ECOSYSTEM-BASED ASSESSMENT OF THE MALINDI-UNGWANA BAY PRAWN FISHERY USING ECOLOGICAL INDICATORS

By

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DECLARATIONS

Declaration by student

I declare that this thesis is my original work and it has not been presented for any other degree award.

Signature__________________________ Date____________________________

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This work has been submitted with our approval as University Supervisors:

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DEDICATION

I would like to dedicate this thesis to my lovely family for the endless moral support and to my fiancé Joe Wanyoike for the encouragement and support accorded to me throughout my study period.

ABSTRACT

The study aimed to generate information that will contribute to development of an Ecosystem-based approach to fisheries management (EAFM) of prawn resources in the Malindi-Ungwana Bay, Kenya. A comprehensive ecosystem-based approach is required to holistically assess and manage fisheries resources and their associated habitats. The study identified and assessed ecological indicators based on the objectives of sustainability of harvests, biodiversity conservation, and maintenance of habitat quality. Data analysed were sourced from; the Fisheries Department's landing records, Research vessels, and on-project fieldwork. Trends in historical landings (1985-2010) of prawns from the Malindi-Ungwana Bay were analysed using LOWESS while future landings trends are forecast using an ARIMA model. Community trend indicators based on size-spectra analyses (e.g. of number, biomass and diversity) were used to assess the ecological state of the bay. Biomass Trophic Level spectra (BTLS) and K-dominance analysis were applied as potential tools for analyzing multi-factor effects on the bay. Indiseas-based ecosystem indicators were used to quantify the impact of prawn fishery on the biodiversity of the bay. Results indicate a long-term series with two peaks (in 1997 and 2000) in historical landings of penaeid shrimps with a monotonous decline in catches from 2002. Forecasts predict a steady decline in catches for the next decade (2010-2020) under the current management strategies. Number, biomass and diversity-size spectra analysis made from artisanal landings (2008-2012) indicated effects of fishing on the ecosystem. The number and biomass-size spectra analysis showed increased fishing mortality with time (2008-2012) and an apparent increase in fisheries productivity of the bay. BTLS analysis using the fish by-catch data indicated reduced levels of biomass across trophic levels and a decline in trophic levels of the fish species caught indicating a fishing-down-the food web effects. Biodiversity and conservation based indicators adopted from the Indiseas program (www.Indiseas.org) showed the Malindi-Ungwana Bay ecosystem to be ecologically degraded in terms of fish sizes, trophic characteristics and proportion of predators. The study recommended adoption of the studied ecological indicators and tools as means of evaluating and monitoring the Malindi-Ungwana Bay resources and ecosystem status. However, there will be need to initiate more long-term monitoring programs in order to strengthen the temporal scale of analysis and application of the results. In addition to the indicators described in this study, additional socio-economic and biological data will be needed to develop a holistic EAFM model for the management of the bay resources.

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Abbreviations and Acronyms

ARIMA	Autoregressive Integrated Moving Average
BTLS	Biomass Trophic Level Spectra
CAFM	Conventional Approach in Fisheries Management
CBD	Convention on Biological Diversity
EAF	Ecosystem Approach to Fisheries
EAFM	Ecosystem Approach to Fisheries Management
EU	European Union
EUR-OCEANS	European Network of Excellence
FAO	Food and Agricultural Organisation
GLM	Generalised Linear Model
IPOAs	International Plans of Action
KCDP	Kenya Coastal Development Project
KG	Kilogramme
KMFRI	Kenya Marine and Fisheries Research Institute
LOWESS	Locally- Weighted Scatter Smoother
NEM	North East Monsoon
MT	Metric Tonnes
SEM	South East Monsoon
SWIOFP	South West Indian Ocean Fisheries Project
UNDP	United Nations Development Programme
WSSD	World Summit of Sustainable Development

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May the Lord bless you all!

CHAPTER ONE

INTRODUCTION

Over the past two decades, the basis for an ecosystem approach to fisheries management (EAFM) has been elaborated and prioritized in terms of both scientific and management rationale (Shin *et al*., 2010). Key frameworks, plans, and commitments have paved the way towards implementation of an EAFM around the world. The 1995 FAO Code of Conduct for Responsible Fisheries (Garcia, 2000) provided a reference framework for incorporating ecosystem considerations into sustainable fisheries management. The 2001 Reykjavı´k Declaration (FAO, 2002), and the 2002 UN Sustainable Fisheries Resolution (Shin *et al*., 2010) committed nations to implementing an EAFM, individually and collectively, with the aim of reinforcing responsible and sustainable fisheries in the marine ecosystem. The concepts of EAFM are no longer new, but practical implementation of an EAFM remains a challenge and is yet to be achieved since the 2010 implementation dateline (Garcia, 2000).

Resource managers, researchers, user groups and other interested parties must be involved in fisheries management process to minimize conflict and maximize commitment to sustainable management of resources (Zhang and Marasco, 2003). The EAFM requires that managers take account of a wide range of fisheries impacts when setting objectives, and attempts to meet these objectives will need to be supported by reliable scientific advice and effective management decision making process (Browman and Stergiou, 2004; Pikitch *et al*., 2004).

Ecosystem Approach to Fisheries Management refers to holistic assessment and management of fisheries resources and their associated habitats (Zhang *et al*., 2009). It is intended to ensure that the planning, development and management of fisheries will meet ecological, social and economic needs, but without jeopardizing the options for future generations to benefit from the full range of goods and services provided by marine ecosystems (FAO, 2003). Management of fish stocks is challenging because of the failure of conventional approaches associated with model uncertainties, enforcement constraints and poor policy frameworks (Mayers and Lilorm, 2003) leading to overfishing of stocks, disruption of ocean ecosystem services and loss of biodiversity (Pauly *et al*., 2000; Curry and Christensen, 2005; Worm *et al*., 2006).

Ecological indicators (Garcia *et al*., 2000) are some of the tools which can be used to overcome problems associated with conventional fisheries management and to support EAFM initiatives. Indicators represent a link between objectives and action in management (Cury and Christensen, 2005; Rice and Rochet, 2005; Spellerberg, 2005). Indicators support the decision making process by describing the pressures affecting the ecosystem, the state of the ecosystem, and the response of managers to ecosystem state (Jennings, 2005). They provide a readily understood set of tools for describing the state of fisheries resources and for assessing trends regarding sustainable development objectives, and performance of fisheries policies and management (Rice and Rochet, 2005).

Several indicators may be needed to directly or indirectly track the state of one ecosystem component and attribute or one indicator may track the state of several components and attributes (Shin *et al*., 2005). Progress towards an EAFM will be fastest if a clear process for selecting these indicators is identified by resource managers (FAO, 2003; ICES, 2005). The aim of this study was, therefore, to develop suitable ecological and biodiversity indicators for possible use in future applications of an EAFM of the penaied prawns and biodiversity resources of the Malindi-Ungwana Bay. The Malindi-Ungwana Bay supports the only known industrial and semi-industrial prawn trawl fishery in Kenya. The fishery of the bay was closed in 2006 because of resource use conflicts between artisanal fishers and commercial trawling vessels (Fulanda *et al*., 2011). There is need for a holistic approach to resource management such as EAFM that will ensure minimal user conflicts, socio-ecological sustainability of resource use and support the improvement of the Prawn Fishery Management Plan of the Malindi-Ungwana Bay.

1.1 Problem Statement

The Malindi-Ungwana Bay is a biodiversity rich ecosystem in coastal Kenya dominated by decapod crustaceans and several commercially important species of fish (Fulanda *et al*., 2011; Munga *et al*., 2013). The Kenyan government suspended bottom trawling in the bay in 2006 when resource use conflicts between commercial trawlers and artisanal fishers due to perceived declining catches, habitat impacts, and destruction of artisanal fishing gear escalated because of the continuous encroachment on the artisanal fishing grounds by the commercial vessels (Ochiewo, 2006). The conflict was partly attributed to competition for the highly abundant shrimp species in the 3–5 nm waters (Fulanda *et al*., 2011). Information on the status of the stocks and the biology of the species, including growth, reproductive cycles and feeding ecology was inadequate to inform management decisions leading to an indefinite suspension of the fishery in 2006. To date, bottom trawling in the bay continues to attract increasing criticism due to both the perceived damage to the environment and threats to fisher livelihoods (Munga *et al*., 2013). This study therefore aims to generate indicators that can be used to develop EAFM initiatives as an alternative to the problem prone conventional fisheries management methods.

1.2 Justification of the Study

The "Prawn Fishery Management Plan" for the Malindi-Ungwana Bay was developed by the State Department of Fisheries to resolve conflicts between the trawlers and artisanal fishermen. This management plan followed the Conventional Approach to Fisheries Management (CAFM) rather than the Ecosystem Approach to Fisheries Management. The CAFM has proved inadequate in managing the prawn fisheries of the bay due to lack of basic biological data on the penaeid prawns useful for setting management guidelines (Fulanda *et al*., 2011). There is need to introduce a holistic approach to managing resources of the bay that involves all stakeholders and the environmental dynamics in order to circumvent the shortcomings of CAFM methods. EAFM is one such approach; it offers benefits of managing fisheries in a manner that takes the overall health of the marine ecosystem into account (Cochrane *et al*., 2004). It takes into consideration ecological relationships between species (harvested or not) and balances the diverse needs and values of all who use, enjoy or depend on the resource (stakeholders) now and in the future. EAFM is now accepted as the preferred approach to managing fisheries (Cochrane *et al.*, 2004). This study aimed to develop relevant ecological and biodiversity indicators of the "state and pressures" on the prawn resources of the bay. This information, together with other datasets will subsequently help in the development of EAFM model of the Malindi-Ungwana Bay prawn fishery that will hopefully reduce the resource user conflicts.

1.3 General Objective

The general objective of this study was to describe and assess ecological indicators useful for development of an EAFM model for the assessment and management of the prawn and biodiversity resources of the Malindi-Ungwana Bay, Kenya.

1.4 Specific Objectives

The specific objectives of this study were:

1. To determine the temporal trends in historical landings of the penaeid shrimps and the environmental correlates of the landings in the Malindi-Ungwana Bay, Kenya.

2. To evaluate the usefulness of; **number, biomass, and diversity-size spectra** as ecological indicators for monitoring the status of the Malindi-Ungwana Bay prawn fishery.

3. To develop and evaluate biodiversity and conservation based indicators and tools for purposes of assessing the status of biodiversity within the Malindi-Ungwana Bay ecosystem.

1.5 Research Hypotheses

This study was guided by the following research hypotheses:

1. The landings of penaeid shrimps in the bay have declined overtime due to exploitation pressures and will continue to decline under the present management regime.

2. The biodiversity and conservation based indicators are useful for describing the species and fishery status of the Malindi-Ungwana Bay.

3. Temporal trends in the Size-based spectra indicators (e.g. diversity, number and biomass) are useful for describing the status of the Malindi-Ungwana Bay prawn fishery.

4. Temporal trends in the biomass-trophic level spectra based on fish by-catch can be used to describe changes in fish community structure within the Malindi-Ungwana Bay and serve as biodiversity indicator for the bay.

CHAPTER TWO

LITERATURE REVIEW

2.1 The Prawn Fishery

Malindi-Ungwana Bay on the Kenyan North coast is an important source of high value penaeid prawns. The estuarine conditions brought about by Rivers Tana and Sabaki make the bay an ideal habitat especially of three migratory species of prawn, *Panaeus monodon, P. indicus and P. semisulcutus* (Wakwabi, 1988). Prawn fishery in Kenya has been practiced for several years. Earliest records indicate active fishery from the mid 1970s through expeditions carried out by the Kenya Government with assistance from UNDP and FAO. It was established that a reasonably equipped prawn trawler could land as much as 3 to 4 tonnes of marketable crustacean per day. According to Matagyera (1984), good trawling grounds exist in the Malindi-Ungwana Bay in shallow waters less than 30 meters. The area available for trawling of shallow water penaeid prawns was estimated at 350 nm^2.

Semi-industrial prawn trawling has been going on in the bay for the last four decades. The number of vessels licensed for the fishery has fluctuated between 4 and 20 (Fennessy, 2002). Since its inception, prawn fishery has been ridden with conflicts, which intensified to serious levels in the 1990s mainly between the trawlers, the small scale artisanal fishers and conservation agencies (Fulanda, 2003; Mwatha, 2005; Fulanda *et al.*, 2009). The main contentious issues among stakeholders surround the contravention of the Fisheries Act, which limits trawling to only areas beyond 5nm offshore, destruction of artisanal fishing gears by the trawlers, wastage of fish by-catch, and alleged killing of other non-target species, especially the sea turtles (Mwatha, 2002).

The prawn trawling fishery grounds on the Kenyan Coast are found in Lamu, Ungwana, Kilifi and Shimoni with the major ground being found in the Ungwana area. The Ungwana Bay is an area known to have the highest concentration of prawns which migrate into the area from both the Northern Kenya banks and the Malindi banks (Brakel, 1984). The maximum potential annual yield has been estimated to be about 350 tonnes of prawns (Anon, 2001). The richest fishing grounds in the Ungwana Bay are within about 6 nm from shore in depths less than 20m. The number of trawlers fishing in the area in the last 20 years have varied between 5-20, however, the average annual catches have hardly exceed 400 tonnes of prawns (Fennessy, 2002). The Kenyan prawn fishery is essentially a semi-industrial type whereby 62% of the prawn total annual catch is caught by commercial fishers using trawlers (Fennessy, 2002). The non-mechanised artisanal fishers account for less than 38% of the total annual prawn catch but from inshore waters whose catch composition is mostly sub-adults (Mwatha, 2002). While artisanal fishers account for 97% of the total annual marine fish landings from shallow waters (to depths of about 20 m) around the reefs and in creeks, the trawlers account for 3% mostly considered as by-catch since their main target is prawn trawling (Fulanda, 2003; Mwatha, 2005). In the shallow water prawn fishery, the *Peneaus indicus, P. monodon, P. semisulcatus, P. monoceros* contribute 46 %, 21%, 12% and 20% of total weight landed by trawlers, respectively (KCDP report, 2002).

Prawns Life cycle

Prawns lay eggs which are fertilized from sperms deposited in a sperm receptacle of the female by the male during copulation. The eggs hatch into a pelagic larvae which metamorphose through nauplius, protozoea, mysis and post larvae stages in 2 to 4 weeks (Wakwabi, 1988).

Post-larvae are then taken by currents into shallow inshore areas where they grow into sub-adults during the northeast monsoon season (King, 1995). The sub-adults then migrate to deeper waters where they grow into adults and subsequently breed. The breeding of these species is continuous but shows remarkable peaks during some periods of the year that coincide with established rainy seasons for the region (Wakwabi, 1988). Post-larvae of prawns enter into shallow inshore areas throughout the year with peaks between August and March (KMFRI, 2002). Studies show that all commercial prawn species in Kenya use inshore shallow waters as nursery areas in different proportions depending on near shore habitat (Mwatha, 2002). The artisanal fishery harvest mainly juvenile prawns in inshore areas. Both sexually mature and juvenile prawns inhabit the trawling area in Malindi-Ungwana Bay. This is an important aspect of research that is needed in terms of productivity of the prawn fishery in terms of strict enforcement of the existing regulations measures, including closed seasons must be implemented and monitored by the State Department of Fisheries and BMUs.

2.2 Ecological Characteristics of the Malindi-Ungwana Bay

The Malindi-Ungwana Bay is a rich ecosystem harboring large numbers of near-shore and offshore species. Apart from the inshore prawn fishery, a substantial fish by-catch composed of snappers, groupers, emperors, grunters and jacks pose a threat to the sustainability of the ecosystem (Fulanda, 2003; Mwatha, 2005).

Other inshore fisheries include that of crab and lobster fisheries as well as octopus in the adjacent patchy reefs. The inshore demersal and pelagic fisheries of the bay support a number of fishers who use non-motorized boats. According to early surveys (e.g. Brusher, 1974) there is an offshore demersal fishery for groupers and snappers as well as deep water prawns and lobsters. The richest grounds of deep water lobsters are offshore Ungwana Bay (Matagyera, 1984). Deep water prawns are abundant off Malindi-Ungwana Bay with highest densities of deep water prawns found off Malindi Bay (FAO/UNDP, 1982). The bay is directly influenced by both natural and anthropogenic forces. Among the natural factors that influence this important ecosystem include freshwater and sediment discharge into the bay (KCDP report, 2002). Land use and anthropogenic activities in areas adjacent to this fishing area affect fishing activities. The bay is also influenced by the reversing Somali current and also the East African coastal current which makes it a unique bio-geographic zone (Brakel, 1984). Trawling is concentrated at the estuaries of Rivers Tana and Sabaki and yields between 300 and 600 metric tonnes of prawns annually, exported mainly to the EU and Asian markets (KCDP report, 2002). Discharges of the Tana and Sabaki Rivers into Ungwana Bay are reasonably variable, the peak flows occur in May during the South East Monsoon (SEM) and December during the North East Monsoon (NEM). The suspended sediment concentrations are also variable for both rivers. In both rivers, most of the sediments in transport are inorganic in nature with very small organic component which increases during the dry season (Schneider, 2000). The heavy discharge of terrigenous sediments and huge volume of nutrient-laden freshwater from Sabaki and Tana Rivers probably explains why the Ungwana Bay is the richest and most productive fishery ground along the Kenyan Coast. This means that the management of the fisheries within Ungwana Bay cannot be successful without the management of landuse and water abstraction within the Tana and Sabaki River basins (Schneider, 2000).

2.3 Ecosystem-based Approach to Fisheries Management

An Ecosystem-based Fisheries Management strategy for marine fisheries is one that reduces potential fishing impacts while at the same time allowing the extraction of fishery resources at levels sustainable for the ecosystem (Ward *et al.*, 2002). Predicting the results of any management action is difficult because the dynamics of ecosystems are complex and poorly understood. When fishery managers understand the complex ecological and socio-economic environments in which fish and fisheries exist, it will improve their ability to anticipate the effects that fishery management will have on the ecosystem, as well as the effects that ecosystem change will have on fisheries (FAO, 2003). Political commitments to an Ecosystem Approach to Fisheries (EAF) are increasingly becoming numerous (FAO, 2003). An EAF is intended to ensure that the planning, development and management of fisheries will meet social and economic needs, without jeopardizing the options for future generations to benefit from the full range of goods and services provided by marine ecosystems (FAO, 2003). An EAF requires that managers take account of a wide range of fisheries impacts when setting objectives, and attempts to meet these objectives will need to be supported by reliable scientific advice and effective management decision making (Murawski, 2000; Pope and Symes, 2000; Link, 2002; Sainsbury and Sumaila, 2003; Browman and Stergiou, 2004). More recently, the FAO-Iceland Conference on Responsible Fisheries in the Marine Ecosystem, (Sinclair *et al.,* 2003) brought the issue to the forefront of fisheries requesting FAO to develop guidelines. Finally, the World Summit on Sustainable Development (WSSD, Johannesburg, (September 2002) "encourage the application by 2010 of the ecosystem approach, noting the Reykjavik Declaration on Responsible Fisheries in the Marine Ecosystem and Decision V/6 of the Convention on Biological Diversity" (Larkin, 1996; Lackey, 1999). The Code of conduct for responsible fisheries offers a synthesis of the requirements of all the above instruments and provides the conceptual basis and institutional requirement for, *inter alia,* ecosystem and habitat protection; accounting for environmental factors and natural variability; reducing impacts of fishing and other activities; biodiversity conservation, multispecies management, protection of endangered species, accounting for relations between populations, reducing land-based impacts and pollution, integration in coastal area management, elimination of ghost-fishing, reduction of waste and discards, precautionary approach, delimitation of ecosystem boundaries and jurisdictions, as well as adapted institutions and governance (Grumbine, 1994). Ecological, economic and social indicators are required to support an EAF, consistent with political aspirations for achieving ecological, economic and social sustainability (WSSD, 2002).

2.4 Ecological Indicators

In the context of an EAF, the ways in which groups of indicators are selected for different purposes can be generalized by considering an ecosystem (or more realistically a spatial management unit) with components and attributes. Indicators support the decision making process by:

(i) Describing the pressures affecting the ecosystem, the state of the ecosystem and the response of managers,

(ii) Tracking progress towards meeting management objectives and

(iii) Communicating trends in complex impacts and management processes to a non specialist audience (Garcia *et al.*, 2002; Rice, 2000, 2003; Rochet and Trenkel, 2003).

For reporting and research, indicators are usually chosen to provide good coverage of the components and attributes, where components are defined as functional or species groups and attributes as properties of the components (Jennings, 2005). ‘Good coverage’ is usually achieved by selecting components and attributes that are considered to be representative of the ecosystem, as knowledge and resources will always be too limited to achieve comprehensive coverage (Jennings, 2005).

Size is recognized as a key feature in marine ecological processes and, because fishing is size-selective, the size distribution of marine populations and assemblages is often used to monitor fishing impacts at various organizational levels (Shin *et al.,* 2005). In particular, the 'size-spectrum theory' has been developed for marine ecosystems (Silvert and Platt 1978; Kerr and Dickie, 2001; Benoît and Rochet, 2004; Andersen and Beyer, 2006). This size-based concept describes the ecological processes underlying the biomass size spectrum, the distribution of biomass across body size classes, where each individual is defined by size regardless of species (Platt *et al.*, 1984). When the energy transfer is governed by size-dependent predation, which determines prey mortality and predator growth, unfished size spectra have two classes of dynamics depending on assumed prey size preferences: either a steady state, or an oscillatory solution in which waves move over time along the size spectrum from small to large body size (Law *et al.*, 2009). The question is, how does fishing affect these dynamics? The shape of the size spectrum is known to be sensitive to fishing intensity (Gislason and Rice, 1998; Shin and Cury, 2004). Intuitively selective fishing might generate gaps that would disturb the biomass flow and potentially favour oscillatory dynamics, creating temporal variations in biomass and catch (Pope *et al.*, 2004; Daan *et al.*, 2005). Clearly both fishing intensity and selectivity interact to determine the impact of fishing on size spectrum dynamics. Measures of fishing impacts include indices of both the spectrum shape and temporal variations in shape (Pope *et al.*, 2004; Daan *et al.*, 2005).

2.5 Application of Indicators to EAF

Indicators are needed to support the implementation of an Ecosystem Approach to Fisheries (EAF). They provide information on the state of the ecosystem, the extent and intensity of effort or mortality and the progress of management in relation to objectives (Cury and Christensen, 2005). Indicators should guide the management of fishing activities that have led to, or are most likely to lead to, unsustainable impacts on ecosystem components or attributes. The numbers and types of indicators used to support an EAF will vary among management regions, depending on resources available for monitoring and enforcement, and actual and potential fishing impacts (Cury and Christensen, 2005; Jennings, 2005; Rice and Rochet, 2005). State indicators provide feedback on the state of ecosystem components or attributes and the extent to which management objectives, which usually relate to state, are met. State can only be managed if the relationships with fishing (pressure) and management (response) are known (Wilson *et al.*, 2003; Nielsen *et al.*, 2004). Predicting such relationships is fundamental to developing a management system that supports the achievement of objectives (Wilson *et al.*, 2003)

In an EAF context, the ways in which groups of indicators are selected for different purposes can be generalized by considering an ecosystem (or more realistically a spatial management unit) with components and attributes. To support an EAF, indicators need to track the state of components and attributes that may be adversely impacted by fishing. Progress towards an EAF will be fastest if a clear process for selecting these indicators is identified (FAO, 2003; ICES, 2005).

In developing a management system it is essential that societal/political aspirations can be translated into operational objectives to achieve sustainability (Sainsbury *et al.,* 2000). The first step in selecting indicators to support management is to identify the fishing impacts most likely to compromise the achievement of operational objectives and to rank them in terms of impact and likelihood of occurrence (Jennings, 2005). From a ranking of impacts most likely to compromise sustainability, state indicators relevant to each of the operational objectives, components and attributes can then be selected, with the number and complexity of indicators reflecting the resources available for management (FAO, 2003).

One indicator or a series of indicators may be used to measure progress towards meeting each operational objective. Progress would be measured by comparing the recorded values or trends in the indicator with a reference point, trajectory or direction (FAO, 2003). The identification of state indicators is only one part of the overall process for selecting indicators to support an EAF. Once state indicators have been selected, the identification of associated pressure and response indicators follow (Degnbol and Jarre, 2004). Understanding and predicting the links between pressure, state and response is fundamental to developing a management system to support the achievement of operational objectives (Clarke, 1990; Hilborn and Walters, 1992). The process will be complex when synergistic and additive impacts occur, especially if the interacting impact is due to sectors other than fisheries (Jennings, 2005). If a management response is not, or cannot, be clearly defined by linking pressure, state and response, then there is little value in adopting a state indicator for management (Jennings, 2005). Taking account of the ecosystem effects of fishing and meeting high level policy commitments such as those agreed at the World Summit on Sustainable Development (WSSD, 2002) clearly imply greater reductions in fishing effort than would be required to meet single species objectives, particularly in mixed fisheries where vulnerable species are taken as by-catch (Hilborn, 2004). Effective implementation of an EAF will not be straightforward, and will be compromised by most of the same issues that led to ineffective single species management, notably the difficulty of meeting social, economic and ecological objectives simultaneously and when the short-term costs of doing so are very high (FAO, 2002). For this reason, the debates over setting reference points, trajectories or directions, and taking management action, will still be long and difficult and heavily influenced by short-term economic and social concerns (Rosenburg and Restrepo, 1994; FAO, 1995; Paterson *et al.,* 2007)

In a management framework supported by pressure, state and response indicators, the relationship between the value of an indicator and a reference point, reference trajectory or direction, provides guidance on the management action to be taken by managers (Jennings, 2005). As pressure and response indicators can link to state indicators in many ways (Jennings and Dulvy, 2005), a management framework might require several pressure and response indicators to measure progress towards meeting the target for one state indicator, or several response indicators may be required to support a single pressure indicator (Jennings, 2005). Despite the prioritization of impacts and indicators, risk is most likely to be managed across all indicators in the early stages of adopting an EAF

(Rice and Rochet, 2005). This will often favour an approach where more management concern is focused on target stocks, with management action being modified to account for ecosystem concerns if reference points, directions or trajectories for ecosystem indicators are not met (Jennings and Dulvy, 2005).

CHAPTER THREE

MATERIALS AND METHODS

3.1 Study Area

The study was based on data collected within Malindi-Ungwana Bay on the northern coast of Kenya (Fig. 1). Malindi-Ungwana Bay lies between latitudes $3^{o}30'S$ and $2^{o}30'S$ and longitudes $40^{0}\ 00'N$ and $41^{0}00'$ N. The bay is within Tana River County and includes areas from Malindi Bay northwards to Tana River delta at Kipini (Fig. 1), covering an estimated 200 km of coastline (Mueni, 2006). The bay is the only known trawlable shallow area of the coastal waters of Kenya (Brusher, 1974; Matagyera, 1984; Mwatha, 2005). The Malindi-Ungwana Bay area has a continental shelf ranging from 15 to 60 km in width (Mwatha, 2005). The continental shelf is wider at Kipini (Fig. 1) where it attains a width of between 20-30 km with waters of less than 20 m depth. The shelf is narrower (5-10 km) at Malindi with deep waters averaging 40 m (Iversen, 1984). Malindi-Ungwana Bay fishing grounds cover an estimated 35,300 km^2 of shelf area (Iversen, 1984; Mwatha, 2005).

The bay is drained by two large rivers; Sabaki/Athi and Tana. An average of 6,000 million m^3 of freshwater and about 3 million tones of sediment are discharged into the bay annually by the two rivers (Tychsen, 2006). The bay is affected by the monsoon seasonality that prevails on the Kenyan coast. The coast is influenced by both north-easterly and south-easterly monsoon winds described in details in McClanahan (1988). Briefly, the northeast monsoon season (NEM, November–March) is a period of calm seas, elevated sea temperatures and higher salinities, while the southeast monsoon (SEM, April–October) is characterized by rough seas, cool weather, and lower salinities.

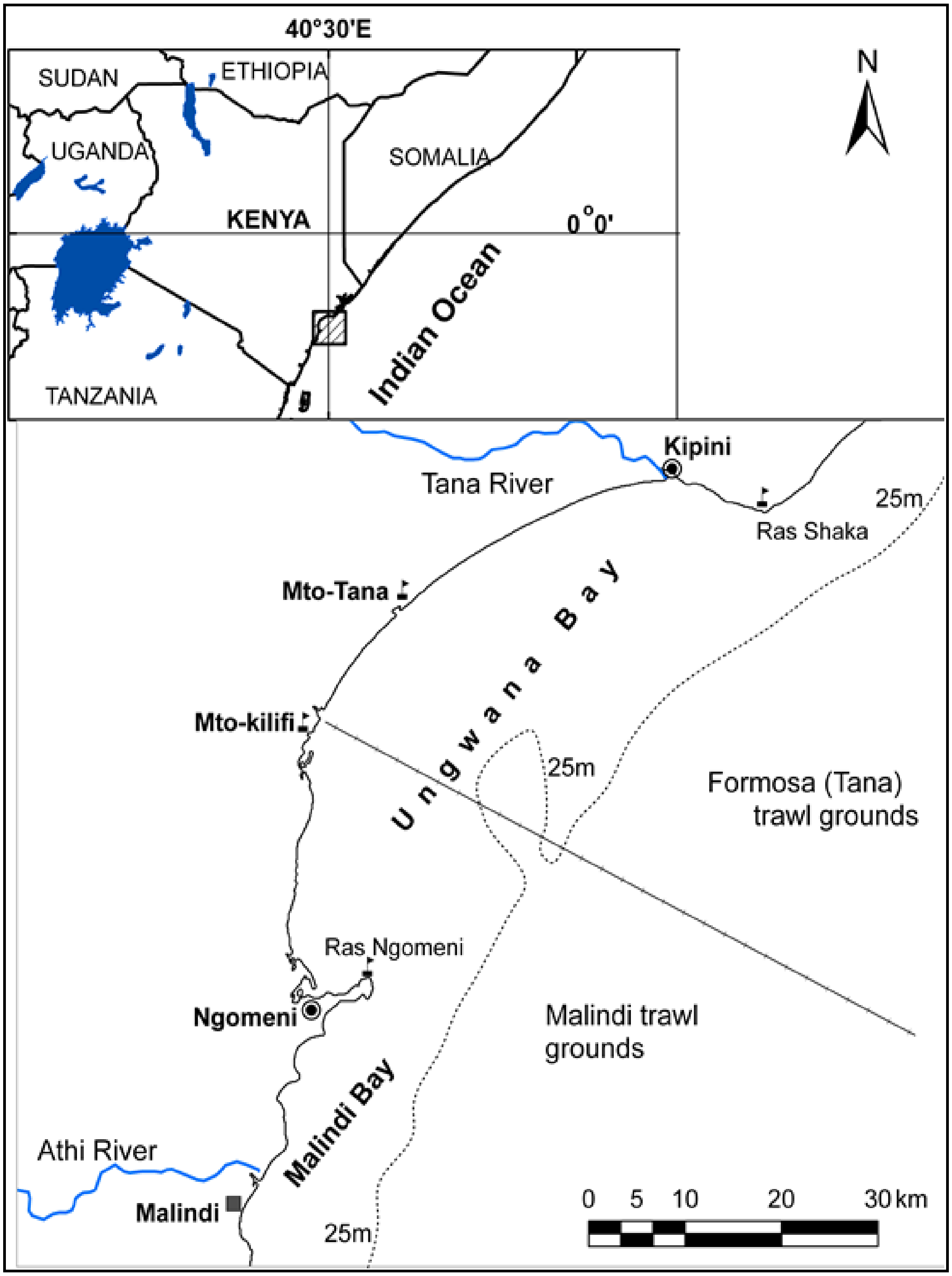

Figure 1: A map of Kenya's coastline showing the location of the Malindi-Ungwana Bay. (Source: Munga *et al.,* 2012)

Highest salinities occur during NEM when air temperatures and solar insolation are high and rainfall and discharge low (MClanahan, 1988). Local runoff is greatest during the SEM season. The total effect is such that the SE monsoon has the greatest influx of freshwater and terrestrial nutrients into the bay. The monsoon seasonality affects chemical and biological processes along the coast. For example, there are higher surface water chlorophyll concentrations and itchthyoplankton productivity during the SEM season (Kaunda-Arara *et al.*, 2009). Malindi-Ungwana Bay area has over the years been a rich fishing ground supporting the artisanal, commercial and semi-industrial fisheries. It has good trawling grounds in the shallow waters less than 30 m deep (Matagyera, 1984; Mwatha, 2005). Semi-industrial prawn trawling has been going on in the area for about 30 years but was banned from 2006 to 2011(Ochiewo, 2006).

3.2 Data sources used in the study

The study used a number of secondary data sets in addition to on-project data as described below:

3.2.1 Time series data on commercial prawn catches

Long-term series of prawn landing data sets extending from 1985-2010 were obtained from the State Department of Fisheries statistical records. The fisheries departmental staff record data on fisheries landings at designated beaches along the coastline. The data is categorized into fish and non-fish groups (mostly families) and recorded as daily landings (in Kg). This data was used to analyze the long-term trends in prawn catches from the Malindi-Ungwana Bay. Data on prawns recorded at the landing sites included; i) total numbers and weights, ii) type and size of fishing gears, iii) number of fishers per fishing craft, and iv) landing site and fishing grounds.

3.2.2 River discharge data sets

The River discharge data sets were obtained from the Ministry of Environment, Water and Natural Resources. Data included; monthly rainfall amounts for various areas in the Malindi–Ungwana Bay region from 1985 - 2010. This data were used to correlate discharge from rivers Tana and Sabaki and rainfall amounts with prawn catches in the bay in order to determine possible environmental drivers of prawn catches.

3.2.3 Prawn Fishery artisanal data sets from 2008-2012

Prawn fishery artisanal data sets extending from 2008-2011 were collected from various landing beaches in the Malindi–Ungwana Bay by technical staff of the Kenya Marine and Fisheries Research Institute (KMFRI). The beaches included: Marereni, Kipini, Malindi and Kanagoni along the Malindi-Ungwana Bay. The beaches were monitored continuously for two weeks after every 3 months for a period of 4 years (2008-2011). The data collected at the beaches included; fishing gear, type of vessel used, total weight by species (Kg) of prawns landed, numbers of each species landed and, carapace length (mm) of individual specimens. Additional data was collected on the project at the same sites during 2012 following the same methodology in order to provide a 5 years (2008-2012) time series of data. The data set was used for; number, diversity and biomass size-spectra analyses (see section 3.3).

3.2.4 Crustacean trawl surveys and fish by-catch data sets

Data on crustacean and fish by-catch were obtained from the Kenya Coast Development Project (KCDP) trawl survey conducted in December 2009 and, the South West Indian Ocean Fisheries Project (SWIOFP) shallow water crustacean trawl surveys conducted in the bay between 26th January to 2nd February and again from 24th May to 30th May 2011 to cover the NEM and SEM seasons, respectively. The trawl surveys were carried out by the commercial trawler FV Vega.

The aim of the surveys was to assess shallow water crustacean resources available in the Malindi-Ungwana Bay, as well as the by-catch levels associated with trawling. A total of 60 trawls were carried out by the surveys in the bay along transects stratified according to four depth strata as: 0-10 m (Zone 1), 10-20 m (Zone 2), 20-40 m (Zone 3) and 40-100 m (Zone 4) (Kimani *et al.,* 2011). Additional data on fish by-catch were also obtained from the demersal trawl survey that was conducted from 31st October - 14th November 2012 (NEM season) by the SWIOFP following the same protocol as well as the vessel and gear as the crustacean surveys. The fish caught during this survey were similar in composition to fish by-catch in the crustacean surveys as the trawl nets used were the same (Kaunda-Arara *et al.,* 2012). The data of the crustacean trawl surveys were used to develop ecological indicators used to assess the status of the prawn fishery. The data on fish by-catch of the crustacean surveys and fish catches of the demersal fish surveys were used to perform biomass-trophic level spectra analysis and construct K-dominance curves (see section 3.3) as biodiversity indicators for the bay.

3.3 DATA ANALYSIS

3.3.1 Trends in historical landings of prawns in the Malindi-Ungwana Bay

A locally-weighted scatter plot smoother (LOWESS) (Cleveland, 1979) was used to fit smoothed trend lines to the full data series of prawn catches (1985 to 2010) using the MINITAB package. The LOWESS is based on a weighted least squares algorithm that gives local weights the most influence while minimizing the effects of outliers (Cleveland, 1979). A smoothness parameter of f = 0.2 was found to adequately smooth the data without distorting the main temporal patterns. An autoregressive integrated moving average (ARIMA) model (Box and Jenkins, 1976) was then used to forecast landings of prawns from Malindi-Ungwana Bay for the next decade (2010–2020). An ARIMA (111) model that integrates first order autoregressive (AR) and moving average (MA) model parameters with first differencing of the annual catches (O'Donovan, 1983; Rothschild, 1996) was used to forecast catches as:

$$y_t = \theta_0 + \phi_1 y_{t-1} + a_t - \theta\, a_{t-1}$$

where, y_t is the first difference of the catches at time t, θ and ϕ model constants for AR and MA parameters, respectively, and "a_t" is a random error term. The model assumes stationary and homogeneity of means and variances, respectively. The means and variances of the output series (catches) were made stationary and homogeneous by first differencing and $\log_e$ transformation of the data, respectively.

3.3.2 Relationship between prawn catches, river discharge and rainfall in the Malindi- Ungwana Bay

Based on annual rainfall and discharge data obtained from the Ministry of Environment, Water and Natural Resources, a stepwise multiple regression analysis was used to determine the environmental variables that explain the greatest variability in prawn catches. The regression model used was:

$$C = \text{constant} + c_1R + c_2D$$

Where C is the catch (tonnes yr^{-1}), R is the annual rainfall (mm) and D is discharge (m^3s^{-1}) into the bay, c_1 and c_2 are coefficients of the independent variables.

3.3.3 Community Trend Indicators

The biological basis for using changes in numbers, diversity and biomass to changes in size composition resulting from fishing is that fishing selectively harvests larger individuals first, while simultaneously increasing the mortality rate for all sizes taken by fishing gear (Rice, 2000). As a result, communities shift to smaller sized fish, and this occurs within and across species and therefore the intercept of the size spectrum (of numbers, diversity and biomass) increases, as does the steepness of the slope of the spectrum (Rice, 2000).

The size-spectra analysis (change of an attribute with size) is based on tropho-dynamic transfer efficiencies, in that although the richness and relative abundances of species in a size series of samples is highly variable, the biomass and numbers of individuals (pooled across all species) decreases log-linearly with size (Rice and Gislason, 1996). The general formula for this log-linear relationship between size (x) and biomass or number (y) within a community is:

$$\ln (y) = a^* \ln (x) + b$$

where, a = slope and b = Intercept of the model. However, the data were fitted to the semi-log relationship as:

$$\ln (y) = a + bx,$$

This relationship was found to provide a better fit to the data than the log-log linear relationship.

From theory, differences in productivity should appear as differences in intercept, whereas differences in transfer efficiencies and mortality rates should appear as temporal or spatial differences in slopes of the above relationships (Thiebaux and Dickie, 1993; Sprules and Goyke, 1994). Steepening of the slope would indicate a decrease in the number of large fish in the population, an increase in the number of small fish, or both as an influence of fishing or other factors (Trenkel and Rochet, 2003).

For diversity-size spectra analysis; the Shannon–Wiener diversity index was used because it is more sensitive to the relative abundances of species (more desirable as a fishing effect) than to the richness of species (Magurran, 1988). For a typical community, fewer organisms are present in larger size classes, this necessarily reduces the difference in abundance between rarest and commonest species, and increases the probability that a species will have zero abundance in larger size classes (Magurran, 1988). Both factors mentioned above cause diversity to decrease across size classes (Rice and Gislason, 1996). Number, biomass, and diversity-size spectra analysis were performed for artisanal landings data extending from 2008-2012. The Shannon-Wiener Diversity Index, H, was calculated using the following equation (Magurran, 1988):

$$H = - \sum P_i (\ln P_i)$$

Where, P_i is the proportion of each species in the samples.

Diversity size-spectra were then constructed by plotting H against the mid-length of length groups. A high value of diversity will occur with a high number of equally abundant species, a low value with a low number of species and a species composition dominated by one or a few species. This means that the overall shape of the diversity spectrum is determined both by the number of species present in each size group as well as by their relative abundance (Pope and Knights, 1982). However, Changes in natural mortality will have an influence on the change in the diversity spectrum (Gislason and Rice, 1998).

3.3.4 Biodiversity and Conservation-based indicators

3.3.4.1 Biomass- trophic level spectra

Cummulative relative biomass trophic level spectra (BTLS) analysis was performed on fish by-catch and fish catch data from the Malindi-Ungwana Bay crustacean and demersal fish trawl surveys, respectively. BTLS analysis was performed on data from two years (2011 and 2012) and changes in the shape of the annual biomass-trophic level spectrum during the two years tested using Kolmagorof-Smirnov test (Zar, 1997). Fish species by-catch and catch data (MT) collected by the SWIOFP during the 14 days shallow water crustacean and demersal trawl surveys, during NEM seasons (January–February, 2011; and November 2012, respectively) were used for the analysis.

Because diet composition data were not available, trophic level (TL) information contained in FishBase (Froese and Pauly, 2003) were used to analyse for annual changes in distribution of biomass across trophic levels as an indicator of multifactor effects in fish community structure and trophodynamics (Cruz-Escalona *et al.,* 2000). Only fish species with relative abundance of ≥ 0.2% were used in the BTLS analysis. Fish biomass-TL spectra were then constructed by combing species by 0.5 TL intervals for each year. Mean TL values were compared between years using Mann-Whitney U-test while, Wilcoxon signed-rank test was used to test the statistical significance of a change (increase or decrease) of each species biomass between 2011 and 2012.

3.3.4.1 K-dominance curves

K-dominance curves (Lambshead *et al.,* 1983) were used to compare changes in fish species diversity in the bay between 2011 and 2012. Fish by-catch (from crustacean survey) and fish catches (from demersal fish survey) of the Malindi-Ungwana Bay were used to plot the K-dominance curves. In constructing the curves, the species were ranked in order of dominance on the x-axis (Logarithmic scale) with percentage dominance (in proportion by number) on the y-axis (cumulative scale) for 2011 and 2012. In the plot, the numbers of individuals of each species were sorted in descending order, and the proportion of the total number of individuals for each species is then plotted on the log-scale against the species rank. The shape of the rank/abundance plot of the assemblages provided an indication of dominance or evenness (Bianchi *et al.,* 2000), For example, steep plots signify assemblages with high dominance (low diversity) and shallower slopes indicate higher evenness or low dominance (high diversity).

3.3.4.2 Application of Indiseas-related ecological and biodiversity status indicators

IndiSeas (Indicators of the seas) is a multi-institutional collaborative program endorsed by IOC-UNESCO and the European Network of Excellence, EUR-OCEANS (Website: www.indiseas.org). It aims at using ecosystem indicators to evaluate the status of the world's exploited marine ecosystems in support of an ecosystem approach to fisheries. Key issues addressed in the Indiseas program relate to the selection and integration of multi-disciplinary indicators including; climate, biodiversity and human dimension indicators, and the development of data and model-based methods to test the performance of ecosystem indicators in providing support for fisheries management (Shin *et al.*, 2010).

This study derived estimates of some of the indicators employed by Indiseas project (see www.indiseas.org) in order to assess the effects of prawn fishing on the ecological and biodiversity status of the Malindi–Ungwana Bay Ecosystem. The indicators (Table 1) are formulated positively so that a low value means a high impact of fishing while a high value indicate a low impact of fishing (derived as in Appendix 1). Similarly, an increase of the indicator (positive trends) means an improving state, whereas a decrease means a deteriorating state (Shin *et al.,* 2010). The indicators were analysed and presented graphically using polar diagrams. The selected ecological and biodiversity indicators (Table 1) were used to evaluate the current state of the bay (S), and the recent trends (T) in the indicators. Data for deriving the Indiseas indicators were obtained from KCDP prawn trawl survey in 2009 and the SWIOFP crustacean prawn survey in 2011.

Polar diagram, a tool for synthesizing multi-factor effects on ecosystem status from a suite of ecosystem indicators (Shin *et al.*, 2010) was used to present the ecosystem state indicators averaged over two years (2009 and 2011) for comparison. Each pie of the diagram corresponds to a selected Indiseas indicator. Each indicator is scaled between a minimum value (centre of the polar diagram) and a maximum value. The purpose of the minimum-maximum boundaries is to scale the indicators for graphical representation, make concentration of strengths and weakness visible, and allow for comparisons across ecosystems or years for support of decision-making in ecosystem approach to fisheries management. The polar diagram was generated on excel spread sheet using KCDP (2009) and SWIOFP (2011) shallow water crustacean trawl surveys data sets.

Table 1: List of selected indiseas-based ecological indicators used to evaluate the ecosystem status of the Malindi-Ungwana Bay (L: length (cm), s: species, N: abundance, B: biomass, Y: catch, TL: trophic level. (Adopted from Shin *et al.*, 2010)

Indicators	Headline label	Calculation, units	Used for State (S), Trend (T) evaluation
Total Biomass of surveyed species	Biomass	B(tons)	T
TL landings	Trophic level	$TL_{land} = \sum TL_s Y_s / Y$	S,T
Total landings Weight of fish/ sub-sample weight of fish	Total landings	Raising Factor = Total catch	S, T
Total biomass of retained species	$B_{retained}$	$B_{retained}$ (tons)	T
1/Landings/Biomass	Inverse fishing pressure	B/Y retained species	T
Mean length of fish in the Community	Fish size	$L = \sum L_i/N$ (cm)	S,T

CHAPTER FOUR

RESULTS

4.1 Trends in prawn landings

The long-term total landings of prawns averaged 395.56 (± 168.58) tonnes yr^{-1} for the period 1985-2010. Annual prawn catches peaked in 1988(535 tonnes yr^{-1}), 1997 (774.2 tonnes yr^{-1}) and in 2000 (686.92 tonnes yr^{-1}) (Fig. 2). Following the peak in 2000, the catches dropped by 48.5 % to 383.63MT in 2002 and remained relatively stable during 2002-2003 when it averaged 383.56 (Fig. 2). Following this period of stability, annual landings declined further to 175.26 tonnes yr^{-1} 2005) and 153.04 tonnes yr^{-1} in 2008 before a slight increase to 217.89 tonnes yr^{-1}in 2010.

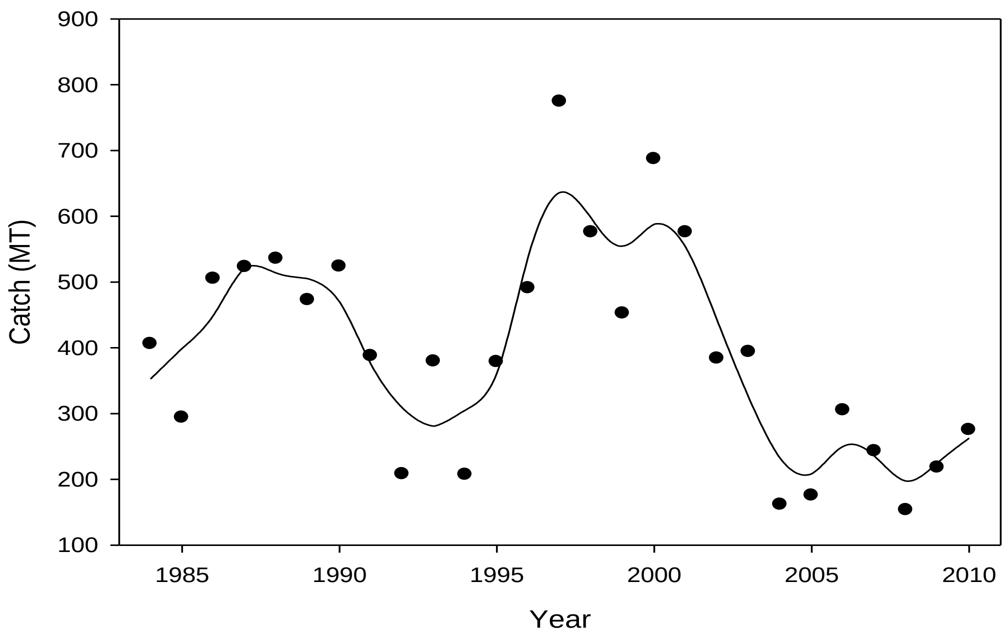

Figure 2: Long-term trends in annual landings (tonnes yr^{-1}) of prawns from 1985 to 2010 in the Malindi-Ungwana Bay, Kenya. Continuous line show the LOWESS trend fit to landings, while (•) are the actual landings.

Based on ARIMA (111) model, the annual prawn catches were used to forecast total landings for the next decade (2010-2020) following the landings last recorded in 2010. The forecast predicted a consistent decline in prawn catches in the next 10 years (2010-2020) under the current management regime (Fig. 3). The 95% prediction limits however, indicated low confidence in the forecasts during the next 10 years. The model generated for the 10 years forecast was:

$$y_t = -3.575 + 0.539y_{t-1} + a_t + 0.95a_{t-1}$$

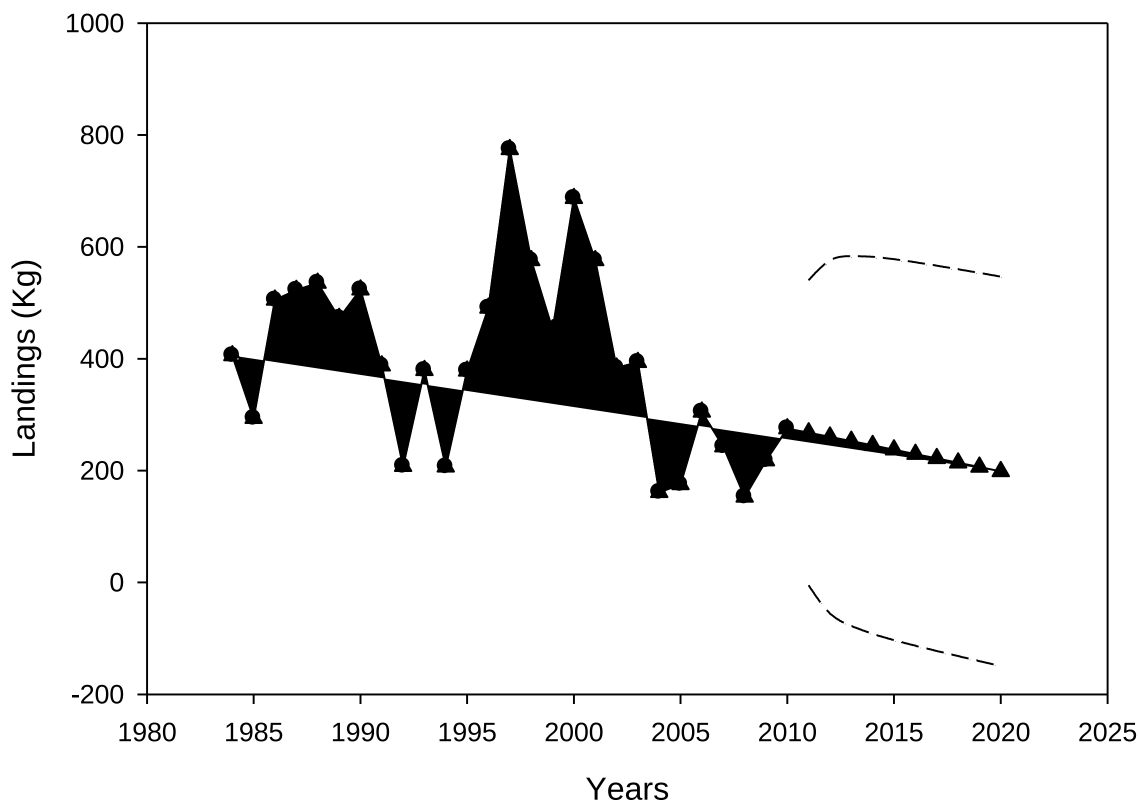

Figure 3: Observed (•) and Forecast landings of prawns in the Malindi-Ungwana Bay for the next ten years (2010–2020). Forecasts (triangles) show catches will decline steadily with time, the upper and lower dashed lines are the corresponding 95% confidence intervals.

4.2 Environmental drivers

There was a general positive correlation between prawn catches and discharge rate (Fig. 4) and rainfall amount (Fig. 5) within the bay, a decrease in discharge rate and rainfall amount resulted to a decrease in catches. A forward selection stepwise multiple regression analysis showed the dependent variable (catches) can be predicted from a linear combination of the independent variables (rainfall amount and discharge rate). 73% of the variability in annual catches (C, tonnes yr^{-1}) can be explained by the annual discharge (D, $p = 0.02$) and rainfall (R, $p = 0.02$) into the bay as:

$$\mathbf{C = 41.24 + 0.255D + 3.04R, \quad r^2 = 0.73}$$

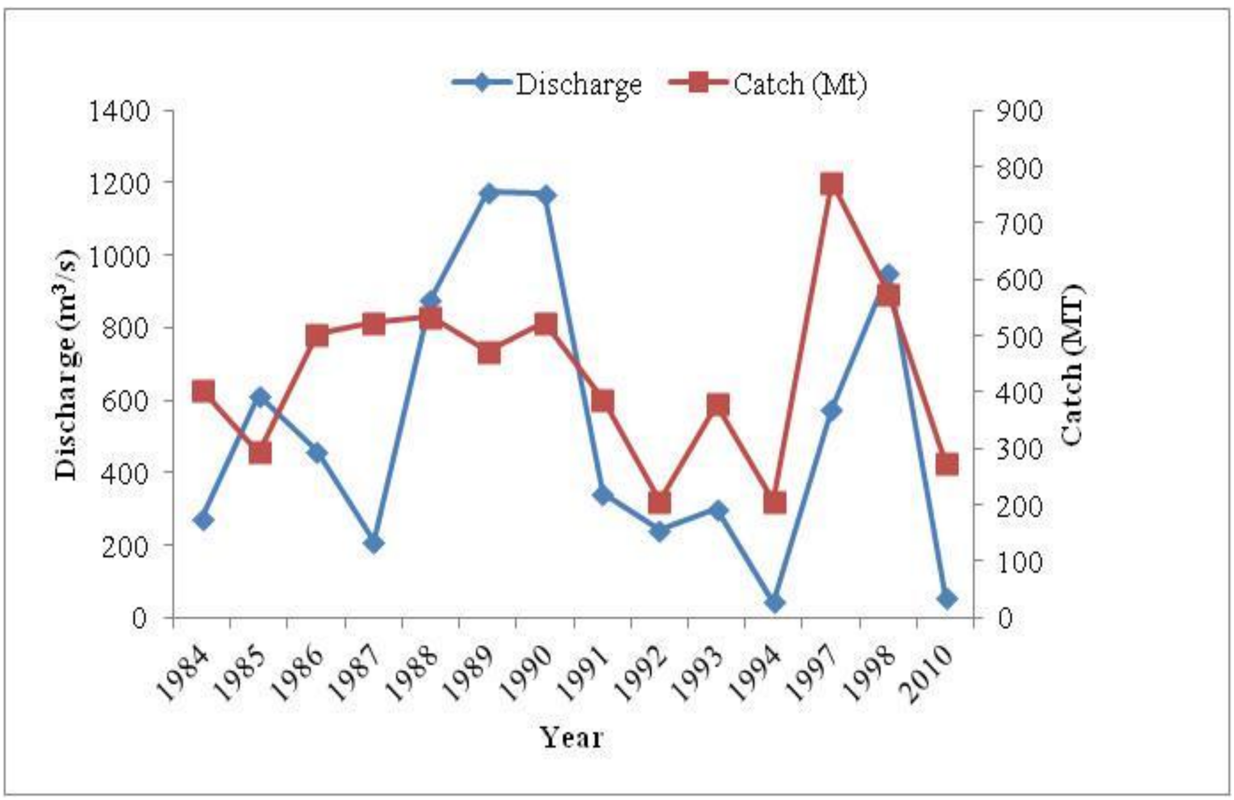

Figure 4: Annual variability in prawn catches in relation to discharge levels in the Malindi-Ungwana Bay, Kenya.

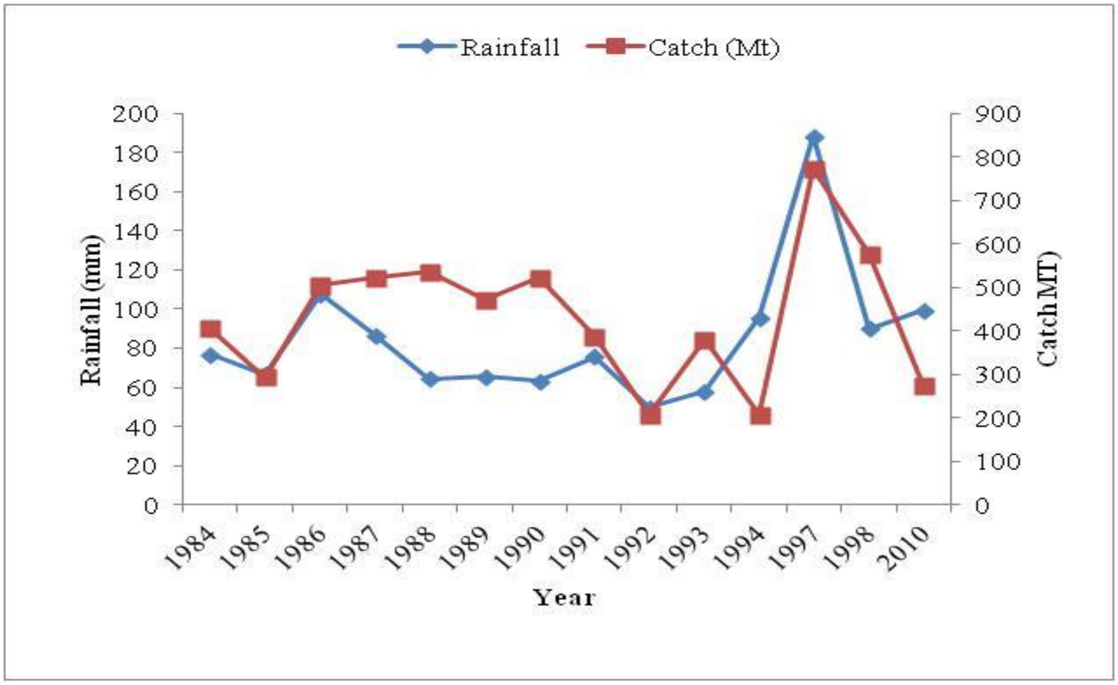

Figure 5: Annual variability in prawn catches in relation to rainfall amount in the Malindi-Ungwana Bay, Kenya.

4.3 Community trend indicators

4.3.1 Number–size spectra analysis

The numbers of crustaceans in the different size classes for the artisanal landings in the period 2008-2012 tended to decrease with size except for the 0-10 mm size class in all the years (Fig. 6). The numbers per size class tended to zero for larger size classes indicating the presence of a “floor effect” which may contribute to difference in transfer efficiencies and mortality rate between cohorts. This effect meant it would be increasingly difficult to detect continued change in slope for larger sizes. The slopes for number size- spectra were negative for all years (Fig. 6).

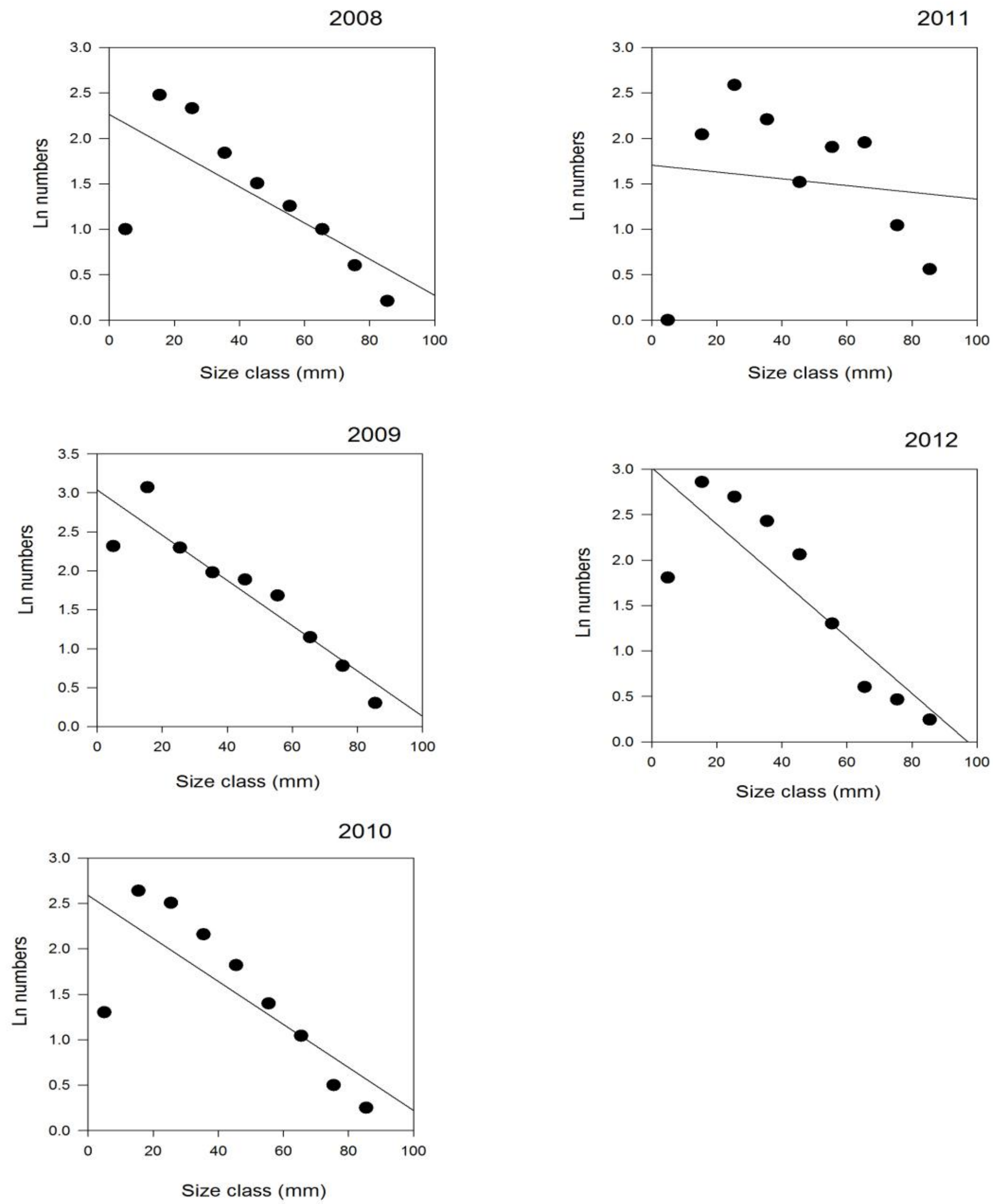

Figure 6: Ln (numbers) against 10 mm size-class for artisanal prawn fishery catch data from 2008-2012. Lines are linear model fits to number size-spectrum.

The linear model of Ln (numbers) against size classes, fitted the annual number-size spectrum very well for 2009 (r^2 = 0.875), moderately for 2010-2012 (r^2 = 0.482, 0.507 and 0.493, respectively) with poor fit for 2008 (r^2 = 0.353) (Table 2). All the intercepts and slopes (Table 2) for the annual number-size spectra analysis were significantly different from zero indicating inter annual difference in productivity and fishing mortality of the stocks.

Table 2: Statistics of fit of linear models to annual number-size spectra using artisanal prawn fishery catch data from 2008-2012.

	Model fit			**Slope**			**Intercept**		
Year	**F**	**P**	**r^2**	**Est**	**S.e**	**T**	**Est**	**s.e.**	**t**
2008	3.280	0.120	0.353	-0.0162	0.00892	-1.811	2.154	0.415	5.192
2009	49.024	0.001	0.875	-0.0291	0.00415	-7.002	3.038	0.217	13.988
2010	5.581	0.056	0.482	-0.0245	0.0104	-2.362	2.597	0.482	5.391
2011	5.137	0.073	0.507	-0.0163	0.00720	-2.266	2.636	0.358	7.366
2012	4.854	0.079	0.493	-0.0260	0.0118	-2.203	2.885	0.480	6.005

A regression of annual slope on year was significant over the five years ($r^2 = 0.875$, $F_{(1, 4)} = 49.024$, p = < 0.001, Fig.7). The slopes of the number-size spectra were more variable but negative in all the years (Fig.7, Table 2) indicating general increase of mortality of the panaeid shrimps over the five year period (2008-2012). However, the increase of the intercepts of the number-size spectra was marginally significant ($r^2= 0.56$, $F_{(1, 4)} = 1.695$, $p = 0.05$) over the five year period with the highest value of 3.038 in 2009 (Table. 2 and Fig.7). This pattern of increase in intercepts suggests an overall increase in productivity of the community with time. However, the correlation between intercept and slopes may confound this interpretation.

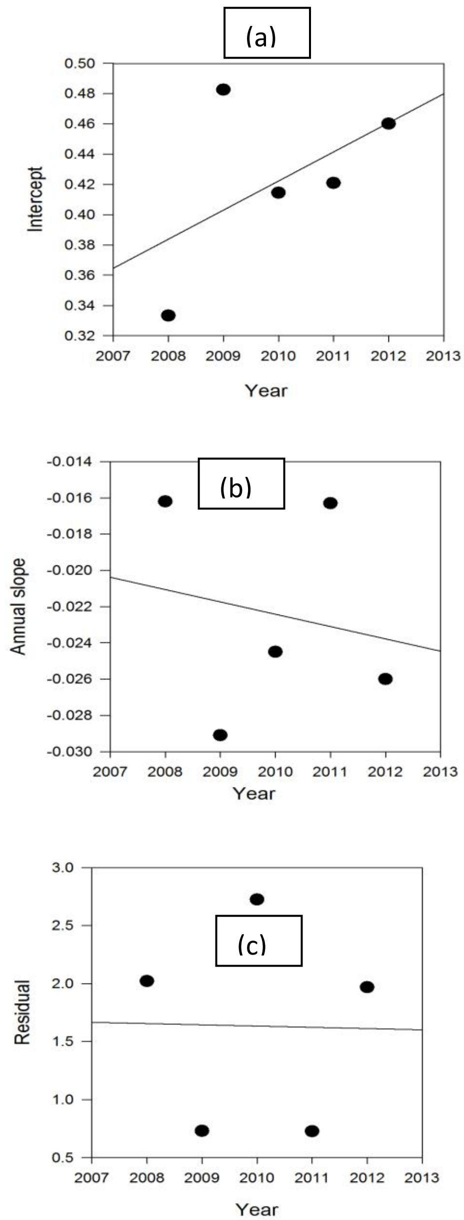

Figure 7: Regression of (a) intercepts, (b) slopes, and (c) residuals of the annual number-size spectra analysed from prawn artisanal landings data sets.

4.3.2 Biomass size-spectra analysis

There was a general increase in biomass of the penaeid shrimps with increase in size classes during 2008, 2009 and 2012 (Fig. 8). The distribution of biomass tended to be low within small size classes for the years; 2008, 2009 and 2010, while in 2011 and 2012 higher biomass occurred in the small and large size-classes with a general decrease in the median sizes-classes (Fig. 8). A regression of annual slope on year was significant over the five years ($r^2 = 0.74$, $F_{(1, 4)} = 20.090$, $p < 0.005$, Fig. 9). The slopes of the biomass-size spectra were variable but negative in 2010 (Fig. 9, Table 3). However, the intercepts of the biomass-size spectra increased but negatively ($r^2 = 0.64$, $F_{(1, 4)} = 12.165$, $p = 0.01$) over the five year period with the highest value of -0.994 in 2009 (Table. 3 and Fig. 9). This pattern of negative increase in intercepts suggests a significant effect of fishing on size composition of exploited fish assemblage.

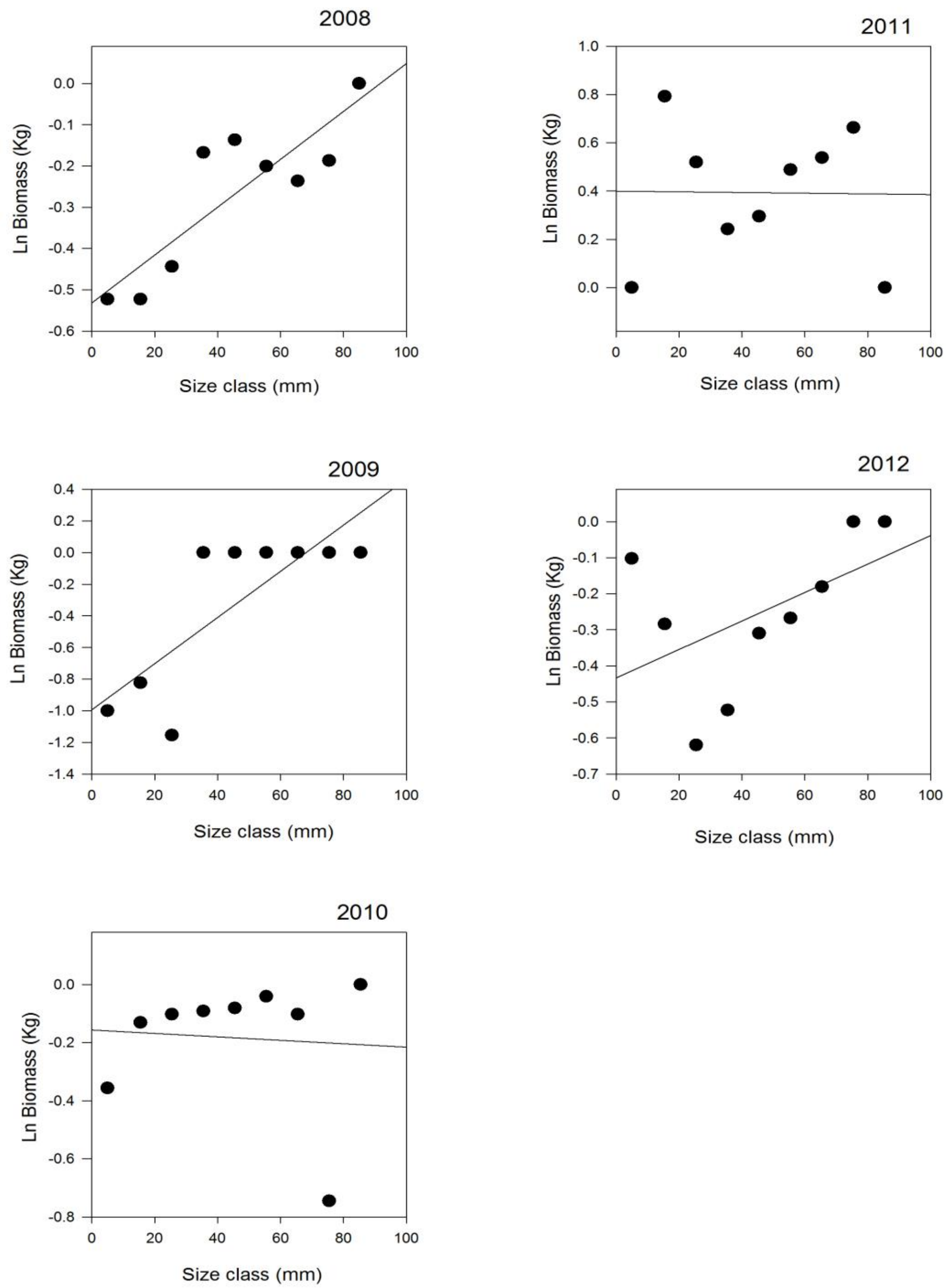

Figure 8: Ln (Biomass, Kg) against 10 mm size-class for artisanal prawn fishery catch data from 2008-2012. Lines are linear model fits to biomass size- spectra.

Biomass-size spectra models were significant during 2008 and 2009 ($p < 0.05$, Table 3) but were not significant for 2010 to 2012 ($p > 0.05$, Table 3). The linear model of biomass against size classes fit the biomass spectrum highly for 2008 ($r^2 = 0.742$) and 2009 ($r^2 = 0.635$), with a poor fit for 2010-2012 ($r^2 = 0.05$, 0.004 and 0.257, respectively) (Table 3).

Table 3: Statistics of fit of linear models to Biomass-size using artisanal prawn fishery catch data from 2008-2012.

	Model Fit			**Slope**			**Intercept**		
Year	**F**	**P**	**r^2**	**Est**	**S.e**	**T**	**Est**	**s.e.**	**t**
2008	20.090	0.003	0.742	0.006	0.001	4.482	-0.532	0.068	-7.873
2009	12.165	0.010	0.635	0.015	0.004	3.488	-0.994	0.219	4.542
2010	0.034	0.859	0.005	-0.001	0.003	-0.185	-0.157	0.167	-0.936
2011	0.029	0.959	0.004	0.0002	0.003	0.054	-0.261	0.154	-1.694
2012	2.417	0.164	0.257	0.004	0.003	1.555	-0.434	0.133	-3.261

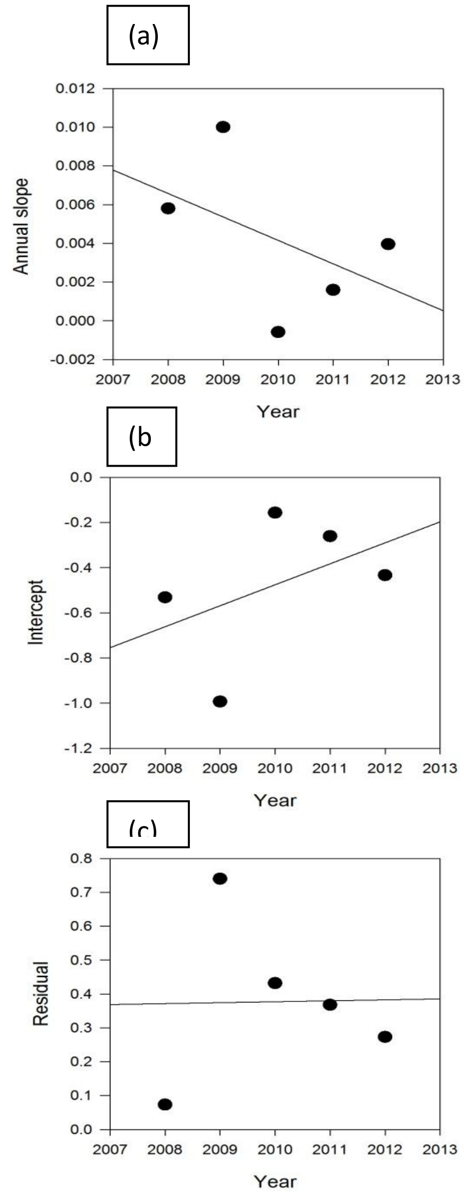

Figure 9: Regression of slopes (a), intercepts (b), and residuals (c) of the annual biomass-size spectra analysed from prawn artisanal landings data sets.

4.3.3 Diversity - size spectra analysis

The Shannon-Weiner Diversity metric (H) peaked in the 10-20 mm size-class of the penaeid shrimps in all the years (Fig. 10). The general decline in diversity as length of the shrimps increased beyond 30 mm was similar between the years. The diversity of the crustaceans within the 0-20 mm size class was consistently lower in all the years producing a consistent arch in the diversity-size distributions. There was a significant negative linear relationship of the diversity-size spectra for all the years except for 2011 (Fig. 10, Table 4). The curvilinearity observed at small size classes (Fig. 10) may be an artifact of differential catchability of various species below 10 mm. The slope of the diversity-size spectra did not change greatly over the five years (Table 4) indicating a near equal change in relative abundance of the species between the years. The dome-shaped pattern in the diversity spectra observed for nearly all the years (Fig. 10) indicated maximum diversity in the 0-20 mm size class. The annual slope of the diversity was significantly for the years except for 2011 (Table 4).

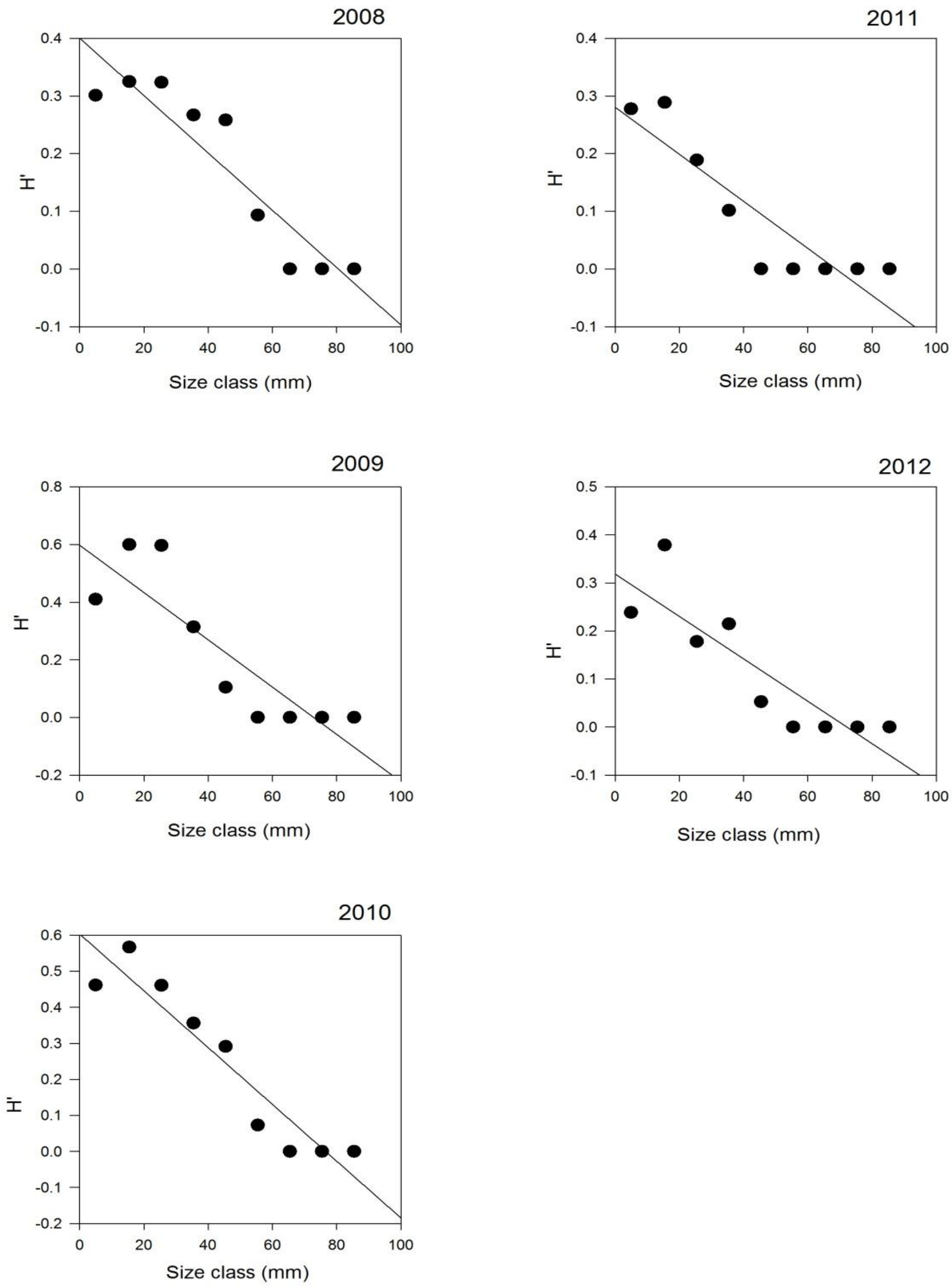

Figure 10: Shannon-Weiner Diversity (H) per 10 mm size-class derived from artisanal prawn fishery catch data from 2008-2012. Line is linear model fit to diversity-size spectrum.

Table 4: Statistics of fit of linear models to the annual diversity-size spectra derived from artisanal prawn fishery catch data from 2008 - 2012.

	Model Fit			Slope			Intercept		
Year	**F**	**P**	**r^2**	**Est**	**S.e**	**t**	**Est**	**s.e.**	**t**
2008	29.474	0.002	0.831	-0.00524	0.000966	-5.429	0.408	0.0449	9.079
2009	21.681	0.002	0.756	-0.00820	0.00176	-4.656	0.597	0.0921	6.486
2010	50.159	<0.001	0.893	-0.00863	0.00122	-7.082	0.625	0.0567	11.025
2011	12.371	0.072	0.861	-0.00627	0.00178	-3.517	0.405	0.0579	6.998
2012	20.101	0.004	0.770	-0.00504	0.00112	-4.483	0.336	0.0523	6.435

4.4 Biodiversity and conservation - based indicators

4.4.1 Biomass-Trophic Level Spectra

According to the Wilcoxon signed-rank test performed on 299 fish species recorded during the crustacean and demersal fish surveys, the biomass of; *Bothus mancus, Leiognathus elongatus, Pellona ditchela, Secutor insidiator, Galeichthys feliceps, Trachinocephalus myops and Upeneus taeniopterus* decreased significantly between 2011 and 2012 (W = 28, p = 0.022, Fig.11). However, the biomass of *Leiognathus equula, Leiognathus daura, Gerres oyena, Johnius amblycephalus, Otolith ruber, Pomadysis maculatum* and *Trichiurus lepturus* increased significantly between 2011 and 2012 (W = 28 p = 0.022, Fig.11). Among the species that showed no significant change between the years included; *Gerres filamentosus, Platycephalus crocodiles, Pelates* *quadrimaculatus, Upeneus sulphureus and Poecilopseta natalensis*. The species that increased in biomass between 2011 and 2012 belonged to the Trophic level (TL) range of 2.5 and 4.5. In contrast, those that decreased in biomass between the years belonged to TL range of 2.8 and 4.4. Differences in mean TL between 2011(0.745.76 ± 0.27) and 2012(0.3 ± 0.04) were not significant (Mann-Whitney, $p = 0.038$). There was higher catches in trophic levels 3.5 and 4.0 during 2011, and higher catches in trophic levels 2.5 and 4.5 in 2012 (Fig. 12). The lowest annual biomasses (2.8 kg) and (2.3 kg) were caught in trophic level 4.0 during 2011 and 2012, respectively (Fig. 12).

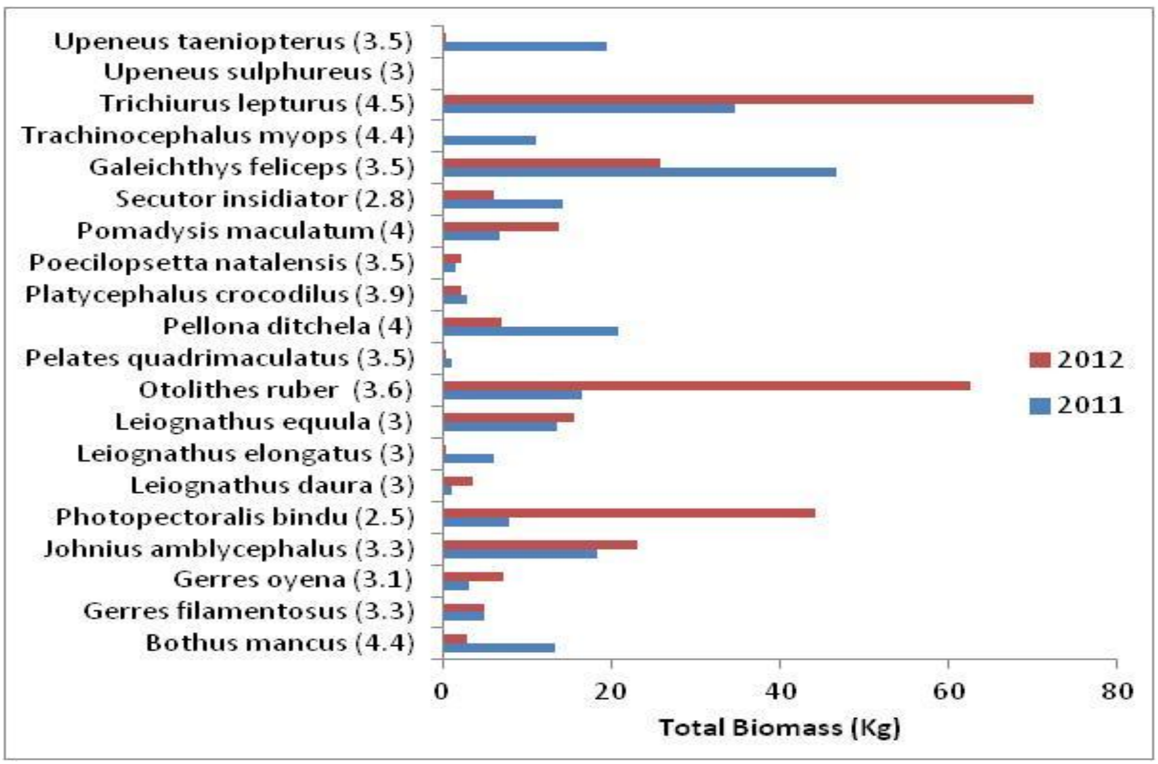

Figure 11: Variation of total fish biomass by year for species with biomass ≥ 0.1 kg (numbers in parenthesis indicate trophic level). Data obtained from crustacean trawls in 2011 and demersal fish trawls in 2012 within the Malindi-Ungwana Bay.

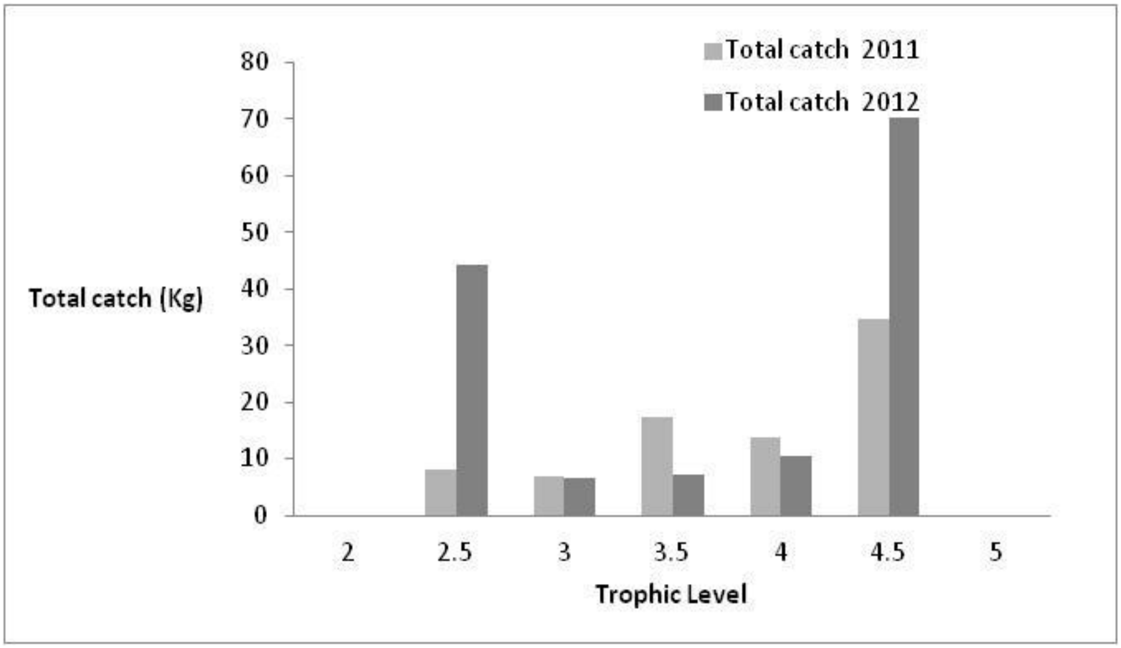

Figure 12: Biomass trophic level spectra (BTLS) for fish catches in the Malindi-Ungwana Bay during 2011 and 2012.

The Cumulative relative biomass trophic level spectra (BTLS) showed highly significant (K-S $p<0.005$) differences in shape of BTLS for 2011 and 2012 (Fig.13a). Pooled data showed the same trend this could be due to increased by-catch of the penaeid shrimp fishery or change in the environmental condition in Malindi-Ungwana Bay (Fig.13b). The K-dominance curves (Fig. 14) analyzed to determine possible changes in diversity of fish catches between 2011 and 2012 showed similar curvature during 2011 and 2012, indicating lack of significant shift in fish species diversity between the years. High relative abundance within low number of species observed in both years 2011 and 2012 indicated minimal inter-annual change in species abundance and number of individuals per species (evenness) in the bay.

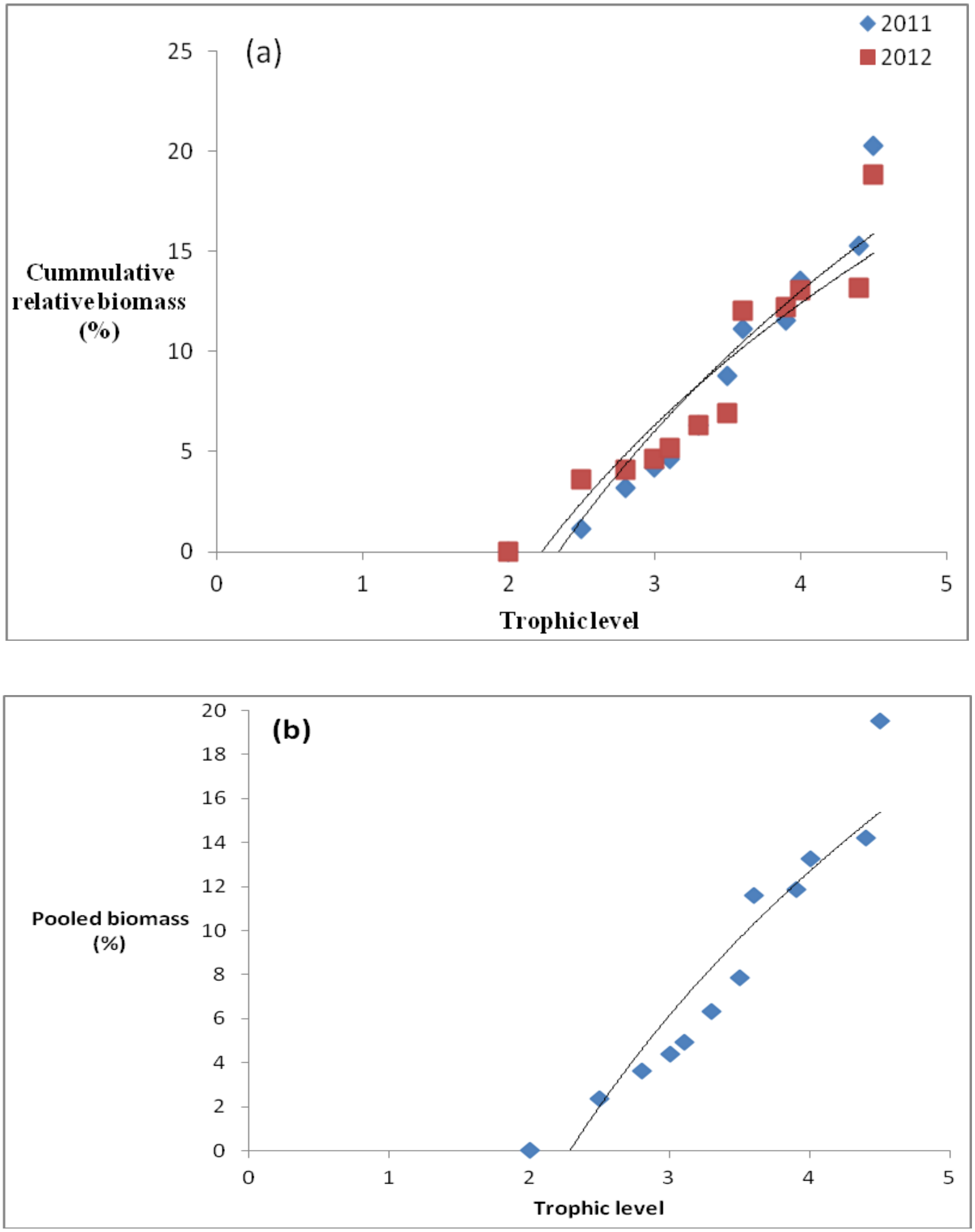

Figure 13: (a) Cumulative relative biomass trophic level spectra, (b) Pooled relative biomass trophic level spectrum for 2011 and 2012 in the Malindi-Ungwana Bay.

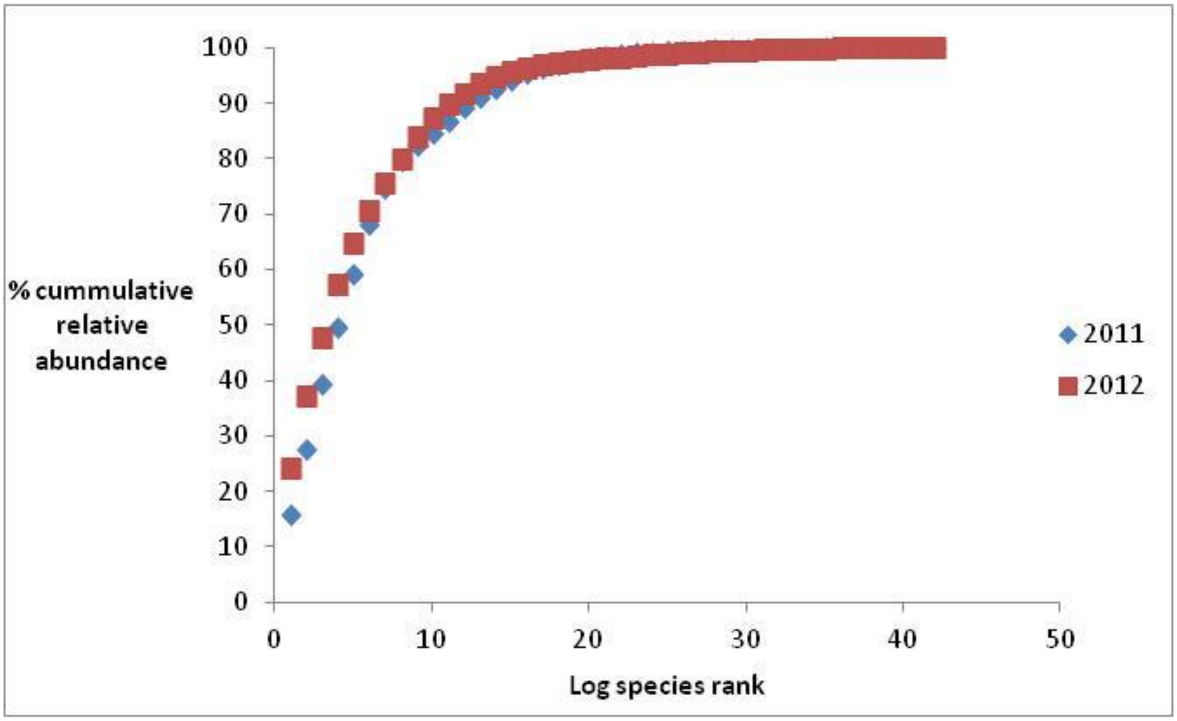

Figure 14: K-dominace curves showing percentage relative abundance of fish species in the Malindi Ungwana Bay for 2011 and 2012.

4.4.2 Indiseas-based ecosystem indicator analysis

Based on the polar diagram (Fig.15) analyzed from the derived Indiseas-based indicators (Table 1), the concentric lines indicate a gradation of the percentage values (0-80%) of the selected indicators on a scale of minimum to maximum and used to scale the derived values for 2009 and 2011. There was a decrease of 8% in the Total Biomass of all the surveyed species of crustaceans between 2009 (8,221 tonnes) and 2011 (6,974 tonnes), while a minor decrease was observed in the Biomass of Retained Species of 2% between 2009 (6,032 tonnes) and 2011 (5,864 tonnes). The Mean Length of the shrimps species showed a large decrease of 38% between 2009 (38cm) and 2011 (17cm).

There was a minor decrease (2%) in Trophic Levels of landings from 3.8 in 2009 to 3.6 in 2011. Total Landings of all the species showed a major decrease of 44% between 2009 (7,218 tonnes) and 2011 (2,860 tonnes). There was a major increase in Inverse Fishing Pressure of 42% between 2009(836 tonnes) and 2011 (2,052 tonnes).

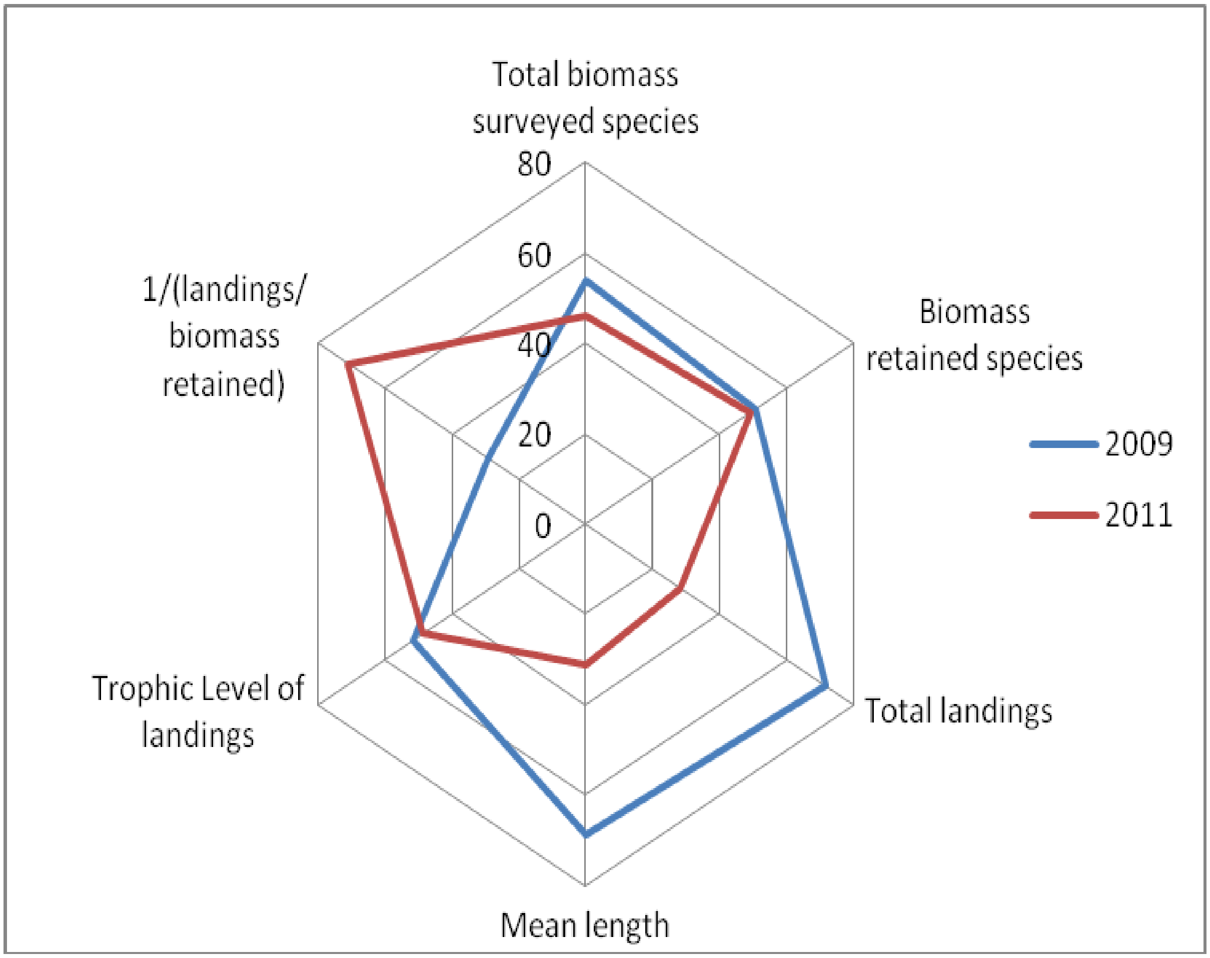

Figure. 15: Polar diagram showing variation of selected Indiseas-based indicators between 2009 and 2011 as a measure of ecosystem status of the Malindi-Ungwana Bay. Figures in the diagram represent the percentage values of the selected indicators used to scale the derived values.

5.0 DISCUSSION

Analysis of historical shrimp landing data indicate that landings from the Malindi-Ungwana Bay have declined over the past 10 years since the peak in 2000 and forecasts indicate that landings will continue to decline under the present management regime. The present management regime is based on the 2010 Prawn Fishery Management Plan. The cause of the trend could be attributed to increased fishing pressure, and perhaps environmental variability as indicated by the correlation between catches, discharge and rainfall. The conventional management strategies of the prawn fishery (e.g. closed seasons and monitoring of fishing gears) have not mitigated the conflict between resource users within the bay (Ochiewo, 2006) or the decline in catches. Resource managers from the State Department of Fisheries will need pragmatic and easy-to-interpret indicators in order to evaluate the state of the resources for management purposes.

This study examined patterns in size-spectra as indicators for ecological state of the bay. The results showed a change in the distribution of numbers across size groups, with larger size groups containing progressively less numbers over time. This effect could be attributed to increasing exploitation rates in the Malindi–Ungwana Bay (Fulanda *et al.*, 2009) affecting adult biomass and hence juvenile recruitment. The results indicate that the size structure of prawn population in the bay is affected by fishing. Slopes of size spectra of communities subject to intensive exploitation tend to become steeper (Gislason and Rice, 1998) similar to what was found in this study, indicating decreasing average length of the individuals caught. The overall trend of the number-size spectra is one of a reduction in large sized prawn species and a relative increase in small fish as suggested by the “floor effect” (*sensu* Gislason and Rice, 1998). Size spectra are believed to be a robust ecosystem metrics for examining the impact of fishing, where fishing causes mortality on larger individuals or species (Rice, 2000; Rochet and Trenkel, 2003). The resultant disproportionate removal of large individuals and theoretical increase in small individuals through prey release (Shin *et al.,* 2005) commonly results in steeper size spectra and lower midpoint height with increased fishing pressure (Dulvy *et al.,* 2004; Graham *et al.,* 2005), as observed in this study. The observed general increase in the intercept of the number size-spectra of the prawns in the bay suggests increased secondary productivity of the ecosystem (Gislason and Rice, 1998).

Biomass is an important parameter for evaluation of the state of a stock (Thurow, 1997). The study therefore adopted this indicator as it is useful for the estimation of recruitment and the effects of both fisheries and environmental factors on the ecosystem (Thurow, 1997). Strong, non-selective fishing can eliminate most of the biomass from an ecosystem, while a low fishing intensity has low impact, irrespective of selectivity. The combinations of fishing selectivity and intensity destabilize the biomass-size spectrum, that is, amplify temporal variations in the biomass flow (Benoît and Rochet, 2004). In this study, the observed general decline in height of the biomass size-spectrum and the decline in annual slopes, indicates decline in abundance of the exploited crustacean community in response to fishing intensity. This pattern may therefore be a useful indicator of the overall abundance-biomass effects of a fishery at the multispecies level (Jennings, 2005). The steepening of the slope in the annual biomass-size spectra can be interpreted as a result of a greater reduction in the abundance of larger shrimps, and not ecological release of small shrimp species from predation or competitive pressure as evident from lack of a arch in the distribution of the juvenile biomass-spectra.

Effects of predatory or competitive release might have been detected if more of the individuals from other species (e.g. Fish, etc) were included in the analysis as they contain most of the total biomass in food webs (López-Martínez *et al.,* 2002).

The diversity-size spectrum was another indicator adopted in this study and it appeared more curvilinear than the number-size spectra for prawn species. The number of species and individuals in the smaller size groups correspondingly raised the diversity in these groups and worsen the performance of linear models as seen in this study. Partial catchability of small-sized prawns would also lower diversity and contribute to a dome-shaped size spectrum (Rice and Gislason, 1996). However, the number–size spectrum was more linear for size groups >10 mm in most of the years. There were statistically significant changes in the linear component of the prawn populations in all the years studied. This produced a widening gap between the diversity of the smallest size groups and the decreasing diversity of the larger size groups. Trophodynamic (energy flow and food web interactions) causes of population regulation (Pope and Knights, 1982) could account for the pattern present in these analyses. The decline in diversity with increasing size of the prawns reflects the effects of removal of target species in the larger size range. This is likely as a result of effects of fishing on population structure as found in a number of studies (e.g. Peters, 1983; Platt, 1985; Beyer, 1989; Murawski and Idoine, 1992) than the effect of predation.

The results of this study, showed evidence of small changes in the trophic structure of the fish community of the bay as a result of trawling. There is a re-allocation of biomass from species characterized by intermediate trophic levels (3.1 - 3.5) to carnivorous (3.9 - 4.5) and herbivorous species (2.5 - 2.8). The shape of the BTLS was consistent between 2011 and 2012 perhaps mainly a result of the short time interval. Similarly, the K-dominance curves showed no significant change in diversity of the fish community in the short-term. Long-term data series on fish catches and by-catch from the bay will make temporal shifts in shapes of BTLS and K-dominance curves robust ecological indicators of the trophic structure of the fish community of the bay. The shape of the BTLS may elucidate the array of effects of ecosystem disturbances that are acting simultaneously and interfere with ecological processes at different trophic levels by triggering top-down or bottom-up controls (Sosa- Lopez *et al.,* 2004).

The Malindi-Ungwana Bay fishery showed significant decrease in trophic level between 2011 and 2012, this is likely due to overall reduction in fish biomass. The Indiseas indicators analyzed for the bay suggests that Malindi-Ungwana Bay is ecologically degraded in terms of fish sizes, trophic characteristics and proportion of predators. The mean length of the shrimps have decreased by 38 % between 2011 and 2012, while the trophic level of landings have declined by 2% in the same period suggesting negative effects of fishing. The lack of long-term data made it difficult to make comparisons of present state of indicators in relation to reference points. The narrow temporal scale of the data for the study area calls for the need to institute long-term monitoring programs by the State Department of Fisheries and other stakeholders. The lack of reference points in this study is a common drawback in many studies employing system indicators (Jennings and Dulvy, 2005) and will be mitigated by long-term ecological monitoring initiatives.

5.1 CONCLUSION

This study found size spectral analysis (number, biomass and diversity) of the collected size-abundance data of the prawns useful in determining trends in the ecological status of the Malindi-Ungwana Bay. More temporal scale date will, however, be required in order to enhance the prediction ability of the datasets and to confirm the existence of the "floor effect" of the annual size-spectra analysis as indicated in this study. The trends in the size-spectra analysis are intuitively simple to interpret and can be adopted by resource managers to determine effects of exploitation on the environment.

The approaches used in this study are likely to be valuable in assessing the status of prawn fishery in the Malindi-Ungwana Bay by the managers. The study used biodiversity (e.g. Indiseas-based indicators) and conservation-based indicators (e.g. Biomass Trophic Level Spectra (BTLS), K-dominance curves) for spatio-temporal analysis of patterns of ecological attributes, and as potential ecosystem indicators of multifactor effects. Although firm conclusions could not be drawn from the patterns of BTLS due to the narrow temporal scale, the analysis was useful for tracking changes in the trophic structure of the bay more than the rather simple comparison of mean changes in trophic levels. Reference points of indicators will need to be developed for the bay using different approaches (e.g. Murawski, 2000; Rice, 2000; ICES, 2001; Rochet and Trenkel, 2003) in order to make application of the Indiseas–related ecological and biodiversity indicators more robust. The indicators derived and assessed in this study will form a suite of inputs in a holistic EAFM framework for the bay.

5.2 RECOMMENDATIONS

Following the results of this study the following recommendations are provided:

1. The ecological indicators described in this study should be adopted as one of the tools in evaluation and monitoring of the Malindi-Ungwana Bay ecosystem. The indicators are easy to conceptualize, have intuitive management interpretation and are based on simple deterministic models that can be applied and interpreted by resource managers.

2. Size spectra (number and biomass) have been shown to be a robust ecosystem metrics for examining the impact of fishing, where fishing causes mortality on larger individuals or species. The temporal decline in slopes and general increase in the intercept of the number size-spectra of the prawns in the bay may be interpreted to suggest effects of fishing on stocks and changes in ecosystem productivity, respectively. However, the contribution of biological interactions and environmental variability to ecosystem changes will have to be determined but are likely less influential in systems with heavy fishing pressure.

3. There is need for identification of thresholds and reference points for the Indiseas based indicators used in this study through modeling and long-term field based studies. Long-term monitoring programs of landings are recommended as a way to determine reference points for the indicators.

4. There is need to provide additional socio-economic data, including stakeholders inputs, and more biological data on the penaied species found in the bay in order to develop a holistic EAFM model for management of the bay resources.

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APPENDIX I

Indiseas –based ecological and biodiversity indicators calculations and formulae.

Indicator	Calculation	Formulae
Total Landings	The actual weights by species were estimated by multiplying the weight sampled by a raising factor for both retained catch (landings) and the discards from the following formula:	Raising Factor = Total catch weight of fish ÷ Sub-sample weight of either fish or prawns
Biomass Surveyed Species	Biomass estimates of fish were calculated using the swept area method. Each distance hauled was estimated in units of nautical miles (nm)	$D = 60 * Sqrt((Lat1-Lat2)^2 + (Lon1-Lon2)^2 * \cos^2(0.5*(Lat1+Lat2))$ where: Lat1= Latitude at start of haul (degrees) Lat2= Latitude at end of haul (degrees) Lon1= Longitude at start of haul (degrees) Lon2= Longitude at end of haul (degrees) The estimated distance was then multiplied by the distance of the head rope of the net to get the hauled area, and a factor X2 = 0.5 as the best compromise

		(Pauly, 1980). swept area, $a = D*hr*X2$ The catch rate per unit area was calculated using the following formula: $\frac{Cw/t}{a/t} = \frac{Cw}{a}$ kg/nm² where: Cw/t = Catch in weight per hour a = Area swept The estimated total biomass, B in an area, A was calculated from the following formula: $B = \frac{Average(Cw/a)*A}{X1}$ where: Cw/a = Catch per unit area of all hauls A = Total size of the area under investigation in nm² X1 = An estimate proportion of fish or prawns present in the area swept (where X1= 1 for prawns, and X1 = 0.5 for the swimming fish) From these calculations, the Total Biomass of Surveyed Species and the Biomass of Retained Species were calculated.

Mean Length	Calculation of the mean length was based on data of all surveyed species. The reason for choosing length is that it is more meaningful to public and length is less directly affected by environmental change.	$L = \sum L_i/N$ (cm) $L = \sum L_i/N$ (cm)
Trophic Level of Landings	The Trophic Level is a fixed non-integer per species. All retained species can be calculated from Ecopath model or diet data. For this study the Trophic Level was calculated for each species based on the mean length (TL_s), the catch of that species (Y_s) the total catch (Y)	$TL_{land} = \sum TL_s Y_s / Y$
1/(Landings/Biomass Retained)	This indicator reflects the global fishing pressure at the community level. The indicator is calculated using total landings and biomass of retained species.	B/Y Retained Species