Megafauna Interactions with East African Small-Scale Fisheries

A thesis submitted by

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Abstract

Small Scale Fisheries (SFF) in the developing southwestern Indian Ocean (SWIO) region employ half a million fishers, contribute over 70% of marine fisheries catch and their importance to coastal communities for food and income cannot be overstated. 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, and play important roles in the structure and function of marine ecosystems. Yet, little is known of the interactions between SWIO SSF and these species. This thesis provides the first independent regional assessment of elasmobranch catch volume and composition based on fisheries landings data and provides evidence for the ongoing catch of marine mammals and sea turtles across SWIO SSF. Elasmobranch catches were estimated at 73% more than reported to the Food and Agriculture Organization of the United Nations in 2016 and 129% more than the 10-year average (2006-16). However, subsequent vulnerability assessments of species is restricted by limited, or absent, life-history data. The thesis provides the first life-history data for the recently described Baraka's whipray (*Maculabatis ambigua*), an important component of the ray catch. The life history results suggest a species of potentially high resilience to fisheries exploitation, although the observed SFF catch demographics indicate a potentially unsustainable catch pattern. The thesis also explores for the first time the dependence of fishers on elasmobranch resources, their use and value. It provides evidence of a specialised livelihood strategy exacerbating elasmobranch dependent fishers' vulnerability to external shocks and highlighting them as a potential target group for livelihood diversification programmes. Lastly, the thesis compares rapid interview-based assessment methods for marine megafauna fisheries catches with data derived from observed fisheries landings and finds that the outputs of these methods show little evidence of equivalence. It indicates the need for a multi-method approach in assessing marine megafauna interactions with SSF in data-poor regions. The thesis demonstrates the challenges of understanding the interactions between small-scale fisheries and marine megafauna and highlights the need for rapid yet effective approaches toward generating priority baseline data for fisheries effort and catch to provide a basis for evidence-based management and feasible solutions.

Dedication

To my parents, William and Dawn Temple, who have supported and encouraged me through all the ups and downs and for which I will be forever grateful.

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Thesis Chapter 2

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Pilot study for thesis Chapter 3

Barrowclift, E., **Temple, A. J**., Stead, S., Jiddawi, N. S. and Berggren, P. (2017) Social, economic and trade characteristics of the elasmobranch fishery on Unguja Island, Zanzibar, East Africa. Marine Policy. 83: 128-136. https://doi.org/10.1016/j.marpol.2017.06.002

Conference Presentations

Temple, A. J., Amir, O. A., Brito, A., Berggren, P., Everett, B., Fennessy, S., Jiddawi, N. S., Kimani, E., Kiszka, J., Ngisiang'e, N., Ong'anda, H., Perez, S., Poonian, C. N. S., Razafindrakoto, Y., Stead, S., Wambiji, N. (2015) Artisanal Vulnerable Megafauna Catch – BYCAM, an Overview. 9th Western Indian Ocean Marine Science Association Scientific Symposium, Wild Coast, South Africa.

Temple, A. J., Kiszka, J. J., Poonian, C. N. S., Wambiji, N., Amir, O. A., Jiddawi, N., Berggren, P. (2017) Assessing marine megafauna captures in small-scale fisheries of the southwestern Indian Ocean: landings data and questionnaire survey disparity. 10th Western Indian Ocean Marine Science Association Scientific Symposium, Dar es Salaam, Tanzania.

Thesis Overview

Background and Rationale

Coastal communities in the southwestern Indian Ocean (SWIO) region, herein referring to the nations of Kenya, Tanzania (including Zanzibar), Mozambique, Seychelles, Comoros, Madagascar, La Réunion and Mauritius, have traditionally relied on marine fisheries as a source of food and income. The SWIO region is undergoing rapid development and the population is set to double to 397.2 million by 2050 (WB 2016b). Coastal cities and island nations are experiencing particularly high rates of population increase, e.g. 5.6% in Dar es Salaam (URT 2013), 3.5% in Mombasa (KNBS 2010), 4.2% in Urban Zanzibar (URT 2013), 5% in Maputo province (INE 2007). Meanwhile migration to coastal areas and interest in marine resources for consumption in inland areas are increasing (Kamulaka 1984; Reuveny 2007). Thus, pressure on marine resources is rising and food security and income generation are major policy drivers. Despite this, little is known of the current state and dynamics of the SWIO's marine fisheries sectors, particularly the small-scale fisheries (SSF) which are generally overlooked despite employing nearly half a million fishers (Temple *et al.* 2018), contributing an estimated 71.9% of marine fisheries catch in 2014 (Pauly and Zeller 2015) and their importance to the livelihoods of coastal communities.

Fisheries present a prominent short to medium-term anthropogenic threat to the survival of numerous marine vertebrates. This is particularly true for those species predominantly displaying classically k-selected life history traits (long-life, high natural survivorship, slow growth, late maturity and low fecundity), such as elasmobranchs (sharks and rays), marine mammals and sea turtles (e.g. Lewison *et al.* 2004; Read *et al.* 2006; Žydelis *et al.* 2009; Wallace *et al.* 2010; Dulvy *et al.* 2014). These taxa are commonly caught, either as target, by-product or bycatch throughout SWIO SSFs and used for income generation, subsistence and as bait (e.g. Razafindrakoto *et al.* 2008; Humber *et al.* 2011). Yet, little is known of the volume or composition of these catches despite widespread indications of overexploitation (e.g. Shehe and Jiddawi 1997; Amir *et al.* 2002; Kiszka 2012; Muir and Kiszka 2012), nor their relative impact on socio-economic drivers that underpin the resilience of coastal communities. Currently, it is therefore difficult to formulate evidence-based management strategies, where required, to ensure sustainable fisheries and prevent species extirpation.

Generally, independent assessments of catch and composition of elasmobranchs, marine mammals and sea turtles have been undertaken in restricted case-studies around the SWIO (e.g. Amir *et al.* 2002; Brito 2012; Robinson and Sauer 2013). Whilst these give important insights into elements of fisheries such as area-specific gear threats and sustainability of catch for specific populations of species, their restricted spatial coverage limits the extrapolation and thus broader application of their findings for management. However, recently a greater research effort has been made to assess these taxa's interactions with, and vulnerability to, SSF at a national and international level within the SWIO (e.g. Poonian *et al.* 2008; Moore *et al.* 2010; Kiszka 2012; Poonian 2015). These efforts have primarily focussed around collection of fisheries catch data through structured or semi-structured interview surveys. However, the outputs of the interview methods in use have yet to be cross-examined against data derived from other traditional scientific methods (such as landings or vessel based monitoring) and thus appropriate interpretation of their results is difficult. Yet, uninformed and inappropriate interpretation of the data used to formulate management strategies risk detrimental impacts to both these resources and the communities reliant upon them.

Interview methods are comparatively cheap and less time-intensive compared to traditional monitoring methods (e.g. landings or vessel based observation) and may be especially valuable given the restrictions in technical expertise and financial capacity faced in SWIO SSF, and SSF elsewhere. They may be particularly helpful in the collection of data for rare and/or illegal catches which are liable to be missed by traditional methods and may further help to access indigenous knowledge that can provide insight not attainable by traditional methods (Johannes *et al.* 2000). Comparison of interview methods against those of traditional methods is a clear priority to allow informed interpretation of their outputs.

Lastly, assessment of the current vulnerability of species to fisheries, and subsequent inference of the risk of overexploitation, are required to inform management priorities. Broadly, such assessments require a combination of fisheries exploitation and species life-history data, and may take the form of fisheries sustainability assessment, full stock assessment, demographic modelling and prediction of exploitation rebound potential (Frisk *et al.* 2001; Cailliet and Goldman 2004; Smith *et al.* 2008). Life-history may vary considerably both among (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), but few regional life-history studies are available for species caught in the SWIO SSF. Generation of life-history data (particularly age-growth, maturity, longevity and reproduction), especially for those species widely exploited and of socio-economic importance to the coastal communities that rely on them, is therefore a priority in the short-term.

Problem Statement

The aim of balancing the conservation of natural resources with their sustainable use by the peoples whom are reliant upon them is a global issue. Achieving this is complicated when little is known of either the current state of the resource, its exploitation, use, and the dependence of people upon it. Despite this, resource managers may be obligated to attempt to achieve such an aim and in doing so risk taking actions that are ineffective or even detrimental to long-term sustainability of both the natural resources and their users.

The aim of this thesis is to address such information gaps in the case of elasmobranch, marine mammal and sea turtle (herein referred to as marine megafauna) exploitation in SWIO SSF and so contribute towards the long-term sustainability of both the marine resources and the communities reliant upon them.

Thesis Summary

Chapter 1 aimed to review the current exploitation status of marine megafauna in SWIO SSF and to identify priority data gaps required to facilitate sustainable long-term management. The review highlighted an urgent need to improve the documentation, monitoring and assessment of SSF at the regional level. SSF data for fisheries effort are generally restricted to vessel numbers with relatively poor quantification of gear prevalence and inconsistent use of metrics among countries, a clear priority for management of SSF at the regional level. Catch data for elasmobranchs in SSF were found to be of poor spatial resolution with limited compositional data. Indeed compositional data were generally anecdotal and/or biased towards easily identifiable species (e.g. Kiszka 2012; Poonian 2015; FAO 2018b). Further, little information was available for marine mammal and sea turtle captures other than in case-study areas (e.g. Amir *et al.* 2002; Poonian *et al.* 2008; Robinson and Sauer 2013). It was concluded that it was not possible to effectively assess the current status of elasmobranchs, marine mammals or sea turtles in SWIO SSF.

Chapter 2 sought to address the clear gap in understanding of both the scale and composition of elasmobranch, marine mammal and sea turtle catch in SWIO SSF as was highlighted in Chapter 1. It was decided that a landings observation approach would be taken. The approach is most suitable for the elasmobranch component of the catch, with the marine mammal and sea turtle components both illegal to land (Temple *et al.* 2018) and thus liable to be hidden from observers. The chapter undertook a cross-sectional approach, with 12 months (2016-17) landing site monitoring at 21 sites across three countries (Kenya, Zanzibar and northern Madagascar). The study focussed primarily on the bottom-set and drift gillnet and longline fisheries, as these have been identified as the main gear threats to marine megafauna in the region (Read *et al.* 2006; Kiszka 2012; Wallace *et al.* 2013), as well as the numerically dominant handline fisheries. The study identified a minimum of 59 species caught in SIWO SSF, including three sea turtles, two small cetaceans and one sirenian (*Dugong dugon*). Overall the catch was dominated by small and moderately sized coastal requiem sharks (Carcharhiniformes) and whiprays (Dasyatidae), and a Productivity-Susceptibility Assessment found that these species were generally most vulnerable to SWIO SSF. Catch of oceanic, deeper-water and large coastal elasmobranchs was also notable and demonstrated the potential for SWIO SSF to impact a range of ecosystems. Catches of elasmobranch in the SWIO SSF were estimated at 35,445 (95%CI 30,478-40,412) tonnes, 72.6% more than reported to the FAO in 2016 and 129.2% more than the 10-year average (2006-16), constituting 2.48 (95%CI 2.20-2.66) million individuals.

Chapter 3 recognises that in order to formulate appropriate management strategies for marine megafauna, resource managers must understand not only the exploitation of these resources but also the dependence of fishers and communities upon them. This chapter aimed to improve the understanding of the broadscale socio-economic context of SWIO elasmobranch SSF, thus informing the vulnerability context (Ashley and Carney 1999) in which dependent fishers and fisher households exist. In order to achieve this aim Chapter 3 combines investigation of dependence, use and value through interviews of resource users with data on use and value of elasmobranch catches directly from landings observations. The study suggests that elasmobranch dependence was linked with fisher experience and financial capital. It also suggests 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 (Allison *et al.* 2006; Béné *et al.* 2007;

Béné 2009). The findings also indicate that infrastructure and access to external, and likely international, markets are linked to commercial demand for elasmobranch products, primarily for shark and shark-like rays which appear to be supply-limited, in the SWIO. A market governance strategy to reduce demand of elasmobranch products (particularly shark fin and *Mobula* spp. gill rakers) may be effective in altering fisher behaviour and dependence upon these resources. However, such a strategy must not be enacted in isolation given its potential impacts on fisher livelihoods and wellbeing (e.g. Lawrence 2001). Targeted programmes to increase livelihoods diversity or provide alternative livelihoods may be an effective pathway to reducing the vulnerability of these fishers whilst reducing pressure on elasmobranch resources. Potential livelihoods programmes must account for fisher and community attitudes and perceptions towards proposed livelihoods (Harrison 1996; Kaiser and Stead 2002; Slater *et al.* 2013) and the effects of personality traits (Barrick and Mount 1991; Mount *et al.* 2005; Schmitt *et al.* 2008) on engagement and success in alternate livelihoods.

In the process of assessing vulnerability risk to species from SWIO SSF in Chapter 2 it became clear that many species affected by these fisheries lack information on basic life-history parameters, despite these being essential in understanding rebound potential, for assessing vulnerability, stock assessment and demographic modelling (Frisk *et al.* 2001; Cailliet and Goldman 2004; Smith *et al.* 2008). Chapter 2 found that the recently described Baraka's Whipray (*Maculabatis ambigua*) (Last *et al.* 2016) was a dominant constituent of the Kenyan SSF ray catch, as well as a common constituent of Zanzibar's SSF catch. Further, in Chapter 3 the species was found to be of commercial value in SWIO SSF, which combined with its commonality in the catch suggests its potential importance to fishers as a source of income. Despite the potential importance of *M. ambigua* in these fisheries, no life-history data are currently available for the species. Chapter 4 investigated the major life-history characteristics of this species in order to aid future assessments and the formulation of any resultant management measures. Specimens were aged using the vertebral sagittal sectioning method and subsequently the size-weight relationship, age-size relationship, maturity and longevity were assessed. The data indicate that *M. ambigua* is a moderately-sized, rapidly growing, early maturing species of whipray with a moderately long lifespan, suggesting that it may be relatively resilient to fisheries exploitation. However, information on fecundity was not possible to collect and therefore the rebound potential of the species is unknown. Further, a cross-sectional Chapman-Robson catch curve was constructed which indicates that *M. ambigua* is exploited across a wide age range, with full recruitment to the fisheries occurring post-maturation, a fisheries exploitation pattern which is generally unsustainable for elasmobranch fishes (Simpfendorfer 1999; Prince 2002).

Chapter 1 identified that, prior to the research carried out in this thesis, the majority of assessments of marine megafauna interactions with SWIO SSF had been carried out in small case-studies (e.g. Amir *et al.* 2002; Poonian *et al.* 2008; Robinson and Sauer 2013). However, some regional-level assessments had been carried out using interview-based methods to access local fisheries knowledge (LFK), to assess these interactions (Moore *et al.* 2010; Kiszka 2012). Further, the interview method used in the previous studies, the Rapid Bycatch Assessment (RBA), is being considered for widespread use to document marine megafauna-SSF interactions in data-poor fisheries. Yet, the outputs of RBAs have not been cross-examined with traditional observation-based methods. In order to allow for the informed use of RBAs, and other similar interview-based methods, it should be a priority to cross-examine their outputs with observation-based methods. Otherwise, if the resulting data are inadequate or wrongly interpreted and used for management, it could have wide-ranging consequences for both resource users, their communities, the marine megafauna resources and the wider environment. LFK is generally considered as a useful indicator of long-term inter-annual trends (Neis *et al.* 1999; Daw *et al.* 2011; O'Donnell *et al.* 2012; Beaudreau and Levin 2014). Thus, Chapter 5 focuses instead on the use of LFK in assessing intra-annual and annual patterns in fisheries effort and catches. This was achieved through the use of RBAs to collect data across the same spatial and temporal scale as the landings based observations conducted in Chapter 2 and by comparing the outputs of the two methods using the Bland-Altman approach (Bland and Altman 1999, 2003). The results demonstrate inconsistency in relationships among spatial and temporal fishing effort and catch patterns from the two methods, with the majority showing no evidence of relationships. Positive relationships appeared more common where patterns in effort and catch displayed a large degree of intra-annual variability and/or for easily identifiable 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.

Finally, an overview of the findings from this thesis is presented in Chapter 6 alongside future considerations for first-step assessments of marine megafauna interactions with SSF in data-poor environments and concluding thoughts on potential management interventions that may be feasible in SSF.

Chapter 1. Marine Megafauna Interactions with Small-Scale Fisheries of the Southwestern Indian Ocean: a Review of Status and Challenges for Research and Management

1.1 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 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.

1.1.1 Key Words

Bycatch; elasmobranch; mammal; turtle; conservation; livelihoods

1.2 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; Žydelis 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 rays), marine mammals (specifically cetaceans and sirenians) and sea turtles represent some of the most threatened animal groups (Table 1.1). 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 rays, 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.

1.3 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) (Figure 1.1; Table 1.2). 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 (Table 1.2). 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

Table 1.1 IUCN Red List global status and population trends for vulnerable megafauna globally and in the western Indian Ocean (FAO Fishing Area 51). Species numbers are broken down by IUCN Red List category. (Red List Categories: DD = Data Deficient, LC = Least Concern, NT = Near Threatened, VU = Vulnerable, EN= Endangered, CR = Critically Endangered. Global Trend: Inc = Increasing, Sta = Stable, Dec = Decreasing, Unk = Unknown.). Source: (IUCN 2016; Weigmann 2016)

Species Category	Spatial	Total Species	IUCN Assessed Species	Red List Status						Total % ≥ VU	Total % DD	Global Trend				Total % Dec	Total % Unk
				DD	LC	NT	VU	EN	CR			Inc	Sta	Dec	Unk		
Chondrichthyans	Global	1188	1067	458	310	118	120	42	19	17.0%	42.9%	7	75	152	834	14.2%	78.2%
	WIO		235	93	34	45	46	12	5	26.8%	39.6%	1	8	57	169	24.3%	71.9%
Cetaceans and Sirenians	Global	94	92	45	22	5	10	7	2	20.7%	48.9%	4	2	12	74	13.0%	80.4%
	WIO		39	19	13	1	3	3	0	15.4%	48.7%	3	1	3	32	7.7%	82.1%
Sea Turtles	Global	7	7	1	0	0	3	1	2	85.7%	14.3%	0	0	5	2	71.4%	28.6%
	WIO		5	0	0	0	3	1	1	100%	0%	0	0	5	0	100%	0%

Table 1.2 Southwestern Indian Ocean country metrics relating to population, development and importance of marine and freshwater fisheries for income and sustenance. Source: (FAO 2012; WB 2012; de Graaf and Garibaldi 2014; UN 2015)

Country	Population 2015 ('000)	Population estimate 2050 ('000)	Average Population Growth (%)	Human Development Index	Average Dietary Fish Protein Intake (% total animal protein)	Fisheries and Mariculture GDP Contribution (%)
Comoros	788	1,502	2.4	Low	51.8	15.00
Kenya	46,050	95,505	2.7	Low	7.6	0.50 - 0.54
Madagascar	24,235	55,294	2.8	Low	15.3	2.76 - 5.50
Mauritius	1,273	1,249	0.4	High	17.2	0.17 - 1.00
Mayotte	240	487	2.8	Very High	-	-
Mozambique	27,978	65,544	2.8	Low	40.3	3.73 - 4.00
La Réunion	861	989	0.7	Very High	5.8	-
Seychelles	96	100	0.7	High	47.6	30.00
Tanzania	53,470	137,136	3.2	Low	21.8	2.70 - 3.07 (Mainland) 6.67 (Zanzibar)

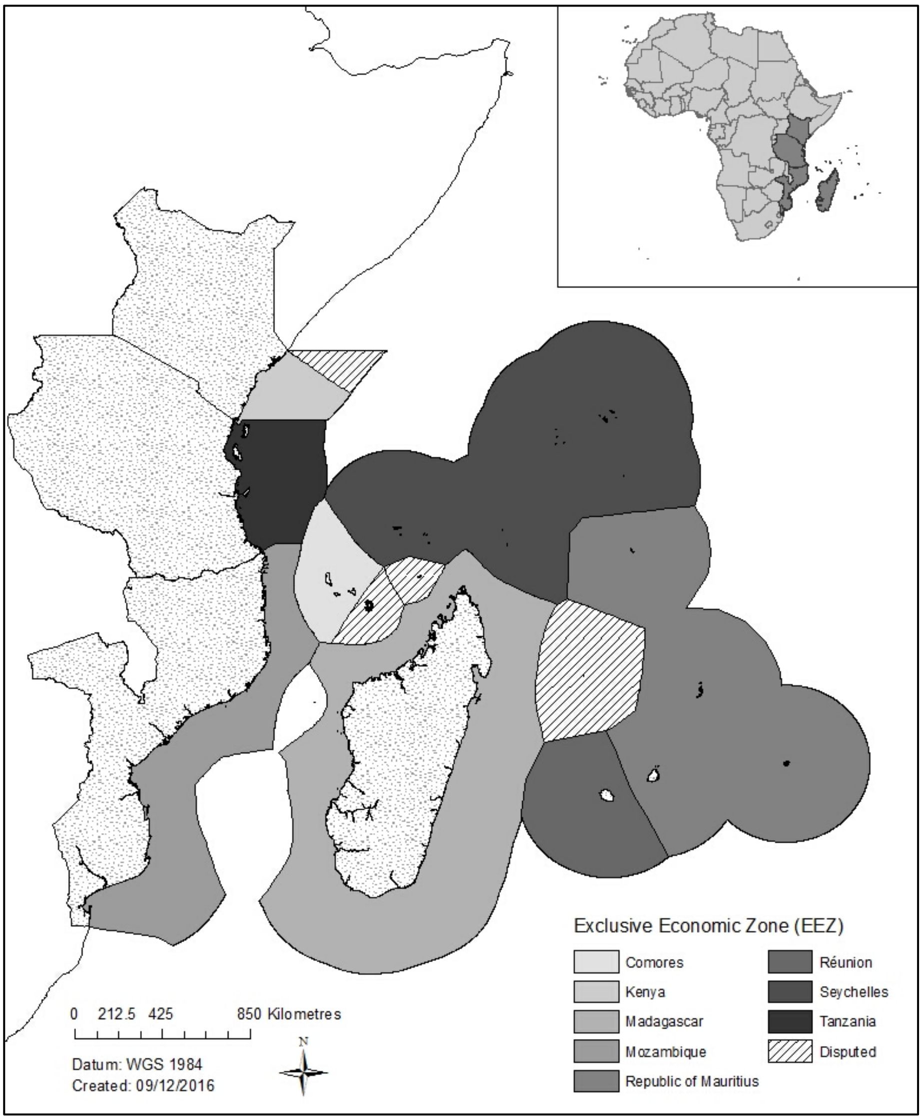

Figure 1.1 The southwestern Indian Ocean: Comoros, Kenya, Madagascar, Mauritius, Mayotte and La Réunion (France), Mozambique, Seychelles and Tanzania (mainland and Zanzibar). Sources: (ESRI 2014; VLIZ 2014)

(Table 1.2). However, other marine income-generating activities also contribute to local economies, particularly marine-tourism activities, worth around $3.95billion, 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 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 (Table 1.1) and regional scale (Kiszka 2015; Kiszka and van der Elst 2015). These uncertainties, coupled with the high proportion of vulnerability and mostly decreasing trends (Table 1.1) 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 345,000-390,000t 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 (Figure 1.2; 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.

1.3.1 Small-Scale Fisheries of the Southwestern Indian Ocean: 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 1.3), 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 1.3), 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 1.3), mostly targeting sharks and pelagic fishes.

Table 1.3 Marine small-scale fishery vessel, fisher and gear data for southwestern Indian Ocean nations. Data sources: (ESAP 2005; Herfaut 2006; MFR 2010; ZMLF 2010; Andriantsoa and Randriamiarisoa 2013; Chavance *et al.* 2014; de Graaf and Garibaldi 2014; KMALF 2014a, 2014b; Soilihi 2014; Chacate and Mutombene 2015; KMALF 2015; Ndegwa 2015; SFA 2015; Albion Fisheries Research Centre unpublished data; L'Institut Français de Recherche pour l'Exploitation de la Mer unpublished data; Instituto Nacional de Investigação Pesqueira personal communication).

Country	**Vessels (Year)**	**Fishers (Year)**	**Gear Prevalence**				
			Measure (Year)	**Handline**	**Longline**	**Gillnet**	**Other/ Unknown**
Comoros	3,601 (2012)	Unknown	Vessels (2012)	23.25% static 23.12% trolled	-	3.11% drift	50.51%
Kenya	2,913/3,500 (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	2,476 (2010)	2,038[c] (2014)	Fishers (2014)	66.00% line&trap 21.05% line/ harpoon/foot[4]	-	5.89% "large net" 0.49% "gillnet"	6.58%
Mayotte	1,132 (2014)	4,800 (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)	7,664 (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]9,609 vessels predicted by 2015

Assessing general landings trends in the SWIO small-scale fisheries is challenging. Whilst long-term data sets are available through the FAO (Figure 1.2) 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 (Figure 1.3; 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 (Figure 1.2). 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 transhipment 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 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 of standardised data and the relatively poor resolution of landings and effort data available. The variability in basic recording metrics (Table 1.4) 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

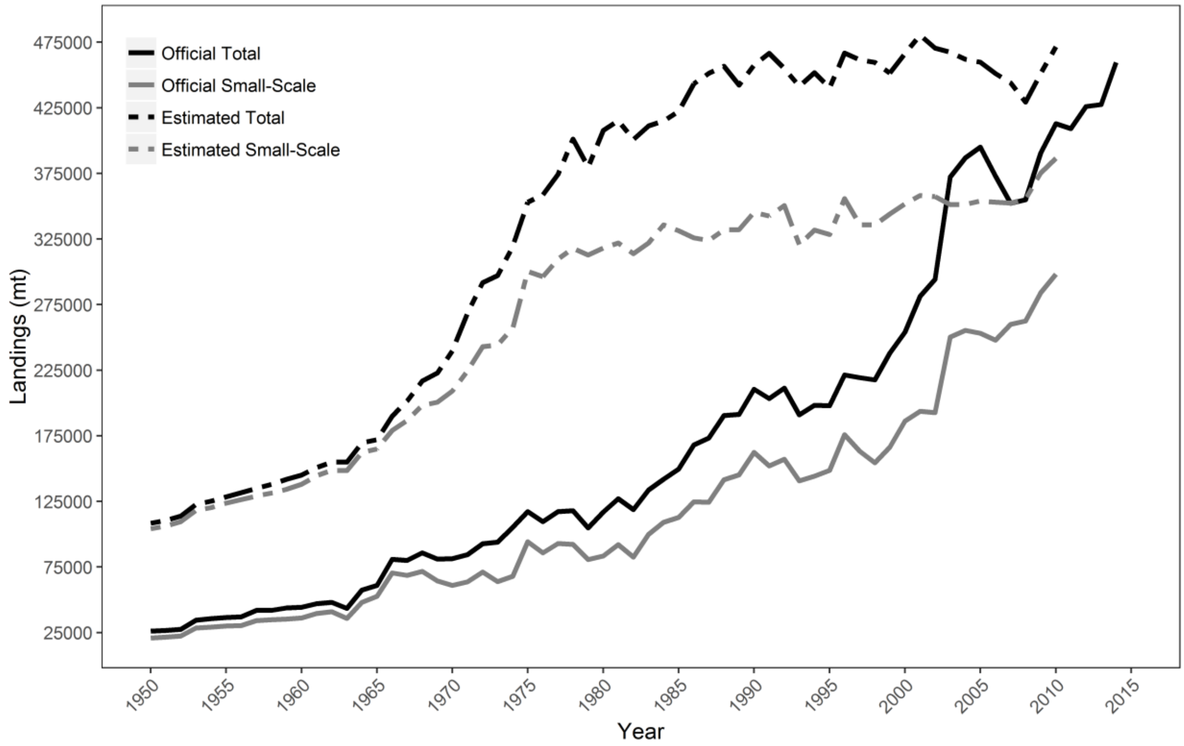

Figure 1.2 Official and estimated landings data for southwestern Indian Ocean nation's fisheries between 1950 and 2013. Data sources: (Pauly and Zeller 2015; FAO 2016b)

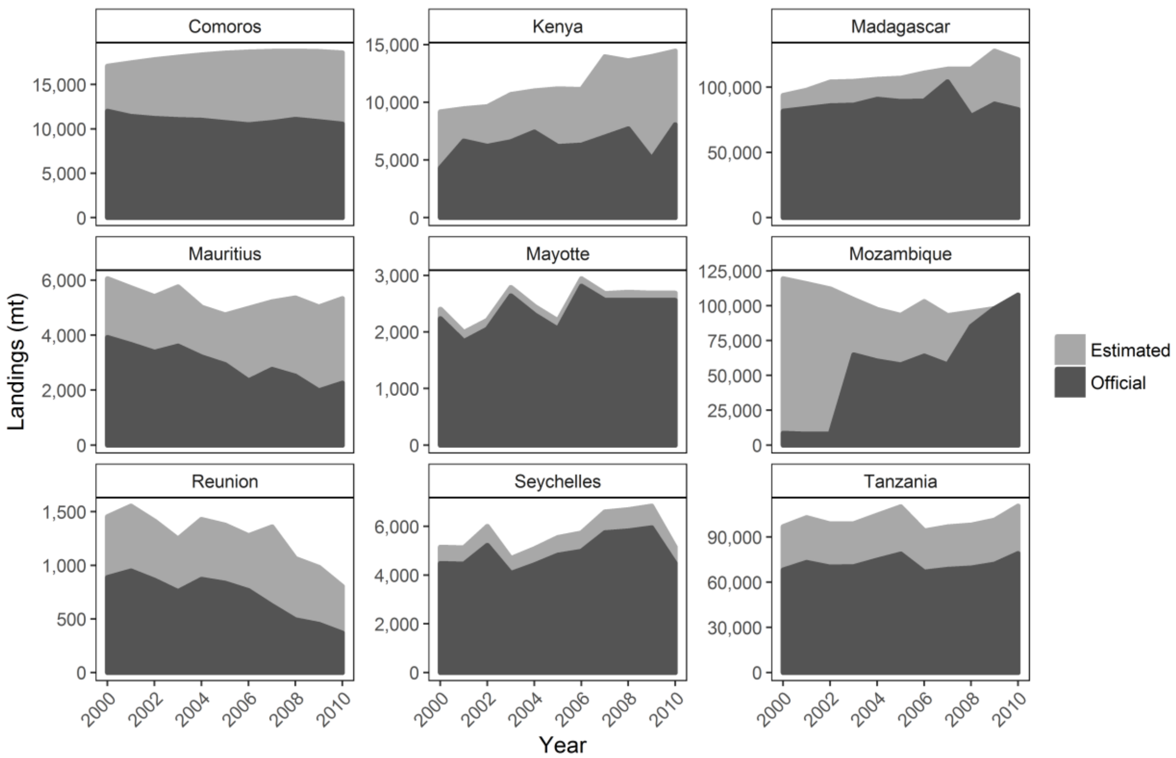

Figure 1.3 Southwestern Indian Ocean small-scale fisheries landings 2000-2010. Note variable scales on the y-axes. Data sources: (Pauly and Zeller 2015)

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 165t of all elasmobranchs landed from artisanal fisheries in its 2007 statistical bulletin, yet reports 174t 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.

Table 1.4 Inconsistency in data metrics used in the reporting of artisanal fisheries of southwestern Indian Ocean countries

Country	Effort Measures			Gear Composition Measure
	Vessel Numbers	Fisher Numbers	Fishing Effort	
Comoros	Total Vessel Count	None	Not Available	Vessels
Kenya	Total Vessel Count	Total Fisher Count	Not Available	Gears
Madagascar	Total Vessel Count	Total Fisher Count	Not Available	Gears
Mauritius	Total Vessel Count	Total Fisher Count	Fisher-Days	Fishers
Mayotte	Total Vessel Count	Total Fisher Count	Active Vessels	Vessels
Mozambique	Total Vessel Count	Total Fisher Count	Active Vessels and Fishing Gears	Gears
La Réunion	Active Vessels	Total Fisher Count	Active Vessels	Vessels
Seychelles	Average Active Vessels/Month	Total Fisher Count	Line = Fisher-Days, Net = Sets	Vessels
Tanzania (mainland)	Total Vessel Count	Total Fisher Count	Not Available	Gears
Zanzibar	Total Vessel Count	Total Fisher Count	None (Trips recorded, not compiled)	Gears

1.4 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 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 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 (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

1.5 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 0t/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 1.5), 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).

1.6 Elasmobranch Interaction with Small-Scale Fisheries

In 2014, 34 countries across all fisheries scales reported 105,969t of elasmobranch landings originating from the western Indian Ocean region, of which only 17,663t were landed by SWIO nations (Figure 1.4; 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 1.3). It is therefore unlikely that the reporting reflects the true proportional contribution of SWIO nations. The regional estimate is likely an underestimate, resulting from under-reporting of landings, illegal fishing and discards, and is consistent with the under-reporting of other landings in the SWIO region

Table 1.5 Existing numerical data for sea turtle catch in the small-scale fisheries of the southwestern Indian Ocean. Data sources: (Rakotonirina and Cooke 1994; Okemwa *et al.* 2004; Muir 2005; Muir and Ngatunga 2007; Pusineri and Quillard 2008; Humber *et al.* 2011; Kiszka 2012)

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	2006/7	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	1,000-2,000/year	Unknown
	National – Incidental Only		Unknown	617-6170/year	Interview Survey

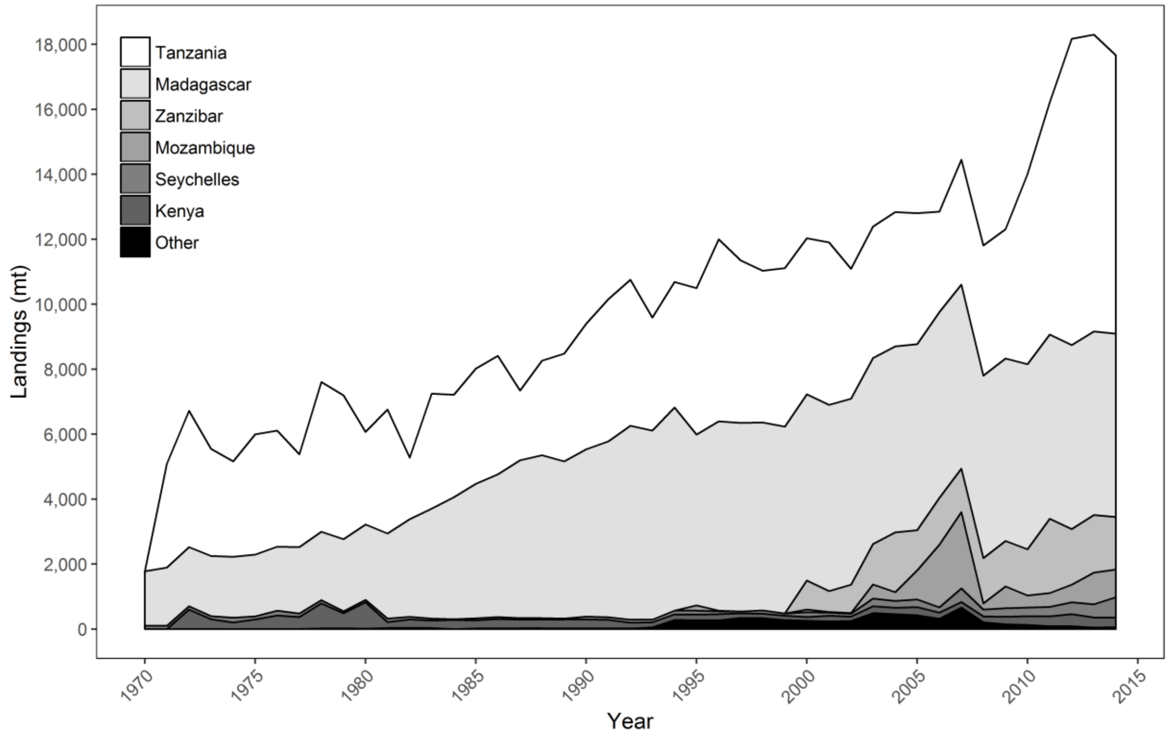

Figure 1.4 Total landings data for elasmobranchs caught by southwestern Indian Ocean nations 1970-2013. Zanzibar and Tanzania are reported separately after 2000. Data source: (FAO 2016b)

(Pauly and Zeller 2015). Despite the high level and year-on-year increase in landings across the SWIO (Figure 1.4) 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 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 1.6). 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:

1.6.1 Comoros

Since first being reported in 1994, elasmobranch landings in the Comoros have dwindled from 230t to 19t 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 19.97t landings in 2012 (Soilihi 2014). Other commonly reported catch includes a variety of oceanic and coastal species;

Table 1.6 Chondrichthyan species common amongst the reported small-scale fisheries landings in the southwestern Indian Ocean, La Réunion has been omitted as no data is available. Data sources: (Barnett 1997; Cooke 1997; Marshall 1997a; Sousa *et al.* 1997; Smale 1998; Nevill *et al.* 2007; Maoulida *et al.* 2009; Kiszka *et al.* 2010; Kiszka 2012; Andriantsoa and Randriamiarisoa 2013; Robinson and Sauer 2013; Poonian 2015; Instituto Nacional de Investigação Pesqueira unpublished data)

Common Name	**Scientific Name**	**Country**							
		Comoros	**Kenya**	**Madagascar**	**Mauritius**	**Mayotte**	**Mozambique**	**Seychelles**	**Tanzania**
Sharks									
Silvertip	*Carcharhinus albimarginatus*			x			x	x	x
Grey reef	*Carcharhinus amblyrhynchus*	x	x	x	x	x		x	x
Spinner	*Carcharhinus brevipinna*			x			x	x	
Silky	*Carcharhinus falciformis*		x	x	x	x	x	x	x
Bull	*Carcharhinus leucas*			x	x		x	x	x
Oceanic whitetip	*Carcharhinus longimanus*	x		x	x	x	x	x	
Blacktip reef	*Carcharhinus melanopterus*		x	x			x	x	x
Sandbar	*Carcharhinus plumbeus*						x	x	x
Spottail	*Carcharhinus sorrah*			x			x	x	x
Tiger	*Galeocerdo cuvier*		x	x	x		x	x	x
Shortfin mako	*Isurus oxyrinchus*		x	x	x	x	x	x	x
Slit-eye	*Loxodon macrorhinus*			x	x		x	x	
Tawny nurse	*Nebrius ferrugineus*		x	x				x	
Blue	*Prionace glauca*		x	x	x	x	x	x	
Whale	*Rhincodon typus*		x	x				x	x
Milk	*Rhizoprionodon acutus*			x			x	x	x
Hammerhead	*Sphyrna* spp.	x	x		x		x		x
Scalloped hammerhead	*Sphyrna lewini*		x	x		x	x	x	x
Great hammerhead	*Sphyrna mokarran*		x	x			x	x	x
Smooth hammerhead	*Sphyrna zygaena*						x	x	x
Whitetip reef	*Triaenodon obesus*		x	x	x		x	x	x

Table 8.6 ctd Chondrichthyan species common amongst the reported small-scale fisheries landings in the SWIO, La Réunion has been omitted as no data is available. Data sources: (Barnett 1997; Cooke 1997; Marshall 1997a; Sousa *et al.* 1997; Smale 1998; Nevill *et al.* 2007; Maoulida *et al.* 2009; Kiszka *et al.* 2010; Kiszka 2012; Andriantsoa and Randriamiarisoa 2013; Robinson and Sauer 2013; Poonian 2015; Instituto Nacional de Investigação Pesqueira unpublished data)

Common Name	**Scientific Name**	**Country**							
		Comoros	**Kenya**	**Madagascar**	**Mauritius**	**Mayotte**	**Mozambique**	**Seychelles**	**Tanzania**
Rays									
Spotted eagle	*Aetobatus* cf. *ocellatus*		x	x			x	x	x
Manta	*Mobula* spp.		x		x		x	x	x
Bluespotted maskray	*Neotrygon caeruleopunctata*		x				x	x	x
Sawfish	*Pristis* spp.			x			x	x	x
Bowmouth wedgefish	*Rhina ancylostoma*			x			x	x	x
Large wedgefish	*Rhynchobatus* spp.		x	x	x		x	x	x
Bluespotted fantail	*Taeniura lymma*		x	x			x	x	x

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 (775kg/vessel/year) and trolled (830kg/vessel/year) handlines than in net fisheries (2.20kg/vessel/year). Indeed, handlines account for nearly 96% of the reported blue shark landings. Given the seemingly greater CPUE for sharks in the handline fisheries specific scrutiny should be placed on these in any future assessments. Conversely, the lack of reporting of ray 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.

1.6.2 Kenya

Elasmobranch landings have fluctuated in recent years, dipping as low as 165t in 2007, peaking at 373t in 2012 and reported at 293t 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 are 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 rays 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).

1.6.3 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 ≥85% of the total shark catch (Le Manach et al. 2011; Randriamiarisoa and Rafomanana 2005) and likely for the vast majority of ray 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 5,650t in 2014 (FAO 2016b), yet estimates suggest small-scale fisheries landings alone are in the region of 7,500t/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 ray 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).

1.6.4 Mauritius

Mauritius reported artisanal landings of 0.456t 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 6,000 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 16.725t 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 disproportionally 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 rays in particular. Hammerhead and tiger sharks are by far the most common species reported (Poonian 2015). Further, *Mobula* spp. were the only ray reported in the catch. Ease of identification for these species likely heavily biases interview responses and so inevitably overestimates their importance in the fisheries.

1.6.5 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 ray 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 1.3) 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 (880t landed in 2014). This suggests that the reported landings of elasmobranchs (11t in 2014), are a significant underestimate of the true catch, perhaps by 75% or more.

1.6.6 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 rays 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 854t 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 200t (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 534.1t, yet in 2012 and 2014 only 142.6t and 141.1t were reported, respectively. In addition, the only known estimate of shark catch in the small-scale fisheries was 2,186t 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.

1.6.7 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.

1.6.8 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 (Figure 1.5). 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).

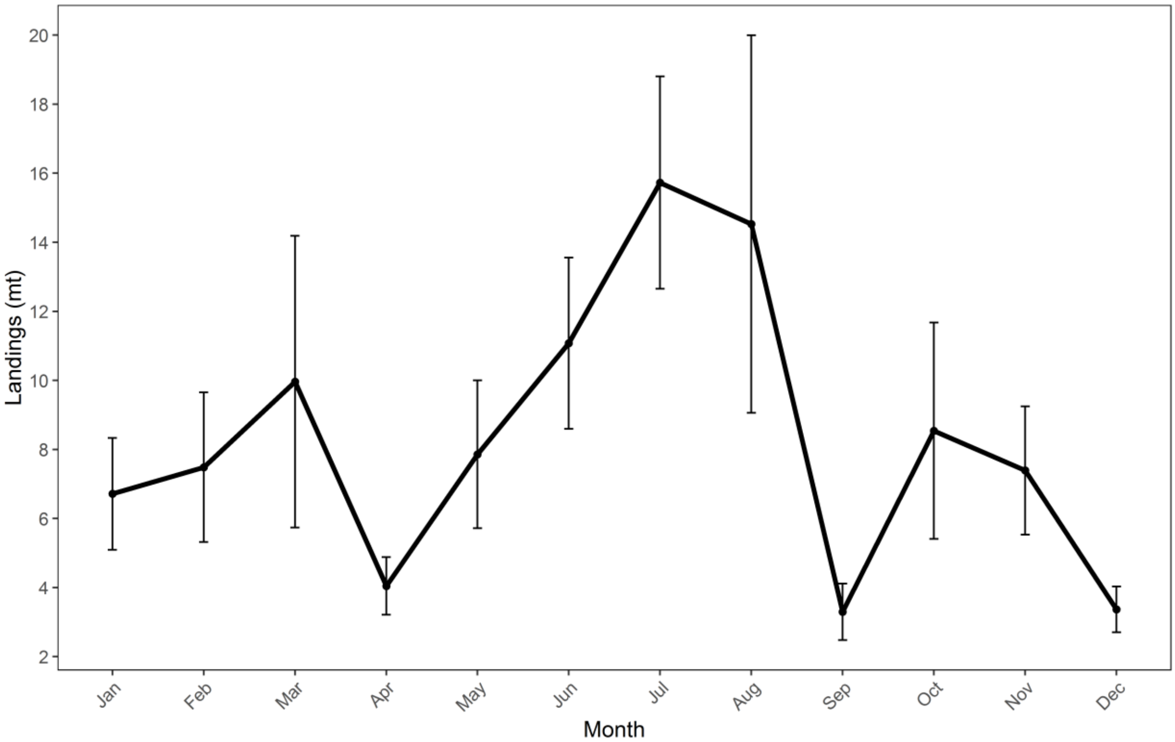

Figure 1.5 Official mean (±S.E) small-scale fisheries landings of elasmobranchs in the Seychelles, based on 2001-2009,2013 and 2014 data. Data sources: (SFA 2001-2013, 2015)

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.

1.6.9 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 (Bamett 1997). These are predominantly requiem sharks, both oceanic and coastal (including coral reef associated), alongside hammerhead, milk (*Rhizoprionodon acutus*) and whitetip reef sharks. Large wedgefish (*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 wedgefish including bottlenose and/or whitespotted and bowmouth (*Rhina ancylostoma*) and Zanzibar guitarfish (*Acroteriobatus zanzibarensis*) also present in relatively high numbers (Schaeffer 2004). Other rays were not recorded, however it is known that various Dasyatidae species dominate the ray 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 (Bamett 1997). In Zanzibar catches of sharks appear seasonal, being highest during the north-east monsoon, particularly between January and May (Bamett 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.

1.6.10 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 (Table 1.7). 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

Table 1.7 Elasmobranch meat and fin sale values from small-scale fisheries and fin export volume from southwestern Indian Ocean nations. Data sources: (Schaeffer 2004; Maoulida *et al.* 2009; MFR 2012; KMALF 2014a; Cripps *et al.* 2015; Kimani *et al.* 2015; TMLFD 2015; Kenya State Department of Fisheries unpublished data; Instituto Nacional de Investigação Pesqueira unpublished data)

Country	Whole Animal Value (USD/kg)	Meat Value (USD/kg)		Fin Value (USD/kg)	Fishery Value (USD) (Year)	Fin Export Volume (t) (Year)
		Fresh	Dried			
Comoros	-	0.5-2	5-6	40-100	-	-
Kenya	1.46	-	-	6.57	457,369 (2013)	6.29 (2014)
Madagascar	-	0.19-0.34	0.45-0.54	24.59-79.36 (shark) 30.17-136.63 (wedgefish)	-	2 (2013)
Mauritius	1.86	-	-	-	-	-
Mayotte	-	-	-	-	-	-
Mozambique	-	0.8-1.31	-	-	-	-
La Réunion	-	-	-	-	-	-
Seychelles	-	-	-	-	-	11 (2013)
Tanzania (mainland)	1.83	-	-	-	7.15 million (2014)	-
Zanzibar	-	0.27-1.28	-	2.24-9.38	-	-

(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 (Table 1.7) in comparison with teleosts (e.g. MFR 2004; MFR 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.

1.7 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.

1.7.1 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 (Table 1.8). 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 1.6; 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, unreported and unregulated (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

Table 1.8 International and national agreements with which southwestern Indian Ocean nations are associated. National Plan of Action (NPOA), CMS (Convention of Migratory Species) MoUs (Memorandum of Understanding), Nairobi Convention (NC), United Nations Fisheries Stock Agreement (UNFSA), Port State Measures Agreement (PSMA). Data sources: (UNEP 2010; CITES 2016; CMS 2016a, 2016c, 2016b; FAO 2016a; IOTC 2016; UN 2016)

Country	NPOA		CMS MoUs			NC	UNFSA	PSMA	CITES
	Sharks	Turtles	IOSEA Turtles	Dugong	Sharks				
Comoros	×	✓ (not implemented)	✓ (2001)	✓ (2008)	✓ (2014)	✓	×	×	✓ (Accession 1994)
Kenya	In Progress	×	✓ (2002)	✓ (2008)	✓ (2010)	✓	✓ (Accession 2004)	✓ (Signatory 2010)	✓ (Ratification 1978)
Madagascar	×	×	✓ (2003)	✓ (2007)	Range State	✓	×	×	✓ (Ratification 1975)
Mauritius	✓ (2015)	×	✓ (2004)	Range State	Range State	✓	✓ (Accession 1997)	✓ (Accession 2015)	✓ (Ratification 1975)
Mayotte (France)	✓ (2009)	✓ (2015)	✓ (2008)	✓ (2007)	Range State	✓	✓ (Ratification 2003)	✓ (Acceptance 2016)	✓ (Approval 1978)
Mozambique	In Progress	In Progress	✓ (2008)	✓ (2011)	Range State	✓	✓ (Accession 2008)	✓ (Ratification 2014)	✓ (Accession 1981)
La Réunion (France)	✓ (2009)	✓ (2015)	✓ (2008)	×	Range State	✓	✓ (Ratification 2003)	✓ (Acceptance 2016)	✓ (Approval 1978)
Seychelles	✓ (2007)	×	✓ (2003)	✓ (2010)	Range State	✓	✓ (Ratification 1998)	✓ (Accession 2013)	✓ (Accession 1977)
Tanzania	In Progress	×	✓ (2001)	✓ (2007)	Range State	✓	×	×	✓ (Ratification 1979)

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, 2016b). Conversely, few SWIO nations have signed the CMS MoU for sharks (Table 1.8), 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 to 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.

1.7.2 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 1.9). 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 1.9) and where they do exist their effectiveness is often questioned. For example, as of 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 1.8), with Kenya and Mozambique projecting completion by the end of 2017, 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.

1.7.3 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

Table 1.9 Legal status and related punishments regarding vulnerable marine megafauna in the small-scale fisheries of the southwestern Indian Ocean

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

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 driftnets (defined there as a net exceeding 250m 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 250m in length) for five 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.

1.8 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 1.6); 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 1.3; Table 1.4). 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 (Table 1.4), 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 1.3). 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; Bamett 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 ray 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 widespread issues (Hauck 2008; Keane et al. 2008; Raakjær 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 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.

1.9 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.

Chapter 2. Marine Megafauna Catch in Southwestern Indian Ocean Small-Scale Fisheries from Landings Data

2.1 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 rays), drift gillnets (marine mammals, rays and sharks) and longlines (sharks). Overall, catch was dominated by small and moderately sized coastal requiem sharks (Carcharhiniformes) and whiprays (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.

2.1.1 Key Words

Elasmobranch; Marine mammal; Sea Turtle; Bycatch; Small-scale fishery; SWIO

2.2 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 rays), marine mammals and sea turtles (e.g. Dulvy et al. 2014; Lewison et al. 2004; Wallace et al. 2010; Žydelis 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 <10m sailing or outboard powered vessels. SSF account for greater than 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. Aragones 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 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 freshwater) 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 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-consumptive 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 underreporting 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 <10m 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.

2.3 Methods

2.3.1 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 2.1) 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 hours 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,

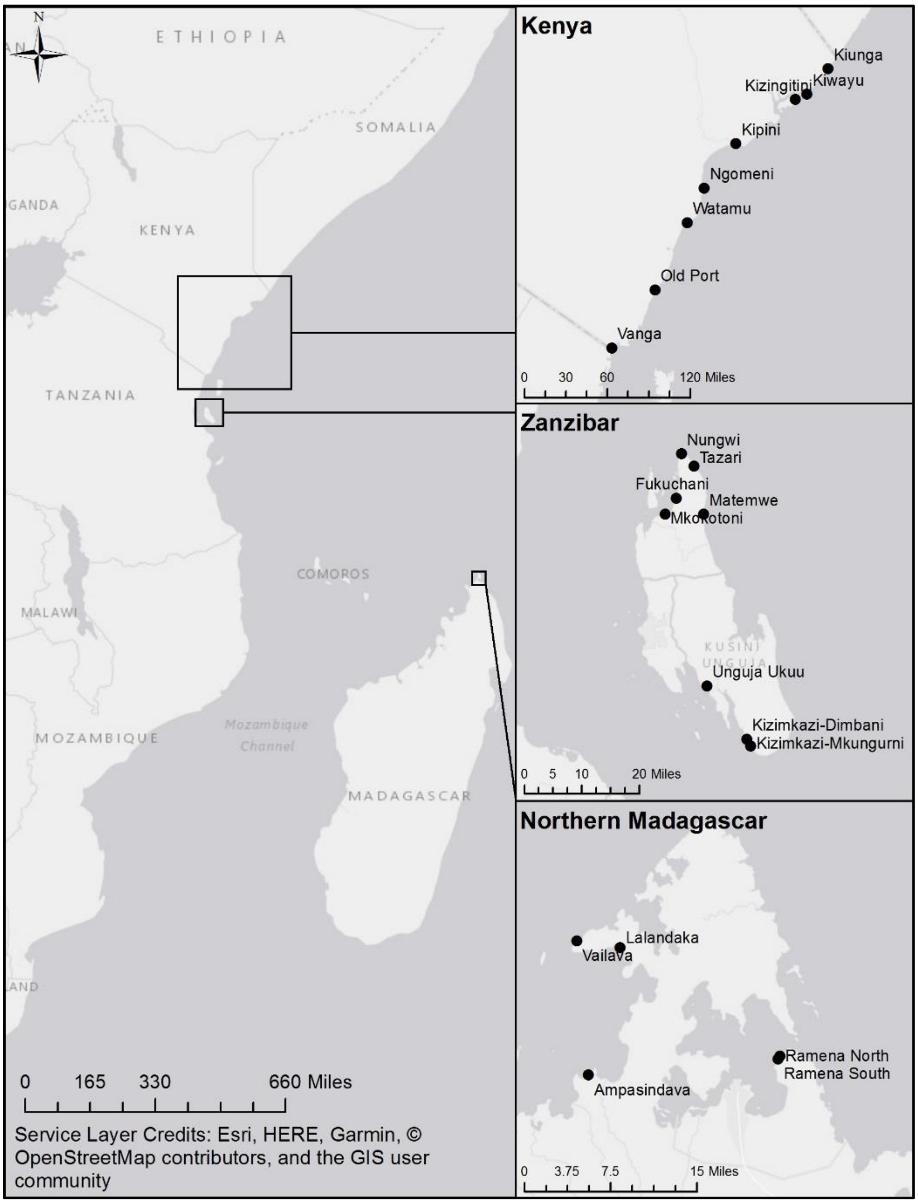

Figure 2.1 Locations of landing sites in the southwestern Indian Ocean monitored for marine megafauna catch between June 2016 and June 2017.

that were not recorded by the landings observer on any given day. Fishers' declarations indicate the potential magnitude of catch underestimate in the study.

2.3.2 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 co-linearity ($r > 0.2$ 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 rays 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 rays. 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 both the size of animals that could be successfully weighed (e.g. larger animals are more difficult to weigh). In order to address this, a weight-fork length and weight-disc width relationships were modelled for sharks and rays respectively. For the purpose of estimating weights, guitarfish and wedgefish (Rhyncobatidae) 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 log 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 (Figure 2.2). 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.

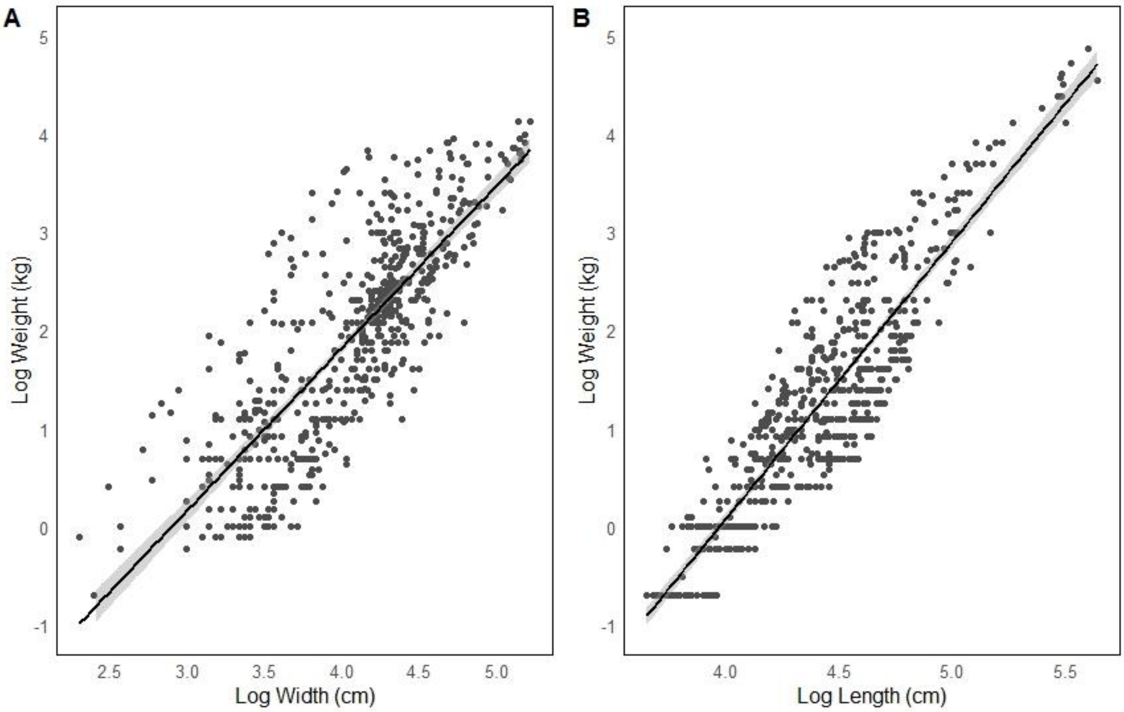

Figure 2.2 Linear model relationships between the natural log (logN) of weight and the natural log of disc width and fork length for A) rays and B) sharks, guitarfish and wedgefish caught in small-scale fisheries of Kenya, Zanzibar and northern Madagascar between June 2016 and June 2017.

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 2.1) 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 (±95% CI) values for productivity and susceptibility as well as confidence in

Table 2.1 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		

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, e.g. maximum size (Juan-Jordá et al. 2015). Thus, direct comparisons should be made between taxa 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).

2.4 Results

2.4.1 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 (χ^2=1453.10, 42.45, 161.48, 380.62, respectively; $p<0.05$). Tidal height significantly influenced driftnet, handline and longline fishing effort (χ^2=41.65, 33.12, 9.76, respectively, $p<0.05$). Wind direction significantly influenced driftnet and handline fishing effort (χ^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 75.0%, 54.2%, 82.2% and 54.9% of the deviance for bottom-set net, driftnet, handline and longline fishery effort, respectively (Figure 2.3).

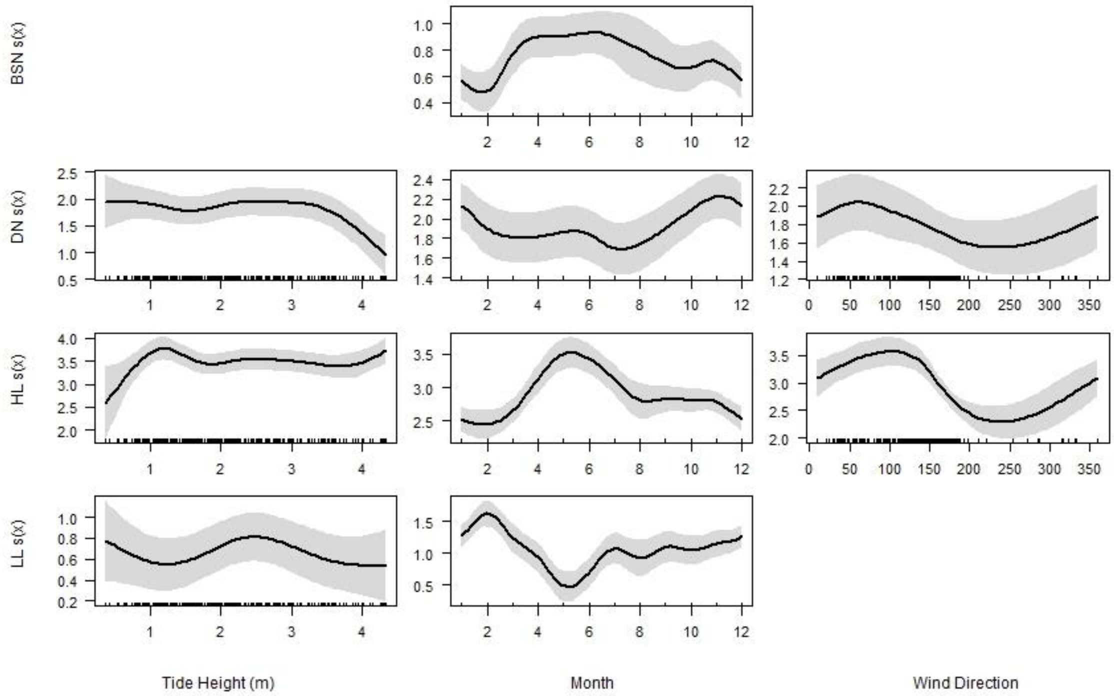

Figure 2.3 Smooths (±95% CI) from a series of generalised additive mixed models (GAMM) describing the cross-sectional relationship between tidal height, month and wind direction (degrees from North) with fishing effort of bottom-set gillnet (BSN), drift gillnets (DN), handline (HL) and longline (LL) small-scale fisheries across Kenya, Zanzibar and northern Madagascar between June 2016-June 2017.

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 driftnet 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 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 gear 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.

2.4.2 Fisheries Catch

Weighted mean CPUE and individual weight is presented by gear for sharks and rays for Kenya, northern Madagascar and Zanzibar (Table 2.2). Using weighted mean CPUEs combined with fishing effort, annual catch was estimated across gear types for Kenya and Zanzibar at the country level (Table 2.3). 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 2.3).

Table 2.2 Weighted mean catch per unit effort (CPUE) (±95% mean CI), with fishing trips as the unit of effort, and mean weight (±95% mean CI) in kilograms estimates for rays and sharks by gear type in Kenya, northern Madagascar and Zanzibar based on observed landings.

Country	Type	Bottom-set gillnets		Drift gillnets		Handlines		Longlines	
		CPUE	Weight	CPUE	Weight	CPUE	Weight	CPUE	Weight
Kenya	Rays	0.055 (0.050 - 0.062)	11.85 (10.04 - 13.66)	0.099 (0.052 - 0.167)	19.00 (16.37 - 21.63)	0.011 (0.011 - 0.012)	12.65 (10.94 - 14.36)	0.019 (0.018 - 0.021)	16.85 (11.73 - 21.96)
	Sharks	0.101 (0.078 - 0.124)	7.28 (6.96 - 7.75)	0.157 (0.023 - 0.291)	6.02 (4.83 - 7.21)	0.029 (0.027 - 0.032)	16.92 (13.36 - 20.48)	0.154 (0.122 - 0.187)	14.05 (12.97 - 15.12)
Northern Madagascar	Rays	0.008 (0.008 - 0.008)	10.62 (5.54 - 15.69)	1.400 (0.259 - 2.565)	8.95 (7.90 - 10.01)	0.072 (0.064 - 0.079)	17.57 (10.37 - 24.76)	0.023 (0.023 - 0.023)	7.86 (5.10 - 10.62)
	Sharks	0.007 (0.007 - 0.007)	4.43 (1.77 - 7.57)	0.549 (-1.300 - 2.398)	10.16 (-4.37 - 49.31)	0.282 (0.235 - 0.329)	12.54 (1.63 - 54.02)	0.050 (0.048 - 0.052)	6.29 (3.47 - 12.05)
Zanzibar	Rays	0.382 (0.361 - 0.414)	11.12 (9.94 - 12.31)	0.075 (0.016 - 0.160)	35.79 (31.04 - 40.54)	0.055 (0.050 - 0.064)	5.52 (3.65 - 7.39)	0.177 (0.147 - 0.220)	10.75 (9.06 - 12.45)
	Sharks	0.046 (0.043 - 0.049)	10.81 (6.76 - 14.86)	0.047 (0.046 - 0.047)	12.54 (9.92 - 15.16)	0.017 (0.017 - 0.018)	6.72 (2.03 - 11.42)	0.104 (0.088 - 0.119)	6.67 (4.27 - 9.08)

Table 2.3 Estimates (±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	Rays	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	Rays	134,384 (110,646 - 158,122)	1,512.5 (900.1 - 2,124.9)
	Sharks	52,575 (48,247 - 56,903)	414.9 (246.2 - 583.5)
	Elasmobranchs	186,959 (162,830 - 211,089)	1,927.4 (1,292.2 - 2,562.6)
SWIO	Rays	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)

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 rays and sharks respectively.

2.4.3 Species Composition and Vulnerability

Landings compositions (frequency of occurrence) for both rays and sharks are presented by country and gear type (Figure 2.4). 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 2.4).

Ray landings (Figure 2.4) 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 across the four gear types of interest, presumably reflecting higher abundance and availability in inshore waters relative to other rays. 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%) bentfin devilrays (*Mobula thurstoni*). These pelagic rays were most commonly caught in drift gillnets, particularly in Zanzibar at sites with access to adjacent deeper waters. Conversely, none of these pelagic rays 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

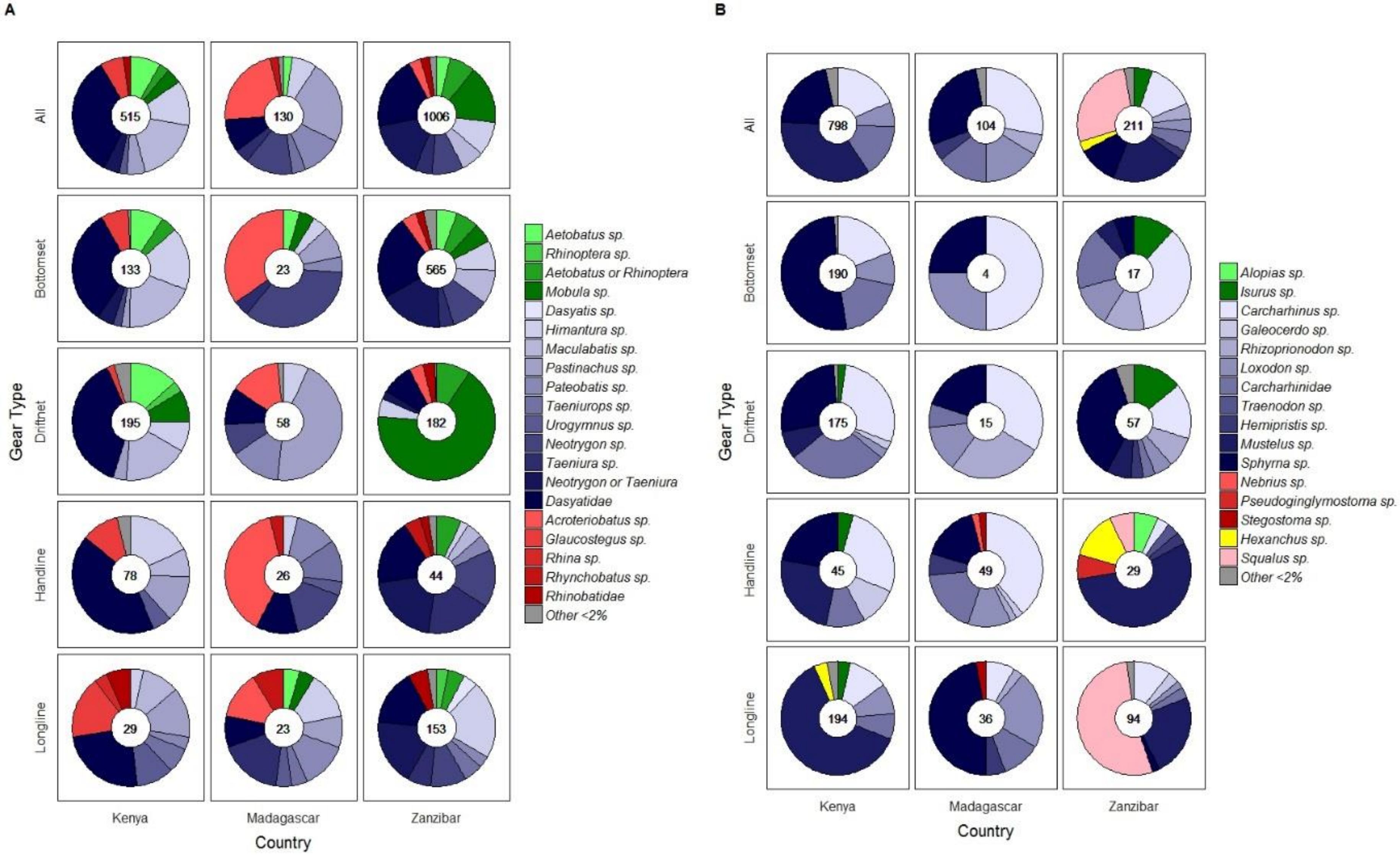

Figure 2.4 Elasmobranch frequency of occurrence at the genus and family level displayed by country and gear type for A) rays and B) sharks respectively. Sample size is displayed in the centre of each chart.

Table 2.4 Elasmobranch species caught in handline, longline, bottom-set and drift gillnet gears in Kenya, Zanzibar and northern Madagascar between June 2016-June 2017.

Common name	Scientific name	Kenya	Zanzibar	Madagascar
Marine Mammal				
Dugong	*Dugong dugon*	X		
Spinner Dolphin	*Stenella longirostris*	X		
Indo-Pacific Bottlenose Dolphin	*Tursiops aduncus*		X	
Sea Turtle				
Loggerhead Turtle	*Caretta caretta*	X		
Green Turtle	*Chelonia mydas*	X	X	
Hawksbill Turtle	*Eretmochelys imbricata*	X	X	
Rays				
Greyspot Guitarfish	*Acroteriobatus leucospilus*			X
Zanzibar Guitarfish	*Acroteriobatus zanzibarensis*		X	
Spotted Eagle Ray	*Aetobatus ocellatus*	X	X	X
Thorntail Ray	*Dasyatis thetidis*		X	
Halavi Guitarfish	*Glaucostegus halavi*	X		
Leopard/Coach Whipray	*Himantura* spp.	X	X	X
Baraka's Whipray	*Maculabatis ambigua*	X	X	X
Bigeye Stingray	*Megatrygon microps*		X	
Kuhl's Devilray	*Mobula kuhlii*	X	X	X
Giant Devilray	*Mobula mobular*		X	X
Bentfin Devilray	*Mobula thurstoni*		X	
Bluespotted Maskray	*Neotrygon caeruleopunctata*	X	X	X
Broad Cowtail Ray	*Pastinachus ater*	X	X	X
Pink Whipray	*Pateobatis fai*			X
Jenkins' Whipray	*Pateobatis jenkinsii*	X	X	
Sharkray	*Rhina ancylostoma*	X		
Shorttail Cownose Ray	*Rhinoptera jayakari*	X	X	
Bottlenose Wedgefish	*Rhynchobatus cf. australiae*	X	X	X
Whitespotted Wedgefish	*Rhynchobatus cf. djiddensis*			X
Bluespotted Fantail Ray	*Taeniura lymma*	X	X	X
Blotched Fantail Ray	*Taeniurops meyeni*	X	X	X
Procupine Whipray	*Urogymnus asperrimus*	X		
Mangrove Whipray	*Urogymnus granulatus*	X	X	
Sharks				
Pelagic Thresher	*Alopias pelagicus*		X	
Silvertip Shark	*Carcharhinus albimarginatus*	X	X	
Grey Reef Shark	*Carcharhinus amblyrhynchos*	X	X	X
Pigeye Shark	*Carcharhinus amboinensis*	X		
Silky Shark	*Carcharhinus falciformis*		X	X
Human's Whaler Shark	*Carcharhinus humanii*	X		

Table 9.4 ctd. Elasmobranch species caught in handline, longline, bottom-set and drift gillnet gears in Kenya, Zanzibar and northern Madagascar between June 2016-June 2017.

Common name	Scientific name	Kenya	Zanzibar	Madagascar
	Sharks			
Bull Shark	*Carcharhinus leucas*	X	X	X
Blacktip Shark	*Carcharhinus limbatus*	X		
Oceanic Whitetip Shark	*Carcharhinus longimanus*	X		
Hardnose Shark	*Carcharhinus macloti*	X		
Blacktip Reef Shark	*Carcharhinus melanopterus*	X	X	X
Dusky Shark	*Carcharhinus obscurus*		X	
Spottail Shark	*Carcharhinus sorrah*	X	X	X
White Shark	*Carcharodon carcharias*		X	
Tiger Shark	*Galeocerdo cuvier*	X	X	X
Snaggletooth Shark	*Hemipristis elongata*	X	X	X
Bigeyed Sixgill Shark	*Hexanchus nakamurai*	X	X	
Mako Shark	*Isurus oxyrinchus*	X	X	
Sliteye Shark	*Loxodon macrorhinus*	X	X	X
Smoothound	*Mustelus* spp.	X	X	
Tawny Nurse Shark	*Nebrius ferrugineus*			X
Shorttail Nurse Shark	*Pseudoginglymostoma brevicaudatum*		X	
Whale Shark	*Rhincodon typus*	X		
Milk Shark	*Rhizoprionodon acutus*	X	X	X
Scalloped Hammerhead Shark	*Sphyrna lewini*	X	X	X
Smooth Hammerhead Shark	*Sphyrna zygaena*		X	
Spurdog	*Squalus* spp.	X	X	
Zebra Shark	*Stegostoma fasciatum*			X
Whitetip Reef Shark	*Traenodon obesus*		X	

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 (Figure 2.4) 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 macloti*), grey reef (*Carcharhinus amblyrhynchos*) 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 oxyrinchus*), silky (*Carcharhinus falciformis*), thresher (*Alopias* spp.) and bigeye sixgill (*Hexanchus nakamurai*) were recorded in relatively low numbers. Other landings of note included a 5.7m 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.

The PSA assessments (Figure 2.5) 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 rays to be: *M. thurstoni*, broad cowtail ray (*Pastinachus ater*), halavi guitarfish (*Glaucostegus halavi*), Zanzibar guitarfish (*Acroteriobatus zanzibarensis*) and *M. ambigua*; and the most vulnerable sharks to be *C. macloti*, blackspot (*Carcharhinus humanii*), *L. macrorhinus*, *Squalus* spp. and *C. sorrah*.

2.5 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 tonnes 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 tonnes (2006-16), and 109.4% higher than the 16,928 tonnes 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 17.3-33.9%). 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 tonnes (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 1631.6 tonnes (2006-16) (FAO 2018) which falls well within the 95% confidence intervals of the study estimate. Such

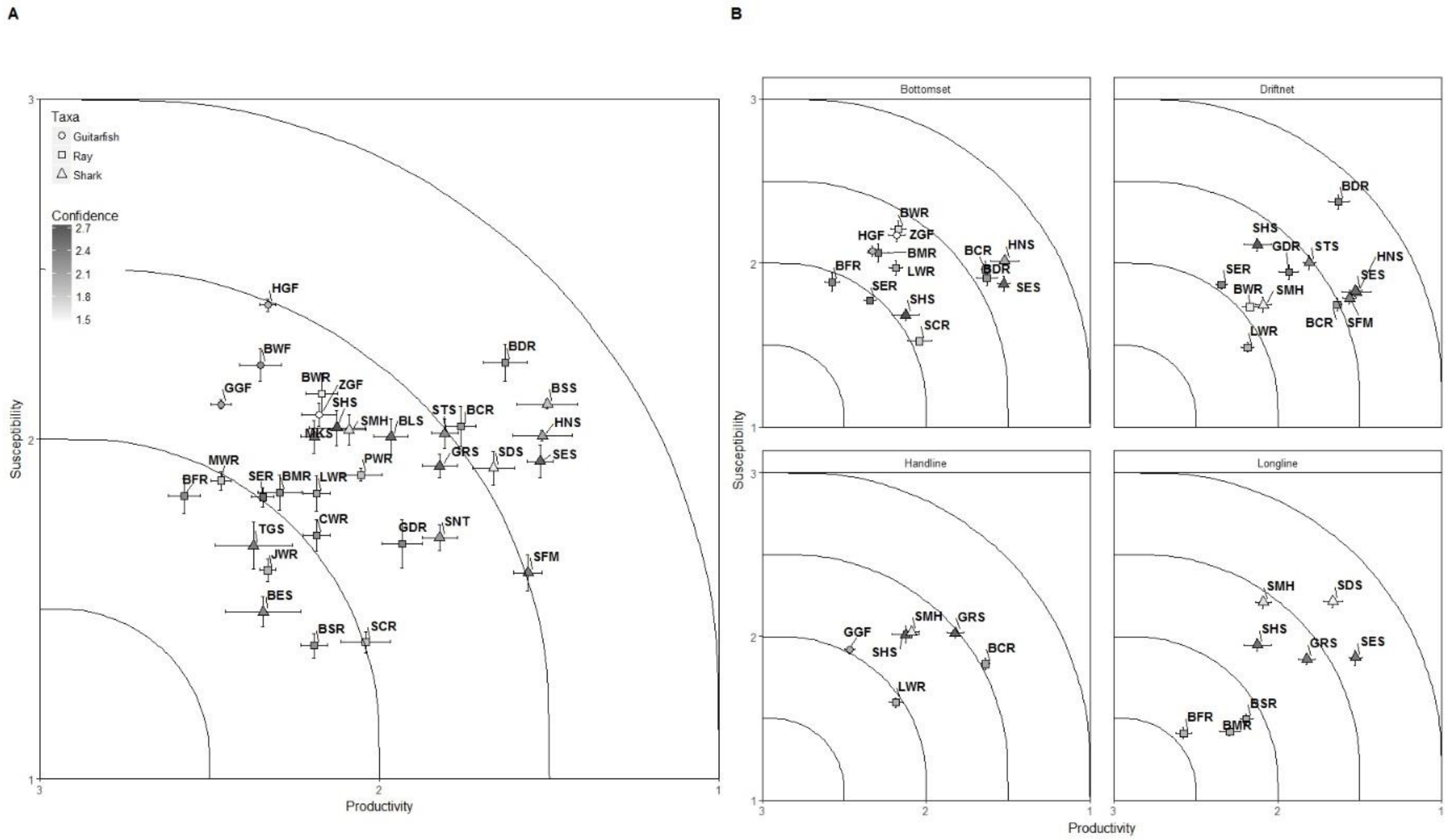

Figure 2.5 Productivity-Susceptibility Assessments for A) combined fishing gears, and B) individual fishing gears. Guitarfish: BWF, *Rhynchobatus australiae*; GGF, *Acroteriobatus leucospilus*; HGF, *Glaucostegus halavi*; ZGF, *Acroteriobatus zanzibarensis*. Rays: BWR, *Maculabatis ambigua*; BDR, *Mobula thurstoni*; BSR, *Taeniurops meyeni*; BFR, *Taeniura lymma*; BMR, *Neotrygon caeruleopunctata*; BCR, *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 sealei;* GRS, *Carcharhinus amblyrhynchos*; HNS, *Carcharhinus macloti*; MKS, *Rhizoprionodon acutus*; SHS, *Sphyrna lewini*; SFM, *Isurus oxyrinchus*; SES, *Loxodon macrorhinus*; SMH, *Mustelus sp.*; SNT, *Hemipristis elongata*; STS, *Carcharhinus sorrah*; SDS, *Squalus* spp.; TGS, *Galeocerdo cuvier*.

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. amblyrhynchos*, 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 ray compositional data often list *Mobula* spp., *Aetobatus* spp., large wedgefish (Rhinidae) 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. lymma*, *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), rays are a significant and sometimes dominant portion of the elasmobranch landings. Rays 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. Rays 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 ray 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 rays, 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, rays appear primarily threatened by bottom-set gillnets, which dominate the ray landings as a whole. Additionally, *M. thurstoni* was identified as the most vulnerable ray, threatened by both bottom-set and drift gillnets. This finding is of particular concern given that *Mobula* spp. are already thought to be in 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 *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 in 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.

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; Žydelis *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 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

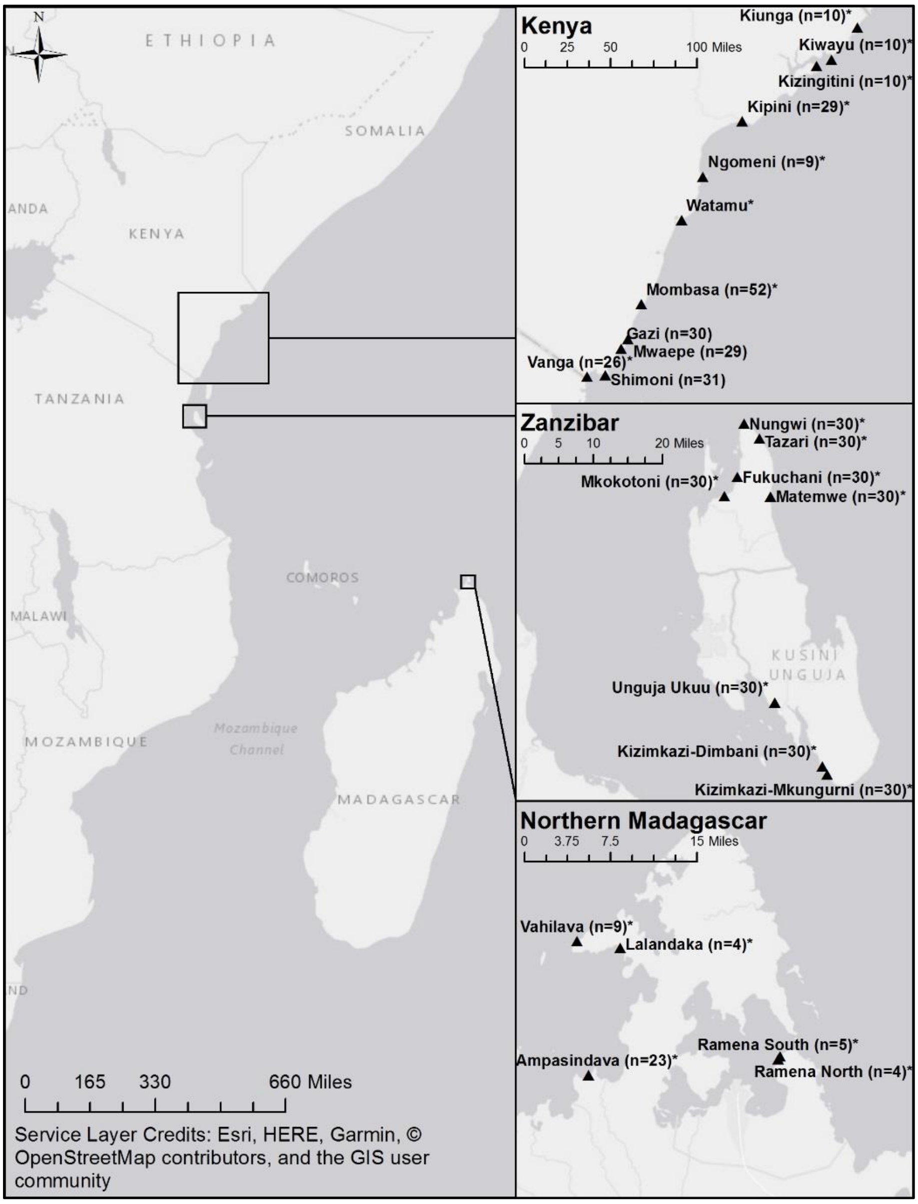

Figure 3.1 Locations of landing sites where surveys and landings observations were undertaken in the southwestern Indian Ocean, sites marked with (*) are those at which landings data was collected, "n=" values represent questionnaire survey sample size and location.

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

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.					

(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.

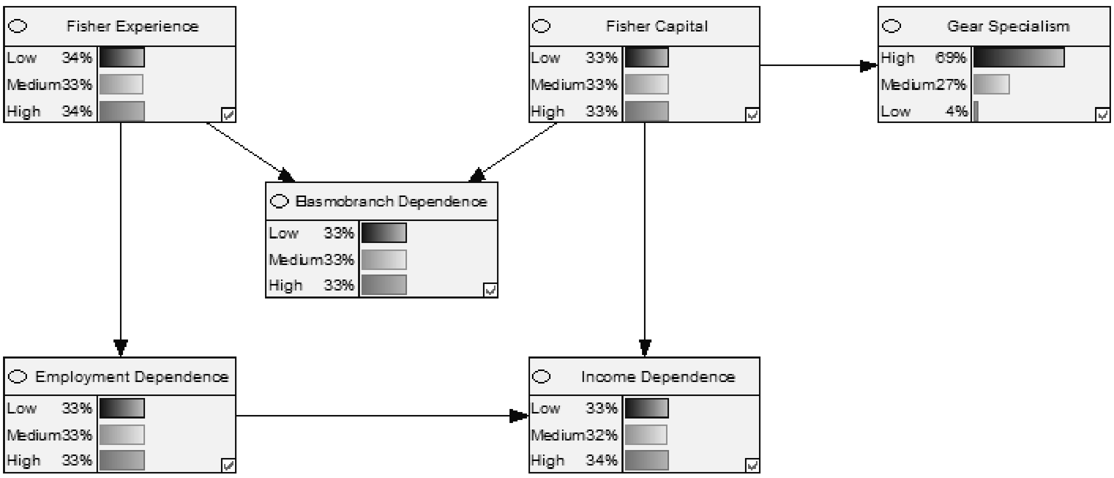

Figure 3.2 Final score-based Bayesian network structure, showing decision nodes and directed influence arcs. “Low”, “Medium” and “High” represent equal discretised quantiles of the underlying continuous factor values identified through exploratory factor analysis, with the exception of Gear Specialism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

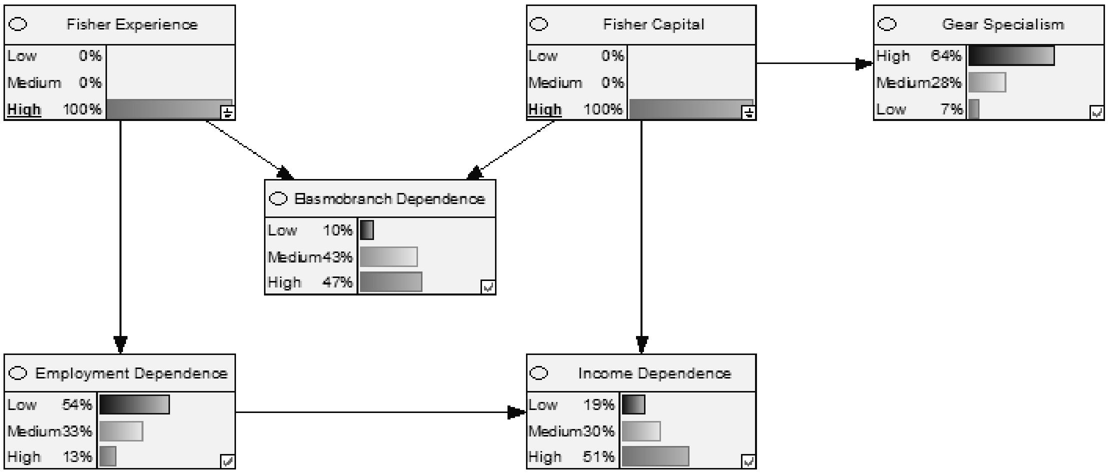

Figure 3.3 Final score-based Bayesian network structure, showing decision nodes and directed influence arcs, post-manipulation of fisher capital and experience factors to "High". "Low", "Medium" and "High" represent equal discretised quantiles of the underlying continuous factor values identified through 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 (χ^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 (χ^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.

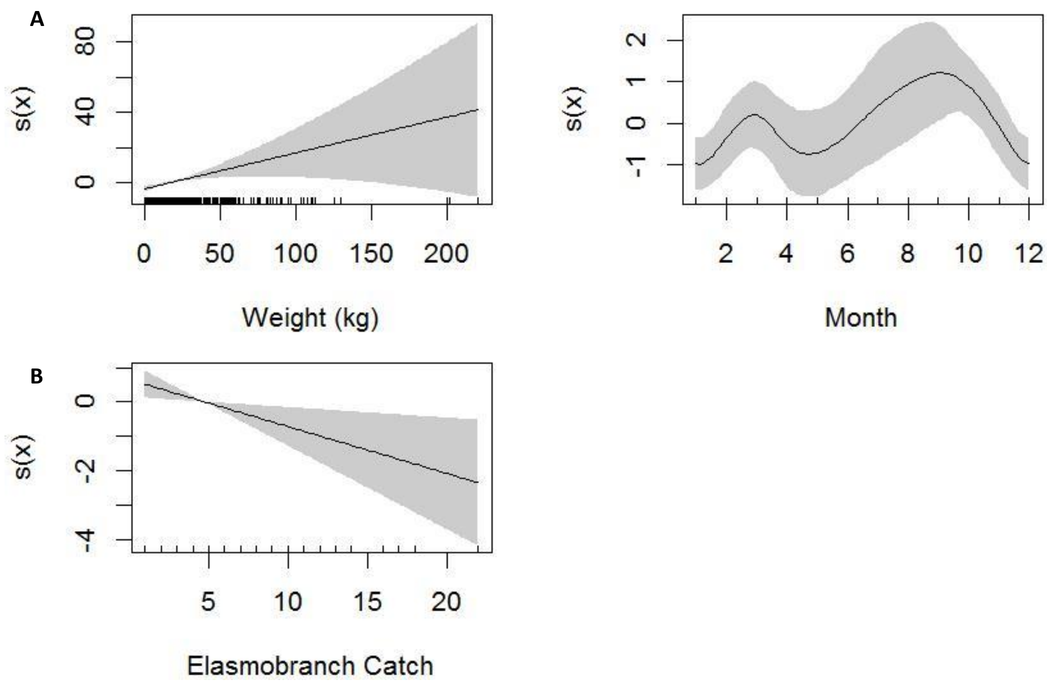

Figure 3.4 Smooths (±95% CI) from generalised additive mixed models (GAMM) describing significant relationships between A) weight and month on ray use, and B) daily elasmobranch catch on shark and shark-like ray use in small-scale fisheries across Kenya, Zanzibar and northern Madagascar between June 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 (χ^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 (χ^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 (χ^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/handing (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

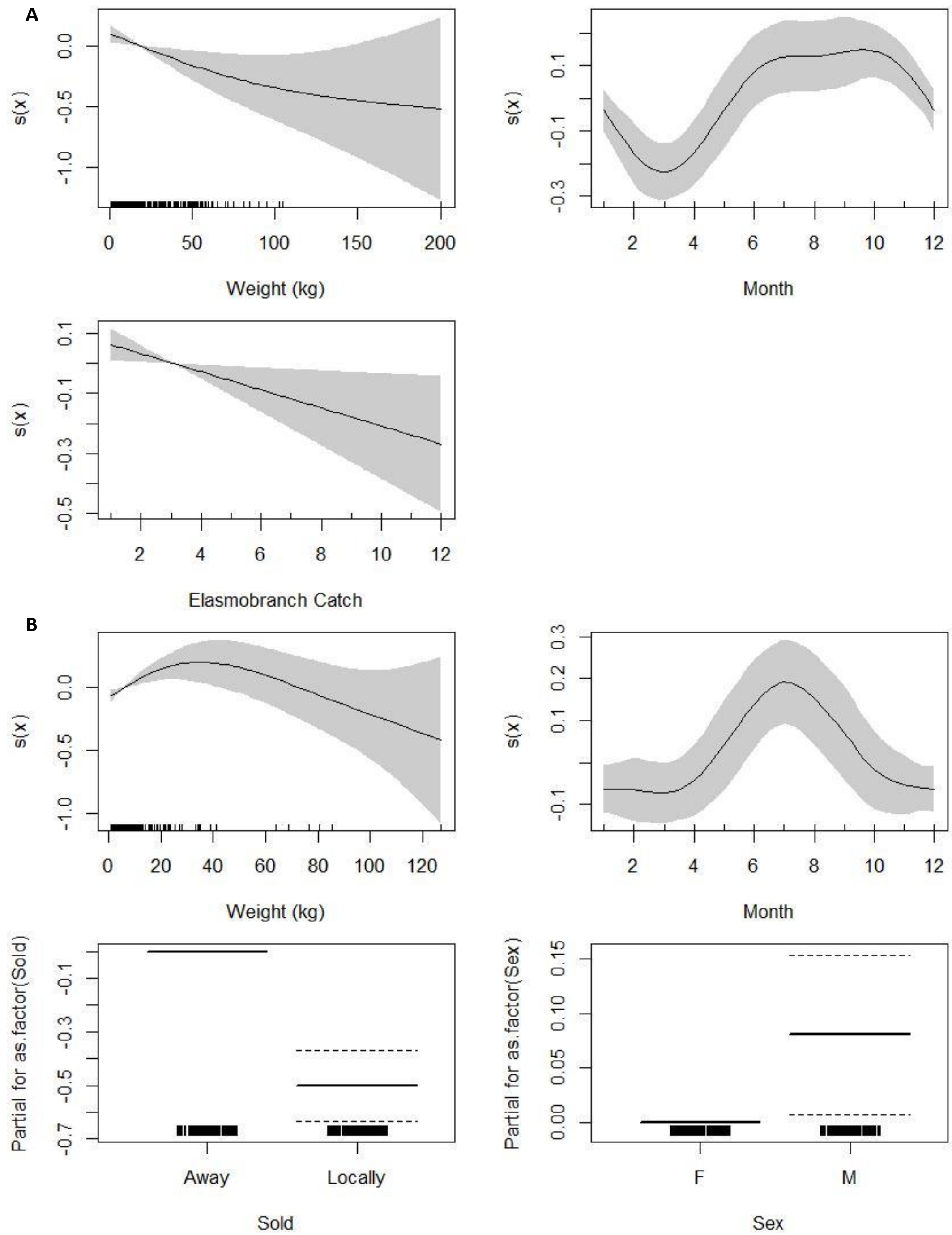

Figure 3.5 Smooths and partial smooths (±95% CI) from generalised additive mixed models (GAMM) describing significant relationships between A) weight, month and total daily elasmobranch catch for rays, and B) weight, month, sale location and sex for sharks and shark-like rays, in small-scale fisheries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

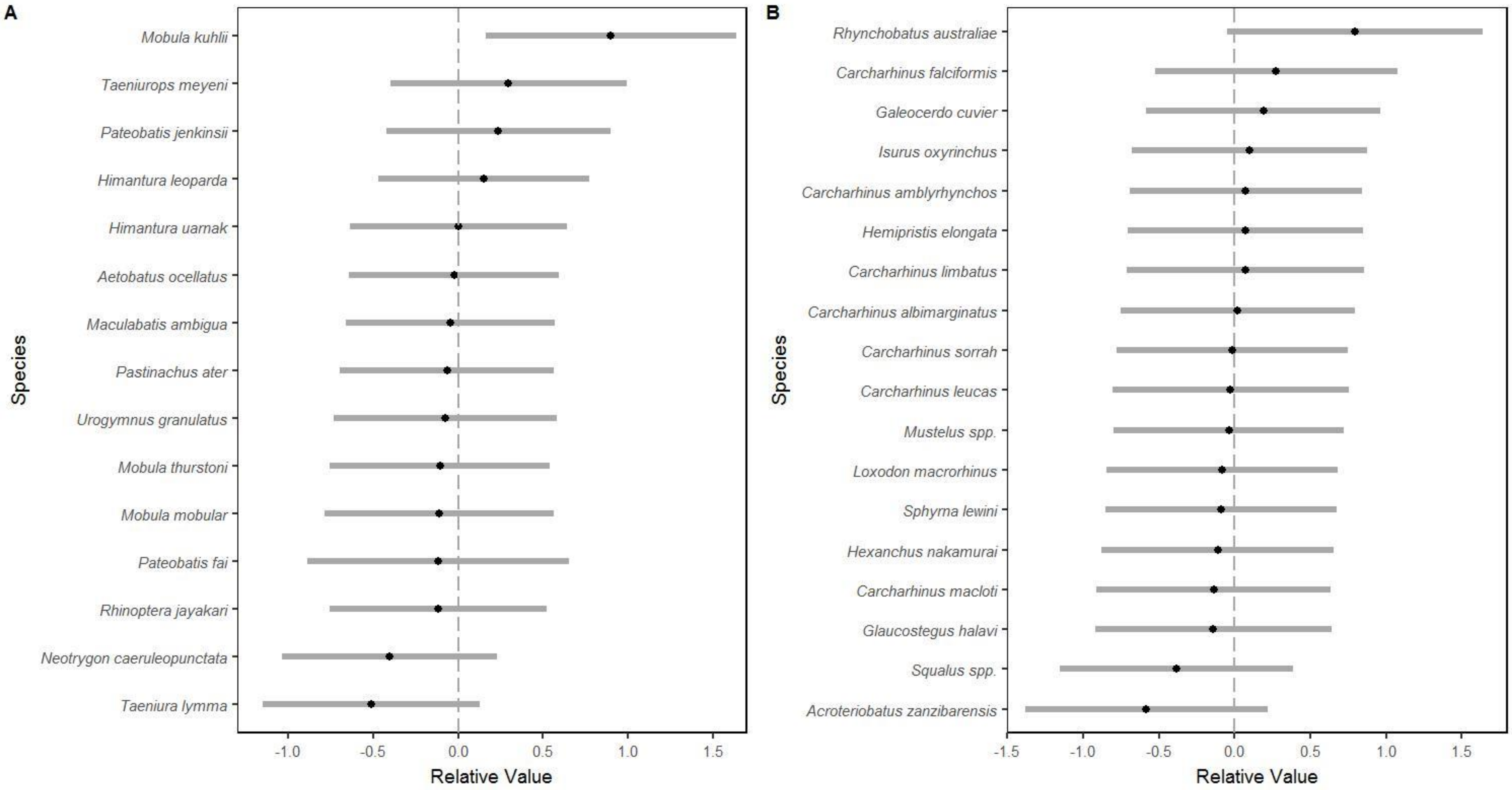

Figure 3.6 Comparisons of relative monetary (price per kg) value (mean ±95% CI) amongst species in theoretical space when known significant influences are controlled, for A) rays, and B) sharks and shark-like rays, in small-scale fisheries across Kenya, Zanzibar and northern Madagascar between June 2016-June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

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 (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 wedgefish 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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Chapter 4. Growth, Maturity and Annual Mortality of Baraka's Whipray (*Maculabatis ambigua*) from Small-Scale Fisheries

4.1 Abstract

The recently described Baraka's whipray (*Maculabatis ambigua*) is a common constituent of the catch in the small-scale fisheries (SSF) of the southwestern Indian Ocean and is the dominant ray in Kenyan SSF. Despite this nothing is known of its life-history. This study investigated life-history parameters of *M. ambigua* 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 disc width (DW)-weight, DW-age and male maturation models for the species, alongside estimates of longevity and indications of female maturation. No evidence was found for differences in growth patterns between males and females and thus male and female data were combined in the analyses. DW-weight (kg) relationship is estimated as logN(Weight) = 2.4912*logN(DW)-8.2875. The best-fit age model, the two parameter von Bertalanffy growth function, provides estimates for DW_{∞}=92.2(95%CI 82.4-112.3)cm, DW_0=33.6(95%CI 25.2-42.5)cm and k=0.234(95%CI 0.120-0.390). Male DW at 50% and 95% maturity was estimated at 56.9(95%CI 53.0-60.8)cm and 67.4(95%CI 53.5-72.4)cm, equivalent to age 2.2(95%CI 1.7-2.7) years and 3.7(95%CI 1.8-4.6) years, respectively. Further, the smallest mature female recorded (n=3) was DW 62cm, indicating a similar size class at maturity to males. Longevity was estimated at 20.7 years. The data indicate that *M. ambigua* is a moderately-sized, fast growing, early maturing species of whiptail stingray with a moderately long lifespan, indicating that the species may be relatively resilient to fisheries exploitation. However, there is no information on the species' fecundity and rebound potential. Further, construction of a Chapman-Robson catch curve, Z=0.373(95%CI 0.233-0.513), equivalent to annual mortality rates of 31.1(95%CI 20.8-40.1)%, indicate that *M. ambigua* is exploited across a wide age range (0-16years) with full recruitment to the fisheries occurring post-maturation. Exploitation across a wide age range with full recruitment to the fishers occurring post-maturation is an exploitation pattern is generally considered unsustainable for elasmobranch fishes and thus raises concern for the long-term survival of this species.

4.1.1 Key Words

Life-history; Dasyatidae; Indian Ocean; Elasmobranch; Ray; Stingray

4.2 Introduction

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. 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, the understanding of the scale and composition of species 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 across SWIO SSF, though there is substantial geographical heterogeneity (Temple *et al.* 2019). Despite this rays have thus far received little consideration in SWIO elasmobranch management formulation, perhaps a result of lower commercial demand (Chapter 3.) making them less visible to managers, an oversight jeopardising the long-term survival of these species.

Despite having only been recently described (Last *et al.* 2016) Baraka's whipray (*Maculabatis ambigua*) is the dominant ray in Kenyan SSF catch, and a common constituent of SSF catch in Zanzibar. They 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.* 2016). The species is believed to be distributed from Zanzibar to the Red Sea and possibly further into the northern Indian Ocean (Last *et al.* 2016). Little is known of the life-history of this species. In this study we begin to investigate aspects of the life-history of the species, including production of age-growth curves, maximum theoretical size, longevity, size at birth, maximum known size and initial estimations of size and age at maturity. Further, catch curves are used to assess current age class selectivity across SWIO SSF for the species and to estimate the rate of total mortality (Z). It is envisaged that these life-history data will serve to assist in future vulnerability, stock and demographic assessments of *M. ambigua* in the SWIO region.

4.3 Methods

4.3.1 Sample Collection

M. ambigua (n=47) were sampled with the consent of fishers and/or merchants from bottom-set gillnet catches at the village of Mkokotoni (n=30) and at a central Darajani Market (gear type used to capture specimens unknown), Stone Town, (n=17) in Zanzibar between 28/07/2015 and 19/08/2015 (Figure 4.1). Disc Width (DW) (cm), weight (kg) and sex (females=29, males=18) were recorded. In a number of cases (n=28) weight was not recorded as the specimen had already had been gutted and the majority of internal organs removed. Maturity status (immature/sub-adult or mature) was also recorded based on calcification of claspers in males (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 internal gonads. Vertebrae were extracted from the mid-disc for all specimens to be used for aging (n=47). All Vertebrae were stored frozen at -20˚C until sectioning.

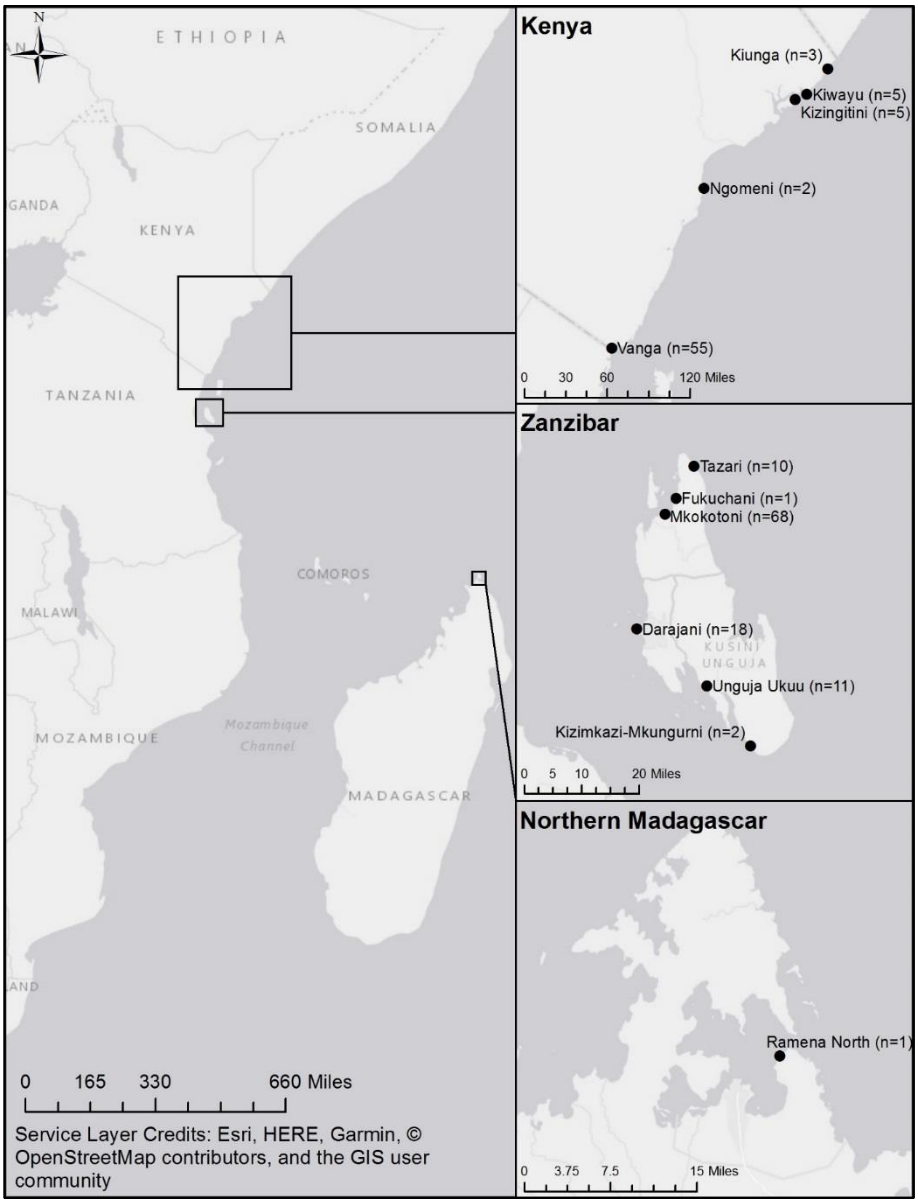

Figure 4.1 Locations of landing sites and markets in the southwestern Indian Ocean where Baraka's whipray (*Maculabatis ambigua*) were sampled between July 2015 and June 2017, sample sizes for those individuals recorded by fisheries observers is displayed.

Supplementary data for *M. ambigua* were recorded by trained fisheries observers during a 12-month landings monitoring programme between June 2016 and June 2017 at 21 sites across Kenya, Zanzibar and northern Madagascar (Temple *et al.* 2019). *M. ambigua* (n=134) were recorded in four sites in Kenya, five in Zanzibar and one in northern Madagascar (Figure 4.1). DW (n=127), weight (n=76), sex (females=55, males=71, unsexed=8) and maturity status for males (n=67) were recorded. Observers also recorded maturity for females opportunistically, females were categorised as either mature (n=3), based on the observed presence of developed eggs, or unclassified (n=52). These specimens were caught across a range of gear types (bottom-set gillnets = 80, drift gillnets=37, handlines=8 and longlines=6, unknown=3).

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

4.3.2 Disc Width and Weight Relationship

DW-weight relationships were modelled for males, females and combined data using linear regression after data were log-transformed. Evidence of sex differences in the relationship between DW and weight was investigated using a linear model with interaction terms. Cook's distance (4/n) was used to identify outliers, likely resulting from measurement and/or data entry errors, which exerted undue influence on linear model(s) for the disc width and weight relationship. Any identified outliers were removed (n=2) and the model(s) subsequently re-run.

4.3.3 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). Sections were subsequently photographed using a high quality digital macro-lens camera (Nikon SLR D7200) and image enhancement for growth band reading carried out in Adobe Photoshop CS3 (Campana 2014).

Ages (to the nearest 0.5 years) for each individual specimen were determined by examination of paired opaque and translucent banding in the *corpus calcareum*, with the half year determined as the presence of 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 4.2). Ages were estimated by the independent examination of both sagittal-sections (images were randomised before reading) from each of the two vertebrae sampled from each specimen by two independent readers. Mean age estimates for each specimen were generated for each reader. These mean estimates were then compared between readers, where they 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.* 2007; Jacobsen and Bennett 2011) in light of the restricted sample size. If differences in mean age estimates between readers were >1 year then 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 ±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 are presented, however these measures are commonly recognised as imperfect (Goldman 2005; Cailliet *et al.* 2006). The authors argue that the Bland-Altman approach (Bland and Altman 1999, 2003), designed primarily for method-comparison, provides improved quantification and visualisation of agreement, precision and bias among reads and readers compared with 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 of age reads 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 CI of the difference between readers. Thus, we apply and display the results of the Bland-Altman method as the primary measure or agreement, precision and bias.

4.3.4 Age-Growth, Longevity and Maturity

Age-DW data was fitted using three growth models (Table 4.1), the two parameter von Bertalanffy growth function (Von Bertalanffy 1938), the two parameter Gompertz growth function (Ricker 1975) and the Logistic growth function (Ricker 1979), using non-linear

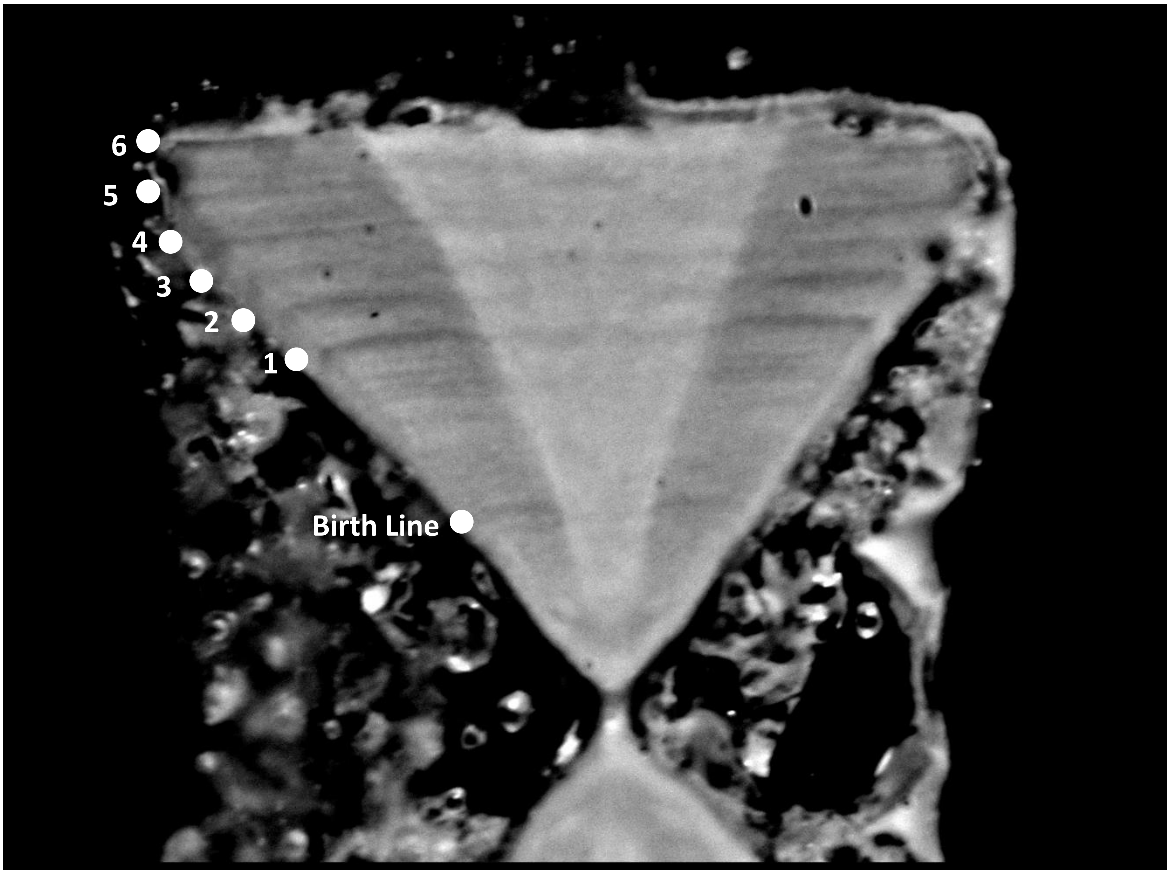

Figure 4.2 Photograph of sectioned vertebrae, with birth line and annuli marked, taken from a 63cm disc-width male Baraka's whipray (*Maculabatis ambigua*) captured in a bottom-set gillnet in August 2015. This individual was aged 6 years.

Table 4.1 Growth models used to fit disc-width at age data for Baraka's whipray (*Maculabatis ambigua*) and formulae for calculation of theoretical longevity. Parameters used are time (t) measured disc-width at known time (DW_t) infinite disc-width (DW_∞), disc-width at time zero (DW_0), growth constant (k), time at which absolute rate of disc-width increase begins to decrease – inflection point (α) and theoretical maximum age (t_{max}).

Function	Type	Equation
Three parameter von Bertalanaffy	Growth	$DW_t = DW_\infty-((DW_\infty-DW_0)*e^{-k*t})$
Two parameter Gompertz	Growth	$DW_t = DW_\infty*e^{-e(-k(t))}$
Logistic	Growth	$DW_t = DW_\infty/(1+exp^{-k(t-\alpha)})$
Age at 95% Disc Width	Longevity	$t_{max} = 5(log_n2)k^{-1}$
Age at 95% Disc Width	Longevity	$t_{max} = (log(1-0.95))k^{-1}$
Age at 99% Disc Width	Longevity	$t_{max} = 7(log_n2)k^{-1}$

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. Evidence for differences between male and female growth curves were investigated using likelihood ratio. Model selection was made through comparison of Akaike's Information Criterion (AIC) selecting for the lowest AIC value, residual differences between the model and data and realism in parameter estimates. The 95% mean CI for growth curves was derived via bootstrapping with 1000 iterations. Longevity was calculated using three growth constant parameter-based formulae (Table 4.1; Taylor 1958; Fabens 1965; Ricker 1979; Smith *et al.* 2007; Pierce and Bennett 2010).

DW (±95%CI) at 50% and 95% of males reach maturity (White 2007) were estimated from logistic regression with bootstrapped confidence intervals. Age (±95%CI) at which 50% and 95% of males are mature was estimated based on conversion of DW estimates for maturity using final growth models. Maturity was not estimated for females because the method of maturity determination cannot distinguish mature females without fertilised eggs from immature females.

4.3.5 Age Validation

Validation of growth band periodicity was not possible within this study. The short temporal period (two months) within which samples were collected in combination with 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.

4.3.6 Catch Curve Analysis

Catch curve analysis for *M. ambigua* catches was carried out, treating all small-scale fisheries monitored as if they were one fishery, to estimate instantaneous mortality (Z) and annual mortality (A) estimates (±95%CI) for *M. ambigua* in southwestern Indian Ocean small-scale fisheries. The Chapman-Robson method was selected as regression estimator catch curves show strong negative bias in mortality estimation (Smith *et al.* 2012). Age was estimated for all recorded catches with DW data (n=127) from the 12-month landings monitoring programme using the age-DW models. Age classes beyond estimated longevity were excluded as confidence in these estimates are low. Age of full recruitment to the fishery was determined as the age with peak abundance and mortality was estimated from one year after the age of peak abundance (Smith *et al.* 2012).

4.4 Results

4.4.1 Disc Width and Weight Relationship

The data showed no evidence for significant differences in the DW-weight relationship between sexes (p>0.05), thus one model was created combining data from both male and female specimens. The linear model describing the significant (p<0.05) relationship between DW and weight, logN(Weight)=2.4912*logN(DW)-8.2875, was found and is presented (Figure 4.3).

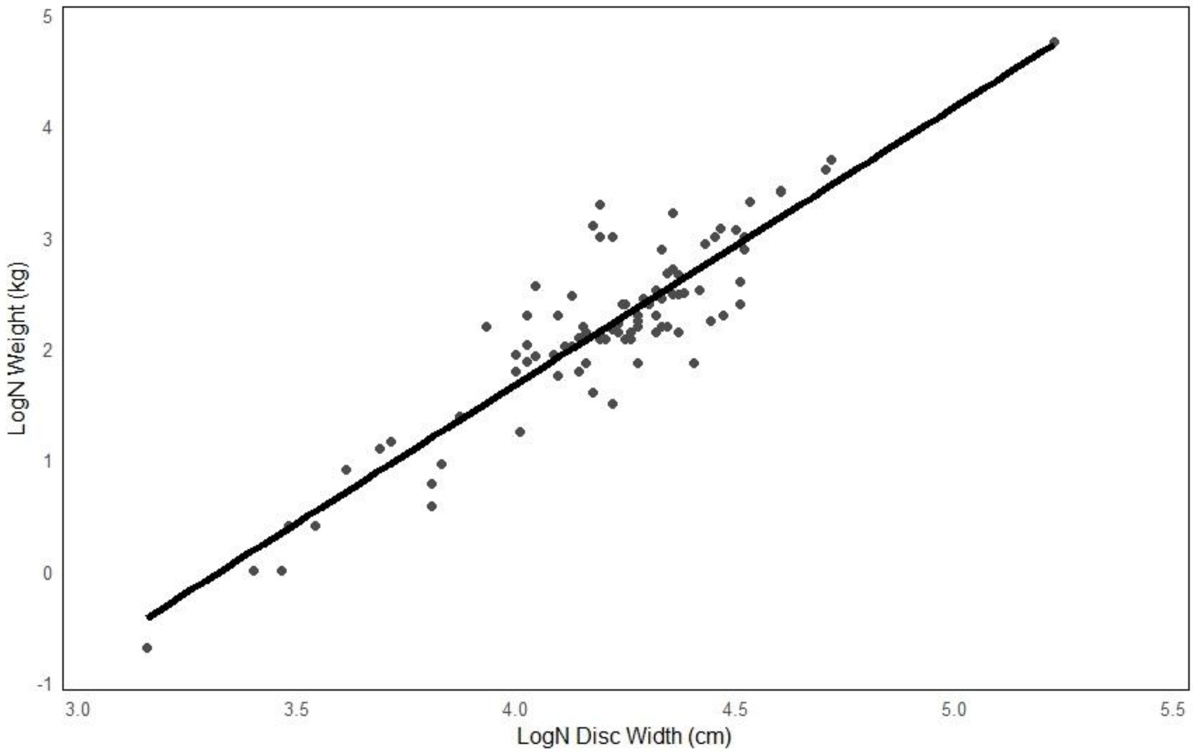

Figure 4.3 Linear relationship for the natural log (LogN) transformations of weight (kg) and disc width (cm) for Baraka's whipray (*Maculabatis ambigua*): logN(Weight)=2.4912*logN(Disc Width)-8.2875.

4.4.2 Age Estimates

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 readers (p>0.05), but evidence of significant bias between readers (p<0.05), with reader 1 producing higher estimates than reader 2 for older specimens (Figure 4.4). LOAs, representing precision between age reads are presented alongside standard precision metrics (Table 4.2).

Table 4.2 Precision values for comparisons of age estimates for Baraka's whipray (*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 ±1year (%)	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

4.4.3 Age-Growth Models and Longevity

Age estimations ranged from 0-12.5 years for males and 0-17 for females. Of the growth models tested, von Bertalanffy was found to provide best fit for both males and females. Likelihood ratio tests showed no evidence ($p>0.05$) for differences in the growth curves between males and females and so data were combined. The von Bertalanffy growth model also showed best fit for the combined male and female data (Figure 4.5). Growth parameter estimates for the combined model were; DW_{∞}=92.2(95%CI 82.4-112.3)cm, DW_0=33.6(95%CI 25.2-42.5)cm and k=0.234(95%CI 0.120-0.390). Longevity estimates based on estimating age at 95% DW_{∞} were both less than those ages observed in this study at 14.8 years (Ricker 1979) and 12.8 years (Taylor 1958). However, longevity estimates based on age at 99% DW_{∞} was estimated at 20.7 years (Fabens 1965). Given that longevity estimates at 95% DW_{∞} were less than ages observed in this study, 20.7 years was considered the best estimate of longevity.

DW (±95%CI) at which 50% of males reach maturity was estimated at 56.9(95%CI 53.0-60.8)cm, and at which 95% of males reach maturity was estimated at 67.4(95%CI 53.5-72.4)cm (Figure 4.6). These estimates equate to 50% of males reaching maturity at 2.2(95%CI 1.7-2.7) years, and 95% of males reaching maturity at 3.7(95%CI 1.8-4.6) years. The smallest of the female specimens classified as mature (n=3) had a DW of 62cm.

4.4.4 Catch Curve

The Chapman-Robson catch curve is displayed (Figure 4.7). Age of full recruitment to the fishery is estimated to be three years. The catch curve estimates total instantaneous mortality (Z) at 0.373(0.233-0.513) and annual mortality rate at 31.1(20.8-40.1)%. However, the data likely violates a number of the assumptions of the catch curve. Specifically the assumption of constant vulnerability. Different age/size classes (after age/size at full recruitment) are unlikely to be equally vulnerable to the fishery as the catch curve presented

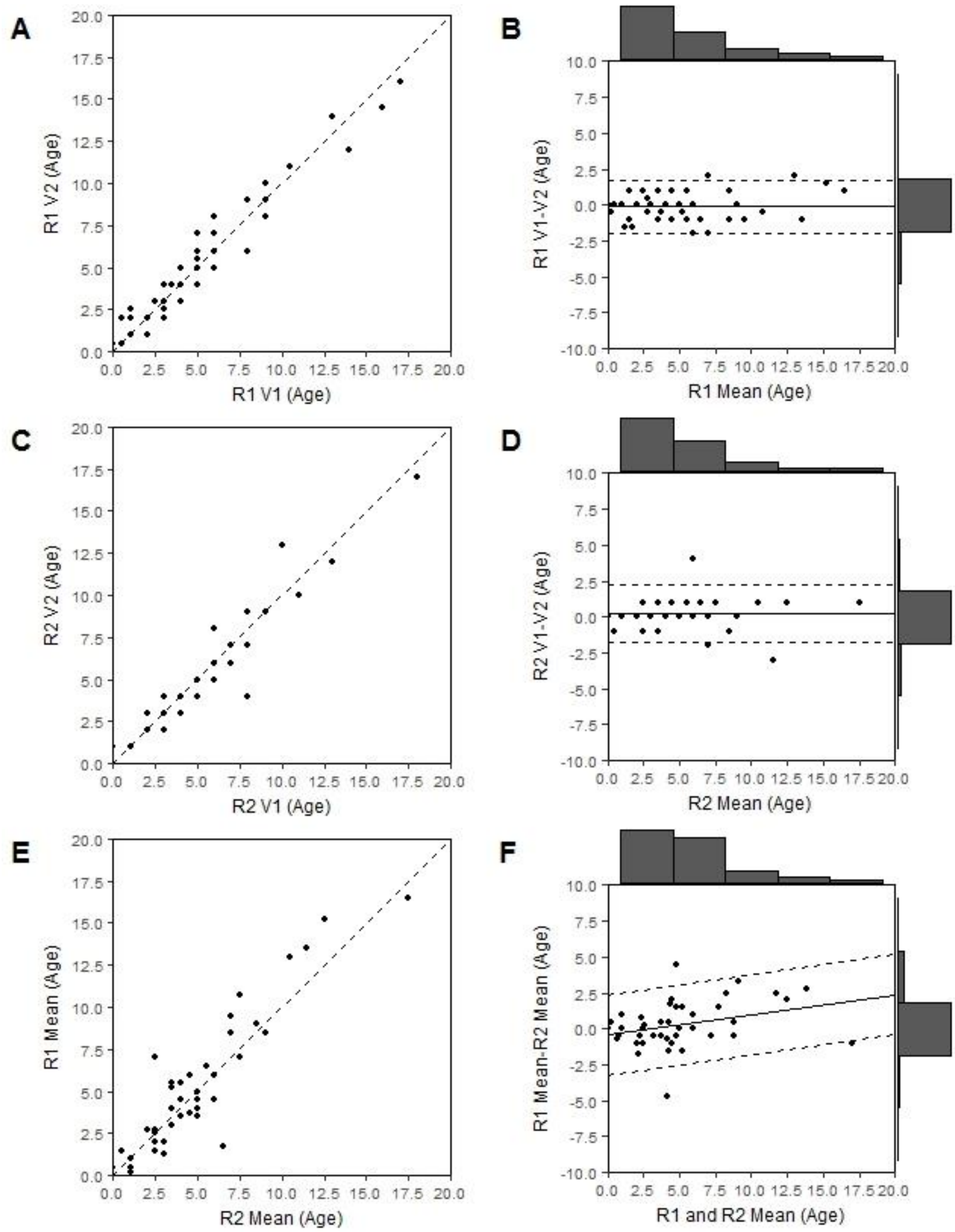

Figure 4.4 Bland-Altman assessments of agreement, precision and bias in age estimates (year ± 0.5) for Baraka's whipray (*Maculabatis ambigua*) within and between readers. A) Relationship between vertebrae age band counts (V1 and V2) for Reader 1 (R1), B) Bland-Altman Plot displaying bias and precision between vertebrae age band counts for Reader 1, C) Relationship between vertebrae age band counts for Reader 2 (R2), 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.

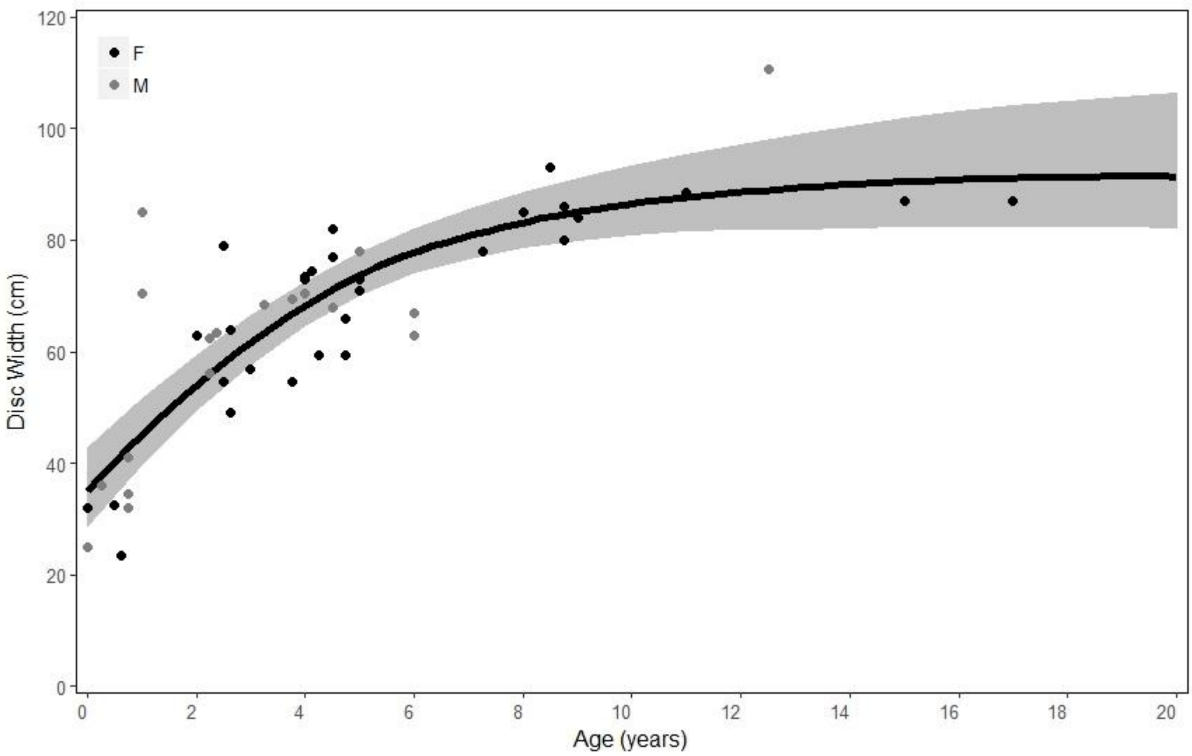

Figure 4.5 Three parameter von Bertalanffy growth curve describing the disc-width to age relationship for male and female Baraka's whipray (*Maculabatis ambigua*) combined, 95%CI is displayed.

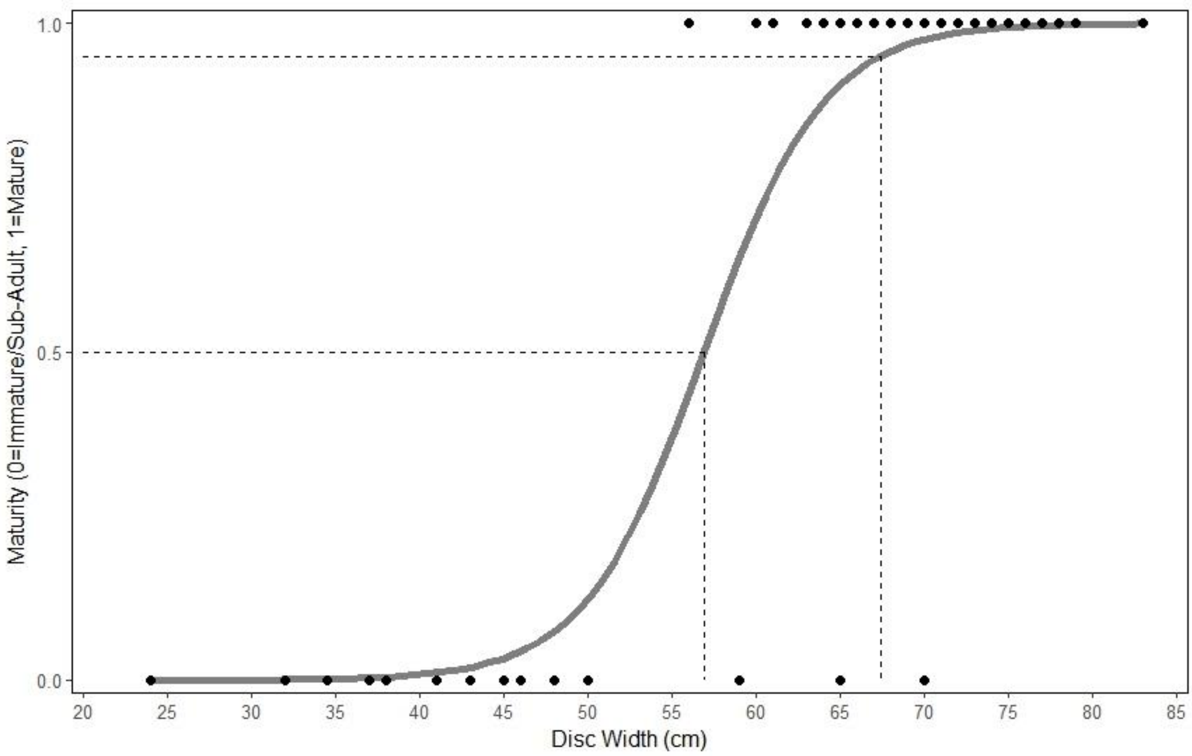

Figure 4.6 Logistic regression describing the relationship between disc width (cm) and maturity status (immature/sub-adult or mature) for male Baraka's whipray (*Maculabatis ambigua*), disc width at 50% maturity and 95% maturity estimates are indicated.

combines data across gear types, and differing gear types are unlikely to follow the same age/size class selectivity. Further we cannot be certain that the assumptions of constant mortality across age/size classes and of a closed population are met. Thus the outputs must therefore be treated with some level of caution.

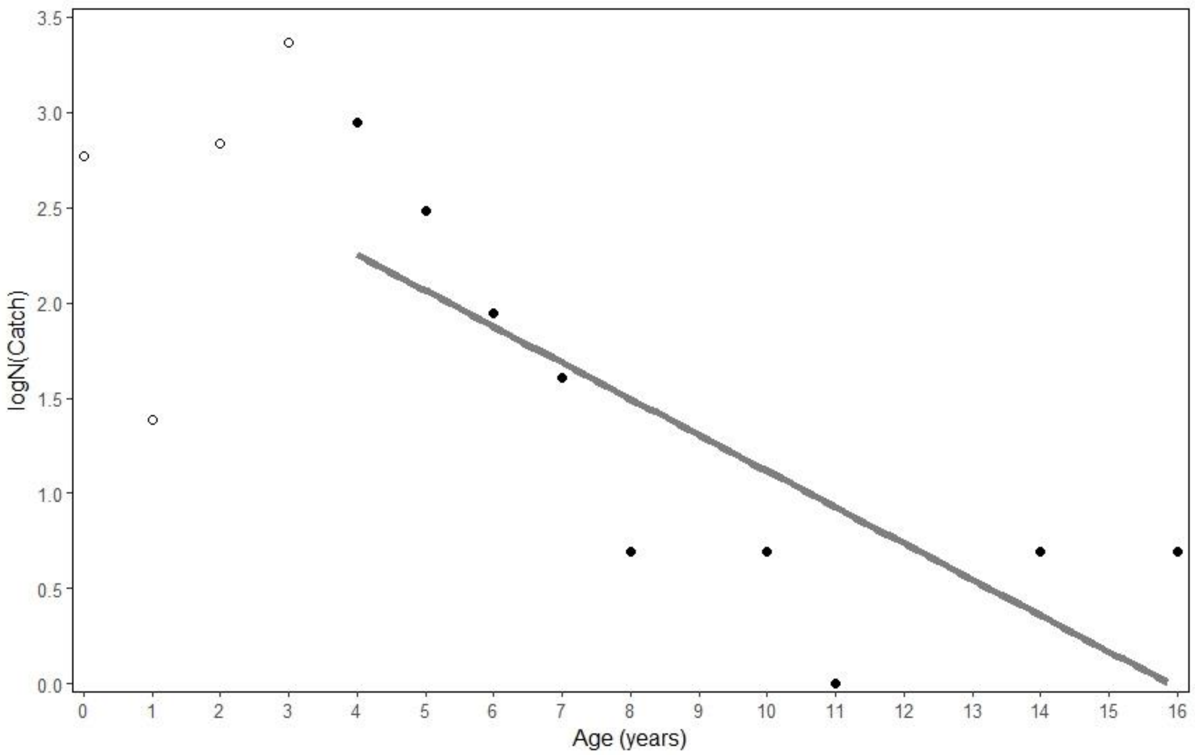

Figure 4.7 Chapman-Robson catch curve for Baraka's whipray (*Maculabatis ambigua*) across southwestern Indian Ocean small-scale fisheries, displaying age class of full recruitment to the fishery (age class 3) and catch curve regression line from age class 4 to 16, Z=0.373(0.233-0.513).

4.5 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. However, fecundity, and therefore rebound potential, remain unknown; SWIO SSF appear to catch *M. ambigua* across a wide range of ages, with full recruitment to the fishery occurring post-maturation; and sex specific growth rates and maturity in females could not be assessed due to limited available sample sizes. Thus, the potential resilience of *M. ambigua* to fisheries exploitation must be treated with caution. The study also indicates larger potential maximum size, DW_∞ at maximum of 112cm, and wider distribution than previously known (Last *et al.* 2016), as evidenced by the presence of *M. ambigua* in the small-scale fisheries 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. However, 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 also presented comparison among and within readers using the Bland-Altman approach (Bland and Altman 1999, 2003), a method designed to assess and illustrate the agreement, precision and bias between measurement outputs from differing sources. The outputs of the Bland-Altman approach showed that there was no evidence for an increase or decrease in discrepancies between reads with increasing age. This indicates consistency in the variability within and among readers and thus increases 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 the 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 therefore 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 (e.g. 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 (e.g. 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 estimates for the combined male and female data indicate that *M. ambigua* is fast growing species with early maturation relative to other moderately sized whiptail stingrays. Studies of other similarly sized whiptail stingray species using the two parameter von Bertalanffy growth curve describe substantially lower growth constants indicating slower growth; e.g. *Maculabatis astra* (male: DW_{∞}=72.3, k=0.03, female: DW_{∞}=82.2, k=0.03) (Jacobsen and Bennett 2011), *Dasyatis chrysonota* (male: DW_{∞}=53.2, k=0.175, female: DW_{∞}=91.3, k=0.07) (Cowley 1997), and *D. dipterura* (male: DW_{∞}=62.2, k=0.1, female: DW_{∞}=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). Yet, the apparently rapid growth rate of *M. ambigua* is also re-enforced by early maturation in males.

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 showed that males mature at ~10.5% of maximum age, in which time they achieve ~61.7% of DW_{∞}. Further, males reached maturity earlier than other whiptail stingrays, e.g. *N. kuhlii* (3.95 years, ~19.3% longevity)(Pierce and Bennett 2010) and *M. astra* (7.32 years, ~40.7% longevity)(Pierce and Bennett 2010), and at similar relative size as other whiptail stingrays, e.g. *M. astra* (~64.9%DW_{∞})(Jacobsen and Bennett 2011), *N. picta* (~63.5%DW_{∞}) and *N. kuhlii* (~64.9%DW_{∞})(Jacobsen and Bennett 2010), *D. dipterura* (~74.7%DW_{∞})(Smith *et al.* 2007) and *D. pastinaca* (~52.1%DW_{∞})(Yigin and Ismen 2012). Female maturation could not be estimated in this study, and male maturation is likely biased by the derivation of growth curves from combined male and female data. However, results from other studies of whiptail stingrays suggest that males and females mature at similar size classes (White 2007; Jacobsen and Bennett 2010, 2011; Da Silva *et al.* 2018) and this is reflected in the smallest known size at maturity recorded for females in this study. Further, female whiptail stingrays generally display slower growth rates but longer lifespans (Cowley 1997; Pierce and Bennett 2010; Jacobsen and Bennett 2011; Da Silva *et al.* 2018). Thus, it would be expected that females would similarly mature at a young age relative to their lifespan and likely at a similar size class. 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). This would suggest that *M. ambigua* may be relatively resilient to fisheries exploitation. However, rebound potentials are intrinsically linked to fecundity. The early maturation of *M. ambigua* combined with a moderately long lifespan relative to other whiptail rays (Jacobsen and Bennett 2011) suggests a life-history strategy aimed at maximising the number of litters over its lifespan. Yet, the smallest animal measured in this study and DW_0 estimated from the growth model, ~25.5% and ~36.4% of DW_∞ respectively, suggest high maternal investment in offspring and thus likely few offspring per litter. Low reproductive rates have strong implications for rebound potential, and subsequently resilience to fisheries exploitation. Thus, a better understanding of the reproductive life-history of this species is critical to allow informed assessment of *M. ambigua* resilience to fisheries exploitation and vulnerability status.

Lastly, of concern from a fisheries perspective is the age and corresponding size class selectivity. 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), though protection of sub and young adults may best maximise the future reproductive potential of the stock (Kindsvater *et al*. 2016). Conversely, Based on the Chapman-Robson catch curve full selectivity to the small-scale fisheries in the southwestern Indian Ocean is estimated around age three, equating to a DW of approximately 63.2cm. Assuming size-at-maturity estimates for males are approximately representative, despite uncertainty generated in the use of a combined male-female growth curve, and that females are likely to mature at a similar size to males (White 2007; Jacobsen and Bennett 2010, 2011; Da Silva *et al.* 2018), the data suggest that recruitment to the fishery occurs primarily in post-maturation life-stages. 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). The data demonstrates the occurrence of broad-age and size class exploitation in SWIO SSF, 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 sustainability of both *M. ambigua* and likely numerous other elasmobranch species caught commonly in the region (Temple *et al.* 2018).

The life-history information presented here for the recently described *M. ambigua* is an important step towards more comprehensive future 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 both males and females in order to allow sex-specific life-history analyses. Further, improving our understanding of female maturation and fecundity is crucial in estimation of the rebound potential of the species. Lastly, life-history traits may vary significantly within species between stocks (e.g. 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), and potentially differing life-histories between stocks have implications for their resilience to fisheries exploitation and subsequent management needs. Given it's prominence in southwestern Indian Ocean elasmobranch fisheries (Temple *et al.* 2019), investigation of stock disaggregation should be considered a priority next step in informing management of *M. ambigua*.

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.

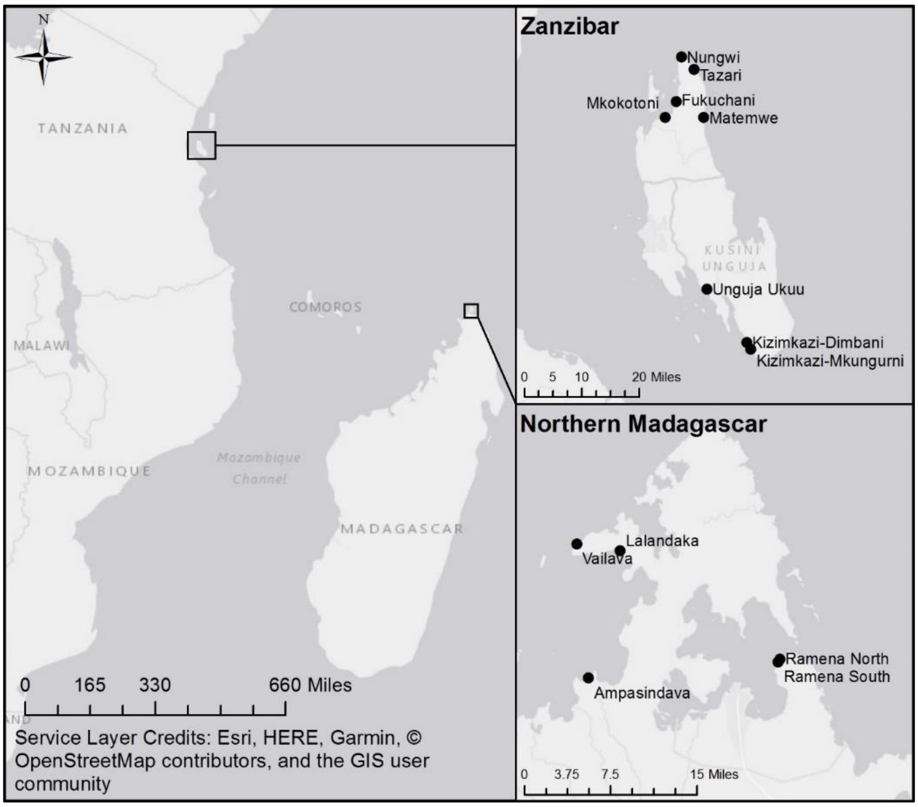

Figure 5.1 Locations of landing sites in the southwestern Indian Ocean monitored by observers for marine megafauna catch between June 2016 and June 2017 where subsequent Rapid Bycatch Assessment interviews were undertaken.

Month significantly influenced bottom-set, driftnet, handline and longline fishing effort (χ^2=1453.10, 42.45, 161.48, 380.62, respectively; p<0.05). Tidal height significantly influenced driftnet, handline and longline fishing effort (χ^2=41.65, 33.12, 9.76, respectively, p<0.05). Wind direction significantly influenced driftnet and handline fishing effort (χ^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 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, 0.715, $p<0.01$) in Madagascar. However, a negative correlation was found for handlines in Madagascar (Pearson, $r=-0.615$, $p<0.05$).

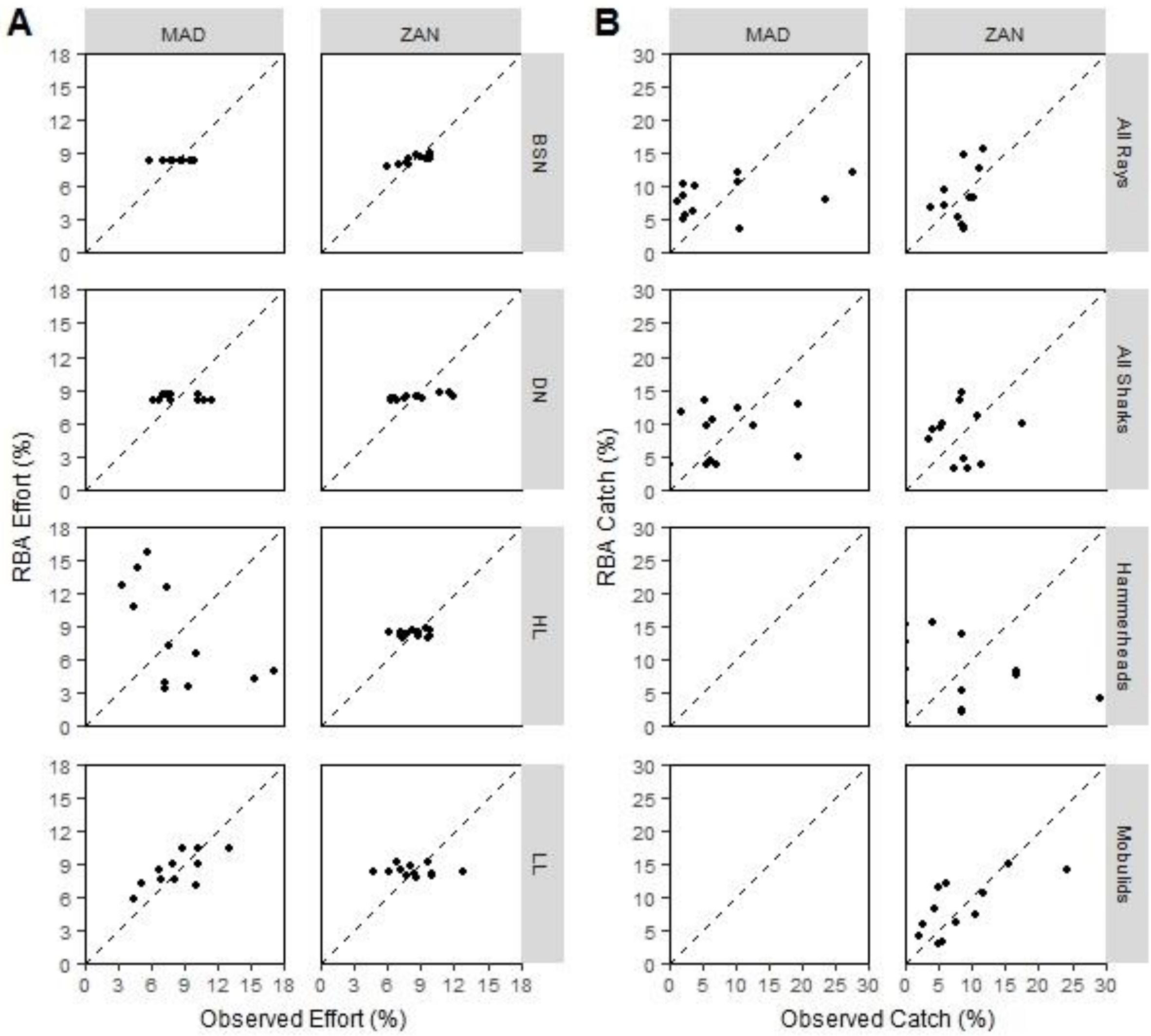

Figure 5.2 Relationships between Rapid Bycatch Assessment (RBA) and landing observer estimates between June 2016-June 2017 of monthly A) fisheries effort, and B) fisheries catch. Data is displayed for bottom-set gillnets (BSN), drift gillnets (DN), handlines (HL) and longlines (LL) for fisheries effort, 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 supressing variability. LOA were calculated at ±1.99%(95%CI 1.51-2.71%) of total effort (Figure 5.3).

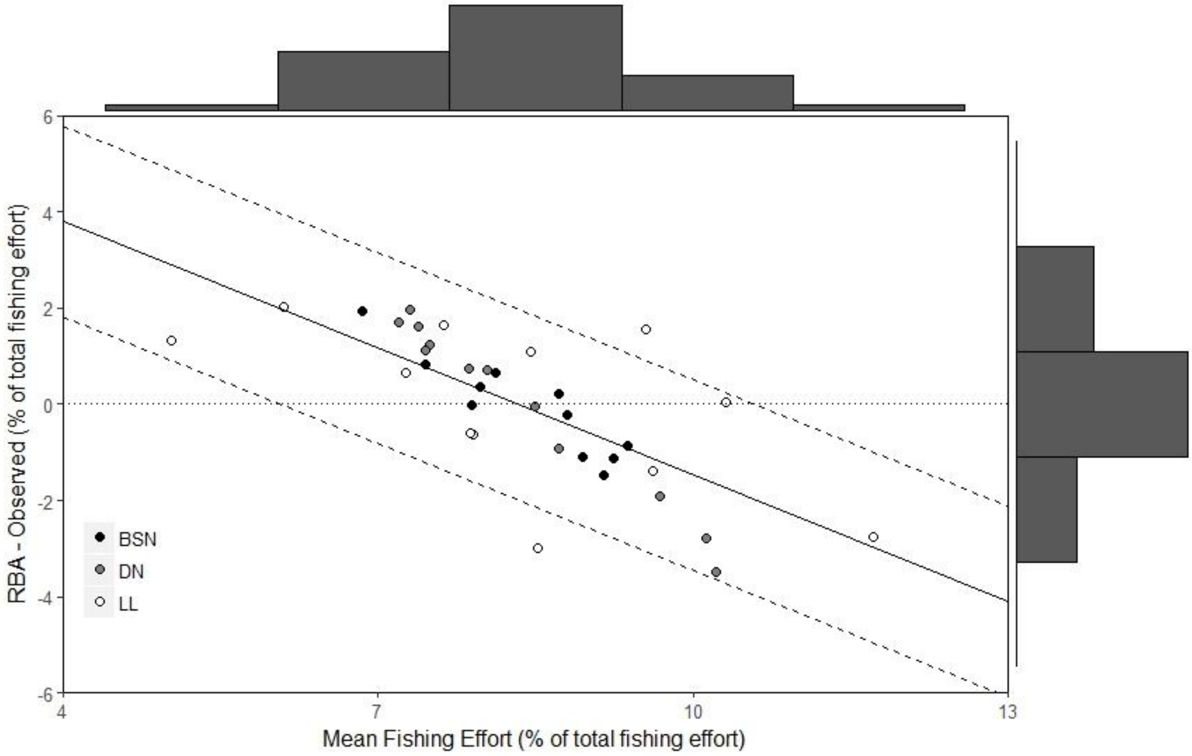

Figure 5.3 Bland-Altman plot for comparison of Rapid Bycatch Assessment (RBA) and landing observer estimates of monthly fishing effort between June 2016-June 2017, linear regression line is displayed alongside the 95% mean confidence interval of residuals, axes histograms display data distribution (BSN = 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 ±9.06%(95%CI 5.63-16.31%) of total catch (Figure 5.4).

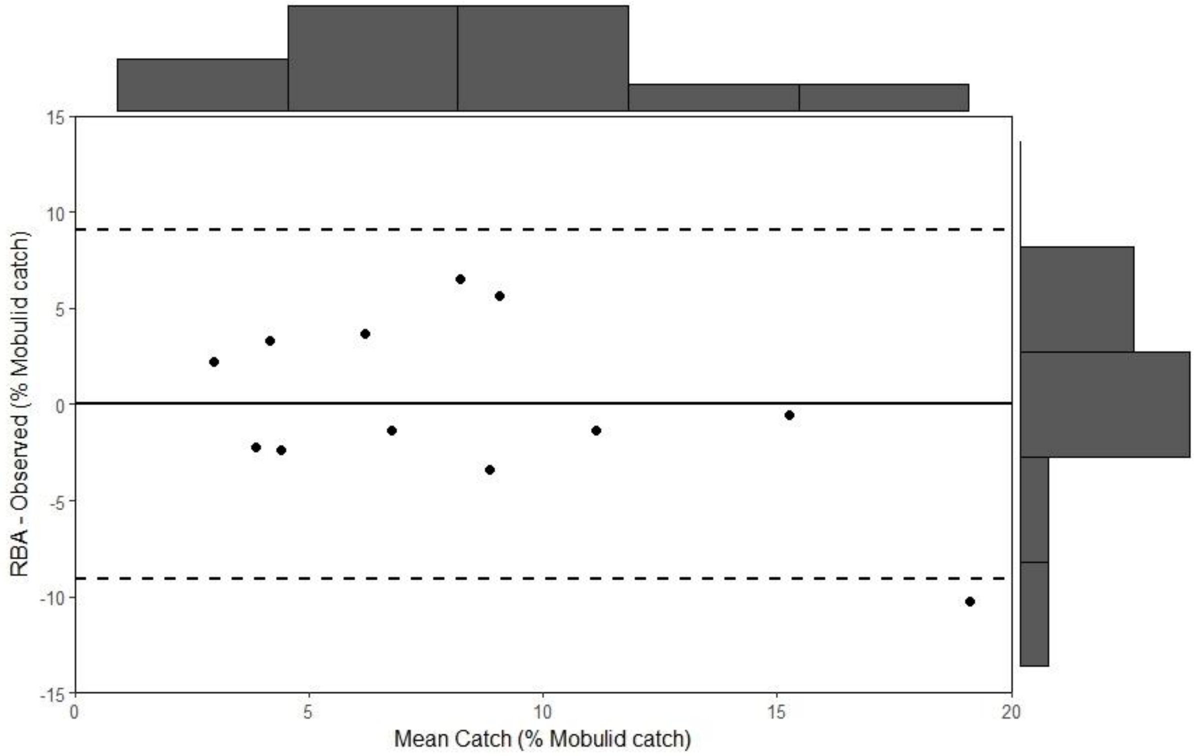

Figure 5.4 Bland-Altman plot for comparison of Rapid Bycatch Assessment (RBA) and landings observer estimates between June 2016-June 2017 of intra-annual patterns in mobulid catch in Zanzibar, 95% mean confidence intervals are displayed, axes display histograms of data distribution.

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 ±16.94%(95%CI 9.43-36.67%) and ±26.87%(95%CI 14.96-58.16%) of total catch respectively (Figure 5.5).

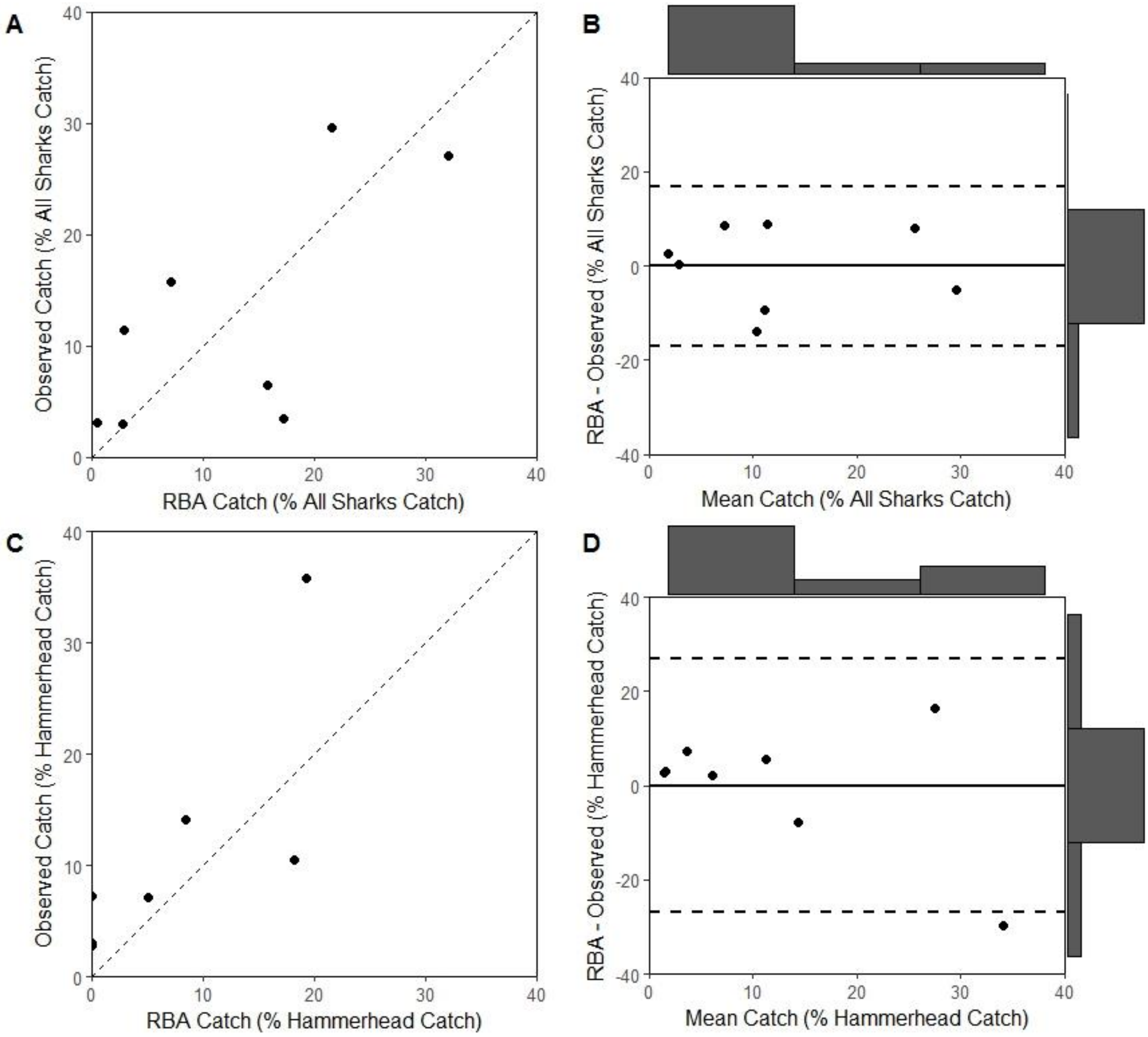

Figure 5.5 Bland-Altman comparison of Rapid Bycatch Assessment (RBA) and observer network estimates of patterns among Zanzibar landing sites between June 2016-June 2017 where significant correlation was found A) sharks (r=0.738) and C) hammerhead sharks (r=0.878), and Bland-Altman plots showing 95% mean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

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).

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
Sawsharks/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 supress 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 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 supressing 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, it’s 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.

Chapter 6. Thesis Conclusion

6.1 Overview

Monitoring the activities of small-scale fisheries (SSF), their impacts on marine megafauna (herein referring to elasmobranchs, marine mammals and sea turtles) and the marine environment more broadly is challenging for researchers and managers. Globally, small-scale fisheries include some 22 million fishers, more than 90% of fishers worldwide (Kelleher *et al.* 2012). Further, SSF are highly diverse, in both fishing methodology and species catch, widely dispersed and most numerous in developing nations, where the financial and human capacity is relatively low. SSF may consequently have substantial negative impacts on the environment (e.g. Hawkins and Roberts 2004; Salas *et al.* 2007; Pinnegar and Engelhard 2008; Moore *et al.* 2010) and their socio-economic importance to coastal communities cannot be overstated (Béné 2006; Pauly 2006).

Prior to this thesis little was known of SSF catch, and in particular catch of marine megafauna, at the regional level in the southwestern Indian Ocean (SWIO). National governmental and regional inter-governmental (e.g. FAO) monitoring of the SWIO SSF currently yield little information of fisheries activities beyond basic vessel and gear counts, and the majority of marine megafauna catch is reported in broad categories (i.e. "sharks and rays") or is entirely unknown (marine mammals and sea turtles) (Chapter 1). Further, what little regional level work has been carried out by independent researchers has been interview based (e.g. Moore *et al.* 2010; Kiszka 2012). The findings of these studies are suspected to be biased towards easily distinguishable and/or memorable species as a result of inherent bias in human cognitive recall (Matlin 2004; Hirst *et al.* 2009) and difficulties in identifying catches to species level for untrained peoples. The outputs (e.g. intra-annual patterns in fisheries catch and effort) from the interview methods used in these studies were found to be neither broadly equivalent nor interchangeable with standard landings observation-based methods (Chapter 5), raising concerns regarding their widespread use in SSF assessment in their current format. The results presented in this thesis, based on landings monitoring, indicate substantial disparity in the composition, volume and vulnerabilities of elasmobranchs, compared to those derived from existing governmental (FAO 2018a) and independent research (e.g. Kiszka 2012; Pauly and Zeller 2015) for SWIO SSF catch (Chapter 2). Additionally, both landings and interview derived data confirm ongoing catches of both marine mammals and sea turtles across the region (Chapter 2 and Chapter 5). The substantial disparities in composition, volume and vulnerabilities between this thesis and existing data indicate that ongoing management efforts from international (e.g. Convention on the Conservation of Migratory Species, International Whaling Commission and Food and Agricultural Organisation of the United Nations), regional (e.g. Indian Ocean Tuna Commission) and national bodies (Chapter 1) are likely to be undermined by the poor reliability and resolution of existing data for SWIO SSF.

The research presented in the thesis also suggests that those fishers who are most dependent on elasmobranch resources display characteristics indicative of a specialised livelihood strategy, relative to others in the fisheries (Chapter 3). This specialism is compounded by the generally low capacity of small-scale fishers to respond to external shocks (Béné *et al.* 2007), the relative vulnerability of the elasmobranch resources they depend upon (Compagno 1990; Žydelis *et al.* 2009; Dulvy *et al.* 2014), the demonstrated underestimation of elasmobranch catches and the lack of informed management (Chapter 2). These findings indicate a greater risk for both elasmobranch resources and the households and communities that rely upon them than previously assumed (Chapter 2 and Chapter 3).

Finally, there is a need for greater regional efforts to document basic elasmobranch species life-history, which vary widely among and with species (Stevens and McLoughlin 1991; Lombardi-Carlson *et al.* 2003; Jacobsen and Bennett 2010, 2011; O'Shea *et al.* 2013) and is critical in future assessments and formulation of evidence-based management for these fisheries resources. This thesis begins this quest by contributing life-history information for a ray species (*Maculabatis ambigua*), a recently described (Last *et al.* 2016), yet important constituent of the ray catch in SWIO SSF (Chapter 4).

6.2 Future Considerations for First-Step Assessments

Beyond the limitations and priority next steps identified within the chapters of this thesis the author wishes to highlight some specific, underlying challenges in the assessment methods used to document SSF interactions with marine megafauna and potential avenues through which these might be mitigated in the future. In the following sections possible methodological adaptations for future first-step assessments of marine megafauna interactions with SSF in data-poor environments are considered.

At the most basic level the assessment of fisheries requires three fundamental components; fisheries activity (which may be as basic as a vessel and gear type census), catch composition (relative contribution of species to the overall catch) and catch volume (which may be absolute or relative). With these components also requiring data for species resilience to exploitation (life-history) in order to achieve basic indicators of species vulnerability.

During the course of the research carried out in this thesis it has become increasingly clear that there is a need to improve first-step methods to document and begin assessments of data-poor or no-data marine megafauna fisheries. Past regional assessments (e.g. Moore *et al.* 2010; Kiszka 2012), and indeed the research carried out within this thesis, attempt to generate more complex data, potentially at the detriment of the identified fundamental components (fisheries activity, catch composition and catch volume). Here I propose adaptations to the two first-step methods used in this thesis (landings monitoring and interview survey) through which to address these three fundamental components. These methods may be adapted to increase complexity depending on the specific context within which the first-step assessment is carried out or form a basic indicator upon which subsequent research may build. The first proposed method may be suitable for assessment of both legal (elasmobranch) and illegal (marine mammals, sea turtles) catches. The second is appropriate only for legal catches.

6.2.1 Re-Development of the Rapid Bycatch Assessment

The primary goal of the Rapid Bycatch Assessment (RBA) (Kiszka 2012), used to document data poor fisheries, is to provide a quick and cheap baseline understanding of the marine megafauna catch in data-poor fisheries. It currently does so by collecting data on fisheries effort and catch, including seasonal variability and composition from a large number of fishers across a range of gear types from a sub-sample of landing sites. It is the belief of the author that, in its current format, the RBA sacrifices spatial coverage in favour of depth of understanding. This is a result of its requirement to sample multiple fishers across gear-types in each landing site, inflating sample-size requirements and thus restricting spatial coverage. The resultant restriction on spatial coverage is counter-productive to its intended use. Given the widely variable nature of SSF, and thus their interactions with marine megafauna that are both legal (elasmobranchs) and illegal (marine mammals, sea turtles), maximising spatial coverage is pivotal to the identification or indication of potential hotspot and/or high risk locations. It is therefore recommended that the RBA is either re-developed, or a new interview based assessment is formulated, so that the intended purpose is better served. The survey should be formulated with the following aims, with the goal of maximising spatial coverage:

- Be short and concise, minimising interview length and so maximising the ability to cover a greater number of landing sites whilst minimising interview fatigue
- Be targeted at key-informants based in landing sites (e.g. village heads, auctioneers, merchants, retired fishers) who can provide a greater overview of fisheries activities than can be derived from individual fishers, thus reducing the sample size required (perhaps 2-3 interviews per landing site)
- Minimise complexity in data e.g. number of vessels by gear type, most common catch groups (from pre-selected easily identifiable groups) and links to gear type in order to reduce complexity for the interviewee
- Minimise complexity of data types collected (i.e. ranking rather than estimation of numerical value where possible) to reduce complexity for the interviewee
- Allow for cross-validation of outputs through the derivation of data for neighbouring landing sites (particularly necessary in the case of illegal catches)

In doing so, this first-step survey would serve as a baseline indicator of potential hotspot and/or high risk areas, assisting in the selection of landing sites of interest for more detailed research.

6.2.2 Simplification of Landings Monitoring and the Use of Metagenomics

The two most prominent difficulties encountered during the course of the landings monitoring programme undertaken in this thesis (Chapter 2) were the identification of landed elasmobranchs and the quality of SSF census data. Species identification from trained observers is generally poor throughout the region (Temple personal observation), photograph quality may be variable, a sizable component of species present in the catch are somewhat cryptic (e.g. *Squalus* and *Himantura* spp.) and a number of species are likely new to science (e.g. the author is currently involved in description of a new sixgill sawshark species and potential new *Squalus* spp. and *Mustelus* spp.). Thus, the author considers methods to maximise the quality of compositional analysis of particular importance. Additionally, fisheries census data (vessels and particularly gears) is a clear regional weakness in the monitoring of SWIO SSF (Chapter 1) and those elsewhere. Improving census data is a priority to generate a minimum level proxy for fisheries effort. Where circumstances permit a more in-depth assessment of marine megafauna catches, particularly as those relate to legal catches (i.e. elasmobranchs), the author would propose the exploration of combining modern molecular techniques (e.g. metagenomics) with observation-based methods and improved fisheries census data.

Metagenomics, a technique primarily used in the study of microbial communities and in biodiversity monitoring through eDNA (e.g. Marco 2011; Bohmann *et al.* 2014; Kelly *et al.* 2014), allows for the analysis of the relative contribution of species to a community. Through the use of standardised tissue sample coring (i.e. extraction of exact quantities of tissue from a standardised location) and combined storage (i.e. samples stored in a singular container) it would be feasible to apply the metagenomics approach to assess relative species composition in fisheries catch, though an appropriate genetic catalogue is first required in order to identify catches consistently to species level. The application of metagenomics to landings-observation based elasmobranch catch sampling could provide a rapid, and simple for the observer, approach to the collation and analysis of species compositional data. Combined with a simple tabulation of total elasmobranch catch data for total catch volume (number of animals) it is possible to derive total counts of catch by species. With representative coverage such data could be combined with fisheries census data, assuming reasonable levels of geographic and temporal coverage, to model and predict total catches and composition of elasmobranchs in SSF at larger geographical scales.

6.3 Conclusion

The research presented in this thesis has helped to identify, and in some areas begin to address, the data gaps surrounding the interactions between SSF and marine megafauna in the SWIO region. It was beyond the scope to explore in detail the avenues through which to mitigate those interactions that result in negative impacts on species affected. What is clear however, is the intractable nature of this problem. The open-access nature of SSF and safety net facility they provide (Béné *et al.* 2010) are currently essential in sustaining coastal communities in developing countries. Impinging upon this may have dire consequences for these communities. Conversely, the role played by SSF in these communities is by its very nature a threat to the long-term sustainability of marine megafauna, and indeed the marine ecosystem as a whole. The issue is compounded further by the disparate and highly abundant nature of SSF, combined with the limited financial capacity of developing nations, making directed management of these fisheries a very challenging task. Thus it seems likely that, at least in the short-term, that negative impacts of SWIO SSF and marine megafauna are liable to increase.

However, there is some hope in the long-term, as SWIO nations continue to undergo rapid economic development, and related elements such as education improve, employment tends to shift between the “mega-sectors”, away from the primary sector (i.e. sectors harvesting natural resources) towards secondary (manufacturing) and tertiary (commercial services) sectors (Joachim 1978). It is the opinion of the author that researchers and managers should consider management interventions and strategies in developing country SSF that aim to act as buffers to marine resource exploitation or reduce dependence on marine resources in the short-to-medium term such as; exchange of species unselective gears, e.g. swapping drift gillnets for pole and line fisheries; development of low cost bycatch mitigation solutions, e.g. cheap acoustic alarms or ways to increase acoustic reflectivity of nets to reduce odontocete bycatch (e.g. Berggren *et al.* 2017a; Berggren *et al.* 2017b; Temple *et al.* 2017); strategies aimed at altering fisher behaviour, such as market governance (Chapter 3); and livelihood programs aimed at diversifying fisher households livelihood strategies and reducing their dependence on fisheries resources. Subsequently, as the financial limitations decrease and enforcement capabilities increase managers may be able to leverage direct strategies, such as limiting fisheries access, quota systems and closed areas, more effectively than is possible at present.

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APPENDIX A. BYCAM Governance and Socio-Economic Fisheries Survey

Survey Ref (e.g ZAN 001)	**Date** (DD/MM/YYYY)	**Interviewer**	**Location/village**	**Occupation** (e.g. Fisher/Captain)

Introduction by interviewer

Hello, my name is [*e.g. Yussuf*]. I am part of an international team that wants to make a difference by helping communities in our country to have better access to food and money from fishing. We believe the best way to do this is to understand how the fishery operates from those involved.

The purpose of this survey is to learn about your experience in fishing and if you are willing to share, to hear about what you think is working, not working and what can be done to improve support for fishing. The senior scientists involved have given their time freely to help train people like me so together we can improve the way we manage our fishery.

I understand a lot of surveys take place and take up your valuable time. We know some people are unhappy if they do not get results of the survey so we will make sure the results are made available for everyone.

We cannot guarantee to change the health of fisheries but you sharing your views is a start in identifying the current issues in fishing and documenting what could be done to improve fishing. All your answers are strictly confidential and cannot be traced back to you.

<u>1. Management</u>

A) Does fishing – catching fish or other animals from the sea to eat or sell – need to be managed, that is should there be rules about fishing? *If yes, why? [List reasons offered by interviewee]*

B) What rules are there for fishing? Do you disagree with any of the rules? If yes, why? [*Prompt for rules regarding who can fish (e.g. licences) what can be caught (e.g. species, size,* *season), how it can be caught (e.g. gear) and what/how it can be sold (e.g. markets)? Provide answers in the table below*]

Rule	Disagree? (Y/N)	If disagree, why?

C) How would you **rate effectiveness** [*1=very effective, 2=effective, 3=ineffective, 4=very ineffective, N/A=non-existent*] of the **people/groups** that **control rules** about fishing activity in order to support healthy fish stocks (i) **today** compared with (ii) **5yrs ago** (iii) **when you started fishing** and (iv) what you **think** they will be like in **10 years time?** [*List different people/groups involved and their roles in Table below*]

Person/Group	Role?	**Effectiveness** [*Rank 1-4 or N/A*]			
		(i) Today	(ii) Past - 5yrs*	(iii) Started fishing	(iv) Future - 10yrs

*Try to find an event in the past that everyone can use as a point in time reference point to compare e.g. MPA introduced or new president or other significant event in time that people will recall before and after.

D) Do you know who are responsible for the following in relation to fishing activities?

Responsibility	Who?
Data/Information Collection	
Data/Information Analysis	
Decision Making about type of fishing rules	
Implementation of fishing rules	
Evaluation of impact of rules (check if rules improve health of fish stock)	

E) Do all boats and fishers have licences? If not, what % of each **do not** have licences?

Boats	Fishers

F) Does rule-breaking of fisheries rules take place? *If yes by whom and why?*

G) If you saw someone breaking rules (i.e. a local, someone from outside the village or someone from outside the country) about fisheries what would you do? *[for the purpose of this question in Zanzibar we consider Tanzanian mainland to be international]*

Local	
Non-local	
International	

H) Can you suggest ways to improve how well rules are followed?

I) Which person, organisation or group do **you think** is most appropriate to make decisions about how best to manage the fishery in a sustainable way so future generations can benefit from access to fish in the oceans? *Why?*

J) What do people think about marine protected areas in relation to fisheries?

2. BYCAM species monitoring data

A) Who *[record their role e.g. beach observer, rather than their name],* if anybody, collects information/data about sharks, rays, dolphins or turtles caught? If data is collected please describe what data and when it is collected.

Species	Who?	What?	When?
Sharks			
Rays			
Dolphins			
Turtles			

B) If data on animals caught from the sea is collected, do you think it is a good reflection of the total catch of these species *[Yes/No/Don't Know]*? How effectively is this information collected i.e. on an average day when information is being collected do any sharks/rays/dolphins/turtles get missed? *If so, what proportion (%) **are missed**?*

Species Group	Sharks	Rays	Dolphin	Turtles
Reflects catch?				
% missed				

3. Governance

A) If a decision is needed about what action to take about a fishing issue (*e.g. if someone has broken a rule related to fishing activity*) then who would be best in finding an effective solution? *Why?*

B) For **effective decision-making** about fisheries activity which of the following **principles** do you think are **most important?** If there are any principles you think are missing please add these to the list **before** ranking (*rank in order of importance, with 1 being most important*)? E.g. openness = 1, accountability = 2 and so on

*[Note to interviewer: If any principles missing please list any volunteered by the interviewee (e.g. one interviewee said **experience in fishing** was most important) these before ranking.]*

Governance principles	Rank
Trust (*individual or group able to make best decision for the good of the fishery for everyone*)	
Transparency (*clear what and why decisions and actions are made)*	
Openness (*willing to explain openly why decisions are made*)	
Participation (*involving others to give their opinion and be involved in decisions*)	
Cohesiveness (*how well do individuals or groups co-operate in deciding on a fishing related issue*)	
Accountability (*taking responsibility and being answerable for their decisions on actions*)	
Respect (*to admire deeply as a result of their achievement, ability or qualities*)	
Effectiveness (*the degree to which someone is successful in making an action work in practice*)	

4. Fisheries activity

A) What types of fishing gears do **you use**? What proportion of **your overall fishing activity** do you do with each gear? What months do you usually use these gears in? *[Circle months in which gear is used below]*

Gear	**Proportion (%)**	**Months Used**
Handline		J F M A M J J A S O N D
Rod+Reel		J F M A M J J A S O N D
Longline (pelagic)		J F M A M J J A S O N D
Longline (demersal)		J F M A M J J A S O N D
Basket Trap		J F M A M J J A S O N D
Multifilament (Bottom set)		J F M A M J J A S O N D
Monofilament (Bottom set)		J F M A M J J A S O N D
Multifilament (Driftnet)		J F M A M J J A S O N D
Monofilament (Driftnet)		J F M A M J J A S O N D
Other? (please describe)		J F M A M J J A S O N D
		J F M A M J J A S O N D
		J F M A M J J A S O N D

B) How many years have you been using **your main gear** for? *[main gear = highest % of overall fishing]*

C) What determines your decisions about your fishing activity? **Why, when, how, where** and **what** you fish? How important are these factors (*1=very important, 2=important, 3=unimportant, 4=very unimportant*) in your decision-making about your type of fishing activity?

Factors in decision-making about fishing activity	Reasons listed	Importance (1-4)
<u>Why</u> do you fish? (e.g. income, food, both or other reason?)		
<u>When</u> (e.g. everyday all seasons, depends on crew availability or ability to cover costs of fishing?)		
<u>How</u> (e.g. why do you use that gear?)		
<u>Where?</u> (e.g. what determines where you fish?)		
<u>What</u> (e.g. do you target certain species, or are you opportunistic – catch anything available?)		

D) Ranking in order of most to least common species caught what species are the top 5 species you catch with **<u>your main and secondary gear</u>**? How much are each of these worth on average (per fish in local currency)? [*Secondary gear = second highest % of overall fishing, for catch species try and get species groups e.g. tuna, snapper, sharks etc. rather than individual species*]

Main Gear		**Secondary Gear**	
Species	**Average price**	**Species**	**Average price**
1.		1.	
2.		2.	
3.		3.	
4.		4.	
5.		5.	

E) Do you ever catch sharks or rays with **<u>your main gear</u>**? If so, what % of your catch are sharks/rays (by weight)? Has this proportion **changed** since you first started using **<u>your main</u>**

gear (*1=increased, 2=same, 3=decreased)*? *If the proportion has changed why do you think this is?* Have there been any changes in the **value** (how much is paid) of sharks/rays since you started fishing? *If so, why do you think this is?*

Type	Proportion of catch			Value of catch	
	% catch	Changed?	Why?	Changed?	Why?
Shark					
Ray					

F) What factors influence differences in **value** (how much is paid) of sharks/rays? (*e.g. meat quality, fin quality, freshness, dried, season, religious holiday etc.*). [*Note for interviewer if a reason is specific to **only sharks** or **only rays** please note*]

Positive (increases price)	Negative (decreases price)

G) Are there any species [*if species is not clear ask for an explanation/description*] of fish, sharks, rays, dolphins or turtles that people in the village (including you) can no longer catch because no longer available in the sea [***this does not*** *mean species they are no longer allowed to catch – we are trying to find out if any species they think could be extinct or no longer available where they fish*]? When was the last time (how many years ago) you caught this species? [*Ask specifically about sawfish – **picture provided see last page***].

H) If a species of shark/ray/dolphin/turtle was likely to **disappear** (go locally extinct) **because of overfishing**, but there was a way to ensure you did not catch these (and so avoid extinction) that **would not affect any of your other catch**, would you be willing to try a method to stop catching them? If not, why not?

I) At what fishing ground do you fish most often (get %) with **<u>your main gear</u>**? How long on average does it take to get there and back? How long do you fish (i.e. gear in the water) for on average once there?

Location	**%**	**Time there**	**Time back**	**Time fishing**

J) What are the **types of costs** you have to pay to go fishing (e.g. fuel, vessel rental etc.)? On an **average fishing trip** how much does each of these cost? *[Note to interviewer: if it is easier to give costs in a different form e.g. yearly vessel rent, rather than per trip, then enter this cost as normal and make a note in the Note column describing the timescale]*

Cost Type (e.g. fuel)	Average	Maximum	Minimum	Note

K) How is any income from fishing trips divided between those individuals involved in fishing (e.g. between fishers, boat owner etc.)?

L) Where do you land most of your fish? Why there? (e.g. they may say near to market or near to home)

M) Do you ever land what you catch anywhere else? If so, where and why?

N) Where are the main markets for you to **<u>sell your catch</u>**?

O) Describe the supply chain for your fishery. *[Example provided **see last page**]*

P) What can help fishers get the best price for catch from a buyer?

<u>5. Personal information and socio-economic data</u>

Are they willing to give their name and some other personal details or would they prefer to have their information recorded confidentially i.e. not linked to them? Assure them that if they give their name no-one outside the research team will be able to access that information. Record answers for those that are ok:

A) Name:

B) Age:

C) Highest school qualification certification?

D) At what age did you leave school (*this is to check above and take into account changes over years*)

E) How long have you been doing your job eg fisher? (*cross-check with age they started*)

F) Where do you live? (e.g. village)

G) Where were you born? (e.g. village)

H) Religion?

I) How many adults, including women, (aged 18 or over) live in your house?

J) How many of these adults, including women, bring in an income?

K) How many adults, including women, in the household are fishers?

L) What proportion of the **<u>total</u>** income for the household is from fishing?

M) What proportion of food is comprised of protein (e.g. fish, meat, beans etc.)?

From this total amount of protein what % is fish/shellfish or other food caught from the sea?

% Food that is Protein	% Protein that is Seafood

O) What other activities do people do in this village for food and income?

P) Is fish important for any special celebrations, traditional activities or religion events? If yes please give examples of species and type of event.

Q) Are you aware of aquaculture (farming of fish, sea cucumbers or other animals and plants)?

R) Would you like to know more about any opportunities in aquaculture that can provide access to food or money? If yes what would be of interest and why?

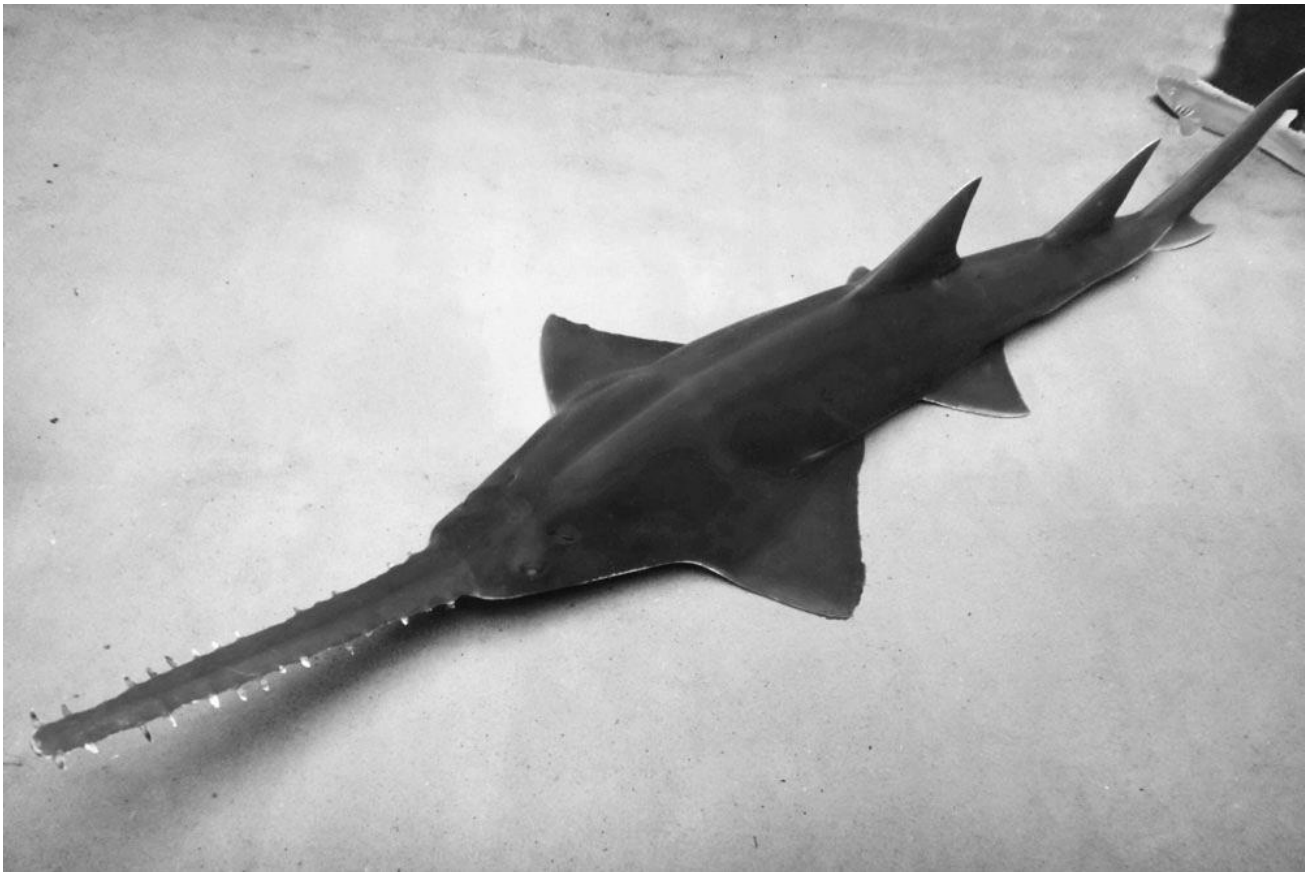
6 Feedback advice

A) Would you like to know about the results of the survey?

B) Can you give suggestions please on what is the best way to feedback the results of this project?

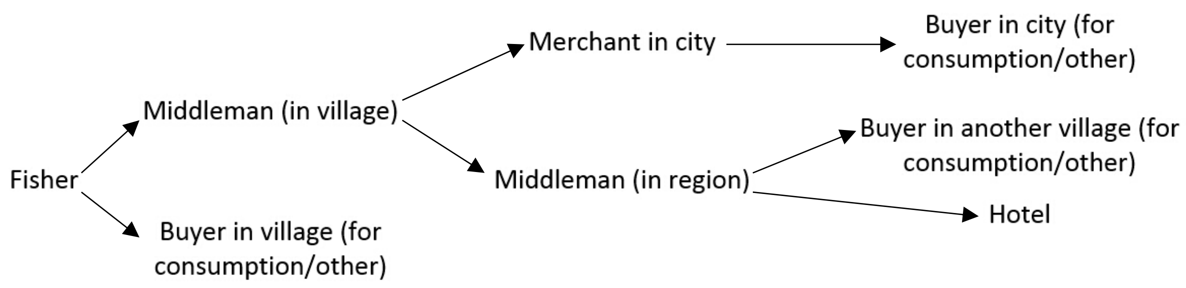
C) Do you have any suggestions on how to improve this survey?

Thank you for giving your time to share your views and experience, this is very much appreciated.



Sawfish

Sawshark – smaller, has barbells, gills on lateral side



Supply Chain (add any steps or notes given by the interviewee)

APPENDIX B. BYCAM Rapid Bycatch Assessment Survey.

Survey Ref (e.g ZAN 001)	**Date** (DD/MM/YYYY)	**Interviewer**	**Location/village**	**Occupation** (e.g. Fisher/Captain)

Introduction by interviewer

Hello, my name is [*e.g. Yussuf*]. I am part of an international team that wants to make a difference by helping communities in our country to have better access to food and money from fishing. We believe the best way to do this is to understand how the fishery operates from those involved. I understand a lot of surveys take place and take up your valuable time. We know some people are unhappy if they do not get results of the survey so we will make sure the end results are made available for everyone here.

The purpose of this survey is to learn about your fishing activities and if you are willing to share, to ask about your catches of certain types of animals from the sea in the past year (July 2016-June 2017) and to follow-up on work that has been done here in the past. This survey forms part of a larger project where we hope to improve support for fishing in this country. The senior scientists involved have given their time freely to help train people like me so together we can improve the way we manage our fishery.

We cannot guarantee to change the health of the fisheries but you sharing your experiences is an important start in identifying the current issues in fishing. Part of this survey will ask about animals which may be illegal to catch, to make sure that your honest answers do not have any consequences for you we want to assure you that any information you provide is confidential and cannot be traced back to you.

1. Fishing Practices

A) What types of fishing gears do **you use** and which months do you usually use these gears in? *[Circle months in which gear is used below]*

During the months you use each of these gears, **on average** how many days per month do you go fishing with them? *[e.g. when using driftnets they might fish 21 days a month]*

On **an average** trip **during these months** how many of these fishing gears do you take with you? *[For example, they may use multiple handlines from one vessel]*

Gear	Months Used	Average Fishing Days per Month	Number of Gears
Handline	J F M A M J J A S O N D		
Rod+Reel	J F M A M J J A S O N D		
Longline (pelagic)	J F M A M J J A S O N D		
Longline (demersal)	J F M A M J J A S O N D		
Basket Trap	J F M A M J J A S O N D		
Multifilament (Bottomset)	J F M A M J J A S O N D		
Monofilament (Bottomset)	J F M A M J J A S O N D		
Multifilament (Driftnet)	J F M A M J J A S O N D		
Monofilament (Driftnet)	J F M A M J J A S O N D		
Other? (please describe)			
	J F M A M J J A S O N D		
	J F M A M J J A S O N D		

2. Ray Questions

A) Have you ever caught rays using **your fishing gears**? *[Circle one, if answer is "No" or "Don't Know" skip to next section]*

Yes No Don't Know

B) Have you ever caught any Mobulid rays *[use local name if known]* using **your fishing gears**? *[Show illustrations, circle one response]*

Yes No Don't Know

C) Do you consider these two rays as the same or different types? *[Circle one response]*

Different Types Same Type

D) For each of the following rays types in the table *[Only ask about devil rays if the fisher said they have caught them. Only ask about Manta rays if the fisher says they consider them as different types to devil rays]*:

In the last year *[July 2016-June 2017]* **what months**, if any, did you catch them in?

How many, if any, did you catch in total last year with **<u>all of your gears</u>**? *[Ask for best estimate]*

How many of these, if any, did you catch with your **<u>main gear</u>**? *[Ask for best estimate]*

Type	Months Caught	All Gears Estimate	Main Gear Estimate
All Rays	J F M A M J J A S O N D		
Devil Rays	J F M A M J J A S O N D		
Manta Rays	J F M A M J J A S O N D		

<u>3. Shark Questions</u>

A) Have you ever caught sharks using **<u>your fishing gears</u>**? *[Circle one, if answer is "No" or "Don't Know" skip to next section]*

Yes No Don't Know

B) Have you ever caught any of these large sharks using **<u>your fishing gears</u>**? *[Show illustrations, circle one response]*.

Yes No Don't Know

C) For each of the following shark types in the table *[Only ask about hammerheads and whale sharks if the fisher said they have caught them]*:

In the last year *[July 2016-June 2017]* **what months**, if any, did you catch them in?

How many, if any, did you catch in total last year with **<u>all of your gears</u>**? *[Ask for best estimate]*

How many of these, if any, did you catch with your **<u>main gear</u>**? *[Ask for best estimate]*

Type	Months Caught	All Gears Estimate	Main Gear Estimate
All Sharks	J F M A M J J A S O N D		
Hammerhead	J F M A M J J A S O N D		
Whale Shark	J F M A M J J A S O N D		

4. Other Sharks/Rays Questions

A) Have you ever caught any of these other sharks and rays using **your fishing gears**? *[Show illustrations, circle one, if answer is "No" or "Don't Know" skip to next section]*

Yes No Don't Know

B) Do you consider these **"saw"** animals as the same or different types? *[Show sawfish and sawshark illustrations, point out size differences (generally sawshark much smaller), gill position differences and barbels in sawsharks, circle one response]*

Different Types Same Type

C) For each of the following other shark and ray types in the table *[Only ask about sawfish and sawsharks if the fisher says they consider them as different types]*:

In the last year *[July 2016-June 2017]* **what months**, if any, did you catch them in?

How many, if any, did you catch in total last year with **all of your gears**? *[Ask for best estimate]*

How many of these, if any, did you catch with your **main gear**? *[Ask for best estimate]*

Type	Months Caught	All Gears Estimate	Main Gear Estimate
Large Wedgefish	J F M A M J J A S O N D		
Sawsharks	J F M A M J J A S O N D		
Sawfish	J F M A M J J A S O N D		

5. Sea Turtle Questions

A) Do you know of any nesting areas *[areas where they lay their eggs]* for sea turtles?

B) Have you ever caught sea turtles using **your fishing gears**? *[Show illustrations, circle one, if answer is "No" or "Don't Know" skip to next section]*

Yes No Don't Know

C) For sea turtles *[Only ask if the fisher says they have caught them]*:

In the last year *[July 2016-June 2017]* **what months**, if any, did you catch them in?

How many, if any, did you catch in total last year with **<u>all of your gears</u>**? *[Ask for best estimate]*

How many of these, if any, did you catch with your **<u>main gear</u>**? *[Ask for best estimate]*

Type	Months Caught	All Gears Estimate	Main Gear Estimate
Sea Turtles	J F M A M J J A S O N D		

<u>6. Dolphin Questions</u>

A) Have you ever caught dolphins using **<u>your fishing gears</u>**? *[Show illustrations, circle one, if answer is "No" or "Don't Know" skip to next section]*

Yes No Don't Know

B) Do you consider these dolphins *[humpback dolphins]* as the same or different to these other types? *[Show illustrations, circle one response]*

Different Types Same Type

C) For each of the following dolphin types in the table *[Only ask about humpback dolphins if the fisher says they consider them as a different type]*:

In the last year *[July 2016-June 2017]* **what months**, if any, did you catch them in?

How many, if any, did you catch in total last year with **<u>all of your gears</u>**? *[Ask for best estimate]*

How many of these, if any, did you catch with your **<u>main gear</u>**? *[Ask for best estimate]*

Type	Months Caught	All Gears Estimate	Main Gear Estimate
All Dolphins	J F M A M J J A S O N D		
Humpback	J F M A M J J A S O N D		

7. Dugong Questions

A) Have you ever caught dugongs using **your fishing gears**? *[Show illustrations, circle one, if answer is "No" or "Don't Know" skip to next section]*

Yes　　No　　Don't Know

B) Have you caught dugongs using **your fishing gears** in the **last 5 years**? *[Show illustrations, circle one, if answer is "No" or "Don't Know" skip to next section]*

Yes　　No　　Don't Know

C) For dugongs *[Only ask if the fisher says they have caught them]*:

In the last year *[July 2016-June 2017]* **what months**, if any, did you catch them in?

How many, if any, did you catch in total last year with **all of your gears**? *[Ask for best estimate]*

How many of these, if any, did you catch with your **main gear**? *[Ask for best estimate]*

Type	Months Caught	All Gears Estimate	Main Gear Estimate
Dugong	J F M A M J J A S O N D		

D) *[Only ask if fisher says they have caught them]* In the last 5 years, how many dugongs have you caught in total with **all of your gears**? Where did you catch each of them? *[Ask for best location estimates]*

8. Other Information

A) Age?

B) Is fishing your primary *[main]* occupation?

C) What are your other occupations, if any?

D) On average, how many fishers are on your vessel when using your **main gear**?

E) What type of boat do you fish from?

F) How long, in metres, is the boat you fish from?

G) Is the boat motorised or it is propelled by other means? *[e.g. sail, paddle/oar]*

H) If your boat is motorised, what horsepower is the engine?

I) Who taught you how to fish/ Who introduced you to fishing?

J) What is the strangest animal you have ever caught or situation you have seen at sea?