., Ferreira, M.A., Orbach, M. and Andrade, F. (2014b). How Sustainable Is Sustainable Marine Spatial Planning? Part II – The Portuguese Experience. *Mar. Policy* 49, pp. 48–58. https://doi.org/10.1016/j.marpol.2014.04.005

Guerreiro, J. (2021). The Blue Growth Challenge to Maritime Governance. *Front. Mar. Sci.* 8:681546. https://doi.org/10.3389/fmars.2021.681546

Guerreiro, J. (2022). Africa Integrated Maritime Policy, Blue Growth and a new Ocean Governance: case studies from the Atlantic to the Indian Ocean. *WIO J. Mar. Sci.* 21(i), pp. 17–32. http://dx.doi.org/10.4314/wiojms.si2022.1.2

UNEP-Nairobi Convention & WIOMSA (2015). The Regional State of the Coast Report: Western Indian Ocean. UNEP (United Nations Environment Programme) and WIOMSA (Western Indian Ocean Marine Science Association). Nairobi, Kenya

UNEP-Nairobi Convention (2020). The state of ocean governance in the Western Indian Ocean. https://wedocs.unep.org/20.500.11822/33767. UN Environment Programme (UNEP)

UN (2003). United Nations General Assembly, Oceans and the law of the sea, 21 February 2003, A/RES/1307. Available on https://www.un.org/en/development/desa/population/migration/generalassembly/docs/globalcompact/A_RES_57_141.pdf Accessed at 24 January 2023

UN (2004). United Nations General Assembly, Oceans and the law of the sea, 5 March 2004, A/RES/58/240. Available on https://www.un.org/en/development/desa/population/migration/generalassembly/docs/globalcompact/A_RES_58_240.pdf Accessed on 24 January 2023

UN (2014). Third International Conference on Small Island Developing States. Available at https://sustainabledevelopment.un.org/sids2014 Accessed on 29 January 2023

UN (2017a). The First Global Integrated Marine Assessment: World Ocean Assessment I. Cambridge: Cambridge University Press. https://doi.org/10.1017/9781108186148

UN (2017b). United Nations General Assembly, Oceans and the law of the sea, 5 December 2017, A/RES/72/73. Available at https://documents-dds-ny.un.org/doc/UNDOC/GEN/N17/421/90/PDF/N1742190.pdf?OpenElement Accessed on 24 January 2023

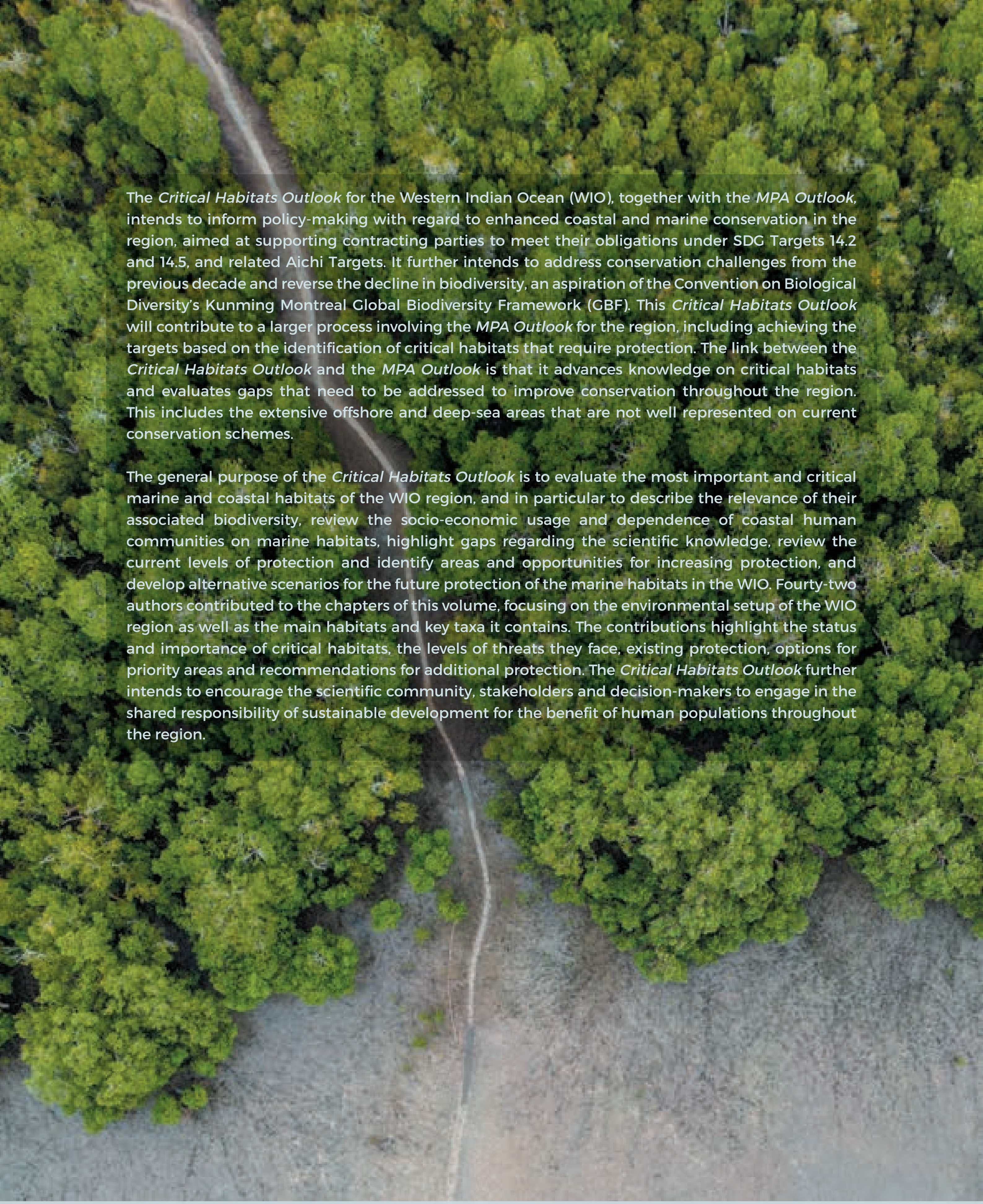
UN (2019). United Nations General Assembly, United Nations Decade on Ecosystem Restoration (2021–2030) 1 March 2019, A/RES/73/284. Available at https://documents-dds-ny.un.org/doc/UNDOC/GEN/N19/060/16/PDF/N1906016.pdf?OpenElement Accessed on 29 January 2023

UN (2021). The Second Global Integrated Marine Assessment, World Ocean Assessment Volumes I&II. United Nations Office of Legal Affairs. ISBN (PDF): 9789216040062

UN (2022). United Nations General Assembly, Further refreshed draft text of an agreement under the United Nations Convention on the Law of the Sea on the conservation and sustainable use of marine biological diversity of areas beyond national jurisdiction.12th December, 2022, A/CONF.232/2023/2 Available at https://documents-dds-ny.un.org/doc/UNDOC/GEN/N22/467/94/PDF/N2246794.pdf?OpenElement Accessed on 6 February 2023

UNESCO-IOC (2022). The United Nations Decade of Ocean Science for Sustainable Development 2021–2030. Ocean Decade Africa Roadmap. The Ocean Decade Series 36. UNESCO, Paris

UNESCO-IOC (2021). The United Nations Decade of Ocean Science for Sustainable Development (2021–2030) Implementation Plan. IOC Ocean Decade Series, 20. UNESCO, Paris



The *Critical Habitats Outlook* for the Western Indian Ocean (WIO), together with the *MPA Outlook*, intends to inform policy-making with regard to enhanced coastal and marine conservation in the region, aimed at supporting contracting parties to meet their obligations under SDG Targets 14.2 and 14.5, and related Aichi Targets. It further intends to address conservation challenges from the previous decade and reverse the decline in biodiversity, an aspiration of the Convention on Biological Diversity's Kunming Montreal Global Biodiversity Framework (GBF). This *Critical Habitats Outlook* will contribute to a larger process involving the *MPA Outlook* for the region, including achieving the targets based on the identification of critical habitats that require protection. The link between the *Critical Habitats Outlook* and the *MPA Outlook* is that it advances knowledge on critical habitats and evaluates gaps that need to be addressed to improve conservation throughout the region. This includes the extensive offshore and deep-sea areas that are not well represented on current conservation schemes.

The general purpose of the *Critical Habitats Outlook* is to evaluate the most important and critical marine and coastal habitats of the WIO region, and in particular to describe the relevance of their associated biodiversity, review the socio-economic usage and dependence of coastal human communities on marine habitats, highlight gaps regarding the scientific knowledge, review the current levels of protection and identify areas and opportunities for increasing protection, and develop alternative scenarios for the future protection of the marine habitats in the WIO. Fourty-two authors contributed to the chapters of this volume, focusing on the environmental setup of the WIO region as well as the main habitats and key taxa it contains. The contributions highlight the status and importance of critical habitats, the levels of threats they face, existing protection, options for priority areas and recommendations for additional protection. The *Critical Habitats Outlook* further intends to encourage the scientific community, stakeholders and decision-makers to engage in the shared responsibility of sustainable development for the benefit of human populations throughout the region.