Assessing the State and Impacts of the Artisanal Reef Fisheries and their Management Implications in Kenyan South Coast

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"Simon Peter saith unto them, I go fishing. They say unto him, we also go with thee. They went forth and entered into a ship immediately, and that night they caught nothing".

John 21:3 (KJV)

Abstract

Artisanal fisheries of tropical waters are estimated to harvest about 25% of the world`s fisheries catch. Despite this importance, a majority of tropical fish stocks remain unassessed and poorly managed. Reasons include a severe under-reporting of catches or the lack of reliable information of the fishery. With the growing concern over overexploitation and the challenge to assess fisheries status in these data-limited situations, a suit of assessment approaches have been proposed. In this study, we explore the usefulness of these data-limited approaches for the multi-species and multi-gear fishery of the Kenyan coast. The primary objective was to evaluate the current level and impacts of the fishery at the species and ecosystem level and to revise current management measures. In a first step, we used the Schaefer and Fox production models to estimate the sustainable catch and effort limits of the pooled catches for the entire coastal fishery and also explored possible changes in the mean trophic level of the catch by analysing officially reported time series data over sixty years. The results indicate that the current fish extraction and effort surpass sustainable limits (MSY) and that the mean trophic level of the catch has continuously declined over the years. In a second step, the size structure of currently obtained catches from the multigear fishery was studied based on a case study area of the Kenyan South coast. Results reveal that the multi-species fisheries' catches are dominated by small to medium-sized species and individuals. While these finding may indicate an unsustainable fishery, where older and larger fish have been serially depleted from the stock leading to a truncation of the size structure of the aggregated catches and a critical removal of large spawners, it is also possible that the observed pattern has emerged because of a fishers shift towards the smaller, more abundant and productive elements of the fished community. In this context, it is important to mention that catches from different gears overlap in species and sizes but also differ due to gear selectivity and spatial differences in gear use (inshore shallow lagoon

versus more offshore waters). In a third step, the exploitation rates of the four commercially most important target species of the fishery were determined using length–based single-species stock assessment approaches. Results suggest moderate to high mean exploitation rates for all species with low spawning potential ratios, supporting the results of the above analysis of an unsustainable fishery, with some species experiencing both growth and recruitment overfishing. In a fourth step, results from the single-species stock assessment were compared to those obtained from a holistic trophic model constructed for the study area. The results from the latter suggest that the system is in a perturbed (immature) state, likely due to the very intense resource exploitation. Overall catch volumes are relatively low (4.6 t Km^{-2} $year^{-1}$), and comparable to other intensively exploited coastal and coral reef ecosystems of the world. Our findings reveal that it may not be sufficient to rely on the current single-species management approaches such as gear restrictions and size limits for sustaining this multispecies fishery. Instead, control and reduction of the fishing effort and the establishment of specific areas closed to some fisheries may be needed if sustainable, ecosystem-based management is to be achieved. This should be done while considering the fishing impacts, the economic and social benefits within the ecosystem context.

Key Words: Artisanal fisheries, data-limited, multi-species, multi-gear, stock assessment, gear selectivity, ecosystem modelling, ecosystem-based management

Zusammenfassung

Schätzungsweise ein Viertel des weltweiten Fischfangs wird durch handwerkliche Fischereien in tropischen Gewässern erbeutet. Trotz dieser Bedeutung ist ein Großteil der tropischen Fischbestände nicht analytisch erfasst oder schlecht bewirtschaftet. Oft trifft sogar beides zu. Gründe hierfür sind unter anderem die Unterschätzung der Fangmengen oder das grundsätzliche Fehlen von Informationen über die Fischerei. Angesichts der wachsenden Besorgnis über die Übernutzung von Fischbeständen und der Herausforderung, den Fischereistatus in diesen datenbeschränkten Situationen zu bewerten, wurde eine Reihe von Bewertungsansätzen vorgeschlagen. In dieser Studie untersuchen wir den Nutzen dieser datenbegrenzten Ansätze für die Mehrartenfischerei vor der kenianischen Küste, in welcher unterschiedliche Fanggeräte zum Einsatz kommen. Das Hauptziel bestand darin, den derzeitigen Fischereidruck und seine Auswirkungen sowohl auf einzelne Arten-, als auch auf Ökosystemniveau zu bewerten und die derzeitigen Managementmaßnahmen vor diesem Hintergrund zu evaluieren. In einem ersten Schritt haben wir basierend auf offiziellen Fischereistatistiken der letzten 60 Jahre Populationsmodelle nach Schaefer und Fox parametrisiert, um nachhaltige Fang- und Aufwandsgrenzen der Gesamtfänge für die gesamte Küstenfischerei abzuschätzen und um mögliche Veränderungen der mittleren trophischen Ebene der Fänge zu analysieren. Die Ergebnisse zeigen, dass die derzeitigen Fangmengen und der Fischereiaufwand das Level des maximalen Dauerertrags (MSY) übersteigen und dass die mittlere trophische Ebene der Fänge im Laufe der Jahre kontinuierlich abgenommen hat. In einem zweiten Schritt wurde die Größenstruktur der letztjährigen Fänge in der gemischten Fischerei der kenianischen Südküste untersucht. Die Ergebnisse zeigen ein größenbezogenes Nutzungsmuster der Mehrartenfischerei, wobei in den Fängen kleine bis mittelgroße Arten und Individuen dominieren. Diese Befunde könnten auf eine nicht nachhaltige Fischerei hindeuten, bei der ältere und größere Fische bereits

aus dem Bestand entfernt wurden, was zu einer Verjüngung der Größenstruktur und der kritischen Überfischung großer Laicher führte. Es ist aber auch möglich, dass das beobachtete Muster entstand, weil sich die Fischerei auf kleinere, häufigere und produktivere Arten und Individuen fokussiert. In diesem Zusammenhang ist es wichtig zu erwähnen, dass sich die Fänge der Fanggeräte in Arten- und Größenzusammensetzung zwar zum Teil überschneiden, sich aufgrund verschiedener Selektivitäten und räumlicher Unterschiede der Fanggerätenutzung aber auch Unterschiede ergeben. In einem dritten Schritt wurden der Fischereidruck auf die vier kommerziell wichtigsten Arten anhand von längenbasierten Einarten-Bestandsberechnungen ermittelt. Die Ergebnisse deuten auf mittlere bis hohe Befischungsraten hin, mit niedrigen Laichpotenzial-Verhältnissen, was die Ergebnisse der oben genannten Analyse einer nicht nachhaltigen Fischerei stützt. Sowohl Rekrutierungs- als auch Wachstums-Überfischung tritt hierbei auf. In einem vierten Schritt wurden die Ergebnisse der Einarten-Bestandsberechnungen mit denen eines Nahrungsnetz-Modells des Untersuchungsgebietes verglichen. Ergebnisse des Nahrungsnetz-Modells deuten darauf hin, dass das System, womöglich aufgrund der intensiven Ausbeutung lebender Meeresressourcen, in einem gestörten Zustand ist. Die Gesamtfangmengen sind im globalen Vergleich relativ gering (4,6 t km^{-2} $Jahr^{-1}$), sie liegen etwa auf dem Niveau anderer intensiv genutzter Küsten- und Korallenriff-Ökosysteme. Unsere Ergebnisse zeigen, dass es zur Erhaltung dieser Mehrartenfischerei möglicherweise nicht ausreicht, sich auf einzelbestands-spezifisches Managements zu verlassen, wie etwa Beschränkungen bestimmten Fanggeschirrs oder Größenbeschränkungen. Für eine nachhaltige, ökosystembasierte Bewirtschaftung scheint vielmehr die Kontrolle und Reduzierung des Fischereiaufwands und die Einrichtung bestimmter Gebiete, die für einige Fischereien gesperrt sind, zielführender sein. Dies sollte unter Berücksichtigung der fischereilichen Auswirkungen und der wirtschaftlichen und sozialen Vorteile im Ökosystemkontext erfolgen.

Schlagwörter: handwerkliche Fischerei, datenlimitiert, Mehrartenfischerei, gemischte Fischerei, Bestandsberechnung, Selektivität von Fanggeräten, Ökosystemmodellierung, ökosystembasiertes Management

Ikisiri

Uvuvi mdogomdogo kwenye maji ya maeneo ya kitropiki unakadiriwa kutoa zaidi ya 25% ya kiasi cha samaki kinachozalishwa duniani. Licha ya umuhimu huu mkubwa, sehemu kubwa ya samaki wanaopatikana kwenye tropiki hawajafanyiwa uchunguzi na usimamizi wao sio mzuri. Sababu kadhaa zimepelekea hali hii ikiwemo kutokuwepo kwa mfumo wa utoaji wa taarifa za kiasi cha samaki wanaovuliwa na ukosefu wa takwimu za kuaminika kuhusu uvuvi. Kuongezeka kwa uhitaji wa kukabiliana na uvunaji uliopitiliza na changamoto za kufuatilia na kuchunguza hali ya uvuvi kwenye mazingira yenye takwimu haba, kumepelekea kuwepo kwa mapendekezo kadhaa ya njia za ufuatiliaji na uchunguzi. Utafiti huu umejielekeza kutazama umuhimu na mchango wa hizo njia zitumiazo takwimu haba kwenye uvuvi wa samaki wa aina tofauti tofauti na utumiaji wa zana mbalimbali kwenye ukanda wa pwani ya Kenya. Lengo kuu la utafiti huu ni kutathmini kiasi cha sasa cha uvuvi na athari zake kwenye aina mbalimbali za samaki na mfumo ikolojia ili kufanya mrejesho wa shughuli za usimamizi wa rasilimali hizo. Hatua ya kwanza ya utafiti huu imehusisha utumiaji wa mfano wa Schaefer na Fox ili kukaridia kiasi endelevu kinachoweza vunwa na kupunguza jitihada za uvuvi kwa ukanda wote wa pwani. Hatua hii imehusisha pia ufuatiliaji wa wastani wa kiwango cha trofiki cha samaki wanaovuliwa kwa kuchanganua takwimu za zaidi ya miaka sitini iliyopita. Matokeo yanaonyesha kuwa kiasi cha sasa cha uvuvi na jitihada zinazotumika kimezidi kiwango endelevu cha juu (MSY) na kwamba wastani wa kiwango cha kitrofiki kimeendelea kupungua kadiri miaka inavyokwenda. Kwenye hatua ya pili ya utafiti huu, muundo wa kiasi cha samaki wanaovuliwa kwa sasa kutokana na zana tofauti tofauti uliangaliwa kulingana na maneneo yaliyochanguliwa kwenye ukanda wa pwani ya Kenya. Matokeo yanaonyesha kuwa ukubwa na kiasi cha uvunaji kwenye uvuvi unaotumia zana anuwai kwa sehemu kubwa unajazwa na samaki wadogo wadogo na wale wa kiasi cha kati. Ingawaje

matokeo yanaonyesha dalili za uvuvi usio endelevu, ambapo samaki wakubwa na waliofikia umri wa kuvuliwa hawapo tena na hivyo kupelekea badiliko kwenye ukubwa wa samaki wanaopatikana na wale walio kwenye kiwango cha juu ya trofiki, huenda pia hali hii imetokea kwa sababu wavuvi wameelekeza shughuli zao kwenye samaki waodogo wadogo na ambao kwa sasa wanapatikana kwa wingi zaidi. Kwa muktadha huu, ni muhimu kueleza kuwa kiasi cha samaki wanaovuliwa na zana tofauti tofauti wanamuingiliano wa kiaina na ukubwa na pia hutofautiana kulingana uwezo wa ukamataji wa zana na mahali zinapotumika (mfano kwenye madogo/rasi na maji makubwa). Hatua ya tatu ya utafiti huu ilihusu uainishaji wa kiwango cha uvunaji wa aina nne za samaki wenye umuhimu mkubwa kibiashara na ambao hutafutwa sana na wavuvi kwa kutumia njia ya urefu wa kwa aina moja tu ya samaki. Matokeo kwenye hatua hii yameonyesha uvunani wa kiwango cha kati hadi wa juu kwa aina za samaki ambao utagaji wao ni wa kiwango ambacho huwa cha chini (SPR). Hali hii inakubaliana na matokeo yaliyoonesha kuwa uvuvi kwenye ukanda huu wa pwani sio endelevu kama ilivyoanishwa hapo juu ikiwemo aina kadhaa za samaki wake wakionekana kukabiliwa na athari ya uvuvi uliopitiliza uhusuyo ukuaji na uzalishwaji. Hatua ya nne ya utafiti huu ilihusisha ulinganishaji wa matokeo yaliyopatikana kwa kutumia njia ya aina pekee ya uchunguzi na yale yaliyopatikana kwenye mfano wa njia sahihi ya kitrofiki kutoka kwenye eneo la utafiti. Kwenye mfano wa njia sahihi ilionekana kuwa mfumo ulikua haujakomaa, sababu kubwa ikiwa ni kutokana na shughuli za uvuvi uliopitiliza kiasi. Kwa ujumla, ukubwa wa kiasi kinachovuliwa ulikuwa mdogo (4.6 t Km^{-2} $mwaka^{-1}$), hali inayoelekea kulinganishwa na sehemu nyingine za pwani na zile za maeneo ya matumbawe zilizovuliwa sana duniani. Matokeo ya utafiti huu yanaonyesha kuwa shughuli za usimamizi wa rasilimali za pwani na hasa zile za uvuvi hazitafanikiwa ikiwa njia itakayotumika itakuwa ni ile ya usimamizi wa aina moja tu ya samaki, kama vile uzuiazi wa zana fulani na kuruhusu ukubwa fulani wa samaki ili kuwa na uvuvi endelevu kwenye uvuvi unaohusisha aina anuwai za samaki. Badala yake, pawepo na usimamizi na upunguzaji wa jitihada za uvuvi na uanzishwaji wa

maeneo ambayo yatafungwa kwa ajili ya uvuvi fulani fulani ikiwa lengo ni kuwa na usimamizi endelevu wa mfumo ikolojia. Hili lifanyike pamoja na kuhakikisha kuwa athariza uvuvi pamoja na faida za kiuchumi na kijamii kwenye mfumo ikolojia unaohusika nazo zinazingatiwa na kupewa kipaumbele.

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from His mouth that I got the knowledge and understanding to have achieved this much.

CHAPTER 1.
General Introduction

1.1 GENERAL STATE OF FISHERIES

Fisheries contribute significantly to the nutrition and economic needs of about 10–12% of the world's population(Lam et al., 2016), yet the majority of fish stocks are under crisis due to increased exploitation and the failures of fisheries management approaches (Beddington et al., 2007; Christensen and Pauly, 1998; Hilborn et al., 2003; Worm and Branch, 2012). While there is much consensus that global limits to exploitation may have been reached (Kleisner et al., 2013; Pitcher and Cheung, 2013; Worm, 2016), there are also differing opinions regarding the exact trend and projection of global fisheries (Costello et al., 2012; Mace, 2004; Pitcher and Cheung, 2013; Worm and Branch, 2012).

These differences relate partly to the reliance and interpretation of different data sources to infer global stock status. For instance, on the basis of catch and landings data accounting for more than 80% of the globally reported fish landings (Pauly et al., 2013), the United Nations Food and Agriculture Organization (FAO), indicate that fishery production has remained relatively static since the late 1980s (Figure 1a). However, the number of fish stocks fished within biologically sustainable levels has decreased (Figure 1) (FAO, 2016). Of all fish stocks assessed, approximately 60% produced lower yields than their biological potential (FAO, 2016). Observed trends suggest that more fish stocks are likely to be overexploited in the future with the proportion of underexploited stocks (10.5%) expected to decrease further.

Despite representing the best possible global view of the fishery status, there are misgivings about the quality of the FAO data (Pauly and Zeller, 2017; Worm and Branch, 2012). A comparison of the global fisheries trend as reported by FAO and that of the catch reconstructed reveal different trajectories in catch trends (Pauly and Zeller, 2016), which helps to highlight the discrepancies that exist in most of the officially reported catches. According to the results of the reconstructed global catch data, the world catches have been declining at a rate of 1.2 million tonnes annually (Pauly and Zeller, 2016). What is perceived as a levelling of catch (1998-2010) in the

FAO dataset is attributed to the biases of the time series data, where reasonably accurate data are compensated with unreliable data from countries with highly questionable data (Pauly and Zeller, 2016; Zeller et al., 2015).

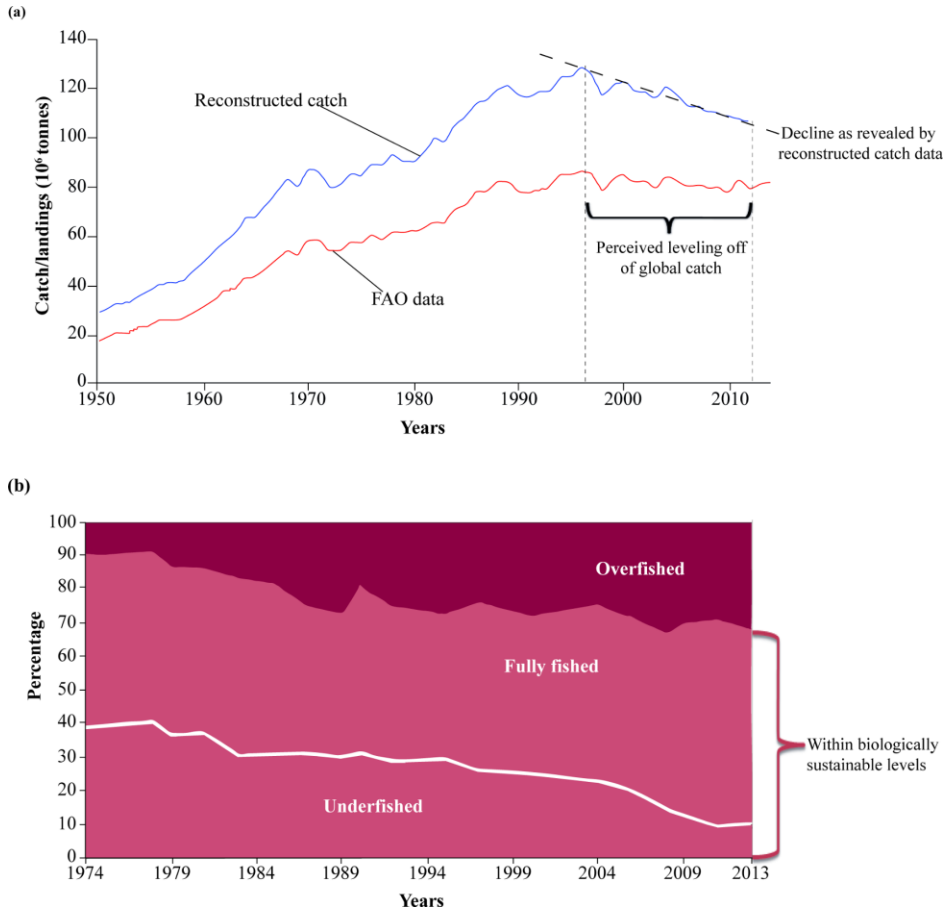

Figure 1. Time series of global marine fisheries catches during the period 1950–2014 from (a) FAO and reconstructed catch for 1950–2010 (Data source: Pauly and Zeller (2016c) and (b) most recent status of marine fisheries stocks as presented by FAO (2016).

The underreporting of catches (amounting to 13 - 32%) is attributed to several factors including the omission of the artisanal fisheries data, bycatch and discards data from the FAO dataset. Therefore, caution should be taken when making global inference on fisheries status exclusively based on catch data, notably when information regarding data acquisition methods are lacking (Gallardo Fernández et al., 2011; Hilborn, 2003). The alternative approach, which relies on independent fishery surveys is widely regarded as the "gold standard" for assessing fish stocks (Carruthers et al., 2014; Kleisner et al., 2013; Pauly et al., 2013). The general impression is that, catch data is not a good proxy for fish abundance and as such the

health of the stocks should, whenever possible, be based on scientific stock assessment data (Hilborn, 2003; Pauly et al., 2013). However, scientific stock assessments are expensive and often limited to fisheries of a few developed countries. It is particularly challenging for developing countries, which despite contributing significantly to the global fisheries, are often data poor (Bray, 2001; Worm and Branch, 2012; Zeller and Pauly, 2007). Therefore, the evaluation of the state of the world's fisheries based on independent data as attempted by Worm and Branch (2012), may also be problematic as only a small percentage of the worlds fish stocks have been scientifically assessed (Apel et al., 2013).

Despite the lack of consensus on the exact status or future projections of the global fishery, there seems to be convergence that the current state of fish stocks is indeed worrying and that there is a need to put measures in place to avert further deterioration (Worm et al., 2009). Therefore, stock rebuilding efforts (Murawski et al., 2007) are undertaken, and alternative ways to improve data collection and assessment of fisheries are explored to ensure that management decisions are guided by reliable information (Pauly et al., 2013). Effective fisheries management ought to be based on reliable information regarding the fisheries status, habitat integrity as well as the information about the resource users (Jennings and Polunin, 1996).

Unfortunately, most tropical fisheries do not have the comprehensive information needed and alternative assessment approaches may have to be adopted that are mindful of the kind of data available. The challenge is to make the best use of all the available fisheries information when assessing fish stocks to inform fisheries management decisions (Apel et al., 2013; Kleisner et al., 2013; Vasconcellos et al., 2005). Such alternatives may be less precise than classical, data-rich stock assessment but would still need to give a fair representation of the fishery status to ensure that appropriate measures are put in place to avoid further deterioration of the fisheries trends (Worm et al. 2009; Kleisner et al., 2013).

1.2 ASSESSING DATA-LIMITED FISHERIES

The growing concern over marine resource overexploitation and the inability to adequately assess the extent of resource exploitation of most fish stocks due to insufficient data has generated much interest in alternative "data-poor" assessment approaches (Apel et al., 2013; Vasconcellos et al., 2005). These approaches are centred on fisheries that lack reliable biological data or/and on those whose collected data are unreliable or under-analysed due to lack of resources and or personnel (Honey et al., 2010). In the case of small-scale tropical fisheries, data limitation may arise from the existence of multiple target species of otherwise mixed fisheries (Pilling et al., 2009). This means that even though the data may be available, the catch statistics are grouped into guilds, which limits the usefulness of the data and application of traditional single-species stock assessment techniques applicable to data-rich fisheries in temperate regions (Caddy and Garibaldi, 2000; Fujita et al., 2014).

The Western Indian Ocean region (WIO) is reported to be one of the regions with the highest percentages of nonspecific landings (i.e., landings not defined to species), where over 60% of the landings are reported in highly aggregated items (Vasconcellos et al., 2005). It is therefore not surprising that it is also one of the regions with the lowest number of fully assessed fisheries and with the highest proportions of (presumably) fully-exploited stocks (Costello et al., 2012; UN, 2010). However, while the overall catch landed in the WIO region has been shown to be increasing, the exact status of many fish stocks is still uncertain given the high number of unreported (and unassessed) fisheries (De Young, 2006). It is therefore difficult to see how the prospect of sustainable fisheries can be achieved without a realistic appraisal of the current fisheries status.

To cope with the uncertainty regarding stock status due to limited or inadequate data, there is an increasing support among fisheries scientists to apply data-limited approaches, provided that at least some relevant and informative data exist (Apel et al., 2013; Fujita et al., 2014; Honey et al., 2010). These approaches are diverse and can be applied to data ranging from simple pooled catch and effort data to more complex

information on species and size composition of the catches (Pilling et al., 2009). Through the novel use of these approaches, the evaluation of the affected fisheries will be enhanced and may lead to a better representation of the tropical fisheries in global estimates.

1.3 CASE STUDY: KENYAN COASTAL FISHERY

1.3.1 Description of the artisanal fisheries

The Kenyan coastline, located in the Western Indian Ocean (WIO) is approximately 640 km long, with the territorial sea and adjacent Exclusive Economic Zone (EEZ) covering some 152,100 Km^2. The Kenyan coast is profoundly influenced by the monsoon winds, which results in two distinct seasons, which in turn affect the fishing activities. During the south-east monsoon season locally referred to as Kusi (May–October) access to fishing grounds are restricted by the strong winds and rough sea conditions as compared to the calmer north-east monsoon locally known as Kaskazi (McClanahan, 1988; Obura, 2001).

The artisanal fishery is one of the most important economic activities providing food and livelihoods opportunities to the coastal communities and is primarily concentrated along the coastal and nearshore areas and is hardly undertaken in the offshore areas. At present, there are approximately 14,000 fishers directly employed in the sector with an additional 20,000 indirectly involved in the service and auxiliary sectors such as net making and boat construction (FID, 2016). However, the contribution of fisheries to the country's gross domestic product (GDP) is low (about 0.5%), which is a major factor, which restricts the development of the fisheries in Kenya (Aloo et al., 2014). This low macro-economic relevance also contributes towards the low priority that the fisheries sector receives at the national level as compared to other sectors. Consequently, very little investment has been allocated to the monitoring and assessment of the fisheries, which has resulted in under-reporting and the lack of measures for sustainable fisheries management

(Kaunda-Arara and Rose, 2004; Le Manach et al., 2015; McClanahan and Kaunda-Arara, 1996).

In general, the fishery is characterised by the use of relatively simple fishing gears and vessels, which limit the fishers to shallow nearshore environments such as the coral reef and the seagrass beds allowing the fishers to travel back and forth in a single day leading to a large pressure on those coastal resources (McClanahan & Muthiga 1988). As a result of the small daily catch per fisherman, the returns from the fisheries are low, and most catches are landed daily at the fish landing sites and are either sold locally or consumed at the household level. Typical fishing gears used include traditional gears such as basket traps as well as more modern gears such as spear guns, hand lines beach seines, ring nets and gill nets (Alidina, 2005; Obura, 2001; Samoilys et al., 2011). Also, the fishery is multi-species with the fishers exploiting over 200 species, which further contributes to the complexity of assessing and managing these resources (McClanahan and Mangi, 2004; Roberts and Nicholas, 1993).

Human activities in the form of fishing are considered as the major threats to the ecosystem and harvested stocks. Two main factors - increased capacity in the fisheries sector, and continued use of gears perceived to be destructive such as the beach seines, spearguns and small-meshed gill nets (Hicks and McClanahan, 2012). These among other factors have been highlighted as directly contributing towards the high exploitation of fisheries resources and thereby contributing towards the decline of Kenya's coastal and marine fisheries (Mangi and Roberts, 2006; McClanahan et al., 1997). According to Kaunda-Arara et al. (2003), there has been a rapid decline in the overall catch landed particularly for the demersal species, which constitute about 38% of the total marine catch.

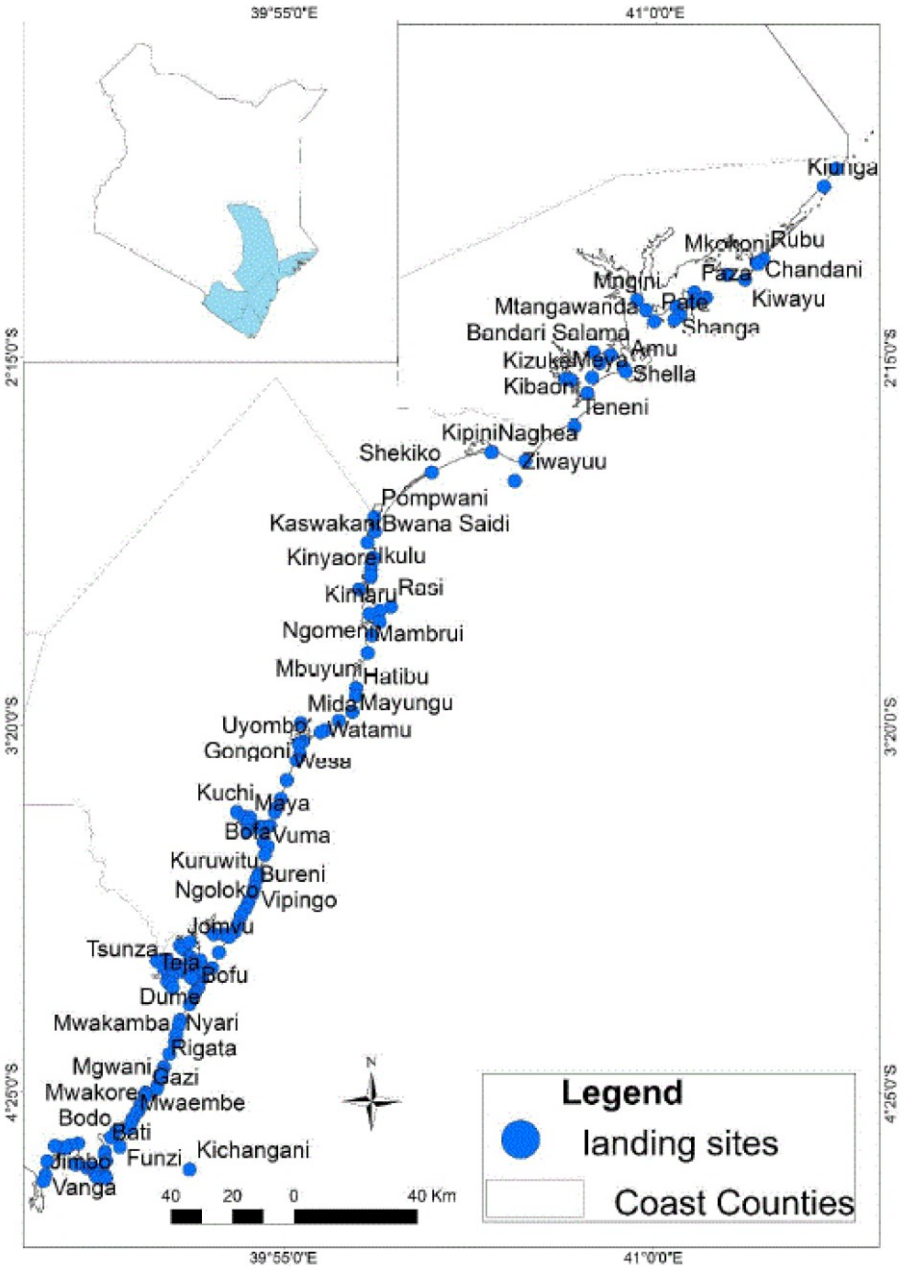

Figure 2. Map of the Kenyan coastline with the mapped fish landing sites. Source: (FID 2016)

Despite the notion of a general deterioration of fish stocks in the country, the precise status of the marine fisheries remains mainly unknown due to lack of or inadequate fisheries statistics (FID, 2014; Fondo, 2004; Kaunda-Arara et al., 2003). Traditionally, the state department of fisheries has direct authority over fisheries resources as well as the mandate to collect the fisheries-related data. However, like in most tropical fisheries in developing countries, the lack of resources and expertise has dramatically hampered the monitoring of the fisheries (Kaunda-Arara and Rose, 2004). Therefore,

the actual state of the coastal artisanal fisheries has not been evaluated sufficiently and alternative approaches or sources of information must be examined (Pilling et al., 2009).

Given the above, the primary objective of this study is thus to assess the current status of the artisanal fisheries along the Kenyan coast based on fisheries dependent data to determine the sustainability of current fishing practices in the artisanal fisheries. As a first step (Chapter 1), the study reviews the current status of the Kenyan artisanal fisheries using fisheries landings statistics from the state department of fisheries of Kenya and describes the evolution of the artisanal fisheries in terms of annual yield catch, and catch per unit of effort using the surplus production methods. Moreover, the mean trophic level of the aggregated catch is used as a proxy for changes in catch composition over the past decades due to fishing. In a second step (Chapter 2), the size structure of the catch is examined based on the length-frequency distributions by species and gear sampled over a one year period (2014-2015) from a localised fishery in Diani-Chale (Figure 3), to examine the impact of gear selectivity. The area is considered one of the most degraded and over-fished coral systems in the East African coast (McClanahan and Muthiga, 1988; McClanahan and Obura, 1995). While it was designated as a marine reserve in the early 1990s, the plan was resisted by the local fishers for fear of losing one of their most important fishing grounds (King, 2000; McClanahan et al., 1997).

1.3.2 Description of the study site

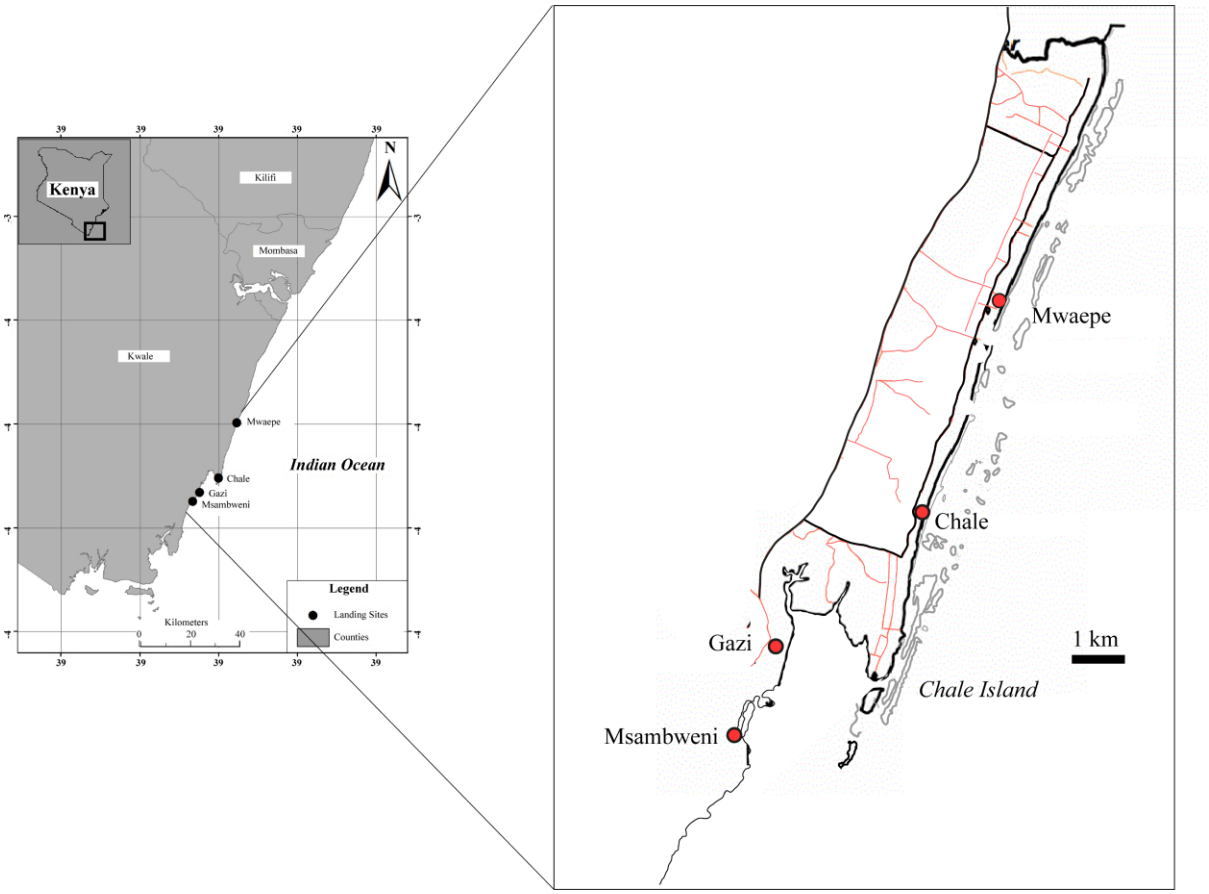

Figure 3. Map of the Diani-Chale area, south of Mombasa, Kenya showing the four selected landing sites. Map adapted from Obura (2001).

Diani-Chale area is an example of an open-access fishery, where the artisanal fishers use both legal and illegal gears targeting multiple species and a range of fishing grounds (McClanahan and Mangi, 2004). In a third step (Chapter 3), we apply length-based stock assessment routines to assess the exploitation levels of those dominant species, which represent a significant portion of the overall catch in terms of abundance and biomass. To deal with the multi-species problem, only these most commercially important, most abundant (in terms of numbers and biomass) species were assessed and the judgement of the overall state of the fishery was based on their status. Finally, (Chapter 4), we explore/demonstrate the usefulness of combining approaches such as the traditional stock assessment and trophic model to quantify fishing impacts on the whole ecosystem and compare the status to set ecological indicators using the Ecopath with Ecosim (EwE) framework.

1.4 RESEARCH QUESTIONS

I. What is the current status of the Kenyan artisanal fisheries? Can we infer the state of the fishery using aggregated catch and effort data? (*Manuscript 1*)

II. What is the role of fishing gears for the exploitation of the different species and sizes in the multispecies multi-gear fishery? (*Manuscript 2*)

III. What is the current status of the commercially important species in the artisanal fisheries? Are they already overexploited? (*Manuscript 3)*

IV. How do results of the single species stock assessment compare to those obtained from holistic ecosystem modelling in assessing fishing impacts? *(Manuscript 4*)

1.4.1 Thesis outline

The thesis has been structured into six chapters, a general introduction that sets the current study in the global context (**chapter 1**), four chapters that deal with specific research questions structured as peer-reviewed articles (**Chapter 2 to Chapter 5**) and a general discussion (**Chapter 6**), which summarises the results, discusses their implications and provides recommendations for future management

In **chapter 1**, the study introduces the global problem of overfishing and the challenges associated with inferring the global fishery status using different data sources. **Chapter 2** (*manuscript 1*), provides a historical context of the Kenyan artisanal fisheries, by describing the changes that have occurred in the catch and effort data. In **chapter 3** (*manuscript 2*), the size structure of the artisanal fishery catches at the southern shores of the Kenyan coast is analysed using multivariate descriptive techniques to highlight the complexity that exists in this multispecies and multi-gear fishery, where fishers use multiple gears to target multiple species of different sizes at multiple sites.

Chapter 4 (*manuscript 3*), represents the comparative application of two length-based single-species fishery stock assessment techniques (length converted catch curve and the length based spawning potential ratio) to infer the current status

of the target species of the artisanal fisheries based on biological thresholds to provide insights on the species' vulnerability to exploitation. C**hapter 5** (*manuscript 4*), uses a holistic trophic modelling approach to assess the ecological impacts of fishing on the ecosystem and to compare the results with those of single-species stock assessment approaches (Chapter 4).

The final chapter, **Chapter 6,** provides a synthesis of the entire study and discusses the main findings in relation to other local and global studies with recommendations for management and avenues for possible future research.

1.4.2 List of manuscripts and authors' contributions

Manuscript I: Tuda, P.M and Wolff, M (2015). Evolving trends in the Kenyan artisanal reef fishery and its implications for fisheries management. ***Ocean & Coastal Management***. 104:36-44.

Contributions: The concept was devised by **PT** and **MW**. **PT** consolidated all the data from the relevant sources and conducted all the data analysis. **PT** wrote the first draft of the manuscript with subsequent contributions from **MW**.

Manuscript II: Tuda, P.M, Wolff, M, Breckwoldt, A (2016). Size structure and gear selectivity of target species in the multispecies multigear fishery of the Kenyan South Coast. *Ocean & Coastal Management*. 130:95-106.

Contributions: The concept was devised by **PT** and **MW**. **PT** planned and conducted the data collection, entry and final analysis. **PT** wrote the first draft of the manuscript with subsequent contributions from **MW and AB**.

Manuscript III: Tuda, P.M, Breckwoldt, A Wolff, M, (2017). Adapting length-based stock assessment for improved management of four coral reef target fish species in Kenya. *Submitted to Fisheries Management and Ecology*

Contributions: The concept was devised by **PT** and **MW**. **PT** planned and conducted the data collection, entry and final analysis. **PT** wrote the first draft of the manuscript with subsequent contributions from **MW** and **AB.**

Manuscript IV: Tuda, P.M and Wolff, M. (2017). Comparing a holistic (trophic modelling) approach with singles species stock assessment: the case of Gazi Bay, Kenya. **Accepted with minor revisions** *Journal of Marine Systems.*

Contributions: The concept was devised by **PT** and **MW**. **PT** compiled the data and constructed the initial model with suggestions from **MW**. **PT** wrote the first draft of the manuscript with improvements from **MW.**

CHAPTER 2.

Evolving trends in the Kenyan artisanal reef fishery and its implications for fisheries management

Paul M. Tuda and Matthias Wolff

This is the author's version of the work. Please cite the final version as:

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ABSTRACT

The contribution of artisanal fisheries to the livelihoods of coastal communities in developing countries is great and is expected to rise given the increasing coastal population. However, the marine capture fishery in Kenya is small and only contributes 4% to the overall fish production. This contribution of the coastal artisanal fisheries in Kenya is small compared to the national fisheries production. In Kenya, the coastal artisanal fisheries have as yet received little attention due to the limited understanding of its contribution to coastal livelihoods. A review of the coastal artisanal fisheries landings for the past sixty years indicates that significant changes have occurred in the fisheries. There has been an increase in effort evidenced by the increased number of fishers, fishing vessels and fishing technology. Overall the landings have remained relatively stable over the past decade fluctuating between 5,000 tones and slightly more than 8,000 tonnes annually which is within the range of the predicted sustainable limit of the fishery based on both the Schaefer and fox model prediction of the MSY (maximum sustainable yield). Our estimate of MSY (8,264-8,543 tonnes) and the corresponding effort of 11,171-15,467 fishers, derived from the Schaefer and Fox models, would suggest that yields higher than the presently obtained levels cannot be expected in future and that the interannual variation in total landings may have to do with environmentally triggered changes in resource productivity. The model results also suggest that the overall effort of the present fishery already exceeds sustainable effort levels by at least 20% suggesting a general state of overfishing.

Keywords: Artisanal fishery, effort, catch data, data-limited, reef-fish

2.1 INTRODUCTION

Artisanal fishery remains one of the most critical livelihoods associated with the coral reef ecosystems in tropical countries. Estimated to yield approximately 6 million tonnes annually, the artisanal fishery contributes significantly towards the livelihoods of over 200 million people (Munro, 1996; Teh et al., 2013). The contribution is particularly high in developing countries and is expected to rise given the predicted further increase in population for most coastal cities (Allison and Ellis, 2001), which is expected to exacerbate the pressure on the coral reefs. Nevertheless, artisanal fisheries remain neglected, and their contribution overlooked (Pauly, 2006; Worm et al., 2009). On the other hand, there is a growing concern over fisheries in general, considering that fish stocks have globally significantly been impacted by fishing (Hilborn et al., 2003; Myers and Worm, 2003).

Artisanal fisheries due to their scale of operation have often been considered benign. Characterised by the use of relatively simple fishing gears and small, often not even motorized vessels, the impacts of artisanal fisheries have often been underestimated in comparison to industrial fisheries (Hawkins and Roberts, 2004). However, there is mounting evidence suggesting that artisanal fisheries can have severe impacts on coral reefs and fish communities (Mcclanahan, 1994). In the East African coast, the explosion of sea urchin populations and the decrease in fish size and biomass is considered to be the direct cause of fishing (Mcclanahan and Muthiga, 1988; Mcclanahan and Shafir, 1990). Similarly, fishers density was found to affect the ecological state of the coral reefs leading to a lower trophic level of the catches and a decrease in the size of target resources (Teh et al., 2013).

However, the lack of independent monitoring data limits the extensive evaluation of the full impacts of artisanal fishing making it difficult to get a realistic understanding of the changes that have occurred in the fishery (Tesfamichael and Pauly, 2011). As such, most of the artisanal reef fisheries remain poorly managed and monitored and their impacts are thus difficult to assess (Pauly, 2006; Sadovy, 2005; Worm et al., 2009). In Kenya, the coastal artisanal fisheries have as yet received little

attention due to the limited understanding of its contribution to coastal livelihoods (Malleret-King et al., 2003). Overall, fish production of Kenya contributes only about 0.5 % to the national Gross Domestic Product (GDP) with an estimated annual production of 200,000t, and a value of over Ksh 4 billion (US$ 50 Million) in foreign exchange earnings (GOK, 2013). However, the marine capture fishery is small and only contributes 4% to the overall fish production (FID, 2014).

The fishery is mainly artisanal and is operated within an area of about 800 km^2 and is based on a small number of demersal coral reef- and seagrass-associated fish species (McClanahan and Mangi, 2004; Obura, 2001). The total annual catch landed has been oscillating between 5,000 and 8,000t and is less than half of the estimated potential yield for the inshore fishery (~20,000t per year) (Odero, 1984; Sanders et al., 1988). The Kenyan reefs are considered to be among the most heavily exploited reefs in East Africa with some considered overfished well above the maximum sustainable yield (MSY) level (Malleret-King et al., 2003; McClanahan et al., 1997; McClanahan and Obura, 1995). There are indications of declining trends in overall landings paralleled by an increase in catch contribution by pelagic fish and invertebrates (Obura, 2001). This mainly reflects increased levels of fishing effort, number of landing sites as well as changes in fishing techniques over the past few decades which may have contributed to the decline in catch volumes and change in target species (McClanahan & Mangi, 2004).

Fisheries are regulated by gear restrictions, and despite the banning of the beach seine and spear gun already in 1990, the use of these gears has continued (and even increased in some areas) putting great pressure on the reefs (McClanahan and Mangi, 2004). Nevertheless, due to a lack of enforcement of these regulations, the use of these gears has continued (and even increased in some areas) putting great pressure on the reefs (GOK, 2013). Compounded by the absence of reliable fisheries data there is some evidence that the declining catch per unit effort (CPUE) recorded in the degraded reefs could be an indicator of a declining fishery (Obura, 2001). Most of the studies on the status of the artisanal fishery have been based on data from

independent research institutions that give an insight into the fishery but are restricted only to specific areas. The catch data collected by the government agencies have been cited to possibly greatly underestimate overall catches by almost half (Malleret-King et al., 2003).

Nevertheless, these data remain the only long-term information available that is representative of the entire fishery, and it is this data that is submitted to the United Nations Food and Agricultural organization (FAO), on which the global fisheries statistics are based. Despite the apparent concern over the reliability of these data, they provide the basis upon which the health of the fishery is to be inferred. The here presented study seeks to analyze the fishery trends from the artisanal fishery in Kenya based on the official catch data to identify the changes that may have occurred over the past decade(s) and to estimate the current harvest level for the target species. Based on these analyses, recommendations for management shall be formulated. Our study responds to the growing interest in the use of simple fisheries analysis tools that are based on catch data from data-poor fisheries. We expect our contribution to be important for policy and management decisions considering that the number of people living in the coastal areas is most probably going to rise, as is the fishing effort.

2.2 METHODS

The usefulness of catch and effort data compared to independent fishery survey data has been widely debated (Pauly et al., 2013). Despite the fact that long-term fishery independent surveys present a better picture of the fishery (Kleisner et al., 2013), they are often expensive and are scarcely applied in developing countries, which, despite contributing significantly to the global fisheries, are often still data poor (Pauly et al., 2013). The Kenyan fishery has been subjected to a number of independent surveys since the early 1970s. They have been undertaken offshore from the grounds fished by the artisanal fishers and include surveys by R/V Prof. Mesyatsev and later R/V Dr Fridtjof Nansen (Sanders et al., 1988). These surveys were undertaken to determine

the country's potential to venture into offshore fishery as a means to reduce the impact on inshore fishery and to increase yield levels.

According to the surveys conducted, the potential annual yield for the demersal and small pelagic species combined was estimated at 20,000t (Odero, 1984). This estimate was based on the offshore surveys undertaken from the R/V Prof. Mesyatsev and R/V Dr Fridtjof Nansen, which estimated the potential annual yields of about 10,000t for each of the demersal and small pelagic species (de Sousa, 1988). The total standing biomass for the trawlable area was estimated to be 32,100t over an estimated area of 12,676 Km^2 (corresponding to about 2.5t/km^2) with a maximum sustainable yield of about 9,000t (Sanders et al., 1988). However, the exploitation of these offshore resources is unlikely, considering the low densities and the poor commercial value of the species present on the narrow continental shelf (Iversen, 1984; Sætersdal et al., 1999). The reef fisheries have not been surveyed, yet they are the biggest contributor to table fish supporting the local livelihoods.

Table 1: Estimated potential yields of the Kenyan coastal fisheries based on estimates from independent surveys.

Year	Survey type/vessel	Resource	Estimated potential yield	Trawled area
1982	Prof. Mesyatsev	Demersal fishery	8,933	10,677 Km^2
1983	Dr. Fridtjof Nansen	Total inshore	20,000	
1984	Prof. Mesyatsev	Demersal fishery	9,790	20,500 Km^2
1984	Dr. Fridtjof Nansen	Small pelagics	10,000	
	FAO/UNDP[3]	offshore	32,100	12,676KM^2

The first catch assessment survey for the artisanal fisheries was set up in 1984 to provide monthly catch estimates by gear, by species and by region for the marine artisanal fishery (Carrara and Coppola, 1985). The results revealed that previous catch estimates had been underestimations by almost half of the actual value. Despite the apparent success of this 1984 assessment and the recommendations that followed to improve data collection, logistic constraints have restricted the continuation of such surveys making it difficult to understand the fisheries and their impacts on the marine ecosystem (Pauly et al., 2013).

2.3 DATA SOURCES

This study is based on a review and analysis of the artisanal fisheries data collected from the more than 197 landing sites at the Kenyan coast (Figure 4). The data is collected daily from the landing sites and aggregated in the form of annual fisheries statistical bulletins. The data has been aggregated from the fisheries statistics for the years 1990-2013 (FID, 2013). Data from previous years from 1950 was consolidated from the United Nations Food and Agricultural Organization database FishStat (FAO, 2006), from the "grey literature" as well as from technical reports from the FAO and other published reports. The data were summarized in a spreadsheet using Microsoft Excel and the analysis and graphs produced using the Sigma plot version 12.5.

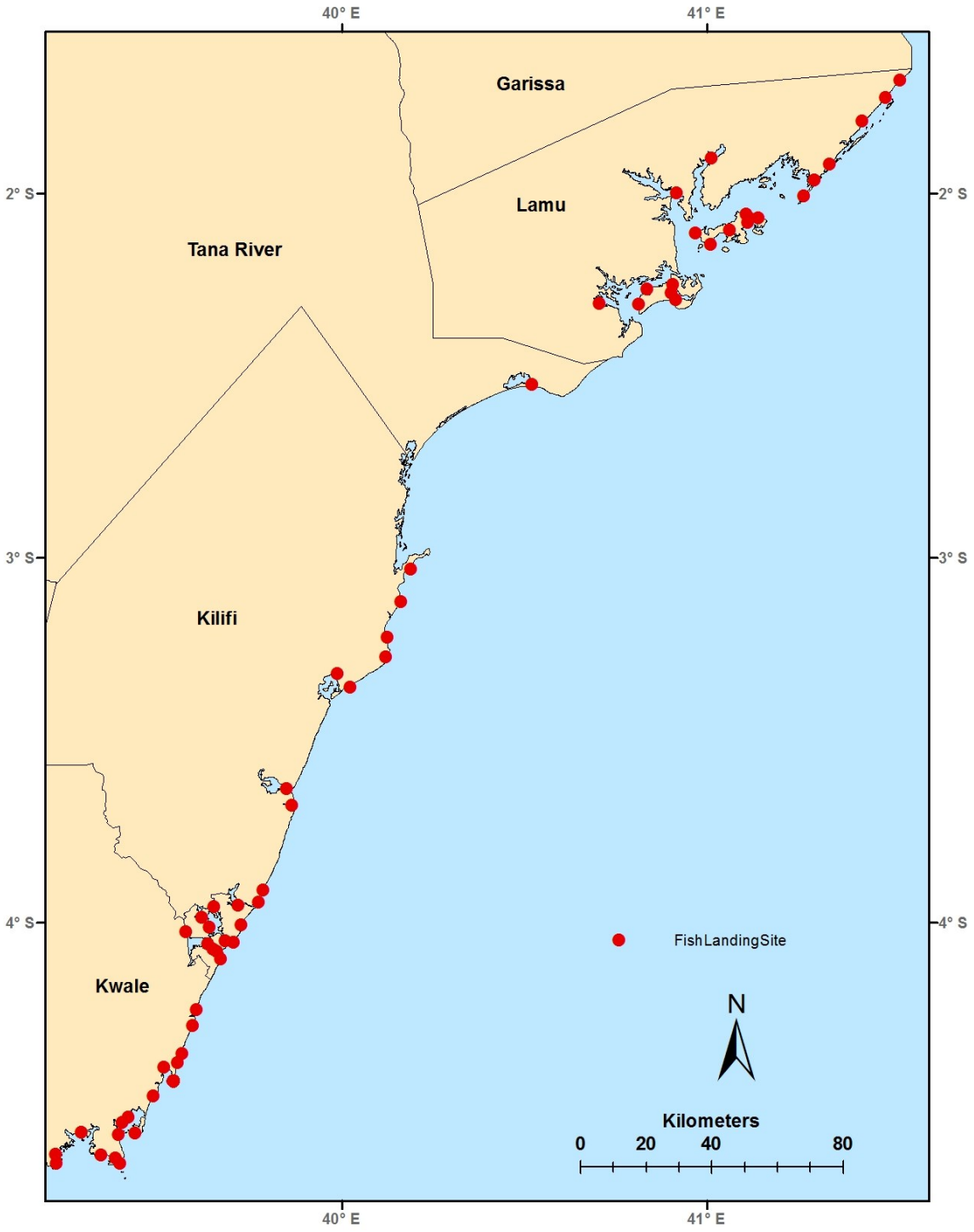

Figure 4. Map of the Kenyan coast showing the fish landing sites. source: FID (2014)

2.4 MEAN TROPHIC LEVEL OF THE CATCH

To determine if there are any changes that have occurred in the trophic level of the target species the mean trophic level of the catch was calculated annually based on the formula by (Pauly et al., 2001):

$$\overline{TL}_k = \sum_{i=1}^{m} Y_{ik}\,\mathrm{TL}_i \Big/ \sum_{i=1}^{m} Y_{ik}$$

where Y_{ik} is the landing of species *i* in year *k* and TL_i is its trophic level. The information on trophic levels was taken from FishBase (Froese and Pauly, 2000). Due to the coarse aggregation of data, only the data for the period 1990 - 2012 was used for the calculation.

2.5 MAXIMUM SUSTAINABLE YIELD (MSY) FROM CATCH AND EFFORT DATA

The surplus production model introduced by Graham (1935) was used to estimate the maximum sustainable yield (MSY) based on the catch and effort data. This approach was selected because it just needs as input, catch and effort data. The model is simple as the stock is treated as one biomass pool and the age and size structure is not taken into account. The approach has been taken up by Schaefer (1954) and Fox (1970) and other authors (Pella and Tomlinson, 1969) who developed specific versions of the "surplus production model". Both the Schaefer and the Fox model were explored for this exercise. The Schaefer model is based on the logistic growth equation expressed as:

$$\frac{dB}{dt} = rB\left(1 - \frac{B}{K}\right) - C$$

where B is the biomass, r the intrinsic growth rate, K is the equilibrium biomass in the absence of fishing and C is the catch.

The catch C is a function of effort E employed over time; the catchability q and the biomass B are expressed linearly in the form,

C=qEB

The model assumes that catch per unit of effort decreases linearly with the fishing effort. The Fox model is similar to the Schaefer model but uses a Gompertz growth model for the fish stock (Winsor, 1932).

$$B_{t+1} = B_{t+}rB_t\left(1 - \frac{ln(B_t)}{ln(K)}\right) - C_{t.}$$

The assumption is that the logarithm of the catch per unit of effort (CPUE) decreases linearly with fishing effort. The important difference between these two models is that the Schaefer model estimates MSY at a fishing level that keeps the stock at 50% and the Fox model at 37% of virgin biomass. These levels correspond to the inflection points of both growth curves. The MSY estimate of the Fox model is usually slightly below that of the Schaefer model.

2.6 RESULTS

2.6.1 The general trend in landings and fishing targets

The artisanal fishery has grown in the past 60 years from about 2,000t of catch in 1950 to about 8,000t during the last years. However, after a sharp increase in landings from 1950 to 1970 there was a first drastic decline from about 8,000t to 3,000t. After a new growth period, a second substantial decline in catches was then observed in the first years of the 1990ties. (Figure 5) Since the middle of that decade landings from the artisanal fishery have been oscillating between 5,000 t and 8,000 t per year showing only marginal fluctuation. On the other hand, the value of the landed fish has continued to increase over the years. For the years where data is available there is evidence that the value of the fish landed has continued to rise from the 1980s except towards the year 2000 where the value steadily declined coinciding with the

declining catch. The value of landed catch in 1977 was estimated at 1.6 million (USD) and this has risen to over 14.2 million (USD).

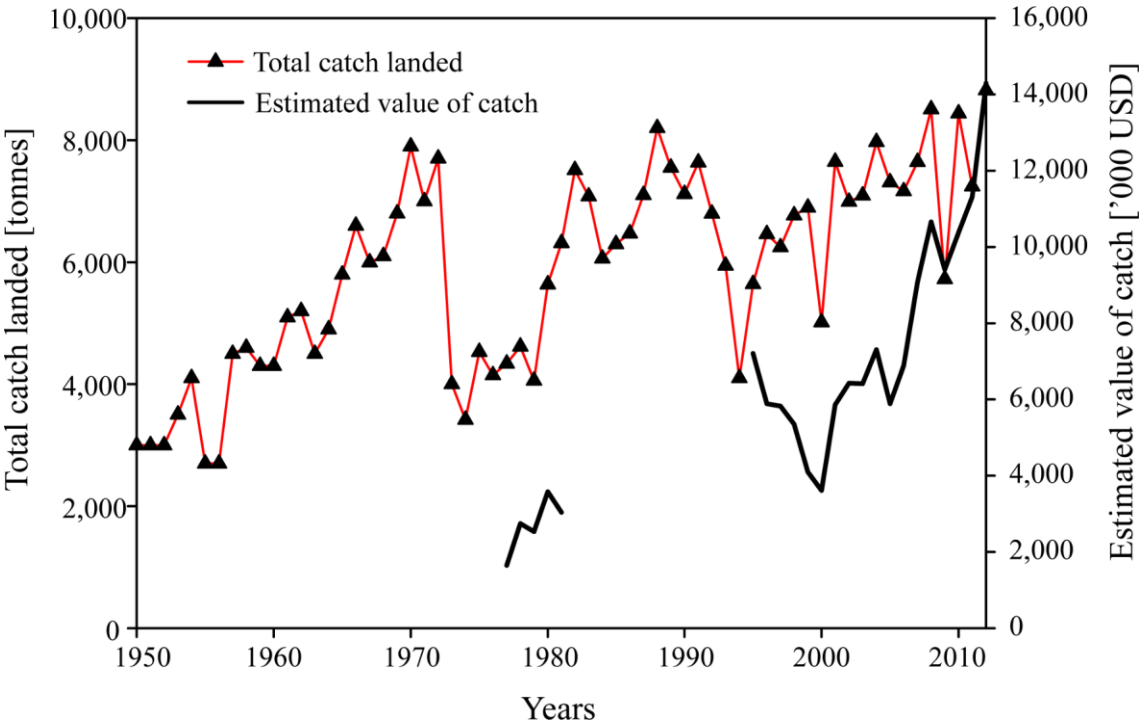

Figure 5. Historical catch trends (tonnes) and the estimated value of the landed catch (USD) from the marine artisanal fisheries for the period 1950 - 2012

Finfish dominates the total landings with only small contributions from invertebrates and other categories including sharks, rays and skates (Figure 6). Demersal and pelagic fish contribute between 30% - 50% and ca. 20% respectively. The scavengers, rabbitfish, parrotfish and the snappers are the major species landed from the demersal fishery, while the mullets are the major contributor from the pelagic fishery.

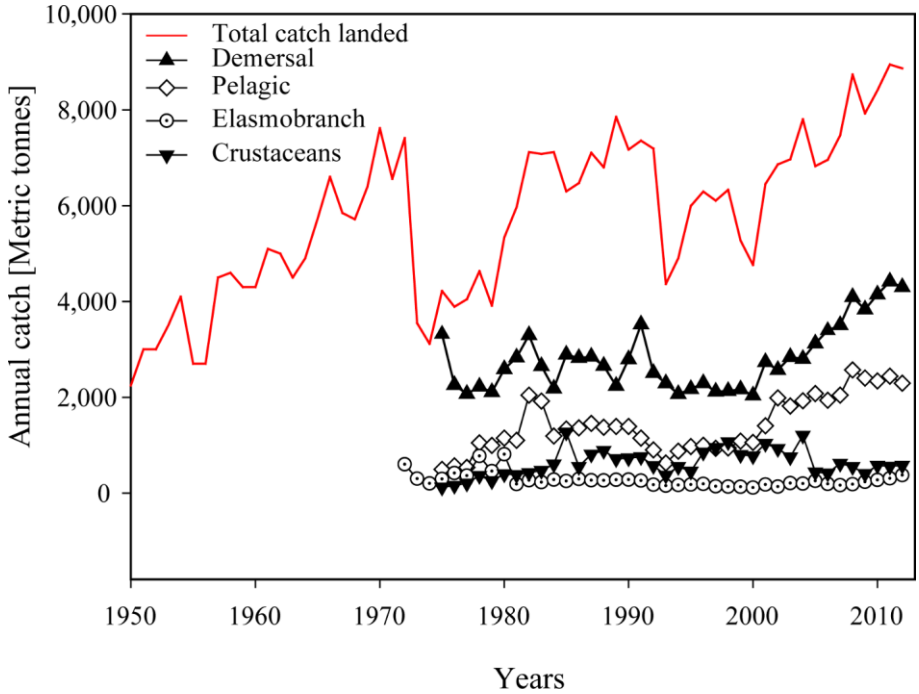

Figure 6. Estimation of the pelagic demersals ratio based on artisanal fisheries landing for the period 1990-2013

The catch per fisher (CPF) has declined tremendously for both the demersal and pelagic fishery. For both the scavengers and the rabbitfish the catch dropped from 160 kg per fisher to almost 40 kg per fisher. For snappers, it also dropped dramatically from 80 kg to 20 kg while the contribution of grunts has remained relatively stable.

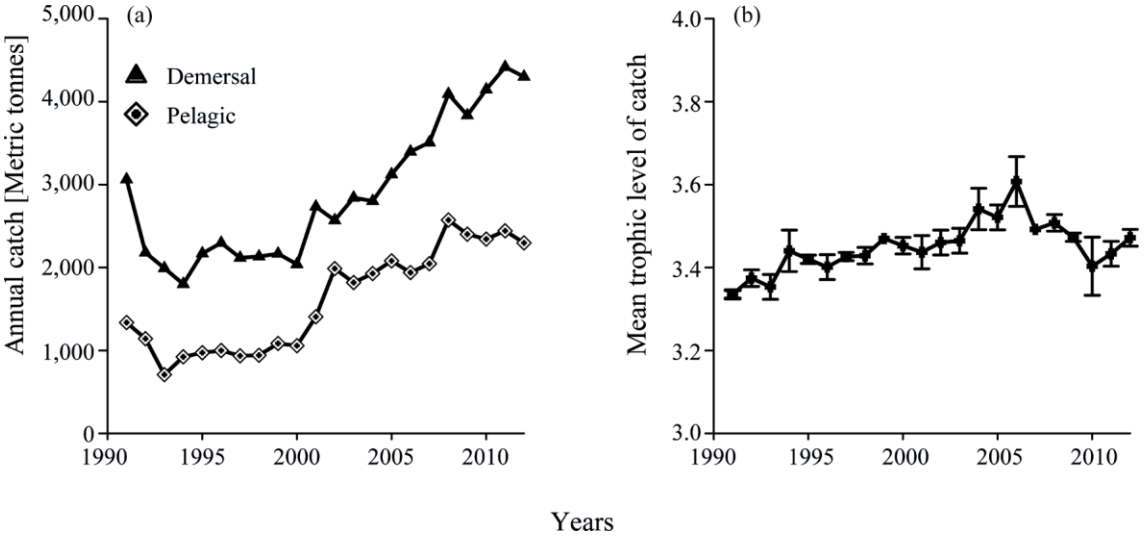

Figure 7. Historical trends in the (a) pelagic-demersal ratio and (b) the mean trophic level of the catch from the year 1990 to 2012.

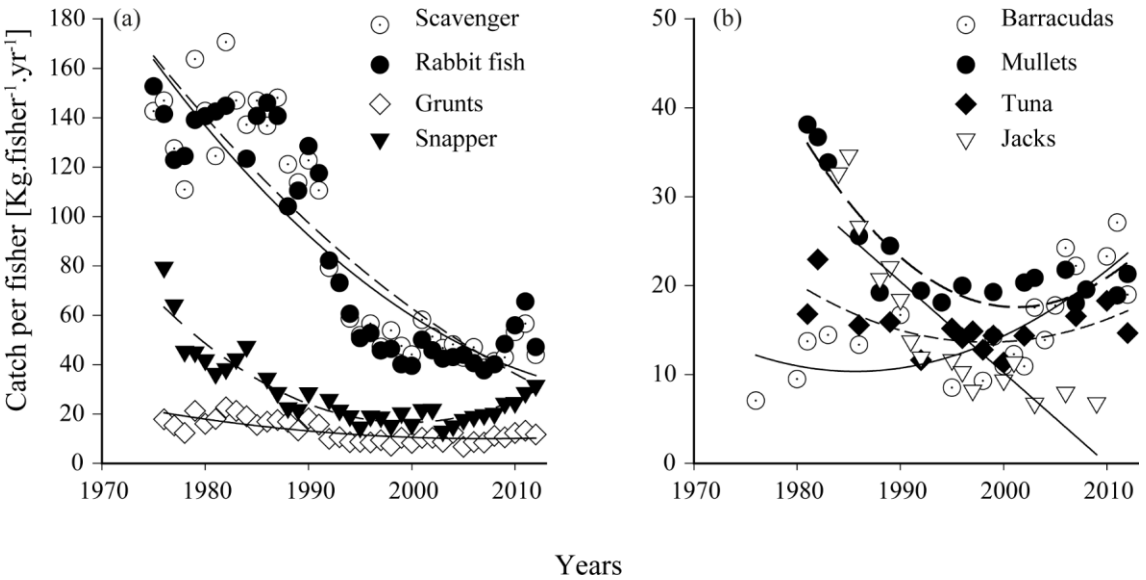

Figure 8. Historical trends in catch landed for (a) selected demersal fish groups and (b) pelagic fish groups landed by the artisanal fisheries.

The contribution of the large pelagic fish is much lower compared to the demersal species. They also showed a general decline over the years. However, while the Jacks continuously declined, the Barracudas increased marginally.

2.6.2 State of the fishery with respect to MSY

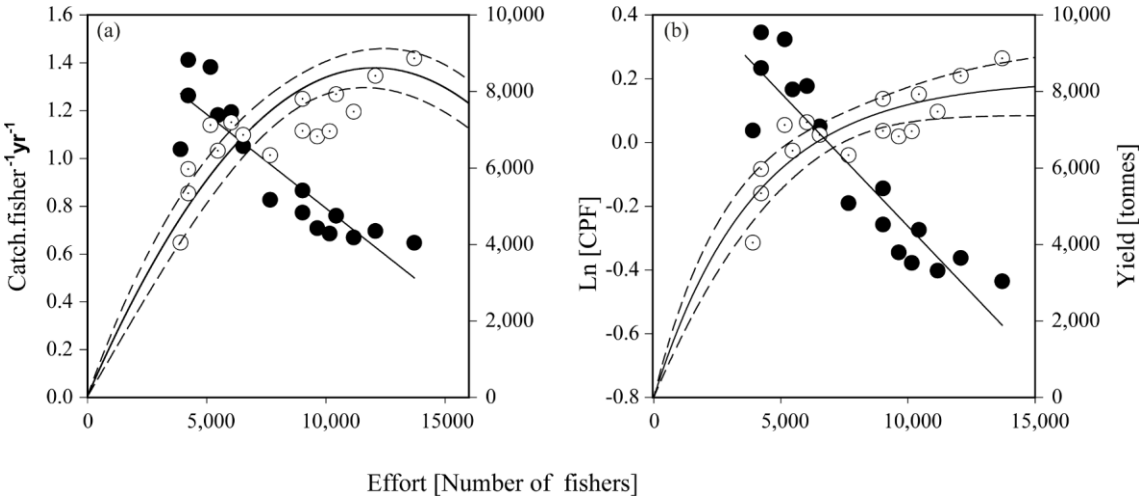

Figure 9. The estimated maximum sustainable yield for the Kenyan Artisanal fisheries based on (a) the Schaefer and (b) the Fox model.

Table 2: Estimated potential yields of the Kenyan coastal fisheries based on Surplus production models (Schaefer and Fox models)

	Schaefer	Fox
Intercept (a)	1.52957028	0.373219938
Slope (b)	-0.0000685	-0.000064652
MSY	8,543.72	8,264.34 (tonnes)
fMSY	11,171	15,467 (fishers)

Overall the landings have remained relatively stable over the past decade fluctuating between 5,000 t and slightly more than 8000 t annually. Based on both the Schaefer and fox model prediction of the MSY, the sustainable limit for the fishery is between 8,500 and 8,200 tonnes with an effort (fisher population) of 11,000-15,000.

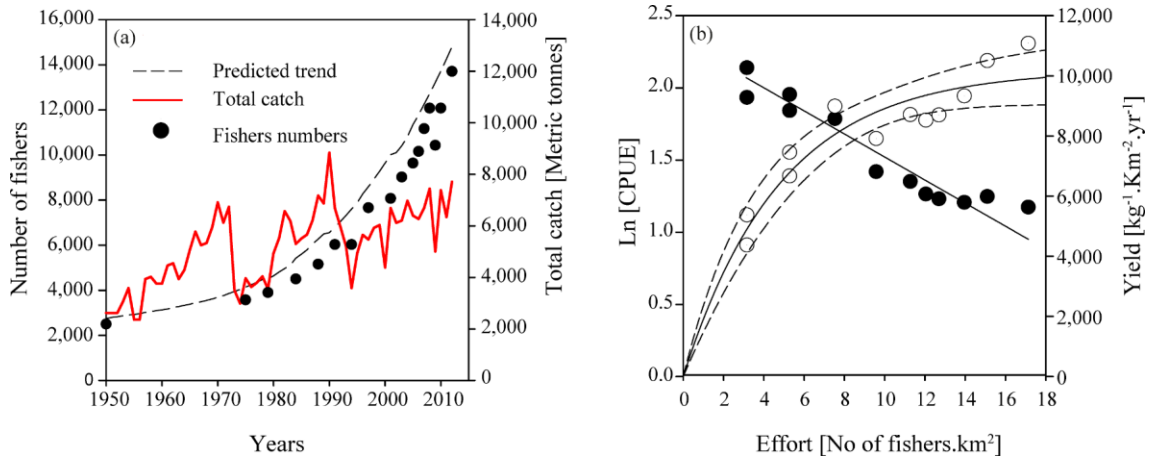

Figure 10. Historical changes in (a) catch trends and fishers' numbers and (b) estimated effort based on current yield.

2.6.3 Development of the fishery regarding fishing gears and vessel

The total number of fishers (with and without vessels) has been increasing based on the biannual fishers survey conducted from 2004 (Figure 11). Among the fishers with vessels, the number using outboard engines has continued to increase from less than 200 in 2004 to slightly over 400. Those using inboard engines have remained relatively constant. In terms of gears employed, the number of long lines increased sharply and then dropped back between 2008 and 2014. There was a reduction in the number of the other gears from 2004, but the number of gears has remained relatively unchanged since 2008. The trolling lines have not shown much variation throughout the years. The use of illegal gears has also increased over the years with the monofilament nets and spear guns tremendously increasing until 2012 when they started to decline. The use of beach seines was slightly reduced since 2006 and remained almost unchanged since then.

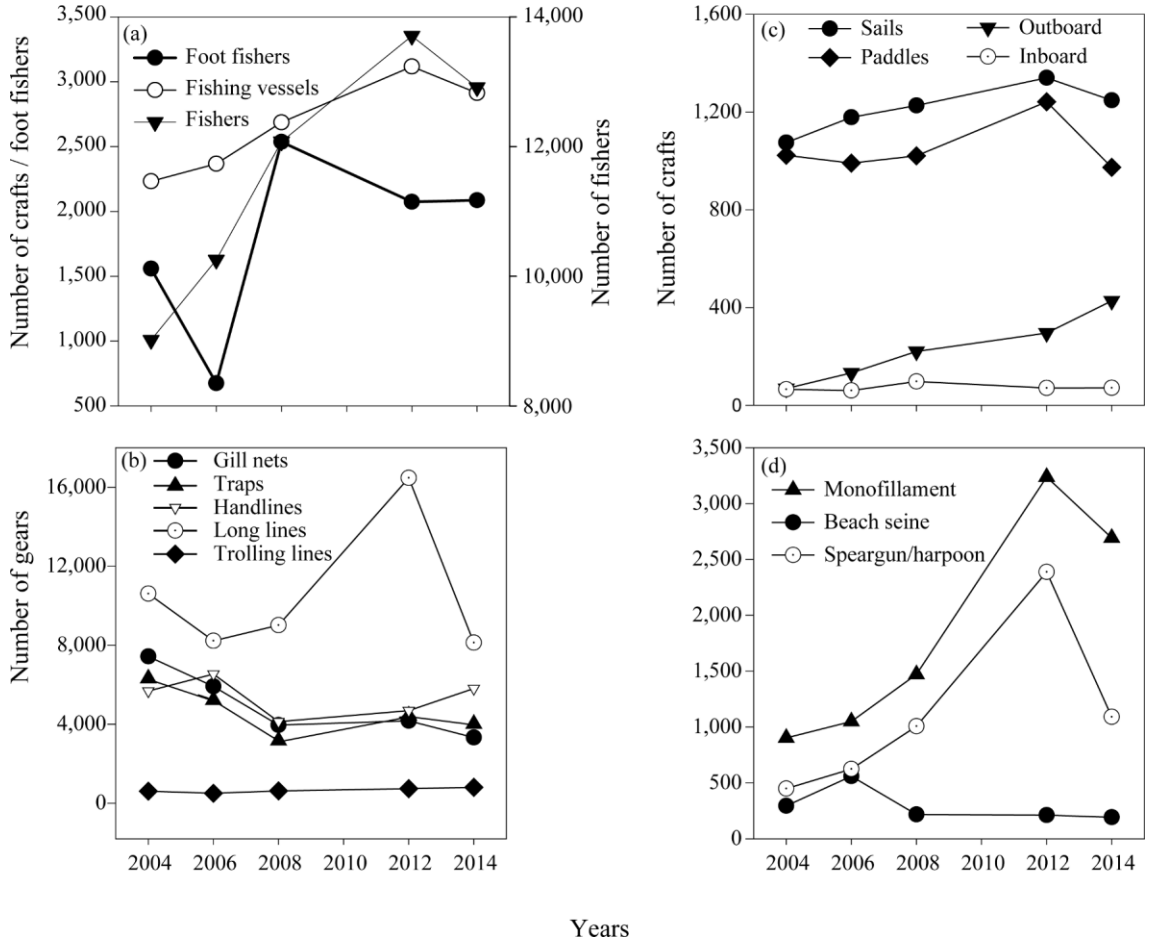

Figure 11. Changes in the number (a) fishers, foot fishers and fishing vessels; (b) fishing gears; (c) fishing vessels with propulsion and (d) illegal fishing gears for the period 2004-2014

2.6.4 Trends in fisheries landings

Variations in landings can be attributed to a number of factors including fluctuations in the marine environment, variations in the fishing intensity, changes in market demand and prices and target species and changes in capture technology (Caddy and Gulland, 1983; de Mutsert et al., 2008). It is important to consider the possibility that observed fluctuations in catch may be driven by other factors unrelated to the effort (Anderson et al., 2012; Branch et al., 2011; de Mutsert et al., 2008). This has been shown for upwelling regions where natural oscillations in the primary production are reflected in the overall catch landed, be it increase or decrease depending on the target fishery (Caddy et al., 1998). Coral reef systems have been shown to be vulnerable to fishing already at a low fishing intensity (Munro, 1996) and significant

changes in yield and species composition have been detected even at the beginning of a fishery (Mcclanahan, 1994, 1999).

Fisheries trends studied globally with landing dates for at least 40 years have shown to exhibit either steady, cyclic, irregular, or spasmodic pattern depending on the natural patterns of variation of the target species (Caddy and Gulland, 1983). These patterns are not always predictable and may overlap between different species. However, they may give a basis upon which fisheries can be assessed to determine the departures from the stable state. In the case of the artisanal fisheries in Kenya, landing trends have exhibited a cyclic pattern with alternating periods of high and low abundance irregularly fixed in time. The earliest documented catch record from the artisanal fisheries dates back to 1949 where the artisanal fishery generated an annual catch of 2,249 tons landed by slightly over 2,500 fishers using 1,019 vessels (Hoorweg, 2009). The fishery experienced a steady increase in landings peaking in 1970 and sharply declining thereafter.

This pattern would seem characteristic for a development phase of an artisanal fishery, where catch levels increase with an increase to a point where the fishery collapses when effort exceeds sustainable levels and the catch breaks down (Pauly, 1994). During these years of steadily increasing catches, the exploitation of the marine resources was high with the incidence of coral harvesting and the use of dynamite fishing (IUCN/UNEP, 1984). According to Ray (1969), the marine resources were finally becoming depleted at an accelerated rate due to intense spearfishing and collection of shells and corals. It is estimated that 1 tonne of corals was exported out of the country in 1978 (Wells, 1981). The number of fishers was estimated to have increased from 4,500 to 12,000 within this period (IUCN/UNEP, 1984; Oduor, 1984), which then resulted in overfishing shown by the declining trend in fisheries landings between 1978 and 1980.

An important contributing factor to the declining trend could have been the introduction of modern equipment and loans given to fishers, enhancing the intensification of the fishery (Sutton, 1975). Elaborate measures were put in place to

improve the data collection system, and the results showed that previous catches had been grossly underreported (Carrara and Coppola, 1985). On a global level, technological advancement in the fishery has been cited as a major contributor to the declining of global marine stocks (Myers and Worm, 2003). Therefore, the question arises if this trend also holds for Kenya. Though artisanal fisheries have been noted to remain relatively unchanged over an extended period, it is evident that the introduction of new or more efficient gears can have significant impacts on the fishery (McClanahan and Mangi, 2004). Due to the nature of data collected it is not possible to quantify the impacts that the introduction of the shark and hand lines had on the fishery, but anecdotal evidence suggests that the shark fishery declined rapidly to unprofitable levels (Hoorweg, 2009).

The concerns over the high exploitation and decline in stocks led to the establishment of the first marine protected areas (MPAs) in Kenya, the Malindi and Watamu marine parks in 1968. Nine MPAs were set up between 1968 and 1990 covering 8.7% of the continental shelf (Wells et al., 2007). This could have had a marked influence on the overall catch landed due to the reduction in the fishing areas accessible to the fishers and thus contributed to a decrease in the overall catch. During the past 20 years, the catch has oscillated between 5,000 tonnes and 8,000 tonnes with no significant increase while the effort has steadily increased from ca. 6,000 to almost 14,000 fishers. While overall catch levels do not show a downward trend during the past two decades, studies conducted on the Kenyan south coast have revealed a declining catch per unit of effort and a decrease in mean trophic level from the landed catch (McClanahan et al., 2008).

Given the nature of data used for this review, only comparable CPUE can be inferred. However, considering that the effort has been increasing while overall catch has stagnated there is no doubt that the catch per fisher has decreased. This is typical of the Malthusian fishing typified by increased effort, reduced fish sizes, reduced catch per unit effort and use of illegal gears (McClanahan et al., 2008; Pauly, 1994). Munro (1984), estimated the yield from coral reefs to usually range between 10-20

tons/Km^2/year. Estimates from the Kenyan artisanal fishers are varied and range from 5.1 to 22 tons/km^2/year (McClanahan and Kaunda-Arara, 1996; Obura, 2001). Based on this study we estimate an annual yield of between 11-12 tons/Km^2/year with a fisher density of 12 fisher/Km^2 (Figure 8).

2.6.5 State of the fishery

While the total catch has as yet not shown a downward trend from the year 2000 (Figure 5), some studies have shown that the catch for the commercially important species has declined and this is also confirmed by this study (Figure 6) (Kaunda-Arara et al., 2003). It is probable that the continued illegal use of beach seines and the spear gun has also contributed towards the declining trend considering that these gears cause direct physical damage to corals (Mangi and Roberts, 2006). The number of fishers using these gears has continued to increase over the past decade (Figure 11) and considering the impacts of beach seine on the Kenyan South coast there is no doubt that they have impacted the fishery to a great extent (McClanahan and Mangi, 2004). In addition, the impacts of the El Niño should not be discounted as this event resulted in bleaching and mortality of corals up to 80% (Obura, 2001). According to McClanahan and Mangi (1999), the El Niño that occurred in 1987 and 1994 caused significant damage to reefs and invertebrates. This could have exacerbated the impacts on the reef already under intense exploitation from the fisheries contributing to decreased landings.

The potential yield for the inshore fishery has been estimated at 20,000 tonnes (Oduor, 1984). Nevertheless, the annual total catch landed has never exceeded 10,000 tonnes with the catch remaining relatively unchanged for the past decades. Our estimate of MSY (8,264-8,543) and the corresponding effort of 11,171-15,467 fishing boats, derived from the Schaefer and Fox models, would suggest that yields higher than the presently obtained levels cannot be expected in future and that the inter-annual variation in total landings may have to do with environmentally triggered

changes in resource productivity. The model results also suggest that the overall effort of the present fishery (1,400 boats) already exceeds sustainable effort levels by at least 20% suggesting a general state of overfishing. The fact that overall catch has increased over the years and has levelled at about 8,000t under conditions of a steady increase in effort is most likely explained by a fishery-induced shift in target species and a relative increase of smaller, more productive species paralleled by the observed decline in many of the former target species. The demersal species, which dominate the landings, have decreased substantially (Figure 8) as shown in our study, while the contribution of the highly productive crustaceans invertebrate species such as the crustaceans has increased slightly as reported in (Obura, 2001). While our review allows painting a general picture of an overfishing situation of the Kenyan artisanal reef fisheries, more detailed stock assessments of the different target resources, as well as socio-economic studies with regard to market demands, prices and costs, are urgently needed to allow for the development of an ecosystem-based sustainable management regime.

2.7 ACKNOWLEDGEMENTS

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CHAPTER 3.

Size structure and gear selectivity of target species in the multispecies multigear fishery of the Kenyan south coast

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ABSTRACT

We investigated the species composition and length frequency distribution of the artisanal reef fishery catch from the Kenyan South Coast with the aim of determining the factors that influence catch composition. Typical to most tropical multi-species fisheries, the artisanal catch was characterized by over 138 species representing 38 families. Of these, 17 species made up 91% of the overall abundance and contributed 70% by weight of the total catch from all gears and contributed most to the observed similarities between the groups. Species selectivity of gear was determined by a classification analysis (hierarchical agglomerative clustering) based on a similarity matrix from a transformed abundance data of the fish species by gear. At a similarity of 50%, five groups were differentiated by the cluster dendrogram. The basket trap and beach seine overlapped in their species and size selectivity and caught smaller individuals than the other gear types. Overall the beach seine landed the smallest individuals in the catch but cumulatively the basket traps targeted similar size range at a higher abundance than the beach seine. The hook and line and the ring net targeted the largest individuals in the catch. The current fishing practices exploit not only fish species of small sizes but also small to medium sized specimens relative to the species potential maximum size. Overall, our findings indicate that fishers, by diversifying their gears and strategies can target if not all but a significant part of the size spectrum and trophic composition of the fish community. This puts a major challenge to managers as the current regulations focusing on gear restrictions are not adequate to manage this complex fishery.

Keyword: Kenya, gear selectivity, artisanal, reef, multispecies, multi-gear

3.1 INTRODUCTION

Artisanal fishing is one of the most important exploitative activities on coral reefs sustaining many coastal communities in the tropics (Russ and Alcala, 1989; Sadovy, 2005). Estimated to account for up to 25% of the world's catch, these artisanal reef fisheries are among the most important direct contributors of fish for human consumption and yet they are still greatly neglected and overlooked (Allison and Ellis, 2001). While the impacts of industrial fishing are widely recognized, marine ecosystems are considered less threatened by artisanal fisheries (Hawkins and Roberts, 2004; Shester and Micheli, 2011). This view is rooted in the fact that artisanal fisheries are small-scale and multispecies and use small quantities of different gears that are often passive and selective and considered to have changed little over the years (Mathew, 2003). Nevertheless, over the past decades, coral reef fish stocks have come under increased pressure at many places, mainly due to growing fishing effort and the use of destructive gears causing moderate to severe declines in valuable tropical marine species (Hawkins and Roberts, 2004; Sadovy, 2005).

Studies on the fishery impacts on tropical reef ecosystems have shown that a decrease in abundance and biomass of target species are the most obvious consequence (Hawkins and Roberts, 2004; Jennings and Kaiser, 1998; Ruttenberg, 2001). The fishery caused reduction in size and yield of target species concomitant with a decrease in their recruitment success may eventually lead to a total species collapse and a gradual shift in fish community structure (Koslow et al., 1994). These declines may be attributed to some interacting factors including increased fishing intensity (Hawkins et al., 1999; Koslow et al., 1994), selective removal of top predators and the use of destructive fishing methods (McManus et al., 1997; Pauly et al., 1989). This has been exacerbated by the limited control over fishing effort and a lack of regular monitoring, which is characteristic of most artisanal fisheries, and which often leads to a rapid growth in artisanal fisheries under open access regime (Mathew, 2003; Tanner et al., 2014).

At the Kenyan coast, the importance of artisanal fisheries cannot be overemphasized; historically they play a vital role in livelihoods and are crucial for nutrition in Kenya. These resources are especially vulnerable to damage due to overfishing, by a growing human population and the frequent use of destructive fishing techniques (McClanahan et al., 2005). According to Tuda and Wolff (2015), the number of fishers has increased as has the number of illegal gears, yet overall catch landed have continued to fluctuate. The situation is further complicated by the fishers' attitudes, some who believe that the amount of fish they catch is what Providence has determined. The level of fishing pressure on Kenyan coral reefs is considered high (Teh et al., 2013) with resources considered exploited at sustainable levels or overfished (Kaunda-Arara et al., 2003). Prior studies that have noted that over-fishing and the use of destructive fishing techniques are the major threats to this fishery (Mangi and Roberts, 2006; McClanahan et al., 2008).

Nevertheless, selective fishing of key species and functional groups from the ecosystem has been identified as the root problem (McClanahan and Kaunda-Arara, 1996; Zhou et al., 2014). An implication of this is the inevitable alteration of the composition of a population or community structure and biodiversity (Garcia et al., 2012). Therefore, understanding the dynamics of these fisheries such as gear selectivity and impacts on coral reef fish assemblages can help to address the challenges involved in fisheries management (Gobert, 1994; Liang et al., 2014). However, these dynamics are still poorly understood especially in the context of artisanal fisheries. This study seeks to assess the size structure of observed catch across a series of fishing gears to evaluate the role and importance of the different fishing gears in place for the exploitation of the different species and sizes. Although this is a simplification, in the absence of gear selectivity studies, our results are expected to be qualitatively informative and contribute towards policy advice.

3.2 MATERIAL AND METHODS

3.2.1 Study site

The study was conducted at the Kenyan South Coast approximately 50 km south of Mombasa to about 80 km north of the Tanzanian border. Four study sites were selected: Mkunguni (Msambweni area), Gazi, Chale and Mwaepe (Figure 12). The area of study was chosen based on the proximity of the landing site to each other and overlap in fishing sites which make the fishing in all these areas relatively similar and easy to access the catch. The fishery is coral reef and lagoon based with fishers employing multiple fishing gears across multiple sites and travel to and from their fishing grounds under the influence of the tides (Alidina, 2005; McClanahan and Mangi, 2004). Fishing typically takes place from the shore to the outer reef and fringing reef lagoon in shallow, hard bottom back reef locations between 0.5–3 m deep at low tide (Hoorweg et al., 2003; McClanahan and Arthur, 2001).

However, the daily fishing patterns are influenced by a range of environmental and weather conditions, which affects the fish migration and the fishers behaviour with regards to the target species (Daw et al., 2011; Mangi et al., 2007). During the Southeast Monsoon associated with strong winds (May – October), the fishers travel less far and are more confined closer to the shore as compared to the calm Northeast Monsoon (November – April). As such, fishing methods which require fishers to spend a substantial amount of time in water such as spearguns may be disadvantaged due to the poor visibility and cold water temperature during the Southeast monsoon and may opt to change to more appropriate fishing gears. Also, fish catch and reproduction have been shown to be highest during the Northeast Monsoon (McClanahan, 1988), making it the preferred fishing season.

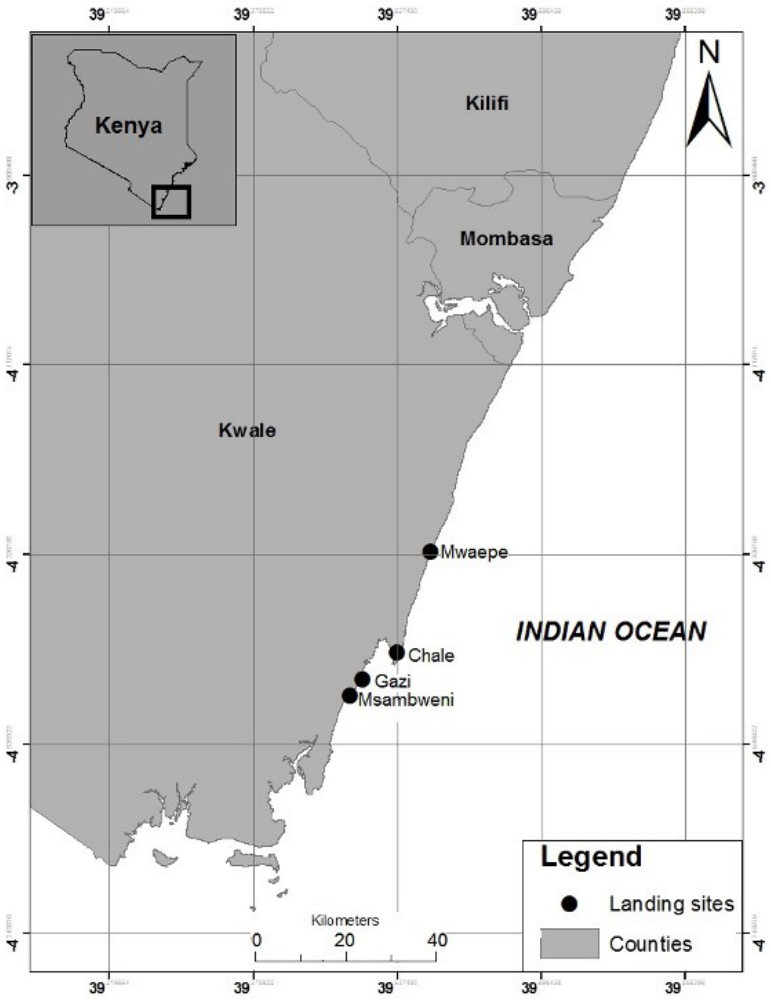

Figure 12. Map of the study area showing the positions of the sampled fish landing sites.

3.2.2 Fishing gears and sampling methods

The artisanal reef fishery in Kenya is multispecies and multi-gear meaning that fishers employ different kinds of fishing gear to harvest the resource. The biannual Marine frame survey in Kenya (FID, 2014), identified up to eighteen different fishing methods at the Kenyan coast notably varying in use depending on the fishing seasons, target fishery and area of fishing. Samoilys et al. (2011), broadly categorized the fishing gears into four groups. These include: (1) the traditional basket trap, (2) hook and lines, (3) spears and harpoons and (4) nets which include gillnet, beach seine, reef seine, ring net and the monofilament net.

Mangi et al. (2007), extensively discuss the factors that determine the choice of fishing gears in Kenya. The results of the study showed that the income earned and the profitability of the gears played a significant role in determining gears choice but varies depending on the experience and age of the fishers. The younger and less experienced fishers prefer the more active fishing gears such as the speargun and the beach seine while the older fishers prefer, the easier and less energy involving basket traps and hook and line fishery (Obura, 2001).

For this study, eight fishing methods were considered, though some are practised at the sites by fewer less regularly. Fish landings were collected from the four landing sites over a period of one year (June 2014 – June 2015) to cover both the North East and the South East Monsoon Season. The fish were identified up to the species level using the field identification guides by Anam and Mostarda (2012) and Lieske and Myers (1994). The total length (TL) (taken from the tip of the snout to the longest caudal lobe) of individual fish species landed by each fisher was recorded to the nearest 0.1cm on a measuring board. Fish lengths were grouped into size-class categories to enable comparative analysis of species and size-class distribution amongst the gears.

3.2.3 Length frequency analysis

The assessment of reef fish resources using size-structure data is complex due to the multi-species nature of the fishery (Gobert, 1994). Therefore, based on a cumulative frequency distribution of the most abundant species, 17 species were found to contribute up to 90% of the total abundance and were retained for further analysis. Biomass estimates were made for each fish by converting the length frequency data using the standard length-weight relationships from FishBase (Froese and Pauly, 2010). The size structure of the selected species was analyzed further based on their length frequencies in the observed samples. The width of the length intervals was determined according to the maximum fish length (Neumann et al., 2012). This

approach proposes a length interval widths of 1cm for fish that reach 30 cm, 2 cm for fish that reaches 60 cm, and 5 cm for fish with maximum lengths greater than 60 cm. This is because the use broad length groups tend to mask length-frequency details while too narrow length groups result in low sample sizes within each length-group (see also Wolff (1989).

3.2.4 Size at maturity

The mean length at which fish of a given population become sexually mature for the first time (L_m), was estimated by the empirical formula given in (Froese and Binohlan, 2000). This is an important management indicator to determine whether enough juveniles in an exploited stock are allowed to mature and spawn (Beverton and Holt, 1956). The relationship between the L_m and the asymptotic length is expressed as:

$$\mathrm{Log}_{10}\, Lm = 0.8979 * \mathrm{Log}_{10} \mathrm{L}\infty - 0.0782$$

In which Lm is the length at first maturity, and L∞ refers to the asymptotic length in cm; expressing the mean length that the fish would reach if they were to grow indefinitely. Estimates of the maximum recorded size of the species were derived from the FishBase (Froese and Pauly, 2010) and size structure of the catches grouped into size classes of between 10-20 cm, 20-30 cm (Gobert, 1994).

3.2.5 Gear selectivity

To determine the species and size selectivity and overlap between the fishing gears, a classification analyses (hierarchical agglomerative clustering) based on a similarity matrix from a transformed abundance data of the fish species by gear was analyzed. Only the species contributing 90% of the overall abundance were analyzed using the Bray-Curtis similarity metric as it does not treat absences to derive similarity

between groups (Clarke, 1993). The software Primer PRIMER 6 (Clarke and Gorley, 2006) was used to perform this analysis.

3.2.6 Trophic level of catch

The mean trophic level of the catch for each gear (*k*) was calculated using the formula by (Pauly et al., 2001):

$$TL_k = \sum_{i=1}^{m} Y_{ik}\, TL / \sum Y_{ik}$$

Where Y_{ik} is the catch of species *i* in gear *k*, TL is the mean trophic level of species *i* for *m* fish. The values of the trophic level estimate for each species were computed from the FISHBASE derived values that are based on the diet of the species.

3.3 RESULTS

3.3.1 Catch composition

A total of 6531 fish individuals representing 138 species from 38 fish families were captured. Of these, 34 families were associated with reef and seagrass species. Nevertheless, the catches were dominated by 17 species making up 90% of the total abundance and 70% by weight of the total catch from all gears (Figure 13). Interestingly, out of the 17 most abundant species, three accounted for 65% and 46 % by abundance and weight respectively. This includes the white-spotted rabbitfish, *Siganus sutor* (Valenciennes), the marbled parrotfish *Leptoscarus vaigiensis* (Quoy & Gaimard) and the dory snapper, *Lutjanus fulviflamma* (Forsskål). The fishery is also composed of large predatory species, which contribute little to abundance but greatly regarding biomass.

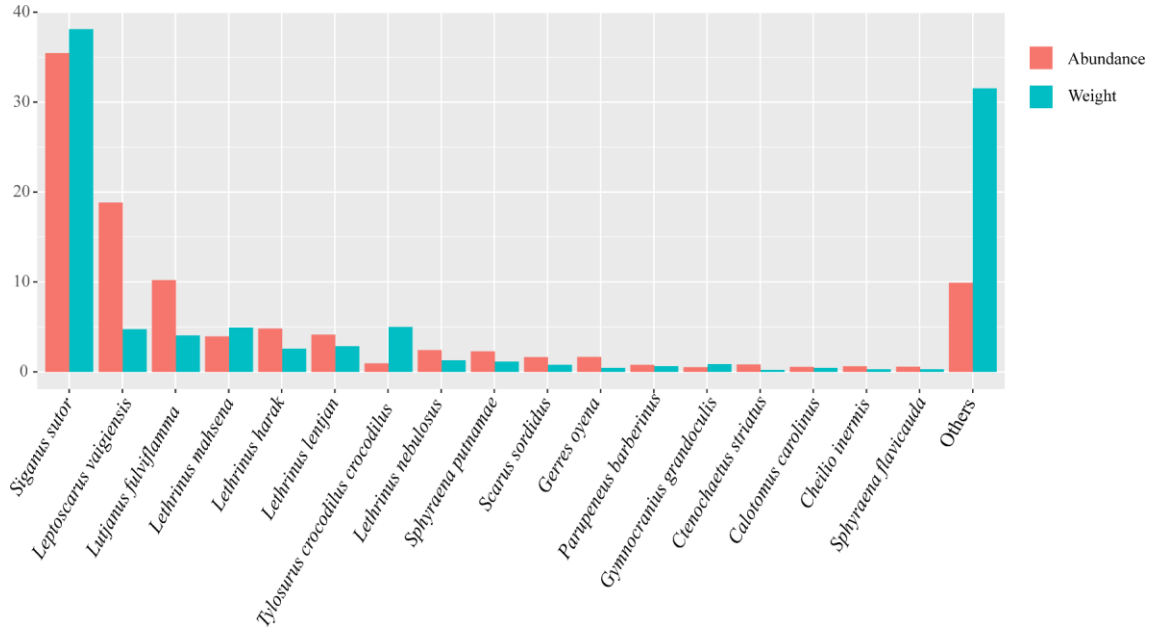

Figure 13. Relative abundance and weight (%) of the total catch for the 17 most commonly caught species from all gears.

3.3.2 Gear selectivity

The relative contribution of the abundant species differed among the various fishing gears (Figure 14). At a similarity of 50%, five groups were differentiated. The basket trap, beach seine and the hook and line targeted similar species (Group I). The basket trap and beach seine were very similar in their relative selectivity with the composition of their catches to a large extent being complementary. *Siganus sutor* and *Leptoscarus vaigiensis* were among the species captured in relatively high proportions of both gears. While the basket trap mostly caught *Siganus sutor* and *Scarus sordidus*, the beach seine mostly captured *Lethrinus nebulosus*, *Lutjanus fulviflamma* and *Leptoscarus vaigiensis*. The hook and line mainly caught Lethrinus *mahsena* and the *Lethrinus lentjan*. Group II consisted of species targeted by the spear gun and reef seine but with some overlap with the hook and line. The speargun landings were dominated by the parrotfish, *Calotomus carolinus* while the reef seine mostly caught *Ctenochaetus striatus*. Group III consisted of two species *Gerres oyena* and *Sphyraena flavicauda* captured mostly by the beach seine and the reef seine respectively while the ring targets characterised Group IV.

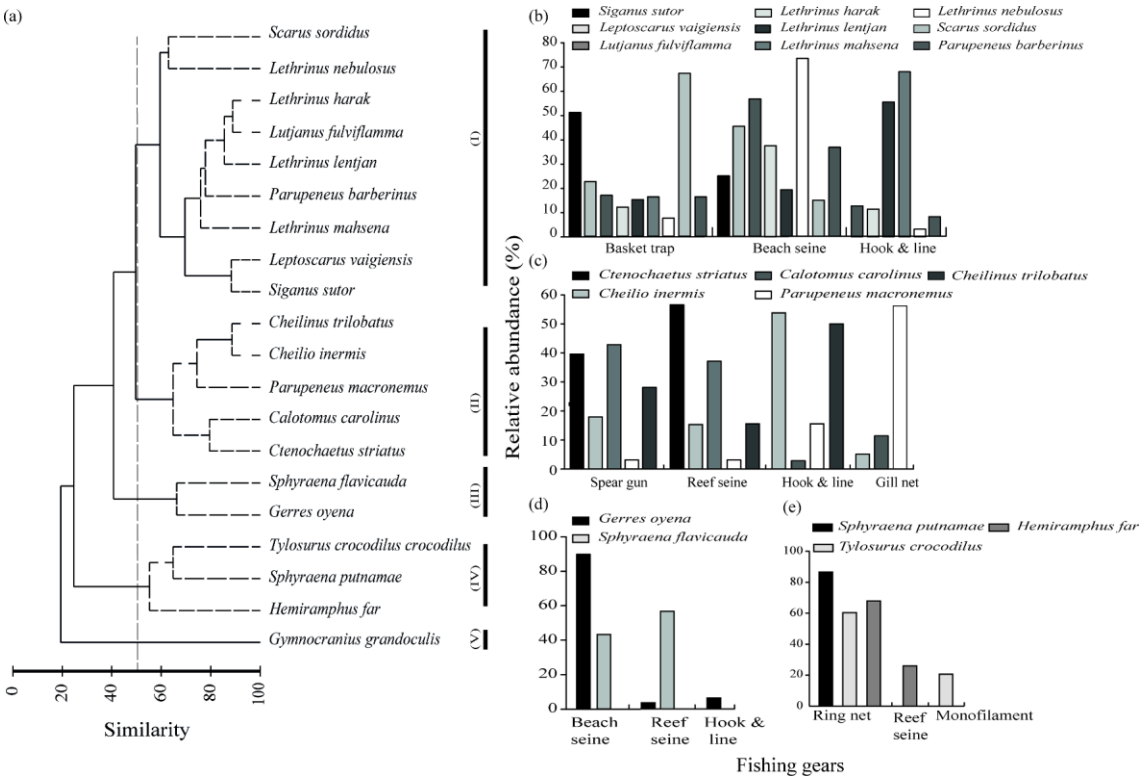

Figure 14. Cluster dendrogram and bar plot of data (abundance of species by gears based on Bray-Curtis metrics and average linkage algorithm) across gears. The dotted line denotes the cut-off of 50% similarity; five groups were identified. In each cluster, the species yielding the highest percentage in catch composition are given.

3.3.3 Size structure and catch per unit of effort

The overall length frequency distribution of the six most abundant species varied across the gears. For instance, the species captured by the beach seine were mostly small in size ranging 15-20 cm (Figure 15). The mean and modal lengths of the *Siganus sutor* and *Leptoscarus vaigiensis* captured by the beach seine (*S.sutor*: 17.1 cm, 17.3cm; *L.vaigiensis*: 15.1cm, 14.1 cm) were comparatively smaller than those caught in the basket trap (*S.sutor*: 24.2 cm, 28.0 cm; *L.vaigiensis*: 16.4 cm, 15.5 cm). Moreover, the sizes were below the size at first maturity L_m of those species. Up to 42% of the *Siganus sutor* landed by the basket traps were between 20cm - 30cm compared to only 16% for the beach seine for the same size range. However, the *Lethrinus harak* captured in the basket trap were relatively smaller in relation to the Length at Maturity. Comparatively, the gill net captured larger *Lethrinus harak* than both beach

seine and basket trap with 84 % all individuals caught by the gear being larger than L_m (Figure 15d). Similarly, individuals landed by the hook and line were much larger compared to the other gears.

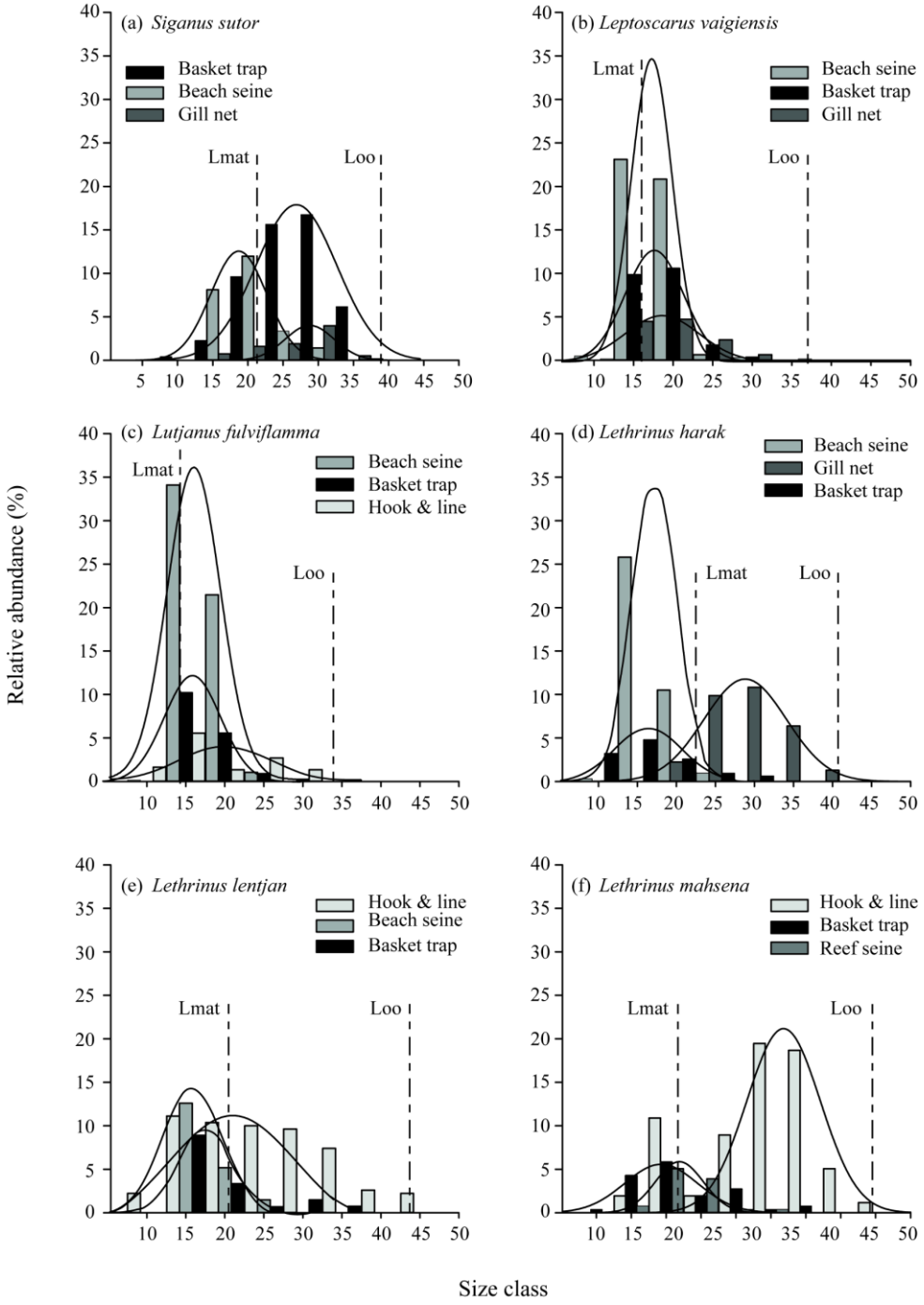

Figure 15. Observed size distribution (TL) and the predicted catches of the six most commonly caught species (by gears). The dashed lines denote the length at first maturity (L_{mat}) and the asymptotic length ($L\infty$).

The size distribution of the cumulative catch is skewed to the left having a dominant mode of 22 cm (with a mean of 21 cm) and a second, but very small mode between 70 and 90cm (Figure 16). Ninety-one percent of the catch is less than 30 cm in length contributing 66% of the total biomass, an indication that the fishery is heavily based on small to medium sized individuals, but some large fish are also caught but in small numbers. The current fishing practices exploit not only fish species of small sizes but also small to medium sized specimens relative to the species potential maximum size (Figure 16). Thirty-six percent of the landings consist of species that can attain a possible maximum size of below 40 cm representing only 12% of the biomass. The most significant contribution is by species that reach a maximum size of between 40-50 cm contributing 42% by abundance and a similar margin towards the overall biomass.

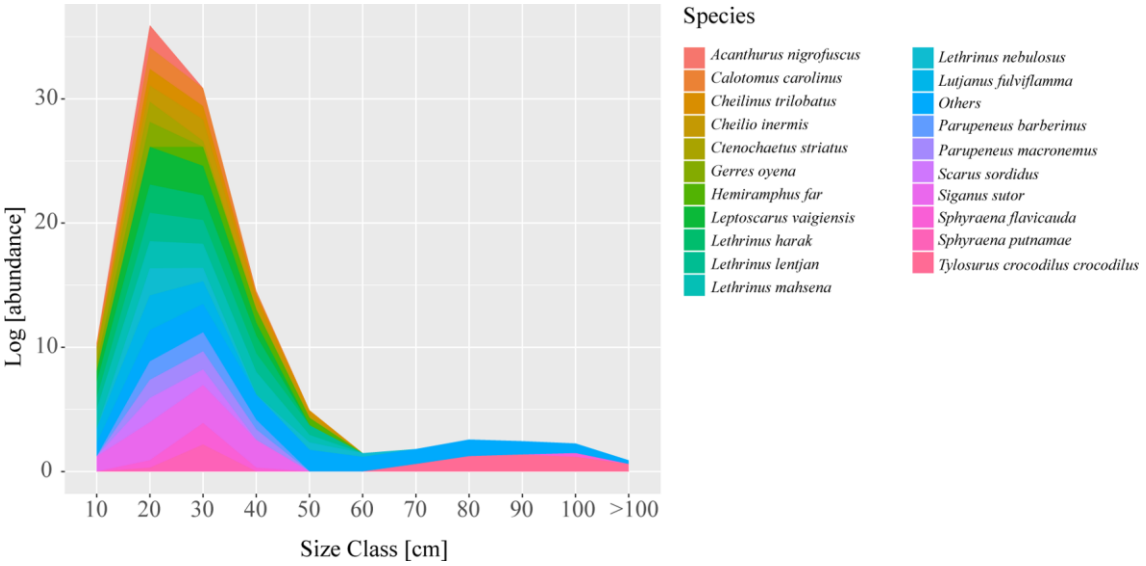

Figure 16. Size distribution of catches (log numbers) for all size classes and all gears.

The beach seine, reef seine, basket trap and the ring net tend to have a sigmoid selection but with the first three gears targeting more of the smaller sized individuals, while the ring net tends to target bigger individuals (Figure 17a). Both the hook and line and gill net have a similar selection, with catches of fish sizes that

correspond well to the chosen mesh and hook size. The selection peak of both gears was similar (50-60 cm) (Figure 17b).

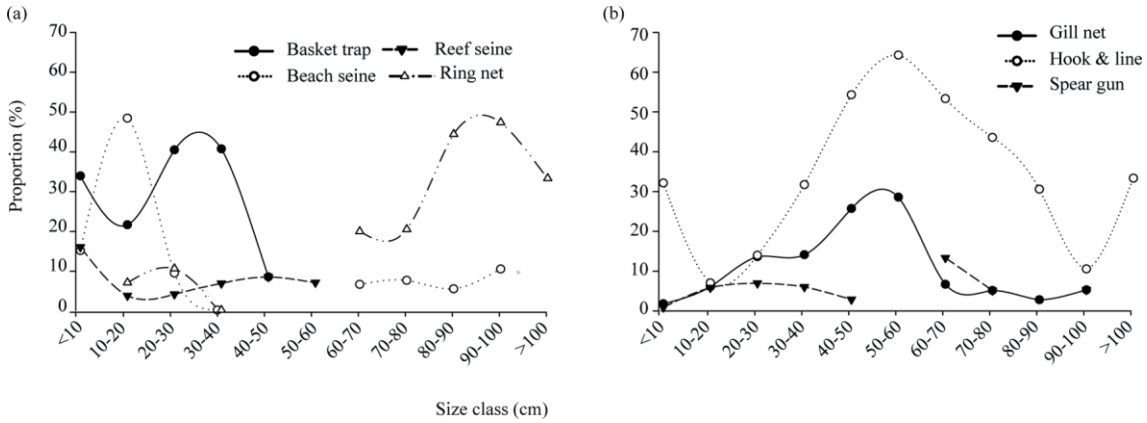

Figure 17. Observed length frequencies of the catches as a proxy for selection for the (a) basket trap, beach seine, reef seine and the ring net; (b) Gill net, hook & line and the spear gun.

Overall, the hook and line contributed slightly 31% of the total biomass, basket trap 29% while the beach seine and gill net each contributed 10% (Figure 18a). Hook and line landed the larger individuals, which translated to higher overall biomass while the basket trap mainly landed individuals less than 50cm in size (Figure 18b)

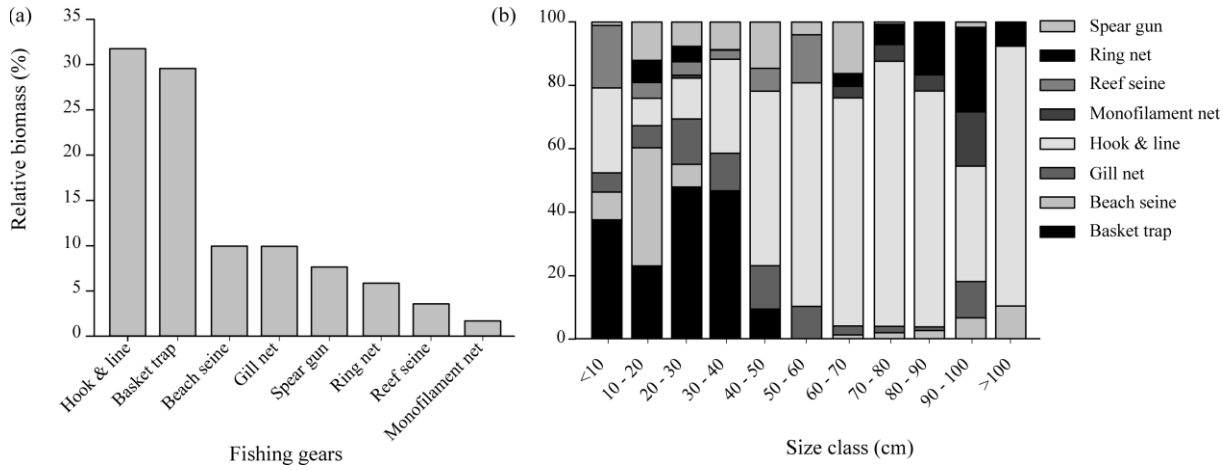

Figure 18. (a) Relative biomass of the landed catch by gears used in the fishery with the error bars denoting standard error and (b) Relative biomass by size class for all gears.

Overall, the beach seine landed the smallest sized individuals with a mean total length of 16.2 ± 0.1 cm, while the hook and line landed the biggest individuals with a mean size of 26.5 ± 0.5. However, there is considerable size overlap in the individuals landed by basket trap, speargun and the reef seine with a mean size of 21.5 ± 0.1, 21.2

± 0.4, and 20.9 ± 0.5 respectively. Regarding the catch per unit of effort (Kg.fisher.trip^{-1}), the hook and line had the highest catch per fisher per trip (CPUE) while the basket traps, reef seine and beach seine had the lowest CPUE of less than 2kg per fisher per trip. Overall, fishers caught between 3-4kg per fisher trip.

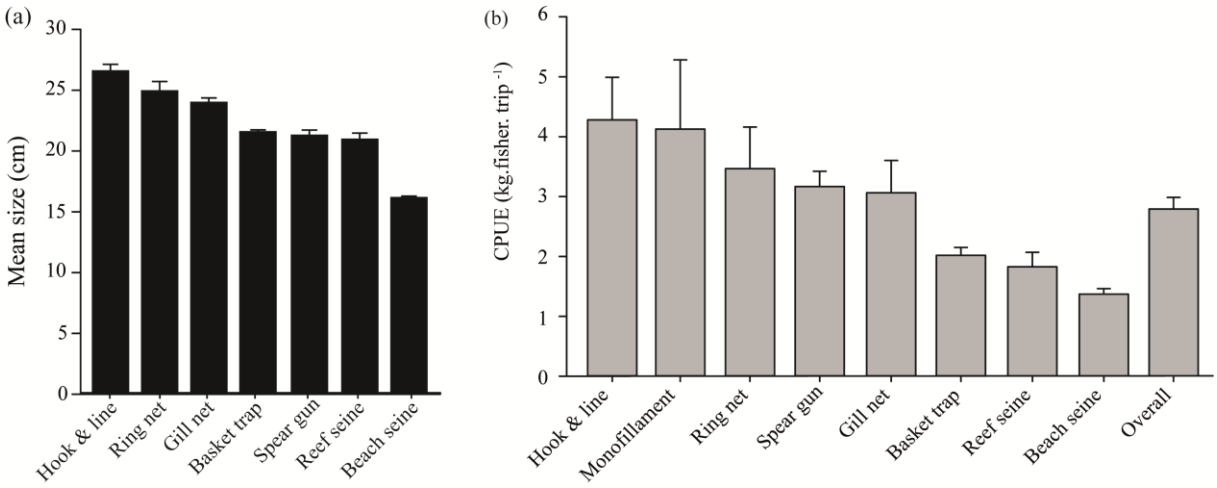

Figure 19. Mean size of individuals landed in the fishery by (a) gear with the error bars denoting standard error and (b) Catch per unit of effort.

3.3.4 Number of species and trophic level by gear

The number of species caught per day by the gears is quite varied. The reef seine caught the highest number of species while the monofilament net caught the lowest. On average ten species per day were captured by the reef seine, eight by the spear gun and seven by the beach seine. On average, the gears caught between three and eight species on a daily fishing trip (Figure 20a). Nevertheless, the hook and line and the monofilament nets targeted species with a higher trophic level compared to other gears. The basket trap and the spear gun targeted species with a lower trophic level as did the beach seine and the gill nets (Figure 20b).

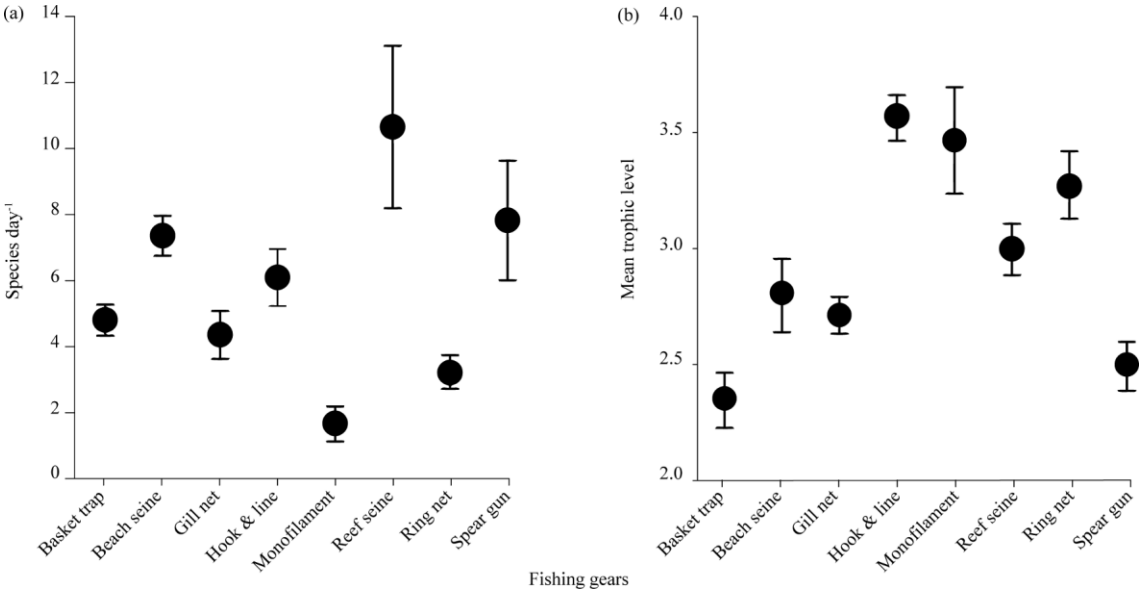

Figure 20. Plots for the (a) number of species and (b) mean trophic level for each of the fishing gears used in the fishery.

3.4 DISCUSSION

3.4.1 Species composition and gear overlap in the multi-gear fishery

Out of the one hundred and thirty eight species recorded from the artisanal fish landings, seventeen species dominated the catch contributing 90% and 70% of the overall abundance and weight respectively. Species belonging to the family Siganidae, Scaridae and Lethrinidae, dominated the catch and contributed most to the similarity between groups. Species belonging to these families represent the most abundant and commercially most important species for the artisanal fisheries (Hicks and McClanahan, 2012; Ntiba and Jaccarini, 1988). This pattern is typical of most tropical reef fisheries, characterized by a high diversity of species, but with a relatively small number dominating the catch (Gell and Whittington, 2002). Similar results have been observed in the Western Indian Ocean Region (WIO), including Zanzibar (de la Torre-Castro et al., 2014), Mozambique (Gell and Whittington, 2002) and Madagascar (Laroche and Ramananarivo, 1995).

According to Oostenbrugge et al. (2001), a number of factors seem to influence the catch composition including distance from shore and resource space available to

fishers. For example, the limited technological capacity of many small-scale in combination with the monsoon winds in the WIO region constraints the fishing activities to the shallow coastal lagoons. Therefore, most of the fishing operations are confined to the seagrass meadows as determined by tidal and diel cycles. As a result, the catches are dominated by seagrass fish assemblages and coral reef affiliated species such as species belonging to the family Siganidae and Lethrinidae, which use the seagrass meadows as a feeding habitat and are therefore more vulnerable to capture (Cinner et al., 2009; Unsworth and Cullen, 2010).

Nevertheless, despite the high diversity and similarities observed between regions, the species composition, size of fishes caught and the proportions of each species caught varies depending on gear type. As observed from this study, most fishing gears captured more than four species a day, an indication that the fishery targets a broad range of species and sizes. Nevertheless, some fishing gears specifically target particular species or groups (Bundy and Pauly, 2001). Fishers may alter their fishing gears, methods or location to target individual fractions or sections of the fish assemblage in response to changes in catch or market preferences (Stergiou et al., 1996; Thompson and Ben-Yami, 1984; Wright and Richards, 1985). These changes are made based on previous information gathered about the species and catch obtained by themselves and their peers (Salas et al., 2004).

According to Sadovy et al. (2003), market is an important driver in the exploitation of coral reefs, and this has been discussed for Zanzibar, Thyresson et al. (2013), Kenya Wamukota (2009) and in other fisheries around the world. Fishers tend to have a strong preference for particular commercial species. For instance, the white spotted rabbitfish, *Siganus sutor* fetches higher prices than the less popular trigger fish at the Kenyan coast (Mangi et al., 2007). In response to such preferences, and the need to minimize uncertainty and maximize income, artisanal fishers tend to employ multiple fishing gears that target various fish species instead of depending on only a single or a few target species. Also, the level of discard is considered low or negligible because there is a growing market for both high and low valued fish

spurred by the need to provide some protein for the majority of the poor population regardless of the size of fish (Obura, 2001; Thyresson et al., 2013). The implication is that the entire fish assemblage (species and size) is under increased pressure with each gear imposing different fishing mortality.

3.4.2 Targeting the small: effects of basket traps, beach seines, reef seine and spear guns

In the current study, we observed that the current fishing practices exploit not only fish species of small sizes but also small to medium sized specimens relative to the species potential maximum size. A comparison of the mean lengths of the various species caught by different gears in the study area indicated that in general, the beach seine caught the smallest individuals when compared to other gears, confirming results from previous studies (McClanahan et al., 1997; McClanahan and Mangi, 2004). This is typical of the beach seines, which are active fishing gears known for efficiently capturing the smallest, immature individuals of a species and are described as one of the most destructive gears to the reefs (King, 2000; Mangi and Roberts, 2006; McClanahan et al., 2005; McClanahan et al., 1997).

Nevertheless, basket traps have the same potential for damage when made with smaller mesh size (Mangi et al., 2007; McClanahan and Mangi, 2004). Our study also showed that the mean and modal lengths of the *Lethrinus harak* and *Lethrinus lentjan* captured by the basket trap were less than the L_m, indicative of their potential to target immature individuals. Though mostly considered benign (Mangi and Roberts, 2006; Shester and Micheli, 2011), traps have been recognized as a problem in the Caribbean notably for capturing numerous immature species that has led to a substantial reductions in size structure, total biomass and total catch of reef fishes (Mahon and Hunte, 2001; Munro, 1983).

Studies on trap selectivity by Bohnsack et al. (1989) and Gomes et al. (2014), have shown that increasing trap mesh size has the potential to reduce the chances of

overfishing and can optimize fishery resource by reducing juvenile and bycatch mortality. However, this is expected to have some short-term loss in revenue for fishers before catches return to the levels before the increase of mesh size and hence lead to non-compliance (Mahon and Hunte, 2001). An alternative measure would be to identify and protect nursery grounds of the most valuable species (Moran and Jenke, 1990).

In this study, we observed that basket traps maximized not only on the smaller size but also the most productive part of the system largely dominated by the smaller fish. Both the beach seine, basket trap, reef seine and the ring net tend to have a sigmoid selection curve, which means that they retain all sizes beyond the size at first capture. Smaller sized individuals dominate beach seines catches, an indication that they are usually used in shallow waters very close to the shore (Gillanders, 1997; Mangi and Roberts, 2006). The basket traps, to the contrary, are set in areas and at depths where particular fish targets are known to occur, hence considered more selective (Obura, 2001; Thompson and Ben-Yami, 1984). Due to the beach seine´s less selective nature combined with the higher overall number of species caught, its current banning appears justified as a measure for the conservation of demersal and inshore biodiversity. However, the damage caused by beach seine fishing of a vast number of species, of which some are targeted as juveniles while others are just species of small adult sizes, will ultimately depend on how much of the population the caught fish represent. Given the current estimates, beach seine only contributed 10% of the overall biomass landed, lower than the hook and line and the basket trap.

Despite the ban on the beach seine, it remains one of the most attractive gear to young fishers entering the fishery because of the low initial investment and reduced risk incurred by the crew (Mangi et al., 2007). Beach seine fishery is labor intensive requiring active crew members, with each crew consisting of between 10 to 15 fishers. Thus, making it more attractive to the young fishers despite the fact that the overall income per fisher is small compared to other gears. Therefore, depending

on the number of beach seine in a landing site, restricting fishers from using beach seine comes with a demand that to provide them with an alternative (Cinner et al., 2009). An alternative is to reallocate the beach seine effort to a legal fishing method such as the ring net fishery which bears similarities to the beach seine fishing regarding crew size, risk, and initial investment. The only potential challenges would have to do with the mode of operation, as ring nets are designed to catch pelagic species offshore at depths above 50 m. Therefore, it would require some level of swimming skills by the fishers and would involve more time investment, but the high catch volume associated with ring net fishery could be an attractive incentive for fishers. Nevertheless, gear exchange has had limited success in the WIO region because fishers have a strong preference to individual gears and are resistant to alternatives that require new skills and investments.

The results of this study show that spear gun catches were dominated by species with a low trophic level similar to the beach seine and basket traps catches. Further, the mean size of captured species between spear gun and basket trap catches were comparable. Nevertheless, the spear gun fishers caught more species per day and had a higher CPUE. This result may be explained by the fact that there is overlap in fishing sites between the fishers using the basket traps, spear gun and beach seine (Obura et al., 2002; Ruddle, 1987). Despite the spatial overlap, selectivity strongest for spear fishing, because the fishers determine the prey captured and have the possibility of targeting larger fishes compared with other fishing gears that explore the same habitats (Coll et al., 2004; Frisch et al., 2012; Meyer, 2007).

3.4.3 Targeting the large: hook and line, gillnet and monofilament fishery

Compared to other gears, the hook and line differed due to its distinct selectivity of bigger sized individuals yet overlapped with the monofilament net for the capture of species with higher trophic level and had higher catch per fisher effort (CPUE). Typically this gear is size-selective and has been shown to target species of the family

Lethrinidae and Labridae, which are carnivorous species of a high trophic level (McClanahan et al., 2008; Welch et al., 2010). This selective removal of species associated with a higher trophic level from a fishery has the potential to alter the food web leading to the phenomenon described as "fishing down the food web"(Pauly et al., 2001). Nonetheless, all fisheries are selective and likely to alter the trophic structure of an ecosystem, but it is essentially the amount of effort, which will decide if this is problematic for the fish species mainly targeted (Bundy et al., 2005).

In comparison, the monofilament nets tend to be monospecific capturing fewer species but larger in size with a corresponding high trophic level. Due to the perceived higher CPUE compared to the typical gill nets and lower relative cost of operation, the monofilament nets have become popular among fishers, and it seems like the gear might be replacing gillnets (FiD, 2015). However, monofilament nets have been banned due to their efficiency in capture attributed to their low visibility in water compared to typical gill nets (Collins, 1979; Maki et al., 2006). Also, the monofilament nets not only have the potential to damage and cause coral mortality in tropical coastal waters but also have a high propensity to generate conflict among resource users (Asoh et al., 2006).

Overall, the catches of hook and line, ring net, and the gill net are dominated by larger and higher trophic level species because they are used in relatively deeper compared to the other gears and their catches are considerably higher. Differences in species composition and biomass have been observed across the Great Barrier Reef, where larger species were associated with offshore reefs while the smaller species dominated inshore (Medley et al., 1993). Therefore, the observed pattern may be simply a result of the difference in fishing effort and the fishing gear used to spatial or temporal changes (Frédou et al., 2006).

3.4.4 Emerging patterns of the multispecies multi-gear fishery

Current fishing practices exploit not only small to medium sized individuals but also small to medium sized species relative to their maximum potential size. This pattern is not surprising considering that most of the catch come from coral reefs that also include vast areas of seagrass and sand habitats (McClanahan and Abunge, 2014). These habitats provide nursery and foraging sites for fish species but are also associated with intense fishing due to their proximity to the shore (Ramos et al., 2015; Teh et al., 2013). Local fishers are well aware of these characteristics and have adapted their fishing strategies to target these resources for particular fish taxa (Obura, 2001).

The variability of the species and size across the fishing gears is related to the characteristics of the fishing areas and their depth where the fishing occurs (Gobert, 2000). Several reef fish species show habitat segregation by size by using inshore nursery habitats such as seagrass beds and mangroves (Mahon and Hunte, 2001). In a previous study by Gillanders (1997), it was observed that smaller individuals were more abundant within near-shore seagrass habitats compared to the coral reefs. This is typical for species of emperors (Lethrinidae), snappers (Lutjanidae), goatfish (Mullidae) and rabbitfish (Siganidae) (Honda et al., 2013), species which contribute significantly to the overall artisanal catch in Kenya (Kaunda-Arara et al., 2003). The dominance of low–trophic level species in the catch such as the *Siganus sutor* and the *Leptoscarus vaigiensis* is reflective of an overexploited fishery because of their fast life history traits, which makes them tolerant to intense fishing (Camilo, 2015; McClanahan and Abunge, 2014).

Previous studies have also reported that majority of the fishing gears used in the Kenyan reef fishery are responsible for the capture of many immature and undersized fish, providing a potential for growth and recruitment overfishing (Mangi and Roberts, 2006). Similarly, our findings reveal that up to 90% of the catch contributing 66% of the total biomass is less than 30 cm in length an indication that the small to medium sized individuals dominate the catch. However, this does not

necessarily imply overfishing but may be attributed to the size-specific spatial and temporal variations in the distribution of fishes, and fishing activities (Jennings and Lock, 1996). Also, this may be an indication of a declining catch and not necessarily a sign of overfishing (Kolding et al., 2014).

It has been shown that so long as a sufficient spawning stock is maintained, the presence of immature fish in catches does not necessarily signify recruitment overfishing (Mahon and Hunte, 2001). A population may thus be subjected to high fishing mortality at certain life stages of the resources but yet does not show signs of overexploitation (Taylor et al., 2014). A recent theoretical exploration of the effect of the use of small gill nets in tropical fisheries (Wolff et al., 2015), has shown that large spawners and mega spawners may well be protected, even if fishing pressure is high. Therefore, as observed in this study, the very much size related and dominance of small and medium-sized fish may only be a pattern of fishing activity taking place.

The selective capture of individuals at certain sizes can alter the age, size and breeding structure of exploited populations in the short term (Garcia et al., 2012; Rodhouse et al., 1998), and further affect the growth traits of fish life history. The implications are varied but may include a smaller size-at-age, or an earlier age-at-maturation among other changes (Liang et al., 2014). However, harvested species respond differently to exploitation (Jennings and Kaiser, 1998; Jensen, 1991), and this makes it complicated to predict the allocation of fishing effort among alternative target species in multispecies fisheries (Salas et al., 2004).

3.4.5 Management considerations

The long-standing aim of fisheries management is to reduce the catch of too small individuals ($<L_m$) of the target species (King, 2007), achievable by either mesh size or gear restrictions. However, non-compliance with regulation has been noted as a major problem undermining the effectiveness of fisheries management in tropical fisheries. Also, the use of size restrictions has been considered an impractical management option due to the difficulty in enforcement of determining the

"optimum" mesh size for the multiple species (Munro, 1996). For instance, it will be difficult to protect species that mature at a larger size if species that mature at smaller sizes are exploited with appropriate mesh sizes (King, 2007; Mangi and Roberts, 2006; Misund et al., 2008). An implication of this is the possibility that a combination of mesh sizes be employed in the fishery because each species would have to have different regulations or catch limits. For developing countries, this may prove a challenge to implement given the number of species and economic constraints to invest in monitoring and enforcement (Hicks and McClanahan, 2012; Jones, 1984; Rhodes et al., 2007).

According to the FAO Code of Conduct for Responsible Fisheries, fishing gears, methods, and practices which are not consistent with responsible fishing should be phased out and replaced with more acceptable alternatives. Despite the fact that most developing countries have endorsed and adopted its guiding principles into their fisheries policy, the objectives have not been fully met (Greenberg and Herrmann, 1994; Hosch et al., 2011). Evidence from other studies has shown that gear restrictions are subject to fishers' creativity and manipulation and as such may lead to unintended consequences (Branch et al., 2006; Cinner et al., 2009). For example, it can be argued that the ban on beach seine in Kenya and its absence in particular fishing sites has provided an incentive for fishers to invest in small meshed traps and gill nets capitalizing on the lack of clearly defined policy to address mesh size for fish traps and gill nets (Waswala-Olewe et al., 2014).

Therefore, although the authorities attempt to restrict the use of illegal gears, poor enforcement coupled with the low incentive by the fishers to police the fishing grounds has only led to an increase in illegal fishing methods (FiD, 2015; Hilborn et al., 2004; Tuda and Wolff, 2015). A possible alternative to reduce non-compliance to illegal fishing has been proposed by Branch et al. (2006), who proposes a reduction in the benefits of illegal activities while providing incentives for legal fishing methods and reducing on unregulated inputs. By this, it is expected that the fishers will be discouraged by the high cost associated with illegal fishing and opt for the legal

fishing options. Since there is no fishing technique equally efficient in catching all sizes of fish (King, 2007). Gear diversification and adjusting fishing gears to target different ecosystem components in proportion to their natural productivity has been proposed as a better way to manage such a multispecies fishery (Bundy et al., 2005; Jones et al., 2009). Within this concept, it is expected that the effort on the selected target species will be greatly reduced while proportionally targeting all utilizable species in the fishery (Garcia et al., 2012; Zhou et al., 2010).

The evidence from this study appears already close to an unplanned balanced harvest as it is likely that fishers have already adapted their fishing techniques and strategies to target if not all but significant components of the available resource and size spectrum, by using different gear dimensions, duration, and depths (Gobert, 1994). However, with the uncertainty surrounding the proportions of each species caught, the results from these findings must always be interpreted with caution as the current data may be too crude to make such a conclusion. Nevertheless, with the already existing market for small fish in developing countries; the balanced harvesting approach is seen an alternative approach particularly for the tropical multispecies and multigear artisanal fisheries in the Asia and Africa. However, one major drawback of this approach is that it would require major policy changes and at the moment the steps to implement balanced harvesting are still difficult to justify (Burgess et al., 2015; Garcia et al., 2012).

Given the current fishing patterns exhibited by this artisanal fishery, it seems that the current fishing regulations emphasizing on size limits and gear restrictions are not adequate for this multispecies and multi-gear fishery. An alternative may be to complement these measures with market-based measures that extend down the value chain (Thyresson et al., 2013). These measures may be rare in tropical fisheries, but they have been shown to have the potential to reduce unsustainable fishing practices while providing an enforcement mechanism for gear restrictions (Rhodes et al., 2007). Also, Frédou et al. (2006), showed that spatial variability played a significant role in differentiating catch composition in reef fishery. Therefore, having

a common harvesting strategy may not be applicable for an entire fishing area, thus further attention should be given to area-based management since the effort may already be spatially distributed (Hilborn and Walters, 1992). This will build on the current policy guidelines that support the establishment of locally managed areas and allow for the participation of the local resource users while promoting conservation. Nevertheless, there has to be an emphasis on enforcement as this is to any successful fisheries management system.

3.5 CONCLUSION

In general, tropical multispecies fisheries exhibit variable fishing practices and fishers target specific species that reflect not only what can be caught, but the desirability of the species to the market (Medley et al., 1993). As shown in the results, the majority of the target species are small with a maximum reported size less than 55 cm or less. Thus, it is not surprising to observe overlap in the catch and size distribution as fishers may have adjusted their effort to exploit these resources based on their availability, ease of capture or market preferences. An interesting question would be to determine whether the currently observed fishing patterns have developed mainly as a response to the resource availability, fisheries regulations or market preferences.

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CHAPTER 4.

Adapting length-based stock assessment for improved management of four coral reef target fish species in Kenya

Tuda, P.M, Breckwoldt, A, Wolff, M. Adapting length-based stock assessment for improved management of four coral reef target fish species in Kenya. *Manuscript submitted to Fisheries Management and Ecology.*

ABSTRACT

The present study makes use of two length-based fishery assessment techniques to infer the status of four target species in an artisanal fishery in Kenya and provide insights on the species' vulnerability to exploitation. In the first approach, we estimated the von Bertalanffy growth parameters K and $L\infty$ based on length frequency data collected over a one-year period to derive annual mortality rates for *Siganus sutor*, *Leptoscarus vaigiensis*, *Lutjanus fulviflamma,* and *Lethrinus harak* using the length converted catch curve (LCC). The estimated growth parameters $L\infty$ and K differed between species but were within the biological limits for the species (i.e., lower K for relatively long-lived and slow growing species and vice versa). However, when contrasted with local and regional estimates, the results revealed a range of possible combinations of K and $L\infty$, with $L\infty$ indicating that these parameters may vary depending on the level of fishing pressure. The length composition of the catch revealed a distinct similarity in the length classes with a skew toward smaller sized individuals (< 20 cm TL), with up to 60% of all individuals caught being immature, indicating an unsustainable fishery. However, the mean exploitation rate (E) derived from a combination of empirical natural mortality (M) estimators for the four species did not support a significant growth overfishing ($E < 0.5$). However, comparing these findings with the length based spawning potential ratio approach (LB-SPR) and setting $SPR_{40\%}$ (a proxy for MSY) as the target for the fishery, the emerging picture is quite different. It would appear that three of our species are overexploited; in fact, the current SPR estimate for *L. fulviflamma* and *L.harak* would suggest both growth and recruitment overfishing. Discordance in results between the two approaches confounds a definitive conclusion regarding the status of these exploited species. Nevertheless, we noted that the outstanding issue was weakness in the use of length-based approaches in estimating growth parameters and, in using these estimates for further analyses of long-lived species in particular. Therefore, applying more conservative estimates resulted in more realistic results that were comparable between the two approaches, suggesting that the high fishing effort and progressive

capture of immature individuals may have contributed to growth and recruitment overfishing of these key species as reflected from previous studies. Therefore, reversing the adverse effect of the fishery on these species would likely require a combination of management tools targeted at regulating fishing mortality. Moreover, it is imperative that any size-based regulation must be applied to fishing effort, but in our view effort reduction would seem to be a more appropriate management option given the challenges of implementing size restriction in a multi-species multigear fishery. Obviously, the above results are presented bearing in mind the data-limited aspect of this study (12 months of data for four species). Nevertheless, our results highlight the possibility of assessing data-limited fisheries and the importance of exploring different approaches in fisheries assessments.

Keywords: Reef fishery, SPR, artisanal, assessment, multispecies, data-limited

4.1 INTRODUCTION

Coral reef fisheries play a vital role as a food source, income, and livelihood for millions of coastal people (Spalding et al. 2001), yet their status remains largely unknown, and many are considered unsustainable and mismanaged (Choat and Robertson, 2002; Newton et al., 2007). According to Costello et al. (2012), close to 80% of the global fisheries catch come from fisheries lacking formal assessment. The majority of these are artisanal, exploiting nearshore tropical coastal ecosystems such as coral reefs and estuaries. Although they support the nutrition of a largest part of the population in these countries, they generate low 'visible' revenues and as such receive only low priority on governmental agendas (Honey et al., 2010; Pauly and Zeller, 2016). Consequently, most of these fisheries still remain inadequately assessed either due to a lack of reliable data of the exploited stock or, if sufficient catch data are available, these are often highly aggregated and inadequate for stock assessment (Hilborn and Walters, 1992; Ricard et al., 2012; Rosenberg et al., 2014). The implication is that overfishing worldwide may go on unrecognized due to management decisions based on lack of adequate information (Apel et al., 2013; Currey et al., 2013; Hordyk et al., 2015; Maunder et al., 2006; Prince et al., 2015).

Proper evaluation and management of a fishery require basic information on the ecology, life history, and population structure of the exploited species (King, 2007; Perry et al., 1999). This information is vital for assessing stock conditions and determining possible shifts in community structure, and is still instrumental in improving fisheries management decisions (Currey et al., 2013; Jennings et al., 1999). However, for most developing countries, the costs and logistic constraints of conducting long-term fisheries monitoring have become major impediments to fisheries management (Prince et al., 2015). In Kenya, like in most tropical countries, artisanal fisheries have played a significant role in the culture and livelihoods of coastal communities, but they still receive low priority at the national level (Hoorweg et al., 2003; Le Manach et al., 2015; Malleret King, 2000).

Recent marine frame and catch assessment surveys (FS and CAS) conducted by the Fisheries Department, Kenya (FiD) estimated that Kenyan artisanal fisheries employ ~14,000 fishers, of which 95 % are considered artisanal (FID, 2014). Over the past decades, increasing evidence of fishery decline has been reported (McClanahan et al., 2008), with some commercially important fishery resources such as groupers (Serranidae) decreasing by up to 80% (Agembe et al., 2010; Kaunda-Arara et al., 2003). An earlier landings data-based assessment of the fisheries by Kaunda-Arara et al. (2003) revealed a decline in yields over time and reported yield levels ranging between 2-4 metric t/km^2/year. These estimates are indicative of a declining fishery when considering that Munro (1983) estimated the potential yield from coral reefs to be within the range of 10-20 t/km^2/year.

Despite the mounting evidence of overfishing and the risk associated with loss of livelihoods, the condition of most reef fish stocks in Kenya is still largely unknown (Kaunda-Arara and Ntiba, 2006; Ntiba and Jaccarini, 1988). Out of the 121 commercially exploited species, only about 45 species have been studied regarding their biology (Fondo et al., 2014). More specifically, information needed for conventional stock assessments are available for only a small number of commercially important fishes (De Young, 2006; Kaunda-Arara and Rose, 2006; Ntiba and Jaccarini, 1988). Given the contribution of these resources to the livelihoods of the coastal communities, it is still critical to assess these fisheries even with the limited data available (Honey et al., 2010). For this reason, a number of data limited approaches have been developed to address the problem of assessing fish stocks under varying levels of data scarcity (Froese et al., 2016; Prince et al., 2015). Such alternatives are not a replacement for stock assessment per se but are meant to elucidate the impacts of fishing while providing a timely and efficient fishery management strategy under increased exploitation settings where data remains scarce or absent (Hordyk et al., 2015; Kleisner et al., 2013).

In the present study, two length-based assessment tools are applied to (1) assess the current status of the artisanal fishery in the in the Kenyan Coast using

length frequency data from a multispecies multigear fishery and (2) make management recommendations for the sustainable use of these stocks. The spawning potential ratio (SPR) approach (Goodyear, 1993) was used to evaluate the regulatory options for the fisheries under different fishing mortalities. Given the high exploitation rates reported for some of the species in this fishery (Hicks and McClanahan, 2012), it is highly likely that a shift may have occurred in the catches from species of low productivity, higher trophic levels, and larger size to those of relatively higher productivity, lower trophic levels, and smaller size. Therefore, the main hypothesis of our study is that the highest exploitation rates with corresponding lowest spawning potential are to be found in the group of fish in the higher trophic level.

4.2 MATERIALS AND METHODS

4.2.1 Study site and field methods

This study was conducted in the Diani-Chale area located approximately 30 km South of Mombasa (Figure 21). The region is known to support one of the oldest artisanal fisheries in Kenya and is considered to be among the most degraded reef areas off the East African coast (McClanahan et al., 1997; McClanahan and Obura, 1995; Tuda et al., 2008). Fishing here is a typical coral reef fishery characterized by multiple species and gears and is carried out close to the shore at a depth between 0.5–3 m at low tide (McClanahan and Mangi, 2004; Tuda et al., 2016). Four landing sites were selected for this study (Mwaepe, Chale, Gazi, and Msambweni; Figure 21), based on their active fishing operations, high fisher density, and proximity to each other, and are considered as representative of the artisanal reef fisheries in Kenya.

Typical fishing gears used in the area include traditional basket traps, gill nets of varying mesh sizes, the more active beach seines and ring nets, speargun, and hook and line. The description, mode of operation, and legal status of these gears have been extensively reported by McClanahan and Mangi (2004) and Samoilys et al.

(2011). These different gears and mesh sizes allow for the capture of specific targets of species and sizes of fish along a large size spectrum of the fish community (Tuda et al. 2016).According to the fisheries department's marine frame survey report FID (2014), there are 16 different gill nets with varying mesh sizes ranging from < 2.5" to 10" (6.35 cm to 25.4 cm) in addition to monofilament gill nets. Other gears include spear guns and hook and line.

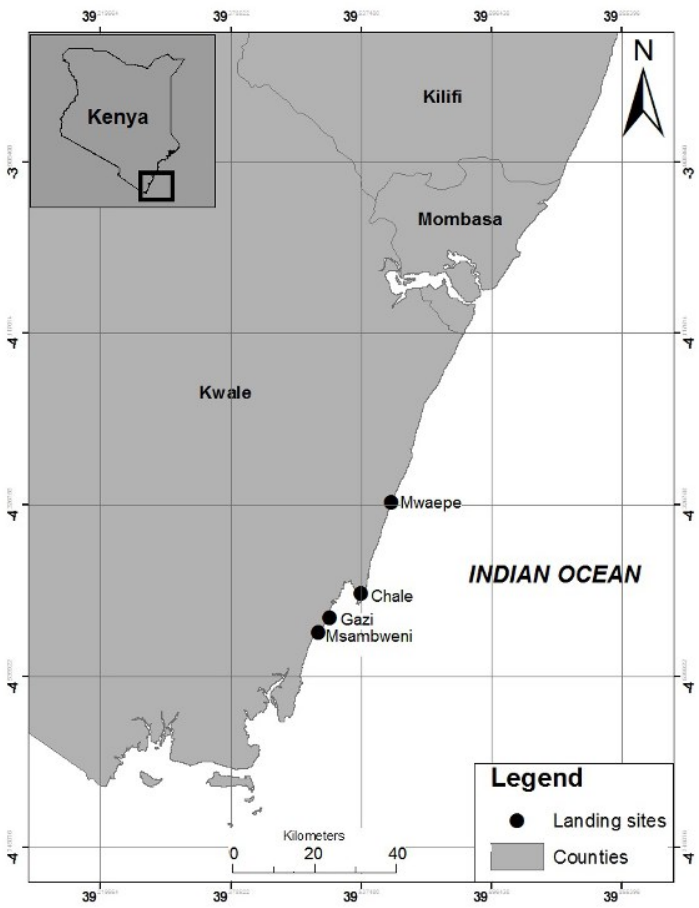

Figure 21. Map of the Kenyan Coast showing the landing sites where data was collected.

Fish sampling was undertaken at the four landing sites over a one year period (June 2014-June 2015) and consisted of 40 visits and 6,531 fish sampled. During this time, the total fish length (to the nearest 0.1 cm) was measured from the snout to tip of the longest caudal lobe using a measuring board. All fish landed by the ~120 fishers using different gears were sampled, but in cases where similar gears landed simultaneously or where large catches were landed, a random sub-sample was

measured and standardized in relation to the total catch landed from that gear. All sampled fish were identified to species level, and the four most abundant species were selected for further analysis – the white spotted rabbitfish *Siganus sutor*, the marbled parrotfish *Leptoscarus vaigiensis*, the dory snapper *Lutjanus fulviflamma*, and the thumbprint emperor *Lethrinus harak*. These four species represented more than 75% by weight of all landings during the period 1997-2008 (Hicks and McClanahan, 2012) and represented 65% by abundance in an earlier study in the same area (Tuda et al., 2016).

4.2.2 Length frequency analysis and estimation of growth parameters

The length data for the four species was grouped into class intervals depending on their maximum size, with interval widths of 1 cm for fish to 30 cm, 2 cm for fish to 60 cm, and 5 cm for fish of lengths > 60 cm TL (Neumann et al., 2012). For this study, the von Bertalanffy growth function (VBGF; (von Bertalanffy, 1938), was fitted using the electronic length frequency analysis routine ELEFAN (Pauly and David (1981). The VBGF is as expressed as:

$$L_t = L_\infty\left[1 - e^{-K(t-t0)}\right]$$

where L_t is the length at time t (years), $L\infty$ is the asymptotic length (cm), K is the growth coefficient (or curvature parameter) (yr^{-1}), and t_0 is the hypothetical age when length would be zero. For a detailed description of the theory and assumptions behind ELEFAN see Pauly (1987) and Pauly and David (1981). The ELEFAN method takes seed values for the growth parameters, from which growth curves are generated that cut through the modal groups of the length frequency samples. Through an iterative procedure, those values for the growth parameters K and $L\infty$ are then varied to find the values which yield the highest goodness of fit index value (Rn) for the growth curve (Gayanilo et al., 2005).

The maximum lifespan (t_{max}) per species was calculated using the empirical relationship of Taylor (1958):

$$T_{max} = t_0 + \frac{2.996}{\mathrm{K}}$$

Derived values of L∞ and K were compared to those from previous stock assessment studies of the same species and the confidence interval was fitted around those values for comparative purposes. The phi-prime index (Φ`) was calculated according to the formula by (Pauly and Munro, 1984). This is an index that allows for comparing growth performance of a species based on previous estimates and is denoted by the formula:

$$\emptyset' = \log \mathrm{K} + 2 * \log \mathrm{L}\infty$$

where K is the growth coefficient and L∞ the asymptotic length as defined above.

4.2.3 Estimation of mortality and exploitation rate

Length-converted catch curves have become the standard for estimating mortality for fish stock assessments from length frequency data when the growth parameters L∞ and K are known (Gulland and Rosenberg, 1992; Pauly et al., 1995). In this method, it is assumed that the population is in a steady state, that the sample represents the population structure of the harvested part of the stock, and that the annual instantaneous rate of total mortality (Z) is constant over all the exploited size classes (Pauly, 1987). The regression line was fitted to the data that exclude the initial ascending data points representing groups of individuals that are either not fully recruited or are too small to be vulnerable to the fishing gear. Moreover, data points close to L∞, where the relationship between length and age becomes uncertain, were also excluded (King, 2007; Sparre and Venema, 1992).

Natural mortality (M) is one of the most important parameters in stock assessments, and yet is difficult to estimate reliably and directly (Kenchington, 2014; Pauly, 1980; Then et al., 2014). Despite the existence of several empirical methods to

indirectly estimate M, Pauly's method based on growth parameters and water temperature (Pauly, 1980) and Hoenig's age-based methods (Hoenig, 1983) remain the most widely applied (Kenchington, 2014). However, there is no consensus on why the estimates of M vary across different methods, or which method should be preferred (Then et al., 2014). In this study, we used two approaches to determine the instantaneous rate of *M*. The first estimate was based on Pauly (1980) approach:

$$\ln \mathrm{M} = -0.0152 - 0.279 * \ln \mathrm{L}_{\infty} + 0.6543 * ln\,\mathrm{K} + 0.4634 * lnT$$

where L∞ and K are the von Bertalanffy growth parameters (Eq. 1), and T is the annual average sea surface temperature given as 27°C for the Kenyan coast and region (Kamukuru et al., 2005; Kaunda-Arara et al., 2003). The second method applied to estimate M was the updated version of Hoenig's formula by Then et al. (2014) based on a review of over 200 species and is given as:

$$M = 4.899 t_{max}^{-0.916}$$

where t_{max} is the maximum age.

Fishing mortality (F) was computed as F=Z-M (Gulland, 1970) and the current exploitation rate was calculated as the proportion of fishing mortality (F) relative to total mortality (Z):

$$\mathrm{E} = \mathrm{F}/\mathrm{Z} \quad . \qquad (6)$$

4.2.4 Spawning potential ratios (SPR)

To evaluate the impact of fishing on the spawning stock, we applied the length-based spawning potential ratio (LB-SPR) model. By definition, SPR is the ratio of the total reproductive production at equilibrium for a given level of fishing mortality divided by the reproductive production in the unfished state (Goodyear, 1993; Mace and Sissenwine, 1993). The method relies on the life history ratios M/K and Lm/L∞, and

the shape of the population's size structure to estimate the ratio of fishing and natural mortality (F/M) for the final calculation of the SPR (Hordyk et al., 2016; Prince et al., 2015).

The model is sensitive to biased estimates of the growth parameters and can lead to an overestimation of the SPR. Therefore, to account for uncertainty in the estimation of the growth parameters, we compiled these estimates based on our current study derived by the ELEFAN routine and contrasted them with several growth parameters derived from local comparative studies for these species to generate lower and upper estimates of the input parameters. The mean between the range of values was assumed to be the best estimate (Table 3).We applied the LB-SPR model to the length frequency data, and an SPR target (SPR_{targ}) was set to 0.40 ($SPR_{40\%}$), which is considered as a conservative proxy for maximum sustainable yield (Hordyk et al., 2015). The lower limit was set at 0.20 ($SPR_{20\%}$), a threshold point when recruitment rates are likely impaired (Mace and Sissenwine, 1993; Prince et al., 2015).

Table 3: Compilation of growth and mortality parameters from previous studies of *Siganus sutor*, *Leptoscarus vaigiensis*, *Lutjanus fulviflamma*, and *Lethrinus harak* exploited by the artisanal fisheries in Kenya, and compared to other regional estimates.

Species	Asymptotic Length (cm) (L∞)	Growth Constant (K)	Mortality Rate (Annual)			Exploitation rate (E)	M/K	F/M	SL50/L∞	Reference
			Natural (M)	Fishing (F)	Total (Z)					
Siganus sutor	45.9	0.87	-	-	-	-	-	-	-	Ntiba and Jaccarini, 1988
Siganus sutor	40.0	0.80	1.45	1.59	3.04	0.52	1.81	1.10	0.62	de Souza, 1988
Siganus sutor	43.6	0.80	1.34	-	-	-	1.68	-	0.46	Nyang'wara, 2002
Siganus sutor	39.9	0.52	1.15	1.44	2.59	0.56	2.21	1.25	-	Kaunda-Arara et al., 2003
Siganus sutor	36.3	1.20	1.87	-	-	-	1.56	-	-	Kaunda-Arara and Rose, 2006
Siganus sutor	36.2	0.87	1.49	1.66	3.15	0.53	1.71	1.11	-	Hicks and McClanahan, 2012
Siganus sutor	37.2	0.72	1.36	1.25	2.58	0.48	1.89	0.92	0.36	This study
Leptoscarus vaigiensis	28.9	1.50	2.30	1.15	3.52	0.33	1.53	0.50	0.40	Mwatha, 1997
Leptoscarus vaigiensis	34.1	1.31	2.02	-	-	-	1.54	-	0.40	Nyang'wara, 2002
Leptoscarus vaigiensis	36.6	0.49	0.98	2.26	3.24	0.70	2.00	2.31	0.37	Hicks and McClanahan, 2012
Leptoscarus vaigiensis	30.8	1.95	2.70	6.45	9.15	0.70	1.38	2.39	-	Locham, 2016
Leptoscarus vaigiensis	29.1	0.76	1.48	1.39	2.87	0.48	1.95	0.94	-	Locham, 2016
Leptoscarus vaigiensis	32.2	0.43	0.88	0.66	1.54	0.43	2.05	0.75	0.40	This study
Lutjanus fulviflamma	35.0	0.49	1.70	0.27	1.97	0.14	3.47	0.16	-	Kaunda and Ntiba 2001
Lutjanus fulviflamma	29.0	0.15	0.27	1.37	1.64	0.84	1.80	5.07	-	Kamukuru et al., 2005
Lutjanus fulviflamma	29.0	0.15	0.27	0.28	0.55	0.51	1.80	1.04	-	Kamukuru et al., 2005
Lutjanus fulviflamma	34.1	0.19	0.41	0.38	0.79	0.48	2.16	0.93	0.37	This study
Lethrinus harak	37.3	0.89	1.50	-	-	-	1.69	-	0.44	Nyang'wara, 2002
Lethrinus harak	36.2	0.21	0.45	0.46	0.91	0.51	2.14	1.02	0.28	This study

4.3 RESULTS

4.3.1 Size structure of catches

The length-frequency distribution was graphically compared between the species based on binned length classes of 2 cm (Figure 22). There was a distinct similarity in the length classes with a skew towards the smaller sized individuals and a peak between length classes (12 - 15 cm TL). The species *L. vaigiensis* and *L. fulviflamma* covered a narrow length distribution (<30 cm TL) while *S. sutor* and *L. harak* had a wider length distribution, showing a bi-modal distribution with peaks at both TL 12 to 15 cm and TL 27 to 30 cm length classes.

There was very little difference between the species in mean length, with the overall mean size of 18.3 ± 2.5 cm TL. The length at first capture SL_{50} was derived using a backward extrapolation of the descending limb of the length-converted catch curve. Estimates of SL_{50} ranged from 10.5 cm TL for *L. harak* to 13.5 cm TL for *S. sutor* (Figure 22 e-h). Most of the individuals recruited to the fishery at a mean size of capture which was considerably smaller (SL_{50} = 12.2 ± 1.3 cm) than the respective mean size at maturity (L_{50} = 22.3 ± 4.7 cm). Overall, the SL_{50} for most of the species was less than 40% of L_{50}, an indication that a large proportion of the individuals caught in all gears were immature (SL_{50}< L_{50} = 60%).

Based on gonadal studies from our study area, it was found that *S. sutor* from Kenyan marine waters reaches 50% maturity at a mean size of ~24.2 cm TL, comparable to *L. fulviflamma* (~24.3 cm TL) and *L. harak* (~25.3 cm TL) respectively. The proportion of individuals with the TL < L_{50}, was greatest for *L. fulviflamma* (TL < L50 = 97.5 %) and lowest for *L. vaigiensis* (TL < L50 = 40 %). In between, *S. sutor* and *L. harak* had TLs < L_{50} for up to 65 % and 77 % of individuals, respectively.

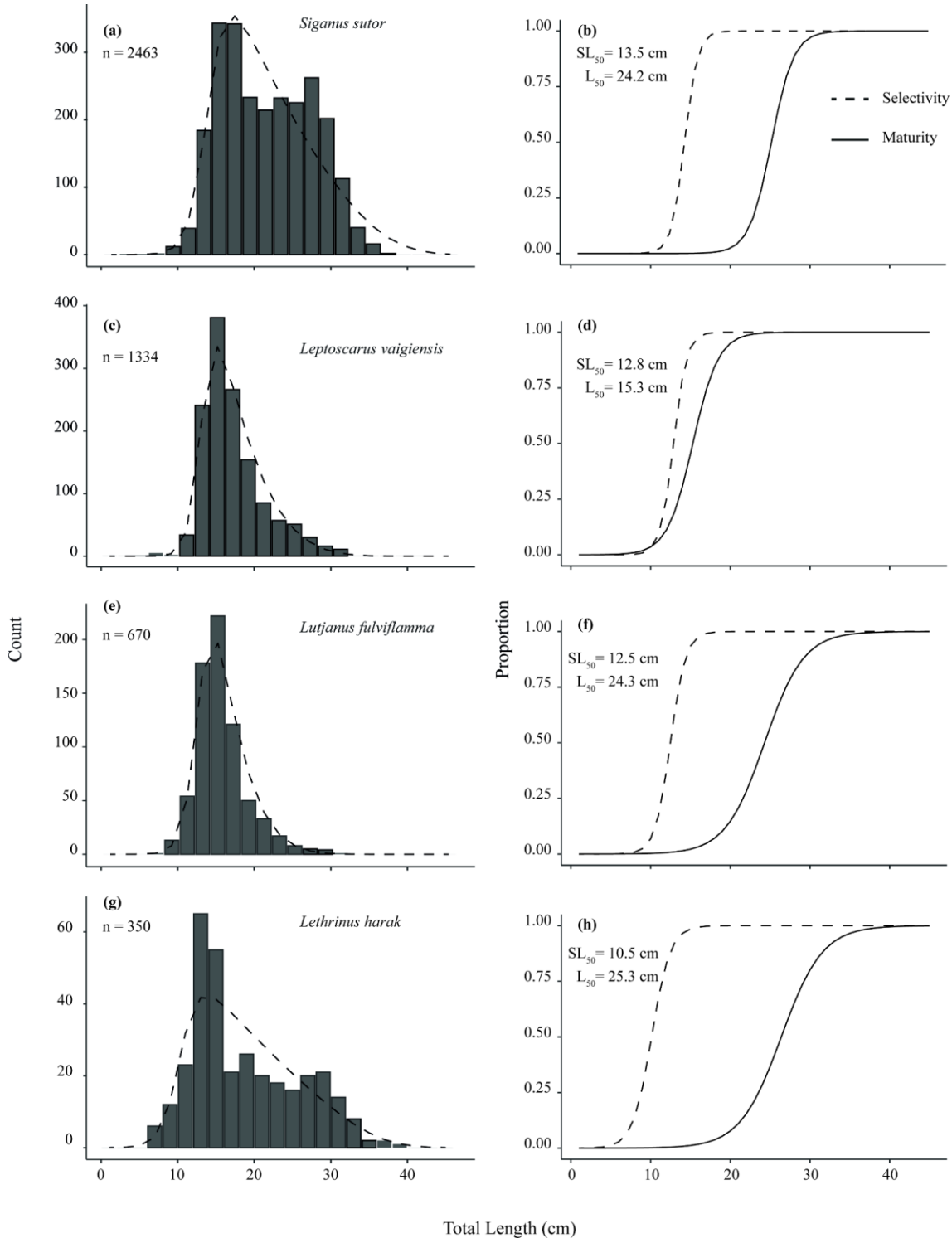

Figure 22. Cumulative length-frequency histograms (left column) with fitted size composition curves (dashed lines) for the four selected species (a) *Siganus sutor*, (c) *Leptoscarus vaigiensis*, (e) *Lutjanus fulviflamma,* and (g) *Lethrinus harak*; and their respective (right column; b, d, f, and h) fitted size at first capture (dotted line) and size at maturity (solid lines) fitted by the LB-SPR assessment software.

4.3.2 Growth parameter analysis

Estimates of the growth parameters generated from the ELEFAN application to the length-frequency data of the four reef species and the derived growth performance index (Ø') are presented in Table 4. Estimates of $L\infty$ and K differed substantially between the species. $L\infty$ ranged from 32.2 cm TL for *L .vaigenesis* to 37.2 cm TL for *S. sutor*. K ranged from 0.19 to 0.72 for *L. fulviflamma* and *S. sutor,* respectively, with comparatively lower K values for *L. fulviflamma and L. harak* (0.19-0.21).

The values of the hypothetical time at which length is equal to 0 (t_0) were all negative, with the values for *S. sutor* and *L. vaigenesis* closer to 0 (-0.21 and -0.37), while those of *L. fulviflamma* and *L. harak* were slightly lower (-0.86 and -0.76).

The growth parameters for the selected species, contrasted with those from other local and regional studies (Table 4), varied, with a wide range of possible combinations of K and $L\infty$. The values of $L\infty$ were more varied compared to K (Figure 23). For instance, estimates for *L. harak* derived in the current study were relatively high (range of $L\infty$ from 28.1 to 37.3). K estimated for *L. fulviflamma* (K=0.19) was relatively lower (range of K from 0.15 to 1.0).

Overall, the growth performance index for *S. sutor* was comparatively higher (Ø' = 2.99) than that of the other species, another indication of a high turnover rate. Estimates of longevity (t_{max}) determined empirically were wide-ranging between the species but comparable to other estimates derived from regional studies (Table 4). The value ranged from four years for *S. sutor* to 14.9 years for *L .fulviflamma*. Those for *L. vaigenesis* and *L. harak* were 6.6 and 13.5 years, respectively.

Table 4: Estimated growth parameters and growth performance indices of the four most abundant species as derived from the ELEFAN analysis in the present study, and range of estimates from comparative studies summarized in Table 3.

Estimated Parameters (this study)							Literature estimates					
Species	$L\infty$	M/K	F/M	t_0	Ø'	t_{max} (yrs)	L_∞ lower	L_∞ Best	L_∞ upper	M/K lower	M/K best	M/K upper
Siganus sutor	37.2	1.9	0.9	-0.21	2.99	4.0	35.7	40.7	45.9	1.6	1.8	2.2
Leptoscarus vaigiensis	32.2	2.0	0.8	-0.37	2.65	6.6	25.9	30.7	36.6	1.5	1.7	2.1
Lutjanus fulviflamma	34.1	2.2	0.5	-0.86	2.35	14.9	29.0	32.7	35.0	1.8	2.4	3.1
Lethrinus harak	36.2	2.1	0.9	-0.76	2.43	13.5	36.2	36.8	37.3	1.7	1.9	2.1

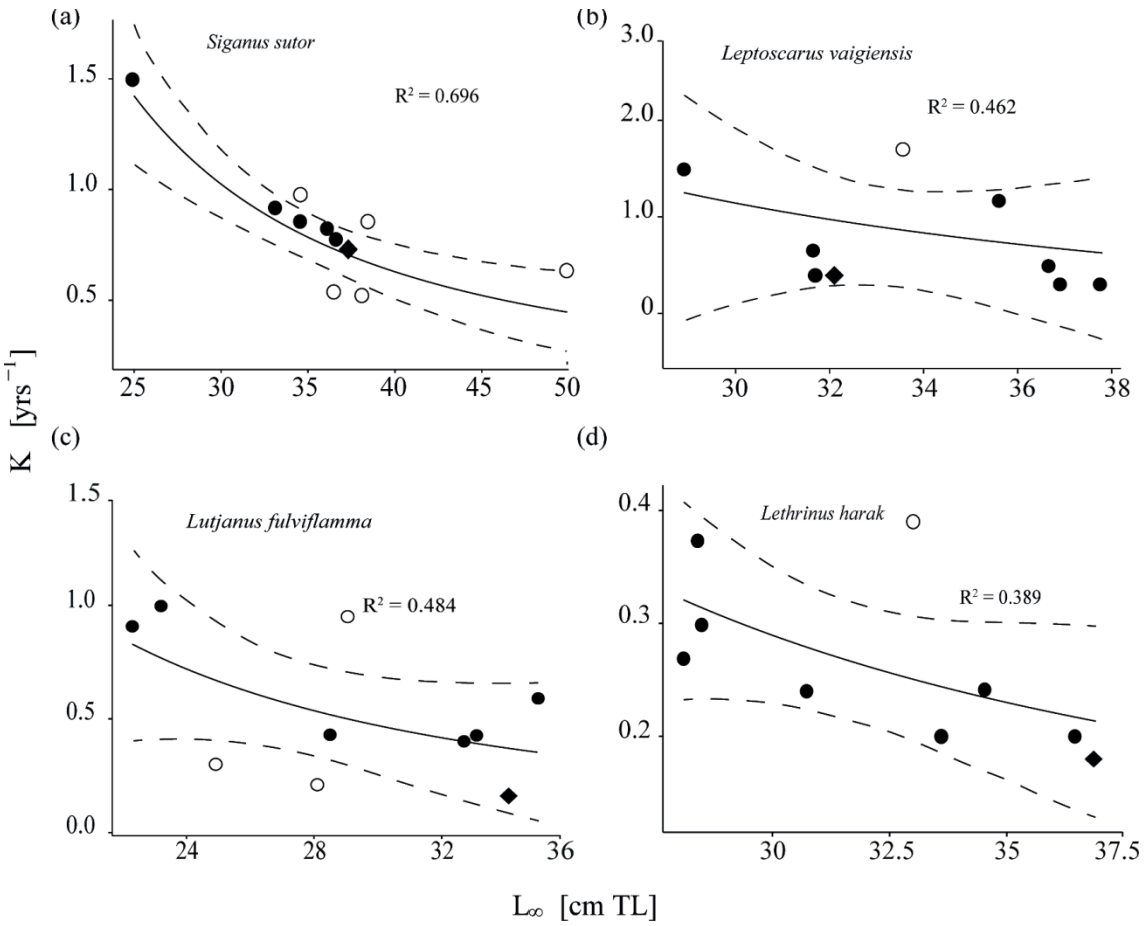

Figure 23. Estimates of growth coefficient (K) and the asymptotic length (L∞) for the four target species derived from published literature and this study, including the 95% confidence intervals surrounding the parameters *K* and *L*∞ (♦); data points included in regression (●); data points not included in regression (○).

4.3.3 Stock assessment

In Table 5, the estimates of total mortality (*Z*) derived from the linearized catch curve, as well as estimates of the natural mortality (M) and fishing mortality (F) are shown. Estimates of total mortality (Z) from the slope of the descending limb of the catch curve differed greatly between the species (Figure 24). *S. sutor* and *L. vaigiensis* had higher total mortality rates compared with *L. fulviflamma* and *L. harak* (Figure 24, Table 6).

By comparison, *M* derived from the two empirical methods did not vary much for *S. sutor* (*M* = 1.33-1.39) and *L. vaigiensis* (*M* = 0.87-0.88) but values varied widely for *L. fulviflamma* (0.41-0.59) and *L. harak* (*M* = 0.45-0.51**)** **(**Table. 5). *M* derived from Pauly's (1980) equation (Eq. 4) were slightly higher for the relatively long-lived species and could represent overestimates. Then's (2014) approach provided a more conservative estimate of *M* for these species.

Variability between the two estimators of *M* resulted in differences in the estimates of *F* and *E* (Figure 25). The estimates of *E* for *S. sutor* ranged from 0.46 ± 0.04 to 0.48 ± 0.04 sd with a mean of 0.47 ± 0.04 sd, giving an indication of a moderate exploitation rate. By comparison, *E* estimates for *L. fulviflamma* were lower and ranged between 0.24 ± 0.13 sd to 0.47 ± 0.09 sd with Eqs. (5) see (Figure 25, Table 5).

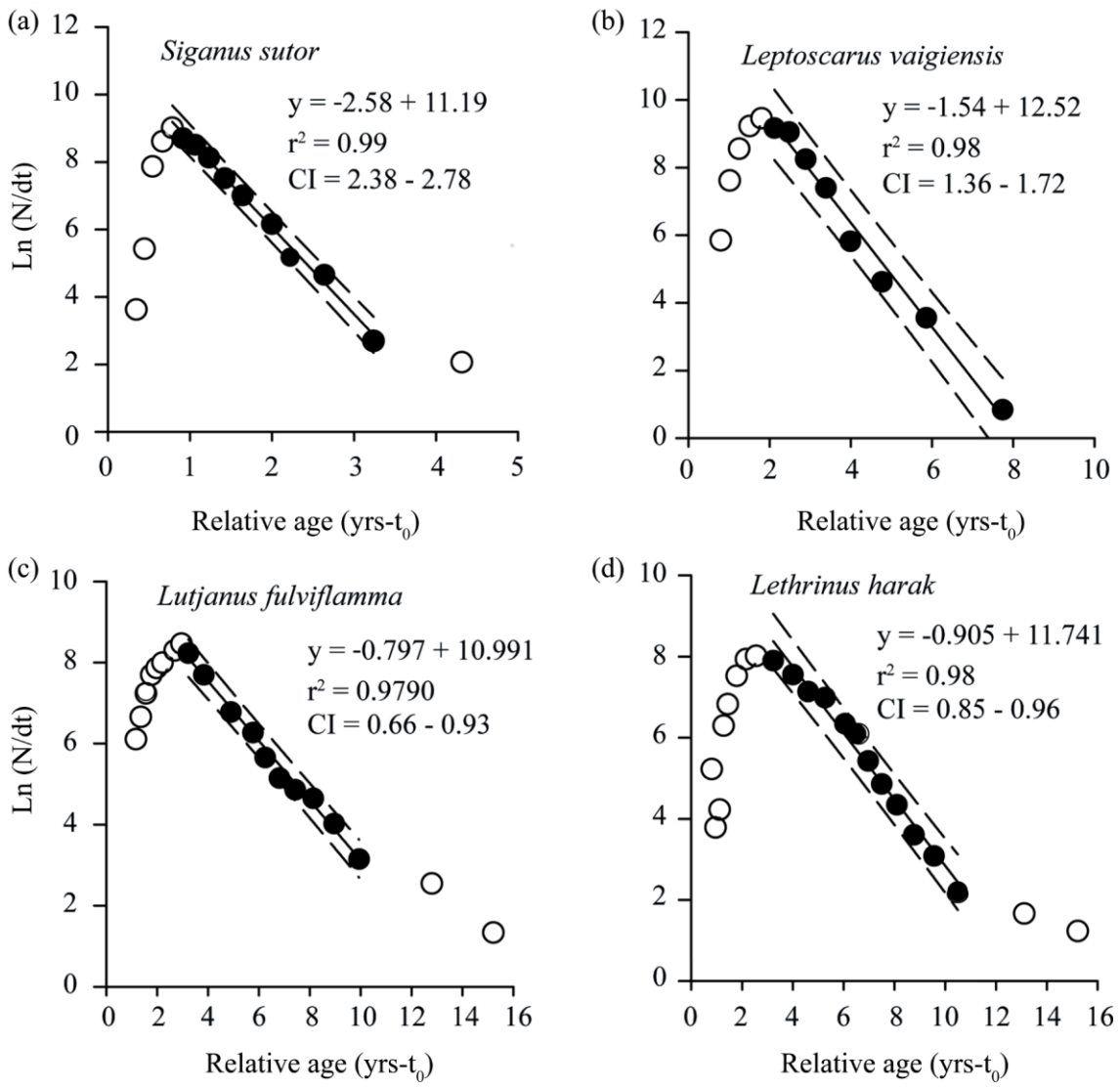

Figure 24. Catch curves for the four most abundant species in the catch from the artisanal fisheries catch data (2013-2014). The slopes of the regressions provide an estimate of the rate of total mortality (*Z*) for each species. The dotted line represents the 95% confidence interval around (Z). Note: only solid data points (●) have been included in the regressions.

Table 5: Mortality and exploitation rates for *S. sutor*, *L. vaigiensis*, *L. fulviflamma*, and *L. harak* with the confidence interval (CI) in parenthesis

Species	Pauly (1980)			Then (2014)	
	Total Mortality (*Z*)	**Natural mortality (*M*)**	**Fishing mortality (*F*)**	**Natural mortality (*M*)**	**Fishing mortality (*F*)**
S. sutor	2.58 (2.38 - 2.78)	1.33	1.25 (1.05 - 1.45)	1.39	1.19 (0.99 - 1.39)
L. vaigiensis	1.54 (1.36 - 1.72)	0.88	0.66 (0.48 - 0.84)	0.87	0.67 (0.49 – 0.85)
L. fulviflamma	0.79 (0.66 - 0.93)	0.59	0.20 (0.07 - 0.34)	0.41	0.38 (0.25 - 0.52)
L. harak	0.91 (0.85 - 0.96)	0.51	0.40 (0.34 – 0.45)	0.45	0.46 (0.40 – 0.51)

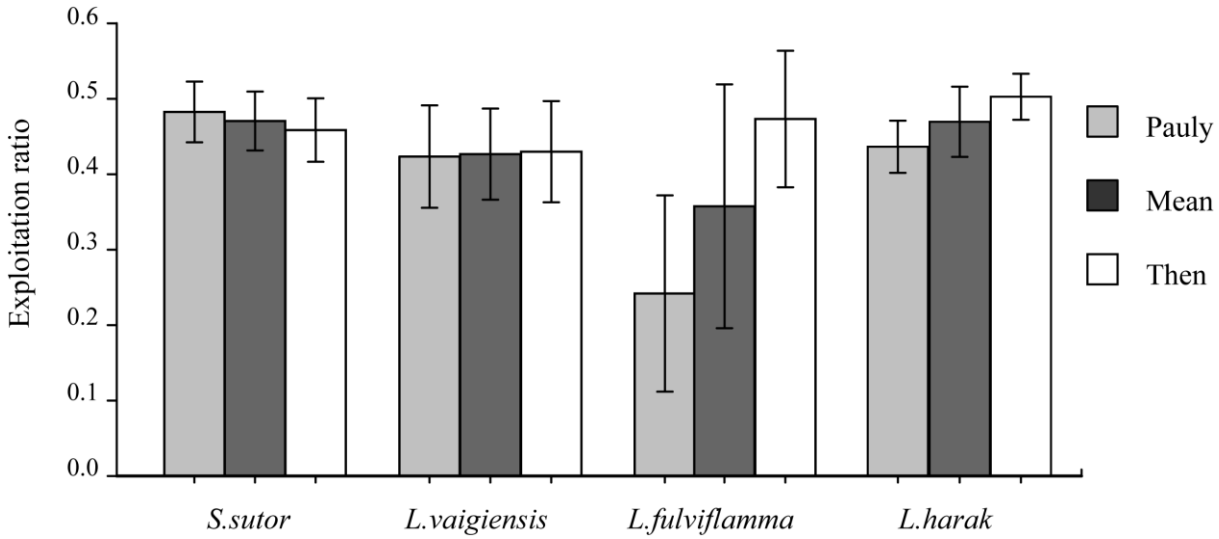

Figure 25. Bar plots showing variance in the estimated exploitation rates for *S. sutor*, *L. vaigiensis*, *L. fulviflamma* and *L. harak* in relation to the method used to calculate natural mortality (right), and the mean value based on the two methods.

The summary of the LB-SPR model for the selected species (Table 6) and Figure 26 reveals that of the four species, three (*L. vaigiensis*, *L. fulviflamma* and *L. harak)* have SPR estimates that are below the target ($SPR_{40\%}$) that constitutes overfishing. In fact, the estimates for *L.fulviflamma* and *L.harak* were even below the critical levels of $SPR_{20\%}$, suggesting that these stocks are also recruitment overfished. According to (Figure 26), the current results based on the SPR, relative yield, and SSB/SSB0 analyses suggest that the current effort (F/M) for each of the species are high and that current fishing mortality should be lowered to levels between 0.5-0.8 (F/M <1) to result in sustainable levels regarding relative yield, SPR and SSB/SSB0. Increasing the

relative effort at levels of F/M > 1 would lead to a steep decrease in SPR, relative yield and the relation SSB/SSB0 for all the species. Therefore, it seems imperative that any size-based fishing regulations must be implemented in relation to fishing effort.

Table 6: The estimated spawning potential ratio (SPR), relative effort F/M and selectivity parameters from the LB-SPR estimation models for the four reef fish species from the Kenyan South Coast.

	Estimated Parameters based on LB-SPR			
Species	SPR (current)	F/M	SL_{50} (cm)	SL_{95} (cm)
S.sutor	0.45 (0.38 – 0.80)	0.4	13.5	16.9
L.vaigiensis	0.39 (0.25 – 0.72)	0.8	12.8	15.0
L.fulviflamma	0.11 (0.05 – 0.20)	1.2	12.5	15.2
L.harak	0.08 (0.05 – 0.11)	2.2	10.0	15.2

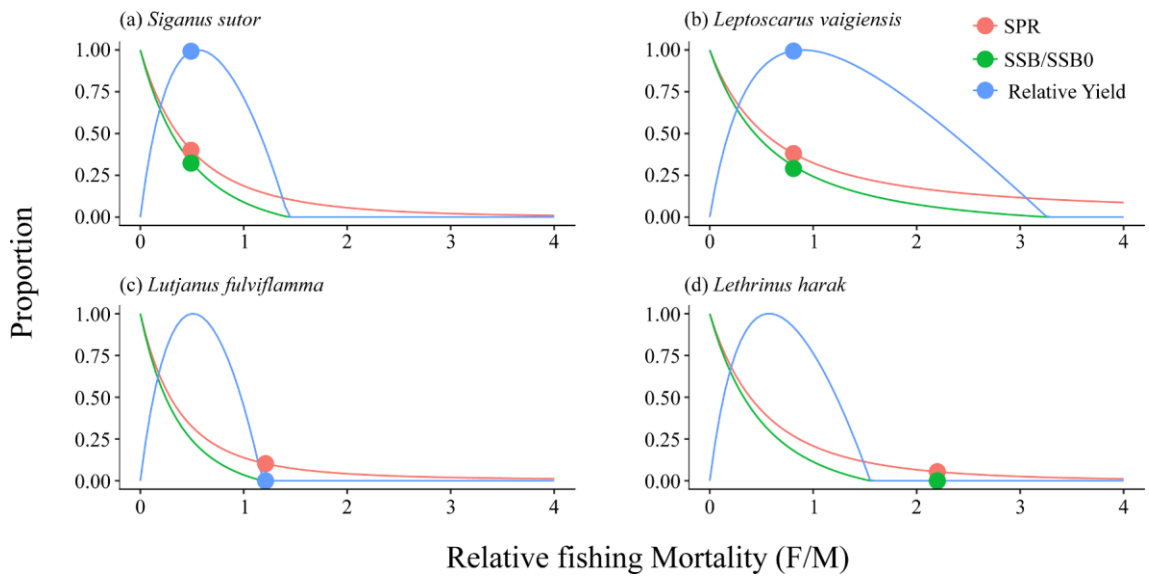

Figure 26. Graphical representation of the spawning-potential ratio (SPR), spawning stock biomass relative to unexploited conditions (SSB/SSB0), and the relative yield in relation to the relative effort for the four selected species as generated by the LB-SPR model.

4.4 DISCUSSION

The present study makes use of two techniques for data-poor fishery systems to understand the status of an artisanal fishery and to provide insights on the species' vulnerability to exploitation. In the first approach, we applied the ELEFAN routine

to estimate the growth parameters K and L_∞ and to generate a range of mortality and exploitation rates. The growth parameters were then contrasted with available growth and mortality parameter values from the literature to generate a range of input parameters used for the length-based spawning potential ratio (LB-SPR) approach.

The length-frequency distribution of the four selected species showed little variation over the sampling period assessed. In all the samples, the size structure of the catch was skewed toward smaller size classes (TL < 20 cm) with a higher proportion of immature individuals (TL (cm) < L_{50} = 60%). This seems to be a clear indication of a likely unsustainable fishery mostly targeting juveniles, which concurs with findings from other studies in the region (Hicks and McClanahan, 2012; Mangi and Roberts, 2006; McClanahan and Mangi, 2004; Silberschneider et al., 2009), and jointly with them underlines the urgency to amend existing fishery regulations and models used for managing these stocks (though this will not solve the persisting challenge of implementing these regulations).

The low volumes of the larger-sized fish in the catch may be evidence of a fishery experiencing juvenescence (Gwinn et al., 2015; Pilling et al., 1999). The general assumption here is that the older and more fecund size classes have increasingly been removed from the system, thus leading to a decrease in the mean size and a truncation of the size structure of the exploited population (Ault et al., 2005; King, 2007; Sparre and Venema, 1998). Therefore, as widely known, a representative sample of the catch, which is primarily compromised of juveniles and with a mean size of catch below the size at maturity, must by definition be subject to an unsustainable level of fishing pressure (Die and Caddy, 1997; Mangi and Roberts, 2006).

However, the exploitation rate estimates derived from the catch curve analysis for our four species does not support a significant growth overfishing. One of the most commonly adopted approaches for inferring optimal exploitation is that of Gulland (1970), which suggests that fisheries are optimally exploited when the rate of fishing

mortality is about equal to the natural mortality (E_{opt} = 0.5). If we apply this approach to our current situation, we would assume that our current estimates of the mean exploitation rates (*S. sutor*, *E*= 0.47 ± 0.04; *L. vaigiensis*, *E*= 0.43 ± 0.06; *L. fulviflamma*= 0.36 ± 0.16 and *L. harak E*= 0.47 ± 0.07) are at or approaching the level of optimum exploitation.

However, one problem with this approach to the question of overfishing is that it assumes constant mortality and probability of capture over all sizes for ages on the descending limb of the catch curve (King, 2007). This assumption appears violated considering the multiplicity of gears used in the fishery, each having a unique selectivity (Tuda et al., 2016). In addition, the sampling across the gears was not uniform, which may have resulted in the under-sampling in the catches. Additionally, using growth parameters which may not apply to a heavily exploited fishery may spread bias in the estimation of natural mortality and thus result in the underestimation of the exploitation rates (Kenchington, 2014; Ramírez et al., 2017). Therefore, our results suggest that determining the status of a fishery, based on E_{opt} = 0.5 as a suitable measure for exploitation, may create dangerous false expectations, and hence cannot be used to establish suitable management recommendations without additional analysis (Die and Caddy, 1997; Prince et al., 2015).

Alternatively, if we compare these findings with the results of the LB-SPR approach, the emerging picture is entirely different: estimates of relative fishing mortality (F/M) are substantially higher, and the spawning potential (SPR) is much lower, which seems to hint at some of the species being overexploited. Discordance in results between the two approaches makes it difficult to come to a definitive conclusion regarding the status of the exploited species and leads to difficulty in formulating fishery management advice (Die and Caddy, 1997). For instance, the estimated F to M ratio for *L. vaigiensis* was comparable between the two methods (F/M = 0.8), suggesting that the current exploitation rate (E = 0.43 ± 0.06) may represent a healthy stock status of the species. The result of SPR (39%) would also seem to confirm a rather uncritical state. Similarly, the results for *S. sutor* (E = 0.47 ±

0.04; SPR = 45%) were consistent and suggested a healthy state of the stock. However, the F to M ratio estimated by the LB-SPR model was 57% lower than that estimated by the catch curve method and 65% lower than the mean value reported for this species in the study area (F/M = 1.15). Given that our estimate for M was within reasonable estimates reported for this species (M = 1.15-1.87), the differences could have been caused by uncertainty in the estimation of fishing mortality.

In contrast to the species *S. sutor* and *L. vaigiensis*, the LB-SPR approach yielded estimates according to which *L. fulviflamma* and *L. harak* are experiencing both growth and recruitment overfishing (SPR< 20%). These results are to be expected given the life-history characteristics of *L. fulviflamma* and *L .harak* (i.e., high longevity, delayed maturity, and large body sizes), which make them susceptible to overfishing (Nadon et al., 2015). However, Kaunda-Arara and Ntiba (2001) assessed the status of *L.fulviflamma* for the Kenyan inshore waters using a length-based approach and found no evidence of overfishing for this species. Their estimated exploitation rate was 14% of the total mortality which suggested that the stock was underexploited. They attributed this to the existence of spatial refugia for the larger sized individuals (mega spawners) (Kaunda-Arara and Ntiba (2001).

Circumstantial evidence exists to support the idea that adult fish of some reef species are spatially separated, with juveniles and adults being more abundant in shallow and deeper waters respectively (Pilling et al., 1999; Williams and Hatcher, 1983). In a way, this pattern would suggest that the larger sized *L. fulviflamma* are further offshore and not fully available to the fishery, and are thus underrepresented in the artisanal fisheries catches. However, two factors influence the adequacy of the results of Kaunda-Arara et al. (2016) for the current issue. First, their study relied only on basket trap samples, a gear type subject to its own sampling bias, which, in turn, might have led to uncertainty in their estimation of growth parameters and resulting exploitation rates (Erisman et al., 2014; Lucena and O'Brien, 2001; Wakeford et al., 2004). Secondly, the results of a recently conducted offshore survey of the fish assemblages in coastal East Africa (Kenya and Tanzania) indicates these species are

not present offshore (Kaunda-Arara et al., 2016). Consequently, the observed lower exploitation rates observed for *L. fulviflamma* could be more due to biased estimates of the growth parameter that may not apply to heavily exploited fisheries, rather than the existence of spatial refugia.

Published work on growth and mortality estimates of species in the families Lethrinidae and Lutjanidae have highlighted the weakness of using length-based approaches (Pilling et al., 1999). It was found that for species with an extended lifespan the length classes of the older fish overlapped, making the modal separation difficult and thus leading to biased estimates (Isaac, 1990; Pilling et al., 1999). Mees and Rousseau (1997), in their study of the deepwater Lutjanid *Pristipomoides filamentosus*, showed that the estimate of Z was positively biased to K and L∞. As a result, the catch curves tended to underestimate the Z when applied to slow growing fish with low mortality rates because of the low contribution of the oldest ages in the catch (Coggins et al., 2013).

A comparison of our most conservative estimate of M for the *L.fulviflamma* (M=0.41) to the age-based estimates by Kamukuru et al. (2005) (M = 0.27) and Grandcourt et al. (2006) (M= 0.29) revealed that our values were possibly overestimated due to the small sample size of the oldest age classes.Therefore, if we used the more conservative natural mortality estimate (M ~0.27) as suggested by Clark (1999), for a data-limited fishery, a different picture would emerge, which would suggest that the *L.fulviflamma* is overexploited (E = 0.66). Thus, the high fishing effort indicated by the LB-SPR (F/M >1.2) would probably be a justified estimate, and likely be the cause of overfishing (Kamukuru et al., 2005). Again, using a more conservative estimate of M for *L.harak* (M= 0.38) as reported by Hilomen (1997), results in a similar conclusion of overexploitation (E = 0.58).

It is interesting to note that for both species, using a more conservative estimate of M would still result in unsustainably low SPR values (SPR< 10%) suggesting, growth and recruitment overfishing related to the selectivity's orientation to catch the juveniles. These results reinforce the argument that species with high longevity,

delayed maturity, and large body sizes have the lowest SPR estimates, making them more susceptible to overfishing (Nadon et al., 2015).

Based on our results, we can deduce that the high fishing effort and progressive capture of immature individuals may have contributed to the growth and recruitment overfishing of our key species. We observed that the dominance of smaller individuals in the catch, the estimated exploitation rates, and the low SPR are indicative of a fishery at or approaching overfishing. Considering the uncertainty surrounding the estimation of the growth parameter, adopting a more precautionary approach to providing stock assessment information would be imperative. According to Patterson (1992), exploitation rates in excess of 0.4 are most likely to lead to stock declines. Thus, if this management strategy would be adopted, the existing exploitation rates as estimated in our study would likely exceed the 40% reference point and would suggest both growth and recruitment overfishing. Therefore, our main recommendation is to adopt a more precautionary approach to maintain exploitation rates at rather conservative levels ($E<0.4$; $F=1/2\ M$), to reduce the risk of overfishing and maintain at least 35% of the spawning stock (Die and Caddy, 1997; Grandcourt et al., 2005; King, 2007; Patterson, 1992).

Reversing this adverse effect of fishing on these four species may require a combination of management tools leading presumably to the reduction of fishing effort. If the aim of fisheries management is to allow the fish to grow to maturity and to spawn at least once before capture (Froese, 2004), then using fishing gears that catch the only largest individuals, or restricting gears that target immature individuals, may mitigate the existing problem (McClanahan and Mangi, 2004). Given the prevalence of immature fish in landed catch and considering that the fishery in this area is dominated by size-selective fishing gears (Tuda et al. 2016), it is plausible to assume that mesh size regulations would be the most feasible and tangible option for managing this fishery (Hicks and McClanahan, 2012; Mangi and Roberts, 2006; Tuda et al., 2016).

However, the challenge with implementing this approach will emerge from the fact that a different selectivity pattern could mean a higher or lower SPR for the same amount of fishing effort for various species. Thus, regulation of SL_{50} achieved through minimum size limits may be ineffective for this multispecies fishery because different mesh/hook sizes may be required for various species. Also, increasing mesh size can theoretically lead to increased yield, but it would require an increase in effort to realize this; the immediate result would be a decrease in catch rates making this measure difficult to enforce (Mees and Rousseau, 1997). However, any increase in fishing effort beyond the current harvest levels (SPR_{40}) would inevitably retard the spawning potential of the stocks and lead to reduced yields.

For a fishery already exhibiting signs of overfishing, any protection aimed at regulating fishing mortality (e.g., temporal closures or size limits) is likely to result in population increases (Gwinn and Allen, 2010), at least temporarily. Related innovative measures such as restricting of fishing locations or setting up locally-managed no-take zones, which is increasingly gaining acceptance among the fishers, might prove to be feasible (McClanahan et al., 2016), potentially as one part of a combination of management tools. If well implemented, such measures have the potential of protecting nursery grounds of the most valuable species, which is key to reducing the capture of immature individuals. Yet in our view, effort reduction would seem to be a more appropriate management option. Currently, there is limited direct control of fishing effort, and the current attempts aimed at regulating the allocation of fishing licenses by number and gear type have failed, given that up to 82% of the fishers are not licensed (GOK, 2013).

To come up with viable ideas on how to do this, effort reallocation may be the next step to lower overall fishing impact but keep the fishers employed. This would require a study on the relative effort caused by the different gears/fleets on the target species since each fishing gear has a different curve of size-specific F. Given the contribution of each component to the total mortality it should be possible to apportion realized SPR among the fisheries allowing each gear to be managed

according to their harvesting strategies (Goodyear, 1993). Thus, management based on SPR could be incorporated into current regulations to test whether management objectives are being met, and to set measures in place for a long-term data collection and more precise assessment approach (Prince et al., 2015; Schmalz et al., 2016).

4.5 CONCLUSION

Given the example of Palau where the application of the SPR approach resulted in the community's acceptance of minimum size limits and change in the state laws (Prince et al., 2015), we believe that the SPR approach can also provide useful information to assess the effectiveness of management actions (Hordyk et al., 2015) for extensive continental coastal waters such as in Kenya. The above results are presented bearing in mind the data limitations of this study (1-year data) for estimating growth parameters of species with potential longevities of >17 years.

Thus, the data may be insufficient in addressing the changes likely to have occurred in response to long-term exploitation and environmental changes, particularly for long-lived species. Also, our estimates are based on data incorporating combined gears and may not be adequate to evaluate the individual effect of each gear and the imposed fishing mortalities on the different species. Nevertheless, given the diversity of life histories that exists in this fishery and the impracticality of performing full size-based or age-based assessments on all stocks, our results highlight the possibility of assessing data-limited fisheries, and the need of exploring different approaches and integrated methodological combinations in fisheries assessments and resulting management conclusions.

4.6 ACKNOWLEDGEMENTS

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CHAPTER 5.

Comparing an ecosystem (trophic modelling) approach with single species stock assessment: the case of Gazi Bay, Kenya

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ABSTRACT

Gazi Bay located in the Kenyan South Coast supports an economically important multispecies and multigear artisanal fishery. However in recent years, local fisheries have experienced a substantial decline due to increased exploitation. Therefore, a trophic mass-balance was built using Ecopath, (1) to characterize Gazi Bay ecosystem structure and functioning, (2) to evaluate the ecological impacts of fishing on the ecosystem and (3) compare the results of the ecosystem assessment to that of previous single-species stock assessments. Using the network analysis, 23 functional groups were aggregated into linear food chain resulting in nine discrete trophic levels *sensu* Lindeman (1942) with a mean transfer efficiency of 12.4%. Results indicate bottom-up control in the ecosystem, with a detritivory/herbivory ratio of 0.72 implying that herbivory dominated energy flow to the higher trophic levels. Measures of ecosystem maturity such as system production to biomass ratio (P_P/B), primary production to respiration ratio (P_P/R) and system biomass to throughput ratio (B/T), indicate that Gazi Bay is an immature, perturbed system likely from the exploitation of the resources through fishing. Artisanal fisheries in Gazi Bay operates at a trophic level of 2.38, similar to that of primary consumers in the system, which likely reflects the shift from fishing large piscivores to lower trophic level species. The results of the multi-species stock assessment presented based on the computed exploitation rates of key resources, though somewhat different with the results of previous evaluations, suggests that this system is heavily exploited (F/Z >0.5). Unlike standard single-species stock assessments, the effects of fishing were examined by quantifying the percentage of primary production required to sustain fisheries and the average trophic level of catch (%PPR–TLc), which allows for among system comparisons. The results indicate that fishing impact in Gazi Bay is comparable to some of the most intensively exploited coastal and coral reef ecosystems of the world. Therefore, relying on single species management approaches such as gear restriction and size limits may not be sufficient for sustaining this multispecies

fishery. Alternative approaches, such as the control and reduction of the fishing effort and the establishment of certain areas closed to some fisheries may be better measures towards ecosystem-based management. This should be done while considering the fishing impacts, the economic and social benefits within the ecosystem context.

Keywords: Ecopath, Gazi Bay, fishery impact, trophic interaction.

5.1 INTRODUCTION

Many fisheries worldwide are showing signs of overfishing, and this has been attributed to the increased fishing pressure (Christensen, 1998; Pauly et al., 1998; Watson and Pauly, 2001; Jackson et al., 2001). Among the most affected and vulnerable fisheries are those in developing countries, which account for over 60% of the global fish catch (FAO 2012). Given these facts, there has been increased attention towards quantifying direct fishing impacts on target and non-target species and the indirect impacts, commonly referred to as ecosystem-based management (Larkin, 1996; Ruckelshaus et al., 2008; Ye et al., 2011).

Despite developing a widespread interest in the past decades, the implementation of the ecosystem-based fisheries management is still underused, and particularly for tropical fisheries, which have traditionally relied on single species approaches (Harvey et al., 2003; Latour et al., 2003; Shin et al., 2010). Nevertheless, the notion is changing, mainly due to the increased application of ecosystem modeling tools, which allow for the integration of ecosystem impacts in the assessment approach. One such tool, which has received widespread application is the Ecopath with Ecosim (EwE)(Christensen and Walters, 2004). The EwE ecosystem modeling approach allows for the evaluation of fishing impacts on the target species and on the ecosystem by providing system feedback about fishery-induced changes in the trophic interactions thereby providing a useful tool for ecosystem-based fisheries management (Coll et al., 2009).

In this study, we develop an ecopath model to investigate the trophic interaction in the Gazi Bay, a tropical coastal Bay located in the Kenyan South Coast and evaluate the fishing impact within the ecosystem context (Coll et al., 2006). Gazi Bay supports diverse fisheries, with artisanal fishers using multiple gears to target the multispecies fishery, which operates between the Bay and the adjacent coral reefs and seagrass beds (Kimani et al., 1996). However, these resources are increasingly threatened by heavy exploitation, and signs of overexploitation have been reported (McClanahan et al., 2008). Previous work on multispecies, multigear fisheries in the study area has focused on single species stock assessment (Hicks and McClanahan, 2012; Kaunda-Arara et al., 2003), gear selectivity, and the monitoring of catch per unit of effort as a basis for defining fishery impacts (McClanahan et al., 2008). In this study, we employ an ecosystem approach to understand the trophic interactions of the Gazi Bay, and to evaluate the impacts of current fishing activities on the ecosystem. This work is meant to complement stock assessments that have been done for some of the target species of the area and is expected to provide guidance towards developing an ecosystem-based management strategy for Gazi Bay.

5.2 STUDY SITE

Gazi Bay is a semi-enclosed and shallow tropical coastal system located approximately 60 km south of Mombasa (4º25´S and 39º50´E) (Figure 27) with a surface area of approximately 10km^2 excluding the mangrove swamp covering about 6·61 km^2 (Ohowa et al., 1997). It is interlinked with a fringing reef (on the seaward side), an extensive mangrove forest on the landward side and seagrass beds interspersed with macroalgae mats visible during low tide. It is inundated twice a day with a maximum tidal height of 4 m (Coppejans and Gallin, 1989). The Bay is relatively shallow with a mean depth of about 5 m and with a wide opening (3,500 m) towards the Indian Ocean in addition to the two small seasonal rivers, the Kidogoweni in the North and the Mkurumu in the South-West.

Extensive research has been conducted in the Gazi Bay (1985-1996) to provide an adequate and comprehensive understanding of the interlinkages between the coral reef, seagrass, and mangrove ecosystems. These studies have highlighted the importance of these ecosystems in enhancing the primary and secondary productivity (Osore et al., 1997), transfer of nutrients (Ohowa et al., 1997) and their role in supporting the growth and survival of the ichthyofaunal community during different life stages (De Troch et al., 1995; De Troch et al., 1998; Wakwabi, 1999).

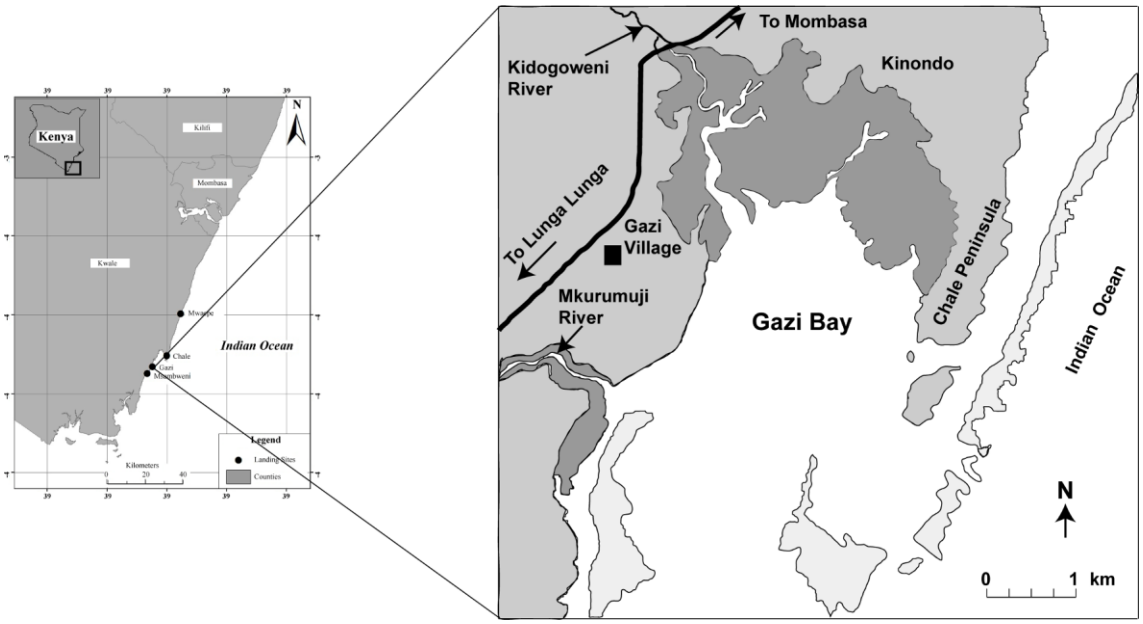

Figure 27. Map of the Study Area showing the two rivers draining in the Bay and the location of the village.

5.2.1 Model construction and data input

A mass-balance trophic model of the Gazi Bay was constructed using the Ecopath with Ecosim software (EwE) (Christensen and Walters, 2004), to represent and examine the trophic flows within the Bay and the adjacent ecosystems. The Ecopath model assumes a steady state over a given period (1 year) and is based on two master equations describing the production and energy balance for each functional group (Christensen and Walters, 2004). The first equation expresses the production term for each trophic group as a function of the catches, predation mortality, biomass accumulation, net migration and other mortality and can be expressed empirically as;

$$\mathbf{B}_i\left(\frac{\mathbf{P}}{\mathbf{B}_i}\right) = \sum_j \mathbf{B}_j\left(\frac{\mathbf{Q}}{\mathbf{B}_j}\right)\mathbf{DC}_{ji} + \left(\frac{\mathbf{P}}{\mathbf{B}_i}\right)\mathbf{B}_i(\mathbf{1}-\mathbf{EE}_i) + \mathbf{EX}_i \qquad (1)$$

where B_i is the biomass of group *i*; P/B_i is the production/biomass ratio of *i*; Q/B_i is the consumption/biomass ratio of predator j; DC_{ji} is the fraction of the prey *i* in the average diet of the predator *j*; EE_i is the ecotrophic efficiency, which expresses the total production that is directly consumed by predators or exported out of the system and EX_i is the export of group *i*. The second equation states that the consumption within a group equals the sum of its production, respiration and unassimilated food;

$$\mathbf{B}\cdot\left(\frac{\mathbf{Q}}{\mathbf{B}}\right) = \mathbf{B}\cdot\left(\frac{\mathbf{P}}{\mathbf{B}}\right) + (\mathbf{1}-\mathbf{GS})\cdot\mathbf{Q} - (\mathbf{1}-\mathbf{TM})\cdot\mathbf{P} + \mathbf{B}\left(\frac{\mathbf{Q}}{\mathbf{B}}\right)\cdot\mathbf{GS} \qquad (2)$$

where GS is the fraction of the food that is not assimilated, TM is the trophic mode expressing the degree of heterotrophy (0 and 1 represent autotrophs and heterotrophs respectively).

5.2.2 Model parameters and functional groups

The Gazi Bay ecosystem was partitioned into 23 functional groups based on the diet and feeding preferences of the ichthyofauna and organisms within the system based

on previous diet studies by De Troch et al. (1998) and Wakwabi (1999) (Table 7). The functional groups represent 10 fish groups and 12 non-fish groups (including detritus) representing either individual species or aggregated species based on feeding strategy.

5.2.2.1. Fish groups

A total of 346 fish species in 72 families has been documented for Gazi Bay based on previous independent surveys (De Troch et al., 1995; Kimani et al., 1996; Wakwabi, 1999). Due to the complexity of representing all the fish groups in the model, the species were reduced to 5 guilds. The aggregation of species into feeding guilds was guided by the food types consumed based on stomach content and carbon isotope analysis of the prey items for the study area (De Troch et al., 1998; Nyunja et al., 2009). The commercially important species were considered separately in the model to allow for further detailed analysis (Tsehaye and Nagelkerke, 2008).

Biomass estimates were obtained from trawl and beach seine surveys of the Gazi Bay, which have been conducted in both mangrove and non-mangrove areas and the surrounding biotopes and were complemented with virtual population analysis (De Troch et al., 1995; Van Der Velde, 1995; Wakwabi, 1999). The estimated P/B ratio of the commercially important species was taken as equivalent to the instantaneous rate of total mortality (Kaunda-Arara and Ntiba, 2001; Kaunda-Arara et al., 2003; Nyang'wara, 2002).

For other species, the P/B values were estimated through Pauly (1980) empirical formula and the Q/B ratios were estimated for each fish species using the empirical formula given by Palomares and Pauly (1989). A diet matrix was developed based on the species stomach content and isotope analysis from the study area for most of the commercially important species and was complemented with literature estimates (De Troch et al., 1998; Nyunja et al., 2009; Wakwabi, 1999).

5.2.2.2. Non-fish groups

The non-fish groups consisted of seabirds, corals, benthic invertebrates, zooplankton, primary producers and the detritus pool. The biomass estimates for the non-fish groups was estimated from previous studies conducted in the study area and the P/B and Q/B ratios were sourced from literature values and previous models of similar characteristics. For a detailed list and compilation of the data sources and estimates see table 8.

5.2.2.3. Fishery

The fishery data for the model were taken from the artisanal catch landings for the study area as reported by Maina et al. (2008) and complemented with recent data (Tuda et al., 2016). The fishery is multi-species and species are targeted by multiple gears, but the major fishing gears included in the model include basket traps, gill nets, speargun, hook and line beach seine and the ring net. The artisanal fisheries of the Kenyan south coast have been described and studied over the years (Mangi and Roberts, 2006; McClanahan and Mangi, 2004). To understand the role of fisheries in the ecosystem, the major fishing grounds in Gazi - estimated to cover ~7 km^2 (Munywoki et al., 2008) and the surrounding fishing sites ~4 km^2 -were included in the modeled area, and the biomass of the input parameters was normalized to a unit surface area using wet weight for biomass expressed in t km^{-2} yr^{-1}.

5.2.3 Balancing the model

To fulfil the condition of a steady state, the model was balanced by ensuring that the values of EE_i and GE_i are within acceptable limits, i.e., $EE_i<1$ and values of (P/Q) GE_i are within the range of 0.1-0.35 of fish species. Input parameters that did not satisfy the mass balance constraints were adjusted following the method described by Christensen et al. (2000), where the least certain parameters were first adjusted and the model rerun until acceptable runs were achieved.

To address the effects of uncertainty in the input parameters, we used the Ecoranger routine in ecopath (Christensen and Walters, 2004). Using a Monte Carlo approach, the 'Ecoranger' routine draws input parameters for each functional group based on a user-defined confidence interval specified in the Pedigree routine of Ecopath. Further, a simple sensitivity analysis routine in Ecopath was applied to evaluate the robustness of the results based on the uncertainty surrounding the input parameters. This was achieved by systematically varying the input parameters in steps of 10 within the range of -50 to + 50% and examining the effects on the estimated parameters.

5.2.4 Network analysis

After achieving mass balance, the network analysis routine in Ecopath (Ulanowicz, 1986; Ulanowicz and Kay, 1991), was used to characterize the system in terms of flows and the food webs were later aggregated into discrete trophic levels to assess flows and distribution among trophic levels (Lindeman, 1942). Other system properties considered in the analysis are those related to ecosystem development and maturity (Christensen, 1995; Odum, 1969), which allows the system to be compared to other marine ecosystems.

To assess the impacts of fishing, the mixed trophic impact analysis (MTI) developed by Ulanowicz and Puccia (1990), implemented in the Ecopath model was used to quantify the direct and indirect impact of each gear on the functional groups. The routine indicates the reciprocal effects of small increases in biomass of one group over the functional groups. Supplementing the above analysis, a relationship between Primary Production Required to sustain the fisheries PPR, (expressed as a unit of catch relative to primary production and detritus of the ecosystem) (PPR%) and the mean Trophic Level of the catch (TL_c) (Pauly and Christensen, 1995; Pauly et al., 1998), were used as indicators of fishing impact as proposed by Tudela (2003) and (Tudela et al., 2005). Further, the estimated exploitation rate (F/Z), was also

calculated for the key commercial species and the results compared with results of the single species stock assessment where F/Z =0.5 was set as the threshold for sustainable fishing (Gulland 1970).

5.2.5 Keystone species

Keystone species are species with a relatively low biomass but with a high and wide impact on the food web (Libralato et al., 2006; Piraino et al., 2002). The Keystone index, which is an output of the network analysis is thus high when the functional group has low biomass but a high overall effect on the ecosystem if its biomass is altered (Christensen et al., 2005).

Table 7: Functional groups and species selected for the Gazi Bay ecosystem

No.	Functional Group	Species/Groups
1.	Detritus	Mangrove litter
2.	Phytoplankton	
3.	Macroalgae	*Gracilaria salicornia, Sargassum spec, Caulerpa racemose, Halimeda opuntia,*
4.	Seagrass	*Thalassodendron cilatum (75%), Cymodocea rotundata, Thalassia hemprichii, Enhalus acoroides, Halodule uninervis*
5.	Zooplankton	Copepoda (~80%): *Acrocalanus spp, Pseudodiaptomus spp., Oithona spp.* Decapoda (10%): Brachyuran zoea, Brachyuran megalopa, fish larvae (3%), Others (7%)
6.	Sea urchin	*Echinometra mathaei, Diadema savignyi, Diadema setosum, Tripneustes gratilla*
7.	Benthic Invertebrates	Polychates,
8.	Sea cucumber	*Holoturia scabra, Stichopus hermani, Holoturia nobilis*
9.	Crabs /Lobster	*Uca lactea annulipes, Uca inversa, Uca chlorophthalmus, Perisesarma guttatum, Panulirus ornatus, Panulirus ongipes*
10.	Corals	*Acropora sp, Porites sp.*
11.	Cephalopods	*Octopus vulgaris*,
12.	**Fish Species**	*Lethrinus harak, Lutjanus fulviflamma, Leptoscarus vaigiensis, Siganus sutor*
	Other Herbivores	*Scarus ghobban, Arothron immaculatus, Acanthurus blochii, Acanthurus leucosternon, Calotomus carolinus*

No.	Functional Group	Species/Groups
	Zooplanktivores	*Caesio xanthonota, Paraplotosus albilabrus, Plotosus lineatus, Pterocaesio tile*
	Benthivores	*Fowleria aurita, Apogon thermalis, Lutjanus argentimaculatus, Leiognathus fasciatus, Paramonacanthus barnardi, Mulloides flavolineatus, Diagramma pictum,*
	Omnivores	*Amblygobius phalaena, Hemiramphus far, Sardinella melanura, Monodactylus argenteus*
	Other Piscivores	*Fistularia commersonii, Sphyraena barracuda, Cheilio inermis, Bothus myriaster, Tylosurus crocodilus crocodilus,*
13.	Turtles	*Chelonia mydas, Eretmochelys imbricate, Caretta caretta, Lepidochelys olivacea* and *Dermochelys coriacea*
14.	Sharks, Skates and Rays	*Raja miraletus, Taeniura lymna, Myliobatis aquila, Dasyatis thetidis, Dasyatis uarnak* and *Dasyatis sephen.*
15.	Seabirds	*Phalacrocorax africanus, Gypohierax angolensis, Ciconia episcopus, Threskiornis aethiopica, Numenius phaeopus, Mycteria ibis, Charadrius mongolus, Ardea alba, Pluvialis squatarola Phalacrocorax africanu.*

5.3 RESULTS

5.3.1 Model output

The trophic flows and estimates from the balanced trophic model of Gazi Bay are depicted in Figure 28 and summarized in Tables 9. The estimated ecotrophic efficiency (EE_i) for all the consumer groups were less than 1.0 and the P/Q and P/R values were within the recommended range (Christensen et al., 2004). Among the primary production groups, phytoplankton exhibited a high EE_i value of 0.83, indicating that over 80% of this resource is consumed compared to only ~11% of the seagrass.

The EE_i of detritus, which is an indication of what flows out and into the detritus was intermediate, implying that only about 42% is used in the system with the difference most probably buried or exported to adjacent systems. Overall, the fish groups exhibited higher EE_i values, which was also reflected by the high exploitation ratio (F/Z = E >0.5). On the contrary, lower EE_i values were estimated for the seabirds (EE_i = 0.225), suggesting low predation on this group (Table 9).

5.3.2 Trophic structure

In the resulting model, the aggregated trophic levels (TL) for the respective functional groups ranged between 1.0 for the primary producers and detritus with the highest TL corresponding to the Piscivores (TL = 3.38). For most of the fish groups trophic levels ranged between 2.77 and 3.38, except for the *Siganus sutor* and the *Leptoscarus vaigiensis*, which had trophic levels below 2.5 due to their herbivory diet.

The bulk of biomass was concentrated in the first two trophic levels (I and II) with 496.7 t km^{-2} at the trophic level I and 115.9 t km^{-2} at trophic level II (Table 11). Seagrass was dominated by the species *Thalassodendron cilatum,* and contributed 51% of the living system biomass (excluding detritus), while the benthic invertebrates and the fish groups only contributed 3.8% and 1.3 % respectively. The fish compartment was dominated by benthivores and herbivores, representing 34% and 19% of the fish biomass respectively.

Of the total net primary production (2897.06 t km^{-2}), about 47 % is grazed upon directly, while the remaining flows to the detritus pool. About 44% of the total system consumption is attributed to the zooplankton, whereas the sea urchin and the fish groups contributed 18% and 5% respectively. Overall, out of the 19 consumer groups, ten were found below trophic levels <2.5.

Table 8: Description of the input data for the Gazi Bay ecosystem model

Functional group	Biomass (tons Km^{-2}	P/B ($year^{-1}$)	Q/B ($year^{-1}$)	Source
1. Seabirds	0.005	0.38	76.5	(Seys et al., 1995; Villanueva and Moreau, 2007) (del Hoyo et al., 1996; Seys et al., 1995)
2. Sharks, Skates and Rays	0.05	0.56	3.5	(Ochumba, 1988; Tesfamichael, 2016)
3. Sea Turtles	0.019	0.1	8.5	(Frazier, 1980; Opitz, 1996)
4. Oth.Piscivores	0.35	0.63	8.45	(De Troch et al., 1998; Locham et al., 2015; Nyunja et al., 2009; Wakwabi, 1999).
5. Oth.Omnivores	1.0	1.06	18.6	(Hicks and McClanahan, 2012; Kaunda-Arara and Ntiba, 2001; Maina et al., 2008; Tuda et al., 2016)
6. Oth.Benthivores	1.90	0.96	12.03	
7. Oth.Zooplanktivores	0.13	1.88	15.17	
8. Oth.Herbivores	1.10	1.07	29.45	
9. *Leptoscarus vaigiensis*	0.25	3.24	23.96	
10. *Lutjanus fulviflamma*	0.175	1.97	10.52	
11. *Lethrinus harak*	0.135	1.83	8.88	
12. *Siganus sutor*	0.51	3.15	21.75	
13. Cephalopods	0.47	3.3	16	(Aliño et al., 1993)
14. Corals	28.41	2.8	12	(Tesfamichael, 2016)
15. Crabs and Lobster	3.73	4.3	20	(Cannicci et al., 2009)
16. Sea Cucumber	0.35	4.4	22.2	(Aliño et al., 1993; Muthiga et al., 2007)
17. Benthic invertebrates	17.45	3.0	15.2	Vanhove et al. (1992); (Schrijvers et al., 1997);Vranken and Heip (1986) (Schrijvers et al., 1997); (Wieser, 1960)
18. Sea urchin	65	0.484	7.86	(McClanahan, 1988) Brey (2001)
19. Zooplankton	7.6	61	165	Mwaluma (1997); (Kitheka et al., 1996) Fetahi et al. (2011); (Osore et al., 1997)
20. Seagrass	230.75	3.5		van Avesaath et al. (1990) Opitz (1996) Ochieng (1995)
21. Macroalgae	76.58	14		van Avesaath et al. (1990) Opitz (1996)
22. Phytoplankton	14.1	70.85		Brown et al. (1991); Brush et al. (2002); Duineveld et al. (1997); Veldhuis et al. (1997)
23. Detritus	160	0.38		Slim et al. (1996)

5.3.3 Trophic flows

Based on the trophic analysis of the ecosystem, the 23 functional groups were aggregated into a simple Lindeman food chain with nine discrete trophic levels (TL). Flows of most of the compartments occurred in the first four levels (Figure 29), with energy transferred to the higher trophic levels mostly derived from primary production (1352 t km^{-2} $year^{-1}$) compared to that coming from the detritus (984 t km^{-2} $year^{-1}$). The greatest flow back to detritus was observed from the primary producers (TL1) 1545 t km^{-2} $year^{-1}$ and from the primary consumers (TL2), 698.1 t km^{-2} $year^{-1}$ (Figure 29). Overall, the trophic efficiencies declined from 12.4% for TL II to 5.0% for TL V, with a mean transfer efficiency for the aggregated food chain (TL I-IX) estimated at 12.6% (Table 11). The low detritivory to herbivory ratio (D:H) ratio of 0.72 implies herbivory dominated energy flow to the higher trophic levels.

Table 9: The compartment and input and resulting parameters biomass (ton km^{-2}), P/B (year−1), Q/B (year^{-1}), EE, GE, P/R and catch (t km^{-2}) for the 23 functional groups of the Ecopath model of the Gazi Bay.

	Functional group	TL	B (t/km²)	P/B (yr^{-1})	Q/B (yr^{-1})	EE$_i$	GE$_i$	P/R	Catch (ton/Km^2)	F$_i$	E$_i$
1	Oth.Piscivores	3.38	0.400	0.630	8.450	**0.900**	**0.075**	**0.103**			
2	Sharks, Skates and Rays	3.26	0.065	0.560	3.500	**0.886**	**0.160**	**0.250**	-	-	-
3	Lethrinus harak	3.23	0.160	1.830	8.880	**0.947**	**0.206**	**0.347**	0.24	1.500	0.82
4	Lutjanus fulviflamma	3.17	0.210	1.970	10.520	**0.994**	**0.187**	**0.306**	0.35	1.648	0.84
5	Oth.Zooplanktivores	3.11	0.150	1.880	15.170	**0.891**	**0.124**	**0.183**	-	-	-
6	Sea Birds	3.09	0.005	0.380	76.500	**0.225**	**0.005**	**0.006**	-	-	-
7	Oth.Omnivores	2.79	**2.158**	1.060	18.600	0.950	**0.057**	**0.077**	-	-	-
8	Oth.Benthivores	2.77	2.137	0.960	12.030	**0.955**	**0.080**	**0.111**	-	-	-
9	Corals	2.59	28.584	2.800	12.000	**0.750**	**0.233**	**0.412**	-	-	-
10	Benthic invert.	2.42	17.450	3.000	15.200	**0.900**	**0.197**	**0.328**	-	-	-
11	Turtles	2.34	**0.026**	0.100	8.500	0.850	**0.012**	**0.015**	-	-	-
12	Oth.Herbivores	2.12	**1.350**	1.070	29.450	0.966	**0.036**	**0.048**	-		
13	Cephalopods	2.10	**1.024**	3.300	16.000	0.900	**0.206**	**0.347**	-	-	-
14	Crabs/lobsters	2.10	14.240	4.300	20.000	**0.204**	**0.215**	**0.368**	-	-	-
15	Siganus sutor	2.07	0.700	3.150	21.750	**0.949**	**0.145**	**0.221**	1.95	2.789	0.89
16	Zooplankton	2.06	7.600	61.000	165.000	**0.727**	**0.370**	**0.859**	-	-	-
17	Leptoscarus vaigiensis	2.01	0.250	3.240	23.960	**0.676**	**0.135**	**0.203**	0.54	2.168	0.67
18	Sea urchins	2.01	65.000	0.484	7.860	**0.447**	**0.062**	**0.083**			
19	Sea cucumber	2.00	0.350	4.400	22.200	**0.429**	**0.198**	**0.329**			
20	Seagrass	1.00	**235.987**	3.500	-	0.105	-	-			
21	Macroalgae	1.00	76.580	14.000	-	**0.408**	-	-			
22	Phytoplankton	1.00	**14.100**	70.850	-	0.829	-	-			
23	Detritus	1.00	160.000	-	-	**0.423**	-	-			

Table 10: Diet composition matrix for the functional groups considered in the model.

Prey	Functional Group	Predator																		
		1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19
1	Oth.Piscivores	0.001	0.188																	
2	Sharks, Skates and Rays	0.003																		
3	Lethrinus harak	0.010	0.009																	
4	Lutjanus fulviflamma	0.001	0.066					0.001												
5	Oth.Zooplanktivores	0.010	0.009											0.070						
6	Sea Birds		0.002																	
7	Oth.Omnivores	0.200	0.010					0.001						0.010	0.004					
8	Oth.Benthivores	0.140	0.019				0.010		0.050					0.020						
9	Corals										0.200		0.047				0.010			
10	Benthic invert.	0.150	0.141	0.850	0.250	0.150	0.230	0.400	0.200		0.050	0.200	0.050	0.400	0.025					
11	Turtles		0.009																	
12	Oth.Herbivores	0.200	0.019	0.100				0.001	0.005											
13	Cephalopods	0.001	0.019					0.060						0.100						
14	Crabs/lobsters	0.100		0.050	0.740	0.150	0.750		0.350			0.010		0.150	0.001					
15	Zooplankton	0.033	0.009		0.010	0.700	0.010	0.080	0.100	0.550	0.001	0.050	0.001	0.200	0.050	0.020				
16	Sea urchins	0.010							0.040		0.049	0.001								
17	Siganus sutor	0.040	0.019																	
18	Leptoscarus vaigiensis	0.001	0.009																	
19	Sea cucumber	0.100	0.094					0.007												
20	Seagrass		0.188					0.010				0.239	0.150				0.150	0.200	0.800	
21	Macroalgae							0.260			0.100	0.500	0.500		0.200		0.600	0.800	0.200	
22	Phytoplankton							0.170	0.010	0.110	0.100		0.250		0.150	0.800				0.200
23	Detritus							0.010	0.245	0.330	0.500		0.002	0.050	0.570	0.180	0.240			0.800
24	Import		0.188							0.010										
25	Sum	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1

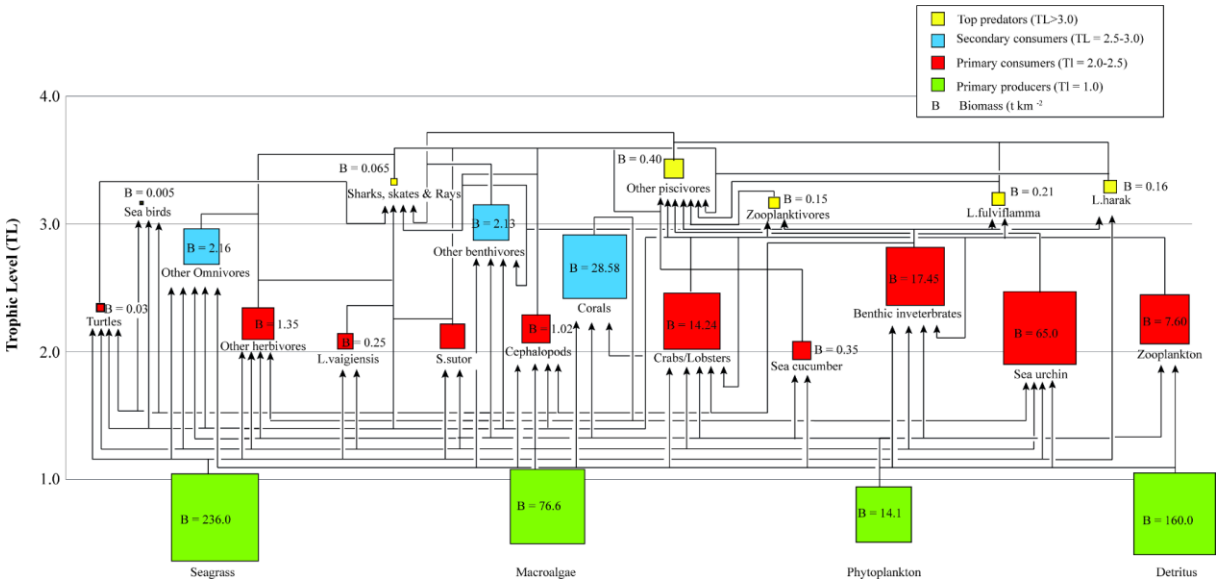

Figure 28. Trophic model of the Gazi Bay ecosystem.

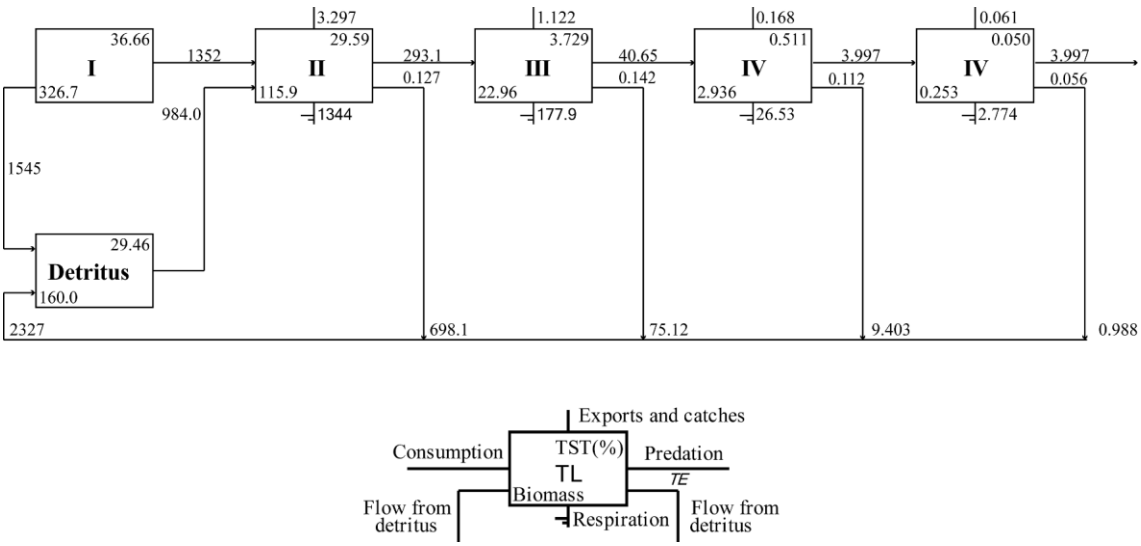

Figure 29. Trophic representation (Lindeman Spine) of the Gazi Bay, showing functional groups aggregated into a single linear food chain and trophic efficiencies.

5.3.4 System properties

The summary statistics of the ECOPATH model are given in Table 11. The total system throughput (TST) was estimated at 8047.953 t $km^{-2}yr^{-1}$, of which about 29% goes into detritus and 19% to respiratory flows. Considering attributes relating to ecosystem maturity based on the work by Odum (1969) and Christensen (1995), the results suggest that Gazi Bay is in a development stage towards maturity as reflected by the high system production to biomass ratio (P_P/B = 6.8), relatively high primary

production to respiration ratio (P_P/R = 1.87) and a low system biomass to system throughput ratio (B/T = 0.056) (Christensen, 1995). The relative low system maturity is also confirmed by the relatively low value of Finn's cycling index (FCI= 7.3%), which is an indication of the fraction of total system throughput that is recycled, and the connectance index (CI = 0.189) and the system omnivory index (SOI = 0.25).

Table 11: Summary system statistics for the Gazi Bay ecosystem model.

Parameter	Units	Gazi Bay (this study
Sum of all consumption	t km^{-2} yr^{-1}	2819.033
Sum of all exports	t km^{-2} yr^{-1}	1348.987
Sum of all respiratory flows	t km^{-2} yr^{-1}	1551.551
Sum of all flows into detritus	t km^{-2} yr^{-1}	2328.382
Total system throughput (TST)	t km^{-2} yr^{-1}	8047.953
Sum of all production	t km^{-2} yr^{-1}	3600.736
Total catch	t km^{-2} yr^{-1}	4.653
Mean trophic level of the catch		2.38
Gross efficiency (catch/net p.p.)		0.002
Calculated total net primary production	t km^{-2} yr^{-1}	2897.060
Total primary production/total respiration (P_p/R)		1.867
Net system production	t km^{-2} yr^{-1}	1345.509
Total primary production/total biomass (P_p/B)		6.183
Total biomass/total throughput (B/T)		0.058
Total biomass (excluding detritus)	t km^{-2} yr^{-1}	468.526
System transfer efficiency (TE, overall)	%	
Finn's Cycling Index (FCI)	% TST	7.3
Finn's mean path length		2.766
Connectance Index (CI)		0.253
System Omnivory Index (SOI)		0.189
Ascendency	%	27
Overhead	%	73
Development Capacity (C)	flowbits	

Table 12: Transfer efficiency (%) and biomass expressed in t km−2 yr-1 of each trophic category in the model.

Source	Trophic category								
	I	II	III	IV	V	VI	VII	VIII	IX
Producer		12.2	14.9	11.5	5.6	5.4	6.1		
Detritus		13.3	13.3	10.7	5.6	5.4	6.3		
All flows		12.7	14.2	11.2	5.6	5.4	6.2	7.4	10.2
Biomass	496.7	115.9	23.0	2.9	0.3	0.01	0.001		

5.3.5 Mixed trophic impacts

The mixed trophic impact (MTI) analysis (Figures 30 and 31) shows the effects of an increase in the biomass of one functional group on the other groups. Based on the trophic relationship among the functional groups, detritus and phytoplankton have a positive impact on most of the higher trophic level groups. For instance, an increase in phytoplankton biomass has a positive influence on the zooplankton and the zooplanktivores. Likewise, an increase in the biomass of macroalgae resulted in a positive impact in the biomass of sea urchin and the sea turtles, which directly graze on this resource.

5.3.6 Fishing impacts

Fishery yields from Gazi Bay were estimated at 4.6 t Km^{-2} $year^{-1}$ with a mean trophic level of the catch of 2.38, slightly lower than the trophic level of benthic invertebrates (TL = 2.42). The gross efficiency of the catch (catch/net p.p) for this system was estimated at 0.002, ten times higher than the weighted global average of 0.0002 (Christensen et al., 2004). *Siganus sutor* and *Leptoscarus vaigiensis* accounted for more

than 50% of the total yield and represented the most important components of the artisanal fishery (Table 9).

The estimated mortality rates from single-species stock assessments and the estimates from the ecopath model are shown and compared in Figure 32. Unlike single-species stock assessment, Ecopath breaks down the natural mortality (M) into predation mortality (M_1) and other mortalities (M_0). A comparative analysis of fishing and predatory impacts revealed that the fishing mortality (F_i) was by far the leading cause of total mortality (Z).

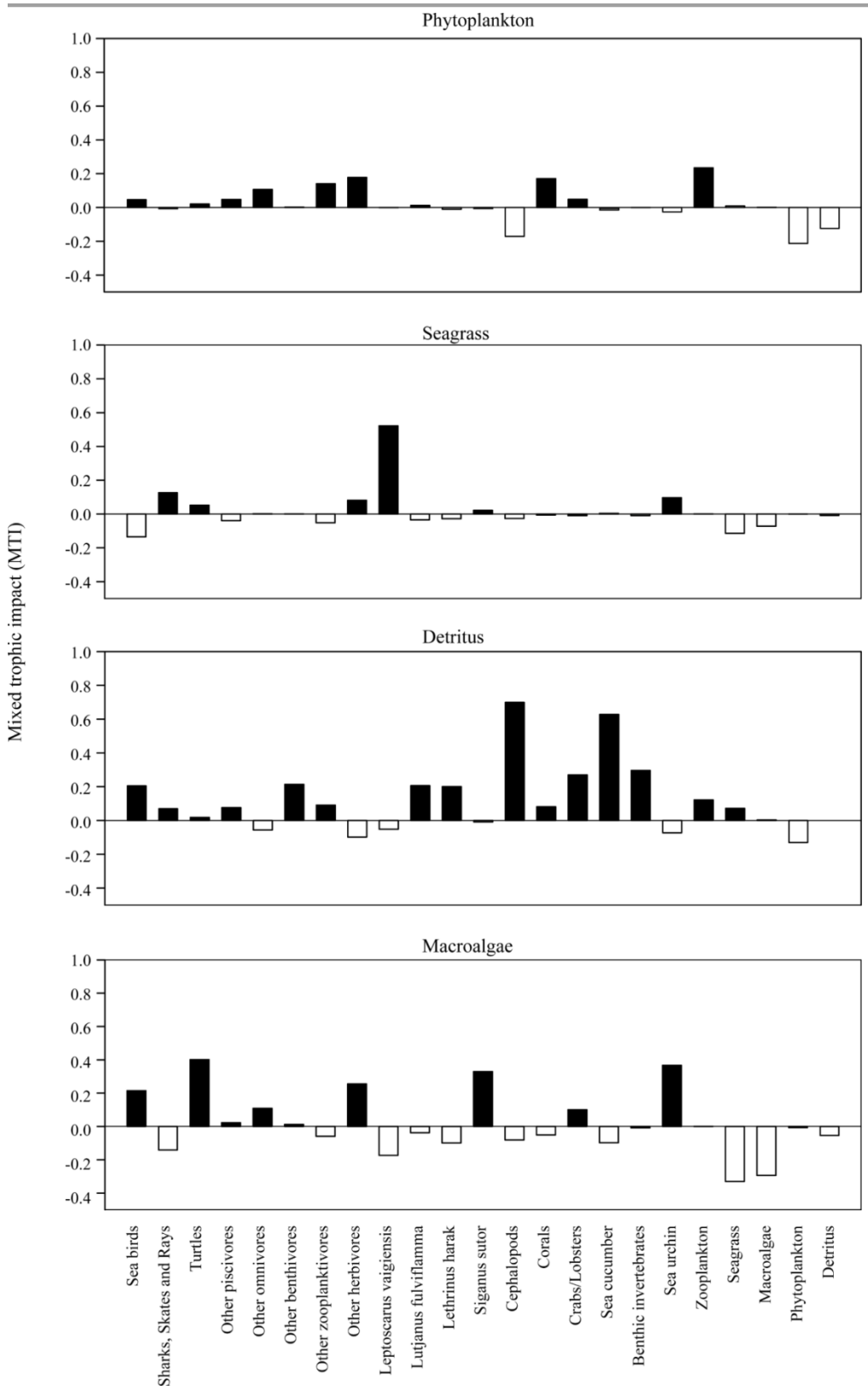

Figure 30. Results of the mixed trophic impacts trophic routine with the bars quantifying the direct and indirect trophic impacts that the primary producers have on other functional groups listed at the bottom. Darker shaded bar represents a positive impact while lighter shaded bars depict a negative impact.

Based on a comparison of the natural mortality estimates, the *M* values estimated by Ecopath were comparatively lower for the *Siganus sutor*, *Lutjanus fulviflamma* and the *Lethrinus harak* but slightly higher for the *Leptoscarus vaigiensis*. Nevertheless, the estimates for the fishing mortality provided by Ecopath were for all species greater than those from the single stock assessment. Overall, all the four key species exhibited signs of overfishing, with the exploitation ratios exceeding the threshold ($E = F/Z > 0.5$).

As expected and shown by the MTI analysis (Figure 31), fishing impact on the target species varies between gears. The hook and line fishery strongly impacts the piscivores, thumbprint emperor *Lethrinus harak* and the dory snapper *Lutjanus fulviflamma* all with a TL > 3, while the speargun has the highest negative impact on the marbled parrotfish *Leptoscarus vaigiensis*, zooplanktivores and other herbivore fish. By contrast, the basket trap greatly impacts the shoe maker spinefoot rabbitfish *Siganus sutor* and to a lesser extent *Leptoscarus vaigiensis*. Overall, the ring net has the least effect on the fish groups in terms of biomass. The gears impacted negatively on each other, which could be explained by the fact that all the gears operate within the same area and as such there is spatial overlap and completion between gears for similar target species.

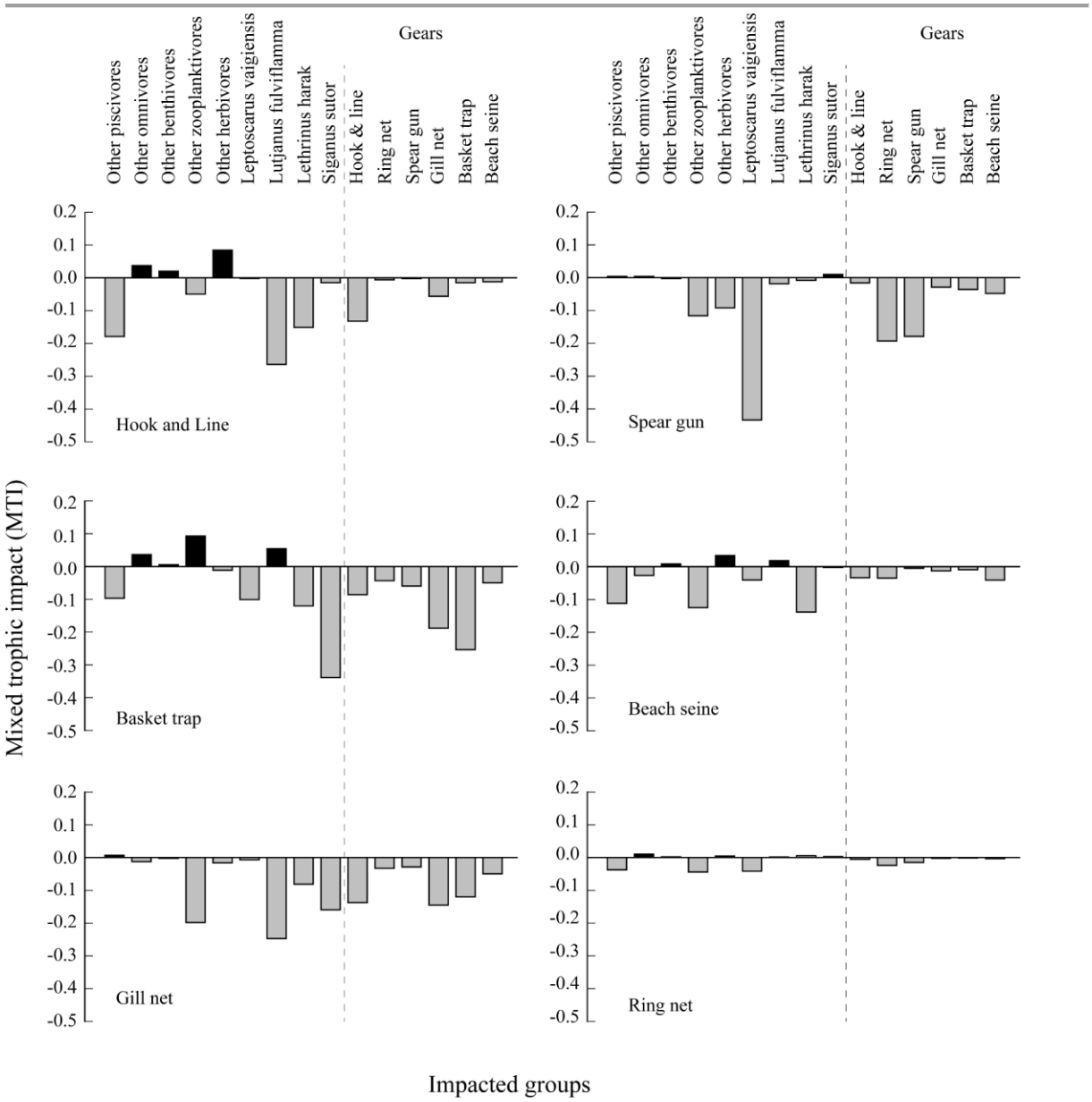

Figure 31. Results of the mixed trophic impacts trophic routine with the bars quantifying the direct and indirect impacts that an increase in fishing effort of the various gears will have on the main commercial species listed at the top.

5.3.7 Keystone species

The estimated keystoneness of the various functional groups in the model is shown in Figure 33. The keystoness analysis showed that the functional group sharks, rays and skates had the highest impact on the food web despite their low biomass (B = 0.065 t km^{-2}). The functional group sharks, rays and skates had the highest keystoneness index (KSi = 0.147) and relative total impact (1.0) and therefore qualified as keystone species (Libralato et al., 2006).

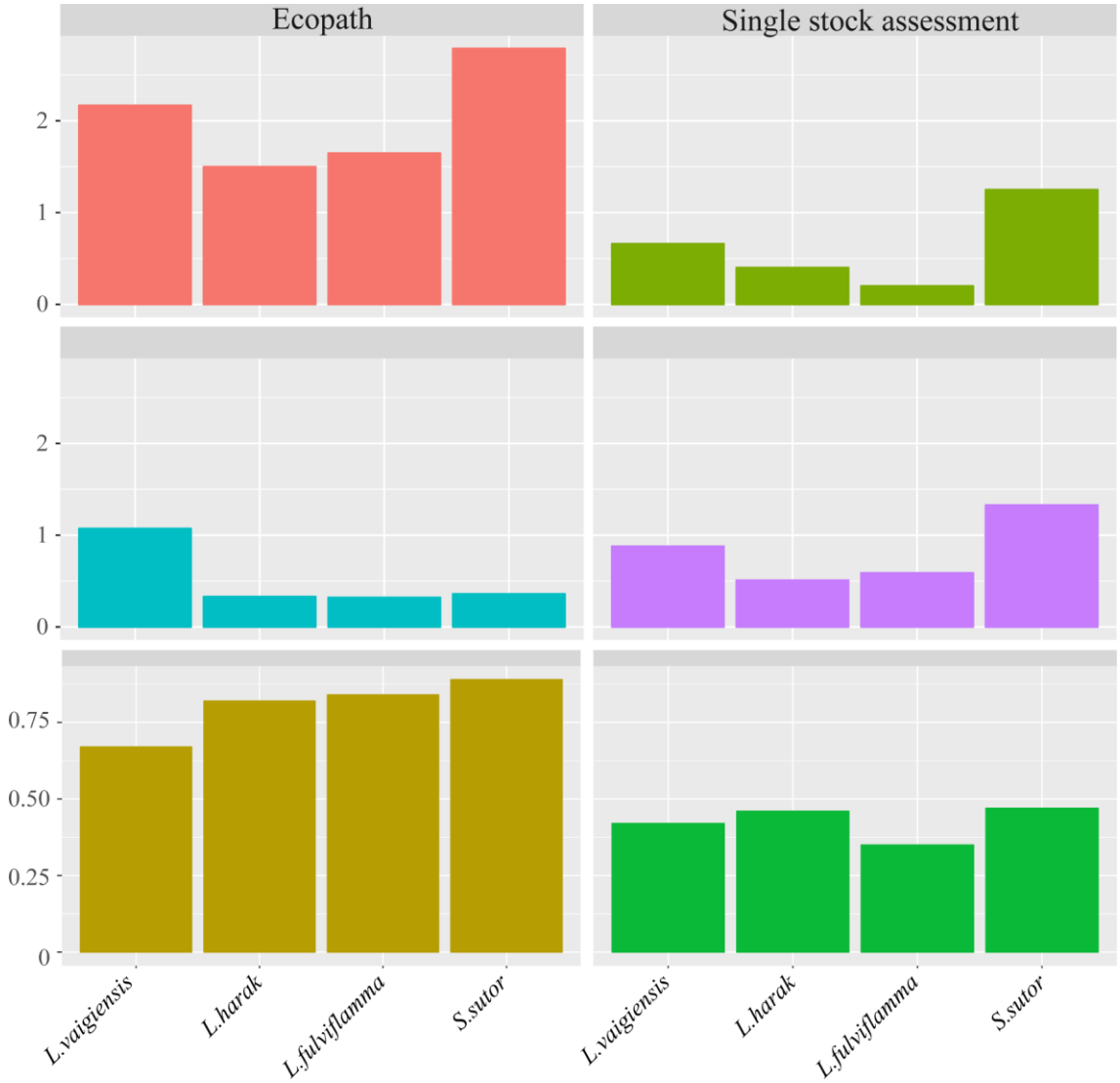

Figure 32. A comparisons of the estimated mortality rates for the selected species based on the single stock assessment and Ecopath model

5.3.8 MODEL UNCERTAINTY

The pedigree index for the Gazi Bay model was estimated at 0.531, which is comparable to that reported for the red sea model by Tsehaye and Nagelkerke (2008), and also within the range of values reported by Morissette (2007) for other trophic models. However, our estimates are comparatively lower than those reported by Villanueva et al. (2006), who reported pedigree indices of 0.75–0.79. Therefore there is still need to refine our model further with contemporarily collected data from the study area.

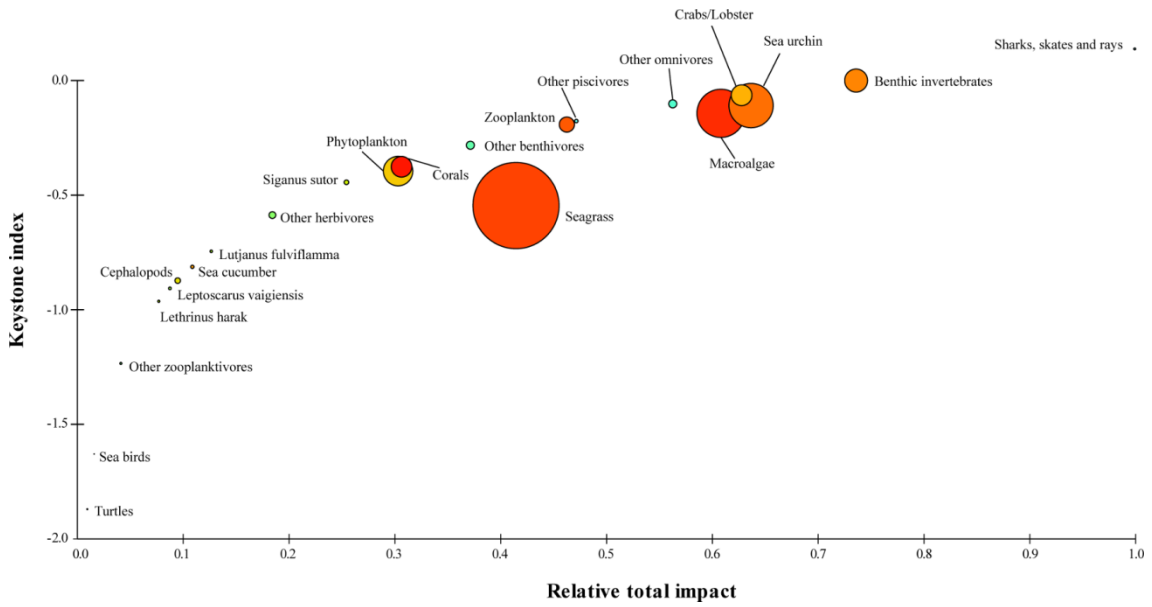

Figure 33. Keystoneness index and the overall effect of each functional group from the Gazi Bay ecosystem model. Keystone species/ groups are those with a higher overall impact (close to 1) and higher Keystoneness index (close to 0).

The results of the Ecoranger routine yielded an average of 76 successful runs out of a total of 10,000, with the least sum of squares of 14.17 suggesting that the model was tightly fitted to the data since there is no major dissimilarity between the output needed to provide mass balance from the original inputs (Gubiani et al., 2011; Tsehaye and Nagelkerke, 2008).

The results of the sensitivity analysis (through the variation of the input parameters in steps of 10% from – 50% to + 50%) reveal substantial differences between model groups in the magnitude of change in the response parameters to variations in the input parameters (Figure 34a). The functional groups are more sensitive to changes in their input parameters, such that a change of 50% in the input parameter would vary the output parameter of that group by 100%. Figure 34b illustrates the sensitivity of selected functional groups to the piscivores biomass input parameter. The ecotrophic efficiencies (EE_i) of the main fish groups undergo a 5-27% decrease when the biomass of the piscivores is reduced by 50% indicating that the fish groups were less sensitive to this parameter.

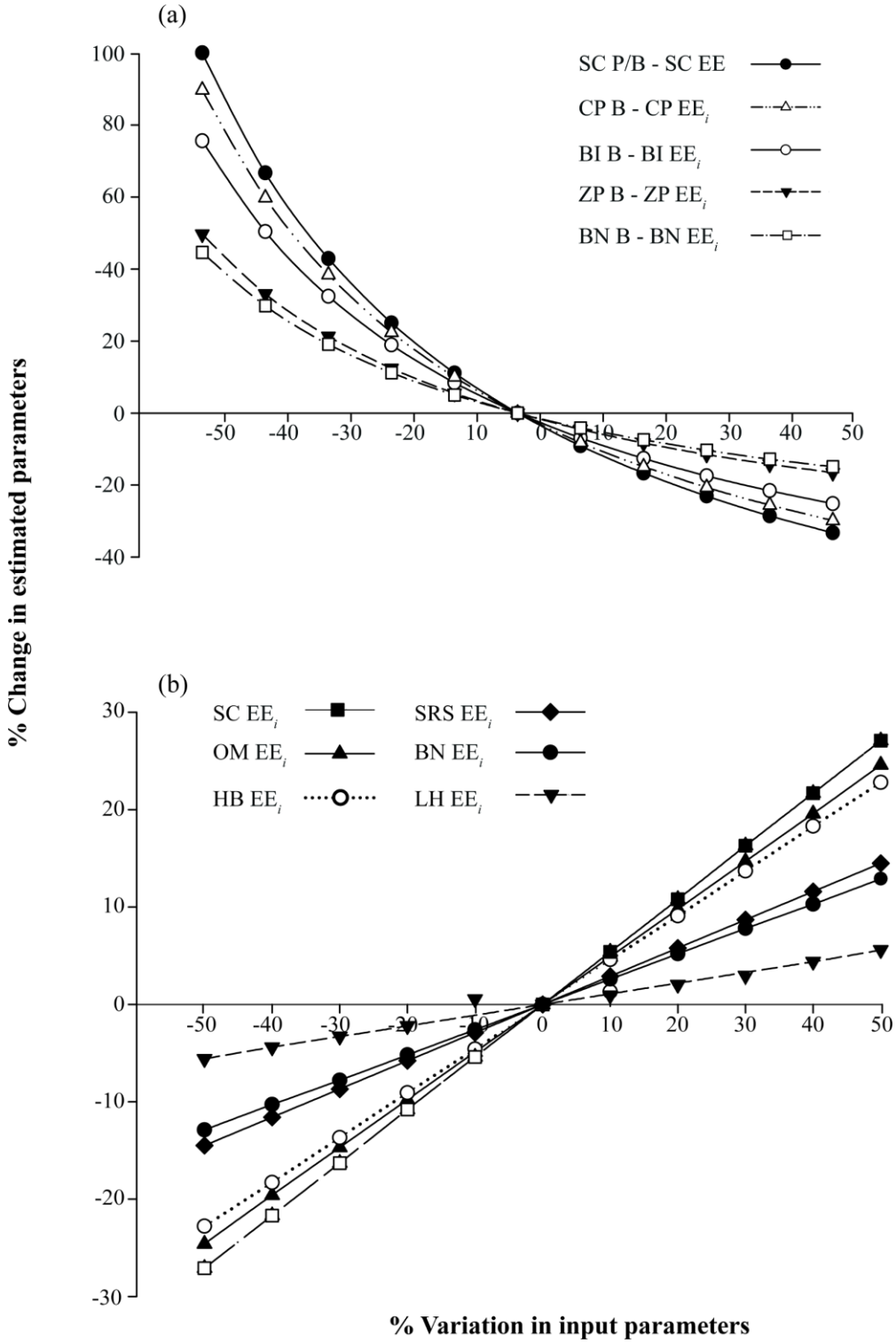

Figure 34. Results from the sensitivity analysis of the Gazi Bay model applied to the input parameters (a) P/B and B; (b) B. see Table 13 for abbreviations.

Table 13: List of abbreviations for Figure 34.

Abbreviation	Full name
B	Biomass
BI	Benthic invertebrates
BN	Benthivores
CP	Cephalopods
EE_i	Ecotrophic efficiency
HB	Herbivores
LH	*Lethrinus harak*
OM	Omnivores
P/B	Production to biomass ratio
SC	Sea cucumber
SRS	Sharks, rays and skates
ZP	Zooplanktons

5.4 DISCUSSION

The ecosystem model developed in this study represents the first attempt to model the trophic components of the Gazi Bay ecosystem using the Ecopath approach and to understand the impacts of fishing on the ecosystem. The model output (based on the mixed trophic analysis) indicates that primary production and detritus are major drivers of the system, which denotes an important bottom-up control of the ecosystem (Hunter and Price, 1992). The importance of the detritus and primary production pathways in such tropical shallow-water ecosystems was also noted by Vega-Cendejas and Arreguín-Sánchez (2001). Due to the low D/H ratio (D/H = 0.75)

found it would appear that for Gazi Bay herbivory plays a major role in the system compared to detritivory. Compared to other systems with a comparativly higher D/H ratio (D/H = 3.4 - 8.6) (Monaco and Ulanowicz, 1997), the low D/H ratio could be indicative of a more effective utilization of the primary production in Gazi Bay. These findings corroborate with previous results obtained from stable isotope analysis on potential food items, which suggest that primary production is by far the the most important organic matter source in the bay (Marguillier et al., 1997; Nyunja et al., 2009).

Like most tropical bay system with extensive mangrove forests, mangrove litter constitutes one of the main sources of detritus in this system (Kihia et al., 2010; Slim et al., 1996). However, being shallow and open, other authors (Kitheka et al., 1996; Osore et al., 1997) have estimated that up to 70% of the detrital pool is exported to the adjacent systems as a consequence of high rates of tidal flushing. This is typically greater than the assumed 50% export rate for other mangrove systems (Jacobi and Schaeffer-Novelli, 1990) and could be the reason for the low cycling index (FCI = 7%) found. According to Finn (1976), the bigger the fraction of the ecosystem throughput that is recycled, the greater is the ecosystem's ability to maintain its structure and integrity (Monaco and Ulanowicz, 1997). Thus, compared to 41 other aquatic ecosystems (Christensen and Pauly, 1993), we can infer that the low FCI for this system suggests a low capacity to recycle detritus, which is typically a sign of disturbance (Christensen, 1995; Ulanowicz, 1986; Vasconcellos et al., 1997).

Meanwhile, systems in their early development stages exhibit a high ratio between total primary production over total respiration ($P_P/R>1$), but this is expected to move towards unity as the system matures (Christensen, 1995). Thus, the deviation from unity is indicative how far a system is from attaining maturity (Odum, 1969). As it is, the ratios between computed production and biomass ($P_P/B = 6.8$), and primary production and respiration ($P_P/R = 1.89$) and between system biomass and throughput ratios (B/T = 0.056) for Gazi Bay are in agreement with the above results, which suggests that the system is in an immature, perturbed (from

ecological/environmental or anthropogenic impacts) state (Christensen, 1995; Fetahi and Mengistou, 2007). However, we should note here that our estimates of total throughput and P/R (and of other system statistics) would differ remarkably if we had included bacterial activity in our model (Christensen and Pauly, 1993; Nyunja et al., 2009). We have not done so because of a lack of knowledge on the bacteria compartments of the system and because any attempt to compare between these results and those of other tropical bay system models (none of them considering bacterial activity) would not be appropriate.

The current status of the Gazi Bay is likely due to the strong human impacts (Huxham et al., 2004) particularly the heavy fishing. Christensen (1995), showed that when ecosystems are disturbed, notably by fishing, the maturity decreases. In their comparative study of two marine ecosystems, Christensen and Pauly (1998) modeled the fished and unfished state of the ecosystem and concluded that for both systems disturbances in the system caused by fishing activity led to a decline in the maturity of the system. Nevertheless, there is need to use additional measures of ecosystem health to ascertain the impact of fishing. Another measure that has been proposed and has been widely used as an indicator of fishery effects in aquatic system relates to the trophic level of the catch (Pauly et al., 1998).

The mean trophic level of catch reflects the fishing strategy in terms of the food web components selected. The very low value for Gazi Bay (TL_c = 2.38, equivalent to primary consumer) suggests that high TL species have increasingly been reduced over the years while low TL species have increased in catch volumes (Jackson et al., 2001). Previous studies have already shown that low TL species dominate a significant portion of the catch (McClanahan and Mangi, 2004). Observed declines in the total catch and decline in the average size of fish landed further support this notion (McClanahan and Mangi, 2001). Thus, in our view, these trends most likely reflects the intensive fishing of the large piscivores and shift in the smaller benthic associated species (Cinner et al., 2009; McClanahan et al., 2008; McClanahan and Mangi, 2004).

Nevertheless, a low TL_c of the catch alone may not reflect critically strong fishing but just a concentration of the fishing effort on low TL species. The gross efficiency of the fishery (GE_f = total catch/primary production) may be an adequate measure (Coll et al., 2009). Values of the GE_f are higher for systems with a high fishing impact and with fishery targets of the low trophic level (Coll et al., 2009). The estimated value of 0.002 is ten times higher than the weighted global average of 0.0002 (Christensen et al., 2005) but comparable to that of the Adriatic sea, which similarly exhibited low TL_c (Coll et al., 2009; Coll et al., 2006). Overall, this supports the notion that the system has been severely modified through intense fishing (Christensen et al., 2004).

Tudela (2003), proposed the use of the relative primary production required by fisheries (%PPR) in combination with the trophic level of catch (TL_c) as an index to capture the effect of fisheries. Based on this approach it is expected that for a given %PPR, a fishery with a higher TL_c would be less disruptive than a fishery with a lower TL_c and vice verse (Tudela et al., 2005). To compare the current status of the Gazi Bay with other coastal and coral reef ecosystems, Figure 35 represents a compilation of % PPR estimates for 25 previous Ecopath models plotted against their respective TL_c (adapted from Tudela et al. (2005)). The original work included 49 previous EwE models, but for this study, we preferentially selected models representing tropical shelves and seas, and the coastal and coral reefs ecosystems (18 and 7 models, respectively). The resulting plot showed that overfished systems exhibited a wider TL_c range of 2.2–3.9 and %PPR of 2.8–89.5. Further, the results highlight the fact that the level of fisheries impact in the Gazi Bay is comparable to some of the most intensively exploited coastal and coral reef ecosystems (Tudela et al., 2005).

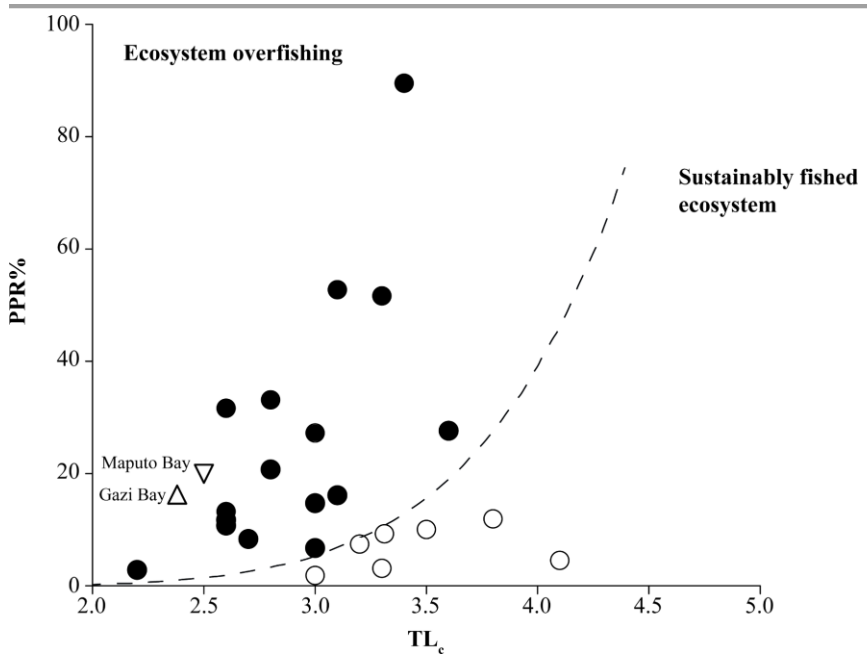

Figure 35. Ecosystem-based reference framework based on %PPR-TLc (Tudela et al., 2005), plotted for the tropical shelves and seas and the coastal areas and coral reefs with reference (dotted line) showing 50% sustainably fished situation.

The above results from the Ecopath model fit the general perception of overfishing in the region (Hicks and McClanahan, 2012; Kaunda-Arara et al., 2003), where the key commercial species exhibited a high exploitation rate (F/Z = 0.5-0.82) above the recommended rate of 0.50 for sustainable fisheries management. A comparative analysis of fishing mortality (F_i) and predatory impacts (*M2*) reveal that fishing mortality is by far the leading cause of total mortality (Z) for the commercially important species. For instance, F_i accounted for 88% and 70% of total mortality for the *Siganus sutor*, and *Leptoscarus vaigiensis* respectively. Previous stock assessment of these species carried by Hicks and McClanahan (2012) following a single species stock assessment approach resulted in lower estimates of *E* for the *Siganus sutor* (*E*= 0.53) compared to that estimated by our model (*E* = 0.89).

In the Ecopath model, fishing mortality (*F*) is calculated based on the catch taken of the overall production (P/B * B) and is thus dependent on the biomass estimate. It is possible that the differences in exploitation rates (F/Z) between our model and previous assessments could have resulted from an underestimation of the stock biomass, leading to an overestimation in the fishing mortality. Nevertheless,

the computed exploitation rates of key resources, - while somewhat differing between our model and previous assessments-, suggests that this system is heavily exploited and that some of the key resources (e.g.Groupers-Serranidae) are already fished beyond MSY (Kaunda-Arara et al., 2003). Therefore, under the current circumstances, the fishery will most likely continue to decline until the fishing effort is reduced or production increased.

A comparison of our estimated yield (4.63 t km^{-2} yr^{-1}) to those reported for the Kenyan coast indicate that the catch estimates from Gazi Bay lie within the range of values reported for the Kenyan coast (3 to 16 $t.km^{-2}.yr^{-1}$) (Kaunda-Arara et al., 2003; McClanahan and Kaunda-Arara, 1996). McClanahan and Mangi (2001), reported catch estimates for the surrounding Galu and Kinondo to be 5.64 $t.km^{2}.yr^{-1}$ with Diani-Mwaepe reporting the lowest catch per area at 3.36 $t.km^{-2}.yr^{-1}$. However, previous trawl surveys conducted within the Gazi Bay have reported much lower fish density for Gazi Bay (De Troch et al., 1995; Huxham et al., 2004; Kimani et al., 1996), reflected in our low B values used for the model. Therefore, it is likely that a significant proportion of the catch comes from fish migrating in from deeper offshore assemblages or that some fishing is taking place in the adjacent systems but landed in Gazi (McClanahan and Mangi, 2001). If so, the fishery impacts exhibited in our model (low TL_c and high exploitation rates) are both a reflection of the system and the adjacent fishing sites (Hicks and McClanahan, 2012; McClanahan and Hicks, 2011).

The mixed trophic impact analysis MTI demonstrated that both, gill nets and the beach seines heavily impact a large number of groups, while the principal targets of basket traps and spear guns are two species: *siganus sutor* and *Leptoscarus vaigiensis*. Effort intensification of those gears would reduce their abundance thereby releasing grazing pressure on the macroalgae. In one of the earliest trophic models of the coral reef ecosystem from the region, McClanahan (1995) simulated the impact that fishing would have on the reef structure and processes. The simulations showed that increasing the harvesting of higher trophic level fish would positively impact the herbivores and corals besides negatively impacting on the macroalgae. It is showed,

however that fishing on both piscivores and herbivores has the potential of increasing fisheries production even at a higher fishing effort. Nevertheless, there is a need for caution as either case, the intensity and selectivity of the fishing gears would structure the ecosystem size and processes (McClanahan, 1995).

Libralato et al. (2006) demonstrated the importance of identifying keystone species in the face of exploitation or stress in the ecosystem. Keystone species are critically important in influencing ecosystem (Power et al., 1996) and their role should be considered when considering effective conservation strategies for species-level prioritization (Valls et al., 2015). For example, McClanahan (1995) identified the red-lined trigger fish *Balistapus undulutus* as an example of keystone species in Kenyan reef lagoons due to their role as sea urchins predators. Intensified fishing pressure on top predators resulted in a release of the sea urchins thereby impacting on the reef ecology including coral and algae interactions (McClanahan, 1995). Thus, the absence or decline of sea urchin predators in the system may account for the high sea urchin biomass (B = 65 t km $^{-2}$) and the corresponding low EE_i (EE_i = 0.45) observed in this model, which is consistent with the studies of McClanahan (1995) and Pinnegar and Polunin (2004).

In this study, the functional group sharks, rays and skates had the highest keystoneness index (0.147) and relative total impact (1.0) and therefore qualified as keystone species based on the theory that keystone species have keystoneness values close to or greater than zero (Libralato et al., 2006). Sharks and rays have been identified as keystone species in many ecosystems (ranking first or second in many models) where they exert strong top-down effects (Libralato et al., 2006). However, in shallow coastal ecosystems, the benthic groups exert a significant bottom-up effect by transferring energy from the detritus to higher trophic levels and hence being potential keystone species (Bustamante et al., 1995; Ortiz et al., 2013). Therefore, it would be a great oversight to ignore food web interactions (at lower levels), as they have the potential to structure the dynamics of the ecosystem of which they are an important part.

5.5 CONCLUSION

The Ecopath model presented here represents the first attempt to analyze trophic structure and functioning of the Gazi Bay and the influence that fishing has on the ecosystem. Two previous studies dealt trophic linkages in the ecosystem by examining the contribution of primary producers in supporting food webs (Nyunja et al., 2009) and food web relationships for commercially important species in Gazi Bay not using EwE (De Troch et al., 1998; Wakwabi, 1999).

The description of system biomass distribution along trophic levels and of energy flow between trophic levels indicated that primary production and detritus exert an overall positive impact on higher trophic levels through a bottom-up control. In contrast, the upper trophic levels -predatory fish- are comprised by little biomass and little throughflow as a result of intense fishery over the past decades. Composite measures such as the P_P/B, $P_P/R/E$ and B/T ratios, (measures of ecosystem maturity, *sensu Odum*) describes an immature ecosystem towards maturity, which may be explained in part by the intensive human exploitation of the resources through fishing (Huxham et al., 2004).

The outcome of the multi-species stock assessment presented here is consistent with previous stock evaluations that have similarly concluded that the current status of Gazi Bay ecosystem is probably due to fisheries impacts, which have resulted in excess fishing mortality (F/Z>0.5). Unlike standard single-species stock assessments, this multi-species approach allowed us to examine trophic relationships and sources of mortality for the commercially important species and enabled us to assess the effects of varying fishing effort on target resources. Fishery impacts are more pronounced in the ecosystem as evidenced by the high gross efficiency, the low mean trophic level of the catch, high exploitation rates and high primary production required to sustain the fishery (Pauly and Christensen, 1995). Our results show that single-species management approaches in places, such as gear restriction and size

limits may not be sufficient for sustaining the multispecies fisheries (Beddington et al., 2007; Tsikliras et al., 2013).

Therefore, alternative approaches, such as the control and reduction of the fishing effort and the establishment of certain areas closed to some fisheries may be better measures towards ecosystem-based management. This should be done while considering the fishing impacts, the economic and social benefits within the ecosystem context. Because Ecopath is a steady-state model representing a snapshot of the ecosystem (1 year), it is not possible to appraise the temporal changes that may occur in the system. Nevertheless, our results provide a scope to further improve the current model by incorporating temporal and spatial analysis to verify the long-term impacts of fishing including seeking optimal management choices. This would necessitate improving the trophic model further by refining the basic input data, particularly for the fisheries groups. Also, the inclusion of seasonality, which is a major driver of both primary and secondary production in the system (Osore et al., 1997), may provide substantial improvements in modeling the Gazi Bay ecosystem.

5.6 ACKNOWLEDGEMENTS

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CHAPTER 6.

General discussion

6.1 OVERVIEW

The primary objective of this study was to assess the status of the artisanal coastal fisheries in Kenya, and the impact of the fishery on the long-term viability of other fisheries and the ecosystem as a whole. The study also explored the usefulness of the current management strategies in addressing the needs of the multispecies and multi-gear fishery, while simultaneously recommending appropriate measures to improve the monitoring and management of the artisanal fisheries. The study relied on fisheries dependent data to describe the overall size structure of catches and gear selectivity by species and size(*chapter 3 and 4*) and compile evidence to determine the status of the fishery (*chapter 4* and *5*) to provide a framework for recommending management options for the fishery (chapter 6). In general, the study highlights the possibility of using relatively simple data-limited approaches to make a reasoned assessment of the likely status of a fishery in a typical data limited tropical fishery.

The results presented suggest that it is possible to infer the status of the Kenyan coastal fishery based on aggregated catch and effort data. As a first general result from this analysis, it appears that the fishery is overexploited with the current yield and effort exceeding the recommended maximum sustainable yield (MSY) (RQ 1, Chapter 2). Secondly, given the multispecies nature of the fisheries, fishers have diversified their strategies and gears to target a significant part of the entire fish assemblage (species and sizes) with each gear imposing different fishing mortalities on the target species. Thus, the overlap in species and size reflects fishers resource use behaviour, which is to target species and sizes that provide highest revenues (due to both their high abundances and market values) (RQ 2, Chapter 3). A more detailed analysis of the length frequency data from the commercially important target species, - used for assessing their stock status- suggests that current fishing pressure in the Diani-Chale area on these species is moderate to high, with the dominance of immature individuals in the catches indicating an unsustainable fishery (RQ 3, Chapter 4).

A comparison of the single-species stock assessment to the holistic ecosystem trophic modelling approach allowed for arriving at a similar conclusion with regard to the heavily exploited state of the ecosystem, thus highlighting the complementarity of both approaches (RQ 3, Chapter 4). Though the current study, only provides a snapshot of the present situation (1-year data), it nevertheless provides a basis for advice towards a more holistic fishery management and for improving current monitoring programs.

6.2 MAJOR FINDINGS AND IMPLICATIONS

6.2.1 State of the fishery as assessed from official catch and effort data

The results of this part of the study were based on the assumption that the official catch statistics as submitted by the state department of fisheries to the FAO were adequate to evaluate the status of the Kenyan coast fishery. The fishery trends and impacts were first analysed by indirect measures (e.g., declining catch, mean trophic level of catch), to determine the maximum catch limits and effort level. In employing both the Schaefer and Fox models to the fisheries data, we assumed that the combined catch represented one big unit of biomass and that the fishing effort had undergone substantial changes over the period covered (Schaefer, 1954; Fox, 1970).
This approach was chosen because of its simplicity and because the data requirements are less demanding (i.e., landings not defined to species) (Vasconcellos et al., 2005). The results of the study allow for some crucial insights into the artisanal fisheries albeit the caveats surrounding data quality (Le Manach et al., 2015; Malleret King, 2000).

The artisanal fishery in Kenya has grown tremendously in the past sixty years as evidenced by the change in the total catch landed and the increase in effort (both fishers and vessels) and the technological advancement (outboard and inboard engines have increased by 50%) in some measure (Tuda and Wolff, 2015). Over the past decade, the annual catch landed has remained relatively stable. However, a

closer look at the trends in the catch per unit of effort (cpue) and the mean trophic level of the catch, shows a decline of the latter (McClanahan et al., 2008). Using the mean trophic level of landings as an indicator of fishing pressure, as recommended by the Convention on Biological Diversity (CBD) (Pauly and Watson, 2005), it seems that the historical changes (decreases) in the mean trophic level of the catch is indicative of a shift from large-sized (low-productive) high trophic level species to previously unexploited resources of lower trophic levels (but higher productivity) (Obura, 2001; Tuda and Wolff, 2015).

Characterized by the introduction of modern fishing gears such as ring and monofilament nets and investment in motorised vessels, it would thus appear that the artisanal fisheries have expanded their resource base, evidenced by the continual increase in the contribution of pelagic and invertebrates species in the annual catch landed (Tuda and Wolff, 2015). According to the state department of fisheries in Kenya, the maximum sustainable yield (MSY) for the Kenyan coastal fisheries is estimated at 8,781 metric tons at an optimal fishing effort of 4,625 boats (FiD, 2016). Compared to the current fish extraction estimated at 9,800 metric tons and an effort of about 5,600 boats, the current level of fishing effort is excessive and unsustainable (FiD, 2016). Given that the current management objective of the Kenyan coastal fishery is embedded in the principle of MSY, our conservative estimate of MSY from this study (8,264-8,543 metric tons) would also seem to suggest an incipient overfishing of the Kenyan artisanal fisheries (FiD, 2016; Tuda and Wolff, 2015). As a result, yields higher than the presently obtained cannot be expected in the future unless, management efforts are geared towards controlling and reducing fishing effort (Branch et al., 2006; Tuda and Wolff, 2015) (Chapter 2).

A comparison of our catch estimates to that of the reconstructed catch by Le Manach et al. (2015) shows a different trend with a more significant decrease in catch volumes from the late 1990 to the year 2000 and a stronger increase in catches over the past decade (Figure 36). The discrepancies between the two datasets results from the inclusion of (discards) of the industrial, artisanal, recreational, and subsistence

fishing sectors in the reconstructed catch, which resulted in a figure that is 2.8 times higher than the official catch reported to the Food and Agriculture Organization of the United Nations (FAO) (Le Manach et al., 2015). The implication is that it is probably most realistic to suggest that the existing situation in the Kenyan coastal fishery could be much more critical than here presented, given that there is incomplete coverage of landing sites, which may lead to underestimation of the national statistics (King, 2000; Le Manach et al., 2015).

However, it was not within the scope of this study to evaluate the quality of the catch statistics. Here, we demonstrate that this first-order assessment of the fishery is useful for retrospective analysis of trends and a diagnosis of a fishery in a data-limited situation. Further, the results from this analysis may assist in clarifying problems, as well as suggest a suitable methodology and sampling technique to suggest solutions for improved data collection (Cheung and Sadovy, 2004; Pilling et al., 2009). Nevertheless, more detailed stock assessment of the different target species was considered necessary to allow for the development of an ecosystem-based management regime (Chapter 3 & Chapter 4).

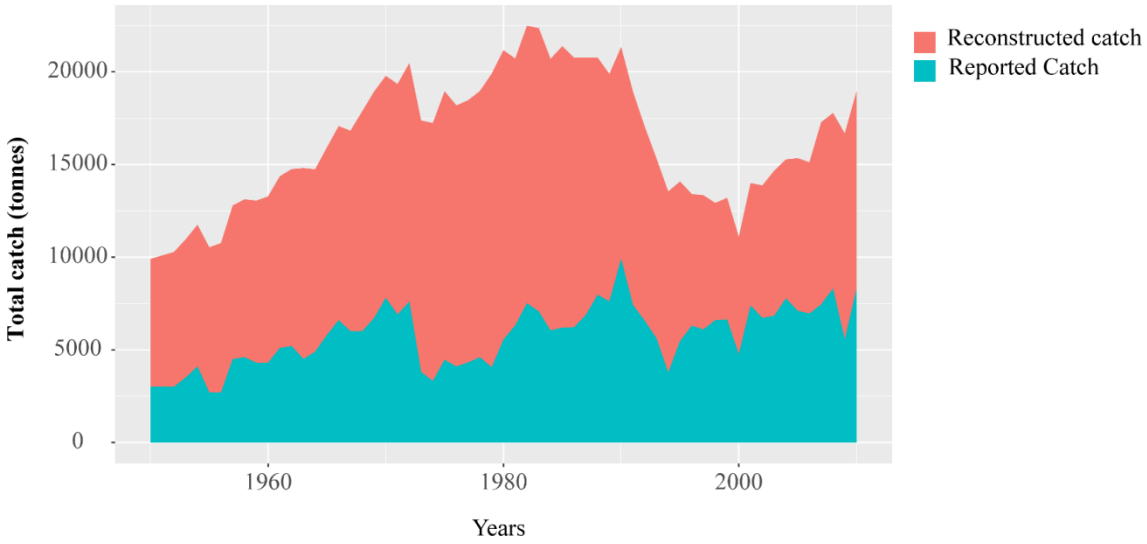

Figure 36. Comparison between the official reported catch statistics from the Kenyan marine artisanal fisheries as reported to FAO and the reconstructed catch (from 1950 to 2010). Data source (Sea Around us).

6.2.2 Characteristic of the fishery: species and size selectivity

Given the course nature of the catch landing data as presented in the first section of this study (Chapter 2), it is only possible to infer general trends of the fishery such as the decline in the estimated biomass/yield as related to the increased overall fishing pressure. These data do not allow, however, to detect possible decreases in catch size and yield of the targeted species. Respective knowledge would complement the general assessment of catch and effort time series and would allow making more informed decisions about fisheries management. Accordingly, this kind of research was conducted for a part of the Kenyan coast, the South coast covering four landing sites, considered representative of the multispecies and multi-gear fishery typical to the Kenyan coast (Tuda et al., 2016).

The results show a size related exploitation of the multispecies fishery with a dominance of small to medium-sized species and individuals with only a few species dominating the catch (Mangi and Roberts, 2006; Tuda et al., 2016). Nevertheless, the species composition and size of the landed catch was greatly influenced by the gear types, with fishers targeting a broad range of species and size range (Tuda et al., 2016). The observed overlap in species and sizes harvested by the different gears may be in part, due to the spatial overlap in fishing sites between gears but also due to the fishers and market preferences (Tuda et al., 2016). For example, some of the fishing gears are specific to a particular depth range or habitat, and if the zone is highly targeted, there is a likelihood of overlap in the observed catch composition.

Therefore, the question emerges here if the overall size structure observed in the catches (dominance of small to medium sized individuals) is a reflection of the gear selectivity pattern of the multigear fishery (i.e. fish are small because small meshes are used) or an indication of an unsustainable fishery (older and larger fish have already been removed from the stock). In other words, are large fish rare in the catches due to overexploitation or because their probability of capture is low for the gears used (Gobert, 1994)? Based on the current results, it is evident that fishers by

diversifying their gears, manage to target the entire assemblage (species and size), with each gear imposing different mortality to different sizes of the population. However, the current fishing practices seem to have resulted in the removal of the older and more fecund size classes, which are not even found in catches of gears with large mesh sizes. As a result, mean sizes landed are small and a truncation of the size structure of the aggregated catches can be observed (Ault et al., 2005; King, 2007; Sparre and Venema, 1998). This general interpretation is supported by previous studies, which have also highlighted the direct impacts of fishing gears in the overexploitation of large groupers (Agembe et al., 2010).

However, given the nature of vessels and gears used by most artisanal fishers along the Kenyan coast, most of the fishing activities are confined to the shallow coastal lagoons, which also double as the nursery and foraging sites for most of the reef-associated species. These habitats are dominated by juveniles and small individuals are therefore targeted and subjected to much higher fishing pressure and in turn high juvenile mortality (Tuda et al., 2016). Therefore, the dominance of the smaller sized fish in the catch may also reflect a shift in the fishers preference for fish at a smaller size (and higher productivity), captured with smaller meshed basket traps. Given that fishing gears such as long lines and ring nets are used further offshore, one would assume that the catch (i.e. fish sizes) from these gears would be different from other gears used inshore. However, the results suggest that the mean size of the individuals landed from all gears, was not significantly different (Tuda et al., 2016).

Thus, the low numbers caught of the larger sized individuals can be seen as evidence of a fishery experiencing juvenescence, where individuals are caught at very small relative sizes. Hence it is likely that both, growth and recruitment overfishing is occurring (Tuda et al., 2016). However, in recent years there has been a growing literature of multispecies fisheries, which suggests that fishing patterns (i.e. the mix of different gears used) often respond to the available biomass rather than selecting for species or sizes (Bundy et al., 2005; Kolding et al., 2016; Kolding and van

Zwieten, 2014). Therefore, the dominance of the small-sized individuals in the catch as observed in this study may be attributed to fishers response towards the smaller elements of the fished community with the highest biomass and turnover rates (Kolding and van Zwieten, 2014). Thus, the observed increase in the number of smaller mesh size gillnets and basket traps in the Kenyan coast may be a strategy by which the fishers maximise individual catch rate under an increased overall effort. Therefore, although the analysis of size structure of the of the catch as presented in (chapter 3) can provide useful insight into the dynamics of the multispecies and multi-gear fishery, it also highlights the weakness of using size structure alone as an indicator for the fishing effects. Further work is required to determine the exploitation rates of the target species along the whole size spectrum.

6.2.3 Status of commercially important species

Further analysis of the length frequency data from landings of commercially important species based on the Gulland (1970) approach for inferring optimal exploitation (E_{opt} = 0.5) suggests that the mean exploitation rates of most of the target species are at or approaching the level of optimum exploitation(Tuda et al., 2017 *under review*). Like many other length based assessment methods, the length based catch curve method has some limitations in that it assumes a constant total mortality (Z) and the probability of capture over all sizes beyond the length at first capture (L_c)(King, 2007). This assumption appears to be violated considering that the different gears are subject to different selectivities, and further the sampling across the gears was not uniform, which may have led to over- or under-sampling in the catches. In addition, the larger size classes are not well represented in the catch curve, which seems to suggest that they are less affected by the fishing gears (Tesfaye and Wolff, 2015).

Given these constraints, there is a need to explore multiple lines of evidence before using this information to recommend strong conservation and regulatory measures (Die and Caddy, 1997; Prince et al., 2015). This necessitated the application

of the second line of assessment using the same length frequency data to estimate the spawning potential ratios (SPR). Out of the four species assessed, three had SPR estimates below the recommended target level (SPR< 40%), with two of the species having SPR estimates even below the lower limit of 20% indicating both growth and recruitment overfishing. Again, in combination with the results of catch curve analysis, the application of the SPR approach demonstrates that recruitment overfishing is occurring and that substantial effort reductions (F/M <1) are required to optimize yield and preserve the spawning stocks to sustainable levels for all the species.

Even though only a small subset of the commercially exploited species (one hundred and thirty-eight species) was analysed in this study, the trends apparent in these data still allow for a diagnosis of a currently declining fishery with signs of overfishing. However, it is important to note that in a data-limited situation as for the here presented case study, fisheries management decisions based on estimated levels of exploitation should be precautionary, and should continuously be revised as new information is gathered. However, the results may be useful in providing some first line of action to avert further deterioration, which includes a reduction in the fishing effort to mitigate the apparent growth and recruitment overfishing conditions in the fishery. However, there is still a need for comparative studies and for the use of more rigorous and ecosystem-based assessment approaches to provide a better understanding of the entire fishery in the ecosystem context (chapter 5).

6.2.4 Ecosystem modelling approach

A major undocumented and often ignored aspect in small-scale fisheries relate to the negative impacts of fishing gears on the habitats and non-target groups, considering that most of the artisanal fishing gears are considered being benign (Shester and Micheli, 2011). Given the complexity that exists in the multispecies and multigear fishery and the move towards the ecosystem-based fisheries management, the mass balance ecosystem modelling routine EwE was incorporated into the study to look beyond the exploitation rates of target resources, but rather to understand the trophic

interactions in the system, and to evaluate the impacts of current fishing activities on the ecosystem as a whole to provide a basis for ecosystem-based fisheries management (Coll et al., 2009). The results from the ecosystem model highlight that the Diani-Chale ecosystem is in a perturbed state of immaturity likely due to the result of the very intense resource exploitation.These results are based on the outcome of the ecosystem maturity indices, such as ascendency and primary production to respiration ratio, which show that when ecosystems are disturbed, notably by fishing, their maturity decreases. Therefore, it seems that fishing activities in the region have resulted in a significant impact on the ecosystem as a whole with the target species showing signs of being fully exploited.

The above results from the trophic model fit the general perception of overfishing in the region, given that the commercially important species exhibit a very high exploitation rate. By quantifying the primary production required to sustain the fishery (%PPR) and the mean trophic level of catches (TLc), a comparison between Diani-Chale with other tropical system revealed that fishing impact in the system is comparable to some of the most intensively exploited coastal and coral reef ecosystems (*chapter 5*). The analysis of the mixed trophic impacts points to an intimately related system in which fisheries impact not only their target species but also other functional groups of lower trophic levels. For instance, increasing harvest of higher trophic level fish (as has been the reality over the past years) positively impacts herbivores and corals but has a negative impact on macroalgae. These results confirm previous concerns about the sustainability of fishing activities in the area. The observed increase in sea urchin biomass seems to be directly linked to intense fishing of the top predators, which has resulted in a release of the predation by intermediate predators on sea urchins, thereby impacting the reef ecology including coral and algae interactions (McClanahan, 1995).

Contrary to the idea that artisanal fisheries have a relatively low impact on ecosystems, the results from our ECOPATH model suggests that the impacts of the fishing activities are pronounced both at the species level (high exploitation rates) as

well as the ecosystem level (ecological indices). The outcome of the multi-species stock assessment was in congruence with previous single stock assessments, which have similarly highlighted a non-negligible risk of ecosystem overfishing, which if not addressed, may lead to further biodiversity loss and associated economic and social benefits. Clearly, the current management efforts such as mesh size regulation and gear restrictions must be integrated with alternative approaches, such as the control and reduction of fishing effort and the establishment of specific protected areas closed from the fishery.

6.3 SYNTHESIS OF MANAGEMENT RECOMMENDATIONS AND FUTURE OUTLOOK

The assessment and management of tropical artisanal fisheries are challenging given the large numbers of exploited species, insufficient/lack of data and a weak enforcement of existing regulations. In addition to the monitoring difficulties caused by this high species diversity, there is an evident lack of well-defined management goals. However, even in such uncertainties, fisheries managers are still compelled to make decisions even with no or limited information, to avert further deterioration of the biological, economic and social environment (Pilling et al., 2009).

Nonetheless, effective fisheries management should be based on sound scientific evidence, which requires some prior information about the fisheries status. Such information can be obtained either from landing, catch and effort data or from biological surveys as well as from the resource users (Krueger and Decker, 1999; Vasconcellos et al., 2005). However, in the likely situation that the scientific evidence is either lacking or inadequate for guiding management decisions, the fisheries may be considered data-poor. Such is the case presented in this study, where out of the possibly 121 commercially exploited species only about 45 species have been studied in terms of their biology (Fondo et al., 2014). Thus, the Kenyan coastal fishery is a typical data-poor fishery where due to the high diversity and varied life histories of

the species exploited, a full based assessments of all stocks is impractical/impossible given the limited resources to monitor and collect data on all target species.

As has been observed in part of this study (chapter 2 and 3), only a few species contribute significantly to the overall biomass, and fishers show commercial preference to only a small portion of the available species spectrum. Therefore, a key recommendation in the monitoring and assessment of this fishery would be to prioritise the major species in the catch with indicator species identified for further biological assessment (Hicks and McClanahan, 2012; Mees, 1996; Pearson, 1994). By focussing on a small subset, patterns can be more quickly distinguished to identify the need for management actions and improve the existing monitoring programs (Pearson, 1994; Pilling et al., 2009). With no defined criteria to assist in the identification of key/indicator species, the application of the ecosystem mass-balance model EwE (chapter 5), has shown that such tools can be used to provide a viable ecological perspective for aggregating important species thus providing an impetus into ecosystem-level assessment and management of multispecies and multigear fisheries.

Unfortunately, despite the fact that the ecosystem approach to fisheries (EAF) is enshrined in most of the fishery policy frameworks, current regulations and management measures, both in terms of stock assessments and management still concentrate on the use of single species approaches (Skern-Mauritzen et al., 2016). Further, threats to fisheries resources are still primarily focused on the problem of destructive fishing gears and juvenile capture, with management responses leading to constraints being placed on fishing operations, most notably through strategies to limit the minimum size of capture directly through mesh size, gear restrictions as well as indirectly through closed areas. In response, fishers have either tended to invest in unregulated input dimensions such as introducing new gears or increasing gear efficiency(technological creep), which has resulted in the current impasse where the number of new gears and the number of nets with diverse mesh sizes have continued to increase (FID, 2016).

Nevertheless, the limitations of these regulations are that they tend to focus too much on the biological resource status and fail to address the fundamental issues, which relates to drivers of resources exploitation (*chapter 2*) (Mangi et al., 2007; Pilling et al., 2009). For instance, the principal act regulating fisheries in Kenya (*Fisheries ACT CAP 378 revised 1991, 2012*) advocates for a biological management system with the MSY as the indicator for resource exploitation. However, despite the fact that the current fishery has been considered overexploited FiD (2016), much effort is still put on gear restriction and mesh size regulation instead of reducing effort. Therefore, the management system has been criticised for its inefficiency to deal with the current multispecies fisheries considering that it was initially designed for the inland fisheries.

Further, the fishers have been resistant to the regulation because it minimizes their economic gains. Therefore, an alternative approach would be to apply an integrated approach, which takes in to account the fishing pattern (how to fish), effort (how much to fish) and information about resource user behaviour (Kolding et al., 2016). This has the advantage of addressing the uncertainty that often results from these conflicting interests and provides a second level of management as a precautionary mechanism should the first line of management fail (Branch et al., 2006). Of course, this has to be adaptive, mindful of the changing aspects of the fishery while at the same time considering the costs and benefits of the strategy adopted.

6.4 CONCLUSION AND PROSPECTS

The findings from this study have highlighted the efficacy of using fisheries dependent data to assess artisanal fisheries in a data-limited context despite the shortcomings and challenges that have been identified in the collection and reporting fisheries data in most developing countries. The principal conclusions from this study are that the state of Kenyan coastal artisanal fisheries appears to be highly exploited, with current effort levels exceeding the sustainable limit and current

fishing practices resulting in the removal of the older and more fecund size classes, which has resulted in the truncation of the size structure of the aggregated catches. The impacts of the high effort and unsustainable fishing practices are evident both at the species level where the commercially important species exhibited high exploitation rate and low spawning potential ratios, indicative of both growth and recruitment overfishing.

Moreover, the impacts are also visible at the ecosystem level where an analysis of the ecosystem maturity indices have pointed to a system that is immature and perturbed likely due to the fishing impacts. Like other tropical artisanal fisheries, the Kenyan coastal fisheries has changed as is evidenced by the rise in fishers number, motorised vessels, modern/efficient fishing gears likely as a response to competition and market demand for the fish. Therefore, if the current management goal, which is to fish within the MSY is to be achieved, then there is an urgent need to not only apply stricter gear restrictions on the unsustainable fishing practices but also the need to regulate effort (new entrants and gears) into the fishery while improving on the collection and monitoring of catch and effort data to better assess and recommend sound policies for responsible fisheries management. Some of the measures to be considered in the next steps should include an ecosim and ecospace simulation to explore the feasibility of gear based or spatial based management option given that community-based spatial closures are likely to have higher compliance due to community participation.

However, at the moment there are no clear guidelines or criteria on the ecological basis for setting them up. Also for the long-term assessment and management of the artisanal fisheries, there is a need to define suitable criteria for identifying the key species for continuous monitoring and assessment to be conducted in selected landing sites. The assessment should focus on both temporal and spatial patterns of exploitation with priority given to the analysis of exploitation across the different size classes. A suitable suite of approaches can be identified and standardized for making stock status determination and reporting in the absence of

more complete assessments.These methods should be adaptive to accommodate new data needs and should be able to lead to an improved assessment and reporting on the fishery status. Thus appropriate training and constant reviews are needed.

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ANNEX

Supplements for Chapter 1

Supplementary table S1.1. The combined time series of the global official catch data as reported by the Food and Agriculture Organization of the United Nations (FAO) and the reconstructed data (combining official reported data and reconstructed estimates of unreported data including major discards) with reference to individual EEZ. (**Data source**: Pauly D. and Zeller D. (2015). (**seaaroundus.org**)

Year	Reported catch	Unreported catch	Reconstructed catch
1950	17.17	11.29	28.47
1951	18.94	11.67	30.60
1952	19.58	12.85	32.43
1953	20.14	13.24	33.39
1954	22.14	14.21	36.35
1955	23.20	15.43	38.62
1956	24.94	16.00	40.93
1957	24.80	15.51	40.30
1958	25.10	15.67	40.77
1959	27.55	16.90	44.45
1960	30.01	18.78	48.79
1961	33.80	20.20	54.00
1962	36.58	21.33	57.91
1963	38.51	23.33	61.83
1964	42.75	24.35	67.10
1965	42.19	25.36	67.55
1966	45.90	26.71	72.61
1967	49.22	29.27	78.49
1968	52.13	32.00	84.14
1969	50.54	30.61	81.15
1970	56.68	31.44	88.13
1971	56.38	30.48	86.86
1972	51.81	28.66	80.48
1973	51.83	30.01	81.84
1974	55.49	31.07	86.56
1975	54.37	30.58	84.95
1976	57.92	30.59	88.51
1977	58.05	31.89	89.94
1978	61.43	32.29	93.72
1979	59.46	31.12	90.58
1980	59.72	30.02	89.74
1981	62.10	31.08	93.18
1982	65.58	34.11	99.70
1983	63.54	34.33	97.87
1984	68.29	36.44	104.72
1985	70.22	38.88	109.09

Supplementary table S1.1 (continued)

Year	Reported Catch	Unreported catch	Reconstructed catch
1986	74.87	39.38	114.26
1987	75.41	41.12	116.54
1988	78.86	41.78	120.64
1989	79.98	42.46	122.44
1990	76.19	41.90	118.08
1991	75.53	41.62	117.15
1992	77.35	41.96	119.31
1993	78.07	41.06	119.13
1994	83.66	41.63	125.29
1995	83.35	41.72	125.07
1996	84.86	43.63	128.49
1997	84.06	41.86	125.92
1998	76.87	39.55	116.42
1999	82.67	39.23	121.90
2000	83.45	40.03	123.48
2001	81.20	38.46	119.65
2002	81.75	37.21	118.95
2003	78.85	37.27	116.12
2004	83.29	37.83	121.12
2005	81.92	36.51	118.43
2006	79.06	34.61	113.67
2007	79.45	34.17	113.62
2008	78.72	32.81	111.53
2009	78.50	32.42	110.92
2010	75.96	32.35	108.31
2011	83.16	31.36	114.51
2012	79.87	30.39	110.27
2013	80.82	31.80	112.62
2014	81.36	30.29	111.64

Supplementary table S1.2. Results of the bi-annual fishers census detailing the number of fish landing sites, total number of fishers, fishing crafts and foot fishers at the Kenyan Coast.(Source: State Department of fisheries, Bi-annual marine frame survey 2014).

Year	Landing sites	Fishers	Fishing crafts	Foot fishers
2004	110	9,017	2,233	1,559
2006	115	10,254	2,368	675
2008	141	12,077	2,687	2,536
2012	160	13,706	3,118	2,074
2014	197	12,915	2,913	2,086

Conference Presentations and Proceedings

Supplementary table S1.3. List of participated conferences and contributions resulting from the PhD work.

Conference Name	Date	Authors	Title of presentation	Presentation type
ZMT Fisheries workshop: Rethinking paradigms & approaches in fisheries research, Bremen, Germany.	20th – 21st March 2014	Tuda and Wolff, M	Assessing Kenyan coral reef fisheries: current status.	Oral
1st Fisherman regional conference: Sustainable fisheries in the South-western Indian ocean., Mahajanga, Madagascar	10th – 11th September 2015	Tuda and Wolff, M	Size structure and gear selectivity of target species in the multi-species multi-gear fishery of the Kenyan South coast.	Oral
9th Western Indian Ocean Marine Science Association scientific symposium, Wild Coast Sun, South Africa.	26th – 31st October 2015	Tuda and Wolff, M	Species and size selectivity in Kenyan multispecies and multi-gear artisanal coral reef fishery.	Poster
Pathways 2016: Integrating Human Dimensions into Fisheries and Wildlife, Nanyuki, Kenya	10th - 13th January 2016	Tuda, Wolff, M, Breckwoldt, A	Assessing the impacts of illegal/destructive fishing operations and their socioeconomic impacts at the Kenyan coast	Oral
ZMT Fisheries Workshop: Tropical Fisheries in a Changing World., Bremen, Germany.	7th – 9th February 2017	Tuda and Wolff, M	Kenyan coastal fishery: challenges for its assessment.	Oral
10th Western Indian Ocean Marine Science Association scientific symposium Dar es Salaam, Tanzania.	30th October – 4th November 2017	Tuda and Munga	A data-limited approach to assessing artisanal fisheries in the Malindi-Ungwana Bay, Kenyan North Coast.	Oral

Supplements for Chapter 2

Supplementary table S2.1. Historical catch of the demersal fish as reported by the State Department of fisheries in Kenya (FiD) (Source: Annual Fisheries Statistical Bulletin 1982-2012).

Years	Snapper	Grunter	Rabbitfish	Scavengers
1982	293	96	612	721
1983	250	90	747	621
1984	177	86	555	617
1985	176	70	633	661
1986	177	76	657	615
1987	154	79	633	666
1988	162	87	536	624
1989	179	69	569	586
1990	213	94	662	632
1991	242	96	707	666
1992	155	61	496	477
1993	129	65	440	441
1994	116	52	365	353
1995	112	67	387	396
1996	147	65	404	433
1997	144	72	350	361
1998	118	55	355	412
1999	155	79	304	360
2000	120	63	299	334
2001	176	84	404	469
2002	177	84	369	414
2003	118	79	382	421
2004	137	112	388	434
2005	171	65	423	412
2006	193	88	412	477
2007	220	94	420	431
2008	244	135	484	499
2009	254	110	504	447
2012	432	161	645	602

Supplementary table S2.2. Historical catch trends (1990-2012) from the marine artisanal fisheries aggregated by counties as reported by the State Department of fisheries in Kenya (FiD) (Source: Annual Fisheries Statistical Bulletin 1996-2012).

Year	Kilifi	Kwale	Lamu	Malindi	Mombasa	Tana River	Grand Total
1996	1,292	1,334	1,123		2,519	24	6,292
1997	1,051	1,563	976		2,493	24	6,107
1998	389	1,379	991	714	2,834	25	6,332
1999	316	1,563	1,000	783	1,575	34	5,271
2000	179	1,471	992	780	1,299	42	4,763
2001	360	1,789	1,435	1,044	1,774	49	6,451
2002	860	1,930	1,478	961	1,517	101	6,847
2003	443	1,951	1,693	1,312	1,370	199	6,968
2004	414	2,062	1,698	1,412	2,025	172	7,783
2005	427	2,416	2,027	1,311	565	77	6,823
2006	521	2,398	2,229	989	661	161	6,959
2007	826	2,236	1,977	1,292	934	203	7,468
2008	899	3,062	2,195	1,509	927	144	8,736
2009	613	2,907	2,092	1,134	1,041	140	7,927
2010	510	2,454	2,270	1,680	1,134	358	8,406
2011	2,331	2,314	2,396		1,116	789	8,946
2012	2,403	2,373	2,279		1,066	743	8,864

Supplements for Chapter 3

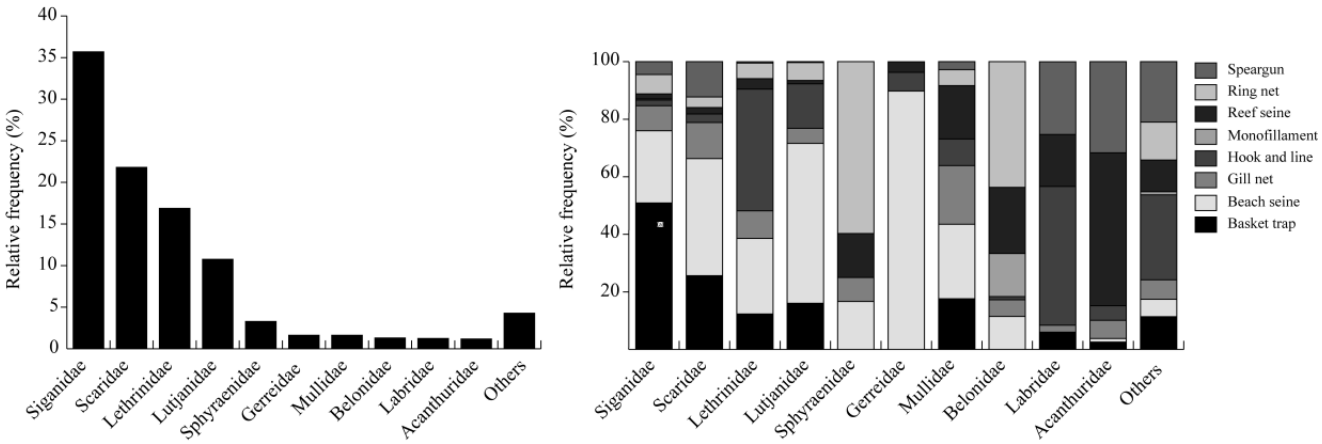

Figure S3.1. The relative contribution of the fish families landed to the overall catch (left) and selectivity of the fish families by gears (right) from the artisanal fisheries in Kenyan South coast.

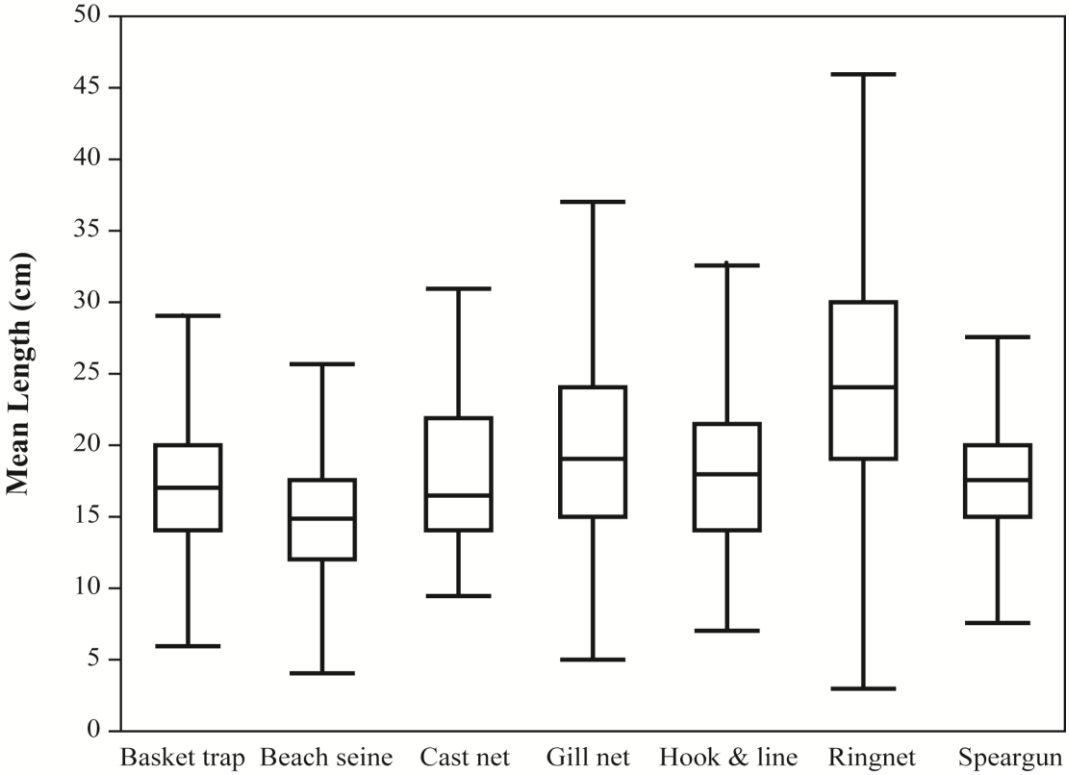

Figure S3.2. Box plot denoting the difference in overall mean size of capture between the fishing gears used in the artisanal fisheries in the Kenyan South coast.

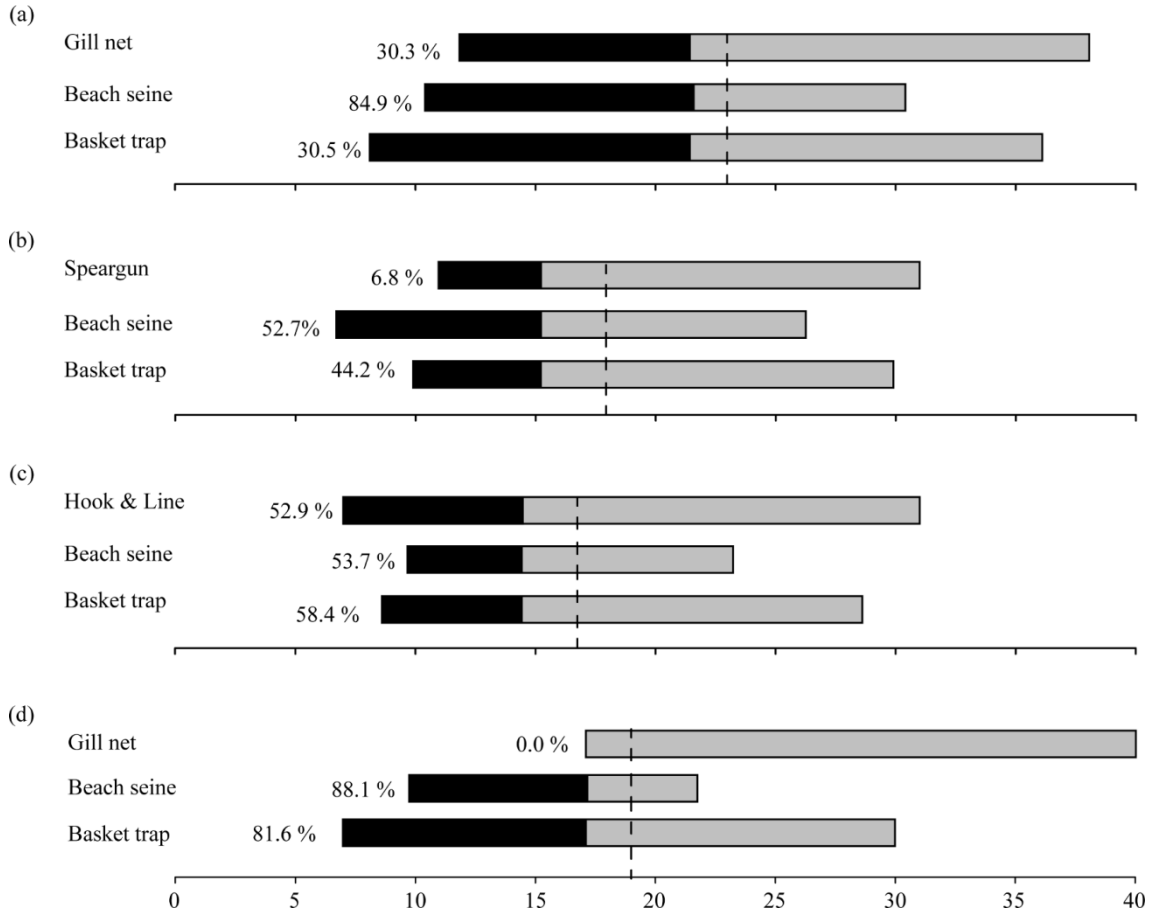

Figure S3.3. Length overlap for different gears used for the four species selected. The bar (black and grey part) represents the total size range of the species' landings, the black part of the bar indicates immature individuals (i.e. those with TL smaller than L_{mat}) of each species and the percentages shows the percentage of immature individuals caught with each gear. The dotted line denotes the optimum exploitation length (L_{opt}). (NB: The size of the dark bar does not represent the actual percentage but is indicative of the size range of the individual caught).

Supplements for Chapter 4

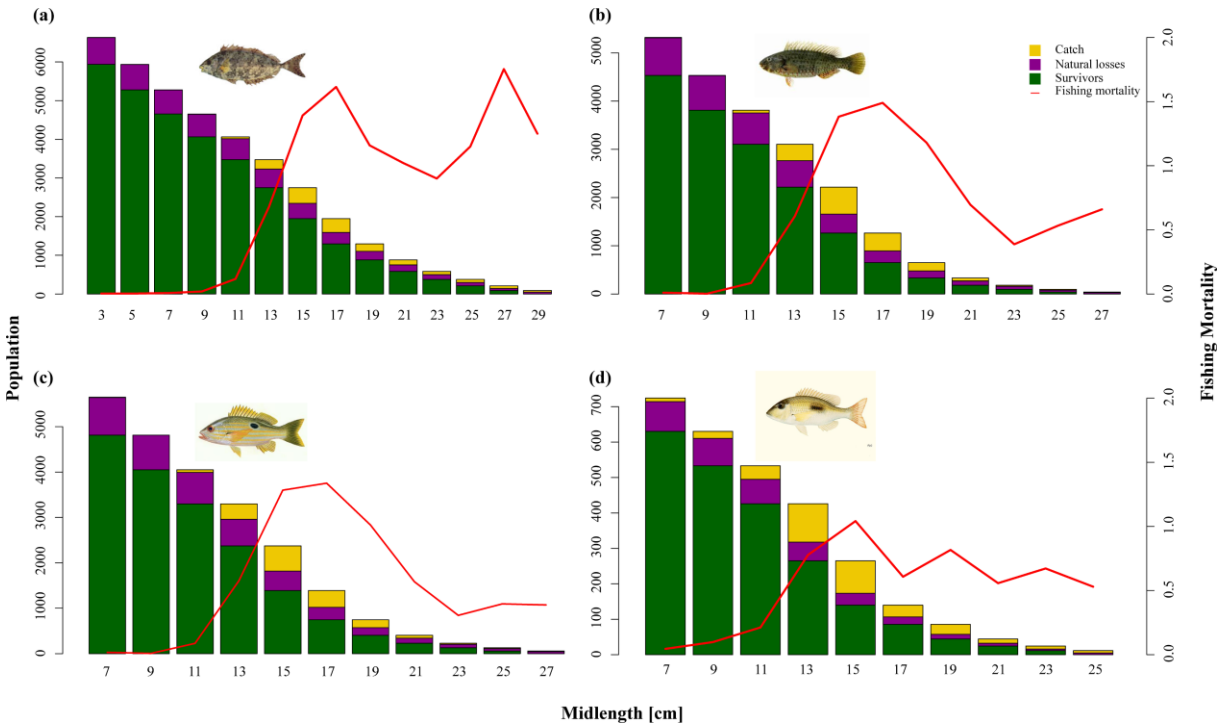

Figure S4.1. The results of the Jones Virtual Population Analysis (VPA), denoting the (i) population as represented by the estimated catch, survivors and natural losses and the (ii) the change in fishing mortality against the binned size classes for the (a) *Siganus sutor*; (b) *Leptoscarus vaigiensis*; (c) *Lutjanus fulviflamma* and the (d) *Lethrinus harak* .

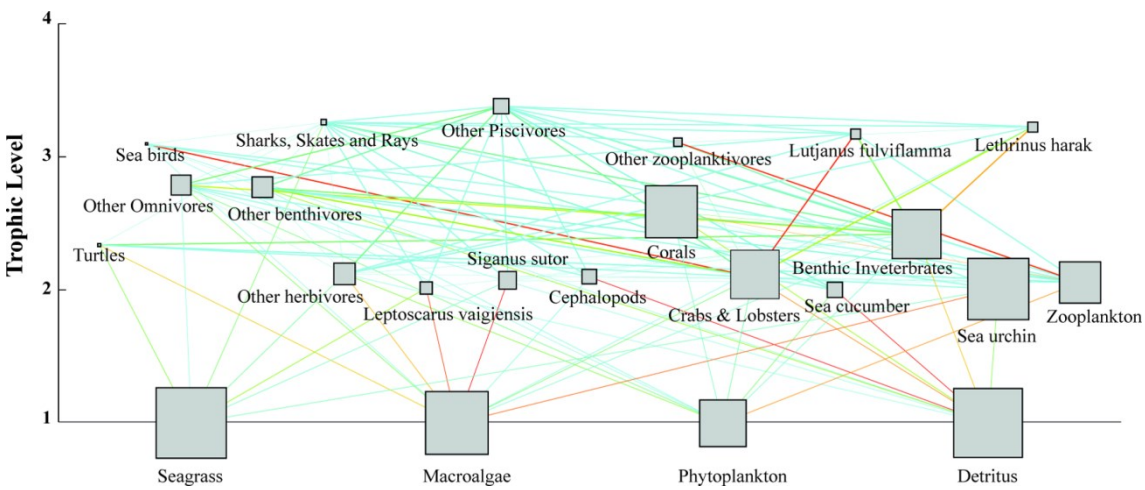

Figure S4.2. The trophic flow diagram for the mass-balance ecosystem model of the Gazi-Bay, Kenya as represented by 23 functional groups. The area of the individual boxes are proportional to the groups biomass in tonnes.km^{-2}.

Eidesstattliche Versicherung

Name: _____ Ort, Datum: _____

Anschrift: _____

ERKLÄRUNG

Hiermit erkläre ich, dass ich die Doktorarbeit mit dem Titel:

- **Assessing the State and Impacts of the Artisanal Reef Fisheries and their Socioeconomic Implications in Kenyan South Coast** -

selbstständig verfasst und geschrieben habe und außer den angegebenen Quellen keine weiteren Hilfsmittel verwendet habe.

Ebenfalls erkläre ich hiermit, dass es sich bei den von mir abgegebenen Arbeiten um drei identische Exemplare handelt.

_____

(Unterschrift)