

**AN EVALUATION OF CORAL REEF FISH
COMMUNITIES IN SOUTH AFRICAN MARINE
PROTECTED AREAS**

by

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ABSTRACT

Differences in coral reef fish assemblages were investigated on six South African and one southern Mozambican reef under varying management regimes. All of the South African reefs fall within marine protected areas (MPA) but are zoned for differing types and intensities of human activity. Reefs where no human activities are allowed were termed Sanctuaries, while those on which restricted fishing and SCUBA diving are permitted were termed Protected. The reef in southern Mozambique is subjected to unrestricted fishing and SCUBA diving and was consequently termed Open.

This study consists of two parts. The first dealt with a community assessment which investigated and provided baseline data on the trophic structure, density, and species diversity of fish assemblages on each of the seven study reefs. The objective was to compare the aforementioned metrics between reefs and thereafter compare them between the different protection zones. The second part of this study focused on assessing the impacts of human activities using 25 fish indicator species. These species were selected *a priori* based on their ecological importance and sensitivity to human activity (fishing and diving). The selection process was then guided by the results of the community assessment. The objective was to use these species as indicators of recreational diving and fishing pressure in the different protection zones. Density, biomass and size frequency analyses comprised the primary metrics in this assessment.

Randomly stratified underwater visual censuses (UVC) were used to collect the fish data and these were conducted on reefs inhabited by a coral community considered to be the core community on South Africa's reefs in terms of biodiversity and coral cover. The fish community assessment consisted of timed counts in which all non-cryptic fish species were quantified. Indicator species counts employed the point count technique with a radius of 10 m. An average of 11 community counts and an average of 62 point counts were conducted per reef. Various environmental variables and habitat characteristics were recorded during the UVCs.

Multivariate analysis of the fish assemblages indicated that the fish community structure differed significantly according to reef protection status. Sanctuary reefs were significantly different from the Open reef in Southern Mozambique. Mean fish abundance was highest on Sanctuary reefs and lowest on the Open reef. In terms of overall species diversity, a total of 284 species belonging to 50 families were recorded, this being comparable to other reefs in the WIO region. Six families

contributed more than 50% towards the fish community composition: Labridae, Acanthuridae, Chaetodontidae, Lutjanidae, Pomacentridae and Serranidae. All predator categories were well represented on Sanctuary reefs, while top-level predators were scarce on the High-Diving and the Open reef. Generalised linear model (GLM) regression analysis indicated that human activities were significant variables in accounting for the variance in fish community structure.

The total fish abundance and biomass of the selected indicator species were significantly higher in Sanctuary zones and lowest in the Open zone. In addition, Sanctuary zones were characterised by high numbers of large predators, while non-Sanctuary zones were characterised by higher abundances of prey species. Target species were also larger and more abundant in Sanctuary zones. The data revealed that recreational fishing and high diving intensity may be influencing the fish community structure on southern African coral reefs, which was confirmed by GLM regression analysis. Long-term monitoring of these fish communities is recommended to confirm the trends observed in this data set.

PREFACE

The experimental work described in this thesis was carried out at the Oceanographic Research Institute in the School of Biological Science, University of KwaZulu-Natal, Durban from March 2007 to May 2010, under the supervision of Professor Michael H. Schleyer and under the co-supervision of Dr Louis Celliers.

These studies represent original work by the author and have not otherwise been submitted in any form for any degree or diploma to any tertiary institution. Where use has been made of the work of others it is duly acknowledged in the text.

Signed _____

Date _____

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CHAPTER 1

GENERAL INTRODUCTION

Human resource use in the marine environment

Human use of marine ecosystems extends back thousands of years (Jackson 1997, Jackson et al. 2001, Myers & Worm 2003). However, only recently have scientists begun examining historical records to assess the extent of this exploitation. The evidence suggests that major structural and functional changes caused by overfishing occurred worldwide in coastal marine ecosystems centuries ago (Salvat 1981, Hay 1984, Birkeland 1997, Jackson et al. 2001, Pandolfi et al. 2005, Sims & Southward 2006). So great was the historical magnitude of overexploitation that many species (e.g. turtles, sea cows, sharks, manatees, dugongs, jewfish, swordfish and shellfish) have failed to regain their former abundance or are absent from most coastal ecosystems (Jackson et al. 2001). This overexploitation preconditioned modern ecological investigations and has thus provided inappropriate reference points for identifying targets for rehabilitation measures (Pauly 1995). This is known as the 'shifting-baseline' theory and is one of the reasons why scientists throughout the past century have been unable to recognise the continual and rapid decline of marine ecosystems (Pauly 1995, Sale 2008).

The lack of baseline models for pristine marine ecosystems is particularly acute in the case of coral reefs. Few, if any, coral reefs remain today that have not been impacted by some form of human activity (Hodgson 1999, Jackson 2001). Without insights into the natural structure and functioning of such biologically complex ecosystems, scientists are left with a limited and poor understanding of undisturbed communities on which to base future management decisions (CoML 2009). In addition, there appears to be a 'focus-shift' in the current marine management ethos. Due to global concern for escalating exploitation rates, gathering baseline data has become secondary to marine science focusing on mitigation of the effects of human activity on marine ecosystems. A consequence is that, due to the escalating intensity of human impacts on the marine environment, considerable biodiversity may have already been lost before it could be documented (Paulay 1997, Reaka-Kudla 1997).

In the 1970-1980s, land-derived sources of pollution were considered the greatest threat to coral reefs (Hatcher et al. 1989) that caused local to regional losses in coral cover and biodiversity. Scientific opinion changed dramatically in 1997-98 when global threats to coral reefs, such as coral

bleaching (Buddemeier 1999, Hoegh-Guldberg 1999, Wilkinson 1999) and chemical imbalances in sea water following increased CO₂ emissions became apparent (Kleypas et al. 1999). The grounds for change were that isolated, previously 'pristine' reefs were being severely damaged (Wilkinson 1999). The global bleaching event of 1998 effectively destroyed 16% of the coral reefs of the world, with losses in the Indian Ocean attaining almost 50% (Wilkinson 2004). It is anticipated that future changes in ocean chemistry due to higher atmospheric carbon dioxide concentrations may cause weakening of coral skeletons and reduce reef accretion (Kleypas et al. 1999, Kleypas & Langdon 2006). The loss of living coral on such a large scale has enormous implications for the biological communities that rely on the architectural complexity of coral reefs (Graham et al. 2006, Flechet 2008, Graham et al. 2008).

Nevertheless, the severity of global threats does not permit one to ignore or underestimate local or regional impacts. Direct anthropogenic impacts listed by Hatcher et al. (1989) continue to threaten coral reefs as both human population and economic growth are increasing exponentially (Wilkinson 1999, Risk et al. 2001). In their analysis of global threats to coral reefs, Donner & Potere (2007) estimated that 10% of the world's population live within 100 kilometres of coral reefs and over 91% live in 'developing' nations. The demand for goods and services derived from coral reefs is thus enormous, given that coral reef fisheries are the main, and, in many instances, the only source of protein for many of the poorest societies in the world (Bryant et al. 1998). Consequently, threats such as destructive fishing practices (Edinger et al. 1998, Fox et al. 2005, Obura et al. 2006b) and overexploitation of marine species (Pauly 1995, Hodgson 1999, Jackson et al. 2001, Knowlton & Jackson 2008, Agnew et al. 2008, Sandin et al. 2008) appear as pervasive as ever and are among the most significant of anthropogenic impacts on coral reefs.

Marine protected areas – old but new concept

Global concern caused by fisheries collapse and a shift from managing single species fisheries to viewing whole communities as ecological units promoted the promulgation of marine protected areas (MPAs) as an alternative management approach to marine conservation (Palumbi 2001). In fact, MPAs were advocated as the 'ideal' management solution because they were perceived to simultaneously address issues of overfishing, habitat degradation, and tourism development (Kelleher & Kenchington 1992). During the late 1950-60s, conservation science and principles for establishing and managing MPAs developed rapidly (Chape et al. 2005). By 1985, 430 MPAs had been proclaimed in 69 countries (De Silva et al. 1986) and, by 1995, the total number of MPAs exceeded 1300 (Kelleher et al. 1995).

Although the concept of marine protected areas (MPAs) is fairly recent, there is some evidence to suggest that the custom of setting areas aside for restricted access and the creation of sanctuaries to facilitate species recovery has its roots in traditional or indigenous communities, particularly in oceanic islands (Johannes 1978, 1981, 2002, Ruddle 1993, Mantjoro 1996). However, the demarcation of areas for aesthetic western values and the establishment of contemporary MPAs also owe much to protected area initiatives which began in the late 1880s and centred on the protection of terrestrial wildlife in the western world (Redford & Sanderson 2000). It was not until 1935 that the first complete marine park was proclaimed at Fort Jefferson National Monument in Florida with the protection of all the underwater areas within its boundaries (Randall 1968, Gare 1975). However, the term ‘marine protected area’ only gained prominence in international marine terminology in the latter half of the twentieth century (Chape et al. 2005).

Over the past two decades, a large literature base has been published on MPAs, their uses and the benefits they provide. Empirical evidence from numerous studies has demonstrated that MPAs can enhance the abundance of target species (Russ & Alcala 1989, Polunin & Roberts 1993, Russ & Alcala 1996a, McClanahan & Arthur 2001, Roberts et al. 2001, Friedlander & Demartini 2002, Unsworth et al. 2007, Lester et al. 2009), increase species diversity or richness (Jennings et al. 1996, McClanahan & Arthur 2001, Barrett et al. 2007), increase total fish densities (McClanahan & Shafir 1990, Lester et al. 2009), increase the size of target species (Russ & Alcala 1996b, Wantiez et al. 1997, Barrett et al. 2007, Watson et al. 2009) and provide export stock for adjacent areas open to fishing (Russ & Alcala 1996a). Although, these studies suggest that MPAs are effective conservation tools from a fisheries management perspective, numerous authors highlight the fact that more MPAs are unsuccessful than successful in achieving their management objectives (Kelleher et al. 1995, Alder 1996, McClanahan 1999, Mora et al. 2006).

MPAs are not ‘cure-alls’

Alder (1996) identified several factors limiting MPA effectiveness when the concept of MPAs gained impetus in the 1960s and developed in the 1970 and 1980s. These included: the lack of a clear definition for MPAs, limited skills in managing the dynamic nature of marine ecosystems and a lack of information about marine resources and their use (Alder 1996). This author noted that, of particular concern, there was a lag between the proclamation of many tropical MPAs and the formulation and implementation of management plans, which may take between 1-2 years to collate. However, many nations have proclaimed at least one MPA every year since the 1970s, which suggests a prevalence of ‘paper parks’ throughout the tropics (Alder 1996).

Currently, there are 5161 MPAs in 176 countries (WDPA-Marine 2008). Nevertheless, the positive impression created by the rapid acceleration in MPA proclamation is tempered by the recognition that, while 18.1% of worldwide coral reefs lie within MPA boundaries, only 1.6% fall within adequately managed MPAs (Mora et al. 2006). It thus appears that the management problems identified by Alder (1996) have resulted in limited progress in the last decade. Most MPAs still face difficulty in implementation and enforcement due to poor governance, and a lack of management guidance and evaluation (White et al. 2006). The mixed success and performance of many current MPAs demonstrate the need to build capacity in MPA management teams. This is necessary to evaluate their effectiveness so that decision-makers can adapt their efforts and enhance their protective strategies over time (Pomeroy et al. 2005).

Defining MPAs

The most commonly used definition of a MPA is that provided by the International World Conservation Union (IUCN), 'any area of intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment' (Kelleher & Kenchington 1992). MPAs are specifically intended to limit human activities in designated locations (Sale et al. 2005, Mora et al. 2006) and the degree to which human activities are limited determines the type of MPA. In most instances, MPAs can be classified into two broad types; areas that are open to resource use and areas closed to resource use. No-take MPAs are areas closed to exploitation and, for the purpose of this study, will be termed sanctuary or no-take zones. Sanctuary areas offer the greatest protection for marine resources and ecosystems (Gell & Roberts 2003, Lester et al. 2009). The second type of MPA allows harvesting of resources, but under protective regulations that pertain to each species being harvested. In addition, the types of fishing or harvesting gear may be restricted. Such MPAs are multiple resource use zones and most often permit recreational activities such as SCUBA diving, snorkelling, whale watching and fishing.

Despite the shortcomings of MPAs, they are still advocated as one of the most viable and useful management tools for conserving coral reefs (McClanahan 1999, Roberts & Hawkins 2000, Lubchenco et al. 2003, IUCN-WCPA 2008). This is particularly true of coral reefs in developing nations where few other fisheries management options are available (Alcala & Russ 2006). In addition, MPAs provide an area that acts as a buffer against unforeseen yet potentially disastrous management mistakes. In science-based conservation, this is termed the precautionary principle. It

is what drives managers to avoid actions that produce irreversible changes to ecosystems at all costs and to err on the side of conservation in the face of scientific uncertainty (Agardy 1994, Sale 2008).

MPA evaluation through monitoring

Well-designed monitoring programmes are needed to gather data about the pathways of biological components and ecosystem rebuilding, to assess the benefits of conservation, increase the knowledge of resources users and scientists, and improve the level of protection (Sumaila et al. 2000). Monitoring programmes should first establish baseline conditions through biodiversity inventories, which then feed directly into stress identification and mitigation programmes (Risk 1999). Inventories form the foundation of any monitoring programme and provide an understanding of how processes such as predation, reproduction and competition regulate marine biodiversity, and aid in predicting the potential consequences of biodiversity loss (Bellwood & Huges 2001). To gain a holistic idea of ecosystem biodiversity, inventories should be assessed in terms of a range of functions that include functional and genetic descriptors as well as species richness and abundance (Ormond & Roberts 1997).

The scope of coral reef monitoring has expanded considerably over the past two decades. Broad-scale efforts by large institutions include regional networks such as the Caribbean Coastal Marine Productivity network (CARICOMP 2002), the Atlantic and Gulf Rapid Reef Assessment (AGRRA 1997-2000; Kramer 2003), and the Global Coral Reef Monitoring Network (GCRMN 2008). Recently, volunteer-based monitoring programmes such as Reef Check (Hodgson 2000) and Reef Watchers (CERMES 2008) have made important advances in informing the plight of coral reefs to the general public and initiating community involvement. In addition, a number of monitoring protocols have been developed by scientists for non-scientists to bridge the gap between academic concepts and reef management (Rogers et al. 1994, McClanahan 2008, Goffredo et al. 2010). These monitoring manuals and protocols have been designed to document changes in various biotic and abiotic variables on coral reefs. However, few are capable of diagnosing what is actually causing a change (Jameson et al. 2001). Without the forensic data linking biological change to causative agents, resource managers and scientists are only able to say that ‘reefs are ill’ or ‘reefs are dying’, but are unable in being able to rectify the situation (Downs et al. 2005).

Indicator-based monitoring

The concept of using indicators to assess ecological condition has become an important tool in coral reef research (see Jameson et al. 2001, Jameson & Kelty 2004 for review). Traditionally, indicators were developed based on single physical, chemical or biological variables or species (ICES 2000).

However, an approach that takes the entire ecosystem into account has recently been advocated as ecosystems are so complex and unpredictable that suites of indicators are needed to provide an adequate representation of reef condition (Rice & Rochet 2005). As ecological understanding has advanced, composite indicators have been developed that are both specific and widely applicable, and that may be based on indirect or direct measures of relevant ecosystem processes (Fabricius 2006). According to Dale & Beyer (2001), the ecological basis of such indices should reflect various key elements of ecosystems, namely their structure (abundance, population composition), function (biomass, tropho-dynamics, reproduction) and composition (diversity, dominance, density). The rationale for this is based on the selection of a suite of representative indicators that will provide a link between these key ecosystem elements and ecosystem processes. The challenge in developing such ecological indicators is in determining which of the numerous measures of ecosystems characterise them but are simple enough to be effectively monitored (Dale & Beyer 2001). In reality, the choice is which taxa to select as indicators, because their presence or absence and fluctuations will reflect changes in the ecological processes (Noon et al. 1999).

Indicators of coral reef health

A broad spectrum of organisms and metrics has been used as indicators to assess coral reef status (Table 1.1). Jameson et al. (1998 & 2001) provide the most recent reviews of these indicators and their metrics used in biomonitoring (indicator-based monitoring programmes) which, according to the authors, have the greatest potential for development. Fish have been successfully used as indicators of environmental change in a variety of aquatic habitats (Karr 1981, 1986, Whitfield 1996, but see Table 1.1 for coral reef references). There are many advantages in focusing on fish species as indicators on coral reefs: 1) they comprise a large proportion of the biomass; 2) they provide ecosystem services to humans; 3) they show clear responses to fishing; 4) extensive life-history information is available for most species; 5) fish are relatively easy to identify; 6) coral reef fish communities include a range of species that represent a variety of trophic levels; 7) they include many life forms and functional groups and are thus likely to reflect changes in most components of coral reefs affected by human activities; 8) they include mobile and sedentary species and thus will reflect stressors with a narrow and broad spatial coverage; and 9) they have high public awareness value such that the general public are more likely to relate to information on fish community condition than on invertebrates or algae (Karr 1981, Whitfield & Elliott 2002, Rice 2003). There are also a number of disadvantages in using fish taxa as indicator; however, these are associated with the use of any major taxon (Karr 1981, Whitefield & Elliot 2002).

Table 1.1 Coral reef bioindicators commonly used to assess coral reef health.

Indicator	Metric	Oceanic region	Reference:
Corals	Colony size structure	Caribbean, Pacific, Indian	Bak & Meesters 1998
	Coral fecundity	Caribbean, Pacific, Indian	Brown 1988, Edinger and Risk 1999
	Coral recruitment	Caribbean, Pacific, Indian	Ward & Harrison 1997, Harrison & Ward 2001
	Coral Damage Index	Red Sea	Jameson et al. 1999
	Disease	Western Atlantic	Richardson 1996, Peters 1997
	Percent coral bleaching	Caribbean, Pacific, Indian	Brown 1988, Jones 1997
	Percent coral cover	Caribbean, Pacific, Indian	Aronson et al. 1994, English et al. 1994, 1997
	Coral growth rates	Caribbean, Pacific	Brown 1988, Cortes & Risk 1985, Brown & Suharsono 1990, Risk et al. 1995
	Bioeroders	Caribbean, Pacific, Indian	Sammarco & Risk 1990, Risk et al. 1995, Holmes et al. 2000, Linton & Warner 2003, Cooper et al. 2009
	Coelobites (cavity dwellers) abundance	Pacific, Indian	Choi 1982, Risk et al 2001, Linton & Warner 2003
	Foraminifera	Western Atlantic	Hallock 1996, 2000
Fish	Corallivore abundance (<i>Acanthaster planci</i> and <i>Drupella</i> sp)	Pacific	
	Butterflyfish abundance and behaviour	Pacific	Reese 1981 & 1995, Hourigan et al. 1988, Crosby & Reese 1996, Erdmann & Caldwell 1997
	Ectoparasites	Caribbean, Pacific, Indian	Evans et al. 1995
	Larval fish assemblages	Caribbean, Pacific, Indian	Doherty 1991
	Target fish species abundance		McManus et al. 1997
	Trophic structure	Pacific	Bozec et al. 2005, Gascuel et al. 2005
Corals, fish, invertebrates	Exploitation rate		Trenkel & Rochet 2003
	Size of target species		Shin et al. 2005
Corals, fish, invertebrates	Target fish species abundance, percent hard coral cover, percent dead coral, sponge, invertebrate abundance, butterflyfish abundance	World	Hodgson 1999 (Reef Check), CERMES 2008 (Reef Watchers)
Coral and fish	Fish abundance, coral cover and colony size, coral bleaching and disease, rugosity,	Atlantic	McField & Kramer 2007 (Healthy Reefs for Health People)
Gastropod	Gastropod imposex	Caribbean, pacific, Indian	Gibbs & Bryan 1994, Evans et al. 1995, Gibson & Wilson 2003
Macrophytes	Macrophytic algal blooms	Caribbean, Pacific, Indian	McManus et al. 1997
Giant clam	Giant clam zooxanthellae, shell growth rates	Pacific, Indian	Ambariyanto & Hoegh-Guldberg 1997
Foraminifera	Sediment constituent analysis, foraminiferal assemblages,	Western Atlantic	Hallock 1996, 2000

Assessing reef fish communities

The most widely used methods to assess reef fish communities are underwater visual census (UVC) techniques. UVCs have been used to estimate fish abundance since the 1950s (Brock 1954) and are believed to be the best method for estimating reef fish abundance and biomass because they are non-destructive, cost-effective and easily implementable in monitoring programmes (Watson & Quinn 1997, Kulbicki et al. 2010). In addition, UVCs are particularly useful because they are independent of fishing (Samoilys & Carlos 2000) and destructive methods such as ichthyocides (Ackerman & Bellwood 2000). All methods of UVCs have inherent biases (Sale 1991) and may be inaccurate through e.g. the underestimation of cryptic species (Brock 1982, Fowler 1987, Kulbicki 1998). Thus, UVCs are most suited to quantify diurnally exposed fish species (Brock 1982, Samoilys & Carlos 2000).

Transect counts and point counts (Bohnsack & Bannerot 1986, Samoilys & Carlos 2000) are the main types of UVC employed to estimate fish densities. Differences between transect and point count UVCs have been highlighted by several authors and, as yet, there is no accepted 'best' method (Samoilys & Carlos 2000, Edgar et al. 2004, Kulbicki et al. 2010). Transect UVCs allow the rapid census of diverse fish assemblages across large spatial scales (Edgar et al. 2004); however, Brock (1982) suggested that UVC accuracy may be increased by conducting a greater number of shorter (20-25 m) rather than longer (100-200 m) transects. In contrast, Samoilys and Carlos (2000) found that transects and point counts were equally effective UVC methods, although point counts were preferred because they could be conducted more quickly and allow for increased replication. Point counts also allow recording of habitat characteristics such as benthic composition and topography within a smaller spatial scale which is particularly advantageous for reefs of varied topography and habitat.

Single-versus multiple-species approach

Chaetodonts or butterflyfish are among the fish species that have received considerable attention as bioindicators of coral reef health. Reese (1977) first proposed that obligate corallivores, such as butterflyfish, could serve as bioindicators, a theory which was then adopted by numerous other authors. The premise is that corallivorous butterflyfish have coevolved with and are intimately linked to the corals on which they feed (Reese 1981, Harmelin-Vivien & Bouchon-Navaro 1983, Reese 1991, Crosby & Reese 1996). However, the effectiveness of butterflyfish as indicators remains unresolved. Roberts et al (1988) argued that butterflyfish are not appropriate indicators to compare coral health between different sites, as pristine sites may naturally have low live coral

cover. In addition, coral declines may be slow and changes in butterflyfish abundance may be even slower.

The proposed use of chaetodons as indicators of coral reef status provides an example of a monitoring approach that is based on parameters associated with individual species or simple community metrics such as abundance. Alternative approaches to ecosystem assessment were proposed more than two decades ago. Karr (1981) introduced the concept of using a representative number of fish species to assess the biotic integrity of stream-fish communities. This author designed an approach that assessed the status of a freshwater stream using twelve fish community parameters; including relative abundance, trophic levels, and species richness. This multi-species and multi-parameter approach has received considerable attention since its conception, particularly with regard to fish communities (Fausch et al. 1984, Karr et al. 1986, Simon & Emery 1995, Hodgson 2000).

The concept of developing a multi-species index to assess community condition formed the basis of this study, which aimed to develop and apply indicators to assess the impacts of human activities on South African coral-inhabited reefs. The fish indicator species were selected using a combination of methods. First, a literature search was conducted to compile a list of potential indicators species that are targeted by fishers, sensitive to diver presence or associated with undisturbed reefs. The indicator list was then validated using the results of a baseline community assessment. A final list of fish indicator species was compiled termed the Fish-index.

South Africa has a long history of MPAs and human resource use in the marine environment and thus provided an opportunity to test the indicator concept and comment on the role of MPAs in coral reef conservation. It is at this point that a review of South African coral reef MPAs is appropriate.

South African MPAs

South Africa has a rich diversity of marine and coastal resources, which has provided important social and economic opportunities for food, commercial gain, recreation and transport (Attwood et al. 2000). South Africa's increasing population density is placing ever-growing demands on marine resources, as is the case in other developing nations in the Western Indian Ocean (WIO) (Tunley 2009). However, unlike many other African countries, South Africa appears to be well-endowed with MPAs (Hockey & Branch 1997, WDPA-Marine 2008).

South Africa is a signatory to several international conventions and protocols that advocate the implementation of MPAs as a tool for marine conservation. These include the Convention on Biological Diversity, and the related Jakarta Mandate, WSSD Johannesburg Accord, Nairobi Convention, and FAO Code of Conduct for Responsible Fisheries (Lemm & Attwood 2003). In addition, South Africa has made a commitment to meet the international target set during the Fifth World Parks Conference of establishing a representative and effectively managed MPA network by 2012 (DEAT 2006).

The first MPA in South Africa was declared in 1964 and, since then, twenty-one MPAs have been promulgated under national legislation – the Marine Living Resources Act No. 18 of 1998 (MLRA). This equates to approximately 18% of South Africa's coastline, with 5% located within no-take or sanctuary zones (Attwood et al. 1997). The MLRA lists three objectives for MPAs, viz.: 1) the protection of marine life; 2) the facilitation of fisheries management; and 3) the reduction of user-conflict. MPAs are thus an important aspect of marine conservation in South Africa and are considered essential for fisheries management (Attwood et al 1997). Numerous types of MPAs have been promulgated, including multiple resource use MPAs, no-take zones or sanctuaries, Ramsar Sites, a World Heritage Site and a UNESCO Biosphere Reserve (Tunley 2009).

The MPAs in South Africa also include a wide variety of ecosystems and range considerably in size. The major ecosystems included in the MPAs are intertidal habitats, estuaries and offshore reefs (coral and rocky reefs).

The South African coral-inhabited reefs

South Africa's coral-inhabited reefs occur at the limits of tropical reef distribution (27-28°S) and are thus considered high-latitude, marginal reefs (Kleypas et al. 1999). Reef formation on marginal reefs such as those in South Africa is said to be constrained by low aragonite saturation state (Ω -arag <3.4), which hinders the creation of the massive biogenic carbonate structures typical of

tropical reefs (Stoddart 1969). Instead, marginal reefs comprise non-accretive communities or veneers of living coral (Goreau 1969). Consequently, the term ‘coral community’ has been suggested as an alternative to ‘coral reef’ because such ecosystems do not conform to the geological definition of typical coral reefs (Kleypas et al. 1999).

Most high-latitude reefs are characterised by large standing crops of macroalgae (see table 2 in Johannes et al. 1983). This is untrue of South Africa’s coral-inhabited reefs where corals are the prominent fauna, constituting between 50-70% of the benthic cover on the reefs (Riegl 1993, Jordan & Samways 2001, Celliers & Schleyer 2008). Furthermore, hermatypic coral diversity is higher than many other reefs at similar latitudes; 95 species representing 46 genera have been recorded on the reefs (Schleyer & Celliers 2003). In comparison, 70 hermatypic coral species representing 32 genera have been identified in the Houtman Abrolhos Islands in Western Australia (Crossland et al. 1984) and 57 species in 33 genera at Lord Howe Island (Veron & Done 1979). In addition, fish diversity on the coral-inhabited reefs is high with 399 recorded species (Chater et al. 1993). This is, however, considered an underestimate as 500 species are expected to occur on the reefs (Dennis King pers. comm.). Approximately 80% of the fish community is comprised of tropical Indo-Pacific reef species (Chater et al. 1993). Thus, although the South African coral-inhabited reefs represent marginal coral communities from a structurally and geological perspective, from an ecological and biological perspective, they appear to represent diverse, high-latitude coral reef ecosystems. This view is shared by other authors who acknowledge that the importance of marginal coral reefs or coral communities should not be considered any less significant than their tropical counterparts because they perform the same ecological function as coral reefs (Spalding et al. 2001). For the purposes of this study, South Africa’s coral-dominated reefs will thus be referred to as coral reefs.

The South African coral reefs are located on the Maputaland coast and are hence known as the Maputaland reefs. Certain biological components of these coral reefs have been well documented. Schleyer and Celliers (2003) provide a review of studies and key events relating to coral community research on the Maputaland coral reefs. An updated literature search contributed an additional eight peer-reviewed articles, bringing the total number of research publications to thirty two. In comparison, there have been only two investigations assessing the fish communities on South African coral reefs (Chater et al. 1993, 1995).

The coral reefs lie within two contiguous MPAs; the St Lucia MPA and the Maputaland MPA. A number of the Maputaland coral reefs have a long history of human resource use, particularly recreational gamefishing and SCUBA diving (Schleyer 2000). In contrast, there are also sanctuary or no-take zones that have been closed to human activities for more than 20 years. The Maputaland

reefs thus presented an opportunity to investigate the nature of the coral reef fish communities exposed to contrasting levels of human resource use. Prior to this study, observations indicated that the reef fish communities varied on reefs subjected to different levels of protection (Michael Schleyer pers comm.). The purpose of this study was to establish whether these observations were real and quantifiable. If this proved the case, it would be useful to include a non-MPA study reef in which unregulated reef resource use occurred. For this reason a reef in southern Mozambique at Ponta Malongane was also included. The major focus of this study; however, remained the Maputaland reefs.

Key questions and research plan

The overall aim of this research was to assess the nature of the fish communities on South African coral reefs relative to their protection from extractive and non-extractive use. The following following key questions were formulated:

1. Do reef fish communities differ between reefs in terms of their abundance, diversity, family composition and trophic structure?
2. Do the above community metrics differ between reefs of varying protection status?
3. Are there any species that appear to be more ecologically important than others i.e. do they have indicator value?
4. Are there differences in abundance, biomass, trophic structure and size of the selected indicator species between the different protection zones?
5. Are there any indicator species that manifested unexpected trends in the presence of human activities?
6. Are these indicator species effective in assessing the impacts of human activity on South African coral reefs?
7. What value, if any, do the indicator species have in assessing the effectiveness of MPA management on South African coral reefs?

Thesis outline

This thesis consists of five chapters and primarily draws on field-based research to achieve the research objectives. Chapter 1 is the general introduction, which provides the theoretical background, rationale, and current literature that contextualises this study and its concepts. Chapter 2 consists of a description of the study sites providing relevant biological, physical and geological information. Detailed 3-D maps of the reefs are included. Chapter 3 provides the baseline data for this study by describing the fish communities on southern African coral reefs. In addition, it

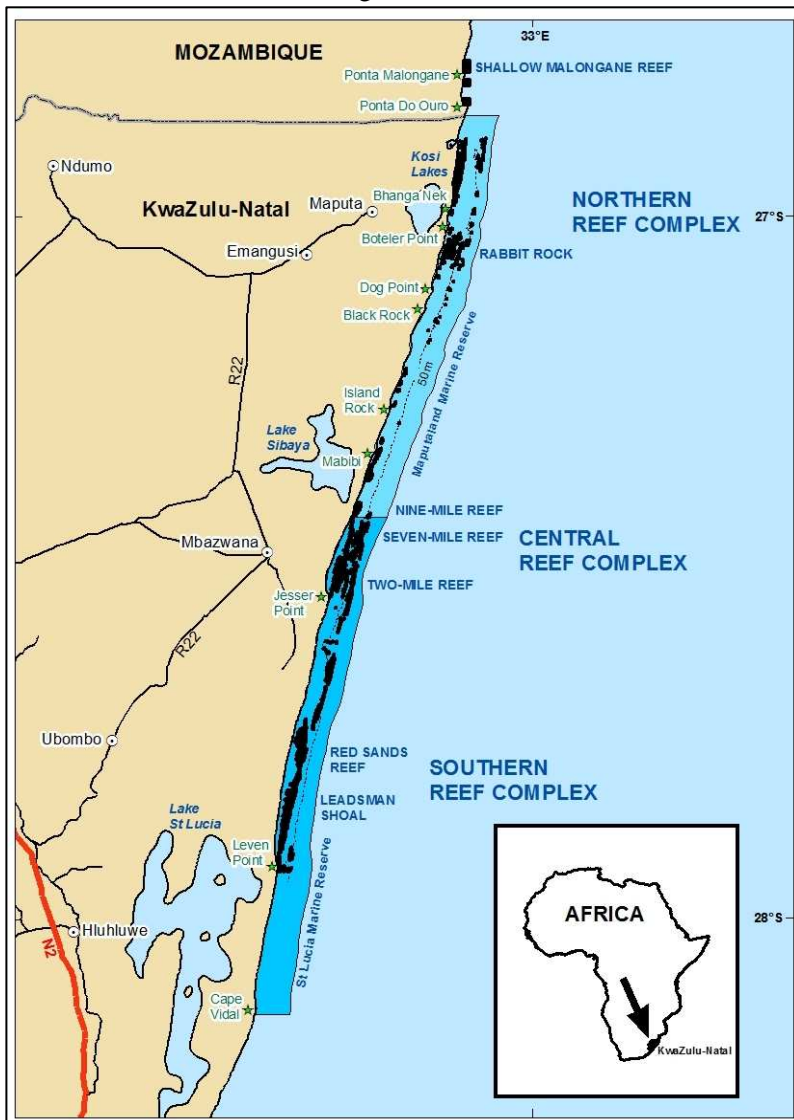
compares the fish communities in terms of reef protection status. Finally, the analysis of the fish community in Chapter 3 identifies a number of ecologically important species. This list of species was used to validate 25 *a priori* selected indicator species that are further investigated in Chapter 4. Chapter 4 used the 25 fish indicator species to assess the impacts of human activities in the different MPA zones. The management implications of these results are discussed. Chapter 5 concludes the thesis and comprises the general discussion. Here, the general findings of the thesis are discussed in a regional and global context. In addition, the strengths and limitations of the study are discussed as well as recommendations for future research.

CHAPTER 2

STUDY SITES

2.1 Study site description

The study area was located along the north-east coast (Maputaland) of South Africa and extended 10 km beyond the South African border into southern Mozambique (Fig. 2.1). The latitudinal extent of the area was from 26°46'S to 27°50'S and covered a distance of 160 km from north to south. Seven separate reefs were included in the study area, six of which are located in the Maputaland region of South Africa and one in southern Mozambique at Ponta Malongane. The GPS co-ordinates are listed in Figure 2.1



Reef	Geographical co-ordinates
Shallow Malongane (SM)	26° 46.903S 32° 54.008E
Rabbit Rock (RR)	27° 02.876S 32° 51.793E
Nine-mile Reef (NMR)	27° 24.689S 32° 43.574E
Seven-mile Reef (SMR)	27° 27.211S 32° 42.678E
Two-mile Reef (TMR)	27° 31.649S 32° 41.061E
Red Sands (RS)	27° 46.176S 32° 37.767E
Leadsman Shoal (LMS)	27° 50.108S 32° 26.200E

Figure 2.1 Location of the seven study reefs along the Maputaland coast of South Africa and at Ponta Malongane in southern Mozambique. The dark shaded section represents the St Lucia MPA and the light shaded area represents the Maputaland MPA.

2.1.2 Climate

The study area has a humid, sub-tropical climate due to the presence of the warm Agulhas Current which feeds moisture to the overlying atmosphere (Jury et al. 1993). The seasonal cycle is unimodal with peak rainfall and temperatures in the summer months (December to February) (Jury 1998). The mean annual rainfall exceeds 800 mm and the mean temperature range along the coast is between 16-25° C for Maputaland and 22-24° C for southern Mozambique (Hunter 1988, Hatton 1995). The predominant winds have a strong north-easterly component with a maximum wind speed of 14-16 knots (Fig. 2.2). The south-westerly winds are less frequent, but have a greater maximum velocity of 20-24 knots.

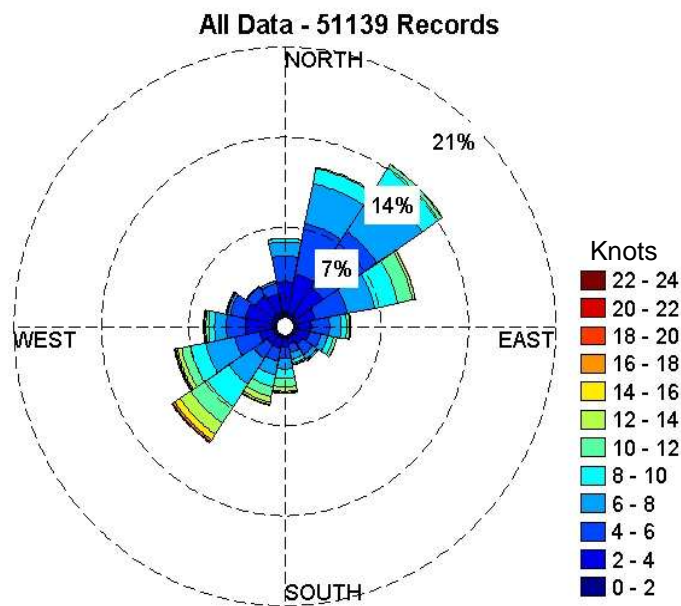


Figure 2.2 Wind rose for the study period January 2007 to February 2009. Data is for Richards Bay, supplied by the CSIR, Stellenbosch, collected on behalf of the Transnet National Port Authority (TNPA).

2.1.3 Oceanography

No major rivers flow into the sea near the reefs; the coastal waters are thus oligotrophic and the average visibility ranges between 10-15 m (Schleyer 2000). The mean seasonal sea surface temperatures (SSTs) range from 22°C in winter to 27 °C in summer (Smith et al. 1996). Celliers and Schleyer (2008) showed an increase in mean sea temperature from 1994 to 2000 of 0.15° C pa, and a decline in temperature of 0.07 °C pa from 2000-2006. These fluctuations were attributed to local, macro-cyclical phenomena (Schleyer & Celliers 2003). Sea temperatures recorded at a fixed monitoring station on Nine-mile Reef during the study period are presented in Figure 2.3.

The Agulhas Current is the predominant regional current carrying warm water southwards at a mean peak velocity of 1.4 m/s (Lutjeharms 2006). The average surface velocity in the region is 0.27 m/s (Morris 2009). This current is a western boundary current within the South West Indian Ocean subgyre, an anti-cyclonic wind-driven circulation system present throughout the year (Lutjeharms 2006). Results of Acoustic Doppler Current Profiler records from Nine-mile Reef showed that southerly currents are predominant, while counter-currents flowing north are infrequent (Morris 2009). The primary driving force of the northerly reversals is southerly winds (Morris 2009). The prevailing north-easterly and south-westerly winds (Fig 2.2) generate considerable swell (Schleyer 2000) with a predominant south-easterly component (Fig. 2.4). An occasional reversal due to southerly winds occurs in conjunction with approaching low-pressure frontal systems (Tyson & Preston-Whyte 2000).

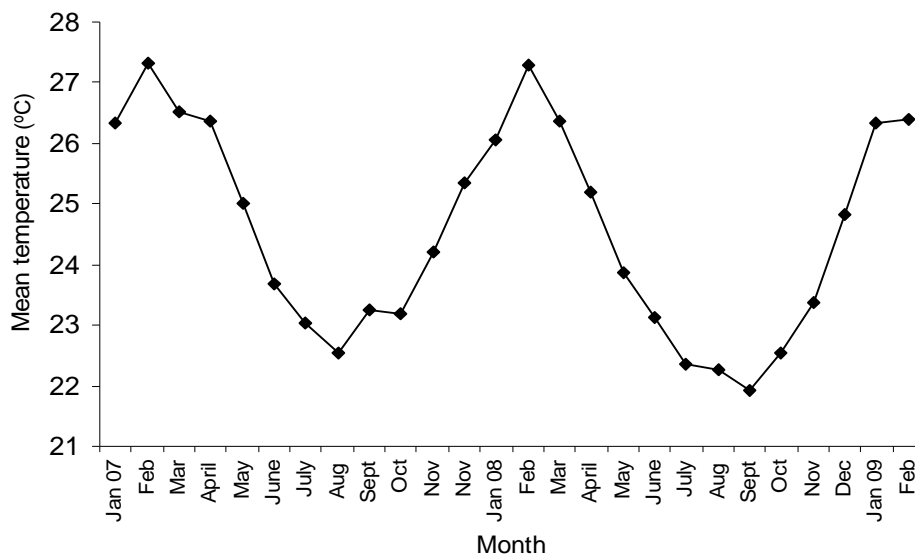


Figure 2.3 Mean monthly temperatures at a depth of 18 m on Nine-mile Reef in the Central Reef Complex for the period January 2007 to February 2009.

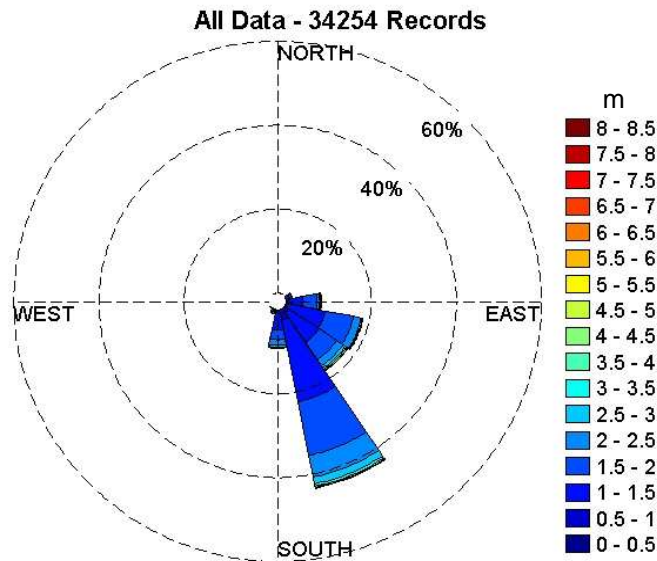


Figure 2.4 Swell rose for the study period January 2007 to February 2009. Data is for Richards Bay, supplied by the CSIR, Stellenbosch, collected on behalf of the Transnet National Port Authority (TNPA).

2.1.4 Geology

The South African and southern Mozambican coral reefs are confined to the narrow continental shelf which extends two to seven km offshore along the length of the coastline. The coast is linear and sandy (Ramsay 1994) but has late-Pleistocene beachrock and aeolianite outcrops comprising the dominant consolidated lithology on the shelf (Ramsay 1996).

The reefs can be classified as patch reefs and lie approximately 1 km offshore (Ramsay & Mason 1990). The size of the reefs varies between 1-2 km in length and between 0.6-1 km in width. None of the reefs reach the surface and the depth range of the reefs is 10-25 m (Schleyer 2000). They are atypical of tropical coral reefs because they are not massive carbonate structures, but instead consist of the aforementioned late-Pleistocene beachrock which originated from submerged coastal sand dunes (Ramsay 1996). Carbon-14 dating of a fossilised coral fragment (*Favia* sp) found in an intertidal beachrock sequence 35 km north of Sodwana Bay date the Maputaland reefs to a minimum age of 3780 ± 60 years BP (Ramsay & Mason 1990).

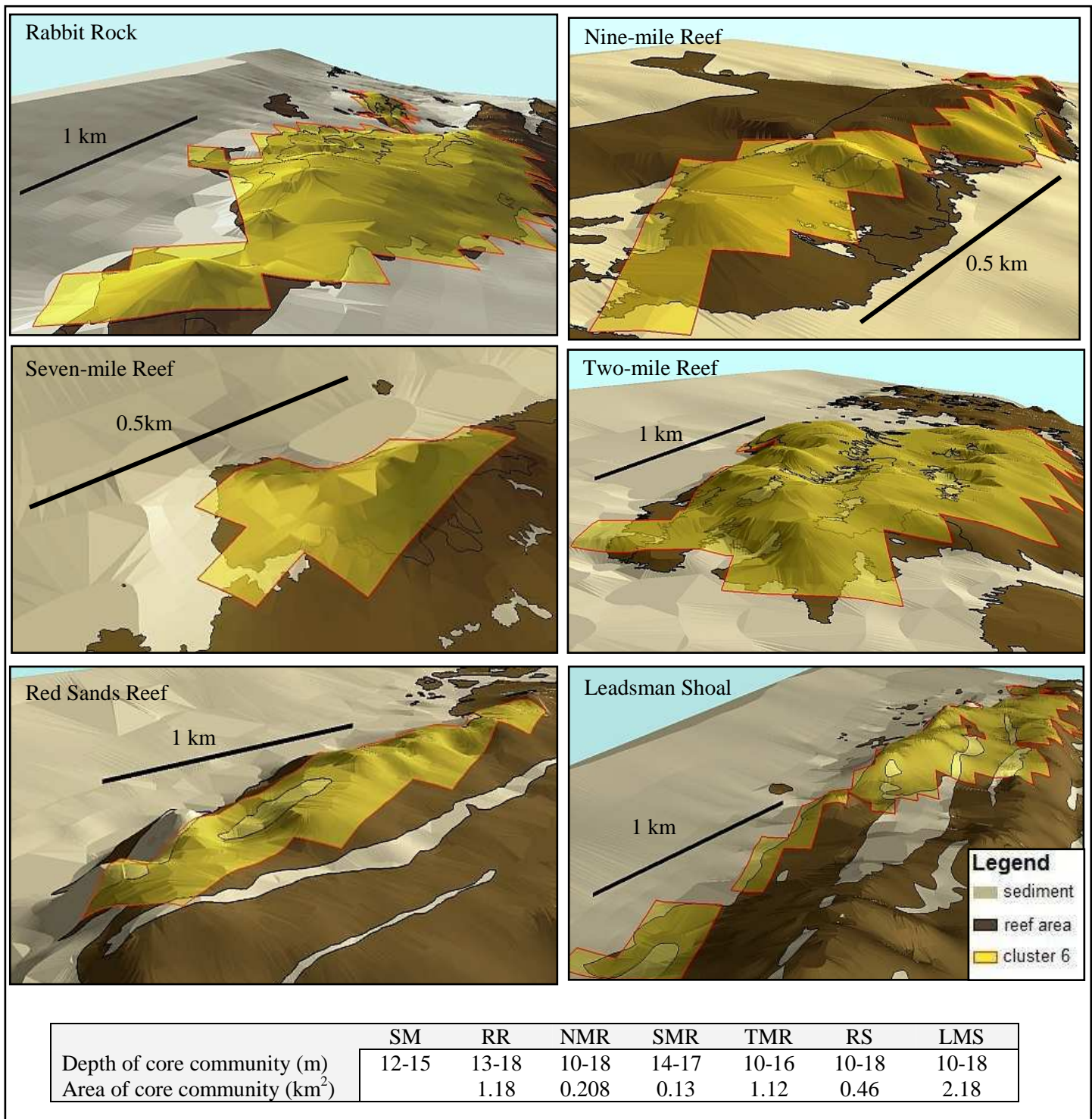


Figure 2.5 Geospatial 3-dimensional maps of the South African study reefs showing the varied beachrock topography (data extracted from Ramsay et al. (2006)) and the extent and area of the dominant coral community (Cluster 6; (Celliers & Schleyer 2008), see 2.1.5 below). Data for Shallow Malongane Reef were unavailable.

The topography of the reefs is variable (Fig 2.5) and lack most geomorphological features of true tropical reefs such as reef crests and steep reef slopes. The major topographical features are gullies, pinnacles and reef tops (Schleyer 2000) (Fig 2.6). Leadsman Shoal, Red Sands Reef and Two-mile Reef consist of shallow pinnacles (8-10 m), extensive deep subtidal reef flats (14-18 m) and a gently a sloping seaward edge (24-27 m) (Celliers & Schleyer 2008). Seven-mile Reef is a small table-like feature (12-15 m deep) with a prominent drop-off from 17 m to a larger, low relief reef at 22 m (Celliers & Schleyer 2008). Nine-mile Reef forms shallow platforms 6-18 m deep, and steep drop-offs from 12–20 m (Riegl et al. 1995, Schleyer 2000). Rabbit Rock is similar to Two-mile Reef, but has a greater average depth of 15 m. Shallow Malongane Reef is similar in topography to Two-mile Reef.



Figure 2.6 Representative reef (Leadsman Shoal) illustrating the typical gully and pinnacle topography of the study reefs.

2.1.5 The benthic communities

The coral reefs of Maputaland and southern Mozambique are the most southerly in the Western Indian Ocean (Riegl et al. 1995). The coral communities colonised the coastal beachrock and aeolianite outcrops subsequent to a glacial maximum sea-level rise (Ramsay 1994). The maximum coral thickness found on the central complex reefs is 30-40 cm (Ramsay and Mason, 1990).

Despite the marginal classification of the Maputaland and southern Mozambique coral reefs, corals are the dominant fauna on the reefs, contributing 50-70% towards the living benthic cover (Jordan & Samways 2001, Pereira 2003, Celliers & Schleyer 2008). The coral communities on these reefs consist of a rich mix of predominantly Indo-Pacific species (Pereira 2003, Celliers & Schleyer 2008). On the South African reefs, 46 hard coral genera and 11 soft coral genera have been recorded, representing a total of 133 coral species (Schleyer 2000, Schleyer & Celliers 2003). Nine of the soft coral species are endemic to the area (Schleyer & Celliers 2003). Thirteen soft coral genera and 40 hard coral genera have been recorded on the Ponta Malongane reefs (Robertson et al. 1996, unpub. data). It is anticipated that further investigations on the reefs will yield similar numbers of hard coral genera and species to the Maputaland reefs.

Celliers & Schleyer (2008) conducted a detailed community structure analysis on the South African coral reefs, which yielded 16 significantly different benthic communities. The most widespread and abundant coral community consists of a diverse blend of hard and soft coral species, with soft corals contributing more than 25% towards the living cover component (Table 2.1). This particular community appears to comprise the 'core' community on South African coral reefs (Celliers & Schleyer 2008) (Fig. 2.7). Colloquially referred to as 'Cluster 6', this coral community constitutes the most abundant benthic community type on the six South African study reefs (Fig 2.5).

Previous studies have reported similarities between the South African reefs and southern Mozambican coral community structure (Robertson et al. 1996, Pereira 2003). A comparison between the South African core coral community and the most abundant coral community on Shallow Malongane supports these observations (Table 2.1).



Category, Genus	Percent cover
Massive corals	9.9
<i>Platygyra</i> spp,	1.9
<i>Favia</i> spp,	1.2
<i>Favities</i> spp	1.7
Submassive corals	1.6
Encrusting corals	10.0
<i>Montipora</i> spp	6.6
<i>Acropora</i> spp	5.0
<i>Lobophytum</i> spp	6.9
<i>Sinularia</i> spp	20.7
<i>Sarcophyton</i> spp	3.0

Figure 2.7 Representative view of the core coral community (Cluster 6; Celliers & Schleyer 2008) on Leadsman Shoal with its rich mix of hard and soft coral species. The predominant coral morphologies and genera are listed in terms of their percent contribution to total benthic cover (Celliers & Schleyer 2008).

Table 2.1 Comparisons of the predominant benthos on the South African and southern Mozambican study reefs (Schleyer et al. 2008). Data for Shallow Malongane are from Schleyer et al. (in prep).

Percentage cover	Hard corals		Soft corals	
South African	25.9		33.2	
Shallow Malongane	23.6		25.6	
Major coral taxa	<i>Sinularia</i> spp	<i>Lobophytum</i> spp	<i>Montipora</i> spp	<i>Acropora</i> spp
South African	20.8	7.6	7.0	5.3
Shallow Malongane	23.0	5.3	6.2	4.3

2.1.6 Marine Protected Area zonation and human resource use

The St Lucia MPA was declared in 1979 (Notice P 35/79), includes a shoreline of 73 km and covers a total area of 414 km². The Maputaland MPA was proclaimed in 1986 (Notice GN 404/86), includes a similar length of shoreline (72 km) and covers a total area of 408 km². Both MPAs extend 3 km offshore. These MPAs constitute the greater marine component of the iSimangaliso Wetland Park, a World Heritage Site (Act No. 49 of 1999). This protected area was formally known as the Greater St Lucia Wetland Park (GSLWP) and underwent a name change in 2007. Although the MPAs are covered by dual legislation, the conservation enforcement authorities (Ezemvelo KwaZulu-Natal Wildlife) in the St Lucia and Maputaland MPAs follow the Marine Living Resources Act (MLRA) regulations because, legally, the MLRA has precedence over the World Heritage Convention Act (Lemm & Attwood 2003).

In terms of the MLRA, the St Lucia and Maputaland MPAs are zoned into restricted and sanctuary zones. The sanctuary zones prohibit all human activities and are considered no-take zones. In the restricted zones, recreational fishing (boat and shore-based), spearfishing and SCUBA diving are permitted. These zones may be classified as multiple resource use zones. All boat-based fishing activities are restricted to gamefish species. According to the MLRA, 'gamefish' are pelagic bony fish of the families Scombridae, Carangidae, Pomatomidae, Coryphaenidae, Rachycentridae, Xiphiidae, Ostiophoridae and Sphyraenidae, the species *Aprion virescens*, as well as pelagic cartilaginous fish of the families Carcharinidae, Isuridae, Sphyrnidae, Alopiidae and Odontaspidae (Section 3.1 (G) Regulation R1429). In line with the MLRA, a permit is required to SCUBA dive in a MPA, while all marine recreational fishing activities require a permit, regardless of the locality in South Africa.

The coral reefs have been divided into three reef complexes; the Northern, Central and Southern Reef Complexes (Riegl et al. 1995) and resource use varies between each reef complex (Table 2.2). The sanctuary zones include all reefs within the Southern Reef Complex and certain reefs in the Northern Reef Complex. The reefs in the Central Reef Complex are subjected to high levels of human extractive and non-extractive resource use. Recreational fishing, spearfishing and SCUBA diving are the most common activities on the reefs. Only SCUBA diving is permitted on Two-mile Reef (TMR); however, the diving intensity on this reef is 18 times higher than on the other reefs. This high diving intensity is attributed to the close proximity of TMR to the launch site, Jesser Point. The remaining study reefs in the Central Reef Complex (Seven-mile and Nine-mile Reef) are subjected to lower levels of diving intensity. Recreational fishing for gamefish species is permitted on these reefs.

The diving intensity (number of divers per annum) for this study was based on 2007-2008 statistics. SCUBA diving numbers for South African reefs were obtained from the conservation authorities (Ezemvelo KwaZulu-Natal Wildlife) responsible for enforcing the MPA regulations (Pieters 2009). Diving intensity in on Shallow Malongane Reef (SM) was obtained from the only dive resort at Ponta Malongane (Parque de Malongane pers.comm). Due to the absence of formal coastal management in southern Mozambique in the past, these are the most accurate data available.

Table 2.2 MPA zonation of the seven study reefs and the types of human activities permitted in each zone (Extracted and adopted from the KwaZulu-Natal Wildlife Nature Conservation Services Marine Zone 2003 Management Plan for the Greater St Lucia Wetland Park).

Location	Reef	MPA zone	Resource use	
			SCUBA diving	Fishing
Ponta Malongane	SM	N/A	SCUBA diving (4500 dives/year)	Unrestricted recreational
Northern Complex	RR	Sanctuary	Nil	Nil
Central Complex	NMR	Multiple use	SCUBA diving (1400 dives/year)	Restricted recreational (gamefish species only)
	SMR	Multiple use	SCUBA diving (2800 dives/year)	Restricted recreational (gamefish species only)
	TMR	Multiple use	SCUBA diving (54 000 dives/year)	Nil
Southern Complex	RS	Sanctuary	Nil	Nil
	LMS	Sanctuary	Nil	Nil

When the present study was undertaken, southern Mozambique had no MPAs protecting the coral reefs at Ponta Malongane. However, two decrees pertaining to marine resources are relevant to the coral reefs. Article 61 (Decree n. 45/2006 of 30 November) prohibits all activities that may damage coral or coral reefs or the biodiversity that is characteristic of coral reefs. The Recreational and Sport Fishing Regulation (Decree 51/99 of 31 August) requires users to obtain a licence in order to participate in such activities. This decree also provides complete protection to vulnerable species and imposes daily bag limits on selected target species (Table 2.3).

Table 2.3 Details of the Recreational and Sport Fishing Regulation in Mozambique (Decree 51/99 of 31 August).

Fully Protected species	
<i>Carcharodon carcharias</i>	<i>Polysteganus undulosus</i>
<i>Epinephelus lanceolatus</i>	<i>Petrus rupestris</i>
<i>Epinephelus tukula</i>	
Restricted species	Daily allowance
Sharks	2
Scarids	1
Serranids	4
Sparids (<i>Chrysoblephus punceus</i> , <i>Cheimereius nufar</i> , <i>Polysteganus coeruleopunctatus</i>)	4

CHAPTER 3

A BASELINE ASSESSMENT OF SOUTH AFRICAN CORAL REEF FISH COMMUNITIES: COMPARISONS BETWEEN REEFS AND BETWEEN REEF PROTECTION STATUS

3.1 Introduction

Monitoring programmes often focus on the most prominent and ecologically important fauna in an ecosystem (Linton & Warner 2003). On coral reefs, these are undoubtedly fish and coral species. Fish are the most diverse vertebrate group on earth (Smith & Heemstra 1986) and are considered by some to be the best-studied marine taxon (Knowlton & Jackson 2008). Yet, fish assemblages in certain parts of the world still remain comparatively under-documented (Garpe & Öhman 2003). One such area is the Western Indian Ocean (WIO); a discrete subregion of the Indian Ocean (Sheppard 1987). Ecosystem and species diversity are high in this region (Sheppard & Wells 1988) which, although largely explored, remains the area in which coral reef fish are the least studied in the world (Heemstra et al. 2004).

In the WIO, coral reefs and their associated fauna provide benefits to tens of millions of people through tourism revenue (Ahamada et al. 2004) and as sources of food (Obura et al. 2004). The estimated economic value in the form of goods and services provided by coastal habitats such as coastal and mangrove forests, coral reefs and seagrass beds is over US\$25 billion per year (WIO-Lab). Certain areas of the WIO have also received more attention than others, as well as certain coral reef taxa. Check-lists of coral species and quantitative descriptions of coral communities in the WIO are well documented in the literature (Pichon 1972, Faure 1977, Hamilton & Brakel 1984, Lemmens 1993, Riegl 1993, Hoeksema & Borel-Best 1994, Riegl et al. 1995, McClanahan et al. 1999, Schleyer 2000, Muhando & Mohammed 2002, Pereira 2003, Schleyer & Celliers 2003, Obura et al. 2006a, Obura et al. 2006b). Numerous studies of fish communities are available. However, many of the data are qualitative in nature and lack detailed ecological information. Inventories of fish assemblages have been compiled for southern Madagascar (Harmelin-Vivien 1979), Maldives (Randall & Anderson 1993), Mauritius (Adjeroud et al. 1998), the Mascarene Archipelago (Fricke 1999), Mozambique (Pereira 2000), Mayotte (Chabanet 2002), Glorieuses Islands (Durville et al. 2003), Reunion Island (Letourneur et al. 2004), Rodrigues (Heemstra et al. 2004), Juan De Nova (Chabanet & Durville 2005) and the sub-tropical east coast of South Africa (Chater et al. 1993,

1995). Some studies have provided quantitative fish data for the region; however, they have largely focused on the affects of anthropogenic impacts on selected families of reef fish and have not yielded species inventories (Jennings et al. 1995, McClanahan & Kaunda-Arara 1996, McClanahan & Arthur 2001, Mohammed 2002).

The coral reefs of the WIO constitute a wide diversity of structures from oceanic atolls and fringing reefs in the tropics to marginal coral reefs at higher latitudes (Sheppard 2000). Marginal coral reefs are those that exist near the limits of their environmental tolerance or latitudinal distribution (Kleypas et al. 1999). In the WIO, high-latitude coral communities are found in marginal environments in South Africa (Kleypas et al. 1999) and southern Mozambique. These coral ecosystems are among the southernmost coral reefs in the world (Ramsay 1996). Despite being at the latitudinal limit of coral distribution in the WIO region, they attain high biodiversity south of the equator (Benayahu & Schleyer 1995, 1998).

The coral communities of South Africa were first investigated in the 1970s (Heydorn 1972, Ballard 1973) and have subsequently received considerable attention (see Schleyer & Celliers 2003 for review). The fish communities by comparison were only investigated 20 years later by Chater et al. (1993) who provided the first check-list of the reef-associated fish assemblages on selected reefs. Realizing the importance of providing reef managers with detailed fish community data, Chater et al. (1995) returned to the reefs to conduct quantitative surveys on the abundance of the reef fish assemblages. In their first study, Chater et al. (1993) highlighted the problems associated with multi-species surveys and only thirteen families of fishes, typical of reefs in the area, were chosen for the subsequent (1995) quantitative study.

Since the end of the Mozambican civil war in 1992, several studies have investigated the biodiversity of coral reefs in the Ponta Malongane area. Robertson et al. (1996) and Pereira (2000) were among the first to collect data on reef fish species. Pereira (2003) conducted a more detailed investigation on the effects of recreational SCUBA diving on selected coral reefs in the region. To date, no studies have examined the effects of SCUBA diving and recreational fishing on the fish communities on coral reefs in southern Mozambique.

The aims of this investigation were three-fold. The first aim was to describe the fish communities on South African coral reefs in term of their abundance, diversity and trophic ecology and compare these parameters between each reef to provide baseline comparisons for long-term monitoring studies. The second aim was to compare these fish community metrics between reefs in terms of their differing protection status. The purpose was to investigate whether patterns observed in the

fish communities may be related to human activities such as fishing and diving or environmental factors. The third aim was to use these baseline data to validate the selection of 25 indicator species that were used to investigate the effects of human activities on reef fish communities. This is dealt with in Chapter 4.

3.2 Materials and methods

3.2.1 Study site description

See Chapter 2 for details.

Levels of reef protection

A description of human resource use and reef protection on the seven study reefs is listed in Table 3.1. The Central Reef Complex is the focal point of marine tourism in the iSimangaliso Wetland Park. For the purpose of this Chapter, these reefs were classified as ‘Protected’ reefs. All the reefs in the Southern Reef Complex and certain parts of the Northern Reef Complex are categorised as no-take zones where no human activities are permitted. In this study, these reefs were termed ‘Sanctuary’ reefs and included Rabbit Rock (RR), Leadsman Shoal (LMS) and Red Sands Reef (RS). Due to the lack of law enforcement on the Ponta Malongane reefs, Shallow Malongane Reef (SM) was termed ‘Open’.

Diving intensity was separated into three categories. Reefs that experienced more than 50 000 dives per year were deemed to have high diving intensity, which included only TMR. Low diving intensity was deemed to occur on reefs that are subjected to less than 10 000 dives per year and this included SM, SMR and NMR. The zero-diving intensity category included Leadsman Shoal (LMS), Red Sands Reef (RS) and Rabbit Rock (RR).

Fishing intensity was also separated into three categories. High fishing intensity occurs on reefs where there are fishing regulations, but the lack of enforcement by marine officers and low compliance by fishers resulted in unrestricted fishing. This category included only SM. Low fishing intensity occurs on reefs where only the removal of bait and gamefish is allowed, which included SMR and NMR. The zero-fishing intensity category included reefs where all types of fishing were prohibited, comprising TMR, LMS, RR and RS.

Table 3.1 Description of human resource use and reef protection on the seven study reefs. SCUBA diving statistics are averages for the period 2007-2008. Reefs are ordered north to south.

Reef name	Reef protection status	Human activities	
		Diving	Fishing
Shallow Malongane (SM)	Open	Low (4500 dives/year)	Unrestricted
Rabbit Rock (RR)	Sanctuary	Nil	Nil
Nine-mile Reef (NMR)	Protected	Low (1500 dives/year)	Restricted (gamefish species only)
Seven-mile Reef (SMR)	Protected	Low (2900 dives/year)	Restricted (gamefish species only)
Two-mile Reef (TMR)	Protected	High (54 000 dives/year)	Nil
Red Sands Reef (RS)	Sanctuary	Nil	Nil
Leadsman Shoal (LMS)	Sanctuary	Nil	Nil

3.2.2 Surveys of reef fish communities

During the period January 2008 to February 2009, five fieldtrips were undertaken to the iSimangaliso Wetland Park and two fieldtrips to southern Mozambique. Data were collected using SCUBA and all diving operations were conducted from a semi-rigid inflatable vessel. Study sites on were randomly selected using geo-spatial data on the reefs from Celliers & Schleyer (2008) (Fig. 3.1). These data allowed the survey sites to be selected within the dominant coral community type ('Cluster 6' see Chapter 2) and required depth range.

Surveys of fish diversity and abundance were undertaken using point count underwater visual census (UVC) techniques adapted from Samoily and Carlos (2000). The point count technique was selected because it is effective in estimating abundances of mobile species (Samoily & Carlos 2000) and is suitable for the varied topography of South African coral reefs. It was anticipated that some degree of underestimation would be incurred due to the nature of UVC sampling techniques (see Chapter 1). Thus the fish assessments were focused on diurnally active fish species.

Each census (community count) consisted of a 60 minute timed swim that covered a large circular area of reef and was equivalent to one dive. Fish abundance was subsequently quantified per unit time (one hour) as opposed to per unit area. The aim of the community counts was to enumerate as many species as possible. The search time was maximised due to the expected high number of fish species. A total of 77 community counts were completed, with 10-12 counts conducted per reef (Table 3.2). All fish taxa observed on a reef and within the water column during the counting period were recorded. Fish were identified to species level. Where necessary, identification of species was aided by identification guides (Smith & Heemstra 1986, King 1996) and websites (Fishbase 2009).

Only two divers entered the water at any one time, a data recorder and a buddy diver. To minimise the impact of diver disturbance caused by the divers entering the water, the data recorder waited five minutes before beginning each count. The same diver conducted all community counts to minimize variation and error incurred by diver bias. Community counts on each reef were separated by at least 100 m. The time of each community count was conducted between 0800 and 1400. All counts were conducted in the depth range of 12-15 m.

Table 3.2 Summary of survey dates and number of fish community counts conducted on the seven study reefs on South African and southern Mozambican coral reefs during 2008-2009. Each community count represents one dive. Reefs are ordered from north to south. The total numbers of community counts per reef are highlighted in bold.

Reef	January 2008	February 2008	June 2008	July 2008	September 2008	December 2008	February 2009	Total
SM	-	-	-	-	5	-	6	11
RR	-	6	-	6	-	-	-	12
NMR	1	1	2	4	-	2	2	12
SMR	1	-	-	4	-	2	3	10
TMR	2	-	-	3	-	2	3	10
RS	3	-	6	-	-	2	1	12
LMS	2	-	6	-	-	-	2	10

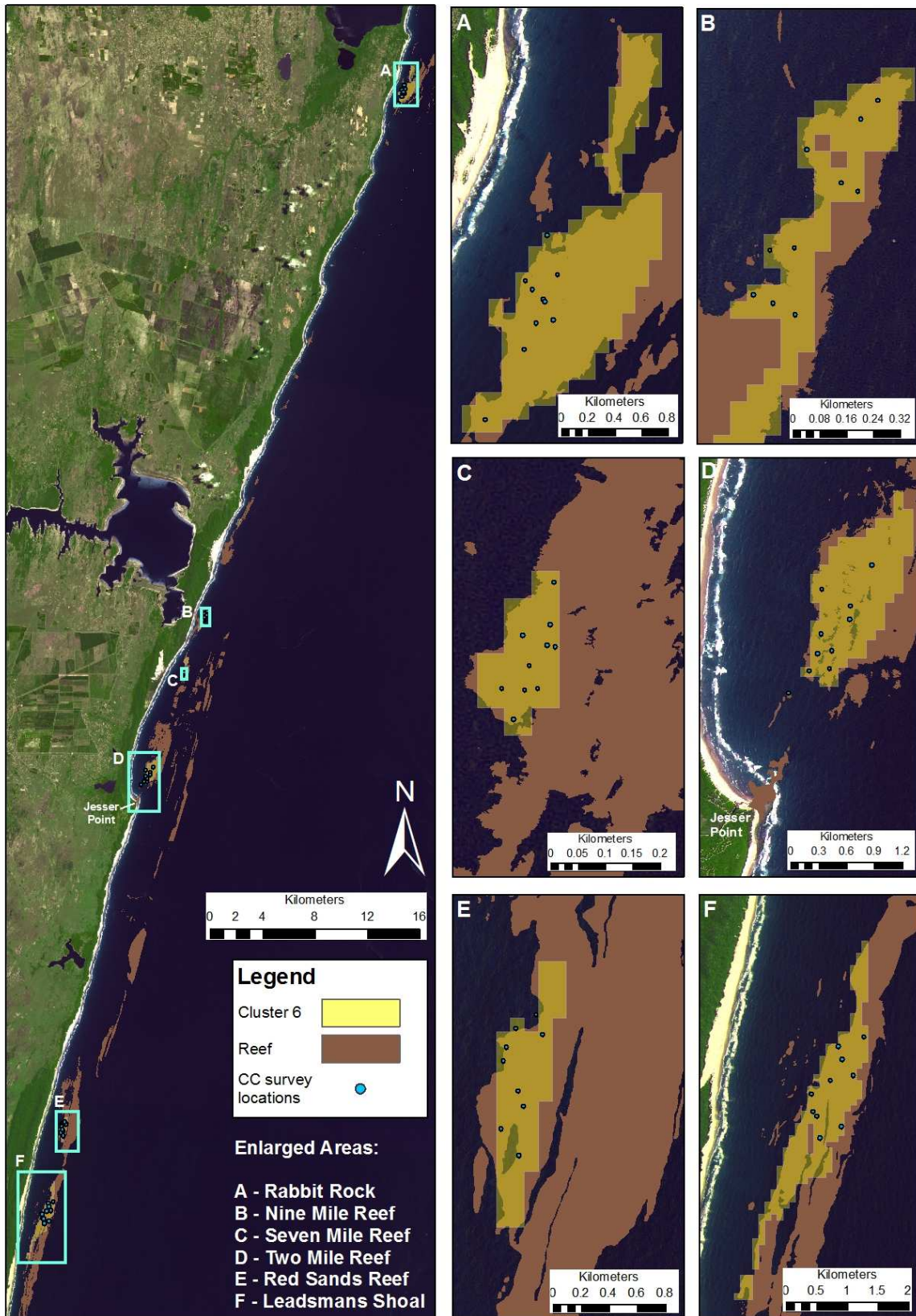


Fig. 3.1. Location of survey sites on the six South African study reefs. Geo-spatial data are from Celliers & Schleyer (2008). No data were available for the southern Mozambican reef.

Trophic levels

Species were allocated to one of nine trophic groups; top-level predators, medium-level predators, low-level predators, planktivores, omnivores, herbivores, benthivores, corallivores and invertivores. The allocation of each species to a trophic guild was done according to diet information retrieved from the FishBase website (<http://www.fishbase.org>) and supplemented by field observations.

Predators were defined as species that feed predominantly on other fish. This category was subdivided into three categories based on body size as it has been proposed to determine the vulnerability of species to exploitation (Gislason 2002). Larger-bodied species, although more fecund, are slower growing and reach sexual maturity at a later age than smaller-bodied species (Hutchings 2002). Not only are larger-bodied species vulnerable to exploitation due to their life history traits (Dulvy et al 2004a) they also tend to be more heavily targeted by fishers due to size selective fishing (Pauly et al. 1998, Pinnegar et al, 2002, Berkeley 2004).

Small predators such as cirrhitids (hawkfish) and small lutjanids (snappers) were included in the low-level predator category, while species such as carangids (excluding *C. ignobilis*) and smaller-bodied serranids (rockcods) were categorised as medium-level predators. Top-level (apex) predators were defined as species with no predator, residing at the top of the food web and included large species such as sharks and giant serranids (e.g. *Epinephelus tukula*).

Benthivores were specialist consumers of sponges and ascidians. Invertivores included species that feed on crustaceans, echinoderms and polychaetes. Planktivores were fish species that feed on plankton in the water column. Omnivorous species were those species that were non-selective in their feeding habits. Herbivores were those species that feed predominantly on algae. Lastly, corallivores were those species that feed exclusively on coral polyps or coral mucus.

Measuring environmental variables and habitat characteristics

Depth and topography were recorded during each fish community count. In addition, hourly water temperatures were obtained from an underwater temperature recorder stationed on NMR at a depth of 18 m. Depth was measured using a dive computer and recorded as a mean of six depth readings during each community count. Topographic complexity for this study referred to the structural variation of the underlying bedrock and was assessed as the height of the substratum above the sand. An area of reef with pinnacles more than 2 m high was categorised as having high topographic complexity. Medium topography complexity described a reef area with substratum between 1-2 m high and low complexity areas had substratum less than 1 m in height. A three point

scale was used because the community counts were conducted over a large area and a rapid method was needed to describe the topography. Coral cover was not included as a habitat variable in these analyses due the large spatial scale and because all fish surveys were conducted within the same benthic community type ('Cluster 6', see Chapter 2 for details).

3.2.3 Statistical analysis

The stepwise process of the statistical analyses used in this study is diagrammatically represented in Figure 3.2.

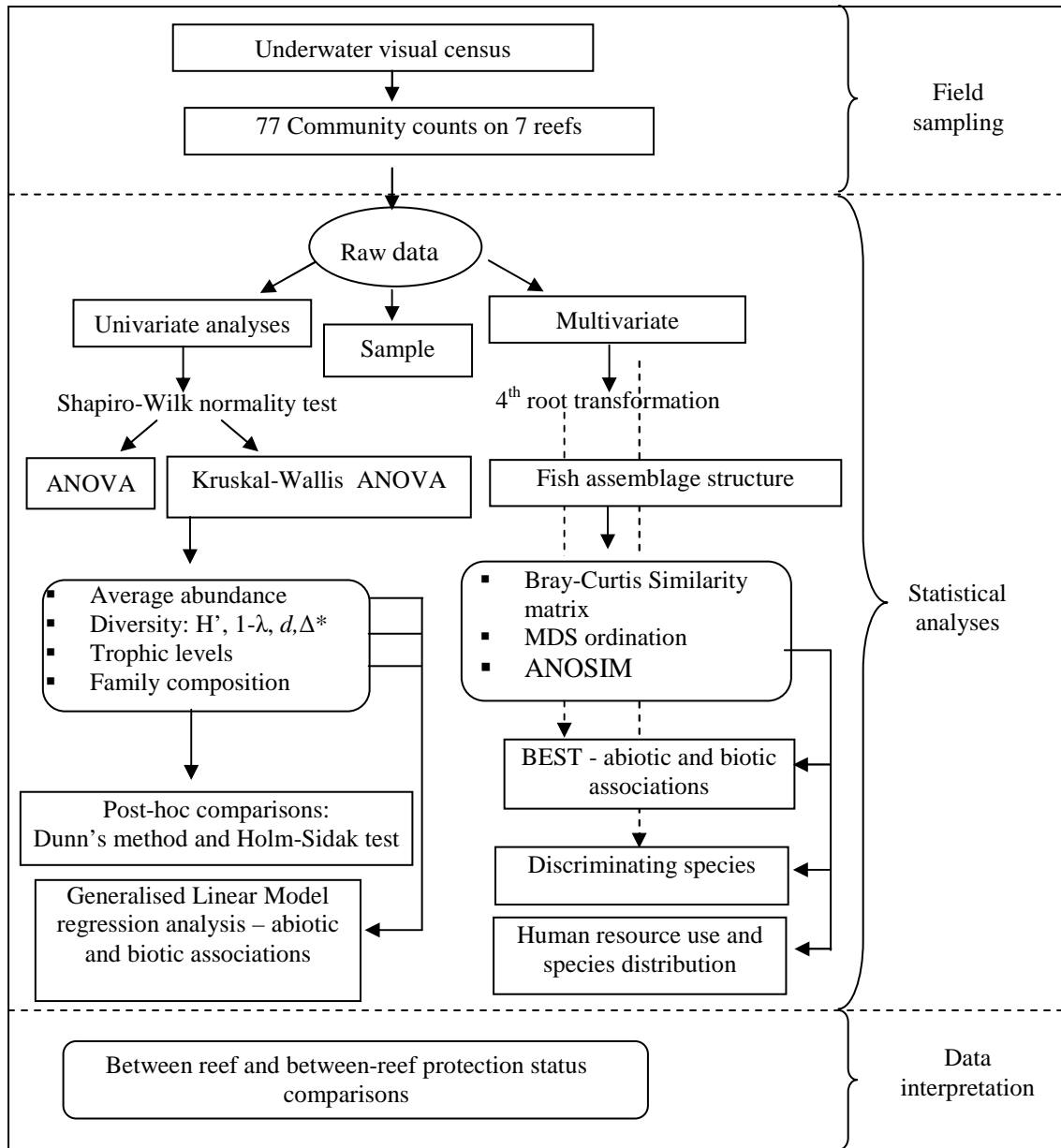


Figure 3.2 Diagrammatic summary of the sampling protocol and data analyses undertaken in the assessment of fish communities on coral reefs in South Africa and southern Mozambique.

Sample size test

Power analysis was used to determine whether the mean of 11 community counts conducted per reef were sufficient to provide the statistical validity needed to detect changes in population estimates. The power analysis technique was used according to Kapadia et al. (2005). The minimum sample size required was calculated as that required to detect a 10% change in the mean estimate at a significance level of 0.05 and a power of 80% (Fairweather 1991, Length 2001).

Univariate analysis

Univariate analyses were carried out using one-way analysis of variance (ANOVA). Before proceeding with an ANOVA, the data were tested for normality using the Shapiro-Wilk test. If data were not normally distributed, non-parametric Kruskal-Wallis (K-W) one-way ANOVA on ranks was used. This method of comparing different groups does not require the assumption that all samples are drawn from normally distributed populations with equal variances (Analyze-It 2008). If differences between fish abundance parameters were detected, pairwise multiple comparison procedures were employed. Two *post-hoc* tests were used to detect differences; Dunn's method and the Holm-Sidak test (Analyze-It 2008). Dunn's method is appropriate for K-W ANOVA on ranks when the sample sizes in the different treatment groups are different. The Holm-Sidak test is more powerful than the Tukey and Bonferroni test and, consequently, it is able to detect differences that the Bonferroni test cannot (Analyze-It 2008). Univariate analyses were conducted using the statistical package Sigma Plot 11.0 (2008), Analyze-It (2008) and GenStat 12.1 (2009) software. The criterion for significance of all tests was $p \leq 0.05$.

All analyses were conducted using abundance data for between-reef and within reef protection status comparisons. Abundance data were estimated as number of fish per unit hour of the survey. Trophic levels and family composition comparisons were examined. To study the changes in species diversity the following indices were used:

Margalef's richness index (d) (Margalef 1958):

$$d = (S-1)/\log_e N,$$

where S is the total number of species and N is the total number of individuals in the sample.

Simpson's evenness index (1-λ) (Simpson 1949):

$$1-\lambda = 1 - [\sum_i N_i(N_i-1)] / [N(N-1)],$$

where N_i is the number of individuals of the i th species in the sample.

(3) Shannon's diversity index (H') (Shannon 1949):

$$H' = -\sum p_i(\log p_i),$$

where p_i is the total count of each sample represented by the i th species.

Generalised linear models (GLM) were used to examine the influence of various habitat characteristics and independent variables on fish community parameters. GLMs were chosen because the data included continuous (temperature and depth) and categorical (topography, fishing intensity and diving intensity) independent variables. Regression models were tested for a number of dependent variables, which included total fish abundance, number of species, Margalef species richness, Shannon diversity, and fish abundance with trophic levels. Total fish abundance data were fourth-root transformed.

Multivariate analysis

All multivariate analyses were analysed using PRIMER v.6 (Clarke & Gorley 2006).

Transformation

All multivariate analyses were conducted using Bray-Curtis dissimilarities on fourth-root transformed abundance data. The degree of transformation was determined by a simple linear regression of the log of the standard deviation of each abundance mean against the log of the mean (Clarke & Warwick 2001). The slope of the linear equation determines the degree of transformation necessary. Figure 3.3 gives an example of the linear regression using data from NMR. Due to the high number of zeros in the abundance data, the purpose of the transformation was to define the balance between contributions from common and rarer species, and not to achieve normality. The fourth-root transformation is a more severe transformation (Clarke & Gorley 2006). However, it provides a means of down-weighting the importance of highly abundant species so that the sample similarities depend equally on less common species (Gillibrand et al. 2007). All comparisons were between individual reefs and between reef protection status.

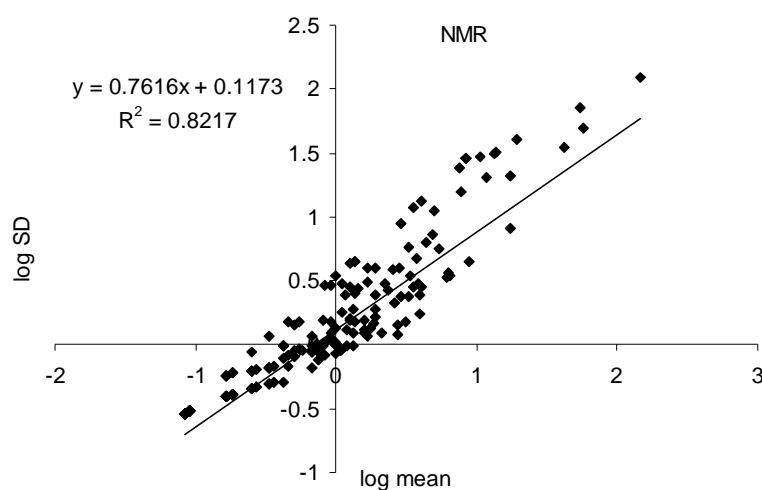


Figure 3.3 Linear regression of the log of the standard deviations of non-zero means against the log of mean fish abundance on NMR

Community analysis

Non-metric multidimensional scaling (MDS) was used to examine differences in community species composition. Stress values in the MDS plots tended to be high (>0.2). In order to present accurate descriptions of the observed patterns, MDS ordinations were included only after being compared with less sensitive multivariate analyses (e.g. cluster analysis). Analysis of similarity (ANOSIM) was also used to investigate differences in assemblage structure. An ANOSIM R-statistic of <0.25 implies that there is too much overlap for sites to be separable (Clarke & Gorley 2006). R-statistics > 0.5 were considered significant. Hierarchical agglomerative clustering with group-average linking was performed on the Bray-Curtis similarity abundance matrix to confirm trends observed in the MDS plots and ANOSIM results.

Discriminating species

SIMPER analysis was used to identify those species responsible for the Bray-Curtis dissimilarity between reefs and between-reef protection status. Due to the high number of species, only those species contributing 33% to the species accumulation were included. SIMPER may also be used to determine discriminating species between sampling areas. The main objective of applying SIMPER to the data was to determine which species were responsible for the differences in fish community structure i.e. which species are good discriminating species. In order to determine this, it is necessary to examine the contribution of each species to the dissimilarity between reef fish populations as well as the average abundance of the species and the dissimilarity-standard deviation ratio (Clarke and Warwick 2001).

Environmental variables, human resource use and community composition

At each sampling site, two environmental variables and three habitat characteristics were recorded: depth, temperature, topography, fishing intensity and diving intensity. Potential relationships between spatial patterns in fish communities and these environmental variables and habitat characteristics were examined using the BEST procedure (PRIMER v6 2006). Among a set of independent variables (and habitat characteristics), the BEST procedure identifies the most influential variable or combination of variables which give rise to the largest rank correlation (p_s) (Clarke & Warwick 2001). All variables were normalised according to the requirements of the BEST procedure.

Human resource use and species distribution patterns

Further MDS plots were generated to examine the relationship between fish abundance and human activities, using the same fishing and diving intensity categories described above. Species abundance MDS plots were factored according to 'human activities' and certain discriminating

species identified by SIMPER were superimposed on the MDS abundance ordinations. Only those species that revealed an abundance pattern linked to human activities have been included in the results.

3.3 Results

3.3.1 Abundance

Figure 3.4 illustrates the total number of fish recorded per unit time on the individual reefs and for the reefs combined according to their protection status. Kruskal-Wallis one-way ANOVA on ranks revealed that there were no significant differences in total fish abundance between the individual reefs. Leadsman Shoal (LMS) had the highest fish densities and Shallow Malongane (SM) the lowest densities. Total fish abundance was highest on Sanctuary reefs and lowest on the Open reef. Differences between protection zones were not significant.

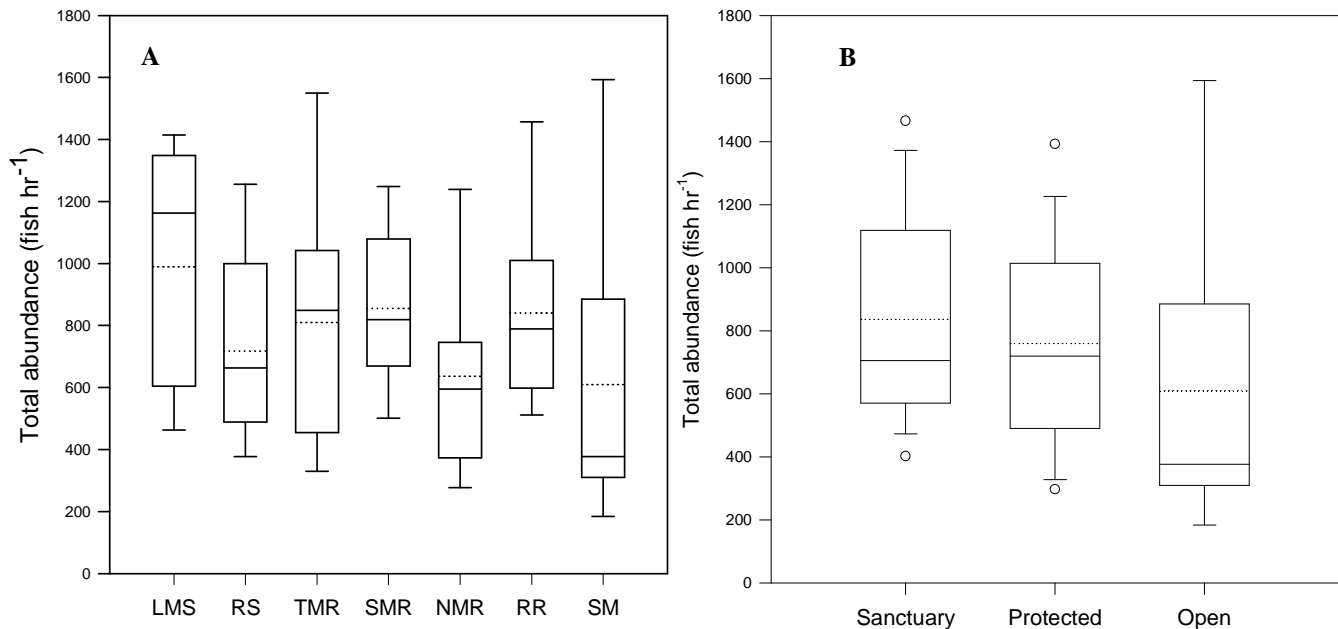


Figure 3.4 Boxplots (5th and 95th percentile) of mean (\pm SD) number of fish recorded in fish communities on the South African and southern Mozambique coral reefs per unit time on A) individual reefs and B) reefs combined according to reef protection status. Means are indicated by dotted lines and medians are indicated by solid lines. Reefs are ordered from south to north.

3.3.2 Spatial distribution of fish assemblages

The ANOSIM multivariate analyses (Table 3.3) demonstrated a large measure of overlap between sites on different reefs and that the separation between certain reefs was more distinct than others. SM was conspicuous as it was significantly different from all reefs except Two-Mile Reef (TMR) and Nine-Mile Reef (NMR) ($R=0.414$, $p<0.0005$). Fish abundance was also significantly different between Seven-Mile Reef (SMR) and all Sanctuary reefs. The multi-dimensional scaling (MDS) ordination echoed this pattern as SM formed a discrete cluster (Figure 3.5A). SMR also formed a cluster, but was only significantly different to LMS. When the factor 'protection status' was

overlaid on the abundance MDS plot (Figure 3.5B), three distinct groupings were evident. ANOSIM was again used to determine whether differences between-reef protection status were significant (Table 3.4). The results confirmed varying degrees of separation, but only Sanctuary and the Open reef were found to be significantly different from each other.

Table 3.3 Results of an analysis of similarity (ANOSIM) run on fourth-root transformed fish species abundance data for between reef differences. Global R=0.414. Significance of Global R <0.00005. Significant differences between reefs are in bold. R-statistics >0.5 were considered significant.

Pairwise Tests	R Statistic	Significance Level %
TMR, SMR	0.323	0.008
TMR, NMR	0.209	0.2
TMR, LMS	0.244	0.1
TMR, RR	0.305	0.003
TMR, RS	0.282	0.04
TMR, SM	0.45	0.001
SMR, NMR	0.355	0.007
SMR, LMS	0.664	0.001
SMR, RR	0.552	0.0002
SMR, RS	0.717	0.0002
SMR, SM	0.621	0.001
NMR, LMS	0.402	0.002
NMR, RR	0.352	0.0001
NMR, RS	0.424	<0.0005
NMR, SM	0.477	0.0003
LMS, RR	0.247	0.4
LMS, RS	0.134	4
LMS, SM	0.612	0.001
RR, RS	0.339	0.008
RR, SM	0.619	0.0002
RS, SM	0.703	0.0002

Table 3.4 Results of ANOSIM run on fourth-root transformed species abundance data for between-reef protection status differences. Global R=0.394. Significance of Global R <0.00005. Significant differences are in bold. P=Protected, S=Sanctuary, O=Open. R-statistics >0.5 were considered significant.

Pairwise Tests	R statistic	Significance level %
P, S	0.283	<0.0002
P, O	0.465	0.002
S, O	0.701	<0.0002

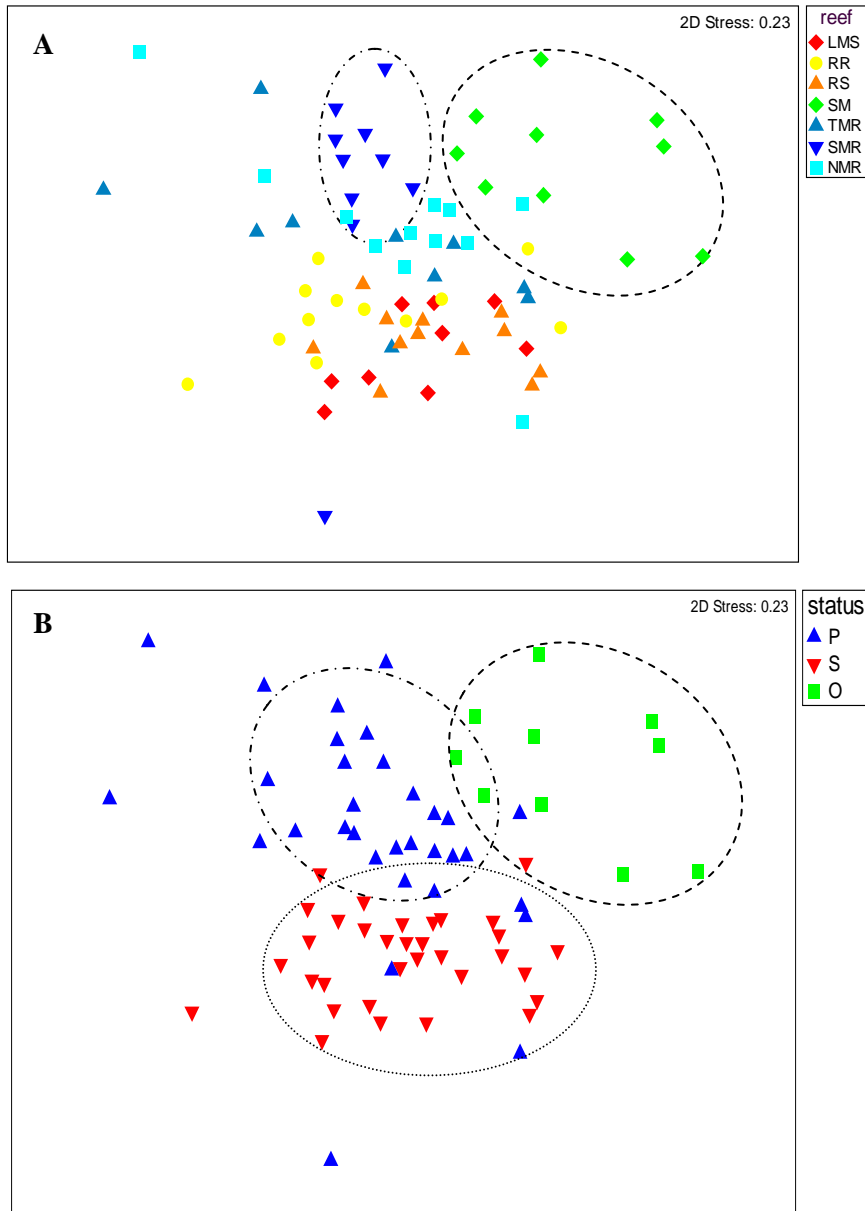


Figure 3.5 Non-metric multidimensional (MDS) ordination of samples based on fourth-root transformed fish species abundance showing groupings according to A) reef and B) protection status. P=Protected, S=Sanctuary and O=Open.

Superimposition of the MDS ordination plots on the Bray-Curtis similarity cluster further emphasized the relationships between the sample sites on each reef. Sanctuary reefs and the Open reef formed distinct groups. Cluster analysis confirmed the trend that sample sites clustered according to their protection status.

3.3.3 Fish community characteristics

Diversity Indices

A total of 284 fish species were recorded on the seven study reefs (Appendix 1). Of these, nine species were cartilaginous (Class Chondrichthyes) and the remaining 275 were bony fish (Class Osteichthyes). There was a large variation in diversity indices between reefs (Table 3.5). The highest total number of species was recorded on RR. This reef also had the highest Margalef's (species richness) index. NMR had the highest diversity and Simpson's evenness index between reefs. SM had the highest taxonomic distinctness, but the lowest total number of species. Despite the large differences in diversity indices, significant differences were found only in terms of taxonomic distinctness between SM-TMR and SMR-TMR.

Table 3.5 Results of diversity indices (number of species, Margalef's richness, Shannon's diversity, Simpson's evenness index and Taxonomic Distinctness) for each individual reef. The highest values for each index are highlighted in bold.

Diversity index	LMS	RS	TMR	SMR	NMR	RR	SM
Number of species (S)	181	168	189	179	188	197	172
Margalef's richness (d)	11.85	10.89	10.93	11.73	11.22	12.14	10.94
Shannon's diversity (H')	2.95	2.94	2.81	2.98	3.04	2.99	2.82
Simpson's evenness (1-λ)	0.88	0.87	0.87	0.89	0.9	0.88	0.82
Taxonomic Distinctness (Delta*)	56.45	58.11	55.61	60.46	58.21	59.81	60.6

Diversity data were combined according to reef protection status and compared using the same diversity indices (Table 3.6). Shannon diversity and Margalef's species richness were highest on Sanctuary reefs. The total number of species and Margalef's richness index was highest for protected reefs. Taxonomic distinctness continued to demonstrate a contrasting pattern with highest Delta* being found on the Open reef, SM. One-way ANOVA revealed that the only significant difference was found in the average number of species between Sanctuary and Open reefs ($p=0.038$).

Table 3.6 Results of diversity indices (number of species, Margalef's richness, Shannon's diversity, Simpson's evenness index and Taxonomic Distinctness) for Sanctuary, Protected and Open reefs. The highest values for each index are highlighted in bold.

Diversity index	Sanctuary	Protected	Open
Number of species (S)	238	242	172
Margalef's richness (d)	35.22	36.34	26.67
Shannon's diversity (H')	3.822	3.684	3.157
Simpson's evenness (1-λ)	0.9499	0.9362	0.8677
Taxonomic Distinctness (Delta*)	58.79	58.77	60.83

The fish abundance on each reef was dominated by a small number of species which, when combined, contributed more than 40% to the over all species abundance: *Chromis dimidiata*, *Pseudanthias squamipinnis*, *Chromis weberi*, *Parapriacanthus ransonneti* and *Nemanthias carberryi*. However, not all of these species were present on each reef. *C. dimidiata* was the most abundant species on RR (12%), TMR (14%), RS (16%), and LMS (18%). The fish abundance on SMR and NMR was dominated by *P. squamipinnis*, which contributed 20% and 23% to the fish abundance respectively. On SM, *P. ransonneti* was the most abundant species contributing 32% to the overall fish abundance. Interestingly, *P. ransonneti* was absent from SMR and RS. Similarly, *N. carberryi* was absent from LMS, RS, RR and SM.

Family composition

A total of 50 fish families were recorded on the seven study reefs. The families showed differences in abundance and species composition of fish between the reefs. Table 3.7 lists the six families that included the most species observed on the reefs, contributing at least 50% to the overall species composition. The greatest number of species belonged to the labrid family, which contained almost twice the number of species than the next biggest family, the Acanthuridae. Of the remaining 50 families, between 11 and 20 families were represented by only one species.

Table 3.7 Percentage of fish species within families on the seven study reefs. Only families with the most species on each reef have been included. The combined species in the listed families contributed 55% to the overall fish species composition.

Family	LMS	RS	TMR	SMR	NMR	RR	SM
Acanthuridae	21	16	18	19	16	20	17
Chaetodontidae	15	13	14	12	13	17	13
Labridae	33	30	30	30	35	34	30
Lutjanidae	10	11	11	13	9	12	8
Pomacentridae	12	12	12	9	12	12	12
Serranidae	12	10	12	12	14	13	7

In terms of abundance, family composition per reef manifested much more variation than the species count per family (Figure 3.6). The Pomacentridae dominated the family community structure on five of the seven reefs; LMS, RS, TMR, NMR and RR. SM was dominated by the Pempheridae and SMR by the Serranidae. The substantial contribution of the Serranidae to the family composition on all reefs except RR and SM was attributable to an abundance of species of the subfamily Anthiinae, which includes seagoldies.

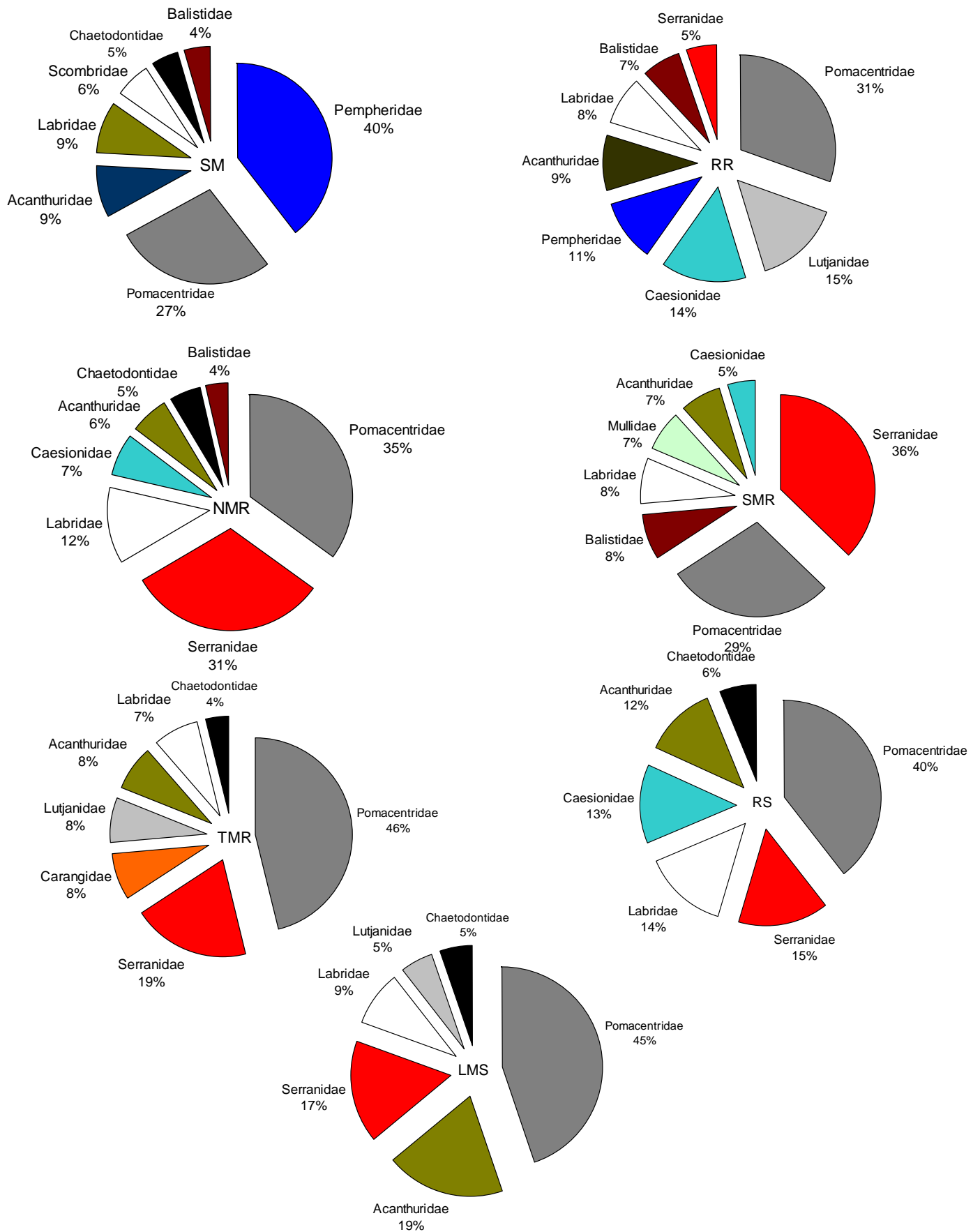


Figure 3.6 Percent composition of coral reef fish communities at family level on the seven study reefs in South Africa and southern Mozambique. Only families contributing 80% towards overall abundance on each reef have been included.

Trophic levels

The most abundant trophic category on all reefs was the planktivores (Table 3.8), ranging from 400 fish/h at NMR to 601 fish/h at LMS. The invertivores were the next most abundant trophic level and abundances in this group varied greatly between reefs. Top predators were most abundant on RR and uncommon on SM. High level predators were also most abundant on RR. High abundances of medium-level predators were recorded on RR and SM. However, the high value found on SM was attributable to a single sighting of a large number of *Scomberoides lysan*. If this species is omitted from the analyses, the abundance of medium-level predators on SM is reduced to 32.6 fish/h. Obligate coral feeders were most abundant on RS and herbivores ranged from very low numbers on SM (31 fish/h) to relatively high abundances on LMS (140 fish/hr).

Table 3.8 Mean abundance of the nine trophic guilds on each of the seven study reefs. Abundance is presented as the number of fish per hour. The highest abundance in each category is in bold.

Trophic level	LMS	RS	TMR	SMR	NMR	RR	SM
Top-level predator	3.11	2.33	0.7	1.6	1.33	4.25	0.2
Medium-level predator	30	20.58	24.4	29	16.67	55.25	62.6
Low-level predator	4.67	10.08	9.2	14.9	8.33	13.33	4.3
Invertivore	141	115.75	137.4	146.7	103.08	177.75	72.8
Benthivore	24.33	9.33	4.7	5.1	8.5	16.92	2.1
Corallivore	6.44	7.08	3.9	1.7	2.25	5.75	5
Planktivore	601.33	453.17	534.9	577.9	433.33	480.42	400.4
Omnivore	38.67	30	34.4	27.3	24.25	27	30.8
Herbivore	140.33	69.67	60.4	51.3	39.08	59.83	31.3

The most common species on the reefs in terms of trophic level were invertivores (Table 3.9). When the three predator levels are combined for each reef, species numbers were highest on SMR and lowest on NMR. The highest number of herbivore species was recorded on LMS and RS. RS, together with TMR, also had the highest number of benthivore species. Planktivore species were most common on NMR and corallivorous species were most abundant on SM.

Table 3.9 Trophic structure of the fish communities on each study reefs according to number of species. Values are expressed as a percentage of total number of species per reef. The highest value in each category is in bold.

Trophic level	LMS	RS	TMR	SMR	NMR	RR	SM
Top-level predator	2.2	1.8	2.1	3.9	1.6	3.1	1.7
Medium-level predator	13.2	12.5	13.6	14.6	11.2	12.7	15.0
Low level predator	3.3	3.6	2.6	4.5	3.7	4.6	4.1
Invertivore	35.7	39.3	39.8	37.1	40.4	39.1	35.8
Benthivore	3.9	4.2	4.2	3.9	3.7	4.1	3.5
Corallivore	3.3	2.4	2.1	1.1	1.6	3.1	3.5
Planktivore	14.8	14.3	14.1	14.6	17.0	12.7	13.9
Omnivore	10.4	8.9	11	8.4	10.1	9.1	12.1
Herbivore	13.2	13.1	10.5	11.8	10.6	11.7	10.4

Trophic category abundances were grouped according to reef protection status (Figure 3.8). Despite large apparent differences between trophic categories in the three reef categories, ANOSIM detected no significant differences according to reef protection status. For most of the trophic categories, abundances were highest in Sanctuaries and decreased as protection status decreased. Benthivores, corallivores, invertivores, herbivores, omnivores and top predators were most abundant on Sanctuary reefs. Medium-level predators were most abundant on the Open reef, where top predators were almost completely absent.

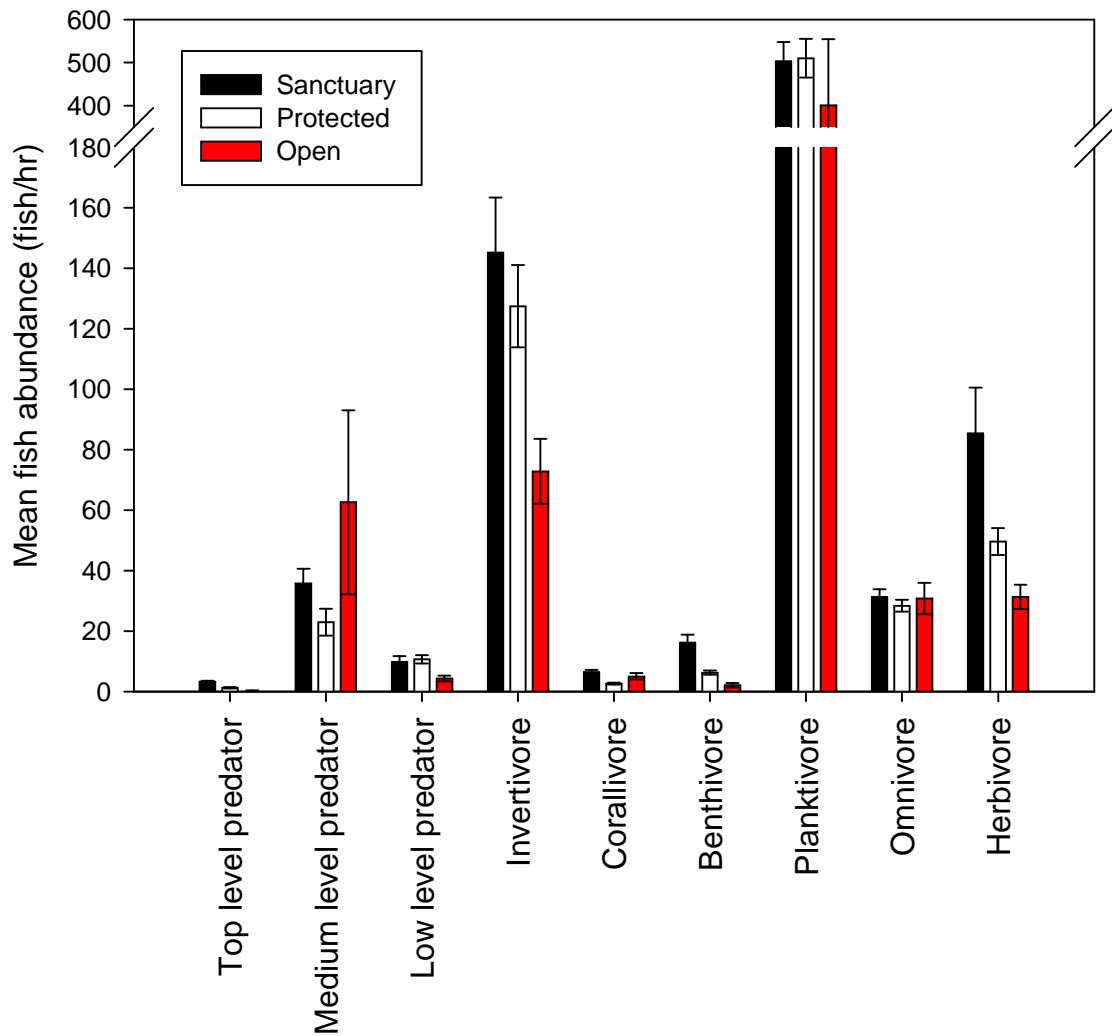


Figure 3.8 Mean (\pm SE) abundance of trophic categories according to reef protection status. Fish abundance is expressed as number of fish per hour.

3.3.4 Discriminating species

The SIMPER algorithm was used to determine the species that made the largest contribution to the dissimilarity in fish abundance between reefs (Appendix 2) and between reef protection status (Appendix 3). Due to the high number of species, only those contributing a cumulative contribution of 33% to the overall dissimilarity have been included in Appendix 3. Not all of the species that contributed the most to the between-reef dissimilarity may constitute good discriminating species. In terms of between-reef comparisons, seven species contributed the most towards the dissimilarity: *Chromis nigrura*, *Pseudanthias squamipinnis*, *Caesio xanthonota*, *Chromis weberi*, *Odonus niger*, *Nemanthias carberryi* and *Lutjanus gibbus*. The dissimilarity contribution of any of these species never exceeded 3% and the average dissimilarity contribution of the remaining species was between 0.95 and 1.05 %. The greatest average dissimilarity was between TMR and SM (54.03%).

In terms of comparisons between reef protection status, four of the already-mentioned species contributed the most towards the dissimilarity: *P. squamipinnis*, *O. niger*, *C. nigrura* and *C. xanthonota*. The contribution of these species towards the dissimilarity ranged from 1.36-2.3%. The greatest average dissimilarity was between Protected reefs and the Open reef (53.49%). The species considered to be potential discriminating species are highlighted in Appendix 2 and 3. As was expected, there was considerable overlap in SIMPER results for between-reef abundance and between-reef protection status.

A total of 26 discriminating species were identified by the SIMPER analyses (Table 3.10). These species were responsible for the dissimilarity in fish assemblage structure between reefs, rendering them potential indicator species of variance between reefs or reef protection status. Preliminary analyses were carried out to investigate relationships between potential indicator species abundance patterns and human activities in section 3.3.6.

Table 3.10 Summary of discriminating species obtained from SIMPER analyses with family, common name and trophic level. Species are arranged alphabetically according to family. Data are for South African and southern Mozambican coral reefs fish communities.

Species	Family	Trophic level
<i>Zebrasoma scopes</i>	Acanthuridae	Herbivore
<i>Balistoides conspicillum</i>	Balistidae	Invertivore
<i>Odonus niger</i>	Balistidae	Planktivore
<i>Caesio xanthonota</i>	Caesionidae	Planktivore
<i>Caranx melampygyus</i>	Carangidae	Medium-level predator
<i>Plectorhinchus flavomaculatus</i>	Haemulidae	Invertivore
<i>Plectorhinchus playfairi</i>	Haemulidae	Invertivore
<i>Sargocentron caudimaculatum</i>	Holocentridae	Invertivore
<i>Thalassoma amblycephalum</i>	Labridae	Planktivore
<i>Lethrinus crocineus</i>	Lethrinidae	Invertivore
<i>Aprion virescens</i>	Lutjanidae	Medium-level predator
<i>Lutjanus bohar</i>	Lutjanidae	Medium-level predator
<i>Lutjanus gibbus</i>	Lutjanidae	Invertivore
<i>Lutjanus kasmira</i>	Lutjanidae	Invertivore
<i>Pervagor janthinosoma</i>	Monacanthidae	Invertivore
<i>Oplegnathus robinsoni</i>	Oplegnathidae	Benthivore
<i>Pomacanthus rhomboids</i>	Pomacanthidae	Benthivore
<i>Amphiprion allardi</i>	Pomacentridae	Omnivore
<i>Chromis dimidiata</i>	Pomacentridae	Planktivore
<i>Plectroglyphidodon dickii</i>	Pomacentridae	Omnivore
<i>Plectroglyphidodon johnstonianus</i>	Pomacentridae	Corallivore
<i>Priacanthus hamrur</i>	Priacanthidae	Invertivore
<i>Cephalopholis miniata</i>	Serranidae	Medium-level predator
<i>Epinephelus tukula</i>	Serranidae	Top-level predator
<i>Nemanthias carberryi</i>	Serranidae	Planktivore
<i>Pseudanthias squamipinnis</i>	Serranidae	Planktivore

3.3.5 Linking environmental variables and habitat characteristics to fish community structure

Multivariate analyses indicated that there were no significant relationships between fish assemblage composition and the abiotic sample variables (Table 3.11). The variable with the highest rank correlation (p_s), and thus the habitat characteristic most likely to be responsible for influencing the fish community structure, was fishing intensity. However, this p_s value is too low to be considered significant and it thus appears that none of the tested environmental variables or habitat characteristics accounted significantly for the variation in fish community composition between reefs. Conversely, univariate analyses identified a number of significant relationships between certain fish assemblage parameters and independent variables (Table 3.12), in particular human activities. According to the Generalised Linear Model (GLM) regression analyses, numerous fish assemblage parameters were significantly influenced by fishing and diving intensity. Combined these variables accounted for 13.3% and 7.5% of the variance in total fish abundance, respectively.

However, in trophic groups such as top-level predators, benthivores and herbivores, human activities accounted for at least 40% of the variance. Depth was the only recorded environmental variable that had a significant influence on the fish assemblages (medium-level predators, 14.2% variance)

Table 3.11 Results of BIO-ENV procedure, taken at k time, yielding the best combinations of abiotic and biotic similarity matrices for each k , as measured by Spearman rank correlation p_s .

k	Best variables	p_s
1	Fishing intensity	0.380
2	Depth, Fishing intensity	0.361
3	Topography, Depth, Fishing intensity,	0.377
4	Topography, Depth, Fishing intensity, diving intensity	0.367
5	Topography, Depth, Diving intensity, Fishing intensity, temperature	0.337

Table 3.12 Results of GLM regression analysis on the influence of various environmental variables and human activities on fish assemblages on the seven study reefs. Only significant interactions have been included.

Dependent variable	Independent variable	P	F	Wald statistic	Percentage variance
Total abundance	Diving intensity	0.05	2.46	9.84	7.3
	Fishing intensity	0.04	3.37	6.73	6
Number of species	Diving intensity	0.05	2.5	10.01	7.5
Top-level predators	Diving intensity	<0.001	14.52	58.01	42.2
	Fishing intensity	<0.001	13.8	27.6	25.7
Medium-level predators	Depth	0.01	3.04	18.24	14.2
Invertivores	Fishing intensity	0.02	4.21	8.41	8
	Diving intensity	0.05	2.5	10.01	7.5
Benthivores	Fishing intensity	<0.001	11.26	35.59	29.9
	Diving intensity	<0.001	8.27	33.1	28.2
Herbivores	Fishing intensity	<0.001	9.74	19.49	19.1
	Diving intensity	<0.001	5.2	20.81	18.5

3.3.6 Linking species abundance distribution to human activities

Multivariate analyses indicated that human activities such as fishing and diving intensity were not significantly associated with fish community composition. However, certain species displayed patterns that relate to these activities when superimposed on the MDS abundance plots (Fig. 3.9-3.16). Although the MDS technique is not an analytical tool, it provided a useful graphical technique illustrating which species may or may not represent potential indicators of human activity. The following MDS plots show a number of selected species abundance distribution patterns in relation to human activities.

Aprion virescens

Aprion virescens is commonly targeted by recreational fishers in South Africa and Mozambique. According to the SIMPER revealed it to be a good discriminating species between reefs (Appendix 3). The abundance of *A. virescens* was consistently higher in areas of no human activity and almost absent from areas of high fishing intensity (Figure 3.9). The low abundance of *A. virescens* at sites open to fishing suggests that fishing intensity has an influence on this species.

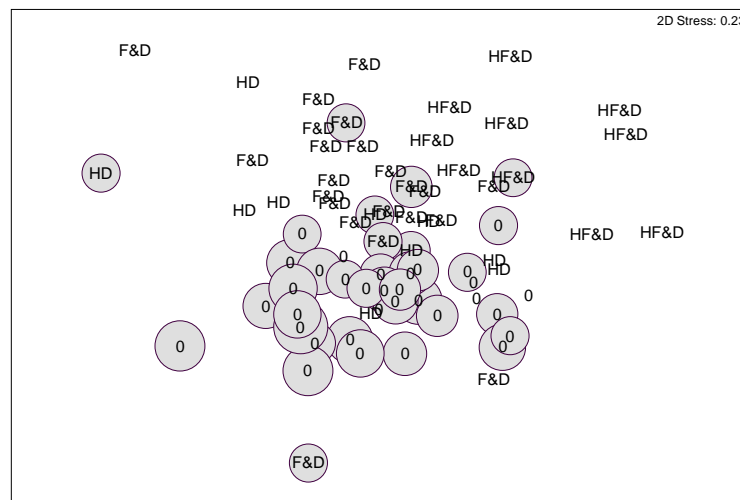


Figure 3.9 *Aprion. virescens* abundance superimposed on non-metric multi-dimensional scaling (MDS) ordination based on 4th root transformed fish abundance and factored for human activities. 0=no fishing, HD=high diving intensity, F=restricted fishing, HF=high fishing intensity, D=diving. The circle size reflects the number of fish at that sampling site.

Caranx melampygus* and *Plectorhincus flavomaculatus

SIMPER identified *Caranx melampygus* as a marginal discriminating species. However, it is a gamefish species commonly targeted by fishers in South Africa and Mozambique. The relationship between the abundance of *C. melampygus* and the main recreational activities on the study reefs is depicted in Figure 3.10A. The abundance of *C. melampygus* was higher at Sanctuary sites. Diving intensity did not appear to have a large effect on the distribution of this species. However, sites with high fishing intensities had considerably lower abundances, emphasizing the importance of this activity. *Plectorhincus flavomaculatus* manifested similar trends to *C. melampygus* (Figure 3.10B).

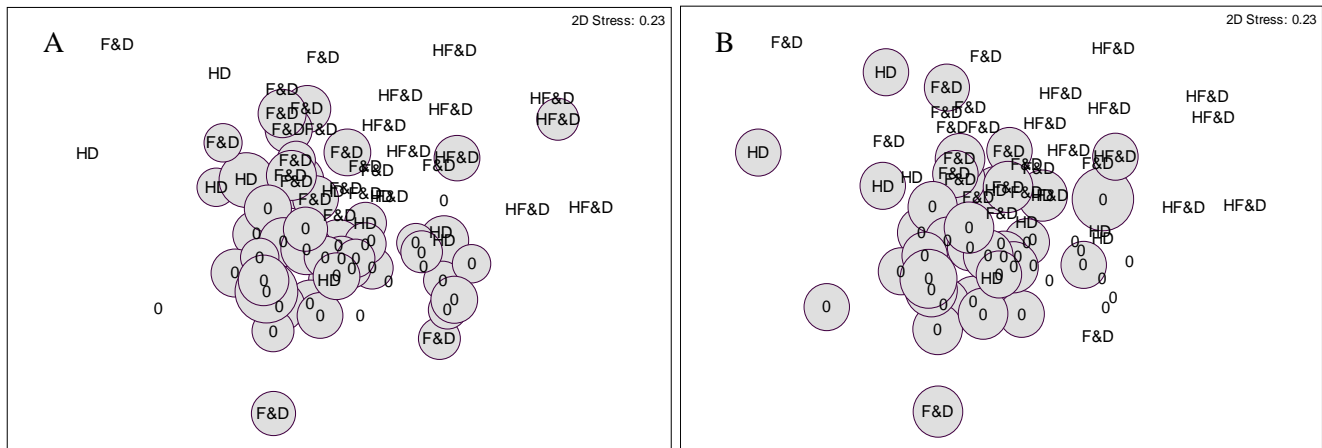


Figure 3.10 A) *Caranx melampygus* and B) *Plectorhincus flavomaculatus* abundance superimposed on non-metric multi-dimensional scaling (MDS) ordination based on 4th root transformed fish abundance and factored for human activities. 0=no fishing, HD=high diving intensity, F=restricted fishing, HF=high fishing intensity, D=diving. The magnitude of the symbols reflects the number of fish at that sampling site

Carcharhinus amblyrhynchos and *Epinephelus tukula*

Carcharhinus amblyrhynchos and *Epinephelus tukula* are two of the largest top-level predators on the study reefs. *C. amblyrhynchos* was not common at the sample sites and thus does not feature in the SIMPER results. However, it is the most common reef shark on the South African coral reefs. The MDS plot (Figure 3.11A) revealed the absence of this species from almost all sites with human activity. The highest abundances were recorded at sanctuary sites. The highest abundance of *E. tukula* was also recorded on reefs with no human influence (Figure 3.11B).

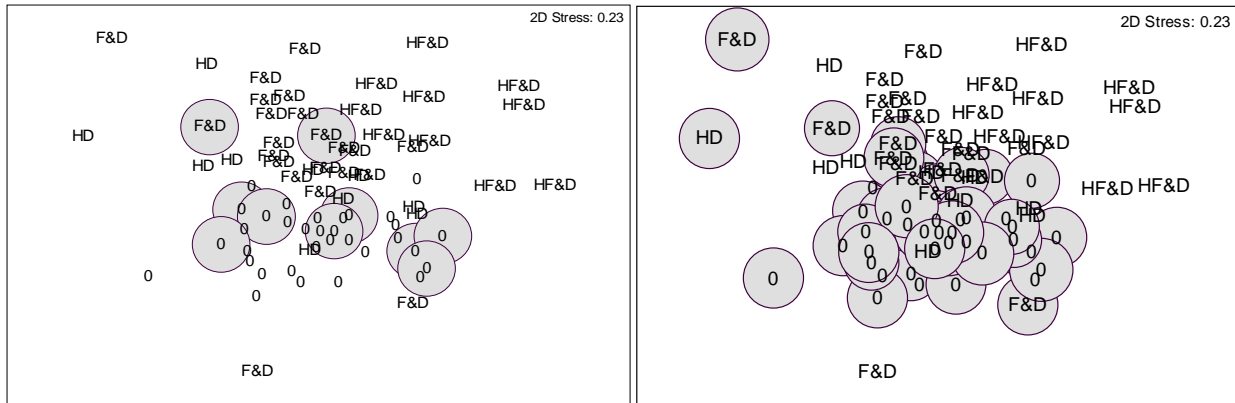


Figure 3.11 A) *Carcharhinus amblyrhynchos* and B) *Epinephelus tukula* abundance superimposed on non-metric multi-dimensional scaling (MDS) ordination based on 4th root transformed fish abundance and factored for human activities. HD=high diving intensity, F=restricted fishing, HF=high fishing intensity, D=diving. The magnitude of the symbols reflects the number of fish at that sampling site

Balistoides conspicillum

Balistoides conspicillum decreased in abundance with increasing diving intensity (Figure 3.12). More fish were observed in the sanctuary areas where human activities are not permitted. This species was recorded at only a few sites at which there is high diving intensity.

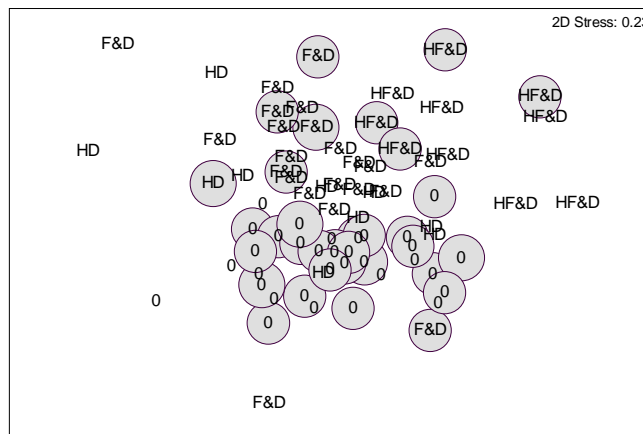


Figure 3.12 *Balistoides conspicillum* abundance superimposed on non-metric multi-dimensional scaling (MDS) ordination based on 4th root transformed fish abundance and factored for human activities. 0=no fishing, HD=high diving intensity, F=restricted fishing, HF=high fishing intensity, D=diving. The magnitude of the symbols reflects the number of fish at that sampling site.

Lethrinus crocineus* and *L. rubrioperculatus

Lethrinus crocineus and *L. rubrioperculatus* were limited in their distribution patterns in terms of abundance (Figure 3.13). Almost all records of these species were in the sanctuary areas where no human activities are allowed. It is difficult to separate the effects of either fishing and diving intensity because so few fish were recorded where such activities occur.

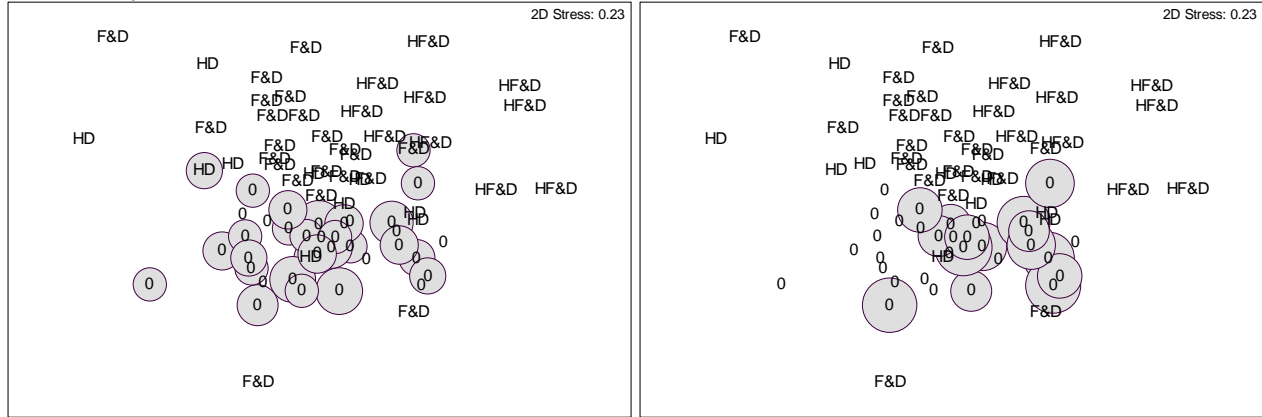


Figure 3.13 A) *Lethrinus crocineus* and B) *L. rubrioperculatus* abundance superimposed on non-metric multi-dimensional scaling (MDS) ordination based on 4th root transformed fish abundance and factored for human activities. 0=no fishing, HD=high diving intensity, F=restricted fishing, HF=high fishing intensity, D=diving. The magnitude of the symbols reflects the number of fish at that sampling site.

Lutjanus bohar* and *Oplegnathus robinsoni

Lutjanus bohar and *Oplegnathus robinsoni* manifested similar trends in their distribution patterns (Figure 3.14). Both species were most common in sanctuary areas. They were also present at sites experiencing high diving and restricted fishing. *O. robinsoni*, however, was less common at these sites. The low abundance of *L. bohar* and *O. robinsoni* suggests that high fishing intensity is a limiting factor on the distribution of these species.

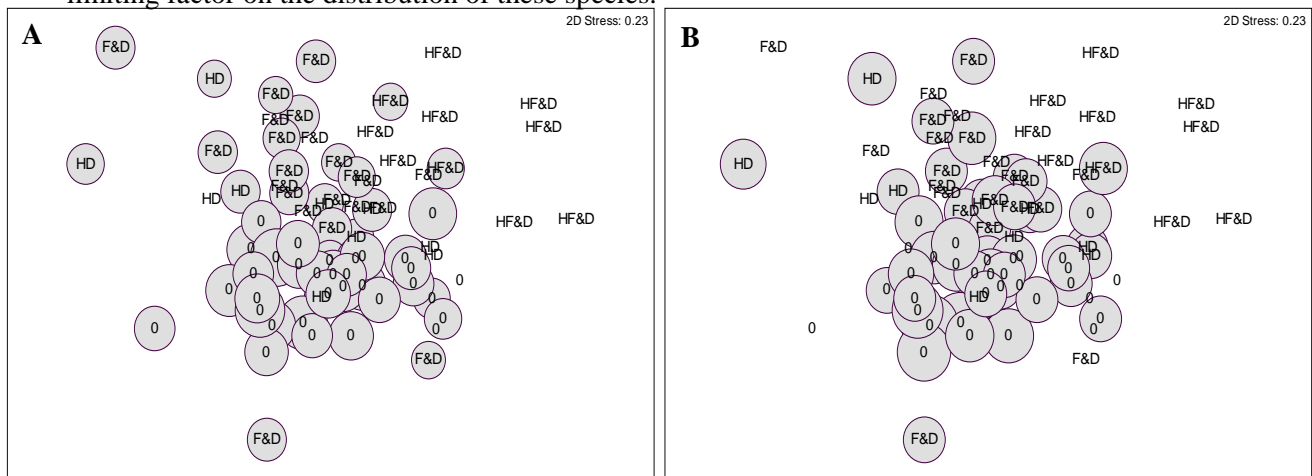


Figure 3.14 A) *L. bohar* and B) *O. robinsoni* abundance superimposed on non-metric multi-dimensional scaling (MDS) ordination based on 4th root transformed fish abundance and factored for human activities. 0=no fishing, HD=high diving intensity, F=restricted fishing, HF=high fishing intensity, D=diving. The magnitude of the symbols reflects the number of fish at that sampling site.

Plectroglyphidodon dickii* and *P. johnstonianus

The damselfish *Plectroglyphidodon dickii* and *P. johnstonianus* had similar distribution patterns as both species were most common at sites with no human activities (Figure 3.15). Sites with high diving intensity had low abundances, while sites at which there is both high fishing and diving had very low numbers or none of these species.

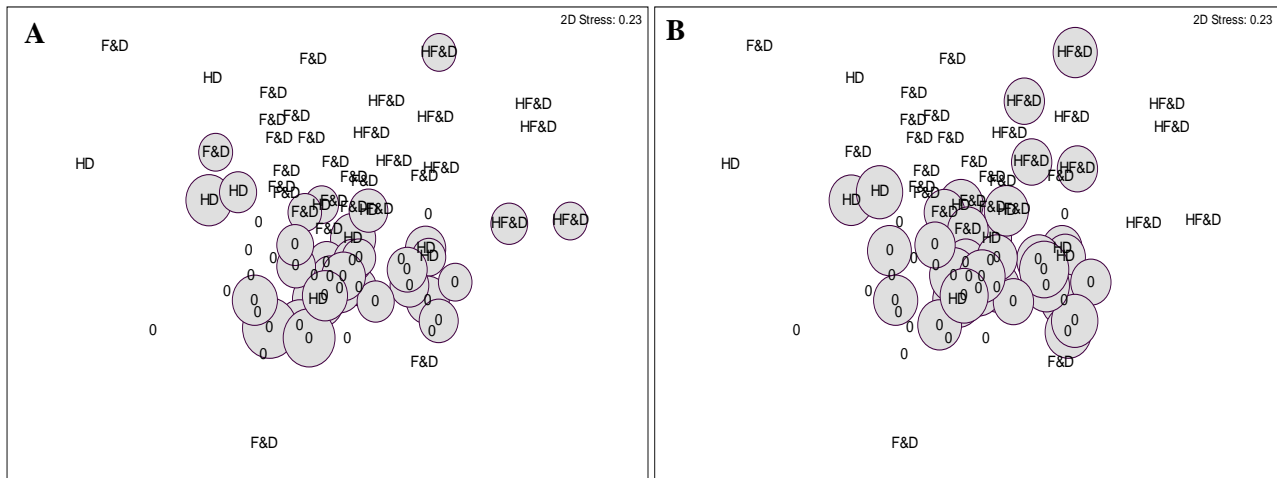


Figure 3.15A) *Plectroglyphidodon dickii* and B) *P. johnstonianus* abundance superimposed on non-metric multi-dimensional scaling (MDS) ordination based on 4th root transformed fish abundance and factored for human activities. 0=no fishing, HD=high diving intensity, F=restricted fishing, HF=high fishing intensity, D=diving. The magnitude of the symbols reflects the number of fish at that sampling site.

Contrasting patterns

A number of species revealed distribution trends that were contrary to the already-mentioned species. *Aulostomus chinensis*, *Lutjanus kasmira*, *Mulloidides vanicolensis* and *Nemanthias carberryi* were recorded in greater abundances at sites experiencing human activities (Figure 3.15). Not only were the abundances of these species low at the sanctuary sites, they were also absent at many of them. *N. carberryi* was the extreme case, being completely absent from all sanctuary sites (Figure 3.16D). In addition, it was also completely absent from all sites with high fishing intensities. *L. kasmira* (Figure 3.16B) and *M. vanicolensis* (Figure 3.16c) were also largely absent from the high fishing intensity sites.

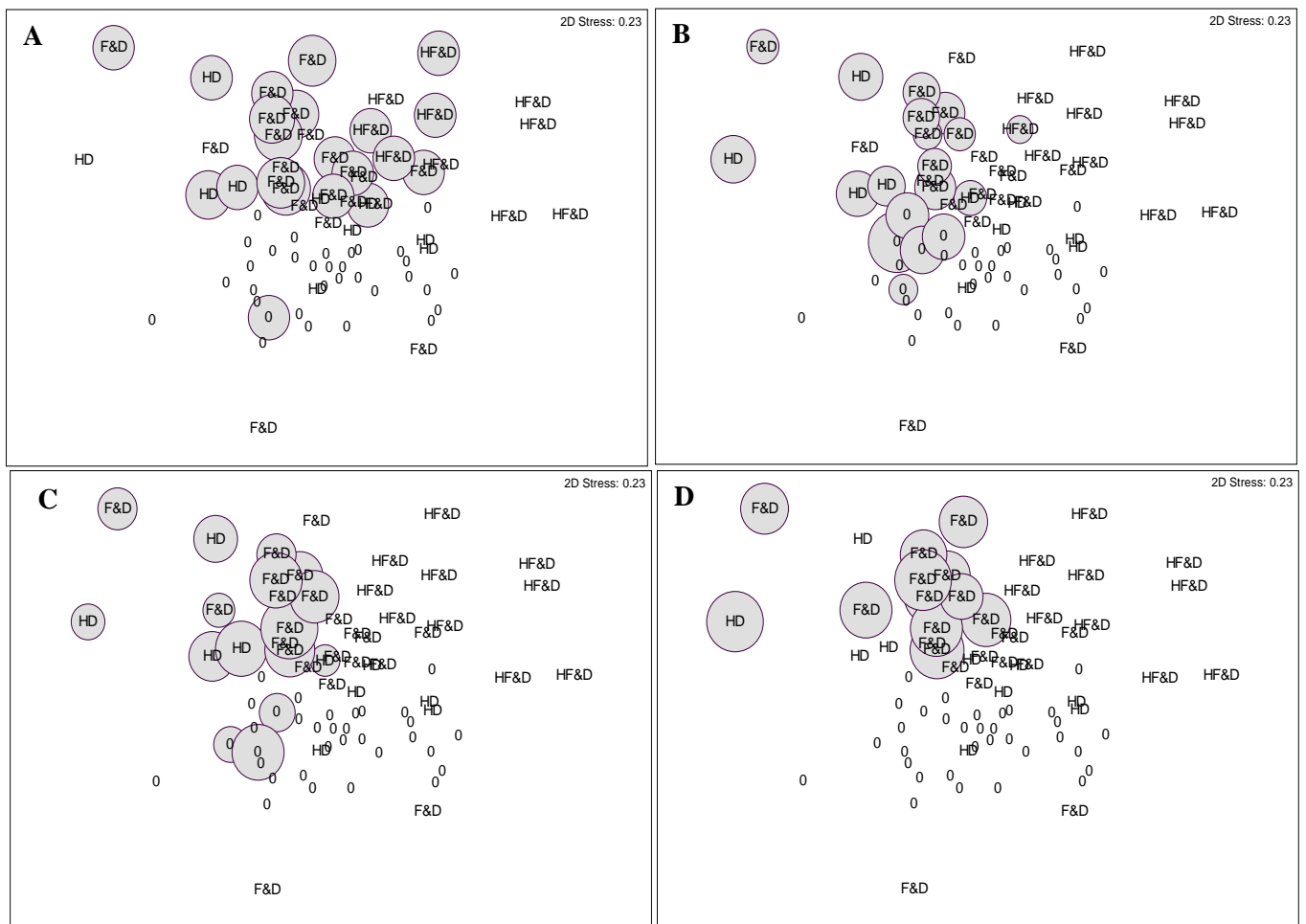


Figure 3.16A) *Aulostomus chinensis*, B) *Lutjanus kasmira*, C) *Mulloidides vanicolensis* and D) *Nemanthias* abundance superimposed on non-metric multi-dimensional scaling (MDS) ordination based on 4th root transformed fish abundance and factored for human activity. 0=no fishing, HD=high diving intensity, F=restricted fishing, HF=high fishing intensity, D=diving. The magnitude of the symbol reflects the number of fish at that sampling site.

3.4 Discussion

Spatial variation in fish assemblages

These surveys constitute one of the few studies in the WIO to document variations in mean fish abundance between reefs on a regional scale. Fish studies in the region have either presented fish inventories that are descriptive in nature (Harmelin-Vivien 1977, Chater et al. 1993, Heemstra et al. 2004, Chabanet & Durville 2005, Gillibrand et al. 2007), have focused on selected fish families or species (Chater et al. 1995, Jennings et al. 1996, Garpe & Öhman 2003, Obura et al. 2006a) or have provided quantitative data for a particular aspect of the fish community such as trophic categories (Chabanet 2002, Durville et al. 2003). While fish abundance data were collected in this study, direct comparisons with the studies referred to above are precluded because abundance was quantified per unit time rather than per unit area. Measures of density of fish per unit area are more suitable for large scale comparisons with reefs in other geographic locations. Unfortunately, the logistical constraints of this study precluded such density measurements. Nevertheless, densities of selected fish species are presented in Chapter 4 where their comparisons with other studies in the WIO will be discussed in detail.

As is often found with underwater visual censuses (Brock 1982, Samoily & Carlos 2000, Thompson & Mapstone 2002), there was considerable variation in mean fish abundance between reefs. All of the South African study reefs had higher fish abundances compared to the southern Mozambican reef. When reefs were grouped according to protection status, the highest abundances were found within Sanctuary zones. There is much evidence for enhanced abundances of target species in marine reserves (Watson & Ormond 1994, Bohnsack 1996, Russ & Alcala 1996, Wantiez et al. 1997, Unsworth et al. 2007, Watson et al. 2007), while non-targeted fish manifest varying abundances in protected areas (Mumby et al. 2006, Watson et al. 2007). In order to show the protective value of a MPA, comparisons of fish communities should be conducted before and after the proclamation of control measures. Halpern (2003) demonstrated that the density of organisms in MPAs roughly doubled and Wantiez et al. (1997) found a 160% increase in fish densities on islands in New Caledonia after five years of protection from fishing. Without historical data, it is difficult to demonstrate whether factors such as habitat, history or larval supply may account for the higher abundances observed on the South African Sanctuary reefs or whether anthropogenic factors are influencing the distribution of fish communities. This study nevertheless confirmed that Sanctuary reefs had higher fish abundances compared to Protected and Open reefs. The benefits of Sanctuary zones along the Maputaland coast are further demonstrated by a study on surf-zone fish in the St

Lucia MPA (Mann 2008); in this the abundance of surf-zone fish assemblages doubled after eight years of closure to shore-based recreational fishing.

Fish community characteristics

Sample size

The number of replicates (community counts) or the sample size is an essential component of any experimental design (Samoilys 1997). In any field-based study there is a trade-off between the minimum sample size required for statistical validity and the maximum number of samples that can be collected within the financial and logistical constraints of a project (Green 1979). Recording data underwater is particularly challenging because of the constraints imposed by SCUBA diving. In this study, depth and distance to the study reefs were factors that limited data collection. The northern Sanctuary site (Rabbit Rock) was particularly difficult to access as it was located 34 km away from the nearest launch site. Conducting pilot studies is useful to determine what sample size is needed; however, few studies have the financial means to include such preliminary surveys. The mean sample size in this study of 11 community counts per reef was initially based on work done by Samoilys and Carlos (2000), who suggested that a minimum sample size of ten counts is recommended based on statistical considerations such as degrees of freedom, which are particularly relevant when dealing with highly variable distributions typical of reef fish.

Power analysis was conducted to determine the effects of increasing the sampling effort per reef. The results revealed that the mean sample size of 11 community counts per reef resulted in a statistical power of 79%. In order to increase the power of the sampling effort to 80% (Fairweather 1991, Lenth 2001), the required sample size would have been 14 community counts per reef. Logistically, this would have been difficult to achieve on the southern African reefs and Samoilys (1997) found no appreciable change in precision in density estimates of fish beyond 10 to 15 replicates. It therefore would appear that the sample size used in this study was sufficient to detect differences in the the fish community assemblages on the South African and southern Mozambican coral reefs.

Diversity

Chater et al. (1993) produced a checklist of 399 fish species belonging to 73 families on South African coral reefs. It differed from the current study in two regards: their fish surveys were carried out using both SCUBA censuses and angling, and the fish assessments were not quantitative. Nevertheless, their results constitute important baseline work that provides many useful comparisons. A total of 284 species of fish belonging to 50 families were recorded in the current

study. This is considerably lower than the number of species recorded by Chater et al. (1993). These authors expanded their sampling capacity with angling and were able to target pelagic and deeper (20-40 m) water species. Angling would account for the absence of 45 such species from the current study, but not for the absence of 126 typically reef-associated species. On the other hand, 44 of the species recorded in the current study were not found by Chater et al. (1993). This highlights the great variability in fish communities and in underwater visual censuses. Underwater visual censuses have further inherent problems (Brock 1982, Sale 1991) and typically underestimate cryptic species (Fowler 1987). Chater et al. (1993) acknowledged this and cryptic species were similarly underestimated in this study. Considerably more underwater time is needed to search for cryptic species and destructive methods such as rotenone must be employed to collect species living in crevices. Logistical constraints precluded employing either of these two techniques here.

An unpublished checklist of fish is available for the Maputaland coral reefs and lists 1257 species (Polack 2007). This checklist, however, includes species from a depth of 200 m upwards, recorded using a variety of collecting techniques such as angling, rotenone, and deep SCUBA diving (100 m). The estimate for the number of reef-associated fish species on the Maputaland coast is roughly 1000 species (Dennis King pers. comm.). The South African coral reefs have a greater minimum depth (>10 m) than typical tropical reefs, but coral cover diminishes rapidly below 25 m. Depth is an important consideration when comparing fish communities between regions, as it is a significant factor influencing habitat partitioning (Sherman et al. 1999). The number of fish species on the South African coral reefs is expected to exceed 399, but it is doubtful that it will reach 1000 in the depth range <25 m in which South African coral reefs are found. To be consistent with reefs in the WIO, but not exclude South Africa's deeper reefs, it is proposed that assessments of reef-associated species be limited to 25 m depth. In this depth range it is estimated that at least 500 of fish may be recorded on South African coral reefs.

The 284 species of fish recorded is comparable to that found in other studies in the WIO (Table 3.13). These studies included a wide range of coral reefs, oceanic islands and fringing reefs along the east coast of Africa. The number of species recorded in this study was in fact higher than certain lower latitude reefs such as Mayotte (225: Chabanet 2002), Aldabra (221: Downing et al. 2004), Reunion (217: Letourneur 1996), southern Mozambique (239: Pereira 2003), Kenya (208: Church & Obura 2006) and Pemba (244: Richmond & Mohammed 2001). Fish communities characteristically decrease in species numbers in a gradient from low to high latitudes (Hobson 1994). However, this pattern may not be as simple as originally thought. Benayahu & Schleyer (1995, 1998) demonstrated that soft coral diversity on South Africa's coral reefs represent a diversity peak south of the equator. The high number of fish species found on these reefs

corroborates their work suggesting that there is a biodiversity peak on South Africa's coral reefs and highlights the unique nature of these high-latitude reefs.

Table 3.13 Number of species and families on WIO coral reefs recorded <25 m. Bold type indicates isolated oceanic islands. Records obtained by angling have not been included. Reefs have been ranked according to the number of species recorded. *Sampling techniques included underwater visual census as well as other techniques such as rotenone and/or explosives.

Country	Region/reef	Family	Species	Reference
Tanzania	Mafia Island	56	394	Garpe & Öhman 2003
Tanzania	Mnazi Bay	47	369	Obura et al. 2006
South Africa	Maputaland	66	354	Chater et al. 1993
Madagascar	Andavadoaka	58	334	Gillibrand et al. 2007
France	Glorieuses Islands	57	332	Durville et al. 2003
France	Basass da India	49	305	van de Elst & Chater 2000*
France	Juan De Nova	55	299	Chabanet & Durville 2005
South Africa	Maputaland	50	284	Current study
Tanzania	Pemba		244	Richmond & Mohammed 2001
Mozambique	Southern Mozambique	71	239	Pereira 2003
Comoros	Mayotte	35	225	Chabanet 2002
Seychelles	Aldabra	45	221	Downing et al. 2004
France	Réunion	44	217	Letourneur 1996
Kenya	Kiunga Bay		208	Church & Obura 2006
Madagascar	Tulear	44	200	Harmelin-Vivien 1977*

The species diversity indices of the current surveys show few trends and the between-reef comparisons were too variable to draw any conclusions. However, when diversity was analysed according to reef protection status, more obvious patterns emerged. Conventional diversity indices such as species richness and Shannon Diversity Index were highest on Sanctuary reefs, lower on Protected reefs and lowest on the Open reef, SM. The number of species per reef was highest on the Sanctuary reefs; however, when combined, the Protected reefs had the highest number of species. The trend of higher biodiversity within MPAs is consistent with the findings of Halpern (2003), who established that the diversity of fish communities was 20–30% higher inside protected areas. In addition, Jennings et al. (1995) demonstrated increased species richness in fish communities in Seychelles's MPAs as did Wantiez et al. (1997) in New Caledonia.

Family composition

The most abundant coral reef fish families (Labridae, Acanthuridae, Chaetodontidae, Pomacentridae, Serranidae and Lutjanidae) were well represented on South Africa's coral reefs in terms of species numbers. SM had fewer species, in comparison, particularly in the serranid and lutjanid families. Both of these families include species that are targeted by fishers for recreational and commercial purposes (Chabanet & Durville 2005). In South Africa, only gamefishing is

permitted on reefs in the Central Reef Complex north of TMR and bottom-fishing for reef species is prohibited. The protective legislation and hence the (near) absence of bottom-fishing on these reefs probably explains the similarity in numbers of serranid species observed between the South African reefs. Within the lutjanid family, only one coral reef-associated species is classified as a gamefish, *Aprion virescens*. With this exception, lutjanids are also largely protected from fishing on South African coral reefs. A contrasting trend was evident on the southern Mozambican coral reef. At the time of data collection, despite regulations specifying daily limits for certain target species, the combination of poor enforcement and low compliance by fishers had led to unrestricted fishing on these reefs. Species commonly removed by fishing from the reefs included gamefish as well as other bottom-dwelling fish species (van der Elst et al. 1996). In addition, illegal semi-industrial fishing vessels have been observed in close proximity to the reefs (Pereira 2003). The low numbers of lutjanid and serranid species on SM are thus probably a consequence of fishing intensity and their lack of protection.

Trophic structure

Trophic structure of fish communities on the South African reefs varied greatly between reefs in terms of abundance and number of species. Benthivores and herbivores manifested no particular distribution pattern, except that these categories were particularly abundant on LMS. Planktivores were the most abundant feeding category (in terms of number of fish) on all the reefs, but were highest on LMS and SMR. Planktivore abundance has been observed to be highest along the reef edges near deeper water (Hobson 1991). Garpe and Öhman (2003) suggested that hydrodynamic factors may regulate fish community composition, particularly if plankton feeders rely on exposure and surge to replenish food resources. A similar distribution pattern emerged on SMR and LMS where planktivores were most abundant near pinnacles and drop-offs. Corallivores were most abundant on RS and LMS and least abundant on SMR. Differences in percentage coral cover between reefs may account for this distribution pattern as coral cover estimates were on average higher on RS and LMS compared to SMR. A positive relationship between fish species diversity, abundance and coral cover has been reported in numerous studies (Chabanet et al. 1997, Adjeroud et al. 1998, McClanahan & Arthur 2001). It is a logical assumption that the abundance of a specialist's food source would have considerable influence on its distribution (Öhman et al. 1997, Graham et al. 2008).

According to Harmelin-Vivien (1979), carnivore levels observed on a healthy reef usually constitute between 60-80% of the trophic composition depending on geographic location (Table 3.14). A comparison of carnivore abundance across the western Indian Ocean supports this theory, as published values for Mayotte (Chabanet et al. 2002), Glorieuses (Durville et al. 2003), south

Madagascar (Harmelin-Vivien 1979), Reunion (Chabanet 1994) and Juan de Nova range from 51 to 74%. The contribution of carnivores to the fish community abundance in these studies was calculated (as per Chabanet 2002) by combining all trophic categories except omnivores and herbivores. When the nine trophic categories in this study are considered in these three simplified groups, the average carnivore levels are close to 80%, making them among the highest on WIO coral reefs and implying that the trophic structure of the fish communities has not been influenced by anthropogenic or natural stresses. This is, however, not the case. While this approach allows comparisons between reefs over large geographic areas, it oversimplifies the trophic structure of reefs on a regional scale. More importantly, it may provide reef managers with spurious information on the number of predators in an ecosystem. As predators are often the species most targeted by fishers, such simplified results may underestimate the effects of fishing.

Table 3.14 Trophic structure of fish communities, expressed as a percentage of total number of species on different WIO coral reefs.

Reef	Carnivore	Omnivore	Herbivore	Reference
Tuléar (Madagascar)	74	13.5	12.5	Harmelin-Vivien 1979
Réunion	51	24	25	Chabanet 1994
Andavadoaka	76	11	11	Gillibrand et al. 2007
Mayotte	69	12.5	18.5	Chabanet 2002
Juan de Nova	73	11	16	Chabanet & Durville 2005
Geyser et Zéléé	69	16	15	Chabanet et al. 2002
Glorieuses	73	12	15	Durville et al. 2003
South Africa	79	9	12	Present study
Shallow Malongane	77	12	11	Present study

Predators are important species in coral reef ecosystems because of their functional role in regulating fish abundance (Connell 1998, Steele et al. 1998, Dulvy et al. 2004a, Pala 2007). Top-level (apex) predators are disproportionately removed by fishing (Koslow et al. 1988, Russ & Alcala 1989) and, as a consequence, have received much attention in the literature (Polunin & Roberts 1993, Watson & Ormond 1994, McClanahan & Arthur 2001, Watson et al. 2007). However, it is important also to understand the dynamics of smaller predators as they consume a significant but smaller proportion of the fish biomass by preying on new recruits, juveniles and other smaller fish (Hixon 1991). In this study, predators were divided into three categories; low, medium, and top-level predators. Medium-level predators were most abundant on SM, but this was only due to a single sighting of a large shoal of the transient species, *Scomberoides lysan*. When this species was omitted from the analyses, the abundance of medium-level predators on SM was reduced by half from 62.6 fish/hr to 32.6 fish/hr. This indicates that the contribution of predators to fish community structure may best expressed as the number of species rather than the number of

individuals as a high abundance of a single species may bias the data. In addition, it may be prudent to exclude shoaling species or species that are not truly reef-associated .

Environmental and habitat variables

Numerous studies have focused on the relationship between the distribution of fish species and factors such as substratum complexity (Luckhurst & Luckhurst 1978, Friedlander 2003), water quality (Rodriquez 2006), wave exposure (Friedlander 2003), substratum composition (Galzin et al. 1994, Holbrook et al. 2006), depth (Friedlander & Parrish 1998) and reef zonation (Williams 1991) to name a few. Live coral cover, in particular, has been shown to be a strong predictor of species abundance and diversity (Chabanet et al. 1997, McClanahan & Arthur 2001). This relationship tends to be positive; however, the correlation is not as linear as initially thought as numerous authors have found conflicting results (Luckhurst & Luckhurst 1978, Öhman & Rajasuriya 1998, Friedlander 2003). The aim of this study was to document and quantify the fish communities on South African coral reefs. Thus, every attempt was made to keep variation in environmental factors to a minimum by standardizing the study area in terms of substratum composition, limiting the depth range at which the counts were conducted and standardizing the seasonality in the sampling trips. Fish censuses were conducted only at study sites that were located within a specific coral community (Celliers & Schleyer 2008). This coral community was selected because it was the most widespread community on the reefs and thus represented the core coral community. It also provided a means of standardizing the benthic composition.

The influence of depth and topography on fish distribution was considered in this study. It is a well-established fact that fish assemblages change along a depth gradient (Williams 1991, Bouchon-Navaro et al. 1997, Gillibrand et al. 2007). In this study, multivariate analyses indicated that depth had no influence on the distribution of the fish communities on the study reefs. Generalised Linear Model (GLM) regression analysis confirmed this as the abundance of only medium-levels predators was associated with depth. However, the depth range of the reefs in this study was narrow (12-15 m) and probably too small to have a significant effect on fish community structure. Several studies have also highlighted the importance of topographic complexity in shaping fish community structure (Luckhurst & Luckhurst 1978, McClanahan 1994, Öhman & Rajasuriya 1998, Friedlander 2003), which has been linked to variations in fish diversity (Roberts & Ormond 1987) and habitat selection during larval settlement (Booth & Wellington 1998). Multivariate analyses (BEST, PRIMER E v6) and GLM regression models indicated that topography did not significantly affect the fish community structure on the reefs.

Fishing and diving intensity were included as habitat characteristics in the multivariate (BEST) and GLM regression analyses. According to the BEST analysis, neither variable had a significant influence on the overall fish community assemblages. However, the GLM regression analysis indicated that fishing and diving intensity accounted significantly for variance in the fish community. The BEST and GLM regression procedures were included in the analyses because they focus on different aspects of the fish community data. Multivariate analyses such as the BEST procedure examine the 'best' match between the multivariate among-sample patterns of an assemblage and associated environmental variables (Clarke & Gorley 2006). GLM regression analysis uses univariate variables such as species abundance, trophic level abundance or diversity indices for comparisons with independent variables and thus do not explore among-sample i.e. variation between species. High species numbers may thus disguise associations in fish assemblage structure and independent variables in such multivariate analyses. This suggests that the BEST procedure is most useful as an exploratory test to investigate subsets of species with similar characteristic such as trophic levels.

Discriminating species, indicators and human activities

The third aim of this chapter was to use the results of the fish community analyses to validate potential indicator species that were identified *a priori* through literature reviews. SIMPER was used to reveal the species most responsible for the dissimilarity between reefs or groups. A total of 26 discriminating species were identified (Table 3.10). These and additional species were superimposed on MDS abundance plots and factored for human activities. A number of the selected species showed some degree of correlation between abundance and human activity, while some species manifested less obvious trends. The list of 26 species was reassessed according to the trends revealed in the MDS plots and through other selective criteria discussed below.

Target species such as *Aprion virescens*, *Caranx melampygus* and *Lutjanus bohar* were low in abundance on reefs with high fishing intensity. Non-target species such as *Balistoides conspicillum* appeared to be affected by high diving intensity as their abundance was low at popular diving sites. These four species were retained as indicators. In terms of reef protection, a number of the discriminating species were more abundant in the Sanctuary areas. However, there were also a number of species that manifested no pattern relative to human activities, while others were more numerous in areas of increased human activity. Species that manifested no correlations or ambiguous correlations with human activities were removed from the list of potential indicators to reduce high variance (*Plectrorhinchus spp*, *Pervagor janthinosoma*, *Nemathias carberryi* and *Sargocentron caudimaculatum*). A further three species were removed from the list because their

shoaling behaviour increased variability in the data (*Caesio xanthonota*, *Lutjanus gibbus* and *L. kasmira*). *P. rhomboides* was replaced with *P. imperator* which was less patchy in its distribution and because of its conspicuous colouration. In two instances, species (*Thalassoma amblycephalum*, and *Cephalopholis miniata*) were replaced with similar species (*T. herbraicum* and *Variola louti*) that were deemed to be of greater importance in terms of trophic function because they were more abundant on the reefs. *Zebrazoma scopas* was replaced with *Z. desjardini* because the latter was more abundant on reefs devoid of human activity and its distribution extends to other WIO coral reefs.

Additional species were added as potential indicators based on numerous criteria such as their ease of identification, vulnerability to exploitation by fishing and hobbyists, and status as rare species. Ease of identification was an important consideration because monitoring may be carried out by non-scientists. Thus, species such as *Acanthaster leucosternon*, *forcipiger flavissimus* and *Chaetodon madagaskariensis* were included because of their trophic status and conspicuous colouration. Species such as *Scarus rubroviolaceus* were included because they are targeted on southern Mozambican reefs, but not on South African reefs. Similarly, *Amphiprion allardi* was selected because it is not actively protected from aquarists in southern Mozambique. The shy or uncommon species, *Pygoplites diacanthus*, was added to the list due to its potential sensitivity to diving intensity. Three corallivorous chaetodons were included in the list due to their reliance on corals for food.

The revised indicator list totaled 25 species and was termed the Fish-index (Table 3.15). This Fish-index formed the basis of a monitoring protocol in which fish indicator species were used to assess the effects of human activities on South African and southern Mozambican coral reefs (Appendix 6). The effectiveness of these indicators will be dealt with in Chapter 4.

Table 3.15. List of indicator species used to assess reef fish community condition in Chapter 4. Species have been ordered alphabetically according to family.

Species	Family	Trophic level
<i>Zebrasoma desjardini</i>	Acanthuridae	Herbivore
<i>Acanthurus leucosternon</i>	Acanthuridae	Herbivore
<i>Balistoides conspicillum</i>	Balistidae	Invertivore
<i>Odonus niger</i>	Balistidae	Planktivore
<i>Caranx melampygus</i>	Carangidae	Medium-level predator
<i>Chaetodon madagaskariensis</i>	Chaetodontidae	Omnivore
<i>Chaetodon trifascialis</i>	Chaetodontidae	Corallivore
<i>Chaetodon trifasciatus</i>	Chaetodontidae	Corallivore
<i>Chaetodon meyeri</i>	Chaetodontidae	Corallivore
<i>Forcipiger flavissimus</i>	Chaetodontidae	Invertivore
<i>Bodianus Diana</i>	Labridae	Invertivore
<i>Labroides dimidiatus</i>	Labridae	Invertivore
<i>Thalassoma herbraicum</i>	Labridae	Invertivore
<i>Aprion virescens</i>	Lutjanidae	Medium-level predator
<i>Lutjanus bohar</i>	Lutjanidae	Medium-level predator
<i>Pomacanthus imperator</i>	Pomacanthidae	Benthivore
<i>Pygoplites diacanthus</i>	Pomacanthidae	Benthivore
<i>Amphiprion allardi</i>	Pomacentridae	Omnivore
<i>Plectroglyphidodon johnstonianus</i>	Pomacentridae	Corallivore
<i>Oplegnathus robinsoni</i>	Oplegnathidae	Benthivore
<i>Epinephelus tukula</i>	Serranidae	Top-level predator
<i>Variola louti</i>	Serranidae	Medium-level predator
<i>Scarus rubroviolaceus</i>	Sacridae	Herbivore
<i>Siganus sutor</i>	Siganidae	Herbivore
<i>Diplodus cervinus hottentotus</i>	Spadidae	Invertivore

Benchmark ecosystems

Due to the advanced state of human impacts on the marine environment, ecosystems that are unaltered by human influence are all but extinct (Jackson et al. 2001, Friedlander & Demartini 2002, DeMartini et al. 2008). However, South Africa is fortunate in that it has coral reefs with long histories of protection from human interference. These relatively undisturbed coral reefs may constitute benchmarks that can provide baseline data for comparisons with similar reefs in the region that are less protected and exposed to human resource use. The potential of South African Sanctuary reefs as benchmark ecosystems thus requires further validation and is dealt with in Chapter 4.

CHAPTER 4

FISH AS INDICATORS OF DIVING AND FISHING PRESSURE ON SOUTH AFRICAN CORAL REEFS.

4.1 Introduction

In the 1980s, reef managers and scientists fervently adopted the marine protected area (MPA) approach in an attempt to find a universal solution to the growing global coral reefs crisis (see Kelleher et al. 1995, McClanahan 1999, Sale 2008 for a review of this subject). Empirical studies are frequently cited as evidence that local resource users gain direct benefits from the proclamation of MPAs (Russ & Alcala 1989, Polunin & Roberts 1993, Russ & Alcala 1996, McClanahan & Arthur 2001, Roberts et al. 2001, Barrett et al. 2007, Lester et al. 2009, Watson et al. 2009). Conversely, the lack of effectiveness in MPAs is also highlighted by many authors (Kelleher et al. 1995, Alder 1996, McClanahan et al. 1999), which raises the question: Why are MPAs not effective? In the past decade, approximately 40 new MPAs were created per year worldwide, which included coral reefs (Mora et al. 2006). Under the current scientific and management ethos of coral reef protection, it appears that the conservation endeavors of a country are measured by the percentage of its coastline that is given such protection. Consequently, there is a growing concern that over-zealous advocacy has led to the approach that MPAs are a 'cure-all' for conserving biodiversity and fisheries management (Jameson et al. 2002, Agardy et al. 2003, Lubchenco et al. 2003). Without the necessary compliance, efficient management, clearly defined objectives and scientific knowledge, many MPAs are doomed to remain 'paper' parks (Burke et al. 2002, Halpern 2003, Burke & Maidens 2004, Pomeroy et al. 2005).

Among the major challenges restricting effective management is a lack of scientific information about the status and nature of conditions operating within a MPA (Kelleher et al. 1995, Pomeroy et al. 2005, Wells et al. 2007). Obtaining such information requires regular assessment of biological and socio-economic processes within the MPA boundaries (Pomeroy et al. 2005). In their investigation of the status of MPAs in three east African countries - Kenya, Tanzania and Mozambique - Wells et al. (2007) revealed that few baseline surveys had been conducted and appropriately designed monitoring programmes were lacking. The authors suggested that better monitoring systems were needed to measure progress and also demonstrate the benefits of MPAs on biodiversity and the livelihoods of people within or near the MPA boundaries. Assessment of MPA effectiveness is a matter of great urgency and importance given the multitude of stressors

threatening the future of coral reefs. This is particularly pertinent as many MPAs are promulgated as multiple resource use zones where extractive activities such as fishing are permitted. Thus, it is imperative that such zones are evaluated to assess the impacts of human activities on the biological components of the coral reef and whether these impacts are consistent with the management objectives of the MPA.

The coastal regions of the Western Indian Ocean (WIO) are characterized by high biodiversity, high human population densities and comparatively low Gross Domestic Product (Keesing & Tennille 2005). MPAs in the WIO are of particular significance because they are advocated as a means of alleviating poverty through fisheries protection and tourism (IUCN 2004). Like many other countries in the Western Indian Ocean, MPAs have been a part of South Africa's marine conservation strategy since the 1960s (Lemm & Attwood 2003). However, MPAs in South Africa include a wider variety of habitats including sandy beaches, subtidal rocky reefs, mangroves, estuaries and coral reefs. South African coral reefs are unique for several reasons: they are the southern most coral reefs in Africa, they are considered marginal reefs (Kleypas et al. 1999) but attain high biodiversity (Benayahu & Schleyer 1995, 1998), and they have not been as heavily affected by stressors such as coral bleaching, destructive fishing practices, or disease, that plague their east African counterparts (Schleyer & Tomalin 2000, Floros et al. 2004, Celliers & Schleyer 2008, Schleyer et al. 2008, Wilkinson 2008).

One of their most distinctive traits is that they are all situated within the boundaries of two long-standing MPAs; the Maputaland Marine Reserve and the St Lucia Marine Reserve. At present, two types of conservation zones are recognized in the MPAs: areas where human activities are prohibited and areas of restricted or controlled human activity. Despite the proclamation of these MPAs more than 20 years ago, a recent assessment of the MPA management strategy revealed that a rigorous monitoring programme evaluating the effectiveness of the MPAs was missing (Lemm & Attwood 2003).

The South African coral communities have been intensively investigated and studies include reef zonation (Riegl et al. 1995), reproduction (Schleyer et al. 1997), diver damage (Schleyer & Tomalin 2000, Walters & Samways 2001), biodiversity (Schleyer & Celliers 2003), coral bleaching (Celliers & Schleyer 2002, Floros et al. 2004) and reef modeling (Schleyer & Celliers 2005). However, the fish communities on the South African coral reefs have received far less attention. Two studies investigating the fish communities were conducted in the early 1990s (Chater et al. 1993, Chater et al. 1995), which provide valuable checklists of fish species on the coral reefs and

abundance data for selected fish species. However, to date, no study has investigated the effects of human activities on the fish communities.

Recreational fishing and SCUBA diving are the most common activities on the South African coral reefs (Schleyer 2000). Fishing is known to have a direct effect on fish communities via the harvest of target and bait species, and the removal of functional groups (Hall 1999, Dulvy et al. 2004a). Even recreational fishing increases mortality rates above the natural mortality threshold and has the potential to cause significant changes in the structure of reef fish communities (Cooke & Cowx 2004, Dulvy et al. 2004b). SCUBA diving on certain reefs is amongst the highest in the world (Hawkins et al. 1999, Tratalos & Austin 2001, Zakai & Chadwick-Furman 2002, Barker & Roberts 2004, Hawkins et al. 2005) and the paucity of empirical studies investigating the effects of high diving intensity on fish communities in the literature represents a 'knowledge-gap'. In light of the many threats facing coral reefs, it is thus both timely and pertinent that the effects of human impacts on the South African coral reef fish assemblages are assessed for efficient MPA management.

The evaluation of the effectiveness with which a MPA achieves its management objectives may be linked to ecological indicators. Ecological indicators include living components of the reef that reflect disturbances that alter the natural functioning of reef processes (Linton & Warner 2003). Fish species have been recommended as effective indicators because they comprise a large proportion of the biomass in marine ecosystems, they provide ecosystem goods to humans, and they show clear responses to human activities (Karr 1981, Whitfield 1996, Rice 2003). The use of indicators has broad applications because it can serve multiple audiences, including policy makers, managers, researchers and conservationists (Pomeroy et al. 2005). Indicator-based monitoring programmes provide an inexpensive method of gathering scientific information that does not require highly skilled personnel. Moreover, interpretation of such data is uncomplicated because most indicator programmes are based on studies that have proven cause-and-effect relationships. Results gained from indicator-based monitoring programmes can be used to evaluate MPA management, assist in identifying strong or weak areas of the MPA management strategy and aid in setting new conservation objectives or goals.

The aim of this study was to investigate the effects of human activities on South African coral reef fish communities using 25 fish indicator species. Termed the 'Fish-index', this multi-species assemblage consisted of fish species selected due to their ecological importance and susceptibility to human activities. Furthermore, the selection of species was validated by the results of the community analyses in Chapter 3.

The main objectives of this chapter were to:

1. Compare the Fish-index in terms of density, biomass, size frequency analysis and trophic structure on reefs of varying reef protection status
2. Document any unusual trends in the Fish-index in the presence of human activities
3. Determine the effectiveness of the Fish-index in assessing the effects of human activities on South African coral reefs

4.2 Materials and methods

4.2.1 Study site description

See Chapter 2 for details.

Reef protection status and human resource use

The reefs were categorised according to the intensity and type of human activity (Table 4.1). These categories were Sanctuary, High-Diving, Diving-Fishing and Open zones. Sanctuary zones consisted of reefs that were not subjected to any human activities and included Leadsman Shoal (LMS), Red Sands (RS) and Rabbit Rock (RR). The High-Diving zone included Two-mile Reef (TMR), where only SCUBA diving is permitted. The Diving-Fishing zone included Seven-mile (SMR) and Nine-mile Reef (NMR), where SCUBA diving and restricted (gamefishing) fishing are permitted. Finally, the Open zone included the shallow reef at Ponta Malongane reef in southern Mozambique, known as Shallow Malongane, where the absence of law enforcement on the reef meant SCUBA diving and fishing was unregulated.

Table 4.1 Categorisation the reefs in the study area based on SCUBA diving and fishing intensities. SCUBA diving statistics are averages for the period 2007-2008.

Zone	Human activities		Reef name
	SCUBA diving intensity	Fishing intensity	
Open	Low (4500 dives/year)	High (unrestricted angling and spearfishing)	Shallow Malongane
Diving-Fishing	Low (~2100 dives/year)	Restricted (angling and spearfishing, gamefish only).	Seven-mile Reef, Nine-mile Reef
High-Diving	High (~54 000 dives/year)	Nil	Two-mile Reef
Sanctuary	Nil	Nil	Leadsman Shoal, Red Sands, Rabbit Rock

4.2.2 Surveys of reef fish communities

During the period August 2007 to February 2009, six fieldtrips were undertaken to the Maputaland coral reefs and two fieldtrips to southern Mozambique. Data were collected using SCUBA and all diving operations were conducted from a semi-rigid inflatable vessel. Study sites on each reef were selected using the same method described in Chapter 3. Figure illustrates the position of each sampling site on the South African study reefs.

An underwater visual census technique was used to estimate the numerical abundance and biomass of the 25 preselected fish-index species. The technique consisted of the point count method adapted from Samoilys and Carlos (2000). Each census (point count) consisted of a five minute count within a circle 10 m in diameter, which enclosed an area of 78 m² for each point count. A minimum of 60 point counts were completed per reef (Table 4.2). Sampling was conducted in summer and winter to test whether patterns in fish assemblages were consistent over time. All Fish-index species observed on the reef and within the water column during each point count were recorded and their sizes were estimated. Size estimates of fish were conducted in 5 cm increments. Estimates of fish length were used to generate biomass using known length-weight regression coefficients from Fishbase (www.fishbase.com). When a length-weight relationship for a given species was not available, values were taken from a species with a similar morphology, always from the same genus. Only two divers entered the water at any one time, a surveyor and a buddy diver. To minimise the impact of diver disturbance caused by divers entering the water, the surveyor waited five minutes before beginning each count. The same diver conducted all point counts to minimize variation and error incurred by diver bias. Point counts within each reef were separated by at least 50 m. The time of each point count was conducted between 0800 and 1400. All counts were conducted at a depth range of 12-15 m.

Table 4.2 Summary of survey dates and number of fish point counts conducted in each resource use zone on South African and southern Mozambican coral reefs during 2007-2009.

Zone	August 2007	January 2008	February 2008	June 2008	July 2008	November 2008	December 2008	February 2009	Total
High-Diving	6	12	0	0	24	9	0	14	65
Diving-Fishing	9	4	5	10	22	0	10	17	77
Sanctuary	11	12	33	0	33	0	10	6	105
Open	0	0	0	0	30	0	0	30	60

Trophic levels

Similar methods to those described in Chapter 3 were used to allocate species to trophic levels, the only difference being that there were reduced to eight and not nine due to the absence of low-level predators in the Fish-Index.

Measuring environmental variables and habitat characteristics

On completion of each visual census, certain aspects of the reef habitat were described within the perimeter of the 10 m diameter circle. These included depth, topography and coral cover. Coral cover was estimated using a rapid visual assessment technique adapted from English et al. (1994). The estimates were divided into three categories according to the percentage of coral (hard and soft) covering the point count area. An area of more than 50% coral was categorised as high. Medium

cover constituted 30-50% coral cover, while areas with less than 30% coral cover were classified as low. Depth was also recorded using similar methods described in Chapter 3. The topographic complexity of the substratum was visually estimated using a six point scale system adapted from Polunin and Roberts (1993). The adaptations were specific to South African coral reefs and included the following categories: 1- flat reef; 2 – low undulating spur and groove; 3 – medium slopes or ridges with no grooves or gullies; 4 – medium spur and groove or pinnacles; 5 – high slopes or ridges; and 6 – high spur and groove, overhangs or pinnacles.

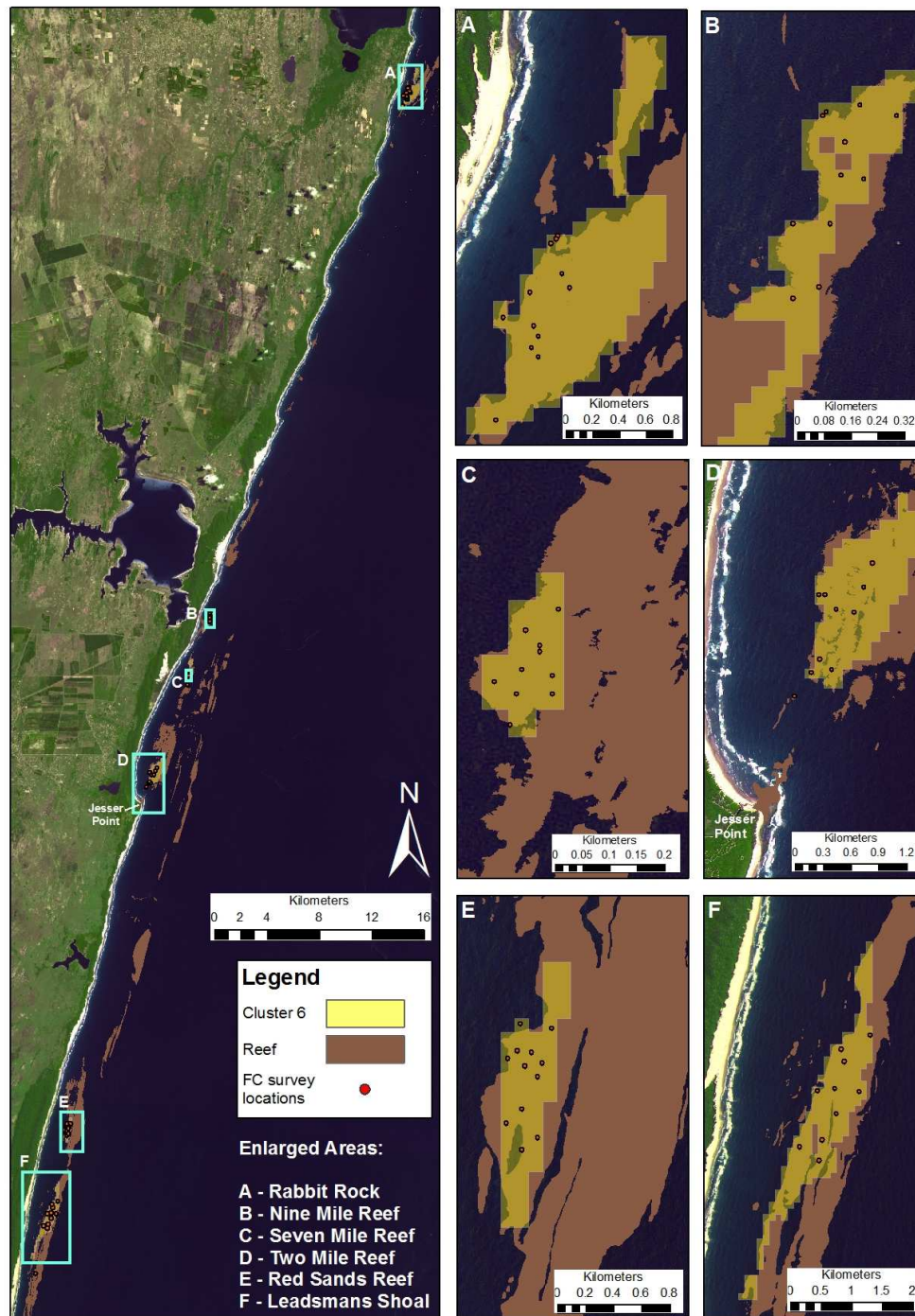


Fig. 4.1. Location of survey sites on the six South African coral reefs. Geo-spatial data are from Celliers & Schleyer (2008). No data were available for the southern Mozambican reef.

Diver-orientated behaviour in a top predator

Epinephelus tukula was observed to display contrasting diver-orientated behaviour between Sanctuary and non-Sanctuary reefs. Comparisons of *E. tukula* density data derived from the two UVC techniques employed in this study were used to investigate whether the behavioural responses could be quantified. Community count estimates were used to represent the total *E. tukula* densities observed in each level of protection, while point count estimates were used to estimate the densities of *E. tukula* approaching divers. The *null* hypothesis was that densities of *E. tukula* would be similar between the two UVC estimates. If community count estimates of *E. tukula* were higher than the point count density estimates, this may indicate that diver presence influences *E. tukula* behaviour. Approach ratios were calculated (count density divided by community count density) to determine the likelihood of *E. tukula* approaching a diver on a scale from 0-1. The closer the ratio approaches 1, the more likely *E. tukula* is unaffected by the presence of divers i.e. it will approach a diver. No *E. tukula* were recorded in the Open zone and the latter site was thus excluded from the analysis.

4.2.3 Statistical analysis.

The stepwise process of the statistical analysis is diagrammatically represented in Figure 4.2.

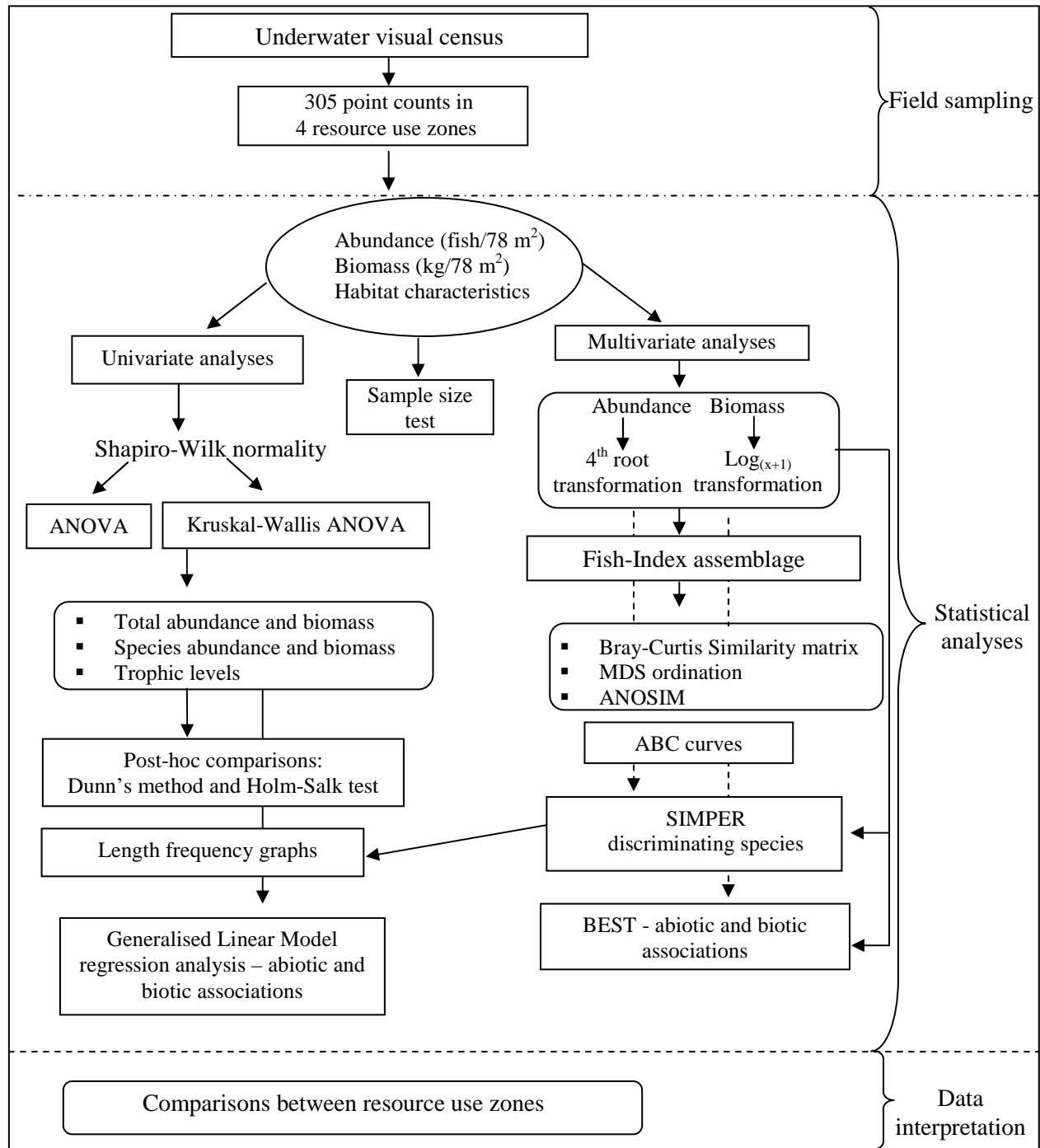


Figure 4.2 Diagrammatic summary of the sampling protocol and data analysis undertaken in the assessment of the Fish-index assemblage on South African and southern Mozambican coral reefs.

Sample size test

Power analysis was used to determine whether the minimum number of 60 point counts per reef were sufficient to provide the statistical validity needed to detect changes in population estimates. The power analysis technique was applied according to Kapadia et al. (2005). The minimum sample size required was calculated as that required to detect a 10% change in the mean estimate at a significance level of 0.05 and a power of 80% (Fairweather 1991, Length 2001).

Univariate analysis

The same univariate analyses were used to analyse the Fish-index abundance and biomass data as those described in Chapter 3.

Multivariate analysis

All multivariate analyses were analysed using PRIMER v.6 (PRIMER-E 2006).

Transformation

Similar methods to those described in Chapter 3 were used to determine the degree of transformation necessary (Fig 4.4). All abundance data were fourth-root transformed, while the $\log_{(x+1)}$ transformation was used for biomass data due to large differences in the values generated by the presence and absence of large predatory fish.

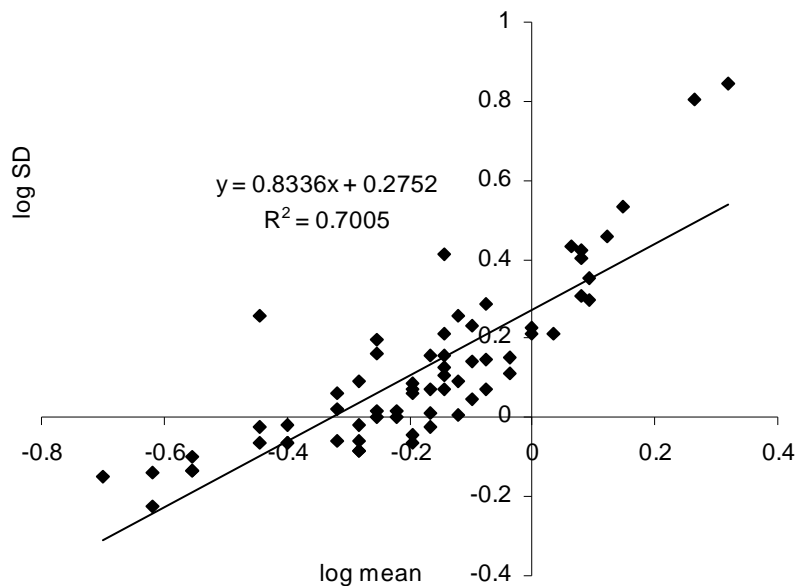


Figure 4.4 Linear regression of the log of the standard deviations of means against the log of mean fish abundance for Sanctuary zones.

Fish-index analysis

Non-metric multidimensional scaling (MDS) was used to examine differences in spatial distribution of Fish-index assemblages across the different resource use zones. Analysis of similarity (ANOSIM) was also used to confirm or refute trends observed in the MDS ordinations. An ANOSIM R-statistic of <0.25 implies that there is too much overlap for sites to be separable (Clarke and Gorley 2006). R-statistics greater than 0.450 were considered significant. Hierarchical agglomerative clustering with group-average linking was performed on the Bray-Curtis similarity abundance matrix to confirm trends observed in the MDS plots and ANOSIM results.

Discriminating species

SIMPER analysis was used to identify those Fish-index species responsible for the Bray-Curtis dissimilarity between resource use zones. Species that contribute consistently towards the average dissimilarity and have a large average dissimilarity/stand deviation ratio are potentially good discriminating species (Warwick & Clarke 2001).

Environmental variables, human resource use and community composition

A suite of two environmental variables and four habitat characteristics were recorded at each sampling site: depth, temperature, topography, coral cover, fishing intensity and diving intensity. Potential relationships between spatial patterns of fish communities and these environmental variables and habitat characteristics were examined using BEST procedure (PRIMER v.6 2006). This procedure identifies the most influential variable or combination of variables which gives rise to the largest rank correlation (p_s) among a set of independent variables (and habitat characteristics) (Warwick & Clarke 2001). All variables were normalised according to the requirements of the BEST procedure. The GLM regression analysis used to investigate the association between environmental parameters, habitat characteristics and fish assemblage metrics were similar to those described in Chapter 3.

4.3 Results

4.3.1 Spatial distribution of species

The multidimensional scaling (MDS) configurations demonstrated similar trends in the abundance and biomass data (Figure 4.5A & B). Sanctuary reefs formed clusters that were distinct from the other protection zones. ANOSIM tests confirmed the trends observed in both abundance and biomass MDS plots. In terms of abundance, the results differed significantly between Sanctuary zones and the High-Diving zone, and between Sanctuary and Diving-fishing zones (Global $R=0.423$ $p<0.001$) (Table 4.3). ANOSIM analysis of biomass data revealed that the Fish-index structure on Sanctuary reefs was significantly different to the other three resource use zones (Global $R=0.423$; $p<0.0001$) (Table 4.4).

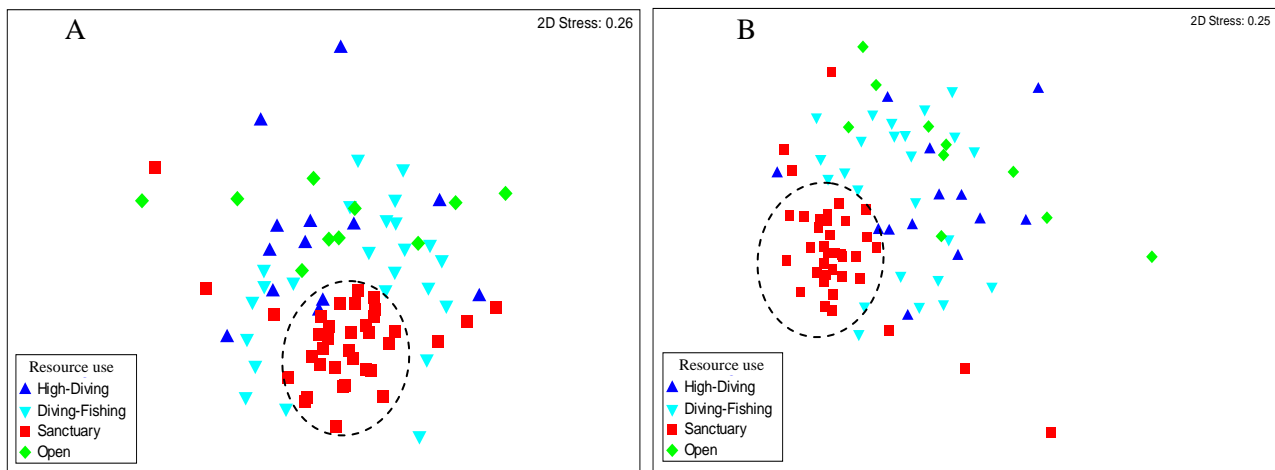


Figure 4.5 Non-metric multidimensional scaling ordination of samples based on A) fourth-root transformed abundance data and B) $\log_{(x+1)}$ transformed biomass data. The sanctuary clusters are demarcated by the dotted circle.

Table 4.3 Results of ANOSIM run on fourth-root transformed species abundance data for differences between resource use zones. Global $R=0.423$. Significance of Global $R<0.000005$. Significant differences are in bold

Pairwise Tests	R Statistic	Significance Level %
High-Diving, Fishing-Diving	0.302	0.006
High-Diving, Sanctuary	0.429	0.005
High-Diving, Open	0.087	9.2
Diving-Fishing, Sanctuary	0.461	<0.0001
Fishing-Diving, Open	0.235	0.6
Sanctuary, Open	0.6	0.0005

Table 4.4 Results of ANOSIM run on fourth-root transformed species biomass data for differences between resource use zones. Global R=0.423. Significance of Global R<0.000005. Significant differences are in bold

Pairwise Tests	R Statistic	Significance level %
High-Diving, Diving-Fishing	0.202	0.2
High-Diving, Sanctuary	0.516	0.0005
High-Diving, Open	0.162	1.7
Diving-Fishing, Sanctuary	0.468	<0.0001
Diving-Fishing, Open	0.236	0.4
Sanctuary, Open	0.713	0.0005

4.3.2 Total abundance and biomass

A combined total of 2072 fish were recorded during the study period. The total mean fish abundance and total mean biomass manifested similar trends across resource use zones with the highest values observed on Sanctuary reefs and the lowest on the Open reef (Fig. 4.6). The total mean abundance per point count ranged from 16.49 (± 4.71 SD) on Sanctuary reefs to 10.0 (± 4.76 SD) on the Open reef. Univariate data analysis revealed differences in mean abundances to be significant between Sanctuary and High-Diving, and Sanctuary and Open reefs ($p < 0.001$). In addition, mean abundances between Diving-Fishing and Open reefs were also significantly different. Total mean biomass on Sanctuary reefs (22.47 kg/78 m², ± 26.54 SD) was significantly greater than in all other resource use zones ($p < 0.001$). The biomass in the other resource use zones was at least three times lower than on Sanctuary reefs.

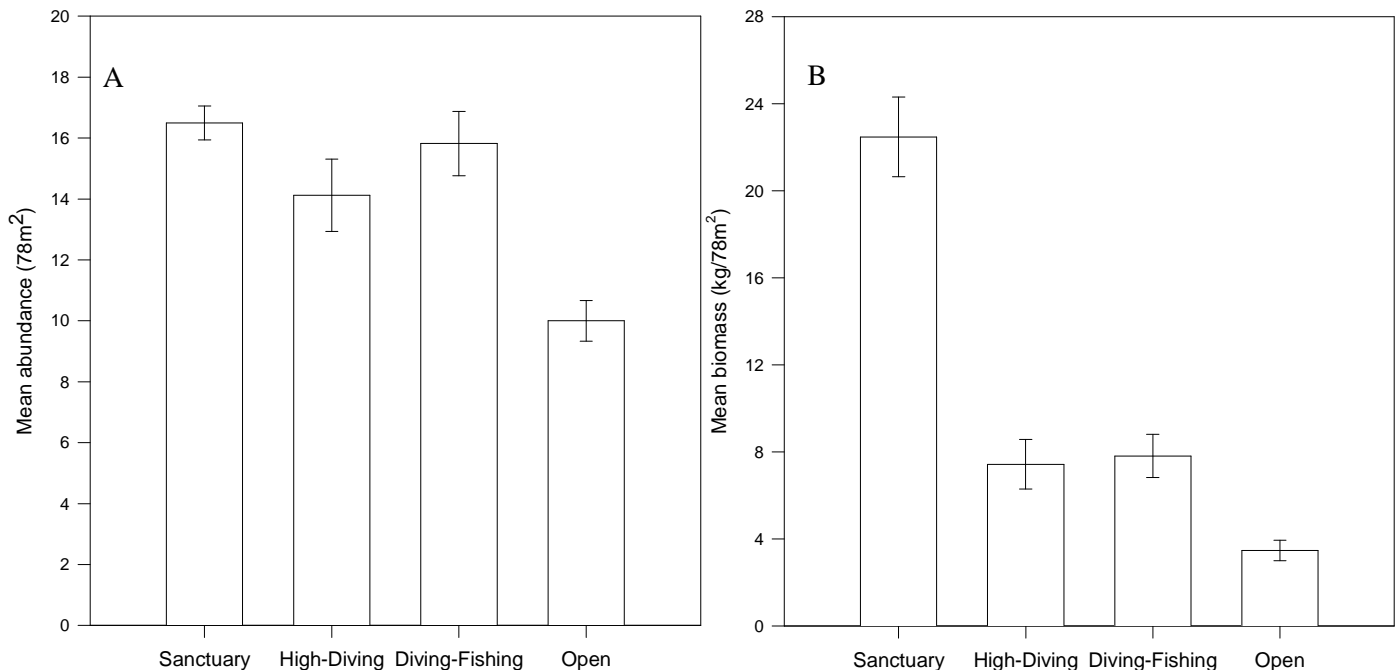


Figure 4.6 Mean values (\pm SE) for total abundance (A) and total biomass (B) per point count for the Fish-index species in each resource use zone on the South African and southern Mozambican study reefs.

4.3.3 Abundance Biomass Comparisons

Abundance biomass comparison (ABC) curves were generated to provide an insight into the relationship between species abundance and biomass across the four resource use zones. When total biomass and abundance were ranked cumulatively for each species, notable differences were evident between resource use zones (Figure 4.7). There was an apparent trend of decreasing biomass relative to abundance from Sanctuary to Open zones, which was confirmed by the decreasing W -statistic, which represents the magnitude of separation between these curves. Sanctuary zones and, to a lesser degree the High-Diving zones, had biomass curves elevated above their respective abundance curve, indicating the predominance of a few large individuals. Conversely, the biomass curves lay above the abundance curves in the Diving-Fishing and Open zones. This reversal of the curves suggested a predominance of a large number of small individuals in these two resource use zones. Despite the large difference between w -statistics in each resource use zone, *post hoc* analysis revealed differences only to be significant between the Sanctuary and Diving-Fishing zones ($p < 0.001$).

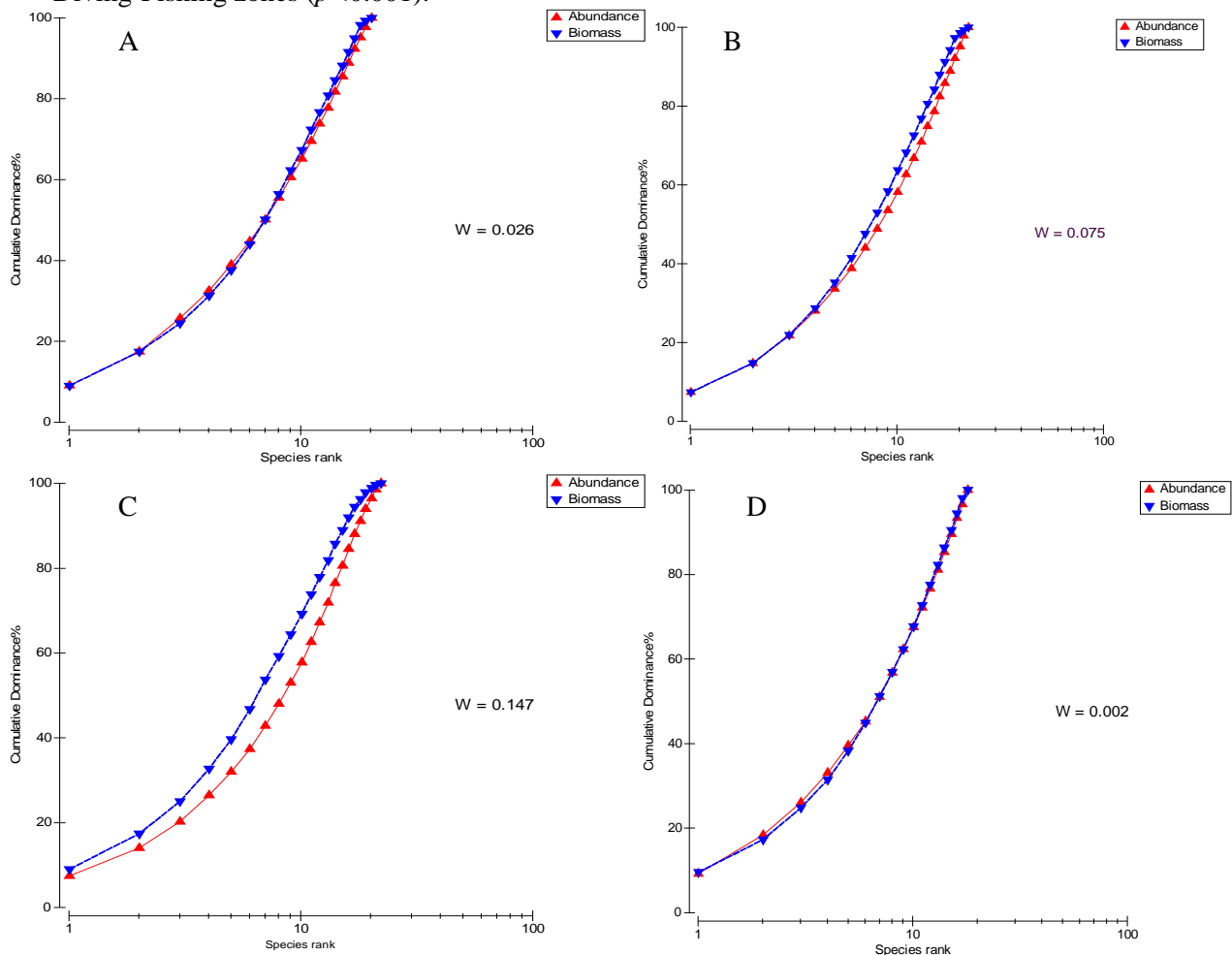


Figure 4.7 Abundance biomass comparison (ABC) plots for species totals in the different resource use zones: A) High-Fishing, B) Diving-Fishing, C) Sanctuary and D) Open. Abundance data was fourth-root transformed and biomass data were $\log_{(x+1)}$ transformed.

4.3.4 Trophic levels

The trophic composition of the Fish-index abundance and biomass data differed across the resource use zones. Six out of the eight trophic levels differed significantly in abundance and biomass in the different zones. Invertivore values were similar across the zones, contributing approximately 30% to the trophic abundance in each zone (Figure 4.8A), which was mainly due to high numbers of *Thalassoma herbraicum*. Herbivores were the second most abundant trophic group across resource use zones, except on Diving-Fishing reefs where planktivores were more numerous. Top and medium-level predators manifested a significant trend of decreasing abundance with increasing human activity ($p < 0.001$) and top predators were absent from the Open zone. Corallivore and benthivore densities also decreased from Sanctuary to Diving-Fishing zones. Corallivore abundance was significantly lower on Diving-Fishing reefs ($P < 0.001$). Planktivores were at least twice as abundant in the Diving-Fishing zones compared to the other zones. Analysis of the trophic contributions to biomass revealed converse trends to the abundance data (Figure 4.8B). Herbivore biomass contributed the most towards the trophic composition on the High-Diving, and Open reefs, while predator and herbivore biomass were equal on the Diving-Fishing reefs. In contrast, predator biomass constituted 80% of the trophic composition in the Sanctuary zones and was at least six times that of the biomass in the other resource use zone. The high biomass values on the Sanctuary reefs were due to *E. tukula*, *A. virescens* and *L. bohar*. Predator biomass varied between 30-40% on Diving-Fishing and High-Diving reefs, while only 10% of the biomass on the Open reefs was attributable to medium predators.

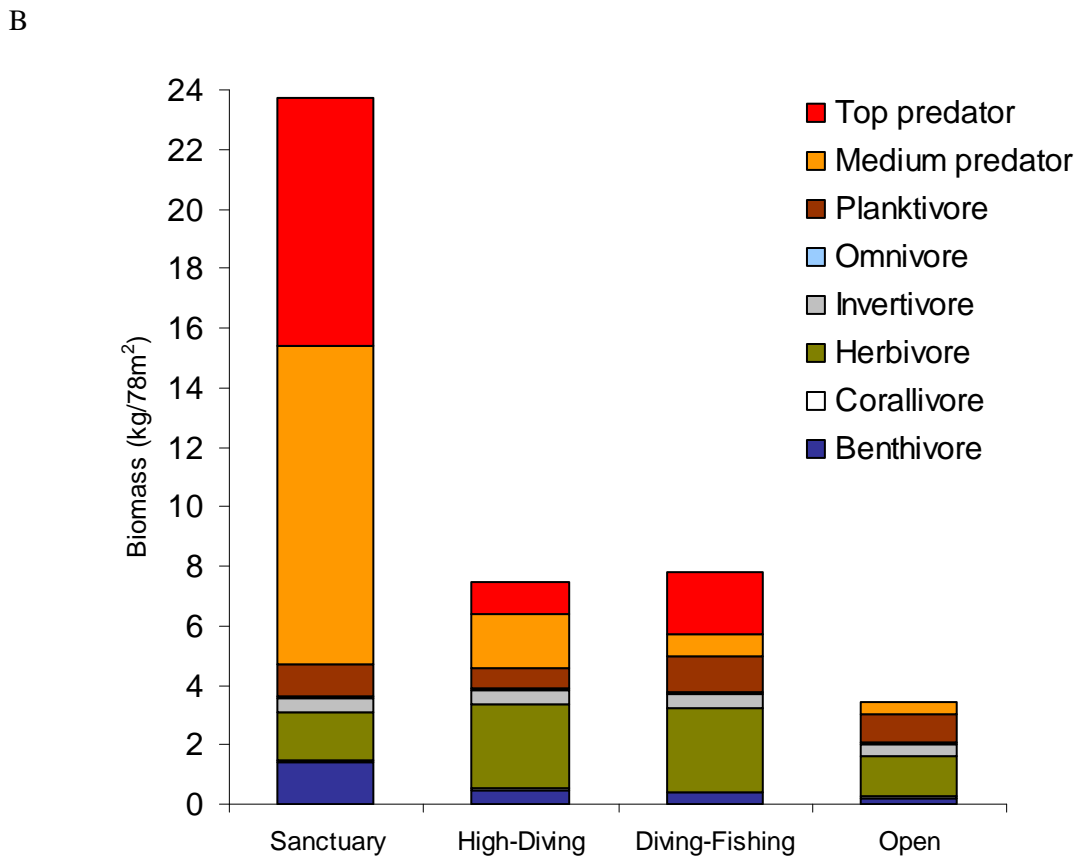
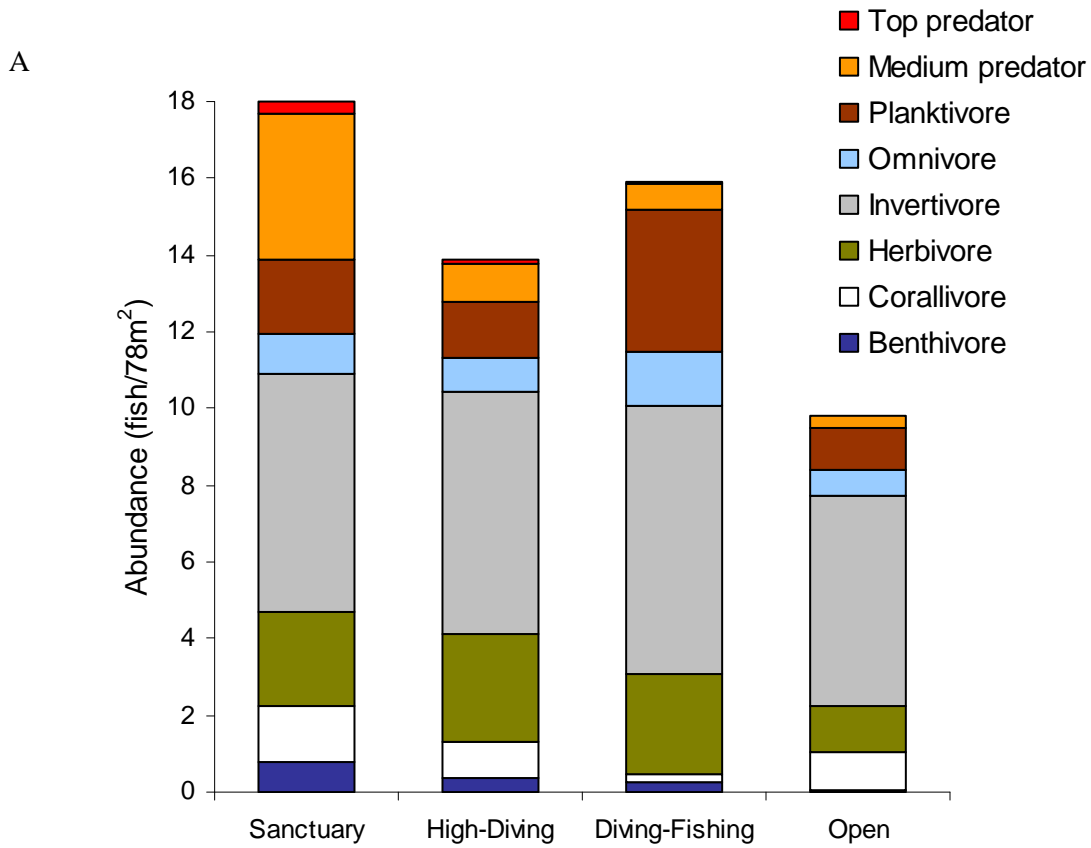


Figure 4.8 Trophic structure of the Fish-index species in the four resource use zones on the South African and southern Mozambican study reefs in terms of A) abundance and B) biomass.

4.3.5 Species distribution, abundance and biomass

The presence and absence of the 25 Fish-index species varied across the different resource use zones (Table 4.5). All species were present in the Sanctuary zones, while a total of seven species were absent from the remaining three resource use zones. The highest number of Fish-index species were absent in the Open zone. The most abundant species in all resource use zones was *Thalassoma hebraicum* and the most uncommon species were *Pygoplites diacanthus* and *Chaetodon trifasciatus*. ANOVA was used to test for differences in mean abundance and biomass between species across the four resource use zones. The statistical analyses of certain uncommon species were inconclusive. Significant differences in abundance between resource use zones were found for ten species, eight of which were most numerous in the Sanctuary reefs. The biomass of twelve species was found to be significantly different between resource use zones (Table 4.6). Eight of the 25 Fish-index species had high biomass values in the Sanctuary zones, the species with the highest biomass being *E. tukula* followed by *L. bohar*.

Table 4.5 Mean abundance (fish/78m²) of the Fish-index species and differences in their mean abundance between resource use zones shown by one-way ANOVA. Highest values are highlighted in bold. *Indicates species that were inconclusive in the ANOVA or *Post Hoc* comparison due to their uncommon status.

	Sanctuary	High-Diving	Diving-Fishing	Open	P
<i>Acanthurus leucosternon</i>	0.904	1.200	1.296	0.353	NS
<i>Amphiprion allardi</i>	0.220	0.077	0.576	0.333	<0.001
<i>Aprion virescens</i>	0.617	0.062	0.024	0	<0.001
<i>Balistoides conspicillum</i>	0.072	0.046	0.464	0.059	0.007*
<i>Bodianus Diana</i>	0.502	0.554	0.464	0.333	NS
<i>Caranx melampygus</i>	0.919	0.615	0.288	0.098	0.003
<i>Chaetodon madagaskariensis</i>	0.804	0.754	0.984	0.706	NS
<i>Chaetodon meyeri</i>	0.885	0.400	0.144	0.549	<0.001
<i>Chaetodon trifascialis</i>	0.038	0.169	0	0.137	0.007*
<i>Chaetodon trifasciatus</i>	0.024	0	0	0.078	0.007*
<i>Diplodus cervinus</i>	0.019	0.385	0	0	NS
<i>Epinephelus tukula</i>	0.292	0.077	0.080	0	<0.001
<i>Forcipiger flavissimus</i>	0.512	0.292	0.352	0.294	NS
<i>Labroides dimidiatus</i>	1.507	2.338	2.704	2.235	0.005
<i>Lutjanus bohar</i>	1.512	0.138	0.184	0.118	<0.001
<i>Odonus niger</i>	2.129	1.754	4.024	1.137	NS
<i>Oplegnathus robinsoni</i>	0.502	0.108	0.104	0.039	0.005
<i>Plectroglyphidodon johnstonianus</i>	0.464	0.369	0.032	0.235	<0.001
<i>Pomacanthus imperator</i>	0.187	0.246	0.144	0.039	NS
<i>Pygoplites diacanthus</i>	0.005	0.015	0.016	0	NS*
<i>Scarus rubroviolceus</i>	1.292	1.569	1.200	0.725	NS
<i>Siganus sutor</i>	0.072	0.215	0.080	0.137	NS*
<i>Thalassoma hebraicum</i>	3.024	2.462	2.576	2.294	NS
<i>Variola louti</i>	0.531	0.185	0.184	0.098	<0.001
<i>Zebrasoma desjardini</i>	0.014	0.092	0.024	0	NS*

Table 4.6 Mean biomass (kg/78m²) of the Fish-index species and differences in their mean biomass between resource use zones shown by one-way ANOVA. Highest values are highlighted in bold. *Indicates species that were inconclusive in the ANOVA or *Post Hoc* comparison due to their uncommon status.

	Sanctuary	High-Diving	Diving-Fishing	Open	P
<i>Acanthurus leucosternon</i>	0.125	0.145	0.193	0.036	0.047
<i>Amphiprion allardi</i>	0.004	0.001	0.016	0.012	<0.001
<i>Aprion virescens</i>	1.914	0.062	0.017	0	<0.001
<i>Balistoides conspicillum</i>	0.102	0.029	0.069	0.119	NS
<i>Bodianus Diana</i>	0.045	0.056	0.038	0.025	NS
<i>Caranx melampygus</i>	2.393	0.828	0.402	0.092	<0.001
<i>Chaetodon madagaskariensis</i>	0.020	0.046	0.031	0.031	NS
<i>Chaetodon meyeri</i>	0.058	0.031	0.017	0.046	<0.001
<i>Chaetodon trifascialis</i>	0.001	0.019	0	0.019	0.011*
<i>Chaetodon trifasciatus</i>	0.001	0	0	0.010	NS*
<i>Diplodus cervinus</i>	0.005	0.029	0	0	NS*
<i>Epinephelus tukula</i>	8.126	1.075	2.077	0	<0.001
<i>Forcipiger flavissimus</i>	0.025	0.015	0.018	0.018	NS
<i>Labroides dimidiatus</i>	0.010	0.016	0.025	0.015	0.004
<i>Lutjanus bohar</i>	4.361	0.407	0.215	0.048	<0.001
<i>Odonus niger</i>	1.079	0.731	1.244	1.000	NS
<i>Oplegnathus robinsoni</i>	1.197	0.134	0.190	0.114	0.04
<i>Plectroglyphidodon johnstonianus</i>	0.002	0.002	0.001	0.003	<0.001
<i>Pomacanthus imperator</i>	0.194	0.329	0.209	0.058	NS
<i>Pygoplites diacanthus</i>	0.000	0.001	0.001	0	NS*
<i>Scarus rubroviolaceus</i>	1.340	2.571	2.508	1.243	<0.001
<i>Siganus sutor</i>	0.050	0.133	0.066	0.125	NS*
<i>Thalassoma hebraicum</i>	0.242	0.258	0.314	0.239	NS
<i>Variola louti</i>	1.170	0.490	0.151	0.215	<0.001
<i>Zebrasoma desjardini</i>	0.005	0.021	0.009	0	NS*

4.4.7 Discriminating species

Simper revealed which species contributed to the largest dissimilarity in abundance and biomass between the four resource use zones (Appendix 4 & 5). Table 4.7 summarises the species that make the greatest contribution to the average dissimilarity between the Sanctuary zones and each of the other resource use zones. Comparisons between Sanctuary zones and the other resource use zones revealed that six species were the top contributors to the dissimilarity in each pairwise comparison. These species were *E. tukula*, *L. bohar*, *A. virescens*, *C. melampygus*, *V. louti* and *O. robinsoni* and their cumulative contribution to the dissimilarity between each comparative group was almost 50%. They comprise important predator species and species targeted by fishing.

Table 4.7 Results of SIMPER analysis. Only species providing the highest % contribution towards the average dissimilarity in abundance and biomass data between resource use zones have been included. Species highlighted in bold are those contributing 50 % to the over all dissimilarity.

	High-Diving vs Sanctuary	Diving-Fishing vs Sanctuary	Open vs Sanctuary
<i>Epinephelus tukula</i>	10.32	9.18	9.85
<i>Lutjanus bohar</i>	9.84	8.65	9.58
<i>Aprion virescens</i>	8.37	9.42	9.51
<i>Caranx melampygus</i>	6.97	6.9	7.73
<i>Variola louti</i>	7.19	6.96	7.18
<i>Oplegnathus robinsoni</i>	6.31	6.63	6.68
<i>Odonus niger</i>	5.88	7.1	6.55
<i>Pomacanthus imperator</i>	5.44	5.45	4.94
<i>Acanthurus leucosternon</i>	4.34	4.31	4.06
<i>Balistoides conspicillum</i>	3.67	4.71	3.86
<i>Chaetodon meyeri</i>	3.65	4.61	3.77
<i>Scarus rubroviolaceus</i>	3.26	3.12	3.63
<i>Bodianus diana</i>	3.74	3.99	3.35
<i>Siganus sutor</i>	2.57	2.77	3.34
<i>Forcipiger flavissimus</i>	3.44	3.25	3.25
<i>Chaetodon trifascialis</i>	2.12	0	2.47
<i>Amphiprion allardi</i>	0	3.28	2.36
Average dissimilarity	45.01	44.11	52.49

Length frequency graphs of core indicator species

Length-frequency graphs were generated to investigate the size class distribution of the six species responsible for the dissimilarities between resource use zones (Table 4.7). Statistical comparisons were inconclusive for certain species between resource use zones due to the low abundances and absence of these species from non-Sanctuary zones. Nevertheless, a clear trend was evident in the monotonic decrease in the abundance and mean body size of all these species from Sanctuary to Open zones. The results revealed varying responses of the species to fishing and diving activities. Most of the species appeared to be susceptible to a combination of fishing and diving activities; however, a few species manifested a direct response to a particular activity.

Target species

Fishing was positively associated with reduced numbers of *A. virescens* as only five individuals were recorded in the Diving-Fishing zone, while no individuals were recorded in the Open zone (Figure 4.9A). In contrast, an average of 48 *A. virescens* was recorded per Sanctuary reef. *A. virescens* abundance was also low on the High-Diving reefs. The mean body size of *A. virescens* on Sanctuary reefs was 1.5 and 1.7 times larger than on High-Diving and Diving-Fishing reefs, respectively. Not surprisingly, fishing and diving were similarly associated with low numbers of another target species, *C. melampygus* (Figure 4.9B). The mean body size for this species was 1.3 times higher on Sanctuary reefs. Low densities of sexually mature individuals were observed in the

High-Diving and Diving-Fishing zones, while none were recorded on the Open reef. The density peak observed on the High-Diving zone was due to a single sighting of a large school of small individuals (30 cm).

Non-target species

The remaining four species were all observed to have higher densities and body size in zones distant from human activities; however, the correlations with diving and fishing activities in this regard appeared to differ. Large individuals of *V. louti* were more numerous on the High-Diving reef compared to the reefs in the other three resource use zones (Figure 4.9D). *E. tukula* was uncommon on the High-Diving reef, but was present in higher densities on the Fishing-Diving reefs (Figure 4.9E). No *E. tukula* were recorded on the Open reef during the study period. The length distribution of *L. bohar* differed in the non-Sanctuary zones (Figure 4.9F). Large individuals were uncommon in all of these zones, and no individuals larger than 20 cm were observed in the Open zone. *O. robinsoni* appeared to be equally affected by diving and fishing as abundances and their mean body size was highest in the Sanctuary zones (Figure 4.9C).

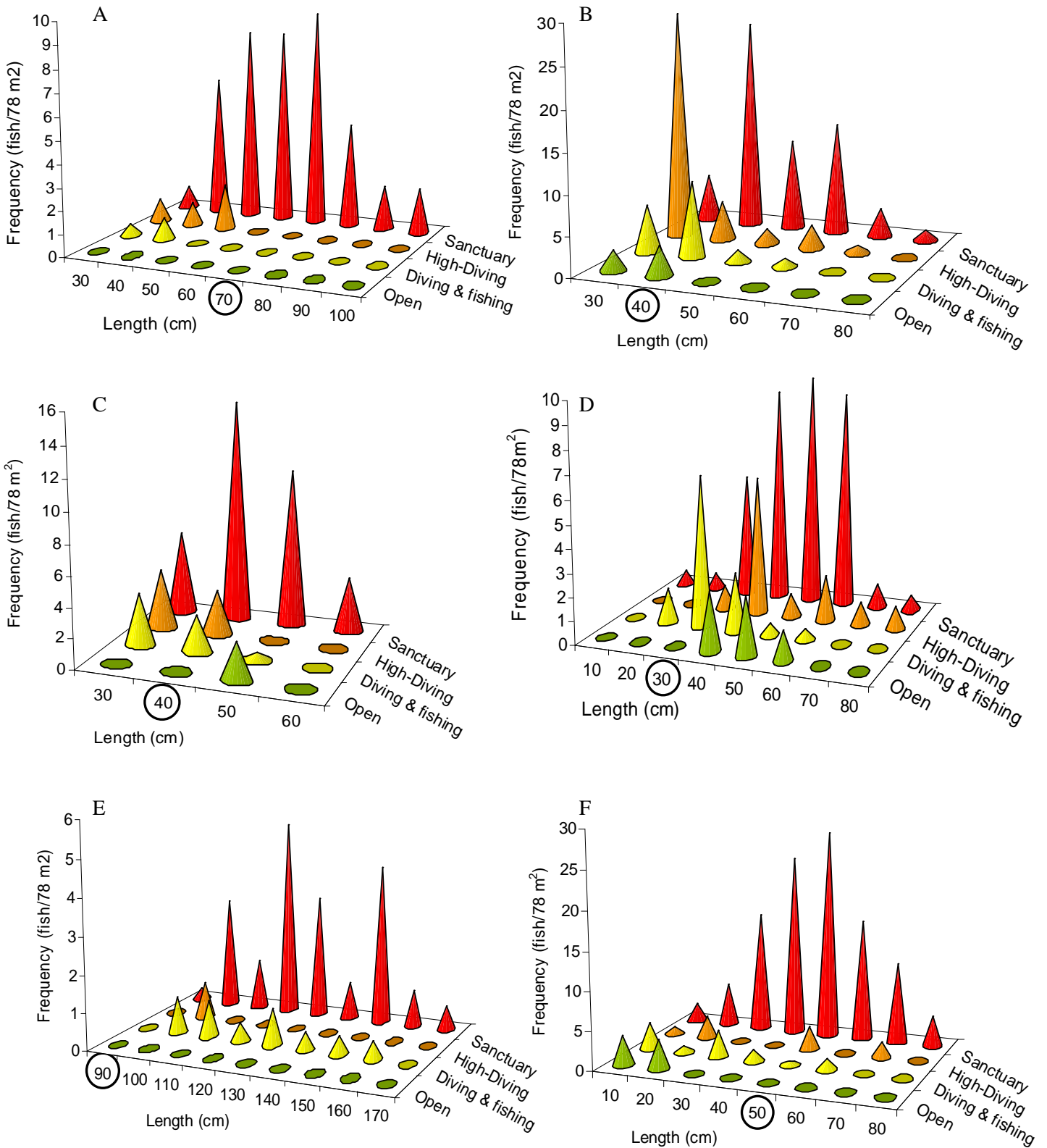


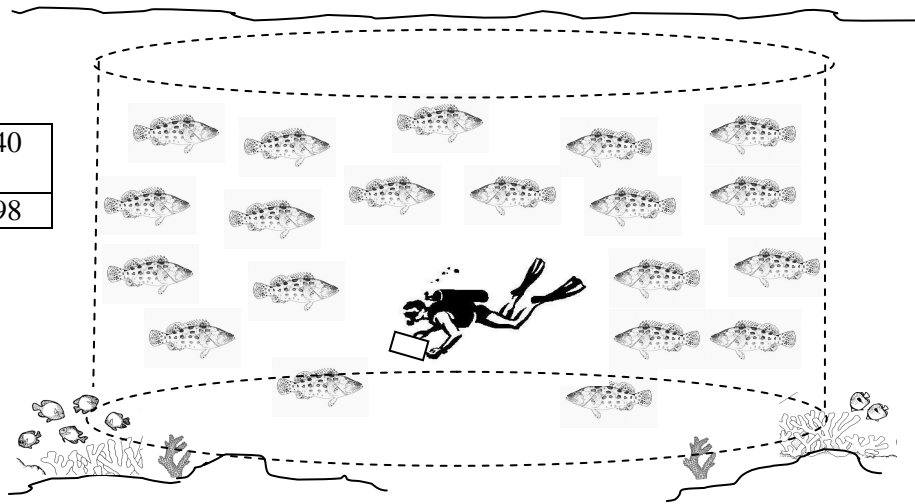
Figure 4.9 Length frequency distribution plots of target species and non target-predators on reefs in the four different resource use zones. A) *Aprion virescens*, B) *Caranx melampygus*, C) *Oplegnathus robinsoni*, D) *Variola louti*, E) *Epinephelus tukula* and F) *Lutjanus bohar*. Circles indicate mean length at sexual maturity.

4.3.8 Investigating diver-orientated response in a top predator

Figure 4.10 illustrates the results of comparisons the community count versus point count density estimates for *E. tukula* in each of the South African protectin zones. On the High-Diving reefs, (~54 000 dives/year), two *E. tukula* were recorded in the community counts, while only one *E. tukula* was recorded in the point count area. On the Diving-Fishing reefs (~ 2300 dives/year), higher *E. tukula* abundances were recorded in the community counts, but only a few individuals were recorded in the point counts. In contrast, at least three times as many *E. tukula* were recorded in the community counts in the Sanctuary zones and almost all of these individuals were recorded in the point counts. These results are mirrored in the approach ratios which were close to one for Sanctuary zones (0.98, Fig. 4.10A).

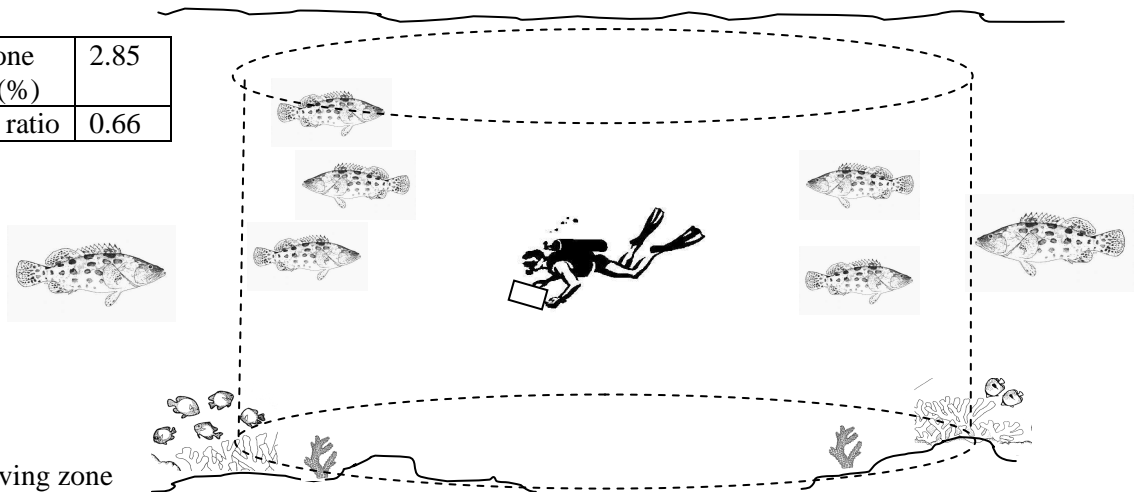
A) Sanctuary zones

Area of zone surveyed (%)	0.40
Approach ratio	0.98



B) Diving-Fishing zones

Area of zone surveyed (%)	2.85
Approach ratio	0.66



C) High-Diving zone

Area of zone surveyed (%)	0.45
Approach ratio	0.5

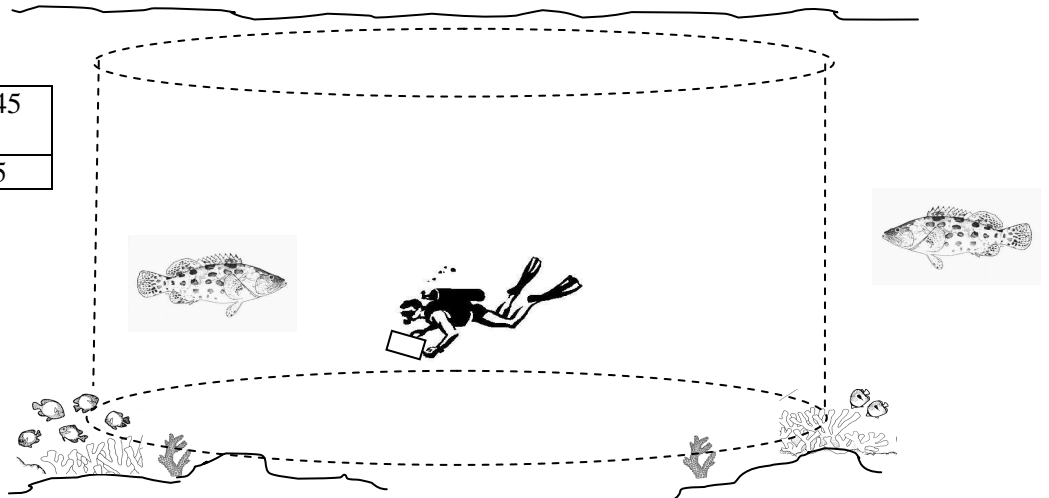


Figure 4.10 Comparison of restricted versus unrestricted visual census data for *E. tukula* on reefs in three different resource use zones. Unrestricted counts are expressed as abundance per hour and restricted counts are expressed as abundance per 78 m².

4.3.9 Factors influencing species distribution: habitat characteristics and human resource use

Synthesis of the effects of habitat characteristics and human use on the fish communities revealed that anthropogenic activities (fishing and diving) appeared to be more accountable for the variance in Fish-index abundance and biomass than depth, topography and coral cover (Table 4.8 & 4.9). The Generalised Linear Model (GLM) regression analysis revealed that both fishing and diving intensity had a significantly negative affect on the Fish-index abundance and biomass. According to the GLM, these activities were responsible for more than 40% of the variance in the total abundance and biomass ($P < 0.001$) between zones. Combined, fishing and diving appeared to account for 46% and 60% of top- and medium-level predator abundance and biomass respectively. These two variables also appeared to explain >50% of the variance observed in corallivore abundance and biomass. The GLM analysis further identified topography as the only significant habitat characteristic accounting for >10% of the variance in any of the recorded fish abundance parameters (herbivore abundance; 10.8%, $P = 0.004$). In terms of biomass parameters, depth was the most influential recorded environmental variable (corallivore biomass; 18.8%, $P = 0.001$).

Table 4.8 Results of Generalised Linear Model regression analysis of habitat characteristics and human use on the Fish-index abundance. Abundance is expressed as fish/78m².

Dependent variable	Independent variable	F statistic	Wald statistic	Percentage variance	P
Total fish abundance	Diving	15.23	30.45	25.5	<0.001
	Fishing	8.61	17.22	15.5	<0.001
Top predator abundance	Diving	12.56	50.2	35.8	<0.001
	Fishing	6.56	13.11	11.8	0.002
Medium predator abundance	Diving	16.01	64.04	42.0	<0.001
	Fishing	15.15	30.31	25.4	0.002
Corallivore abundance	Fishing	22.6	44.11	33.7	<0.001
	Diving	17.03	34.07	27.9	<0.001
Benthivore abundance	Fishing	8.25	16.5	14.9	<0.001
	Diving	4.44	17.78	14.2	0.003
Herbivore abundance	Diving	5.9	23.59	19.1	<0.001
	Topography	6	12	10.8	0.004
	Fishing	4.01	8.01	6.8	0.022
Planktivore abundance	Diving	2.58	11.52	8.3	0.028
Omnivore abundance	Diving	2.61	10.42	7.2	0.04
	Fishing	3.7	7.39	6.1	0.03

Table 4.9 Results of Generalised Linear Model regression analysis of habitat characteristics and human use on the Fish-index biomass. Biomass is expressed as kg/78m².

Dependent variable	Independent variable	F statistic	Wald statistic	Percentage variance	<i>P</i>
Total fish biomass	Diving	12.19	48.38	34.8	<0.001
	Fishing	11.79	23.58	20.6	<0.001
Top predator biomass	Diving	13.55	54.22	37.7	<0.001
	Fishing	6.46	12.92	11.6	0.002
Medium predator biomass	Diving	12.74	50.98	36.1	<0.001
	Fishing	13.72	27.45	23.5	<0.001
Corallivore biomass	Diving	9.4	37.61	28.8	<0.001
	Fishing	15.15	30.31	25.4	<0.001
	Depth	4.21	25.27	18.8	0.001
Benthivore biomass	Fishing	7.08	14.16	12.8	0.001
Herbivore biomass	Diving	3.17	12.68	9.5	0.018
	Topography	4.79	9.58	8.4	0.011
	Fishing	3.67	7.34	6	0.019
Omnivore biomass	Topography	3.95	7.9	6.6	0.02
	Fishing	3.15	6.29	5	0.05

4.4 Discussion

The effect of human activities

Fishing

The results represent the first investigation of the effects of recreational fishing on fish communities on South African and southern Mozambiquen coral reefs. The fishing intensity on the coral reefs in these two regions differs considerably. Although recreational fishing is the only form of fishing permitted in both regions, only gamefish species may be legally targeted on South African coral reefs, while the only reef-associated species completely protected from recreational fishing in southern Mozambique is *E. tukula* (Decree Nr 51/99). Each of the six core indicator species identified through the SIMPER analysis showed a decrease in abundance and biomass from Sanctuary reefs to the Open reef, highlighting the differences between protected and unprotected fish communities. Increased mortality and low densities of target fish species are expected consequences of fishing (Russ & Alcalá 1989). Carangids, lutjanids and serranids are families commonly targeted by fishers on coral reefs throughout the world due to their large size (Jennings et al. 1999) and historical records from Ponta Malongane support this trend as 24% of the angling catch-composition in 1996 constituted species from these three families (van der Elst et al. 1996). Thus, it is most likely that the low densities of the core indicator species in southern Mozambique are a result of unrestricted fishing.

The absence of two conspicuous reef-associated predatory species, *A. virescens* and *E. tukula*, on the Open reef poses numerous questions. Only one reef was sampled in southern Mozambique which raises the possibility that low sampling effort rather than human activity may account for the low abundance of these species. However, similar sampling effort on the Sanctuary reefs yielded high abundances of *A. virescens*. This species is a preferred target species for spearfishers and anglers (van der Elst et al. 1996). Thus the absence of this species on the Open reef is most likely related to high levels of human activity.

It is, however, unclear why *E. tukula* was also absent from the southern Mozambique study reef. Pereira (2003) recorded *E. tukula* on two out of six study reefs at Ponta Malongane and a popular dive site named 'Bass city' confirms the presence of *E. tukula* in the region. A number of possibilities may explain this result. Fishing cannot be completely excluded because industrial, semi-industrial, artisanal and subsistence fishers have been noted to catch this and other reef fish species in the southern Mozambique (Marcos Pereira pers. coms). Alternatively, the differences in *E. tukula* densities may be related to the inherent variability associated with UVC techniques

(Samoilys & Carlos 2000). However, similar sampling effort was conducted on each of the study reefs, some of which yielded high abundances of *E. tukula*. In addition, the results of the power analysis revealed that the sample size per reef yielded a power of 0.84, suggesting that a minimum of 60 point counts was sufficient to detect differences in fish communities between the resource use zones. It is possible that the presence of only one Open reef represented unbalanced sampling in the study. This shortcoming was unavoidable due to limited access to the reefs in southern Mozambique and it is recommended that future studies should include other reefs in southern Mozambique to refute or confirm these findings.

The low densities of sexually mature target species (Figure 4.9) on the southern Mozambican reef highlights the need for marine regulations protecting the coral reef fish communities. In July 2009, a new Marine Protected Area was proclaimed (Decree 42/2009 of 21 August): the Ponta do Ouro Partial Marine Reserve (PPMR). The PPMR is effectively an extension of the Maputaland MPA (Figure 4.11) and stretches north along the coast to Inhaca Island in Delagoa Bay. The PPMR has been zoned to include multiple resource use areas as well as sanctuary and restricted areas. Although marine laws are now in place to provide legal protection to the various ecosystems such as coral reefs, there is still no specific regulation of fishing activities other than the prohibition of bottom-fishing. Additional regulations such as daily quotas and minimum size limits will be necessary to protect the fish communities on southern Mozambican coral reefs. It is thus anticipated that the fish data from this study will provide valuable baseline information for the Mozambican authorities in the development of the PPMR management plan.

Efforts have been made to regulate fishing on the South African coral reefs by restricting the extraction of species to gamefish. In addition to the species restrictions, the Marine Living Resources Act (MLRA) stipulates daily fishing quotas for each target species. *C. melampygyus* has a daily quota of five, while *A. virescens* has a daily quota of ten. However, a total bag limit of ten fish per day may not be exceeded. There are no minimum size limits for these species. Despite the daily quotas, the results presented here suggest that fishing is having an effect on populations of the aforementioned target fish species. Low densities and small mean body sizes of target species are detectable effects of over-fishing on coral reefs (Russ & Alcala 1989). In the Diving-Fishing zones, the reduced mean size of target species corresponded closely to theoretical predictions (see Dulvy et al. 2004b, Graham et al. 2005), with all target species showing significantly smaller mean sizes.

A reduction in the mean size of targeted species in fished areas is termed size selective fishing or 'age truncation' and is due to fishers targeting larger individuals (Berkeley et al. 2004). Age truncation can have an important effect on fish assemblage structure and function, potentially affecting the productivity and resilience of fish populations (Baskett et al. 2005). Body size is related to life history traits such as fecundity, growth rate and age at maturity (Hutchings 2002). Consequently, fecundity is a power function of body length in fishes (Harmelin et al. 1995). Thus, resident populations lacking sufficient densities of large individuals may not be able to reproduce, particularly with increased mortality due to fishing.

The low abundance of sexually mature individuals recorded in the Diving-Fishing zones could have significant ramifications for future generations of targeted species and indirect implications for non-target species. In the Diving-Fishing zones, sexually mature *C. melampygyus* occurred at low densities, while only immature *A. virescens* were recorded. Thus, populations of either of these species may be reliant on juveniles from surrounding no-take zones to replenish stocks. Sanctuary or no-take zones have been advocated as areas of high reproductive output because there are greater densities of larger, sexually mature fish present. It has been further suggested that the increased reproductive output, whether in the form of eggs, larvae or juvenile fish, may repopulate areas open to fishing (Berkeley et al. 2004, Francini-Filho & Moura 2008). Whether this occurs, depends on the location of Sanctuary zones, the oceanographic conditions in the region, larval dispersal and larval life history characteristics (Watson et al. 2009). It was not within the scope of this study to demonstrate whether 'spill-over' is taking place from the Sanctuary zones. However, it does highlight the need for such studies in light of the extractive resource use that is taking place within the non-Sanctuary zones.

The results of this study suggest that, despite the location of South Africa's coral reefs within a MPA, certain targeted or gamefish species are of reduced mean size and density in areas open to fishing. The MLRA states that gamefish include pelagic bony fish of the families Scombridae, Carangidae, Pomatomidae, Coryphaenidae, Rachycentridae, Xiphiidae, Ostiophoridae and Sphyrnidae, the species *Aprion virescens*, as well as pelagic cartilaginous fish of the families Carcharinidae, Isuridae, Sphyrnidae, Alopiidae and Odontaspidae (Section 3.1 (G) Regulation R1429). There is no definition for gamefish in this regulation, which has allowed for much speculation on the reasoning behind the inclusion of these particular species and families. Of the two target Fish-index species, *A. virescens* appeared to be most susceptible to fishing pressure. Commonly known as kaakap in South African waters, this species is frequently targeted by anglers and spear-fishers. However, the reasons for its inclusion as a gamefish species are unclear as it is the only lutjanid included in this category.

According to van der Elst (2008), the definition of ‘gamefish’ is a pelagic fish that is actively pursued by anglers because of its fighting ability. Pelagic, by definition, refers to species living in the open ocean (Allaby 1992) and suggests species that are transient in their movements. Recent studies have found *A. virescens* to have site fidelity to core areas on atolls in the Northwestern Hawaiian Islands (Meyer et al. 2007). Furthermore, Pilling et al (2000) suggested that length-based growth estimates for *A. virescens* may have overestimated mean growth rates and asymptotic length. This implies that lutjanids may have slower growth rates and a longer life-span than indicated by previous length-based assessments and they may therefore be vulnerable to overfishing. With these new insights on the ecology and biology of *A. virescens*, it is recommended that the classification of this species as a gamefish be reassessed due to its vulnerability as a target species.

The other target species, *C. melampyus*, has also been shown to display strong site fidelity on reefs (Holland et al. 1996). Acoustic telemetry studies provided empirical evidence that the dispersal of *C. melampyus* is much less than might have been predicted for a highly mobile, predatory species (Holland et al. 1996). The growth rate of *C. melampyus* is faster than that of *A. virescens* (Fishbase 2009); however, the resident nature of *C. melampyus* warrants implementation of a minimum size limit for this species to prevent further population depletion. The trends manifested by these target species emphasise the need to include biologically-based criteria for the selection of species as gamefish. The new definition and recommendations for effective management of reef-associated target species will be dealt with in the Management Guidelines section in Chapter 5.

SCUBA diving

Due to increased mortality of target species associated with fishing, it was anticipated that the densities and biomass of these species would be lowest in the Diving-Fishing zones and similar in the Sanctuary and High-Diving zones. Differences between Sanctuary and Diving-Fishing zones were significant; however, so too were the differences between Sanctuary and High-Diving zones. The High-Diving zone is designated as a no-take area and thus the differences in densities of the target species require closer examination. There are two possible explanations. Firstly, there may be a lack of compliance by the fishers in the High-Diving zone. The lack of ‘fenced’ boundaries in marine reserves makes it difficult to clearly demarcate restricted areas. Illegal fishing may be occurring along the northern perimeter of the High-Diving reef, but the high volume of divers on this reef would make it likely that suspicious fishing activity would be reported to the authorities. Secondly, and more likely, the high diving intensity in this zone may be having a negative effect on the abundance of the target species.

Chater et al. (1995) conducted a comparison of selected fish species on the reefs categorized as High-Diving and Sanctuary zones. Although different visual techniques were employed to this study, 13 of the Fish-index species were included in the Chater et al. (1995) study. This allows for a temporal comparison of mean fish abundance between High-Diving and one of the Sanctuary reefs (Leadsman Shoal) (Table 4.10). Almost twenty years ago, six of the Fish-index species were more abundant on the High-Diving reef compared to the Sanctuary reef. At the time of this study, only three species were more abundant in the High-Diving zone.

Whether this is due to an increase in species densities in the Sanctuary zones or a decrease in abundances in the High-Diving zones is unclear and the reasons for the changes must be speculative. The reefs are approximately 30 km apart so it is likely that any oceanographic changes over the past 30 years would have affected both sites equally. Both of the reefs in question are situated within the St Lucia MPA which was proclaimed in 1979 (Notice P 35/79). Fishing is prohibited on both of the reefs, while SCUBA diving is permitted only on the High-Diving reef. SCUBA diving intensity increased by a factor of six between 1987 and 1996, when it reached approximately 100 000 dives/year (Schleyer 2000), but it currently stands at 54 000 dives/year (Pieters 2009). The changes in SCUBA diving intensity may not be directly linked to the observed changes in the species abundances; however, it does stress the need for long-term monitoring studies on these reefs to assess trends in fish assemblages in the different resource use zones.

Table 4.10 Temporal comparisons of selected fish species in High-Diving and Sanctuary zones on South African coral reefs. Years in parenthesis are dates of fieldwork. The symbol x denotes in which zone the mean abundance of a species was highest in the study period.

Species	Chater et al. 1995 (1987-1992)		This study (2007-2009)	
	High-Diving zone	Sanctuary Zone	High-Diving zone	Sanctuary zone
<i>Acanthurus leucosternon</i>	x		x	
<i>Amphiprion allardi</i>		x		x
<i>Aprion virescens</i>		x		x
<i>Bodianus diana</i>	x		x	
<i>Chaetodon madagaskariensis</i>	x			x
<i>Caranx melampygus</i>	x			x
<i>Diplodus cervinus cervinus</i>		x	x	
<i>Epinephelus tukula</i>	x			x
<i>Forcipiger flavissimus</i>	x			x
<i>Lutjanus bohar</i>		x		x
<i>Oplegnathus robinsoni</i>		x		x
<i>Thalassoma herbraicum</i>		x		x
<i>Variola louti</i>		x		x

Of the six core indicator species, sexually mature *Variola louti* occurred in greater densities on the High-Diving reef; however, these were by no means similar to values recorded in the Sanctuary zones. The low densities of the other predatory species suggest that factors such as high diving pressure may be influencing the fish assemblage in this zone. The low densities of the top-level predator, *E. tukula* on the High-Diving reef, were of particular concern on the High-Diving reef. As one of the most significant large-bodied predators on South African coral reefs, *E. tukula* or potato bass play a key role in structuring coral reef communities and a large population is indicative of a thriving and productive reef community (Bohnsack et al. 1994, Costa et al. 2003). Serranids and particularly *E. tukula* are highly sought after for the live-fish food trade and reefs in many oceanic regions have depleted or no serranids due to over-fishing (Beet & Friedlander 1992, Sadovy & Colin 1995, Costa et al. 2003). In contrast, it is unlikely that the low densities of *E. tukula* can be attributable to fishing on the High-Diving reefs for two reasons. *E. tukula* is classified as a no-take species and the High-Diving reef is a no-take zone. Nevertheless, historic records and anecdotal accounts of higher abundances of *E. tukula* on the High-Diving reef during the 1980s to early 1990s (Koornhof 1991, Chater et al. 1995) indicate that a reduction in population numbers has occurred during the last three last decades. It is suggested that this decline is linked to persistent high diving intensity. In spite of the fact that diving intensity has decreased in the last decade, the current diving level on the High-Diving reef is still amongst the highest in the world. *E. tukula* is a large resident predator known to show aggressive territorial behaviour towards divers on reefs isolated from human activities (Delbelius 2001, Peschak 2009). SCUBA divers may be viewed as large predators by *E. tukula* and thus seen as a threat. In this study, all *E. tukula* encountered on Sanctuary reefs displayed aggressive or curious behaviour towards the divers which included open mouth displays, bumping of divers, biting of the buoy-reel and stalking of divers throughout the dive. In contrast, *E. tukula* behaviour on the Diving-Fishing reef was cautious and the divers were seldom approached. Furthermore, *E. tukula* were most commonly observed at the edge of diver visibility where they moved from one overhang to another.

The comparison of density data from the two UVC techniques provided insight into the different behavioural responses of *E. tukula* to varying levels of diving intensity. It is suggested that similar numbers of *E. tukula* recorded in the two methods were indicative of a natural behavioural response, while *E. tukula* recorded only in the community count were indicative of individuals affected by diver presence. On the High-Diving reef, a total of two *E. tukula* were recorded in the community counts despite the same percentage of reef being surveyed as on the Sanctuary reefs where 20 of these fish were recorded. Similar numbers of *E. tukula* on the Sanctuary reefs were recorded in the point counts, while one of two *E. tukula* approached the diver on the High-Diving reef. The decreasing gradient of *E. tukula* densities with increasing diving pressure coupled with the

different behavioural responses suggests that high diving intensity may be causing diver-orientated behaviour in this large resident fish species. The almost constant presence of divers on the High-Diving reef (approximately 150 dives per day) may be influencing the territorial nature of *E. tukula* and appears to be affecting its abundance. .

Few studies have documented the behavioural responses of fish to diver presence alone. In their investigation of the biases induced by underwater visual census techniques, Kulbicki (1998) measured the behavioural response of 293 coral reef species to diver presence on reefs of different disturbance levels. This author reported that most species avoided the observer as the disturbance intensity increased. Other studies have observed altered fish behaviour indirectly. Bohnsack (1983) reported how spearfishing not only reduces the number of large predatory species by extraction, but that target species become agitated and secretive in their habits in the presence of a diver. Similarly, Friedlander & DeMartini (2002) observed large, highly-prized parrotfish to show conditioned aversion to divers in the main Hawaiian islands, which the authors concluded was a response to the high spear-fishing intensity in the region. In an assessment of the effects of recreational SCUBA diving on Caribbean fish communities, Hawkins et al. (1999) reported a significant difference in serranid abundance in high-versus low-use areas. However, when fish size was converted to biomass, the difference was not significant.

The paucity of studies documenting the effects of diving on fish behaviour makes the interpretation of trends difficult. Recreational activities, particularly diving, are increasing in popularity on most reefs throughout the world (Davis & Tisdell 1995). It is important that reef managers and scientists are cognisant of the fact that escalating diving intensity has the potential to become a threat to coral reef fish communities and make allowance for this in their future management strategies. The results on *E. tukula* constitute an important finding in this regard.

Prey species and reef protection

Marine Protected Areas are known to protect marine ecosystems against direct and indirect effects of fishing (Botsford et al. 1997, Pauly et al. 1998). Predatory fish species are the most significant consumers of fish biomass on coral reefs, and their removal by fishing has been suggested to influence the abundance and body size distribution of their prey (Hixon & Beets 1993, Graham et al. 2003). Fishing down food webs by removing higher trophic levels can result in relaxation of top-down control and can lead to an ecosystem dominated by lower trophic guilds, which is termed a phase shift (Pauly et al 1998). Evidence for an increase in prey abundance (prey-release) in the absence of predators has been well documented for closed systems such as lakes (Jones 1982), but

the evidence in coral reef fish communities is equivocal (Jennings & Polunin 1997, Russ & Alcala 1998).

Russ and Alcala (1989) suggest two possible reasons why the removal of top predators will not necessarily result in prey-release. Firstly, many predators are generalists and the high diversity of these species on coral reefs will restrain any populations of prey species from becoming abundant. Secondly, predation may not be the ultimate process governing population densities of coral reef fish. The southern Mozambican coral reef had the lowest densities of predatory fish species; however, the only potential prey species with a high abundance on this reef was *Chaetodon trifasciatus*. *C. trifasciatus* is considered uncommon on South African reefs because it is at the southernmost extreme of its distribution (King 1996), which may explain why its abundance was slightly higher on the southern Mozambican reef. Thus, the absence of large predators such as *E. tukula* and *A. virescens* did not appear to result in prey-release on the Open reef.

On South African coral reefs, there appeared to be some compensatory response to low densities of predators as 14 potential prey species occurred at higher abundances and biomass outside the Sanctuary zones. Both of the High-Diving and Diving-Fishing zones contained larger numbers of smaller-bodied species such as labrids, chaetodons and acanthurids. Similarly Graham et al. (2003) reported the negative effect of protection on a selection of non-target species, which they attributed to increased predation. Interestingly, *Scarus rubroviolaceus* were more abundant and of significantly greater mean length in the High-Diving zone. Large-bodied scarids at the Abrolhas Islands were also found to more abundant in the non-fished areas (Watson et al. 2007). The authors attributed the higher abundances of these large-bodied parrotfish species to a reduction in the abundance of their small-bodied, non-target competitors for habitat and food. There was no such reduction in the abundance of other herbivores such as *Siganus sutor* and *Acanthurus leucosternon* on the South African reefs, making such an explanation unlikely for the presence of large scarids in the High-Diving zone. It is more likely that *S. rubroviolaceus* was able to attain greater abundances and length due to reduced predator densities in the High-Diving zone. The reason that *S. rubroviolaceus* did not attain such a large body size on the Open reef may be due to size-selective fishing. The low abundances of potential prey species in Sanctuary zones and their higher abundances in zones of reduced predation may constitute evidence of an important structuring force in predator-prey relationships on South African coral reefs.

Habitat characteristics

In order to show the effect of protection on fish community assemblages, it is necessary to assess fish communities prior to and after the implementation of any protective legislation (Francini-Filho

& Moura 2008). No studies were conducted on South African coral reefs prior to the declaration of the Maputaland and St Lucia Marine Reserves, and the differences observed in the Fish-index assemblages between the resource use zones may be attributable to factors other than the different intensities of human activity. Among the most important documented factors that influence fish community structure are benthic composition (Galzin et al. 1994, Jennings et al. 1996, Garpe & Öhman 2003, Wilson et al. 2009), topographic complexity (Öhman et al. 1997, Friedlander & Parrish 1998, Öhman & Rajasuriya 1998) and depth (Friedlander & Parrish 1998, Sherman et al. 1999).

Assessment of the fish community structure on South African coral reefs (see Chapter 3) revealed that habitat characteristics such as topography, coral cover and depth were not significant contributors to the variation in fish assemblage structure between reefs. However, investigating potential abiotic-biotic relationships using a large number of species involves a high level taxonomic complexity, which may cause certain correlations to appear ambiguous. Yet, pooling of the data to lower the taxonomic complexity (family and trophic level) did not change the significance of the habitat effects on the fish community parameters. Multiple regression analysis of interactions between the habitat variables and the Fish-index species confirmed these findings. Thus, the fish community assessment and the Fish-index study suggest that, while habitat characteristics are important in structuring fish communities, the influence of human activities on South African and southern Mozambican reef fish populations are at present more significant. Nevertheless, long-term research on the links between fish assemblage structure and habitat characteristics would further elucidate trends in the fish communities on the South African coral reefs.

Inverted biomass pyramids and Benchmark ecosystems

Data collected in the Sanctuary zones provided insight into the fish community structure on South African coral reefs in the absence of human activities. The biomass densities of predators in these zones (Figure 4.7B) constituted 80% of the total fish biomass, which were three-fold greater than those on the non-Sanctuary zones where prey biomass was dominant. A typical fish biomass pyramid on most coral reefs throughout the world, including the reefs open to diving and fishing in South Africa, has high densities of small prey species and low densities of medium- or large-sized predators (Sandin et al. 2008). The significantly greater biomass of predators recorded on South African Sanctuary reefs may represent an example of an inverted biomass pyramid. Such a reversal of predator-prey biomass relationships has only been documented on certain coral reefs isolated from human disturbance in the Northwestern Hawaiian Islands (Friedlander & DeMartini 2002) and

Northern Line Islands (DeMartini et al. 2008, Sandin et al. 2008). In these studies, the biomass of top-level (apex) predators such as sharks, carangids and lutjanids relative to the total standing fish biomass was 54 and 85%, respectively. Excluding sharks, the predators observed in these studies were the same as those included in the Fish-index. The high biomass of predators in South African Sanctuary zones is thus higher than that on the undisturbed coral reefs in the North Western Hawaiian Islands and comparable to that on coral reefs in the Northern Line Islands.

Biomass pyramids dominated by predators are rare in ecological studies (DeMartini et al. 2008). It has been suggested that this is an artefact of the generally degraded condition of most coral reefs worldwide, where reductions of top-level predators due to over-fishing have drastically altered fish assemblage dynamics (DeMartini et al. 2008). The paucity of inverted biomass pyramids in coral reef ecosystems highlights the uniqueness of ecosystems dominated by predators. South African Sanctuary zones appear to have the high predator biomass characteristic of inverted pyramids. However, in depth studies investigating population growth, death and turnover rates of major predator and prey species (DeMartini et al. 2008) are required to validate this suggestion.

Demartini et al (2008) proposed that high estimates of predator biomass on the reefs in the Northern Line Islands should be viewed as representative of undisturbed central Pacific reefs and may constitute baselines for other reef fish assemblages in the region. The results in Chapter 3 similarly suggest that the fish communities on South African Sanctuary reefs represent benchmark communities for South African and southern Mozambican coral reefs. The high biomass values presented in this study provide support for this suggestion and promote the use of Sanctuary zones as relatively undisturbed ecosystems on which future management plans should be based. Thus, it is suggested that these Sanctuary zones should continue to be no-take zones in which human activities are excluded. Considering their long history of closure to human activities and their higher abundance of sexually mature fish, the Sanctuary reefs may also be conserving spawning stocks and acting breeding refugia for non-Sanctuary zones (Francini-Filho & Moura 2008). The increased size of target species within Sanctuary zones is highly significant because they represent higher biomass and fecundity (Watson et al. 2009). Thus fish populations within Sanctuary zones have older, larger and more fecund individuals, capable of producing larvae with greater survival potential than their smaller counterparts (Berkeley et al. 2004).

Morris (2009) suggested that the predominantly south-flowing Agulhas Current could transport spawned coral propagules and larvae from reefs southwards from the northern reaches of the Maputaland coast. Similarly, fish larvae or fish eggs may be conveyed from the northern Sanctuary reefs to the High-Diving and Diving-Fishing reefs in the central complex. It is important to establish

an understanding of larval connectivity between MPA sites and adjacent regions (Mora & Sale 2002, Palumbi 2003). Further studies are needed to investigate the network relationships between the different reefs along the north-south gradient to determine whether the reef fish communities are self-seeding or acting as valuable breeding reservoirs. The Sanctuary reefs in the Northern Reef Complex are potentially the most important no-take coral reefs in South Africa because of their strategic position and it is suggested that they should continue to receive the highest level of protection.

CHAPTER 5

GENERAL DISCUSSION

The overall aim of this study was to assess the nature of southern African coral reef fish communities in the context of human activities. Three fundamental principles of coral reef research guided this study. The first of these was baseline community assessment. As stated by Green et al. (2006), conservation planning that targets ecosystem-based protection objectives cannot be achieved without the baseline data needed to monitor progress. Many baseline surveys may be limited in their application because they focus on species inventories alone. The present fish community investigation incorporated baseline information such as species diversity and species abundance with process-orientated data such as trophic functioning. In addition, assessments were made of the structure of the fish communities on reefs subjected to different human resource use. Such data can be integrated with previous studies (Chater et al. 1993, Chater et al. 1995) to form the basis for long-term monitoring programmes.

Results of the present fish community surveys were also compared with those on other coral reef fish communities in the Western Indian Ocean (WIO) bioregion. The results indicated that despite the marginal distribution of these coral reefs, the fish communities are, in fact, quite similar to their East African and WIO counter parts. In terms of species numbers (Table 3.13), the Maputaland fish communities were similar to and more specios than certain tropical WIO reefs. This trend mirrors that of the Maputaland coral communities which represent a biodiversity peak south of the equator (Benayahu & Schleyer 1995, 1998). In addition, the fish communities are comprised of predominantly Indo-Pacific species (80%), demonstrating considerable overlap in species composition with other tropical reefs in the WIO and on certain Indo-Pacific reefs (Kenya: McClanahan 1994, New Caledonia: Wantiez et al. 1997, north-western Hawaii Islands: Friedlander & Martini 2002, Tanzania: Garpe & Öhman 2003, Glorieuses Islands: Durville et al. 2003, Reunion: Letourneur et al. 2004, Juan De Nova: Chabanet & Durville 2005, Andavadoaka: Gillibrand et al. 2007, Bazaruto: Maggs et al. 2010).

This taxonomic 'sharing' of fish species extends to key functional species such as top and medium-level predators as well as large herbivorous species and implies that the trophic structure on the Maputaland reefs may be similar to that of many tropical coral reefs. Trophic relationships are among the major forces that structure biological communities (Polunin & Pinnegar 2002, Cury et al.

2003), and the nature of the fish communities on the Maputaland reefs thus suggests that their ecological functioning is more similar to that of coral reefs and less similar to that of other marginal coral communities (e.g. Lord Howe Island: Francis 1993, Harriott et al. 1995; Houtman Abrolhos: Hutchins 1997, Crossland et al. 1984; Gulf of Aqaba: Khalaf & Kochzius 2002). Despite not conforming to the true geological definition of tropical coral reefs and not being as accretive (Kleypas et al. 1999), one should not underestimate the functional importance of these diverse ecosystems.

Application of the Fish-index

The second guiding principle was the use of indicators as tools in ecosystem monitoring. The need for indicators to assess coral reef health has grown out of the notion that resource managers and stakeholders require relatively simple ways of assessing the impacts that humans have on natural resources (Alban et al. 2004). Although the indicator species were developed for southern African coral reefs, it is not the species but the concepts that they represent which are central to the effectiveness of the Fish-index. The Fish-index was developed to assess the effects of human activities and the major stresses affecting coral reefs are comparable among nations (Risk 1999). The Fish-index species were selected as indicators that may act as links between biological change and causative agents. While the dominant species may change from one region to the next, coral reef fish families are universal and manifest similar characteristics in their responses to human activities (Clua et al. 2005).

To elucidate this further, consider the overexploitation of fish species, which is one of the most severe threats posed to coral reef ecosystems worldwide. Certain reef-associated species such as the lutjanid, *Aprion virescens*, were included in the Fish-index to assess the effects of recreational fishing on the study reefs in question. Using abundance and biomass data, this species manifested trends that suggested high levels of fishing are occurring on certain South African and southern Mozambican reefs. Although *A. virescens* may not occur on reefs in other oceanic provinces, its incorporation in the Fish-index may be substituted with species targeted at a similar trophic level on other coral reefs. Similarly, the species selected to represent the functional groups in this study may be substituted with other, similarly important species. Those species which are prominent trophic contributors need only be identified and incorporated into a site-specific Fish-index. This should be done through fish community surveys and highlights the importance of conducting baseline studies. These principles of developing a Fish-index relate to processes that structure coral reefs fish communities such as predation, competition and reproduction. Species that signal changes in the

functioning of these processes will provide an insight into the nature of and changes in the coral reef fish communities.

The concepts employed in the development of the Fish-index are not novel as there is a large literature documenting and promoting the use of ecological (diversity, trophic levels) and biological (biomass, abundance and length-frequency) indicator metrics to assess coral reef fish communities (see Pelletier et al. 2005 for review). In addition, there are a number of monitoring manuals that have been developed for non-scientists (Reef Check, AGRRRA 2005, CRAMP, McClanahan 2008, Wilkinson et al. 2003). There are also other metrics that may be used to investigate human impacts on reef fish communities, particularly fishing. Size metrics are a useful ecosystem metric of exploitation effects (Dulvy et al. 2004). Thus size-frequency distributions for each of the six coral indicator species were calculated to investigate trends in body size in each protection zone. Alternatively, size-spectra analysis may be employed as it has been shown to be a useful indicator of the effects of fishing on reef fish assemblages (Graham et al. 2005; Friedlander et al. 2010). The inclusion of this type of analysis could have provided a direct measure of fishing effects on the fish communities in this study. However, such analyses require quantitative data on fishing intensities for each reef in question to provide upper and lower limits of extractive resource use. Such data are not always available, particularly in developing countries where the necessary infrastructure is missing or in countries like South Africa where monitoring of coral reef fishing is not yet this detailed. Nevertheless, it is recommended that future studies make every effort to obtain such information.

The United Nations Commission on Sustainable Development (CSD) (1993 and 1994) emphasized the need for a 'Menu of Indicators' to provide the basis for early warning systems, a cost-effective means of data collection, monitoring and assessment of trends, and informed decision-making, particularly for natural resource systems (Garcia & Staples 2000). While every attempt has been made to ensure the results of this study were statistically and scientifically grounded, the fact that only one Open reef could be included is recognized as a limitation that should be addressed in future studies with the inclusion of more Open sites. Nevertheless, the Fish-index has been developed for scientists or reef managers to rapidly assess differences in fish communities and the results of this study suggest it to be a useful monitoring and management tool.

The effectiveness of protected area management on South African coral reefs

The third key principle was MPA assessment in the context of coral reef conservation. Fish communities on South African coral reefs have not been exposed to the same stresses that plague coral reefs in the Western Indian Ocean and Indo-Pacific (Wilkinson 2008). This is most likely attributable to the protection afforded by the St Lucia and Maputaland MPAs for more than 20 years and their relative inaccessibility prior to this. However, the results of Chapter 3 revealed that the fish communities differed on reefs of varying protection status. Furthermore, Chapter 4 focused the study by using 25 fish indicator species to elucidate trends evident in the community data. While the results were not unexpected, the trends manifested by the indicator species suggest that recreational fishing and SCUBA diving intensity are having an effect on the fish community structure. The question may thus be asked: Is coral reef protection within our MPAs effective? Chapter 4 dealt with this issue and deduced that reefs within South African MPAs had higher fish abundance, biomass and species diversity than the non-MPA reef in southern Mozambique. However, further significant differences became apparent when the fish communities between the South African study reefs were compared. As mentioned in Chapter 1 and 4, many MPAs are not achieving their management objectives with regard to the reef-associated fish communities. The following discussion deals with this issue in the context of South African coral reef MPA management.

At present the theoretical management framework surrounding South African coral reef MPAs is in an indeterminate state due to an over-emphasis of legislative Acts and a lack of an adaptive management plan for the different MPA zones. Over the past decade, the changes brought about by the proclamation of the iSimangaliso Wetland Park as a World Heritage Site has not resulted in concurrent changes in the relevant management plans. The most recent management plan available for the St Lucia and Maputaland MPAs is for the Greater St Lucia Wetland Park (GSWLP, its earlier name) dated 2003. Another significant complication is the dual legislative protection afforded to the reefs by the Marine Living Resources Act (MLRA) and World Heritage Convention Act (WHCA) (see Attwood et al. 1997 for details). This has introduced considerable confusion with respect to the number of zones in the park, their boundaries and the activities permitted in them (Lemm & Attwood 2003). Under the MLRA, there are two types of zones in the MPAs; sanctuary and restricted (multiple resource use) zones. According to the WHCA, there are three types of zones; Sanctuary (no-take), Restricted (Diving-Fishing) and Controlled (High-Diving). To ensure consistency between these legislations, the conservation enforcement agency, Ezemvelo KwaZulu-Natal Wildlife, adopted the WHCA zonation under the MLRA regulations in their GSLWP 2003

management plan (Fig. 5.1). The objectives presented in the 2003 GSLWP zonation management plan will be used as in the ensuing discussion to assess the effectiveness of coral reef protection and, hence, MPA management

According to the GSLWP 2003 management plan, the goals for the MPAs are classified in three functional categories: protection, fisheries management and utilization, these differing according to habitat type, species under protection and human activity. All objectives relevant to coral reefs and coral reef fish communities were extracted and assessed according to the reef zonation (Table 5.1). Several trends are evident in Table 5.1. The no-take, no-entry policy applied to the Sanctuary zones has ensured that all the objectives and goals for the St Lucia and Maputaland MPAs have been successfully achieved. The prohibition of any type of human activity in these areas has provided benchmark areas for scientific research, protected vulnerable life-histories of reef fish (targeted and non-targeted), maintained spawner biomass, and provided potential breeding reservoirs of fish species, supplying recruits to adjacent areas where harvesting is permitted. This highlights the importance of these areas in maintaining natural processes in reef fish communities. The Controlled and Restricted zones appeared to achieve only five of the thirteen management objectives (Table 5.1). These two zones are effective in protecting the coral reef habitat for fish communities; however, the results of this study suggest that, in terms of fisheries management, the Controlled and Restricted zones are of questionable value. The low numbers of sexually mature targeted fish species indicate that high levels of resource use are occurring in the Restricted and Controlled zones. Despite the fact that only restricted fishing is permitted in these zones, there is growing evidence that the effects of recreational fishing are greater than previously thought. Recent research in New Zealand revealed that partial closure that still permitted recreational fishing was ineffective as a conservation tool and that angling effort in partially closed areas was equal to that in sites afforded no protection (Westera et al. 2003). In addition, the low densities of sexually mature, non-targeted predatory species in the Controlled zones suggest that high diving intensity may also be influencing the fish community structure to some degree. Thus, neither the Restricted nor Controlled zones appear to be providing sufficient protection for the reefs under the current levels of human consumptive and non-consumptive recreational activity.

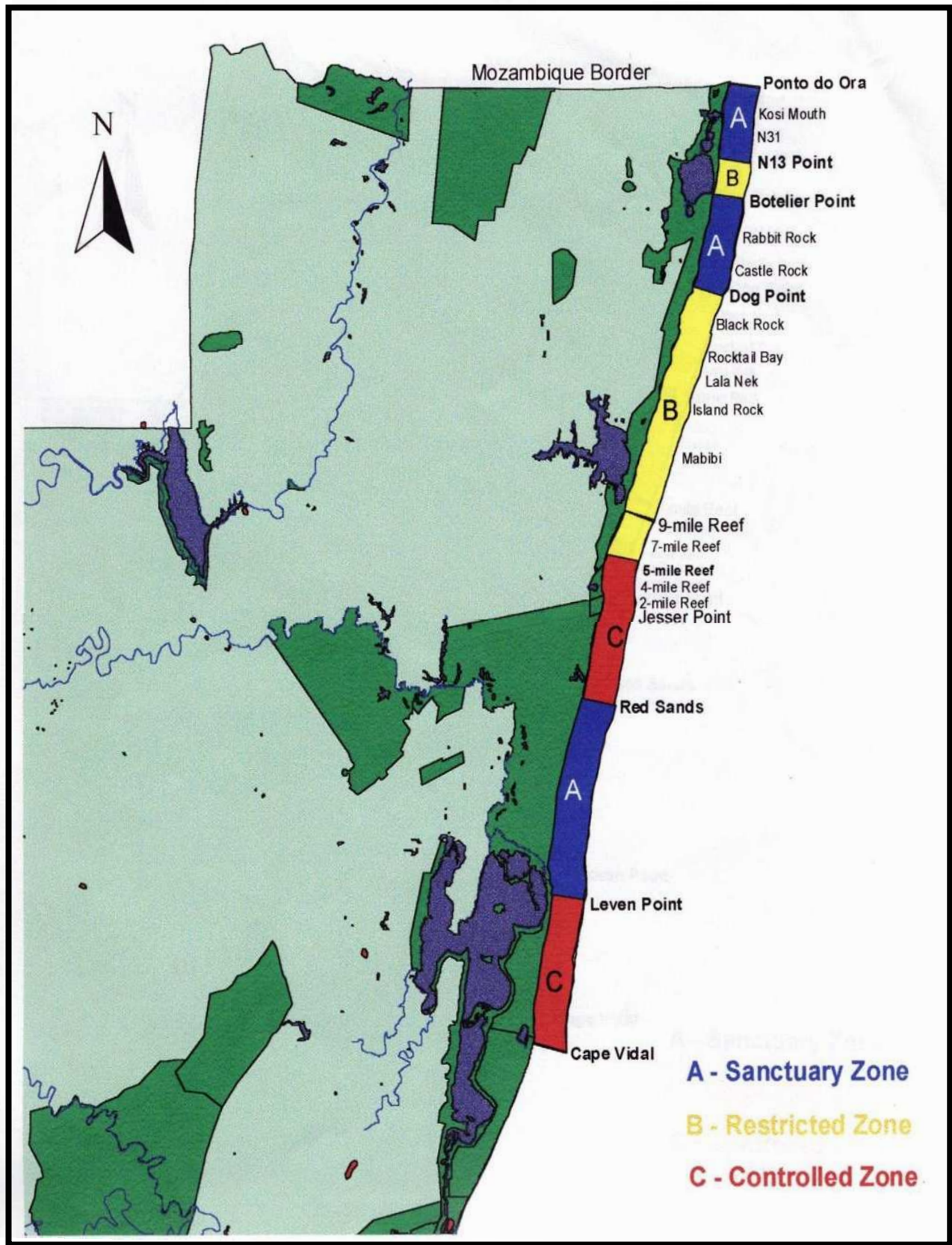


Figure 5.1 The offshore zonation plan for the St Lucia and Maputaland MPAs according to the Greater St Lucia Wetland Park management plan 2003.

Table 5.1 The conservation objectives relevant to coral reefs in the St Lucia and Maputaland MPA in the three different zonation areas. Zone A – Sanctuary, zone B – Restricted and zone C – Controlled. (Extracted from the KZMW Marine Zone Management Plan for the GSLWP 2003).

Objective		Function	Objective achieved		
			Zone A	Zone B	Zone C
1	Ensure protection of representative sections of the marine environment in the eastern Indo-Pacific biogeographic region	Protection	Yes	Yes	Yes
2	To maximize habitat diversity	Protection	Yes	Yes	Yes
3	To ensure protection of rare, localized or endemic species through protection of their habitats	Protection	Yes	Yes	Yes
4	To protect areas essential for the completion of vulnerable life stages	Protection	Yes	?	?
5	To prevent over exploitation by providing refuge areas for exploited sedentary species	Protection	Yes	Yes	Yes
6	To protect vulnerable life-history stages of mobile or migratory exploited species	Protection	Yes	?	?
7	To improve or sustain yields in adjacent areas	Fisheries management	Yes	?	?
8	To maintain spawner biomass	Fisheries management	Yes	?	?
9	To provide undisturbed localities, populations and communities for research	Fisheries management	Yes	?	?
10	To provide sites in which monitoring can be conducted	Fisheries management	Yes	?	?
11	To promote and facilitate the development of tourism in South Africa	Utilisation	N/A	Yes	Yes
12	To provide sites for low impact, non consumptive recreation	Utilisation	N/A	?	?
13	To allow exploitation of selected taxa at a sustainable level	Utilisation	N/A	?	?

Inclusion of a southern Mozambican reef provided a valuable comparison for the overall assessment of the effectiveness of reef conservation in the different MPA zones. The large difference in fish communities between the Sanctuary and Open zones was significant, but not unexpected. On the other hand, the lack of a clear separation between the fish communities in the High-Diving zone (Controlled zone), Diving-Fishing (Restricted zone) zones and the Open zone was not anticipated. Two possibilities may explain the unexpected similarities in fish communities between these zones. Firstly, factors other than human resource use may be influencing the fish communities in these zones; this seems unlikely in view of the similarity of the reefs. Secondly, and alternatively, the levels and effects of human resource use may be similar in the High-Diving, Diving-Fishing zones and the Open zone, despite the differences in protective legislation. Studies to investigate these

possibilities further would be beneficial for to assess the effectiveness of management of these MPAs.

The results of this study thus suggest that non-Sanctuary zones in the St Lucia and Maputaland MPAs may not be achieving all of their current management objectives. However, this does not imply that such MPA zones cannot be effective. Multiple resource use zones are important aspects of MPAs because they can offer benefits to a wide variety of stakeholders or constituents and provide a frame-work for resolving user-conflict (Agardy et al. 2003). In addition, they offer a basis for encouraging public awareness and promoting responsible attitudes and resource use. Although multiple resource use areas permit access to marine resources to a wider range of stakeholders, they represent a more complex management zone and thus require a multi-disciplinary management approach to balance the trade-off between sustainable resource use, conservation objectives and environmental change.

It appears that more rigorous measures may be needed to monitor and assess the current levels of human activities on the Maputaland reefs. The aspects of the MLRA recreational fishing regulations that warrant modification were discussed in detail in Chapter 4. In particular, it was recommended that biologically-based criteria for the selection of 'gamefish' be incorporated into the MLRA recreational fishing regulations so species vulnerable to overexploitation, such as *Aprion virescens*, may be excluded. These criteria are presented in the final management recommendations below. In addition, reduced daily bag limits and the implementation of minimum size limits were suggested to prevent the extraction of potential brood stock. The recommended minimum size limits were set higher (10 cm) than the mean length at 50% sexual maturity to ensure that at least half the individuals of a cohort caught have had a chance to spawn at least once (Caddy & Mahon 1995). This recommendation pertains to *Caranx melampygus* and the details are included in the management recommendations below.

Concomitant with the proposed adaptations to the regulations, a new management approach is also put forward for the Maputaland reef fish communities. As stated by Bellwood et al. (2004), the rapid decline of reef systems calls for more rigorous, innovative and adaptive management strategies. The application of adaptive management was advocated more than a decade ago (Agardy 1994) and numerous monitoring tools are available that support adaptive management (see IUCN-WCPA 2008 Table 16). Adaptive management differs from conventional strategies in that it has a strong emphasis on forward-looking components, with the aim that managers become proactive rather than reactive in their decision making (Biggs & Rogers 2003, Nyström 2006, IUCN-WCPA 2008).

The suggested approach for the Maputaland reef fish communities is based on the management strategy employed in South Africa's largest terrestrial reserve, the Kruger National Park (KNP), where goal-setting and monitoring end-points have been combined to discern thresholds of potential concern (TPC) (Biggs & Rogers 2003). A TPC is an operational goal towards which a component of an ecosystem is managed with upper or lower limits set along a continuum of change according to selected ecosystem indicators. TPCs are predicted levels of acceptable or unacceptable change in ecosystem structure, function and composition (Foxcroft 2009), and provide managers with an understanding of the possible implications of change, and the response-measures needed to sustain or mitigate these changes.

Suggested TPCs for the southern African coral reef fish communities are presented in Table 5.2 and integrate the results of Chapter 4 to determine the upper and lower limits of acceptable change. They thus aim to provide a link between science, monitoring and adaptive management at three levels for a number of indicator metrics. TPC-1 represents a benchmark reference point based on the fish communities in the Sanctuary zones because these fish communities were considered undisturbed. TPC-2 is a threshold limit that signals acceptable change has been exceeded and management intervention is needed to return to TPC-1. TPC-3 represents overexploitation of an indicator metric and signals that immediate management intervention is needed.

Predatory and targeted species were most affected by human activities in the different zones and were thus considered appropriate indicators of fish community change. Herbivores have been included as they represent a key functional group that is a critical source of resilience and vulnerability to ecosystem phase shifts from coral- to algal-dominated states (Bellwood et al. 2004). Although herbivores such as scarids are protected on South African reefs, adaptive management requires an ecosystem approach that goes beyond the traditional concepts of monitoring targeted species alone for sustainable fisheries management (Bellwood et al. 2004). Thus indicators have been included that may promote the early prediction of possible threats, allowing proactive rather than reactive management. Continued monitoring of the fish communities will allow modification of indicators if deemed necessary.

The trophic level (TL) metric has been included because the trophic structure in coral reef fish assemblages may provide an insight into the functioning of the entire reef ecosystem (Bozec et al. 2005). The TL score was generated by summing each of the Fish-Index species TL (Appendix 6.1.2), providing useful comparisons of zones of different human resource use. The sum of TLs per reef was used rather than the average TL per reef (as per Pauly et al. 1998) because the low

abundance of fish in the Open zone produced spurious results. The TL score is useful for relative comparisons between reefs and not between regions.

Table 5.2 Thresholds of potential concern (TPC) for southern African coral reefs fish communities within the zones of present human resource use. The total trophic level category constitutes the sum of each Fish-index species trophic level.

Zone	Reef	Indicator metric				Implication			Management action
		Number of top predators	Number of herbivores	Number of target & predatory species	Total Trophic level	Reproductive potential			
						Top predators	Herbivores	Target species	
Sanctuary	Rabbit Rock Red Sands Reef Leadsman Shoal	abundant	abundant	abundant	60	normal	normal	normal	Continued monitoring to assess long-term changes
High-Diving	Two-Mile Reef	rare	abundant	low	42	impaired	normal	reduced	Reduce diving intensity
Diving-Fishing	Seven-Mile Reef Nine-Mile Reef	present but wary	abundant	very low	50	reduced	normal	impaired	Reassess fishing restrictions
Unrestricted fishing & diving	Shallow Malongane	absent	low	rare	32	absent	reduced	inhibited	Implement relevant fishing restrictions and active law enforcement

TPC-1	TPC-2	TPC-3
Benchmark	Threshold	Overexploited

Management guidelines and recommendations

The results of this study highlight the following key points:

1. Total fish abundance and biomass was highest in Sanctuary zones.
2. Top-level predators were more abundant and larger in Sanctuary zones.
3. The above points suggest that Sanctuary zones are achieving their management objectives.
4. The Open reef had the lowest total fish abundance and biomass.
5. Similarities between the Open zone, Controlled and Restricted zones suggest that certain MPA zones may not be fully achieving their management objectives.
6. A low abundance of sexually mature target species may indicate that fishing is influencing fish communities on the Diving-Fishing reefs.
7. The low abundance of large no-take predators on the High-Diving Reef requires further investigation.
8. The designation of certain species and families as gamefish by the MLRA requires re-examination.

With regards to the above points, the following management recommendations are proposed:

1. Sanctuary areas should remain no-take and no-entry areas where all forms of human activity are excluded apart from essential monitoring and research
2. Further research is required to assess whether fish communities on South African coral reefs are self-seeding or inter-connected. Such studies should include larval dispersal, genetic and tagging studies.
3. The MPA management plan should include measures to investigate and address the low numbers of sexually mature resident target species such as *Caranx melampygus* and *Aprion virescens*. Such measures may include:
 - A - The implementation of a minimum size limit and a daily bag limit for *Caranx melampygus*. The minimum recommended forklength would be 50 cm, which is 10 cm larger than the mean length at 50% sexual maturity. The recommended bag limit for *C. melampygus* is three.
 - B - The declaration of a moratorium on the capture of *Aprion virescens*.
4. Reevaluation of zonation in the Central Reef Complex and implementation of diving carrying capacities for each reef.
5. The development of biologically-based criteria for the designation of fish species as gamefish in the MLRA, e.g.:

Criteria	Include	Exclude
Movement	Species that are nomadic or migratory	Species that aggregate for spawning or mating, or that show site fidelity including residency and/or territoriality
Stock status	Species that are under-exploited or optimally exploited (see Griffiths et al. 1999 for biological reference points)	Species that are over-exploited or where the stock has collapsed (see Griffiths et al. 1999 for biological reference points)
Fertilisation and fecundity	Species that are external fertilisers and broadcast spawners	Species that are viviparous or ovoviviparous with low fecundity (i.e. sharkds)
Growth and maturation	Species that are early maturing and fast growing (low vulnerability index)	Species that are slow growing and slow maturing (high vulnerable index)

Conclusion

This study has demonstrated that the southern African coral reef fish communities have several unique traits. They are associated with high-latitude coral communities that occur at the limit of coral reef distribution, yet they are diverse and appear to have strong functional similarities with coral reef fish communities on other WIO and Indo-Pacific reefs. In addition, the Sanctuary zones appear to maintain relatively undisturbed fish communities, which is uncommon in the current global environment of human resource use and climate change. The results contribute valuable information to the growing body of literature demonstrating the importance of no-take zones as conservation and management tools for coral reef fish communities. However, the results also demonstrated that multiple resource use zones may not be providing the same levels of protection to the reef fish communities as Sanctuary zones, despite the fact that they share the same MPA management objectives. Such zones have the potential to be effective in coral reef conservation. Yet, it is critical that managers set aside the notion of a ‘one size fits all’ approach and that they align their management objectives to suit the different types of MPA zones and their respective resource use.

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Appendix 1

Mean abundance of all fish taxa recorded on the seven study reefs, expressed as fish/hr. Species have been ordered alphabetically. Distribution (D) refers to the species range. IP = Indo-Pacific, WIO = Western Indian Ocean, I = Indian Ocean, C = cosmopolitan, E = endemic.

Species	D	LMS	RS	TMR	SMR	NMR	RR	SM
Acanthuridae								
<i>Acanthurus blochii</i> Cuvier and Valenciennes, 1835	IP	0	3.58	0	1.2	1.17	1.17	0.2
<i>Acanthurus dussumierii</i> Valenciennes, 1835	IP	2.33	0.92	0.3	0.4	1.58	3.5	0.6
<i>Acanthurus leucocheilus</i> Herre 1927	IP	0.22	0	0	0.4	0	0	0
<i>Acanthurus leucosternon</i> Bennet, 1833	IP	6.78	5.25	5.5	8.6	1.92	2.67	1.4
<i>Acanthurus lineatus</i> (Linnaeus, 1758)	IP	0.11	0.08	0.2	0	0	0.17	0
<i>Acanthurus mata</i> (Cuvier, 1829)	IP	2.22	1.42	1.1	0.4	0.17	11.08	4.3
<i>Acanthurus nigrofuscus</i> (Forsskål, 1775)	IP	61.44	25.58	15.8	21.1	17.42	17.5	11.5
<i>Acanthurus tennentii</i> Günther, 1861	I	19.22	11.42	7.3	2.4	3.33	5.75	3.7
<i>Acanthurus thompsoni</i> Fowler, 1923	IP	6.78	8.58	4.4	3	1.42	4.42	1
<i>Acanthurus triostegus</i> (Linnaeus, 1758)	IP	0	0	0.3	0	0	0.5	0
<i>Acanthurus xanthopterus</i> Cuvier & Valenciennes, 1835	IP	1.44	0	1.5	1.1	0.42	3.67	0
<i>Ctenochaetus binotatus</i> Randall, 1955	IP	0.78	0.75	2.1	1.2	1.25	0.25	1.5
<i>Ctenochaetus striatus</i> (Quoy & Gaimard, 125)	IP	0.89	0.25	0.6	0.2	0.25	0.08	0.1
<i>Ctenochaetus truncatus</i> Randall & Clements, 2001	IP	23.44	7.42	6.3	2.5	0.92	2	3.8
<i>Naso brachycentron</i> (Cuvier & Valenciennes, 1835)	IP	0.89	0	0.2	0.1	0	0.33	0.2
<i>Naso brevirostris</i> (Cuvier, 1829)	IP	1.33	0	0	0	0	0	5.5
<i>Naso hexacanthus</i> (Bleeker, 1855)	IP	17.67	1.27	0.6	2.1	0.08	0.75	9.5
<i>Naso lituratus</i> (Forster in Bloch & Schneider, 1801)	IP	2.67	2.5	2.3	1.7	1.25	2	1.1
<i>Naso unicornis</i> (Forsskål, 1775)	IP	1.22	0.75	0.7	0.2	0.67	6.42	0.2
<i>Paracanthurus hepatus</i> (Linnaeus, 1766)	IP	6	0	0	0	0	0	0.3
<i>Zebrasoma gemmatum</i> (Valenciennes, 1835)	I	0.44	0.17	0	0.2	0.17	0.5	0
<i>Zebrasoma scopes</i> (Cuvier, 1829)	IP	0.78	2.17	1.8	1.1	0.08	2.58	1.1
<i>Zebrasoma desjardini</i> (Bennett, 1836)	WIO	0.11	0	0.1	0.1	0	0.33	0
Apogonidae								
<i>Apogon angustatus</i> (Smith & Radcliffe in Radcliffe, 1911)	IP	0	0	0	0	0.08	0	0
<i>Apogon taeniophorus</i> Regan, 1908	IP	0	0	0	0	0.08	0	0.1
<i>Apogon apogonoides</i> (Bleeker, 1856)	IP	0	0	3.1	0	0	1.58	0
<i>Cheilodipterus artus</i> Smith, 1961	IP	0	0	0	0	0.33	0.08	0
Aulostomidae								
<i>Aulostomus chinensis</i> Linnaeus, 1766	IP	0.11	0	0.5	1.6	0.58	0	0.4
Balistidae								
<i>Balistapus undulates</i> (Mungo Park, 1797)	IP	1.44	1.33	1.2	0.7	0.83	1.58	1.6
<i>Balistoides conspicillum</i> (Bloch & Schneider, 1801)	IP	0.89	0.67	0.2	0.4	0.17	1	0.4
<i>Balistoides viridescens</i> (Bloch & Schneider, 1801)	IP	0.22	0.09	0.4	0.1	0	0.08	0.2
<i>Melichthys indicus</i> Randall & Klausewitz, 1973	I	0.56	1.08	1.1	2.2	0.83	1.25	1.9
<i>Odonus niger</i> (Rüppel, 1836)	IP	24.44	24.5	5.5	47.1	13.42	38.17	14
<i>Pseudobalistes flavimarginatus</i> (Rüppel, 1829)	IP	0.11	0	0	0	0	0	0
<i>Pseudobalistes fuscus</i> (Bloch & Schneider, 1801)	IP	0.11	0	0	0	0	0	0
<i>Sufflamen bursa</i> (Bloch & Schneider, 1801)	IP	0.22	0.17	1.2	1.2	1.08	0.92	0.4
<i>Sufflamen chrysopterus</i> (Bloch & Schneider, 1801)	IP	1.78	2.08	1.2	2.5	1.75	2.08	3.1
<i>Sufflamen fraenatus</i> (Latreille, 1804)	IP	1.11	0.67	0.9	0.5	0.67	0.83	1.1
Blennidae								
<i>Ecsenius midas</i> Stark, 1969	IP	0	0	0.1	0	0.08	0	0
<i>Plagiotremus rhinorhynchus</i> (Bleeker, 1852)	IP	0.11	0.09	0.3	0.3	0.33	0.58	0

Species	D	LMS	RS	TMR	SMR	NMR	RR	SM
<i>Plagiotremus tapeinosoma</i> (Bleeker, 1852)	IP	0.11	1.17	0.3	0.9	2.33	2.5	2.4
<i>Exallias brevis</i> (Kner, 1868)	IP	0.56	0	0.1	0	0	0.08	0.1
Caesionidae								
<i>Caesio caeruleaurea</i> (Lacepède, 1801)	IP	0	45.45	13	3	9.75	27.5	0
<i>Caesio lunaris</i> Cuvier, 1830	IP	6.44	0	0	0	1.25	0.08	0
<i>Caesio xanthonota</i> Bleeker, 1853	IP	4.44	18.58	8.8	15.5	19.33	71.75	3.3
<i>Pterocaesio tile</i> (Cuvier, 1830)	IP	0	18.27	3.5	14.5	4.58	0	0
Caracanthidae								
<i>Caracanthus madagascariensis</i> (Guichenot, 1869)	WIO	0	0	0.1	0.3	0	0	0
Carangidae								
<i>Carangoides coeruleopinnatus</i> (Rüppell, 1830)	IP	0	0	0.1	0	0	0	2.5
<i>Carangoides fulvoguttatus</i> (Forsskål, 1775)	IP	2.56	0.17	0.5	2.5	4.92	16.67	0.5
<i>Caranx ignobilis</i> (Forsskål, 1775)	IP	0.33	0	0	0	0	0	0
<i>Caranx melampygus</i> Cuvier & Valenciennes, 1833	IP	3.56	2.75	2.9	4.2	1.25	7.58	0.6
<i>Caranx heberi</i> (Bennet, 1830)	IP	3.56	0	0	1	0	0.33	1.8
<i>Caranx sexfasciatus</i> Quoy & Gaimard, 1825	IP	0	0	0	3.3	0	0	0
<i>Decapterus macrosoma</i> Bleeker, 1851	IP	0	2.5	50	0	8.33	0	0
<i>Elagatis bipinnulata</i> (Quoy & Gaimard, 1825)	IP	0	1.67	0	0	0	0	0
<i>Pseudocaranx dentex</i> (Bloch & Schneider, 1801)	IP/C	0	0	0	0	0.08	0	0
<i>Scomberoides lysan</i> (Forsskål, 1775)	IP	0.11	0	0	0	0	0	30
<i>Seriola rivoliana</i> Valenciennes in Cuvier & Valenciennes, 1833	IP	0	0	0.3	0.5	0	0	0
<i>Seriolina nigrofasciata</i> (Rüppell, 1829)	IP	0	0	0.1	0	0	0	0.3
Carcharhinidae								
<i>Carcharhinus amblyrhynchos</i>	IP	0.11	0.18	0	0.1	0.08	0.42	0
<i>Triaenodon obesus</i> (Rüppell, 1837)	IP	0	0.09	0	0	0	0.08	0
Chaetodontidae								
<i>Chaetodon auriga</i> (Forsskål, 1775)	IP	2.11	1.75	0.7	1.5	1	1.25	0.9
<i>Chaetodon blackburnii</i> Desjardins, 1836	WIO	1.67	0.42	0.8	2.5	1.58	0.5	0.3
<i>Chaetodon guttatissimus</i> (Bennet, 1832)	I	5	3.92	3.1	3.8	3.08	3	2.2
<i>Chaetodon interruptus</i> Ahl, 1923	IP	5.89	6.92	5.4	3.8	3.83	8.17	7.3
<i>Chaetodon kleinii</i> Bloch, 1790	IP	5.78	5	3.7	4.2	3.25	2.58	2.2
<i>Chaetodon lineolatus</i> Cuvier in Cuvier & Valenciennes, 1831	IP	0	0	0	0	0	0.17	0
<i>Chaetodon lunula</i> (Lacepède, 1802)	IP	1.44	0.92	3.4	0.7	0.75	1.17	0.4
<i>Chaetodon madagaskariensis</i> (Ahl, 1923)	I	4.22	3.17	3.5	3.8	1.92	4.25	3.1
<i>Chaetodon meyeri</i> (Bloch & Schneider, 1801)	IP	4.22	3.5	1.6	1.6	1.58	4.5	3
<i>Chaetodon trifascialis</i> (Quoy & Gaimard, 1825)	IP	0.44	0.67	0.6	0	0	0.33	0.8
<i>Chaetodon trifasciatus</i> (Mungo Park, 1797)	IP	0.22	0	0	0	0	0.33	0.5
<i>Chaetodon vagabundus</i> Linnaeus, 1758	IP	1.22	0.42	0.5	0	0.33	0.5	0.7
<i>Chaetodon xanthocephalus</i> Bennet, 1832	I	0.33	0.25	0.4	0	0	0	0
<i>Chaetodon zanzibarensis</i> Playfair in Playfair & Günther, 1867	WIO	0	0	0	0	0	0.17	0
<i>Forcipiger flavissimus</i> Jordan & McGregor, 1898	IP	2.78	2.83	1.5	2.1	2.75	2.67	1.5
<i>Hemitaurichthys zoster</i> (Bennet, 1831)	I	6.67	7.17	0.3	4.4	2.33	4.25	1.1
<i>Heniochus acuminatus</i> (Linnaeus, 1758)	IP	0.44	0	0.2	0.1	0.42	0.25	0
<i>Heniochus diphreutes</i> Jordan, 1903	IP	0	0	0	0	3.67	0	0
<i>Heniochus monoceros</i> (Cuvier in Cuvier & Valenciennes, 1831)	IP	0	0	0	0.4	0	0.17	0
Cheilodactylidae								
<i>Chirodactylus jessicalenorum</i> Smith, 1980	E	0.56	0	0.1	0.5	0.17	0	0
Cirrhitidae								
<i>Amblycirrhitus bimacula</i> (Jenkins, 1903)	IP	0.11	0	0	0	0	0.08	0.3

Species	D	LMS	RS	TMR	SMR	NMR	RR	SM
<i>Cirrhitichthys oxycephalus</i> (Bleeker, 1855)	IP	2.33	3.25	2.9	3.4	3.58	4.17	2.4
<i>Paracirrhites arcatus</i> Cuvier in Cuvier & Valenciennes, 1829	IP	0.78	1.58	2.2	3.3	1.92	0.42	0.5
<i>Paracirrhites forsteri</i> (Bloch & Schneider, 1801)	IP	1.44	1.33	1.4	0.9	1.25	1.08	0.5
Dasyatidae								
<i>Dasyatis kuhlii</i> (Müller & Henle, 1841)	IP	0	0	0	0.1	0.08	0	0.1
<i>Himantura gerrardi</i> (Gray, 1851)	IP	0	0.09	0.1	0	0	0	0.2
<i>Taeniura lymma</i> (Forsskål, 1775)	IP	0	0	0	0.1	0	0	0
Dinopercaidae								
<i>Dinoperca petersi</i> (Day, 1875)	WIO	0	0	0.1	0	0.25	0.42	0
Diodontidae								
<i>Diodon hystrix</i> Linnaeus, 1758	C	0	0.17	0.2	0	0.17	0.08	0.2
<i>Diodon holocanthus</i> Linnaeus, 1758	IP	0	0	0.1	0	0	0	0
<i>Diodon liturosus</i> Shaw, 1804	IP	0.11	0	0.1	0.1	0	0	0.3
Echeneidae								
<i>Echeneis naucrates</i> Linnaeus, 1758	C	0	0.17	0	0.1	0.08	0	0
Ephippidae								
<i>Tripteron orbis</i> Playfair, 1876	WIO	0	0	0.4	1.8	2.67	0	0
Fistulariidae								
<i>Fistularia commersonii</i> Rüppell, 1838	IP	0	0	0.6	1.1	0	0	0.1
Gobiidae								
<i>Nemateleotris magnifica</i> Fowler, 1938	IP	1	0.17	0.5	1.4	2.25	0.67	0.9
<i>Ptereleotris evides</i> (Jordan & Hubbs, 1925)	IP	0.78	2.33	0	0.7	1.75	0	8
<i>Ptereleotris heteroptera</i> (Bleeker, 1855)	IP	1	0.5	0	0	0.08	0.25	0.5
<i>Valenciennea strigata</i> (Jordan & Hubbs, 1925)	IP	0	0.75	0.6	0.8	1.67	1.08	1.6
Haemulidae								
<i>Diagramma pictum</i> (Thunberg, 1792)	IP	0	0	0	0	1.25	0	0
<i>Plectorhinchus chubby</i> (Thunber, 1792)	WIO	0	0.25	0.8	0	0.33	0.25	0
<i>Plectorhinchus flavomaculatus</i> (Cuvier, 1830)	IP	1	1.33	0.6	0.8	0.25	2.08	0.1
<i>Plectorhinchus playfairi</i> (Pellegrin, 1914)	WIO	0.44	0.17	0.8	0.5	1.08	1.58	0.2
<i>Plectorhinchus schotaf</i> (Forsskål, 1775)	IP	0.22	0	0	0	0.08	0	0
Holocentridae								
<i>Myripristis murdjan</i> Frosskål, 1775	IP	0.78	1.55	4.8	8	2.58	2.5	1.2
<i>Sargocentron caudimaculatum</i> Rüppell, 1838	IP	0.89	1.42	0.6	0.6	1.67	1.58	0.1
<i>Sargocentron diadema</i> Lacepède, 1802	IP	0.89	0.42	1.3	0.6	0.5	2	0.7
<i>Sargocentron spiniferum</i> Frosskål, 1775	IP	0	0	0.1	0	0	0.25	0
Kyphosidae								
<i>Kyphosus cinerascens</i> Forsskål, 1775	IP	0.56	0.17	0	0	0	0	0
<i>Kyphosus vaigiensis</i> (Quoy & Gaimard, 1825)	IP	3.44	1	8	0	1.25	1.83	0
Labridae								
<i>Anampses caeruleopunctatus</i> Rüppell, 1829	IP	2.22	2.58	1.6	2	4	1.58	2
<i>Anampses lineatus</i> Randall, 1972	I	0.11	0	0.4	0.1	0.08	0	0
<i>Anampses meleagrides</i> (Valenciennes, 1840)	IP	1.33	0.83	1.3	0.9	0.92	1.33	0.9
<i>Anampses twistii</i> Bleeker, 1856	IP	0.22	0	0.4	0	0	0.5	0.2
<i>Bodianus anthoides</i> Bennet, 1832	IP	0	0	0	0.3	0.08	0	0.2
<i>Bodianus axillaries</i> Bennet, 1832	IP	0.89	0.5	0.3	0.2	0.33	1.17	1.1
<i>Bodianus bilunulatus</i> (Lacepède, 1801)	IP	2.56	3.67	0.9	0.7	1.33	1.92	0.4
<i>Bodianus diana</i> Lacepède, 1802	IP	2.33	2	2.1	3.1	3.92	2	0.9
<i>Bodianus perditio</i> (Quoy & Gaimard, 1834)	IP	0.22	0.67	0	0	0.08	0.5	0
<i>Cheilinus fasciatus</i> (Bloch, 1791)	IP	0.22	0.42	0	0	0.08	0	0
<i>Cheilinus trilobatus</i> Lacepède, 1801	IP	0.56	0	0	0	0	1	0
<i>Cheilio inermis</i> Frosskål, 1775	IP	0	0	0	0	0	0.17	0.1

Species	D	LMS	RS	TMR	SMR	NMR	RR	SM
<i>Cirrhilabrus exquisitus</i> Smith, 1957	IP	1.78	1.08	0.1	4.4	3.25	1.83	9.9
<i>Coris aygula</i> (Lacepède, 1801)	IP	0.56	0.17	0.4	0	0.25	0.42	0.1
<i>Coris caudimacula</i> (Quoy & Gaimard, 1834)	I	1.56	5.42	1	0.9	1.75	1.42	2.5
<i>Coris cuvieri</i> (Bennett, 1831)	I	0.56	0.5	0.3	0	0.5	0.33	0.3
<i>Coris formosa</i> (Bennett, 1830)	I	0.11	0	0.1	0.2	0.17	0.25	0
<i>Gomphosus caeruleus</i> Lacepède, 1801	I	2.22	2.83	2.5	0.8	1.83	2.25	1.8
<i>Halichoeres cosmetus</i> Randall & Smith, 1982	I	2.33	0.92	2.5	3.1	3.58	1.33	0.9
<i>Halichoeres hortulanus</i> Lacepède, 1801	IP	3.22	3.17	2.4	0.4	2.75	2.67	2.4
<i>Halichoeres iridis</i> Randall & Smith, 1982	WIO	0	0	0	0.1	0.08	0	0
<i>Halichoeres lapillus</i> Smith, 1947	WIO	0.44	0.25	0	0.1	0.17	0	0
<i>Halichoeres nebulosus</i> (Valenciennes in Cuvier & Valenciennes, 1839)	IP	1.22	0.75	0.3	0.5	0.33	0.42	0.6
<i>Halichoeres scapularis</i> Bennet, 1832	IP	0	0	0	0	0	0.08	0
<i>Hemigymnus fasciatus</i> Bloch, 1792	IP	0.56	1.18	0.2	0.3	0.92	0.67	0.6
<i>Hologymnosus annulatus</i> Lacepède, 1801	IP	0	0.18	0.1	0.1	0.17	1.42	0
<i>Hologymnosus doliatus</i> Lacepède, 1802	IP	0	0.18	0	0	0	0.17	0.5
<i>Labroides bicolor</i> Fowler & Bean, 1928	IP	0.22	0.27	0.8	0.4	0.67	1.5	0.4
<i>Labroides dimidiatus</i> Valenciennes, 1839	IP	8.33	6.5	8.5	7.6	6.42	8	7.3
<i>Labropsis xanthonota</i> Randall, 1981	IP	0.33	0.08	0	0.1	0.5	0	0.1
<i>Macropharyngodon bipartitus</i> Smith, 1957	WIO	0	0.58	0.5	0.4	1.17	0.58	1
<i>Macropharyngodon cyanoguttatus</i> Randall, 1978	I	0	0.09	0.2	0.2	0.33	0.08	0.4
<i>Novaculichthys taeniourus</i> (Lacepède, 1801)	IP	0.11	0	0.1	0	0	0.42	0
<i>Pseudocheilinus evanidus</i> Jordan & Evermann, 1903	IP	0	0	0	0	0	0	0.1
<i>Pseudocheilinus hexataenia</i> (Bleeker 1857)	IP	1.78	1.17	0.8	0.7	1.08	1.25	2.2
<i>Pseudodax moluccanus</i> (Valenciennes, 1840)	IP	0.67	0.67	0.6	0.3	0.17	1.25	0
<i>Pseudojuloides cerasinus</i> (Snyder, 1904)	IP	0.56	0	0	0	0	0	0
<i>Stethojulis albobittata</i> (Bonnaterre, 1788)	I	0	0	0	0	0.08	0	0
<i>Stethojulis interrupta</i> (Bleeker, 1851)	IP	0	0	0.1	0.1	0.17	0.08	0.1
<i>Thalassoma amblycephalum</i> Bleeker, 1856	IP	6.78	16.42	10.2	17.5	17.25	6.08	0.8
<i>Thalassoma genivittatum</i> (Valenciennes in Cuvier & Valenciennes, 1839)	I	0.56	0.33	0.2	0.4	0.42	0.58	0.1
<i>Thalassoma hardwicke</i> Bennet, 1830	IP	0.22	0	0	0	0	0	0
<i>Thalassoma hebraicum</i> Lacepède, 1801	I	24.22	30.67	9	6.5	8.92	13.83	6.1
<i>Thalassoma lunare</i> Linnaeus, 1758	IP	1.78	1.08	2.9	0.6	1	0.33	1.3
Lethrinidae								
<i>Gnathodentex aureolineatus</i> (Lacepède, 1802)	IP	0	0	0	0	0	0.17	0
<i>Gymnocranius griseus</i> (Temminck & Schlegel, 1843)	IP	0.33	0.25	0.4	0	0.08	0	0.1
<i>Lethrinus crocineus</i> Smith, 1959	WIO	7.33	1.92	0.3	0	0.08	0.75	0
<i>Lethrinus microdon</i> Valenciennes in Cuvier & Valenciennes, 1830)	IP	0	0	0.1	0	0	0.08	0
<i>Lethrinus nebulosus</i> (Forsskål, 1775)	IP	0	0	0.3	0	0	0.58	0
<i>Lethrinus rubrioperculatus</i> Sato, 1978	IP	3.33	3.17	0	0	0	0.33	0
<i>Lethrinus mahsena</i> (Forsskål, 1775)	WIO	0	0	0	0	0	0.08	0
<i>Monotaxis grandoculis</i> (Forsskål, 1775)	IP	1.67	0.42	0.2	1	0.5	3	0.5
Lutjanidae								
<i>Aphareus furca</i> (Lacepède, 1801)	IP	0.33	0.55	1.3	1	0.42	0.25	0.2
<i>Aphareus rutilans</i> Cuvier in Cuvier & Valenciennes, 1830	IP	0.11	0	0	3	0	0.92	0.5
<i>Aprion virescens</i> Valenciennes, 1830	IP	3.67	2.25	0.3	0.2	0.25	4.67	0.1
<i>Lutjanus argentimaculatus</i> (Forsskål, 1775)	IP	0	0	0	0.1	0	0.08	0
<i>Lutjanus bohar</i> (Forsskål, 1775)	IP	7.78	5.08	1.3	1.9	1	10.75	0.3
<i>Lutjanus fulviflamma</i> (Forsskål, 1775)	IP	0.11	0.08	6	8.2	0	24.17	0
<i>Lutjanus gibbus</i> (Forsskål, 1775)	IP	27.11	0.33	1	1.1	6.92	6.67	0.1

Species	D	LMS	RS	TMR	SMR	NMR	RR	SM
<i>Lutjanus kasmira</i> (Forsskål, 1775)	IP	0	4.33	20.7	10.4	0.42	51.33	0.1
<i>Lutjanus Lutjanus</i> Bloch, 1790	IP	0	0	10	7.2	0	0	0
<i>Lutjanus rivulatus</i> (Cuvier in Cuvier & Valenciennes, 1828)	IP	0.56	0.33	0.5	0.1	0.25	0.58	0
<i>Lutjanus russelli</i> (Bleeker, 1849)	IP	2	0.33	5	6.6	0.17	0	0.3
<i>Lutjanus sebae</i> (Cuvier, 1816)	IP	0	0	0	0	0	0.08	0
<i>Macolor niger</i> (Forsskål, 1775)	IP	0.89	0.91	1.6	1.6	0	1.75	0.1
<i>Paracaesio sordidus</i> Abe & Shinohara, 1962	IP	1.44	8.33	3.5	2.3	1.67	1.33	0
<i>Paracaesio xanthura</i> (Bleeker, 1869)	IP	0	0.08	0	0	0.08	0	0
Malacanthidae								
<i>Malacanthus brevisrostris</i> Guichenot, 1848	IP	0.44	0.17	0	0.1	0	0.33	0
<i>Malacanthus latovittatus</i> (Lacepède, 1801)	IP	0	0	0	0	0.08	0	0
Mobulidae								
<i>Manta birostris</i> (Walbaum, 1792)	IP	0.11	0	0	0	0	0	0
<i>Mobula kuhlii</i> (Müller & Henle, 1841)	IP	0	0	0	0.4	0	0	0
Monacanthidae								
<i>Cantherhines dumerilii</i> (Hollard, 1854)	IP	0	0.42	0.1	0	0.08	0.08	0.5
<i>Cantherhines pardalis</i> (Rüppell, 1837)	IP	0.33	0.58	0.3	0.9	0.75	0.08	1
<i>Paraluteres prionurus</i> (Bleeker, 1851)	IP	0	0.08	0	0.4	0.25	0	0
<i>Pervagor janthinosoma</i> (Bleeker, 1854)	IP	0.78	0.08	0.5	2.4	0.17	0.67	1.7
Mugiloididae								
<i>Paraperca punctulata</i> Cuvier in Cuvier & Valenciennes, 1829	I	0.67	0.42	0.1	0	0.08	0.08	0.3
Mullidae								
<i>Mulloidichthys vanicolensis</i> (Valenciennes, 1831)	IP	0	9.09	15.9	47	1	0.67	0
<i>Parupeneus bifasciatus</i> (Lacepède, 1801)	IP	0.22	0.58	0.4	0.5	0.25	1.75	0.2
<i>Parupeneus cyclostomus</i> (Lacepède, 1801)	IP	0.44	0.92	0.6	1	0.67	3.08	1.7
<i>Parupeneus indicus</i> (Shaw, 1803)	IP	1.78	0.08	0	0	0	0	0.1
<i>Parupeneus macronema</i> (Lacepède, 1801)	IP	0	2	3.2	1.8	1.75	4.08	12.4
<i>Parupeneus rubescens</i> (Lacepède, 1801)	IP	2.33	0	0	0	0.17	0.17	0
<i>Parupeneus pleurostigma</i> (Bennett, 1832)	IP	0	0	0.6	0.1	0	0	0.1
Muraenidae								
<i>Gymnomuraena zebra</i> (Shaw, 1797)	IP	0	0	0	0	0	0	0.1
<i>Gymnothorax brederi</i> McCosker & Randall, 1977	IP	0	0	0.2	0.3	0	0.25	0
<i>Gymnothorax eurostus</i> (Abbott, 1860)	IP	0	0	0	0.1	0	0	0.1
<i>Gymnothorax favagineus</i> (Bloch & Schneider, 1801)	IP	0	0.09	0.1	0.2	0	0	0
<i>Gymnocranius griseus</i> (Temminch & Schlegel, 1843)	WIO	0	0	0	0.2	0.08	0	0.1
<i>Gymnothorax meleagris</i> (Shaw, 1795)	IP	0	0	0.1	0.3	0	0.17	0.2
<i>Gymnothorax undulates</i> (Lacepède, 1801)	IP	0	0	0.1	0.2	0	0	0.1
Odontaspidae								
<i>Carcharias Taurus</i> Rafinesque, 1810	IP	0	0	0	0.1	0	0	0
Oplegnathidae								
<i>Oplegnathus robinsoni</i> Regan, 1916	WIO	4.22	1.25	1.2	0.7	0.67	1.75	0.3
Ostraciidae								
<i>Ostracion cubicus</i> (Linnaeus, 1758)	IP	0	0	0.4	0.1	0.08	0.25	0.1
<i>Ostracion meleagris</i> Shaw, 1796	IP	0	0	0.4	0.4	0	0.08	0.2
Pempheridae								
<i>Pempheris adusta</i> Bleeker, 1877	IP	33.33	0	7.5	0.1	7.08	5.17	0
<i>Parapriacanthus ransonneti</i> Steindachner, 1870	IP	2.22	0	3	0	8.33	69.17	200
Pomacanthidae								
<i>Apolemichthys trimaculatus</i> (Lacepède, 1801)	IP	2.11	2	1.2	1.1	0.17	0.83	0.3
<i>Centropyge acanthops</i> (Norman, 1922)	WIO	5.44	2.92	1.2	1.6	0.67	2.33	1.9

Species	D	LMS	RS	TMR	SMR	NMR	RR	SM
<i>Centropyge multispinis</i> (Playfair, 1867)	IP	0.33	2.08	3.5	2.9	2.58	3.33	5
<i>Centropyge bispinosa</i> (Günther, 1860)	IP	5.44	0	0.3	0	0.08	0.42	0.5
<i>Pomacanthus imperator</i> (Bloch, 1787)	IP	1	1.67	0.8	1.3	1.33	2.75	0.5
<i>Pomacanthus rhomboids</i> (Gilchrist & Thompson, 1908)	WIO	11.33	3.33	0.8	0.9	3.75	9.33	0.3
<i>Pomacanthus semicirculatus</i> (Cuvier, 1831)	IP	0.78	0.25	0.3	0.7	1.08	0.5	0.4
<i>Pygoplites diacanthus</i> (Boddaert, 1772)	IP	0	0	0.1	0	0.17	0.08	0
Pomcentridae								
<i>Abudefduf natalensis</i> (Hensley & Randall, 1983)	I	2.78	2.5	8.8	1.8	10.83	1.5	0
<i>Abudefduf vaigiensis</i> (Quoy & Gaimard)	IP	0	0	4.9	0	0	0	0
<i>Amphiprion akallopisos</i> (Bleeker 1853)	I	0.89	0	0	1.5	0.5	0	1.9
<i>Amphiprion allardi</i> Klausewitz, 1970	I	1.67	0.75	1	2.7	6.17	3.5	2.9
<i>Chromis dimidiata</i> (Klunzinger, 1871)	I	185.2	116.7	111.5	55.3	57.17	104.75	74.3
<i>Chromis nigrura</i> (Smith, 1960)	I	54.11	73.83	66.1	37.8	42.42	76.08	1.7
<i>Chromis opercularis</i> (Günther, 1867)	I	0	0.17	0.1	2.4	0.92	0.17	3.4
<i>Chromis weberi</i> (Fowler & Bean, 1928)	IP	94.44	19.5	104.6	94.1	54.67	23.08	39
<i>Dascyllus trimaculatus</i> (Rüppell, 1829)	IP	13.44	3	2.6	4.1	5.42	1.17	6.8
<i>Lepidozygus tapeinosoma</i> (Bleeker, 1856)	IP	0	0	4.9	0	0.83	0	0
<i>Neopomacentrus cyanomos</i> (Bleeker, 1856)	IP	1.67	0	0	0	0	0.33	0.5
<i>Plectroglyphidodon dickii</i> (Liènard, 1839)	IP	0.22	6.92	6.3	0	0.17	0.58	0.4
<i>Plectroglyphidodon johnstonianus</i> Fowler & Ball, 1924	IP	7.56	2.83	1.7	0	0.17	0.33	0.5
<i>Plectroglyphidodon lacrymatus</i> (Quoy & Gaimard, 1825)	IP	1.11	0	0	0	0	0.17	0.8
<i>Plectroglyphidodon leucozonus</i> (Bleeker, 1859)	IP	0.11	0.25	0	0	0	0	0
<i>Pomacentrus caeruleus</i> (Quoy & Gaumard, 1825)	I	0	7.42	1.3	1.7	4.42	1.08	6
<i>Stegastes nigricans</i> (Lacepède, 1802)	IP	0	0.17	0	0	0	0	0
Priacanthidae								
<i>Priacanthus hamrur</i> (Forsskål, 1775)	IP	1.22	0.17	4.4	7.6	0.75	0.08	1.7
Pseudochromidae								
<i>Pseudochromis dutoiti</i> Smith, 1955	WIO	0	0.17	0.5	2.1	0.42	0.5	0.3
<i>Pseudochromis natalensis</i> Regan, 1916	WIO	0	0	0	0	0	0.08	0
Rhincodontidae								
<i>Rhincodon typus</i> Smith, 1828	C	0	0	0	0	0	0	0.1
Scaridae								
<i>Calotomus carolinus</i> (Valenciennes, 1840)	IP	0.56	0.08	0.4	0.6	0.42	0.42	0.3
<i>Chlorurus atrilunula</i> (Randall & Bruce, 1983)	I	0.78	0.08	0.1	0.1	0	0.58	0.5
<i>Scarus cyanescens</i> Valenciennes in Cuvier & Valenciennes, 1840	I	0.44	0.58	0	0	0	0.17	0
<i>Scarus frenatus</i> (Lacepède, 1801)	IP	0	0	0	0	0.08	0	0
<i>Scarus ghobban</i> (Forsskål, 1775)	IP	0	0	0	0	0.42	0	0.1
<i>Scarus rubroviolaceus</i> Bleeker, 1847	IP	9.22	6.17	5.8	5.3	6.25	6.75	3.4
<i>Scarus tricolor</i> Bleeker, 1847	IP	0.56	0.33	0	0.7	0.08	0	0.3
Scombridae								
<i>Euthynnus affinis</i> (Cantor, 1849)	IP	0	0	0	0	0	0.83	0
<i>Scomberomorus commerson</i> (Lacepède, 1801)	IP	0	0	0	0	0.08	0.25	0
Scorpaenidae								
<i>Pterois miles</i> (Bennett, 1825)	IP	0	0	0	0	0	0	0.1
<i>Scorpaenopsis diabolus</i> (Cuvier, 1829)	IP	0	0.08	0	0	0	0	0
<i>Scorpaenopsis venosa</i> (Cuvier, 1829)	IP	0.11	0	0	0.1	0	0.08	0.3
<i>Scorpaenopsis oxycephala</i> (Bleeker, 1849)	IP	0.11	0.25	0.5	0	0	0.08	0.1
<i>Sebastapistes cyanostigma</i> (Bleeker, 1856)	IP	0.11	0.17	0	0	0.17	0.08	0

Species	D	LMS	RS	TMR	SMR	NMR	RR	SM
Serranidae								
<i>Aethaloperca rogae</i> (Forsskål, 1775)	IP	0.78	0.67	1.2	0.5	0.33	0.5	0.3
<i>Cephalopholis argus</i> Bloch & Schneider, 1801	IP	0.33	0.17	0.1	0	0.67	0.17	0
Species								
<i>Cephalopholis miniata</i> (Forsskål, 1775)	IP	0.22	0.17	1	1.7	1	0.75	0.3
<i>Cephalopholis urodeta</i> (Forster in Bloch & Schneider, 1801)	IP	0.22	0.92	0.1	0.6	0.25	0.67	1
<i>Dermatolepis striolatus</i> (Playfair in Playfair & Günther, 1867)	WIO	0	0	0	0	0	0.08	0
<i>Epinephelus caeruleopunctatus</i> (Bloch, 1790)	IP	0	0.08	0	0	0	0	0.1
<i>Epinephelus fasciatus</i> (Forsskål, 1775)	IP	1.22	0.75	0.9	0	0.83	0.25	0
<i>Epinephelus flavocaeruleus</i> (Lacepède, 1801)	IP	0	0.25	0.2	0	0.25	0.17	0
<i>Epinephelus macrospilos</i> (Bleeker, 1855)	IP	0	0	0	0.1	0	0	0
<i>Epinephelus marginatus</i> (Lowe, 1834)	C	0.11	0	0	0	0.17	0	0
<i>Epinephelus posteli</i> Fourmanoir & Crosnier, 1964	WIO	0.22	0	0	0	0.08	0.08	0
<i>Epinephelus tukula</i> Morgans, 1959	IP	1.78	2.08	0.2	0.3	1.08	1.58	0
<i>Grammistes sexlineatus</i> (Thunber, 1782)	IP	0	0	0	0.2	0	0	0
<i>Nemanthias carberryi</i> Smith, 1954	WIO	0	0	19	72	12.5	0	0
<i>Plectropomus punctatus</i> Quoy & Gaimard, 1824	I	0.89	0	0.2	0.2	0.17	1.33	0.1
<i>Pseudanthias cooperi</i> (Regan, 1902)	IP	14.44	0	17	12.6	1	0	0
<i>Pseudanthias squamipinnis</i> Peters, 1855	IP	112.6	79.92	90.9	173	145.3	28.42	8.3
<i>Pseudanthias evansi</i> (Smith, 1954)	I	0	0	0	0.1	0	0.33	0
<i>Variola louti</i> (Forsskål, 1775)	IP	3.56	4.92	1.3	0.6	0.75	2.33	1.2
Siganidae								
<i>Siganus sutor</i> (Valenciennes in Cuvier & Valenciennes, 1835)	WIO	2	0.25	1.1	2.1	0.17	0.67	1.3
Sparidae								
<i>Chrysoblephus puniceus</i> (Gilchrist & Thompson, 1908)	WIO	0	0	0.8	0	8.33	0	0
<i>Diplodus cervinus hottentotus</i> (Bleeker, 1844)	E	1.22	0	2.4	0	0	0	0
<i>Polyamblyodon gibbosum</i> (Pellegrin, 1914)	WIO	4.78	0.67	0.2	0.2	1.33	1.58	0
<i>Polysteganus praeorbitalis</i> (Günther, 1859)	WIO	0.22	0.08	0	0	0	0	0
Sphyraenidae								
<i>Sphyraena jello</i> Cuvier in Cuvier & Valenciennes, 1829	IP	0.22	0	0.1	0.2	3.25	7.5	20
<i>Sphyraena putnamiae</i> Jordan & Seale, 1905	IP	0	0	0	0.3	0	0	0
Synodontidae								
<i>Synodus dermatogenys</i> Fowler, 1912	IP	0	0	0	0.2	0	0	0.1
Tetraodontidae								
<i>Arothron hispidus</i> (Linnaeus, 1758)	IP	0.11	0.08	0.1	0.3	0.17	0	0.1
<i>Arothron meleagris</i> (Lacepède, 1798)	IP	0.11	0	0.1	0.2	0.17	0.17	0.1
<i>Arothron nigropunctatus</i> Bloch & Schneider 1801	IP	0.11	0.17	0.1	0.2	0	0.08	0.3
<i>Canthigaster amboinensis</i> (Bleeker, 1865)	IP	0	0	0	0	0	0	0.1
<i>Canthigaster coronata</i> (Vaillant & Sauvage, 1875)	IP	0	0	0	0	0	0	0.1
<i>Canthigaster valentini</i> (Bleeker, 1853)	IP	0	0	0.5	0	0.25	0	0.9
Zanclidae								
<i>Zanclus canescens</i> (Linnaeus, 1758)	IP	2.78	2.75	2.3	4.4	2.92	3.42	1.5

Appendix 2

SIMPER results of percent contribution of each species to overall dissimilarity between reefs. Cumulative cut-off to exclude species with low contributions was 33%. Species in bold are considered potentially good discriminating species according to criteria discussed in Clarke and Warwick (2001)

Species	TMR Average abundance	NMR Average abundance	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
<i>Chromis weberi</i>	3.07	2.18	0.62	1.25	1.25	1.25
<i>Caesio xanthonota</i>	1.09	0.94	0.61	1.22	1.25	2.5
<i>Chromis nigrura</i>	2.23	2.26	0.59	1.37	1.19	3.69
<i>Lutjanus kasmira</i>	1.2	0.12	0.58	0.97	1.19	4.88
<i>Thalassoma amblycephalum</i>	1.15	1.63	0.56	1.25	1.15	6.03
<i>Plectroglyphidodon dickii</i>	1.19	0.17	0.56	1.5	1.13	7.16
<i>Amphiprion allardi</i>	0.5	1.45	0.53	1.5	1.09	8.25
<i>Abudefduf natalensis</i>	0.51	0.95	0.52	1	1.07	9.32
<i>Odonus niger</i>	0.52	0.88	0.51	0.96	1.04	10.36
<i>Mulloides vanicolensis</i>	1.01	0.25	0.5	0.97	1.03	11.39
<i>Ctenochaetus truncatus</i>	1.11	0.42	0.48	1.29	0.98	12.37
<i>Chromis dimidiata</i>	2.89	2.59	0.46	1.53	0.93	13.3
<i>Pempheris adusta</i>	0.64	0.56	0.45	0.84	0.92	14.22
<i>Carangoides fulvoguttatus</i>	0.15	0.85	0.45	0.97	0.91	15.13
<i>Pseudanthias squamipinnis</i>	2.88	3.25	0.44	1.21	0.89	16.03
<i>Myripristis murdjan</i>	0.97	0.72	0.43	1.29	0.87	16.9
<i>Plectroglyphidodon johnstonianus</i>	0.94	0.17	0.43	1.59	0.87	17.77
<i>Pomacentrus caeruleus</i>	0.42	0.82	0.43	1.08	0.87	18.64
<i>Cirrhilabrus exquisitus</i>	0.1	0.86	0.42	1.2	0.86	19.5
<i>Caesio caeruleus</i>	0.55	0.55	0.42	0.74	0.86	20.36
<i>Acanthurus thompsoni</i>	0.78	0.52	0.41	1.09	0.84	21.2
<i>Acanthurus dussumieri</i>	0.22	0.89	0.4	1.37	0.82	22.02
<i>Pomacanthus rhomboides</i>	0.55	1.19	0.4	1.29	0.81	22.83
<i>Dascyllus trimaculatus</i>	0.85	1.3	0.4	1.28	0.81	23.64
<i>Pomacanthus semicirculatus</i>	0.13	0.81	0.39	1.56	0.8	24.44
<i>Priacanthus hamrur</i>	0.75	0.39	0.39	1.05	0.79	25.23
<i>Hemitaurichthys zoster</i>	0.3	0.87	0.39	1.28	0.79	26.02
<i>Lutjanus fulviflamma</i>	0.76	0	0.38	0.78	0.77	26.79
<i>Paracaesio sordidus</i>	0.57	0.55	0.38	1	0.77	27.56
<i>Caranx melampygus</i>	0.7	0.44	0.37	1.07	0.76	28.32
<i>Nemanthias carberryi</i>	0.37	0.49	0.37	0.55	0.74	29.06
<i>Epinephelus tukula</i>	0.12	0.74	0.36	1.3	0.74	29.8
<i>Chaetodon lunula</i>	0.7	0.55	0.36	1.11	0.73	30.53
<i>Sargocentron caudimaculatum</i>	0.44	0.97	0.36	1.27	0.73	31.26
<i>Thalassoma lunare</i>	0.54	0.73	0.36	1.22	0.73	31.98
<i>Plagiotremus tapeinosoma</i>	0.3	0.75	0.36	1.1	0.72	32.71
<i>Zebrasoma scopas</i>	0.73	0.08	0.35	1.1	0.72	33.42

Average dissimilarity = 49.16

Species	SMR Average abundance	NMR Average abundance	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
<i>Nemanthias carberryi</i>	2.59	0.49	1.08	1.92	2.28	2.28
<i>Odonus niger</i>	2.57	0.88	0.87	1.81	1.83	4.11
<i>Mulloides vanicolensis</i>	1.67	0.25	0.75	1.22	1.59	5.7
<i>Chromis weberi</i>	2.66	2.18	0.62	1.3	1.32	7.02
<i>Pseudanthias cooperi</i>	1.28	0.16	0.6	1.26	1.28	8.3
<i>Lutjanus kasmira</i>	1.28	0.12	0.58	1.37	1.23	9.53

<i>Caesio xanthonota</i>	0.76	0.94	0.56	0.97	1.19	10.73
<i>Priacanthus hamrur</i>	1.27	0.39	0.51	1.39	1.08	11.81
<i>Pseudanthias squamipinnis</i>	3.32	3.25	0.47	1.13	1	12.81
<i>Myripristis murdjan</i>	1.21	0.72	0.47	1.24	0.99	13.8
<i>Abudefduf natalensis</i>	0.58	0.95	0.46	1.14	0.98	14.78
<i>Carangoides fulvoguttatus</i>	0.22	0.85	0.45	1.01	0.94	15.72
<i>Caranx melampygus</i>	1.04	0.44	0.43	1.31	0.91	16.63
<i>Pterocaesio tile</i>	0.75	0.38	0.42	0.76	0.9	17.53
<i>Halichoeres hortulanus</i>	0.4	1.19	0.41	1.58	0.87	18.4
<i>Cirrhilabrus exquisitus</i>	0.75	0.86	0.41	1.24	0.87	19.27
<i>Pomacentrus caeruleus</i>	0.65	0.82	0.4	1.19	0.86	20.13
<i>Pervagor janthinosoma</i>	0.92	0.17	0.4	1.38	0.84	20.97
<i>Thalassoma amblycephalum</i>	1.82	1.63	0.4	1.1	0.84	21.81
<i>Macolor niger</i>	0.83	0	0.39	1.48	0.83	22.64
<i>Lutjanus lutjanus</i>	0.86	0	0.39	0.91	0.83	23.47
<i>Cephalopholis miniata</i>	1.03	0.42	0.39	1.58	0.82	24.29
<i>Hemitaurichthys zoster</i>	1.05	0.87	0.38	1.23	0.81	25.1
<i>Siganus sutor</i>	0.86	0.17	0.38	1.31	0.8	25.9
<i>Paracaesio sordidus</i>	0.98	0.55	0.38	1.37	0.8	26.7
<i>Ctenochaetus truncatus</i>	0.93	0.42	0.38	1.28	0.8	27.5
<i>Lutjanus gibbus</i>	0.76	0.5	0.37	1.13	0.79	28.3
<i>Acanthurus thompsoni</i>	0.81	0.52	0.37	1.16	0.78	29.08
<i>Acanthurus dussumieri</i>	0.32	0.89	0.36	1.3	0.76	29.83
<i>Chromis dimidiata</i>	2.42	2.59	0.36	1.46	0.76	30.59
<i>Nemateleotris magnifica</i>	0.64	0.63	0.35	1.12	0.75	31.34
<i>Lutjanus fulviflamma</i>	0.76	0	0.35	0.76	0.75	32.09
<i>Lutjanus bohar</i>	0.98	0.51	0.35	1.22	0.73	32.83
<i>Plagiotremus tapeinosoma</i>	0.47	0.75	0.34	1.14	0.73	33.55

Average dissimilarity = 47.24

Species	TMR	LMS	Average diissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Lutjanus gibbus</i>	0.37	1.64	0.66	1.37	1.36	1.36
<i>Chromis weberi</i>	3.07	2.26	0.61	1.15	1.26	2.62
<i>Naso hexacanthus</i>	0.16	1.27	0.59	1.08	1.21	3.83
<i>Lethrinus crocineus</i>	0.22	1.4	0.59	1.8	1.2	5.03
<i>Thalassoma amblycephalum</i>	1.15	0.62	0.53	1.21	1.1	6.13
<i>Lutjanus kasmira</i>	1.2	0	0.53	0.94	1.09	7.22
<i>Chromis dimidiata</i>	2.89	3.46	0.51	1.39	1.05	8.27
<i>Pomacanthus rhomboides</i>	0.55	1.37	0.5	1.39	1.02	9.29
<i>Caesio xanthonota</i>	1.09	0.59	0.49	1.18	1	10.29
<i>Dascyllus trimaculatus</i>	0.85	1.11	0.49	1.29	1	11.29
<i>Ctenochaetus truncatus</i>	1.11	1.49	0.49	1.3	1	12.29
<i>Chromis nigrura</i>	2.23	2.69	0.48	1.41	0.99	13.28
<i>Aprion virescens</i>	0.3	1.33	0.48	2	0.99	14.27
<i>Acanthurus thompsoni</i>	0.78	0.84	0.45	1.11	0.92	15.19
<i>Mulloides vanicolensis</i>	1.01	0	0.45	0.88	0.92	16.11
<i>Odonus niger</i>	0.52	0.69	0.45	0.79	0.91	17.02
<i>Centropyge acanthops</i>	0.51	1.31	0.45	1.44	0.91	17.94
<i>Lethrinus rubrioperculatus</i>	0	0.94	0.44	1.29	0.91	18.85
<i>Acanthurus tennentii</i>	1.16	1.92	0.43	1.48	0.88	19.73
<i>Epinephelus tukula</i>	0.12	0.94	0.41	1.58	0.85	20.58
<i>Plectrogliphidodon dickii</i>	1.19	1.06	0.41	1.3	0.85	21.42
<i>Myripristis murdjan</i>	0.97	0.3	0.41	1.31	0.84	22.27
<i>Pseudanthias squamipinnis</i>	2.88	2.94	0.41	1.28	0.84	23.1

<i>Caranx melampygus</i>	0.7	1.17	0.39	1.39	0.79	23.9
<i>Lutjanus bohar</i>	0.8	1.59	0.38	1.29	0.78	24.67
<i>Thalassoma lunare</i>	0.54	0.95	0.38	1.41	0.77	25.44
<i>Priacanthus hamrur</i>	0.75	0.31	0.36	0.99	0.74	26.19
<i>Acanthurus dussumieri</i>	0.22	0.86	0.36	1.25	0.74	26.93
<i>Lutjanus fulviflamma</i>	0.76	0.11	0.36	0.85	0.74	27.66
<i>Balistoides conspicillum</i>	0.12	0.8	0.36	1.65	0.73	28.39
<i>Kyphosus vaigiensis</i>	0.66	0.37	0.35	0.81	0.73	29.12
<i>Pempheris adusta</i>	0.64	0.23	0.35	0.73	0.72	29.84
<i>Hemitaurichthys zoster</i>	0.3	0.73	0.35	0.94	0.72	30.56
<i>Cirrhilabrus exquisitus</i>	0.1	0.74	0.34	1.13	0.7	31.26
<i>Chaetodon lunula</i>	0.7	0.79	0.34	1.2	0.7	31.96
<i>Pomacentrus caeruleus</i>	0.42	0.66	0.33	1.13	0.69	32.65
<i>Chaetodon auriga</i>	0.46	0.97	0.33	1.25	0.68	33.33

Average dissimilarity = 48.70

Species	SMR	LMS	Average diissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Nemanthias carberryi</i>	2.59	0	1.13	2.68	2.24	2.24
<i>Odonus niger</i>	2.57	0.69	0.94	2.48	1.88	4.12
<i>Mulloides vanicolensis</i>	1.67	0	0.71	1.16	1.41	5.54
<i>Thalassoma amblycephalum</i>	1.82	0.62	0.65	1.71	1.29	6.83
<i>Lethrinus crocineus</i>	0	1.4	0.63	2.26	1.25	8.08
<i>Chromis weberi</i>	2.66	2.26	0.63	1.18	1.25	9.32
<i>Pseudanthias cooperi</i>	1.28	0.38	0.61	1.31	1.22	10.54
<i>Naso hexacanthus</i>	0.21	1.27	0.56	1.07	1.11	11.66
<i>Chromis dimidiata</i>	2.42	3.46	0.55	1.53	1.11	12.76
<i>Lutjanus kasmira</i>	1.28	0	0.55	1.4	1.1	13.87
<i>Priacanthus hamrur</i>	1.27	0.31	0.51	1.46	1.01	14.88
<i>Pseudanthias squamipinnis</i>	3.32	2.94	0.5	1.31	1.01	15.88
<i>Aprion virescens</i>	0.2	1.33	0.49	2.56	0.98	16.86
<i>Lutjanus gibbus</i>	0.76	1.64	0.48	1.14	0.96	17.82
<i>Myripristis murdjan</i>	1.21	0.3	0.48	1.32	0.95	18.77
<i>Dascyllus trimaculatus</i>	1.09	1.11	0.47	1.49	0.93	19.71
<i>Plectroglyphidodon dickii</i>	0	1.06	0.46	1.25	0.91	20.62
<i>Hemitaurichthys zoster</i>	1.05	0.73	0.44	1.32	0.87	21.49
<i>Caesio xanthonota</i>	0.76	0.59	0.44	0.92	0.87	22.36
<i>Paracaesio sordidus</i>	0.98	0.21	0.43	1.75	0.86	23.23
<i>Ctenochaetus truncatus</i>	0.93	1.49	0.43	1.17	0.86	24.09
<i>Lethrinus rubrioperculatus</i>	0	0.94	0.42	1.3	0.84	24.93
<i>Acanthurus thompsoni</i>	0.81	0.84	0.42	1.26	0.84	25.77
<i>Acanthurus tennentii</i>	1.02	1.92	0.41	1.29	0.82	26.59
<i>Pomacanthus rhomboides</i>	0.82	1.37	0.4	1.48	0.8	27.39
<i>Variola louti</i>	0.52	1.37	0.39	1.51	0.77	28.16
<i>Cephalopholis miniata</i>	1.03	0.22	0.37	1.74	0.75	28.9
<i>Halichoeres hortulanus</i>	0.4	1.16	0.37	1.5	0.74	29.64
<i>Epinephelus tukula</i>	0.22	0.94	0.37	1.47	0.74	30.38
<i>Cirrhilabrus exquisitus</i>	0.75	0.74	0.36	1.22	0.72	31.1
<i>Oplegnathus robinsoni</i>	0.53	1.12	0.36	1.4	0.72	31.82
<i>Lutjanus lutjanus</i>	0.86	0	0.36	0.91	0.72	32.55
<i>Aulostomus chinensis</i>	0.84	0.11	0.36	1.39	0.71	33.25
<i>Amphiprion allardi</i>	1.17	0.61	0.35	1.27	0.69	33.95

Average dissimilarity = 50.17

Species	NMR Average abundance	LMS Average abundance	Average diissimilarity	Diss/SD	Percent contribution	Cumulative percent
<i>Lutjanus gibbus</i>	0.5	1.64	0.68	1.4	1.38	1.38
<i>Chromis weberi</i>	2.18	2.26	0.66	1.31	1.36	2.74
<i>Thalassoma amblycephalum</i>	1.63	0.62	0.65	1.51	1.34	4.08
<i>Lethrinus crocineus</i>	0.08	1.4	0.64	2.09	1.3	5.38
<i>Naso hexacanthus</i>	0.08	1.27	0.6	1.1	1.23	6.61
<i>Ctenochaetus truncatus</i>	0.42	1.49	0.58	1.32	1.18	7.79
<i>Odonus niger</i>	0.88	0.69	0.56	0.93	1.14	8.93
<i>Aprion virescens</i>	0.18	1.33	0.54	2.53	1.11	10.04
<i>Lutjanus bohar</i>	0.51	1.59	0.51	1.67	1.05	11.09
<i>Chromis dimidiata</i>	2.59	3.46	0.51	1.5	1.04	12.13
<i>Caesio xanthonota</i>	0.94	0.59	0.5	1	1.02	13.15
<i>Dascyllus trimaculatus</i>	1.3	1.11	0.5	1.69	1.02	14.17
<i>Plectroglyphidodon dickii</i>	0.17	1.06	0.46	1.24	0.95	15.12
<i>Amphiprion allardi</i>	1.45	0.61	0.46	1.35	0.94	16.06
<i>Centropyge acanthops</i>	0.46	1.31	0.46	1.48	0.94	17
<i>Lethrinus rubrioperculatus</i>	0	0.94	0.45	1.3	0.92	17.92
<i>Abudefduf natalensis</i>	0.95	0.25	0.45	0.95	0.91	18.83
<i>Pseudanthias squamipinnis</i>	3.25	2.94	0.43	1.4	0.89	19.72
<i>Apolemichthys trimaculatus</i>	0.17	1.04	0.43	1.77	0.89	20.61
<i>Caranx melampygus</i>	0.44	1.17	0.43	1.57	0.89	21.5
<i>Carangoides fulvoguttatus</i>	0.85	0.5	0.43	1.11	0.87	22.37
<i>Acanthurus thompsoni</i>	0.52	0.84	0.42	1.12	0.86	23.24
<i>Hemitaurichthys zoster</i>	0.87	0.73	0.42	1.29	0.86	24.1
<i>Pomacentrus caeruleus</i>	0.82	0.66	0.4	1.25	0.81	24.91
<i>Pomacanthus rhomboides</i>	1.19	1.37	0.39	1.45	0.79	25.71
<i>Polyamblyodon gibbosum</i>	0.62	0.47	0.37	1.11	0.77	26.48
<i>Sargocentron caudimaculatum</i>	0.97	0.41	0.37	1.42	0.75	27.22
<i>Oplegnathus robinsoni</i>	0.6	1.12	0.37	1.37	0.75	27.97
<i>Variola louti</i>	0.61	1.37	0.36	1.34	0.74	28.71
<i>Acanthurus tennentii</i>	1.2	1.92	0.36	1.35	0.74	29.45
<i>Cirrhilabrus exquisitus</i>	0.86	0.74	0.36	1.31	0.74	30.19
<i>Pomacanthus semicircularatus</i>	0.81	0.29	0.35	1.48	0.71	30.9
<i>Coris caudimacula</i>	0.89	0.58	0.34	1.3	0.7	31.6
<i>Nemateleotris magnifica</i>	0.63	0.53	0.34	1.12	0.69	32.29
<i>Chaetodon madagaskariensis</i>	0.76	1.37	0.34	1.21	0.69	32.98
<i>Myripristis murdjan</i>	0.72	0.3	0.33	1.02	0.69	33.67

Average dissimilarity = 48.83

Species	TMR Average abundance	RR Average abundance	Average diissimilarity	Diss/SD	Percent contribution	Cumulative percent
<i>Caesio xanthonota</i>	1.09	2.38	0.79	1.44	1.6	1.6
<i>Odonus niger</i>	0.52	1.63	0.7	1.29	1.43	3.03
<i>Lutjanus kasmira</i>	1.2	0.85	0.66	1.02	1.35	4.38
<i>Chromis weberi</i>	3.07	1.93	0.56	1.34	1.14	5.52
<i>Chromis nigrura</i>	2.23	2.67	0.55	1.35	1.11	6.63
<i>Pomacanthus rhomboides</i>	0.55	1.46	0.52	1.52	1.06	7.7
<i>Lutjanus fulviflamma</i>	0.76	0.56	0.49	0.89	0.99	8.69
<i>Parapriacanthus ransonneti</i>	0.23	0.94	0.49	0.64	0.99	9.68
<i>Aprion virescens</i>	0.3	1.28	0.49	1.66	0.99	10.67
<i>Chromis dimidiata</i>	2.89	2.9	0.47	1.27	0.97	11.64
<i>Thalassoma amblycephalum</i>	1.15	1.16	0.47	1.32	0.96	12.6
<i>Pseudanthias squamipinnis</i>	2.88	2.1	0.46	1.34	0.94	13.54

<i>Plectroglyphidodon dickii</i>	1.19	0.38	0.46	1.38	0.94	14.48
<i>Mulloides vanicolensis</i>	1.01	0.23	0.46	0.97	0.94	15.42
<i>Lutjanus bohar</i>	0.8	1.62	0.45	1.48	0.92	16.35
<i>Epinephelus tukula</i>	0.12	1.04	0.45	2.2	0.92	17.26
<i>Naso unicornis</i>	0.46	1.19	0.45	1.34	0.91	18.17
<i>Lutjanus gibbus</i>	0.37	1.01	0.45	1.15	0.91	19.08
<i>Caranx melampygus</i>	0.7	1.17	0.44	1.27	0.9	19.98
<i>Acanthurus mata</i>	0.41	0.78	0.43	0.9	0.88	20.86
<i>Acanthurus thompsoni</i>	0.78	1.09	0.43	1.32	0.88	21.74
<i>Acanthurus dussumieri</i>	0.22	0.93	0.41	1.24	0.83	22.57
<i>Pempheris adusta</i>	0.64	0.47	0.41	0.84	0.82	23.39
<i>Caesio caeruleus</i>	0.55	0.54	0.4	0.65	0.82	24.21
<i>Amphiprion allardi</i>	0.5	1.09	0.4	1.27	0.81	25.03
<i>Kyphosus vaigiensis</i>	0.66	0.56	0.39	1.03	0.79	25.82
<i>Myripristis murdjan</i>	0.97	0.8	0.38	1.28	0.77	26.59
<i>Acanthurus leucosternon</i>	1.31	0.84	0.38	1.19	0.77	27.37
<i>Parupeneus cyclostomus</i>	0.44	0.95	0.37	1.3	0.76	28.13
<i>Ctenochaetus truncatus</i>	1.11	0.92	0.37	1.31	0.75	28.88
<i>Plectroglyphidodon</i>	0.94	0.27	0.37	1.47	0.75	29.63
<i>Paracirrhites arcatus</i>	0.91	0.35	0.36	1.28	0.74	30.37
<i>Balistoides conspicillum</i>	0.12	0.8	0.35	1.52	0.71	31.08
<i>Abudefduf natalensis</i>	0.51	0.45	0.35	0.84	0.71	31.79
<i>Acanthurus tennenti</i>	1.16	1.41	0.35	1.35	0.7	32.5
<i>Zebrasoma scopas</i>	0.73	0.79	0.35	1.18	0.7	33.2
<i>Dascyllus trimaculatus</i>	0.85	0.61	0.35	1.15	0.7	33.91

Average dissimilarity = 49.17

Species	SMR	RR	Average diissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Nemanthias carberryi</i>	2.59	0	1.14	2.68	2.29	2.29
<i>Caesio xanthonota</i>	0.76	2.38	0.87	1.53	1.76	4.05
<i>Pseudanthias squamipinnis</i>	3.32	2.1	0.73	2.14	1.46	5.51
<i>Mulloides vanicolensis</i>	1.67	0.23	0.7	1.22	1.41	6.91
<i>Lutjanus kasmira</i>	1.28	0.85	0.65	1.37	1.31	8.23
<i>Odonus niger</i>	2.57	1.63	0.59	1.31	1.19	9.42
<i>Pseudanthias cooperi</i>	1.28	0	0.57	1.27	1.16	10.57
<i>Priacanthus hamrur</i>	1.27	0.08	0.54	1.58	1.1	11.67
<i>Chromis weberi</i>	2.66	1.93	0.54	1.33	1.09	12.76
<i>Naso unicornis</i>	0.12	1.19	0.5	1.43	1.02	13.77
<i>Aprion virescens</i>	0.2	1.28	0.49	1.91	0.99	14.76
<i>Lutjanus fulviflamma</i>	0.76	0.56	0.46	0.87	0.93	15.69
<i>Chromis dimidiata</i>	2.42	2.9	0.46	1.43	0.92	16.61
<i>Thalassoma amblycephalum</i>	1.82	1.16	0.43	1.23	0.87	17.48
<i>Paracirrhites arcatus</i>	1.26	0.35	0.42	1.65	0.84	18.32
<i>Acanthurus mata</i>	0.4	0.78	0.41	0.98	0.83	19.15
<i>Myripristis murdjan</i>	1.21	0.8	0.41	1.28	0.83	19.98
<i>Pomacanthus rhomboides</i>	0.82	1.46	0.41	1.7	0.82	20.8
<i>Parapriacanthus ransonneti</i>	0	0.94	0.4	0.55	0.81	21.62
<i>Acanthurus leucosternon</i>	1.66	0.84	0.4	1.24	0.81	22.43
<i>Hemitaurichthys zoster</i>	1.05	0.8	0.39	1.3	0.79	23.22
<i>Epinephelus tukula</i>	0.22	1.04	0.39	1.82	0.79	24.01
<i>Paracaesio sordidus</i>	0.98	0.37	0.39	1.54	0.79	24.79
<i>Halichoeres hortulanus</i>	0.4	1.25	0.38	1.62	0.77	25.56
<i>Aulostomus chinensis</i>	0.84	0	0.38	1.47	0.76	26.32

<i>Acanthurus dussumieri</i>	0.32	0.93	0.37	1.23	0.75	27.07
<i>Acanthurus thompsoni</i>	0.81	1.09	0.37	1.28	0.75	27.81
<i>Caranx melampygus</i>	1.04	1.17	0.37	1.16	0.74	28.56
<i>Lutjanus lutjanus</i>	0.86	0	0.36	0.91	0.73	29.29
<i>Cirrhilabrus exquisitus</i>	0.75	0.57	0.36	1.1	0.73	30.02
<i>Dascyllus trimaculatus</i>	1.09	0.61	0.36	1.36	0.72	30.74
<i>Lutjanus gibbus</i>	0.76	1.01	0.36	1.21	0.72	31.46
<i>Lutjanus bohar</i>	0.98	1.62	0.35	1.42	0.71	32.17
<i>Centropyge acanthops</i>	0.82	0.61	0.35	1.33	0.7	32.87
<i>Parupeneus cyclostomus</i>	0.58	0.95	0.34	1.22	0.68	33.55

Average dissimilarity = 49.69

Species	NMR Average abundance	RR Average abundance	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
<i>Caesio xanthonota</i>	0.94	2.38	0.9	1.53	1.86	1.86
<i>Odonus niger</i>	0.88	1.63	0.7	1.28	1.44	3.31
<i>Lutjanus bohar</i>	0.51	1.62	0.56	1.67	1.17	4.48
<i>Pseudanthias squamipinnis</i>	3.25	2.1	0.56	1.31	1.16	5.63
<i>Aprion virescens</i>	0.18	1.28	0.54	1.92	1.12	6.75
<i>Carangoides fulvoguttatus</i>	0.85	0.53	0.54	1.05	1.11	7.86
<i>Parapriacanthus ransonneti</i>	0.26	0.94	0.5	0.63	1.03	8.89
<i>Chromis weberi</i>	2.18	1.93	0.49	1.3	1.02	9.91
<i>Lutjanus gibbus</i>	0.5	1.01	0.48	1.17	1	10.91
<i>Thalassoma amblycephalum</i>	1.63	1.16	0.47	1.22	0.97	11.89
<i>Caranx melampygus</i>	0.44	1.17	0.47	1.3	0.97	12.86
<i>Naso unicornis</i>	0.46	1.19	0.46	1.32	0.96	13.82
<i>Abudefduf natalensis</i>	0.95	0.45	0.45	1.08	0.93	14.75
<i>Chromis dimidiata</i>	2.59	2.9	0.44	1.53	0.9	15.66
<i>Chromis nigrura</i>	2.26	2.67	0.43	1.15	0.89	16.55
<i>Sargocentron diadema</i>	0.21	1	0.43	1.7	0.89	17.44
<i>Acanthurus thompsoni</i>	0.52	1.09	0.42	1.32	0.87	18.31
<i>Plectorhinchus flavomaculatus</i>	0.18	0.98	0.42	1.55	0.86	19.17
<i>Lutjanus kasmira</i>	0.12	0.85	0.42	0.61	0.86	20.03
<i>Caesio caeruleus</i>	0.55	0.54	0.41	0.69	0.86	20.89
<i>Dascyllus trimaculatus</i>	1.3	0.61	0.4	1.43	0.83	21.72
<i>Pomacentrus caeruleus</i>	0.82	0.36	0.4	1.05	0.82	22.54
<i>Acanthurus mata</i>	0.1	0.78	0.4	0.75	0.82	23.36
<i>Anampses caeruleopunctatus</i>	1.27	0.65	0.39	1.29	0.81	24.17
<i>Cirrhilabrus exquisitus</i>	0.86	0.57	0.38	1.26	0.79	24.96
<i>Plagiotremus tapeinosoma</i>	0.75	0.6	0.37	1.18	0.77	25.74
<i>Hemitaurichthys zoster</i>	0.87	0.8	0.37	1.26	0.77	26.51
<i>Paracirrhites arcatus</i>	1.01	0.35	0.37	1.41	0.77	27.28
<i>Pempheris adusta</i>	0.56	0.47	0.37	0.79	0.77	28.06
<i>Myripristis murdjan</i>	0.72	0.8	0.37	1.16	0.77	28.83
<i>Ctenochaetus truncatus</i>	0.42	0.92	0.37	1.33	0.76	29.59
<i>Parupeneus cyclostomus</i>	0.53	0.95	0.36	1.33	0.75	30.34
<i>Zebrasoma scopas</i>	0.08	0.79	0.35	1.13	0.73	31.08
<i>Pomacanthus rhomboides</i>	1.19	1.46	0.35	1.4	0.72	31.79
<i>Chaetodon madagaskariensis</i>	0.76	1.38	0.34	1.2	0.7	32.49
<i>Acanthurus leucosternon</i>	0.99	0.84	0.34	1.28	0.7	33.19
<i>Balistoides conspicillum</i>	0.17	0.8	0.33	1.42	0.69	33.88

Average dissimilarity = 48.37

Species	LMS	RR	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Caesio xanthonota</i>	0.59	2.38	0.87	1.68	1.89	1.89
<i>Odonus niger</i>	0.69	1.63	0.7	1.27	1.53	3.42
<i>Naso hexacanthus</i>	1.27	0.24	0.55	1.12	1.2	4.62
<i>Chromis weberi</i>	2.26	1.93	0.55	1.19	1.19	5.81
<i>Parapriacanthus ransonneti</i>	0.46	0.94	0.51	0.65	1.11	6.92
<i>Lutjanus gibbus</i>	1.64	1.01	0.5	1.31	1.09	8.02
<i>Thalassoma amblycephalum</i>	0.62	1.16	0.48	1.43	1.04	9.05
<i>Naso unicornis</i>	0.33	1.19	0.47	1.33	1.02	10.07
<i>Pseudanthias squamipinnis</i>	2.94	2.1	0.46	1.24	1.01	11.08
<i>Chromis dimidiata</i>	3.46	2.9	0.46	1.25	1.01	12.09
<i>Dascyllus trimaculatus</i>	1.11	0.61	0.45	1.26	0.98	13.07
<i>Acanthurus thompsoni</i>	0.84	1.09	0.44	1.42	0.96	14.03
<i>Centropyge acanthops</i>	1.31	0.61	0.42	1.38	0.92	14.95
<i>Plectroglyphidodon dickii</i>	1.06	0.38	0.41	1.23	0.88	15.84
<i>Lethrinus crocineus</i>	1.4	0.61	0.4	1.53	0.88	16.72
<i>Lethrinus rubrioperculatus</i>	0.94	0.12	0.4	1.29	0.88	17.6
<i>Ctenochaetus truncatus</i>	1.49	0.92	0.4	1.12	0.87	18.47
<i>Hemitaurichthys zoster</i>	0.73	0.8	0.4	1.17	0.86	19.33
<i>Pomacanthus rhomboides</i>	1.37	1.46	0.39	1.21	0.85	20.18
<i>Carangoides fulvoguttatus</i>	0.5	0.53	0.39	0.77	0.85	21.03
<i>Acanthurus mata</i>	0.23	0.78	0.38	0.76	0.83	21.86
<i>Acanthurus leucosternon</i>	1.32	0.84	0.38	1.3	0.83	22.69
<i>Acanthurus nigrofuscus</i>	2.4	1.84	0.37	1.06	0.8	23.49
<i>Amphiprion allardi</i>	0.61	1.09	0.36	1.2	0.78	24.27
<i>Lutjanus kasmira</i>	0	0.85	0.36	0.55	0.77	25.05
<i>Labroides bicolor</i>	0.22	0.94	0.34	1.54	0.75	25.79
<i>Myripristis murdjan</i>	0.3	0.8	0.34	1.11	0.74	26.54
<i>Polyamblyodon gibbosum</i>	0.47	0.55	0.33	0.97	0.73	27.26
<i>Thalassoma lunare</i>	0.95	0.33	0.33	1.42	0.72	27.99
<i>Apolemichthys trimaculatus</i>	1.04	0.49	0.33	1.36	0.72	28.71
<i>Monotaxis grandoculis</i>	0.35	0.7	0.33	1.03	0.72	29.42
<i>Acanthurus dussumieri</i>	0.86	0.93	0.33	1.2	0.71	30.14
<i>Caranx melampygus</i>	1.17	1.17	0.32	1.21	0.7	30.84
<i>Cirrhitilabrus exquisitus</i>	0.74	0.57	0.32	1.13	0.7	31.54
<i>Anampses caeruleopunctatus</i>	0.97	0.65	0.32	1.22	0.69	32.23
<i>Acanthurus xanthopterus</i>	0.47	0.57	0.32	0.93	0.69	32.92
<i>Epinephelus fasciatus</i>	0.76	0.18	0.32	1.26	0.69	33.61

Average dissimilarity = 45.91

Species	TMR	RS	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Chromis weberi</i>	3.07	1.54	0.82	1.44	1.72	1.72
<i>Odonus niger</i>	0.52	1.18	0.62	1.04	1.3	3.02
<i>Caesio xanthonota</i>	1.09	1.05	0.62	1.22	1.29	4.31
<i>Lutjanus kasmira</i>	1.2	0.32	0.59	0.99	1.25	5.56
<i>Chromis nigrura</i>	2.23	2.75	0.58	1.36	1.21	6.77
<i>Thalassoma amblycephalum</i>	1.15	1.54	0.56	1.22	1.16	7.93
<i>Pseudanthias squamipinnis</i>	2.88	2.45	0.54	1.25	1.14	9.07
<i>Mulloides vanicolensis</i>	1.01	0.26	0.53	0.93	1.11	10.18
<i>Acanthurus thompsoni</i>	0.78	1.02	0.5	1.19	1.04	11.22
<i>Epinephelus tukula</i>	0.12	1.03	0.48	1.78	1.01	12.23
<i>Chromis dimidiata</i>	2.89	3.1	0.46	1.38	0.96	13.2

<i>Lethrinus rubrioperculatus</i>	0	0.9	0.46	1.27	0.96	14.15
<i>Dascyllus trimaculatus</i>	0.85	0.42	0.44	1.18	0.92	15.07
<i>Pomacentrus caeruleus</i>	0.42	0.84	0.44	1.01	0.92	15.99
<i>Myripristis murdjan</i>	0.97	0.55	0.42	1.27	0.88	16.87
<i>Pterocaesio tile</i>	0.38	0.69	0.42	0.74	0.87	17.74
<i>Ctenochaetus truncatus</i>	1.11	1.43	0.41	1.24	0.86	18.6
<i>Acanthurus tennenti</i>	1.16	1.69	0.4	1.5	0.85	19.45
<i>Lethrinus crocineus</i>	0.22	0.91	0.4	1.4	0.85	20.29
<i>Caranx melampygus</i>	0.7	1.07	0.4	1.36	0.84	21.14
<i>Centropyge acanthops</i>	0.51	0.84	0.4	1.18	0.83	21.97
<i>Caesio caeruleaureus</i>	0.55	0.39	0.39	0.56	0.83	22.79
<i>Aprion virescens</i>	0.3	0.87	0.39	1.27	0.81	23.6
<i>Lutjanus fulviflamma</i>	0.76	0.08	0.38	0.83	0.81	24.41
<i>Coris caudimacula</i>	0.66	1.04	0.38	1.1	0.8	25.21
<i>Plectroglyphidodon dickii</i>	1.19	1.32	0.38	1.21	0.8	26.01
<i>Pomacanthus rhomboides</i>	0.55	0.99	0.38	1.2	0.79	26.8
<i>Thalassoma hebraicum</i>	1.55	2.16	0.37	1.3	0.78	27.59
<i>Priacanthus hamrur</i>	0.75	0.1	0.37	0.93	0.78	28.36
<i>Paracaesio sordidus</i>	0.57	0.26	0.37	0.75	0.77	29.13
<i>Lutjanus bohar</i>	0.8	1.37	0.36	1.21	0.76	29.89
<i>Parupeneus macronema</i>	0.95	0.85	0.36	1.2	0.76	30.65
<i>Chaetodon lunula</i>	0.7	0.58	0.36	1.09	0.75	31.4
<i>Thalassoma lunare</i>	0.54	0.67	0.35	1.17	0.74	32.15
<i>Zebрасoma scopas</i>	0.73	0.96	0.35	1.18	0.73	32.88
<i>Plectorhinchus playfairi</i>	0.72	0.1	0.35	1.42	0.73	33.61

Average dissimilarity = 47.72

Species	SMR	RS	Average diissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Nemanthias carberryi</i>	2.59	0	1.21	2.69	2.41	2.41
<i>Odonus niger</i>	2.57	1.18	0.77	1.59	1.53	3.94
<i>Mulloides vanicolensis</i>	1.67	0.26	0.76	1.17	1.52	5.46
<i>Chromis weberi</i>	2.66	1.54	0.73	1.46	1.46	6.92
<i>Pseudanthias squamipinnis</i>	3.32	2.45	0.73	1.56	1.45	8.37
<i>Plectroglyphidodon dickii</i>	0	1.32	0.62	1.98	1.23	9.6
<i>Pseudanthias cooperi</i>	1.28	0	0.61	1.27	1.22	10.82
<i>Lutjanus kasmira</i>	1.28	0.32	0.59	1.38	1.17	11.98
<i>Caesio xanthonota</i>	0.76	1.05	0.58	0.99	1.16	13.14
<i>Priacanthus hamrur</i>	1.27	0.1	0.58	1.55	1.15	14.29
<i>Plectroglyphidodon</i>	0	1.19	0.57	2.92	1.13	15.42
<i>Paracaesio sordidus</i>	0.98	0.26	0.52	1.69	1.03	16.45
<i>Pterocaesio tile</i>	0.75	0.69	0.5	0.84	0.99	17.44
<i>Hemitaurchichthys zoster</i>	1.05	0.59	0.5	1.36	0.99	18.43
<i>Dascyllus trimaculatus</i>	1.09	0.42	0.48	1.52	0.95	19.38
<i>Myripristis murdjan</i>	1.21	0.55	0.48	1.28	0.95	20.33
<i>Chromis dimidiata</i>	2.42	3.1	0.47	1.43	0.93	21.26
<i>Cephalopholis miniata</i>	1.03	0.1	0.46	2.24	0.91	22.17
<i>Acanthurus thompsoni</i>	0.81	1.02	0.45	1.24	0.89	23.06
<i>Halichoeres hortulanus</i>	0.4	1.32	0.44	1.78	0.88	23.93
<i>Variola louti</i>	0.52	1.4	0.43	1.4	0.86	24.8
<i>Lethrinus rubrioperculatus</i>	0	0.9	0.43	1.28	0.86	25.65
<i>Epinephelus tukula</i>	0.22	1.03	0.43	1.62	0.85	26.5
<i>Lethrinus crocineus</i>	0	0.91	0.42	1.63	0.84	27.34
<i>Amphiprion allardi</i>	1.17	0.4	0.42	1.48	0.84	28.18
<i>Thalassoma amblycephalum</i>	1.82	1.54	0.42	1.09	0.83	29.01

<i>Pomacentrus caeruleus</i>	0.65	0.84	0.42	1.12	0.83	29.84
<i>Pervagor janthinosoma</i>	0.92	0.08	0.41	1.43	0.81	30.65
<i>Aulostomus chinensis</i>	0.84	0	0.4	1.47	0.8	31.45
<i>Macolor niger</i>	0.83	0.15	0.39	1.49	0.78	32.23
<i>Siganus sutor</i>	0.86	0.11	0.39	1.36	0.78	33.01
<i>Chaetodon blackburnii</i>	1.05	0.35	0.39	1.48	0.78	33.79

Average dissimilarity = 50.27

Species	NMR	RS	Average diissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Odonus niger</i>	0.88	1.18	0.67	1.12	1.4	1.4
<i>Caesio xanthonota</i>	0.94	1.05	0.65	1.08	1.35	2.75
<i>Chromis weberi</i>	2.18	1.54	0.64	1.34	1.34	4.09
<i>Plectroglyphidodon dickii</i>	0.17	1.32	0.61	1.78	1.27	5.36
<i>Pseudanthias squamipinnis</i>	3.25	2.45	0.61	1.21	1.26	6.63
<i>Dascyllus trimaculatus</i>	1.3	0.42	0.58	1.89	1.21	7.84
<i>Amphiprion allardi</i>	1.45	0.4	0.58	1.71	1.2	9.04
<i>Ctenochaetus truncatus</i>	0.42	1.43	0.57	1.59	1.2	10.23
<i>Plectroglyphidodon</i>	0.17	1.19	0.54	2.08	1.14	11.37
<i>Thalassoma amblycephalum</i>	1.63	1.54	0.5	1.18	1.05	12.42
<i>Pomacentrus caeruleus</i>	0.82	0.84	0.49	1.15	1.02	13.44
<i>Lutjanus bohar</i>	0.51	1.37	0.48	1.47	1.01	14.45
<i>Acanthurus thompsoni</i>	0.52	1.02	0.48	1.17	1	15.45
<i>Abudefduf natalensis</i>	0.95	0.2	0.48	0.94	0.99	16.44
<i>Hemitaurichthys zoster</i>	0.87	0.59	0.47	1.3	0.98	17.42
<i>Apolemichthys trimaculatus</i>	0.17	1.02	0.46	1.7	0.97	18.39
<i>Lethrinus rubrioperculatus</i>	0	0.9	0.46	1.28	0.96	19.35
<i>Zebрасoma scopas</i>	0.08	0.96	0.46	1.6	0.96	20.31
<i>Carangoides fulvoguttatus</i>	0.85	0.1	0.45	1	0.94	21.25
<i>Caranx melampygus</i>	0.44	1.07	0.44	1.48	0.91	22.16
<i>Chromis nigrura</i>	2.26	2.75	0.44	1.09	0.91	23.08
<i>Lethrinus crocineus</i>	0.08	0.91	0.43	1.55	0.9	23.98
<i>Pterocaesio tile</i>	0.38	0.69	0.43	0.71	0.89	24.87
<i>Cirrhilabrus exquisitus</i>	0.86	0.35	0.42	1.27	0.89	25.76
<i>Aprion virescens</i>	0.18	0.87	0.41	1.29	0.87	26.62
<i>Variola louti</i>	0.61	1.4	0.41	1.25	0.86	27.48
<i>Caesio caeruleus</i>	0.55	0.39	0.41	0.6	0.85	28.34
<i>Chromis dimidiata</i>	2.59	3.1	0.4	1.3	0.83	29.17
<i>Centropyge acanthops</i>	0.46	0.84	0.4	1.23	0.83	30
<i>Plectorhinchus playfairi</i>	0.81	0.1	0.39	1.58	0.82	30.82
<i>Halichoeres cosmetus</i>	1.17	0.63	0.38	1.38	0.8	31.61
<i>Sargocentron caudimaculatum</i>	0.97	0.52	0.38	1.36	0.79	32.4
<i>Myripristis murdjan</i>	0.72	0.55	0.37	1.1	0.78	33.18
<i>Paracaesio sordidus</i>	0.55	0.26	0.37	0.82	0.77	33.95

Average dissimilarity = 47.88

Species	LMS	RS	Average diissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Chromis weberi</i>	2.26	1.54	0.7	1.32	1.63	1.63
<i>Lutjanus gibbus</i>	1.64	0.27	0.67	1.4	1.56	3.19
<i>Odonus niger</i>	0.69	1.18	0.64	1.03	1.49	4.68
<i>Thalassoma amblycephalum</i>	0.62	1.54	0.63	1.46	1.46	6.15
<i>Naso hexacanthus</i>	1.27	0.52	0.57	1.27	1.33	7.48

<i>Pseudanthias squamipinnis</i>	2.94	2.45	0.53	1.24	1.25	8.72
<i>Caesio xanthonota</i>	0.59	1.05	0.53	1.05	1.23	9.96
<i>Dascyllus trimaculatus</i>	1.11	0.42	0.51	1.09	1.19	11.15
<i>Acanthurus thompsoni</i>	0.84	1.02	0.49	1.21	1.14	12.29
<i>Pomacanthus rhomboides</i>	1.37	0.99	0.43	1.35	1	13.29
<i>Hemitaurichthys zoster</i>	0.73	0.59	0.42	0.98	0.99	14.28
<i>Ctenochaetus truncatus</i>	1.49	1.43	0.41	1.37	0.97	15.25
<i>Chromis dimidiata</i>	3.46	3.1	0.41	1.46	0.95	16.2
<i>Pomacentrus caeruleus</i>	0.66	0.84	0.41	1.17	0.95	17.16
<i>Plectroglyphidodon dickii</i>	1.06	1.32	0.41	1.28	0.95	18.11
<i>Coris caudimacula</i>	0.58	1.04	0.4	1.23	0.93	19.04
<i>Acanthurus nigrofuscus</i>	2.4	2.01	0.38	1.11	0.89	19.93
<i>Centropyge acanthops</i>	1.31	0.84	0.38	1.21	0.88	20.81
<i>Lethrinus rubrioperculatus</i>	0.94	0.9	0.36	1.28	0.85	21.66
<i>Lethrinus crocineus</i>	1.4	0.91	0.36	1.22	0.83	22.49
<i>Cirrhilabrus exquisitus</i>	0.74	0.35	0.34	1.1	0.8	23.29
<i>Chaetodon blackburnii</i>	0.92	0.35	0.34	1.34	0.79	24.09
<i>Cirrhitichthys oxycephalus</i>	0.88	0.96	0.34	1.18	0.78	24.87
<i>Zebrasoma scopas</i>	0.51	0.96	0.33	1.22	0.78	25.64
<i>Acanthurus leucosternon</i>	1.32	1.23	0.32	1.21	0.76	26.4
<i>Acanthurus dussumieri</i>	0.86	0.57	0.32	1.2	0.75	27.15
<i>Plectroglyphidodon</i>	0.65	1.19	0.31	1.19	0.73	27.88
<i>Halichoeres cosmetus</i>	0.86	0.63	0.31	1.19	0.73	28.61
<i>Chaetodon vagabundus</i>	0.75	0.28	0.31	1.26	0.72	29.34
<i>Anampses meleagrides</i>	0.67	0.46	0.31	1.14	0.72	30.06
<i>Amphiprion allardi</i>	0.61	0.4	0.3	1.01	0.71	30.77
<i>Bodianus axillaris</i>	0.71	0.28	0.3	1.25	0.71	31.48
<i>Forcipiger flavissimus</i>	1.01	1.03	0.3	1.09	0.71	32.18
<i>Pterocaesio tile</i>	0	0.69	0.3	0.56	0.7	32.88
<i>Macolor niger</i>	0.62	0.15	0.3	1.14	0.7	33.58

Average dissimilarity = 42.85

Species	RR	RS	Average diissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Caesio xanthonota</i>	2.38	1.05	0.84	1.39	1.81	1.81
<i>Odonus niger</i>	1.63	1.18	0.7	1.3	1.52	3.33
<i>Plectroglyphidodon dickii</i>	0.38	1.32	0.5	1.54	1.08	4.41
<i>Chromis weberi</i>	1.93	1.54	0.49	1.31	1.07	5.48
<i>Thalassoma amblycephalum</i>	1.16	1.54	0.48	1.3	1.04	6.52
<i>Plectroglyphidodon</i>	0.27	1.19	0.46	1.82	1	7.51
<i>Pseudanthias squamipinnis</i>	2.1	2.45	0.46	1.04	0.99	8.5
<i>Lutjanus kasmira</i>	0.85	0.32	0.45	0.67	0.98	9.48
<i>Naso unicornis</i>	1.19	0.47	0.45	1.31	0.98	10.47
<i>Lutjanus gibbus</i>	1.01	0.27	0.44	1.16	0.96	11.42
<i>Acanthurus thompsoni</i>	1.09	1.02	0.43	1.26	0.94	12.36
<i>Hemitaurichthys zoster</i>	0.8	0.59	0.43	1.13	0.92	13.29
<i>Parapriacanthus ransonneti</i>	0.94	0	0.43	0.55	0.92	14.21
<i>Amphiprion allardi</i>	1.09	0.4	0.42	1.37	0.91	15.12
<i>Chromis dimidiata</i>	2.9	3.1	0.42	1.25	0.91	16.03
<i>Lethrinus rubrioperculatus</i>	0.12	0.9	0.42	1.28	0.9	16.93
<i>Pomacanthus rhomboides</i>	1.46	0.99	0.41	1.32	0.89	17.82
<i>Pomacentrus caeruleus</i>	0.36	0.84	0.41	0.99	0.88	18.7
<i>Acanthurus mata</i>	0.78	0.17	0.4	0.75	0.87	19.57
<i>Sargocentron diadema</i>	1	0.28	0.39	1.53	0.85	20.42
<i>Plectorhinchus playfairi</i>	0.88	0.1	0.39	1.54	0.85	21.27
<i>Centropyge acanthops</i>	0.61	0.84	0.39	1.15	0.84	22.11

<i>Acanthurus leucosternon</i>	0.84	1.23	0.39	1.22	0.83	22.94
<i>Labroides bicolor</i>	0.94	0.18	0.39	1.69	0.83	23.78
<i>Coris caudimacula</i>	0.61	1.04	0.39	1.21	0.83	24.61
<i>Caesio caeruleaureus</i>	0.54	0.39	0.38	0.52	0.82	25.43
<i>Chromis nigrura</i>	2.67	2.75	0.37	1.46	0.79	26.22
<i>Acanthurus dussumieri</i>	0.93	0.57	0.36	1.23	0.78	27.01
<i>Paracirrhites arcatus</i>	0.35	1.02	0.36	1.45	0.78	27.78
<i>Myripristis murdjan</i>	0.8	0.55	0.36	1.14	0.78	28.56
<i>Anampses caeruleopunctatus</i>	0.65	0.99	0.35	1.2	0.76	29.32
<i>Caranx melampygus</i>	1.17	1.07	0.35	1.17	0.76	30.08
<i>Apolemichthys trimaculatus</i>	0.49	1.02	0.35	1.27	0.76	30.84
<i>Parupeneus cyclostomus</i>	0.95	0.64	0.35	1.29	0.76	31.6
<i>Dascyllus trimaculatus</i>	0.61	0.42	0.34	1.08	0.74	32.34
<i>Plagiotremus tapeinosoma</i>	0.6	0.59	0.34	1.12	0.74	33.09
<i>Zebrasoma scopas</i>	0.79	0.96	0.34	1.22	0.74	33.83

Average dissimilarity = 46.21

Species	TMR	SM	Average diissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Pseudanthias squamipinnis</i>	2.88	0.91	1.08	1.8	2	2
<i>Chromis nigrura</i>	2.23	0.63	0.93	1.53	1.72	3.72
<i>Chromis weberi</i>	3.07	2.07	0.64	1.38	1.19	4.91
<i>Odonus niger</i>	0.52	1.26	0.64	1.26	1.18	6.09
<i>Parapriacanthus ransonneti</i>	0.23	1.12	0.63	0.59	1.17	7.26
<i>Lutjanus kasmira</i>	1.2	0.1	0.61	0.98	1.13	8.39
<i>Ptereleotris evides</i>	0	1.13	0.61	1.24	1.13	9.52
<i>Cirrhilabrus exquisitus</i>	0.1	1.14	0.6	1.23	1.1	10.62
<i>Thalassoma amblycephalum</i>	1.15	0.45	0.57	1.26	1.06	11.68
<i>Pomacentrus caeruleus</i>	0.42	1.26	0.55	1.39	1.02	12.7
<i>Caesio xanthonota</i>	1.09	0.63	0.54	1.21	1.01	13.71
<i>Chromis dimidiata</i>	2.89	2.63	0.53	1.24	0.99	14.7
<i>Plectroglyphidodon dickii</i>	1.19	0.32	0.53	1.42	0.99	15.69
<i>Mulloides vanicolensis</i>	1.01	0	0.51	0.88	0.94	16.63
<i>Naso brevirostris</i>	0	0.97	0.5	1.12	0.93	17.56
<i>Dascyllus trimaculatus</i>	0.85	1.21	0.48	1.21	0.88	18.44
<i>Myripristis murdjan</i>	0.97	0.38	0.46	1.3	0.84	19.29
<i>Naso hexacanthus</i>	0.16	0.83	0.45	0.82	0.83	20.12
<i>Parupeneus macronema</i>	0.95	1.71	0.45	1.25	0.83	20.95
<i>Oplegnathus robinsoni</i>	0.87	0.13	0.44	1.69	0.82	21.77
<i>Ctenochaetus binotatus</i>	0.42	0.91	0.43	1.52	0.8	22.58
<i>Priacanthus hamrur</i>	0.75	0.73	0.43	1.22	0.8	23.38
<i>Ctenochaetus truncatus</i>	1.11	1.1	0.43	1.29	0.79	24.17
<i>Amphiprion allardi</i>	0.5	1.06	0.43	1.26	0.79	24.96
<i>Acanthurus thompsoni</i>	0.78	0.38	0.42	1.02	0.78	25.75
<i>Plagiotremus tapeinosoma</i>	0.3	0.91	0.42	1.31	0.78	26.53
<i>Lutjanus fulviflamma</i>	0.76	0	0.4	0.78	0.74	27.27
<i>Paracirrhites arcatus</i>	0.91	0.42	0.4	1.27	0.74	28
<i>Acanthurus leucosternon</i>	1.31	0.8	0.39	1.19	0.72	28.73
<i>Siganus sutor</i>	0.28	0.7	0.39	1.2	0.72	29.44
<i>Thalassoma lunare</i>	0.54	0.71	0.38	1.2	0.71	30.15
<i>Pervagor janthinosa</i>	0.42	1.02	0.38	1.32	0.71	30.86
<i>Centropyge acanthops</i>	0.51	0.67	0.38	1.1	0.7	31.57
<i>Chaetodon lunula</i>	0.7	0.32	0.38	1.02	0.7	32.27
<i>Lutjanus bohar</i>	0.8	0.22	0.38	1.32	0.7	32.97
<i>Caranx melampygus</i>	0.7	0.26	0.38	1.02	0.7	33.67

Average dissimilarity = 54.03

Species	SMR Average abundance	SM Average abundance	Average diissimilarity	Diss/SD	Percent contribution	Cumulative percent
<i>Pseudanthias squamipinnis</i>	3.32	0.91	1.32	2.05	2.46	2.46
<i>Nemanthias carberryi</i>	2.59	0	1.28	2.68	2.38	4.84
<i>Chromis nigrura</i>	2.39	0.63	0.89	2.14	1.65	6.5
<i>Mulloides vanicolensis</i>	1.67	0	0.8	1.17	1.49	7.99
<i>Thalassoma amblycephalum</i>	1.82	0.45	0.74	1.88	1.39	9.38
<i>Odonus niger</i>	2.57	1.26	0.7	1.47	1.3	10.68
<i>Pseudanthias cooperi</i>	1.28	0	0.65	1.26	1.21	11.88
<i>Chromis weberi</i>	2.66	2.07	0.63	1.41	1.17	13.06
<i>Lutjanus kasmira</i>	1.28	0.1	0.61	1.4	1.14	14.19
<i>Myripristis murdjan</i>	1.21	0.38	0.54	1.39	1	15.19
<i>Parapriacanthus ransonneti</i>	0	1.12	0.52	0.5	0.96	16.15
<i>Ptereleotris evides</i>	0.37	1.13	0.51	1.18	0.96	17.11
<i>Cirrhilabrus exquisitus</i>	0.75	1.14	0.51	1.25	0.95	18.07
<i>Caesio xanthonota</i>	0.76	0.63	0.51	1	0.94	19.01
<i>Paracaesio sordidus</i>	0.98	0	0.5	1.73	0.93	19.94
<i>Naso brevirostris</i>	0	0.97	0.47	1.13	0.88	20.82
<i>Chromis dimidiata</i>	2.42	2.63	0.47	1.4	0.87	21.69
<i>Caranx melampygus</i>	1.04	0.26	0.47	1.35	0.87	22.56
<i>Pomacentrus caeruleus</i>	0.65	1.26	0.46	1.3	0.85	23.42
<i>Priacanthus hamrur</i>	1.27	0.73	0.45	1.3	0.83	24.25
<i>Acanthurus leucosternon</i>	1.66	0.8	0.44	1.43	0.83	25.08
<i>Naso hexacanthus</i>	0.21	0.83	0.44	0.83	0.83	25.9
<i>Paracirrhites arcatus</i>	1.26	0.42	0.44	1.49	0.81	26.71
<i>Lutjanus bohar</i>	0.98	0.22	0.44	1.53	0.81	27.52
<i>Cephalopholis miniata</i>	1.03	0.22	0.43	1.8	0.81	28.33
<i>Hemitaurichthys zoster</i>	1.05	0.58	0.43	1.28	0.81	29.14
<i>Chaetodon blackburnii</i>	1.05	0.3	0.43	1.52	0.8	29.94
<i>Dascyllus trimaculatus</i>	1.09	1.21	0.42	1.25	0.79	30.73
<i>Halichoeres hortulanus</i>	0.4	1.22	0.41	1.53	0.77	31.5
<i>Lutjanus lutjanus</i>	0.86	0	0.41	0.91	0.76	32.26
<i>Parupeneus macronema</i>	0.94	1.71	0.4	1.21	0.74	33
<i>Bodianus diana</i>	1.21	0.56	0.4	1.33	0.74	33.74

Average dissimilarity = 53.74

Species	NMR Average abundance	SM Average abundance	Average diissimilarity	Diss/SD	Percent contribution	Cumulative percent
<i>Pseudanthias squamipinnis</i>	3.25	0.91	1.27	1.88	2.41	2.41
<i>Chromis nigrura</i>	2.26	0.63	0.94	1.91	1.78	4.19
<i>Thalassoma amblycephalum</i>	1.63	0.45	0.74	1.53	1.39	5.58
<i>Odonus niger</i>	0.88	1.26	0.67	1.27	1.26	6.84
<i>Parapriacanthus ransonneti</i>	0.26	1.12	0.64	0.57	1.21	8.05
<i>Chromis weberi</i>	2.18	2.07	0.6	1.39	1.13	9.19
<i>Pomacentrus rhomboides</i>	1.19	0.13	0.6	2.21	1.13	10.32
<i>Ptereleotris evides</i>	0.39	1.13	0.57	1.21	1.09	11.4
<i>Caesio xanthonota</i>	0.94	0.63	0.57	1.05	1.08	12.48
<i>Naso brevirostris</i>	0	0.97	0.51	1.13	0.96	13.44
<i>Pomacentrus caeruleus</i>	0.82	1.26	0.51	1.28	0.96	14.4
<i>Cirrhilabrus exquisitus</i>	0.86	1.14	0.5	1.28	0.94	15.35
<i>Ctenochaetus truncatus</i>	0.42	1.1	0.49	1.41	0.92	16.27
<i>Sargocentron caudimaculatum</i>	0.97	0.1	0.48	1.82	0.92	17.18
<i>Pervagor janthinosoma</i>	0.17	1.02	0.48	1.87	0.91	18.09
<i>Abudefduf natalensis</i>	0.95	0	0.48	0.9	0.9	19
<i>Carangoides fulvoguttatus</i>	0.85	0.15	0.48	0.99	0.9	19.9
<i>Bodianus diana</i>	1.39	0.56	0.47	1.41	0.88	20.79

<i>Chromis dimidiata</i>	2.59	2.63	0.46	1.51	0.87	21.66
<i>Naso hexacanthus</i>	0.08	0.83	0.45	0.82	0.85	22.51
<i>Acanthurus dussumieri</i>	0.89	0.25	0.44	1.46	0.83	23.34
<i>Dascyllus trimaculatus</i>	1.3	1.21	0.42	1.31	0.8	24.14
<i>Parupeneus macronema</i>	0.98	1.71	0.41	1.17	0.77	24.9
<i>Plagiotremus tapeinosoma</i>	0.75	0.91	0.4	1.25	0.76	25.66
<i>Halichoeres cosmetus</i>	1.17	0.66	0.4	1.26	0.76	26.42
<i>Paracirrhites arcatus</i>	1.01	0.42	0.4	1.33	0.75	27.17
<i>Sphyaena jello</i>	0.21	0.63	0.4	0.57	0.75	27.92
<i>Myripristis murdjan</i>	0.72	0.38	0.4	1.11	0.75	28.68
<i>Epinephelus tukula</i>	0.74	0	0.4	1.37	0.75	29.42
<i>Ctenochaetus binotatus</i>	0.5	0.91	0.39	1.31	0.74	30.17
<i>Forcipiger flavissimus</i>	1.26	0.65	0.39	1.21	0.74	30.91
<i>Chaetodon blackburnii</i>	0.87	0.3	0.39	1.29	0.74	31.66
<i>Nemateleotris magnifica</i>	0.63	0.58	0.39	1.16	0.74	32.39
<i>Hemitaenichthys zoster</i>	0.87	0.58	0.39	1.22	0.74	33.13
<i>Priacanthus hamrur</i>	0.39	0.73	0.38	1.13	0.72	33.85

Average dissimilarity = 52.82

Species	RR	SM	Average diissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Chromis nigrura</i>	2.67	0.63	1.03	1.99	1.91	1.91
<i>Caesio xanthonota</i>	2.38	0.63	0.97	1.75	1.8	3.71
<i>Parapriacanthus ransonneti</i>	0.94	1.12	0.8	0.74	1.48	5.19
<i>Pseudanthias squamipinnis</i>	2.1	0.91	0.72	1.4	1.34	6.53
<i>Lutjanus bohar</i>	1.62	0.22	0.71	2.21	1.32	7.85
<i>Pomacanthus rhomboides</i>	1.46	0.13	0.68	1.79	1.27	9.12
<i>Odonus niger</i>	1.63	1.26	0.68	1.38	1.26	10.37
<i>Aprion virescens</i>	1.28	0.1	0.59	2.12	1.1	11.48
<i>Ptereleotris evides</i>	0	1.13	0.57	1.26	1.05	12.53
<i>Naso unicornis</i>	1.19	0.2	0.54	1.45	1	13.54
<i>Pomacentrus caeruleus</i>	0.36	1.26	0.53	1.42	0.98	14.52
<i>Caranx melampygus</i>	1.17	0.26	0.52	1.36	0.96	15.48
<i>Epinephelus tukula</i>	1.04	0	0.51	3.06	0.95	16.44
<i>Cirrhilabrus exquisitus</i>	0.57	1.14	0.5	1.21	0.92	17.36
<i>Chromis dimidiata</i>	2.9	2.63	0.49	1.16	0.91	18.27
<i>Lutjanus gibbus</i>	1.01	0.1	0.49	1.16	0.91	19.19
<i>Dascyllus trimaculatus</i>	0.61	1.21	0.48	1.43	0.89	20.08
<i>Thalassoma amblycephalum</i>	1.16	0.45	0.47	1.36	0.88	20.96
<i>Sphyaena jello</i>	0.54	0.63	0.47	0.73	0.88	21.84
<i>Acanthurus mata</i>	0.78	0.42	0.47	0.83	0.87	22.71
<i>Naso brevirostris</i>	0	0.97	0.47	1.13	0.87	23.58
<i>Acanthurus thompsoni</i>	1.09	0.38	0.47	1.37	0.87	24.45
<i>Chromis weberi</i>	1.93	2.07	0.46	1.39	0.86	25.31
<i>Plectorhinchus flavomaculatus</i>	0.98	0.1	0.45	1.67	0.84	26.15
<i>Acanthurus dussumieri</i>	0.93	0.25	0.43	1.26	0.81	26.96
<i>Naso hexacanthus</i>	0.24	0.83	0.43	0.86	0.8	27.76
<i>Lutjanus kasmira</i>	0.85	0.1	0.43	0.61	0.79	28.55
<i>Plagiotremus tapeinosoma</i>	0.6	0.91	0.4	1.29	0.75	29.3
<i>Pomacanthus imperator</i>	1.13	0.42	0.4	1.36	0.75	30.05
<i>Oplegnathus robinsoni</i>	0.83	0.13	0.39	1.31	0.73	30.78
<i>Myripristis murdjan</i>	0.8	0.38	0.39	1.18	0.73	31.51
<i>Plectorhinchus playfairi</i>	0.88	0.2	0.39	1.43	0.72	32.22
<i>Centropyge acanthops</i>	0.61	0.67	0.38	1.1	0.71	32.93
<i>Ctenochaetus binotatus</i>	0.25	0.91	0.38	1.44	0.7	33.64

Average dissimilarity = 53.77

Species	LMS	SM	Average diissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Pseudanthias squamipinnis</i>	2.94	0.91	1.04	1.7	1.97	1.97
<i>Chromis nigrura</i>	2.69	0.63	1.02	2.69	1.93	3.89
<i>Lutjanus gibbus</i>	1.64	0.1	0.76	1.53	1.44	5.34
<i>Lethrinus crocineus</i>	1.4	0	0.7	2.26	1.32	6.66
<i>Odonus niger</i>	0.69	1.26	0.68	1.32	1.29	7.95
<i>Lutjanus bohar</i>	1.59	0.22	0.67	2.63	1.27	9.22
<i>Chromis weberi</i>	2.26	2.07	0.66	1.36	1.25	10.47
<i>Parapriacanthus ransonneti</i>	0.46	1.12	0.64	0.61	1.22	11.69
<i>Pomacanthus rhomboides</i>	1.37	0.13	0.63	1.53	1.18	12.87
<i>Naso hexacanthus</i>	1.27	0.83	0.62	1.13	1.17	14.04
<i>Aprion virescens</i>	1.33	0.1	0.6	3.41	1.14	15.18
<i>Chromis dimidiata</i>	3.46	2.63	0.58	1.33	1.1	16.28
<i>Dascyllus trimaculatus</i>	1.11	1.21	0.54	1.38	1.03	17.31
<i>Ptereleotris evides</i>	0.3	1.13	0.51	1.22	0.97	18.28
<i>Oplegnathus robinsoni</i>	1.12	0.13	0.51	1.63	0.97	19.25
<i>Caranx melampygus</i>	1.17	0.26	0.49	1.8	0.93	20.18
<i>Lethrinus rubrioperculatus</i>	0.94	0	0.47	1.3	0.89	21.07
<i>Epinephelus tukula</i>	0.94	0	0.46	1.8	0.88	21.95
<i>Plectroglyphidodon dickii</i>	1.06	0.32	0.46	1.25	0.87	22.82
<i>Naso brevirostris</i>	0.31	0.97	0.46	1.15	0.86	23.68
<i>Ctenochaetus truncatus</i>	1.49	1.1	0.45	1.2	0.85	24.53
<i>Pomacentrus caeruleus</i>	0.66	1.26	0.45	1.38	0.85	25.37
<i>Cirrhilabrus exquisitus</i>	0.74	1.14	0.45	1.2	0.85	26.22
<i>Centropyge acanthops</i>	1.31	0.67	0.43	1.31	0.82	27.04
<i>Acanthurus thompsoni</i>	0.84	0.38	0.43	1.03	0.81	27.85
<i>Macropharyngodon bipartitus</i>	0	0.84	0.41	1.91	0.78	28.63
<i>Caesio xanthonota</i>	0.59	0.63	0.41	1.02	0.78	29.41
<i>Apolemichthys trimaculatus</i>	1.04	0.3	0.4	1.48	0.76	30.17
<i>Hemitaurichthys zoster</i>	0.73	0.58	0.4	1.1	0.76	30.93
<i>Acanthurus tennentii</i>	1.92	1.2	0.39	1.31	0.74	31.67
<i>Amphiprion allardi</i>	0.61	1.06	0.39	1.26	0.74	32.41
<i>Acanthurus dussumieri</i>	0.86	0.25	0.39	1.3	0.73	33.14
<i>Coris caudimacula</i>	0.58	1.14	0.39	1.4	0.73	33.87

Average dissimilarity = 52.88

Species	RS	SM	Average diissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Chromis nigrura</i>	2.75	0.63	1.15	2.21	2.18	2.18
<i>Pseudanthias squamipinnis</i>	2.45	0.91	0.94	1.44	1.79	3.96
<i>Odonus niger</i>	1.18	1.26	0.71	1.33	1.34	5.3
<i>Thalassoma amblycephalum</i>	1.54	0.45	0.7	1.54	1.33	6.63
<i>Lutjanus bohar</i>	1.37	0.22	0.63	2.12	1.19	7.83
<i>Caesio xanthonota</i>	1.05	0.63	0.61	1.13	1.15	8.98
<i>Dascyllus trimaculatus</i>	0.42	1.21	0.6	1.35	1.14	10.12
<i>Chromis weberi</i>	1.54	2.07	0.6	1.29	1.13	11.25
<i>Plectroglyphidodon dickii</i>	1.32	0.32	0.58	1.64	1.1	12.35
<i>Cirrhilabrus exquisitus</i>	0.35	1.14	0.56	1.23	1.07	13.42
<i>Ptereleotris evides</i>	0.5	1.13	0.55	1.21	1.05	14.46
<i>Parapriacanthus ransonneti</i>	0	1.12	0.55	0.5	1.04	15.5
<i>Epinephelus tukula</i>	1.03	0	0.55	2.11	1.03	16.53
<i>Pomacentrus caeruleus</i>	0.84	1.26	0.53	1.33	1.01	17.54
<i>Acanthurus thompsoni</i>	1.02	0.38	0.51	1.13	0.96	18.5
<i>Chromis dimidiata</i>	3.1	2.63	0.51	1.23	0.96	19.46

<i>Pervagor janthinosoma</i>	0.08	1.02	0.5	2.23	0.95	20.41
<i>Naso brevirostris</i>	0	0.97	0.5	1.13	0.95	21.36
<i>Pomacanthus rhomboides</i>	0.99	0.13	0.5	1.45	0.94	22.31
<i>Naso hexacanthus</i>	0.52	0.83	0.5	1.06	0.94	23.24
<i>Caranx melampygus</i>	1.07	0.26	0.49	1.63	0.93	24.17
<i>Lethrinus rubrioperculatus</i>	0.9	0	0.48	1.28	0.92	25.09
<i>Oplegnathus robinsoni</i>	0.97	0.13	0.48	2.43	0.91	26
<i>Lethrinus crocineus</i>	0.91	0	0.47	1.64	0.9	26.9
<i>Parupeneus macronema</i>	0.85	1.71	0.47	1.29	0.89	27.79
<i>Bodianus bilunulatus</i>	1.22	0.4	0.47	1.54	0.89	28.67
<i>Plectroglyphidodon</i>	1.19	0.42	0.46	1.5	0.87	29.55
<i>Amphiprion allardi</i>	0.4	1.06	0.45	1.37	0.86	30.4
<i>Aprion virescens</i>	0.87	0.1	0.44	1.33	0.84	31.25
<i>Apolemichthys trimaculatus</i>	1.02	0.3	0.44	1.5	0.83	32.08
<i>Hemitaenichthys zoster</i>	0.59	0.58	0.42	1.03	0.8	32.88
<i>Ctenochaetus binotatus</i>	0.31	0.91	0.42	1.53	0.8	33.67

Average dissimilarity = 52.83

Appendix 3

SIMPER results for percent contribution of each species to overall dissimilarity between reefs protection status. Cumulative cut-off to exclude species with low contributions was 33%. Species in bold are considered potentially good discriminating species according to criteria discussed in Clarke and Warwick (2001).

Species	Protected Average abundance	Open Average abundance	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
<i>Pseudanthias squamipinnis</i>	3.16	0.91	1.23	1.89	2.3	2.3
<i>Chromis nigrura</i>	2.29	0.63	0.92	1.81	1.72	4.02
<i>Thalassoma amblycephalum</i>	1.54	0.45	0.69	1.52	1.28	5.3
<i>Odonus niger</i>	1.29	1.26	0.67	1.33	1.25	6.55
<i>Chromis weberi</i>	2.61	2.07	0.62	1.4	1.16	7.72
<i>Parapriacanthus ransonneti</i>	0.17	1.12	0.6	0.56	1.12	8.83
<i>Ptereleotris evides</i>	0.26	1.13	0.57	1.21	1.06	9.89
<i>Nemanthias carberryi</i>	1.11	0	0.55	0.76	1.03	10.92
<i>Caesio xanthonota</i>	0.93	0.63	0.54	1.08	1.01	11.94
<i>Cirrhilabrus exquisitus</i>	0.59	1.14	0.53	1.24	1	12.93
<i>Pomacentrus caeruleus</i>	0.64	1.26	0.51	1.32	0.95	13.88
<i>Naso brevirostris</i>	0	0.97	0.5	1.13	0.93	14.81
<i>Chromis dimidiata</i>	2.63	2.63	0.49	1.36	0.91	15.72
<i>Mulloides vanicolensis</i>	0.93	0	0.46	0.76	0.86	16.58
<i>Myripristis murdjan</i>	0.95	0.38	0.46	1.25	0.86	17.43
<i>Naso hexacanthus</i>	0.15	0.83	0.45	0.82	0.84	18.27
<i>Dascyllus trimaculatus</i>	1.09	1.21	0.44	1.25	0.82	19.09
<i>Pomacanthus rhomboides</i>	0.88	0.13	0.44	1.5	0.82	19.91
<i>Ctenochaetus truncatus</i>	0.79	1.1	0.43	1.28	0.79	20.7
<i>Lutjanus kasmira</i>	0.82	0.1	0.42	0.83	0.79	21.5
<i>Priacanthus hamrur</i>	0.78	0.73	0.42	1.21	0.78	22.28
<i>Parupeneus macronema</i>	0.96	1.71	0.42	1.21	0.78	23.06
<i>Bodianus diana</i>	1.2	0.56	0.41	1.32	0.77	23.83
<i>Paracirrhites arcatus</i>	1.06	0.42	0.41	1.36	0.77	24.6
<i>Ctenochaetus binotatus</i>	0.44	0.91	0.41	1.43	0.76	25.36
<i>Plagiotremus tapeinosoma</i>	0.52	0.91	0.4	1.28	0.75	26.11
<i>Pervagor janthinosoma</i>	0.48	1.02	0.39	1.4	0.73	26.84
<i>Chaetodon blackburnii</i>	0.85	0.3	0.38	1.29	0.71	27.55
<i>Hemitaurichthys zoster</i>	0.75	0.58	0.38	1.17	0.71	28.25
<i>Acanthurus thompsoni</i>	0.69	0.38	0.38	1.03	0.7	28.96
<i>Acanthurus leucosternon</i>	1.3	0.8	0.37	1.19	0.7	29.66
<i>Sphyaena jello</i>	0.15	0.63	0.37	0.57	0.7	30.35
<i>Caranx melampygus</i>	0.71	0.26	0.37	1.03	0.69	31.05
<i>Centropyge acanthops</i>	0.59	0.67	0.37	1.15	0.69	31.74
<i>Forcipiger flavissimus</i>	1.07	0.65	0.37	1.2	0.69	32.43
<i>Bodianus axillaris</i>	0.26	0.85	0.37	1.39	0.69	33.12
<i>Lutjanus bohar</i>	0.75	0.22	0.36	1.2	0.68	33.8

Average dissimilarity = 53.49

Species	Sanctuary Average abundance	Open Average abundance	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
<i>Chromis nigrura</i>	2.7	0.63	1.07	2.2	2.01	2.01
<i>Pseudanthias squamipinnis</i>	2.46	0.91	0.89	1.46	1.67	3.68
<i>Odonus niger</i>	1.21	1.26	0.69	1.35	1.3	4.98
<i>Caesio xanthonota</i>	1.41	0.63	0.69	1.23	1.29	6.27
<i>Lutjanus bohar</i>	1.52	0.22	0.67	2.26	1.26	7.53
<i>Parapriacanthus ransonneti</i>	0.47	1.12	0.66	0.61	1.25	8.77
<i>Pomacanthus rhomboides</i>	1.26	0.13	0.6	1.57	1.13	9.9
<i>Chromis weberi</i>	1.88	2.07	0.57	1.31	1.07	10.97
<i>Ptereleotris evides</i>	0.26	1.13	0.55	1.23	1.03	12
<i>Aprion virescens</i>	1.15	0.1	0.54	1.88	1.02	13.02
<i>Dascyllus trimaculatus</i>	0.68	1.21	0.54	1.37	1.02	14.04
<i>Thalassoma amblycephalum</i>	1.15	0.45	0.53	1.25	1	15.04
<i>Chromis dimidiata</i>	3.12	2.63	0.52	1.23	0.98	16.02
<i>Epinephelus tukula</i>	1.01	0	0.51	2.21	0.96	16.98
<i>Cirrhilabrus exquisitus</i>	0.54	1.14	0.51	1.21	0.96	17.93
<i>Pomacentrus caeruleus</i>	0.62	1.26	0.51	1.37	0.95	18.89
<i>Naso hexacanthus</i>	0.63	0.83	0.5	1	0.95	19.84
<i>Caranx melampygus</i>	1.13	0.26	0.5	1.54	0.94	20.78
<i>Naso brevirostris</i>	0.09	0.97	0.48	1.14	0.9	21.67
<i>Lethrinus crocineus</i>	0.94	0	0.48	1.47	0.9	22.57
<i>Acanthurus thompsoni</i>	1	0.38	0.47	1.17	0.89	23.46
<i>Oplegnathus robinsoni</i>	0.96	0.13	0.46	1.66	0.86	24.32
<i>Lutjanus gibbus</i>	0.91	0.1	0.45	0.97	0.84	25.16
<i>Plectroglyphidodon dickii</i>	0.9	0.32	0.42	1.18	0.8	25.95
<i>Centropyge acanthops</i>	0.89	0.67	0.41	1.17	0.77	26.72
<i>Bodianus bilunulatus</i>	1.08	0.4	0.4	1.42	0.75	27.48
<i>Pervagor janthinosoma</i>	0.33	1.02	0.4	1.55	0.75	28.23
<i>Hemitaurichthys zoster</i>	0.7	0.58	0.4	1.08	0.75	28.98
<i>Sphyaena jello</i>	0.23	0.63	0.39	0.6	0.73	29.7
<i>Parupeneus macronema</i>	1.03	1.71	0.39	1.16	0.73	30.43
<i>Plagiotremus tapeinosoma</i>	0.56	0.91	0.38	1.28	0.72	31.15
<i>Amphiprion allardi</i>	0.71	1.06	0.38	1.22	0.72	31.87
<i>Acanthurus leucosternon</i>	1.11	0.8	0.38	1.25	0.71	32.59
<i>Forcipiger flavissimus</i>	1.07	0.65	0.38	1.21	0.71	33.3

Average dissimilarity = 53.18

Species	Protected Average abundance	Sanctuary Average abundance	Average dissimilarity	Diss/SD	Percent Contribution	Cumulative percent
<i>Odonus niger</i>	1.29	1.21	0.67	1.21	1.36	1.36
<i>Caesio xanthonota</i>	0.93	1.41	0.67	1.18	1.36	2.72
<i>Chromis weberi</i>	2.61	1.88	0.63	1.29	1.29	4.01
<i>Pseudanthias squamipinnis</i>	3.16	2.46	0.56	1.34	1.15	5.16
<i>Thalassoma amblycephalum</i>	1.54	1.15	0.51	1.26	1.05	6.21
<i>Nemanthias carberryi</i>	1.11	0	0.5	0.76	1.02	7.23
<i>Chromis dimidiata</i>	2.63	3.12	0.47	1.4	0.96	8.19
<i>Aprion virescens</i>	0.22	1.15	0.47	1.67	0.95	9.14
<i>Lutjanus kasmira</i>	0.82	0.43	0.47	0.85	0.95	10.1
<i>Dascyllus trimaculatus</i>	1.09	0.68	0.45	1.41	0.92	11.02

<i>Mulloides vanicolensis</i>	0.93	0.18	0.45	0.82	0.91	11.93
<i>Acanthurus thompsoni</i>	0.69	1	0.44	1.21	0.9	12.82
<i>Ctenochaetus truncatus</i>	0.79	1.26	0.43	1.24	0.88	13.7
<i>Lutjanus gibbus</i>	0.54	0.91	0.43	1.04	0.87	14.57
<i>Lethrinus crocineus</i>	0.1	0.94	0.43	1.4	0.87	15.44
<i>Plectroglyphidodon dickii</i>	0.43	0.9	0.42	1.19	0.85	16.29
<i>Lutjanus bohar</i>	0.75	1.52	0.41	1.34	0.85	17.14
<i>Chromis nigrura</i>	2.29	2.7	0.41	1.14	0.84	17.97
<i>Hemitaurichthys zoster</i>	0.75	0.7	0.4	1.16	0.83	18.8
<i>Myripristis murdjan</i>	0.95	0.57	0.4	1.21	0.83	19.63
<i>Caranx melampygus</i>	0.71	1.13	0.4	1.3	0.82	20.45
<i>Pomacanthus rhomboides</i>	0.88	1.26	0.39	1.32	0.79	21.24
<i>Pomacentrus caeruleus</i>	0.64	0.62	0.38	1.05	0.78	22.02
<i>Centropyge acanthops</i>	0.59	0.89	0.37	1.22	0.76	22.78
<i>Amphiprion allardi</i>	1.07	0.71	0.37	1.17	0.76	23.55
<i>Paracaesio sordidus</i>	0.69	0.29	0.37	1	0.76	24.31
<i>Priacanthus hamrur</i>	0.78	0.15	0.37	1	0.75	25.05
<i>Abudefduf natalensis</i>	0.7	0.3	0.36	0.84	0.74	25.8
<i>Epinephelus tukula</i>	0.38	1.01	0.36	1.38	0.74	26.54
<i>Cirrhilabrus exquisitus</i>	0.59	0.54	0.35	1.07	0.71	27.25
<i>Acanthurus tennentii</i>	1.13	1.65	0.34	1.22	0.69	27.94
<i>Naso unicornis</i>	0.35	0.7	0.34	1.04	0.69	28.62
<i>Zebrasoma scopas</i>	0.47	0.77	0.33	1.17	0.68	29.31
<i>Variola louti</i>	0.64	1.32	0.33	1.18	0.68	29.99
<i>Coris caudimacula</i>	0.72	0.76	0.33	1.15	0.68	30.67
<i>Acanthurus dussumieri</i>	0.5	0.78	0.33	1.15	0.68	31.35
<i>Acanthurus leucosternon</i>	1.3	1.11	0.33	1.18	0.67	32.02
<i>Caesio caeruleus</i>	0.45	0.34	0.32	0.55	0.66	32.68
<i>Carangoides fulvoguttatus</i>	0.44	0.36	0.32	0.7	0.66	33.34

Average dissimilarity = 48.91

Appendix 4

Results of SIMPER analysis on the abundance of Fish-index species for Sanctuary, Diving, Diving & Fishing and Open zones.

Species	Diving Average abundance	Diving - Fishing Average abundance	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
<i>Odonus niger</i>	2.01	4.11	3.6	1.18	8.16	8.16
<i>Caranx melampygus</i>	3.34	3.18	3.21	1.14	7.26	15.43
<i>Variola louti</i>	3.09	2.74	3.05	1.14	6.92	22.34
<i>Pomacanthus imperator</i>	4.03	2.55	3.05	1.17	6.91	29.25
<i>Oplegnathus robinsoni</i>	2.92	2.3	2.79	1.08	6.32	35.57
<i>Epinephelus tukula</i>	0.72	2.85	2.75	0.74	6.23	41.8
<i>Lutjanus bohar</i>	1.92	2.66	2.75	1.09	6.23	48.03
<i>Balistoides conspicillum</i>	0.65	2.44	2.26	1.1	5.12	53.15
<i>Bodianus diana</i>	2.56	2.08	2.09	1.28	4.74	57.89
<i>Chaetodon meyeri</i>	2.33	1.39	1.97	1.19	4.47	62.36
<i>Amphiprion allardi</i>	0.22	2.2	1.94	1.58	4.39	66.75
<i>Acanthurus leucosternon</i>	4.07	4.35	1.87	1.26	4.24	70.99
<i>Forcipiger flavissimus</i>	1.49	2.1	1.71	1.28	3.88	74.87
<i>Aprion virescens</i>	1.58	0.61	1.66	0.74	3.76	78.63
<i>Siganus sutor</i>	0.95	1.13	1.49	0.65	3.37	82
<i>Chaetodon madagaskariensis</i>	2.81	2.92	1.41	1.06	3.19	85.19
<i>Zebrasoma desjardini</i>	1.05	0.48	1.37	0.6	3.11	88.3
<i>Labroides dimidiatus</i>	2.34	2.92	1.05	1.07	2.37	90.68

Average dissimilarity = 44.13

Species	Diving Average abundance	Sanctuary Average abundance	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
<i>Epinephelus tukula</i>	0.72	6.41	4.64	1.4	10.32	10.32
<i>Lutjanus bohar</i>	1.92	7.08	4.43	1.71	9.84	20.16
<i>Aprion virescens</i>	1.58	5.82	3.77	1.53	8.37	28.53
<i>Variola louti</i>	3.09	6.04	3.24	1.25	7.19	35.72
<i>Caranx melampygus</i>	3.34	5.52	3.14	1.25	6.97	42.69
<i>Oplegnathus robinsoni</i>	2.92	4.45	2.84	1.21	6.31	49.01
<i>Odonus niger</i>	2.01	2.83	2.65	0.94	5.88	54.89
<i>Pomacanthus imperator</i>	4.03	3.26	2.45	1.06	5.44	60.33
<i>Acanthurus leucosternon</i>	4.07	3.41	1.96	1.07	4.34	64.67
<i>Bodianus diana</i>	2.56	2.81	1.68	1.22	3.74	68.41
<i>Balistoides conspicillum</i>	0.65	1.94	1.65	0.81	3.67	72.08
<i>Chaetodon meyeri</i>	2.33	3.72	1.64	1.2	3.65	75.73
<i>Forcipiger flavissimus</i>	1.49	2.27	1.55	1.25	3.44	79.17
<i>Scarus rubroviolaceus m</i>	7.68	6.41	1.47	0.67	3.26	82.44
<i>Siganus sutor</i>	0.95	0.89	1.16	0.62	2.57	85.01
<i>Chaetodon madagaskariensis</i>	2.81	2.65	1.14	1.05	2.54	87.54
<i>Zebrasoma gemmatum</i>	1.05	0.31	1.09	0.55	2.42	89.96
<i>Chaetodon trifascialis</i>	1.11	0.4	0.96	0.76	2.12	92.09

Average dissimilarity = 45.01

Species	Diving Average abundance	Open Average abundance	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
<i>Odonus niger</i>	2.01	4.24	4.02	1.16	8.39	8.39
<i>Pomacanthus imperator</i>	4.03	1.14	3.73	1.26	7.77	16.16
<i>Caranx melampygus</i>	3.34	1.19	3.47	1.01	7.23	23.39
<i>Variola louti</i>	3.09	2.51	3.46	1.01	7.22	30.61
<i>Oplegnathus robinsoni</i>	2.92	0.69	3.02	1.05	6.29	36.9
<i>Acanthurus leucosternon</i>	4.07	2.39	2.7	1.25	5.63	42.53
<i>Chaetodon meyeri</i>	2.33	2.29	2.38	1.21	4.97	47.49
<i>Lutjanus bohar</i>	1.92	1.36	2.36	0.87	4.92	52.42
<i>Scarus rubroviolaceus m</i>	7.68	6.27	2.17	0.63	4.53	56.95
<i>Siganus sutor</i>	0.95	1.77	2.15	0.75	4.48	61.43
<i>Bodianus diana</i>	2.56	2.28	2.14	1.22	4.46	65.89
<i>Forcipiger flavissimus</i>	1.49	1.75	1.94	1.14	4.04	69.92
<i>Chaetodon trifascialis</i>	1.11	1.56	1.81	0.94	3.77	73.7
<i>Balistoides conspicillum</i>	0.65	1.24	1.68	0.62	3.51	77.21
<i>Aprion virescens</i>	1.58	0	1.64	0.64	3.43	80.64
<i>Chaetodon madagaskariensis</i>	2.81	2.99	1.6	0.97	3.34	83.98
<i>Amphiprion allardi</i>	0.22	1.47	1.38	1.01	2.87	86.85
<i>Zebrasoma gemmatum</i>	1.05	0	1.36	0.48	2.84	89.69
<i>Labroides dimidiatus</i>	2.34	2.63	1.12	1	2.34	92.03

Average dissimilarity = 47.95

Species	Diving & fishing Average abundance	Sanctuary Average abundance	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
<i>Aprion virescens</i>	0.61	5.82	4.16	1.78	9.42	9.42
<i>Epinephelus tukula</i>	2.85	6.41	4.05	1.21	9.18	18.61
<i>Lutjanus bohar</i>	2.66	7.08	3.82	1.64	8.65	27.26
<i>Odonus niger</i>	4.11	2.83	3.13	1.16	7.1	34.36
<i>Variola louti</i>	2.74	6.04	3.07	1.36	6.96	41.32
<i>Caranx melampygus</i>	3.18	5.52	3.04	1.2	6.9	48.22
<i>Oplegnathus robinsoni</i>	2.3	4.45	2.92	1.19	6.63	54.85
<i>Pomacanthus imperator</i>	2.55	3.26	2.4	1.06	5.45	60.3
<i>Balistoides conspicillum</i>	2.44	1.94	2.08	1.13	4.71	65.01
<i>Chaetodon meyeri</i>	1.39	3.72	2.04	1.54	4.61	69.62
<i>Acanthurus leucosternon</i>	4.35	3.41	1.9	1.05	4.31	73.93
<i>Bodianus diana</i>	2.08	2.81	1.76	1.22	3.99	77.92
<i>Amphiprion allardi</i>	2.2	0.79	1.45	1.4	3.28	81.21
<i>Forcipiger flavissimus</i>	2.1	2.27	1.43	1.16	3.25	84.46
<i>Scarus rubroviolaceus</i>	7.72	6.41	1.37	0.66	3.12	87.57
<i>Siganus sutor</i>	1.13	0.89	1.22	0.68	2.77	90.34

Average dissimilarity = 44.11

Species	Diving - Fishing	Open	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Odonus niger</i>	4.11	4.24	3.97	1.15	8.64	8.64
<i>Caranx melampygus</i>	3.18	1.19	3.16	0.99	6.87	15.5
<i>Variola louti</i>	2.74	2.51	3.11	1.08	6.76	22.27
<i>Epinephelus tukula</i>	2.85	0	2.78	0.68	6.05	28.31
<i>Lutjanus bohar</i>	2.66	1.36	2.76	1.05	5.99	34.31
<i>Acanthurus leucosternon</i>	4.35	2.39	2.74	1.42	5.97	40.28
<i>Balistoides conspicillum</i>	2.44	1.24	2.72	1.1	5.91	46.19
<i>Pomacanthus imperator</i>	2.55	1.14	2.71	0.89	5.91	52.1
<i>Oplegnathus robinsoni</i>	2.3	0.69	2.5	0.79	5.45	57.55
<i>Chaetodon meyeri</i>	1.39	2.29	2.25	1.17	4.89	62.44
<i>Bodianus diana</i>	2.08	2.28	2.19	1.19	4.77	67.21
<i>Siganus sutor</i>	1.13	1.77	2.15	0.79	4.68	71.89
<i>Scarus rubroviolaceus</i>	7.72	6.27	2	0.62	4.35	76.25
<i>Forcipiger flavissimus</i>	2.1	1.75	1.94	1.15	4.21	80.46
<i>Amphiprion allardi</i>	2.2	1.47	1.82	1.26	3.96	84.42
<i>Chaetodon madagaskariensis</i>	2.92	2.99	1.41	0.93	3.07	87.49
<i>Chaetodon trifascialis</i>	0	1.56	1.39	0.8	3.01	90.51

Average dissimilarity = 45.96

Species	Sanctuary	Open	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Epinephelus tukula</i>	6.41	0	5.17	1.45	9.85	9.85
<i>Lutjanus bohar</i>	7.08	1.36	5.03	1.96	9.58	19.43
<i>Aprion virescens</i>	5.82	0	4.99	1.81	9.51	28.94
<i>Caranx melampygus</i>	5.52	1.19	4.06	1.47	7.73	36.67
<i>Variola louti</i>	6.04	2.51	3.77	1.31	7.18	43.85
<i>Oplegnathus robinsoni</i>	4.45	0.69	3.51	1.25	6.68	50.53
<i>Odonus niger</i>	2.83	4.24	3.44	1.08	6.55	57.08
<i>Pomacanthus imperator</i>	3.26	1.14	2.59	1.06	4.94	62.02
<i>Acanthurus leucosternon</i>	3.41	2.39	2.13	1.37	4.06	66.08
<i>Balistoides conspicillum</i>	1.94	1.24	2.03	0.82	3.86	69.94
<i>Chaetodon meyeri</i>	3.72	2.29	1.98	1.3	3.77	73.72
<i>Scarus rubroviolaceus</i>	6.41	6.27	1.91	0.69	3.63	77.35
<i>Bodianus diana</i>	2.81	2.28	1.76	1.17	3.35	80.7
<i>Siganus sutor</i>	0.89	1.77	1.75	0.78	3.34	84.03
<i>Forcipiger flavissimus</i>	2.27	1.75	1.71	1.12	3.25	87.29
<i>Chaetodon trifascialis</i>	0.4	1.56	1.3	0.92	2.47	89.76
<i>Amphiprion allardi</i>	0.79	1.47	1.24	1.11	2.36	92.12

Average dissimilarity = 52.49

Appendix 5

Results of SIMPER analysis on the biomass of Fish-index species for Sanctuary, Diving, Diving & Fishing and Open zones.

Species	Diving Average abundance	Diving & fishing Average abundance	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
<i>Odonus niger</i>	2.01	4.11	3.6	1.18	8.16	8.16
<i>Caranx melampygus</i>	3.34	3.18	3.21	1.14	7.26	15.43
<i>Variola louti</i>	3.09	2.74	3.05	1.14	6.92	22.34
<i>Pomacanthus imperator</i>	4.03	2.55	3.05	1.17	6.91	29.25
<i>Oplegnathus robinsoni</i>	2.92	2.3	2.79	1.08	6.32	35.57
<i>Epinephelus tukula</i>	0.72	2.85	2.75	0.74	6.23	41.8
<i>Lutjanus bohar</i>	1.92	2.66	2.75	1.09	6.23	48.03
<i>Balistoides conspicillum</i>	0.65	2.44	2.26	1.1	5.12	53.15
<i>Bodianus diana</i>	2.56	2.08	2.09	1.28	4.74	57.89
<i>Chaetodon meyeri</i>	2.33	1.39	1.97	1.19	4.47	62.36
<i>Amphiprion allardi</i>	0.22	2.2	1.94	1.58	4.39	66.75
<i>Acanthurus leucosternon</i>	4.07	4.35	1.87	1.26	4.24	70.99
<i>Forcipiger flavissimus</i>	1.49	2.1	1.71	1.28	3.88	74.87
<i>Aprion virescens</i>	1.58	0.61	1.66	0.74	3.76	78.63
<i>Siganus sutor</i>	0.95	1.13	1.49	0.65	3.37	82
<i>Chaetodon madagaskariensis</i>	2.81	2.92	1.41	1.06	3.19	85.19
<i>Zebrasoma gemmatum</i>	1.05	0.48	1.37	0.6	3.11	88.3
<i>Labroides dimidiatus</i>	2.34	2.92	1.05	1.07	2.37	90.68

Average dissimilarity = 44.13

Species	Diving Average abundance	Sanctuary Average abundance	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
<i>Epinephelus tukula</i>	0.72	6.41	4.64	1.4	10.32	10.32
<i>Lutjanus bohar</i>	1.92	7.08	4.43	1.71	9.84	20.16
<i>Aprion virescens</i>	1.58	5.82	3.77	1.53	8.37	28.53
<i>Variola louti</i>	3.09	6.04	3.24	1.25	7.19	35.72
<i>Caranx melampygus</i>	3.34	5.52	3.14	1.25	6.97	42.69
<i>Oplegnathus robinsoni</i>	2.92	4.45	2.84	1.21	6.31	49.01
<i>Odonus niger</i>	2.01	2.83	2.65	0.94	5.88	54.89
<i>Pomacanthus imperator</i>	4.03	3.26	2.45	1.06	5.44	60.33
<i>Acanthurus leucosternon</i>	4.07	3.41	1.96	1.07	4.34	64.67
<i>Bodianus diana</i>	2.56	2.81	1.68	1.22	3.74	68.41
<i>Balistoides conspicillum</i>	0.65	1.94	1.65	0.81	3.67	72.08
<i>Chaetodon meyeri</i>	2.33	3.72	1.64	1.2	3.65	75.73
<i>Forcipiger flavissimus</i>	1.49	2.27	1.55	1.25	3.44	79.17
<i>Scarus rubroviolaceus m</i>	7.68	6.41	1.47	0.67	3.26	82.44
<i>Siganus sutor</i>	0.95	0.89	1.16	0.62	2.57	85.01
<i>Chaetodon madagaskariensis</i>	2.81	2.65	1.14	1.05	2.54	87.54
<i>Zebrasoma desjardini</i>	1.05	0.31	1.09	0.55	2.42	89.96
<i>Chaetodon trifascialis</i>	1.11	0.4	0.96	0.76	2.12	92.09

Average dissimilarity = 45.01

Species	Diving & fishing	Sanctuary	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Aprion virescens</i>	0.61	5.82	4.16	1.78	9.42	9.42
<i>Epinephelus tukula</i>	2.85	6.41	4.05	1.21	9.18	18.61
<i>Lutjanus bohar</i>	2.66	7.08	3.82	1.64	8.65	27.26
<i>Odonus niger</i>	4.11	2.83	3.13	1.16	7.1	34.36
<i>Variola louti</i>	2.74	6.04	3.07	1.36	6.96	41.32
<i>Caranx melampygus</i>	3.18	5.52	3.04	1.2	6.9	48.22
<i>Oplegnathus robinsoni</i>	2.3	4.45	2.92	1.19	6.63	54.85
<i>Pomacanthus imperator</i>	2.55	3.26	2.4	1.06	5.45	60.3
<i>Balistoides conspicillum</i>	2.44	1.94	2.08	1.13	4.71	65.01
<i>Chaetodon meyeri</i>	1.39	3.72	2.04	1.54	4.61	69.62
<i>Acanthurus leucosternon</i>	4.35	3.41	1.9	1.05	4.31	73.93
<i>Bodianus diana</i>	2.08	2.81	1.76	1.22	3.99	77.92
<i>Amphiprion allardi</i>	2.2	0.79	1.45	1.4	3.28	81.21
<i>Forcipiger flavissimus</i>	2.1	2.27	1.43	1.16	3.25	84.46
<i>Scarus rubroviolaceus m</i>	7.72	6.41	1.37	0.66	3.12	87.57
<i>Siganus sutor</i>	1.13	0.89	1.22	0.68	2.77	90.34

Average dissimilarity = 44.11

Species	Diving	Open	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Odonus niger</i>	2.01	4.24	4.02	1.16	8.39	8.39
<i>Pomacanthus imperator</i>	4.03	1.14	3.73	1.26	7.77	16.16
<i>Caranx melampygus</i>	3.34	1.19	3.47	1.01	7.23	23.39
<i>Variola louti</i>	3.09	2.51	3.46	1.01	7.22	30.61
<i>Oplegnathus robinsoni</i>	2.92	0.69	3.02	1.05	6.29	36.9
<i>Acanthurus leucosternon</i>	4.07	2.39	2.7	1.25	5.63	42.53
<i>Chaetodon meyeri</i>	2.33	2.29	2.38	1.21	4.97	47.49
<i>Lutjanus bohar</i>	1.92	1.36	2.36	0.87	4.92	52.42
<i>Scarus rubroviolaceus m</i>	7.68	6.27	2.17	0.63	4.53	56.95
<i>Siganus sutor</i>	0.95	1.77	2.15	0.75	4.48	61.43
<i>Bodianus diana</i>	2.56	2.28	2.14	1.22	4.46	65.89
<i>Forcipiger flavissimus</i>	1.49	1.75	1.94	1.14	4.04	69.92
<i>Chaetodon trifascialis</i>	1.11	1.56	1.81	0.94	3.77	73.7
<i>Balistoides conspicillum</i>	0.65	1.24	1.68	0.62	3.51	77.21
<i>Aprion virescens</i>	1.58	0	1.64	0.64	3.43	80.64
<i>Chaetodon madagaskariensis</i>	2.81	2.99	1.6	0.97	3.34	83.98
<i>Amphiprion allardi</i>	0.22	1.47	1.38	1.01	2.87	86.85
<i>Zebrasoma desjardini</i>	1.05	0	1.36	0.48	2.84	89.69
<i>Labroides dimidiatus</i>	2.34	2.63	1.12	1	2.34	92.03

Average dissimilarity = 47.95

Species	Diving & fishing	Open	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Odonus niger</i>	4.11	4.24	3.97	1.15	8.64	8.64
<i>Caranx melampygus</i>	3.18	1.19	3.16	0.99	6.87	15.5
<i>Variola louti</i>	2.74	2.51	3.11	1.08	6.76	22.27
<i>Epinephelus tukula</i>	2.85	0	2.78	0.68	6.05	28.31
<i>Lutjanus bohar</i>	2.66	1.36	2.76	1.05	5.99	34.31
<i>Acanthurus leucosternon</i>	4.35	2.39	2.74	1.42	5.97	40.28
<i>Balistoides conspicillum</i>	2.44	1.24	2.72	1.1	5.91	46.19
<i>Pomacanthus imperator</i>	2.55	1.14	2.71	0.89	5.91	52.1
<i>Oplegnathus robinsoni</i>	2.3	0.69	2.5	0.79	5.45	57.55
<i>Chaetodon meyeri</i>	1.39	2.29	2.25	1.17	4.89	62.44
<i>Bodianus diana</i>	2.08	2.28	2.19	1.19	4.77	67.21
<i>Siganus sutor</i>	1.13	1.77	2.15	0.79	4.68	71.89
<i>Scarus rubroviolaceus</i>	7.72	6.27	2	0.62	4.35	76.25
<i>Forcipiger flavissimus</i>	2.1	1.75	1.94	1.15	4.21	80.46
<i>Amphiprion allardi</i>	2.2	1.47	1.82	1.26	3.96	84.42
<i>Chaetodon madagaskariensis</i>	2.92	2.99	1.41	0.93	3.07	87.49
<i>Chaetodon trifascialis</i>	0	1.56	1.39	0.8	3.01	90.51

Average dissimilarity = 45.96

Species	Sanctuary	Open	Average dissimilarity	Diss/SD	Percent contribution	Cumulative percent
	Average abundance	Average abundance				
<i>Epinephelus tukula</i>	6.41	0	5.17	1.45	9.85	9.85
<i>Lutjanus bohar</i>	7.08	1.36	5.03	1.96	9.58	19.43
<i>Aprion virescens</i>	5.82	0	4.99	1.81	9.51	28.94
<i>Caranx melampygus</i>	5.52	1.19	4.06	1.47	7.73	36.67
<i>Variola louti</i>	6.04	2.51	3.77	1.31	7.18	43.85
<i>Oplegnathus robinsoni</i>	4.45	0.69	3.51	1.25	6.68	50.53
<i>Odonus niger</i>	2.83	4.24	3.44	1.08	6.55	57.08
<i>Pomacanthus imperator</i>	3.26	1.14	2.59	1.06	4.94	62.02
<i>Acanthurus leucosternon</i>	3.41	2.39	2.13	1.37	4.06	66.08
<i>Balistoides conspicillum</i>	1.94	1.24	2.03	0.82	3.86	69.94
<i>Chaetodon meyeri</i>	3.72	2.29	1.98	1.3	3.77	73.72
<i>Scarus rubroviolaceus m</i>	6.41	6.27	1.91	0.69	3.63	77.35
<i>Bodianus diana</i>	2.81	2.28	1.76	1.17	3.35	80.7
<i>Siganus sutor</i>	0.89	1.77	1.75	0.78	3.34	84.03
<i>Forcipiger flavissimus</i>	2.27	1.75	1.71	1.12	3.25	87.29
<i>Chaetodon trifascialis</i>	0.4	1.56	1.3	0.92	2.47	89.76
<i>Amphiprion allardi</i>	0.79	1.47	1.24	1.11	2.36	92.12

Average dissimilarity = 52.49

Appendix 6

An indicator-based manual for assessing fish communities on coral reefs

6.1. Introduction

This manual has been developed as a practical guide for scientists and coral reef managers to assess and document changes in coral reef fish communities over time. A rigorous process preceded the development of this manual and involved numerous layers of research that included comprehensive data collection, manipulation and interpretation. Baseline fish community data were collected and synthesised to generate fish indicator species. Further scrutiny of these fish indicator species led to the development of a Fish-Index that consists of 25 indicator species. This Fish-Index was applied to assess the fish communities on southern African coral reefs subjected to differing human resource use. Significant relationships emerged between key indicator species and human activities, which facilitated the development of a monitoring protocol with the Fish-Index as its central tool.

Although this manual is the product of various strata of intensive research, it has been designed and packaged into a user-friendly guide that is both relevant to and practical for coral reef scientists, managers and conservationists.

This protocol highlights the importance of long-term monitoring, which represents the repeated surveying of organisms or environmental parameters over time (Rogers et al. 1994). Monitoring may imply data collection alone; however, for the purposes of this manual, it refers to data collection, analysis and validation. Coral reef fish are highly variable in abundance and movement, both temporally and spatially. Consequently, to gain an insight into reef processes such as predation, reproduction, recruitment and competition it is necessary to conduct long-term rather than short-term monitoring programmes. Furthermore, long-term assessment provides valuable information on ecosystem function. This manual focuses on coral reef fish communities alone. In order to identify and comprehend coral reef processes from an ecosystem perspective, it is recommended that fish community monitoring is complemented by a benthic monitoring programme such as that initiated by the Oceanographic Research Institute (Schleyer & Celliers 2003). Benthic monitoring conducted in the same area as the fish community monitoring will provide valuable information on the relationships between the mobile and sessile biological communities in the coral reef ecosystem.

Long-term monitoring is essential for adaptive management of coral reefs (Wells & Mangubhai 2004). For coral reefs that are situated in marine protected areas (MPA), long-term monitoring

allows continual updates on the status of coral reef fish communities and facilitates performance evaluation on the effectiveness of their protective management. Most MPAs are implemented for the protection of biodiversity and sustainable resource utilization (Wilkinson et al. 2003, Pomeroy et al. 2004). Thus, monitoring of changes in selected biological components such as coral reef fish communities can provide an insight as to whether an MPA is achieving its goals.

Obtaining reliable estimates of fish assemblages may be limited by financial and logistical constraints. In addition, extensive and ongoing training is often required because field teams are not static. To mitigate the need for highly skilled personal, the use of selected indicators can reduce the number of fish species that field surveyors need to recognise, increase the accuracy of data collection and reveal links between the breakdown in community processes and the causative agents. Underwater visual censuses are the most common techniques employed to assess fish communities (Bohnsack & Bannerot 1986, Samoily 1997) and counting fewer species enables greater accuracy in assessing abundance and estimating fish length. However, reducing the number of species may raise concern as to how representative the selected species are of the fish community (Kulbicki et al. 2007). The Fish-Index used here includes species from each of the main functional trophic groups, and has been demonstrated to be representative of the functional diversity on southern African coral reef fish communities. Nevertheless, a survey that includes as many fish species as possible has many advantages, particularly when monitoring species diversity. In such instances it may be practical to contract experts to carry out the surveys because the recognition of over 300 species of fish may be required. However, such surveys need only be conducted once every 2-3 years.

This manual does not present a stand-alone method for the assessment of the health of a coral reef ecosystem or the effectiveness of MPA management. It is a practical guide that has been scientifically tested and summarised into a management support tool. Its aim is to provide a greater understanding of the human impacts on one biological component of a complex ecosystem and, consequently, aid reef managers in their conservation decisions. Although the fish indicator species were developed for southern African coral reefs, the concepts are applicable to reefs in other regions.

6.2 Objective of monitoring protocol:

To assess the nature of fish communities on coral reefs subjected to different human extractive and non-extractive resource use.

6.3 Logistics

Personnel

To minimise time spent in the field and to maximise the number of surveys carried out per field excursion, the fish survey team should consist of four people; two surveyors (fish counters) and two buddy divers; one of which should also be a skipper. Only two divers will be in the water at one time; a fish counter and a buddy diver. Thus, use of time at sea will be maximised as one dive pair will conduct fish counts while the other pair has a prescribed surface interval.

Monitoring equipment

The fish surveys should be conducted using SCUBA from a semi-inflatable boat, with two outboard motors and the necessary safety equipment. To record the data, pencils and several A4 perspex slates are required. The best practice for attaching the pencil is with a wrist lanyard made from rubber tubing. The pencil is cut in half and inserted into the end of the rubber tube. Thin speargun rubber tubing is suitable because it is flexible and does not perish in salt water. The rubber tube must be fitted loosely around the fish counter's wrist and fastened with a cable tie. Pencils often break and it is important to have a second pencil as a backup on every dive. The slates must be prepared prior to the dive following the data sheet layout in Figure 6.5. The necessary equipment to record fish data is illustrated in Figure 6.1. All divers must wear a dive computer to record dive time, depth and to log the details of each dive. A weighted 10 m rope is needed to delineate the survey site on the substratum.

Training of the dive team

Before the fish survey team can begin monitoring fish communities, training in fish identification and fish-length estimation is necessary. Each fish counter needs to be proficient in identifying the 25 Fish-Index species. Identification photographs of each species are presented in Appendix 6.1.1 Fish identification books such as 'Reef fishes and Corals' (King 1996) and 'Two Oceans' (Branch et al. 1994) may be useful to improve and test fish identification skills. Fish-length estimation training is of great importance because such measurements will be used to investigate size frequency distributions of the Fish-Index species. In addition, length estimates will be used to generate biomass data, which are important both ecologically and from a fisheries perspective.

Consequently, consistency and accuracy in fish-length estimates is of great importance for data analyses.

Training in fish-length estimation should be conducted using fish models that range in size. Models should mimic actual body shapes of various indicator fish species such as surgeonfish, kingfish, butterflyfish and rockcods. Examples of model fish and their sizes are presented in Appendix 6.1.3. Fish-lengths are estimated in 5 cm increments using forklength measurements from the tip of the upper jaw to the end of the caudal rays (Fig 6.2). Training must be conducted initially on land and then underwater. Training underwater is most important because many factors such as visibility and distance from the fish can influence fish-length estimation.



Figure 6.1 Equipment needed to conduct fish surveys; a 10 m weighted rope, two small pencils secured in rubber tubing and several perspex slates, pre-prepared with a list of the indicator species.

Training in fish-length estimation

Creating representative fish models is central to the fish-length estimation training. Sheets of any material that is waterproof such as plastic or marine-ply can be used for cut-out models of the representative fish species. However, the material should not be so thin that it loses its shape underwater and not so thick that it is too cumbersome to manoeuvre. Material that sinks is easiest to handle underwater. However, floating material is recommended because it can be weighted down with dive weights and simulate fish swimming in the water column. Cut-outs should resemble the shapes of the various fish species as closely as possible. The fish shapes in Appendix 6.1.3 may be enlarged and used as templates to create various sizes. Once each cut-out is prepared, the actual size

should be written on the back of the model. Each model should float in the water when attached by string to a weight so that it will float appropriately in the water above the substratum during training sessions.

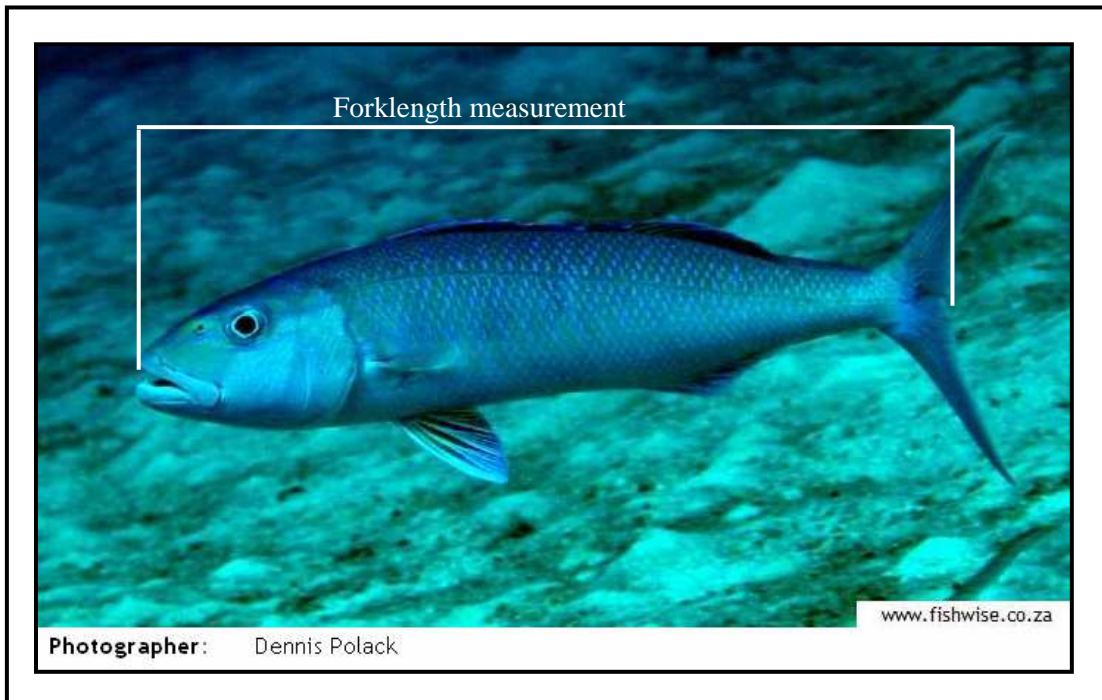


Figure 6.2 Lateral view of a green jobfish illustrating its forklength.

Perspex slates and pencils are needed to record fish lengths during the training sessions. Numerous fish-length estimation training methods include detailed analysis of the training records to monitor the progress of the fish counters (Samoilys 1997, Brown et al. 2004, Mous 2006). However, few field teams have the time or finances to dedicate entire fieldtrips to training or to conduct post-training analysis of the data. Thus, to maximise training in fish-length estimation and to take into account long periods of time that often elapse between field excursions, it is recommended that fish length training should be conducted on the first day of each field trip. In their manual for assessing fish stocks, Samoilys and Carlos (1992) stated that fish-length training involved approximately six trials before fish counters achieved acceptable accuracy. Consequently, it is recommended that each fish counter should commit two full dives to fish-length training, completing three trials per dive.

For the practical training, each fish counter and buddy diver will work as a team. Fish estimates begin on land with the buddy diver randomly selecting 20 different fish models of different shapes and sizes to spread in a circle on the ground. The fish counter then starts by estimating the fish length of a model and systematically estimating the forklength of each consecutive fish model until

all 20 models have been assessed. Each size is written on a slate. Once the lengths of all the fish models have been estimated the fish models are turned over to reveal their actual sizes. An estimate is considered acceptable if it falls within 5 cm of the actual size. The fish counter is given a score out of 20. The practice trials should continue until the fish counter achieves at least 18 correct size estimates per trial. The next stage is to conduct the same trials underwater. This should take place on a sandy area away from the reef to prevent any unnecessary contact with coral. On the boat, the buddy should randomly select 20 fish models that will be weighted and arranged on the sand in a circle around the fish counter. Once the fish counters are confident in fish-length estimation in their trial-runs, the monitoring can begin.

6.4 Sampling design

A well-designed sampling approach will ensure that the data collected are comprehensive and accurate. The sampling design for a long-term monitoring study requires careful consideration because it determines the type of data that are collected and the statistical analyses that can be performed. Sampling design is also determined by the questions that are being asked (See 6.7 for possible questions). The present monitoring protocol aims to assess the long-term trends in fish communities on reefs subjected to different resource use. Thus, it is important to select sites in each different resource use zone and to ensure sufficient replication of the sampling. The use of replicates is required to minimise variability and increase the confidence level of the results so that they reflect actual conditions (Pomeroy et al. 2004). Sampling should be conducted at least once a year, at similar times each day and preferably in the same month or at least the same season. According to tests conducted on South African coral reefs, the minimum number of samples needed to detect variation in fish communities is 10 replicates per reef. However, to increase statistical power it is recommended that at least 18 replicates are conducted per reef (Table 6.1). Due to the constraints of collecting data using SCUBA, the maximum number of replicates possible per dive is six. Thus, a total of 3 dives per reef will yield 18 replicates. Each replicate should be approximately 50 m apart. Figure 6.3 provides an example of a survey layout on a representative South African reef.

Table 6.1 Recommended sampling strategy for the long-term Fish-Index assessment.

Complex	Reef	Minimum number of sites	Number of replicates at each site
Northern	Rabbit Rock Reef	3	6
Central	Nine-mile Reef	3	6
	Seven-mile Reef	3	6
	Two-mile Reef	3	6
Southern	Red Sands Reef	3	6
	Leadsman Shoal	3	6

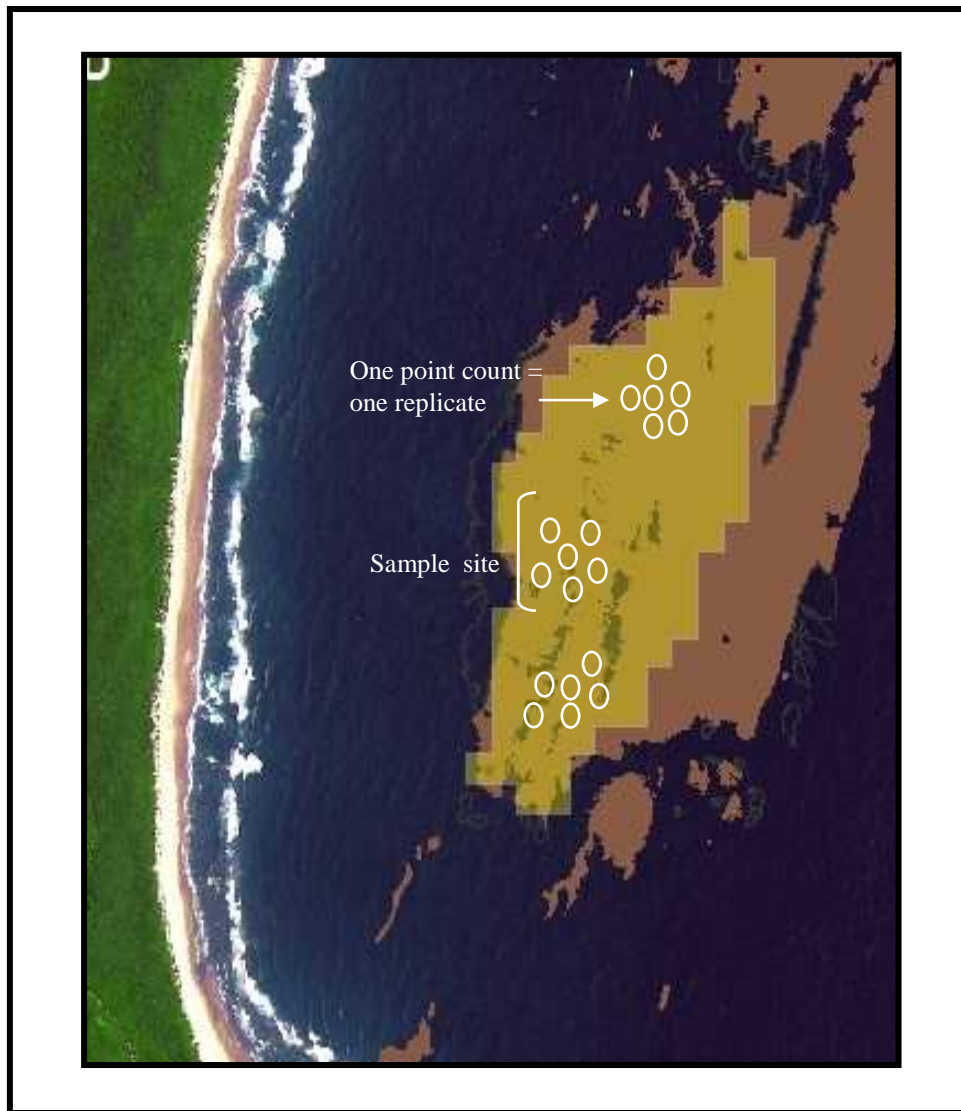


Figure 6.3 Typical sample site and replicate layout on a reef. Six replicate point counts are conducted at each sampling site.

6.5 Site selection

Due to the long-term nature of this monitoring protocol, site selection is of great importance and it is suggested that permanent monitoring sites are established on each reef. It is necessary to conduct the fish surveys in areas of similar topography, depth and benthic community composition to eliminate confounding variables. It is important that reconnaissance surveys are conducted to confirm whether the habitat is suitable for comparisons within and between the reefs. Once a suitable site has been located, the exact position should be recorded using a GPS and a fixed marker should be located on the reef for future surveys; a stainless steel rod may be suitable for this. The sites on a particular reef must be separated by a minimum of 100 m. A separate fieldtrip to establish the fixed sites may be necessary because of the difficulties in fixing permanent markers on the reef.

6.6 Methods

The Fish-index species

The 25 Fish-Index species consist of Indo-Pacific coral reef fish species and one southern African endemic species (Table 6.2). The species were chosen according to numerous criteria. The most important was a rigorous scientific process that identified core indicator species that manifested correlations with human activities. Following that, species were chosen according to their key functional role in trophic groups such as large herbivores or predators on the reefs. Ease of identification was a criterion for all potential indicator species. From an ecosystem perspective it was critical to have a Fish-Index that was representative of the fish community structure on southern African coral reefs.

Table 6.2 Common and scientific names, as well as family and trophic level of the 25 Fish-index species. Fish are ordered alphabetically according to their common names.

Common name	Species	Family	Trophic level
Bluefin kingfish	<i>Caranx melampygus</i>	Carangidae	Medium-level predator
Bohar snapper	<i>Lutjanus bohar</i>	Lutjanidae	Medium-level predator
Clown triggerfish	<i>Balistoides conspicillum</i>	Balistidae	Invertivore
Cleaner wrasse	<i>Labroides dimidiatus</i>	Labridae	Invertivore
Diana's hogfish	<i>Bodianus diana</i>	Labridae	Invertivore
Ember parrotfish	<i>Scarus rubroviolaceus</i>	Scaridae	Herbivore
Emperor angelfish	<i>Pomacanthus imperator</i>	Pomacanthidae	Benthivore
Goldbar wrasse	<i>Thalassoma hebraicum</i>	Labridae	Invertivore
Green jobfish	<i>Aprion virescens</i>	Lutjanidae	Medium-level predator
Natal knifejaw	<i>Oplegnathus robinsoni</i>	Oplegnathidae	Benthivore
Longnose butterflyfish	<i>Forcipiger flavissimus</i>	Chaetodontidae	Invertivore
Lyretail rockcod	<i>Variola louti</i>	Serranidae	Medium-level predator
Maypole butterflyfish	<i>Chaetodon meyeri</i>	Chaetodontidae	Corallivore
Pearly butterfly	<i>Chaetodon madagaskariensis</i>	Chaetodontidae	Omnivore
Powderblue surgeon	<i>Acanthurus leucosternon</i>	Acanthuridae	Herbivore
Potato bass	<i>Epinephelus tukula</i>	Serranidae	Top-level predator
Purple butterflyfish	<i>Chaetodon trifasciatus</i>	Chaetodontidae	Corallivore
Redfang trigger	<i>Odonus niger</i>	Balistidae	Planktivore
Regal angelfish	<i>Pygoplites diacanthus</i>	Pomacanthidae	Benthivore
Rightangle butterflyfish	<i>Chaetodon trifascialis</i>	Chaetodontidae	Corallivore
Sailfin tang	<i>Zebrasoma desjardini</i>	Acanthuridae	Herbivore
Two-bar clownfish	<i>Amphiprion allardi</i>	Pomacentridae	Omnivore
Widebar damselfish	<i>Plectroglyphidodon johnstonianus</i>	Pomacentridae	Corallivore
Whitespotted rabbitfish	<i>Siganus sutor</i>	Siganidae	Herbivore
Zebra	<i>Diplodus cervinus hottentotus</i>	Sparidae	Invertivore

The fish survey technique

The point count method adapted from Samoilys and Carlos (2000) is recommended as the most appropriate technique to survey fish communities because it is quick, requires simple equipment and is suitable for variable topography. The point count technique consists of a five minute timed count within a restricted point count area of 78 m² (the area of a circle with a diameter of 10 m). Each indicator fish species entering the area is counted and the forklength is estimated. It is suggested that common names are used for the fish counts as they are easier to remember. One point count represents one replicate on a reef and there are thus six point counts per site.

Recording fish data

Two divers enter the water together to record the fish counts; a fish counter and a buddy diver. The purpose of the buddy diver is to hold the buoy line and for safety. The diver pair descends onto the site as quickly as possible to reduce disturbance of the fish. The fish counter lays the 10 m rope along the substratum (Fig 6.4). The point count area is an imaginary circle that encompasses the 10 m rope as its diameter. Once the rope is laid onto the substratum, the fish counter takes up position in the middle of the point count circle and waits for 2-3 minutes before beginning the count. During this time, habitat characteristics (coral cover, topography and depth) are recorded. During the point count, the fish counter spins slowly within the circle, trying to remain in its centre but at all times avoiding contact with the substratum. All the selected indicator species that enter the designated area are counted but avoiding enumeration of the same fish twice. The duration of each point count is five minutes, regardless of whether fish are still being counted or whether no new fish have entered the designated area. When large schools are present, the number of fish may be estimated in multiple of 10s. Fish on or near the substratum as well as in the water column must be included in the count. Depth of the substratum is measured using a dive computer.

Figure 6.5 provides an example of how the perspex slates should be prepared prior to each dive. The fish indicator species are written on the left hand side and prompts on the physical parameters such as topography, depth and coral cover as well as date, site and reef are listed at the top of each replicate point count. Due to limited space, only three point counts can be completed per side on an A4 slate. The other side must be prepared in the identical manner.

A



B

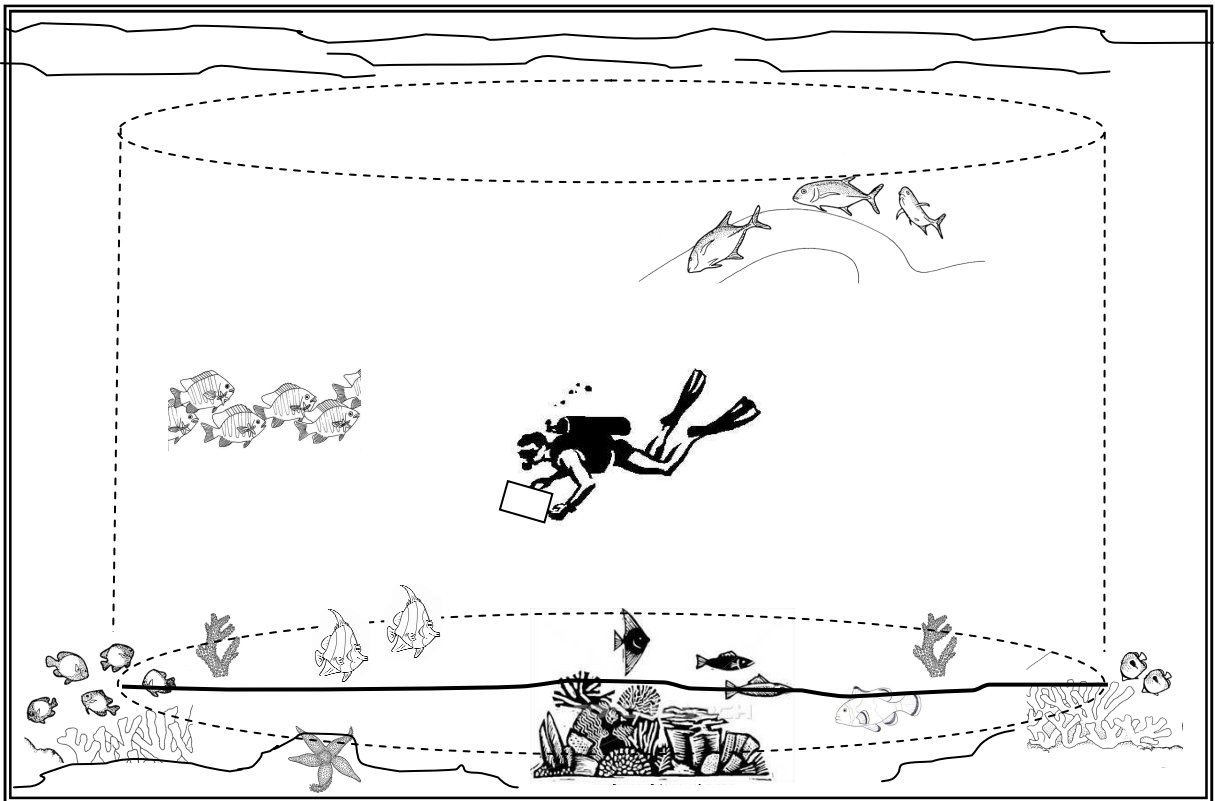


Figure 6.4 The point count fish census method illustrating: A – an in situ example of a point count on a representative South African reef, and B - theoretical positioning of fish counter and layout of the point count.

Date	16/5/2010	Coral cover	L	Coral cover	
Reef	TMR	Depth	12.5	Depth	
Site	coral gardens	Topography	M	Topography	
Bluefin kingfish		1x50			
Bohar snapper					
Cleaner wrasse					
Clown triggerfish					
Diana's hogfish		1x50			
Ember parrotfish					
Emperor angelfish					
Goldbar wrasse					
Green jobfish					
Maypole butterflyfish					
Longnose butterflyfish					
Lyretail rockcod					
Natal knifejaw					
Pearly butterfly		2x15			
Powder blue					
Potato bass		1x45			
Purple butterflyfish					
Redfang trigger		1x35			
Regal angelfish					
Rightangle butterflyfish					
Sailfin tang					
Twobar clownfish					
Widebar damselfish		1x10			
Whitespotted rabbitfish					
Zebra					

Figure 6.5 Recommended layout of the point count perspex slate and data sheet, including typical data. Common names are used for the fish and are ordered alphabetically.

Estimation of coral cover

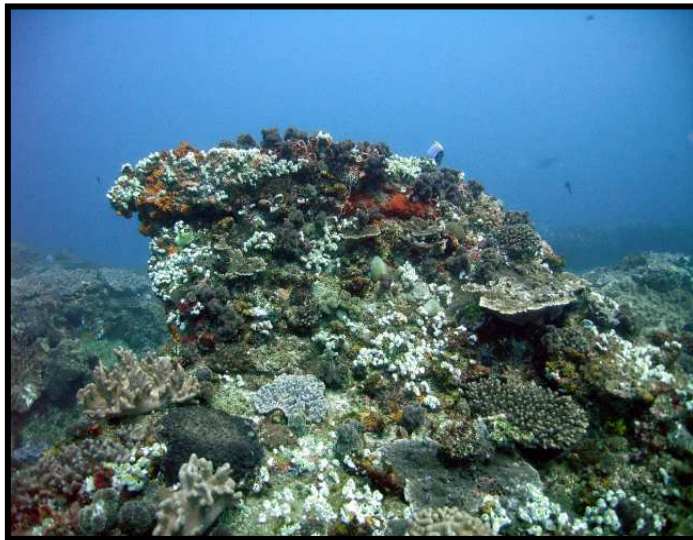
The amount of coral cover is estimated using a rapid assessment technique adapted from English et al. (1997). Due to the time constraints of conducting point counts with SCUBA, coral cover is estimated using a simplified scale of three categories; low, medium and high. Coral cover includes soft and hard coral species. Low coral cover has less than 30% of the point count area covered in coral. Medium coral cover has between 30-50% coral covering the point count area. High coral cover has more than 50% of the point count area covered by coral. Examples of each category are illustrated in Figure 6.6. It is recommended that two replicates are conducted in each coral cover category.

Estimation of topography

The reef topography is estimated using a rapid assessment technique into three categories; low, medium and high. High topography includes pinnacles, spurs, grooves, gullies and over hangs. These features should be elevated more than 1.5 m. Medium topography includes spurs, grooves and gullies. These features should be elevated than 1.5 m. Low topography may consist of very low spurs and shallow grooves, but not gullies or pinnacles; the reef is generally flat. Examples of each category are illustrated in Figure 6.7. Again, it is recommended that two replicates are conducted in each topography category on each reef.



A. High coral cover



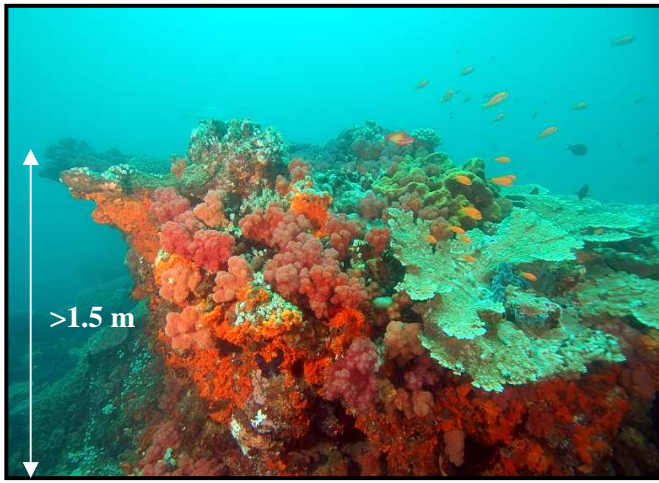
B. Medium coral cover



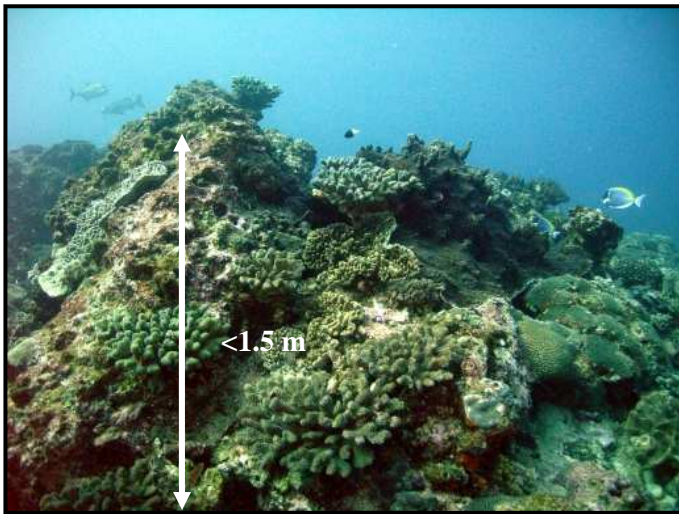
C. Low coral cover



Figure 6.6. Representative examples of the three different coral cover categories: A – high coral cover, B – medium coral cover, and C – low coral cover. Photos are taken from representative southern African coral reefs.



A. High topography.



A. Medium topography



C. Low topography

Figure 6.7 Representative examples illustrating the three topography categories: A – high topography, B – medium topography, and C – low topography. Photos are taken from representative southern African coral reefs.

6.7 Data manipulation and evaluation

Data capture

At the end of each dive, data are entered on data sheets that are identical in layout to the perspex slates. It is efficient to have a data reader and a scribe complete the transcription from slate to paper. Slates can be cleaned using an eraser or by scrubbing with a scouring sponge. However, slates must not be cleaned until each data sheet has been checked twice, by the fish counter and by the buddy diver. Once a hard copy of the data is on file, it is necessary to transfer the data onto a computer spreadsheet for analysis. Excel is a user-friendly spreadsheet and it has a range of basic statistical analyses that are suitable to investigate and describe the fish count data. The spreadsheet must be prepared in a similar fashion to the example in Figure 6.8 in which point counts from three different reefs are illustrated with representative data. Each row represents one fish species, so, if 20 bluefin kingfish are observed in the point count area, each fish must be entered in a separate row. In addition, each point count should be allocated an identification (ID) number. This can be recorded as the date of the count plus a letter to separate between the different point count replicates completed in a dive. Categories 'a' and 'b' are the length-weight parameters required to generate the biomass for each fish species. The relationship between total length (L) and total weight (W) for nearly all species of fish is expressed by the equation:

$$W = aL^b$$

The length-weight parameters for each Fish-index species are presented in Appendix 6.1.1.

In addition to the data sheets, it is recommended that a detailed log of each dive is recorded. Supplementary information, such as unusual species sightings, water temperature, visibility and surge may be recorded for use at a later stage.

Data extraction and analysis

Excel has an efficient function called 'pivot table' that creates a table according to the extraction requirement and allows rapid summation of data. Pivot tables are located in the pull-down menu under the data tab on the top toolbar. The process of creating a pivot table using the pivot table wizard is provided in Figure 6.9. Once data are extracted, they can be manipulated accordingly. Chapter 4 presents a detailed explanation of the statistical analyses that may be applied to the Fish-Index data. Figure 6.10 provides examples of univariate statistical analyses that may be used to explore trends in different abundance and biomass parameters. Once the data have been described and trends become evident, multivariate software such as PRIMER E may be used to conduct more complex manipulations. Sigma Plot is also recommended as a user-friendly statistical software package that offers more sophisticated analyses than Excel.

Microsoft Excel - Book1

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K5 Chaetodontidae

	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q
1	ID	Date	Complex	Reef	Site	Depth	Topog	Y coordina	X coordina	Species	Family	Tropic leve	size	coral cove	a	b	biomass
2	310707a	31/07/200	Central	2MR	valpre	11.6	medium	27 31.193	32 41.234	Thalassoma f	Labridae	invertivore	20	High	0.0271	3	216.8
3	310707a	31/07/200	Central	2MR	valpre	11.6	medium	27 31.193	32 41.234	Thalassoma f	Labridae	invertivore	15	High	0.0271	3	91.4625
4	310707a	31/07/200	Central	2MR	valpre	11.6	medium	27 31.193	32 41.234	Acanthurus le	Acanthuridae	herbivore	20	High	0.0286	2.921	180.5821
5	310707a	31/07/200	Central	2MR	valpre	11.6	medium	27 31.193	32 41.234	Chaetodon tri	Chaetodontidae	corallivore	15	High	0.0258	2.969	80.06352
6	310707a	31/07/200	Central	2MR	valpre	11.6	medium	27 31.193	32 41.234	Chaetodon tri	Chaetodontidae	corallivore	15	High	0.0258	2.969	80.06352
7	310707b	31/07/200	Central	2MR	valpre	11.5	medium	27 31.193	32 41.234	Labroides dirr	Labridae	invertivore	10	High	0.0059	3.17	8.726739
8	310707b	31/07/200	Central	2MR	valpre	11.5	medium	27 31.193	32 41.234	Caranx melan	Carangidae	predator	30	High	0.0237	2.941	523.5546
9	310707b	31/07/200	Central	2MR	valpre	11.5	medium	27 31.193	32 41.234	Caranx melan	Carangidae	predator	30	High	0.0237	2.941	523.5546
10	180907a	18/09/200	Central	7MR	Middle	17	medium	27 27.155	32 42.726	Labroides dirr	Labridae	invertivore	5	low	0.0059	3.17	0.969587
11	180907a	18/09/200	Central	7MR	Middle	17	medium	27 27.155	32 42.726	Labroides dirr	Labridae	invertivore	5	low	0.0059	3.17	0.969587
12	180907a	18/09/200	Central	7MR	Middle	17	medium	27 27.155	32 42.726	Scarus rubrov	Scaridae	herbivore	50	low	0.0136	3.109	2603.974
13	180907a	18/09/200	Central	7MR	Middle	17	medium	27 27.155	32 42.726	Thalassoma f	Labridae	invertivore	15	low	0.0271	3	91.4625
14	180907a	18/09/200	Central	7MR	Middle	17	medium	27 27.155	32 42.726	Thalassoma f	Labridae	invertivore	15	low	0.0271	3	91.4625
15	180907a	18/09/200	Central	7MR	Middle	17	medium	27 27.155	32 42.726	Chaetodon m	Chaetodontidae	omnivore	10	low	0.0311	2.976	29.42798
16	180907a	18/09/200	Central	7MR	Middle	17	medium	27 27.155	32 42.726	Chaetodon m	Chaetodontidae	omnivore	15	low	0.0311	2.976	98.35761
17	180907a	18/09/200	Central	7MR	Middle	17	medium	27 27.155	32 42.726	Odonus niger	Balistoide	invertivore	20	low	0.0242	3	193.6
18	180907b	18/09/200	Central	7MR	Middle	17	medium	27 27.155	32 42.726	Odonus niger	Balistoide	invertivore	20	low	0.0242	3	193.6
19	180907b	18/09/200	Central	7MR	Middle	17	medium	27 27.155	32 42.726	Odonus niger	Balistoide	invertivore	20	low	0.0242	3	193.6
20	180907b	18/09/200	Central	7MR	Middle	17	medium	27 27.155	32 42.726	Odonus niger	Balistoide	invertivore	20	low	0.0242	3	193.6
21	180907b	18/09/200	Central	7MR	Middle	16.6	low	27 27.155	32 42.726	Bodianus diar	Labridae	invertivore	15	low	0.0201	2.992	66.38365
22	180907b	18/09/200	Central	7MR	Middle	16.6	low	27 27.155	32 42.726	Thalassoma f	Labridae	invertivore	10	low	0.0271	3	27.1
23	201107a	20/11/200	Southern	Redsar	Outer	12	high	27 45.690	32 37.910	Labroides dirr	Labridae	invertivore	10	high	0.0059	3.17	8.726739
24	201107a	20/11/200	Southern	Redsar	Outer	12	high	27 45.690	32 37.910	Thalassoma f	Labridae	invertivore	10	high	0.0271	3	27.1
25	201107a	20/11/200	Southern	Redsar	Outer	12	high	27 45.690	32 37.910	Thalassoma f	Labridae	invertivore	10	high	0.0271	3	27.1
26	201107a	20/11/200	Southern	Redsar	Outer	12	high	27 45.690	32 37.910	Thalassoma f	Labridae	invertivore	10	high	0.0271	3	27.1
27	201107a	20/11/200	Southern	Redsar	Outer	12	high	27 45.690	32 37.910	Thalassoma f	Labridae	invertivore	10	high	0.0271	3	27.1
28	201107a	20/11/200	Southern	Redsar	Outer	12	high	27 45.690	32 37.910	Thalassoma f	Labridae	invertivore	10	high	0.0271	3	27.1
29	201107a	20/11/200	Southern	Redsar	Outer	12	high	27 45.690	32 37.910	Thalassoma f	Labridae	invertivore	10	high	0.0271	3	27.1
30	201107a	20/11/200	Southern	Redsar	Outer	12	high	27 45.690	32 37.910	Thalassoma f	Labridae	invertivore	10	high	0.0271	3	27.1
31	201107a	20/11/200	Southern	Redsar	Outer	12	high	27 45.690	32 37.910	Thalassoma f	Labridae	invertivore	10	high	0.0271	3	27.1
32	201107a	20/11/200	Southern	Redsar	Outer	12	high	27 45.690	32 37.910	Thalassoma f	Labridae	invertivore	15	high	0.0271	3	91.4625
33	201107a	20/11/200	Southern	Redsar	Outer	12	high	27 45.690	32 37.910	Thalassoma f	Labridae	invertivore	15	high	0.0271	3	91.4625
34																	
35																	

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Figure 6.8. Excel spreadsheet template with representative data illustrating the layout needed to capture the Fish-Index point count data.

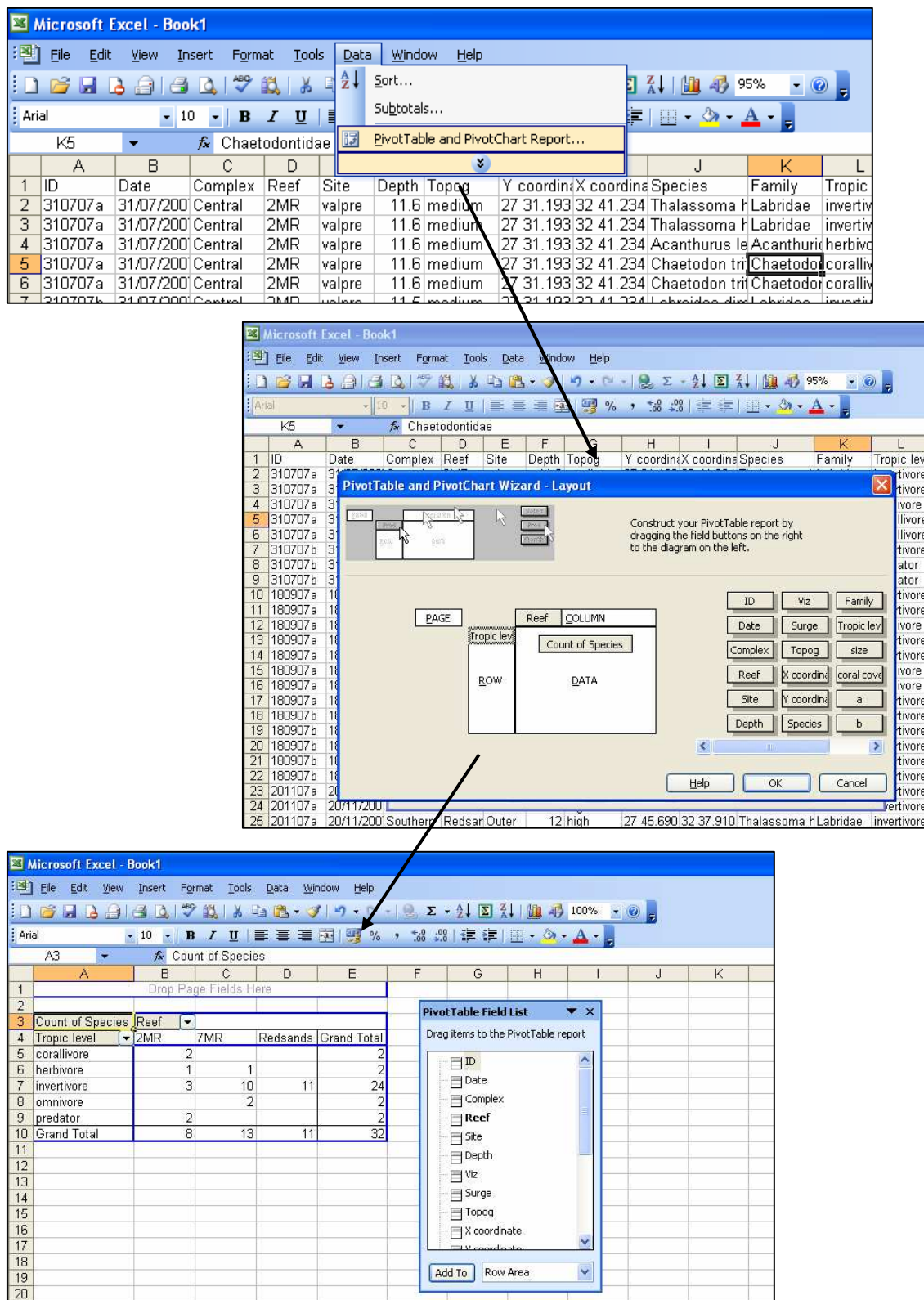


Figure 6.9. Creating a pivot table using the pivot table wizard in Excel.

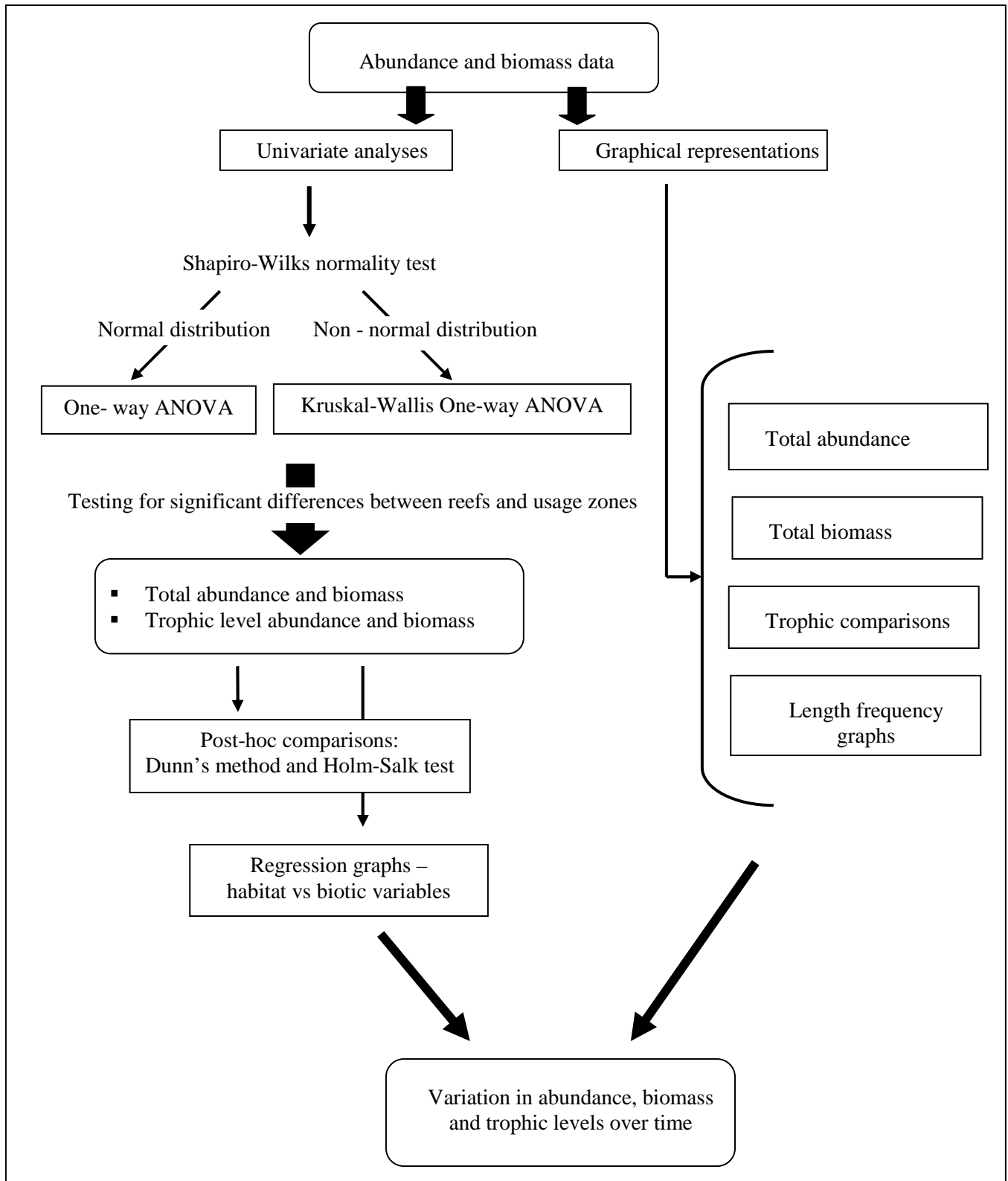


Figure 6.10 Flow chart illustrating the univariate analyses and graphs used to explore trends and compare the Fish-index results.








Possible questions to assess fish community status

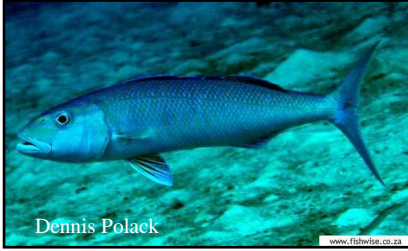
1. What trends are evident in total abundance and biomass between the different usage zones?
See Chapter 4 Figure 4.5.
2. Are the functional (trophic) groups present in equal abundance and biomass between the different usage zones? See Chapter 4 Figure 4.7.
3. Are there significant differences between individual key species such as target species or top-level (apex) predators? See Chapter 4 Figure 4.8.

Additional resources and references

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Appendix 6.1.1 Identification photographs of the Fish-index species. Notes on the significance of each species have been included.

 <p>Bluefin kingfish <i>Caranx melampygus</i></p> <p>Target species: low numbers may indicate overfishing</p>	 <p>Bohar snapper <i>Lutjanus bodar</i></p> <p>Key reef predator. Low numbers may high fishing intensity or disturbance in reef functioning</p>	 <p>Cleaner wrasse <i>Labroides dimidiatus</i></p>
 <p>Clown triggerfish <i>Balistoides conspicillum</i></p> <p>Low abundances may be indicative of high diving intensity</p>	 <p>Diana's hogfish <i>Bodianus Diana</i></p> <p>Prey species. Can be used to analyse predator-prey relationships</p>	 <p>Ember parrotfish - Female <i>Scarus rubroviolaceus</i></p> <p>Key herbivore</p>
 <p>Ember parrotfish – Male <i>Scarus rubroviolaceus</i></p> <p>Key herbivore.</p>	 <p>Emperor angelfish <i>Pomacanthus imperator</i></p> <p>Specialist feeder of sponges and tunicates.</p>	 <p>Goldbar wrasse <i>Thalassoma hebraicum</i></p> <p>Prey species. May be used to assess predator-prey ratios.</p>



Green jobfish
Aprion virescens

Key reef predator. Target species. Sensitive to overfishing.



Longnose butterflyfish
Forcipiger flavissimus

Prey species.



Lyretail rockcod
Variola louti

Key reef predator. Important for trophic comparisons



Maypole butterflyfish
Chaetodon meyeri

Specialist feeder – Corallivore. High abundances may indicate high coral cover.



Natal knifejaw
Oglegnathus robinsoni

Benthivore. Important for trophic comparisons



Potato bass
Epinephelus tukula

Top reef predator. May be sensitive to high diving intensity



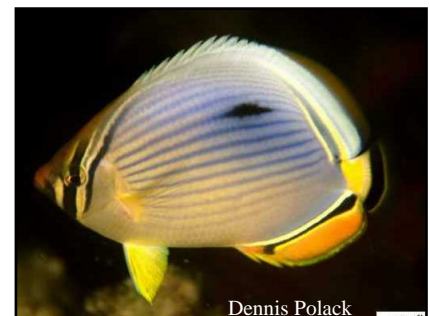
Powderblue surgeonfish
Acanthurus leucosternon

Key herbivore species. Important for trophic comparisons



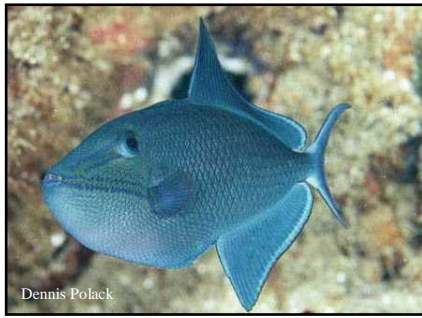
Pearly butterflyfish
Chaetodon madagaskariensis

Prey species. May be used to assess predator-prey ratios



Purple butterflyfish
Chaetodon trifasciatus

Specialist feeder – Corallivore. High abundances may indicate high coral cover.



Redfang trigger
Odonus niger

Key planktivore species.
Important for trophic
comparisons



Regal angelfish
Pygoplites diacanthus

Specialist feeder – sponges
Uncommon species. May be
sensitive to high diving
intensity



Rightangle butterflyfish
Chaetodon trifascialis

Specialist feeder –
Corallivore. High abundances
may indicate high coral cover.



Sailfin tang
Zebrasoma desjardini

Herbivorous species



Two-bar clownfish
Amphiprion allardi

Low numbers may indicate
removal by hobbyists.



Widebar damselfish
Plectroglyphidodon johnstonianus

Specialist feeder – Corallivore.
High abundances may indicate the
presence of high branching coral
cover.



Whitespotted rabbitfish
Siganus sutor

Key herbivore species



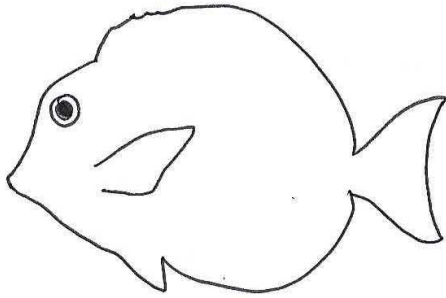
Zebra
Diplodus cervinus hottentotus

Endemic species

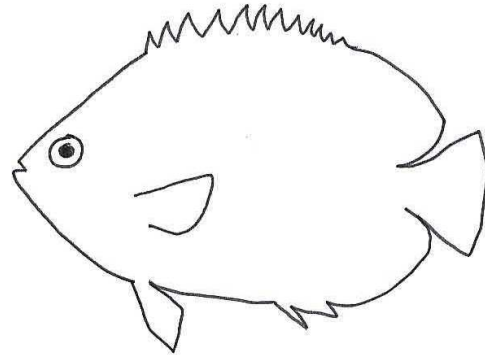
Appendix 6.1.2 Length-weight parameters, maximum sizes (forklength in cm) and trophic level (TL) of the 25 Fish-Index species extracted from www.fishbase.com.

common names	a	b	maximum size	Trophic level
Bluefin kingfish	0.0237	2.941	100	4.28
Bohar snapper	0.0156	3.0587	90	4.11
Cleaner wrasse	0.0059	3.17	10	3.49
Clown triggerfish	0.0244	3.018	50	3.31
Diana's hogfish	0.0201	2.992	25	3.5
Ember parrotfish	0.0136	3.109	60	2
Emperor angelfish	0.0276	3	40	2.7
Goldbar wrasse	0.0271	3	25	3.5
Green jobfish	0.0294	2.76	110	3.98
Longnose butterflyfish	0.0167	3	20	3.38
Lyretail rockcod	0.0122	3.079	80	4
Maypole butterflyfish	0.0296	2.895	20	3.34
Natal knifejaw	0.0232	3	60	3.17
Pearly butterflyfish	0.0311	2.976	15	2.76
Potato bass	0.106	2.56	200	4.4
Powderblue surgeonfish	0.0286	2.921	25	2
Purple butterflyfish	0.0311	2.976	15	3.34
Redfang triggerfish	0.0242	3	50	3.22
Regal angelfish	0.0163	3	25	2.69
Rightangle butterflyfish	0.0258	2.969	18	3.34
Sailfin tang	0.0285	2.992	22	2
Twobar clownfish	0.0239	2.982	15	2.69
Whitespotted rabbitfish	0.0597	2.754	45	2
Widebar damselfish	0.0612	2.635	10	3.31
Zebra	0.0116	3.14	50	3.64

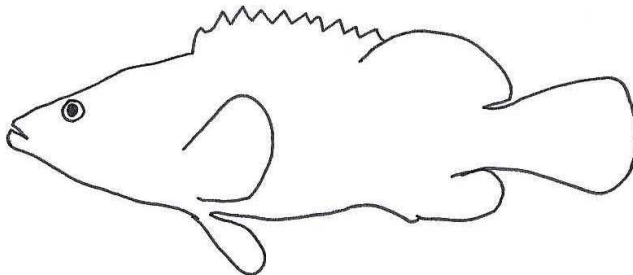
Appendix 6.1.3 Examples body shapes of selected fish indicator species that can be enlarged to different sizes and in different lengths to create fish models for fish-length estimate training.



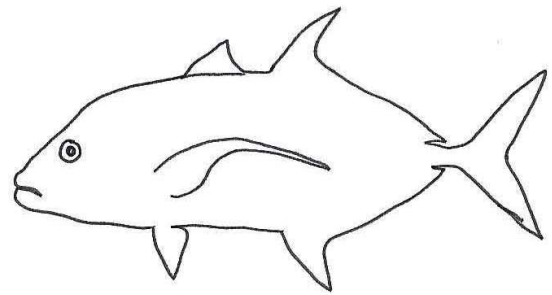
Powder blue surgeonfish
10 cm, 15 cm, 25 cm



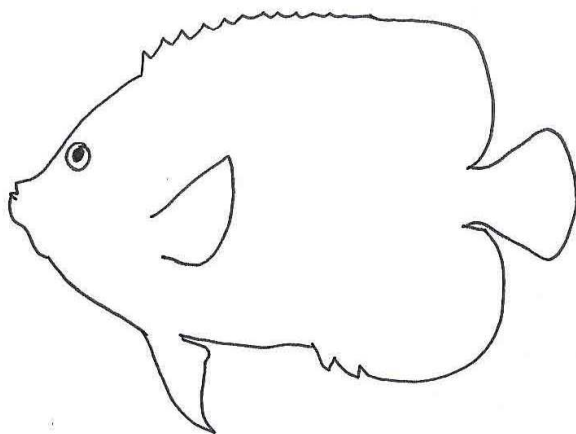
Purple butterflyfish
5 cm, 10 cm, 15 cm



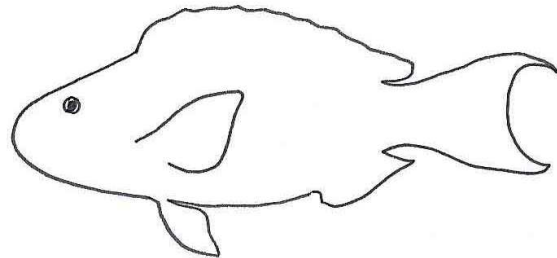
Potato bass
90 cm, 110 cm, 140 cm, 150 cm, 170 cm



Bluefin kingfish
35 cm, 40 cm, 55 cm, 60 cm, 70 cm



Emperor angelfish
30 cm, 35 cm, 45 cm, 50 cm



Ember parrotfish
20 cm, 35 cm, 40 cm, 55 cm, 60 cm