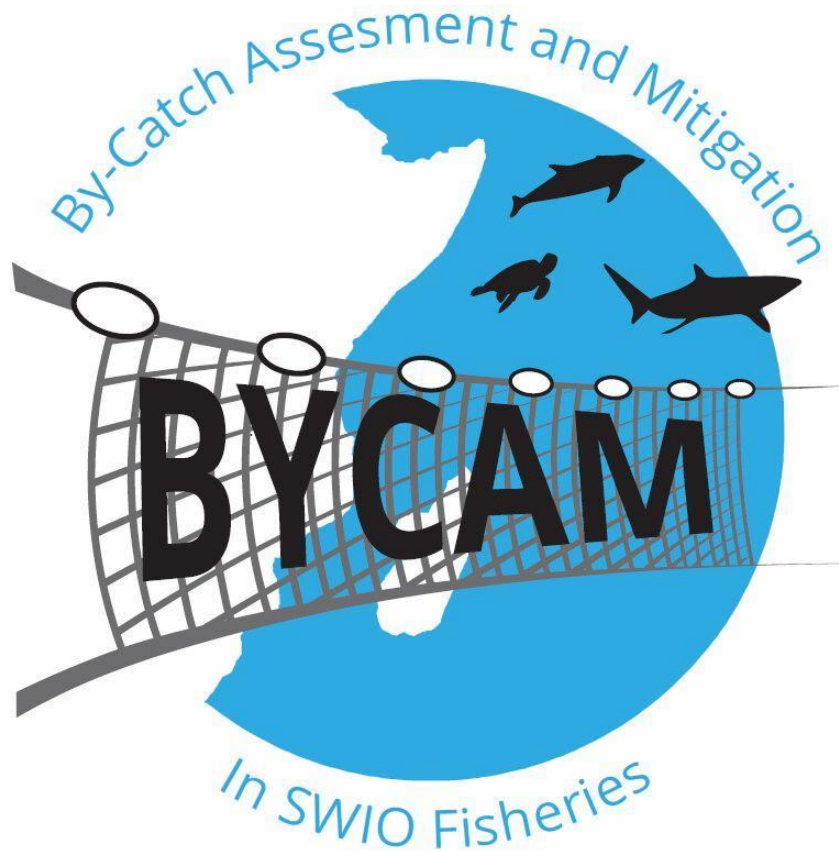


Final Report

Western Indian Ocean Marine Science Association: Grant Number MASMA/CP/2014/01

By-Catch Assessment and Mitigation in the Western Indian Ocean (BYCAM)



September 2019

Abstract

While there are considerable industrial marine fisheries in the south-western Indian Ocean (SWIO), notably for tunas and crustaceans, regional Small Scale Fisheries (SSF) are of greater social importance, employing > 0.5 million fishers and contributing > 70% of regional marine fisheries' catches. Despite often being overlooked, SSF may impact marine ecosystems, thus they reflect the global challenge of balancing conservation goals with communities' dependence on marine resources. Marine megafauna (elasmobranchs, marine mammals and sea turtles) are vulnerable to fisheries, and play important roles in marine ecosystems. Yet, little is known of the interactions between SWIO SSF and these species. This lack of information was the main driver for the WIOMSA/MASMA funded BYCAM project. The overall aim of BYCAM was to provide for future sustainable fisheries by creating a baseline estimate of current marine megafauna exploitation and subsequently develop and test methods for mitigation of vulnerable megafauna catches, and to provide recommendations for future governance and management. BYCAM focussed on three fisheries with known bycatch problems: semi-industrial prawn trawls, and small scale coastal longlines and gillnets.

BYCAM provides a much needed first independent quantitative assessment of SSF catch in the SWIO region and demonstrates that current FAO data for elasmobranch catches are seriously underreported, whilst catches of both marine mammals and sea turtles are ongoing. The implication of catch underestimates in official data is that vulnerable elasmobranch species may be overexploited and further that the specialist livelihood strategies of those relying on this catch are under threat. The project also demonstrates the need for a multi-method approach towards assessing marine megafauna interactions with SSF in the SWIO and other data-poor areas. The low cost mitigation methods developed and trialled in the project to reduce bycatch of vulnerable megafauna are promising. Funding has been secured and new trials with the bottle alarm and acoustic reflectors are underway. Further, there is considerable scope for more wide-spread use and implementation of TEDs in SWIO prawn trawls. The project has taken the first steps towards building governance frameworks to promote sustainable fisheries management within the WIO.

Project Investigators

Nina Wambiji – Kenya Marine and Fisheries Research Institute, Kenya

Edward Kimani - Kenya Marine and Fisheries Research Institute, Kenya

Harrison Ong'anda - Kenya Marine and Fisheries Research Institute, Kenya

Noah Ngisiang'e - Kenya Marine and Fisheries Research Institute, Kenya

Narriman Jiddawi – Institute of Fisheries Research, Zanzibar, Tanzania

Omar Amir – Deep Sea Fishing Authority, Tanzania

Sean Fennessey – Oceanographic Research Institute, South Africa

Bernadine Everett - Oceanographic Research Institute, South Africa

Chris Poonian – Community Centred Conservation, Madagascar

Jeremy Kiszka – Florida International University, United States of America

Sergi Pérez Jorge - Institute of Marine Research, Azores, Portugal

Atanasio Brito - University of Washington/ITECHMOZ, Mozambique

Per Berggren – Newcastle University, United Kingdom

Selina Stead – Newcastle University, United Kingdom

Andrew Temple – Newcastle University, United Kingdom

BYCAM: Existing Outputs, Students and Studentships Facilitated

This draft report represents the final framework of the BYCAM project where the results from all project activities have been summarised in the Table below.

Authors	Title	Type of output	Year	Submitted to	Outcome/progress
Existing Outputs					
FENNESSY ST, EVERETT BI, KAGURU BL, MWAKOSYA CA, MUNGA CN, SITOE S, DA CONCEICAO K, RAZAFINDRAKOTO HL, BRITO A.	Report of a workshop on the analysis of existing data from prawn trawl nets with Turtle Excluder Device (TED) and without TED (non-TED) in the Western Indian Ocean	Unpublished report	2015	WIOMSA	Finalized
TEMPLE AJ, AMIR OA, BRITO A, BERGGREN P, EVERETT BI, FENNESSY ST, JIDDAWI NS, KIMANI E, KISZKA J, NGISIANG'E N, ONG'ANDA H, PEREZ S, POONIAN C, RAZAFINDRAKOTO Y, STEAD S, WAMBIJI NN	Artisanal Vulnerable Megafauna Catch – BYCAM, an Overview	Conference - oral	2015	WIOMSA symposium	Presented
AMIR OA, BRITO A, BERGGREN P, EVERETT BI, FENNESSY ST, JIDDAWI NS, KIMANI E, KISZKA J, NGISIANG'E N, ONG'ANDA H, PEREZ S, POONIAN C, RAZAFINDRAKOTO Y, STEAD S, TEMPLE A, WAMBIJI NN	Artisanal Vulnerable Megafauna Catch – BYCAM, an Overview	Conference - poster	2015	WIOMSA symposium	Presented
BERGGREN P, TEMPLE AJ, SHARPE M, YANG L, AMIR OA, JIDDAWI N, NEASHAM J	URGENT need for no/low cost methods to reduce marine mammal bycatch in artisanal fisheries	Conference - oral	2015	Society for Marine Mammalogy	Presented
BERGGREN P, KISZKA J, READ A	Low cost methods to reduce marine mammal bycatch in global artisanal fisheries	Conference - Workshop	2015	Society for Marine Mammalogy	Conducted
ANON	Mise en place du projet BYCAM	Newspaper article	2015	L'Observateur	Published
FENNESSY ST, EVERETT BI, KAGURU BL, MWAKOSYA CA, MUNGA CN, SITOE S, DA	Effectiveness of a Turtle Excluder Device (TED) in prawn trawl nets in the Western Indian Ocean	Paper	2016	African Journal of Marine Science	Rejected, substantial changes

CONCEICAO K, RAZAFINDRAKOTO HL AND BRITO A					needed; will be submitted to another journal
BARROWCLIFT E, TEMPLE AJ, STEAD S, JIDDAWI NS, BERGGREN P.	Social, economic and trade characteristics of the elasmobranch fishery on Unguja Island, Zanzibar, East Africa.	Paper	2017	Marine Policy	Published 2017
TEMPLE AJ, KISZKA JJ, STEAD SM, WAMBIJI N, BRITO A, POONIAN CNS, AMIR OA, JIDDAWI J, FENNESSY ST, PÉREZ-JORGE S, BERGGREN P.	Marine megafauna interactions with small-scale fisheries in the southwestern Indian Ocean: a review of status and challenges for research and management	Paper	2017	Reviews in Fish Biology and Fisheries	Published 2018
TEMPLE AJ, BERGGREN P, SHARPE M, YANG L, AMIR OA, JIDDAWI N, NEASHAM J	Recycled bottles offer a potential low-cost solution to marine mammal bycatch.	Conference - oral	2017	WIOMSA symposium	Presented
TEMPLE AJ, KISZKA JJ, POONIAN CNS, WAMBIJI N, AMIR OA, JIDDAWI N, BERGGREN P	Assessing marine megafauna captures in small-scale fisheries of the southwestern Indian Ocean: landings data and questionnaire survey disparity.	Conference - oral	2017	WIOMSA symposium	Presented
BERGGREN P, TEMPLE AJ, SHARPE M, YANG L, AMIR O, JIDDAWI N, NEASHAM J	Recycled bottles offer potential low-cost solution to marine mammal bycatch	Conference - oral	2017	Society for Marine Mammalogy	Presented
PER BERGGREN, ANDREW TEMPLE, SEAN FENNESSY, NARRIMAN JIDDAWI, CHRIS POONIAN, SELINA STEAD, JEREMY KISZKA, SERGI PÉREZ JORGE, EDWARD KIMANI & NINA WAMBIJI	By-Catch Assessment and Mitigation in the Western Indian Ocean (BYCAM)	Conference - oral	2019	WIOMSA symposium	Presented
FENNESSY ST, EVERETT BE, KIMANI E, WAMBIJI N, ANAM R NALOVIC MA	Progress towards implementation of Turtle Excluder Devices in the WIO	Conference - oral	2019	WIOMSA symposium	Presented
NINA WAMBIJI, ISIGI KADAGI, NELLY, EDWARD KIMANI	Can circle hooks mitigate bycatch of marine megafauna? A case study of Kenyan artisanal handline and longline fisheries?	Conference - oral	2019	WIOMSA symposium	Presented

BERGGREN P, KISZKA J	Development and implementation of low-cost methods to reduce cetacean bycatch in small scale gillnet fisheries	Conference - Workshop	2017	Society for Marine Mammalogy	Conducted
STEAD SM	Rethinking marine resource governance for the United Nations Sustainable Development Goals	Paper	2018	Current Opinion in Environmental Sustainability	Published 2018
TEMPLE AJ, WAMBIJI N, POONIAN CNS, JIDDAWI N, STEAD SM, KISZKA JJ, BERGGREN P	Marine megafauna catch in southwestern Indian Ocean small-scale fisheries from landings data.	Paper	2018	Biological Conservation	Published 2019
PIROG A, RAVIGNÉ V, FONTAINE M, RIEUX A, GILABERT A, CLIFF G, CLUA E, DALY R, HEITHAUS M, KISZKA J, MATICH P, NEVILL J, TEMPLE A, BERGGREN P, JAQUEMET S, MAGALON H.	Population structure, connectivity and demographic history of an apex marine predator, the bull shark (<i>Carcharhinus leucas</i>)	Paper	2018	Ecology and Evolution	Accepted Aug 2019
PIROG A, JAQUEMET S, RAVIGNÉ V, CLIFF G, CLUA E, HOLMES BJ, HUSSEY NE, NEVILL J, TEMPLE A, BERGGREN P, VIGLIOLA L, MAGALON H.	Genetic population structure and demography of an apex predator, the tiger shark <i>Galeocerdo cuvier</i>	Paper	2018	Ecology and Evolution	Published 2019
BERGGREN P, TEMPLE AJ, SHARPE M, YANG L, NEASHAM J.	Drink bottle acoustic alarms – recycled glass and plastic bottles provide potential low cost solution to end 50 years of cetacean fisheries bycatch.	Paper	2018	Scientific Reports	In Review
SHARPE M, BERGGREN P	New abundance estimate for the Indian Ocean humpback dolphin in Menai Bay off the south coast of Zanzibar, East Africa predicts risk of extirpation.	Paper	2018	Aquatic Conservation – Marine and Freshwater Ecosystems	Accepted Aug 2019
BERGGREN P	Low-cost solution to marine mammal bycatch.	Presentation - Oral	2018	Permanent Commission for the South Pacific	Presented
BERGGREN P	Glass and plastic bottles as low-cost solutions to dolphin bycatch.	Presentation - Oral	2018	FAO Workshop on Means and Methods for Reducing Marine Mammal Mortality	Presented

BERGGREN P, SHARPE M, TEMPLE AJ, YANG L, AMIR OA, JIDDAWI N, NEASHAM J	Recycled bottles offer potential low-cost solution to marine mammal bycatch.	Conference - oral	2017	Society for Marine Mammalogy	Presented
SALMIN Y, JIDDAWI N, GRAY T, TEMPLE A, STEAD S	Improving bycatch mitigation measures for marine megafauna in Zanzibar, Tanzania	Paper	2018	WIOMSA Journal of Marine Science	In Review
WEIGMANN S, GON O, LEENEY RH, TEMPLE AJ, BARROWCLIFT E, SWEET M	<i>Pliotrema kaja</i> , a new six-gilled sawshark (Chondrichthyes, Pristiophoriformes) from the western Indian Ocean	Paper	2019	Marine Biodiversity	In Review
TEMPLE AJ, JIDDAWI N, WAMBIJI N, STEAD SM, BARROWCLIFT E, BERGGREN P	Life-history, exploitation and relative extinction risk of Baraka's whiplay (<i>Maculabatis ambigua</i>) in small-scale fisheries.	Paper	2019	ICES Journal of Marine Science.	In Submission
TEMPLE A, STEAD S, JIDDAWI N, BERGGREN P	Comparison of local knowledge and observation-derived methods for use in wildlife exploitation assessment and management	Paper	2019	Biodiversity and Conservation	In Submission
NINA WAMBIJI, ISIGI KADAGI NELLY, ANDREW J. TEMPLE, EDWARD KIMANI, JEREMY J. KISZKA, PER BERGGREN	Integrating long-term citizen science data and contemporary artisanal fishery survey data to investigate recreational and small-scale shark fisheries in Kenya	Paper	2019	To be determined	Manuscript
BERGGREN P, SHARPE M, JIDDAWI N	Menai Bay Conservation Area Zanzibar – Important Marine Mammal Area (IMMA)	Documents to support nomination	2019	IUCN Marine Mammal Protected Areas Task Force	Accepted July 2019
BYCAM Studentships					
MOHAMED AHMAN MOHAMED	By-catch assessment in Kenyan coast fisheries	MSc Thesis	2017	Pwani University	Passed
YUSSUF SALMIN	Improving bycatch mitigation measures for marine megafauna in Zanzibar, Tanzania	MSc Thesis	2018	State University of Zanzibar	Passed
MATTHEW SHARPE	Abundance and Conservation Assessment of Indo-Pacific Bottlenose and Indian Ocean Humpback Dolphins off the South Coast of Zanzibar	MPhil Thesis	2018	Newcastle University	Passed
ANDREW TEMPLE	Marine Megafauna Interactions with East African Small-Scale Fisheries	PhD Thesis	2018	Newcastle University	Passed

BYCAM Facilitated Studentships (i.e. using existing BYCAM data at no cost)					
ELLEN BARROWCLIFT	Investigating the socio-economy of the elasmobranch fishery in Zanzibar: a pilot study on Unguja Island	BSc Thesis	2016	Newcastle University	1 st Class - highest mark in year
ELLEN BARROWCLIFT	Life history of select elasmobranch species common in the small-scale fisheries of Unguja Island, Zanzibar	MPhil Thesis	2018	Newcastle University	Passed
ANNA POTTICARY	Elasmobranch fisheries on Unguja Island, Zanzibar	BSc Thesis	2016	Newcastle University	2.1 Class
DAISY SHARPE	Age-Growth Models of Baraka's Whipray	BSc Thesis	2017	Newcastle University	2.1 Class
LOUISE HOPKINS	The effect of two low-cost bycatch mitigation devices on the catch of small-scale fisheries in Zanzibar, Tanzania	MSc Thesis	2018	Newcastle University	Passed
SHANNON HORROCKS	Age-Growth Models of Smoothound species	BSc Thesis	2019	Newcastle University	2.1 Class
CHARLIE VAGG	Age-Growth Models of Scalloped Hammerheads	BSc Thesis	2019	Newcastle University	2.1 Class

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Summary Table

Summary of all BYCAM activities. Progress colour codes: **Red** = Not Achieved, **Yellow** = Partially Achieved, **Green** = Achieved, **Blue** = Achieved and Exceeded

Objectives	Progress		Comments
Activity 1.1			
Aim: Review of WIO fisheries statistics and bycatch for small-scale and semi-industrial fisheries			
Review of WIO fisheries statistics and bycatch for small-scale fisheries			Assessed for SWIO region, including official and research based data. Published open-access in Reviews in Fish Biology and Fisheries
Review of WIO fisheries statistics and bycatch for semi-industrial fisheries			Forms part of Activity 2.4, please see relevant section
Activity 2.1			
Aim: Understanding the socio-economic importance of bycatch species to stakeholders is vital in the process of bycatch mitigation and in determining priority species.			
Survey of socio-economic importance of species and perceptions of stakeholders towards species.			- Fieldwork carried out in Kenya, Zanzibar and Madagascar, but not in Mozambique. - Outputs of pilot study conducted in Zanzibar published in Marine Policy (see Annex 2), and main data in preparation (see Annex 3).
Activity 2.2 and 2.3			
Aim: Assess Vulnerable Megafauna (Elasmobranchs, Sea Turtles and Marine Mammals) Catch in Small-Scale Gillnet and Longline Fisheries			
Assess Drivers of Fisheries Effort in Gillnet and Longline Fisheries Using Landing Site Monitoring			- Fieldwork carried out in Kenya, Zanzibar and Madagascar, but not in mainland Tanzania - Assessed the Handline Fishery in Addition to Gillnets and Longlines - Published in Biological Conservation (see Annex 7)
Assess Catch and Composition of Vulnerable Megafauna in Gillnet and Longline Fisheries Using Landing Site Monitoring			- Fieldwork carried out in Kenya, Zanzibar and Madagascar, but not in mainland Tanzania - Assessed the Handline Fishery in Addition to Gillnets and Longlines
Assess Catch Demographics of Species, with a View to Assessing Vulnerability			- Achieved Species Vulnerability Assessment Using Known Life-History, Fisheries Catch and Catch Demographics in a Productivity-Susceptibility Analysis Overall and for Each Gear - Published in Biological Conservation (see Annex 7)
Explore Species Life-History for Selected Species Important to the Fishery			- Age-Based Life History Traits (Age-Growth, Age and Maturity, Longevity) Assessed for <i>Maculabatis ambigua</i>, ongoing for <i>Squalus</i> sp., <i>Mustelus</i> sp. and <i>Sphyrna lewini</i>. <i>M. ambigua</i> submitted to ICES Journal of Marine Sciences (see Annex 8)

Compare Outputs of a Common Interview Method (Rapid Bycatch Assessment) with Landings Monitoring		- Achieved Through a Bland-Altman Approach to Method Comparison - Manuscript draft in preparation (see Annex 9)
Activity 2.4		
Aim: Existing catch and effort data from those countries which have been using TEDs (Kenya, Tanzania, and Madagascar) will be analysed to examine their effectiveness.		
To assess whether existing TED usage is effective		Effectiveness of TEDs in SWIOFP surveys in Kenya and Tanzania assessed; no equivalent data from Madagascar available
Activity 3.1		
Aim: Creation and redevelopment of bycatch mitigation technologies		
Creation and redevelopment of bycatch mitigation technologies for gillnets		Glass bottle alarm (pinger) and plastic bottle acoustic reflector were developed to mitigate marine mammal catch in drift gillnets, paper in review with Scientific Reports (see Annex 10)
Activity 3.2		
Aim: Trial studies of pingers and other redesigned mitigation methods' efficiency in reducing bycatch of marine mammals in gillnets		
Trials of glass and plastic bottle pingers and acoustic reflectors to reduce dolphin bycatch in gillnets		Trials with bottle alarms and acoustic reflectors conducted in Zanzibar and Kenya
Assess the efficacy of the mitigation technologies to reduce bycatch of megafauna		There was no observed catch of marine mammals during 1107 observed hauls in Zanzibar comparing nets equipped with bottle alarms, reflectors and unmodified nets. Further, no marine mammals were caught in additional trials conducted in Kenya with nets equipped with bottles and unmodified nets.
Assess the effect of the mitigation technologies on target catch species		There was a significant difference in the Tuna catch between treatments, with plastic bottle reflector hauls showing a significantly higher catch rate of Tuna than the control hauls.
Activity 3.3		
Aim: Trial studies with circle hooks as potential mitigation method to reduce bycatch in coastal longline fisheries		
Trials with Circle hooks to reduce marine megafauna bycatch in longlines		Trials conducted with Circle hooks and traditional J hooks in longline fisheries in Kenya
Assess the efficacy of the mitigation technologies to reduce bycatch of megafauna		The circle, j-hooks and the combination of both hooks, "combo" landed a variety of megafauna species. J-hooks had greatest landings and average catch per unit effort (CPUE), but circle hooks but that circle hooks had higher CPUE than J-hooks in some months.
Assess the effect of the mitigation technologies on target catch species		The use of circle hooks reduced the CPUE of teleosts and thus may be used if reduction of catch for these taxa is required.
Activity 3.4		

Aim: Prawn trawling industry representatives in Madagascar will be consulted to determine the most appropriate TED design and construction material, as used in TEDs in that country		
Determine the most appropriate TED design and construction material based on Madagascar experience		Madagascar uses a tried and tested NOAA design, which is readily available; TEDs are made of materials easily available
Activity 3.5		
Aim: Trials of TED efficiency in reducing bycatch will be conducted in the Mozambique fishery		
To undertake TED trials so as to assess practicality and feasibility of using TEDs		Trials could not be undertaken in Mozambique owing to industry reluctance, so were undertaken in Kenya instead. The trials showed that TEDs can be used without loss of target catch, and with slight reduction in small fish bycatch. There was some loss of large fish bycatch. Megafauna bycatch was minimal in the trials, but the TEDs will exclude such catch because of their design.
		The TEDs were made based on the National Oceanic and Atmospheric Administration (NOAA). NOAA officers led by Joseph Fette visit KMFRI for an introductory visit and field testing of the TEDs.
Activity 3.6		
Aim: Understanding the governance processes in study countries and how these may contribute to, or inhibit the introduction and widespread use of bycatch mitigation where required is essential in building towards future sustainable management of the fisheries.		
Initial desktop reviews.		Critical review of recent published literature. Components of which are published in Marine Policy (see Annex 2) and Current Opinions in Environmental Sustainability (see Annex 4) and included in one paper in submission (see Annex 5) and in preparation (see Annex 3).
Pros and cons of different management measures considered alongside policy, governance, socioeconomic behavioural drivers, markets and livelihood issues.		- Fieldwork carried out in Kenya, Zanzibar and Madagascar, but not in Mozambique. - Combine face-to-face surveys conducted to understand efficacy of management measures with national policy and governance workshops. - Workshop policy briefs drafted (see Annex 6)
Constraints to management measures and critical factors affecting economic and social performance indices will be identified and used to inform development of good governance support and policy reform so they are developed at the same time and not at the end as an 'add-on'.		- Fieldwork carried out in Kenya, Zanzibar and Madagascar, but not in Mozambique - Combine face-to-face surveys conducted to understand efficacy of management measures, national policy and governance workshops and fisheries data collected in activities 2.2 and 2.3. - Workshop policy briefs drafted (see Annex 6)
Activity 4		
Aims: Data mapping: Creation of an interactive updatable spatial database including baseline data from previous surveys in the WIO and data collected during the project duration.		

Creation of an interactive updatable spatial database including baseline data available in the WIO from previous surveys carried out by SWIOFP and National Fisheries datasets (e.g. CAS data) and data collected during the project duration.		• Database developed with global fisheries databases • Local datasets yet to be mapped
Output		• Officers trained on ArcGIS.
Time Frame		• Whole duration of the project
Geographic description		• Whole project
Activity 5 Aims: Comprehensive Final Framework		
The final framework is represented by this draft final report which presents the results from all project activities.		This draft report represents the final framework of the BYCAM project where the results from all project activities have been summarised in the Table (above) of Existing Outputs, Students and Studentships Facilitated

Activity 1.1: Temple AJ

Background and Method

In developing regions, coastal communities are particularly dependent on small-scale fisheries for food security and income. However, information on the scale and impacts of small-scale fisheries on coastal marine ecosystems are frequently lacking. Large marine vertebrates (marine mammals, sea turtles and elasmobranchs) are often among the first species to experience declines due to fisheries. Our aim is to review existing knowledge and status regarding vulnerable marine megafauna interactions with marine Small Scale Fisheries (SFF) in the south-western Indian Ocean (SWIO) region. Data were gathered from a range of sources including, but not limited to, information requests from relevant government departments in SWIO nations, scientific and non-governmental organisations (NGO) publications and reports, international and national annual reports and databases. We discuss the likely implications of the current situation, highlighting vital knowledge gaps that need to be addressed and challenges for future research and management across the region.

Results and Discussion

We highlight an urgent need for proper documentation, monitoring and assessment at the regional level of SFF and the megafauna affected by them to inform evidence-based fisheries management. Catch and landings data are generally of poor quality and resolution with compositional data, where available, mostly anecdotal or heavily biased towards easily identifiable species. There is also limited understanding of fisheries effort, most of which relies on metrics unsuitable for proper assessment. Management strategies (where they exist) are often created without strong evidence bases or understanding of the reliance of fishers on resources. Consequently, it is not possible to effectively assess the current status and ensure the sustainability of megafauna species groups; with indications of overexploitation in several areas. To address these issues, a regionally collaborative approach between government and non-governmental organisations, independent researchers and institutions, and SSFF stakeholders is required. In combination with good governance practices, appropriate and effective, evidence-based management can be formulated to sustain these resources, the marine ecosystems they are intrinsically linked to and the livelihoods of coastal communities that are tied to them.

Further Information

For more detailed information on the analysis and outputs resulting from Activity 1.1 please see Annex 1.

Activities 2.1 and 3.6: Stead SM

Background

Governments such as those in the WIO are facing mounting pressure to 'show how they are making progress on the United Nations Sustainable Development Goal (SDG) 14' (Stead, 2018). A major constraint to many Nations' achieving SDG14 targets, especially in supporting sustainable fisheries is fragmented governance, where the natural environment is commonly managed separately to fisheries. Furthermore, the socio-economic impacts of environmental conservation measures on fishers, such as marine protected areas, is not fully considered in some regions such as the WIO, when developing policy, due to a lack of expertise and capability in how to do this (Stead, 2018).

Central to developing evidenced based recommendations to improve fisheries management is understanding how and why decisions are made by fishers. In our BYCAM project, surveys and workshops were designed to collect information on fishers' fishing practices, behaviour and compliance with management rules (Peterson and Stead, 2011; Bergseth et al., 2018). This is because fishers' perceptions of the health of a resource, e.g. indicated by catch size of targeted species, and their attitudes to management can influence their behavioural responses to regulations. Thus knowledge of drivers (e.g., environmental, economic and social) underpinning decision-making (governance) can be useful in ensuring that resultant policy recommendations reflects behavioural drivers more likely to motivate compliance for management measures associated with sustainable fisheries management practice.

Small Scale Fisheries (SSF) in the developing south-western Indian Ocean (SWIO) region employ half a million fishers and coastal communities' dependence on SSF for supplying food and income across the region is high (Barrowclift et al. 2017). However, marine megafauna are particularly vulnerable to fisheries impacts, due to their classically k-selected life-histories (e.g. Dulvy et al. 2014), and play important roles in the structure and function of marine ecosystems. Little is known of the interactions, both in terms of volume and composition, between SWIO SSF and these species (Temple et al. 2018). Similarly, there is a dearth of scientific evidence on the value of these species at local and national levels (Barrowclift et al. 2017). Training the next generation of scientists (e.g. BYCAM post graduate students Barrowclift, Mohamed, Salmin and Temple) in developing scientific evidence-led management and policy making was an important component of Activities 2.1 and 3.6.

An increasing number of studies consider the socio-economic aspects of fishing communities with environmental challenges in the WIO but few have teams with the practical expertise on linking scientific evidence to inform policy using context-specific governance systems. A unique contribution of the BYCAM project was to apply the experience of a UK Chief Scientific Advisor (Professor Selina Stead, PI of Activities 2.1 and 3.6) in starting to build capacity for the WIO in science-led evidence for policy to support sustainable fisheries development.

Methods

The BYCAM project brought together expert biologists, social scientists, interdisciplinary researchers and stakeholders, including fisheries managers, fishers, and government and non-government representatives. The team developed an interdisciplinary approach drawing on mixed methods from the natural and social sciences to collect both quantitative and qualitative data to assess the extent of bycatch in WIO fisheries. The intended outcome from the outputs is to provide scientific evidence which can be applied by different countries in the WIO to develop strategies for mitigating bycatch which account for social and economic constraints and opportunities within a multi-level governance framework. In recognising few studies in the WIO have examined the relationship between efficient governance and fisheries management efficacy, this project has extended earlier socio-economic research undertaken by the collaborators among others in the WIO through running national policy development workshops for addressing strengths and weaknesses in existing multi-level governance systems.

Details on the study sites, sampling approaches, survey instruments, descriptive and statistical statistics used in Activity 2.1 are provided in Annexes 2, 3 and 5. Noteworthy is that the MSc student, Yussuf Salmin

from the Institute of Marine Sciences, Zanzibar, had an environmental BSc and his willingness to engage in the social sciences for which he successfully defended his MSc provides this expertise locally in the WIO.

Results

Full details of the results for each task are provided in the relevant Annexes. Below highlight the main results from each of the national fisheries policy workshops held in Kenya, Madagascar and Zanzibar during August 2018.

Kenya

Positive and negative impacts of historical changes on fisheries in Kenya

In Kenya, conservation action involving marine protected areas was considered to have the most positive impacts, however, the initial gazettement of a MPA in the 1950s was also found to have negative impacts. Part of the problem, was the lack of transitional arrangements to help fishers to adapt to changes introduced which impacted where they could fish.

Those fishers less adaptable to change, such as older fishers, need to be considered and offered help to bridge time required to find alternative or supplementary ways to source fish for food or generate income when management interventions are implemented.

In Kenya, most human and physical resources for fisheries management activity are allocated to collecting data and information. Least effort is directed towards reviewing and evaluating fisheries management practices. This is a common problem in complex fisheries governance systems where action at a local level is not fully integrated nor aligned into regional and national policy objectives. Thus resource allocation to support sustainable management of fisheries is not optimised.

Recommendations for action in Kenya

When introducing management interventions such as MPAs, full consideration should be given to the socio-economic aspects as well as the environmental considerations. This is so those directly impacted can be given assistance to mitigate any unforeseen negative consequences, e.g. less income will impact negatively on quality of life.

Goals and objectives of fisheries policy should be evaluated regularly against local capacity to operationalise fisheries management strategies.

Good governance principles – cohesion, openness, participation, effectiveness and accountability – is a prerequisite for building transparency and trust in fisheries policy and associated management interventions. Thus involving relevant stakeholders requires identification of who does what and linkages made so that knowledge exchange occurs at each stage of the policy cycle review.

Madagascar

Positive and negative impacts of historical changes on fisheries in Madagascar

In Madagascar, the most positive change to impact fisheries was perceived to be the introduction of Ordinance 93 for regulating fisheries and aquaculture.

In contrast, enforcing regulations associated with appropriate marine resource use in the different zones was perceived as having a negative impact on the health of fisheries. Weak governance was offered as one of the reasons underpinning rule breaking.

In Madagascar, most human and physical resources for fisheries management related activity are directed towards the collection of data and information. Least effort is put towards reviewing and evaluating fisheries legislation and management. This is a common problem in complex fisheries governance systems where action at a local level is not fully integrated nor aligned into regional and national policy objectives. Thus resource allocation to support sustainable management of fisheries is not optimised.

Recommendations for action in Madagascar

A review and evaluation of both positive and negative impacts on fisheries identified by participants would be useful in terms of highlighting best practice which could be rolled out more widely. Updating relevant legislation was highlighted as a priority action which could help address shortcomings in existing legislation and management.

Engaging locals in fisheries management decisions at an earlier stage could lead to better support for compliance with rules through formal and informal governance mechanisms. Participants recommended better engagement between those involved in enforcement, management and policy of fisheries with marine resource users who are impacted by any changes to fisheries practices.

Zanzibar

Positive and negative impacts of historical changes on fisheries in Zanzibar

In Zanzibar, the most positive changes to impact fisheries were the establishment of the First Fisheries Act and the set-up of the Department of the Environment. Similarly, the implementation of the Menai Bay Conservation Area was considered to have positive influences on fisheries.

In contrast, the Village Community Banks (VICOBA) were perceived to have negative impacts, as was social law and the introduction of non-sustainable fishing methods including dynamite.

A recurring theme highlighted as a major issue which needed to be tackled is the lack of enforcement of fisheries regulations.

In Zanzibar, most human and physical resources for fisheries management activity are allocated to the environmental aspects of fisheries management yet enforcement of associated rules such as bans was perceived to be weak. Least effort is directed towards governance. Participants highlighted a lack of opportunity to be engaged in decision-making about fisheries policy and management rules.

Recommendations for action in Zanzibar

A review and evaluation of both positive and negative impacts on fisheries identified by participants would be useful in terms of highlighting best practice which could be rolled out more widely and areas needing to be prioritised for action to address shortcomings. In particular, understanding the reasons underpinning success and failure would allow a more targeted approach to developing future fisheries policy that is context-specific to the local issues.

Engaging locals in fisheries management decisions at an earlier stage could lead to better support for compliance with rules through informal governance mechanisms. Participants suggested strengthening the local community fisheries committees could be one way to improve fisheries governance and reduce rule-breaking.

Further Information

For more detailed information on the analysis and outputs resulting from Activity 2.1 please see Annex 2 and 3. For more detailed information on the analysis and outputs of activity 3.6 please see Annex 4, 5 and 6.

Activities 2.2 and 2.3: Temple AJ

Background

Landings Assessment

Small Scale Fisheries (SSF) in the developing south-western Indian Ocean (SWIO) region employ half a million fishers and contribute 72% of the marine fisheries catch (Pauly and Zeller 2015) and their importance to coastal communities for supplying food and income across the region cannot be overstated (e.g. Razafindrakoto *et al.* 2008; Humber *et al.* 2011; Barrowclift *et al.* 2017). However, SSF may also have substantial negative impacts on marine ecosystems, reflecting the global challenge of balancing conservation goals with the needs of communities reliant on natural resources. Marine megafauna (here referring to elasmobranchs, marine mammals and sea turtles) are particularly vulnerable to fisheries impacts, due to their classically k-selected life-histories (e.g. Lewison *et al.* 2004; Read *et al.* 2006; Žydelis *et al.* 2009; Wallace *et al.* 2010; Dulvy *et al.* 2014), and play important roles in the structure and function of marine ecosystems. Little is known of the interactions, both in terms of volume and composition, between SWIO SSF and these species (Temple *et al.* 2018). Improving fisheries catch and composition data, paired with fishing effort data, are essential in informing robust evidence-based management strategies to safeguard the future sustainability of marine megafauna fisheries and those communities whose livelihoods are dependent on them. Further, they act as first-steps towards understanding these fisheries stocks and thus form a baseline for future stock assessment and management.

RBA and Landings comparison

The use of local fisheries knowledge (LFK) to monitor various aspects of fisheries, either in isolation or in combination with other methods, is increasingly common (e.g. Moore *et al.* 2010; Beaudreau and Levin 2014; Pilcher *et al.* 2017). Relative to observer-based methods, LFK is often considered a cheap but effective way to generate fisheries data (Neis *et al.* 1999; Anadón *et al.* 2009; Rist *et al.* 2010). Consequently, widespread use of LFK is proposed as a solution with which to rapidly assess fisheries in data-poor and capacity-limited situations. Such situations are most evident in small-scale fisheries (SSF), which account for approximately 95% of fishers, 23% of catch by weight and 27% of catch by value at the global level (Pauly 2006; Pauly and Zeller 2015). The majority of SSF are found in capacity-limited developing nations, making the use of LFK particularly attractive. Additionally, LFK may be advantageous in monitoring unusual events, such as catches of rare species, which equivalent observation-based methods are likely to miss if sampling effort is not high enough. However, LFK is vulnerable to a number of pitfalls and biases which may undermine confidence in their outputs. These may be malicious, such as supply of intentionally false information, or malign, for instance biases in human cognitive recall (Matlin 2004; Hirst *et al.* 2009).

Despite the uncertainties and biases surrounding both observer-based and LFK methods there are relatively few studies in fisheries science that have cross-examined their outputs thoroughly. The majority of studies have been restricted to identifying evidence of relationships between methods (e.g. Fox and Starr 1996; Lunn and Dearden 2006; Anadón *et al.* 2009; Rist *et al.* 2010; Daw *et al.* 2011; Sampson 2011; Mion *et al.* 2015; Lima *et al.* 2017) and consistently failed to properly assess patterns in bias and precision. Yet bias and precision are vital components in assessing and understanding the variable structure of relationships among methods. Evidence for relationships between LFK and observation-based data are mixed (e.g. Anadón *et al.* 2009; Rist *et al.* 2010; O'Donnell *et al.* 2012). LFK is generally considered to be a useful indicator for assessing long-term trends in species and animal harvesting activities (Neis *et al.* 1999; Daw *et al.* 2011; O'Donnell *et al.* 2012; Beaudreau and Levin 2014), although this is difficult to validate. Comparatively, the use of LFK as a tool to assess harvesting activities across shorter temporal ranges, such as intra-annual trends, and among locations has received limited attention. Yet, these are important components in the understanding of fisheries and the formulation of management strategies.

Life-History

Unlike teleost fishes, elasmobranchs (sharks and rays) generally display classically k-selected life history traits, slow growth, late reproduction, long gestation and low fecundity (Compagno 1990). These traits

exacerbate their vulnerability to non-natural mortalities and limit their recovery potential (Žydelis *et al.* 2009; Dulvy *et al.* 2014). However there is a considerable variation in the life history traits both among species (Stevens and McLoughlin 1991; Jacobsen and Bennett 2011) and within species (Lombardi-Carlson *et al.* 2003; Jacobsen and Bennett 2010; O'Shea *et al.* 2013). Fisheries are the most prominent source of non-natural mortalities for elasmobranchs at the global level (Worm *et al.* 2013; Dulvy *et al.* 2014). Understanding species and stock specific life-history traits is important when undertaking assessment of the sustainability of fisheries exploitation, conducting accurate stock assessment, producing demographics models and predicting rebound potential (Frisk *et al.* 2001; Cailliet and Goldman 2004; Smith *et al.* 2008). Thus, this information is also pivotal in the formulation of evidence-based fisheries management. Management plans in data-poor fisheries formulated despite a lack of species-specific life-history and fisheries exploitation data risk being ineffective or even detrimental to long-term sustainability of elasmobranchs

Methods

Landings Assessment

Trained observers recorded landings of marine megafauna (elasmobranchs, marine mammals and sea turtles) from select fisheries gears at landings sites in Kenya (n=8), Zanzibar (n=8) and northern Madagascar (n=5) for a period of 12 months between June 2016 and June 2017. Gears monitored were drift and bottom-set gillnets, longlines (demersal and pelagic) and handlines (including rod and reel gears). Observers at each site collected data for 147 simultaneous sampling days. Sampling days were selected using a stratified-random approach: the year was divided into lunar months which were subdivided into four lunar phases and sampling days randomly generated within each lunar phase. This regime ensured that the study accounted for potential lunar-driven patterns in fishing effort and species availability to the fishery (e.g. variability in vertically migrating species), and subsequent effects on catches. Landing sites were selected accounting for three major factors: prevalence of longline and gillnet gears (maximising representation), geographic spread (maximising geographic coverage and potential links to species availability) and logistical constraints (e.g. sites needed to be accessible by road). Observers recorded data for landed marine megafauna including photographs for species identification, morphometric data (fork length, disc width and weight), sex, gear used and local species name. Observers also recorded daily fishing effort as total number of vessels trips by gear type.

Patterns in fisheries effort were modelled and combined with CPUE estimations derived from monitored catch and effort and fleet numbers derived from official data in order to predict overall catches in the SWIO, as well as at the country level for Zanzibar and Kenya. Landings composition by country and gear type was achieved through species identification from photographs taken by observers, to species or nearest taxonomic level where possible. Local species names were also recorded. Where local names corresponded to specific species groups they were used to identify non-photographed individuals to genus level or higher, with the majority being to family level, minimising the risk of wrongful identification. Lastly, in order to identify priority species and gear-species interactions for future research, a series Productivity-Susceptibility Analysis (PSA) (Hobday *et al.* 2011), confined within the context of SWIO SSF, were carried out for gear types in isolation and combination.

RBA and landings monitoring comparison

Face-to-face RBA questionnaire interviews were carried out with fishers, targeting vessel captains where possible, in the same sites and covering the same gears in Zanzibar (n=204, captains = 99) and Madagascar (n=36, captains = 9) as had the landings observers. The RBA recorded fisher declarations of fisheries effort and catch including: months of gear use; average days fished across those months; months in which specified taxa were caught (sharks, batoids, sea turtles, dolphins) and selected easily identified species groups within these (hammerhead sharks *Sphyrna spp.*, whale sharks *Rhincodon typus*, mobulids *Mobula spp.* and sawfish/sawsharks *Pristidae/Pristiophoridae*); and average number of each group caught across those months. RBA derived data were scaled using total known vessel numbers by gear at each site, derived from frame census survey data in Zanzibar (ZDFD 2018) and counts by the project team in Madagascar, to both monthly and annual levels.

In order to comprehensively compare the outputs of both observed landings and RBA methods and thus assess their comparability the Bland-Altman approach was taken (Bland and Altman 1999, 2003). Outputs assessed using this method were: patterns in fisheries effort by month, measured as a proportion of total effort observed or declared; patterns in taxa catch by month, measured as the total proportion of catch observed or declared (groups with extremely few declarations or observed landings were excluded from these analysis – dolphins, sea turtles, whale sharks and sawshark/sawfish); and patterns in taxa catch among sites, as a proportion of catch declared or observed (groups with extremely few declarations or observed landings were excluded from these analysis – dolphins, sea turtles, whale sharks and sawshark/sawfish). All aforementioned analyses are carried out at the country level, i.e. RBA and observed landings outputs are summed across sites, to reflect the design purpose of RBAs. Lastly, total site-by-site estimates for catch of taxa generated from RBA declarations and observed landings were directly compared to one another, with mean differences by site (for those sites where catch was reported from both RBAs and observed landings) also calculated.

Life-History

Baraka's whiplay (*Maculabatis ambigua*) was found to be an important catch constituent for commercial use in Kenya and Zanzibari fisheries. The species was first described in 2016 (Last *et al.* 2016) and no life-history data is available. In order to generate life-history data required for assessment we create age-growth models, maturity and longevity estimates based on aging data from vertebrae sagittal sections, derived from animals sampled in Zanzibar (n=47). Further we describe size-weight relationships from animals sampled during the 12month observer network. Using the life-history information data generated we derive estimates of total (Z), fisheries (F) and natural (M) mortality from catch curve analysis, as well as estimating the exploitation ratio (E). We also calculate the credible range for the maximum intrinsic rate of population increase (r_{max}). Combining Z with estimates of E and r_{max} we infer the current extinction risk and likely sustainability status of this species in SIWO SSF.

Similar data was recorded for a wide-range of other elasmobranch species and assessment for these is currently in progress, with ongoing MPhil and BSc student projects at Newcastle University facilitating analysis for *Squalus* sp., *Mustelus* sp. and *Sphyrna lewini*. These results will be made available to WIOMSA upon completion, as will any other resultant works in future.

Results and Discussion

Landings

Landings (n=4666) identified 59 species, including three sea turtles, two small cetaceans and one sirenian (Dugong dugon). Primary gear threats to investigated taxa were identified as bottom-set gillnets (marine mammals, sea turtles and batoids), drift gillnets (marine mammals, batoids and sharks) and longlines (sharks). Overall, catch was dominated by small and moderately sized coastal requiem sharks (Carcharhiniformes) and whiplays (Dasyatidae). Larger coastal and oceanic elasmobranchs were also recorded in substantial numbers as were a number of deeper-water species. The diversity of catch demonstrates the potential for small-scale fisheries to have impacts across a number of ecosystems. From the observed catch rates we calculated annual regional elasmobranch landings to be 35,445 (95%CI 30,478-40,412) tonnes, 72.6% more than officially reported in 2016 and 129.2% more than the 10-year average (2006-16) (FAO 2018), constituting 2.48 (95%CI 2.20-2.66) million individuals. Productivity-Susceptibility Analyses indicate that small and moderately sized elasmobranchs are most vulnerable in the small-scale fisheries. The study demonstrates substantial underreporting of catches in small-scale fisheries and highlights the need to expand efforts globally to assess the extent and impact of small-scale fisheries on vulnerable marine species and their respective ecosystems.

RBA and landings monitoring comparison

The results demonstrate inconsistency in relationships among spatial and temporal fishing effort and catch patterns between these methods, with the majority showing no evidence of relationships. Positive relationships appeared more common where patterns in effort and catch showed greater levels of variability and/or potentially for distinct species groups, but precision between methods was low and in

some cases evidence of bias was found between methods. Thus, outputs of the two methods cannot be considered broadly equivalent nor interchangeable. The findings support the need for multi-method approaches to natural resource monitoring in order to better inform management decisions and highlight areas of contention where further works may be required.

Specific considerations from this study as they apply to future works are:

- A need for future research in methodological development and comparison to assess not only the relationship among outputs of methodological approaches but also to quantify biases and precision between methods. Thus, improving the informed use and interpretation of multi or mono method approaches and their outputs.

And

- A need to assess how specific species in fisheries catch (e.g. memorable or distinctive species such as hammerhead sharks or mobulid rays) drive fisher perceptions of overall fisheries activity and the consequences this has on fisher declarations.

***Maculabatis ambigua* life-history**

The outputs represent the first estimates of disc width-weight relationships, growth models, maturation and longevity for this species, as well as demonstrating a wider distribution range than previously reported. From these data we derive the first plausible estimates for maximum intrinsic population growth rate $r_{\max}$ and these are combined with estimates of total Z , fisheries F and natural M mortality, derived from Chapman-Robson catch curve analysis, as well as estimation of the exploitation ratio E in order to assess relative extinction risk. The data indicate that *M. ambigua* is a moderately-sized, fast growing and early maturing species of whiptail stingray (Family Dasyatidae), with a relatively long lifespan. This life-history results in a high $r_{\max}$ relative to many other elasmobranch species which, in conjunction with estimates of F and E , suggest that the current fishery is likely sustainable at near optimal exploitation. However, data is lacking with regard to fecundity (litter size and inter-birth interval) and this information is highlighted as a priority in order to improve confidence in future assessments. Further, full recruitment to the fishery occurs post-maturation and fisheries exploitation occurs across broad age and size classes which raises concern for the long-term sustainability of these fisheries if fishing pressure were to increase.

Further Information

For more detailed information on the analysis and outputs resulting from Activity 2.2 and 2.3 please see Annexes 7, 8 and 9.

Activity 2.4: Fennessy ST

Background

The first effort to introduce TEDs in the region took place in Kenya (Wamukoya and Salm 1997), with legislation requiring their use introduced in 2001. Compliance was poor however, until the formalisation of the prawn fishery management plan in 2010, when regular use of TEDs commenced. Since 2005, TEDs have been in use in the industrial fleet in Madagascar, with good compliance (Fennessy et al. 2008). Conversely, the Tanzanian prawn management plan only makes reference to TED use i.e. they are not compulsory. There have been no efforts to assess effectiveness of TEDs in the region. In the South West Indian Ocean Fisheries Project (SWIOFP) which ran from 2010-2015, consideration was given to assessing the effectiveness of TEDs by collecting data on TED vs non-TED catches by prawn trawl surveys in Kenya and Tanzania, during the course of SWIOFP. These data were available for use by BYCAM and were used to address this project objective.

Methods

As part of SWIOFP, shallow-water prawn surveys were undertaken in Kenya and Tanzania in 2011 to determine, amongst others, species composition and distribution of prioritized crustacean species. These surveys presented opportunities to test the effectiveness of TEDs in these two countries. The surveys occurred in the North East Monsoon (NEM) and South East Monsoon (SEM), from 22nd January to 4th February and May 21st to June 4th in Kenya, while the Tanzanian surveys occurred from February 9th to 23rd and June 13th to 25th. The same prawn trawler fitted with two outrigger trawl nets was used for both countries. The vessel's crew remained the same for all surveys, with the exception of the addition of a different fishing master in Tanzania; the scientific crew comprised staff from KMFRI and TAFIRI for the surveys in Kenya and Tanzania, respectively. One net was fitted with a TED; trawls were of one-hour duration and standard data collection (trawl speed, location etc) was undertaken. Catches from TED and non-TED trawls were kept separate; small catches were sampled in their entirety while larger catches were subsampled and scaled up to determine total catch. Numbers and weights of individual taxa were enumerated for each trawl. Greater details on the sampling and data collection are available in SWIOFP research reports by Kimani et al. (2012a, 2012b) and Mwakosya et al. (2012).

Also available was a set of TED vs non-TED trawl catch records from Madagascar which were provided by CEDP who had been sent the data from an industrial trawling company in Mahajunga. The company conducted trials on one vessel during commercial operations on the west coast from 6th to 25th of November 2013. The vessel had four nets (two per outrigger), two with TEDs and two without. Catch weights of prawns and total retained fish on a per-trawl basis were provided for the combined pairs of TED and non-TED nets. Industrial fleet catch data from the pre-TED period (to 2005) and post-TED implementation were not made available.

Sean Fennessy and Bernadine Everett of the Oceanographic Research Institute (ORI) undertook the analyses and a joint workshop was held in Maputo from 10-11 June 2015 to present the results and to consider implications. This was attended by project participants from the Tanzanian Fisheries Research Institute (TAFIRI), Kenya Marine and Fisheries Research Institute (KMFRI), Administração Nacional das Pescas (ADNAP), Universidade Eduardo Mondlane (UEM), Centre d'Études et de Développement des Pêches (CEDP) and Instituto de Investigação Pesqueira.

For the Kenyan and Tanzanian TED vs non-TED data analysis, all prawns were aggregated as Target catch, while all other taxa (mainly teleosts) were considered as Bycatch. Nominal catch data were standardized to total kg and number of individuals of Target and Bycatch organisms per trawl hour. Examination of comparative TED/non-TED mean catch rates per country and season suggested considerable differences, and Generalized Linear Models (GLMs) were run in R to compare comparative TED/non-TED catch rates and, subsequently, to investigate potential factors which could influence TED functioning. After consideration, two influencing factors were assessed, namely trawl speed and size of catch (Broadhurst 2000). Country was not obviously an influential factor, since the vessel, gear and methodology were the same for both countries. Catch sizes were categorized as Large (> 100 kg) and Small (< 100 kg). All other

factors, such as season (NEM/SEM), depth, salinity, etc. were either deemed non-influential on TED versus non-TED catch rates i.e. their influence was the same for both trawl types, or were considered to be correlates of the catch size factor. Details are available in the full technical report produced for BYCAM (Fennessy et al. 2016).

For the commercial trawl data from Madagascar, the number of trawls (165) was sufficiently high to overcome the apparent skewness of the data (non-normal), so paired two-sample t-tests for differences in means were run in Microsoft Excel.

Results and Discussion

In total, 30 Kenyan trawls (15 NEM and 15 SEM) and 68 Tanzanian trawls (31 NEM and 37 SEM) were undertaken using a TED. No turtles were caught in the surveys, and elasmobranch catches were too infrequent (18 positive trawls out of 98) and were too small to test for TED/non-TED differences, except for the Kenyan SEM survey in which 15 trawls were undertaken, in which the TED trawl had 17 elasmobranchs and the non-TED 24. There were marked contrasts between mean catch rates of Target and Bycatch (in weight and/or in number), as a function of country (Kenya and Tanzania), and/or season (NEM and SEM). However, the potential role of season in TED effectiveness was considered to be mediated via catch size, so season was ignored in the GLMs.

Nominal catch rate results suggest that, in both Kenya and Tanzania, catch rates of numbers of prawns in TED trawls were lower than in non-TED trawls, particularly on the first survey in each country, but the prawn catch rates by weight in TED and non-TED trawls were very similar in each case. In Kenya, bycatch catch rates by weight and number in TED trawls were considerably lower than in non-TED trawls, except on the second survey when bycatch rates by number in TED trawls were greater than in non-TED trawls. In Tanzania, bycatch rates by weight and number in TED trawls were similar to those in non-TED trawls, except on the second survey when bycatch rates by number in TED trawls were considerably lower than in non-TED trawls. The nominal results thus present some conflicting findings.

The initial GLMs, incorporating only the presence or absence of a TED as an explanatory factor for both countries combined, showed that the TED reduced bycatch only slightly, and that prawn catch rates were increased slightly by weight and decreased slightly by number in TED trawls (all differences significant). This suggests that the TED was not functioning optimally on the surveys.

Subsequent GLMs were used to investigate potential factors which may have affected TED functioning. All differences between TED and non-TED trawls reported below are significant unless indicated. TED trawls in Kenya had higher bycatch rates than non-TED trawls, conversely, non-TED trawls in Tanzania had slightly higher bycatch rates than TED trawls. However, target catch rates in non-TED trawls in Tanzania were much higher than in non-TED trawls in Kenya, and were also much higher than in TED trawls in Tanzania. Target catch rates in non-TED trawls in Kenya were slightly higher than in TED trawls. The effect of the TED on bycatch rates was not affected by catch size, although non-TED catch rates were slightly higher than TED catch rates when catches were large; however, non-TED catch rates of target prawns were higher than TED catch rates for both large and small catches, and particularly when catches were large. Trawl speed showed variable effects on TED function with respect to bycatch catch rates, but, for target prawns, non-TED catch rates were greater than TED catch rates regardless of speed, and were particularly greater at slower speeds.

When catch rates were examined on a per-country basis, it was broadly observed that, in Tanzania, both bycatch and target catch rates were higher in TED trawls, and this was particularly the case for target catch rates at slower speeds and when catches were large. In Kenya, the opposite was broadly seen: TED catch rates (target and bycatch) were generally higher than non-TED catch rates, although this was less marked when catches were small. Full results are available in the technical report produced (Fennessy et al. 2016).

In summary, it was concluded that the TED used in the Kenyan and Tanzanian surveys was not functioning very well, as bycatch rates were not much reduced by the TED (in Tanzania) and were increased relative to non-TED catch rates (in Kenya); it was anticipated that the TED would reduce bycatch to some extent.

Prawn catch rates were reduced in Tanzanian TED trawls (relative to non-TED trawls) but were increased slightly in Kenyan TED trawls.

The commercial trawl data from Madagascar showed that the trawls with TEDs caught considerably fewer prawns and fish (both significant) than non-TED trawls. Two turtles were caught in this period, in a net without a TED. The Madagascan information was not verified independently, but suggests that the TEDs used in that survey are also not functioning correctly, as prawn catches were considerably reduced in the TED trawls. It may be expected that retained bycatch is reduced in TED trawls relative to non-TED trawls owing to the release of larger fishes by the TEDs.

It was not the intention of the workshop to compare TED effectiveness between Kenya and Tanzania – the data were only separated in order to try and understand if the TED was working and what could be affecting it. Unfortunately, the observed differences cannot be explained with the information available. One possible factor is that the distances trawled in Tanzania were derived from haul duration and estimated speed, whereas in Kenya they were derived from haul duration and grid reference positions at the beginning and end of the trawls. Inaccuracies in trawl speed estimation may have affected catch rate calculation, with consequences for the assessment of the relative performance of TED versus non-TED trawls. Other uncertainties exist in the consistency of hauling of trawls without reducing forward motion of the trawler – a “haul-back delay” can cause the catch to wash forward in the trawl resulting in losses via the TED (Watson 1989), or in the amount of time the net is on the surface after hauling, which may cause loss of catch. It also appears there was no comparison of the catches by the two nets before the TED was installed i.e. to account for possible pre-existing biases in catch efficiency.

Previous trials with correctly-rigged TEDs have reduced bycatch relative to non-TED trawl nets, and TEDs should not reduce prawn catches substantially – prawn catches may even be increased relative to non-TED catches owing to increased gear efficiency (Isaksen 1992, Brewer et al. 1998, Broadhurst 2000, Fennessy and Isaksen 2007). It is not apparent to what extent the rigging of the TED used on the MF Vega in the SWIOFP surveys was checked and adjusted, but this is something which needs to be undertaken frequently, as the trawl nets change over time because of stretching, resulting in changes to the TED angle (FAO 2007).

These findings emphasized that, for the testing of TEDs in BYCAM, the potentially influential factors which could compromise analysis of results, as described above, be minimised as much as possible. Testing procedures and conditions must be standardized to avoid bias, while providing adequate replication to enable rigorous statistical analysis to satisfy both managers and the industry as to whether the TEDs are functioning correctly. This will require skills transfer from a fishing gear specialist, co-operation from industry, well-trained observers and capable scientists.

Further Information

See

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3.1 Creation and redevelopment of bycatch mitigation technologies

Bycatch in gillnet fisheries is considered the most significant threat to marine mammals globally (Read *et al.* 2006) and in the south-western Indian Ocean (Kiszka *et al.* 2012). Approximately 10% of the population of Indo-Pacific bottlenose dolphins (*Tursiops aduncus*), resident in Menai Bay, Zanzibar (Fig.1), was removed by fisheries bycatch in driftnets in 2003/2004 and 2007/2008 (Stensland *et al.* 2006; Amir, 2010). The removals were documented by onboard observer programmes conducted in the fishery and the data collected indicated a bycatch rate of 0.02 dolphin/driftnet haul (Amir 2010). During 2007/2008, trials were also conducted in the same fishery using electronic alarms (Fumunda, 10kHz, 132dB) which resulted in a reduction of dolphin bycatch (Amir 2010). Toothed whales (odontocetes) navigate and detect objects in their environment using echolocation (the process of transmitting and receiving high frequency sound for target detection) (Cranford *et al.* 2009). However, successful detection is influenced by the sound frequency and acoustic backscatter properties of an object. Gillnets' backscatter is typically too weak for dolphins to detect in time to avoid entanglement. There are few examples where bycatch rates have been estimated, trials conducted and mitigation implemented to reduce bycatch. Electronic acoustic alarms (pingers) have been trialled and used successfully in some fisheries to reduce bycatch of some dolphin species in gillnet fisheries (Amir 2010; Mangel *et al.* 2013). However, to date little mitigation has been implemented. The main driver behind low uptake of pingers is cost (commercially available pingers cost US\$35-\$100 per 100m net). To address this, we have developed and trialled recycled glass and plastic drink bottles to create low-cost (US\$0-0.50 per 100m net) mechanical pingers and passive acoustic reflectors as potential solutions to reduce dolphin bycatch in gillnets. Representatives from project BYCAM also organised workshops in low cost methods to reduce marine mammal bycatch in global artisanal fisheries at the Society for Marine Mammalogy Biennial meetings in 2015 (Berggren and Kiszka, 2017) and 2017 (Anon. 2017) which provided very useful feedback for the trials conducted as part of BYCAM.

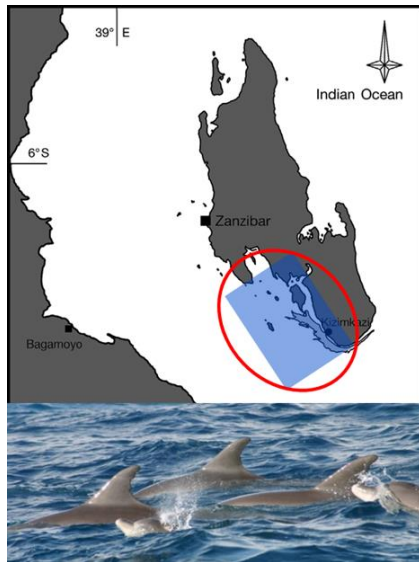


Fig. 1. The Menai Bay (indicated by red circle) where trials were conducted with glass bottle pingers and plastic bottle acoustic reflectors as potential mitigation to reduce dolphin bycatch in driftnets.

Bottle pinger and acoustic reflector development

The bottle pinger uses recycled 350ml glass drink bottles with a metal bolt inside that creates a clinking sound as the bolt hits the sides when the bottles are moved. The initial design had a bolt with a nut placed in the bottle for mounting horizontally on the headrope of a gillnet. A redesign (Fig. 2a) of the bottle pinger instead has the bolt suspended on a string inside the bottle, attached by a screw to a cork manufactured from disused styrofoam floats (see Appendix X for full description, Berggren *et al. In Review*). This design is mounted vertically in the net (Fig. 2b) with the neck of the bottle tied to the head rope and the bottom

to the net mesh. When the bottle moves, the string inside acts as a pendulum making the bolt hit the side of the bottle which produces a “clinking” sound. The acoustic sound emissions of the glass bottle were measured in an anechoic testing tank at Newcastle University, UK. The bottles produced short, frequent but random, sounds with a spectral density of 130-160dB re 1uPa/VHz @ 1m below 10 kHz with additional audible components up to 30 kHz. For comparison, we tested one of the most commonly used 10 kHz commercial electronic pingers which has been used to reduce dolphin bycatch in drift gillnet fisheries (Amir 2010; Mangel *et al.* 2013). This pinger emitted a 10 kHz sound, every 4 seconds and the maximum spectral density did not exceed 140 dB re 1uPa/VHz @ 1m. The sounds produced by the bottle alarms should facilitate gillnet detection by dolphins at sufficient distance to avoid entanglement.



Fig. 2a and b. Glass bottle with pendulum metal bolt produce a sound similar to electronic pingers designed to reduce dolphin bycatch in gillnets.

One of the strongest sonar targets in water is an air bubble due to the acoustic impedance mismatch at the air-water interface. We investigated the use of air-filled plastic water bottles (500ml) as a sonar target/acoustic reflector (see Appendix 10 for full description, Berggren *et al.* *In Review*). The plastic material of the bottle has similar acoustic impedance to water and hence the reflection would be produced by the air inside the bottle. We synthesized a typical dolphin echolocation signal, a broadband “click” signal centred on 70 kHz, which we transmitted in the test tank at Newcastle University and recorded the sound reflected by the plastic bottle (Fig. 3). The same recording equipment was used as described above for the glass bottle. The plastic bottle produced a strong reflection (-27 dB target strength) of the simulated dolphin click. The reflection is comparable to that of a 100cm yellowfin tuna (*Tunnus albacares*). The recommendation is to deploy the plastic bottles on the bottom part of driftnets on the sink line to maximise acoustic reflection away from additional noise from wave action at the water surface. The bottle acoustic reflectors should facilitate gillnet detection by echolocating dolphins at sufficient distance to avoid entanglement.

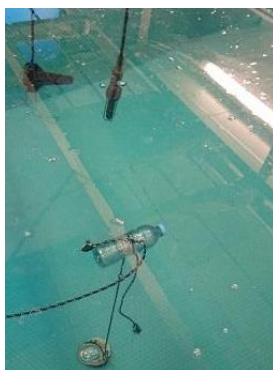


Fig. 3. Measuring the acoustic impedance (reflection) by plastic bottle from a dolphin click signal in the test tank at Newcastle University.

Activity 3.2 Trials of bottle pingers and acoustic reflectors to reduce dolphin bycatch in gillnets

The small scale (artisanal) Zanzibar driftnet fishery uses small (6-10m) sail or outboard powered boats fishing with a single 400-1100m multi-filament (7x7cm mesh size) driftnet which is set and hauled by hand (Fig. 4). When fishing, the net is attached to the vessel drifting with the tide. Fishing is only conducted during the night and the soak time is 5-8 hours. Each vessel normally set and haul their net once during a night. The main target species for the fishery are large pelagic fish e.g. tuna, swordfish & kingfish.



Fig. 4. Driftnets used in Zanzibar small scale fishery are typically 400-1100m multi-filament gillnets used at night to catch (7x7cm) driftnet large pelagic fish e.g. tuna, swordfish & kingfish.

2016 pilot trials in Zanzibar

We had planned to conduct the full bycatch mitigation field trial of the recycled bottles in the Zanzibar driftnet fishery in Menai Bay between April and June 2016. The planned total effort for the trial was 1800 driftnet hauls (one haul = one vessel fishing for one night): 600 control hauls (vessels fishing with unmodified nets, 600 hauls with glass bottle pingers (in this trial we used the initial glass bottle design which had a bolt with a nut inside and the bottle mounted horizontally on the head rope of the gillnet every 50m), and 600 hauls with plastic bottle acoustic reflectors (plastic bottles mounted at the bottom of the net on the sink line every 50m). The expected dolphin bycatch in the 600 control net hauls was 600×0.02 dolphins per haul = 12 dolphins.

Before arriving in Zanzibar we sourced and brought with all the necessary field equipment from the UK (e.g. cameras and field forms for observers to document fishing effort and catch). We held meetings with our local collaborators, arranged the necessary permits and set-up meetings with the fishermen. We engaged 42 vessels and 42 observers from six villages (Uzi, Ng'ambwa, Ugunja Ukuu, Kikungwi, Kizimkazi Dimbani and Kizimkazi M'kunguni) in Menai Bay. We trained the fishermen and observers, during several workshops held in participating villages, in how to use the bottles in their nets, conducted extensive pilot trials in the driftnets. Vessels were randomly assigned to be control, or to equip their net with either glass bottle alarms or plastic bottle reflectors. Vessels were also randomly assigned an independent observer to record fishing operation and catch. However, it later turned out that the vessels swapped observers to have the observers they knew. The start of the experiment was repeatedly delayed by the aftermath of the re-run of the Zanzibar general election followed by Ramadan. The trial was eventually started but unfavourable weather conditions resulted in very low fishing effort. We discussed the situation with our local collaborators and eventually decided that we should postpone the experiment until early January 2017, the time of the year with the most favourable weather conditions and highest fishing effort with driftnets. We organised meetings with the fishermen and explained the situation and they agreed that it would be best to postpone the trials. The total effort that was realised before cancelling the trial was 84 hauls. No dolphin catch was recorded during this pilot trial. The cancelled trial produced very useful information which was used to redesign the glass bottle pingers and to improve the logistics for implementing the trial in 2017.

Trials in Zanzibar January-March 2017

We returned to Zanzibar in early January 2017 with the redesigned glass bottle pinger (see above) and after discussion with our local collaborators it was decided to implement the trial in only two villages, Kizimkazi Dimbani and Kizimkazi M'kunguni, where we engaged 36 driftnet vessels. We planned to have

the same trial effort as in 2016 with a total of 1800 hauls (600 control, 600 with glass bottle pingers and 600 with plastic bottle acoustic reflectors). This time we made a stipulation that participating vessels had to agree to have a randomly assigned independent observer to record fishing operation, effort and catch. Observers were asked to take pictures of the net at the start and end of the haul for verification of treatment (control, pinger or reflector equipped net) and to calculate soaktime (recorded on data form and time on digital images). Vessels were randomly assigned to be control, or to equip their net with either glass bottle alarms every 50m on the headline, or plastic bottles reflectors every 50m on the sinkline (Fig. 5). In addition, six vessels (2 control, 2 with pingers and 2 with acoustic reflectors) were also supplied passive acoustic recorders (CPODs, www.chelonia.co.uk) to be deployed at the end of their driftnet to record occurrence of dolphins as indicated by recorded dolphin echolocation click trains (Temple *et al.* 2015). Finally, we had one trial coordinator in each village who met the observers after each night's fishing to download camera images and to collect data forms.

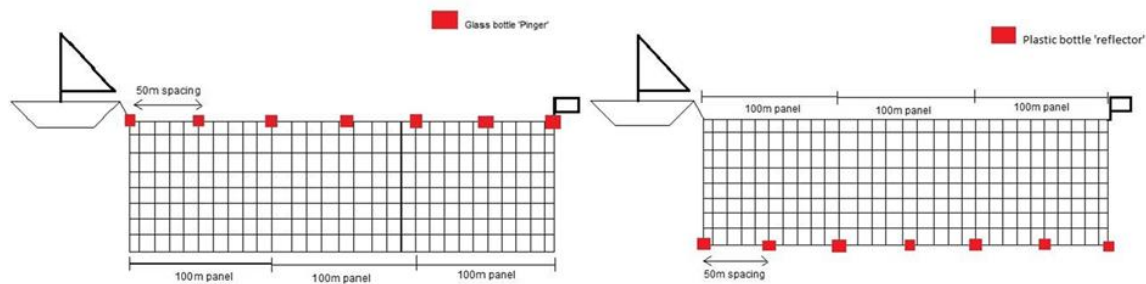


Fig. 5. Driftnet vessels were randomly assigned to be control, or to equip their net with either glass bottle alarms every 50m on the headline, or plastic bottles reflectors every 50m on the sinkline.

Data analysis

To ensure correct species identification, photographs were used to identify the species as close to species level as possible (Appendix 1). If photographs taken of catch conflicted with the information on the catch form, then the digital image was used to inform on the catch composition (species and number caught) for that haul. Where it was not possible to identify the catch due to a lack of photographs or low image quality, the haul was not used in the analyses. Once identified, the target catch was grouped as: tuna (*Euthynnus affinis*; *Thunnus albacore*; *Katsuwonus pelamis*); billfish (*Istiophorus platypterus*; *Makaira indica*; *Xiphius gladius*); and kingfish (*Scomberomorus commerson*).

Net length (km), set and haul times, and soak time were recorded/calculated for each haul. Catch data were compared among treatments as catch per unit effort (CPUE) by dividing catch for each haul by net length (km) multiplied by the soak time (h).

The effect of the two mitigation devices upon the target catch was analysed using a generalised linear model (GLM) using the R statistical software, with the *MASS* package. Tuna models were fitted with the negative binomial family, Billfish and Kingfish models were fitted with a binomial distribution, model fit was checked with likelihood ratio.

Results

The trial resulted in 1107 driftnet hauls (401 control, 394 pingers & 312 reflectors) for analysis of dolphin bycatch. The expected dolphin bycatch in the 401 control hauls based on previous observer programme data in the same fishery was 8 (401 x 0.02 dolphin catch/haul). Despite dolphin presence in the area during the experiment (as indicated by limited data collected using the CPODs and anecdotal information from fishermen), no dolphins were caught in the nets during the trial. Thus no conclusions can be drawn as to the effect of mitigation devices on dolphin bycatch.

A total of 26 different species (2389 specimens) were identified (Annex 1). The trial had sufficient information (e.g. photographs of sufficient quality) for 860 hauls to analyse target species catch among treatments (321 control, 292 pingers and 247 reflectors hauls). There was a significant difference in the

Tuna catch between treatments, with reflector hauls showing a significantly higher catch rate of Tuna than the control hauls ($Z=5.029$, $p<0.001$). There was no significant difference between the treatments for the catch rates of Billfish or Kingfish. Increasing net length ($Z=-2.213$, $p=0.027$) and soak time ($Z=-6.244$, $p>0.001$) had a negative impact on the CPUE of Tuna, conversely soak time had a positive impact on CPUE of Billfish ($Z=3.803$, $p<0.001$).

2017 Kenya trials: Wambiji N

Additional trials were conducted in Ngomeni, Kenya between August and October 2017. The observer equipment used in Zanzibar (digital cameras, protocols for recording catch and the corks with metal bolts that make the sound in the glass bottle pingers) were shipped to Kenya from Zanzibar. We engaged 15 vessels/fishermen that were instructed to use the same experimental design as in Zanzibar. Unfortunately, when the trial was implemented in Kenya independent observers were not placed aboard the vessels to record effort and catch. When reviewing the collected data from the 397 hauls fished during the Kenya trial it became apparent that there were several issues regarding how the pingers and reflectors were used in the nets and how the data were recorded. No further data analyses were therefore conducted on these data.

Conclusions and recommendations

Without further trials we are unable to conclude whether the glass bottle alarms and/or plastic bottle reflectors reduce dolphin bycatch in gillnets. However, we can conclude that the plastic bottle reflectors had a positive effect on target fish catch (tuna) and that participating fishermen in general were positive to using bottles in their nets. We have received new funding from the US Marine Mammal Commission and NOAA BREP to continue this research and will commence new trials with both the glass bottle alarms and plastic bottle reflectors in a driftnet fishery Peru (2019-2020), a bottom set gillnet fishery in Argentina (October 2019-2020), both with very high recorded dolphin catch that should facilitate enough power in the trials to conclusively test the bottles efficacy to reduce dolphin bycatch. We will also conduct further work in Zanzibar as part of this new project but the details of this will depend on the outcomes of the trials in Argentina and Peru.

Further Information

For more detailed information on the analysis and outputs resulting from Activity 3.1 and 3.2 please see Annexes 10 and 11.

Activity 3.3: Wambiji N and Kadagi NI

Abstract

Bycatch in Small Scale Fisheries (SFF) remains poorly defined, however in the context of this study we refer to the following taxa when using the term bycatch: Whales, Dolphins, Sharks, Rays, Sea Turtles and Billfish.

Several studies have suggested the use of circle hooks for bycatch mitigation in longline fisheries. This study examined the efficacy of circle hooks as a mitigation tool, undertaking trials in SFF longline fisheries in Kenya between February and April 2018. Circle hooks, J-hook styles and a combination of both were tested to evaluate their catch performance with target and non-target species. The circle, j-hooks and the combination of both hooks, “combo” landed a variety of megafauna species. J-hooks had greatest landings and average catch per unit effort (CPUE), but circle hooks but that circle hooks had higher CPUE than J-hooks in some months.

Our results show that haul number, depth fished, hook type and hours fished had significant impacts on the CPUE of megafauna species. Further, our findings indicate that the use of circle hooks reduced the CPUE of teleosts and thus may be used if reduction of catch for these taxa is required. We found no evidence for any impact of hook type on the CPUE for sea turtles and dolphins.

Introduction

Bycatch of marine megafauna populations is increasingly becoming a major concern on a global scale as a result of human fishing activities (Ayala et al., 2018, Teh et al., 2015, Lewis et al., 2014). Bycatch comprises several species, from lower trophic-level species to marine megafauna that are important for the functioning of marine ecosystems and services. The low productivity coupled with the long-life cycles of megafauna species such as whales, dolphins, sharks, rays, and sea turtles make their populations vulnerable to overexploitation resulting in declining populations (Lewis et al., 2004). There is need for management strategies that mitigate the adverse impacts of bycatch on marine megafauna especially in areas characterized by the presence of a high overlap between marine megafauna species and fishing activities such as the Western Indian Ocean (Kiszka et al., 2009, Kiszka and Muir, 2006).

The Western Indian Ocean (WIO) is characterized by a high diversity of marine megafauna though limited information on the catch status and extent of threats at regional, national and local scale exists (Kiszka, 2015). The lack of adequate information inhibits the prioritization and development of management measures to reduce bycatch which may have ecological consequences on vulnerable marine megafauna and lower trophic populations. Furthermore, the continued growth of SFFs in the WIO region, which are mostly under-documented and unregulated, may limit the effectiveness of bycatch mitigation especially where food security is a priority.

Artisanal fisheries account for about 75–85% of marine landings across WIO nations, with annual landings reportedly higher than what is reported to FAO, (Pauly and Zeller, 2016). Megafauna species can be indirectly or directly captured in a wide range of artisanal fishing gears including driftnets, long lines and purse seines among others which may contribute to declines the populations of these species (Mangel et al, 2010, Kiszka 2012, UNEP-Nairobi Convention and WIOMSA 2015, Temple et al, 2018). Thus, there is a need to evaluate bycatch of marine megafauna in artisanal fisheries and develop measures to reduce bycatch mortality whilst providing support for sustainable fisheries and food systems. In addition, conservation and management efforts that increase the survival of marine megafauna should be emphasized to reduce the effects of fisheries on marine megafauna.

Several studies have assessed bycatch of marine megafauna and proposed guidance to reduce the capture of these species especially in large industrial fisheries (e.g., Gilman et al. 2008). However, there are limited studies on bycatch in artisanal fisheries and mitigations methods in the WIO region (Kiszka, 2012).

Evidently, there has been the trial use of pingers to reduce dolphin bycatch in drift- and bottom-set gillnets off Zanzibar, Tanzania, although that represents the only significant mitigation measure trialled in the region (Amir, 2010). Recent research through the BYCAM project extended the use of bottle pingers through trials in Zanzibar and Kenya. Although no dolphins were recorded in the trials, fishers reported the presence of dolphins in the area which could be indicative of the effectiveness of the bottle pingers (see Section 3.2). However, additional and long-term studies are required to ascertain bycatch selectivity with regards to bottom pingers.

In the current study, we compared the efficacy of circle and J-hooks in mitigating megafauna bycatch. Studies conducted on circle hooks have elucidated that they are effective as a bycatch mitigation tool in pelagic longline fisheries, especially for marine turtles and to some extent elasmobranchs. For example, an exchange programme involving replacement of traditional J-type hooks or tuna hooks with circle hooks for fishers was championed by US NOAA in several countries Peru, Ecuador, Colombia, Panama, Costa Rica, El Salvador and Guatemala, Nicaragua and Mexico (Andraka, et al, 2013). The programme focused on two major longline fisheries in the region, which targeted dorado (*Coryphaena hippurus*) tunas, billfishes, and sharks respectively. Hence, the goal of this BYCAM component was to conduct trials for bycatch mitigation in artisanal fisheries that catch marine megafauna. Specifically, activity 3.3 compared landings and bycatch rates in coastal artisanal longline fisheries when J and Circle hooks were deployed.

Most pelagic artisanal longline use J-hooks and tuna hooks for catching various types of species. In some cases, artisanal longline fishers deploy circle hooks although the J and tuna hooks are the common types. J-hooks as its name implies, are J-shaped with its point oriented parallel to the hook shaft (see Fig 2a). Circle hooks are characterized by a rounded shape with the point oriented perpendicular to the shank (Fig 2b), while tuna hooks are almost similar in shape to either circle or a J hook, however, the point of the tuna hook is not guarded by the shaft, as is the case for J hooks. The point on a circle hook is turned in, towards the hook shank.

In line with BYCAM objective 3.1, we assessed overall landings by number and by weight from both circle hooks and J-hooks, deployed by 14 artisanal longline boats during January to May 2018 in Ngomeni, located on the North coast of Kenya.

Materials and methods

Artisanal longline and gillnet data

Fisheries-independent data from artisanal fisheries was collected from one pilot landing site at Ngomeni (Figure 1).

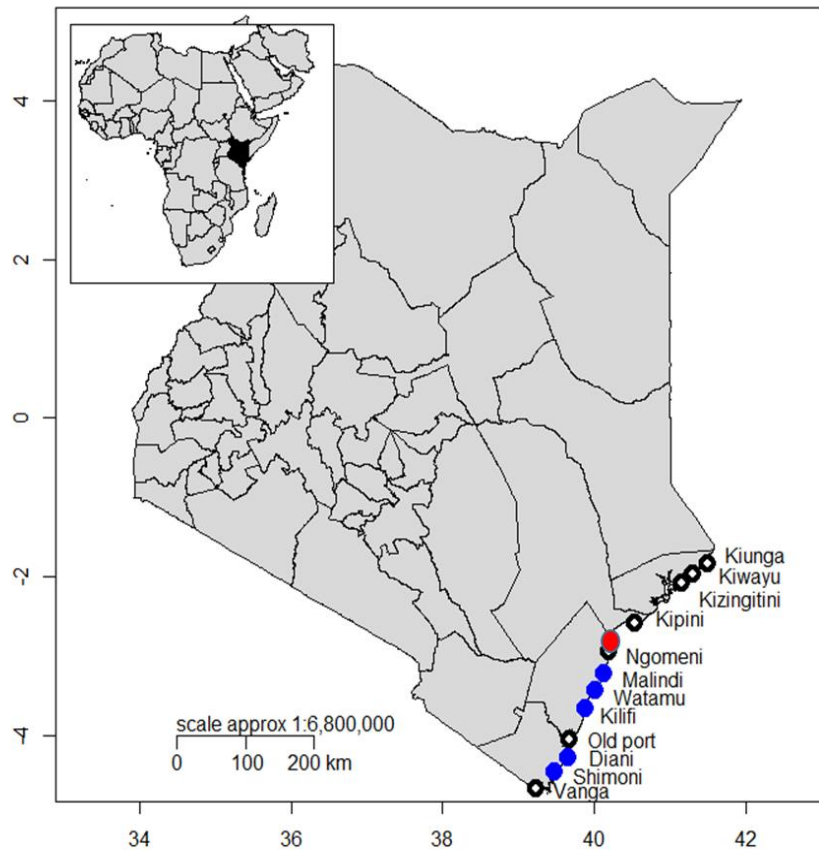
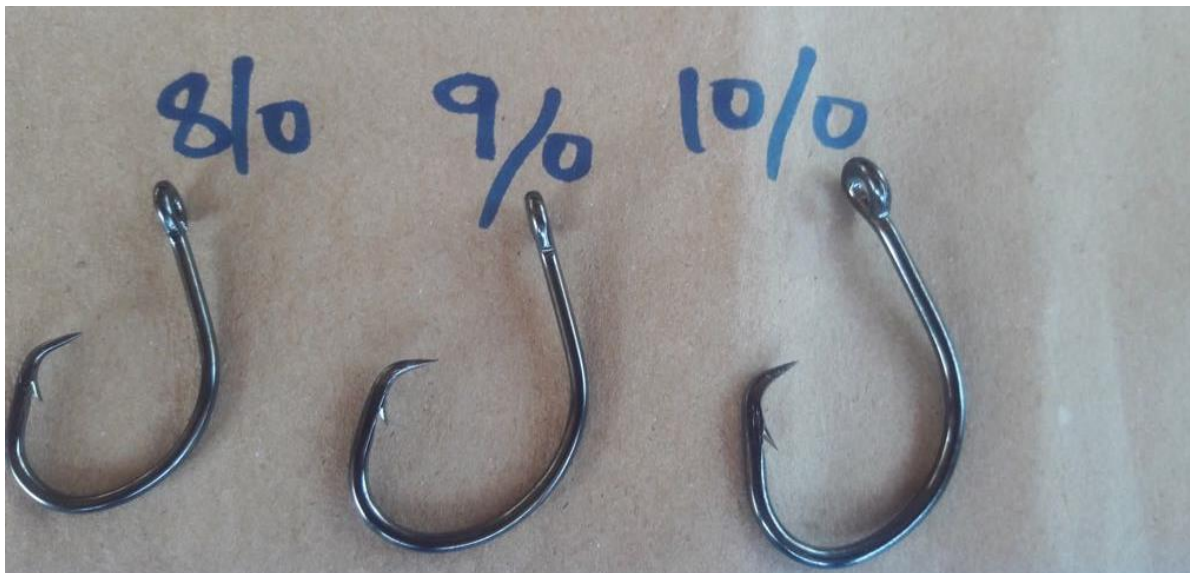


Figure 1 Map of the Kenya coast showing 8 artisanal longline and gillnet data collection sites (Kipini, Kiunga, Kiwayu, Kizingitini, Ngomeni, Watamu, Old Port, and Vanga). Pilot area Ngomeni marked in red.

Sampling design

For the purpose of this study the following taxa were considered as bycatch: whales, dolphins, sharks, rays, and sea turtles. All other catches were considered to be target catch. Three sizes of circle hooks (8/0, 9/0, and 10/0) and two sizes of J-hooks (5/0 and 6/0) were tested respectively with each type of hook deployed separately or combined. Fishers deployed each type of hook separately (either J or Circle hooks), or a combination of both ("Combo"). All landed catch was identified and measured to the weight and total fork length, the time of haul back, hook type, number of hours spent fishing, and the number of hooks used in each haul. Data on the socio-economic backgrounds of fishers were also collected.



2a



2b

Figure 2a) Images of the two J-hooks (Mustard Big gun, size #5 and #6) experiment and b) the circle hook (Bkk Bait Packs Circle Super Slide Fishing Hooks, size #8.0, #9.0, #10)

2.3 Data analyses

Catch rates were expressed as catch-per-unit effort (CPUE) for both number and weight of catches per hook per hour fished. The median CPUEs by hook type were calculated and are presented. We then used a generalised linear regression model to determine whether there were statistically significant differences in catch rates between the hook types for each taxon.

We used generalised linear regression to assess the effect of variables on the performance of the CPUE from each hook type. Other independent variables considered include hook type, depth of fishing, distance from shore, soak time, and month. CPUE was set as a response variable. The model for the relationship between CPUE, denoted as Y and model coefficients was as follows:

$Y = \beta_0 + \beta_1 X_1 + \beta_2 X_2 \dots \beta_k X_k + \varepsilon$	
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where Y is the response variable (CPUE of taxa), X_k are the explanatory variables (haul number, hook type, depth of fishing, time fished, and area of fishing), β_0 is the intercept, β_k are the coefficients that were estimated by maximum likelihood, and ϵ represents the error structure. The variables were fitted as fixed effects by using the lme4 library in R. The model was fit using gaussian error structure given that the average CPUE had been log transformed to meet the assumption of normality.

Results

Overall, fourteen boats participated in the study and effort totalled 5289 hook hours for J-hooks and 3200 hook hours for circle hooks. Ten fishing grounds were visited by the fishers over the sampling period, although the North Kenya Banks was the most fished, followed by Ras Ngomeni, Ziwayu, Mbwanyongo, Kiwayu, Lamu, Ngomeni, Watamu, Mambui, and Kiunga. The average distance fished from the shore from was 21.8 nautical miles while the average depth fished ranged between 63 m and 283 m depending of the fishing ground. Results from interviews showed that most of the fishers had seen circle hooks before, but they did not use them since they thought that they were expensive and perceived that using circle hooks would reduce their catch rates. All fishers used mostly J hooks which had been locally made from wires. A variety of marine megafauna (sharks, rays, turtles and a dolphin) were caught and landed during the study (Table 1 & 2).

Table 1: Species caught during the trials using circle hooks and J-hooks

Taxa	Circle hooks	J-hooks	Grand Total
Teleost (All)	965 (5187kg)	3226 (15,607kg)	4191 (20,794kg)
<i>Billfish</i>	13 (275kg)	14 (474kg)	27 (749kg)
<i>Tuna</i>	16 (218kg)	12 (176kg)	28 (394kg)
Dolphin	1	0	1
Ray	19 (294kg)	0 (0kg)	19 (294kg)
Shark	17 (282kg)	60 (895kg)	77 (1,177kg)
Turtle	2	0	2

Table 2: Mean ($\pm$ SD) Catch Per Unit Effort (hook/hour) of notable marine megafauna taxa and cumulative teleost catch in the study trial.

Taxa	Circle Hook		J-Hook	
	Number	Weight (kg)	Number	Weight (kg)
Teleost (All)	11.8 $\pm$ 26.1	63.3 $\pm$ 113.1	33.6 $\pm$ 81.6	162.6 $\pm$ 344.6
<i>Billfish</i>	0.16 $\pm$ 0.40	3.35 $\pm$ 9.46	0.15 $\pm$ 0.43	4.94 $\pm$ 17.36

Tuna	0.20±0.82	2.66±12.59	0.13±0.76	1.83±10.69
Ray	0.23±1.01	3.59±18.80	0±0	0±0
Shark	0.21±0.56	3.44±10.81	0.63±1.53	9.32±35.71

Landings from both J and circle hooks had a similar species composition. A total of 42 different families were recorded (Figure 3 & 4). The top families caught by circle hooks were Lutjanidae (55.0%), Nemipteridae (19.0%) and Serranidae (7.8%), while the J-hooks mostly caught Lutjanidae (61.7%), Nemipteridae (19.1%) and Serranidae (3.7%) (Figure 3 and 4).

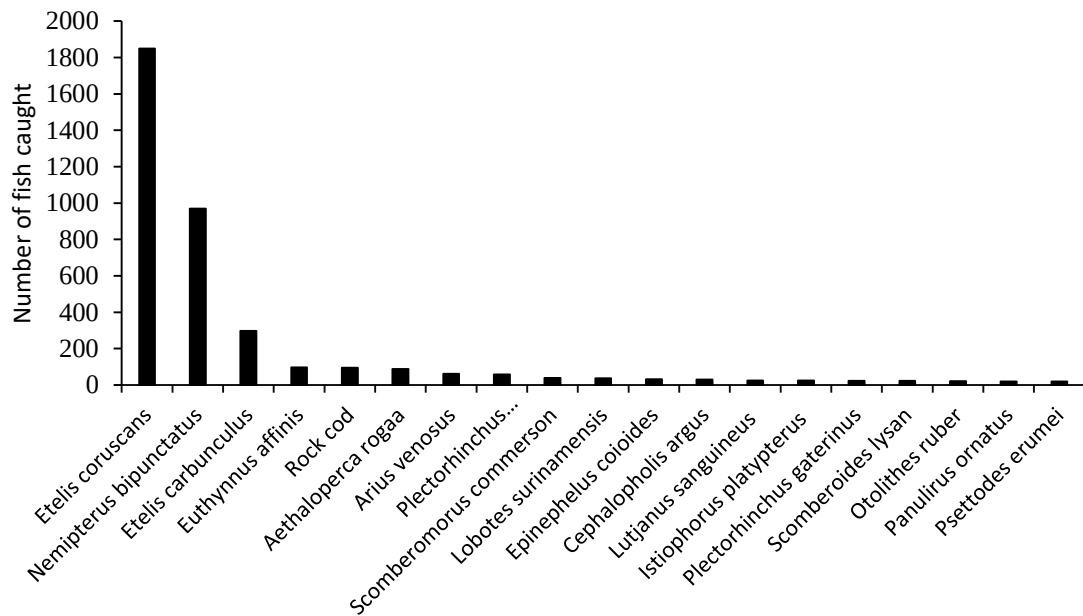


Figure 3: Number of species landed using J hooks, with small and medium pelagic fish species dominated the landings.

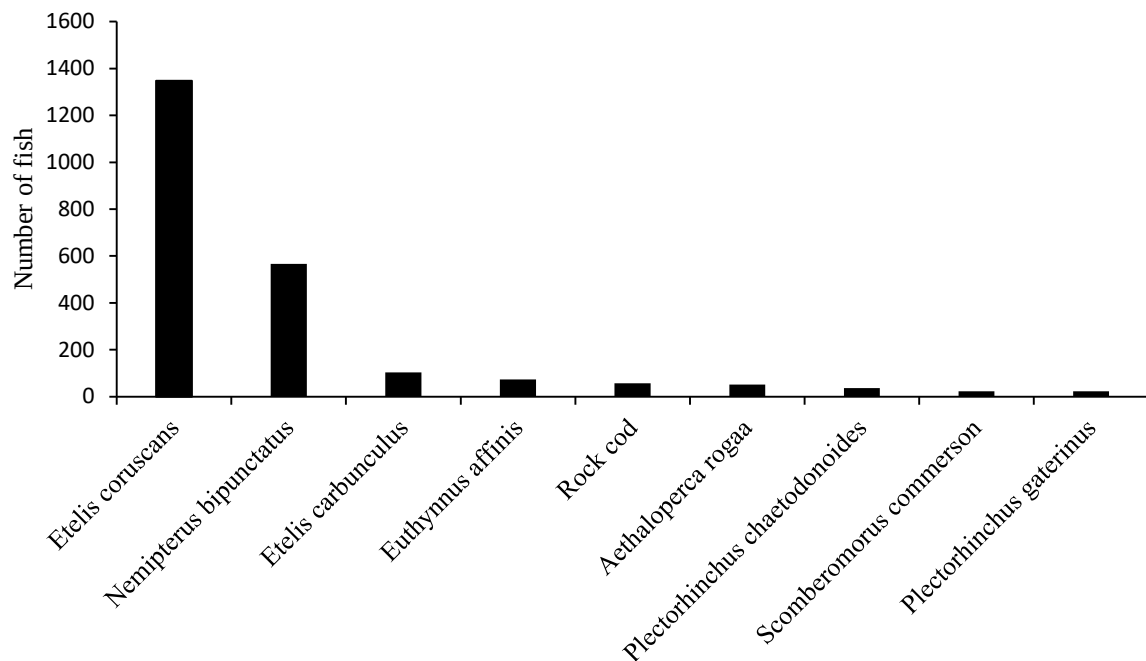


Figure 4: Number of species landed using circle hooks, with red snapper (*Etelis coruscans*), and the delagoa threadfin bream - *Nemipterus bipunctatus* dominating the catch.

The most common species caught in circle hooks were the long tail red snapper - *Etelis coruscans* (42.5%), the Delagoa threadfin bream - *Nemipterus bipunctatus* (22.3%), deep-water red snapper - *Etelis carbunculus* (6.8%), and kawakawa - *Euthynnus affinis* (2.2%).

Notably, the trial included 77 sharks dominated by species belonging to family Carcharhinidae, 19 rays (including 6 *Aetobatus narinari*), two sea turtles (loggerhead turtle *Eretmochelys imbricata*) and one dolphin (*Tursiops truncatus*). Specifically, the presence of billfish species was important for our collaborators, the African Billfish Foundation (ABF), who had previously tested the circle hooks in the recreational fishery in Kenya. The billfish encountered were *Istiophorus platypterus*, *Makaira indica* and *Xiphias gladius*.

Generalised linear models were run for both teleosts catches and shark catches, for other taxa the occurrence of catch events (i.e. hauls with 1 or more individuals caught) was too few for reliable comparison. Our results showed no significant effects of hook type on CPUE (for either number or weight) for teleost or shark catches. However, there was a potential trend in CPUE by weight of teleosts in J-hooks being greater than that of circle hooks ($Z=1.874$, $p=0.061$). Month was the only factor that significantly influenced any of the models, With CPUE by weight of teleosts decreasing in March ($Z=-3.273$, $p=0.001$) and April ($Z=-2.161$, $p=0.031$) relative to February. Similarly, CPUE by weight of sharks decreased in April ($Z=-2.180$, $p=0.029$) relative to February and March.

Our study also revealed a specific challenge in any future mitigation of bycatch in artisanal and longline fisheries in Kenya. Out of 20 fishers interviewed, 16 were migrant fishers from Dar es Salaam, Pemba and Tanga in Tanzania, while others were from Msambweni located on the south coast of Kenya. Only four were residents of Ngomeni. Although this is a small sample of respondents to make substantial inferences, the number of migrant fishers involved in the artisanal longline and gillnet fishery is a factor determining the application of bycatch mitigation methods. Ultimately, integrating any impactful mitigation method into the existing systems would require the involvement of migrant fishers rather than just residents. This could be particularly challenging given that migrant fishers are potentially less concerned with the health of ecosystems in areas where they do not reside.

Discussion

Findings from this first-time study on the use of circle hooks for mitigating bycatch showed no significant effects of circle hooks on either teleost or shark catches. Further, whilst sample sizes were too small to assess hook impacts on dolphin and turtle catch statistically, the catch of both turtles and a dolphin on circle hooks suggests that their use will certainly not stop the bycatch of these species and may have little or no impact. This was contrary to our expectations given that circle hooks have been found effective in mitigating catch and reducing post-release mortality for sea turtles and large pelagic species such as billfish (Curran et al., 2009, Prince et al., 2002). However, the evidence for a statistical trend in reduced CPUE by weight of teleosts suggests that circle hooks may reduce the average size of catch in longline fisheries, but a larger trial would be required to clarify this trend. Reductions in CPUE by weight would be in-line with many previous studies (Woll et al., 2001; Prince et al., 2002; Skomal et al., 2002; Cooke et al., 2003; Alós et al., 2009; Afonso et al., 2011; Curran and Bigelow, 2011; Hannan et al., 2013).

While the results from this study are inconclusive, the work presented here opens a discussion on the effectiveness and application potential of various bycatch mitigation measures in regard to small-scale fisheries that are multi-species and non-selective. Particular complexities for bycatch mitigation are presented by the difficulty of defining bycatch in these fisheries and the extent of migratory fishers' participation in fisheries and its implications for the application and enforcement of bycatch mitigation strategies.

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Activity 3.4: Fennessy ST

Background

TEDs have been successfully used in the industrial prawn trawling fleet in Madagascar since 2005, following an experimental and consultation phase in a collaborative exercise with scientists, industry and fisheries managers (Fennessy et al. 2008). Since the fishery targets the same species as those on the East African coast, and has many bycatch species in common, the aim was to gather information on TED use in Madagascar, to inform the planned TED trials in Mozambique. The objective was to assess the suitability of TEDs used in Madagascar by consulting with stakeholders involved in their implementation in that country, and to further identify issues which could affect TED trials and implementation in Mozambique.

Methods

The mission participants were Sean Fennessy (ORI), Lizette Sousa (IIP) and Samuel Siteo (ADNAP), and interviews with fishing companies and researchers took place in Mahajunga and Antananarivo, from 22-27 February 2015. Interviews and discussion were held with Jean-Jacques BE and Herimamy Razafindrakoto from the Centre d'Études et de Développement des Pêches (CEDP) of the Madagascar Ministry of Fisheries and Marine Resources, and directors and/or trawler captains from four fishing companies (RefrigePeche, PêcheExport, UNIMA, SomaPêche) - all in Mahajunga. In Antananarivo, Sean Fennessy also met with representatives of GAPCM (Groupement des Armateurs à la Pêche Crevettière de Madagascar), a society comprised of representatives of the fishing industry, aquaculture and various government bodies, which was instrumental in TED implementation.

Results and Discussion

Regarding the similarity of shrimp trawl specifications in Madagascar and Mozambique, numbers of trawlers in both countries are very similar. Mesh size in the body of trawls in both Madagascar and Mozambique is 60 mm, and all vessels use four nets on booms; trawl duration is 1-2 hours. Apart from the use of 50 mm codend mesh in Madagascar versus 55 mm in Mozambique, the fishing gear and its configuration in the two countries is very similar, and according to one of the captains, using the Malagasy TEDs in Mozambican trawls will not be a problem.

TEDs were inspected on trawlers in the fishing harbour at two fishing companies (Refrigepêche and PêcheExport); TEDs were already installed in the nets in preparation for the new season commencing 5 March 2015. Companies reported that they all use the same TED design (Figure 1) which uses United States specifications (although the first TEDs used in Madagascar were imported from Australia), with minor modifications (e.g. rectangular or round bars – Figure 2). Some of the first TEDs used in 2005 are still being used currently – the companies repair them, or import new TEDs, or build them at their workshops using the same design. Aluminium is mainly used; although Innox (stainless steel) is sometimes used. Bycatch Reduction Devices (big-eyes) are also compulsory although these were not observed. The design specifications are not included in legislation and could not be provided to us by the companies or GAPCM; however, obtaining the specifications was not seen to be a problem as they are based on the National Oceanic and Atmospheric Administration design, available on the internet.

TEDs are installed in an extension piece in the beginning of the codend. The main criterion is that a 45° grid angle is the maximum allowed. Individual captains make small modifications when installing the TEDs in their nets. All TEDs observed used a double flap cover for the escape opening (Figure 2), to prevent excessive loss of prawns. Some TEDs use one large float while some use two smaller floats to help grid alignment and buoyancy.



Figure 1: Typical TEDs used in Madagascar



Figure 2: Double flaps (the dark green mesh which is folded back) are used to cover the escape opening; a single large float is used on this TED; there is a mixture of round and square bars in this TED

Regarding industry perspectives on TED use in Madagascar, the director of one of the companies said that when they started to use TEDs, 50% of the catch was lost, but this improved later with experience; these days they believe the TEDs lose 5-10% of big shrimp. The TEDs and BRDs also lose big fish, which reduces profitability; in 2005-2007, shrimp catches were good and profitability was OK, but this has changed (fuel price increase, shrimp catch rates reduced) so now they need the bycatch to help profitability. The TEDs do not improve shrimp quality; TEDs helped them to be able to use eco-labels on their product, but this did not improve their access to markets (there are several eco-labels - Red Label, MSC, EcoPêche, with different criteria). The markets in Europe have improved now, so consumers have less concern about eco-labels. The company said that if they were allowed, they would remove the TEDs although they do understand that the TEDs help the ecosystem. In 2004, before TEDs were used, they caught 8 turtles (1 dead) in that season. They do not have a problem with the physical operation of the TEDs i.e. the TEDs do not cause problems to the operation of the trawl nets.

Another company did their own experiment to test the effect of the TEDs in 2013 (reported earlier under Activity 2.4), but there was no independent observer onboard; without a TED, small shrimp catch increased 5-10%, large shrimp increased 20% and retained fish increased by 30%. The director believes that reducing the predatory fish (which are released by the TEDs) has resulted in reduced prawn catch; they have disrupted the balance of the ecosystem. However, he was not able to say why prawn catches have also declined in Mozambique where TEDs are not used. They have only caught two turtles since 2009. This company reportedly met with MSC representatives who said that TEDs are not essential for MSC certification. They are concerned about profitability – some companies have closed as a result of the

poor catches. Apparently this company has recently invested in large (45 m) trawlers and their bycatch is exported; they need to show profitability.

The third company's director had no problems with using TEDs, and said they only lose 5% of catch. Eco-labelling of their products is not a concern. They use TEDs because of ecosystem concerns, and it is a legal requirement. One of his trawler captains said TEDs cause 20% loss in catch. When fishing in Nosy-Be area (some years ago), he once caught 15 turtles. The company does not use any eco-labelling and don't have any problems with marketing the product. One of the local Malagasy captains said it takes about two hours to change a damaged TED onboard (they carry four spares). They normally install the TED into the extension before leaving port, and this extension is later attached to the main trawl net in front of the codend. Branches sometimes clog the TEDs, especially on the east coast in the beginning of season.

We only met a trawler captain from the fourth company as the director was not available. He did not have problems with the operation of the TEDs, although he had only been a captain for two years, and could not say whether the TEDs caused a loss in catch.

The Malagasy legislation only says that TEDs and BRDs must be used – there are no specifications given. GAPCM also said that there are no specifications given to the companies – they are still using the same design which was implemented in 2005. The 45° grid angle is generally known and used, and bar spacing is about 11 cm. There has been no independent testing of TEDs in Madagascar after IFREMER's assistance in 2003-2005, apart from a one-day trial during an FAO TED workshop in Mahajunga in 2007. In the workshop report, the FAO and US gear experts noted that the Malagasy double flap system could be improved, and that their TED angles were less than the optimum; the grid angles need to be checked frequently, as the nets stretch with time. The perspective from GAPCM is to use TEDs so as to reduce environmental impacts by trawling; they would like to improve the operation of the TEDs so as to minimize loss of shrimp, as the requirement for the use of TEDs causes conflict at their meetings.

Regarding compliance, there is inspection of fishing gear at sea – but not on specifics such as the optimum angle for the grid. One of the captains said his boat was inspected at sea 3-4 times per season. All boats observed in the harbour appeared to have TEDs installed, or were at least ready for installation. One of the people interviewed suggested that it is possible that the trawler crews sometimes remove the TEDs at sea.

A very experienced trawler captain said that he was available to come to Mozambique to train local captains in the installation and use of TEDs; he was available from September to January. The cost of this training needs to be established. The timing of this training would have to be appropriate for the Mozambique industry, so that trawler captains and/or fishing masters were available (the best time is close to the opening of the season). However, following discussions between BYCAM participants, it was decided that since the specifications/installation of TEDs in Madagascar can be improved, it was preferred to rather obtain practical advice from another gear expert with more recent information, so as to maximize the chances of success in the planned TED trials in Mozambique.

In conclusion, the TEDs being used in Madagascar are likely to be suitable for use in Mozambican trials and their specifications are available. Opinions by the Malagasy industry varied as to the extent they considered TEDs to be affecting their fishing, and it was apparent that there was no regular checking of TED installations to optimise their use.

Activity 3.5: Fennessy ST, Kimani, EN

Introduction

Initially TED trials were to occur in Mozambique, which has a large trawling fleet and likely large numbers of turtle catches, but resistance from industry to the testing of TEDs led to BYCAM project funds being shifted to Kenya, whose companies were open to TED evaluations. Because the practicalities of TED use in Kenya had been identified as problematic (see Section 2.4), and since there was no local capacity for TED installation, a TED specialist was identified in 2016 by the then BYCAM PI Atanasio Brito from Mozambique. The specialist even went to Mozambique to discuss the feasibility of TED trials there with the stakeholders, including fishing companies – to no avail. The requirements necessary for the trials to assess the catch efficiency of a TED on a Kenyan prawn trawler were finally agreed on by the various parties (WIOMSA, KMFRI, the fishing company and the gear specialist) in April 2018, shortly before the trials commenced. The primary objective was to assess TED effectiveness in prawn trawls by conducting comparative tows of trawls equipped with TEDs against non-TED equipped trawls, identifying operational problems with TED use specific to the vessel participating in the trials. Notwithstanding their compulsory use in Kenya, the fishing industry has complained about TED use, stating that 1) not enough testing and reconfiguration of TEDs has been carried out; 2) a significant part of the target catch and valuable by-catch is lost by TEDs; 3) the TED poses a hazard to crew members especially in rough weather where it is difficult to handle a cod-end with a TED and 4) there is a lack of knowledge and capacity of the vessel owners to build or procure TEDs and lack of expertise to inspect and maintain working TEDs. Hence the objectives of the BYCAM TED trawls in Kenya were to assess these four stated concerns.

Methods

Once in Kenya, the specialist conferred with the boat owners and decided to build two different TED models in case one model was not compatible with the trawls used by the chartered vessel, F/V Roberto - all trawlers were at sea when he arrived. One was a standard super shooter model TED of 122cm height and 101.6 cm width with 10 cm bar spacing, the second a rectangular grid of 165 cm in height and 114.3 cm in width with 10cm bar spacing. Fortunately, the trawling company also has interests in a very well-equipped aluminium fabrication company, so they were able to create drawings and build the TEDs using the templates provided by the specialist. In the end, the TEDs had to be sewn into their extensions at sea since the grids were only finished being built two days before the cruise started. On inspecting the net on board the vessel, the TED design selected for the study was the flat bar super-shooter style TED (Figure 1).



Figure 1: A flat bar super shooter TED grid being mounted into a TED extension

Although there was some indication of differences in catch between the two nets before the TED was installed (see Nalovic et al 2019), the pairing of the trawls tends to control, to a large extent, between-haul variability and also variability resulting from alterations to the normal trawl configuration (Mitchell

et al. 2002). Consistent with past comparative towing experiments, the specialist sought to further reduce potential side bias, with resulting differences in catch between one side of the vessel and the other and hence compromised results, by swapping the TED position between port and starboard trawls after every five tows, to collect data from an equal number of tows on each side. A minimum of 30 successful comparative tows was required to generate sufficient statistical power for the comparison.

Prior to the formal experimental tows, a combination of different TED configurations was trialled so as to expose the captain and crew members to the different possibilities and the advantages/ disadvantages of each configuration. The different configurations consisted of using the TED with an escape opening at the bottom vs the top, single vs double TED escape hole flap covers, and the use or absence of an accelerator netting funnel in front of the TED. Also prior to the experimental 30 tows, and while the TED was being trialled, there were a series of modifications done with the objective of optimizing the gear configuration before the trials proper started. The following gear performance problems were addressed: chafing of gear on an obsolete structure of the boat, potential loss of catch due to incorrect TED angle, the use of a double flap escape hole cover, differences in the diameter of the trawl and the diameter of the TED extension, and the taper of the nets not being uniform between the port and starboard nets.

Because the objective of the study was to evaluate TED performance under conditions characteristic of the Kenyan trawl industry, the vessel captain decided where and when to fish. It was, however, agreed that fishing operations would be interrupted to make necessary trawl adjustments in order to maintain uniform fishing performance and to properly maintain the TED. Tows were conducted at fishing depths ranging from 45.5 m to 5.6 m with a majority of tows (60%) occurring at depth ranging from 8 m to 15 m. All the tows occurred in the Malindi-Ungwana Bay area. Data collected during each tow included tow duration, fishing depth, vessel position, vessel speed, and gear operation observations (i.e. torn/damaged trawls, mud tows, important seaweed or bycatch catch, twisted bags or TED problems). Tows occurred during daylight hours only and the trawls were retrieved approximately every 2 hours. The various catch categories from the starboard and port trawls were kept completely separate. The trials proper (after all modifications had been undertaken, which took 23 tows to finalize) comprised 30 consecutive comparative tows with the gear specialist on board, and a further 20 tows once he had disembarked, which were supervised by the KMFRI technician who remained on board until the end of the trip. Paired T-tests tested the null hypothesis of no difference in mean catch weights by catch category between experimental (TED) and control (non-TED) trawls.

Results

It soon became apparent that this is not just a prawn trawl fishery, since the fish catches are considered important to the company, and much of the fish catch is packed, which is atypical for conventional prawn trawling. Approximately 75% of the catch discharged at the end of the trip comprised fishes. Approximately 18 species of the category Small fish (those <0.2 kg) were retained, as well as 14 species of large fish (dominated by Pawizi = *Lobotes surinamensis*) and reported in the results below - very few individuals were larger than 0.5 kg. Unwanted Bycatch consisted of small unmarketable fish (11 species regularly seen), and small individuals of marketable species, unwanted crustaceans/crabs, small cephalopods, and jellies. This Bycatch was generally thrown overboard. Two species of prawn – called Banana shrimp (*Penaeus indicus*) and Tiger shrimp (*Penaeus monodon*) in the results shown below - were retained. No turtles were caught in any of the tows. Elasmobranchs mostly comprised very small milk sharks (*Rhizoprionodon acutus*), and seven other species were also caught in very low numbers.

The TED net catches showed a slight reduction in Small Fish (~7%, significant), a marked reduction (~68%, significant) in Big fish (Pawizi) and a slight reduction in unwanted Bycatch (~7%, significant); differences in prawn catches between the TED and Control nets were neither substantial nor significant (Table 1). Milk sharks occurred in 35 of the 50 trial tows, mostly occurring in both TED and Control nets in the same tow; in 6 tows they only occurred in the Control net, in 7 only in the TED net; catches were very similar (total catch weights in the 35 positive tows were 37.2 and 39.9 kg respectively).

The full results are available in the report (Nalovic et al. 2019) submitted by the specialist to the BYCAM team in November 2018, and which was reviewed by them.

Table 1: Results of paired t-test comparisons of Control vs TED net catches (in kg) per trawl; see text for definitions of catch categories. NB: % Diff for Small Fish should be -6.97.

50 tows								
	N	Control Naked Net		Experimental 10cm Flat TED		% Diff	95%CI	p-value
		Mean	SD	Mean	SD			
Small Fish	50	55.84	39.86	51.94	33.74	6.97	-13.96 to 0.02	0.0372
Banana Shrimp	50	10.89	7.82	10.71	7.54	-1.47	-6.87 to 3.93	0.2958
Tiger Shrimp	50	7.95	4.70	8.08	4.90	1.61	-4.58 to 7.81	0.3009
Pawizi	31	6.22	4.88	2.01	2.45	-67.7	-85.58 to -49.78	0.0001
Bycatch	49	86.45	35.37	80.14	34.57	-7.29	-13.53 to -1.06	0.0137

Discussion

Over the duration of the cruise, the captain and crew experienced very few problems in handling and operating the TED-equipped trawl. There were, however, difficulties to overcome since the trawl net's configuration was not optimal in the beginning of the experiment. After a few gear and procedural adjustments agreed upon by specialist and the Captain of the vessel, these issues were overcome. This is normal since these trawls were not initially designed to incorporate TEDs and the TED also needed some adjustments before it began to work effectively. At sea, the vessel crew worked patiently with the specialist to install the TED grids, adjust the TED angle and swap the TEDs between different sides of the boat. By the end of the first week, several crew members were very competent in this.

It was initially necessary to adjust the trawl so as to accommodate the chosen TED. With these adjustments, the net with the TED consistently maintained prawn catches similar to that of the Control net. Due to a lack of time, the other larger rectangular TED was not trialled; this larger TED may be a better fit for the trawls used by the other vessels, but this would need to be established. Although no turtles were caught, given the approved specifications used for the TED and its success elsewhere, there is no reason to believe turtles would not be excluded if these TEDs were implemented in this fishery. Larger elasmobranchs would also be excluded; the low incidence of sharks and rays in trawl catches is concerning and likely reflects the substantial artisanal fishing effort and catches, dealt with elsewhere in this report.

With the realization that retained fish were of considerable consequence to the company, the TED was rigged accordingly i.e. to ensure that turtles (and hence large elasmobranchs) would not be retained, that prawn catches would not be affected and that fish catches would be compromised as little as possible. The results of the study indicate that the use of TEDs in the Kenyan trawl fleet is feasible because there was very little difference in catch rates of prawns between TED-equipped trawls and non-TED trawls. Considering only the first 30 trial tows with the specialist on board (results not shown separately), the difference in catch rates of small retained fish between TED trawls and non-TED trawls is minor. However, the low but statistically significant difference between small retained fish catches based on all 50 observed tows does not correspond with the results from the initial 30 observed tows, suggesting that monitoring of the final 20 tows may have been problematic – particularly since the swapping of the TED between sides was not done consistently for the final 20 tows. So the results of the first 30 tows may need to be considered separately from the final 20 tows.

Attempts to reduce the loss of Big fish (Pawizi in particular) by modifying the guiding accelerator netting funnel showed some promise in this regard, but requires further consideration. The economic consequences of the loss of larger fishes due to the TED did not elicit comment from the industry at the restitution meeting held after the trials, but could be further investigated (see recommendations).

One of the objectives of this project was to build capacity in relation to TED use. This was accomplished at a variety of levels:

A local company learned how to design and manufacture quality TEDs which could help other WIO nations interested in procuring TEDs for their fisheries.

The electronic design plans of the TED manufacturing specifications are available as a result of this project.

A KMFRI technician (Rashid Anam) was trained to be able to complete comparative towing experiments, beyond TED testing e.g. for mesh size and other experiments aimed at reducing bycatch in the Kenyan fleet.

Crew members of the R/V Roberto were familiarized with the functionality of the TED and the basics of trouble-shooting TED issues. A crew member on the Roberto is capable of inserting a TED grid into a TED extension, changing the angle of the TED, changing the flap of the TED, attaching floats to the TED, inserting and removing an accelerator funnel, switching a TED from top opening to bottom opening.

It would be worthwhile to reinforce this capacity with further training which could take the form of an official exchange with NOAA in the framework of their TED program that occurs in the Mississippi Harvesting Lab facility (<https://www.sefsc.noaa.gov/labs/mississippi/ted/>). The gear specialist can facilitate this.

Recommendations

There are five actions that should be prioritized:

- KMFRI should evaluate the total catch of large fish caught by trawlers operating in Kenya to determine if the indicated lowered catch using TEDs would hinder TED implementation (see reference to management plan below).
- Reinforcement of the capacity in Kenya to:
- Conduct comparative towing experiments
- Conduct further experiments with TEDs e.g. larger TEDs for other vessels
- Install TEDs
- Maintain TEDs in good working order.

This should be done by a request from KMFRI for NOAA support for TED technology transfer in accordance with the US TED program, by requesting US TED certification.

KMFRI should consider the importance of fish catches in the trawl fishery, and adjust the prawn management plan accordingly; this must include specifications for TED installation.

KMFRI should continue to engage with the industry and the fisheries licensing department via follow-up meetings, to press for the installation of approved TEDs in the Kenyan fleet.

WIOMSA should facilitate communication with Tanzanian stakeholders on the results of the Kenya TED trials, so as to advance TED implementation there.

Further Information

Nalovic MA, Kimani E, Wambiji N, Anam R, Fennessy ST. 2019. Report on TED efficiency trials aboard a Kenyan shrimp trawler, April 5th -May 5th 2018. Unpublished BYCAM Report to WIOMSA. 62 p.

Introduction

Through the BYCAM project it has been clear that marine megafauna are caught especially in gillnet, handline and longline fisheries. The landings although only from Kenya, Tanzania and Madagascar can be extrapolated to the other WIO countries. Thus a fully queryable database is necessary to highlight areas of importance and hotspots, both in terms of risk of overfishing and population declines, and also areas where more research is required (research gaps). Creation of an interactive updatable spatial database builds upon regional, national and local programs such as SWIOFP and National Fisheries datasets (e.g. CAS data) within the WIO. These data would be complimented with large-scale habitat mapping using remote sensing data (SST, Primary Production, derived bathymetry etc.) sourced from the Natural Environment Research Council Earth Observation Data Acquisition and Analysis Service (NEODAAS).

The Indian Ocean covers about 30 per cent of the total global ocean area and being predominantly a tropical ocean, accounts for a significant part of tropical coastal biodiversity and deep-sea oceanic biodiversity in various marine ecosystems. Based on FAO statistical fishing areas, the Indian Ocean is divided into two major parts: The Western Indian Ocean (WIO) and Eastern Indian Ocean (EIO) (FAO, 1990-2015). There are two monsoonal seasons, but it is common to have a third inter-monsoonal season: North East Monsoon (NEM), from February to May; South West Monsoon (SWM) from June to October and an Inter-Monsoon Season (IMS) from November to January.

Focussing on Fisheries, bycatch has threatened megafauna species and their trends have been dipping. It is important to determine the drivers to their survival especially their distribution patterns for better management and bycatch mitigation. Other major threats included anthropogenic pressures, especially habitat loss and degradation, overexploitation or unsustainable exploitation, pollution and climate change, whose root causes are a major concern (Chaturvedi et al., 2013; Bellard et al., 2014).

Likelihood occurrence models provides for an opportunity to study species distribution using different methods (Royle et al., 2012). The aim of this study is to show that occurrence probability of species over time can be estimated by environmental parameters using minimum and maximum values hence be used for bycatch mitigation through coalition avoidance.

Methodology

Decadal averages data were downloaded from National Oceanic and Atmospheric administration (NOAA) site which host data from their ocean climatic laboratory. One-degree statistical mean data including temperature, dissolved oxygen, salinity, nitrates, phosphates and silicates were downloaded in netCDF format. The data were then pre-processed using Integrated Data viewer, a Java-based software framework for analysing and visualising geoscience data and converted to hierarchical data format (hdf) using Coastwatch Data analysis tool by NOAA ready for Analysis.

Image Processing

Averaged data for the above mentioned parameters favourable for sharks in the WIO region were used for image analysis. Each image representing a parameter for each month (January to December) was classified based on minimum and maximum values. Seasonality was also considered. The resulting raster images were then combined to perform likelihood occurrence ranking. The resulting criteria used were high, medium and low.

Environmental Variables

Statistical mean environmental data averaged in decades at one-degree resolution were obtained from NOAA data site (<https://www.nodc.noaa.gov/cgi-bin/OC5/woa13/woa13.pl?parameter=t>)

Temperature, salinity, nitrates, silicates, phosphates and Dissolved oxygen were used. The data were for the months of January to December from 1955 to 2012. In addition, range data were obtained from Ocean

Biogeographic Information System (OBIS) data site (<http://www.iobis.org/>) as shown in table 1. These range data were from peer reviewed publication of the most common sharks along the WIO region which include the Alopius, Sphyrnidae, Prionace and Hexanchus groups.

Table 1 showing the range for the different environmental parameters used.

Parameter	Min	Max	Units
Temperature	0.381	28.7	°C
Salinity	28.8	37.2	PSS
Dissolved Oxygen	0.73	7.7	ml/l
Nitrates	0.006	31.7	umol/l
Phosphates	0.03	2.6	umol/l
Silicates	0.33	80.1	umol/l

Likelihood Occurrence Model

The model was generated by classifying the raster images for all the environmental parameters using minmax ranges to obtain the suitability classes. This was done by use of raster calculator in ArcGIS where all the areas that fit the range were classified as one and the no fit area classified as zero.

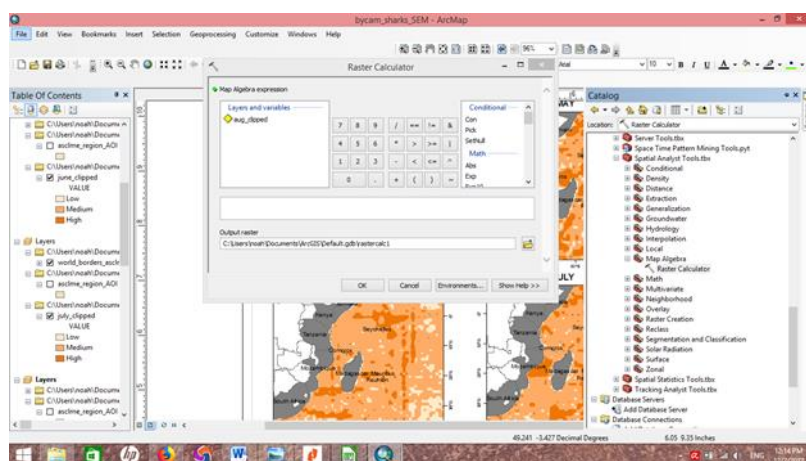


Figure 1 showing ongoing classification process using ArcGIS 10.4 arcmap.

The resulting raster images were the combined to produce range classes of the most likelihood area of occurrence to the least are for all the different months in the two seasons.

Results and Discussion

During the North East Monsoon (NEM) the month of March indicates low likelihood occurrence of sharks and same is observed during the season changeover i.e. the month of April. This period we expect low shark bycatch. Madagascar will experience increased shark bycatch during the month of October and November. During South East Monsoon (SEM), the months of June, July September will experience medium likelihood of shark occurrence. Notably we observe occurrences during the month of September in Kenya, Tanzania, Mozambique and Seychelles. NEM has highest likelihood of shark bycatch from October to February with Kenya highly affected during the month of January. High likelihood of shark occurrences is also observed during SEM in the months of May and August.

NORTH EAST MONSOON SHARKS LIKELIHOOD OCCURENCE MAP FOR WIO REGION

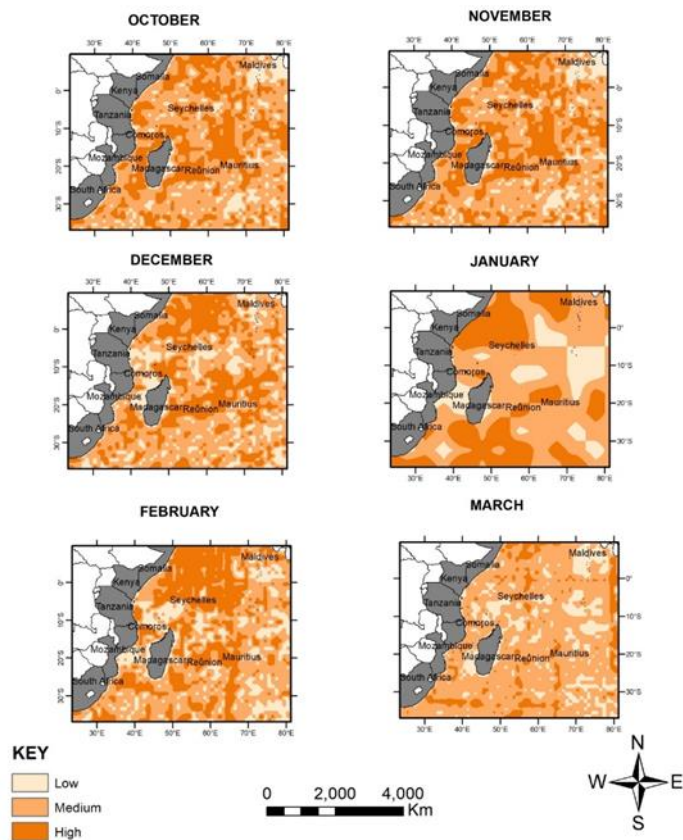


Figure 2 showing the shark likelihood occurrences during NEM.

SOUTH EAST MONSOON SHARKS LIKELIHOOD OCCURENCE MAP FOR WIO REGION

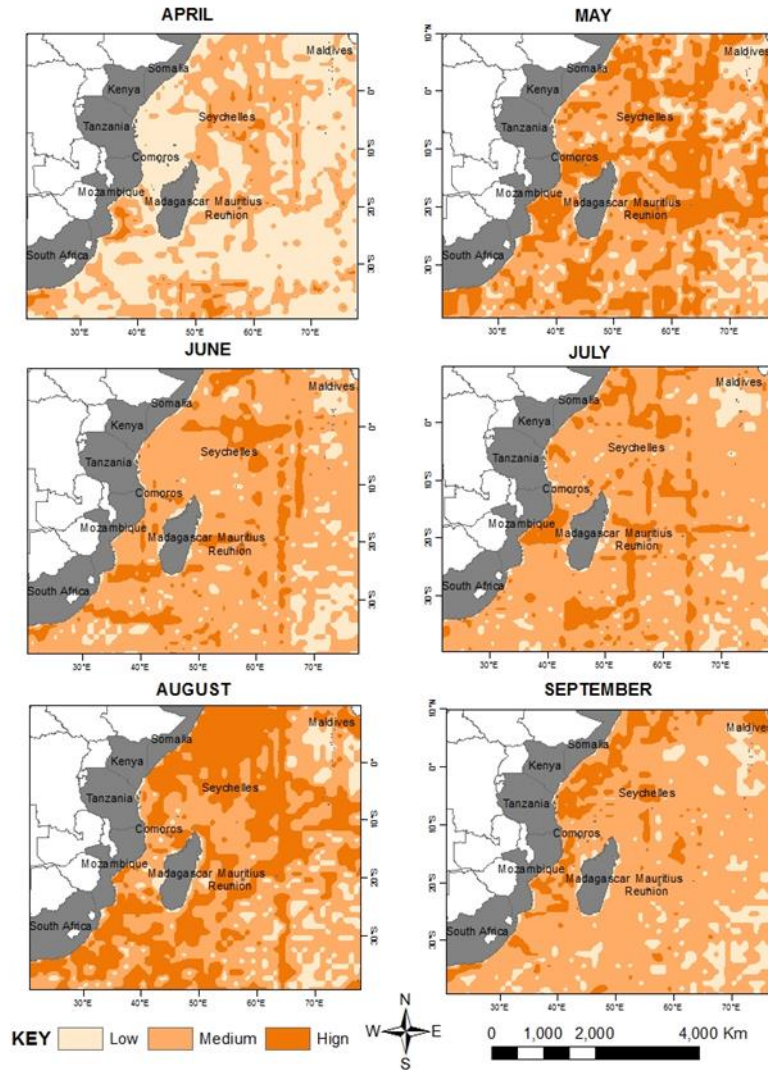


Figure 3 showing the shark likelihood occurrences during SEM.

Conclusion: It was clear that shark occurrence was seasonal and affected by environmental parameters. Once datasets are linked across the WIO region assessments can be properly conducted to give the true status of marine megafauna.

Challenges

1. Due to lack of geographical latitude and longitude data of the fishing grounds, they could not be mapped
2. A fully updatable spatial database running on a web platform relational database management system.
3. A status atlas product for the project encompassing all the components was not finalized

Recommendations

With all components of the BYCAM project having completed their field work, thematic maps will be produced (e.g. fishing grounds, migratory patterns, survey routes, fisheries effort, gear usage, bycatch species etc.) including web/online atlas maps beyond the project time. A fully queryable database will be used to identify areas of importance and hotspots, both in terms of risk of overfishing and population

declines, and also areas where more research is required (research gaps). This will also be used to extrapolate the project findings to the entire WIO region and give a regional assessment of bycatch.

Outputs

Several datasets and maps from known datasets were linked to the website. These included

- Artisanal Fisheries Catch both reported and unreported including Sharks and Rays
- Industrial Fisheries Catch both reported and unreported including sharks and Rays
- Subsistence Fisheries Catch both reported and unreported including sharks and Rays
- Recreational Fisheries Catch for EEZs
- Country Level Datasets
- Kenya Artisanal Fisheries data
- Mozambique Artisanal Fisheries Data
- Other Fisheries Databases
- Statbase
- Chimaera
- Sea Around Us

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Annex 1: Marine megafauna interactions with small-scale fisheries in the south-western Indian Ocean: a review of status and challenges for research and management

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REVIEWS

Marine megafauna interactions with small-scale fisheries in the southwestern Indian Ocean: a review of status and challenges for research and management

Andrew J. Temple · Jeremy J. Kiszka · Selina M. Stead · Nina Wambiji ·
Atanásio Brito · Christopher N. S. Poonian · Omar A. Amir · Narriman Jiddawi ·
Sean T. Fennessy · Sergi Pérez-Jorge · Per Berggren

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Abstract In developing regions, coastal communities are particularly dependent on small-scale fisheries for food security and income. However, information on the scale and impacts of small-scale fisheries on coastal marine ecosystems are frequently lacking. Large marine vertebrates (marine mammals, sea turtles and chondrichthyans) are often among the first species to experience declines due to fisheries. This paper reviews the interactions between small-scale fisheries and vulnerable marine megafauna in the southwestern Indian Ocean. We highlight an urgent

need for proper documentation, monitoring and assessment at the regional level of small-scale fisheries and the megafauna affected by them to inform evidence-based fisheries management. Catch and landings data are generally of poor quality and resolution with compositional data, where available, mostly anecdotal or heavily biased towards easily identifiable species. There is also limited understanding of fisheries effort, most of which relies on metrics unsuitable for proper assessment. Management strategies (where they exist) are often created without strong evidence bases or understanding of the reliance of fishers on resources. Consequently, it is not possible to effectively assess the current status and ensure the sustainability of these species groups; with indications

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A. J. Temple (✉) · S. M. Stead · P. Berggren
School of Marine Science and Technology, Newcastle
University, Newcastle-upon-Tyne, UK
e-mail: andrew.temple@ncl.ac.uk

J. J. Kiszka
Department of Biological Sciences, Florida International
University, Miami, FL, USA

N. Wambiji
Kenya Marine and Fisheries Research Institute, Mombasa,
Kenya

A. Brito
Fisheries Research Institute, Maputo, Mozambique

C. N. S. Poonian
Community Centred Conservation (C3), London, UK

O. A. Amir
Ministry of Agriculture, Natural Resources, Livestock and
Fisheries, Zanzibar, United Republic of Tanzania

N. Jiddawi
Institute of Marine Sciences, University of Dar es Salaam,
Zanzibar, United Republic of Tanzania

S. T. Fennessy
Oceanographic Research Institute, Durban, South Africa

S. Pérez-Jorge
Watamu Marine Association, Watamu, Kenya

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of overexploitation in several areas. To address these issues, a regionally collaborative approach between government and non-governmental organisations, independent researchers and institutions, and small-scale fisheries stakeholders is required. In combination with good governance practices, appropriate and effective, evidence-based management can be formulated to sustain these resources, the marine ecosystems they are intrinsically linked to and the livelihoods of coastal communities that are tied to them.

Keywords Bycatch · Conservation · Elasmobranch · Livelihoods · Mammal · Turtle

Introduction

Large marine vertebrates such as marine mammals, sea turtles and chondrichthyans are highly vulnerable to non-natural mortalities resulting from anthropogenic activities, especially fisheries (Lewison et al. 2004; Read et al. 2006; Żydelis et al. 2009). This is a result of the mainly k-selected life history displayed by these species groups: comparatively long-life, high natural survivorship, slow growth, late maturity and low fecundity. Chondrichthyans, comprising the chimeras and elasmobranchs (sharks and batoids), marine mammals (specifically cetaceans and sirenians) and sea turtles represent some of the most threatened animal groups (Online Resource 1). Using IUCN Red List criteria, both marine mammals and sea turtles represent a relatively small number of species (92 and 7, respectively) with high levels of vulnerability. In contrast, chondrichthyans combine large numbers of species (546 batoids, 475 sharks and 46 chimeras) and high levels of vulnerability with the highest proportions of Data Deficient and lowest of Least Concern status of any vertebrate class (Dulvy et al. 2014; Hoffmann et al. 2010; IUCN 2016). These species provide vital marine ecosystem services at various levels. As apex and meso-predators across a number of food webs they affect community structure and dynamics (Heithaus et al. 2008; Kiszka et al. 2015), and as grazers impact seagrass systems and nutrient cycling (Aragones et al. 2006; Burkholder et al. 2013; Preen 1995). Therefore, the loss of vulnerable marine megafauna has potential consequences for ecosystem structure and function, with

implications and impacts across multiple spatiotemporal scales.

The complex interrelationships between marine megafauna and human impacts on the marine ecosystem make simultaneously managing the use of marine resources and protection of these species especially challenging. Fisheries are widely considered the greatest threat to vulnerable marine megafauna (Dulvy et al. 2014; Lewison et al. 2004; Read et al. 2006; Wallace et al. 2010). Many are non-target species, widely perceived to be of low value and are often viewed as a nuisance by fishermen, especially in industrial fisheries. In contrast, others, mainly elasmobranchs, are targeted in a range of coastal and oceanic fisheries, particularly for their fins and other products including meat and gill plates (Couturier et al. 2012; Musick 2005). For many fisheries, particularly small-scale fisheries in developing nations, vulnerable marine megafauna species may constitute both target and non-target catch. Indeed, their categorisation as target or by-catch species may vary on a fisher-by-fisher and trip-by-trip basis. As such we herein refer to their presence in the fisheries simply as 'catch'.

The multi-gear nature of many fisheries, the perceptions of many vulnerable marine megafauna as either a nuisance or of low value, together with the illegality of catching certain species and the sometimes-secretive nature of fishermen mean that catch is largely under-reported and data are sparse in many regions, making accurate estimation of global catch exceedingly difficult. However, available estimates indicate that catches are likely unsustainable, with an estimated 0.53–0.82 million marine mammals, 0.85–8.5 million sea turtles and 63–273 million sharks caught worldwide annually (Read et al. 2006; Wallace et al. 2010; Worm et al. 2013). Gillnet and line fisheries account for the majority of marine mammal, elasmobranch and sea turtle catch (Lewison et al. 2004; Read et al. 2006). These fishing methods are relatively inexpensive, simple and effective with widespread usability.

While vulnerable marine megafauna interactions with industrial and commercial fisheries have received some attention, less is known of the magnitude and mechanisms of interaction with small-scale fisheries, herein defined as those fisheries operating either for subsistence or for income generation (artisanal) but not as part of a commercial company, particularly in

the developing regions. Globally, small-scale fisheries include some 50 million fishers (FAO 2016c), more than 95% of fishers worldwide (Pauly 2006). They are especially prevalent in the developing regions of South and Central America, Africa and the Indo-Pacific. Given their prevalence and widespread occurrence the environmental impacts of small-scale fisheries are likely significant, though they are often overlooked (e.g. Hawkins and Roberts 2004; Moore et al. 2010; Salas et al. 2007). With continued unregulated exploitation, small-scale fisheries can negatively impact the abundance, distribution and species composition of vulnerable taxa (Pinnegar and Engelhard 2008), including vulnerable marine megafauna. Thus, small-scale fisheries may lead to declines of these key species with consequences for the broader food web and ecosystem, including other species that are critical to local livelihoods.

In developing regions small-scale fisheries are of considerable socio-economic importance, particularly in rural areas where they are important contributors to the local economy (Béné 2006; Pauly 2006) and to food security. In these regions elasmobranchs, sea turtles and marine mammals were historically important sources of human sustenance and remain so in many areas (Robards and Reeves 2011; Vannuccini 1999). Elasmobranchs are most important in this respect, but hunts still exist for both sea turtles and marine mammals, often in spite of national or international laws and regulations banning these practices (Cerchio et al. 2009; Hart et al. 2013; Humber et al. 2014; Kasuya 2007; Riedmiller 2013). Vulnerable marine megafauna are an important source of income, both in fisheries and increasingly from ecotourism activities (Cisneros-Montemayor et al. 2013; O'Connor et al. 2009). However, there is a lack of information regarding the non-monetary, including cultural, value of vulnerable marine megafauna to fishers, which has implications both for decision-making regarding catch and conservation of these species and for the full understanding of their societal value.

Our aim in this paper is to review existing knowledge and status regarding vulnerable marine megafauna interactions with marine small-scale fisheries in the southwestern Indian Ocean (SWIO) region. Data were gathered from a range of sources including, but not limited to, information requests from relevant government departments in SWIO

nations, scientific and non-governmental organisations (NGO) publications and reports, international and national annual reports and databases. We discuss the likely implications of the current situation, highlighting vital knowledge gaps that need to be addressed and challenges for future research and management across the region.

Southwestern Indian Ocean profile

The SWIO, as considered in this review, consists of 8 countries (and their Economic Exclusive Zones) with broadly comparable fisheries: Comoros, Kenya, Madagascar, Mauritius, Mayotte and La Réunion (France), Mozambique, the Seychelles and Tanzania (including Zanzibar) (Fig. 1, Online Resource 2). The region's human population is projected to more than double from 155 million to 357.3 million by 2050 (WB 2016b), with coastal cities and island nations experiencing particularly high rates of population increase (Online Resource 2). Food security and income generation are therefore major policy drivers requiring sustainable solutions built on sound management practices. Coastal communities have traditionally relied on marine fishes (including elasmobranchs) as their main sources of protein, with some also making use of marine mammals and sea turtles for sustenance or as bait (Church and Palin 2003; Humber et al. 2011; Razafindrakoto et al. 2008). Marine fisheries (including mariculture) account for 0.5–30% of Gross Domestic Product (GDP), with island nations particularly reliant (Online Resource 2). However, other marine income-generating activities also contribute to local economies, particularly marine-tourism activities, worth around \$3.95 billion, including recreational fishing, whale and dolphin-watching and dive tourism (Amir and Jiddawi 2001; Divetime 2016; Gallagher and Hammerschlag 2011; Obura 2017; O'Connor et al. 2009; O'Malley et al. 2013; Pérez-Jorge et al. 2016). Some countries in the SWIO, like the Seychelles, are focusing on expanding their other food security sectors (e.g. mariculture) to reduce reliance on vulnerable fisheries especially in the face of climate change (Stead et al. 2015).

The SWIO (part of FAO Fishing Area 51) has among the highest marine species richness worldwide (Tittensor et al. 2010; Worm and Branch 2012). This diversity is threatened by increasing anthropogenic

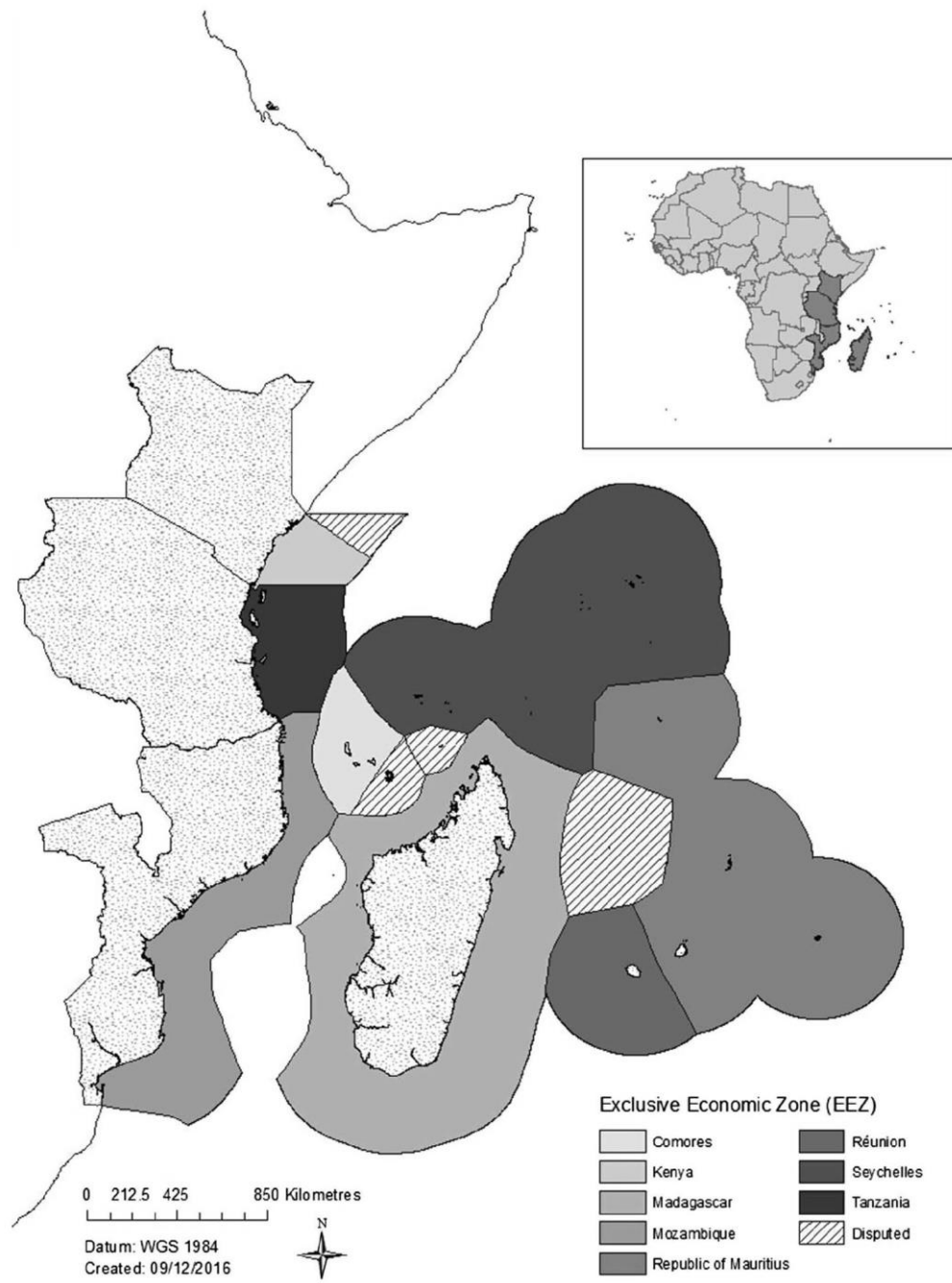


Fig. 1 The southwestern Indian Ocean: Comoros, Kenya, Madagascar, Mauritius, Mayotte and La Réunion (France), Mozambique, Seychelles and Tanzania (mainland and Zanzibar). Sources: ESRI (2014), VLIZ (2014)

pressure, especially from fisheries, and limited management effectiveness where it exists (Mora et al. 2009; Worm et al. 2013). The area contains a high diversity of vulnerable marine megafauna species but there is large uncertainty regarding status, catch and trends of many of these at both a global (Online Resource 1) and regional scale (Kiszka 2015; Kiszka and van der Elst 2015). These uncertainties, coupled with the high proportion of vulnerability and mostly decreasing trends (Online Resource 1) in assessed species is a major concern.

The risk to vulnerable marine megafauna species in the SWIO is further exacerbated by the continued expansion of the dominant yet largely poorly documented and unregulated small-scale fisheries. Small-scale fisheries officially account for 75–85% of marine landings across SWIO nations (Pauly and Zeller 2015), with annual landings reportedly 345,000–390,000 mt as of 2014 (FAO 2016b). However, independent estimates suggest gross under-reporting of landings and effort, with total SWIO landings estimated to average 165% greater than reported figures for 1950–2010 (Fig. 2; Pauly and Zeller 2015). Although these are retrospective estimates, they provide an improved assessment of the landings magnitude and serve as a useful reference point.

Small-scale fisheries of the southwestern Indian Ocean: main features and data quality

Currently the SWIO small-scale fisheries employ more than 495,000 fishers operating 150,000 assorted vessels across the SWIO (Table 1), with the largest fleets in Madagascar and Mozambique. However, this does not account for many of the unlicensed small-scale fisheries fishers, which are of substantial number (Teh and Sumaila 2013). Unlicensed and open-access fishing is a major issue for SWIO small-scale fisheries, with direct implications for the assessment of catch and socio-economic value of these fisheries and so inhibiting effective stock management. Additionally, small-scale fisheries support various other livelihoods, including: auctioneers, fish mongers, middlemen, gear repairers and fish fryers among others.

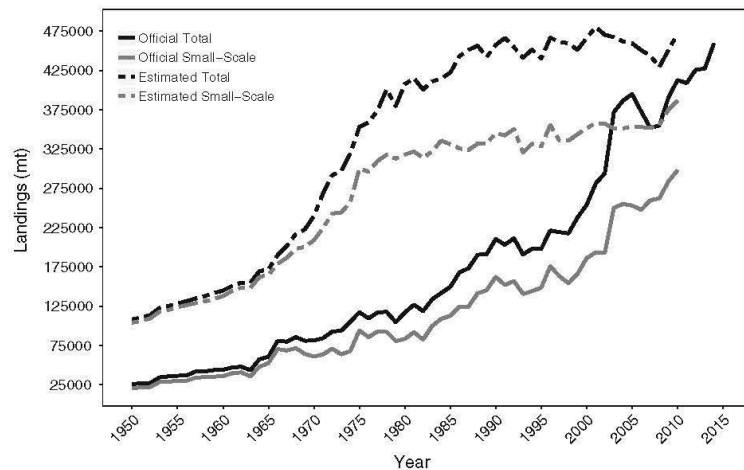
In terms of fisher participation the regional fisheries are dominated by handlines (Table 1), with simplicity, ease of use and affordability as likely drivers. However, more advanced gears are increasingly used. Specifically, fisheries in Kenya, Madagascar, Mozambique, the Seychelles and Tanzania (including Zanzibar) are using longline (demersal and pelagic) and gillnet (drift and bottom set) gears (Table 1), mostly targeting sharks and pelagic fishes.

Assessing general landings trends in the SWIO small-scale fisheries is challenging. Whilst long-term data sets are available through the FAO (Fig. 2) and national reports, the validity and quality of these are questionable given the lack of standardised and systematically collected historical data, particularly regarding effort, and unlicensed fishing. However, if we consider only data from recent years (Fig. 3; KMALF 2015; MFR 2012; SFA 2015; L'Institut Français de Recherche pour l'Exploitation de la Mer unpublished data; Tanzania Ministry of Livestock and Fisheries Development unpublished data; Seychelles Fishing Authority unpublished data) landings appear relatively stable in most nations, with a marked overall regional increase (Fig. 2). Decreasing trends are seen in the official data from Mauritius and La Réunion, reflecting the declining effort (vessel numbers and fisher days respectively) in these fisheries (MFR 2010; L'Institut Français de Recherche pour l'Exploitation de la Mer unpublished data). Conversely a rapid increase in official landings has been observed for Mozambique, likely driven by improvements in both monitoring programmes and proper extrapolation of data to the national level (Doherty et al. 2015).

Compounding the issues regarding monitoring efficacy and accuracy is the widespread commonality of national and international migrant fisheries in the region (WIOMSA 2011). These catches may be taken in one nation and declared in another, declared in both or in neither. Undeclared transshipment of catches to neighbouring markets is also common. For example catches in Zanzibar are often landed, compiled and shipped directly to markets in mainland Tanzania or southern Kenya, often following a seasonal pattern (Fowler et al. 2005; Wanyonyi et al. 2016; A. Temple personal observation). These may have significant impact on landings data and could have consequences for stock management.

The biggest stumbling block in the monitoring and management of SWIO small-scale fisheries is the lack

Fig. 2 Official and estimated landings data for SWIO nation's fisheries between 1950 and 2013. Data sources: FAO (2016b), Pauly and Zeller (2015)



of standardised data and the relatively poor resolution of landings and effort data available. The variability in basic recording metrics (Online Resource 3) hinders the comparability and summation of data at the regional scale, with only simple measures feasible for use i.e. effort can only be regionally derived through vessel count data. Measures and definitions of gear type, effort metrics and vessel types differ between countries and even within countries between years, whilst the breakdown of data by geographic region, gear and vessel types is often inconsistent over time. Data reports have variable formats and contents, and are often unclear as to whether data presented are that observed or whether they have been extrapolated to country level. Most notably data are inconsistent among reports. For example, there are a number of years (1990, 1993, 1994, 2000 and 2007) where artisanal handline and troll line numbers reported by the Seychelles to the Indian Ocean Tuna Commission (IOTC) are greater than the numbers reported by the Seychelles Fishing Authority for its entire artisanal fleet (IOTC 2016a; SFA 2001–2013). Similarly Kenya reports 165 mt of all elasmobranchs landed from artisanal fisheries in its 2007 statistical bulletin, yet reports 174 mt of sharks alone from the same fishery and year to the IOTC (IOTC 2016a; KMALF 2008). Clearly, these issues must be addressed, both at national and the regional levels, if small-scale fisheries are to be sustainably managed and vital livelihoods protected across coastal areas of the SWIO region.

Marine mammal interaction with small-scale fisheries

There is limited information available on marine mammal populations and their interaction with the small-scale fisheries of the SWIO (Kiszka 2015; Kiszka et al. 2009), but where data exists there is evidence of both targeted and incidental catch. Catches, mostly incidental, have been documented in the Comoros and Mayotte (Kiszka et al. 2007, 2010; Poonian et al. 2008; Pusineri et al. 2013; Pusineri and Quillard 2008), Zanzibar and Tanzania (Amir 2010; Amir et al. 2002; Muir and Kiszka 2012), Kenya (Kiszka 2012), Madagascar (Cerchio et al. 2009; Razafindrakoto et al. 2004, 2008), Mozambique (Guissamulo and Cockcroft 1997; Kiszka 2012) and La Réunion (Kiszka et al. 2009). To date no marine mammal catch has been reported in Mauritius or the Seychelles (Kiszka et al. 2009). Published studies identify coastal gillnet fisheries (both drift and set nets) as the main threat to marine mammals across the region, although interactions have also been documented in longline fisheries (Kiszka et al. 2009, 2010).

Understanding the true impacts of these catches requires data on capture rates and abundance estimates for any given population investigated. To date population abundances have only been estimated for two cetacean species in restricted areas of the SWIO: Indian Ocean humpback (*Sousa plumbea*) and Indo-Pacific bottlenose dolphins (*Tursiops aduncus*) in the

Table 1 Marine small-scale fishery vessel, fisher and gear data for South Western Indian Ocean nations. *Data Source:* Andriantsoa and Randriamiarisoa (2013), Chacate and Mutombene (2015), Chavance et al. (2014), ESAP (2005), de Graaf and Garibaldi (2014), Herfaut (2006), KMALF (2014a, b, 2015), MFR (2010), Ndegwa (2015), SFA (2015), Soilihi (2014), ZMLF (2010), Albion Fisheries Research Centre unpublished data; L'Institut Français de Recherche pour l'Exploitation de la Mer unpublished data, Instituto Nacional de Investigación Pesqueira personal communication)

Country	Vessels (year)	Fishers (year)	Gear prevalence				
			Measure (year)	Handline	Longline	Gillnet	Other/ unknown (%)
Comoros	3601 (2012)	Unknown	Vessels (2012)	23.25% static 23.12% trolled	—	3.11% drift	50.51
Kenya	2913/3500 (2013/2014)	12,915 (2013)	Gears (2014)	20.35% static 2.81% trolled	28.48% (strings)	9.44% mono- filament 8.41% set 5.14% drift 1.07% active	24.31
Madagascar	78,787 ^a (2012)	119,334 (2011)	Gears (2012)	67.69% “lines” ^b	—	27.23%	5.08
Mauritius	2476 (2010)	2,038 ^c (2014)	Fishers (2014)	66.00% line&trap 21.05% line/ harpoon/foot ^d	—	5.89% “large net” 0.49% “gillnet”	6.58
Mayotte	1132 (2014)	4800 (2003)	Landings (2005)	57% static 32% trolled	—	~10% encircling	~1
Mozambique	45,805 ^d (2013)	285,000 (2012)	Gears (2012)	25.75%	2.00%	37.57%	34.68
La Réunion	172 (2014)	340 (2014)	Vessels (2014)	88.37%	8.72%	—	2.91
Seychelles	424 ^e (2014)	Unknown	Landings (2014)	56.28% line 3.71% line&trap	—	20.86% encircling	19.15
Tanzania (mainland)	7664 (2009–2014)	36,321 (2014)	Gears (2014)	25.24%	17.07%	36.06% set 6.75% drift	14.88
Zanzibar	8,639 ^f (2010)	34,571 (2010)	Gears (2010)	44.11%	1.76%	13.46% drift 4.07% set	36.60
Total	151,613	495,319					

^a Interim results for 9 regions, of 22^b not broken down^c registered fishers^d vessel licenses not vessel numbers^e average vessels active/month^f 9609 vessels predicted by 2015

Menai Bay Conservation Area off the south coast south of Unguja Island, Zanzibar and in the Kisite-Mpunguti Marine Protected Area (MPA), Kenya (Meyler et al. 2012; Pérez-Jorge et al. 2015, 2016; Stensland et al. 2006); and *T. aduncus* off the south-west of Mauritius (Webster et al. 2014), around Mayotte (Pusineri et al. 2014) and La Réunion (Dulau

et al. 2017). Capture rate estimates are only available for Zanzibar, with these showing unsustainable levels of fisheries mortality for both species (Amir 2010; Amir et al. 2002). Numbers of the dugong (*Dugon dugon*) have significantly reduced across the SWIO region with only relict populations remaining in the region, the largest of which exists in Mozambique

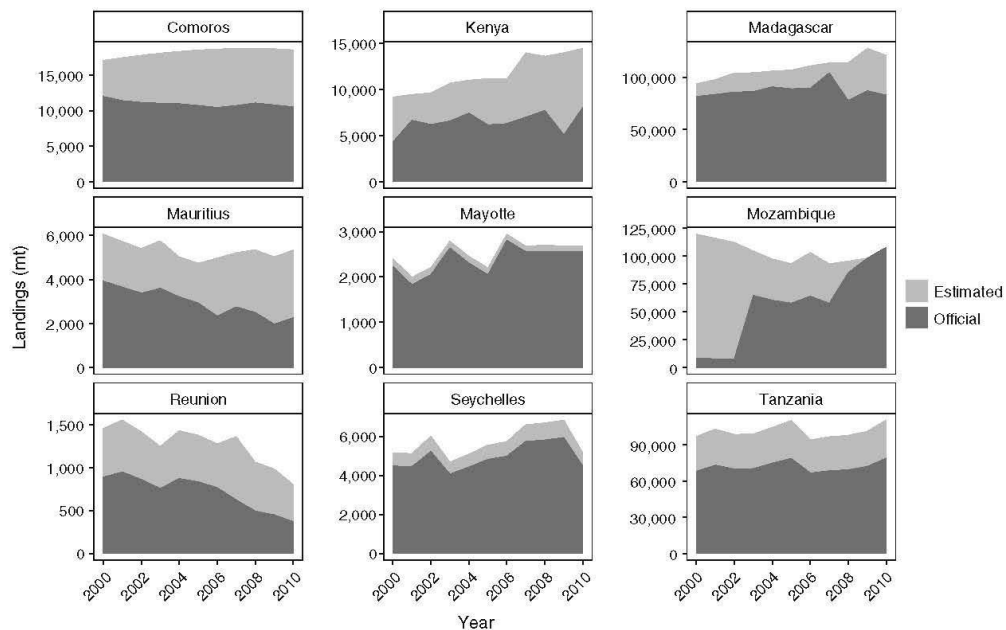


Fig. 3 SWIO small-scale fisheries landings 2000–2010. Note different scales on the y-axes. Data sources Pauly and Zeller (2015)

(WWF-EAME 2004). Evidence of ongoing catches has led to serious concern for the future viability of this species (Kiszka 2015; Kiszka et al. 2007). Of further concern is the on-going illegal hunt for marine mammals in Madagascar (Cerchio et al. 2009; Razafindrakoto et al. 2008) and Tanzania (Riedmiller 2013), possibly in Mayotte (Kiszka et al. 2009) and likely other parts of the region.

Currently there are no annual statistics relating to the catch or landings of marine mammals in the SWIO region. Minimal attention is given to these species as a component of the fisheries at a national level and there is likely an inherent reluctance to report any such catch given its illegality. Yet, this is hardly a problem restricted to this region, rather it is one at the global level.

Sea turtles interaction with small-scale fisheries

Five species of sea turtles are known to occur in the SWIO, but green (*Chelonia mydas*), loggerhead (*Caretta caretta*) and hawksbill turtles (*Eretmochelys*

imbricata) are the most common and widely distributed in the region (Bourjea 2015). Sea turtles have attracted both long-term and intensive studies in the SWIO relative to other vulnerable marine megafauna species. Nevertheless, there are still major data gaps e.g. unreliable nesting data and a lack of species abundance estimates, partially as a result of their highly mobile and complex life history preventing comprehensive population level assessment for most species in the SWIO region (Bourjea 2015). However, qualitative global assessments rank loggerhead, leatherback (*Dermochelys coriacea*) and olive ridley (*Lepidochelys olivacea*) as high risk species in the western Indian Ocean, with olive ridley and green considered to face the greatest levels of threat to survival (Wallace et al. 2011, 2013).

Three gear-types have been identified as catching substantial numbers of sea turtle, namely gillnets, prawn/shrimp trawls and longlines (Bourjea et al. 2008; FAO 2010; Wallace et al. 2013). Yet, in most countries of the region, the extent and impact of fisheries on sea turtles is poorly known, except for open ocean fisheries (Bourjea et al. 2014). Both

incidental and targeted catch of sea turtles appear widespread in small-scale fisheries (for review see Bourjea (2015) and Bourjea et al. (2008)), with the threat posed by gillnet and line gears across the region to sea turtles well established (Bourjea et al. 2008; Kiszka et al. 2010; Poisson and Taquet 2001; Poonian et al. 2008). However, there are no annual statistics of note for sea turtle capture in the SWIO. Kenya is the only reporting nation, for which it has reported 0 mt/year since 1964 (FAO 2016b). Whilst other sources of quantitative data are sparse, and for areas where data exist they are rarely comprehensive, it appears annual regional small-scale fisheries catch is in the order of tens or even hundreds of thousands (Table 2), representing a serious threat to the survival of sea turtles in the SWIO. This is compounded further by alterations and destruction of nesting beaches in some areas, sizeable egg poaching activities and hunting of nesting females, which are common across the region (Bourjea 2015).

Elasmobranch interaction with small-scale fisheries

In 2014, 34 countries across all fisheries scales reported 105,969 mt of elasmobranch landings

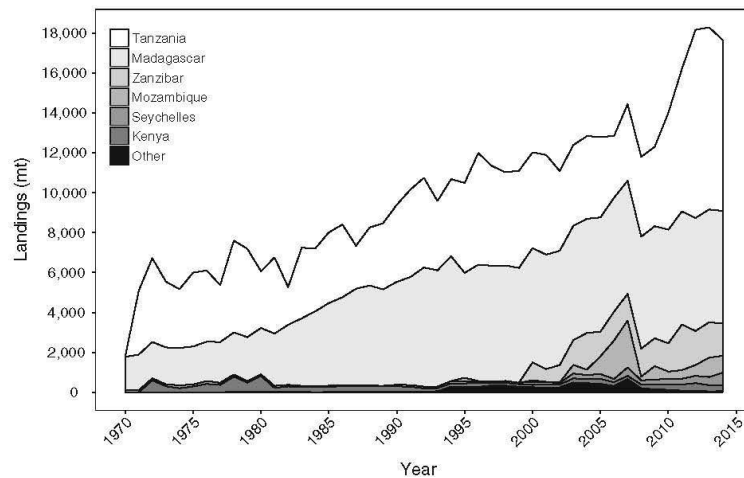
originating from the western Indian Ocean region, of which only 17,663 mt were landed by SWIO nations (Fig. 4; FAO 2016b). Of this a disproportionate amount (89.6%) was accounted for by Tanzania (including Zanzibar) and Madagascar, which together account for 62% of known SWIO small-scale vessels (Table 1). It is therefore unlikely that the reporting reflects the true proportional contribution of SWIO nations. The regional estimate is likely an underestimation, resulting from under-reporting of landings, illegal fishing and discards, and is consistent with the under-reporting of other landings in the SWIO region (Pauly and Zeller 2015). Despite the high level and year-on-year increase in landings across the SWIO (Fig. 4) little independent research has been undertaken on these fisheries.

There is a notable imbalance (>80% of peer-reviewed papers) between studies focussing on elasmobranchs in the industrial and semi-industrial fisheries e.g. (Fennessy 1994; Huang and Liu 2010; Kiszka and van der Elst 2015; Romanov 2002) and the small-scale fisheries (Molina and Cooke 2012). Published information for small-scale fisheries is generally sparsely quantified and limited to target species (e.g. Marshall 1997b; McVean et al. 2006; Schaeffer 2004). Grey literature, in the form of unpublished theses, governmental and consultancy reports, data

Table 2 Existing numerical data for sea turtle catch in the small-scale fisheries of the South Western Indian Ocean. *Data Source:* Humber et al. (2011), Kiszka (2012), Muir (2005), Muir and Ngatunga (2007), Okemwa et al. (2004), Pusineri and Quillard (2008), Rakotonirina and Cooke (1994)

Country	Scale	Gears	Year	Catch estimate	Method
Kenya	Regional— Watamu Kiunga	Net	Unknown	~600/year	Catch data
Madagascar	National—Subset of Fishers	Terrestrial trap, Harpoon, Diving, Net, Longline, Poison	1987	11,061/year (17 target fishers) 215/year (16 incidental fishers)	Interview survey
	Regional— Southwest	Net, Line, Spear	2006/ 2007	10,000–16,000/year	Landings data
Mauritius	National	Beach Seine, Bottom-set Gillnet, Line under FAD, Handline	2010	570/year	Interview Survey
Mayotte	National	All	2007	111–256/year	Interview Survey
Tanzania Mainland	Regional—Mafia Island	Gillnet	Unknown	1000–2000/year	Unknown
	National— Incidental Only		Unknown	617–6170/year	Interview survey

Fig. 4 Total landings data for elasmobranchs caught by South Western Indian Ocean nations 1970–2013. Zanzibar and Tanzania are reported separately after 2000. Data Source: FAO (2016b)



collected by NGO's and other such works exist. However, much of this information is not easily accessible and in some cases remains confidential. This highlights a need to better the flow of information to responsible organisations and into the public domain.

A variety of species are regularly taken in the region's fisheries, with the most commonly reported being blue (*Prionace glauca*) and silky (*Carcharhinus falciformis*) sharks from the industrial longline and purse seine fisheries (Smale 2008). Concurrently, a number of species are known to appear in the region's small-scale fisheries (Table 3). The species listed are influenced by ease of identification and observation bias. As such they are unlikely to accurately reflect fisheries composition but do provide evidence for a level of regional species homogeneity. Most elasmobranch landings in the SWIO region are not identified beyond basic taxonomic level and are simply grouped as "sharks and/or rays" (FAO 2016b). Thus, the data cannot support effective stock management, at either local or regional levels. This is a major constraint to decision-making on fisheries management measures as there is no reliable data for population dynamics, given that that small-scale fishers are reporting significant declines in elasmobranch abundance and catch this demonstrates a clear information gap (Kiszka and van der Elst 2015). Specific data is required to properly document both which species, and in what volume, elasmobranchs are interacting with

SWIO small-scale fisheries. Below, current understanding of elasmobranch catch and landings in SWIO small-scale fisheries is summarised by country:

Comoros

Since first being reported in 1994, elasmobranch landings in the Comoros have dwindled from 230 to 19 mt by 2013 (FAO 2016b), despite the apparent increases in fishing effort (IOTC 2016a; Soilihi 2014). Import of dried shark meat from Madagascar (Cooke 1997) is one possible driver, with Comorian fisheries known to be unable to meet the domestic demand for dried fish products (WB 2016a). Alternatively, these declines may reflect stock collapses. Blue shark is a major constituent, accounting for 26% of the 19.97 mt landings in 2012 (Soilihi 2014). Other commonly reported catch includes a variety of oceanic and coastal species; primarily oceanic whitetip (*Carcharhinus longimanus*), silky, grey reef (*C. amblyrhynchos*) and hammerhead (*Sphyrna* spp.) sharks (Maoulida et al. 2009).

Based on reported landings and gear composition from IOTC National Reports (Soilihi 2014), effective catch per unit effort (CPUE) for sharks is much higher in both static (775 kg/vessel/year) and trolled (830 kg/vessel/year) handlines than in net fisheries (2.20 kg/vessel/year). Indeed, handlines account for nearly 96% of the reported blue shark landings. Given the seemingly greater CPUE for sharks in the handline

Table 3 Chondrichthyan species common amongst the reported small-scale fisheries landings in the SWIO. La Réunion has been omitted as no data is available. *Data Source:* Andrianisoa and Randriamiasoa (2013), Barnett (1997), Cooke (1997), Kiszka et al. (2010), Kiszka (2012), Maoulida et al. (2009), Marshall (1997a), Nevill et al. (2007), Poonian (2015), Robinson and Sauer (2013), Smale (1998), Sousa et al. (1997), Instituto Nacional de Investigación Pesqueira unpublished data)

Common name	Scientific name	Country									
		Comoros	Kenya	Madagascar	Mauritius	Mayotte	Mozambique	Seychelles	Tanzania		
<i>Sharks</i>											
Silvertip	<i>Carcharhinus albimarginatus</i>			X			X	X	X		
Grey reef	<i>Carcharhinus amblyrhynchus</i>	X	X	X	X	X		X	X		
Spinner	<i>Carcharhinus brevipinna</i>			X			X	X			
Silky	<i>Carcharhinus falciformis</i>		X	X	X	X	X	X	X		
Bull	<i>Carcharhinus leucas</i>			X	X		X	X	X		
Oceanic whitetip	<i>Carcharhinus longimanus</i>	X		X	X	X		X			
Blacktip reef	<i>Carcharhinus melanopterus</i>		X	X			X	X	X		
Sandbar	<i>Carcharhinus plumbeus</i>			X			X	X	X		
Spottail	<i>Carcharhinus sorrah</i>			X			X	X	X		
Tiger	<i>Galeocerdo caviar</i>		X	X	X		X	X	X		
Shortfin mako	<i>Isurus paucus</i>		X	X	X	X	X	X	X		
Slit-eye	<i>Loxodon macrorhinus</i>			X	X		X	X	X		
Tawny nurse	<i>Nebrius ferrugineus</i>		X	X				X			
Blue	<i>Prionace glauca</i>		X	X	X	X	X	X			
Whale	<i>Rhincodon typus</i>		X	X				X	X		
Milk	<i>Rhizoprionodon acutus</i>			X			X	X	X		
Hammerhead	<i>Sphyrna spp.</i>	X	X		X		X	X	X		
Scalloped hammerhead	<i>Sphyrna lewini</i>		X	X		X	X	X	X		
Great hammerhead	<i>Sphyrna mokarran</i>		X	X			X	X	X		
Smooth hammerhead	<i>Sphyrna zygaena</i>						X	X	X		
Whitetip reef	<i>Triacodon obesus</i>		X	X	X		X	X	X		
<i>Batoids</i>											
Spotted eagle	<i>Aetobatus cf. ocellatus</i>		X	X			X	X	X		
Manta	<i>Mobula spp.</i>		X		X		X	X	X		
Bluespotted maskray	<i>Neotrygon caeruleopunctata</i>		X				X	X	X		
Sawfish	<i>Pristis spp.</i>			X			X	X	X		
Bowmouth wedgetfish	<i>Rhina ancylostoma</i>			X			X	X	X		
Large wedgetfish	<i>Rhynchobatus spp.</i>		X	X	X		X	X	X		
Bluespotted fantail	<i>Taeniura lymna</i>		X	X			X	X	X		

fisheries specific scrutiny should be placed on these in any future assessments. Conversely, the lack of reporting of batoid catches and/or landings combined with their known susceptibility to net gears means the potential contribution of these gears to elasmobranch catches should not be overlooked.

Kenya

Elasmobranch landings have fluctuated in recent years, dipping as low as 165 mt in 2007, peaking at 373 mt in 2012 and reported at 293 mt in 2014 (Ndegwa 2015). Curiously, between 2011 and 2013 Tana River province had the highest contribution to the total elasmobranch landings, despite it having the lowest overall reported fisheries landings (KMALF 2015). This is possibly the result of the much greater longline prevalence in this area, although use of this gear type has subsequently dramatically decreased (KMALF 2014b).

Landings of elasmobranchs are not reported to species level. However, recent studies suggest that hammerheads, tiger (*Galeocerdo cuvier*), blacktip reef (*Carcharhinus melanopterus*), whitetip reef (*Triaenodon obesus*) and grey reef sharks feature prominently, together with a number of batoids including Mobulid rays (*Mobula* spp.), spotted eagle (*Aetobatus* cf. *ocellatus*) and bluespotted fantail rays (*Taeniura lymma*) as well as large wedgefish (*Rhynchobatus* spp.) (Ndegwa 2015; J. Kiszka unpublished data). Other rays belonging to Dasyatidae and Myliobatidae families also appear common, having been recorded in catch assessment surveys (State Department of Fisheries and the Blue Economy unpublished data).

Madagascar

Sharks are exploited throughout Madagascan waters. However, catch and landing data are mostly available for the west coast, particularly in the southwestern (Toliara) and northwestern (Mahajunga) regions, where conditions are more favourable for fishing (Cooke 1997; Cripps et al. 2015; McVean et al. 2006; Robinson and Sauer 2013). Data relating to the volume of elasmobranch catch are scarce. Traditional fishing (i.e. from sail-powered dug-out canoes) is estimated to produce $\geq 85\%$ of the total shark catch (Le Manach et al. 2011; Randriamiarisoa and Rafomanana 2005) and likely for the vast majority of batoid catch. In

some areas elasmobranchs may account for as high as 50–60% of the overall catch (Andriantsoa and Randriamiarisoa 2013; de Feu 1998). Elasmobranch landings across all Madagascan fisheries was reported as 5650 mt in 2014 (FAO 2016b), yet estimates suggest small-scale fisheries landings alone are in the region of 7500 mt/year (Le Manach et al. 2011). Shark catches are reportedly decreasing in Madagascar (Cooke 1997; McVean et al. 2006) possibly a response to declining shark fin demand (Whitcraft et al. 2014) and/or intensive overfishing.

A breakdown of species composition is not available for the small-scale fisheries, however a number of case studies have been undertaken. These studies list hammerheads, silky, tiger, spottail (*Carcharhinus sorrah*), sliteye (*Loxodon macrorhinus*), whitetip reef, blacktip reef and grey reef sharks as common species in a number of areas (Andriantsoa and Randriamiarisoa 2013; Cooke 1997; Robinson and Sauer 2013; Short 2011; Smale 1998). Information regarding batoid catch is more limited, but spotted eagle, thornback (*Raja clavata*) and bluespotted fantail rays, various guitarfish (*Rhinobatidae* spp.) and large wedgefish (*Rhynchobatus* spp.) are also captured (Cooke 1997), with *Mobula* spp. reported anecdotally (Heinrichs et al. 2011).

Mauritius

Mauritius reported artisanal landings of 0.456 mt of elasmobranchs in 2013 (Albion Fisheries Research Centre unpublished data), representing around 0.8% of the total fisheries landings by weight. In contrast, interview surveys suggest around 6000 elasmobranchs are caught annually (Poonian 2015). Whilst interview surveys are likely an unreliable way to estimate total catch effectively, the magnitude of difference suggests official reports are substantial underestimates.

The reported landings in 2013 is a 97% decrease from the 16.725 mt in 2000 and follows general declines in landings across the Mauritian fisheries since the mid-2000's (FAO 2016b). Over this period around 85% of elasmobranch landings originated from line gears (representing 0.95% of the total line fisheries landings), whereas landings from net gears was disproportionately lower at around 7% of total elasmobranch landings (representing 0.1% of the total net fisheries landings). Basket traps account for the remaining portion, probably impacting smaller,

benthic species, and may represent a threat to small batoids in particular. Hammerhead and tiger sharks are by far the most common species reported (Poonian 2015). Further, *Mobula* spp. were the only batoid reported in the catch. Ease of identification for these species likely heavily biases interview responses and so inevitably overestimates their importance in the fisheries.

Mayotte

Traditionally fishers have exploited the species-rich lagoon surrounding the island. However, decreasing catches in reef habitats (Guézel et al. 2009) and modernisation of fishing gears have resulted in a shift towards offshore pelagic resources, evidenced by the proliferation of pelagic teleost and elasmobranch species in their landings (FAO 2016b). It seems likely that significant loss of coastal/inshore elasmobranch species may have already taken place and remaining lagoon-based small-scale fisheries may be continuing to impact on or at least hindering the recovery of these stocks.

The shift towards a pelagic fishery means both competition and shared stock resources with the regional industrial fisheries. An investigation of the expanding longline small-scale fisheries of Mayotte revealed high abundance of silky, blue, scalloped hammerhead (*Sphyrna lewini*) and oceanic whitetip sharks in the catch, with pelagic stingrays (*Pteroplatytrygon violacea*) being the only batoid representative (Kiszka et al. 2010). The study showed that elasmobranchs comprised 24.6% of the catch. Whilst the fisheries themselves are dominated by handlines (both static and trolled) (Table 1) the composition is potentially similar to that of the longlines and so these may reflect the fisheries as a whole. Most, if not all, elasmobranch catch is discarded, of which 16.1% were dead (Kiszka et al. 2010). If this is representative of the fisheries as a whole, elasmobranchs discarded as dead would represent 5% of total catch of pelagic species (880 mt landed in 2014). This suggests that the reported landings of elasmobranchs (11 mt in 2014), are a significant underestimate of the true catch, perhaps by 75% or more.

Mozambique

Significant improvement in the monitoring and estimation of small-scale fisheries landings has been made

in Mozambique (Dias and Afonso 2011; Doherty et al. 2015). Partial elasmobranch disaggregation in overall fisheries data is available through the FAO, with shortfin mako (*Isurus oxyrinchus*) and blue shark landings available separately in 2014 accounting for 26.3% of elasmobranch landings (FAO 2016b). Previously, separate landings data for copper (*Carcharhinus brachyurus*), silky and oceanic whitetip sharks have also been provided. However, at least for the small-scale fisheries in some provinces, data are available for a further 15 shark species and for spotted eagle ray (Instituto Nacional de Investigação Pesqueira unpublished data). This represents a significant step forward in the fisheries landings data resolution compared to other SWIO nations, though further disaggregation of species, and batoids in particular, is desirable. This is especially true for *Mobula* spp., for which Mozambique has a targeted sustenance fishery (Marshall et al. 2011) and possible gill plate trade (Heinrichs et al. 2011).

Over 98.8% of the 854 mt of elasmobranch landings reported in 2014 originates from small-scale fisheries (Instituto Nacional de Investigação Pesqueira unpublished data). The majority comes from the dominant beach seine fishery, of which 92% of landings were reportedly spotted eagle ray, clearly highlighting this fishery as a specific threat to this species. Whilst the beach seine fishery is the most important component by virtue of its size (accounting for 46.4% of small-scale fisheries landings in 2014), much higher CPUE rates for elasmobranchs are seen in the bottom set gillnet sector (Instituto Nacional de Investigação Pesqueira unpublished data). This further emphasizes the apparent threat this gear poses to elasmobranchs at a regional level.

The validity and accuracy of both the FAO and the official Mozambique small-scale fisheries datasets, is difficult to assess. Aside from the probable significant under-reporting, there are serious discrepancies both between and within data sets. The small-scale fisheries elasmobranch landings data are often much larger than that of the total reported amounts through the FAO, in the case of 2011 by over 200 mt (FAO 2016b; Instituto Nacional de Investigação Pesqueira unpublished data). There are also significant variations between years e.g. in 2013 bottom set gillnet landings were reported as 534.1 mt, yet in 2012 and 2014 only 142.6 and 141.1 mt were reported, respectively. In addition, the only known estimate of shark catch in the small-

scale fisheries was 2186 mt from 1993, far in excess of both current and historical landings figures for the whole fisheries sector (Sousa et al. 1997). It is clear that these inconsistencies and the identification of their underlying drivers must be resolved as an urgent priority.

La Réunion

La Réunion does not currently record the landings of elasmobranchs in its small-scale fisheries; therefore, no official estimates exist (L'Institut Français de Recherche pour l'Exploitation de la Mer personal communication). The fisheries are dominated by longline and trolled handline (~77.5% of catch) and so it is possible that elasmobranchs, particularly sharks, are an important constituent of the catch. The restricted shelf system around La Réunion is thought overfished and has been exploited historically (Le Manach et al. 2015). Since 2011, tiger and bull sharks (*Carcharhinus leucas*) have been the cause of increasing numbers of reported attacks on bathers and surfers, resulting in increasing efforts to reduce numbers of these species (Lemahieu et al. 2017; L'Institut de Recherche pour le Développement personal communication). However, no statistics are currently available and the magnitude of culling is unknown.

Seychelles

Fishing for elasmobranchs, primarily for finning, has been ongoing since the 1920s. Since declines in the 1950s, elasmobranchs have shifted towards incidental catch (Fowler et al. 2005), though targeted fisheries still exist. In 2014 elasmobranchs accounted for approximately 1% of total small-scale fisheries landings (SFA 2015). Landings appear to be seasonal, peaking during the months of July and August (Online Resource 6). This pattern is likely driven by the increased catches of hammerhead sharks during this time (particularly *S. lewini* and *S. mokarran*), a fishery that is believed to be sustainable (Nevill et al. 2007). Breakdown of effort and landings by gear in the official reports are insufficient to allow for analyses of historical CPUE across the fisheries. A lack of species level identification in landings data has been identified as impeding effective management in the Seychelles (Nevill et al. 2007). It is suggested that the most commonly caught are spottail and grey reef sharks in

inshore waters (Fowler et al. 2005), though interviewed fishers reported tiger and sandbar sharks (*Carcharhinus plumbeus*) as most common (Nevill et al. 2007).

Currently there is an encouraged expansion of small-scale fisheries towards targeting pelagic finfish. This includes a development fund providing access to loans for purchasing/upgrading to longlines (Seychelles Fishing Authority personal communication). This could increase pressure on oceanic elasmobranchs, with catch composition mirroring that of the semi-industrial pelagic longliners, which report high landings of silvertip (*Carcharhinus albimarginatus*) and oceanic whitetip sharks (Nevill et al. 2007). Given the current lack of detailed catch statistics these developments need to be monitored closely.

Tanzania (including Zanzibar)

The catch of elasmobranchs in small-scale fisheries is significant and, at least in Zanzibar, shows signs of overexploitation and partial collapse (Jiddawi and Shehe 1999). Information regarding the catch composition of these fisheries is limited. In mainland small-scale fisheries, 11 species are commonly caught (Barnett 1997). These are predominantly requiem sharks, both oceanic and coastal (including coral reef associated), alongside hammerhead, milk (*Rhizoprionodon acutus*) and whitetip reef sharks. Large wedgetfish (*Rhynchobatus* spp.) have also been reported. In Zanzibar, at least 21 elasmobranch species are caught (Barrowclift et al. 2017). A market sampling survey in 2004 identified milk, grey reef and black tip reef sharks as the most common species, with various wedgetfish including bottlenose and/or whitespotted and bowmouth (*Rhina ancylostoma*) and Zanzibar guitarfish (*Acroteriobatus zanzibarensis*) also present in relatively high numbers (Schaeffer 2004). Other batoids were not recorded, however it is known that various Dasyatidae species dominate the batoid catch (Barrowclift et al. 2017). *Mobula* spp. are caught in both mainland and Zanzibari small-scale fisheries (Heinrichs et al. 2011; A. Temple unpublished data).

In mainland small-scale fisheries, shark catches are common through most of the year, reduced only when the weather restricts fishing activity (Barnett 1997). In Zanzibar catches of sharks appear seasonal, being highest during the north-east monsoon, particularly

between January and May (Barnett 1997; Schaeffer 2004). However, there is insufficient information available to suggest that elasmobranch abundance is related to season. More likely is that seasonal weather precludes the use of certain gears and/or fishing locations, so impacting elasmobranch captures.

Use and value

Elasmobranchs are generally considered a target species throughout SWIO small-scale fisheries (Cooke 1997; Jiddawi and Shehe 1999; Maoulida et al. 2009; Poonian 2015; Wekesa 2013). Often they are taken as a desired constituent of a multi-species fishery also targeting moderate-to-large pelagic or reef fish species, rather than in a dedicated elasmobranch fishery. Finning is relatively rare in the region's small-scale fisheries, with the majority of catch landed whole and fully utilised (Wekesa 2013; A. Temple personal observation). However, finning does occur in parts of Mozambique and Madagascar where dedicated fisheries with formal processing and export markets exist (Cripps et al. 2015; Pierce et al. 2008). These practices present further difficulties in documenting and assessing catch. Currently, there is little evidence of demand for *Mobula* spp. gill plates emanating from the SWIO region (Cisneros-Montemayor et al. 2013), with the exception of small exports from Mozambique (Dent and Clarke 2015). However, should this change it will likely increase fisheries pressure exerted on these species. Where export markets for elasmobranch products do exist, shark and wedgefish fins hold substantially higher value relative to the rest of the body (Online Resource 4). However, readily available data on landings volume and value are limited e.g. the only reports to the FAO were from Madagascar and Seychelles in 2013 (FAO 2016b). Further, there is a lack of information regarding supply chains and the contribution from small-scale fisheries to these.

As a source of protein elasmobranch meat is relatively cheap (Online Resource 4) in comparison with teleosts (e.g. MFR 2004, 2012), and may form an important nutritional component in the diets of those supplied by and dependant on small-scale fisheries. Lack of access to cold storage facilities requires that meat is often either sold fresh locally or air/salt dried in preparation for sale at distant markets or export (Cripps et al. 2015). Value of elasmobranch products

depends on a number of factors including perceived quality (species and preparation related), route of sale and cultural and religious influences, such as Ramadan, which affect both supply and demand (Barrowclift et al. 2017; Cripps et al. 2015). It is therefore important to further our understanding of elasmobranch value in SWIO small-scale fisheries, and their markets and drivers, if we are to assess the socio-economic importance of these fisheries and their component species. Ultimately, a proper understanding of the socio-economic value of the fisheries is vital to the design and implementation of any successful management strategy.

Fisheries policy and management in the southwestern Indian Ocean: implications for vulnerable marine megafauna in small-scale fisheries

Formal governance arrangements in the fisheries sector are a fundamental component in the sustainable use of fisheries stocks. Good governance, policies and resultant effective management of fisheries stand to create a strong platform from which sustainable species harvest can be achieved and controlled. It is therefore vital that we understand both the principles upon which policies are built and what management is in place to achieve these goals with regard to vulnerable marine megafauna in the SWIO.

International

Many SWIO nations are party to international fisheries-specific agreements that have implications for both general fisheries policy and management and vulnerable marine megafauna specifically (Online Resource 5). Primary amongst these is the United Nations Fish Stocks Agreement (UNFSA) 1995, which obligates that parties undertake ecosystem-based and precautionary approaches to migratory fish stock management (UN 2016). This agreement increases the responsibility of nations over their fisheries and their enforcement of laws within them, and strengthens the roles of regional fisheries bodies. Given the overlap in vulnerable marine megafauna species between SWIO nations (Table 3; Bourjea 2015; Kiszka 2015; Kiszka and van der Elst 2015), and absence of stock delineations for the majority of

species, there is a clear need to consider these as shared resources until clarification can be achieved. Concurrently, the signatory status, ratification and accession to the Port State Measures Agreement by many SWIO nations (FAO 2016a), signifies increasing regional efforts to tackle illegal, unlicensed and unreported (IUU) fisheries. These fisheries are rife in the SWIO region (Agnew et al. 2009) and so likely have substantial impacts on vulnerable marine megafauna species and marine ecosystems in general. Better control and documentation of these IUU fisheries will therefore be vital in managing both vulnerable marine megafauna fisheries and the fisheries as a whole in the SIWO region.

At a regional level, SWIO marine environmental policies are largely outlined by a number of conventions and agreements. The most ubiquitous of these is the Nairobi Convention 2010 (formally, the Amended Convention of the Protection, Management and Development of the Marine and Coastal Environment of the Western Indian Ocean) to which all SWIO nations are party. Broadly, the convention places onus on the party states to work, both individually and co-operatively, in an effort to sustainably maintain, manage and develop their marine and coastal ecosystems (UNEP 2010). It highlights a recognition and willingness of SWIO nations to view the marine environment as an inter-linked and shared resource. This outlook is pivotal to any meaningful management of the region's fisheries, including those which catch vulnerable marine megafauna.

There are also specific international agreements dealing with vulnerable marine megafauna to which SWIO nations are contracted. The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) (CITES 2016) make international trade of any marine mammal or sea turtle illegal. Further, trade of 17 species of elasmobranch (12 of which are known to occur in the SWIO) are controlled to varying extents. In this regard these species receive some level of protection through control over commercial exploitation for international trade. The Indian Ocean Cetacean Sanctuary, designated and established in 1979 by the International Whaling Commission (IWC), covers the entirety of the Indian Ocean south to 55°S and prohibits commercial whaling. However, it does not provide protection for smaller cetaceans nor does it identify critical habitats for cetaceans (IWC 1980), a

significant roadblock to achieve effective implementation. Currently Kenya and Tanzania are the only SWIO nation members of the IWC. The Conservation of Migratory Species Memoranda of Understandings (CMS MoUs) seeking to encourage protection of and promote stock recovery for sea turtles, dugongs and sharks, have been effective since 2001, 2007 and 2010 respectively. Both the sea turtle and dugong agreements boast wide coverage, with only La Réunion absent as a signatory on the dugong MoU, as the species is absent from its waters (CMS 2016a, b). Conversely, few SWIO nations have signed the CMS MoU for sharks (Online Resource 5), raising some concern over the political willingness of SWIO nations to sustainably manage these species, though it must be considered that this MoU has only been created very recently (CMS 2016c). Those SWIO countries yet to sign the CMS MoUs should be encouraged too do so, as the commission and these agreements represent a pathways for facilitating the conservation (in its widest sense) of vulnerable marine megafauna in the region.

National directed management

At a national level there are varying degrees of directed management relating to vulnerable marine megafauna species. Dugongs and cetaceans are protected by law throughout the SWIO and for sea turtles, intentional catch, egg poaching and sale is widely prohibited (Table 4). Further protection is offered through anti-disturbance regulations in some areas, both site-specific (in various MPAs) and nationally in the case of Mozambique (Marine and Coastal Environment Regulation, Decree 45/2006). Only the Seychelles do not offer complete protection, with green and hawksbill protected but not loggerhead, leatherback or olive ridley, despite their presence (although not nesting) in the SWIO region (Frazier 1984; Remie and Mortimer 2007). A National Plan of Action (NPOA) for sea turtles has been implemented by France, covering Mayotte, the French dispersed islands (Tromelin, Glorieuses, Juan de Nova, Bassas da India and Europa) and La Réunion, and a NPOA is under discussion for Tanzania (Igulu and El Kharousy 2015).

Conversely, current regulations regarding elasmobranchs are very limited for the small-scale fisheries of the region (Table 4) and where they do exist their effectiveness is often questioned. For example, as of

Table 4 Legal status and related punishments regarding vulnerable marine megafauna in the small-scale fisheries of the South Western Indian Ocean

Country	Sea Turtle	Cetaceans	Dugong	Chondrichthyans
Comoros	Prohibited— <i>Punishable by imprisonment</i> (Soilihi 2014)	Prohibited	Prohibited	Partial— <i>Thresher shark prohibited</i> (Soilihi 2014)
Kenya	Prohibited— <i>The Wildlife (Conservation and Management) Act of 2013 (revised), The Fisheries Act Cap 378 revised 2012</i>	Prohibited— <i>Kenya Fisheries Act 2012, The Fisheries Management and Development Bill, 2014—1 year imprisonment and/or 100,000 KES fine</i>	Prohibited— <i>Kenya Fisheries Act 2012, The Fisheries Management and Development Bill, 2014—1 year imprisonment and/or 100,000 KES fine</i>	None
Madagascar	Prohibited— <i>Décret no 2006</i>	Prohibited	Prohibited	None
Mauritius	Prohibited— <i>Mauritian Fisheries and Marine Resources Act 2007—100,000 MUR fine</i>	Prohibited— <i>Mauritian Fisheries and Marine Resources Act 2007—100,000 MUR fine</i>	Prohibited— <i>Mauritian Fisheries and Marine Resources Act 2007—100,000 MUR fine</i>	Partial— <i>Fishing licence not granted for targeting sharks</i> (Soondron et al. 2013)
Mayotte	Prohibited— <i>National decree (October 14th 2005)</i>	Prohibited— <i>National decree (July 27th 1995)</i>	Prohibited— <i>National decree (July 27th 1995)</i>	None
Mozambique	Prohibited— <i>Law of Regulation Forests and Wildlife (Decree No. 12/2002)—25,000 MZN fine</i>	Prohibited	Prohibited— <i>Law of Regulation Forests and Wildlife (Decree No. 12/2002)—50,000 MZN fine</i>	None
La Réunion	Prohibited— <i>National decree (October 14th 2005)—5000 EUR or 6 month imprisonment</i>	Prohibited— <i>National decree (July 27th 1995)—5000 EUR or 6 month imprisonment</i>	Prohibited— <i>National decree (July 27th 1995)—5,000 EUR or 6 month imprisonment</i>	Partial— <i>Préfecture de La Réunion, arrêté no 06—2412/SG/DRCTCV 2006—Due to ciguatera poisoning risk, hammerhead sharks (Sphyrna spp.) cannot be commercialised</i>
Seychelles	Prohibited— <i>Fisheries Act 2014 Regulations—green and hawksbill turtles only</i>	Prohibited	Prohibited	Partial— <i>Fisheries Act 2014 Regulations—Baiting and chumming for shark illegal 450,000 SCR fine. Gillnetting for shark prohibited</i>
Tanzania (mainland)	Prohibited— <i>The Fisheries Act, 2003 (Regulations 2005)—200,000 TZS fine or 3 month imprisonment</i>	Prohibited	Prohibited	Partial— <i>Export of meat and fins not permitted. Whale shark catch prohibited.</i>
Tanzania (Zanzibar)	Prohibited	Prohibited	Prohibited	Partial— <i>Licences for fin export no longer issued</i>

2016, licences are no longer being distributed for shark fin export from Zanzibar (Ministry of Agriculture, Livestock and Fisheries, personal communication), effectively making the trade illegal. However, post-

harvest removal of fins from landed elasmobranchs to supply international trade (including from CITES listed species) still occurs in spite of these restrictions (A. Temple personal observation). Unlicensed

shipping of shark fins to Kenya is thought common (Fowler et al. 2005) and may be continuing to provide the export route out of the country. In spite of the general lack of regulations for the elasmobranch small-scale fisheries, NPOAs have been or are currently in development for most SWIO nations (Table 4), with Kenya and Mozambique projecting completion by the end of 2017 (Chacate and Mutombene 2015, Wekesa 2014), suggesting widespread recognition of the threats faced by these species and a movement towards addressing the issues. As with other regulations pertaining to many small-scale fisheries in the region, implementation is challenged by management authorities' lack of infrastructure and resources to ensure compliance, compounded by the generally open-access nature of these fisheries.

National indirect management

There are also some regulations and initiatives which likely indirectly impact interactions between small-scale fisheries and vulnerable marine megafauna in the SWIO (see <http://www.wiofish.org> for comprehensive information on prohibited fisheries and gear restrictions in the SWIO). For example, in Mauritius, the Fisheries and Marine Resources Act 2007 prohibits the use of certain fishing gears. Most relevant to vulnerable marine megafauna is the ban on the use of drift-nets (defined there as a net exceeding 250 m in length, fitted with floats or weights to make it hang vertically in the water column) with a fine of up to 20,000 USD if found in breach of the Act. Further it prohibits the use of shorter net gears ("large nets" and "gillnets" less than 250 m in length) for 5 months of the year (October–February, 500,000 MUR/~14,000 USD fine) and licences may only be issued for use of 10 "large nets" and 5 "gillnets" at any one time in the lagoon waters of Mauritius. Given these nets form the principal gear threats to vulnerable marine megafauna at the global scale, this likely has a major impact on their interactions with the fisheries in Mauritius. In Kenya (Fisheries Act Cap 378, Kenya Gazette Notice No. 7565) and Tanzania (Fisheries Act Regulations, 2003) the use of mono-filament nets is prohibited, though compliance is poor (KMALF 2014b; A. Temple personal observation), resultant changes in gear use could impact the catches of vulnerable marine megafauna species susceptible to alternate gears whilst protecting those species susceptible to the mono-filament nets.

Beach seine nets are prohibited in Kenya, Tanzania and Comoros, which reduces fishing pressure on some coastal elasmobranchs, but again, ensuring compliance is problematic.

Discussion

This review highlights the severely limited understanding of vulnerable marine megafauna and their interactions with the small-scale fisheries of the SWIO resulting from a lack of robust data. Yet, where evidence exists, there are indications of population declines due to fisheries interactions (Amir et al. 2002; Cooke 1997; FAO 2016b; Jiddawi and Shehe 1999; McVean et al. 2006; Muir and Kiszka 2012). Furthermore, it is clear that at both national and regional levels, current small-scale fisheries monitoring practices and management are insufficient to assess and ensure the long-term sustainability of small-scale fisheries and the vulnerable marine megafauna species impacted by them. Therefore, there is a clear regional priority to collect much more information on small-scale fisheries characteristics, species catch, landings and composition, as well as data regarding vulnerable marine megafauna gear-interactions. These data are required to undertake a robust and detailed analysis at both regional and area-specific spatial scales, without which informed, evidence-based management and facilitating policies cannot be achieved effectively. As such, and in the absence of truly precautionary management, vulnerable marine megafauna must be considered at high risk of on-going overexploitation at both the national and regional scale in the SWIO.

Improved monitoring and assessment of small-scale fisheries both at national and regional levels is critical in achieving sustainable harvest of fisheries resources, including vulnerable marine megafauna. It is probable that the majority of these fisheries stocks are shared given: the highly mobile, transboundary nature of many species (e.g. Table 3); few geographical barriers in the SWIO; migrant fishing; and the notion of Economic Exclusive Zones being at best flexible when regarding small-scale fisheries. In light of this there is a clear need for SWIO nations to begin identification and delineation of stocks and to devise a joint strategy defining protocols for collection and reporting of small-scale fisheries data. At the heart of any such strategy must be a consensus on minimum

data requirements (landings, effort, gear composition and their breakdowns), standardised metrics and methodologies used for collecting the data. Standardised reporting procedures and formats would also be of great benefit, allowing data to be compiled and synthesised with greater ease. Beyond any agreed minimum, nations should be encouraged to collect further data as far as is feasible, particularly where these data address specific issues or interests of each party or the region. Where such issues may have wider applicability an open discussion regarding metrics and methods would benefit all parties. Ultimately such changes would aid the understanding and management of small-scale fisheries and assist the decision-making processes, with implications for the long-term regional sustainability of the sector. However, currently there is no regional body with the ability to make binding decisions on such issues, the Indian Ocean Tuna Commission's (IOTC) mandate is too restricted and the Southern Indian Ocean Fisheries Agreement's (SIOFA) membership only has partial SWIO coverage. Perhaps best placed to facilitate such changes is the South West Indian Ocean Fisheries Commission (SWIOFC), through its role as an advisory body, in conjunction with expanding the mandate of the IOTC and through the ability of IOTC and SIOFA to make binding decisions (van der Geest 2017).

CPUE is a fundamental measure used to monitor stock health and fisheries sustainability (Maunder and Punt 2004; Sparre and Venema 1998), yet for most SWIO small-scale fisheries such data cannot be generated. To create an accurate and usable CPUE time series, effort data must accommodate changes in fisheries dynamics, behaviour and power. To this end measurement of fishing effort requires four main data types: gear type including specificity (e.g. mesh size, hook size, mono/multifilament etc.); fishing mode (e.g. active or passive) to allow proper categorisation of the fisheries; gear characteristics (e.g. net dimensions, number of hooks etc.); and active fishing effort (e.g. soak times, trawl speeds/distance etc.). Currently data collected for small-scale fisheries in SWIO nations consistently lacks one or more of these aspects (Table 1, Online Resource 3). For example, in five nations (Comoros, Kenya, Madagascar, Tanzania mainland and Zanzibar) effort is either recorded and not compiled or not recorded at all. Whereas, in others fishing effort is given in trips, sets, active vessels and fisher-days (Online Resource 3), none of which are

suited for accurate estimation of fishing effort due to large potential variation within these measures e.g. trips may last for hours or days. Thus the utilisation of fisheries data for informing management objectives, targets and strategies is restricted.

Some of these effort monitoring weaknesses could be addressed through relatively minor changes to current monitoring protocols. Detailed gear specifications and fishing modes could be incorporated into national census/frame surveys in which data on gear type are already routinely collected (Table 1). Active fishing effort data are however more difficult to obtain, especially given the informal nature of many small-scale fisheries. It is perhaps inevitable that in the short term active fishing effort data may need to be generated through declarations by fishers along with evidence-based assumptions until a more formalised system is possible. Alternatively, CPUE estimates could be generated through fisheries-independent data (Sparre and Venema 1998), however this can be costly and so is likely unfeasible for most SWIO nations.

Regarding vulnerable marine megafauna in SWIO small-scale fisheries, generally catch and landings data are relatively poor and where available are often lacking in both species composition and catch-by-gear data. This lack of information severely limits the ability to identify and manage at-risk species and stocks, including assessment of gear and area-specific threats. Despite the general paucity of data, it is clear that in a number of areas catches are in decline and some populations are known to be overexploited (Amir et al. 2002; Cooke 1997; FAO 2016b; Jiddawi and Shehe 1999; McVean et al. 2006; Muir and Kiszka 2012; Nevill et al. 2007). Thus, there is an urgent need for proper assessment of vulnerable marine megafauna in SWIO small-scale fisheries. Catch and landings data which do exist show substantial numbers of large oceanic shark species in the small-scale fisheries (Andriantsoa and Randriamiarisoa 2013; Barnett 1997; Cooke 1997; FAO 2016b; Kiszka et al. 2010; Maoulida et al. 2009; Ndegwa 2015; Nevill et al. 2007; Poonian 2015; Robinson and Sauer 2013; Schaeffer 2004; Smale 1998; Soilihi 2014), indicating increasing competition for resources with the industrial fisheries. Increasing competition for these stocks is concerning for two main reasons: increased pressure on stocks that are already thought to be overharvested and at high extinction risk (García et al. 2008); and it indicates a shift towards fishing further offshore by the small-

scale fisheries sector, a phenomenon seen elsewhere in instances where inshore stocks may have become depleted.

Comparatively little information on the marine mammal, sea turtle and batoid and chimera catch components of the fisheries are available, though given the general confinement of chimera to deeper waters (Kyne and Simpfendorfer 2007) they are unlikely affected by small-scale fisheries in the SWIO region. There is therefore a clear need to address this data vacuum. With regards to marine mammals and sea turtles specifically, understanding of catch is further limited as a result of their legal status, creating a reluctance to declare catches. Governmental departments are therefore poorly placed to generate these data and so they should represent a priority focus for collaborative work with independent researchers and NGOs.

Currently the data regarding SWIO small-scale fisheries interactions with vulnerable marine megafauna are unable to support management aimed at safe sustainable exploitation of these resources, yet there is clear evidence that vulnerable marine megafauna species are at risk from these fisheries. As such, and in accordance with the UN Fisheries Stocks Agreement 1995 and the Nairobi Convention, a precautionary conservation-minded approach is mandated to safeguard vulnerable marine megafauna until such time as robust evidence-based management strategies for sustainable exploitation can be achieved. If such measures are not taken, SWIO nations will be failing in their duty of care to both to the fishers and communities that rely on these resources and to the vulnerable marine megafauna themselves by failing to protect the biodiversity of their marine environment.

At the national scale, effective management strategies require a proper understanding of the human elements of the fisheries (Gray 2005; Kooiman et al. 2005). Fishers face an increasing variety of changing socio-economic conditions related to overexploitation, climate change, globalization, and conservation of marine biodiversity. Understanding the socio-economic importance of vulnerable marine megafauna species across stakeholder groups, including perceptions and attitudes (both cultural and individual) towards these species, as well as how fishers will respond to potential ecosystem and institutional changes is critical to better managing these fisheries, achieving fishers acceptance of and compliance with

management strategies and improving the livelihoods of those dependent on fisheries supply chains (Daw et al. 2012). Further, effective and appropriate enforcement is vital for sustainable management practices to be implemented, whether this be establishment or community driven, and presents a significant challenge for SWIO governments. Without this, dissent and non-compliance (Peterson and Stead 2011) can become wide-spread issues (Hauck 2008; Keane et al. 2008; Nielsen and Mathiesen 2003) and management strategies can be rendered ineffective. This review highlights that a significant gap in information exists in describing existing and shifting dependency of fishers in SWIO small-scale fisheries which, if collected alongside the much needed ecological data on fisheries, could provide better context for introducing effective management measures for vulnerable marine megafauna.

In the face of the numerous data gaps and taking into context infrastructure and the resource constraints present in SWIO nations, there is a clear need to identify appropriate low-cost methods to assess the magnitude of vulnerable marine megafauna catch in small-scale fisheries and to mitigate these when they are unsustainable. Critical assessment of various data collection methods (e.g. vessel-based observer programs, interview surveys, landing site data collection), incorporating time and cost factors, is vital in facilitating informed decision making, through which challenges can be addressed in both the short and long term using the appropriate methodological tools. Further, for species at highest risk, precautionary mitigation strategies need to be considered. Sea turtles, marine mammals and several species of elasmobranchs are of primary concern in this regard and catch mitigation methods (e.g. turtle excluding devices, weak links for nets and acoustic alarms) are already available for some of them (e.g. Barlow and Cameron 2003; Gilman et al. 2006; Ward et al. 2008). However, many of these methods are costly, and so there is a clear need and an opportunity to develop minimum-cost methods that are feasible for implementation in small-scale fisheries in the SWIO and globally. As a Regional Fisheries Authority, the SWIOFC and its working groups are well placed to facilitate the promotion and co-ordination of these initiatives, and to undertake and/or guide regular assessments of small-scale fisheries in the SWIO, in order to address questions of sustainability of vulnerable megafauna, and of the fisheries themselves.

Conclusions

This review highlights information needed to reconcile vulnerable marine megafauna conservation with small-scale fisheries demands and where it is lacking in the SWIO. Here the SWIO acts as both subject and case study in the broader issue of marine species conservation goals and the needs of those communities that rely on them, especially in data poor and developing regions. In addressing these issues, it is essential that solutions be built properly upon principles that balance environmental and human (economic and social) needs and are grounded in realism. Both funding and the time-scales in which to find effective solutions are limited, particularly in developing regions, and so research must be strictly prioritised towards practical and goal-oriented outputs that properly account for and engage stakeholders. Given the potentially dire situation for several vulnerable marine megafauna species it is critical to address priority baseline data gaps and their associated challenges. Governments, NGOs, independent researchers and research institutions and other stakeholders must act collaboratively to achieve common goals and ensure implementation of findings into effective evidence-based management, thus facilitating for a sustainable future for vulnerable marine megafauna species, the marine ecosystem and the associated livelihoods in coastal communities.

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Compliance with ethical standards

Conflict of interest Authors N. Wambiji, A. Brito and O. A. Amir are currently and/or have previously worked for in-region fisheries management (or related) institutions, however the authors do not believe that these current and/or past affiliations have had undue influence over the contents or interpretation of this review.

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Annex 2: Social, economic and trade characteristics of the elasmobranch fishery on Unguja Island, Zanzibar, East Africa

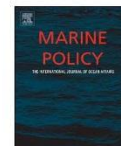
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Social, economic and trade characteristics of the elasmobranch fishery on Unguja Island, Zanzibar, East Africa



E. Barrowclift^{a,*}, A.J. Temple^a, S. Stead^a, N.S. Jiddawi^b, P. Berggren^a

^a Newcastle University, School of Marine Science and Technology, NE1 7RU, UK

^b Institute of Marine Sciences, Dar es Salaam University, Zanzibar, Tanzania

ABSTRACT

Understanding the socio-economic drivers underpinning fishers' decisions to target elasmobranchs is considered vital in determining sustainable management objectives for these species, yet limited empirical data is collected. This study presents an overview of elasmobranch catch, trade and socio-economic characteristics of Zanzibar's small-scale, artisanal fishery. The value of applying this information to future elasmobranch fisheries policy is demonstrated. In August 2015, interviews were conducted with fishers ($n = 39$) and merchants ($n = 16$) at two landing sites, Kizimkazi-Dimbani and Mkokotoni, along with the main market site in Stone Town. Additionally, elasmobranch catches were recorded across the same locations between June and August 2015. Elasmobranchs were listed as target species by 49% of fishers interviewed. Whilst most fishers ($n = 30$) stated that 76–100% of their household income came from fishing, there was variation in how elasmobranch catch and trade contributed. One-third of fishers ($n = 36$) that caught and sold elasmobranchs reported that 41–60% of their income came from elasmobranch catch. However, for some fishers ($n = 8$) elasmobranch catch represented 0–20% of their income, whilst for others ($n = 4$) it represented 81–100%. Differences in fisheries income and elasmobranch price could be attributed to several interacting factors including season, weather, fishing effort, fishing gear, target catch and consumer demand. Further, elasmobranch price was influenced by size and species. The study revealed information on catch, trade, markets and socio-economy that is important for future research, conservation and management of elasmobranchs and fisheries in Zanzibar. The methods utilised have potential for broader application to understudied, artisanal elasmobranch fisheries in the western Indian Ocean.

1. Introduction

The global decline of elasmobranch populations is both a species conservation and socio-economic concern. As apex and meso-predators, loss of elasmobranchs could have severe ecological impacts [1–3], with wider implications for the health of marine ecosystems and the associated dependent livelihoods. Furthermore, elasmobranchs are of social and economic importance in many parts of the world, as a source of income and dietary protein through fisheries catch and trade [4], as well as income generated through tourism [5]. Elasmobranchs' life history (slow growth, late maturity and low fecundity) means they are particularly vulnerable to fisheries mortalities and typically have a lower chance of recovery from population decline, making sustainable harvest problematic [6–8]. Fisheries are widely considered the primary driver of elasmobranch population decline worldwide, both through targeted and incidental catch [9–11], but in many regions the data and understanding necessary to effectively manage them are lacking. Specifically, limited information exists on elasmobranch catch

trends and the socio-economic drivers influencing these fishing practices; both of which are needed to design policy to support appropriate, context specific management measures.

Given the anthropogenic impacts elasmobranchs face, the wider effects on marine ecosystem health and the socio-economic importance of this marine resource, it is difficult to both successfully protect and sustainably manage these species at the same time. Thus, appropriate management that is locally context specific is crucial for future sustainability of elasmobranch fisheries. By addressing gaps in the data and tailoring policy to account for socio-economic characteristics, elasmobranch fisheries could be better managed [12,13]. One approach is through interviewing fishers to collate their local knowledge of the fishery [14,15]. Fisheries management should balance the social dimensions of fisher behaviour with reliable catch and species composition information. Currently, ineffective management, poor enforcement and unregulated fishing mean elasmobranch populations are still declining [16].

Presently, elasmobranch catch trends are widely underreported

* Correspondence to: Newcastle University, School of Marine Science and Technology, Ridley Building 2, NE1 7RU, UK.
E-mail address: e.barrowclift-mahon@newcastle.ac.uk (E. Barrowclift).

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making it difficult to accurately assess these fisheries [6,17]. Of the 1041 elasmobranch species assessed on the IUCN Red List, 18.1% are classified as critically endangered, endangered or vulnerable, whilst 43.1% are data deficient [18]. In some regions, such as the western Indian Ocean (WIO), where coastal communities are highly dependent on small-scale fisheries (SSF) for food security, nutrition, income generation and well-being, the extent to which elasmobranchs are targeted compared with incidental catches is unclear. Despite an estimated 137 million people worldwide involved in SSF [19], their importance in poverty alleviation and food security, as well as their likely significant contribution to catch, they are often overlooked compared to industrial fisheries [20–23].

Zanzibar is part of the United Republic of Tanzania and has jurisdiction over the fishing grounds located within its 12 nautical mile zone [21]. Zanzibar's fisheries are primarily artisanal, supporting approximately 27,200 artisanal fishermen, a further 27,200 subsistence fisherwomen and 2140 fish traders [24]. In 2010, the most recent year data are available for, there were a total of 212 fisheries landing sites and an estimated 8600 fishing vessels in Zanzibar. Fishing occurs throughout the year, with peak times relating to two monsoon seasons, generating an estimated annual yield of 31000 t, mostly consumed locally. Fish is the most important source of dietary protein [25]. Zanzibar's fisheries are comprised mainly of traditional wooden fishing boats, typically powered by sails, but increasingly common are out-board engine powered fibreglass boats. Fishing gears commonly used include lines (longline, hand line and troll line), nets (drift gill, set gill, shark, scoop, cast and ring), seine (purse and beach), traps, fences and spears [24]. Target species include large pelagic fish such as tuna (*Thunnus* spp.); small pelagic fish such as sardines (*Sardinella* spp.); coral reef fish such as grouper (*Epinephelus* spp.); lobster (*Nephropidae*) and octopus (*Octopodidae*). Sharks are targeted for their fins, meat, liver, skin, cartilage, jaws and teeth [26] and [Temple, pers. obs.]. Batoids are targeted for their meat and liver, with fins also harvested from guitarfish (*Rhinobatidae*). The practice of finning where only fins are taken and the rest of the animal is discarded appears rare [26]. A market for mobulid ray gill plates [27] does not yet appear to have developed (Temple, pers. obs.). Capture production of sharks, batoids and chimaeras for 2013 in Zanzibar was reported by the FAO as 1776mt but available data does not distinguish between species [28].

This study aimed to assess the dependence of local communities on Zanzibar's artisanal elasmobranch fishery by investigating catch, trade and socio-economic characteristics. Further, the socio-economic importance of the elasmobranch fishery, relative to fishers' and merchants' occupations in general was also assessed. The objectives were to explore how this was affected by the seasonality of catch availability in Zanzibar's fisheries, fishing practices and variation in elasmobranch price. The way in which this information can be utilised for future elasmobranch fisheries policy is also demonstrated.

2. Methods

2.1. Study area

Zanzibar is located in the western Indian Ocean off the coast of Tanzania and consists of two islands: Pemba and Unguja. The majority of fishing vessels and fisheries landing sites are located in Unguja (Fig. 1) [24]. The tropical climate of the islands is defined by two monsoon seasons and a longer rainy season in April/May. The South-east monsoon season ('Kusi') occurs from June to September and is characterised by strong winds, whilst the Northeast monsoon season ('Kaskazi') between November and March is characterised by lighter winds and short rains [29].

Data were collected using face-to-face questionnaire-based interviews between 7 and 21 August 2015 in Unguja. Interviews were carried out with both fishers and merchants and included questions that covered both the Kusi and Kaskazi seasons. Fishers' (n = 39) and

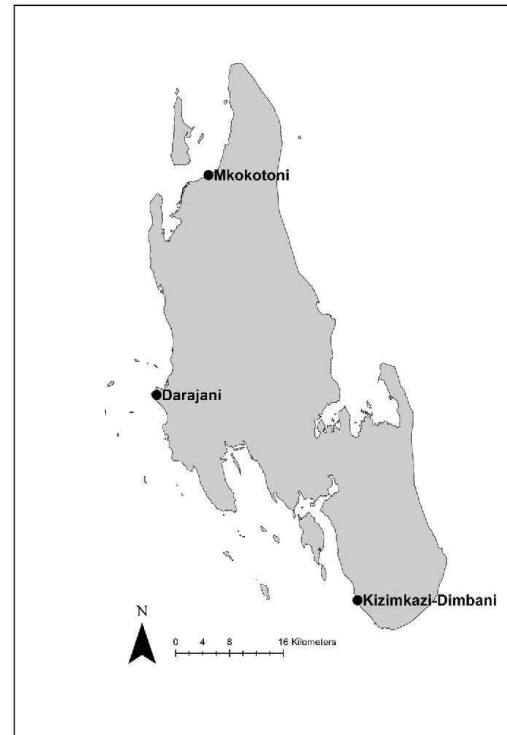


Fig. 1. Locations of the study sites (Kizimkazi-Dimbani: 6°16'12"S, 39°27'12"E; Darajani market, Stone Town: 6°9'44"S, 39°11'36"E; and Mkokotoni: 5°52'30"S, 39°15'18"E) on Unguja Island, Zanzibar, East Africa, where sampling took place between 28 June and 22 August 2015.

merchants interviews (n = 2) were undertaken in Kizimkazi-Dimbani (6°16'12"S, 39°27'12"E), with merchants' interviews also conducted in Mkokotoni (n = 10) (5°52'30"S, 39°15'18"E) and the Darajani market in Stone Town (n = 4) (6°9'44"S, 39°11'36"E) (Fig. 1). In addition to interview data, elasmobranch catch was sampled between 28 June and 22 August 2015 during the Kusi season. This included elasmobranchs sold at the Darajani market and those landed in Kizimkazi-Dimbani and Mkokotoni.

2.2. Survey design

Fishers and merchants were asked for verbal consent to participate in the survey and anonymity was guaranteed before undertaking the face-to-face interview. Interviews were carried out in private with only the interviewee, translator and interviewer present. Main interview questions recorded qualitative and quantitative data on fishers' target catch and fishing gear; the primary fish species sold and fish landing sites used to source elasmobranchs for sale by merchants; and income and historic changes in the abundance and value of elasmobranchs. Fishers and merchants were asked to list the elasmobranchs they commonly caught or bought by their local Swahili name. Local names were matched with the elasmobranch species by direct identification of catch to record the correct scientific name. Interviews were conducted in Swahili, directly translated and recorded in English. Interviews lasted 15–20 min and were arranged at the convenience of the fishers and merchants.

2.3. Elasmobranch catches data and trade information

Elasmobranch catches data were collected during daylight hours at the two landing sites, Kizimkazi-Dimbani and Mkokotoni. The data recorded included: species, local common name, fishing gear, weight in kg, size in cm (shark and guitarfish total length; batoid disc width) and sale price in Tanzanian Shillings (TZS; 1 TZS = 0.7 British pounds or 0.9 US dollars). Data were collected with permission and assistance from local fishers. A total of 480 elasmobranchs were sampled (Kizimkazi-Dimbani, $n = 225$; Mkokotoni, $n = 146$; Darajani market, $n = 109$). Sale prices were recorded for 153 of the elasmobranchs sampled in TZS (Kizimkazi-Dimbani, $n = 78$; Mkokotoni, $n = 36$; Darajani market, $n = 39$). The trade of two thirds ($n = 150$) of the elasmobranchs landed at Kizimkazi-Dimbani ($n = 225$) was also recorded.

2.4. Data analysis

Maximum income for the Kusi (June–September) and Kaskazi (November–March) seasons stated during fishers' interviews in Kizimkazi-Dimbani was used to investigate if there was a significant difference in income between the two seasons for fishers interviewed. Elasmobranchs used for subsistence purposes had no monetary value and were excluded from price data analyses. Guitarfish were excluded from ray price data analysis because their body form is more similar to that of sharks, which affects their monetary value and could skew the results. Price per unit weight (TZS/kg) was calculated for a total of 86 elasmobranchs sold from landings in Kizimkazi-Dimbani and Mkokotoni. Price at first sale (TZS/kg) and weight (kg) data of elasmobranchs were used to investigate if there was a significant correlation for sharks or rays. Elasmobranch species with less than four sale values recorded were excluded from species comparisons due to the low sample size. Price at first sale data was also used to investigate if there was a significant difference in median price among four shark species and among five ray species.

The Kolmogorov-Smirnov test was used to test for normality and Levene's test was used to test for equality of variance. If the assumptions for a parametric test were not met and data could not be transformed, then an alternative non-parametric test was utilised. Where post hoc analysis was required, Bonferroni adjustment was used to control for type 1 error. Statistical tests were conducted using SPSS Statistics 22. A significance level of 5% was used.

3. Results

3.1. Interview data

All fishers ($n = 39$) interviewed in Kizimkazi-Dimbani stated that fishing was their primary occupation, with the majority ($n = 38$) fishing for both subsistence and sale purposes, and only one fisher stating he fished solely for subsistence purposes. The majority of fishers ($n = 30$) stated that 76–100% of their household income came from fishing illustrating a high level of dependence on this activity for income generation. The household income of some fishers ($n = 7$) was 100% dependent on their fishing occupation as the sole income for their household, whilst others reported they had additional income from supplementary occupations ($n = 20$) or their wives' occupation ($n = 8$). Supplementary occupations included agriculture, tourism and maintenance of boat engines, whilst wives' occupations included tailoring, agriculture and beachcombing. There was no statistically significant difference in mean annual income of fishers in Kizimkazi-Dimbani between the Kusi (mean = 577,077 $\pm$ 662,218 TZS S.D.) and the Kaskazi seasons (mean = 764,513 $\pm$ 1,085,776 TZS S.D.) (t -test, $t = -0.885$, $df = 76$, $P > 0.05$). Nevertheless on an individual basis, fishers stated different reasons for earning more in a particular season (Table 1). The reasons that were given covered weather, fishing effort, gear, target catch and consumer demand. The majority of merchants

interviewed at the three locations stated that trading fish was their primary occupation ($n = 15$) and that 76–100% of their household income came from this occupation ($n = 11$). Merchants reported supplementary occupations ($n = 7$) such as agriculture and auctioneering, whilst one merchant also reported that his wife's occupation of baking and agriculture supported their household income. Those that did not have supplementary incomes or additional income from their wives stated that their household income was 100% dependent on their occupation as a merchant ($n = 9$). Like the fishers, there was variation in whether merchants earned more during the Kusi or Kaskazi seasons (Table 1). Of the fishers ($n = 36$) that caught and sold elasmobranchs, one third reported that 41–60% of their income came from elasmobranch catch. Of the merchants ($n = 16$) that sold elasmobranchs, 31% reported that 61–80% of their income came from selling elasmobranchs. However, for some fishers ($n = 8$) and merchants ($n = 1$) elasmobranch sale represented 0–20%, whilst for other fishers ($n = 4$) and merchants ($n = 2$) it represented 81–100% of their income. This was supported by variation in the reported number of elasmobranchs caught by fishers in the past year. For example, 36% of fishers that stated they caught sharks ($n = 36$), reported they had caught between 1 and 10 sharks in the past year, whilst 19% reported they had caught over 50 sharks. Similarly, 34% of fishers that stated they caught rays ($n = 35$), reported they had caught between 1 and 10 rays in the past year, whilst 17% reported they had caught over 50 rays.

Of fishers ($n = 39$) surveyed, 69% and 84% considered shark and batoid numbers respectively to have decreased in their fishing grounds, whilst 69% and 87% considered the sale value of sharks and batoids respectively to have increased since they had first started fishing, which ranged from 3 to 51 years. Fishers offered reasons such as overfishing ($n = 5$) for the stated decrease in elasmobranch numbers and reasons such as lower catch ($n = 3$) and increased demand ($n = 8$) for the stated increase in elasmobranch value (Table 2). It was also reported that price fluctuated with supply and demand. For example, throughout the holy month of Ramadan (17 June–18 July 2015; the majority of Zanzibar's population follow Islam) and during periods of bad weather, lower fishing effort and catch increased the sale value of fish caught. Some fishers commented that they took advantage of this to obtain higher prices for their catch when there was lower fishing effort or poorer sea state. Despite the fluctuating price of elasmobranch catch, fishers had consistent costs, including vessel fuel, public transport and bait. Salaries for those who helped transport elasmobranchs, either from the vessel to the landing site or to the Darajani market in Stone Town (the main location where fish was sold) were an additional cost. It was also reported that money was put aside for vessel and gear repairs. All fishers stated that the remaining profit from the sale of catch was split equally between all boat crew including the captain.

Of fishers ($n = 39$) interviewed in Kizimkazi-Dimbani, 69% stated that they used their main gear 6–7 days per week as a yearly average. Some fishers commented that poor weather conditions ($n = 2$) and lack of bait ($n = 4$) were reasons why they might fish less, with many also reporting that they did not fish on Fridays and that there was less fishing during the holy month of Ramadan. Longline was used by 59% of the fishers interviewed but responses indicated that fishers' main gear altered at different times of the year depending on the season, weather constraints and target preferences. Top targets were mainly large pelagic fish such as tuna as well as coral reef fish including grouper. Elasmobranchs were listed by 49% of fishers as target species; these fishers mostly used longline as their main gear. Handline, bottom-set gillnets and drift gillnets were reported as additional gear that sometimes caught elasmobranchs, despite not targeting them.

The majority of the 39 fishers caught batoids ($n = 35$) and sharks ($n = 36$), with 38 fishers reporting that they sold the majority of the catch. The price range stated in interviews for shark or guitarfish fins was 10–35,000 TZS/kg with bottlenose wedgefish (*Rhynchobatus australiae*) fins considered the most valuable and highest quality for consumption. The fins of small sharks were discarded. The value of meat

Table 1

Reasons stated by fishers (n = 27) and merchants (n = 14) for earning more during the 'Kusi' or 'Kaskazi' monsoon season during questionnaire surveys conducted between 7 and 21 August 2015 in Kizimkazi-Dimbani, Mkokotoni and Stone Town, Unguja Island, Zanzibar. The number of fishers that stated each reason is given in parentheses.

Higher income in Kusi		Higher income in Kaskazi	
Fishers	Merchants	Fishers	Merchants
Higher elasmobranch price/demand: earns more from elasmobranchs (5)	Fish less abundant so higher value (1) Elasmobranchs targeted have a higher sale price than other fish species caught, generating a higher income (1)	Calmer seas: increased fishing effort and higher catch (7) Fish more abundant (4) Fish species caught have a higher sale price (2) More bait (1) Increased demand & abundance of fish (1)	Fish more abundant & larger (5) Calmer seas: higher fish catch (3) More boats/fishers: higher fish catch (2) Season where fish are more abundant (2)
Calmer seas: increased fishing effort (3) Cooler seas: greater abundance of fish (3) Increased fishing effort (3) Gear used catches more fish (1) Gear used & greater abundance of fish (1) Strong winds: fewer fish caught so increased sale price (1) Fish species caught have a higher sale price (1)	Calmer weather: increased number of fishing vessels active (1)		

also varied for different species depending on the perceived quality. For example, the meat of hammerhead sharks (*Sphyrna* spp.) was considered of high quality whilst tiger sharks (*Galeocerdo cuvier*) and blotched stingrays (*Taeniurops meyeri*) were viewed as lower quality, due to their 'consumption of anything' and 'watery flesh', respectively. The mean price of two smooth hammerhead sharks (*Sphyrna zygaena*) recorded in this study was 2995 TZS/kg compared to 2778 TZS/kg recorded for two tiger sharks and 562 TZS/kg for four blotched stingrays (Table 3).

3.2. Elasmobranch landings and trade data

The trade of 150 elasmobranchs landed at Kizimkazi-Dimbani was documented; 86% were sold whilst the remainder were used for subsistence purposes (food or bait) (Fig. 2). This equated to 98.7% of the total weight landed being sold. One third of those used for subsistence purposes were smooth-hound sharks (*Mustelus* spp.), whilst another third were bluespotted maskrays (*Neotrygon caeruleopunctata*) and all were ≤ 7 kg. Although 43% of elasmobranch landings were sold locally in Kizimkazi-Dimbani, this only equated to 4.3% of the total weight sold. This was compared to 92.1% of the total weight landed (50% of total landings) being sold at the Darajani market in Stone Town. Of those sold at the Darajani Market (n = 65), 62% (50.5% of the total weight sold) were traded through a local merchant who bought elasmobranchs at Kizimkazi-Dimbani landing site and transported them to the Darajani market for sale. Some elasmobranchs (n = 9) were also traded through a second merchant who sold them in a nearby town, Makunduchi (Fig. 2). Generally, elasmobranchs taken for sale outside the village were bought whole. At the Darajani market, large elasmobranchs were fanned, with the remainder auctioned whole or in parts, typically quartered for sharks and halved for batoids. For example, a quarter of a bull shark (*Carcharhinus leucas*), approximately 2.5 m in length (based on two specimen: 242 cm and 252 cm total length) ranged from 100 to 160,000 TZS, whilst its fins sold for 105,000 TZS (17–19% of the total price). Merchants with stalls at the Darajani market would sell slices of meat or smaller whole specimens.

Of elasmobranchs (n = 208) landed in Kizimkazi-Dimbani with recorded weights, 57.7% were caught by longline, making it the predominant fishing gear for catching elasmobranchs during the sampling period (Table 4). The prices at first sale (TZS/kg) of elasmobranchs from the two landing sites were analysed to indicate whether the weight of elasmobranchs influenced their price. There was a statistically significant positive correlation between the first sale price/kg and weight of sharks (Spearman Rank Correlation Coefficient, $r_s = 0.705$, n = 29, $P < 0.05$) and a statistically significant negative correlation between the first sale price/kg and weight of rays (Spearman Rank Correlation Coefficient, $r_s = -0.652$, n = 57, $P < 0.05$) (Fig. 3). Three

specimens of shark ray (*Rhina ancylostoma*) (total lengths (cm) = 167, 194, 196; total weights (kg) = 60, 69, 55; first sale price (TZS/kg) = 1417, 1159, 1455) and two specimens of bottlenose wedgefish (total lengths (cm) = 276, 301; total weights (kg) = 147, 169; first sale price (TZS/kg) = 4286, 3728) were also recorded at Kizimkazi-Dimbani landing site.

Elasmobranchs were further analysed to investigate if there were any species differences in the price at first sale among sharks and rays landed. There was a statistically significant difference in the median price of bull sharks (median = 2500 ± 2575 TZS/kg range), grey reef sharks (*C. amblyrhynchos*) (median = 2628 ± 2259 TZS/kg range), snaggletooth sharks (*Hemipristis elongata*) (median = 2655 ± 1036 TZS/kg range) and milk sharks (*Rhizoprionodon acutus*) (median = 1250 ± 667 TZS/kg range) (Kruskal-Wallis test, $K = 13.7$, $df = 3$, $P < 0.05$) (Fig. 4). Bull sharks, grey reef sharks and snaggletooth sharks all had a significantly greater median price/kg compared to milk sharks (Mann-Whitney U-test, $P < 0.008$) but were not significantly different in median price/kg from each other (Mann-Whitney U-test, $P > 0.008$). There was a statistically significant difference in the median price of thornail stingrays (*Dasyatis thetidis*) (median = 1408 ± 1088 TZS/kg range), coach whips (Himantura uarnak) (median = 1656 ± 3574 TZS/kg range), Javan cownose rays (*Rhinoptera javanica*) (median = 2000 ± 2935 TZS/kg range), Baraka's whips (Maculabatis ambigua) (median = 3000 ± 5703 TZS/kg range) and spotted eagle rays (*Aetobatus ocellatus*) (median = 3556 ± 2711 TZS/kg range) (Kruskal-Wallis test, $K = 16.6$, $df = 4$, $P < 0.05$) (Fig. 4). Baraka's whips were significantly greater in price/kg than thornail stingrays and coach whips (Mann-Whitney U-test, $P < 0.005$) but were not significantly different in price/kg to Javan cownose rays or spotted eagle rays (Mann-Whitney U-test, $P > 0.005$). Thornail stingrays, coach whips, Javan cownose rays and spotted eagle rays were not significantly different in price/kg from each other (Mann-Whitney U-test, $P > 0.005$).

4. Discussion

This study investigated the dependence of local communities in Zanzibar on elasmobranch catches and trade, as a source of income and subsistence. This represents the first initiative to assess empirical data for determining the socio-economic value of an artisanal elasmobranch fishery in the western Indian Ocean. This information is useful for understanding the broader implications of fisheries management measures on fishers and their communities especially in a livelihood dependence and policy development context. The household incomes of fishers and merchants interviewed were highly dependent on fishing as their primary occupation, with 77% of fishers (n = 39) and 69% of merchants

Table 2
Reasons stated by fishers ($n = 30$) for changes in elasmobranch abundance and sale value since they started fishing (range 3–51 years) during questionnaire interviews conducted between 7 and 21 August 2015 in Kizimkazi, Dindani, Unguja Island, Zanzibar. The number of fishers that stated each reason is given in parentheses.

Elasmobranch	Reported change in abundance			Reported change in sale value		
	Increased	Decreased	No change	Increased	Decreased	
Sharks	Low fishing pressure; not highly targeted (1)	Highly fished (4) Increased technology, numbers of vessels and improved gears (3) Decrease of larger sharks (2) Climatic change (2) Large foreign vessels (2)	No change in smaller sharks (2) Catch has always fluctuated but no overall change (1)	Lower catch so higher price (3) Higher cost of living; natural increase in price (3) Increased demand for fins so higher sale price (2) Increased demand (2) Increased price of meat (2) Seasonal fluctuation in sale value but overall increase (1) Increased tourism (1) Increased consumption and therefore demand (1) Increased consumption and therefore demand (6)	Reduced demand for fins so lower price (4) Reduced consumption and therefore demand (2) Reduced demand (2) Restrictions on exporting fins (1)	
Baroids	Climatic change (2) Increased numbers due to pup release (1) Highly fished (1) Gears used more efficient at catching rays so increased catch (1)	Fishers from northern Zanzibar extending their fishing grounds (1) Poor technology: decreased catch (1) Less bait to target fish; decreased catch (1) Highly fished (5) Increased technology & improved gears (2) Fishers from northern Zanzibar use bottom-trawl so overfished (2) Decrease in all fish (1) Climatic change (1) No longer use fence traps so lower catch (1) Not targeted as much so lower catch (1)	Still abundant in known areas (1)			

Table 3

Mean weight (kg) and price (TZS/kg) of elasmobranch species ($n = 13$) recorded in Kizimkazi-Dimbani (KI), Mkokotoni (MK) and the Darajani market (DA) on Unguja Island, Zanzibar between 28 June and 22 August 2015. Local swahili names in bold were reported in fisher and merchant interviews as the primary elasmobranchs caught and sold. The number of specimens used to calculate mean values are reported in parentheses. An additional 15 species were recorded with a sample size < 10 specimens. ND = no data.

Elasmobranchs: scientific name (common name), local Swahili name	Mean weight (kg)			Mean price (TZS/kg)		
	KI	MK	DA	KI	MK	DA
<i>Neotrygon caeruleopunctata</i> (Bluespotted maskray), Katwe mweupe	1.0 (28)	1.5 (32)	1.5 (1)	471 (8)	ND	ND
<i>Himaniura uarnak</i> (Coach whiplay), Taa chui	44 (17)	26 (14)	31.25 (4)	1401 (10)	2169 (6)	1299 (2)
<i>Taeniana lymna</i> (Bluespotted fantail ray), Katwe blu	1.5 (19)	1.5 (26)	1.0 (2)	0 (1)	1333 (2)	1600 (1)
<i>Maculabatis ambigua</i> (Baraka's whiplay), Taa mweupe , Nyenga	ND	10 (40)	18 (7)	ND	2681 (15)	1579 (1)
<i>Mustelus</i> spp. (Smooth hound shark), Papa kinengwe (sg.)	5.0 (36)	ND	ND	143 (7)	ND	ND
<i>Rhinochimaera acutus</i> (Milk shark), Papa vinengwe (pl.)	4.0 (14)	2.0 (3)	4.0 (10)	1321 (7)	ND	4167 (4)
<i>Squalus</i> spp. (Spurdog shark), Papa kinengwe (sg.)	20 (8)	ND	3.0 (18)	0 (1)	ND	ND
<i>Hemipristis elongata</i> (Snaggletooth shark), Papa manyu	33 (9)	29 (1)	8.0 (11)	2598 (4)	2655 (1)	3625 (3)
<i>Carcharias leucas</i> (Bull shark), Papa Sumbwe	95 (13)	ND	80 (1)	3677 (5)	ND	2475 (1)
<i>Dasyatis thetidis</i> (Thorntail stingray), Taa mweupe	72.5 (11)	ND	ND	1577 (6)	ND	ND
<i>Acroteriobatus zanzibarensis</i> (Zanzibar guitarfish), Barrobarro	2.0 (3)	1.0 (3)	2.0 (5)	0 (2)	4000 (1)	2000 (1)
<i>Aetobatus ocellatus</i> (Spotted eagle ray), Pungu	ND	23.69 (8)	21.75 (2)	ND	23.69 (8)	2000 (2)
<i>Taeniaraps meyeri</i> (Blotched stingray), Taa maji	40 (7)	21 (1)	22.5 (2)	543 (2)	581 (1)	581 (1)

($n = 16$) stating that 76–100% of their income came from this activity. The relative importance of elasmobranch catch and trade to the income generated by being a fisher and merchant was influenced by a combination of interacting factors including season, weather, fishing effort, fishing gear, target catch and consumer demand. These factors, along with size and species, caused variation in elasmobranch price per kilo.

Often, seasonal differences were cited as main reasons for variation in the sale value of fisheries catch and income generated. However, the situation was much more complex than comparing the two monsoon seasons, with fluctuations in catch and price on smaller temporal scales, as well as individual differences in gear use and target catch preferences. This may account for contradicting reasons given by different fishers and merchants for which season their income was higher (Table 1). Whilst this can make it difficult to interpret results, it also highlights the importance of fishers' knowledge in understanding the complex drivers influencing their fishing practices. Some fishers reported that seasonal differences in elasmobranch catch and price influenced in which season their income was higher (Table 2). For example, five fishers reported that they earned more in the Kusi season due to a higher demand and therefore higher price for elasmobranchs. Fluctuation in elasmobranch price, and income generally, was evident over smaller temporal scales due to factors including weather, fishing effort, bait availability and cultural reasons. Findings from this study suggested that fishers interviewed utilised different fishing gears and altered their target catch with these temporal fluctuations. Engaging in multiple fisheries provides social and economic resilience by adapting to temporal changes in climate and available catch, and should be accounted for in fisheries management [30].

Fluctuation in elasmobranch price was evident historically,

Table 4

Number and weight (kg) of elasmobranchs landed by different gears in Kizimkazi-Dimbani, Unguja Island, Zanzibar between 28 June and 22 August 2015 including the percentage of total number of elasmobranchs landed ($n = 208$) and percentage of total weight (5243 kg) landed by all gears.

Gear	No. elasmobranchs landed	% of total no. landed	Weight landed (kg)	% of total weight landed
Longline, big hooks	36	17.30	2175	41.50
Longline, small hooks	30	14.40	575	10.95
Longline, unspecified	54	26.0	2069	39.45
Bottom-set gillnet	38	18.30	83	1.60
Unspecified net	2	0.95	5	0.10
Handline	2	0.95	47	0.90
Spear	12	5.70	16	0.30
Unspecified gear	34	16.40	273	5.20
Total	208	100	5243	100

seasonally and over smaller temporal scales. The majority of fishers interviewed in this study reported that elasmobranch numbers had decreased in their fishing grounds, whilst value had increased since they first began fishing (Table 2). Fishers perceived changes in elasmobranch catch as the result of a range of factors such as overfishing, with an increased number of vessels and more efficient gear. Changes in value were perceived due to combined factors of lower catch and increased demand as well as a higher cost of living. However, some of the

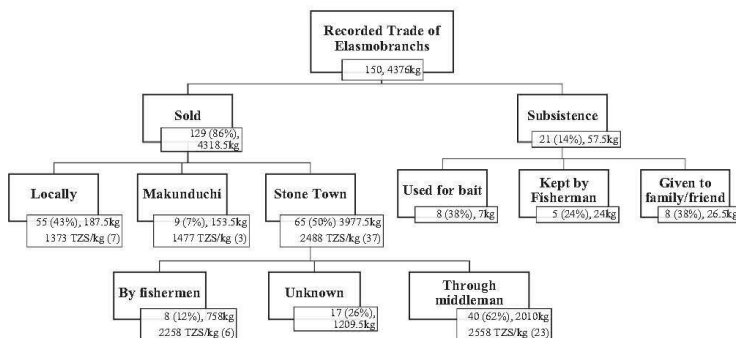


Fig. 2. Recorded trade of elasmobranchs ($n = 150$) from Kizimkazi-Dimbani, Unguja Island, Zanzibar between 28 June and 22 August 2015 utilised for subsistence or sale purposes. The number of elasmobranchs is stated with the percentage of total number recorded in parentheses, followed by total weight (kg). Mean sale prices (TZS/kg) are also indicated with the number of recorded specimens' values are based on given in parentheses.

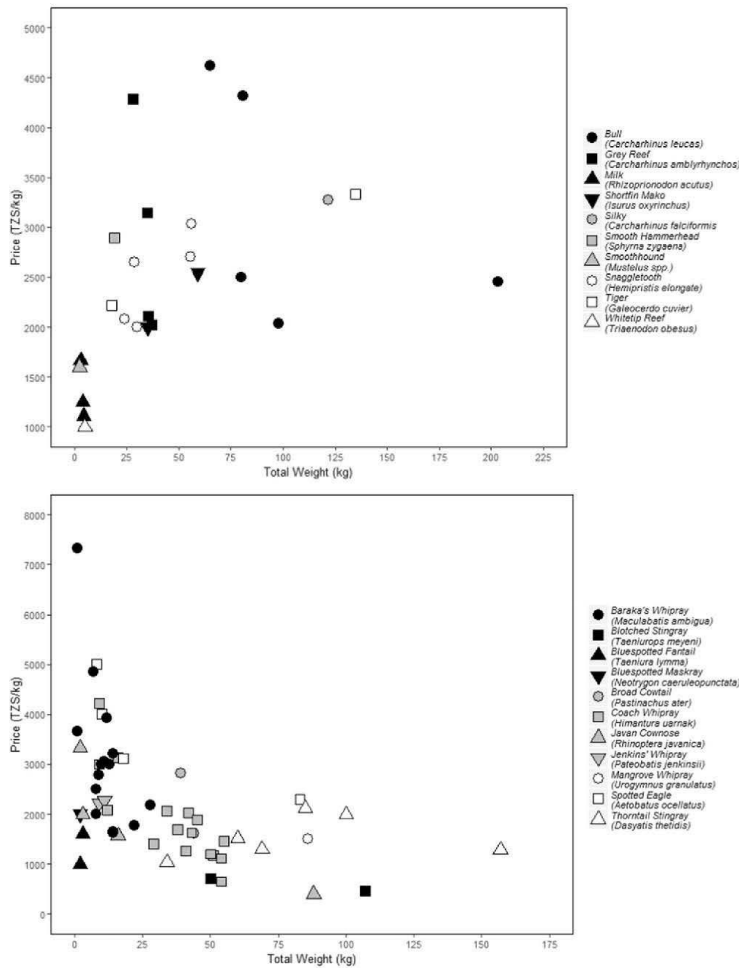


Fig. 3. First sale price (TZS/kg) and total weight (kg) of a) ten species of sharks ($n = 29$) and b) eleven species of rays ($n = 57$) landed in Kizimkazi-Dimbani and Mkokotoni, Unguja Island, Zanzibar between 28 June and 22 August 2015.

reasons cited by different fishers were contradictory, for example, some thought the demand for elasmobranch products had increased, whilst others thought it had decreased along with the sale value (Table 2). This could result from the large variation in the number of years fishers had been fishing and requires further investigation. Some fishers commented that larger sharks had decreased in number whilst smaller sharks had remained the same (Table 2). This may be a result of the typically greater vulnerability of larger elasmobranch species and corroborates the likely overexploitation and partial collapse of Zanzibar's elasmobranch fishery [31]. Further research is needed to increase understanding of these temporal changes in Zanzibar, the adaptive approach fishers and merchants employ, and the influence it has on their livelihoods, in order to aid future fisheries management.

The recorded trade of elasmobranchs from landings in Kizimkazi-Dimbani indicated that size influenced the market for elasmobranchs. Smaller specimens were kept for subsistence or sold locally, whilst larger specimens were generally sold at the Darajani market in the capital city, Stone Town (Fig. 2). Whilst there was a domestic market for elasmobranch meat in Zanzibar, the high price of fins was driven by

an international export market with reports of fin traders operating from the Darajani market. Analysis of the landings data showed that first sale price/kg had a significant positive correlation with weight of sharks but a significant negative correlation with weight of rays (Fig. 3). In global markets, fins are disproportionately higher in value than meat [32]. Interview responses from this study indicated that the price of sharks and guitarfish increased with size due to large, valuable fins. For example, the fins of two bull shark specimens recorded in this study were worth nearly 20% of the total shark price. Whilst sale price is likely to vary depending on the season and availability, our findings suggest that the high fin value offsets any reduction in meat value per kilo. Any management measures would need to account for the disproportionately higher value of large sharks compared to small sharks and provide incentives to replace the market for highly valued, large fins.

Results showed that there was variation in the first sale price/kg amongst different elasmobranch species (Fig. 4). In some cases, this was likely due to inherent size differences between species, given price/kg was found to increase with weight (Table 1) (Fig. 3). For example, milk

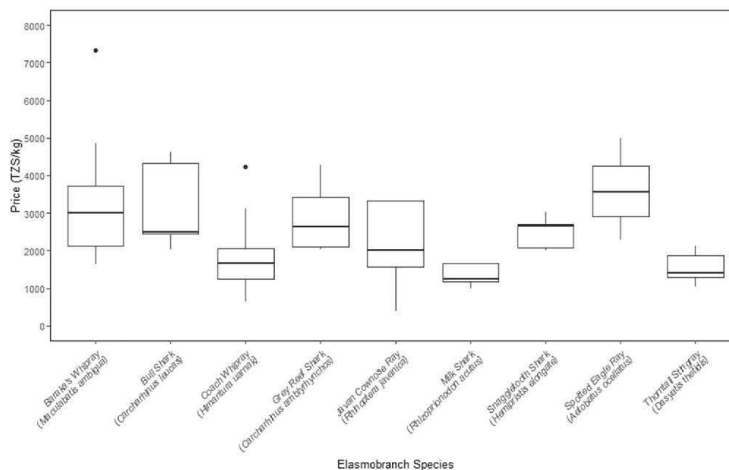


Fig. 4. Median ($\pm$ lower and upper quartiles, $\pm$ range) price at first sale (TZS/kg) of four species of sharks ($n = 21$) and five species of rays ($n = 47$) landed in Kizimkazi-Dimbani and Mkokotoni, Unguja Island, Zanzibar between 28 June and 22 August 2015. Baraka's whips (Maculabatis ambigua) were significantly more expensive than thorntail stingrays (Dasyatis thetidis) and coach whips (Himantura uarnak). Milk sharks (Rhizoprionodon terraenovae) were significantly cheaper than the other three shark species.

sharks were significantly cheaper per kilo compared to other shark species analysed, which could be due to their typically smaller size. Similarly, one longline fisher reported only fishing elasmobranchs for consumption due to only catching bluespotted fantail rays (*Taeniura lymna*), which were typically low in value due to their small size. However, in other cases, the 'quality' of the meat and fins for consumption was reported to vary among species and therefore to influence the price. For example, the meat of hammerhead sharks and the fins of guitarfish were considered of high quality. The lower caudal fins of hammerhead sharks and guitarfish have been reported by traders in China as a source of high quality fin needles for consumption [33], which could drive a high export value for these species in Zanzibar. Understanding the drivers of demand for elasmobranch products and the factors that influence price through methods utilised in this study is important to inform the management of elasmobranch fisheries and markets. For example, recording fin values once they are removed from specimens at the Darajani market in Stone Town would give the proportion of the total sale value that fins account for. Further, following their trade would provide insight into the influence of the international market for different products on domestic elasmobranch catch.

Understanding the variation in price among elasmobranch size classes or species could help inform a more tailored approach to fisheries management to meet both conservation and socio-economic requirements. For example, limiting fishing to particular juvenile age classes, as oppose to targeting all age classes, is a more strategic approach to addressing recruitment issues and has been shown as a potential management strategy to improve the sustainability of elasmobranch fisheries [34]. This was shown to be effective for species with lower productivity and lower natural mortality rates. This would require appropriate fisheries management, adapting fishing practices to protect adult elasmobranchs, for example through gear modification, but also appropriate knowledge of elasmobranch behaviour, for example habitat utilisation. Given that the first sale price/kg was found to decrease with increasing ray weight, fisheries may be more economically resilient to catch restrictions on larger rays since price/kg did not increase with size. It is also important to understand any variation in price between different elasmobranch species, how this influences fishers' target catch and how this could contribute to protecting more vulnerable species. However, the current lack of species-specific information on elasmobranch biology and catch composition of Zanzibar's elasmobranch fishery needs to be addressed in order to assess elasmobranch population status and recommend sustainable fishing

practices.

5. Conclusions

The socio-economic importance of Zanzibar's elasmobranch fishery to local communities has not previously been determined. This is the case for the majority of countries worldwide despite wide recognition that this information can inform development of more tailored fisheries management strategies. Here, findings show that elasmobranchs caught provide a source of income, as well as subsistence. The importance of elasmobranch catch and trade in supporting fishers' and merchants' livelihoods varied according to a number of key interacting factors including season, weather constraints, fishing effort, fishing gear, target catch preferences and consumer demand. These factors contributed to fluctuation in elasmobranch catch biomass and price historically, seasonally and over shorter temporal scales. The price at first sale was also shown to vary among different elasmobranch species and size classes. This information could be utilised in tailoring locally context specific, elasmobranch fisheries management.

As a first initiative to assess the socio-economic value of an artisanal elasmobranch fishery in the western Indian Ocean, this study highlights how social, economic and trade characteristics can help inform the structure of future elasmobranch fisheries policy. However, it is important to recognise the limited spatial and temporal coverage; a larger more comprehensive study is required to accurately assess Zanzibar's elasmobranch fishery and to regionally assess elasmobranch populations. The vulnerability of many of the elasmobranch species recorded, according to their IUCN red list classification, emphasise the need for species-specific catch and biological data to be examined within the socio-economic context, so as to better understand drivers influencing catch and effort. This study has highlighted how future socio-economic research combined with population dynamic studies can identify the actions needed to better understand understudied elasmobranch fisheries locally, regionally and globally. This broader knowledge can then be used to ensure that any fisheries management methods employed will more likely have a positive impact on the health of elasmobranch stocks and the sustainability of the fisheries that target them.

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Chapter 3. Fisher Dependence, Use and Value of Elasmobranchs in Southwestern Indian Ocean Small-Scale Fisheries

3.1 Abstract

The vulnerability of small-scale elasmobranch (ray and shark) fishers in the southwestern Indian Ocean is exacerbated by convergence of the inherent vulnerabilities of fishers and fisher households, elasmobranch resources and the coastal ecosystems upon which they all rely. This study provides an assessment of the dependence of fishers, and fisher households, on elasmobranch resources and the broadscale socio-economic context of these fisheries. Face-to-face interviews (n=521) with fishers were conducted at 23 sites across Kenya, Zanzibar and northern Madagascar, collecting data relating to fisher and fisher household demographics, fisheries activity and perceived drivers of elasmobranch use and value. In addition, use (sale, sustenance or bait) and value data (n=2908) were collected at landing sites over 12 months in 2016–17. Fisher dependency on elasmobranch resources was linked to financial capital and fishing experience, which were also found to influence proportional adult (≥ 18 years) engagement in income generating employment and relative household dependence on fisheries for income. The findings suggest that elasmobranch dependent households tend towards specialist livelihood strategies relative to the rest of the fishery, and may therefore be less resilient to social, economic and environmental shocks. Further, the findings suggest that infrastructure and access to external markets are linked to commercial demand for elasmobranch products, primarily for shark and shark-like rays which appear to be supply-limited, in the region. A management strategy based on market governance targeted above the fisher level may be effective in altering fisher behaviour and dependence on elasmobranch resources. However, such strategies risk impoverishing those fishers most dependent on these resources given their specialised livelihoods. Targeted programs to increase livelihood diversity, including alternative livelihoods may provide a pathway through which to decrease the vulnerability of these fishers to external shocks as well as decreasing the fisheries pressure on the regions vulnerable elasmobranch resources.

3.1.1 Key Words

Capital; Livelihoods; Vulnerability; Resilience; Market Governance

3.2 Introduction

The Sustainable Livelihoods Framework (SLF) seeks to take a holistic approach to livelihood development and poverty reduction through better accounting for the varied complexities of household livelihood strategies (Ashley and Carney 1999). Additionally, SLF can also be applied to improving wellbeing, a more rounded approach partially removed from traditional poverty-centric thinking, wherein basic human needs are met and a satisfactory quality of life achieved (Coulthard *et al.* 2011; Weeratunge *et al.* 2014), which may be a more appropriate concept in a small-scale fisheries context. A major component in SLF is the understanding of the vulnerability context in which livelihood strategies are undertaken and how these relate to their livelihood strategy, with the goal of promoting increased resilience. Vulnerability refers to the perceived exposure to risk to livelihoods from external shocks, e.g. economic, social or environmental, that are beyond their control. Livelihoods displaying high degrees of vulnerability to various shocks, which may themselves be interrelated and result in cumulative effects, are at particular risk and should be considered as priorities for intervention.

Small-scale fishers, particularly in developing countries, are generally considered amongst the most vulnerable socio-economic groups as a result of their exposure to a large number of potential economic, social and environmental shocks (Allison *et al.* 2006; Béné *et al.* 2007; Béné 2009). They combine high levels of exposure, high sensitivity and low adaptive capacity to respond to such shocks (Béné *et al.* 2007), including limited ability to mobilise between sectors. This is because they have restricted capital, education and skills with which to do so. Concurrently, and somewhat paradoxically, open-access small-scale fisheries (SSF) play an important role in reducing community vulnerability through provision of a safety net employment facility (Béné *et al.* 2010). SSF are also important for wealth generation and food security at both the individual and community levels (Allison and Ellis 2001; Béné *et al.* 2007). Thus, reducing the vulnerability of small-scale fishers is imperative in supporting the wellbeing and long-term sustainability of fishers and their communities.

The southwestern Indian Ocean (SWIO), here defined as Kenya, Tanzania, Mozambique, Seychelles, Comoros, Mayotte, Madagascar, La Réunion and Mauritius, is a rapidly developing region, with a population set to more than double to 357.3 million by 2050 (WB 2016b). Currently, fish proteins (marine and freshwater) range among countries from 7.3–49.7% of animal protein and 1.9–23.1% of total protein consumed (FAO 2017). The marine SSF sector alone employs nearly half a million fishers across the region, though this is likely a significant underestimate (Temple *et al.* 2018). As the region continues to develop increasing pressure on fish resources for sustenance and income present major challenges for long-term resource, income and food security sustainability.

As with many SSF, SWIO SSF tend to operate in near-shore coastal environments, limited by the technological capacity of fishing vessels and gears. Within near-shore coastal zones much of the fish biomass and diversity is supported either directly or indirectly by productive habitats such as coral reefs, seagrass beds and mangroves (Connell 1978; Mumby *et al.* 2004; Heck *et al.* 2008), with fishing pressure often focussed in or around such habitats. These habitats are themselves vulnerable to a range of both natural and anthropogenically mediated impacts (e.g. McClanahan 1995; Orth *et al.* 2006; Hoegh-Guldberg *et al.* 2007), including from fisheries pressure, which have implications for the stability of these ecosystems and the abundance and composition of species within them. The vulnerability of these habitats to external shocks, including from the SSF themselves, exacerbates the vulnerability context within which these livelihoods exist.

Within SWIO SSF the livelihood of some small-scale fishers may be particularly at risk due to their dependence on species particularly vulnerable to fisheries impacts. Elasmobranchs (sharks and rays) are a common constituent of SWIO SSF catch, as both target and by-product species (Temple *et al.* 2018; Temple *et al.* 2019). Elasmobranchs are particularly vulnerable to non-natural mortalities, of which fisheries are the greatest source at the global level, as a result of their classically k-selected life history traits (Compagno 1990; Żydelis *et al.* 2009; Dulvy *et al.* 2014) with indications of over-exploitation in many areas of the SWIO (Kiszka and van der Elst 2015). Further, as apex and meso-predators loss of elasmobranchs have implications for the structure, function and productivity of ecosystems (Heithaus *et al.* 2008). Thus, the dependence of fishers on these species impacts both the vulnerability context of their own livelihoods and may also impact those of others reliant on the marine environment.

Traditionally SWIO coastal communities use elasmobranchs as sources of income, sustenance and bait (Marshall and Barnett 1997; Barrowclift *et al.* 2017). The majority of elasmobranchs are landed whole and fully utilised (Wekesa 2013; Barrowclift *et al.* 2017; Temple personal observation). Post-harvest finning for the international market is common

(Schaeffer 2004; Maoulida *et al.* 2009; Cripps *et al.* 2015), with a limited fin-and-discard industry at specific locations in Madagascar and Mozambique (Pierce *et al.* 2008; Cripps *et al.* 2015). Elasmobranch meat is a cheap source of protein compared to large teleosts, such as Scombrinae and Xiphiidae species, often caught in the same fisheries (e.g. MFR 2004, 2012) and as such may constitute an important source of protein for some fishers or fishing communities. Despite the potential commercial and subsistence uses of elasmobranchs, little is known of the varied extents and drivers of elasmobranch use and value, nor the dependence of fishers upon them.

The use of SLF to improve the long-term sustainability of fishers livelihoods and wellbeing must be built on an understanding of the strategies in use and the vulnerability context in which they exist (Allison and Ellis 2001). In this study we aim to begin addressing vulnerability context as it relates to elasmobranch fishers in SWIO SSF. Firstly, we aim to assess whether characteristics of fisher and fisher household demographics and/or fisheries activities relate to their dependence on these resources and whether these in turn are linked to other dependencies on fisheries for income or employment. Secondly, we aim to improve the understanding of the use and value of elasmobranch resources to SWIO SSF and the factors that influence these and thus help to define the broadscale socio-economic context in which these vulnerable fisheries exist. The outputs are discussed in the context of future management strategies for SWIO SSF elasmobranch fisheries and the potential for application of targeted livelihoods programs to those most vulnerable within them.

3.3 Methods

Socio-economic data relating to fishers, their households and livelihoods strategies were collected through face-to-face resource user questionnaires (n=521) with fisher and fisher captains. Data were collected at 23 landing sites in Zanzibar (n=8, January-March 2017), Kenya (n=10, January-February 2018) and northern Madagascar (n=5, October 2017-March 2018) (Figure 3.1). Sites were selected based on access to pre-existing data on the vessel number and gear composition of the SSF. Site selection was biased towards those sites where gillnet (bottomset and driftnet) and longline gears were in use, the main threats to elasmobranchs in these fisheries (Kiszka and van der Elst 2015; Temple *et al.* 2018), in combination with maximising geographic spread while balancing logistical constraints (e.g. site accessibility). Questionnaires were carried out in Swahili and Malagasy as appropriate by teams of trained interviewers from each country. Interviewees were selected opportunistically, avoiding repeat interviews and interviews with multiple crew from the same vessel. Surveys were relatively long (approximately 1 hour) and covered a range of socio-economic and governance related topics (Appendix A.). Data relevant to the present study include those on fisher and fisher household demographics and fisheries activity (Table 3.1), and perceived drivers of elasmobranch use and value.

Complimentary data on elasmobranch landings were collected by trained observers at the same landing sites in Zanzibar and northern Madagascar and seven of the same sites in Kenya plus one additional site (Figure 3.1) for a period of 12 months between June 2016 and June 2017. Sampling days were selected using a stratified-random approach: the year was divided into lunar months which were subdivided into four lunar phases (new moon, first quarter, full moon, third quarter) and sampling days randomly generated within each lunar phase. This sampling regime ensured that the study accounted for potential lunar-driven patterns in fishing effort, catch and species composition in the fishery, and thus subsequent effects on catch use and value. Observers recorded data for catch use (sold, kept and/or gifted for sustenance, or bait), sale location (local or external market), price at first sale (in

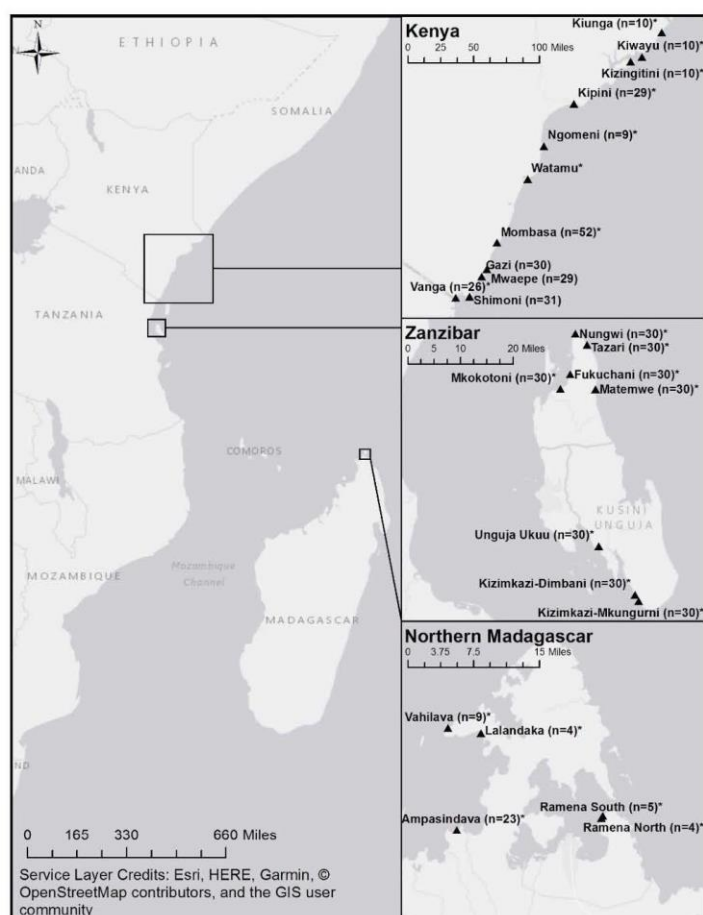


Figure 3.1 Locations of landing sites where surveys and landings observations were undertaken in the southwestern Indian Ocean, sites marked with (*) are those at which landings data was collected, "n=" values represent questionnaire survey sample size and location.

local currency), morphometric data (fork length, disc width and weight), sex and fishing gear used.

3.3.1 Analysis

Unless otherwise stated all analyses and data visualisations were carried out and produced using the R statistical software, version x64 3.4.0 (R Core Team 2017).

To investigate fisher elasmobranch resource dependence a Bayesian network was created incorporating data from interview surveys relating to fisher and fisher household demographics and fisheries activity. A Bayesian network is considered appropriate because it can account for the complex, often confounding direct and indirect interrelationships between factors that generally are poorly accounted for in regression models (Slater *et al.* 2013) and also allows for scenario-testing which can aid decision-making for managers. Data from interview surveys were manipulated into binary and numeric form, scaled between 0 and 1 within each country, as appropriate and subjected to exploratory factor analysis (EFA) (R packages *psych* and *GPArotation*). Data from all interviews were incorporated into the

Table 3.1 Exploratory Factor Analysis outputs, variables with factor standardised loadings of ± 0.4 or greater are detailed and iteratively excluded variables are displayed, sum squared loadings and proportion of variability explained by each identified factors are presented.

Variable	Factor (standardised loadings)					
	F1: Fisher Experience	F2: Fisher Capital	F3: Employment Dependence	F4: Elasmobranch Dependence	F5: Gear Specialism	F6: Income Dependence
Fisher Age	0.85					
Years fishing	0.79					
Years fishing with main gear	0.75					
Reason to fish (income > both > food)		0.53				
Proportion of food that is protein		0.45				
Proportion of protein that is seafood		0.76				
Commonly land catch elsewhere?		-0.52				
Occupation (captain > fisher)		-0.43				
Proportion of adults (≥ 18 years) with an income			0.75			
Proportion of adults (≥ 18 years) who fish			0.88			
Proportion of catch that is shark				1		
Proportion of catch that is ray				0.54		
Proportion of fishing effort with main gear					-0.42	
Number of gears used					1	
Proportion of household income from fishing						0.63
Sum squared loadings	1.96	1.57	1.4	1.33	1.24	0.62
Proportion of variance explained	0.13	0.10	0.09	0.09	0.08	0.04
Variables iteratively excluded from the Exploratory Factor Analysis:	Attitude towards fisheries management; Estimated time to primary fishing ground; Years in education; Religion; Number of adults (≥ 18 years) with income; Number of adults (≥ 18 years) who fish; Primary residence in village or elsewhere; Born in village or elsewhere.					

model, missing data were imputed using the median. The EFA used an *oblimin* rotation, as opposed to the standard *varimax*, as latent factors were not expected to be independent of one another. The number of factors sought were identified using parallel analysis (Hayton *et al.* 2004). Backwards elimination was used to remove variables that did not contribute to any identified factors at a level of 0.4 or greater. From the 25 variables subjected to EFA six latent factors were identified, composed of 15 variables (Table 3.1). These identified latent factors correspond to fisher experience, capital (reflecting a combination of fisher and location financial capital), employment dependence (income generating employment), elasmobranch dependence, gear specialism and fisheries income dependence. Factors

(which formed continuous distributions of data) were discretised into three quantiles of equal representation “High”, “Medium” and “Low”, with the exception of gear specialism which had formed a distinct tri-modal distribution and so was discretised following that instead. The Bayesian network was then formed using averaged network structures from both *hill-climbing* and *tabu* score-based structure learning algorithms with random starting

nodes and 2000 iterations (R package *bnlearn*). Network arcs were included if they appeared in 5% or more of the iterations. Both *hill-climbing* and *tabu* produced the same network. The model was visualised in the software geNie 2.2 Academic and validated for dependence variable nodes using k-fold cross-validation with 100 folds.

Patterns in both elasmobranch use (direct commercialisation defined herein as sale, or for sustenance and/or bait defined herein as subsistence) and value were assessed using a Generalised Additive Mixed Model (GAMM) approach (R packages *nlme* and *mgcv*). Site and species were assigned as random effect variables throughout in order to allow consistent underlying drivers and patterns to be observed. Where independent variables showed evidence of co-linearity ($r > 0.2$ or < -0.2), those with least explanatory power were removed from the model for subsequent iterations. Subsequently, the effects of species on use and value were assessed using a Generalised Linear Mixed Model (GLMM) approach (R package *lme4*). In GLMM models all variables, including site, that had been found significant in the respective GAMM, were assigned as random effect variables in order to allow observation of underlying variation not associated to these variables. Analysis was carried out for rays and sharks separately, with the exception of guitarfish and wedgefish (Order Rhinopristiformes) which were included in the shark analysis. This was considered appropriate as these shark-like rays were considered to be, and treated as, sharks by fishers and merchants. Variables considered in the GAMM and GLMM analyses were informed by fisher declarations of factors affecting species value. They included species, weight (kg), sex, daily elasmobranch catch (as a proxy for supply), month (to explore intra-annual patterns) and location of sale (local or external markets).

3.3.2 Ethics Statement

Survey participants were informed of both the survey motivation and the intended use of the data collected and subsequently verbal consent was sought before the survey was undertaken. Participant's names were recorded but anonymity of their responses was assured. Further, participants were informed of their right to decline any question with which they were unwilling or unsure about answering and, that should they so wish, the interview could be ended at any time. Interviews were not facilitated with either monetary or material motivation. Ethical approval for the survey was sought from and granted by the ethics review board at Newcastle University.

3.4 Results

3.4.1 Elasmobranch Dependence

The three identified latent dependence factors, elasmobranch dependence, employment dependence and income dependence, were found to be directly affected by fisher experience and fisher capital, fisher experience, and fisher capital and employment dependence, respectively. Gear specialism was not found to directly affect any of the other factors, but was affected by fisher capital. The data-driven Bayesian network describing these relationships is displayed (Figure 3.2). k-fold cross-validation of the network found predictive rates of 39.3%, 46.3% and 36.1% for elasmobranch dependence, employment dependence and income dependence respectively, suggesting that addition of further factors in future would stand to improve the predictive power of the network.

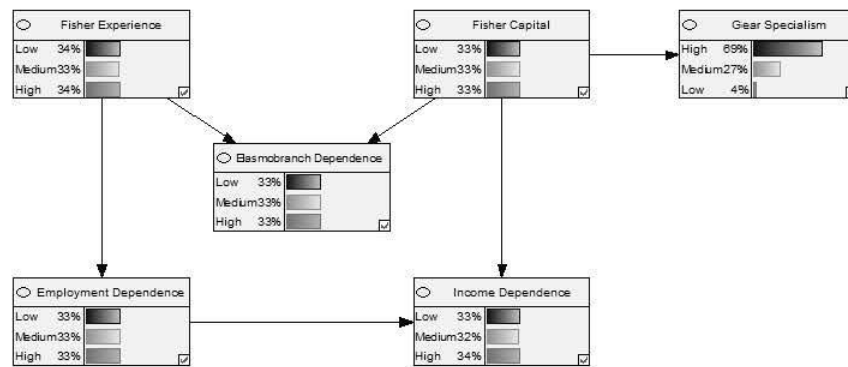


Figure 3.2 Final score-based Bayesian network structure, showing decision nodes and directed influence arcs. “Low”, “Medium” and “High” represent equal discretised quantiles of the underlying continuous factor values identified through exploratory factor analysis, with the exception of Gear Specialism where “Low”, “Medium” and “High” represent the distinct tri-modal distribution displayed by this factor.

In order to identify factors of greatest influence on elasmobranch dependence and how these affected other dependence variables, a sensitivity analysis was carried out with elasmobranch dependence as the target node. Fisher capital was found to have the greatest effect on elasmobranch dependence, followed by fisher experience. Manipulation of the network to select only ‘High’ fisher capital increased elasmobranch dependence (Low 13.5%, Medium 44.1%, High 42.4%) and income dependence (Low 21.7%, Medium 29.9%, High 48.4%). High fisher capital was also found to decrease gear specialism (Low 7.4%, Medium 28.2%, High 64.4%). Subsequently the network was manipulated to select only “High” fisher experience. Elasmobranch dependence increased further (Low 9.8%, Medium 43.1%, High 47.1%) and employment dependence decreased (Low 53.7%, Medium 33.1%, High 13.1%). The alteration to employment dependence also further increased income dependence (Low 18.6%, Medium 30.2%, High 51.2%). The manipulated final network is displayed (Figure 3.3).

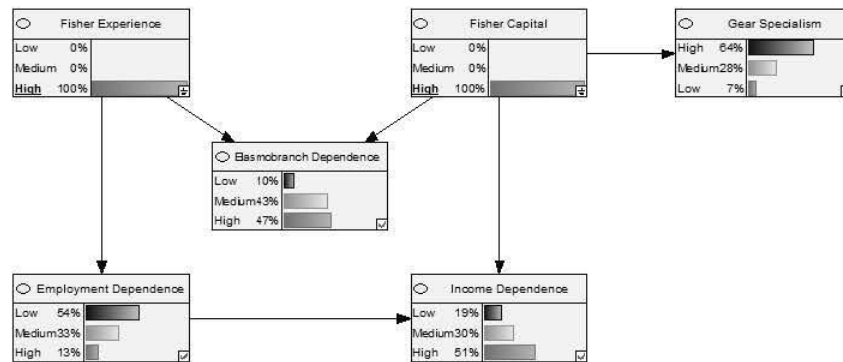


Figure 3.3 Final score-based Bayesian network structure, showing decision nodes and directed influence arcs, post-manipulation of fisher capital and experience factors to “High”. “Low”, “Medium” and “High” represent equal discretised quantiles of the underlying continuous factor values identified through exploratory factor analysis, with the exception of Gear Specialism where “Low”, “Medium” and “High” represent the distinct tri-modal distribution displayed by this factor.

3.4.2 Elasmobranch Use and Value

Usage patterns of elasmobranchs showed substantial variation among countries and sites, Elasmobranchs caught in Kenya and Zanzibar during the period of this study were primarily

used commercially (rays=91.7%, 98.0%, sharks=99.6%, 94.2%, respectively) with limited use for sustenance (rays=8.3%, 2.0%, sharks=0.3%, 5.8%, respectively) and no evidence of use as bait. However, in northern Madagascar commercial use was much less common (rays=28.3%, sharks=24.8%), with catch primarily used for sustenance (rays=63.7%, sharks=65.7%) and the remainder as bait (rays=8.1%, sharks=9.5%).

After co-linear independent variables were iteratively excluded, GAMM analyses showed significant effects ($p < 0.05$) of three variables on ray and shark use (Figure 3.4). Increases in weight reduced the likelihood of rays being retained for subsistence use ($\chi^2 = 24.24$). Further, annual patterns in ray use were found, with the likelihood of sale increasing between April-June and November-February. Shark use was only found to be significantly affected by daily elasmobranch catch ($\chi^2 = 6.64$), with increased supply leading to increased likelihood of catch being retained for subsistence purposes. Assessment of the variable use of elasmobranchs amongst species was attempted but was inhibited by low occurrence of non-commercial use for the majority of species, particularly in Kenya and Zanzibar, thus no meaningful comparisons could be drawn.

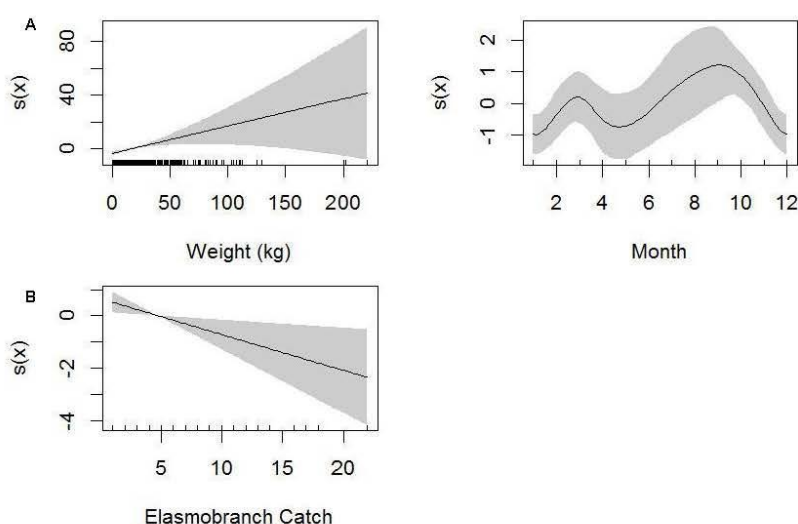


Figure 3.4 Smooths ($\pm 95\%$ CI) from generalised additive mixed models (GAMM) describing significant relationships between A) weight and month on ray use, and B) daily elasmobranch catch on shark and shark-like ray use in small-scale fisheries across Kenya, Zanzibar and northern Madagascar between June 2016-June 2017.

However it follows that as species increase in value they are more likely to be used for commercial gain. Fishers declared a large number of perceived influences on relative species value (Table 3.2). Quantitative assessment of potential influences of species value (price per kg) corroborated a number of those declared by fishers. After co-linear independent variables were iteratively excluded, GAMM analyses showed significant effects ($p < 0.05$) of five variables on ray and shark value (Figure 3.5). Increasing weight resulted in a steady reduction in both ray and shark price per kg ($\chi^2 = 7.58$ and 12.57 , respectively). Though, sharks showed an initial increase in value with weight before the decline, likely linked to the sizes at which shark fins become a saleable asset. However, the effect of weight was small in comparison to annual patterns in value of rays and sharks ($\chi^2 = 978.94$ and 3151.56 , respectively). Both showed increasing value during high tourist seasons (June-September). Ray value declined with increasing daily elasmobranch catch ($\chi^2 = 5.64$), whereas shark value

did not. Simultaneously shark value significantly increased when sold at external markets ($Z=7.58$), whereas ray value did not. Lastly, value of male sharks appeared to be larger than that of females ($Z=2.23$). However, it is noted that there was a higher proportion of moderate-to-large (e.g. 20kg or greater) male sharks (20.8%) than was seen in females (10.9%), which likely contributes to this discrepancy. Subsequent comparison of value (price per kg), whilst controlling variables previously identified as significant, demonstrated significant differences amongst species (Figure 3.6).

Table 3.2 Fisher declarations (n=431) of factors influencing elasmobranch first sale value (price per kg) either in a positive (increasing value) or negative (decreasing value) manner, number of fishers declaring each factor is displayed in brackets.

Positive	Negative
Demand (82): High (75), Festivities (7)	Demand (58): Low (49), High (4), Festivities (5)
Availability (58): Low (44), High (11)	Availability (64): High (48), Low (14), Stable (2)
Season (20): High tourism (11), NE Monsoon (4), SE Monsoon (3)	Season (34): Low tourism (11), SE Monsoon (4), NE Monsoon (3)
Markets (16): Commercial/export buyers (5), Stability (2), City markets (1), Skipping middleman (1)	Markets (15): Poor Access (10), Unstable (1)
Economy (14): Growth and inflation (8), Economic stability (5)	Economy (9): Economic Instability (2), Inflation (2), Politics (1)
Strong local co-operation (6): Among fishers (5), with beach management unit (1)	Weak local co-operation (1)
Species (12)	Species (3)
Quality (44): Freshness (26)	Quality (45): Not fresh (18); Bad preparation/handling (3), Poor storage (2)
Meat (116): Dried (34), Low water content (24), Low water content in rays (47), Good taste (11)	Meat (96): Dried (3), Undried (4), High water content (39), High water content in rays (51), Bad taste (3)
Fins (113): Large size (90), Quality (11), Species (9), Hammerhead (6), Wedgefish (2)	Fins (64): Small Size (62)
Size (35): Large (23), Small (3)	Size (19): Small (19)
Organs (25): Size (1), Large liver (7), Large intestines (1)	
Large Teeth (4)	

3.5 Discussion

This study presents the first assessment of fisher dependence on elasmobranch resources and the patterns and drivers of elasmobranch use and value in SWIO SSF at the fisher level. It begins to explore the broadscale socio-economic context within which elasmobranchs and fishers interact, a vital step in shaping regional-level strategies for the long-term sustainable use of these resources and the coastal communities that rely upon them.

Elasmobranch dependent fishers showed evidence of specialised income livelihood strategies, increasing their vulnerability to economic, social and environmental shocks relative to other fishers in SWIO SSF. Those fishers most dependent on the elasmobranch resources combined increased dependencies on fisheries for income with proportionally lower levels of income generating employment. This was mediated by increasing fisher financial capital and fisher experience, suggesting that such strategies were more common in less impoverished landing sites (potentially linked to market access and greater commercialisation of fishing products) and that such households were specialised in fishing as a livelihood, rather than those using fisheries as a source of short-term or safety net employment (e.g. Béné *et al.* 2010). Low income diversity increases vulnerability to shocks

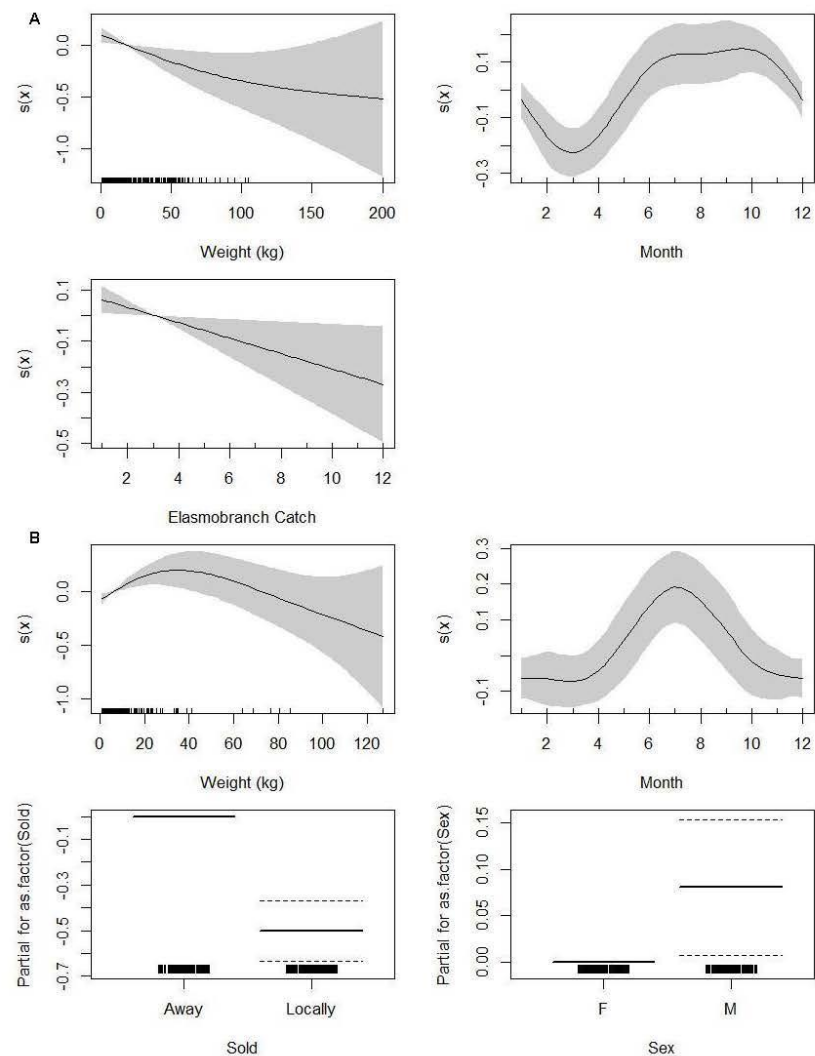


Figure 3.5 Smooths and partial smooths ($\pm 95\%$ CI) from generalised additive mixed models (GAMM) describing significant relationships between A) weight, month and total daily elasmobranch catch for rays, and B) weight, month, sale location and sex for sharks and shark-like rays, in small-scale fisheries across Kenya, Zanzibar and northern Madagascar between June 2016-June 2017.

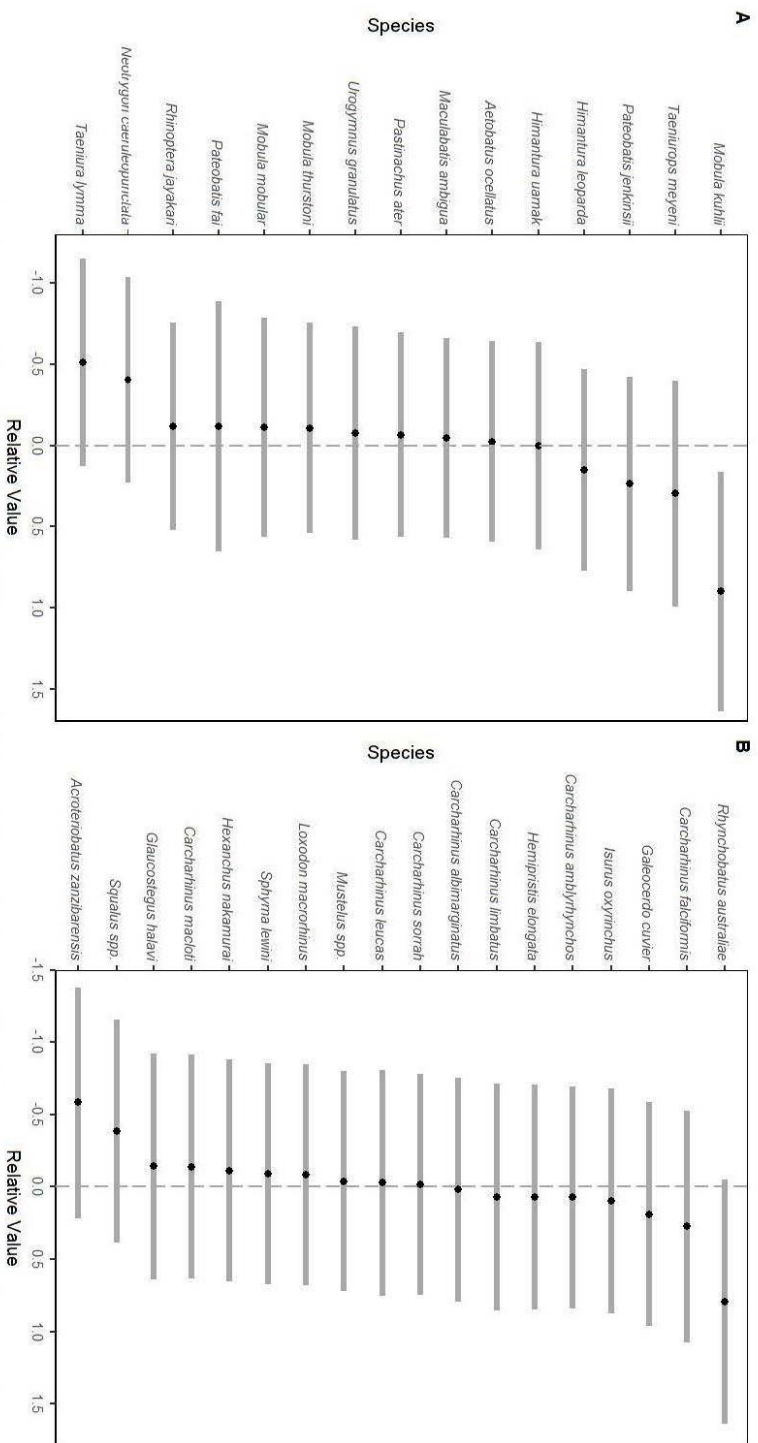


Figure 3.6 Comparisons of relative monetary (price per kg) value (mean $\pm$ 95% CI) amongst species in theoretical space when known significant influences are controlled, for A) rays, and B) sharks and shark-like rays, in small-scale fisheries across Kenya, Zanzibar and northern Madagascar between June 2016–June 2017.

(Allison and Ellis 2001; Béné *et al.* 2007), this is particularly concerning in the context of elasmobranch dependent fishers, given that elasmobranchs are themselves vulnerable to environmentally and anthropogenically mediated shocks and have relatively low rebound potential (Compagno 1990; Žydelis *et al.* 2009; Dulvy *et al.* 2014). Whilst, elasmobranch dependent households potentially have a greater number of supplementary, non-monetary income streams (such as farming), as indicated by their decreased dependence on income based employment, their low income stream diversity suggests they may be priority candidates for livelihood diversification programmes and initiatives. Further, improving income stream diversity may have two-fold benefits, reducing household vulnerability whilst potentially also indirectly reducing fisheries pressure (Allison and Ellis 2001).

Livelihood diversification initiatives present an avenue through which vulnerability of specialised livelihood strategies and pressure on resources may be alleviated, but their success is contingent on appropriate selection of target communities. Aquaculture is a commonly proposed alternative livelihood for fishers, particularly in small-scale developing fisheries. However, successful initiatives for aquaculture, and so also other alternative livelihoods, may be undermined by a poor understanding of localised socio-economic contexts, attitudes and perceptions towards them (Harrison 1996; Kaiser and Stead 2002; Slater *et al.* 2013). It also should be considered that the variable personality traits of those persons targeted by livelihood initiatives, be they fishers or other members of the household, will likely affect their interest in the proposed livelihood, their performance within the livelihood and the satisfaction they derive from conducting the activity (Barrick and Mount 1991; Mount *et al.* 2005; Schmitt *et al.* 2008), with implications for engagement. Thorough investigation of these aspects is therefore a priority if such strategies are to be considered for implementation in elasmobranch fisher households.

More generally the study notes that despite the multi-gear nature of many SSF, fishers in this study showed a high degree of gear specialism. This specialism may result from low financial capital, inability to purchase multiple gears, or external ownership of fishing vessels, often by merchants, where gear use is prescribed. Regardless, high gear specialism likely further constrains their adaptive capacity and may therefore increase their vulnerability (Béné 2009). Assistance with such adaptation is possible through government backed gear exchange and similar schemes. However, such schemes may be costly if fishers are to be properly trained in the use of new gears or methods and as such are likely to be difficult to implement at the regional scale. Further, if handled poorly these schemes may compound existing issues and result in damaging unintended consequences. An example of which is the proliferation of gillnets in Zanzibar, further facilitated by gear replacement schemes, which have contributed to the current issues for elasmobranchs, as well as likely exacerbating an already unsustainable marine mammal bycatch (Amir *et al.* 2002).

Initiatives to decrease fisher vulnerability, adaptive capacity and resilience also require an understanding of the socio-economic context of the fishery they seek to alter. Use of elasmobranchs in SWIO SSF differed among countries and sites. Broadly, in Kenya and Zanzibar ray and shark catches were primarily used for commercial gain at the fisher level, whereas catches in northern Madagascar were predominantly for subsistence. Whilst there are a range of potential socio-cultural reasons for such differences in elasmobranch usage, access to markets have significant effects on resource usage and fisheries pressure (Sierra *et al.* 1999; Brewer *et al.* 2012). Road infrastructure, and thus access to markets, is substantially better in both Kenya and Zanzibar compared to Madagascar. Given that access

to market is vital in minimising spoilage of fisheries products and access to wider markets allows for a greater degree of commercialisation of fisheries products, these infrastructure differences likely explain much of the polarisation in elasmobranch usage between these countries. Further, increasing access to markets, which may further connect to international markets, lead to market-driven influences on supplier behaviour - as is corroborated by fisher's declarations of commercial and export buyers, market access and shark fins as drivers of elasmobranch value. It follows that as SWIO nations continue to develop and thus demand from external and foreign markets increases, elasmobranch fisheries, where they haven't already done so, are liable to become increasingly income driven.

The demand from international markets for elasmobranch products is already evident in the SWIO region, as is reflected in the findings of species value in this study. Primary international demand for elasmobranch products revolve around the fin and gill plate trade from East Asia (Dent and Clarke 2015), and trade within the SWIO for meat (Cooke 1997). Shark fin trade and export is evident throughout the SWIO region (Marshall and Barnett 1997; Temple *et al.* 2018), though its magnitude is poorly quantified. Large shark and wedgetfish species are highly sought after, and this is reflected in their relative value evidenced in this study. Similarly, given the high catch rates in certain areas (Temple *et al.* 2019), the growing international trade in Mobulid (*Mobula* spp.) ray gill plates has potential implications for these fisheries. Whilst the relative value of mobulids evidenced in this study suggests that the trade has not yet affected the study areas covered, there is evidence of the trade in near-by Mozambique (Dent and Clarke 2015), where populations are thought to be in steep decline as a result of fisheries pressure (Rohner *et al.* 2017). High market demand, increasing the likelihood of targeted fishing, is of particular concern given a number of these prized species are considered to already be at significant risk of overexploitation in SWIO SSF (Temple *et al.* 2019).

Within the SWIO region elasmobranch catches appear to be primarily driven by the shark and shark-like ray sub-components. Shark and shark-like ray value increased when sold at external markets and there was no evidence of value decreasing with increasing supply, indicative of a supply limited market. Conversely, ray value showed no evidence of increases at external markets and showed significant decrease with supply, suggestive of oversupply. Both sharks and rays showed evidence of other market-drivers, such as decreased value per kg with weight and fluctuation in value with annual changes in relative supply-demand levels, possibly driven by increased demand for fisheries products (particularly large teleosts) inflating market prices during high tourism seasons and annual fluctuation in supply of fisheries products. Shark value per unit weight does increase initially, peaking at around 40kg, likely reflecting the initial value increase as fins reach marketable size. In spite of the apparent inverses in supply and demand relationship, fisher's ray and shark catches were found to be positively correlated. Further, both taxa are commonly caught in the same fishing gears, primarily bottom-set and drift gillnets (Temple *et al.* 2019). Thus, the two fisheries cannot be broadly considered as independent of one another, though this may be the case in some areas. As such, fisheries management interventions are liable to have cross-taxa implications, with opposition to changes most likely to be encountered in the context of their effect on shark catches. Thus management strategies and livelihood interventions targeting elasmobranch fishers are likely to be most effective when formulated primarily around the shark component, but may be leveraged for indirect effects on rays.

The likely increasingly market-driven nature of elasmobranch catches in SWIO SSF, presents an opportunity for the use of broadscale formal market governance aimed at curbing demand for elasmobranch products but targeted above the fisher level. An example of such measures might be bans, restrictions or incentivising sustainable shark fin exports. Zanzibar ceased issuing shark fin export permits in 2016 (Temple *et al.* 2018) and fishers report a subsequent decrease in the value of shark and wedgefish fins (various fishers, personal communication). Whilst the fin market still exists in spite of this (Temple personal observation) the financial incentive for fishers has decreased. These effects are likely bolstered by the decreasing demand for shark fin in East Asia (Dent and Clarke 2015). Similar market governance strategies may also prove effective in curbing the increasing demand for mobulid ray gill plates both globally and within the SWIO region. Such indirect effects on elasmobranch value have the potential to alter fisher behaviour and may help to decrease pressure on the resource.

Market governance has been increasingly popular as a means of changing supplier behaviour through profit incentives, rather than via penalisation of behaviours deemed negative (e.g. Stavins 2003). In isolation this form of approach has come under criticism for its potential to exacerbate poverty, especially in the absence of alternative livelihood initiatives, and effectively restrict access to a resource (e.g. Lawrence 2001), which clashes with the open-access nature of many small-scale fisheries. Yet, it must be acknowledged that, in the context of the developing and dispersed small-scale fisheries, traditional regulatory approaches are generally not feasible to enforce at the fisher level, primarily a result of lacking the resources to do so, and so supplementary market governance may be a viable alternative. Efficacy of any such measures to alter fisher behaviour in SSF would likely require regional-level co-ordination and co-operation. The migratory (Wanyonyi *et al.* 2016), open access and open-border nature of these fisheries, within which trans-location of catch is common (Temple, personal observation), mean such strategies enacted by nations in isolation would likely be undermined.

3.6 Conclusion

In accordance with the Sustainable Livelihood Framework this study begins to examine the dependence of fishers on elasmobranch resources and the broadscale socio-economic context of these fisheries, thus better informing the vulnerability context of these fishery-dependent livelihoods. The study indicates that fishers whom are most dependent on these vulnerable resources also display increased fisheries income and income-generating employment dependencies, likely compounding their vulnerability to social, economic and environmental shocks relative to other elements of SWIO SSF. Further, the findings suggest that improving infrastructure and access to external markets is linked to the demand for elasmobranch, particularly shark and shark-like ray, products and thus will likely continue to increase, at least in the short-term future. Though little is known of their current status, elasmobranch resources are showing initial signs of decline across the region (Kiszka and van der Elst 2015) and so fisheries managers must prioritise means to alter fisher behaviour in order to reduce pressure on these vulnerable taxa and so facilitate their long-term sustainability. In tandem managers must support the most dependent fisher households in appropriate diversification (within and out with fisheries) of their livelihood strategies. Thereby reducing their vulnerability to external shocks. In order to facilitate for the long-

term wellbeing of fishers, their households and communities, and sustainable use of elasmobranch resources.



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Rethinking marine resource governance for the United Nations Sustainable Development Goals

Selina Marguerite Stead



Governments are facing mounting pressure to 'do something on the United Nations Sustainable Development Goal (SDG) 14'. The SDG 14 comprises targets and indicators for countries to show progress in achieving conservation and sustainable use of oceans, seas and marine resources for sustainable development. One novel approach experiencing traction in improving growth performance of marine resource sectors — particularly marine biotechnology and fisheries — is open innovation. This review assesses the potential impacts of using open innovation approaches more widely with good governance principles to promote sustainable management of marine resource use. Recent research filled the gap in creating a quantitative marine open innovation measure to monitor and manage marine sector innovation process. Prior 2018, formulation of marine policies — aimed at building economic prosperity from sustainable use of marine environments — were constrained by the absence of a tool to provide an index to compare marine sectors' innovation performance. This review highlights a need to broaden the measures used to determine marine management effectiveness especially in the context of achieving the SDGs. Governments are urged to pay more attention to new governance tools including open innovation when formulating new policy aimed at building future scenarios of economic resilience involving marine resource use.

Address

School of Natural and Environmental Sciences, Newcastle University, Newcastle upon Tyne, NE1 7RU, UK

Corresponding author:

Stead, Selina Marguerite (Selina.stead@ncl.ac.uk)

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Introduction

Oceans comprise over 90% of earth's habitable space and endemic phyla. Evidence for the 6th mass extinction [1] influenced the United Nations to hold its first Ocean Conference in 2017 [2]. This was in response to documented declines in fauna and flora. Simultaneously, concern was rising about the extent of undocumented declines in the vast under sampled ocean and risks to the future health of our planet including species redistribution [3]. Reconciling use of oceans, seas and marine resources, with conservation goals and livelihood dependency, is at the heart of the United Nations Sustainable Development Goal (SDG) 14 — life below water [4].

In 2015, the United Nations Development Programme (UNDP) agreed to implement a global strategic plan comprising 17 SDGs. The aim was to bring governments, industry, researchers and non-government representatives together to agree priority actions for achieving the SDG's 169 targets. While a vast volume of literature is building on how governments can meet the targets outlined in the SDGs, considerable debate continues around implementing actions to balance the relationship between marine resource management effectiveness and governance efficacy. This is in part because of the difficulty in attributing social and economic impacts beyond proximate drivers [5] to specific marine environmental management interventions including marine protected areas (MPAs) [6].

Many Nations' main marine resource strategy for our seas and oceans has been to manage the natural environment separately to fisheries. This creates a lack of join-up between Ministerial departments tasked to protect the natural environment with those responsible for job creation and marine industry development. This can lead to a fragmented approach to marine management. Approaches are diverse, including top down management involving government-controlled MPAs [7] for conserving biodiversity to informal governance systems, such as territorial user rights for fisheries. This paper reviews efficacy in marine resource governance using fisheries and MPAs as case study themes, against a background of institutional and political challenges in meeting the SDGs. Future research in marine open innovation [8,9] is recommended to demonstrate wider benefits associated with effective engagement of governments with universities and industry, in co-development of marine policy that is fit to support marine management goals.

The author's practical experience as a Chief Scientific Advisor for the UK and overseas governments, is combined with current opinions in participatory governance reviewed from published literature. For the first time, the potential for adopting open innovation approaches to improve marine governance and accelerate meeting the targets for SDG 14 — life below water — is described [8,9]. The paper begins with an introduction and review on the concept of open innovation and its application to growing marine resource sectors. Fisheries is described as a traditional sector to illustrate the institutional and political challenges governments face in meeting the SDGs. It then explains the issues and difficulty in reconciling conservation goals using marine protected areas with sustainable fisher livelihoods and building marine industries. The implications of the review findings for policy makers, managers and researchers are set out in the conclusions.

The United Nations Sustainable Development Goals (SDGs) and open innovation

This section reviews contemporary literature in the context of rethinking how marine resource governance — using open innovation — can assist governments facing mounting pressure to 'do something on SDG 14'. In a world of nine billion people and a widening income gap between the rich and poor, many maritime nations are looking for new ways to develop strategies to meet the SDGs and improve the quality of people's lives. An emerging approach for developing marine knowledge-based economies is open innovation [8,9]. The concept of open innovation was introduced in 2003 [10], and for the first time, used in 2015 [8] to develop policy to enhance competitiveness in the global market of the marine biotechnology sector [9]. Open innovation can be defined as 'the use of purposive inflows and outflows of knowledge to accelerate internal innovation, and expand the markets for external use of innovation, respectively' [11]. In other words, open innovation is a strategy by which organizations allow knowledge to flow across their boundaries to enhance their capability to innovate [12].

A major challenge for governments is obtaining evidence-based decisions on which marine resource sectors should be prioritised for investment programmes to strengthen the national economy. The absence of a quantitative measure to monitor and manage marine sector innovation process has constrained formulation of marine policies aimed at building economic prosperity [8]. In 2018, a newly created measure of inbound open innovation was developed, validated and applied to the marine bio-industry sector of Oman [9]. This new metric received positive feedback from managers and policy advisers as a valuable tool in providing an index to compare performance of different marine sectors [9].

Open innovation is gaining momentum as a strategic tool to develop marine resources sectors which should, in time, help to diversify a country's economy [8]. This is especially important to developing countries that rely on traditionally low-tech marine industry dependent on vulnerable marine resources such as fisheries. Although fisheries can be important for domestic and export markets thus acting as a major source of foreign currency, the financial benefits of these exports is commonly low since they consist largely of fresh unprocessed fish with no added value [13]. However, introducing collaboration strategies such as open innovation at the local and national business levels, can offer beneficial opportunities for development. This can vary from traditional marine resource industries sourcing new seafood to marine sectors built on added-value products, for example, high-end niche-value aquaculture (e.g. sea cucumbers [14]), functional food, nutraceuticals, cosmetics and healthcare [15,16]. One of the success factors for the aquaculture industry is its collaboration with the marine biotechnology sector such as vaccine producing companies, fish-feed development firms, marine management operations and fish breeding units [17]. Karlsen *et al.* [18] explained that in Norway, collaboration between universities and vaccine producing marine biotechnology companies has radically reduced the incidence of disease in fish farming. Similarly, in New Zealand, effective linkages between fisheries companies and chemical companies led to the extraction of bioactive compounds from marine organisms for commercial use, a good example of which is Lyprinol, an anti-inflammatory extracted from the New Zealand Green-Lipped Mussel (*Perna canaliculus*) [16,19].

Development of the open innovation measurement tool in Oman [9] is helping the government to make informed decisions about policy support needed to drive investment strategies for more effective national economic growth through strengthening the marine biotechnology sector. This reflected the government's increasing awareness about marine organisms being more diverse, unique and complex in their biophysical, chemical and genetic characteristics than terrestrial organisms [20]. Recent scientific and technological developments in marine biology, has produced many new discoveries [21], leading to the emergence of new marine biotechnology sectors, all of which depend on open innovation for their successful development [15,16,18,20,22,23]. A key contemporary consideration for maritime countries when deciding which marine sectors should be selected for investment and growth, is do they have the potential to meet the targets of the SDGs? In order to realize these goals, governments need to provide the institutional and political environments that support open innovation in old and new marine sectors. In a turbulent and uncertain climate, innovation is considered a risky business [11,12] yet many nations believe open innovation approaches offer a way

forward in addressing the grand global challenges of population health, food and energy security [8,9]. While a vast volume of literature is building on how governments can realise the targets outlined in SDG2 (end hunger, achieve food security and improve nutrition); SDG3 (ensure healthy lives and promote well-being), and; SDG8 (promote sustained, inclusive and sustainable economic growth), considerable debate continues around marine resource management effectiveness and governance efficacy. This is in part because of the difficulty in attributing social and economic impacts to specific marine environmental management interventions including MPAs [7] and fisheries measures [24*], such as fishing bans [25].

Marine Protected Areas (MPAs)

One methodological aspect in marine conservation efforts that deserves more attention is the set-up of effective participation from a wide range of interest groups to understand how MPAs will impact the social characteristics of their marine resource activity and their coastal community more broadly [26**]. Traditionally, scholarly studies on MPAs focussed mostly on gaining an understanding about the environmental and economic variables that can affect the success of an MPA [6]. An increasing number of authors around the world are questioning what a successful MPA is in practice especially in relation to how MPAs impact those people mostly affected by its implementation [6,7,27,28]. Despite growing evidence to show engagement of natural resource users is linked with support for marine management [29**,30] participatory marine resource governance should be promoted better by governments to improve buy-in from those targeted [25,26**]. This is especially true in relation to policy development to support how best MPAs can achieve the targets of SDG 14. Thus I argue that the SDG 14 will not be achieved in practice by the 2030 deadline nor maintained in the longer term until governments rethink their approaches to marine resource governance. This review recommends governments explore the adoption of open innovation approaches [8] to help accelerate growth in traditional and new marine resource sectors [9] as a way to increase likelihood of meeting the targets set out in the SDG 14.

To improve the management of marine resources we need to better integrate social science survey data into marine policy by involving coastal communities through improved participatory governance [31]. Methods that take account of information beyond proximate drivers can provide a more detailed understanding of the local context to better inform the policy needs [5]. In particular, using governance methods to co-develop management decisions impacting future livelihoods, well-being and quality of life [25]. However, the relationship between data generated from the Global Ocean Observing System (GOOS), poor ocean monitoring and social

science evidence is unclear. Therefore, to fully evaluate the potential benefits of MPAs to wider society in the context of marine environmental carrying capacity, there needs to be greater involvement of social scientists in participatory governance forums to evaluate marine resource management outcomes using GOOS and future scenario predictive models. Advances in measuring peoples' perceived proximate drivers (e.g. fewer fish) to include linkages with ultimate drivers (e.g. poor governance leading to poaching) [5] of MPA efficacy over time, would benefit from a new module of GOOS to help tailor-specific management action required to tackle an identified issue. The Large Marine Ecosystems Governance Toolkit [32] launched in 2018, makes a start in bringing together experts to share lessons learned in developing an integrated, collaborative approach to ocean and coastal management.

The SDG target 14.5 which states, 'conserve at least 10 per cent of coastal and marine areas, consistent with national and international law and based on the best available scientific information' continues to influence scholarly debates on the pros and cons of MPAs [26**,33,34–37]. To progress the SDGs in coastal communities, governments need to improve multi-level governance to support marine resource management effectiveness. Why? Nations worldwide are prioritising management options, primarily the set-up of MPAs for oceans, without effective engagement of those impacted. This is especially at the initial design and evaluation stages. Countries weak in engaging people in marine resource management decision-making generally have poor coastal governance [30], which is linked to non-compliance of MPA regulations [7,27,28]. I argue governments need to better integrate scientific evidence on social indicators [38] in decision-making about MPA design and location. The global focus on increasing the number and size of MPAs to enhance environmental biodiversity is failing to fully capture important unintended social (e.g. poaching) and economic (e.g. loss of income) consequences. This can lead to 'marine paper parks' where regulations are not enforced and people break MPA rules [7,27,28]. The MPA amounts to no more than words on paper. Over time a lack of confidence in marine management is created with locals disengaging with the goals of the MPA. Few recent journal articles describe the social impacts of MPAs on coastal communities [39,40], however, growing evidence for linking MPA efficacy with effective governance mechanisms is emerging [6,41–44,45*,46,47].

Restricting access to marine resources which provide food or income without consideration of alternatives or transitional livelihood arrangements contributes to conflict over use of contested marine space [48,49]. Mitigating negative impacts of MPAs on people has received insufficient consideration given the proposed rate at which

governments are setting-up MPAs to meet the targets of SDG 14.5. This paper highlights the need and opportunity to improve participatory governance through effective engagement of local people in the decision-making process of marine management options such as MPAs [6,26^{••},27]. Failure to achieve participation of people in management decisions can build a growing divergence between those prioritising marine conservation goals and others prioritising, for example, food security resilience from the same marine resources. When MPAs impact negatively on sustainability of coastal livelihoods and in the absence of alternative ways to source food and generate cash, rule-breaking of MPAs is common [6,7,27]. The next section briefly reviews the role of Integrated Coastal Zone Management in promoting effective engagement of local people in the marine planning process which can mitigate rule-breaking [31,50].

Marine planning and integrated coastal zone management

Although Integrated Coastal Zone Management (ICZM) is a well-established concept its application to the rapid development of MPAs globally, which promotes horizontal (e.g. sectors) and vertical (e.g. local versus national) 'join-up' of decision-making on marine resource management matters, is arguably overlooked [51–53].

This is despite introduction of marine spatial planning (MSP) statutory approaches in some regions, for example, Europe, which is built on ICZM principles [50]. Part of the problem is that stakeholder participation exercises [54] and communication is frequently done as consultation which is not the same as effective engagement [29^{••}]. ICZM emphasises the benefits associated with effectively involving people in the marine management decision-making cycle. MPA supporters heralded MSP as a way to overcome the policy vacuum associated with earlier forms of ICZM albeit the latter placed a greater emphasis on implementing good governance processes using a principle-based approach [52] to underpin effective engagement. The advantage of using principles when developing good governance for deciding on the type of natural resource management, is the choice of principles can match values of local peoples' cultures. These in turn when translated to human behavioural traits can be used to tailor the context to help explain the importance of MPA rules to those affected by their introduction. With growing awareness of implications for gaps in social science data constraining governments in achieving the SDG 14 targets by 2030, policy makers are advised to better integrate socio-economic scientific evidence with marine environmental data to improve relevance to local contexts. This should be approached using participatory governance (SDG 16.7) forums relevant in engaging locals impacted by proposed introductions of MPAs.

A major constraint to governments realising their full potential in contributing positively to the SDGs in many parts of the world, is over reliance on incomplete environmental datasets. This 'patchiness' could be improved through combining local ecological knowledge [36] to fill in gaps where trends can be determined through existing datasets like GOOS. Comprehensive scientific information, especially longitudinal data on sea bed habitats is frequently limited or unavailable. Despite this shortfall in evidence, marine policy decisions on natural resource use remain biased towards using environmental evidence to influence who has preferential access to marine areas. This is undertaken more reactively than proactively, that is, an environmental impact assessment or similar is done as part of the decision-making as to whether a licence or access will be granted for a particular use or activity of a marine area. What is often lacking is robust and systematically collected evidence on the health and status of the marine environment over spatial and temporal scales where natural versus anthropogenic impacts can be considered well before any plans to disrupt marine areas occurs. Without confidence in the historical trend analysis for likely impacts to the marine ecosystem, which terrestrial ecosystems more commonly have information, then predicting the pros and cons of an MPA can be limited in terms of future forecasting and future scenario planning. Some countries like the Seychelles are thus looking for new ways to inform their decision making on priority use of marine resources to underpin national economic prosperity and have embraced the concept of the Blue Economy, the subject of the next section.

Blue growth economy and fisheries

The development of the Blue Growth Strategy [55,56] has led to many governments approaching the SDG 14 by identifying initiatives aimed at increasing economic prosperity using the contested 'blue economy' concept [56]. This is especially in relation to reducing dependence on fisheries [57,58]. To illustrate, the Seychelles has adopted Blue Bonds to generate financial support — including ca. US\$20million from the World Bank — to preserve its natural capital, by expanding its offshore MPAs and improving fisheries governance [59]. The author was tasked to lead a team of international scientists in 2015 to advise the Seychelles Government about the risks associated with introducing a marine aquaculture sector in connection with its Blue Economy Strategy. The rationale being a way for the Seychelles to diversify sourcing protein away from heavy dependence on its marine capture fisheries sector. This was aimed at improving marine food security through development of a new aquaculture sector, thus reducing vulnerability to over exploited fish stocks and decreasing reliance on its fisheries export trade for income. Lessons learnt from traditional fisheries management approaches in prioritizing economic objectives (e.g. species value) over environmental dimensions (e.g. stock size) are better balanced

with consideration of social impacts in contemporary fisheries governance [25,31].

Governments have used MPAs in their various guises, especially as tools to enhance biodiversity or as fishing bans [6]. When fishing is allowed in an MPA and data is available, fishing effort control measures such as catch limits are commonly based on the maximum sustainable yield (MSY). This is used as a reference point for advising on the amount of fishing allowed (quotas) and maintaining healthy stock levels (avoiding over fishing). In data poor fishery-dependent countries such as those in the tropics, MPAs are sometimes used as a banning measure to ensure zero catch or to restrict access to marine fishery resources [27]. The science base underpinning decisions about determining MSY-effort (where catches peak), rely mostly on ecological indicators (e.g. abundance, distribution, life history strategy and stock size). The tropics are characterised by mixed-fisheries and limited technical capacity to collect robust scientific fishery evidence [13]. Hence the popularity of MPAs in tackling declining fishery resources is often seen as an easier solution to manage fishers, that is, restrict access rather than rely on comprehensive ecological datasets to determine catch quotas. More recently, governments are turning to the economic value of marine resources to inform management decisions. Nonetheless, lessons from the fisheries sector [60] should be considered in terms of SDG target 14.5 in whether achieving this can make a difference to ocean health and peoples' associated livelihoods.

Economic fishery data has traditionally been used to show the value of a particular marine resource, for example, the Spiny lobster (*Panulirus argus*) fishery in the Turks and Caicos Islands was valued at 10% of the GDP [61]. Despite efforts to apply Maximum Economic Yield (MEY) models over MSY, Christensen [62] showed MEY to be misleading. This was due to MEY not capturing wider societal benefits beyond biological (e.g. source of food) and economic (e.g. income) aspects of a fishery. Christensen [62] concluded MEY excludes the governance culture in which decisions are made locally about who has the right to fish what, when and where. More recent research claims bioeconomic models when combined with multiple management approaches can help policy makers to better understand trade-offs between food provision, fishery profits and fish biomass in the sea [24*,63,64,65*]. Understanding marine resource governance mechanisms in diverse contexts [66–68] and at different local, national and international levels (SDG 16.7) will facilitate integration of scientific bioeconomic evidence with societal priorities [69*]. In some coastal fishing nations, informal governance mechanisms prevail, for example, qoli-qoli systems in Fiji [70] and prefecture systems in Japan [71].

Conclusions and future directions

Against the popularity of growing MPAs worldwide [6,72] as the preferred approach to meeting the targets of SDG

14, this review advises caution. Governments are recommended to evaluate the lessons learned from the marine fisheries sector. In particular, this paper highlights the limitations in focussing on environmental and economic goals in designing marine policy to develop sustainable management options and recommends wider consideration of open innovation as a way to enhance growth performance of old and new marine resource sectors [8,9]. Although 94% of MPAs allow fishing [73], in scenarios where fishing is not compatible with conservation goals fishing is commonly prohibited, referred to as No Take Zones. However, as reported by Christensen [62], fishers and their associated coastal communities may seek wider benefits from marine resource use, for example, food security, employment, local control and the right to continue traditional cultural activities. Thus in developing MPAs it is important governments collect robust scientific data on society's interests as a whole that go beyond bio-economic models [74]. Furthermore, if fishing bans are being implemented there needs to be transitional arrangements put in place before this happens to allow those impacted time to adjust. Failure to consider alternative or supplementary livelihood options can lead to poverty (SDG 1) and hunger (SDG 2) in vulnerable coastal communities. If MPAs are to contribute positively to the SDG 14 targets, governments are encouraged to be creative about involving community members including fishers and/or those likely to be impacted in their design [66,75–77]. This can best be achieved through employing responsive, inclusive, participatory and representative decision-making at all levels (local, national and international) — SDG target 16.7. The digital revolution offers a variety of new ways to improve understanding of the relationship between participation and effective engagement of stakeholders to improve marine resource governance. Importantly governance mechanisms designed to foster engagement of locals must take account of local decision-making culture which can be more informal (e.g. codes of conduct, self-regulation) than formal (regulations).

The relationship between efficacy of MPAs and sustainable fisheries is equivocal in many parts of the world despite the huge investment made by some countries in starting MPAs [78]. A greater understanding of the links between locals' perceptions and compliance for MPA rules such as fishing bans would go some way to addressing how effective MPAs are as a fisheries or broader conservation tool [7,27,79]. The inclusion of social science data will help to enhance knowledge about the marine resource management options likely to be supported by those targeted. Effective multi-level governance (SDG 16.7) is key to achieving the targets outlined in SDG 14. Participatory governance processes that promote meaningful engagement of locals [80] with decision-makers [29*,30] should be applied more widely in future marine resource management scenario planning.

Combining this approach with data on proximate and ultimate drivers underpinning human behaviour will help build context-specific management measures to achieve the targets outlined in SDG 14 [5] in relevant cultural contexts.

Perhaps the biggest constraint to achieving the full suite of benefits that could arise from translating SDG 14 theory into practice is a lack of interdisciplinary [81] marine governance expertise on SDG processes. This review recommends financial capacity is increased for training and capacity needs of developing tomorrow's ocean leaders. Rethinking marine resource governance approaches including the wider use of open innovation to tackle the SDGs, especially number 14, will require an interdisciplinary licence to be bold and creative. This could be the decade scientists work more closely with governments and industry to co-implement marine sustainable management with those impacted being at the heart of decision-making so there are enough marine resources for all, forever.

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Annex 5: Yussuf Salmin MSc

IMPROVING BYCATCH MITIGATION MEASURES FOR MARINE MEGafauna IN ZANZIBAR, TANZANIA

AUTHORS:

Yussuf N. Salmin¹, Narriman Jiddawi², Tim Gray³, Andrew J. Temple^{4a}, Selina Stead^{4b}

1: State University of Zanzibar, Vuga Road, P.O.Box 146, Zanzibar, United Republic of Tanzania. Email: yussufsalmin@gmail.com. Tel: +255772679967 (**Corresponding Author**)

2: Institute of Marine Sciences, University of Dar es Salaam, P.O.Box 668 Zanzibar, United Republic of Tanzania. Email: jiddawi@ims.udsm.ac.tz. Tel: +255777423183

3: School of Geography, Politics and Sociology, Newcastle University, Newcastle-upon-Tyne, NE1 7RU United Kingdom. Email: tim.gray@ncl.ac.uk. Tel: +44 (0) 191 208 7528

4a: School of Natural and Environmental Sciences, Newcastle University, Newcastle-upon-Tyne, NE1 7RU, United Kingdom. Andrew.temple@newcastle.ac.uk.

4b: School of Natural and Environmental Sciences, Newcastle University, Newcastle-upon-Tyne, NE1 7RU, United Kingdom. Selina.stead@ncl.ac.uk. Tel: +44 (0) 191 208 6832

ABSTRACT

This study was conducted to explore the governance processes and socio-economic factors relevant to the potential implementation of bycatch mitigation for various vulnerable marine megafauna (batoids, sharks, marine mammals and turtles) in Zanzibar, Tanzania. Questionnaire-based interviews were conducted between February and April 2017 with fishers (n= 240) at eight landing sites. One focus group discussion was held in each site and eleven key informant interviews were carried out. The study showed that current measures to manage bycatch rates are not explicit; no rules govern batoids and sharks bycatch; and rules regarding marine mammal and sea turtle bycatch are poorly enforced. Binary logistic regression was used to determine the effect of five selected socio-economic factors (education, age, proportional fishing income, fishing experience, and the number of adults who bring income into the household) on the willingness of fishers to participate in potential future bycatch mitigation measures for marine megafauna. The results indicate that only one factor (the number of adults who bring income into the household) had any significant effect ($p= 0.016$). These findings could benefit the future governance and management of marine megafauna in Zanzibar through a better understanding of what mitigation measures are more likely to be supported.

Key words: Bycatch, Governance, megafauna, mitigation measures, socio-economic

1. INTRODUCTION

Marine megafauna play major roles in ecosystem structure and function (Bowen, 1997). Their status as apex and meso-predators and as mega-grazers mean they directly influence community structure, community dynamics and nutrient cycling (Preen, 1995; Aragones, *et al.*, 2006; Heithaus, *et al.*, 2008). Therefore, threats to the survival of these species have potentially wide-ranging consequences for marine ecosystems and those who rely upon them. In the past, loss of their natural habitats contributed to considerable mortality (Pusineri & Quillard, 2008). However, currently, fisheries are the greatest anthropogenic threat to these species at the global level (Lewison, *et al.*, 2004; Kiszka *et al.*, 2009; Riskas, *et al.*, 2016), where they may present as both targeted catch and bycatch. Due to persistent growth in human activities, increased interactions between megafauna and humans have contributed to injuries, damage and finally death to megafauna (Capietto *et al.*, 2014). Thus, in order to preserve these species, the ecosystems they affect, and the people who rely upon the marine environment, fisheries bycatch requires immediate action (Reeves *et al.*, 2013).

Bycatch in artisanal fisheries receives little attention from either local or global fisheries authorities (Moore *et al.*, 2010). Although artisanal fishers use simple gears for fishing, some types of gears contribute seriously to bycatch problems for marine megafauna (Adimey *et al.*, 2014). A growing number of researchers believe that marine megafauna bycatch in these small-scale fisheries might be as extensive or even greater than in large-scale commercial fisheries (Alfaro-Shigueto *et al.*, 2011; López-Barrera *et al.*, 2012; Mancini *et al.*, 2012). In the South Western Indian Ocean, where artisanal fisheries employ over 495,000 people, bycatch is reported to take place all over the region (Temple *et al.*, 2017). Kiszka (2012) found that 31 species of marine megafauna were regarded as bycatch by artisanal fishers in Zanzibar: five species of sea turtles, five species of marine mammals, and 21 species of elasmobranchs.

Despite the fact that several studies have been conducted in the past that contributed to our knowledge of the bycatch problem in industrial fisheries (Komoroske & Lewison, 2015), the bycatch problem in artisanal fisheries remains largely ignored (Curtis *et al.*, 2015). This is partly because there is insufficient information about bycatch in the artisanal fisheries of Africa since most megafauna species at risk of incidental catch live in the open sea far from public eyes, and this makes examination of their bycatch situation difficult and costly (Read, 2008). But it is also because it is hard to work out the best way of achieving bycatch mitigation in artisanal fisheries since it is multifaceted, involving cultural, economic, social and environmental issues which all need to be considered in any management plans. This complexity is reflected in the growing recognition of the role of social and economic research approaches in facilitating the implementation of mitigation plans (Komoroske & Lewison, 2015). Social and economic factors can affect the chances of bycatch alleviation measures being effective in mitigating bycatch, because fishers' dependence on a fishery will influence how likely they will follow rules which may reduce their social and economic well-being (Peterson & Stead, 2011; Teh, *et al.*, 2015). Knowledge of socio-economic factors such as the numbers of people in certain areas, their beliefs, and their age can contribute to our understanding of how fishers can impact the sustainability of the megafauna populations (Stead *et al.*, 2006; Brewer *et al.*, 2012). Adequate understandings of social and economic features of fisher's communities are also essential requirements for good governance (Kittinger, 2013; Turner *et al.*, 2014).

Good governance and appropriate management are acutely relevant to the bycatch problem. Government intervention is needed to assist widespread bycatch reduction, whether through coercion or incentives, and so understanding fishers' perceptions of current governance processes and their effects on fisher behaviour is vital (Eriksson *et al.*, 2015; Turner *et al.*, 2017). The term 'governance' is a more comprehensive term than 'management', and it goes further than imposing controls or creating opportunities (Chuenpagdee & Sumaila, 2010). Good governance entails having accountability, participation, predictability, transparency, the rule of law and strong institutions (Lockwood *et al.*, 2010; Turner *et al.*, 2014). In order to reduce bycatch problem in the Western Indian Ocean region all of these characteristics of good governance are needed in the fisheries sector. Good governance can also help sustainable natural resource management by securing the availability of food, strengthening the rural economy, safeguarding the marine sustainable ecology, and promoting alternative livelihoods (Finkbeiner & Basurto, 2015).

The aim of this study is to identify governance processes within a socio-economic context that may hinder, or contribute to, the introduction and widespread use of bycatch mitigation methods. The outputs of the research are intended to include recommendations about how to mitigate bycatch through a better understanding of the human dimension of fishing.

2. MATERIALS AND METHODS

The Zanzibar archipelago is part of the United Republic of Tanzania, consisting of many small islands and two large ones, Unguja and Pemba. Like many other African nations, Zanzibar is considered as (part of) a developing state. It has a GDP of \$ 675 million (Murphy *et al.*, 2016) and a total population of 1,303,569 (Population and Housing Census, 2012). This study was conducted in Unguja Island which is located at 6° 13'S and 39°13' E, situated approximately 40 miles off the coast of mainland Tanzania. Nearly 70% of Zanzibar's population is found in Unguja Island. In this study, data were collected from eight fisheries landing sites (See Figure 1). These sites were chosen on the basis of geographic spread, fishing gear composition (with a bias toward sites with high numbers of long line, drift nets and bottom-set gillnets) and logistical constraints. Data collection took place between February and April 2017. A mixed-methods approach was used to obtain qualitative and quantitative information from different stakeholders. This approach was taken so as to reduce the weakness of mono-method research (Place & Kelle, 2008), and allow for triangulation of information (reinforcement of findings).

(PUT FIGURE 1 HERE)

Face-to-face structured questionnaires were administered in a survey of 240 fishers to collect data on (i) Socio-economic factors comprising education (years spent in school), age, proportion of income from fishing to household, number of adults who bring income to the house, and fishing experience with gears (ii) Willingness of fishers to participate in potential future bycatch mitigation, (iii) Perceptions of current management in relation to principles of good governance, and (iv) Appropriate persons/organisations to involve when making decisions on marine megafauna bycatch management. Simple random sampling was used to select fishers, and survey questionnaires were administered at landing sites when fishers returned from fishing trips, repaired their fishing gears, or relaxed at landing sites, or at their homes. Interviews were conducted face-to-face. Fishers were asked for their consent before the interview was conducted,

anonymity was assured, fishers were free to choose not to answer any questions that they did not feel comfortable with and could end the interview at any time.

Focus group discussions (FGD) were conducted in each village where the moderator led different stakeholders such as leaders of the villages, fishers and members of shehia fisheries committees. Each group contained six participants who were selected on the basis of their expert knowledge, their fisheries experience and the length of time they had lived in the area, thus taking account of historical context. Information obtained from the fishers included fishers' perceptions of catching marine megafauna, current laws regarding marine megafauna, their enforcement, and ways to conserve marine megafauna. The discussions were tape-recorded with the permission of the participants. Charlesworth & Rodwell (1997) suggested that FGDs should be comparatively small in size not less than five and not more than eight participants to give them more time to discuss their views, experiences, and enable moderators to manage active discussions better than with larger groups.

Eleven key informant (KI) interviews were carried out, comprising one stakeholder from each study site and three from the fisheries department (a lawyer, a fisheries officer, and the Manager of Menai Bay conservation area). These participants were selected for their knowledge, role in the setting, and willingness and ability to provide useful information on the topic. These KI interviews were conducted to obtain a more synoptic perspective on the marine megafauna bycatch problem. The interviews were tape-recorded with the consent of the participants.

Quantitative data from the survey questionnaire returns were analysed by using statistical software SPSS version 20 wherein binary logistic regression was employed to assess the effect of socio-economic factors on the willingness of fishers to implement bycatch mitigation measures of marine megafauna. The socio-economic factors of the level of education, age, proportion of income from fishing to household, number of adults who bring income to the house, and fishing experience with the gears were taken as independent variables. A significance level (α) of 0.05 was used. Content analysis was employed to analyse qualitative data from focus group discussions and key informant interviews, where opinions recorded were listened to carefully, coded and interpreted to provide meaningful data which are presented below in the form of tables.

3. RESULTS

3.1 Governance of marine megafauna bycatch

3.1.1 Perceptions of fishers about governance principles:

From the survey questionnaire returns, the results showed that trust is the most important governance principle for effective decision-making on bycatch and fisheries issues since it was ranked number one by most fishers (32%), followed by accountability and then effectiveness as shown in Figure 2.

(PUT FIGURE 2 HERE)

3.1.2 Perceptions of fishers about appropriate persons/organisations for making decisions on marine megafauna's bycatch management:

In the survey, 38% of respondents (n=240) perceived that Shehia fisheries committees are the most suitable organisations for making decisions on management of marine megafauna bycatch in Zanzibar, followed by the fisheries department and leaders of the villages, both of which were rated as the most suitable by 14.6% of respondents (See Figure 3). These Shehia fisheries committees are found in each village.

(PUT FIGURE 3 HERE)

3.1.3 Rules for catching marine megafauna in Zanzibar:

There are rules that forbid catching, landing or using products such as eggs and shells of some marine megafauna in Zanzibar like sea turtles, whales and dolphins, and if these marine megafauna are caught accidentally in the gears they must be released. These rules are set by fisheries department in collaboration with shehia fisheries committees. However, there are no such rules for elasmobranch species (batoids and sharks) in Zanzibar. Nevertheless, results from FGDs showed that awareness of fishers about these rules is high due to the considerable efforts made by governments to educate fishers about the rules, but the level of enforcement is very low for reasons explained in Table 1.

(PUT TABLE 1 HERE)

3.2 Socio-economics of marine megafauna

3.2.1 Perception of fishers about marine megafauna and their mitigation measures:

Results from FGDs revealed that most fishers believe that catching dolphins, whales and sea turtles is wrong because they are more valuable if kept alive in the ocean since they are important for tourism. Moreover, FGDs showed that most fishers consider them as bycatch because targeting them is illegal, whereas some did not consider them as bycatch and targeted them for food and bait. However, elasmobranchs are not considered as bycatch by most fishers and they believe is not a bad thing to catch them since they provide marketable products such as fins, teeth, meat and livers for anti-fouling paint on boats and also it is still legal to catch them.

The majority of fishers surveyed in face-to-face interview (84%) perceived that implementing mitigation measures would not affect their livelihood and they were willing to implement those mitigation measures, while a small minority (16%) perceived that implementing mitigation measure would affect their livelihood by reducing their catch and therefore they were not willing to comply with them.

3.2.2 Effects of socio-economic factors on the willingness of fishers to implement mitigation measures:

Education: From the survey 43.4% of all fishers interviewed had reached ordinary secondary school which is about 10 years of school, but only 0.4% had reached higher education level (university). Statistical results revealed that education levels had no significant effect on fishers' willingness to implement mitigation measure ($p>0.05$) (Table 2).

Age: 45.5% of all fishers surveyed were aged within the range of 41 to 63 years, while 8.7% were aged above 63 years. The statistical results showed that the age of the fishers had no significant effect on their willingness to implement mitigation measure ($p>0.05$) (Table 2).

Proportion of household income from fishing: Fishing activity was the main source of income in most households: 47.5% of all fishers surveyed said that fishing activities contributed 81-100% of household income; 35% of the fishers said fishing activity contributed between 61-80% of the household income, and 17.5% of fishers said fishing activities contributed 40-60% to their household income. Statistical results indicated that the proportion of household income from fishing had no significant effect on the willingness of fishers to implement mitigation measures ($p>0.05$).

Experience of fishers with main fishing gear: On experience with the main fishing gears, 37.9% of fishers said that they had experience of between 1-10 years, while 0.8% of interviewed fishers said they had experience of greater than 60 years. This factor also had no significant effect on the willingness of fishers to implement mitigation measures ($p>0.05$).

Number of adults who bring income into the household: The average number of adults bringing income into a household is two people, and the survey showed that most households (87.9%) have 1-3 of adults who contributed to the income of the household; while 0.5% of respondents said they had more than six people who bring income in their households. The numbers of adults bringing income to the household had a statistically significant effect on fishers' willingness to implement mitigation measure ($p<0.05$).

(PUT TABLE 2 HERE)

4. DISCUSSION

4.1 Governance of marine megafauna bycatch

This study shows that management actions to reduce bycatch of marine megafauna in Zanzibar are ineffective. There are no laws governing either catch or bycatch of elasmobranchs, and while laws do exist for marine mammals and sea turtles, they are poorly enforced. Fishers know about the laws that are in place, a result of improved ways used by the government to educate fishers, yet rule-breaking by catching illegal species appears to be common despite their relatively low market value. On the lack of rules on elasmobranch species, the results found that there are no rules about them in small-scale fisheries, and fishers target them for their meat and fins. However, fishers said that the price they obtained for sharks fins had fallen dramatically since the shark fins trade (including exportation) was prohibited in Zanzibar. These findings support the observation of Temple *et al.*, (2017) when they reported the government cancelled the sharks fins export licence in Zanzibar.

On understanding why there is poor enforcement of the rules governing other marine megafauna, there were three main reasons given by those surveyed. First, there were insufficient resources for enforcing the current laws. Several studies show that lack of human resources, fewer patrol trips and less investment in equipment like boats, trigger rule-breaking events and undermine the effectiveness of conservation law enforcement (Ehler, 2003; Gilman, 2011; Peterson & Stead,

2011 Gilman, *et al.*, 2014). Second, there was a lack of trust in the people who are responsible for governing and managing fisheries activities. For example, respondents claimed rule enforcers like fisheries officers who carried out patrols, took bribes from rule-breakers. This meant that fishers stopped reporting rule-breaking activities, since when they did report, no action was seen to be taken. Smith & Walpole (2005) in their study indicated that corruption can seriously reduce the efficiency of conservation measures and lead to over-exploitation of vulnerable species. Third, there was no system to make government officers accountable for their actions (or inactions). For example, there were inadequate mechanisms for supervising and monitoring the work of the rule enforcers. Lockwood *et al.*, (2010) consider accountability to be the crucial governance principle for effective conservation of natural resources.

Results of the perceptions of fishers about appropriate persons or organisations for decision-making on fisheries management as presented in Figure 3 show that 38% of the fishers surveyed perceived that Shehia fisheries committees are the appropriate organisations for decision-making on fisheries issues including mitigating megafauna bycatch. The reason behind this perception is that Shehia fisheries committees involve local stakeholders, including ten fishers from the village, the leader of the village, and the beach recorder who represents the fisheries department. Carlsson & Berkes (2005) in their study found that the cooperative approach is the best governance approach in decision making of common pool resources since it reduces the marginalization of many stakeholders, empowers them, enables them to share their knowledge, and facilitates their sense of collective strength through unity. As for the 14.6% of fishers who perceived the fisheries department is the most appropriate organisation to make decisions on fisheries management, their main reason was that the fisheries department is responsible for all fisheries activity in the country and they have resources for implementing their decisions.

4.2 Socio-economic considerations of marine megafauna bycatch mitigation

In the past, fishers targeted dolphins and used them as bait for sharks. However, nowadays, most fishers perceived that dolphins, whales and turtles are less valuable to them as meat than kept alive as tourism attractions. Although 47.8% of interviewed fishers depend on fishing for 81-100% of their household income, most young fishers in coastal villages like Kizimkazi Mkunguni, Kizimkazi Dimbani and Nungwi are also involved with marine ecotourism which is a lucrative source of income and has led to a reduction in the number of fishers targeting these marine megafauna species in those villages. However, this kind of tourism activity itself has some negative impacts on marine megafauna because of disturbance from boats and swimmers, so it needs to be well managed as recommended by Stensland & Berggren (2007) and Christiansen *et al.* (2010).

The majority of fishers (84%) perceived that implementing mitigation measures will not affect their income, while only 16% perceived that implementing mitigation measures will have a negative impact on their income. The latter believed that such measures will cause catch reductions not only of marine megafauna species but also of other marine species, hence reducing their income. These results are in line with the findings of Bennett & Dearden (2014), who reported that some communities had negative perceptions about conservation measures since they believed such measures would harm their livelihoods, therefore they did not provide any support for them. Fishers who had a positive perception of mitigation measures on their

livelihoods stated that these marine megafauna have less value to them when they catch them compared to other species, so mitigation measures will not reduce their income since they will catch other more valuable species. Fishers from Kizimkazi Dimbani, Kizimkazi Mkunguni and Unguja Ukuu stated that when they did pilot trials of pingers to avoid catching marine mammals they did not experience any reduction in their catches of their target species so the results encouraged most fishers to be willing to implement the measures.

Results from binary logistic regression indicated that the willingness of fishers was significantly affected by the number of adults who bring income to the household. Households with a higher number of adults who bring income were more willing to implement mitigation measures. The magnitude of effect of this socio-economic factor was higher than that of other socio-economic factors studied (Table 2). For an additional one adult bringing income in the household, the odds of willingness rose by 1.5. The assumption is that households with more adults who bring in income have a higher income compared to those with fewer adults, and Liobikiene & Juknys (2016) found that income levels have a positive influence on environmental concern. In support of this finding, the 'social class hypothesis' proposed by Liere & Dunlap (1980) argues that the households with higher incomes are more concerned about the environment since they already satisfy their basic needs unlike households with less income who will do whatever they can to get their basic needs satisfied even by activities which destroy their environment.

Another finding from this study is that all the socio-economic factors studied had positive coefficients except education which had a negative coefficient (Table 2), for every additional year fishers spent in school the odds of willingness to implement mitigation measures is lowered. This implies that fishers who had a low level of education were more willing to implement mitigation measures than those with higher education. This finding is contrary to the conclusion of Liobikiene & Juknys (2016) who found that concern for the environment increases with the number of years that people spent in school. On their hypothesis, education has a major contribution in making people understand environmental issues, and therefore increases their awareness and encourages in them a greater sense of environmental responsibility. However this is not always the case (Kollmuss & Agyeman, 2002). In this study, fishers with low levels of education explained that employment opportunities outside fishing are lower for them compared to those with higher education thus they are willing to implement mitigation measures to sustain their jobs.

In conclusion, marine megafauna are ecologically, socially and economically important for most coastal communities. However, the population size of marine megafauna is falling sharply because of unprecedented depletion rates by bycatch globally. In order to reduce or to eliminate this decline, rules and regulations for catching elasmobranch species should be established and those for other megafauna species should be strictly enforced. Furthermore, fishers must be encouraged to implement bycatch mitigation measures, and to achieve this encouragement we need to understand socio-economic factors that influence fishers' willingness to collaborate with the authorities in introducing regulations and to then to comply with those regulations. The present study can form a basis for understanding these socio-economic factors and the educational processes needed to encourage the willingness of fishers, but further studies are

needed to understand the institutional governance of bycatch and to find alternative livelihoods in order to reduce pressure on the marine resources.

5. ACKNOWLEDGEMENTS

We would like to acknowledge beach recorders, fishers and leaders of the villages of all eight landing sites for their generosity and hospitality during data collection: without their support this study would not have been successful. Also our gratitude goes to WIOMSA through the BYCAM project for their financial support to conduct this study.

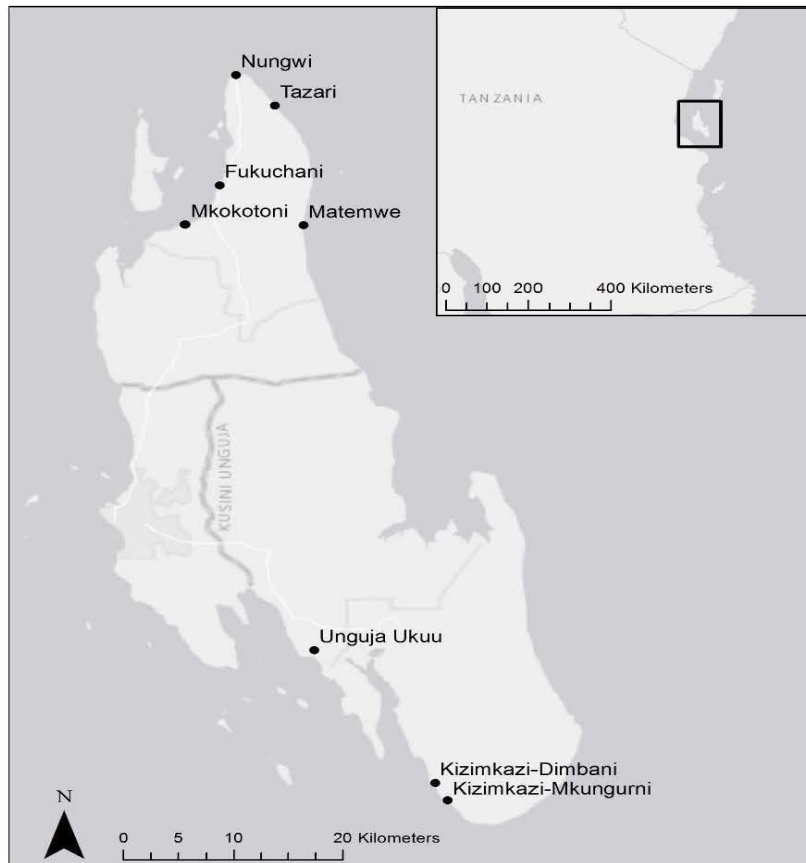


Figure 1: Map of Unguja Island highlighting the study areas.

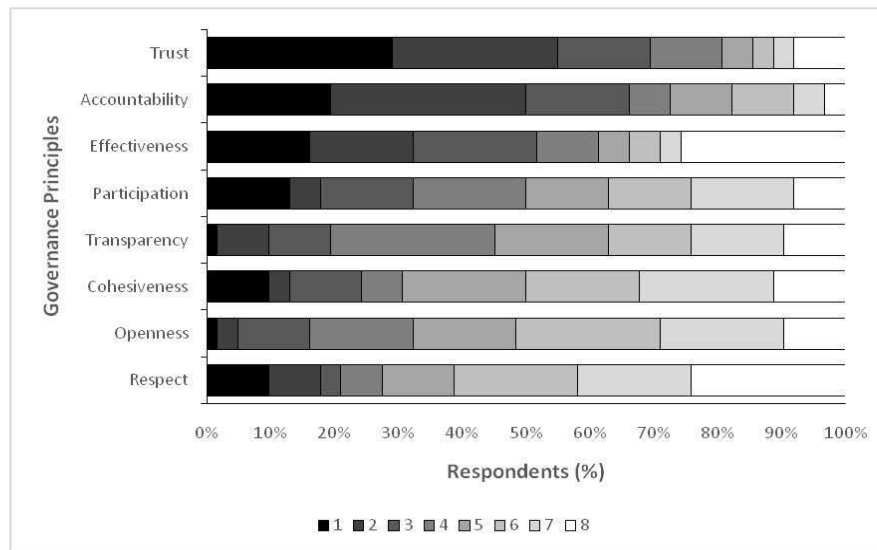


Figure 2: Perceptions of fishers about governance principles

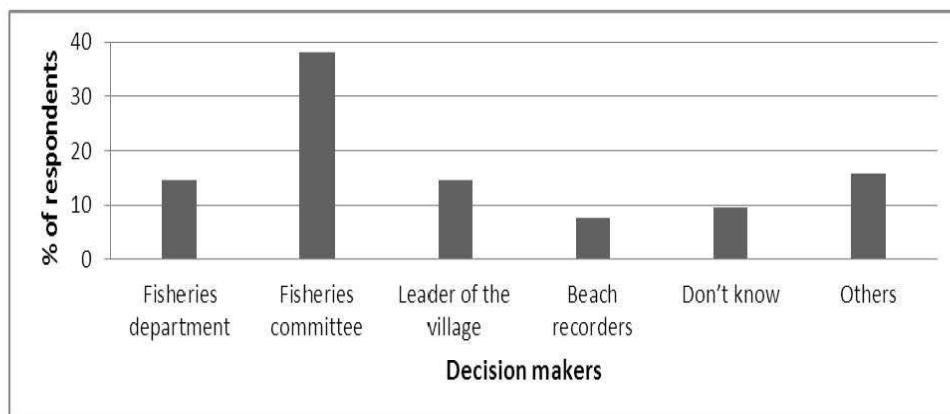


Figure 3: Perceptions of fishers about appropriate persons/organisations for making decisions on management of marine megafauna's bycatch

Table 1: Reasons stated by stakeholders for low enforcement of rules for and ways of improving conservation of marine megafauna

Reasons for low level of rules enforcement	Ways of improving conservation of marine megafauna	Ways used to educate fishers about the rules
In adequate resources for rule enforcement, for example, there are few patrol boats and insufficient fuel for them	More patrols and stricter enforcement of the rules	Through fisheries officers
corruption between rule enforcers and rule - breakers	Establish marine protected areas to reduce fishing in biodiversity hotspots	Through Shehia fisheries committee
Rule enforcers often come from the same villages and even the same families as the rule-breakers	Accountability of managers and rule enforcers	Through mass media such as television, radio and newspapers
There is a poor system for supervising and monitoring the work of the rule enforcers	Improve environmental awareness of the importance of marine megafauna	
Fishers are skilled at concealing their illegal activities by hiding when they catch dolphin and sea turtle.	Require fishers to use more selective fishing gears to avoid unwanted bycatch	
	Suggest fishers move out to deeper water where there is less marine megafauna	
	More cooperation between management and fishers, making better use of fishers' knowledge	

Table 2: Binary logistic regression analysis on socio-economic factors effect willingness of fishers to implement mitigation measures

Socio-economic factors	Coefficient (β)	SE	Exponent of (β)	<i>p</i> -value
Age	0.011	0.014	1.011	0.438
Education	-0.071	0.046	0.931	0.120
Income proportion from fishing	0.017	0.013	1.017	0.215
Adults bring income to the household	0.419	0.174	1.520	0.016
Experience with fishing gear	0.007	0.016	1.007	0.657



Overview of the project

The BYCAM project, a four-year initiative funded by WIOMSA, commenced in 2015 and brings together a unique multi-disciplinary team of 16 researchers from 7 countries.

The project has provided scientific evidence to address questions about how changes will affect communities' dependent on fisheries for livelihoods.

People in the WIO rely heavily on fisheries, which contribute to food security, income and ocean health.

Local people in the WIO have reflected on their experience to assess how to better manage and govern fisheries sustainably.

KENYA NATIONAL FISHERIES POLICY MEETING

2nd August 2018, KEMFRI Mombasa, Kenya

NATIONAL BYCAM MEETING

The **objectives** were to:

1. Identify key changes in fisheries with respect to political, economic, social, technological, legal and environmental systems;
2. Examine how these changes impact people, and;
3. Discuss who is involved in fisheries governance and management, both formally (regulations) and informally.

Methods

1. Historical timeline to map changes, and;
2. Policy cycle review to show how organisations are involved in fisheries.

Contents

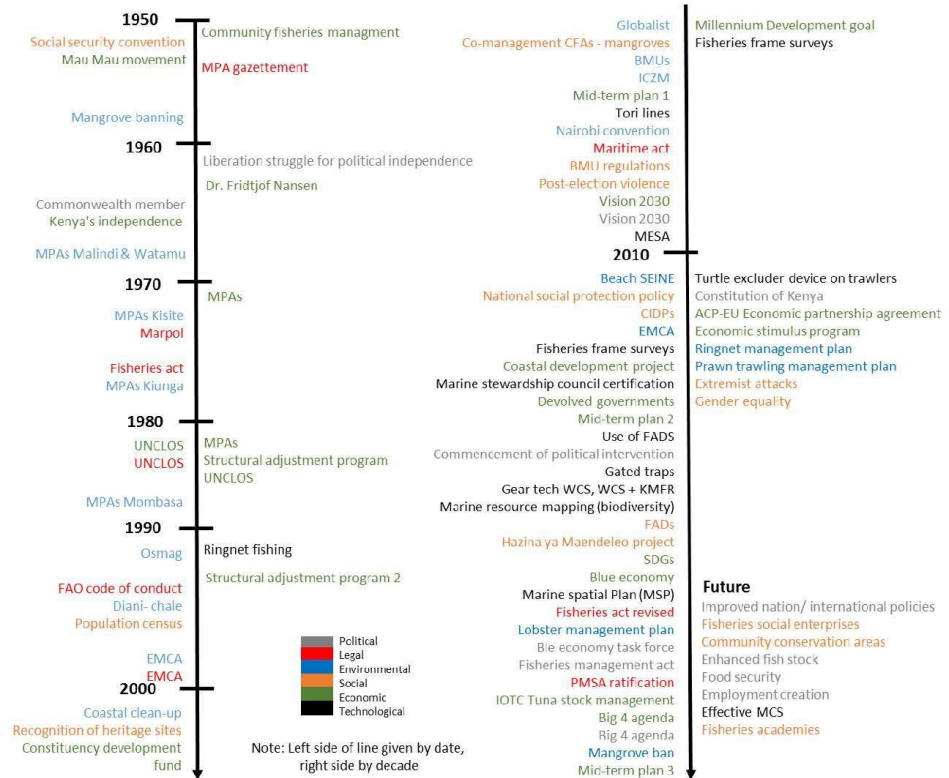
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Participants	6

Professor Selina Stead explaining fisheries policy options



Historical scan

Results of the historical timeline exercise showing key events and changes affecting fisheries in Kenya identified by participants based on their collective experiences



Acronyms: PLEASE COMPLETE BY ADDING ACRONYM NAMES FROM POLICY CYCLE BELOW

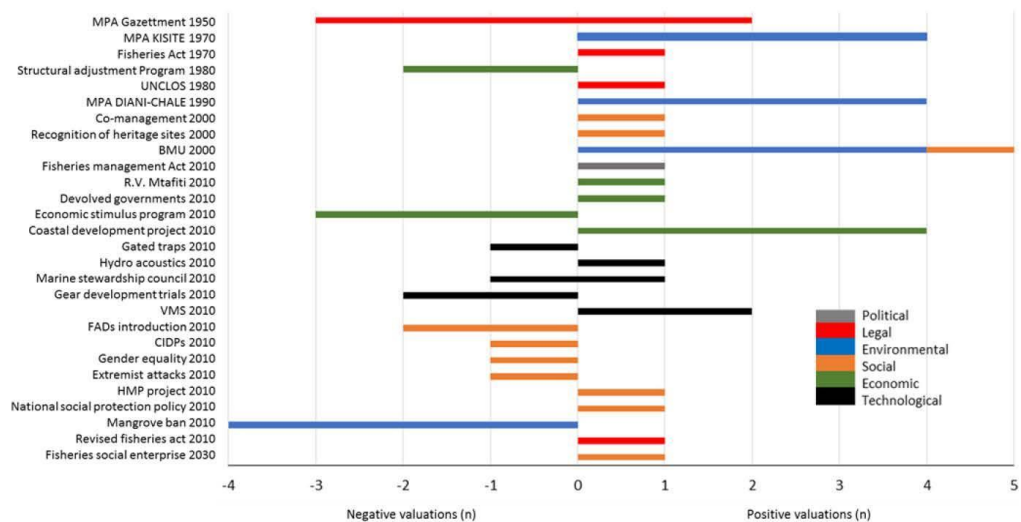
MPA
UNCLOS
FAO
EMCA
BMUs
ICZM
MESA
CIDPs
FADS
WCS
KMFR
SDGs
MSP
PMSA
MCS

Participants building the historical timeline



Results of key changes impacting fisheries positively and negatively

Changes identified in the historical timeline were ranked by participants as being most significant in affecting fisheries, are shown below. Colours reflect the theme (political, legal, environmental, social, economic and technological) participants were working on in the focus groups. Positive changes received valuations above 1 and negative changes are indicated with scores less than one.



Summary of the three key positive and negative impacts on fisheries

The main **positive changes** (ranked as scores above 3) to affect fisheries as perceived by the participants were related to: **marine protected areas (MPA)**, **beach management units (BMU)** and **gated traps**. The key **negative impacts** (scores below -3) on fisheries were related to the initial **MPA gazettment**, the **economic stimulus programme** and the **mangrove ban**.

Policy Cycle Review: who is involved in fisheries and what do they do

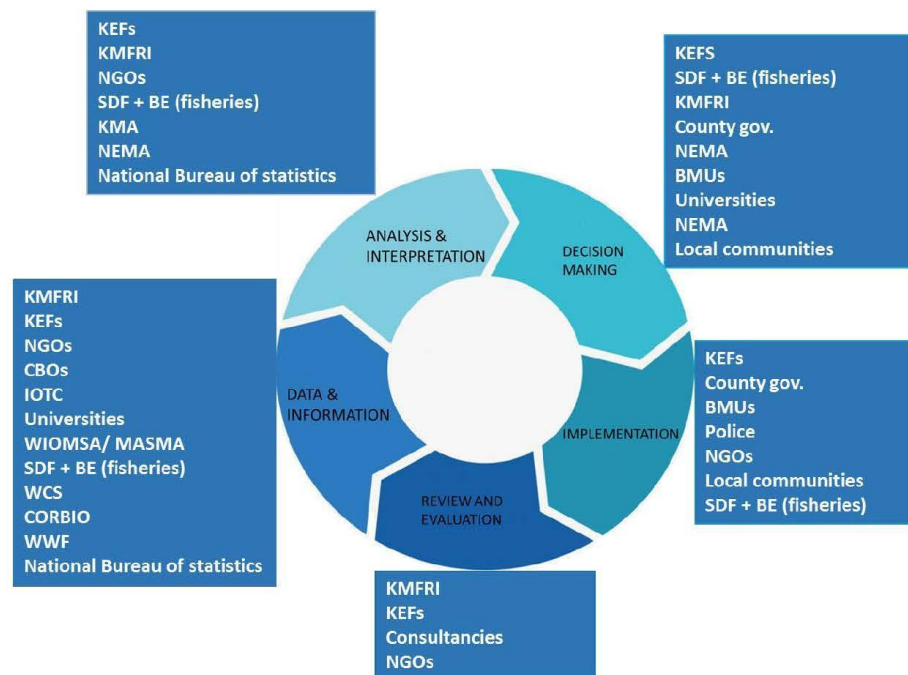
The policy cycle review was designed to show which groups and organisations are involved in fisheries governance (how decisions are made) and management (making and enforcing rules).

Theory states that a country has good governance if each of the below five stages of a simple policy cycle are complete and each stage is integrated.

Acronyms

PLEASE COMPLETE BY ADDING ACRONYM NAMES FROM POLICY CYCLE BELOW

Participants listed the organisations and groups involved in each stage of the policy cycle:



Key results from the policy cycle review exercise

Participants constructed the above policy cycle which shows **most resources** are allocated to **collecting data and information**. **Least effort and resources** is directed towards **reviewing and evaluating fisheries management**.

Conclusions and recommendations

Conclusions

The BYCAM project has provided scientific evidence to show experts' opinions on changes impacting fisheries in positive and negative ways. Similarly, the national policy workshop have highlighted who is involved in fisheries management activity. This is a first step required for evaluating adequacy of resources against achieving sustainable fisheries goals such as the targets outlined in the United Nations Sustainable Development Goal 14.

Main recommendation from historical timeline exercise

In Kenya, conservation action involving marine protected areas was considered to have the most positive impacts, however, the initial gazettement of a MPA in the 1950s was also found to have negative impacts. Part of the problem, was the lack of transitional arrangements to help fishers to adapt to changes introduced which impacted where they could fish.

Those fishers less adaptable to change, such as older fishers, need to be considered and offered help to bridge time required to find alternative or supplementary ways to source fish for food or generate income when management interventions are implemented.

Recommendation: when introducing management interventions such as MPAs, full consideration should be given to the socio-economic aspects as well as the environmental considerations. This is so those directly impacted can be given assistance to mitigate any unforeseen negative consequences, e.g. less income will impact negatively on quality of life.

Main recommendation from policy review exercise

In Kenya, most human and physical resources for fisheries management activity are allocated to collecting data and information. Least effort is directed towards reviewing and evaluating fisheries management practices. This is a common problem in complex fisheries governance systems where action at a local level is not fully integrated nor aligned into regional and national policy objectives. Thus resource allocation to support sustainable management of fisheries is not optimised.

Recommendation: goals and objectives of fisheries policy should be evaluated regularly against local capacity to operationalise fisheries management strategies.

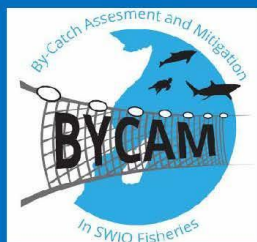
Good governance principles – cohesion, openness, participation, effectiveness and accountability – is a prerequisite for building transparency and trust in fisheries policy and associated management interventions. Thus involving relevant stakeholders requires identification of who does what and linkages made so that knowledge exchange occurs at each stage of the policy cycle review.

Participants

BYCAM National Policy Team	Attendees continued
Professor Selina Stead	
Dr Nina Wambiji	
Rosie Wilson	
Please add attendees	

Participants at the meeting





BYCAM



BY-Catch Assessment and Mitigation in Western Indian Ocean Fisheries

Overview of the project

The BYCAM project, a four-year initiative funded by WIOMSA, commenced in 2015 and brings together a unique multi-disciplinary team of 16 researchers from 7 countries.

The project has provided scientific evidence to address questions about how changes will affect communities' dependent on fisheries for livelihoods.

People in the WIO rely heavily on fisheries, which contribute to food security, income and ocean health.

Local people in the WIO have reflected on their experience to assess how to better manage and govern fisheries sustainably.

MADAGASCAR NATIONAL FISHERIES POLICY MEETING

14th August 2018, Antsiranana, Madagascar

NATIONAL BYCAM MEETING

The **objectives** were to:

1. Identify key changes in fisheries with respect to political, economic, social, technological, legal and environmental systems;
2. Examine how these changes impact people, and;
3. Discuss who is involved in fisheries governance and management, both formally (regulations) and informally.

Methods

1. Historical timeline to map changes, and;
2. Policy cycle review to show how organisations are involved in fisheries.

Contents

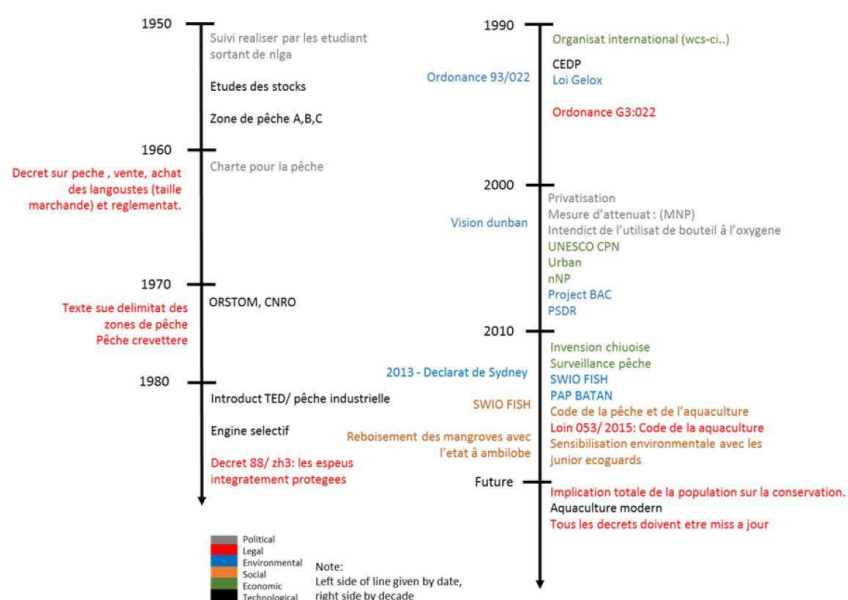
Historical timeline	2
Impacts of change	3
Policy Cycle	4
Recommendations	5
Participants	6

Professor Selina Stead with workshop facilitation team



Historical scan

Results of the historical timeline exercise showing key events and changes affecting fisheries in Madagascar identified by participants based on their collective experiences



Acronyms: PLEASE COMPLETE BY ADDING ACRONYM NAMES FROM POLICY CYCLE BELOW

ORSTOM

CNRO

TED

UNESCO

nNP

PSDR

SWIOFish

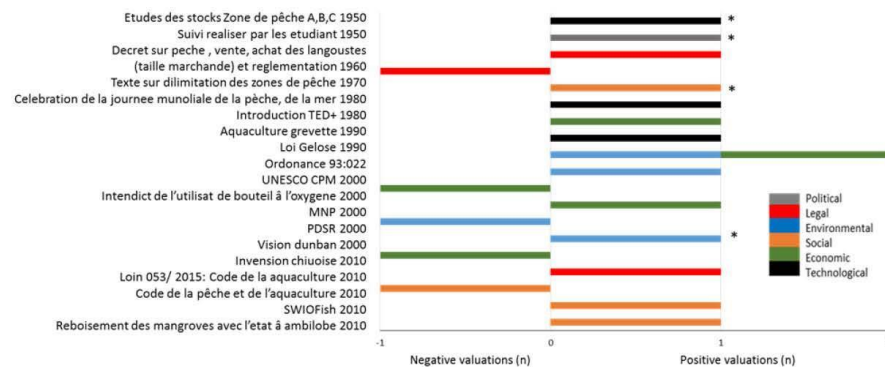
CEDP

Participants building the historical timeline



Results of key changes impacting fisheries positively and negatively

Changes identified in the historical timeline were ranked by participants as being most significant in affecting fisheries, are shown below. Colours reflect the theme (political, legal, environmental, social, economic and technological) participants were working on in the focus groups. Positive changes received valuations above 1 and negative changes are indicated with scores less than one.



*Indicates event was prioritised by participants, but impact not specified as positive or negative.

Summary of the key positive and negative impacts on fisheries

The main **positive changes** (ranked as scores above 1) to affect fisheries as perceived by the participants were related to: **Ordinance 93** and **Loi Gelosse**. The key **negative impacts** (scores below zero) on fisheries were related to **regulations associated with zonation, use of oxygen bottles, Marine National Park** and the **Code of fisheries and aquaculture**.

Policy Cycle Review: who is involved in fisheries and what do they do

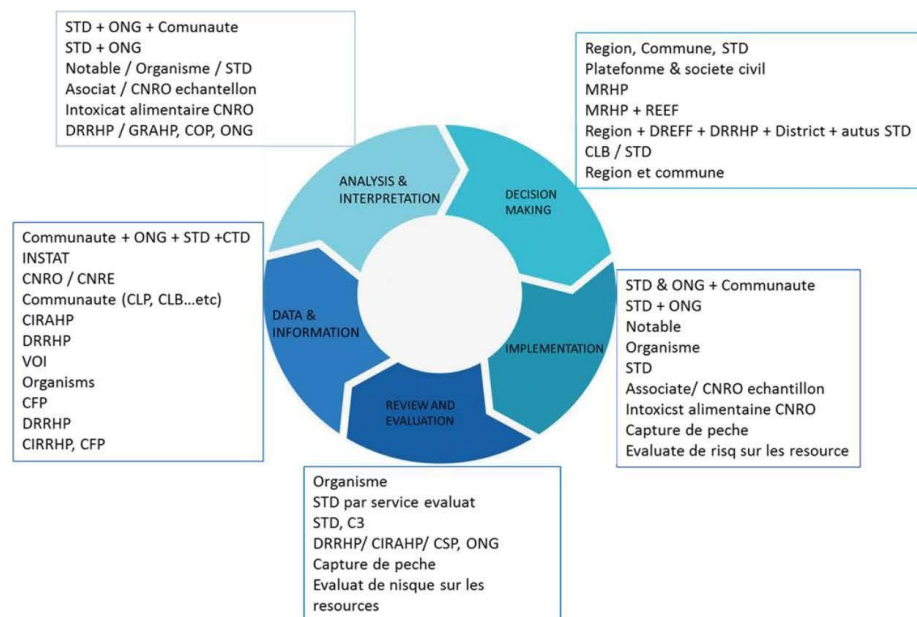
The policy cycle review was designed to show which groups and organisations are involved in fisheries governance (how decisions are made) and management (making and enforcing rules).

Theory states that a country has good governance if each of the below five stages of a simple policy cycle are complete and each stage is integrated.

Acronyms

PLEASE COMPLETE BY
ADDING ACRONYM
NAMES FROM POLICY
CYCLE BELOW

Participants listed the organisations and groups involved in each stage of the policy cycle:



Key results from the policy cycle review exercise

Participants constructed the above policy cycle which shows **most resources** are allocated to **collection of data and information for fisheries management**. **Least effort and resources** is directed towards fisheries **review and evaluation**.

Conclusions and recommendations

Conclusions

The BYCAM project has provided scientific evidence to show experts' opinions on changes impacting fisheries in positive and negative ways. Similarly, the national policy workshop have highlighted who is involved in fisheries management activity. This is a first step required for evaluating adequacy of resources against achieving sustainable fisheries goals such as the targets outlined in the United Nations Sustainable Development Goal 14.

Main recommendation from historical timeline exercise

In Madagascar, the most positive change to impact fisheries was perceived to be the introduction of Ordinance 93 for regulating fisheries and aquaculture.

In contrast, enforcing regulations associated with appropriate marine resource use in the different zones was perceived as having a negative impact on the health of fisheries. Weak governance was offered as one of the reasons underpinning rule breaking.

Recommendation: a review and evaluation of both positive and negative impacts on fisheries identified by participants would be useful in terms of highlighting best practice which could be rolled out more widely. Updating relevant legislation was highlighted as a priority action which could help address shortcomings in existing legislation and management.

Main recommendation from policy review exercise

In Madagascar, most human and physical resources for fisheries management related activity are directed towards the collection of data and information. Least effort is put towards reviewing and evaluating fisheries legislation and management. This is a common problem in complex fisheries governance systems where action at a local level is not fully integrated nor aligned into regional and national policy objectives. Thus resource allocation to support sustainable management of fisheries is not optimised.

Recommendation: engaging locals in fisheries management decisions at an earlier stage could lead to better support for compliance with rules through formal and informal governance mechanisms. Participants recommended better engagement between those involved in enforcement, management and policy of fisheries with marine resource users who are impacted by any changes to fisheries practices.

Participants

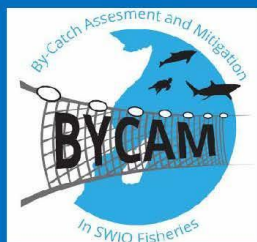
BYCAM National Policy Team	Attendees continued
Professor Selina Stead	
Lala Rakotoarimino	
Rosie Wilson	
Please add attendees	



CONTACT:

Lala Rakotoarimino

Email: lala@c-3.org.uk



BYCAM



BY-Catch Assessment and Mitigation in Western Indian Ocean Fisheries

Overview of the project

The BYCAM project, a four-year initiative funded by WIOMSA, commenced in 2015 and brings together a unique multi-disciplinary team of 16 researchers from 7 countries.

The project has provided scientific evidence to address questions about how changes will affect communities' dependent on fisheries for livelihoods.

People in the WIO rely heavily on fisheries, which contribute to food security, income and ocean health.

Local people in the WIO have reflected on their experience to assess how to better manage and govern fisheries sustainably.

ZANZIBAR NATIONAL FISHERIES POLICY MEETING

4th August 2018, Stone Town, Zanzibar, Tanzania

NATIONAL BYCAM MEETING

The **objectives** were to:

1. Identify key changes in fisheries with respect to political, economic, social, technological, legal and environmental systems;
2. Examine how these changes impact people, and;
3. Discuss who is involved in fisheries governance and management, both formally (regulations) and informally.

Methods

1. Historical timeline to map changes, and;
2. Policy cycle review to show how organisations are involved in fisheries.

Contents

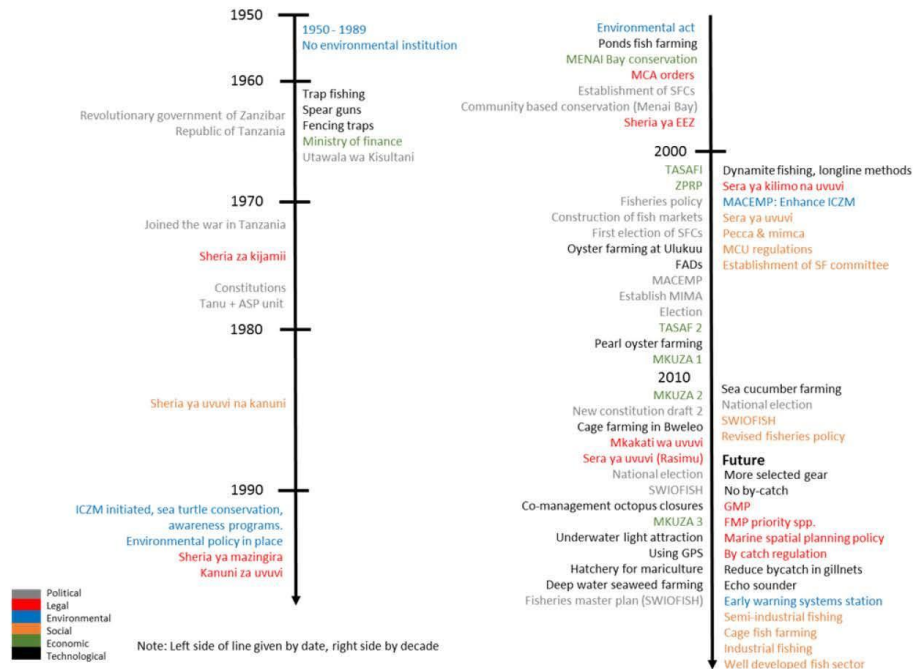
Historical timeline	2
Impacts of change	3
Policy Cycle	4
Recommendations	5
Participants	6

Drs Amir, Jiddawi & Prof Stead Selina Stead explaining aims



Historical scan

Results of the historical timeline exercise showing key events and changes affecting fisheries in Zanzibar identified by participants based on their collective experiences



Acronyms: PLEASE COMPLETE BY ADDING ACRONYM NAMES FROM POLICY CYCLE BELOW

MCA

EEZ

TASAFI

ZPRP

FADs

MACEMP

MKUZA

SWIOFish

MCU

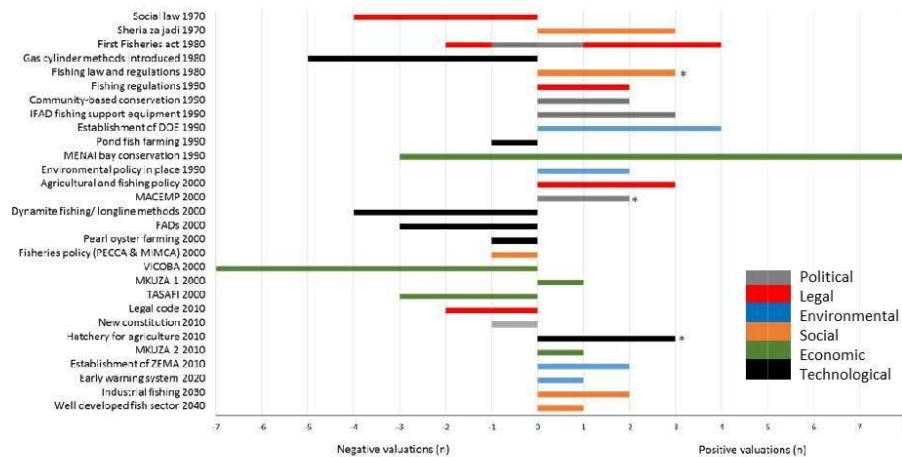
GMP

Participants building the historical timeline



Results of key changes impacting fisheries positively and negatively

Changes identified in the historical timeline were ranked by participants as being most significant in affecting fisheries, are shown below. Colours reflect the theme (political, legal, environmental, social, economic and technological) participants were working on in the focus groups. Positive changes received valuations above 1 and negative changes are indicated with scores less than one.



*Indicates event was prioritised by participants, but impact not specified as positive or negative.

Summary of the three key positive and negative impacts on fisheries

The main **positive changes** (ranked as scores above 3) to affect fisheries as perceived by the participants were related to: the **First Fisheries Act**, **MENAI Bay conservation** and establishment of the **Department of the Environment (DOE)**. The key **negative impacts** (scores below -3) on fisheries were related to the **social law**, the **introduction of gas cylinder/dynamite/longline methods** and **VICOBA**.

Policy Cycle Review: who is involved in fisheries and what do they do

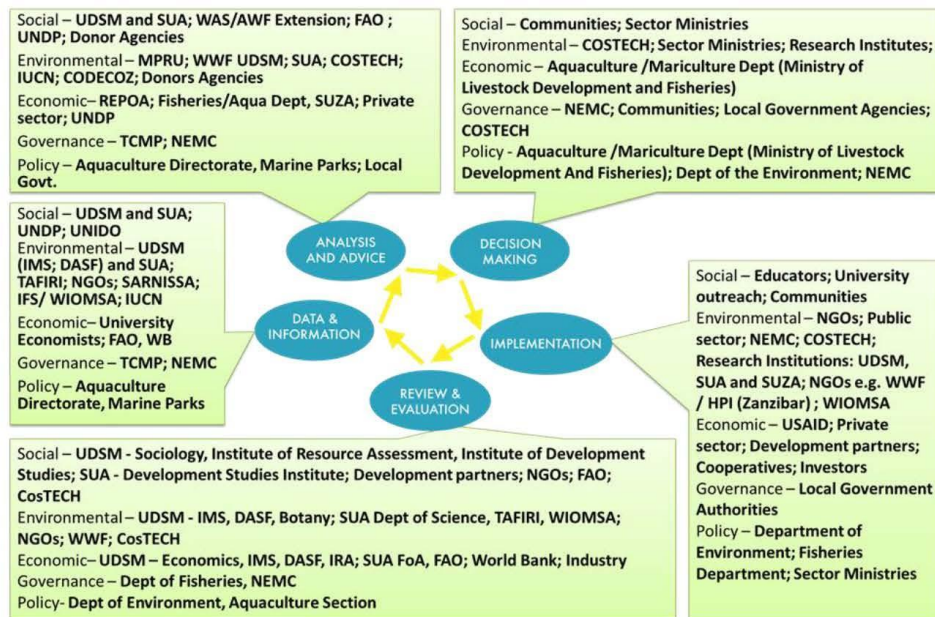
The policy cycle review was designed to show which groups and organisations are involved in fisheries governance (how decisions are made) and management (making and enforcing rules).

Theory states that a country has good governance if each of the below five stages of a simple policy cycle are complete and each stage is integrated.

Acronyms

PLEASE COMPLETE BY
ADDING ACRONYM
NAMES FROM POLICY
CYCLE BELOW

Participants listed the organisations and groups involved in each stage of the policy cycle:



Key results from the policy cycle review exercise

Participants constructed the above policy cycle which shows **most resources** are allocated to **environmental aspects of fisheries management**. **Least effort and resources** is directed towards fisheries **governance**.

Conclusions and recommendations

Conclusions

The BYCAM project has provided scientific evidence to show experts' opinions on changes impacting fisheries in positive and negative ways. Similarly, the national policy workshop have highlighted who is involved in fisheries management activity. This is a first step required for evaluating adequacy of resources against achieving sustainable fisheries goals such as the targets outlined in the United Nations Sustainable Development Goal 14.

Main recommendation from historical timeline exercise

In Zanzibar, the most positive changes to impact fisheries were the establishment of the First Fisheries Act and the set-up of the Department of the Environment. Similarly, the implementation of the Menai Bay Conservation Area was considered to have positive influences on fisheries.

In contrast, the Village Community Banks (VICOPA) were perceived to have negative impacts, as was social law and the introduction of non-sustainable fishing methods including dynamite.

A recurring theme highlighted as a major issue which needed to be tackled is the lack of enforcement of fisheries regulations.

Recommendation: a review and evaluation of both positive and negative impacts on fisheries identified by participants would be useful in terms of highlighting best practice which could be rolled out more widely and areas needing to be prioritised for action to address shortcomings. In particular, understanding the reasons underpinning success and failure would allow a more targeted approach to developing future fisheries policy that is context-specific to the local issues.

Main recommendation from policy review exercise

In Zanzibar, most human and physical resources for fisheries management activity are allocated to the environmental aspects of fisheries management yet enforcement of associated rules such as bans was perceived to be weak. Least effort is directed towards governance. Participants highlighted a lack of opportunity to be engaged in decision-making about fisheries policy and management rules.

Recommendation: engaging locals in fisheries management decisions at an earlier stage could lead to better support for compliance with rules through informal governance mechanisms. Participants suggested strengthening the local community fisheries committees could be one way to improve fisheries governance and reduce rule-breaking.

Participants

BYCAM National Policy Team	Attendees continued
Professor Selina Stead	
Dr Narriman Jiddawi	
Dr Omar Amir	
Yussuf Salmin	
Rosie Wilson	
Please add attendees	

Participants at the meeting



Annex 7: Marine megafauna catch in south-western Indian Ocean small-scale fisheries from landings data

Biological Conservation 230 (2019) 113–121



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Marine megafauna catch in southwestern Indian Ocean small-scale fisheries from landings data



Andrew J. Temple^{a,*}, Nina Wambiji^b, Chris N.S. Poonian^c, Narriman Jiddawi^d, Selina M. Stead^a, Jeremy J. Kiszka^e, Per Berggren^a

^a School of Natural and Environmental Sciences, Newcastle University, Newcastle-upon-Tyne, NE1 7RU, UK

^b Kenya Marine and Fisheries Research Institute, Mombasa, P.O. Box 80100, Kenya

^c Community Centred Conservation (C3), London, WC1N 3AX, UK

^d Institute of Marine Sciences, University of Dar es Salaam, Zanzibar, P.O. Box 668, United Republic of Tanzania

^e Department of Biological Sciences, Florida International University, Miami, FL 33181, USA

ARTICLE INFO

Keywords:

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Marine mammal
Sea turtle
Bycatch
Small-scale fishery
SWIO

ABSTRACT

The measurable impacts of small-scale fisheries on coastal marine ecosystems and vulnerable megafauna species (elasmobranchs, marine mammals and sea turtles) within them are largely unknown, particularly in developing countries. This study assesses megafauna catch and composition in handline, longline, bottom-set and drift gillnet fisheries of the southwestern Indian Ocean. Observers monitored 21 landing sites across Kenya, Zanzibar and northern Madagascar for 12 months in 2016–17. Landings ($n = 4666$) identified 59 species, including three sea turtles, two small cetaceans and one sirenian (*Dugong dugon*). Primary gear threats to investigated taxa were identified as bottom-set gillnets (marine mammals, sea turtles and batoids), drift gillnets (marine mammals, batoids and sharks) and longlines (sharks). Overall, catch was dominated by small and moderately sized coastal requiem sharks (Carcharhiniformes) and whiptails (Dasyatidae). Larger coastal and oceanic elasmobranchs were also recorded in substantial numbers as were a number of deeper-water species. The diversity of catch demonstrates the potential for small-scale fisheries to have impacts across a number of ecosystems. From the observed catch rates we calculated annual regional elasmobranch landings to be 35,445 (95%CI 30,478–40,412) tonnes, 72.6% more than officially reported in 2016 and 129.2% more than the 10-year average (2006–16), constituting 2.48 (95%CI 2.20–2.66) million individuals. Productivity-Susceptibility Analyses indicate that small and moderately sized elasmobranchs are most vulnerable in the small-scale fisheries. The study demonstrates substantial underreporting of catches in small-scale fisheries and highlights the need to expand efforts globally to assess the extent and impact of small-scale fisheries on vulnerable marine species and their respective ecosystems.

1. Introduction

Fisheries present the greatest short to medium-term anthropogenic threat to the survival of numerous marine vertebrates. This is particularly true for those species predominantly displaying classic k-selected life history traits (long-life, high natural survivorship, slow growth, late maturity and low fecundity), such as elasmobranchs (sharks and batoids), marine mammals and sea turtles (e.g. Dulvy et al., 2014; Lewison et al., 2004; Wallace et al., 2010; Zydalis et al., 2009). In industrial fisheries elasmobranchs, marine mammals and sea turtles (herein referred to as marine megafauna) are usually considered as bycatch, whereas in many small-scale fisheries (SSF) these taxa may constitute target or by-product species. SSF are herein defined broadly

as those as those fisheries operating either for subsistence or for income generation (artisanal) but not as part of a commercial company, generally < 10 m sailing or outboard powered vessels. SSF account for > 95% of fishers at the global level (Pauly, 2006) and 32% of fisheries catch (Pauly and Zeller, 2015). Despite this, SSF have received disproportionately little attention (Molina and Cooke, 2012), especially in developing countries where SSF are most prevalent. Losses of elasmobranchs, marine mammals and sea turtles may have implications for the structure, function and productivity of ecosystems (e.g. Aragonés et al., 2006; Heithaus et al., 2008; Kiszka et al., 2015). These implications are especially concerning in SSF dominated regions, as it is there that coastal communities rely most heavily upon near-shore environments for their survival and livelihoods, with limited adaptive capacity to

* Corresponding author at: Room 4.71, Ridley Building 2, Newcastle University, Newcastle-upon-Tyne, Tyne and Wear, NE1 7RU, UK.
E-mail address: andrew.temple@ncl.ac.uk (A.J. Temple).

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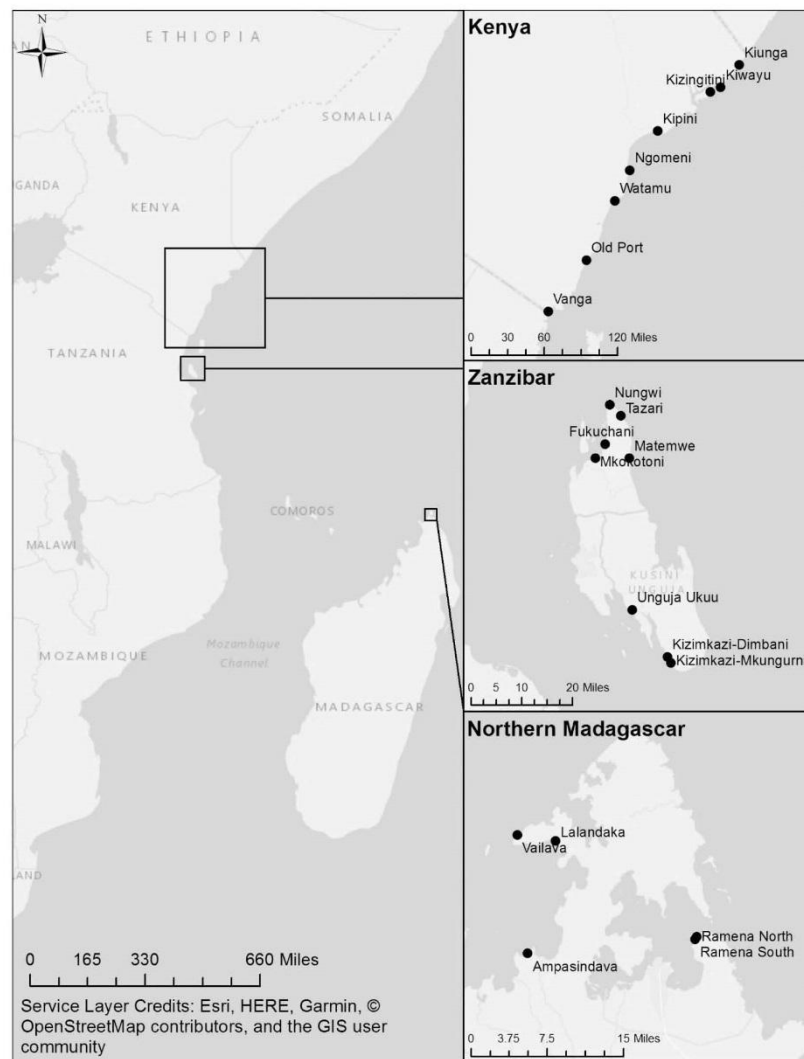


Fig. 1. Locations of landing sites in the southwestern Indian Ocean monitored for marine megafauna catch between June 2016 and June 2017.

respond to ecosystem change.

The southwestern Indian Ocean (SWIO), consisting of East Africa and the associated islands, represents a SSF dominated developing region, where catch of marine megafauna is common (for review see Temple et al., 2018). The SWIO has upwards of 0.5 million SSF fishers (Temple et al., 2018) contributing around 66.4% of the annual catch (Pauly and Zeller, 2015). The region is undergoing rapid population growth, with the human population expected to double to 357 million by 2050 (WB, 2016), and migration from rural inland areas to coastal regions. Thus, increasingly pressure is placed on marine resources for food and income generation. Indeed, fish proteins (marine and fresh-water) range from 7.3–49.7% of animal protein and 1.9–23.1% of total protein consumed (FAO, 2017), with means of 20.7% and 3.6%

respectively across the region, once population size is accounted for. Traditionally marine megafauna in the SWIO has been used for both subsistence and commercially marketed as food and bait (e.g. Barrowclift et al., 2017; Humber et al., 2011; Razafindrakoto et al., 2008). Elasmobranchs may be targeted (particularly for fins and meat), but are mostly considered as by-product species and are commonly caught in gillnet and longline gears (Kiszka and van der Elst, 2015; Temple et al., 2018). Little in the way of management exists for elasmobranchs in the SWIO region, though National Plans of Action are either in place or under development throughout the region, suggesting widespread recognition of the potential threat to elasmobranchs (Temple et al., 2018). Conversely, as prohibited species sea turtles and marine mammals are rarely targeted, but are frequently captured in

gillnets throughout the region (Bourjea et al., 2008; Kiszka et al., 2009; Muir and Kiszka, 2012). However, a burgeoning range of ecotourism activities focused around marine megafauna species are present across the region (Gallagher and Hammerschlag, 2011; O'Connor et al., 2009; O'Malley et al., 2013) and may incentivise increasingly non-sustainable uses for these species for livelihoods in future.

Despite the prevalence of marine megafauna in SWIO SSF, there is limited understanding of catch and composition in these fisheries (Temple et al., 2018). Official catch statistics show systemic under-reporting and are often inconsistent. Elasmobranchs are primarily reported under generalised categories with limited species level information (FAO, 2018; Temple et al., 2018). Moreover, limited independent data are available, with the majority being geographically restricted case studies (Temple et al., 2018). Data for sea turtles and marine mammals are generally lacking (FAO, 2018). Yet, there is evidence of sea turtle captures in several countries across the SWIO (e.g. Humber et al., 2011; Okemwa et al., 2004; Pusineri and Quillard, 2008) and for marine mammals wherever they encounter fisheries (Kiszka et al., 2009).

Improving fisheries catch and composition data, paired with fishing effort data, are essential in informing robust evidence-based management strategies to safeguard the future sustainability of marine megafauna fisheries and those communities whose livelihoods are dependent upon them. Further, such data act as first-steps towards understanding fisheries and thus form a baseline for future stock assessment and management. In this study, we aim to provide detailed, multi-country cross-sectional data on the scale and composition of SWIO SSF marine megafauna landings in high-risk gears, specifically, gillnets (bottom-set and drift) and longlines, as well as those of the numerically dominant handlines. SSF are herein defined broadly as those as those fisheries operating either for subsistence or for income generation (artisanal) but not as part of a commercial company, generally < 10 m sailing or outboard powered vessels. Further, we analyse the fisheries effort, patterns and drivers and use this to predict total annual landings of marine megafauna in SWIO SSF both at select national scales and estimate landings at the regional level.

2. Methods

2.1. Data collection

Trained land-based observers recorded landings of marine megafauna (elasmobranchs, marine mammals and sea turtles) from select fisheries gears at landings sites in Kenya ($n = 8$), Zanzibar ($n = 8$) and northern Madagascar ($n = 5$) (Figure 1) for a period of 12 months between June 2016 and June 2017. Gears monitored were drift and bottom-set gillnets, longlines (demersal and pelagic) and handlines (including rod and reel gears). However, within each gear category there is variability in specifications (e.g. mesh size, net length, hook size, number of hooks etc.), the impacts of which are not considered in this study. Observers at each site collected data for 147 simultaneous sampling days. Sampling days were selected using a stratified-random approach: the year was divided into lunar months which were subdivided into four lunar phases (new moon, first quarter, full moon, third quarter) and three sampling days randomly generated within each lunar phase. This sampling regime ensured that the study accounted for potential lunar-driven patterns in fishing effort and species availability to the fishery (e.g. variability in vertically migrating species), and subsequent effects on catches. Landing sites were selected accounting for three major factors: prevalence of longline and gillnet gears (maximising representation), geographic spread (maximising geographic coverage and potential links to species availability) and logistical constraints (e.g. sites needed to be accessible by road).

Observers recorded data for landed marine megafauna including photographs for species identification, morphometric data (fork length, disc width and weight), sex, vessel primary gear used and local species

name. Observers also recorded fishing effort as total number of vessels active per day by primary gear type, fishing trips were < 24 h and a single vessel may make more than one trip per day, multiple trips are not counted separately for the purpose of this monitoring fishing activity. In the event of vessels also employing a secondary gear type, catch was assigned to the primary gear type used by the vessels, secondary gears were rarely employed.

In order to validate landings observations, fishers ($n = 521$) at each site were independently asked to give an anonymous opinion on the efficacy of observers. Specifically, fishers gave an estimate, for each recorded megafauna taxon, of the average proportion of landings, if any, that were not recorded by the landings observer on any given day. Fishers' declarations indicate the potential magnitude of catch under-estimate in the study.

2.2. Analysis

Patterns in fisheries effort data by gear type were assessed using a Generalised Additive Mixed Modelling (GAMM) approach, with landing site as a random-effect variable. This meant that cross-sectional patterns in fisheries effort could be drawn across the whole study area whilst also accounting for landing site-specific patterns when using the GAMM to predict effort for days where sampling did not occur. Independent variables input to the GAMM model included daily data on precipitation, wind speed, wind direction, temperature, maximum tidal height (as a proxy for lunar phase) and month. Environmental data were extracted from the NCEP/NCAR Reanalysis database (ESRL, 2017). Cyclical variables (e.g. wind direction, month) were fitted with cyclic splines. Where independent variables showed evidence of collinearity ($r > 0.2$ or < -0.2), those with least explanatory power were removed from the model for subsequent iterations.

An annual weighted mean Catch Per Unit effort (CPUE) was calculated, as the number of individuals caught per active vessel, for sharks and batoids separately, by gear type, across landing sites for each country. This was achieved by first calculating respective CPUEs at each site and subsequently weighting CPUEs based on the total predicted effort for each site (sourced from the GAMM model) as this was considered the best estimate of their relative contribution to the overall fisheries effort in their respective country. Where data were missing for either megafauna group or the gear type in which an animal was caught, missing data were retrospectively assigned proportional to known catches, on a site-by-site basis. The weighted mean CPUE was then multiplied by the total predicted effort to create estimates of total catch across sites within each country. This catch was subsequently scaled to the national level for Zanzibar and Kenya. This scaling was achieved by dividing the total predicted catch across sites by the total number of vessels of respective gear type present and multiplying this by the total number of vessels of each gear in the respective country. Vessel by gear type data was sourced from existing frame survey data (KMALF, 2017; ZDFD, 2018) as these data represent the best estimates currently available.

An estimate of total catch weight was also produced for both sharks and batoids. Substantial error is likely in the weight data collected, this is primarily the result of much of the catch being landed in partially dressed states (e.g. organs and/or fins removed) and potential biases in the size of animals that could be successfully weighed (e.g. larger animals are more difficult to weigh). In order to address this, weight-fork length and weight-disc width relationships were modelled for sharks and batoids respectively. For the purpose of estimating weights, guitarfish and wedgefish (Rhynchobatidae) were included with the shark data. Weights for specimens which were known to be in a partially dressed state were excluded from the analysis. A linear model was used to assess weight-fork length/disc width relationships, after data were logN transformed. Cook's distance was used to identify data outliers which exerted undue influence on the linear model, likely a result of measurement and/or data entry errors, and these outliers were

removed. The linear models were then re-run and the relationship described (Fig. A1). The model was used to predict the weight for all specimens with a known fork length or disc width as appropriate, back-transformation of weights was done using Sprugel's correction factor method (Sprugel, 1983). The revised weight dataset was assumed to be representative of the weight distributions of species in respective sites and gear types. Mean weight for each elasmobranch taxa for each gear type was calculated and subsequently multiplied by the total estimated megafauna catch to produce a total catch weight by gear for each elasmobranch taxa at each site.

An estimate for SSF elasmobranch landings across the SWIO was also generated. This estimate was calculated by dividing the sum total of predicted catch across all 21 sites by both the number of vessels across all sites and the number of gears across all sites for respective gear types, generating two new CPUE values. CPUE values were scaled using existing SSF fleet data for total vessel counts by gear and/or gear counts, as available, for all SWIO countries to create a regional estimate (Chacate and Mutombene, 2016; IOTC, 2018; Kiszka et al., 2009; KMALF, 2017; MFR, 2010; SFA, 2015; UDC, 2017; URT, 2017; WIOFish, 2018; ZDFD, 2018). Bottom-set and drift gillnets are combined into a singular category in the fisheries statistics of a number of SWIO nations. Thus it is assumed that the proportion of drift and bottom-set gillnets monitored in this study is representative of these gears at the SWIO level.

Landings composition by country and gear type was achieved through species identification from photographs taken by observers, to species or nearest taxonomic level where possible. Local species names were also recorded. Where local names corresponded to specific species groups they were used to identify non-photographed individuals to genus level or higher, with the majority being to family level, minimising the risk of wrongful identification.

Lastly, in order to identify priority species and gear-species interactions for future research, a series Productivity-Susceptibility Analysis (PSA) (Hobday et al., 2011), confined within the context of SWIO SSF, were carried out for gear types in isolation and combination. Species were included in the assessment only if 10 or more individuals were identified in the catch. The PSA (Table 1) was based on existing MRAG and NOAA designs (Patrick et al., 2009; Rosenberg et al., 2009). Categories were excluded, added or modified where they were either irrelevant, required adaptation to be applicable or were needed to better address the classically k-selected life history, biological and ecological characteristics of the species considered in this study. When carrying

out the PSA across gear types the “Gear Interaction Risk” attribute was excluded. Further, “Management Strategy” and “Management Regulation” attributes were ultimately removed from all PSA assessments as there was no variability in these attributes among species, a result of the limited monitoring and regulation of elasmobranch catch throughout SWIO SSF (Temple et al., 2018). Attributes were scored on a scale of 1 (Low) to 3 (High) with intervals of 0.5 allowed. Attributes originating from available quantitative data were scored by scaling the data between one and three. In cases where data were skewed by outlier values (e.g. extreme size) these were first logN transformed before scaling. Quality of the data used for each combination of attribute and species, was represented by assigning a confidence scores between 1 (Low) and 3 (High) with intervals of 0.5 allowed. PSA scoring was carried out independently by three of the authors and the mean values of these scores were taken. Weightings reflecting relative importance of attributes as indicators of the respective vulnerability aspect (productivity or susceptibility) were sought independently from a range of experts (with nine respondents, four of whom are authors) with backgrounds in fisheries modelling, fisheries social science and fisheries-marine megafauna interactions. Weightings were combined to calculate mean ($\pm$ 95% CI) values for productivity and susceptibility as well as confidence in the data used. Mean values are plotted and the overall vulnerability calculated as the Euclidean distance from the origin. Some attributes used in the PSA are unlikely equable measures of vulnerability aspects across taxa, e.g. maximum size (Juan-Jordá et al., 2015). Thus, direct comparisons should be made between sharks and batoids with caution.

All analyses and data visualisations were carried out and produced using the R statistical software, version x64 3.4.0 (R Core Team, 2017). Ethical approval for this project was sought from, and approved by, Newcastle University's animal welfare ethics review board (ID 426).

3. Results

3.1. Fisheries effort

After co-linear independent variables were iteratively excluded, GAMM analyses showed significant effects ($p < 0.05$) of three independent variables on fishing effort: maximum tidal height, month and wind direction. Month significantly influenced bottom-set, driftnet, handline and longline fishing effort ($\chi^2 = 1453.10$, 42.45, 161.48, 380.62, respectively; $p < 0.05$). Tidal height significantly influenced

Table 1
Attributes and scoring used in the Productivity-Susceptibility Analysis, confined within the southwestern Indian Ocean small-scale fisheries context.

Attribute	Low (1)	Medium (2)	High (3)
Productivity			
Maximum size	Scaled LogN transformation of fork length OR disc width as appropriate. Larger size = lower score.		
Fecundity	Scaled LogN transformation of offspring per annum. Higher fecundity = higher score.		
Mode of reproduction	Viviparous	-	Ovoviviparous
Size at maturity	Scaled size at maturity as a percentage of maximum size. Higher proportion = lower score.		
Susceptibility			
Geographic spread	Regional	Sub-regional	Endemic
Overlap with small-scale fisheries	Oceanic exclusive	Semi-Oceanic	Coastal
			Shallow water restricted
Gear interaction risk	Low risk of gear interaction (e.g. pelagic species and demersal gear)	Moderate risk of gear interaction	High risk of gear interaction (e.g. pelagic species and pelagic gear)
Desirability of catch for consumption or sale	Low value	Moderate value	High value
Management strategy	Appropriate monitoring	Limited monitoring	No monitoring
Management regulations	Regulated	Partially regulated	No Regulation
Catch relative to productivity	Scaled LogN transformation of catch relative to productivity score. Observed catch was divided by the exponential of the mean productivity score, high productivity species are assumed to sustain much higher catch rates. Higher rate = higher score.		
Female mortality	Scaled proportion of captures that are female. Higher proportion = higher score		

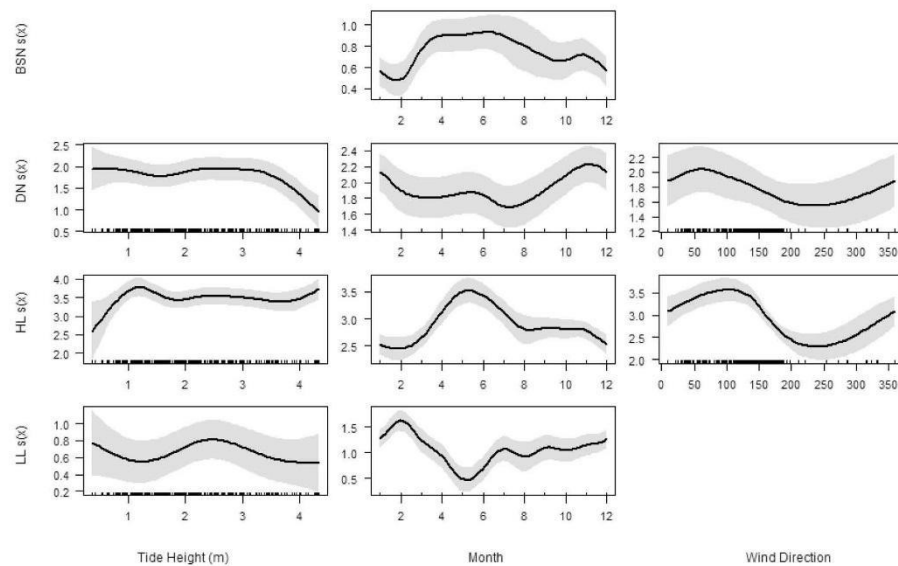


Fig. 2. Smooths ($\pm$ 95% CI) from a series of generalised additive mixed models describing the cross-sectional relationship between tidal height, month and wind direction (degrees from North) with fishing effort of bottom-set gillnet (BSN), drift gillnets (DN), handline (HL) and longline (LL) small-scale fisheries across Kenya, Zanzibar and northern Madagascar between June 2016–June 2017.

driftnet, handline and longline fishing effort ($\chi^2 = 41.65, 33.12, 9.76$, respectively, $p < 0.05$). Wind direction significantly influenced driftnet and handline fishing effort ($\chi^2 = 14.57, 175.12$, respectively; $p < 0.05$). Wind speed was not found to affect fishing effort for any gear type, despite being expected to impact sea conditions. The models explained 75.0%, 54.2%, 82.2% and 54.9% of the deviance for bottom-set net, driftnet, handline and longline fishery effort, respectively (Figure 2).

Patterns of fisheries effort across gears show intra-annual variability and are suggestive of inverse relationships between efforts from bottom-set gillnet and handline gears with those of drift gillnet and longline gears. Fishers often operate multiple gear types over the course of the year and inverse effort relationships likely reflect fishers transitioning between gears to maximise yield as influenced by external factors. The peaks in longline and drift gillnet use correspond closely to the seasonal monsoons in the SWIO region. SWIO SSF catches are highest during the north-east monsoon (November–March) (Jury et al., 2010; Lan et al., 2013; McClanahan, 1988). This seasonality coincides with increases in coastal availability of migratory oceanic species, such as yellowfin tuna (Lan et al., 2013), which are often target species for longline and drift gillnet gears. Similarly, anecdotal evidence from fishers suggesting a reduction in drift gillnet use during the brightest phases of the moon, because “the fish can see the nets”, is supported and similar patterns may also be emerging in longline gear. Handline gear usage follows an inverse pattern, further reflecting the multi-gear nature of the SSF.

3.2. Fisheries catch

Weighted mean CPUE and individual weight is presented by gear for sharks and batoids for Kenya, northern Madagascar and Zanzibar (Table A1). Using weighted mean CPUEs combined with fishing effort, annual catch was estimated across gear types for Kenya and Zanzibar at the country level (Table 2). An annual estimate for Madagascar is not calculated as the sampling sites are not considered representative of the

country as a whole. Further, reliable data for fishing effort metrics (i.e. number of vessels by gear type) was not available for the provinces covered. A total estimate for SWIO level catch was also calculated (Table 2).

Fisher estimates of observer efficacy in recording elasmobranch catch suggest a weighted mean underreporting rate across sites of 21.1% and 33.9% in Kenya, 23.3% and 19.2% in Zanzibar and 17.3% and 19.1% in Madagascar, for batoids and sharks respectively.

3.3. Species composition and vulnerability

Landings compositions (frequency of occurrence) for both batoids and sharks are presented by country and gear type (Fig. 3). Composition is presented at the genus or family levels depending on the level at which catch could be identified. Additionally, in Kenya, bottom-set nets landed one loggerhead (*Caretta caretta*), 24 green (*Chelonia mydas*), 43 hawksbill (*Eretmochelys imbricata*) and one unidentified sea turtle; drift gillnets landed one spinner dolphin (*Stenella longirostris*); and handlines landed four *C. mydas* and one *E. imbricata*. In Zanzibar, bottom-set gillnets landed two *C. mydas*, two *E. imbricata*, two unidentified sea turtles and one unidentified dolphin; and drift gillnets landed one *E. imbricata*, two unidentified sea turtles, one Indo-Pacific bottlenose dolphin (*Tursiops aduncus*) and one unidentified dolphin. In Madagascar, one unidentified sea turtle was landed from a longline. Whilst most fishers declined to declare the subsequent use of landed sea turtles and dolphins, at least seven of the turtles were sold for human consumption and two of the dolphins for use as fisheries bait, indicating an existing market for these species. The full list of species caught is available (Table A2).

Batoid landings (Fig. 3) across the three countries were primarily dominated by whiprays (Dasyatidae). The largest contributors were small and moderately sized, benthic, coastal species such as the bluespotted maskray (*Neotrygon caeruleopunctata*), bluespotted fantail ray (*Taeniura lymma*), leopard whipray (*Himantura leoparda*) and Baraka's whipray (*Maculabatis ambigua*). The whiprays were commonly captured

Table 2

Estimates ($\pm$ 95% CI) for total individuals and weight of elasmobranchs landed from handline, longline, bottom-set and drift gillnet gears in Kenya, Zanzibar and in the southwestern Indian Ocean (SWIO) between June 2016–June 2017.

Scale	Type	Individuals	Weight (tonnes)
Kenya	Batoids	17,393 (13,680–21,106)	264.9 (184.8–344.9)
	Sharks	34,354 (22,436–46,273)	327.0 (222.0–432.9)
	Elasmobranchs	51,748 (39,265–64,231)	591.8 (459.8–723.8)
Zanzibar	Batoids	134,384 (110,646–158,122)	1512.5 (900.1–2124.9)
	Sharks	52,575 (48,247–56,903)	414.9 (246.2–583.5)
	Elasmobranchs	186,959 (162,830–211,089)	1927.4 (1292.2–2562.6)
SWIO	Batoids	1,148,467 (1,016,745–1,280,189)	17,040.3 (12,567.7–21,512.8)
	Sharks	1,332,971 (1,210,680–1,455,261)	18,404.8 (16,244.7–20,565.0)
	Elasmobranchs	2,481,437 (2,301,700–2,661,175)	35,445.1 (30,478.3–40,412.0)

across the four gear types of interest, presumably reflecting higher abundance and availability in inshore waters relative to other batoids. There was also notable landings of spotted eagle rays (*Aetobatus ocellatus*), shorttail cownose rays (*Rhinoptera jayakari*) and various mobulids (*Mobula* spp.), which were predominantly (68.1%) benthic devilrays (*Mobula thurstoni*). These pelagic batoids were most commonly caught in drift gillnets, particularly in Zanzibar at sites with access to adjacent deeper waters. Conversely, none of these pelagic batoids were caught in drift gillnets in northern Madagascar, although these species did appear in both bottom-set gillnet and longline landings. Drift gillnet catches in northern Madagascar were almost entirely comprised of benthic species, suggesting that unlike other areas drift gillnets in northern Madagascan sites operated primarily in shallow water environments. Various species of guitarfish and wedgefish were also landed.

Shark landings (Fig. 3) across the three sampled countries were dominated by ground sharks (Carcharhiniformes), within which requiem (Carcharhinidae), hammerhead (Sphyrnidae) and hound

(Triakidae) sharks were most common. The largest contributors were small and moderately sized species occurring in a range of coastal, oceanic and deep-sea habitats, particularly smoothhounds (*Mustelus* spp.), sliteye (*Loxodon macrorhinus*), spurdog (*Squalus* spp.), hardnose (*Carcharhinus macroti*), grey reef (*Carcharhinus amblyrhynchus*) and spottail (*Carcharhinus sorrah*) sharks. Scalloped hammerheads (*Sphyrna lewini*) were also common. Larger species, such as bull (*Carcharhinus leucas*) and tiger (*Galeocerdo cuvier*) sharks, were recorded in limited numbers. Oceanic and deep-water species, including shortfin mako (*Isurus paucus*), silky (*Carcharhinus falciformis*), thresher (*Alopias* spp.) and bigeye sixgill (*Hexanchus nakamurai*) were recorded in relatively low numbers. Other landings of note included a 5.7 m male whale shark (*Rhincodon typus*) caught in a bottom-set gillnet in Kenya, and a large female white shark (*Carcharodon carcharias*) in a drift gillnet in Zanzibar, one of few records in East Africa (Cliff et al., 2000). Both *Rhincodon typus* and *Carcharodon carcharias* appear rare in the catch, and the magnitude of SSF impacts on these species in the SWIO is likely to be limited.

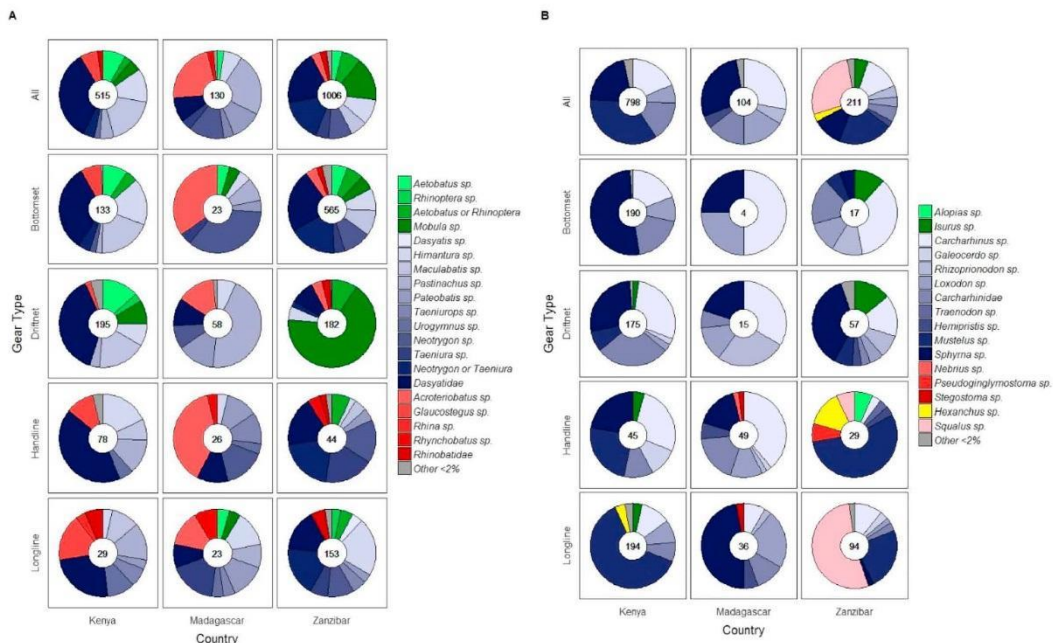


Fig. 3. Elasmobranch frequency of occurrence at the genus and family level displayed by country and gear type for A) batoids and B) sharks respectively. Sample size is displayed in the centre of each chart.

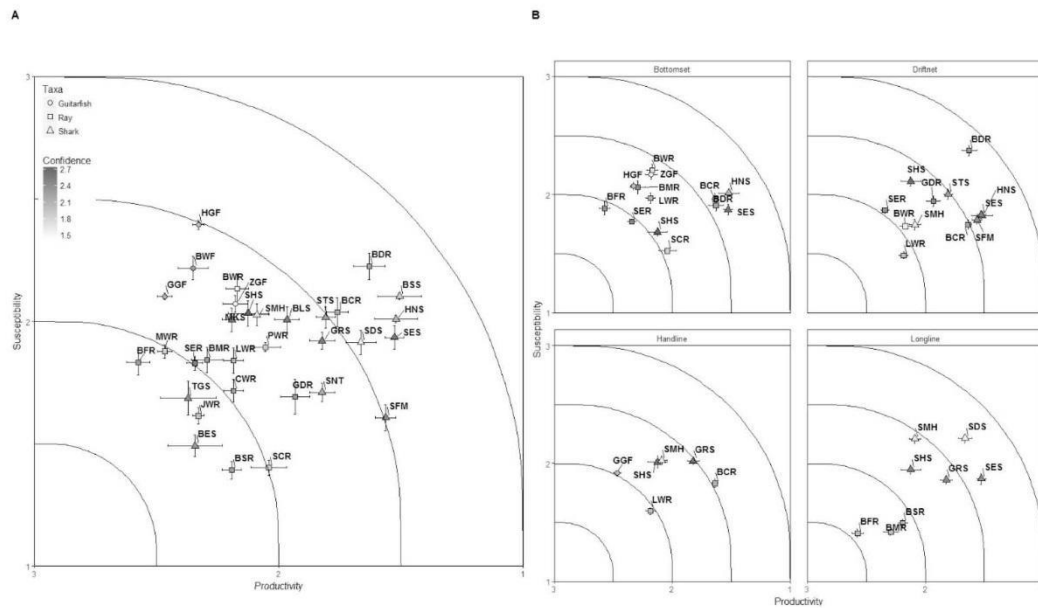


Fig. 4. Productivity-Susceptibility Assessments for A) combined fishing gears, and B) individual fishing gears. Guitarfish: BWF, *Rhynchobatus australiae*; GGF, *Acroteriobatus leucospilus*; HGF, *Glaucoctegus halavi*; ZGF, *Acroteriobatus zanzibarensis*. Rays: BWR, *Maculabatis ambigua*; BDR, *Mobula thurstoni*; BSR, *Taenirops meyeri*; BFR, *Taeniura lymna*; BMR, *Neotrygon caeruleopunctata*; BSR, *Pastinachus ater*; CWR, *Himantura uarnak*; GDR, *Mobula mobular*; JWR, *Pateobatis jenkinsii*; LWR, *Himantura leoparda*; MWR, *Urogymnus granulatus*; PWR, *Pateobatis fai*; SCR, *Rhinoptera jayakari*; SER, *Aetobatus ocellatus*. Sharks: BES, *Hexanchus nakamurai*; BLS, *Carcharhinus leucas*; BSS, *Carcharhinus humani*; GRS, *Carcharhinus amblyrhynchus*; HNS, *Carcharhinus macroti*; MKS, *Rhizoprionodon acutus*; SHS, *Sphyrna lewini*; SFM, *Isurus paucus*; SES, *Loxodon macrorhinus*; SMH, *Mustelus sp.*; SNT, *Hemipristis elongata*; STS, *Carcharhinus sorrah*; SDS, *Squalus spp.*; TGS, *Galeocerdo cuvier*.

The PSA assessments (Fig. 4) for a total of 32 species give initial insight into relative species vulnerability across and within gear type. The overall assessments indicate the most vulnerable batoids to be: *M. thurstoni*, broad cowtail ray (*Pastinachus ater*), halavi guitarfish (*Glaucoctegus halavi*), Zanzibar guitarfish (*Acroteriobatus zanzibarensis*) and *M. ambigua*; and the most vulnerable sharks to be *C. macroti*, blackspot (*Carcharhinus humani*), *L. macrorhinus*, *Squalus* spp. and *C. sorrah*.

4. Discussion

This study presents the first independent estimates of elasmobranch landings in SWIO SSF handline, longline, bottom-set and drift gillnet gears. Despite the study covering only the aforementioned gear types, landings estimates are 72.6% higher than the cumulative total of 20,547 t reported by SWIO nations (which included large-scale fisheries) to the FAO in 2016, 129.2% more than the 10 year average of 15,468 t (2006–16), and 109.4% higher than the 16,928 t estimated in catch reconstructions for SWIO SSF in 2010 (FAO, 2018; Pauly and Zeller, 2015). Further, this estimate does not account for the landings (number of individuals) missed by observers at study sites, which were potentially substantial (weighted means between 17.3 and 33.9%). Overall, the results clearly demonstrate that the landings of elasmobranch species originating from SSF are likely to be substantially underrepresented in fisheries statistics outputs from the SWIO region.

The level of underreporting was variable at the country level. In Kenya, annual elasmobranch landings were estimated at 80.7% higher than their 10 year average of 327.5 t (2006–16) (FAO, 2018). Conversely, landings in Zanzibar are more similar to reported FAO data, with an estimate at 18.1% higher than the 10 year average of 1,631.6 t (2006–16) (FAO, 2018) which falls well within the 95% confidence

intervals of the study estimate. Such differences reflect the varied efficacy of official landings data collection programmes among SWIO nations. Identifying and ameliorating low-efficacy observation programs is a clear priority if data derived from them is to be relied upon for evidence-based management, not only of elasmobranch resources but SSF resources as a whole. Other priority countries in the SWIO for such work include Madagascar and Mozambique, given their size and the relative disparity between elasmobranch landing declarations and the size of their SSF fleets (Chacate and Mutombene, 2016; FAO, 2018; WIOFish, 2018).

Previous studies have given some insight into elasmobranch composition of SWIO SSF, but are heavily biased towards the shark component (Temple et al., 2018). These studies suggest that the shark landings were comprised mainly of coastal and coral reef associated species such as *C. amblyrhynchus*, blacktip reef (*Carcharhinus melanopterus*), *T. obesus*, *Sphyrna* spp. and *G. cuvier* (e.g. Cooke, 1997; Kiszka et al., 2010; Maoulida et al., 2009; Robinson and Sauer, 2013). Though, particularly in the oceanic island nations, oceanic species such as blue shark (*Prionace glauca*) and *C. falciformis* also appear to be common (Kiszka et al., 2010; Maoulida et al., 2009; Soilihi, 2014). Pre-existing batoid compositional data often list *Mobula* spp., *Aetobatus* spp., large wedgefish (Rhinoidea) and sawfish (*Pristis* spp.) (e.g. Cooke, 1997; Heinrichs et al., 2011; Poonian, 2015). This study, alongside other recent works (Barrowclift et al., 2017; Robinson and Sauer, 2013), clearly demonstrates that much of the pre-existing data is heavily biased by ease of identification and memorability. The vast majority of landings are composed of species that are either not easy to discern for untrained observers from more iconic species (e.g. *C. sorrah*) are more numerous in the catch than *C. melanopterus*, or smaller species (e.g. *N. caeruleopunctata*, *T. lymna*, *Mustelus* spp., *Squalus* spp. and *L. macrorhinus*).

Further, the range of species impacted, from coastal, oceanic and deep-sea habitats demonstrate the potential for SSF to have impacts across a wide-range of ecosystems. It is clear that historically data for SWIO SSF elasmobranch catch composition is deficient. Whilst this study contributes substantially to our understanding, further improvement of compositional understanding is a clear priority in facilitating risk-assessment and sustainable management of elasmobranchs.

This study also demonstrates that despite receiving limited attention by SWIO SSF research (Temple et al., 2018), batoids are a significant and sometimes dominant portion of the elasmobranch landings. Batoids represented 43.3%, 30.7% and 74.7% by number, and 45.4%, 38.9% and 82.1% by weight of the landings in Kenya, northern Madagascar and Zanzibar, respectively. Batoids have a particularly high representation in Zanzibar, which may reflect fishing practices, market demand or the suspected decline and partial collapse of shark stocks (Barrowclift et al., 2017; Jiddawi and Shehe, 1999). The high level of batoid landings, combined with limited understanding of the ecology and life history of many of the species recorded, demonstrate a need to allocate research efforts to document life history parameters for this taxa.

Vulnerability assessment of the species identified in the study suggest that many of the large elasmobranch species (e.g. *G. cuvier*, *C. leucas*, *Sphyrna* spp., Rhinidae and *Mobula* spp.) achieve only moderate vulnerability scores. Instead many species that have gone unreported in previous works are identified as most vulnerable to SSF impacts. For sharks, highest vulnerability was identified for small coastal and continental shelf species (*C. malcoti*, *C. humanii*, *L. macrorhinus* and *C. sorrah*) and, surprisingly, deeper-water sharks (*Squalus* spp.). High vulnerability scores for sharks were observed in gillnets and longlines, with drift gillnets appearing to also threaten large oceanic species such as *I. oxyrinchus* and *Sphyrna* spp. In batoids, high vulnerability was identified in a number of geographically-restricted coastal species, including *G. halavi*, *A. zanzibarensis*, the recently described *M. ambigua* (Last et al., 2016) and *P. ater*. Unlike sharks, batoids appear primarily threatened by bottom-set gillnets, which dominate the batoid landings as a whole. Additionally, *M. thurstoni* was identified as the most vulnerable batoid, threatened by both bottom-set and drift gillnets. This finding is of particular concern given that *Mobula* spp. are already thought to be in steep decline in parts of the SWIO (Rohner et al., 2017) and in the Indian Ocean at large (Walls et al., 2016). These unexpected vulnerability outputs further highlight the need to properly assess the catch composition of SSF, rather than relying on declarations of species composition, if catches are to be made sustainable through evidenced-based management in the long-term.

Whilst the study is not able to provide an estimate for the catch of marine mammals and sea turtles in SWIO SSF, it does reinforce the threats presented to both taxa by both bottom-set and drift gillnets across the region. Both species of delphinid (*S. longirostris* and *T. aduncus*) caught during this study have been reported or implicated in gillnet fisheries historically in the region (e.g. Amir et al., 2002; Pusineri and Quillard, 2008; Razafindrakoto et al., 2008), as have both commonly caught species of sea turtle (*E. imbricata* and *C. mydas*) (Okemwa et al., 2004). *E. imbricata* and *C. mydas* are the most common species in the tropical SWIO (Bourjea, 2015) and represented 60.3% and 38.5% of catch in this study, respectively. Bottom-set gillnets dominate sea-turtle captures (90.2%). However, the majority of landings (87.8%) were reported from one site (Watamu, Kenya), biasing the compositional results. Therefore, sea turtle composition presented here may not be representative of composition across SWIO SSF. The high level of reporting from this site is likely a result of the long-term collaboration, and thus trust, between fishers and one of the study partners. Further, in the few cases where usage data for sea turtle and dolphin landings were obtained their respective sales for human consumption and fisheries bait suggest that there continues to be notable markets for these taxa in spite of the illegality of their capture. Additionally, a single dugong (*Dugong dugon*) is known to have been

landed during the study period in a drift gillnet in Msambweni, Kenya. Whilst not one of the sites monitored, the incident does highlight the ongoing threat from gillnet gears to the relict populations of this species in the SWIO region (Muir and Kiszka, 2012).

Though this study has begun to address the gap in understanding of marine megafauna catch it is important to acknowledge its limitations. Substantial heterogeneity in composition and catch rates was seen both among countries and among sites within countries. This heterogeneity may result from a large number of factors influencing fishing selectivity and species availability, including variability in gear specifications and fishing methods, exploited habitats and ecosystems and historical exploitation. This suggests that a larger number of sampling sites are required to ensure that outputs are representative. Further, it is important to recognise the likely biases of the methodology. Both marine mammal and sea turtle catch are universally prohibited across the study area (Temple et al., 2018), and so the likelihood of these catches being declared is low, inevitably leading to substantial underestimates of their catch. Identification bias will have an effect on species composition data presented here. This is primarily driven by the variable difficulty in identifying species from images of varying quality. Further, where appropriate we identified some landings by the local names given. Both elements will result in overrepresentation of distinctive species, such as *Sphyrna* spp. and *Mobula* spp., which are easier to identify and often have specific local names. The methodology also likely under-samples smaller elasmobranch species which are often stored and transported mixed with various fish and other landings, making smaller elasmobranchs less likely to be recorded by observers. Lastly, because landings were monitored on a representative sub-sample of days from those available, there is a risk that landings of rare but highly vulnerable species, such as sawfish (*Pristis* spp.), may have been missed. Yet, even limited catches may be highly significant to the long-term sustainability of highly vulnerable species. Conversely, catch discards in SWIO SSF are thought to be relatively low and so likely have limited influence on the overall results. Additionally, none of the sites in this study are known to operate fin-and-discard practices which would lead to underrepresentation of large shark and wedgefish landings.

The outcomes of this study clearly show the potential effects from SSF to a diverse range of coastal, oceanic, and even deep-water marine megafauna species, reinforcing SSFs potential to impact across multiple ecosystems. We provide the first cross-sectional assessment of marine megafauna catch and composition within SWIO SSF and demonstrate the underreporting of catch and the overrepresentation of large, iconic species in most existing assessments. Indeed, in many cases we show that historically under-represented species are likely those at most immediate threat from SSF. Further, we reinforce the cross-taxa threats posed by gillnet gears, and note their particular proclivity for impacting iconic species with implications for the growing ecotourism activities (Gallagher and Hammerschlag, 2011; O'Connor et al., 2009; O'Malley et al., 2013) in this and other regions. However, we recognise that this study represents a limited single-year time scale and, within the context of the SWIO, a limited geographic range. Thus, there is a clear need for further work in other areas and over longer time periods in order to improve assessments at the SWIO scale and inform evidence-based management of SWIO SSF. Similarly, this study focussed primarily on gear types which are thought to pose the greatest threat to marine megafauna and as such may overlook the impacts from other widespread gears such as purse and beach seines, these gears should be considered in future works. However, what is clear is that for the future sustainability of marine megafauna resources, further focus must be placed on the dominant but often overlooked SSF. Researchers and managers must face the challenges of working in SSF head-on, rather than seeking the relative comfort of the industrialised sectors, if touted aims of sustainability are to be met.

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.biocon.2018.12.024>.

Annex 8: Life-history, exploitation and relative extinction risk of Baraka's whiplay (*Maculabatis ambigua*) in small-scale fisheries

Life-history, exploitation and relative extinction risk of Baraka's whiplay (*Maculabatis ambigua*) in small-scale fisheries

Temple AJ, Jiddawi N, Wambiji N, Stead SM, Barrowclift E & Berggren P

Abstract

The recently described Baraka's whiplay (*Maculabatis ambigua*) is a major constituent of small-scale fisheries (SSF) catch in the southwestern Indian Ocean. Despite this, nothing is known of its life-history. This study investigated the life-history parameters of *M. ambigua* collected from SSF catch in Kenya, Zanzibar and northern Madagascar (n=171). Specimens were aged using vertebrae sagittal sectioning (n=47). The outputs represent the first estimates of disc width-weight relationships, growth models, maturation and longevity for this species, as well as demonstrating a wider distribution range than previously reported. From these data we derive the first plausible estimates for maximum intrinsic population growth rate r_{max} and these are combined with estimates of total Z , fisheries F and natural M mortality, derived from Chapman-Robson catch curve analysis, as well as estimation of the exploitation ratio E in order to assess relative extinction risk. The data indicate that *M. ambigua* is a moderately-sized, fast growing and early maturing species of whiptail stingray (Family Dasyatidae), with a relatively long lifespan. This life-history results in a high r_{max} relative to many other elasmobranch species which, in conjunction with estimates of F and E , suggest that the current fishery is likely sustainable at near optimal exploitation. However, data is lacking with regard to fecundity (litter size and inter-birth interval) and this information is highlighted as a priority in order to improve confidence in future assessments. Further, full recruitment to the fishery occurs post-maturation and fisheries exploitation occurs across broad age and size classes which raises concern for the long-term sustainability of these fisheries if fishing pressure were to increase.

Key Words

Life History; Batoid; Indian Ocean; Elasmobranch; r_{max} ; Catch curve

Introduction

Relative to teleosts, elasmobranchs (sharks and rays) generally display slow growth, late maturity and low fecundity (Compagno, 1990). These life-history traits exacerbate their vulnerability to non-natural mortalities and limit their recovery potential (Žydelis *et al.*, 2009; Dulvy *et al.*, 2014a). However, there is considerable variation in the life-history traits both among species (Stevens and McLoughlin, 1991; Jacobsen and Bennett, 2011) and within species (Lombardi-Carlson *et al.*, 2003; Jacobsen and Bennett, 2010; O'Shea *et al.*, 2013). Fisheries are the most prominent source of non-natural mortalities for elasmobranchs at the global level (Worm *et al.*, 2013; Dulvy *et al.*, 2014a). Understanding species and stock-specific life-history traits is therefore important when assessing the sustainability of fisheries exploitation, conducting accurate stock assessment, producing demographics models and predicting rebound potential (Frisk *et al.*, 2001; Cailliet and Goldman, 2004; Smith *et al.*, 2008). Thus, species and stock specific life-history traits are pivotal in the formulation of evidence-based fisheries management.

Formulating management plans in data-poor fisheries without species-specific life-history and fisheries exploitation data risk being ineffective or even detrimental to long-term sustainability of elasmobranchs. Throughout the southwestern Indian Ocean (SWIO) region (here consisting of Kenya, Tanzania – including Zanzibar, Mozambique, Seychelles, Comoros, Mayotte, Madagascar, La Réunion and Mauritius) efforts to manage shark resources have been initiated (e.g. widespread development of national plans of action for sharks) (Temple *et al.*, 2018). Yet, understanding of the scale and composition of elasmobranchs exploited in SWIO fisheries is extremely poor. Recent vulnerability assessments based on small-scale fisheries (SSF) landings across the SWIO suggest that a number of coastal rays, primarily whiptail stingrays (Family Dasyatidae), are potentially at risk from SWIO SSF (Temple *et al.*, 2019). Many of these species have either limited or no regional life-history data available. Rays contribute nearly half of SWIO SSF landed elasmobranch catch by weight and number and originate from many of the same fisheries as sharks, though there is substantial geographical heterogeneity (Temple *et al.*, 2019). Despite this, rays have received little consideration in SWIO elasmobranch management formulation, potentially jeopardising the long-term survival of these species.

Despite having only been recently described (Last *et al.*, 2016b), Baraka's whipray (*Maculabatis ambigua*) is the dominant ray in Kenyan SSF catch, and a common constituent of SSF catch in Zanzibar. *M. ambigua* are caught primarily in bottom-set gillnets (Barrowclift *et al.*, 2017; Temple *et al.*, 2019) and appear commonly as bycatch in trawl fisheries in other areas within their range, such as the Red Sea (Last *et al.*, 2016b). *M. ambigua* is believed to be distributed from Zanzibar to the Red Sea and possibly further into the northern Indian Ocean (Last *et al.*, 2016b). Little is known of the life-history of this species. In this study, we investigate aspects of the life-history of *M. ambigua* through the production of disc width-weight relationships, age structured growth models, maturation and longevity. Subsequently, we produce plausible estimates for maximum intrinsic population growth rate, which are compared to instantaneous fisheries mortality of females in order to assess relative extinction risk for this species.

Methods

Sample Collection

M. ambigua (n=47) were sampled from bottom-set gillnet catches, with the consent of fishers and/or merchants, at the village of Mkokotoni (n=30) and the central Darajani Market (unknown gear type used to capture specimens), Stone Town, (n=17) in Zanzibar between 28/07/2015 and 19/08/2015 (Figure 1). Disc width (cm), weight (kg) and sex (females=29, males=18) were recorded. Where the specimen had already been gutted and the majority of internal organs removed, weight was not recorded (n=28). Maturity status of males (immature or mature) was recorded based on calcification of claspers (Walker, 2005), with only those specimens exhibiting complete calcification considered as mature. Maturity status was not recorded for females as fishers and merchants did not consent to examination of the reproductive tract. Vertebrae were extracted from the mid-disc for all specimens to be used for aging (n=47) and stored at -20°C until sectioning. Samples were collected under research permit from the Office of Chief Government Statistician Zanzibar (No. 0697), exported under the provision of the then Ministry of Livestock and Fisheries Zanzibar and imported to the UK under IMP/GEN/2014/06.

Supplementary data for *M. ambigua* were recorded by trained fisheries observers at 21 sites across Kenya, Zanzibar and northern Madagascar during a 12-month landings monitoring programme between June 2016 and June 2017 (Temple *et al.*, 2019). *M. ambigua* (n=134) were recorded at four sites in Kenya, five in Zanzibar and one in northern Madagascar (Figure 1). Disc width (n=127), weight (n=76), sex (females=55, males=71, unsexed=8) and maturity status for males (n=67) were recorded. Observers also opportunistically recorded maturity for females, recording only those observer with fertilised eggs as mature (n=3). These specimens were caught across a range of gear types (bottom-set gillnets=80, drift gillnets=37, handlines=8, longlines=6 and unknown=3). Sampling was carried out by relevant in-country institutions and did not require specific permits.

In the absence of data relating to stock delineation, analyses in this study assume that all catches originate from the same fisheries stock. Specimens where disc width was not recorded were disregarded from analyses. Data analyses and visualisations were carried out and produced using R statistical software, version x64 3.4.0 (R Core Team, 2017). Ethical consent was granted by Newcastle University's animal welfare ethics review board.

Disc width-weight relationship

Disc width and weight data were natural log (ln) transformed and their relationship modelled using linear regression. Sex was included as an interaction variable and compared to the null model using ANOVA to determine whether the effect of sex was significant. Cook's distance ($4/n$) was used to identify data points exerting undue influence on the linear models, likely resulting from measurement and/or data entry errors, these points were removed and the models subsequently re-run.

Age estimation

Two vertebrae from each sampled specimen (n=47) were cleaned of excess muscle and connective tissue and both neural and haemal arches were removed. Subsequently, vertebrae were immersed in a 5% sodium hydrochloride solution for 10-30min, dependent on the vertebrae size and quantity of remaining tissues. Samples were then immersed in water, towel and air dried. Cleaned vertebrae were embedded in clear epoxy resin (Buehler EpoxiCure). A single sagittal-plane section was taken from each vertebrae using a slow-speed precision saw with a diamond wafering blade (Buhler IsoMet Low Speed Precision Cutter). Several section widths were initially trialled (600µm, 450µm, 300µm, 200µm and 150µm), with 200µm producing highest readability. Sections were mounted permanently onto glass slides using DPX mounting medium (Fisher Chemical DPX Phthalate Free Mounting Media) and photographed using a high quality digital macro-lens camera (Nikon SLR D7200). Image enhancement for growth band reading was carried out in Adobe Photoshop CS3 (Campana, 2014).

Age (to the nearest 0.5 years) for each individual specimen was determined by examination of paired opaque and translucent banding in the *corpus calcareum*, with the half year determined as the presence of a translucent band after the final band pair. The birth band was distinct but did not show a clear change in angle on the *corpus calcareum* (Figure 2). Ages were estimated by the independent examination of both sagittal-sections (images were randomised before reading) by two independent readers. Mean age estimates for each specimen were generated for each reader and compared. Where reader mean estimates differed by <1 year the mean was taken as the best estimate of age. This allowable difference was more conservative than in other studies (Smith *et al.*, 2008;

Jacobsen and Bennett, 2011) in light of the restricted sample size. If differences in mean age estimates between readers were >1 year ages were re-estimated with both readers present. If readers could not agree (n=0) then samples would have been discarded (Goldman, 2005). Commonly, measures of precision in agreement and bias in reads, both within (individual vertebrae estimates) and among (mean age estimates) readers, are given as Percentage Agreement (PA), PA $\pm$ 1 year, the Coefficient of Variation (CV) and the Average Percentage Error (APE) (Beamish and Fournier, 1981; Chang, 1982; Goldman, 2005; Cailliet *et al.*, 2006). These measures are presented, however they are commonly recognised as imperfect (Goldman, 2005; Cailliet *et al.*, 2006). The authors argue that the Bland-Altman approach (Bland and Altman, 1999; Bland and Altman, 2003), designed for method-comparison, provides improved quantification and visualisation of agreement, precision and bias among reads and readers compared to the standard methods used in aging studies. Potential bias in the relationship between reads (within and among readers) is assessed through linear regression of the mean age read for each specimen against the difference between reads for each specimen, with significance indicating bias. Precision in age reads (within and among readers) is described by the Limits of Agreement (LOA) defined by the 95% mean confidence interval of the difference between readers. Thus, we apply and display the results of the Bland-Altman method as the primary measure of agreement, precision and bias.

Validation of growth band periodicity was not possible within this study. The short temporal period (two months) within which samples were collected and the restricted sample sizes meant that both marginal incremental analysis and edge analysis, which are the most common validation methods for elasmobranchs (Cailliet *et al.*, 2006), were not possible to conduct. Further, mark-recapture of chemically marked or captive reared individuals were not feasible within the constraints of this study.

Growth modelling

Age and disc width data were fit using three separate growth models using non-linear regression (R package *nls*). Starting values for respective growth function parameters were estimated before beginning growth curve fitting (R packages *FSAtools* and *stats*). Given the restricted sample size, models were run for males and females both separately and combined. Model selection was made through comparison of Akaike's Information Criterion (AIC) and realism in parameter estimates. The 95% mean confidence intervals (95%CI) for growth curves were derived via bootstrapping with 1,000 iterations. The models used were:

the three parameter von Bertalanffy growth function (Von Bertalanffy 1938),

$$DW_t = DW_{\infty} - ((DW_{\infty} - DW_0)e^{(-kt)})$$

the three parameter Gompertz growth function (Ricker 1975),

$$DW_t = DW_{\infty}e^{e^{-k(t-\alpha)}}$$

and the Logistic growth function (Ricker 1979),

$$DW_t = \frac{DW_{\infty}}{1 + e^{-k(t-\alpha)}}$$

where DW_t is the disc width at age t , DW_{∞} is the asymptotic disc width, DW_0 is disc width at age zero, k is a growth constant and α is the inflection point.

Estimating maximum intrinsic population growth rate

The maximum intrinsic population growth rate r_{max} , for female *M. ambigua* was estimated using the simplified Euler-Lotka equation, accounting for juvenile mortality (Cortés, 2016; Pardo *et al.*, 2016a; Pardo *et al.*, 2016b):

$$l_{t_{mat}} b = e^{r_{max}} - e^{-M}(e^{r_{max}})^{t_{mat}-1}$$

where $l_{t_{mat}}$ is survival to maturity, b is the annual reproductive output of female offspring, t_{mat} is age at maturity and M is the instantaneous rate of natural mortality. $l_{t_{mat}}$ is calculated as

$$l_{t_{mat}} = (e^{-M})^{t_{mat}}$$

There is no data for the annual reproductive output of *M. ambigua*, thus b is instead estimated from related species. *M. gerradi* and *M. astra* litters are reported as 1-4 (White *et al.*, 2006) and 1-3 (Last *et al.*, 2016a) pups respectively, and reproduction is assumed to be annual. Assuming a 1:1 sex ratio, we estimate the range of b as 0.5-2 for the purposes of estimating r_{max} .

t_{mat} cannot be estimated directly for female *M. ambigua* because observations of female maturity were both opportunistic and reliant upon the presence of fertilised eggs. However, results from other studies of whiptail stingrays suggest that males and females mature at similar size classes (White, 2007; Jacobsen and Bennett, 2010; Jacobsen and Bennett, 2011; Da Silva *et al.*, 2018). As such, we estimate size at maturity using male maturity data. Disc width ($\pm 95\%$ CI) at which 50% of males reach maturity (White, 2007) were estimated from logistic regression and bootstrapping with 1,000 iterations used to define confidence intervals. This size range was then converted into age using the disc width and age relationship from the selected growth model for females, thus providing a range estimate for t_{mat} . Disc width ($\pm 95\%$ CI) at which 95% of males reached maturity was also estimated.

Lastly, in order to estimate M we first need to estimate longevity, t_{max} . Estimates for t_{max} were bound by the oldest female specimen aged in this study (17 years) and the theoretical age at which 95% DW_{∞} or 99% DW_{∞} is reached (Fabens, 1965; Ricker, 1979) dependant on how these results compare to the maximum age observed. 95% DW_{∞} is calculated as $5\ln(2)k^{-1}$ and 99% DW_{∞} calculated as $7\ln(2)k^{-1}$. Using these limits a median and range of M was calculated as

$$M = \left(\frac{t_{max} - t_{mat}}{2}\right)^{-1}$$

The simplified Euler-Lotka equation, solving for r_{max} , was then optimised using the *nlinb* function (R package *stats*). 10,000 data sets for optimisation were drawn from the estimated ranges of b , t_{mat} , and M , assuming a uniform probability distribution within these ranges in order to explore the full range of potential estimates for r_{max} , particularly given the uncertainty in these parameters (Dulvy *et al.*, 2014b).

Estimating total mortality, fisheries mortality and the exploitation ratio

In order to estimate instantaneous fisheries mortality F for female *M. ambigua* we first estimated total instantaneous mortality Z ($\pm 95\%$ CI), using the Chapman-Robson catch curve method (R package *FSA*). The analysis was carried out treating all SSF monitored as if they were one fishery. The Chapman-Robson method was selected as the regression estimator because traditional catch curves may show negative bias in mortality estimation (Smith *et*

al., 2012). Age was estimated for all recorded catches with disc width data ($n=127$) from the 12-month landings monitoring programme using the best fitting growth model. Age classes beyond estimated t_{max} were excluded from the catch curve. Age of full recruitment to the fishery was determined as the age with peak abundance and Z was estimated from one year after the age of peak abundance (Smith *et al.*, 2012). From the estimates range of Z we subtract M in order to produce an estimate of F . F is subsequently compared to r_{max} and the exploitation rate E , calculated as F/Z , is compared to its optimum value of 0.5 (Gulland, 1971; Pauly, 1983). Subsequently the likely sustainability status of *M. ambigua* is inferred.

Results

Disc width-weight relationship

ANOVA comparison of models with and without sex as an interaction term provide evidence for a moderating effect of sex on the disc width-weight relationship (ANOVA, $F=3.212$, $p=0.043$). Thus, separate disc width-weight relationships were modelled for males and females. Linear regression models describing the disc width-weight relationship for females ($R^2=0.882$, $F=531$, $p<0.001$), $\ln(\text{weight})=2.62*\ln(\text{disc width})-8.88$, and males ($R^2=0.705$, $F=186$, $p<0.001$), $\ln(\text{weight})=2.42*\ln(\text{disc width})-7.96$, were found and presented (Figure 3A).

Age estimation

Age estimates were successfully made for all 47 specimens used in this study. Bland-Altman analysis of agreement, precision and bias within and among readers showed no evidence of bias within reader 1 ($R^2=0.038$, $F=1.78$, $p=0.189$; Figure 4A, Figure 4B) or reader 2 ($R^2=0.001$, $F=0.494$, $p=0.825$, Figure 4C, Figure 4D), but evidence of significant bias between readers ($R^2=0.118$, $F=6.04$, $p=0.018$, Figure 4E, Figure 4F), with reader 1 producing higher estimates than reader 2 for older specimens. LOAs, representing precision between age reads are presented alongside standard precision metrics (**Error! Reference source not found. 1**).

Growth modelling

Age estimations for specimens ranged from 0-12.5 years for males and 0-17 for females. Of the growth models tested the three parameter von Bertalanffy was found to provide best fit for females, males and combined female and male data (Table 2; Figure 3B). The von Bertalanffy growth parameter estimates for females were; $DW_{\infty}=89.4(95\%CI\ 82.2-100.9)\text{cm}$, $DW_0=26.1(95\%CI\ 15.9-36.6)\text{cm}$ and $k=0.28(95\%CI\ 0.17-0.41)$. The von Bertalanffy growth parameter estimates for males were; $DW_{\infty}=139.7(95\%CI\ 84.1-598.0)\text{cm}$, $DW_0=40.9(95\%CI\ 27.4-50.1)\text{cm}$ and $k=0.08(95\%CI\ 0.01-0.43)$. The von Bertalanffy growth parameter estimates for combined females and males were; $DW_{\infty}=92.2(95\%CI\ 82.9-113.4)\text{cm}$, $DW_0=33.6(95\%CI\ 25.4-42.5)\text{cm}$ and $k=0.23(95\%CI\ 0.12-0.38)$. The wide confidence intervals for parameters in the male growth curve clearly demonstrate a need for greater sample size in order to stabilise the male age-growth model. As such, estimations of life-history parameters for males are instead made conservatively from the combined female and male growth curve.

Estimating maximum intrinsic population growth rate

Disc width at which 50% and 95% of males reach maturity was estimated at 54.5(95%CI 49.0-60.0)cm and 67.5(95%CI 49.2-73.1)cm, respectively (Figure 3C). These estimates equate to 50% of males reaching maturity at 1.9(95%CI 1.3-2.6) years, and 95% of males reaching maturity at 3.8(95%CI 1.3-4.9) years. The smallest of the female specimens classified as mature ($n=3$) had a disc width of 62cm, equivalent to an age of 3.0 years. If it is assumed that females mature at similar size ranges as males then the approximate age ($\pm 95\%$ CI) at which 50% and 95% of females reach maturity would be 2.1(1.6-2.7) years and 3.7(1.6-4.8) years, respectively. For the purpose of estimating r_{max} we estimate the range of female t_{mat} to be 1.6-2.7 years. t_{max} estimates based on age at 95% DW_{∞} (Ricker, 1979) were less than those ages observed in this study at 12.4 years for females and 15.1 years for males. However, longevity based on age at 99% DW_{∞} (Fabens, 1965) were estimated at 17.3 years for females and 21.1 years for males. Given that longevity estimates at 95% DW_{∞} were less than ages observed in this study, those based on 99% DW_{∞} were considered to be the best estimates. For the purpose of estimating r_{max} we estimate the range of female t_{max} to be 17-17.3 years. Using the estimates of t_{max} and t_{mat} we calculate M for *M. ambigua* to be 0.133 (95th percentiles=0.128-0.139). Using the estimates for b , t_{max} and t_{mat} we calculate the r_{max} for *M. ambigua* to be 0.410 (95th percentiles=0.210-0.622).

Estimating total mortality, fisheries mortality and the exploitation ratio

Age at full recruitment to the fishery is estimated at three years from the Chapman-Robson catch curve (Figure 5) and provides an estimate for Z of 0.265(95%CI 0.202-0.328), equivalent to an annual mortality rate of 23.3(95%CI 18.3-27.9)%. Given this and the estimate for M , we estimate F to be 0.132 (95th percentiles=0.074-0.189), considerably lower than estimates for r_{max} . E is estimated from F and Z at 0.498, approximately similar to the optimal value of $E=0.5$ (Gulland, 1971; Pauly, 1983). It must be noted that the data used in the current assessment likely violates a number of the assumptions of the catch curve and thus the outputs must be treated with a level of caution. Whilst there is no known reason, such as migration, that age classes of *M. ambigua* over the age at full recruitment would become unavailable SWIO SSF, different age/size classes are unlikely to be equally vulnerable to the fisheries as the catch curve presented here combines data across several gear types, which will not represent equable selectivity. Further, we cannot be certain that the assumptions of constant mortality across age/size classes and of a closed population are met.

Discussion

This study provides the first estimates of life-history parameters for the recently described *M. ambigua*. The findings suggest that *M. ambigua* is a moderately-sized, fast growing and early maturing species of whiptail stingray that exhibits a moderately long lifespan, relative to other whiptail stingrays. The species is therefore potentially more resilient to fisheries pressure than many other widely exploited whiptail stingrays. This resilience is reflected in the relatively high r_{max} (with r_{max} being equivalent to the fisheries mortality rate that would lead to extinction F_{ext}). The disparity between r_{max} and F estimates and E being approximately similar to its optimal value indicate that *M. ambigua* catch in SWIO SSF can likely be considered among the sustainable elasmobranch fisheries at the global level (Simpfendorfer and Dulvy, 2017) at this time, despite prominent and widespread

exploitation in SWIO SSF (Temple *et al.*, 2019). The study also demonstrates a larger maximum size, with a maximum recorded disc width of 112cm, and wider distribution than previously known (Last *et al.*, 2016b), as evidenced by the presence of *M. ambigua* in the SSF catch of northern Madagascar.

Agreement between age readers was relatively low (APE=17.07%, CV=24.14% and PA±1=68.09%) in comparison with some (e.g. Jacobsen and Bennett, 2010, APE=7.1%, CV=10.1%), but not all (e.g. Gutteridge *et al.*, 2013, APE=17.1%, CV=24.2%, PA±1=85.1%) similar studies. Agreement was likely influenced in part by presence of substantial sub-annuli banding which was prominent in young specimens. Further, the authors agree that traditional metrics to compare readers are imperfect (Goldman, 2005; Cailliet *et al.*, 2006) and do not adequately reflect the potentially variable nature of agreement among and within readers across age classes. We therefore also presented comparison among and within readers using the Bland-Altman approach (Bland and Altman, 1999; Bland and Altman, 2003). The outputs of the Bland-Altman approach demonstrated no evidence for an increase or decrease in discrepancies between reads with increasing age, indicating consistency in the variability within and among readers and thus increasing confidence in the validity of band reads across age-spectra. Further, the LOA estimate from the Bland-Altman approach provides a more interpretable metric of precision which, we believe, better encapsulates the variability among reads. The comparison does however reveal a significant bias among readers, with reader 1 estimating higher band counts for older specimens. We recommend that future aging studies consider the use of this method when presenting the results of within and among reader agreement, precision and bias in band counts.

As validation of banding periodicity was not possible within the scope of this study, the results presented assume that the banding observed in *M. ambigua* are deposited on a consistent, annual and continual basis. This was considered a reasonable assumption given that annual band deposition in whiptail stingrays has been validated (Cowley, 1997; Smith *et al.*, 2007; Jacobsen and Bennett, 2010; Pierce and Bennett, 2010). However, such assumptions may lead to misclassification of age in a number of elasmobranch species (Harry, 2018). Banding periodicity is increasingly seen to violate the consistent, annual and continual assumptions. Under or over estimation of age may result from violating these assumptions (Natanson and Cailliet, 1990; Kinney *et al.*, 2016; Harry, 2018), difficulty counting increasingly small bands in older animals, or even variability in band formation between vertebrae within the same individual (Natanson *et al.*, 2018). Whilst such issues have thus far been most common in large and longer-lived species, applications of the results from this and other studies must be carried out with these in mind.

Growth models from this study indicate that *M. ambigua* is a fast growing species with early maturation relative to other moderately sized whiptail stingrays. von Bertalanffy growth curve parameters for other similarly sized whiptail stingray species describe substantially lower growth constant; e.g. *Maculabatis astra* (male: $DW_{\infty}=72.3$, $k=0.03$, female: $DW_{\infty}=82.2$, $k=0.03$) (Jacobsen and Bennett, 2011), *Dasyatis chrysonota* (male: $DW_{\infty}=53.2$, $k=0.175$, female: $DW_{\infty}=91.3$, $k=0.07$) (Cowley, 1997), and *D. dipterura* (male: $DW_{\infty}=62.2$, $k=0.1$, female: $DW_{\infty}=92.4$, $k=0.05$) (Smith *et al.*, 2007). However, it must be considered that k may exhibit considerable variability, a product of the constraints of both the study, e.g. sample size and representation of size classes, and the growth-models used (Cailliet and Goldman, 2004; Smith *et al.*, 2007). This may result in wide variability of k even within

species, as demonstrated in studies of *Neotrygon kuhlii* where k ranges between 0.08 and 0.38 (Jacobsen and Bennett, 2010; O'Shea *et al.*, 2013). This rapid growth is re-enforced by the early maturation seen in males and females.

Whiptail stingrays, and *M. ambigua* in particular, display earlier maturation relative to both longevity and size compared to other elasmobranchs (Frisk *et al.*, 2001). Estimates from this study indicate that males mature at $\sim 9\% t_{max}$, in which time they achieve $\sim 59\% DW_{\infty}$. Further, males reached maturity earlier than other whiptail stingrays, e.g. *N. kuhlii* (3.95 years, $\sim 19\% t_{max}$) (Pierce and Bennett, 2010) and *M. astra* (7.32 years, $\sim 41\% t_{max}$) (Jacobsen and Bennett, 2011), and at similar relative size as other whiptail stingrays, e.g. *M. astra* ($\sim 65\% DW_{\infty}$) (Jacobsen and Bennett, 2011), *N. picta* ($\sim 64\% DW_{\infty}$) and *N. kuhlii* ($\sim 65\% DW_{\infty}$) (Jacobsen and Bennett, 2010) and *D. pastinaca* ($\sim 52\% DW_{\infty}$) (Yigin and Ismen, 2012). Female maturation could not be assessed directly in this study. However, working on the assumption that males and females mature at similar size classes (White, 2007; Jacobsen and Bennett, 2010; Jacobsen and Bennett, 2011; Da Silva *et al.*, 2018), estimates suggest they also mature relatively early at $\sim 12\% t_{max}$ and $\sim 61\% DW_{\infty}$. Greater sample sizes for both males and females, alongside thorough examination of female reproductive status at all size classes are clear priorities to improve current estimates of life-history traits for these species.

Relatively fast growth rates ($k > 0.1$) and early maturation (Branstetter, 1990; Musick, 1999; Frisk *et al.*, 2001) are generally associated with higher potential rates of population increase and thus higher rebound potentials (Frisk *et al.*, 2001). The early maturation of *M. ambigua* combined with a moderately long lifespan relative to other whiptail rays (Jacobsen and Bennett, 2011) suggests a high reproductive potential over the lifespan of *M. ambigua*. This life-history is reflected in the high r_{max} estimates presented here, relative to those of many other elasmobranchs (Dulvy *et al.*, 2014b). However, uncertainty remains around fecundity (litter size and inter-birth interval). The smallest specimens measured in this study and DW_0 estimated from the joint male and female growth model (23.5cm and 33.6cm disc width, respectively) are large relative to DW_{∞} estimates, suggesting high maternal investment in offspring and thus likely few offspring per litter. Given that fecundity is a critical life-history parameter, collation of this data (litter size and inter-birth interval) is therefore critical to improve confidence in future exploitation assessments of *M. ambigua* and other exploited elasmobranchs in SWIO and global fisheries.

Despite the relatively high resilience and likely sustainability of *M. ambigua* in SWIO SSF, the age and size class selectivity of the fisheries is of potential concern. Restricting elasmobranch fisheries to catches of non-adult age classes is considered an effective management strategy for the sustainability of these taxa (Simpfendorfer, 1999; Prince, 2002). Conversely, protection of sub and young adults may be optimal to maximise the future reproductive potential of the stock (Kindsvater *et al.*, 2016). Further, it is broadly accepted that fishing mortalities occurring across a wide range of age classes make age-specific catch management strategies for elasmobranchs difficult (Prince, 2002). Based on the Chapman-Robson catch curve, full selectivity to SWIO SSF is estimated around age class three. Assuming size-at-maturity estimates are approximately representative, the data suggest that recruitment to the fishery occurs primarily in post-maturation life-stages. Additionally, exploitation occurs across a broad range of age and size classes, indicative of a non-selective fishery. Whilst this is not unexpected given the multi-gear, multi-species, and highly diverse nature of these fisheries, the non-selective nature of catches raise concerns

for the future sustainability of both *M. ambigua* and numerous other elasmobranch species caught in these fisheries (Kiszka and van der Elst, 2015; Temple *et al.*, 2019) in the face of increasing regional fisheries pressure (Temple *et al.*, 2018).

The life-history, r_{max} , exploitation and mortality data information presented here for the recently described *M. ambigua* is an important step towards future comprehensive quantitative assessment of the stock status and vulnerability of this species. However, the limitations of the study also outline the priorities for future works. Greater sample sizes are required for males in particular in order to stabilise the current growth model. Further, improving our understanding of female maturation and fecundity is crucial in estimation of exploitation resilience. Lastly, life-history traits may vary significantly within species and between stocks (Lombardi-Carlson *et al.*, 2003; Jacobsen and Bennett, 2010; O'Shea *et al.*, 2013), yet nothing is known of the delineation, if any, of *M. ambigua* stocks within its range. This delineation is vital in defining management units for fisheries (Pita *et al.*, 2016), with potentially differing life-histories between stocks having implications for their resilience to fisheries exploitation and subsequent management needs. Given the prominence of *M. ambigua* in SWIO SSF (Temple *et al.*, 2019), investigation of stock disaggregation should be considered a priority next step in informing management of this species.

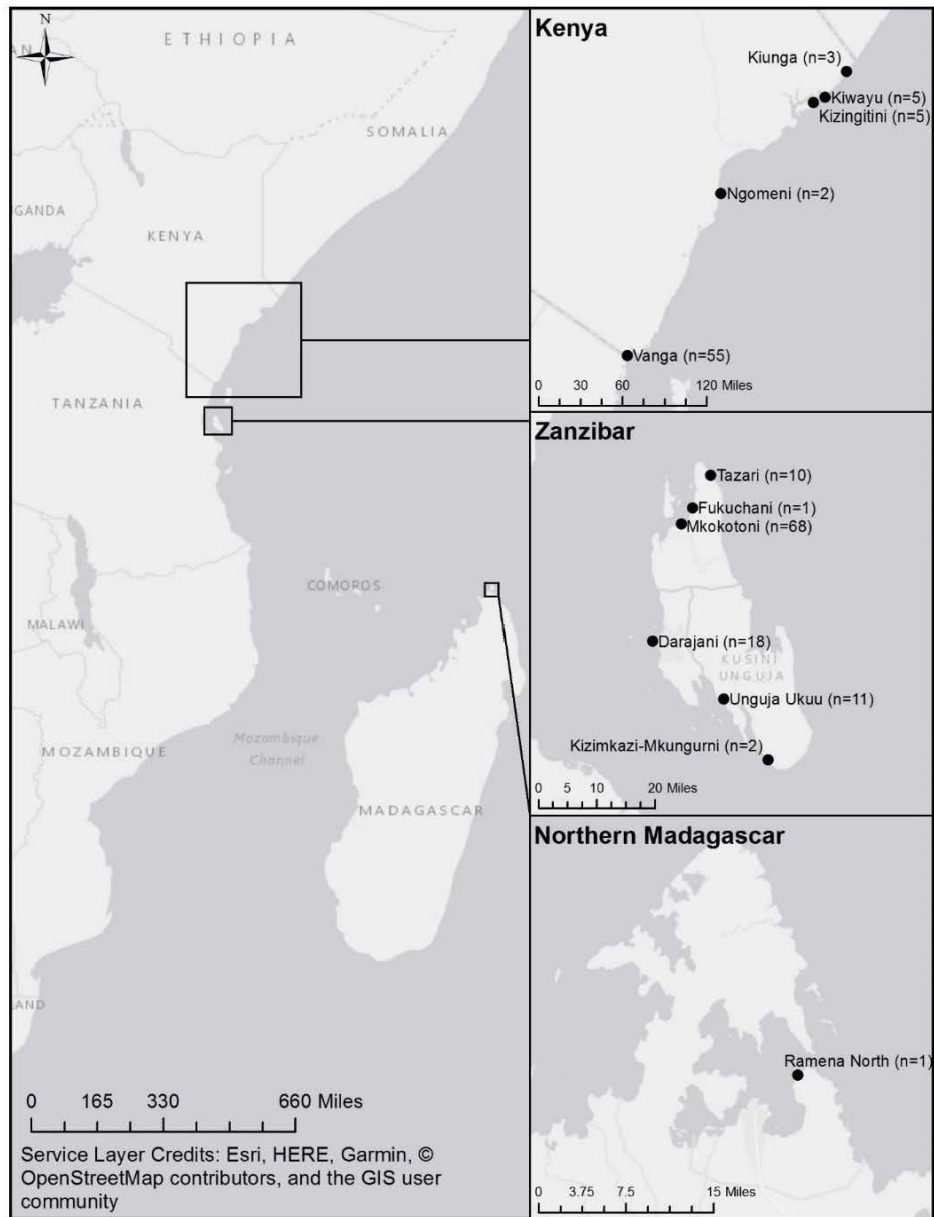


Figure 1. Locations of landing sites and markets in the southwestern Indian Ocean where Baraka's whipray (*Maculabatis ambigua*) were sampled between July 2015 and June 2017, sample sizes for those individuals recorded by fisheries observers is displayed.

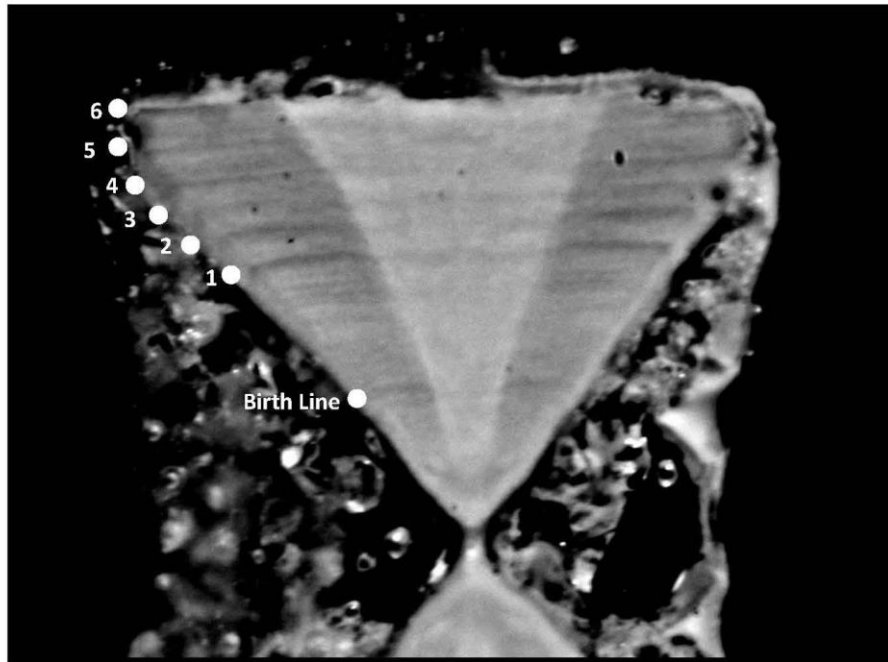


Figure 2. Photograph of sectioned vertebrae, with birth line and annuli marked, taken from a 63cm disc-width male Baraka's whiplay (*Maculabatis ambigua*) captured in a bottom-set gillnet in August 2015. This individual was aged 6 years.

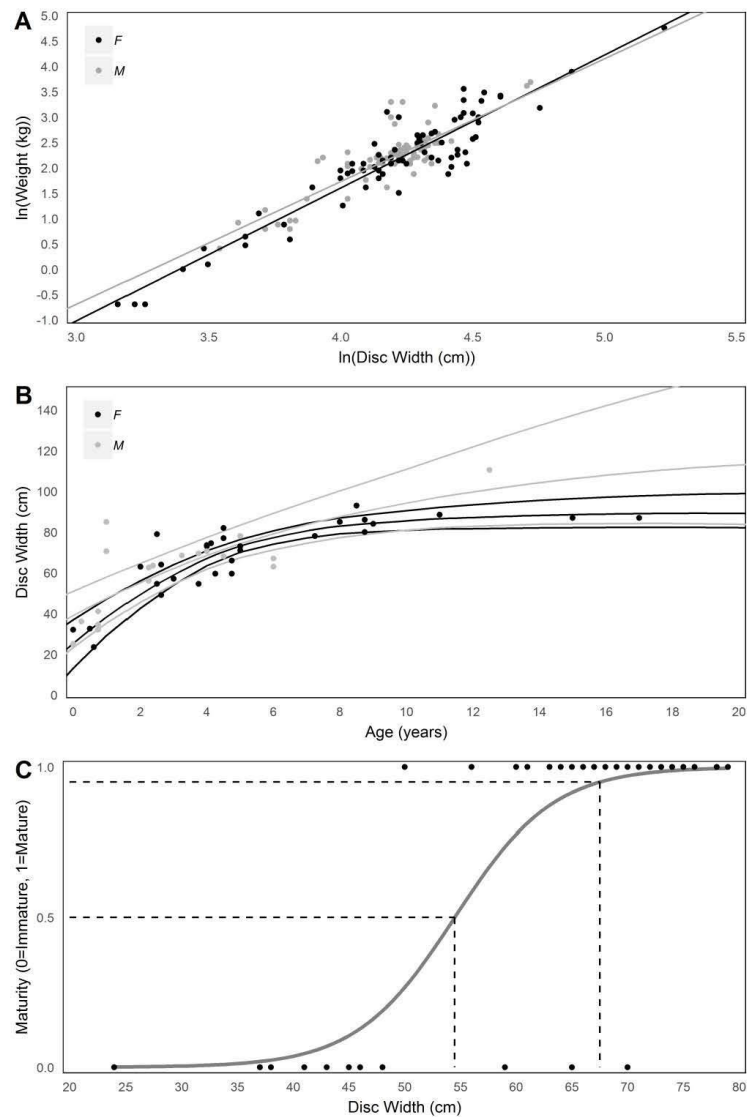


Figure 3. A) Linear relationships between the natural log transformed disc width (cm) and weight (kg) for female, $\ln(\text{weight}) = 2.62 \cdot \ln(\text{disc width}) - 8.88$, and male, $\ln(\text{weight}) = 2.42 \cdot \ln(\text{disc width}) - 7.96$, Baraka's whipray (*Maculabatis ambigua*). B) Three parameter von Bertalanffy growth curve (±95%CI) describing the disc width and age relationship for female, $DW_{\infty} = 89.4$ (95%CI 82.2-100.9)cm, $DW_0 = 26.1$ (95%CI 15.9-36.6)cm and $k = 0.28$ (95%CI 0.17-0.41), and male, $DW_{\infty} = 139.7$ (95%CI 84.1-198.0)cm, $DW_0 = 40.9$ (95%CI 27.4-50.1)cm and $k = 0.08$ (95%CI 0.01-0.43), *M. ambigua*. C) Logistic regression describing the relationship between disc width and maturity status (immature or mature) for male *M. ambigua*, disc width at 50% maturity and 95% maturity are indicated.

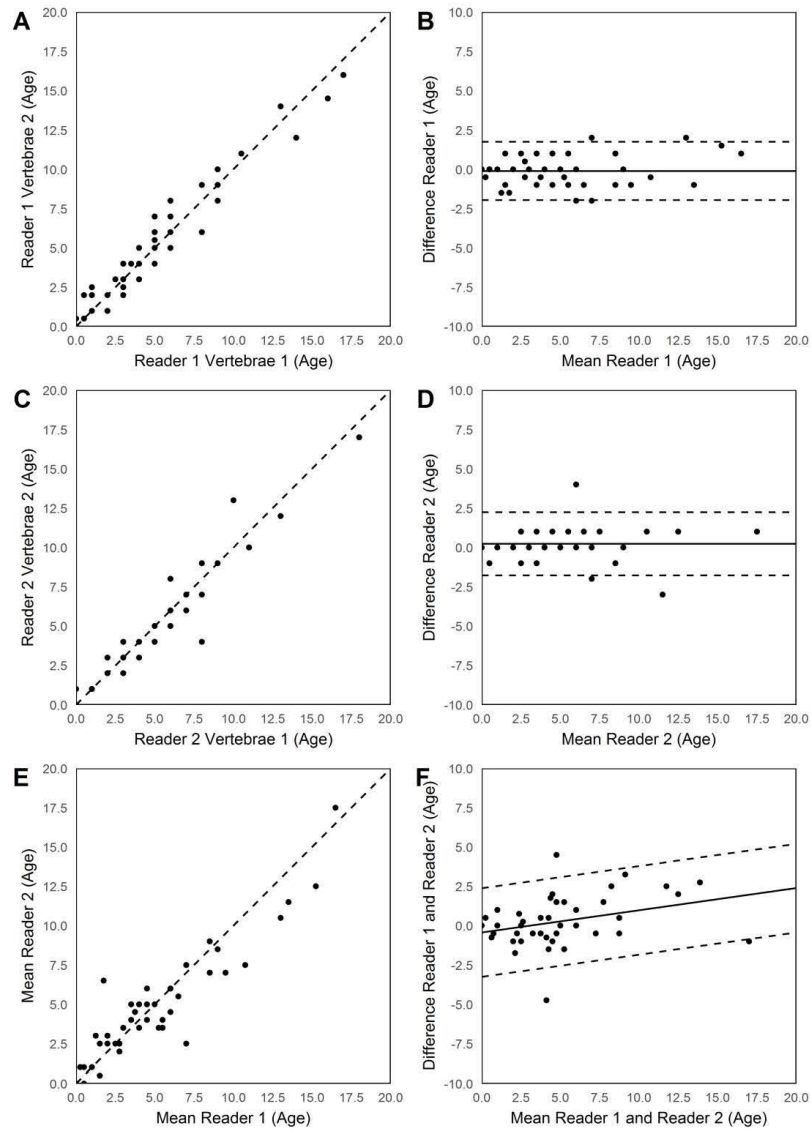


Figure 4. Bland-Altman assessments of agreement, precision and bias in age estimates (year $\pm$ 0.5) for Baraka's whipray (*Maculabatis ambigua*) within and between readers. A) Relationship between vertebrae age band counts for Reader 1. B) Bland-Altman Plot displaying bias and precision between vertebrae age band counts for Reader 1. C) Relationship between vertebrae age band counts for Reader 2. D) Bland-Altman Plot displaying bias and precision between vertebrae age band counts for Reader 2. E) Relationship between mean vertebrae age band counts from Reader 1 and Reader 2. F) Bland-Altman Plot displaying bias and precision between mean vertebrae age band counts from Reader 1 and Reader 2.

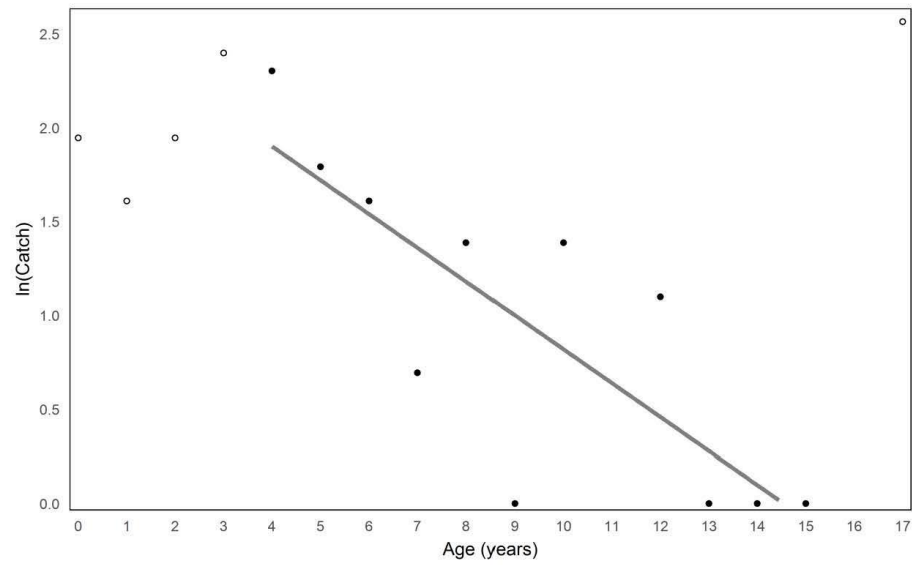


Figure 5. Chapman-Robson catch curve for female Baraka's whiplay (*Maculabatis ambigua*) across southwestern Indian Ocean small-scale fisheries, displaying age class of full recruitment to the fishery (age class 3) and catch curve regression line from age class 4 to 16, total mortality Z is estimated at 0.265(95%CI 0.202-0.328).

Table 1. Precision values for comparisons of age estimates for Baraka's whiplay (*Maculabatis ambigua*) within and between readers, Percentage Agreement (PA), Percentage Agreement ± 1 year, Coefficient of Variation (CV), Average Percentage Error (APE) and Bland-Altman Limits Of Agreement (LOA)

Comparison	PA (%)	PA ± 1 year (%)	CV (%)	APE (%)	LOA (years)
Within Reader 1	34.04	85.11	15.38	10.87	± 1.81
Within Reader 2	46.81	93.61	12.87	9.10	± 2.01
Between Reader 1 and Reader 2	12.77	68.09	24.14	17.07	± 2.82

Table 2. Growth model fit and parameters for male, female and combined male and female Baraka's whiptail (*Moculabatis ambigua*). Estimated values for asymptotic disc width (DW_{∞}) disc width at age zero (DW_0), growth constant (k) and inflection point (α) are displayed as appropriate, as are their associated standard errors (S.E.). Values for Akaike's Information Criterion (AIC), residual standard error (r.s.e.) and sample number (n) relating to fitted models are also displayed.

Model	Parameter	Female					Male					Combined				
		Estimate	S.E.	AIC	r.s.e.	n	Estimate	S.E.	AIC	r.s.e.	n	Estimate	S.E.	AIC	r.s.e.	n
Three parameter von Bertalanffy	DW_{∞}	89.35	4.60	209.1	8.19	29	139.69	90.42	151.5	14.29	18	92.21	6.47	362.7	10.88	47
	DW_0	26.08	5.59				40.92	6.90				33.58	4.52			
Three parameter Gompertz	k	0.279	0.061				0.081	0.108				0.234	0.064			
	DW_{∞}	87.54	3.90	210.6	8.26	29	130.64	61.12	152.0	14.47	18	90.87	5.70	364.1	11.05	47
Logistic	α	0.30	0.08				0.84	3.42				-0.25	0.41			
	k	0.304	0.354				0.131	0.109				0.302	0.073			
	DW_{∞}	86.54	3.59	209.6	8.40	29	125.38	47.60	152.3	14.59	18	90.07	5.26	365.4	11.20	47
	α	1.23	0.35				3.30	4.43				0.87	0.44			
	k	0.487	0.095				0.187	0.115				0.371	0.083			

Chapter 5. Baseline Assessment of Natural Resource Exploitation: a Methodological Comparison in a Data-Poor, Capacity-Limited Environment

5.1 Abstract

Understanding the exploitation of natural resources is a global challenge for environmental managers. Mismanagement resulting from inadequate or wrongly interpreted data can have wide-ranging consequences for resource users, their communities and the wider environment. Methodological approaches used in monitoring exploitation are subject to inherent biases which must be properly understood to facilitate informed and appropriate use. However, in data-poor and capacity limited (technologically or financially) environments, traditional observer-based methods are sometimes infeasible and the use of local knowledge is proposed as a first-step in generating data upon which evidence-based management can be formulated. However, comparison of outputs derived from local knowledge with those of traditional methodologies are generally lacking and often incomplete, undermining their robust interpretation. Methodological comparisons should assess not only the relationship but also the nature of biases and the levels of precision between the outputs. In this study we use the Bland-Altman approach to compare two commonly used methods to estimate fishing effort and catch of elasmobranchs (sharks and rays), sea turtles and dolphins. Landings observations covering a 12-month period and rapid bycatch assessment interviews (n=348) covering identical spatial and temporal scales were conducted at small-scale fisheries landing sites in Madagascar (n=5) and Zanzibar (n=8). The results demonstrate inconsistency in relationships among spatial and temporal fishing effort and catch patterns between these methods, with the majority showing no evidence of relationships. Positive relationships appeared more common where patterns in effort and catch showed greater levels of variability and/or potentially for distinct species groups, but precision between methods was low and in some cases evidence of bias was found between methods. Thus, outputs of the two methods cannot be considered broadly equivalent nor interchangeable. The findings support the need for multi-method approaches to natural resource monitoring in order to better inform management decisions and highlight areas of contention where further works may be required.

5.1.1 Key Words

Hunting; Local Ecological Knowledge; Local Fishing Knowledge; Rapid Bycatch Assessment

5.2 Introduction

Historically, the majority of wildlife harvest monitoring and assessments have been observer-based and/or through formal declarations. Common methodological examples for collection of effort and catch data in fisheries research include landings observation, vessel based observation and log books. A number of these methods have been extensively cross-examined (e.g. Hill and Barnes 1998; Walsh *et al.* 2002; Faunce 2011; O'Donnell *et al.* 2012; Silva *et al.* 2012). Direct observations (e.g. vessel based) generally produce higher estimates of catch and bycatch than landings observations and may alter fisher behaviour, and declaration based data (e.g. log books) generally produce lower catch and higher fisheries effort than landings observations. However, appropriate application of these methods is time and labour intensive, and thus expensive, especially in numerous and/or disparate

fisheries. Further, fishers have often contested conclusions derived from scientific works and in a number of cases (e.g. Arctic bowhead whale *Balaena mysticetes* stocks and Canadian Atlantic cod *Gadus morhua* collapse) ignoring these contests has led to inappropriate fisheries management (Freeman 1989; Neis 1992; Johannes *et al.* 2000). In the case of the bowhead whale, restrictions on aboriginal whaling were put in place as a result of researchers underestimating population size as a result of poorly informed observer programmes (Freeman 1989) and in the case of Atlantic cod, fishers observed decreases in spawning stocks but were ignored by managers (Neis 1992).

The use of local fisheries knowledge (LFK) to monitor various aspects of fisheries, either in isolation or in combination with other methods, is increasingly common (e.g. Moore *et al.* 2010; Beaudreau and Levin 2014; Pilcher *et al.* 2017). Relative to observer-based methods, LFK is often considered a cheap but effective way to generate fisheries data (Neis *et al.* 1999; Anadón *et al.* 2009; Rist *et al.* 2010). Consequently, widespread use of LFK is proposed as a solution with which to rapidly assess fisheries in data-poor and capacity-limited situations. Such situations are most evident in small-scale fisheries (SSF), which account for approximately 95% of fishers, 23% of catch by weight and 27% of catch by value at the global level (Pauly 2006; Pauly and Zeller 2015). The majority of SSF are found in capacity-limited developing nations, making the use of LFK particularly attractive. Additionally, LFK may be advantageous in monitoring unusual events, such as catches of rare species, which equivalent observation-based methods are likely to miss if sampling effort is not high enough. However, LFK is vulnerable to a number of pitfalls and biases which may undermine confidence in its outputs. These may be malicious, such as supply of intentionally false information, or malign, for instance biases in human cognitive recall (Matlin 2004; Hirst *et al.* 2009).

Despite the uncertainties and biases surrounding both observer-based and LFK methods there are relatively few studies in fisheries science that have cross-examined their outputs thoroughly. The majority of studies have been restricted to identifying evidence of relationships between methods (e.g. Fox and Starr 1996; Lunn and Dearden 2006; Anadón *et al.* 2009; Rist *et al.* 2010; Daw *et al.* 2011; Sampson 2011; Mion *et al.* 2015; Lima *et al.* 2017) and consistently failed to properly assess patterns in bias and precision. Yet bias and precision are vital components in assessing and understanding the variable structure of relationships among methods. Evidence for relationships between LFK and observation-based data are mixed (e.g. Anadón *et al.* 2009; Rist *et al.* 2010; O'Donnell *et al.* 2012). LFK is generally considered to be a useful indicator for assessing long-term trends in species and animal harvesting activities (Neis *et al.* 1999; Daw *et al.* 2011; O'Donnell *et al.* 2012; Beaudreau and Levin 2014), although this is difficult to validate. Comparatively, the use of LFK as a tool to assess harvesting activities across shorter temporal ranges, such as intra-annual trends, and among locations has received limited attention. Yet, these are important components in the understanding of fisheries and the formulation of management strategies.

The aim of this study is to compare data on intra-annual and annual fishing effort and catch derived from LFK and observer-based methods. Thus, helping to inform the use of these methods, both in combination or in isolation, as a means of generating usable information for fisheries managers in data-poor and capacity-limited situations. We use a case-study from the handline, longline, bottom-set and drift-gillnet small-scale fisheries in two developing countries, Zanzibar and Madagascar, and their catch of various marine

megafauna, which include both legal (sharks and rays) and illegal (sea turtles and dolphins) catch components (Temple *et al.* 2018). LFK data were collected through fisher interviews using a modified Rapid Bycatch Assessment (RBA) (Poonian *et al.* 2008; Moore *et al.* 2010; Whitty *et al.* 2010; Kiszka 2012; Poonian 2015; Pilcher *et al.* 2017; Braulik *et al.* 2018), which are designed primarily to generate data at the country level. The observation-derived data were from landing site monitoring. The data comparisons included are: patterns in intra-annual variation in fisheries effort and catch; patterns in catch among sites; and total catch estimates among sites.

5.3 Methods

Trained observers collected data on fisheries effort (active fishing vessels per day – with no evidence of multi-day fishing trips occurring) and landed catch (number of individuals) of marine megafauna (herein referring to sharks, rays, marine mammals and sea turtles) from bottom-set and drift gillnet, longline and handline gears at landings sites in Zanzibar (n=8) and Madagascar (n=5) over a complete 12-month period (total landings recorded, n=3101) between June 2016 and June 2017 (Figure 5.1). Observers simultaneously collected data over 147 pre-determined sampling days, selected using a stratified-random approach. The year was divided into lunar months and then subdivided into four lunar phases (new moon, first quarter, full moon, third quarter), three sampling days were then randomly generated within each lunar phase. This sampling regime ensured that the study could account for lunar-driven patterns in both fishing effort and species landings as well as being representative of other potentially significant effects such as varying environmental conditions and cultural celebrations (e.g. Ramadan). Landings sites were selected accounting for three major factors: prevalence of longline and gillnet gears (maximising representation of the main gear-threats), geographic spread (maximising geographic coverage and potential linked species assemblage changes) and logistical constraints (e.g. site accessibility).

Fisheries effort data collected by observers was successfully modelled using a Generalised Additive Mixed Modelling (GAMM) approach. Independent variables input to the GAMM model included daily data on precipitation, wind speed, wind direction, temperature, maximum tidal height (as a proxy for lunar phase) and month, as well as site as a random-effect variable. Environmental data were extracted from the NCEP/NCAR Reanalysis database (ESRL 2017). Cyclical variables (e.g. wind direction, month) were fitted with cyclic splines. Where independent variables showed evidence of co-linearity ($r > 0.2$ or < -0.2), those with least explanatory power were removed from the model for subsequent iterations. The final models used to predict daily fisheries effort showed significant effects of $p < 0.05$ of three independent variables: maximum tidal height, month and wind direction.

Month significantly influenced bottom-set, driftnet, handline and longline fishing effort ($\chi^2=1453.10, 42.45, 161.48, 380.62$, respectively; $p < 0.05$). Tidal height significantly influenced driftnet, handline and longline fishing effort ($\chi^2=41.65, 33.12, 9.76$, respectively, $p < 0.05$). Wind direction significantly influenced driftnet and handline fishing effort ($\chi^2=14.57, 175.12$, respectively; $p < 0.05$). Wind speed was not found to affect fishing effort for any gear type, despite being expected impact sea conditions. The models explained

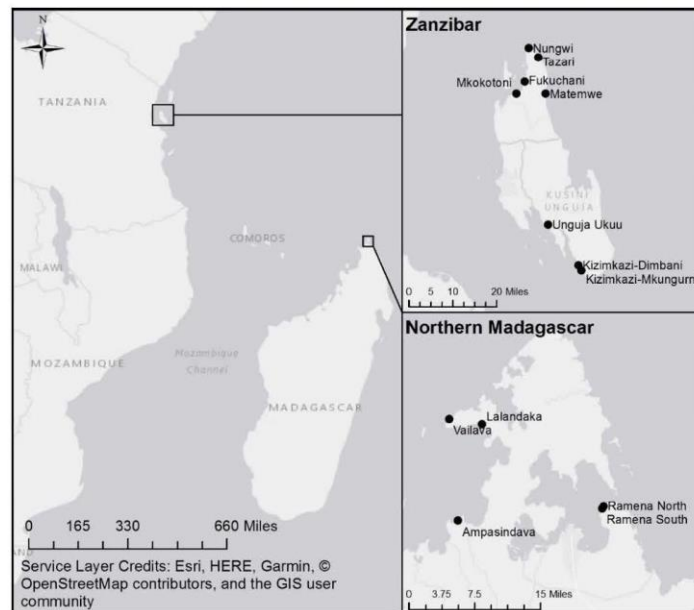


Figure 5.1 Locations of landing sites in the southwestern Indian Ocean monitored by observers for marine megafauna catch between June 2016 and June 2017 where subsequent Rapid Bycatch Assessment interviews were undertaken.

75.0%, 54.2%, 82.2% and 54.9% of the deviance for bottom-set net, driftnet, handline and longline fishery effort. Predicted effort was used to scale CPUE, calculated as mean catch per vessel active per day, to monthly and annual timescales for each landing site. For further details see Temple *et al.* 2019.

Subsequently, face-to-face RBA questionnaire interviews were carried out with fishers (Appendix B.), targeting vessel captains where possible, in the same sites and covering the same gears in Zanzibar ($n=204$, captains = 99) and Madagascar ($n=36$, captains = 9). A minimum sample size of a quarter of the total known number of vessels of respective gear types in each site was targeted as a means of achieving an overall representative sample for the fishery across landing sites, and was achieved. RBAs were carried out in Swahili and Malagasy as appropriate by teams of trained interviewers from each country. Interviewees were selected opportunistically, avoiding interviews with multiple crew from the same vessel. RBAs took approximately 20 minutes to complete. The participant was informed of both the motivation and the intended use of the data collected after which verbal consent from participants was sought, before the RBA was undertaken. Participant's names were recorded but anonymity of their responses was assured. Further, participants were informed of their right to decline any question which they were unwilling or unsure about answering and, should they so wish, that the interview could be ended at any time. RBAs were not facilitated with either monetary or material motivation. Ethical approval for the survey was sought from and granted by the ethics review board at Newcastle University.

The RBA records fisher declarations of fisheries effort and catch including: months of gear use; average days fished across those months; months in which specified taxa were caught (sharks, rays, sea turtles, dolphins) and selected easily identified species groups within these (hammerhead sharks *Sphyrna* spp., whale sharks *Rhincodon typus*, mobulids *Mobula* spp.

and sawfish/sawsharks *Pristidae/Pristiophoridae*); and average number of each group caught across those months. RBA derived data were scaled using total known vessel numbers by gear at each site, derived from frame census survey data in Zanzibar (ZDFD 2018) and counts by the project team in Madagascar, to both monthly and annual levels.

5.3.1 Comparative Analysis

In order to comprehensively compare the outputs of both observed landings and RBA methods and thus assess their comparability the Bland-Altman approach was taken (Bland and Altman 1999, 2003). This method firstly assesses the relationship between methodologies, which was done by repeated measures and Pearson correlation as appropriate. Secondly, any potential bias in the relationship between the two methods is assessed using a linear mixed effect model, with country and gear type as random effect variables as appropriate, in cases where random effect variables were not significant a linear model were instead used. Thirdly, the precision of methods relative to one another is described by the limits of agreement (LOA), which are the 95% mean CI of the differences between methods and these are presented as exact limits of agreement, LOA (inner LOA – outer LOA), in accordance with Carkeet and Goh (2018). The Bland-Altman method does not assume that either methodology being assessed represents the absolute reality. Thus the approach was considered appropriate in this case, where both methods represent facets of absolute reality with individual intrinsic biases.

The results of the analysis presented do not attempt to make judgement on the value of either method in measuring the absolute reality. Outputs assessed using this method were: patterns in fisheries effort among months, measured as a proportion of total effort observed or declared; patterns in taxa catch among months, measured as the total proportion of catch observed or declared (groups with extremely few declarations or observed landings were excluded from these analysis – dolphins, sea turtles, whale sharks and sawshark/sawfish); and patterns in taxa catch among sites, as a proportion of catch declared or observed (groups with extremely few declarations or observed landings were excluded from these analysis – dolphins, sea turtles, whale sharks and sawshark/sawfish). All aforementioned analyses are carried out at the country level, i.e. RBA and observed landings outputs are summed across sites, to reflect the design purpose of RBAs. Lastly, total site-by-site estimates for catch of taxa generated from RBA declarations and observed landings were directly compared to one another, with mean differences by site (for those sites where catch was reported from both RBAs and observed landings) also calculated.

5.4 Results

5.4.1 Intra-Annual Patterns in Fishing Effort

Intra-annual patterns in effort derived from RBA and observer networks were regressed across gear types using a linear mixed effects model, with gear type nested within country as random effect variables; no significant relationship was found ($t=0.962$, $p>0.05$). Subsequently relationships were sought for Madagascar and Zanzibar independently. Repeated measures correlation models found no significant correlation in intra-annual patterns of effort derived from RBA and observer networks for either country ($p>0.05$). Consequently relationships for intra-annual patterns in effort derived from RBA and observer networks were sought separately for each combination of country and gear type (Figure 5.2). No evidence of correlation was found for handlines or longlines in Zanzibar, and

bottom-set or drift gillnets in Madagascar (Pearson, $p > 0.05$). Positive correlations were found for bottom-set (Pearson, $r = 0.796$, $p < 0.005$) and drift gillnets (Pearson, $r = 0.821$, $p < 0.005$) in Zanzibar, and longlines (Pearson, $r = 0.715$, $p < 0.01$) in Madagascar. However, a negative correlation was found for handlines in Madagascar (Pearson, $r = -0.615$, $p < 0.05$).

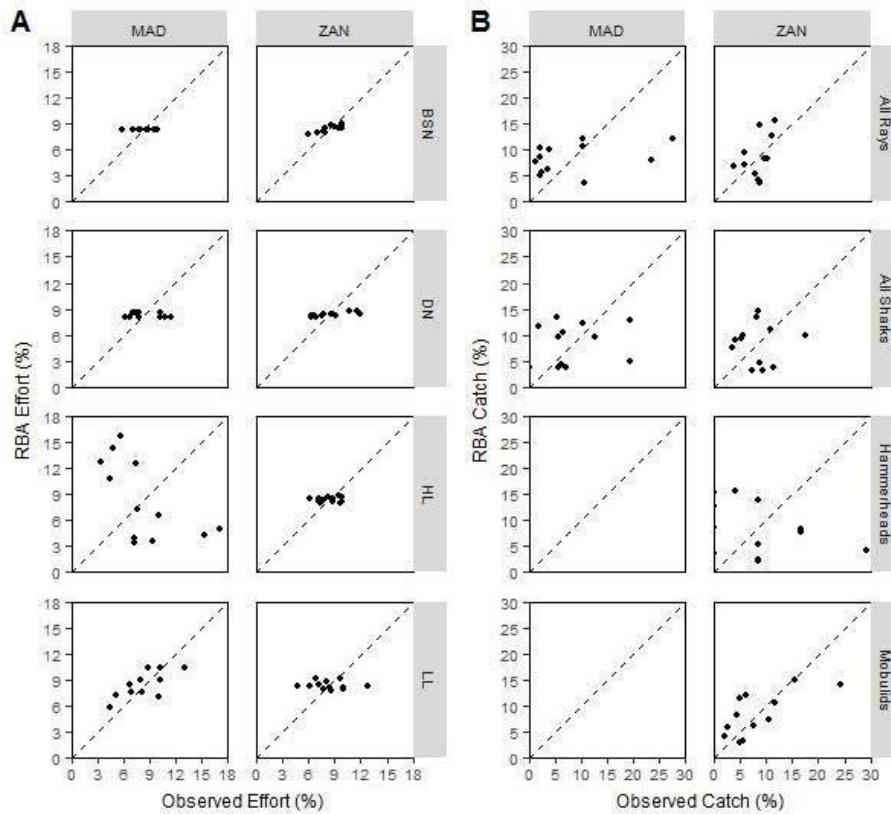


Figure 5.2 Relationships between Rapid Bycatch Assessment (RBA) and landing observer estimates between June 2016–June 2017 of monthly A) fisheries effort, and B) fisheries catch. Data is displayed for bottom-set gillnets (BSN), drift gillnets (DN), handlines (HL) and longlines (LL) for fisheries effort, and species group for fisheries catch, in Madagascar (MAD) and Zanzibar (ZAN). Correlations were found to be significant for fisheries effort in Zanzibar bottom-set gillnets ($r = 0.796$), Zanzibar drift gillnets ($r = 0.821$), Madagascar handlines ($r = -0.615$), and Madagascar longlines ($r = 0.715$). Correlations were only significant for mobulid catch in Zanzibar ($r = 0.691$).

For those combinations of country and gear type showing positive relationships between RBA and observation derived fisheries effort, bias in agreement was assessed. Evidence of bias in agreement was found when regressing the difference in proportional effort against the mean of proportional effort between method outputs. Both a linear mixed effect model, with gear type nested within country as a random effect variable, and a linear model were run ($t = 2.23$, $p < 0.05$ and $t = 6.39$, $p < 0.001$, respectively). No significant difference between models was found (ANOVA, $p > 0.05$) suggesting that neither country nor gear type significantly affected the nature of bias, thus the linear model was used. The linear model indicates a bias whereby RBAs estimate higher effort than observer networks at low effort levels; conversely RBAs estimate lower effort than observer networks at high effort levels.

This likely reflects the nature of the RBAs employed, which record average fishing days per month rather than daily effort, thus suppressing variability. LOA were calculated at $\pm 1.99\%$ (95%CI 1.51-2.71%) of total effort (Figure 5.3).

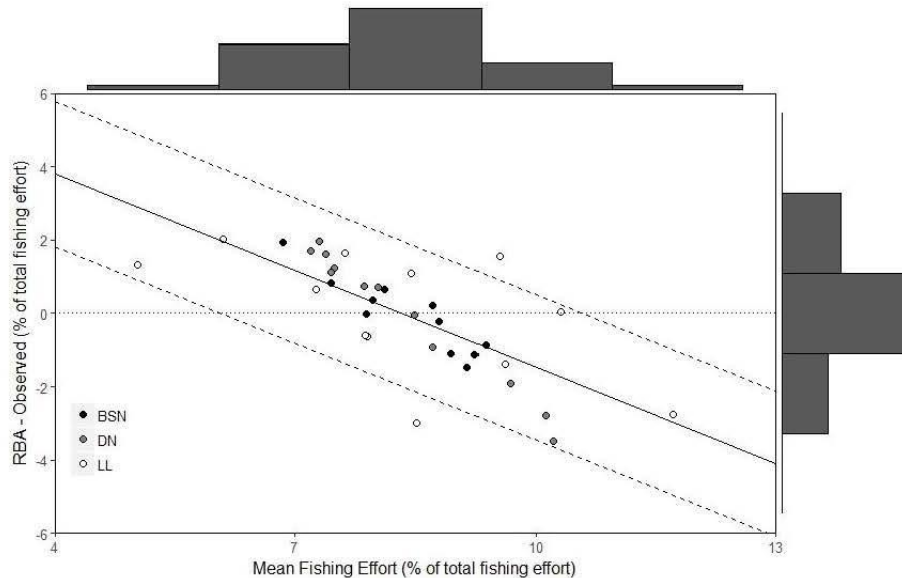


Figure 5.3 Bland-Altman plot for comparison of Rapid Bycatch Assessment (RBA) and landing observer estimates of monthly fishing effort between June 2016-June 2017, linear regression line is displayed alongside the 95% mean confidence interval of residuals, axes histograms display data distribution (BSN= Zanzibar bottom-set gillnets, DN = Zanzibar drift gillnets, LL = Madagascar longline).

5.4.2 Intra-Annual Patterns in Catch

Repeated measures correlation models, with country as a random effect variable, found no evidence of a relationship between RBA and observer-derived data ($p > 0.05$) for either shark or ray catches. Landing sample sizes for sub-groups (hammerhead sharks, mobulids, whale sharks and sawsharks/sawfish) in Madagascar were too small ($n=29, 2, 0$ and 0 respectively) to draw comparisons with any confidence and thus these groups were not assessed. Consequently, relationships between RBA and observer-derived catch data were sought separately for each combination of country and grouping. No significant correlations were found for sharks or rays in either Zanzibar or Madagascar, nor for hammerhead sharks in Zanzibar (Pearson, $p > 0.05$). However, a significant moderately strong correlation (Pearson, $r=0.691$, $p < 0.05$) was found for mobulids in Zanzibar (Figure 5.2). Subsequent Bland-Altman assessment of RBA and observer monthly estimates of mobulid catch showed no evidence of bias (Linear model, $t=1.67$, $p > 0.05$) and LOA were estimated at $\pm 9.06\%$ (95%CI 5.63-16.31%) of total catch (Figure 5.4).

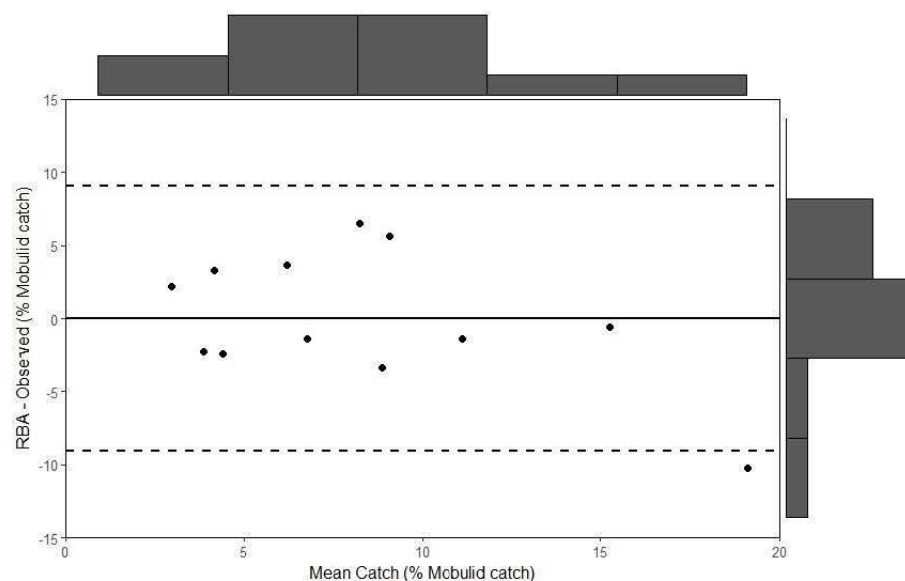


Figure 5.4 Bland-Altman plot for comparison of Rapid Bycatch Assessment (RBA) and landings observer estimates between June 2016-June 2017 of intra-annual patterns in mobulid catch in Zanzibar, 95% mean confidence intervals are displayed, axes display histograms of data distribution.

5.4.3 Annual Patterns in Catch Among Sites

Correlations in patterns of catch among sites in Zanzibar derived from RBA and observer networks were assessed for all taxa (sharks, rays, sea turtles, dolphins) and sub-groups with large enough sample sizes (hammerhead sharks, mobulids). No correlations (Spearman, $p > 0.05$) were found for either rays, mobulids, sea turtles or dolphins. However, strong positive correlations were found for both sharks (Spearman, $r = 0.738$, $p < 0.05$) and hammerhead sharks (Spearman, $r = 0.878$, $p < 0.005$). Subsequent Bland-Altman assessment of RBA and observer-derived data among sites for all shark and hammerhead shark catch showed no evidence of bias (Linear models, $t = 0.088$, $p > 0.05$, and $t = 1.300$, $p > 0.05$ respectively) and LOA were estimated at $\pm 16.94\%$ (95%CI 9.43-36.67%) and $\pm 26.87\%$ (95%CI 14.96-58.16%) of total catch respectively (Figure 5.5).

5.4.4 Total Catch Estimates

Zanzibar's total catches (number of individuals) by grouping were compared between methods in order to assess how closely the two method's estimates compared (Table 5.1). Total catch estimates for legal taxa (sharks, rays) were substantially lower from RBA data than for those from the observer network. Conversely, catch of sub-groups within these (hammerhead sharks, mobulids), rare but legal sub-groups (whale sharks and sawsharks/sawfish) and the illegal taxa (sea turtles, dolphins) were substantially higher from the RBA data than those from the observer network. The contribution to overall taxa (sharks and rays) catch from sub-groups (hammerhead sharks and mobulids) was much larger in the RBA assessment (66.01% and 42.27% respectively) than was seen in the observer network (7.23% and 14.20% respectively).

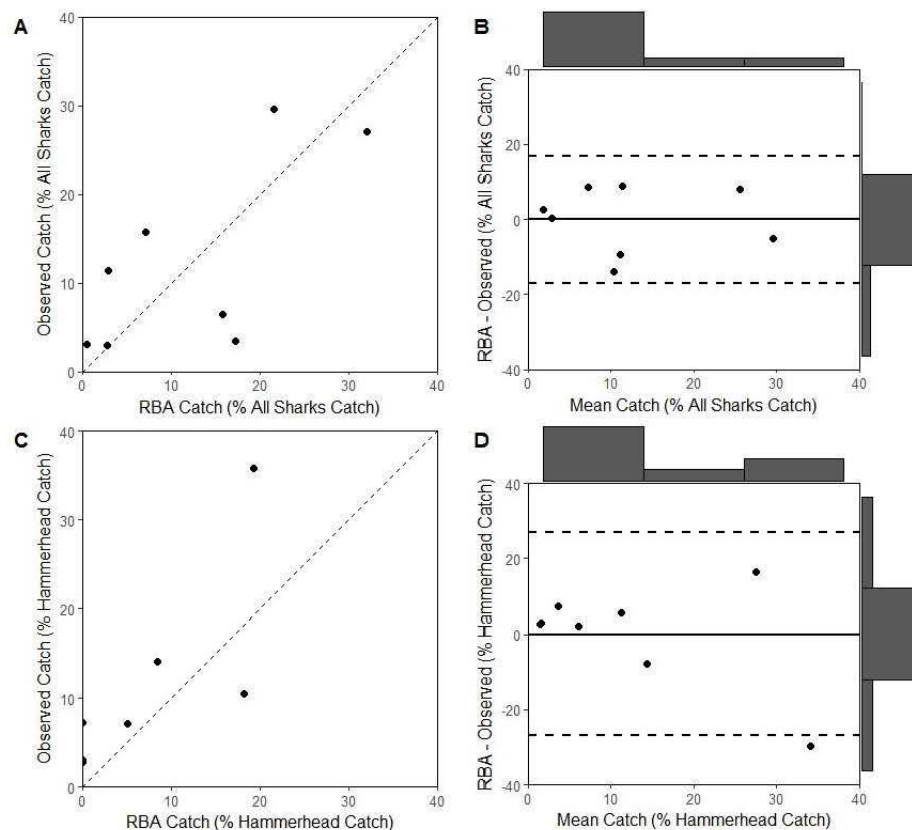


Figure 5.5 Bland-Altman comparison of Rapid Bycatch Assessment (RBA) and observer network estimates of patterns among Zanzibar landing sites between June 2016-June 2017 where significant correlation was found A) sharks ($r=0.738$) and C) hammerhead sharks ($r=0.878$), and Bland-Altman plots showing 95% mean confidence interval with axes histograms showing data spread for B) sharks and D) hammerhead sharks.

Table 5.1 Comparison of catch (number of individuals) of various megafauna species groups estimated from Rapid Bycatch Assessment (RBA) and landings observer network methods for Zanzibari fisheries between June 2016-June 2017.

Species Group	RBA as % Landings Total	RBA as % Landings Site Mean	RBA as % Landings Site 95%CI	Landings Total	RBA Total
Sharks	64.3	79.5	13.1-481.5	2150.9	1383.7
Hammerheads	586.9	584.7	149.0-2294.9	155.6	913.5
Rays	41.2	29.2	5.3-150.9	6353.3	2616.0
Mobulids	122.6	158.0	9.8-2532.7	902.3	1105.9
Sea Turtles	276.8	127.3	24.0-675.1	48.7	134.7
Dolphins	344.2	-	-	11.1	38.2
Whale Sharks	Inf	-	-	0	27.7
Saw sharks/Sawfish	Inf	-	-	0	150.3

5.5 Discussion

Observer-derived data, often landings or vessel based, have been the primary method used to monitor and assess effort and catch across many developed and developing countries' fisheries. Yet, there have been instances where reliance on such methods has failed to result in appropriate management of fisheries, despite LFK offering contradictory information (e.g.

Neis 1992; Johannes *et al.* 2000). Use of LFK has been proposed as an effective means of generating usable baseline information for fisheries managers in data-poor and capacity-limited situations. LFK is usually considered to provide relatively robust indicators of long-term trends in fisheries (Neis *et al.* 1999; Daw *et al.* 2011; O'Donnell *et al.* 2012; Beaudreau and Levin 2014), but little attention has been paid to LFK's feasibility in documenting intra-annual and annual patterns. Given that no current method is capable of capturing fisheries data without bias nor interference with fisher behaviour all methods are inherently fallible. Thus neither LFK nor observer based methods can be assumed to represent the true reality and so both should be interpreted with caution and proper consideration of their failings. This study indicates that relationships between LFK and observer-derived data for fisheries effort and catch at relatively short timescales may be inconsistent. The findings indicate that observed and LFK derived data should not be assumed equivalent nor interchangeable over such timescales and that underlying factors undermining potential equivalency require further attention. Independent use of either observer or LFK derived data therefore risks overconfidence in their outputs with potentially damaging consequences for both fisheries resources and dependent communities.

Where there was evidence of agreement between RBA and landings observations, the LOA between methods were large, indicating low precision. Such low precision indicates that one, or both, methods are not an accurate indication of the reality or one another and thus dependence on the finer levels of detail from their outputs as evidence bases for management should be treated cautiously or risk resultant mismanagement. Further, in the case of fisheries effort comparisons there was evidence of a bias in the pattern of agreement. The limitations of this study mean that the drivers of this bias could not be assessed, however the conflicting pitfalls fundamental to either method, limitations of their specific designs (e.g. temporal and spatial specificity and resolution) and their implementation likely contributed. Future comparative studies should seek to quantify the potential drivers of biases in agreement. Such understanding may assist towards future developments of LFK and observer based methods with the goal of improving consensus among methods.

Efficacy of fishers' ability to recall effort and catch from memory is a likely driver of disparity between the outputs derived from the two methods. Human recall is more effective for events that are considered particularly unusual, emotive or displaying prominent and consistent trends (Matlin 2004; Hirst *et al.* 2009). Catches that are particularly rare (e.g. dolphins, whale sharks, sea turtles) or otherwise distinctive (e.g. large in volume or size) and/or catch of particularly high value (e.g. large and/or valuable animals, for example large sharks with high fin value) may therefore be more easily recalled by fishers than other catches. Similarly, catches of species that are illegal (e.g. dolphins, turtles) or those displaying substantial intra-annual variability may also be easier to recall, though the threat of persecution for illegal catches may influence their willingness to report these. It would be expected that fishers were more accurately able to recall patterns in fisheries effort or catch that were strongly seasonal, especially where these relate to distinctive and/or valuable catches. This is reflected in agreements between RBA and observer-derived outputs in a number of cases, e.g. Zanzibari bottom and drift gillnets and Madagascan longlines, which show substantial intra-annual variability, and mobulid catch in Zanzibar, which combine substantial intra-annual variability with ease of identification and relative rarity. However, a number of other examples of effort and catch data displaying similar strong seasonal

patterns in observer-derived data (e.g. Madagascan ray catches and longline effort in Zanzibar) were not reflected in RBA outputs.

Given that patterns in fisheries effort are likely to be relatively consistent among years, at least in the short-term, the limited agreement between methods was unexpected. However, daily fishing effort is impacted by environmental conditions, such as heavy rains and strong winds (Temple personal observation). Fluctuations in these conditions at relatively short temporal scales may inhibit fishers' ability to accurately retrospectively identify monthly patterns in their fishing effort over the year. Specifically, perceptions of effort within a given month may be unduly influenced by prominent events, such as extremely poor conditions, which are not representative of the time-period as a whole. Further, such events are likely to influence perceptions across the fisheries as a whole, and so as a result of their consistency have strong influences on outputs and consequently undermine the agreement between methods. Conversely, in the case of fishers, where memorable events, such as prominent catches, are less universal, overall correlations in data are less influenced by individual differences. However, if variability among fishers' declarations is high this may obscure overall patterns at the fishery scale (O'Donnell *et al.* 2012) and thus similarly suppress relationships between methods. Regardless, these outcomes suggest that factors affecting a fisher's ability to recall events are not, in isolation, a satisfactory explanation for the disparity between outputs presented here.

A second element that likely affects outputs of fisher declarations is the way in which fishers estimate their previous catches. Previous studies examining fisher estimations of their catch rates, i.e. their catch per unit effort (CPUE), suggest that fishers tend to overestimate average CPUE when compared to other methods, including landings, and log books (Lunn and Dearden 2006; Daw *et al.* 2011; O'Donnell *et al.* 2012). Further, the degree to which they overestimate catch may be exacerbated when considering greater temporal scales (O'Donnell *et al.* 2012). If fishers are basing their catch estimates on perceived average CPUEs, it would be expected that fishers provide overestimates for overall catch, resulting in large discrepancies between outputs. However, this was not consistent in the current study. Fishers' declarations appear to substantially underestimate the catches of the broad taxa groups (64.3% and 41.2% of shark and ray catch estimated from landings observations). However, fisher estimates of catches for distinctive sub-groups far exceeded the catches estimated from landings. Further, they accounted for a far higher proportion of overall catch from fisher declarations, than was seen from observer data. However, it must be noted that there was substantial variability in discrepancies between methods in the total catch estimate among sites, suggesting this comparison would benefit from a larger sample size. Yet, the results indicate that easily identifiable and memorable species, particularly relatively common ones like hammerheads and mobulids in the current study, may play a vital role in driving fisher's perceptions of catches of the larger taxa groups. Thus we suggest that greater consideration must be given to the role of individual or groups of species in shaping fisher perceptions of their fishing activities.

There are also a number of influences on observer efficacy that may undermine or further alter agreement among methods. Firstly, the illegality of landing both dolphin and sea turtles (Temple *et al.* 2018) mean that these catches are less likely to be landed openly and thus less likely to be recorded by observers, resulting in underestimates from observation-derived data. Though, illegal catches are also likely to be underrepresented in fisher declarations due to fear of prosecution. Despite this declarations were substantially higher

than observations. Secondly, observation efficacy will differ as a result of observer competence and the nature of the sites themselves (e.g. size, level of formal organisation), which may specifically impact the comparison of data amongst sites. Thirdly, biases exist in observation efficacy for specific components of the catch. Smaller individuals are less likely to be observed, as they are often grouped with other fish catches of similar size. Lastly, observer data based on sampling programmes may underrepresent rare catches if they happen to fall on non-sampling dates and this is likely reflected in the declaration of whale shark and sawfish/sawshark catches from RBA interviews, but none in the observed landings.

For the purpose of the analyses presented here it was assumed that RBA and observer data are directly comparable. However, beyond the influences of method efficacy, differing assumptions and biases within each method likely undermine equivalency. A lack of true equivalency between measurements has been observed in other studies which seek to draw information for varying sources (e.g. Jennings and Polunin 1995; Daw *et al.* 2011), where factors such as selectivity, temporal and spatial coverage have undermined equivalency of results. With regard to this study there are a number of prominent factors which contribute to a potential lack of equivalency and thus may undermine the relationships between outputs. Firstly, whilst thought to be minimal in SWIO SSF, discards and/or loss of catch at sea will result in underestimates in observation derived data. Fishers declared catches may not consistently differentiate discards from the total catch, yet discards are not accounted for in landings observations. Discards are likely to be most prevalent with illegal catch (e.g. sea turtles, dolphins), which may be discarded for fear of prosecution, or those species most difficult or dangerous to bring aboard (large predatory sharks, large rays, whale sharks etc.), especially in gears that are not suited to their capture (e.g. handlines, longlines with small hooks sizes). Such instances may contribute to the higher estimates from fisher declarations of these groupings. Secondly, fishers often land catches at different sites depending on local market conditions and demands for specific catch (Temple personal observation). Thus, catches may have site-specific under and overrepresentation in observation data. Lastly, the migratory nature of many sub-components of the SSF in the SWIO (Wanyonyi *et al.* 2016) and other regions means fishers may still be active in other fishing grounds at times of the year when activity from their home-port is low. Additionally, the analysis assumes that the two methods compared are independent of one another. However, in practice fishers participating in RBA interviews were aware that data on catch and effort had been recorded over the preceding year. This knowledge may be expected to reduce the likelihood of fishers providing false information, potentially suppressing the level of disagreement between methods. It is therefore possible that, had this not been the case, levels of agreement and precision between methods may have been further reduced.

5.6 Conclusion

This study adds credence to the growing body of literature advocating for a multi-methodological approach to fisheries monitoring and management. Specifically, through quantitatively demonstrating the potential for disparity between formal methods of fisheries monitoring (landings observation) and LFK derived data covering identical spatio-temporal scales, we hope to further the conversation around continued development of multi-method approaches aimed at accounting for the fundamental weaknesses and flaws inherent to mono-method approaches. Through increased use and development of multi-

methodological approaches to fisheries management it is hoped that managers and resource users can reach closer consensus on the state of fisheries stocks and thus increase the probability of appropriate, effective management to facilitate sustainable fisheries.

Specific considerations from this study as they apply to future works are:

- A need for future research in methodological development and comparison to assess not only the relationship among outputs of methodological approaches but also to quantify biases and precision between methods. Thus, improving the informed use and interpretation of multi or mono method approaches and their outputs.

and

- A need to assess how specific species in fisheries catch (e.g. memorable or distinctive species such as hammerhead sharks or mobulid rays) drive fisher perceptions of overall fisheries activity and the consequences this has on fisher declarations.

In the more specific context of data-poor and capacity-limited (be that technological or financial) environments where LFK is being considered as a proxy for traditional observation-based methods we urge caution in its use to generate high resolution data. Our findings suggest that whilst LFK may be a useful indicator of fisheries activities over larger temporal and/or spatial scales, its use at intra-annual temporal scales and small spatial scales (e.g. among individual sites) does not consistently produce equivalent, or similar, patterns to those derived from observation based methods. Thus, we do not recommend that data be collected through this method alone, and that further development and comparative assessment of interview methodologies for this purpose are required before widespread use can be recommended.

Annex 10: Drink bottle acoustic alarms – recycled glass and plastic bottles provide potential low cost solution to end 50 years of cetacean fisheries bycatch.

Berggren, Temple, Sharpe, Yang & Neasham. Drink bottle acoustic alarms – recycled glass and plastic bottles provide potential low cost solution to end 50 years of cetacean fisheries bycatch. In Review *Scientific Reports* (SREP-18-36647-T).

Abstract

Cetaceans have been threatened by incidental catch (bycatch) in gillnet fisheries for 50 years. An unsustainable proportion ($\geq 3\%$) of the global cetacean population is lost annually through bycatch in developed countries gillnets alone. Toothed whales (odontocetes) navigate and detect objects in their environment using echolocation sounds, although successful detection depends on the acoustic backscatter properties of an object. Gillnets typically have too weak backscatter for dolphins to detect the fishing gear in time to avoid entanglement. Electronic acoustic alarms (pingers) have been used successfully in some fisheries to reduce bycatch of some dolphin and porpoise species. However, to date little mitigation has been implemented. The main driver behind low uptake of pingers is cost, between US\$35-\$100 per 100m net. To address this, we have developed very low cost (US\$0-0.50 per 100m net) solutions using recycled plastic and glass bottles to create passive acoustic reflectors and mechanical pingers with similar acoustic emissions to existing, commercially available devices. We anticipate that these novel, low cost solutions will reduce cetacean bycatch globally and it is hoped that by publishing these results, fishermen and fisheries managers will be encouraged to take up these techniques and contribute to global conservation of cetacean species.

Introduction

Cetacean bycatch was first discovered in the late 1960s^{1,2}, investigated in the 1980s and 90s³ and a global estimate was provided in the 2000s⁴. The global ($\geq 3\%$ population mortality) estimate is not sustainable for dolphins and porpoises which have a limited recruitment of 1-4% per year⁵ and in the case of the Baiji-Yangtze River dolphin (*Lipotes vexifer*) contributed to the species' extinction⁶. Furthermore, the global bycatch estimate is likely a serious underestimate because it considers only 216,000 industrial gillnet vessels and does not include bycatch in small scale artisanal fisheries. There are around 3 million artisanal fishing vessels globally of which a large proportion use gillnets (www.fao.org). In addition, globally there are a very large number of illegal and unregistered vessels using gillnets for which no data are available^{7,8}. Wherever distributions of gillnets and cetacean populations overlap there is evidence of cetacean bycatch¹. Unlike many other threats facing cetaceans (e.g. pollutants or climate change) bycatch could be mitigated and reduced in the short-term.

Toothed whales (odontocetes) navigate and detect objects in their environment using echolocation (the process of transmitting and receiving high frequency sound for target detection)⁹. However, successful detection is influenced by the sound frequency and acoustic backscatter properties of an object. Gillnets' backscatter is typically too weak for dolphins to detect in time to avoid entanglement. Electronic acoustic alarms (pingers) have been demonstrated to reduce bycatch of several dolphin and porpoise species in developed country and small scale artisanal drift gillnet fisheries^{10,11}.

Pingers are currently mandatory in two US fisheries, the Gulf of Maine groundfish gillnet fishery (<http://www.greateratlantic.fisheries.noaa.gov/protected/porptrp/>) and the California/Oregon thresher shark/swordfish drift gillnet fishery (<http://www.nmfs.noaa.gov/pr/interactions/trt/poctrp.html>). In Europe, EC Regulation 812/2004 legislates that pingers should be used on gillnets fished by EU vessels larger than 12m. However, compliance is very low in both EU fisheries and in the US New England fishery, most likely driven by cost. This is also why currently available pingers will never be a viable solution in developing countries where fishermen have little financial means to pay for bycatch mitigation.

Artisanal fishers use drift nets at the surface and sink/set gillnets anchored near the seabed¹². Both are large mesh (10-20cm stretched mesh) and made of mono-or multi-filament nylon which has acoustic impedance similar to sea water and presents limited acoustic backscatter for detection by cetaceans. Both

are often set at night, limiting visual detection. There is therefore a strong incentive to develop minimum cost methods to mitigate cetacean bycatch in artisanal gillnet fisheries with potential implications for developed country fisheries.

Empty, used, glass and plastic drink bottles are available at minimum cost globally and this study demonstrates how they can be used to construct effective acoustic alarms to make cetaceans aware of the presence of gillnets.

Methods, Results & Application

The first and simplest design placed scrap metal bolts in empty glass bottles (350ml) which created high frequency acoustic emissions as the bolts moved inside the bottles. Initial trials showed that 40mm bolts with hexagonal heads produced useful sound levels with little risk of getting stuck in the bottle. The bottles were sealed by reusing the original cap secured with electrical tape. These bottles would be attached horizontally along the top of the net between floats and emit sound as the net and bottles are moved by surface waves and currents. A second glass bottle design, has the bolt instead suspended on a string inside the bottle, attached by a screw to a cork manufactured from disused styrofoam floats (Figure 1a). In this design, the bottles are mounted vertically with the neck of the bottle tied to the net head rope and the bottom of the bottle to the net mesh (Figure 1b). When the bottle moves, the string inside acts as a pendulum to provide mechanical amplification. Hence the bolt hits the side of the bottle with greater force and produces higher acoustic emission levels than in the initial design.

The acoustic emissions of the glass bottle were measured by a broad-band calibrated hydrophone in an anechoic testing tank (3mx3mx1.5m) at Newcastle University, UK. Both designs were suspended in the tank 0.5m from a Reson TC4032 hydrophone, located 0.5m below the water surface, connected to a VP2000 preamplifier and digital storage oscilloscope. Motions were then manually induced to simulate being attached to the headrope of a driftnet in low sea state.

The initial glass bottle design produced random “clinking” sounds of up to 10 milliseconds duration with a spectral density of up to 130 dB re 1uPa/VHz @ 1m below 10 kHz and audible components up to 30 kHz. The second, pendulum based design produced similar signals but with a higher overall output level and spectral density approaching 160 dB re 1uPa/VHz @ 1m as shown in Figures 2a and 2b. For comparison, we tested one of the most commonly used 10 kHz commercial electronic pingers. This pinger emitted a 300ms duration harmonic series with 10 kHz fundamental, every 4 seconds and the maximum spectral density did not exceed 140 dB re 1uPa/VHz @ 1m (Figures 2c and 2d). Other commercial pingers emit 50ms - 1s duration sounds at 3-160 kHz, continuous or random, in 1-30s intervals¹³.

Dolphins are able to detect gillnets with an estimated target strength of -62 to -46 dB at distances between 25 and 55m in low noise environments^{14,15}, but the detection distance will likely be significantly reduced at sea depending on environmental conditions (e.g. natural and anthropogenic noise levels) and whether and how the animal is using its echolocation. The nature of some entanglements indicates that dolphins may detect the nets, but that the detection is too close to the net for animals travelling at speed to avoid entanglement^{14,15}. This suggests that any increase in the sonar target strength of the net would likely result in a reduced bycatch risk.

One of the strongest sonar targets in water is an air bubble due to the acoustic impedance mismatch at the air-water interface. We investigated the use of air-filled plastic water bottles (500ml and 2l) as a sonar target. The plastic material of the bottle has similar acoustic impedance to water and hence the reflection would be produced by the air inside the bottle. We synthesized a typical dolphin echolocation signal, a broadband “click” signal centred on 70 kHz (Figure 3), which we transmitted in the test tank and then recorded the sound reflected by the plastic bottle. The same recording equipment and distance was used as described above for the glass bottles.

Both plastic bottles produced strong reflections of the simulated dolphin click (Figures 4 and 4b). As expected the larger 2l bottle was a stronger target (-16 dB target strength) compared to the smaller 500ml bottle (-27 dB target strength). The reflection by the smaller bottle is comparable to that of a 100cm yellowfin tuna (*Thunnus albacares*)¹⁶.

The sonar target produced by the plastic bottles in the nets should be large enough to be easily detectable by a dolphin's echolocation. The smaller plastic bottles used here (0.5l) represent a 100-1000-fold increase (given that dB is a logarithmic scale) in target strength compared to gillnets' typical target strength (-27dB for the plastic bottle compared to -62 to -46dB for gillnet).

The logistics of using the glass and plastic bottle alarms were tested on artisanal driftnets in Zanzibar, East Africa. The results provided information for attachment and deployment of the bottles in the fishery and confirmed the feasibility of the mitigation approach. The next step will be to trial the bottles in selected fisheries to evaluate their effect on target and bycatch species catch.

The production cost for one glass bottle pendulum pinger is \$0-0.50 depending on refund value for empty bottles, one scrap metal bolt and nut, a single small screw, and string for the pendulum and to tie the bottle to the headrope of the net. The production cost of the plastic bottle sonar target (\$0-0.25) also depends on refund value and the cost of a piece of string to fit the bottle to the bottom of the net. This represents a vast cost saving compared to using commercially available pingers and would remove the current economic obstacle for fishermen to implement bycatch mitigation. Given the acoustic output of the glass bottle pinger and the target strength of the plastic bottle, we recommend that both deterrent devices be spaced at intervals of 50m along a drift net.

The glass bottles emit sound in a similar frequency band to commonly used electronic pingers¹⁶. Although the duration of each emission is significantly shorter (10 ms compared to 300 ms), the sound produced has a higher peak power and spectral density, emissions are more frequent and random, and energy is spread more evenly over a wider spectrum compared to the tones/harmonic series produced by electronic pingers. Hence, in addition to the cost saving, the glass bottle pingers have many signal characteristics found in more sophisticated electronic deterrent devices and the potential to mitigate bycatch of a range of cetacean species.

Competing interests

The authors declare no competing interests.

Acknowledgements

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Figure Legends:

Figure 1. Glass bottle acoustic alarm (pinger), a) glass bottle pinger with a pendulum made from a single bolt and string attached to the cork at the top of the bottle, b) glass bottle pinger attached to the head rope of a drift gillnet.

Figure 2. Tank testing of pinger devices: a) Hydrophone signal from glass bottle pinger b) spectrum of glass bottle pinger c) hydrophone signal from electronic pinger d) spectrum of electronic pinger.

Figure 3. Time domain plot showing the synthesised 70 kHz broadband pulse waveform used to simulate a typical dolphin echolocation click.

Figure 4. Time domain plots of the backscatter signals from; a) small 0.5l plastic bottle (-27 dB target strength), and b) large 2l plastic bottle (-16 dB target strength).



Fig. 1a

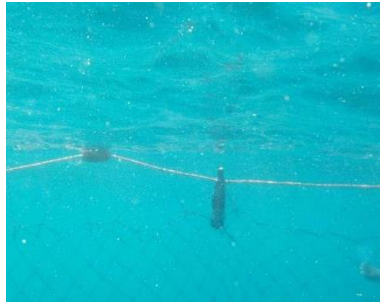


Fig. 1b

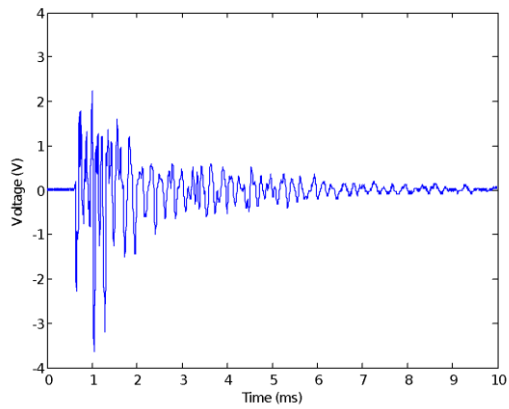


Fig. 2a

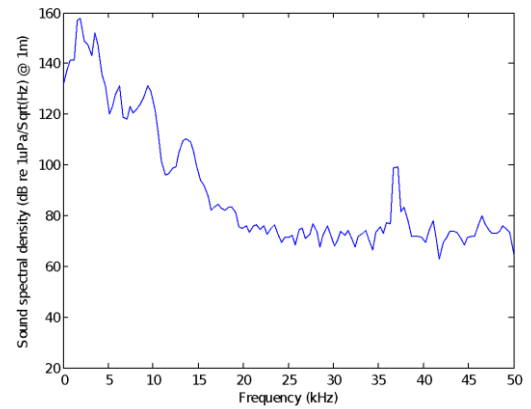


Fig. 2b

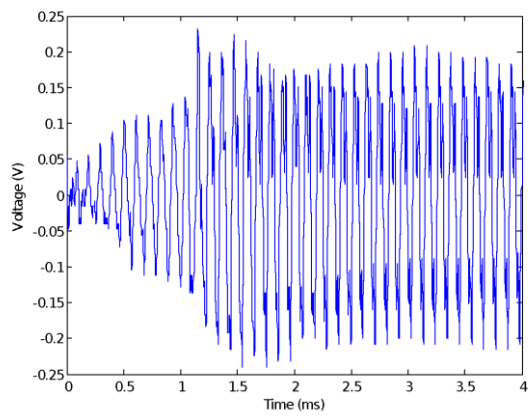


Fig. 2c

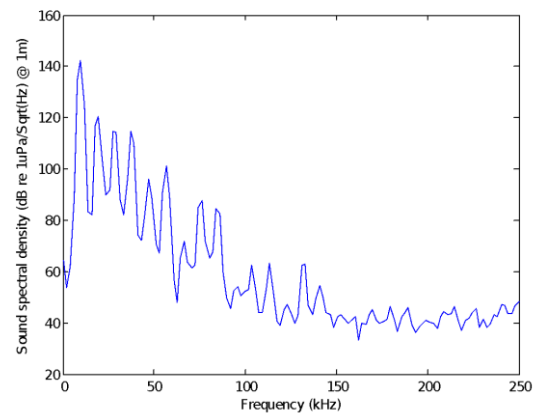


Fig. 2d

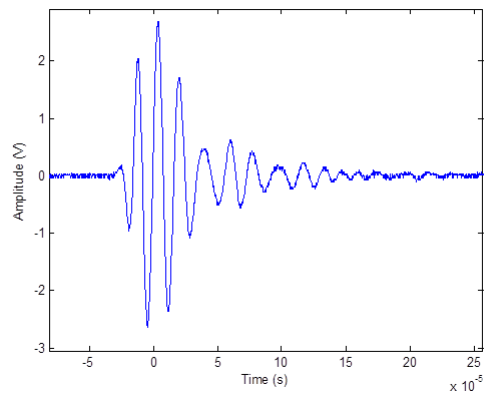


Fig. 3

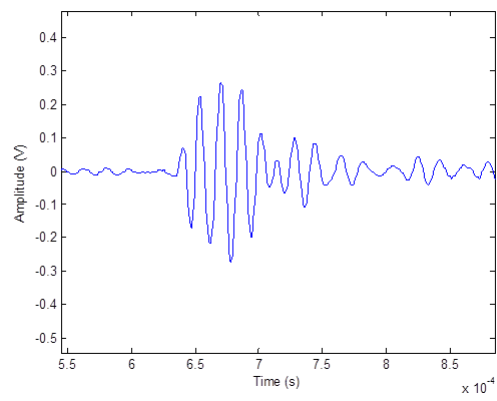


Fig. 4a

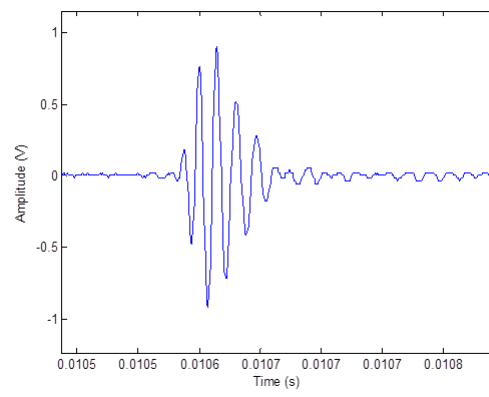


Fig. 4b



The Effect of Two Low-Cost Bycatch Mitigation Devices on the Catch of Small-Scale Fisheries in Zanzibar, Tanzania.

Louise Hopkins BSc

140385786

MSc Wildlife Management

BIO8060: Wildlife Research Project

Supervisor: Dr Per Berggren

September 2018

Abstract

Bycatch is known to be one of the greatest threats to marine mammal populations, with increasing recognition that small-scale fisheries play a large role in this. The use of bycatch mitigation devices is mandatory in many areas of the world for larger vessels; however, challenges such as high cost of these devices prevent them from being adequately utilised by small-scale fisheries. Further challenges arise where fishermen are unwilling to use devices if they are believed to have a detrimental impact upon their catch. This study looked at the impacts of two low-cost bycatch mitigation devices upon the target catch in small-scale fisheries of Zanzibar during January 2017 – March 2017. The devices were made from recycled glass and plastic bottles, creating an acoustic alarm and an acoustic reflector. Results indicated that there was no significant difference between the catch from the two experimental treatments and the control. Providing evidence that such devices will not have a detrimental effect on fisherman. They are also cheap to implement thus increasing the likelihood of implementation and compliance with regulations by small-scale fisheries.

Introduction

The incidental catch, or bycatch, is recognised as one of the biggest threats facing marine mammals (Read et al, 2006; Peckham et al, 2007; Reeves et al, 2013). An estimate made by Read et al (2006) put the number of marine mammals being killed each year worldwide by incidental catch at more than 650,000 individuals. A number of studies have identified that gillnet fisheries cause more mortalities than other fishing methods such as trawling (Read et al, 2006; Yousuf et al, 2008; Reeves et al, 2013).

Being a K-selected species, marine mammals are particularly vulnerable to bycatch due to having a long life history, late maturity and a long life span (Read, 2008; Brown et al, 2014; Capietto et al, 2014; Temple et al, 2017). As a result, any non-natural mortalities can have significant impacts on a populations' potential growth and ability to recover from such events (Read, 2008; Temple et al, 2017). The removal of individuals impacts small populations more heavily than it does large populations, due to a larger proportion of the population being removed (Read, 2008; Komoroske and Lewison, 2015). Such non-natural mortalities from anthropogenic impacts can greatly reduce and threaten populations, and in some instances lead to extirpation or extinction. This has already occurred with the Yangtze River dolphin (*Lipotes vexillifer*), and other species are at a great risk of following such as the vaquita porpoise (*Phocoena sinus*) and the North-Atlantic right whale (*Eubalaena glacialis*). (Cooke, 2018; Smith et al, 2017)

Marine mammal species feed in most trophic levels; as such they play an important role in the wider ecosystem, removal of these species not only impacts upon their own populations but can have a number of top-down effects on the food chain (Bowen, 1997; Capietto et al, 2014; Kiszka et al, 2015; Komoroske and Lewison, 2015). This is especially important in areas with high species richness such as the South Western Indian Ocean, an area utilised heavily by small-scale fisheries (Temple et al, 2017). Little is known regarding interactions between marine mammal species and small-scale fisheries, these fisheries are mainly found in developing countries where data and management of bycatch is scarce (Peckham et al, 2007; Mangel et al, 2013; Komoroske and Lewison, 2015). Much more is known about bycatch caused by commercial and industrial fishing (Peckham et al, 2007; Temple et al, 2017). These data are often extrapolated to try and understand global trends in bycatch (Read et al, 2006), yet 99% of the 51 million fishers worldwide are employed by small-scale fisheries, meaning these estimates are unlikely to be accurate (Peckham et al, 2007).

Most small-scale fisheries operate in coastal habitats often frequented by many marine mammal species (Read et al 2006; Peckham et al, 2007). Thanks to their low cost and easy deploy-ability from small vessels, gillnets are common fishing gear used in small-scale fisheries (Read, 2008; Yousuf et al, 2008). A study in Zanzibar, an area highly utilised by small-scale fisheries, found increased occurrence and foraging activity of dolphins occurring at night (Temple et al, 2016; Waring et al, 1990). In Zanzibar, the target catch of the drift gillnet fisheries here, such as tuna species, are in competition with dolphins for this prey. Here, nets are deployed in the evening and hauled in before dawn (Amir et al, 2002). These fishing methods are utilised as the target catch come closer to the surface during the hours of darkness in search of their prey. Unfortunately, as do their competitors, thus there is high potential for interaction between these fisheries and marine mammal species, with small-scale fishers now being recognised as having a significant impact upon them in many coastal areas (Amir et al, 2002; Peckham et al, 2007; Mangel et al, 2013; Komoroske and Lewison, 2015; Temple et al, 2017).

To reduce bycatch of cetacean species, some fisheries use acoustic deterrent devices known as “pingers” (Carretta et al, 2008; Mangel et al, 2013; Stone et al, 1997). These devices are attached to the nets and work by producing a 10-70kHz frequency sound, alerting nearby cetaceans to the presence of fishing gear, thus deterring them from the fishing gear preventing entanglement (Stone et al, 1997; Kastelein et al, 2007; Goetz et al, 2014). Previous studies have shown such devices to be effective in reducing bycatch for a wide range of different cetacean species (Stone et al, 1997; Carretta et al, 2008; Larsen and Eigaard, 2014). With bycatch decreasing from between 37% to 86%, variations in effectiveness result from differences in fishing characteristic (net type, target catch etc)

and pinger specifications (Mangel et al, 2013; Barlow and Cameron, 2003). Importantly, such devices have worked effectively to reduce bycatch in drift gillnet fisheries (Barlow and Cameron, 2006).

Acoustic pingers have become compulsory in many corners of the world. The EU introduced regulations in 2001 making the use of acoustic deterrent devices compulsory across the EU by 2007 (European Commission, 2003), similar regulations have been imposed in the USA (Marine Mammal Commission, 2018). These regulations stipulate that vessels above a certain size and using specific gears must fit and use acoustic deterrent devices, often stating the spacing requirements between each device, indicating how many are required (Marine Management Office, 2017). A number of studies have found that using more pingers and placing them closer together increases the effectiveness of this mitigation method in reducing bycatch (Barlow and Cameron, 2003; Mangel et al, 2013). It is important that any mitigation method remains as effective as possible, Carretta and Barlow (2011) found that cetaceans showed no signs of habituation to acoustic pingers over a 14 year period.

When it comes to using such devices in small-scale fisheries there are a number of barriers. Firstly, small-scale fisheries are poorly regulated, making it harder to enforce such regulations that make acoustic deterrent devices mandatory (Peckham et al, 2007; Read, 2008). There are two main issues with bycatch mitigation devices for small-scale fisheries, the first being the cost; acoustic pingers come at a high cost especially when considering the level of income for small-scale fisheries (Read, 2008; Mangel et al, 2013; Temple et al, 2017). This cost to small-scale fisheries further increases when a number of devices are required to make this mitigation method effective (Mangel et al, 2013).

Compliance with any such regulations is unlikely where it is seen to result in high costs to the fishermen and as such, are unpopular solutions (Read, 2008; Komoroske and Lewison, 2015). The second issue making compliance more challenging is if it is believed that the use of such devices could negatively affect the target catch (Komoroske and Lewison, 2015). Small-scale fisheries play a vital role in coastal communities, relying on the ocean for their main source of protein, three quarter of countries where this is the case are low income, food deficient countries (Teh et al, 2016; Temple et al, 2017). In Zanzibar increasing pressure upon coastal resources has resulted in overfishing leading to a decline in the productivity of the coastal ecosystems (Mkenda and Folmer, 2001; Silva, 2006). Thus any solutions to the issues of bycatch that will put further impact upon the utilisation of this resource are understandably unpopular.

A number of studies have looked at how acoustic pingers affect target species. A study by Kastelein et al (2007), conducted in a large tank, showed a variety of responses depending on the device being used and the frequency of this device, finding that higher frequencies had less effect on fish behaviour. This study looked at a range of species, for some such as cod, no response was observed to any of the devices (Kastelein et al, 2007). In contrast, a study conducted in the North Sea gillnet fisheries found a slightly higher number of cod being caught where acoustic pingers were used, although not significant it could suggest they were attracted to the devices (Larsen and Eigaard, 2014). One study that was conducted in small-scale fisheries of Peru found that the target catch of shark and ray species were unaffected (Mangel et al, 2013). This study used acoustic devices donated to the study. However, none of these studies have addressed the issue of cost, although they show the devices do not affect target catch, they are still costly to purchase. Studies have shown that acoustic pingers also reduce net damage caused by cetaceans, but given the low cost of gill nets this benefit is unlikely to outweigh the cost for these small-scale fisheries (Yousuf et al, 2008; Mangel et al, 2013). Given the income in such areas utilisation of such devices becomes more challenging.

To address such issues, this study aims to investigate if and how two low-cost mitigation devices impact upon the target catch of small-scale fisheries. The devices used here is an acoustic pinger made from a recycled glass bottle and an acoustic reflector made from a recycled plastic bottle. The fisheries involved in this study are based in Zanzibar, an area previously identified to have high levels of bycatch, high enough to be having a significant impact upon the local cetacean populations (Amir et al, 2002). The fisheries involved in this study are driftnets used to target species such as tuna, billfish and kingfish; although billfish species are not as commonly caught they are highly sought after due to their high value. Given the findings of previous studies it is hypothesised that there will be no difference found between the catch gained from the hauls from the control treatment and the two experimental treatments. This study does not look to investigate the effectiveness of the mitigation devices themselves for the reduction of bycatch.

To investigate whether the use of such devices resulting in the loss of time for the fishermen, this study will also include handling time of setting and hauling the nets in the analysis. Where the time difference is significant, this could result in lost fishing time for the fishermen thus making them an unpopular solution. It is hypothesised that there will be no significant difference in the time taken to deploy and haul in the nets.

Methods

STUDY SITE

The study was conducted between January and March 2017 on the south west coast of Unguja Island, Zanzibar around 40 miles off the coast of Tanzania (Figure 1). Fishermen involved in the study were from the villages of Mkunguni and Dimbani in the Kizimkazi area. Fishermen in this area use multi-filament driftnets with a mesh size of 7x7 cm. The length of the net used by each vessel varies between 400 – 1100m. These driftnets are used to target large pelagic species such as tuna, billfish and kingfish. Fishing is conducted at night, nets are set before sunset and hauled before dawn, the nets are attached to a buoy at one end and the other to the vessel, enabling the net and vessel to drift with the current. The study period was split into three seasons that coincided with the darker phases of the moon.

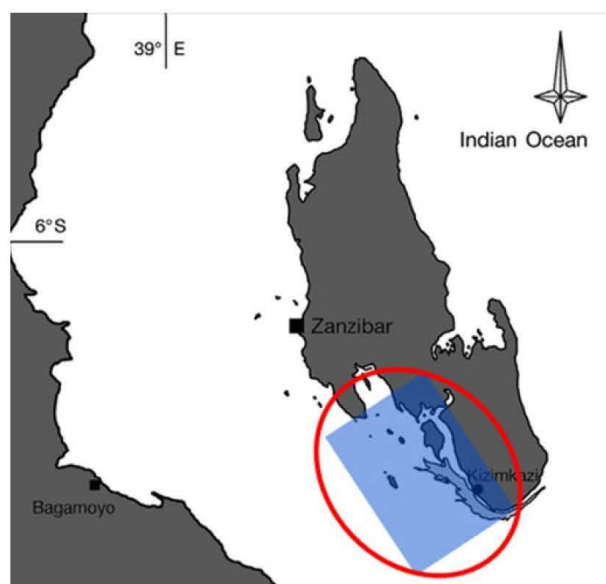


Figure 1 – Map of Unguja Island, Zanzibar, the highlighted area shows the location of where fishing took place, fishermen involved in the study were from Dimbani and Mkunguni, two villages in the Kizimkazi area in the south of Unguja Island. Fishing took place between January and March 2017.

MITIGATION DEVICES

Two different mitigation devices were made for this study using recycled glass (Figure 2) and plastic bottles (Figure 3) to create a passive acoustic reflector and an acoustic alarm. The sound and acoustic reflections produced by the bottles, respectively, should allow dolphins to detect them at

sufficient distances to prevent entanglement in the nets. A metal bolt fitted inside of the glass bottle worked by producing random “clinking” sounds with a frequency ranging between 3-15 kHz. The sound produced from this device lasts a few milliseconds with a source level at <10kHz of 130-160dB re 1uPa/VHz @ 1m. As a comparison, commercially available devices such as the Future Ocean 10kHz acoustic alarm produce a 300 ms ping every 4 seconds with a source level of 132dB re 1uPa/VHz @ 1m (McGarry et al, 2018). Exposing the plastic bottle acoustic reflectors to a transmitted sound of 70kHz broadband pulse, to simulate a dolphin echolocation click, produced a -27dB reflection. Given that the acoustic reflection created by gillnets is -62 to -46 kHz, this represents a 100 – 1000 fold increase in target strength.



Figure 2 - Glass bottle alarm device attached to the headline, device is fitted with a metal bolt producing random “clinking” noise to deter marine mammals from the nets.



Figure 3 - Plastic bottle reflector attached to the sinkline, the device reflects echolocation clicks allowing detection of the nets by marine mammals.

EXPERIMENTAL DESIGN

Thirty-six fishermen were involved in the trials; each randomly assigned to one of the three treatments: control, glass bottles or plastic bottles. After each month the fisherman were reassigned to a different treatment. Those assigned to the control group fished as normal, those assigned to the glass bottles were instructed to attach the devices at 50m intervals along the headline of the net. Those assigned to the plastic bottles devices were instructed to attach the devices to their nets at 50m intervals along the sinkline of the net. Fishermen in both the glass and plastic treatments otherwise fished as normal.

DATA COLLECTION

Each fisherman was assigned an independent observer, after each season the observers were reassigned to a different fisherman to avoid bias. Observers were instructed to document the start and finish time of the fishermen setting and hauling the nets on an effort form for each haul. As well as documenting the time this occurred observers were also provided with a camera to take a photograph of this occurring allowing visual verification of the correct treatment being used. Further to this the length of net was also documented, the information collected here allowed calculation of the soak time and catch per unit effort (CPUE).

Observers also documented what species were caught (using local names) and the number of each caught on a catch form for each haul. Length measurements were also documented for each individual, where a school of fish were caught only one individual was measured. Observers were instructed to take a picture showing the lateral side of the species caught to allow species identification to be completed.

DATA VALIDATION

The use of cameras by the observers enabled verification of the information documented for each haul. Each camera was individually identified and the camera being used was documented for each haul. To verify the catch forms, data for each haul was checked against the relevant photographs. The catch form was deemed to be accurate if the photograph taken confirmed the data documented, or if the photograph was unable to verify or refute the information documented on the catch form, i.e. there was no photograph taken, or the image quality was low. In the event that the photographs taken conflicted with the information on the catch form, i.e. more individuals / species in the photograph than documented, then the photography was used to inform on the catch composition (species and number caught) for that haul.

All species noted on to the catch forms were identified by their local name, some of these local names are often used for multiple species and some species have multiple local names, further to this the name used for a species can often differ between villages. To ensure correct identification, the photographs were used in order to identify the species as close to species level as possible (Appendix 1). Where it was not possible to identify the catch due to a lack of photographs or low image quality, the haul in question was removed prior to data analysis. Once identified, the catch was then grouped into the three target catch groups: tuna (*Euthynnus affinis*; *Thunnus albacore*; *katsuwonus pelamis*); billfish (*Istiophorus platypterus*; *Makaira indica*; *Xiphias gladius*); and kingfish

(*Scomberomorus commerson*). The total number of hauls from the study was 1107, due to the aforementioned reasons, it was possible to analyse the catch composition of 860 hauls.

From the remaining 860 hauls the time it took to set the net and haul them in was calculate for each haul as well as the soak time. Soak time was calculated using the set finish time and the haul start time. These were used as during these two times there is no interference with the net from setting or hauling it. Where any of the times were not documented for the set and haul times, the mean set and haul times were calculated and used to fill these data gaps, this was done for each of the three treatments to enable analysis of set and haul times for each treatment. This was also done were the data documented gave exceptionally high or low set, haul of soak times. For set and haul times, lower values were left as they were as some nets were smaller requiring less time to set or haul, however any values high than three hours were changed to the mean. For the soak time, any values lower than one hour and any higher than 15 hours were changed to the mean. Of the 860 hauls this only had to be done for 60.

Prior to data analysis, catch per unit effort (CPUE) was determined by multiplying the net length (Km) by the soak time (h), the number of individuals caught was then divided by this. This was done for each haul, calculating the CPUE for the total catch and each of the three target catch groups. This was done to control for any variance in the effort between the hauls caused by high variance in the soak time and the length of the nets used.

DATA ANALYSIS

Visualisation of the data identified a small number of outliers (n=5), such as those highlighted in Figure 4, these were due to a large school of fish being caught in a number of hauls. These outliers were retained for the analysis as they are a true reflection of the catch composition in the area.

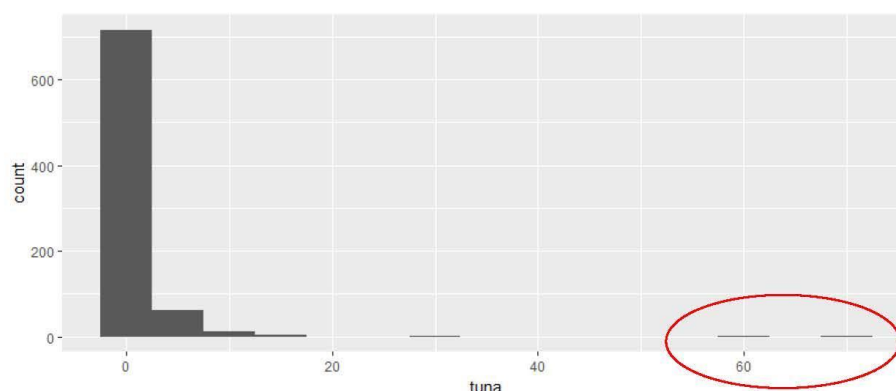


Figure 4 – Frequency of tuna haul catch size gained from tuna catch per unit effort data. Outliers from this data are circled, these represent two large hauls of tuna (n=71 and 90 individuals).

The effect of the two mitigation devices upon the target catch was analysed using a generalised linear mixed effect model (GLMER) created in R studio version 0.99.902, using the R programming language. Preliminary analysis was done using a generalised linear model (GLM) in order to identify which of the random effects were significant, where this was not the case they were removed. Using this, a GLMER was created for each of the following:

- **Total catch** (fixed effect: Treatment; random effects: fisher and soak time/Km; family: negative binomial)
- **Tuna** (fixed effect: Treatment; random effects: fisher; family: negative binomial);
- **Billfish** (fixed effect: Treatment; random effects: fisher; family: binomial);
- **Kingfish** (fixed effect: Treatment; random effects: fisher; family: negative binomial);
- **Set time** (fixed effect: Treatment; random effects: fisher; family: negative binomial);
- **Haul time** (fixed effect: Treatment; random effects: fisher; family: negative binomial).

The R package “emmeans” was then used to investigate potential differences between each of the three methods for the catch in terms of CPUE and time for the set and haul times; this is done by comparing the estimated marginal means. Goodness of fit was assessed using QQ-plots to plot the model residuals against the normal distribution showing a good fit.

Results

Of the 860 hauls that could be analysed 321 were in the control treatment, 292 in the glass treatment and 247 hauls for the plastic treatment. A total of 26 different species were identified, 19

of which were caught in the control treatment, 21 in the glass treatment and 23 in the plastic. The results from testing for differences in the catch among these three treatments are shown in Table 1.

Table1: Results from GLMER analysis and comparison of estimated marginal means testing for differences in catch between the three treatments, including standard error values and p values.

	<i>Estimate (km h⁻¹)</i>			<i>P-value</i>
Total Catch				
<i>Control - Glass</i>	-0.290	(-0.448 - -0.132)		0.066
<i>Control - Plastic</i>	-0.259	(-0.435 - -0.082)		0.142
<i>Glass - Plastic</i>	0.031	(-0.137 - 0.199)		0.853
Tuna				
<i>Control - Glass</i>	-0.168	(-0.379 - 0.043)		0.426
<i>Control - Plastic</i>	-0.200	(-0.431 - 0.031)		0.387
<i>Glass - Plastic</i>	-0.032	(-0.253 - 0.189)		0.885
Billfish				
<i>Control - Glass</i>	-0.368	(-0.692 - -0.044)		0.256
<i>Control - Plastic</i>	-0.325	(-0.691 - 0.041)		0.375
<i>Glass - Plastic</i>	0.043	(-0.310 - 0.396)		0.903
Kingfish				
<i>Control - Glass</i>	-0.586	(-0.974 - -0.197)		0.132
<i>Control - Plastic</i>	-0.374	(-0.792 - 0.044)		0.371
<i>Glass - Plastic</i>	0.212	(-0.162 - 0.586)		0.571

For the overall total catch from all 860 hauls, the results in Table 1 show that no significant difference was identified between the experimental treatments and the control treatments. Similarly, no significant difference was found between the experimental treatments for any of the target catch species groups either. Although a statistical trend ($p=0.05-0.10$) was present where the total catch in the glass bottle treatment was higher than that of the control for the overall catch ($p=0.066$). However, no other such statistical trends were identified.

Further to this no significant difference was identified between the two experimental treatments themselves, for both the overall catch and each of the three target catch groups.

Table 2: Results from GLMER analysis and comparison of estimated marginal means testing for differences in net set and haul time between the three treatments, including standard error values and p values

	<i>Estimate (h)</i>		<i>± 1 SE</i>		<i>P-value</i>
Set Time					
<i>Control - Glass</i>	-0.085	-0.171	-	0.001	0.322
<i>Control - Plastic</i>	-0.104	-0.197	-	-0.011	0.264
<i>Glass - Plastic</i>	-0.018	-0.113	-	0.077	0.846
Haul Time					
<i>Control - Glass</i>	-0.043	-0.117	-	0.031	0.564
<i>Control - Plastic</i>	-0.010	-0.090	-	0.070	0.898
<i>Glass - Plastic</i>	0.033	-0.048	-	0.113	0.687

The results shown in Table 2 indicate that there was no significant difference between the two experimental treatments and the control treatment in the time taken to either set or haul the nets. Further to this, no significant difference was identified between the two experimental treatments themselves and no statistical trends found.

Discussion

CATCH COMPOSITION

Previous studies have been conducted to look at the impacts of mitigation devices on the target catch of small-scale fisheries (Mangel et al, 2013). However this study is the first to investigate this in regards to the low-cost devices tested here. Compared to the control treatment, neither the glass bottle devices nor the plastic bottle devices have significantly impacted upon the catch. Further to this, when comparing the glass bottles devices with the plastic bottle devices again there was no significant difference found for both the total catch and all three of the target catch groups. Only one statistical trend ($p=0.05-0.10$) was found for the overall catch in the glass bottle treatment being higher than that of the control treatment ($p=0.066$).

Although no significant difference was found between the two experimental treatments used here, it should be noted that all outliers resulted from large schools of kawakawa tuna being caught in nets utilising the plastic bottle devices, with haul ($n=5$) sizes ranging from 37 to 90 individuals being caught across. Further research is necessary to investigate whether this occurred by chance or if this resulted from the use of the plastic bottle devices.

As for the glass bottle devices, previous studies investigating the reaction of different fish species to electronic pinger devices have found mixed results where some show there was no impact (Bordino et al, 2002; Carlstrom et al, 2002; Gazo et al, 2008; Gonener and Bilgin, 2009; Mangel et al, 2013; Trippel et al, 1999) and others found a negative impact with fewer fish being caught (Kraus et al, 1997; Trippel et al, 1999). In contrast, some other studies have found that where pingers are used, there was a significant increase in the number of fish caught, suggesting they were attracted to the net by the noise (Busciano et al, 2009). This study provides further support showing that acoustic pinger devices do not affect the catch of fish species such as tuna, billfish and kingfish.

This is not surprising for species such as tuna given that the hearing of tuna species such as the yellowfin tuna (*Thunnus albacares*) is most sensitive between 200 – 800 Hz (Finneran et al, 2000) and the sound produced by the glass bottle devices had a frequency ranging between 3-15 kHz. Further to this, of all the species caught during this study, the most common was the kawakawa tuna (*Euthynnus affinis*) a species likely to be less sensitive to sounds given the lack of a swim bladder (Dale et al, 2015; Luna, 2018). In comparison, yellowfin tuna do have a swim bladder and, especially larger individuals; this could result in greater sensitivity of hearing (Finneran et al, 2000; Schaefer and Oliver, 2000).

METHODOLOGIES

A notable strength of this study was the use of independent observers, previous studies have identified the importance of using observers to collect fisheries data such as bycatch and catch composition in order to reduce any bias that may result from this being collected by the fisheries themselves (Reeves et al, 2013). Any bias was further reduced by the requirement of photographs. Although training was provided to the observer on the use of the camera, the photographs gained identified that further training was required. Low quality images resulted from the photographs being taken at night without utilisation of the on camera flash further complicated where sea conditions were unfavourable. This did introduce an element of bias to the catch composition documented here as it is likely that species may have been missed due to not being able to identify them as a result of the image quality. This mainly affected species that were not caught as often, thus there are only a limited number of photographs. As such, it is likely that those species missed from this study were rarely caught and thus unlikely to be species of importance to the fisheries. However, low image quality also affected the hard to identify species, where a number of species are hard to differentiate from one another, grouping in to target catch groups eliminated this issue at the data analysis stage.

UTILISATION

As mentioned earlier, the high cost of commercially available bycatch mitigation devices and the possibility of them impacting upon a fisheries target catch are the two main issues for fishermen when it comes to the reduction of marine mammal bycatch (Read, 2008; Komoroske and Lewison, 2015). These issues make it unlikely that fishermen will utilise such devices and comply with regulation put in place for the protection of marine mammal species. This study has successfully shown that this affordable solution to bycatch mitigation will not have any detrimental effect to the target catch of small-scale fisheries.

Further to this, the analysis of net set and haul times found no significant difference between the two experimental treatments and the control treatment. These results show that the use of these two low-cost devices will not result in lost time to the fisherman. This is unsurprising, attaching the devices to the nets may be time consuming, however, given the ease at which driftnets can be deployed (Read, 2008; Yousuf et al, 2008), there is no reason for the mitigation devices to result in a significantly higher handling time, and once they're attached there is no need to be removed. Although, were these devices to be used on more complex fishing gear such as longlines, this could result in increased time required to set and haul the nets.

These results further reinforce that the utilisation of both the glass and plastic bottle devices will not negatively impact the fishermen improving the chances of such devices being utilised. Given that fisheries in Zanzibar are primarily small-scale (Barrowclift et al, 2017), it is unlikely that commercially available mitigation devices are being utilised here given their cost. The bycatch mitigation devices tested here provide an important step forward in the mitigation of bycatch for small-scale fisheries, especially in developing regions such as Zanzibar. It is important, however, that further studies are conducted to further understand the impacts of such devices on a range of other target catch species. More importantly, throughout this study no dolphins were caught in any of the three treatments preventing any evaluation of their ability to reduce bycatch, further trials are required to allow analysis of whether these devices are capable of reducing dolphin bycatch.

During this study a number of sharks and rays were caught, but not substantial enough numbers to conduct an analysis. Zanzibar has a number of small-scale fisheries that target sharks and ray species, particularly in austral summer (Kiszka and van der Elst, 2015). To target such species, bottom-set gillnets are often used as well as longlines. Therefore, further studies are required to investigate the capability of the glass and plastic bottle devices in reducing bycatch in other types of fishing gear commonly used by small-scale fisheries, as well as whether or not they impact upon the target catch. Implementing the use of the glass and plastic bottle devices is likely to be time-

consuming where longlines are used and thus unfavourable, however, they may be useful for bottom-set gillnets.

The results gained here show great promise for the implementation of both the glass bottle acoustic alarm and the plastic bottle acoustic reflector devices in small-scale fisheries, given that small-scale fisheries are so numerous world-wide, this is likely to be a huge step forward in the mitigation of marine mammal bycatch globally. These devices allow fishermen in developing countries to easily comply with regulations in place to protect marine-mammal populations. It is important that improvements are also made to the enforcement of regulations governing the management of bycatch and marine mammal protection to ensure the use of such devices is taken up by these fisheries.

Acknowledgements

I would like to thank my supervisor Dr Per Berggren for all of his help alongside Andrew Temple. I would also like to thank Matthew Sharpe for collating and organising the data used for this project. As well as those mentioned, I would like to thank the fishermen and independent observer involved in the research.

Appendix

Appendix 1: Species identified from the local names documented on catch forms, using photographs for accurate species identification. Including number of each species caught in each of the three treatments and overall. (Anam and Mostarda, 2012; Bianchi, 1985)

Local Name	Common Name	Family	Species	Control	Glass	Plastic	Total
<i>Papa sp.</i>	Thresher Shark	Alopiidae	<i>Alopius vulpinus</i>	0	1	0	1
<i>Mchobwa / Mkule</i>	Flat Needlefish	Belontiidae	<i>Ablennes hians</i>	3	1	2	6
<i>Mchobwa / Mkule</i>	Hound Needlefish	Belontiidae	<i>Tylosurus crocodilus crocodilus</i>	1	2	4	7
<i>Hubwi / Mbonga /Kolekole</i>	Bigeye Trevally	Carangidae	<i>Caranx sexfasciatus</i>	4	3	10	17
<i>Kihahua / Kihamama</i>	Blackbanded Trevally	Carangidae	<i>Seriolina nigrofasciata</i>	3	5	26	34
<i>TanTan</i>	Shrimp Scad	Carangidae	<i>Alepes djedaba</i>	5	0	21	26
<i>Papa sp. / Kinegwwe</i>	Silky Shark	Carcharinidae	<i>Carcharhinus falciformis</i>	6	6	1	13
<i>Mkongwe</i>	Dorab Wolf Herring	Chirocentridae	<i>Chirocentrus dorab</i>	1	2	1	4
<i>Panje</i>	Common Dolphinfish	Coryphaenidae	<i>Coryphaena hippurus</i>	53	62	26	141
<i>Nungu</i>	Spot-fin Porcupinefish	Diodontidae	<i>Diodon hystrix</i>	0	1	7	8
<i>Chazo</i>	Live Sharksucker	Echeneidae	<i>Echeneis naucrates</i>	0	0	2	2
<i>Hanisi</i>	Tenpounder	Elopiidae	<i>Elops machnata</i>	0	0	1	1
<i>Panzi Maji</i>	Pharao Flyingfish	Exocoetidae	<i>Cypselurus naresii</i>	0	1	0	1
<i>Papa sp.</i>	Snaggletooth Shark	Hemigaleidae	<i>Hemipristis elongata</i>	0	1	1	2
<i>Mbase</i>	Sailfish	Istiophoridae	<i>Istiophorus platypterus</i>	22	32	23	77
<i>Nduaro</i>	Black Marlin	Istiophoridae	<i>Makaira indica</i>	4	4	1	9
<i>Kanda</i>	Tripletail	Lobotidae	<i>Lobotes surinamensis</i>	11	8	5	24
<i>Pungu / Jeti /Taa sp. / Tenga</i>	Bentfin Devil Ray	Myliobatidae	<i>Mobula thurstoni</i>	1	1	1	3
<i>Pungu</i>	Flapnose Ray	Myliobatidae	<i>Rhinoptera javanica</i>	0	1	0	1
<i>Songoro</i>	Cobia	Rachycentridae	<i>Rachycentron canadum</i>	6	16	13	35
<i>Chape / Sehewa</i>	Kawakawa Tuna	Scombridae	<i>Euthynnus affinis</i>	416	428	779	1623
<i>Jodari</i>	Yellowfin Tuna	Scombridae	<i>Thunnus albacares</i>	63	81	69	213
<i>Nguru</i>	Narrow-barred Spanish Mackerel	Scombridae	<i>Scomberomorus commerson</i>	30	34	27	91
<i>Zanuba / Sehewa</i>	Skipjack Tuna	Scombridae	<i>Katsuwonus pelamis</i>	18	25	1	44
<i>Mzia</i>	Blackfin Barracuda	Sphyraenidae	<i>Sphyraena genie</i>	2	0	2	4
<i>Samsuri</i>	Swordfish	Xiphiidae	<i>Xiphias gladius</i>	1	0	1	2
	Total			650	715	1024	2389

Annex 12: Mohamed Athman Mohamed MSc Dissertation manuscript

ASSESSMENT OF MARINE MAMMALS AND SEA TURTLES BY-CATCH IN THE ARTISANAL FISHERY ALONG NORTH COAST KENYA

Mohamed Athman Mohamed ¹, Mlewa C. Mwatete ², Nina N. Wambiji ³, Sergi Pérez-Jorge ⁴

Corresponding Author: Department of Biological Sciences, Pwani University,

P.O. Box 195-80108, Kilifi, Kenya.

Mobile No. +254 721500621 or +254 763500621

Email: mohammadathman@yahoo.com

School of Graduate Studies, Pwani University,

P.O. Box 195- 80108, Kilifi, Kenya.

Mobile No. +254 714356322/ 734970121

Email: m.mwatete@pu.ac.ke

Kenya Marine and Fisheries Research Institute,

P.O. Box 81561-80100, Mombasa, Kenya.

Mobile No. +254 706214929

Email: nwambiji@gmail.com

Marine and Environmental Sciences

Centre and Institute of Marine Research, University of the Azores,

Rua Frederico Machado 4, 9901-862 Horta, Portugal.

Email: sergiperezjorge@gmail.com

Abstract

The populations of marine mammals and sea turtles are declining around the world because of their high vulnerability to fishing mortality and they have been reported as by-catch globally. However, specific information on the magnitude of by-catch is still largely lacking in artisanal fisheries. The aim of this study was to assess marine mammals and sea turtles by-catch in artisanal bottom-set and drift gillnet fisheries along the north-coast of Kenya. The study was conducted between May and November 2016 in the small scale fisheries of Mombasa, Malindi and Lamu. Questionnaires were used to gather data and guide books for identification of species. Twelve (12) species were identified as by-catch including 7 species of marine mammals and 5 species of sea turtles. The most frequently caught species of sea turtles were the green turtles *Chelonia mydas* (50%), hawksbill *Eretmochelys coriacea* (18%), olive ridley *Lepidochelys olivacea* (18%) and leatherback *Dermochelys imbricata* (9%). By-catch incidences were significantly ($p < 0.05$) higher during north-east monsoon than south-east monsoon. Drift gillnet fisheries reported higher by-catch incidences of marine mammals while bottom-set gillnets reported higher by-catch incidences of sea turtles and are, therefore, both considered as threat to these marine mega-fauna along the Kenya coast.

Keywords: By-catch, marine mammals, sea turtles, Western Indian Ocean, north-coast of Kenya, gillnets.

Introduction

Several species of marine mammals have been recorded in a number of coastal areas and in various studies (e.g. Wamukoya et al., 1996; Husar, 1975; DeMaster et al., 2001; Kiszka et al., 2008; Kiszka et al., 2009a; Kiszka et al., 2009b; Kiszka, 2012a; Kiszka, 2012b; Pérez-Jorge et al., 2015; Pérez-Jorge et al., 2016). Marine mammals are considered very important because they contribute high flow of energy in the marine ecosystem. Cetaceans as a group and sperm whale alone, can consume a greater quantity of prey than all teleost fish combined. Some marine mammals such as cetaceans have an important ecological role in the recycling of nutrients by feeding at depth and then defecating in the euphotic zone. Large cetaceans also continue to play an important ecological role even after death through the downward transfer of nutrients to benthic communities (Bowen 1997).

In Kenya, marine mammals are facing extraordinary threats due to fisheries and many species have been listed as endangered or vulnerable and data deficient, while others have been reported to have almost completely disappeared from the Kenyan waters (Kiszka et al., 2009b). The Indo-Pacific humpback (*Sousa chinensis*) and bottlenose dolphins (*Tursiops aduncus* and *Tursiops truncatus*), as well as the dugong (*Dugong dugon*) have been reported by previous studies as the most affected marine mammal species through by-catch in gillnet fishery (Kiszka et al., 2009a). Historically, the dugong occurred in large numbers off the Kenyan coast before the 1970's with large groups of about 500 reported along the south coast in 1967 (Husar, 1975; WWF EAME, 2004). A study by Kiszka et al. (2009a) attributed the incidental capture of dugong in the Indian Ocean to gillnets and hunting which has reduced their population. The study also indicated that dugongs still occur, though in small numbers, off the Tana Delta area, in the Lamu Archipelago and in the Kiunga Marine reserve in the extreme north.

On the other hand, sea turtles spend most of their lives in coastal or pelagic waters, making uninterrupted coastal waters critical to their population stability. They play a significant role in the marine ecosystem and can transfer substantial quantities of nutrients and energy from nutrient-rich foraging grounds to nutrient-poor nesting beaches for other marine organisms to benefit. The loggerhead turtle (*C. caretta*) can modify the physical structure of their habitat in a number of ways, including digging trenches through soft substrates in search of in-fauna prey (Bjorndal & Jackson, 2003). Sea turtles have been negatively impacted by a number of anthropogenic factors which include oil spills, contaminants and other types of marine pollution. Direct offtake and habitat modification have also shown to affect sea turtles. In spite of this, the human activity that has the greatest impact on sea turtles is fisheries' by-catch. Because fishing is an important source of protein and the livelihood for millions of people worldwide, by-catch of sea turtles continues to be the most pressing human impact on sea turtle populations globally (Lewison et al., 2013).

Sea turtles are facing unmatched threats worldwide and are listed as endangered species under the International Union for Conservation of Nature (IUCN) Red list with varied vulnerability. Kenya is home to five species: loggerhead turtle (*Caretta caretta*), leatherback turtle (*Dermochelys coriacea*), olive ridley (*Lepidochelys olivacea*), hawksbill (*Eretmochelys imbricata*) and green turtle (*Chelonia mydas*) out of the seven which are found in the world oceans. The hawksbill turtle is considered to be the most critically endangered species among all the existing five species in the Kenyan coast (Kiszka, 2012a). High sea turtle mortalities have so far been reported in the country by various studies with majority of kills linked to by-catch (Frazier, 1975; Frazier, 1980; Wamukoya et al., 1995; Wamukoya et al., 1998; Mueni & Mwangi, 2001; Okemwa et al., 2004; Bourjea et al., 2008; Kiszka, 2012a; Olendo & Mwasi, 2017).

Kenya marine fisheries comprise both artisanal and semi-industrial sectors and are of major socio-economic importance especially to the coastal communities. Artisanal fisheries are confined to shallow coastal waters but account for 90% of the annual total marine fish landed (Kiszka & van der Elst, 2015).

Various studies have been conducted on by-catch on marine mammals and sea turtles but very few of them focused on the Kenyan coast artisanal fisheries (Wamukoya & Mbendo, 1995; Wamukoya et al., 1998; Bourjea et al., 2008; Kiszka et al., 2008; Kiszka et al., 2009a; Kiszka et al., 2009b; Kiszka, 2012a; Kiszka, 2012b; Lewison et al., 2013). Therefore, the information on the impacts of by-catch on marine mammals and sea turtles in the small-scale artisanal fisheries is still scanty. Consequently, this study was aimed at assessing the by-catch of marine mammals and sea turtles in small-scale artisanal fisheries along the Kenyan coast off Lamu, Malindi and Mombasa in order to generate data and information necessary in the effective management and conservation of these marine resources.

Materials and Methods

Sampling sites

The study was conducted along the Kenya coast focusing on landing sites in Lamu, Malindi and Mombasa towns (Figure 1), two landing sites were selected for each of the three towns: Kiwayuu and Kizingitini fish landing sites in Lamu; Mbuyuni and Ngomeni landing sites in Malindi; and Nyali and Old town landing sites in Mombasa. The landing sites were selected for sampling because they are easily accessible and have high numbers of gillnet fishers based on data from the state department of fisheries (Government of Kenya, 2014).

Research Design

This study was part of a bigger project investigating marine mega-fauna by-catch in Lamu, Malindi and Mombasa. Structured interviews with questions have also been used in the study similar to various studies done in the past (Moore et al., 2010; Kiszka, 2012a; Kiszka, 2012b). The questionnaire used in the present study was developed by Kenya Marine and Fisheries Research Institute as the implementing institution of the bigger BYCAM project. However, the questionnaires were modified and adapted to fit the purpose of the present study which focused only on the gillnet fishery, in contrast to the bigger project that covered several artisanal fisheries. Questionnaires were administered to fishers in Kiwayuu and Kizingitini landing sites in Lamu; Mbuyuni and Ngomeni landing sites in Malindi; and old town and Nyali landing sites in Mombasa. Only fishers confirmed to be using gillnet fishery for their fishing activities were interviewed at the study site.

Sampling Procedures

The study was conducted from late May through early November 2016. Data was collected once in every month using structured interviews (questionnaires). Interviews were administered to the fisher available at the landing sites during the field visits. Interviews were conducted at the landing sites and at the fisher camps under varying circumstances, including when fishers were repairing their fishing gears/boats, at landing sites, or when they were coming back/going to fishing (Plate 1 & 2). Depending on the availability of fishers on the landing site, between three and five fishers from different fishing vessels were interviewed per day. Although the questionnaires were prepared in English, they were often translated to Kiswahili (the national language) for the respondents. Each participant in the study was issued with a consent form that aimed at ensuring that the fishers participated in the research voluntarily. The forms were then taken back after being signed by the fishers. By-catch data was recorded by the researcher with the help of field assistants. Species identification was done using species identification keys (Plate 1) adopted from the FAO species catalogues and a field guide (Anam & Mostrada, 2012).

Data Analysis

Data on the species of sea turtles, marine mammals and elasmobranchs identified, number of times (frequency) a species was recorded over the study period were transferred from the questionnaires to

Microsoft Excel for analysis. By-catch composition was analyzed descriptively and presented in graphs or tables for drift and bottom set gill nets. Differences in numbers of sea turtles, marine mammals and elasmobranchs were compared between landing sites and seasons. Since the count data was not normally distributed, square root data transformation was performed (McDonald, 2014). The transformed data was later tested for homoscedacity using the Levene's test (Levene, 1960) prior to statistical analysis. 2-way Analysis of Variance (ANOVA) was performed on the data that conformed to the assumption of homoscedacity while Kruskal-Wallis test was performed on the data which did not meet the requirements for ANOVA test.

By-catch incidences were calculated for each species group using the following equation:

$$BI = (N \text{ species/study period} / N \text{ fishers}).$$

Where BI is by-catch incidence, Nspecies/study period is the number of individual species during the whole study period and Nfishers is the total number of fishermen interviewed during the entire study period (Kiszka, 2012a). Differences in the by-catch incidences between bottom-set and drift gillnet fishery during NEM and SEM seasons and at different landing sites were compared using 2-Way ANOVA.

Results

Fishermen Characteristics and Fishing Gears Used

A total of 92 interviews were conducted. All respondents were males. At least 26% of respondents had already been involved in questionnaire surveys in the past. The age of interviewed fishermen had a mean of $46.3 \pm 13.01SD$. Their mean fishing experience was (mean $\pm$ SD) 23.8 ± 11.5 years. For all interviewed fishermen, fishing was their primary activity.

Ngomeni had the highest number of fishing vessels using gillnets (40%, $n = 37$), while Nyali had the lowest (2%, $n = 2$) (Figure 2). About 61% of fishing vessels ($n = 56$) were using bottom-set gillnets, 15% ($n = 14$) of them were using drift gillnets, while 24% ($n = 22$) were using both gillnet types.

Fishing Effort

The number of days the fishers go fishing per week were between 5 and 7, with most fishermen (90%) reporting durations of 6 to 7 days. This applied to those fishermen who go fished and returned to the landing site on a daily basis. For the night fishers referred to as "Ago", all the respondents declared that they spend between 23 and 24 days (depending on sighting of the moon) at sea (from 17th to 10th of every month of lunar calendar) and rest for 6 days of travelling back at the landing site. Based on fishermen declarations, lowest effort (4%) was observed during the Southeast monsoon (Kusi), i.e. from May to August, and highest effort (59%) was observed during Northeast monsoon (Kaskazi), i.e. from November to February. Some respondents (37%) declared that they were fishing throughout the year. Five vessel types were commonly used along the coast of Kenya; dhow (91.3%), fiber re-enforced plastic (FRP) (5.4%), dau (1.1%), motaboti (modern boats with outboard engines) (1.1%) and mtori (1.1%).

By-catch Composition of Sea Turtles

Five species of sea turtles were recorded as by-catch in all landing sites (*Eretmochelys imbricata*, *Chelonia mydas*, *Lepidochelys olivacea*, *Caretta* and *Dermochelys coriacea*). The green turtle *C. mydas* was the most recorded by-catch species ($n = 63$) with Ngomeni reporting the highest number of by-catch (19.6%; $n = 32$) followed by *E. imbricata* with 17.2% in Ngomeni ($n=28$) and 7.4% in Kiwayuu landing site ($n = 12$). *Caretta* recorded reportedly the lowest by-catch ($n = 13$) (Figure 3). There was no significant difference in

the by-catch of sea turtles between different landing sites (Kruskal-Wallis test: $H = 8.886$ $p = 0.064$). However, there was a significant difference in the by-catch of sea turtles between the seasons (Figure 4) (Kruskal-Wallis test: $H = 60.461$; $p < 0.05$).

By-catch Composition of Marine Mammals

A total of seven species of marine mammals (Figure 5) were recorded as by-catch in all landing sites; (Common minke whale *Balaenoptera acutorostrata*, Spinner dolphin *Stenella longirostris*, Fraser's dolphin *Lagenodelphis hosei*, Indo-pacific humpback dolphin *Sousa chinensis*, Rissor's dolphin *Grampus griseus*, Indian Ocean bottlenose dolphin *Tursiops aduncus* and Common bottlenose dolphin *Tursiops truncatus*). Overall by-catch of these species was low in all landing sites with the highest by-catch reported in Kiwayuu ($n = 9$) with *T. aduncus* having the highest number of by-catch (37.5%; $n = 9$) followed by *G. griseus* and *S. chinensis* both at 20.8% ($n = 5$). A significant difference in by-catch of marine mammals across the landing sites existed as Ngomeni reported significantly the highest by-catch composition (46.7%, $n = 21$) and Nyali had the lowest (2.2%, $n = 1$) (Kruskal-Wallis test: $H = 60.081$; $p < 0.05$).

There was no seasonal significant difference in the by-catch of marine mammals in different seasons (Kruskal-Wallis test: $H = 9.388$, $p = 0.402$) (Figure 6).

Gillnet Fishery

About 74% of respondents declared by-catch of sea turtles in bottom-set gillnet fishery while 19% of them declared by-catch of sea turtles in drift gillnet fishery. In bottom-set gillnet *C. mydas* (38.4%, $n = 26$) had the highest number of by-catch while *C. caretta* (7.4%, $n = 5$) had the lowest. Over 16% of fishermen declared that they caught 1-10 individuals in the year prior to the interview while only 3.2% of the respondents declared by-catch of between 21-50 individuals of sea turtle species on the same period. In drift gillnet, *C. mydas* (50%, $n = 17$) was the highest sea turtle species reported as by-catch while the lowest was *C. caretta* (6%, $n = 2$; Figure 7).

The result of 2-Way ANOVA shows significant difference in the by-catch of different species of sea turtles in the bottom-set and drift gillnet fisheries ($df = 1$; $f = 29.8$; $p < 0.05$). No by-catch of marine mammals in bottom-set gillnet fishery was reported. Marine mammal by-catch was declared by 51% of fishermen using drift gillnets. The most commonly involved species were *T. aduncus* (42%, $n = 19$) and *T. truncatus* (20%, $n = 9$), while the least involved species, *S. longirostris* (4%, $n = 2$), *L. hosei* (4%, $n = 2$), *B. acutorostrata* (2%, $n = 1$), and *D. dugon* (2%, $n = 1$). Number of marine mammals' by-catch in the previous year varied from 0 to 10 individuals. Six percent ($n = 17$) of the respondents declared that they caught none of marine mammals in the year prior to the interview.

When fishermen were asked about how many species of the marine mammals and sea turtles they caught during the previous year before the study was conducted (the range was given in the questionnaire as follows: 0, 1-10, 11-20, 21-50 and over 50), the reply was as follows (Table 1):

Bottom-set and drift gillnets were used differently (Figure 8) during different seasons (NEM and SEM) and the difference was statistically significant ($df = 1$; $f = 26.3$; $p < 0.05$). The difference in the use of bottom-set and drift gillnets along different landing sites was also statistically significant ($df = 5$; $f = 5.6$; $p < 0.05$).

The result of 2-Way ANOVA indicated no significant difference in the by-catch of sea turtles and marine mammals between bottom-set and drift gillnet fisheries ($df = 2$; $f = 8.8$; $p = 0.102$).

By-catch Incidence

By-catch incidence (number of individual species reported as by-catch during the study period per boat) was calculated for each fishery and each taxonomic group (elasmobranchs, marine mammals and sea turtles) between different seasons and landing sites. All values are presented in Table 2, 3 and 4.

There was a significant difference in the by-catch incidences of marine mammals and sea turtles between NEM and SEM seasons at the six landing sites ($df = 11$; $f = 3.2$; $p < 0.05$) as shown in Table 2. The results of 2-Way ANOVA further showed a significant difference in the by-catch incidences between the use of bottom-set and drift gillnet fishery across different seasons at the six landing sites ($df = 1$; $f = 21.66$; $p < 0.05$). By-catch incidences of bottom-set and drift gillnets across different seasons and landing sites are presented in Table 3.

Based on the by-catch incidences, sea turtles appear to have higher by-catch incidences in bottom-set gillnet than in drift gillnet fishery. Marine mammal by-catch was higher in drift gillnet than bottom-set gillnets (Table 4). However, these differences were not statistically significant (Kruskal-Wallis test: $H = 4.5$, $p = 0.212$).

Use of By-catch Species

All interviewed fishers declared that they released all sea turtles caught incidentally in the nets. Marine mammals were systematically discarded or released alive, except in some cases e.g. where the by-catch species is dugong, the meat was consumed secretly.

Perceptions of By-catch and the Status of Population

Perceptions of the status of population of marine mammals and sea turtles were recorded from key informant respondents. According to the fishers' perceptions, there is an increase in the population of marine mammals and sea turtles over time. However, *D. dugon* was considered to have completely disappeared from the coastal waters of Kenya (41%, $n = 38$). On the other hand, 4% of the respondents ($n = 4$) claimed that the status of marine mammals and sea turtles remains the same and is only affected by seasonality.

Discussion

This study investigated by-catch of marine mammals and sea turtles along the north-coast of Kenya. It is based on 92 interviews with artisanal fishers conducted in Lamu, Ngomeni, Malindi and Mombasa, where very little was known about the by-catch of these mega-fauna in the artisanal fisheries. This is one of the very few studies on marine mammals and sea turtle by-catch in the north-coast of Kenya. The major finding of this study reveals a high extent of marine mammals and sea turtles by-catch in artisanal fisheries in both bottom-set and drift gillnets. A total of 12 species were identified as by-catch, including five (5) species of sea turtles and seven (7) species of marine mammals.

The sea turtle species reported as by-catch in this study were *C. mydas*, *E. imbricata*, *L. olivacea*, *D. coriacea* and *C. caretta*. Marine mammal species were *T. aduncus*, *T. truncatus*, *G. griseus*, *S. chinensis*, *S. longirostris*, *L. hosei* and *B. acutorostrata*. The results of this study are consistent with the earlier study by Wamukoya et al., (1996) and Kiszka (2012a) in the area, both in terms of species involved and by-catch incidences respectively. Among the species which were not recorded by Kiszka (2012a) but were recorded in this study include *G. griseus*, *B. acutorostrata*, *L. hosei* and *T. truncatus*. Kiszka (2012a) recorded by-

catch of *Stenella attenuate*, *Megaera novaeangliae* and *Delphinus delphis* which were not recorded in this present study.

One of the most threatened and vulnerable species in the region, the dugong, was reported to have declined by majority of fishers. They reported that they used to capture dugongs a lot in the past but presently the species is reported to have disappeared and they were not reported as by-catch anymore. This is in contrast to previous studies that reported higher presence, by-catch and mortality rates of this species (Jarman & Kariba, 1961; Husar, 1975; Wamukoya et al., 1996; WWF EAME, 2004) and its presence in reasonably small numbers in Lamu archipelago and in Kiunga (Kiszka et al., 2009a). Fishermen in Kizingitini village reported that there used to be many dugongs in their fishing grounds, especially in a fishing area called “dugong hill” (they call it “Mlima wa nguva” in Swahili). This area had rich grass meadows and dugongs used to graze. This study recorded no dugong by-catch in the areas around dugong hill. This may be attributed to the current low populations of this species along the East African coast and its rapid decline since the early 2000s due to incidental capture in gillnets (Kiszka et al., 2009a).

As indicated by this present study, there is a difference in the extent and effect of by-catch of marine mammals and sea turtles among gears. By-catch levels of sea turtles were higher in bottom-set gillnets than in drift gillnet fisheries. By-catch levels of marine mammals were higher in drift gillnets than in bottom-set gillnet fishery. However, the difference in the by-catch incidences between the two gear types was not statistically significant ($p = 0.102$). The difference in the use of bottom-set and drift gillnet fishery between different seasons was significant because fishers preferred using bottom-set gillnet during SEM season (because they can't venture offshore fishing) and drift gillnet during NEM as they can venture offshore fishing. The adverse effects of gillnets, including bottom-set and drift gillnets, have already been highlighted in previous studies along the Kenyan coast (Kiszka et al., 2009a; Kiszka, 2012a) and in the region for particularly vulnerable species, such as the dugong (WWF EAME, 2004; Kiszka, 2012b; Pusineri et al., 2013).

By-catch incidences (or by-catch rate per fishing vessel) for marine mammals were higher in drift gillnets than in bottom-set gillnets. This could be attributed to the fact that marine mammals (specifically dolphins) escape from the bottom-set gillnet as it is set for a long time but they don't get time to do so when a drift gillnet is used. On the other hand, by-catch incidences for sea turtles were higher in bottom-set gillnet than in drift gillnet fisheries. This could be attributed to the fact that sea turtles are demersal species and are more susceptible to bottom-set gillnet which is set at the bottom of the sea.

This study reports higher by-catch of sea turtles especially at Ngomeni landing site. However, the difference in the by-catch of sea turtles across the six landing sites was not statistically significant ($p = 0.06$). This study also highlighted new results on sea turtle by-catch in gillnets with high by-catch levels observed for leatherback turtles (*D. coriacea*) in the Kenyan coastal waters compared to the previous study by Kiszka (2012a) where this was relatively rare. Loggerhead turtles were reported previously by the same study as caught frequently in the Kenyan coast, but the present study reports a very low by-catch of this species (only 1.5% of the artisanal fishers) in the Kenya coast. It should be given serious redress since this species is severely declining in the region (Kiszka, 2012a). The seasonal variations in by-catch were directly related to fishing effort. The highest fishing effort was recorded during Northeast monsoon, i.e. from October/November to February/March and lowest effort was observed during the Southeast monsoon, i.e. from May to August. This phenomenon is linked to better sea conditions, especially between October and March. However, this was not evident in all gears as bottom-set gillnet was reported to be preferred during Southeast monsoon when venturing offshore is impossible. This could be attributed to the fact that bottom-set gillnet can just be set near the sea shore and there is no need to venture offshore or sheltered fishing grounds.

Artisanal fishermen declared that sea turtles and marine mammals by-catch was on the increase. While it is difficult to take these witnesses as real population trends, it is interesting to see that they are correlated to previous studies which have highlighted decline of dugongs and at the regional level (Wamukoya & Salrn, 1997; WWF EAME, 2004; Kiszka, 2012a; Kiszka & van der Elst, 2015). In all study sites, marine mammals and sea turtles were considered as by-catch and were not commonly used for food in the region. When still alive, marine mammals and sea turtles were generally released. Overall, marine mammals and sea turtle consumption may be underestimated as fishers may fear being reprimanded for killing and consumption of these species which is prohibited in the country by law.

Conclusions and Recommendations

This study clearly highlighted that an important diversity of marine mammals and sea turtles were exposed to artisanal fisheries as by-catch in Kenya. Dugongs, most likely the most vulnerable species according to IUCN conservation status, are still in decline and are even reported in this study to have almost completely disappeared from the Kenyan waters. With regards to by-catch of sea turtles, high rates are reported by this study especially at Ngomeni landing site. Fishermen also perceive the tough penalties against fishing of sea turtles as oppression against them while the species (in their perception) increases abundantly when sustainably utilized.

The results of this study also highlighted that many coastal populations of dolphins, especially Indian Ocean bottlenose dolphin, common bottlenose dolphin, risso's dolphin and Indo-Pacific humpback dolphin are facing detrimental interactions with drift gillnet fishery. It can also be concluded by this study that sea turtles are facing an exceptional threat in bottom-set gillnet fishery.

Based on this study, some recommendations for future research and management initiatives are highlighted below:

Extend Rapid By-catch Assessments (RBAs) to other artisanal fisheries such as artisanal long-lines, trawl nets, beach seines, cast nets and other types of nets which can pose a threat to these vulnerable marine mega-fauna in Kenya. However, a special focus should be directed to drift gillnet fishery which was reported in this study to have a great impact.

Create awareness, through trainings, to fishermen in the country on the importance of protecting the resources we have with regards to these marine mega-fauna and empower them increase their socio-economic value. Conduct further studies that would include collection of biological samples for laboratory identification of unidentified species as this present study reported ten of them.

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Tables and Figures

Table 1: Summary showing total number of by-catch by category reported during the year before the study was conducted in 2016.

Group	0	Range 1–10	Range 11–20	Range 21–50	Range >50	Sub- total
Sea turtles	6	47	7	9	0	69
Marine mammals	17	14	0	0	0	31

Table 2: By-catch incidences (BI: rate of by-catch per one fishing vessel) of marine mammals, sea turtles, sharks and rays at six different landing sites during NEM and SEM seasons.

BI = (N species/study period / N fishers).

	Kiwayuu		Kizingitini		Ngomeni		Malindi		Old town		Nyali	
Marine mega-fauna	NEM	SEM	NEM	SEM	NEM	SEM	NEM	SEM	NEM	SEM	NEM	SEM
Marine mammals	0	0	0	0	0	0	0	0	0	1	1	0
Sea turtles	1	1	0	1	1	1	1	0	0	0	2	1

Table 3: By-catch incidences (BI: rate of by-catch per fishing vessel) of bottom-set and drift gillnet fisheries by landing site during NEM and SEM seasons.

BI = (N species/study period / N fishers).

	Kiwayuu		Kizingitini		Ngomeni		Malindi		Old town		Nyali	
Gillnet fishery	NEM	SEM	NEM	SEM	NEM	SEM	NEM	SEM	NEM	SEM	NEM	SEM
Bottom- set	3	3	3	5	4	4	3	2	4	4	7	3
Drift	3	1	1	1	2	1	3	1	2	2	3	1

Table 4: By-catch incidences (BI) of sea turtles, marine mammals, rays and sharks in bottom-set and drift gillnet fishery along the north coast of Kenya in all landing sites.

BI = (N species/study period / N fishers).

Gillnet fishery	Sea turtles	Marine mammals
Bottom-set gillnet	2	0

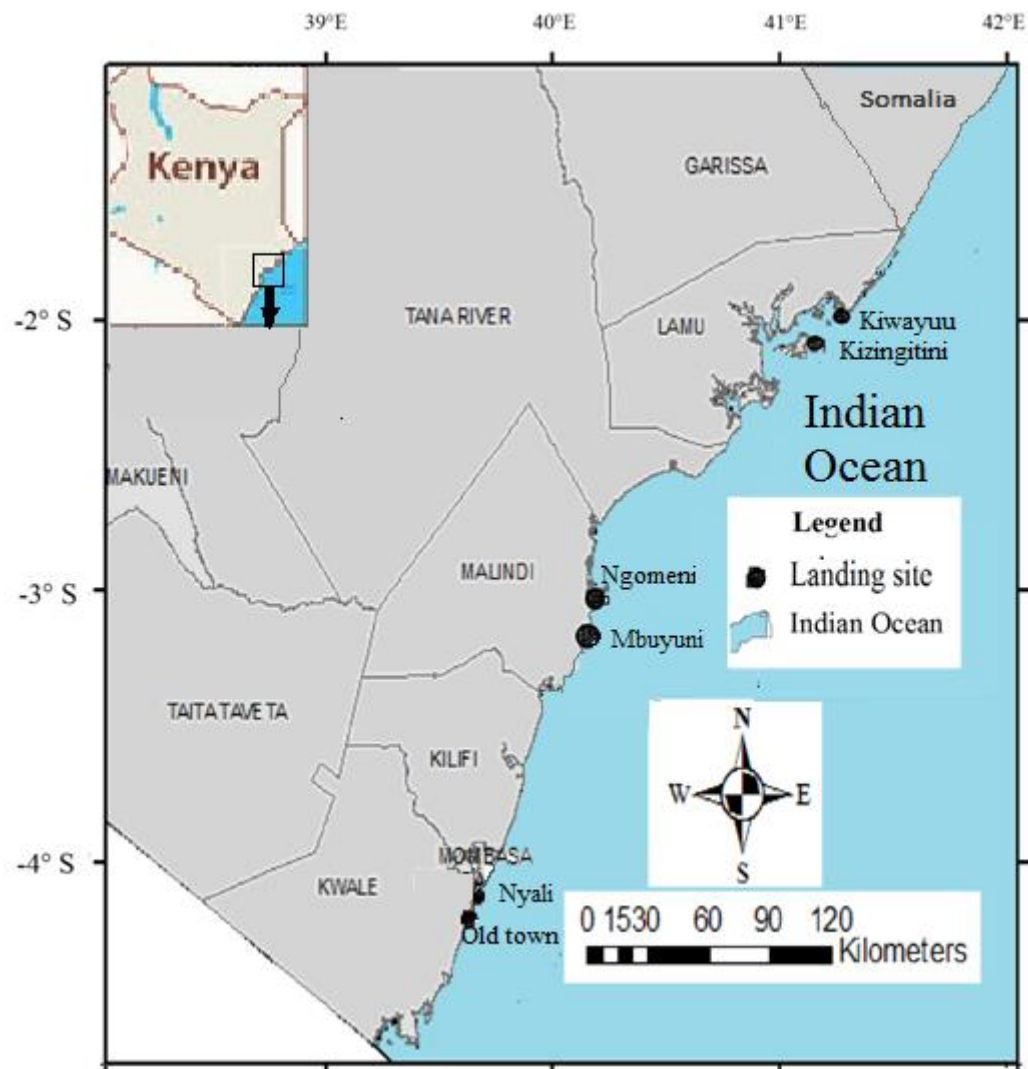


Figure 1: A map of Kenya (inset) showing the location of the sampled landing sites along north coast Kenya.



Plate1: Training of the fishermen on the use of species identification keys at Kiwayuu landing site during the survey.



Plate 2: Researcher interviewing a fisherman who was repairing his fishing gear at Old town landing site, Mombasa.

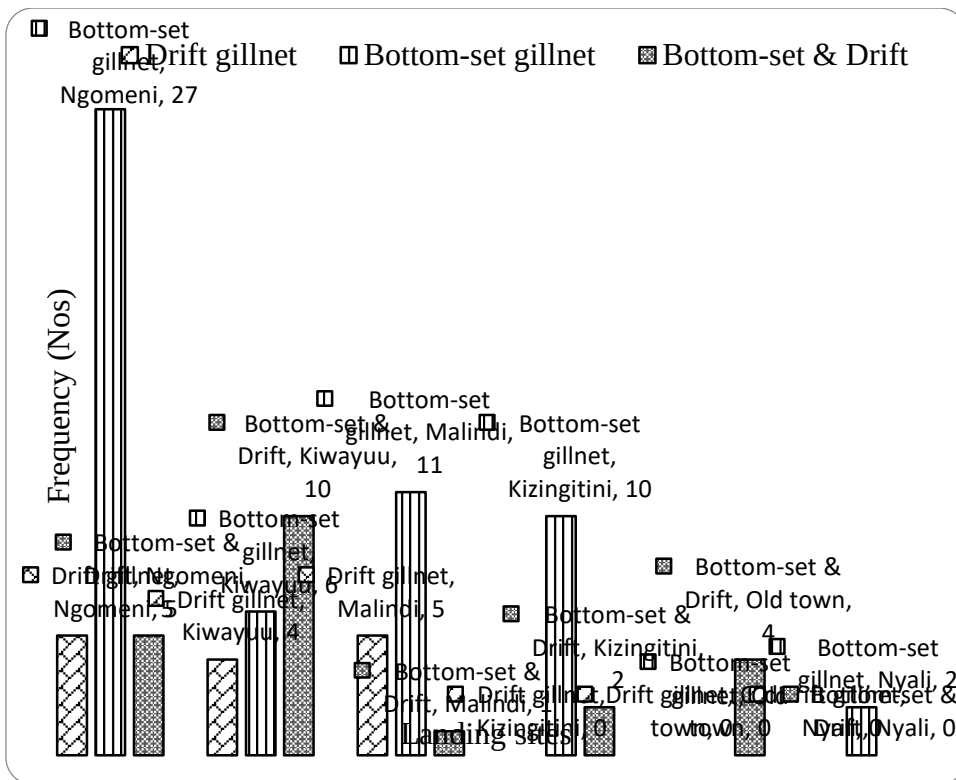


Figure 2: Distribution of the fisheries at the studied landing sites along north coast Kenya.

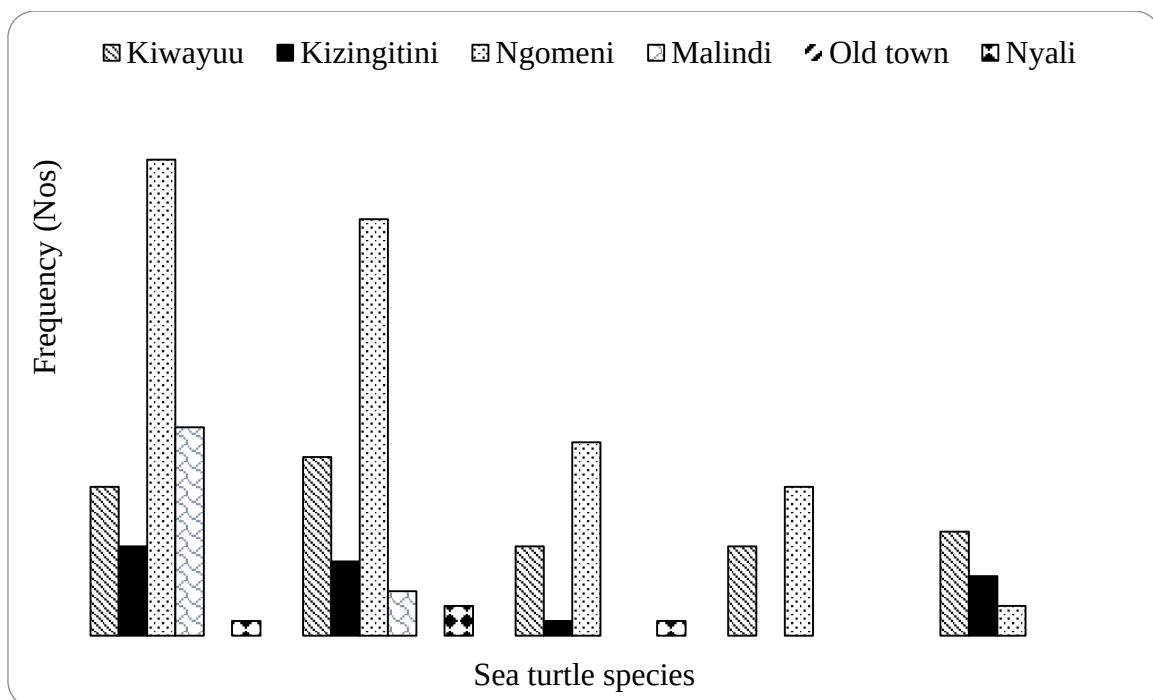
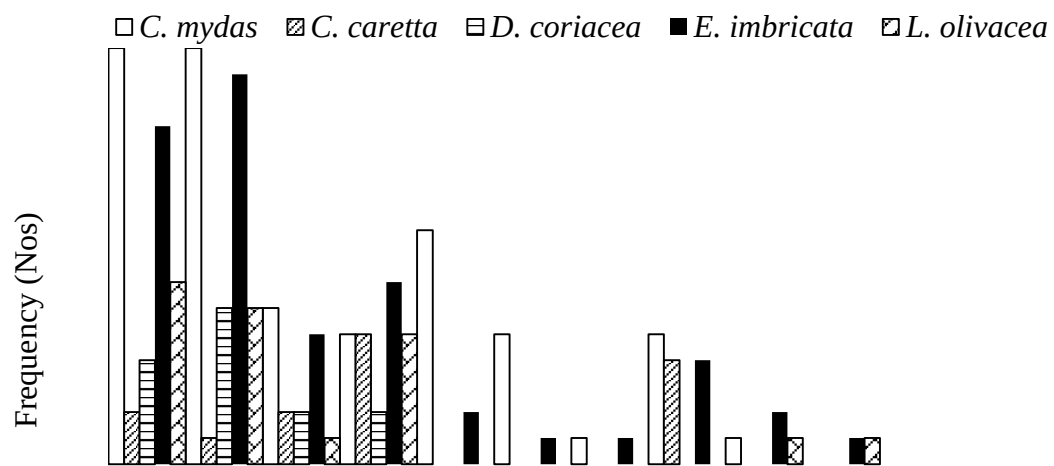


Figure 3: Spatial occurrence of by-catch of sea turtles at the sampling sites along north coast Kenya.



Seasons and Landing sites

Figure 4: Temporal occurrence of sea turtles by-catch by season (NEM and SEM) at the sampling sites along north coast Kenya.

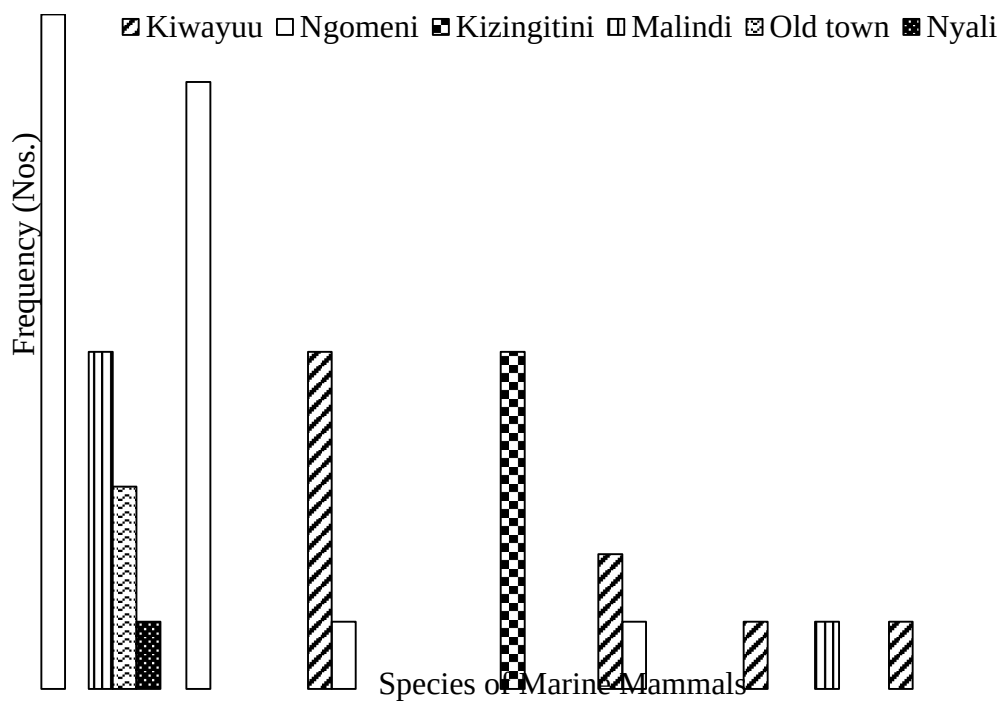


Figure 5: Occurrence of by-catch of marine mammals at the sampling sites along north coast Kenya.

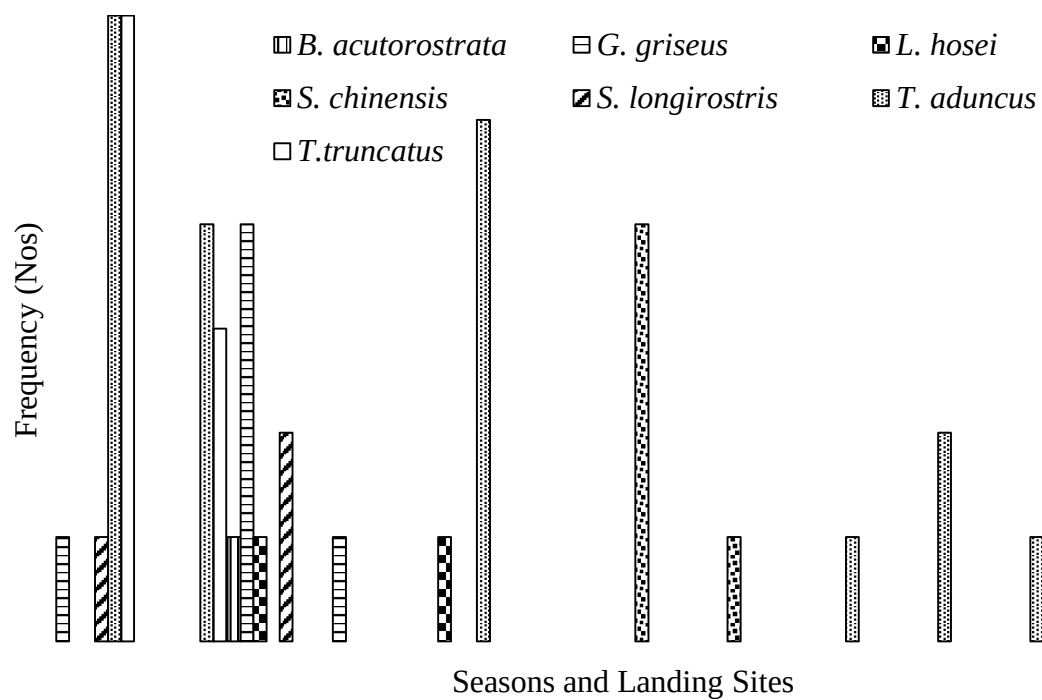


Figure 6: Temporal occurrence of marine mammals' by-catch by season at the sampling sites along north coast Kenya.

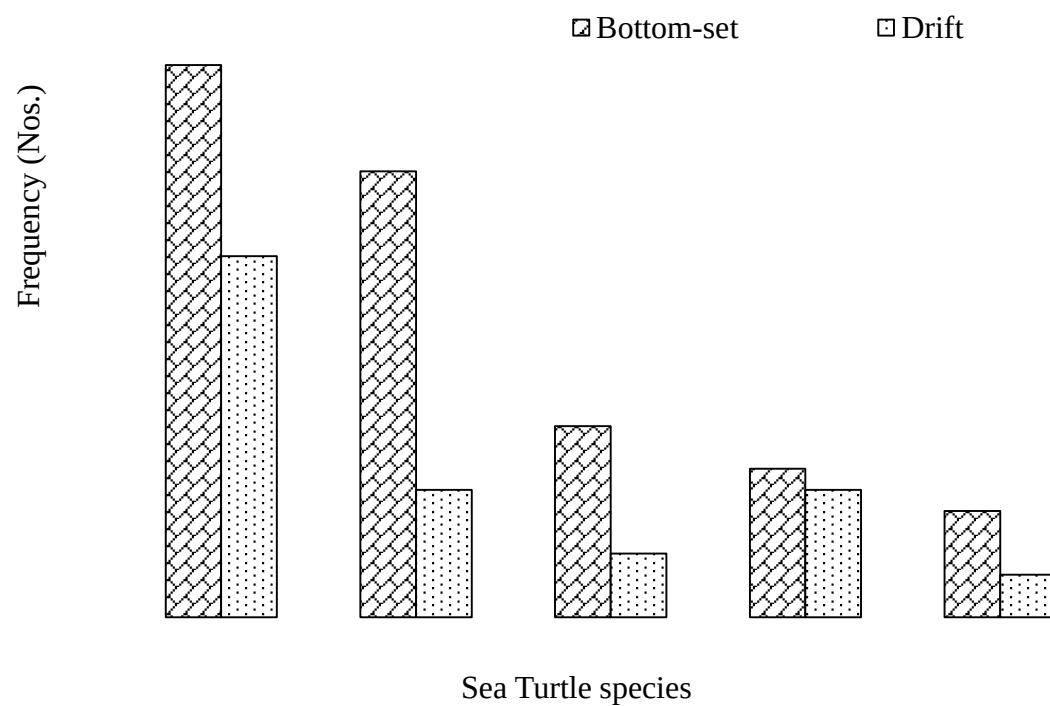
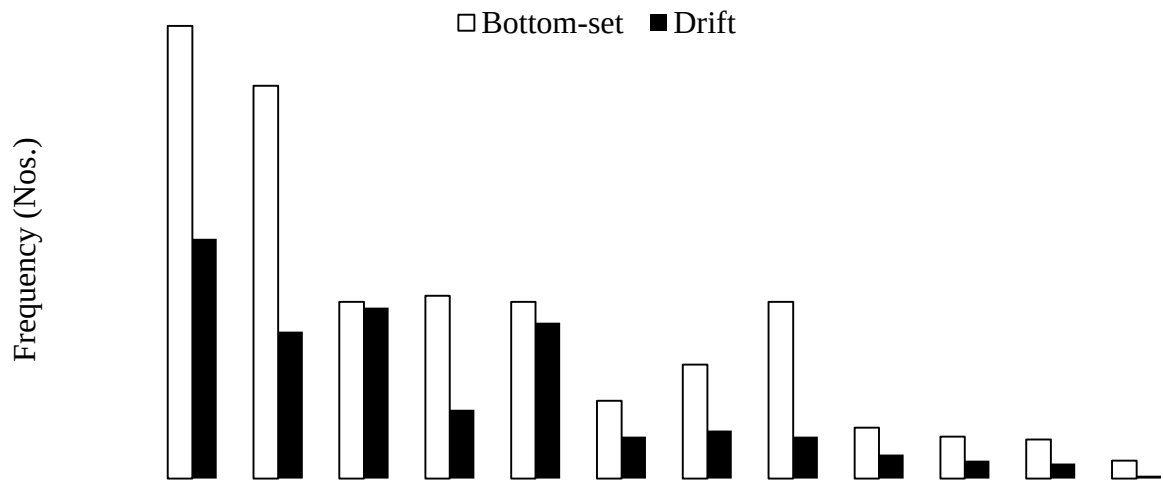


Figure 7: Occurrence of by-catch of sea turtles (Counts) in both bottom-set and drift gillnet fisheries along north coast Kenya.



Seasons and Landing sites

Figure 8: Temporal occurrence of bottom-set and drift gillnet fisheries by season (NEM and SEM) and sampling sites along north coast Kenya.

Annex 13: Trends in Kenyan artisanal and recreational shark landings manuscript

Trends in Kenyan artisanal and recreational shark landings

1. Introduction

Sharks are caught either as target or bycatch species in different fishing sectors including artisanal, recreational and large-scale commercial fleet globally (Dharmadi et al. 2015, Worm et al. 2013, Romanov et al. 2011). However, most studies tend to focus exclusively on the impact of commercial fisheries on shark populations while ignoring the effects of artisanal fisheries and recreational fisheries (Stevens et al. 2000, Kiszka and Heithaus, 2014). Nevertheless, recent evidence has shown that a significant number of sharks are targeted by these fisheries in tropical countries, yet few studies have been undertaken to assess the catch composition, trends and magnitude of shark fishing, whether targeted catch or as bycatch (Yulianto et al., 2018; Baeta et al., 2010; Cartamil et al., 2011; Van der Elst and Everett, 2015).

Although commonly considered benign compared to industrial fisheries, artisanal fisheries may have the potential to profoundly affect fish stocks and on ocean ecosystems than previously thought (Purcell and Pomeroy, 2015; Smith and Basurto, 2019; Stallings, 2009). For instance, in the Western Indian Ocean (WIO) region, studies have shown that sharks are commonly captured as targeted species in artisanal fisheries, particularly for the fin trade but also for local consumption (Ferretti et al., 2018; Humber et al., 2017; McVean et al., 2006). However, the lack of monitoring and absence of biological information on sharks and species-specific catches complicates the quantification and assessment of the actual impacts of these fisheries on shark populations in the region, particularly along the East African coast (Temple et al. 2018, 2019, Kiszka and van der Elst 2015, Humber et al. 2016).

The capture of sharks is also typical among marine recreational fishers (Kiszka and van der Elst, 2015, Anderson and Simpfendorfer, 2005, Lucifora et al., 2012, Davidson et al., 2016, McVean et al., 2006, Temple et al. 2018) and like in artisanal fisheries, some evidence posits that recreational fishing (RF) have substantial negative impact on sharks yet often remain under-reported and unregulated in many developing countries (Belhabib et al., 2016). Since artisanal fishers comprise more than 65% of the WIO fishers and are likely to account for a considerable proportion of shark bycatch, there is an urgent need for accurate baseline information and data to address population declines that are likely to go unnoticed. Further, understanding the status of directed and non-directed shark fisheries catalyses efforts to create and implement effective plans for protecting and managing shark resources.

Artisanal fisheries and the recreational fisheries represent an important sector in the marine fisheries with the artisanal fisheries in Kenya, constituting about 80% of the fishing effort (Marine Frame Survey 2016). Like in most SSF, artisanal fisheries in Kenya are characterized by the use of relatively simple gears, are often remote and dispersed, making them difficult to monitor and manage (Glaus et al., 2019; Smith and Basurto, 2019).

Various studies have described characteristics coastal artisanal fisheries landings for small and medium pelagics) in terms of gear selectivity, species and size composition, seasonality, and the stock status (Kiilu et al. 2019, Kyalo and Ndegwa, 2013, Fulanda et al. 2011, Munga et al. 2013, Swaleh et al. 2015, Tuda and Wolff, 2015), but information on the artisanal shark landings is critically lacking.

The recreational fishery, also termed as sport fishing or big game fishing has a long history which dates to at least the early 1950s and has a mixture of both private and charter sporting operations (Howard & Starcke II, 1975, Williams, 1970). Fishing gears in the recreational fisheries are limited to handline or rod and reel accompanied with techniques such as light and heavy tackle or fly rods using live baits, dead baits or lures. Sportfishing particularly for billfish has put Kenya on the world map as one of the few game fishing destinations where an angler has a likelihood of catching a 'Fantasy slam' (sailfish, blue marlin, black marlin, striped marlin and swordfish) with the first record achieved in 2005 and two other fantasy slams in 2009. Additional game fish species include sharks, tuna, kingfish, wahoo, and giant trevally among others. From mid-1980's, the tradition of catch, tag and release became prominent in Kenya's sport fishing industry with the establishment of the African Billfish Foundation (ABF) (a private tagging programme based in Kenya) (Harris et al. 2013, Kadagi et al. 2011). The tagging data presents evidence that sharks are landed in recreational fisheries, but information is lacking on the composition of shark landings and magnitude of impact.

In this study, we explore data on the composition of sharks in the Kenyan artisanal and recreational fisheries. Our approach provides an informative way to conceptualize the types of species, composition and proportions in the two fisheries at different time scales. We then applied a multiple linear regression model to examine factors most affecting the shark landings in the artisanal fishery. By examining the composition of sharks in the two fisheries and the factors influencing mean shark landings, the study contributes to an understanding of differential fishing impacts on shark species and informs the process of developing a national plan of action for protection and management.

2. Materials and methods

2.1. Study Area and Fishery Context

2.2. Data sources and collection

To evaluate the trends in shark landings in Kenya, data for this study were obtained from primary and secondary sources as follows:

2.3. Primary data

2.3.1. Artisanal longline and gillnet data

Fisheries-dependent data from artisanal fisheries were collected from eight landing sites occurring along the Kenya coast. Kipini, Kiunga, Kiwayu, Kizingitini, Ngomeni, Watamu, were located in the north coast while Old Port, and Vanga are located in the South coast) (Figure 1). These study areas are characterized by the presence of artisanal longline and gillnet (bottom-set and drift) fisheries that catch a variety of target and by-catch species (Marine Frame survey, 2016). The geographical spread of these landing sites and prior knowledge from bi-annual surveys of fishing effort distribution were considered in the selection of the study sites (Marine Frame survey, 2016; CAS SOP's), to provide a representative sample of artisanal fish landing sites along the Kenya coast.

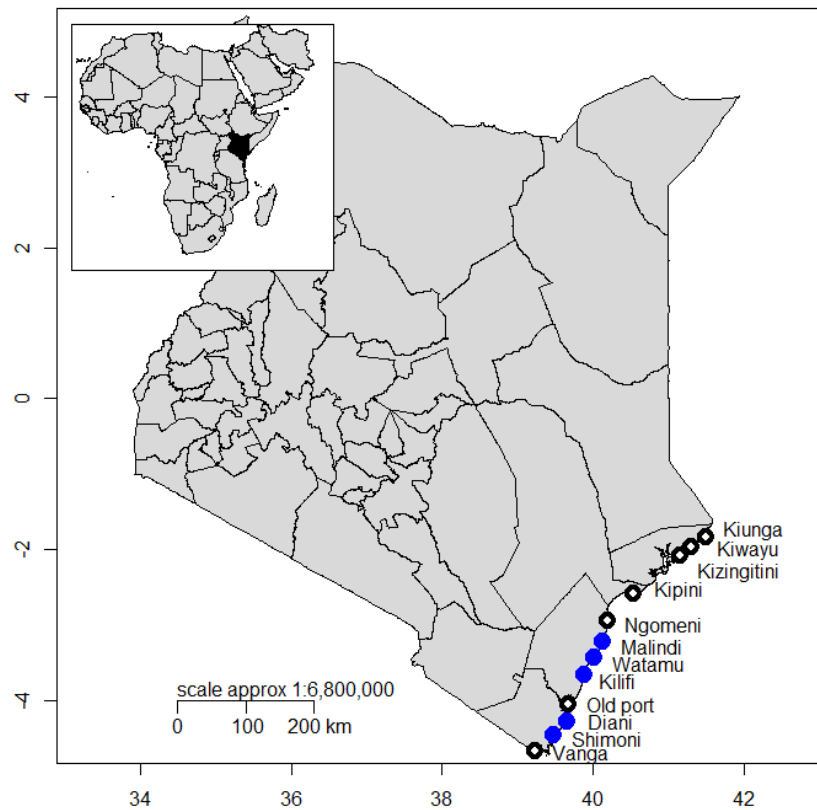


Figure 1: Map of the Kenya coast showing 8 artisanal longline and gillnet data collection sites (Kipini, Kiunga, Kiwayu, Kizingitini, Ngomeni, Watamu, Old Port, and Vanga). Major recreational fishing centres are indicated in blue (Malindi, Watamu, Kilifi, Diani and Shimoni).

Data were collected by trained land-based observers who recorded landings of sharks from select fisheries gears at each of the study locations for a period of 12 months between June 2016 and June 2017, this covered the two distinct fishing season typical to this region (McClanahan, 1988). Seasons were defined based on two monsoon winds: The North East Monsoon (NEM) and South East Monsoon (SEM) winds which have been known to coincide with the level of fishing effort and activity along the Kenya coast (McClanahan & Mangi, 2001; Ndegwa & Herrera, 2011; Munga et al., 2013). In the NEM season, the sea is calm and thus the intensity of fishing increased during this period. In contrast, the SEM season causes roughness in the sea resulting in a reduction in the level of fishing activity.

Shark landings data included fleet components, for example, fishing craft and type of gear used, fishing site, species landed and fishing days. The gears sampled included longline (LL)

(demersal and pelagic), drift net (DN), handline (HL) (including rod and reels), bottom-set nets (BSN) and combination of the three gears (referred to as “Combo”).

The landed sharks were photographed by taking a lateral photograph and the images utilized to identify specimens to species level. In cases where identification to species level was not possible (i.e. due to poor image quality or lack of detail), specimens were grouped to genus level. The total length (TL) of individual sharks were measured to the nearest cm using a standard measuring board. Other recorded information included the seasonal changes which were categorized based on the wind patterns.

2.4. Secondary data

2.4.1. *Historical, recreational fisheries tagging data*

Kenya is one of the key game fishing destinations along the coast of East Africa. The main sport fishing areas/centres include Diani, Shimoni, Kilifi, Watamu, and Malindi (Figure 1). However, the majority of sport fishing activity such as tournaments and competitions are undertaken in Malindi and Watamu. Marine recreational fishers also referred to as sporting anglers are mostly fishing tourists from Europe and North America and in some cases local tourists. Since the mid-1980s and early 1990s, the tradition of catch, tag and release have gained prominence in Kenya’s sport fishing industry. Initially, most billfish were boated and brought to the gantry. Recent years have seen a change in attitude towards releasing billfish, whereby about 95% of billfish are released (Harris et al. 2013; Kadagi et al. 2011). Tagging involves the use of hydroscopic nylon tags which are inserted below the dorsal fin of the fish.

Recreational fisheries historical tagging data on sharks were sourced from the African Billfish Foundation (ABF) tagging database (www.africanbillfish.org). ABF has overseen the tagging of billfish and other gamefish species in the WIO since the late 1980’s through its network of recreational fishing tourists and fishers. The Kenyan recreational fishery is designated as mostly catch and release (except few gamefish species). Participants in the ABF tagging programme primarily focus on billfish, but other species such as sharks, rays and giant trevally are also tagged. For this study, landings data on sharks were available for the period between 1987 - 2016. The shark tagging data was from 48 boats that operated off the main recreational sport fishing

centres along the Kenya coast although the number of boats fluctuated over time. Data included dates, year, type of shark and fishing gear used. It should be noted that the landings data for sharks only considers the species tagged and released by fishers participating in the tagging programme.

2.5. Data analyses

2.5.1. *Composition of shark landings in artisanal and recreational fisheries*

All data were analysed using R statistical software. To determine the composition of shark landings in both artisanal and recreational fisheries, proportions of each family and species were calculated. Additionally, the length composition of landed sharks and the effect of gear and seasonality of the shark landings were determined for both fisheries.

The historical tagging data for recreational fisheries from 1987 to 2016 were grouped into 10-year time series. These data were used to analyze differences in the composition of shark landings (families) in three distinct periods (1987 -1997, 1998 -2008, and 2008 -2016). The dates of tagging were used to determine the season in which the sharks were landed.

The differences in the number and proportion of shark landings according to types of season and gears in artisanal fisheries were investigated using the Kruskal–Wallis non-parametric analysis of variance.

2.5.2. *Influence of gear types and seasonality changes on the number of shark landings in artisanal fisheries*

A generalized linear model (GLM) (McCullagh and Nelder, 1989, Chambers and Hastie, 1992) was used to analyse the influence of seasonal changes (SEM and NEM) and gear types on the number of shark landings. The counts of artisanal landings were the dependent variable, while season and gear were factors. A poisson GLM with logarithm as the link function, which is appropriate for count data was used.

$$Y = \beta_0 + \beta_1 X_1 + \beta_2 X_2 + \beta_i X_i + \varepsilon Y$$

is the response variable (counts of shark landings)?

X_i are the explanatory variables (season and gear)?

β_0 is the intercept,

β_i are the coefficients that were estimated by maximum likelihood,

ε represents the error structure.

The best model was selected using the Akaike Information Criterion (AIC) method. The AIC scores were used to evaluate the influence of the predictor variables on the proportions of shark counts. In addition, the p-value and 95% confidence intervals (CI) were also extracted. Statistical significance of the variables in predicting the number of shark landings was considered to exist if $p < 0.05$.

3. Results

Trends in landing numbers in both the artisanal and recreational fisheries were presented based on the composition, and the effect of gear and seasonality. The study also provides comparisons of GLMs (based on AIC scores) and the results of the p-values and 95% CI for the best model explaining the effect of seasonal changes and gear on counts of shark landings in artisanal fisheries.

3.1. Shark landings in artisanal fisheries

A total of 1215 sharks across 21 species were recorded in artisanal fisheries. Most landed sharks belonged to family Carcharhinidae (33%) followed by Triakidae (29%), Lamnidae (20%), Sphyrnidae (17%). The proportions of *Hexanchidae*, *Hemigaleidae*, *Rhincodontidae* and *Squalidae* represented less than 1% of landed sharks. Specifically, *Mustelus* species featured prominently with the highest number (n=275) landed mainly in the NEM season, and accounting for 23% of the landings recorded while *Sphyrna lewini* (n=166, 16%) was mostly landed in SEM (Table 1). Results showed that the number of shark species landed were homogeneous (Table 1), although some species were frequently landed in either SEM or NEM.

More than half of the 21 species found fall under the IUCN RedList threatened categories Endangered (2) and Vulnerable (3), Data Deficient (2), Near Threatened (8), and six are in the lowest risk category (Table 6, appendix 1). The identification of the some of the species such as the *Mustelus* spp. was difficult, and were therefore reported to genus level owing to uncertainty in regional taxonomy. Additionally, about 35% of shark species were un-identified due to lack of corresponding photographic evidence and the cumulative use of similar common local names for different families and species.

Table 1: Reported number of shark landings (identified to species level) from the artisanal gillnet and longline fishery for the period between April 2016 -March 2017.

Species	Common name	NEM	SEM	Grand Total
Unidentified sharks	NA	193	227	420
<i>Mustelus sp.</i>	Smooth-hound	275		275
<i>Sphyrna lewini</i>	Scalloped hammerhead shark	36	130	166
<i>Carcharhinidae</i>	Carcharhinid shark	48	55	103
<i>Carcharhinus macroti</i>	Hardnose shark		65	65
<i>Loxodon macrorhinus</i>	Sliteye shark	30	26	56
<i>Carcharhinus sorrah</i>	Spottail shark	15	13	28
<i>Carcharhinus amblyrhynchos</i>	Grey Reef shark	13	2	15
<i>Isurus oxyrinchus</i>	Shortfin Mako	9	5	14
<i>Carcharhinus sealei</i>	Blackspot shark	2	8	10
<i>Galeocerdo cuvier</i>	Tiger shark	6	4	10

<i>Hexanchus nakamurai</i>	Bigeyed sixgill shark	9		9
<i>Carcharhinus leucas</i>	Bull shark	5	1	6
<i>Carcharhinus limbatus</i>	Blacktip shark	5	1	6
<i>Carcharhinus albimarginatus</i>	Silvertip shark	5		5
<i>Carcharhinus amboinensis</i>	Pigeye shark	5		5
<i>Carcharhinus melanopterus</i>	Blacktip Reef shark		3	3
<i>Carcharhinus sp.</i>	Carcharhinid shark	3		3
<i>Squalus sp.</i>	Spurdog	3		3
<i>Hemipistris elongata</i>	Snaggletooth shark		2	2
<i>Rhizoprionodon acutus</i>	Milk shark		2	2
<i>Carcharhinus longimanus</i>	Oceanic Whitetip shark	1		1
<i>Rhincodon typus</i>	Whale shark	1		1

The size distribution of shark landings in artisanal fisheries showed mixed sizes ranging from smaller individuals of <100 cm (52%) and medium to large sized individuals of >100 cm (48%). The largest size recorded for the captures of elasmobranch species in the artisanal fisheries was 568 cm for a whale shark, *Rhincodon typus*. The landings comprised of a wide range of size composition between 10 cm and 300 cm (Figure 2). Whilst the driftnets caught more sharks of <100 cm, the longlines were selective in catching bigger sized sharks. Gillnets (bottomset and drifting), longline and handline were used to landed over 85% of the sharks. longlines landed over 41% (n = 411), followed by driftnets (n = 303, 30%), bottom set (21%, n = 212) and handline (8%, n=81) and a combination of gears hereby named (Combo) had the lowest proportion of landings (17%, n = 208).

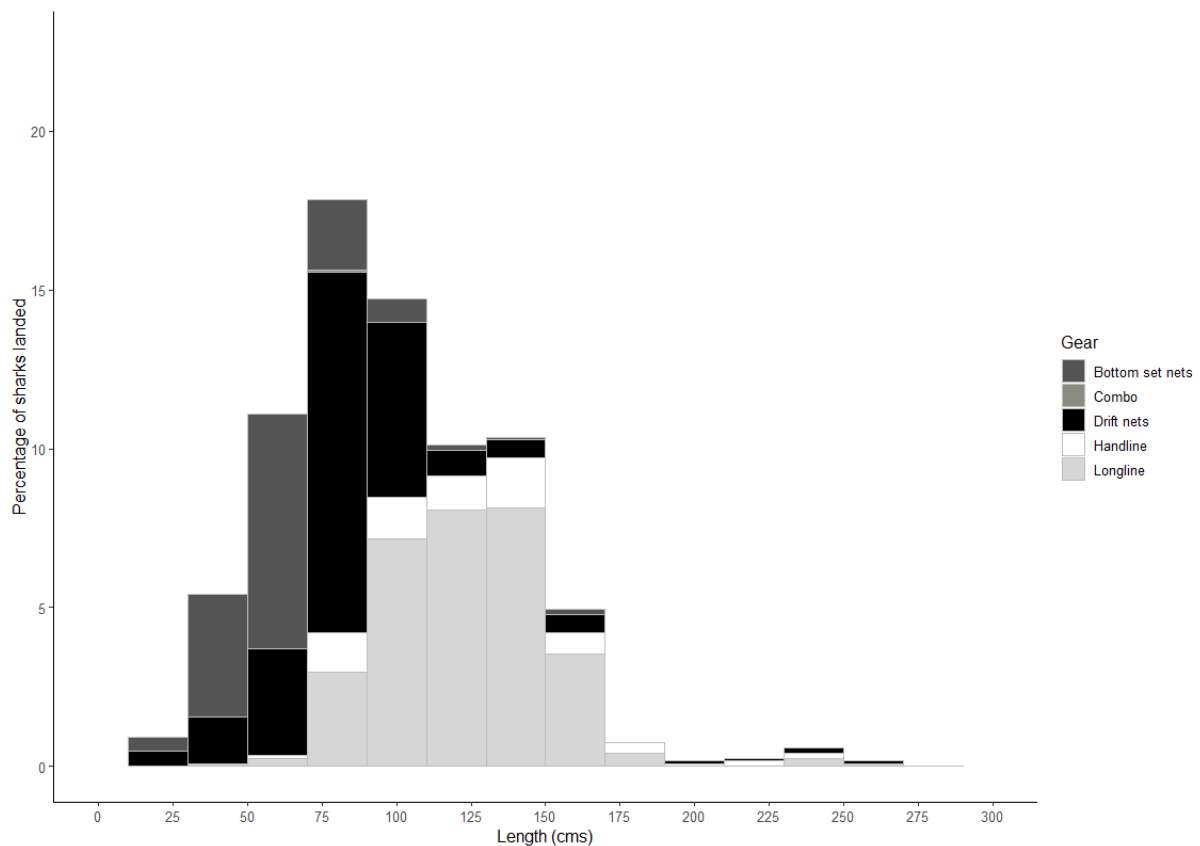


Figure 2: Percentage total length distribution histogram of sharks ($n = 1216$) examined at the 8 landing sites (Kiunga, Faza, Kizingitini, Kipini, Ngomeni, Mombasa and Vanga) in 25 cm size classes for the period between April 2016 and March 2017. The colours correspond to the percentage of shark landings in bottom set nets, combo, drift nets, handline, and longline respectively.

3.2. Proportions of artisanal shark landings in relation to gear types

Changes in proportions of landings (by species) were apparent based on the type of gear used (Fig 3). Longline and driftnets had the highest proportion of landed species (33.8% and 24.9 %), whereas bottom set nets, a combination of gears hereby named (Combo), and handline had the lowest proportion of landings (17.6%, 16.9% and 0.07% respectively). There were significant differences in the number of shark landings based on the types of gears ($X^2 = 9.23$, $p = 0.05$, $df = 4$)

An examination of estimated species proportions by shark landings by species (Figure 3) showed that some species were landed in specific gears. For example, *Rhizoprionodon acutus* and

Rhinconodon typus were primarily landed in bottom set nets while *Hemipistris elongata* was recorded in driftnets during the study period. Additionally, *Carcharhinus longimanus* was landed using handlines. The other 16 species were landed using the different types of gears.

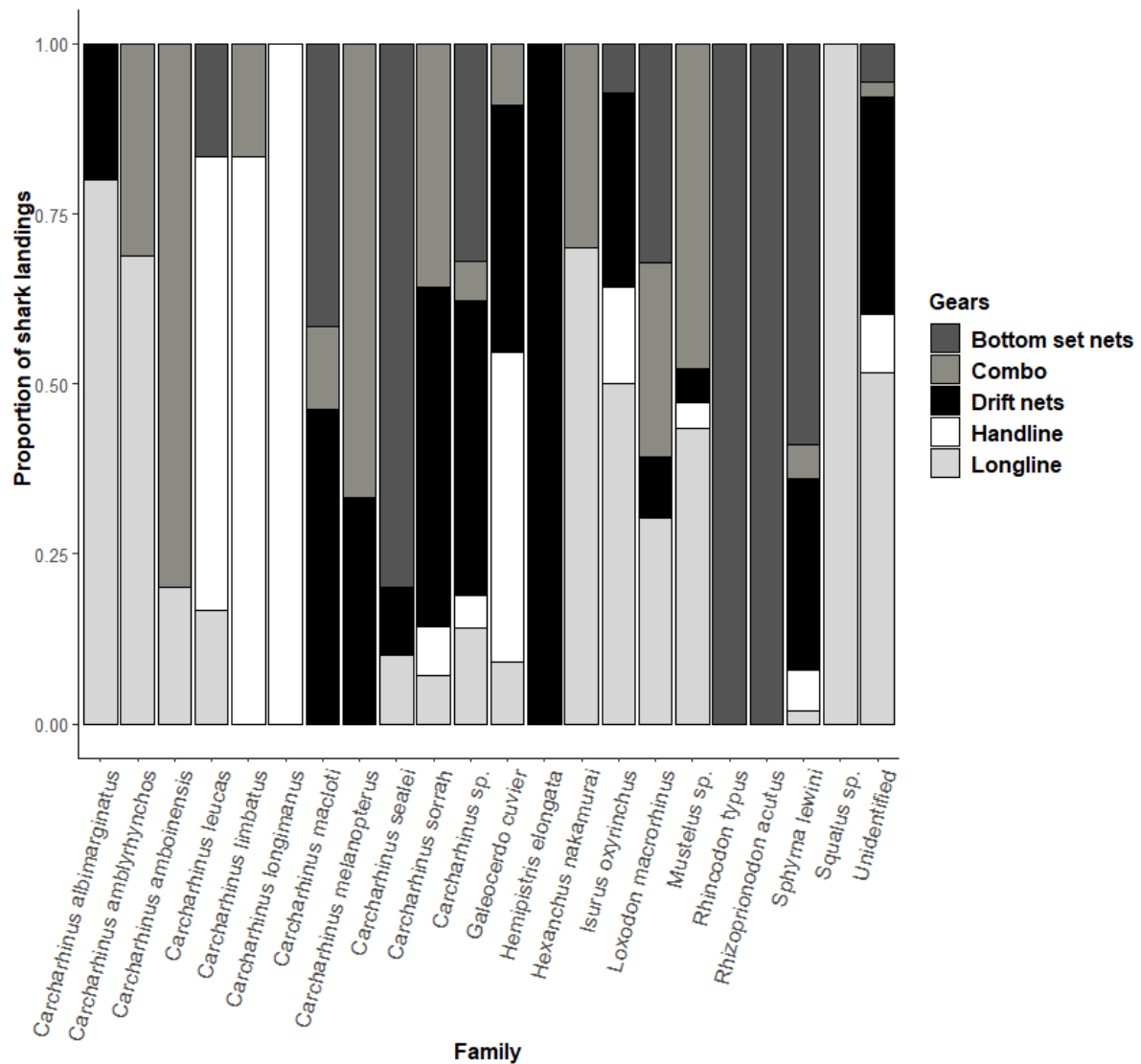


Figure 3: Proportions of shark species landed based on artisanal gear-type: Longline, Driftnet, Handline, Bottom-set nets and a combination of either of the three gears (referred to as “Combo”) for the period between April 2016 and March 2017.

3.3. Proportions of artisanal shark landings in relation to seasonal changes

Sharks were landed during both NEM and SEM season (Fig 4). A total of 664 individuals representing 17 species (54.7%) were landed during NEM season which runs from October to March. On the other hand, 551 individuals representing 16 species (45.3%) were landed in the SEM season which runs from April to September.

Even though more than 50% of shark landings were recorded in the NEM season, the top 4 species including *Mustelus* spp. (275), *Carcharhinus* spp. (51), *Sphyrna lewini* (36), *Loxodon macrorhinus* (30) were less than those fished during SEM (*Sphyrna lewini* (130), *Carcharhinus* sp. (55), *Carcharhinus macroti* (65), *Loxodon macrorhinus* (26). In addition, some of the landed species were only recorded either in the NEM or SEM season. For example, the *Mustelus* spp. were only caught in the NEM. Other species such as *Carcharhinus melanopterus*, *Carcharhinus macroti*, *Hemipistris elongata* and *Rhizoprionodon acutus* were only observed during NEM whereas *Carcharhinus albimarginatus*, *Carcharhinus amboinensis*, *Carcharhinus longimanus*, *Rhincodon typus* and *Squalus* sp. were recorded during the SEM. Significant differences in the number of shark landings ($X^2 = 0.34$, $p = 0.05546$, $df = 4$) were found among the two categories of seasons.

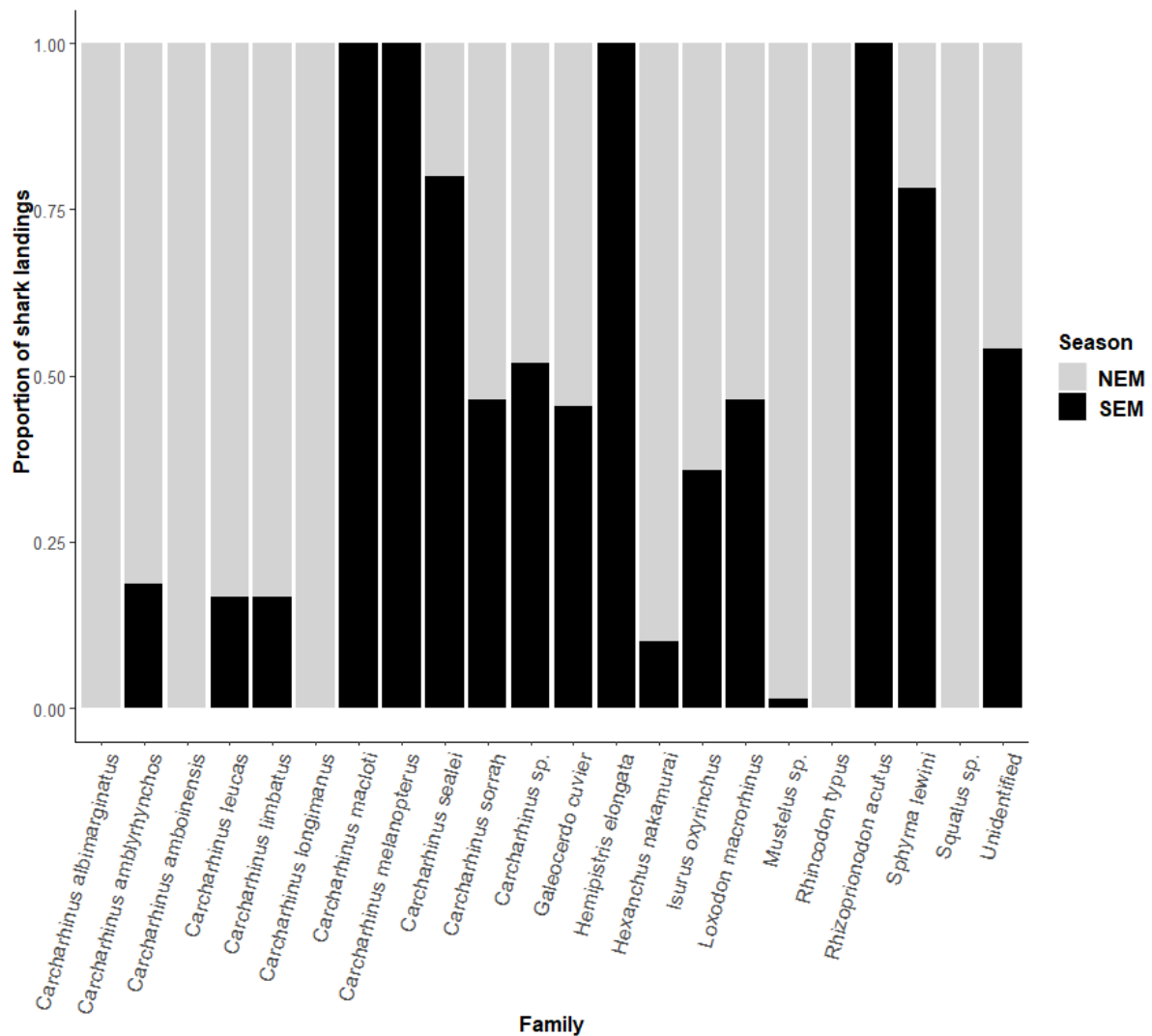


Fig 4: Proportions of shark species landed in during the North East Monsoon (NEM) and the South East Monsoon (SEM) in coastal artisanal fisheries between April 2016 to March 2017.

3.4. Shark landings in recreational fisheries

A total of sixteen (16) species belonging to seven families (7) were recorded in the recreational fisheries between 1987 and 2016. *Carcharhinus limbatus* was the most common species followed by *Sphyrna lewini* and *Carcharhinus longimata* in the three distinct 10-year time series for the historical recreational tagging data (Figure 5).

3.5. Proportions of recreational shark landings based on seasonal changes and gear types

Majority of the species (above 90%) were landed during the NEM season, with only a few species recorded in the SEM for the two distinct periods: 1987 to 1997 and 1998 -2008 (Figure

5). No sharks were recorded in the SEM for the period between 2009 -2016. Results of recreational shark landings in relation to gear type (Fig. 6 and 7) showed no apparent differences in size distribution of sharks when fishing rods and troll lines were used. Interestingly, both gears equally landed a diverse range of weight categories. The size distribution by weight ranged from smaller individuals of 10 kg to large sized individuals of 250 kg.

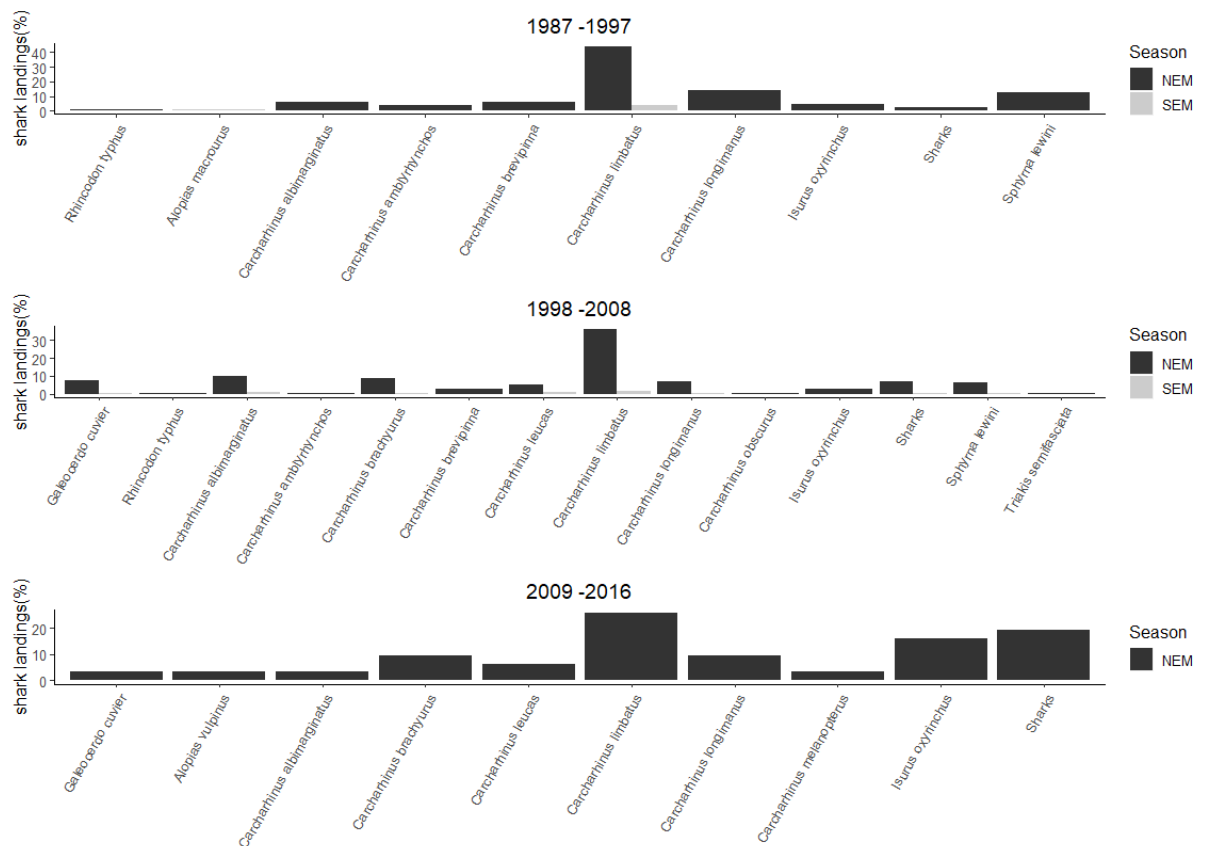


Figure 5. Percentage of shark of each species reported as landed by season in Kenyan recreational fishery through the billfish tagging program for the period between 1987 and 2016.

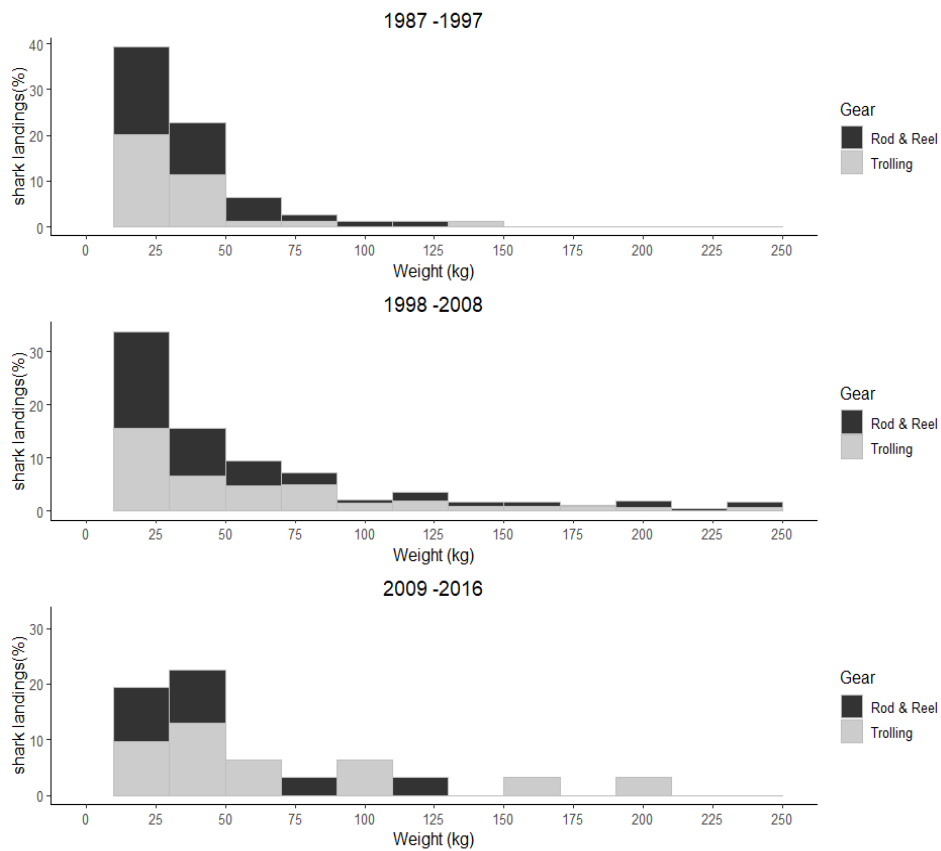


Figure 6: Percentage of shark species by weight (Kg) reported as landed in Kenyan recreational fishery using rod and reel, and trolling lines for the period between 1987 and 2016.

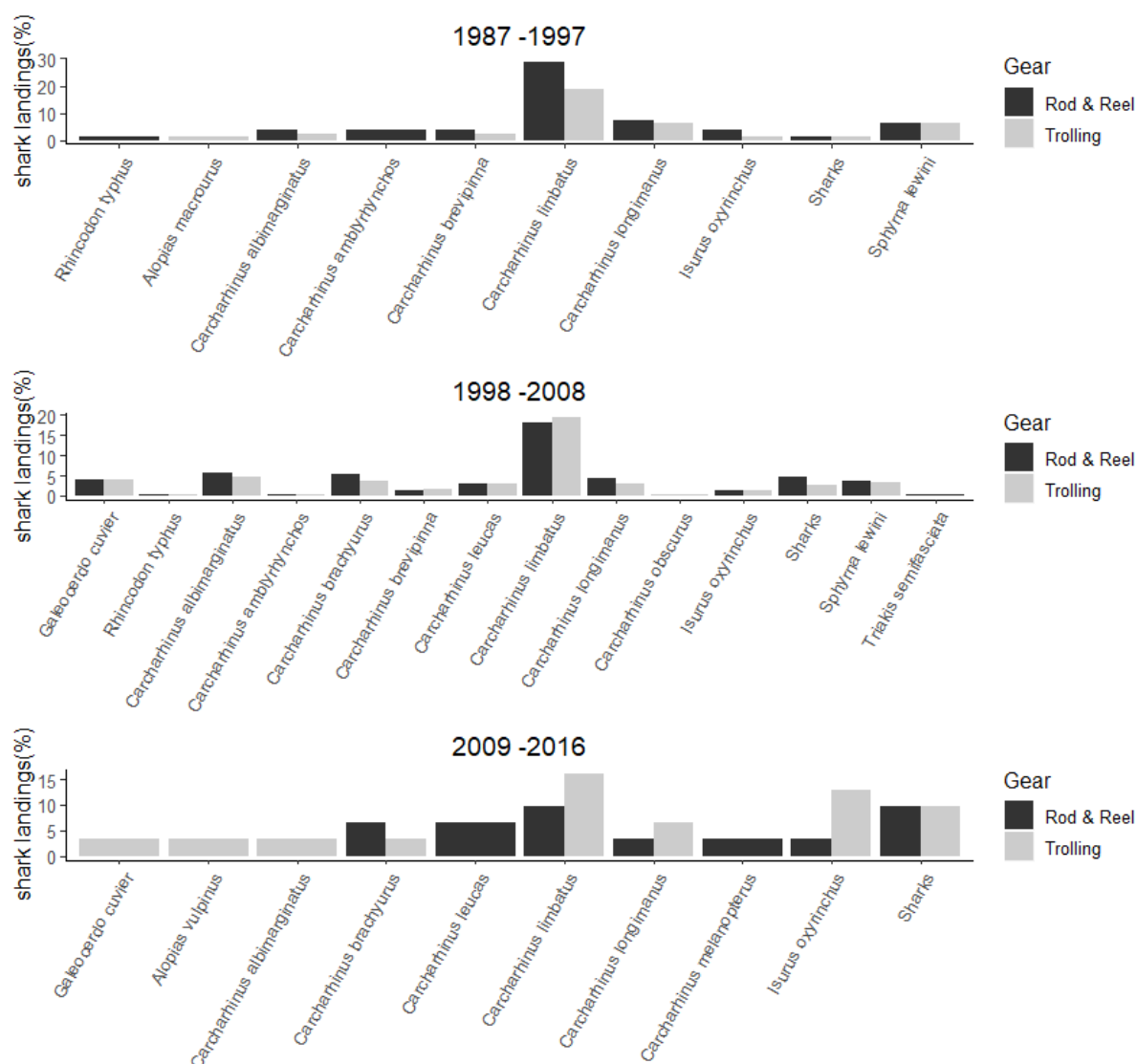


Figure 7: Percentage of shark landings by species in Kenyan recreational fishery using rod and reel, and trolling lines for the period between 1987 and 2016.

3.6. Influence of gear types and seasonal changes on the number of shark landings in artisanal fisheries

The GLM incorporating season, gear and the interaction effect of gear and season outperformed the other models (based on the lowest AIC value) and was thus selected as the best model (Table 2).

Table 2: Summary of the model selection information from three GLM analyses.

Model_names	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Season+Gear+Season*Gear		2678.099	0	1	1	-1328.54	1
Season+Gea	6	2826.082	147.9827	7.34e ⁻³³	7.34e ⁻³³	-1406.85	1
Gear	5	2841.411	163.3114	3.45e ⁻³⁶	3.45e ⁻³⁶	-1415.57	1
Season	2	3117.992	439.893	3.01e ⁻⁹⁶	3.01e ⁻⁹⁶	-1556.97	1

The single factor model with type gear as the predictor variable showed significant differences ($p < 0.05$) in the influence of gear types on the number of artisanal shark landings (Table 3). Conversely, the model with season as predictor variable showed no significant differences ($p > 0.05$) between SEM and NEM in relation to artisanal landings. In addition, the interactions between gear types and season were significant ($p < 0.05$) for (either combo, longline or driftnets vs. bottom seine nets). However, there were no significant differences ($p > 0.05$) in the number of shark landings between handline and bottom seine nets based on season. The significant results for these predictors (both gear and the interaction term) were likely as a product of the narrow confidence interval (Table 3).

Table 3: Coefficients and 95% CI from the best fit GLM fitted to artisanal shark landings count data with gear, seasonal changes and the interaction terms as the main effects. The interaction effect between variables is denoted by a colon. Significance levels of model terms are as follows: * $p < 0.05$, ** $p < 0.01$; * $p < 0.001$.**

Coefficients	Estimate	Std. Error	z value	Pr(> z)	2.50%	97.50%
(Intercept)	1.7148	0.14142	12.125 5	$<2e^{-16}$ ***	1.3416	1.8917
GearCombo	1.3238	0.16121	8.2115	$<2e^{-16}$ ***	1.1038	1.7323
GearDrift nets	-0.5171	0.17156	-3.014	0.0113e**	-0.7598	-0.0902
GearHandline	-0.8546	0.18116	-4.7172	0.0000195***	-1.206	-0.494
GearLongline	0.4863	0.15433	3.1508	0.0002 ***	0.1935	0.7998
SeasonSEM	-0.3587	0.16166	-2.2189	0.0887	-0.5797	0.0507
GearCombo: SeasonSEM	-1.5812	0.24034	-6.5791	$3.43e^{-12}$ ***	-2.1477	-1.2075
GearDrift nets: SeasonSEM	0.66	0.20161	3.2739	0.0041**	0.1761	0.0637
GearHandline : SeasonSEM	-0.096	0.61015	-0.1574	0.7659	-1.6166	0.857
GearLongline: SeasonSEM	0.5226	0.19148	2.7291	0.0218*	0.0565	0.8041

4. Discussion

There is a growing recognition that recreational (Gallagher et al. 2017, Molina and Cooke, 2012) and artisanal fisheries (Gurshin et al. 2004, Blaber et al. 2009) have an influence on the diversity and abundance of elasmobranch populations. Though evidence of the effects of these fisheries on shark populations is sparse and, in some cases, the information exists in grey literature in most regions. This study found that different shark's species were landed by artisanal and recreational fisheries, and that the number of landings were dependent on types of gears and seasonal changes. Findings highlight the importance of artisanal and recreational fisheries in assessing the status of shark populations and in the development of sustainable management approaches especially for vulnerable species.

4.1. Composition of shark landings in artisanal and recreational fisheries

Diverse species of sharks were recorded in both artisanal and recreational fishing sectors. A total of 21 species were recorded in artisanal gears, while 17 species were noted in the recreational tagging data. These findings corroborate with an assessment conducted along the SWIO region, which showed high diversity of species (Van der Elst and Everett, 2015, Fowler et al. 2005).

Findings show the effect of interaction between artisanal gears and shark species especially with more than half of the recorded species falling under the IUCN RedList threatened categories. The two endangered species *Rhincodon typus* and *Sphyrna lewini* encountered during this study highlights the impact that these fisheries may have on vulnerable shark populations. Further, the documentation of vulnerable, data deficient and near threatened species in this study is likely to reveal the threats posed by gears such as the gillnets and longlines. The difficulties in identifying some of the species is of particular concern given that more than 50% of these species are listed as threatened, thus amplifying the increasing need to improve species identification methods through training of data collectors. Most shark landings belonged to the genus *Mustelus* and could not be further identified to species level for the morphometric similarities they possessed. In some cases, the data enumerators assigned local names to a wide range of species or families e.g. the Carcharinids. Hence only photographic evidence would be used to identify the right species to the landings reported in this study. In the light of this, standardized data collection

methods and species identification protocols are required to improve the quality and applicability of data particularly in facilitating national and regional management of shark fisheries.

The wide range of size composition in the artisanal landings (e.g., milk shark, (*Rhizoprionodon acutus*), and blacktip reef shark (*Carcharhinus melanopterus*), and to some extent in the recreational fisheries (blacktip shark (*Carcharhinus limbatus*) and silver tip shark (*Carcharhinus albimarginatus*) are consistent with studies in Mexico and Costa Rica that show that a mixture of juvenile and immature sharks species are caught (Cartamil et al. 2011, Dapp et al. 2013). Although this study did not assess the magnitude of fishing pressure on the size composition, these findings are likely to support the premise that there is no size selectivity in shark landings from both fisheries. Hence, it is likely that the small-sized sharks are highly susceptible. Non-selectivity of shark sizes based on length could also be linked to the multi-gear nature of the artisanal fishery. There were differences in the composition of shark species and size structure in the historical recreational tagging data suggesting a strong influence of gear type in the landings. Although the recreational shark fisheries accounts for landings reported from boats that participated in the ABF tagging programme, above 90% of recreational boats that operate off the Kenya coast have actively participated in the tagging programme (pers. communication). Thus, the recreational shark landings data represents the best available dataset to make inferences on recreational shark fishery in Kenya due to the prevailing difficulties in collecting adequate data. Further investigations on size distributions and changes in size over a long period of time for artisanal fisheries are required to make substantial inferences and determine the susceptibility of species due to the type of fishery.

4.2. Seasonal changes and gear types on shark landings

This study also identified differences in the number of shark landings and species composition between NEM and SEM in both fisheries. A variety of species occurred during the NEM especially in artisanal fisheries. These results were in line with our initial hypothesis that shark landings in artisanal fisheries varied by season and fishing gear. The species diversity and reported landings seemed to be related to the different gears in the artisanal fisheries, which in turn increased the capability to catch more species. Studies on the south coast of Kenya found that artisanal fishers

were more likely to catch a wide array of species because of the use of multi-gears (Hicks & McClanahan, 2012, Okemwa et al. 2015, Tuda and Wolf, 2015, Tuda et al. 2016). The presence of multi-gears increases the potential for fishers to switch fishing gears based on the season. The ability to switch gears enables artisanal fishers to obtain a considerable amount of landings in the SEM regardless of the strong winds and the reduction in the fishing effort. This explains the observed all-year shark landings in artisanal fisheries compared to recreational fisheries that use mainly rod and reel as a gear deployed on boats. In contrast, most recreational charter boats reduce their operations in the SEM which seems plausible that the decrease in landings and species diversity might be related to the reduction in fishing effort. The landings in artisanal fisheries could also signify a growing interest to target sharks especially by engaging in shark finning which is perceived to have high returns. Assessing the effects of artisanal shark finning on the abundance of sharks and size classes will require accurate, extensive studies in future.

The seasonal variation of species with the lowest diversity recorded in the SEM and highest in NEM, could be related to the reproductive or latitudinal migratory behaviour of shark species. Patterns of habitat use due to inter-annual variability have been studied in several shark species (For example, latitudinal range extension in the summer months by tiger sharks in the south western Pacific Ocean (Holmes et al. 2014), and evidences of seasonal site fidelity in white sharks (*Carcharodon carcharias*) (Domier and Nasby-Lucas, 2013, Kock et al. 2013). Additionally, studies in artisanal shark fishery in Mexico indicated that landings were highly seasonal and varied among regions which was attributed to the migratory behavior of shark species (Castillo-Geniz et al. 1998). It possible that the observed influence of season on the composition and size structure of landings in the two fisheries could be as a result of seasonal shifts in habitat use. However, the scope of this study is insufficient to provide a strong support for this conclusion. Research should focus on identifying critical areas which are priority for creating a management framework and conservation of sharks in Kenya and the WIO.

4.3. Influence of gear types and seasonal changes on the number of shark landings in artisanal fisheries and recreational fisheries

The GLM model provided insights about the interactions between shark landings and variables such as fishing gears and seasonal changes, which is crucial in assessing underlying mechanisms determining the differences in the landings in Kenya. The gear type and seasonal influence were important factors in explaining the differences in artisanal shark landings. These findings were

consistent with our prediction that type of fishing gear and seasonal changes influenced the landings of sharks. Although there were no significant differences in the shark landings based on season, previous studies have shown a reduction in fishing effort during the SEM (Ndegwa and Herrera, 2011; Munga et al., 2013). The landings recorded in the SEM could be due to shifts in gear use among the artisanal fishers resulting in all year fishing. Further, the effect of seasonal changes has potential to influence the spawning of the sharks, their growth and development and movement. Similar findings have been observed in a gillnet fishery in Goa, India where the monsoons resulted in lower diversity of fish as compared to post-monsoon and pre-monsoons seasons where the abundance increased. Therefore, indicating that seasons had a direct effect on the abundance, richness and evenness (Sreekanth et al. 2018). Testing the effect of gear type and seasonal patterns on shark landings was possible through comparing the performance of different models. However, including other variables such as temperature and salinity, depth of fishing, day and night fishing, and location of fishing that have been shown to affect shark catch rates could improve the model outputs.

4.4. Management implications and recommendations

The outcomes presented in this study demonstrates that recreational and artisanal shark fisheries have influence on shark landings, and highlights the need to evaluate the implications of these fisheries on vulnerable shark's species. The study also reveals that fishers could play a vital role as citizen scientists in collecting information particularly for data deficient and data-limited fish species. The availability of 30 plus years of historical recreational conventional shark tagging database is one example.

On a broad scale, several key points for management consideration can be drawn from these findings as follows: Firstly, although the Kenya artisanal fishery for sharks may be considered as "low- medium sophisticated", there is possibility of growth and expansion to include highly motorized boats. The possible shift by artisanal fisheries from small, mostly traditional fishing gears and crafts to modern ones may challenge the local definition of artisanal fisheries with regards to highly susceptible species such as sharks. Secondly, the presence of recreational shark fisheries based on historical tagging data presents evidence of a shared fishery resource thus a unique aspect and priority in the process of developing the Kenya's National Plan of Action for the Conservation and Management of Sharks (NPOA- Sharks); and thirdly, the ability of some shark species to transverse across jurisdictions of different countries could imply that the

dynamics of landings are more “regional” than “localized” introducing the need for a regional management approach both from artisanal, recreational and commercial fisheries perspectives. Although the scope of this study cannot comprehend if these fisheries implications on sharks are on a wider scale, the question of endangered, vulnerable, data deficient, near threatened and unknown category of shark species in the face of competing resource users (fishers and countries) is critical for Kenya and the WIO region (Kiszka, and van der Elst., 2015). Considerable efforts should be made to obtain detailed analyses for specific species within these fisheries. This must also include education and awareness among different resource users on the IUCN categories of the species to protect and conserve them.

5. Conclusion

Our study provides an important baseline for further research while contributing to the ongoing efforts to develop a Kenya National Plan of Action for Sharks (NPOA-Sharks). The additional information on recreational and artisanal shark fisheries accentuates the need for awareness and education to strengthen the capacity of the public and government agencies to collaborate in citizen science. Given the importance of historical shark tagging data and fishers knowledge on sharks, promoting participation of fishers in a collaborative monitoring program will increase the understanding of the risks and threats to shark populations especially for endangered species. Furthermore, there is a need to prioritize management actions to monitor fishing pressure in both NEM and SEM particularly for sharks given that they are caught throughout the year in different gears. This is to ensure the preservation of the remaining shark populations and long term sustainable fisheries.

This work is also timely as Kenya is in the process of developing a National Plan of Action for sharks and rays. Thus, these findings will contribute to these efforts and provide a new baseline for future research on the implications of recreational and artisanal fisheries on sharks in the Kenya and the WIO region.

AUTHOR CONTRIBUTIONS

Andrew Temple, Nina Wambiji and Nelly Isigi Kadagi completed data cleaning. Nelly Isigi Kadagi and Nina Wambiji analyzed the data. Nina Wambiji and Nelly Isigi Kadagi wrote the manuscript. Andrew Temple contributed country-specific information for artisanal shark catch used in the

manuscript. Paul Tuda, Andrew Temple, Per Berggren, Jeremy Kiszka, Edward Kimani, Chris Poonian, Narriman Jiddawi advised on methods, guided analysis, contributed to the manuscript and reviewed and edited the manuscript.

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ANNEXES

Table 2. A summary of landings by species including estimated biomass and standard length

Common name	Latin name	IUCN Redlist status
<i>Carcharhinus albimarginatus</i>	Silvertip Shark	V
<i>Carcharhinus amblyrhynchos</i>	Grey Reef Shark	NT
<i>Carcharhinus amboinensis</i>	Pigeye Shark	DD
<i>Carcharhinus dussumieri</i>	Whitecheek Shark	NT
<i>Carcharhinus leucas</i>	Bull Shark	NT
<i>Carcharhinus limbatus</i>	Blacktip Shark	NT
<i>Carcharhinus longimanus</i>	Oceanic Whitetip	V
<i>Carcharhinus macroti</i>	Hardnose Shark	NT
<i>Carcharhinus melanopterus</i>	Blacktip Reef Shark	NT
<i>Carcharhinus sorrah</i>	Spottail Shark	NT
<i>Carcharhinus</i> spp.	Carcharhinid Shark	
<i>Galeocerdo cuvier</i>	Tiger shark	NT
<i>Hemipistris elongata</i>	Snaggletooth Shark	V

<i>Hexanchus nakamurai</i>	Bigeyed sixgill shark	DD
<i>Isurus oxyrinchus</i>	Shortfin Mako	V
<i>Loxodon macrorhinus</i>	Sliteye Shark	LC
<i>Mustelus spp.</i>	Smoothhound	
<i>Rhincodon typus</i>	Whale Shark	E
<i>Rhizoprionodon acutus</i>	Milk Shark	LC
<i>Sphyrna lewini</i>	Scalloped hammerhead shark	E
<i>Squalus spp.</i>	Spurdog	
-	Un-identified sharks	

NB: IUCN Red List status (V= vulnerable, NT = near threatened, E= endangered, LC= least concern).

Different species of shark were caught in the different seasons. For example, and *Rhizoprionodon acutus* were not caught during NEM. On the other hand, *Carcharhinus albimarginatus*, *Carcharhinus amboinensis*, *Carcharhinus longimanus* and *Rhincodon typus* were not caught in SEM. Other species were exclusively caught by certain gears in the different seasons such as *Carcharhinus longimanus*, *Carcharhinus limbatus* were only caught by handlines during NEM while *Carcharhinus limbatus* was caught by various gears during SEM. Handlines and longlines were mostly used during NEM than SEM which was dominated by bottom set nets, driftnets and combos (Figure 8 in the Annex)