FISH SURVEYS IN EXPLOITED AND PROTECTED AREAS OF THE PONDOLAND MARINE PROTECTED AREA WITH CONSIDERATION OF THE IMPACT OF THE MPA ON COASTAL FISHERIES

by

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ABSTRACT

The line-fishery in KwaZulu-Natal, on the east coast of South Africa, experienced a rapid decline during the 20th century. The stocks of some important fishery species collapsed due to over-fishing, which led to the Minister of Environmental Affairs and Tourism declaring a state of emergency in the line-fishery in 2000. More restrictive regulations were implemented and the importance of marine protected areas became recognised.

In June 2004, 80 km of the Pondoland coast was declared a multiple-use marine protected area. This coastline is a zone of transition between the Natal and Agulhas biogeographic regions and had been identified as a critical gap in South Africa's marine protected area network. Part of the protected area is a 40x10 km no-take area, which is closed to offshore exploitation. The primary objectives of the closure, as gazetted, are the conservation of biodiversity and the rebuilding of commercially important line-fish stocks, which have been depleted by over-fishing. Not mentioned in the legislation, but of significant importance is the benefit of the Pondoland Marine Protected Area to nearby fisheries through the export of fish biomass.

Between 2006 and 2010, experiments were conducted to evaluate the outcome of the fishery closure. The state of previously exploited line-fish species was assessed by means of a controlled fishing experiment where the abundance and biomass of key line-fish in the Pondoland no-take area were compared to that in the adjacent fished area. Some of South Africa's most important line-fish species, which have been over-fished in the past, were significantly more abundant and larger in the no-take area. Slinger *Chrysoblephus puniceus* is a small sparid, which is heavily fished in KwaZulu-Natal where it accounts for 60% of the commercial line-fish catch. During this study and in the absence of fishing in the no-take area, this species increased in number and body size. Scotsman *Polysteganus praeorbitalis* is a larger piscivorous sparid, highly esteemed by the ski-boat fishery in KZN. The stock status of this species was reported to have collapsed due to over-fishing, but in the no-take area, scotsman was significantly more abundant and of a larger mean size. An increase in the mean size of this species was also recorded during the four years of study.

With an increase in the number and size of previously depleted carnivorous line-fish in the no-take area, it is anticipated that the wider fish community may be indirectly affected through competition or predation. An underwater visual census was used to obtain a wider perspective of the fish community than was possible with line-fishing which limits sampling to line-fish species. The fish community in the Pondoland no-take area was notably different from that in the adjacent exploited area, but it was difficult to attribute this conclusively to protection from fishing. Most important was the dominance of large carnivorous species in the no-take area, in direct contrast to the abundance of small planktivores in the exploited area. This suggested a healthy predator-dominated community following protection from fishing in the no-take area.

The potential benefit of the Pondoland Marine Protected Area on nearby fisheries was investigated by means of a tag-recapture experiment, which assessed fish movement within and between the no-take area and the adjacent exploited area. Tag-recapture observations indicated a great degree of site-fidelity (station-keeping) with an element of nomadic behaviour (ranging). Ninety percent of recaptured fishes were within 250 m of the original release-site, suggesting vulnerability to over-fishing. This study also presents some of the first known estimates of home-range size in certain species. Besides station-keeping behaviour, some fishes undertook ranging movements, between 150 and 1059 km from their release-site, making them available to adjacent fisheries. This distinction in movement behaviour was noted in the sparids, scotsman and slinger, as well as in the serranids, yellowbelly rockcod *Epinephelus marginatus* and catface rockcod *E. andersoni*. However, no ranging movements were recorded amongst poenskop *Cymatoceps nasutus* (Sparidae), halfmoon rockcod *E. rivulatus* (Serranidae) or Natal seacatfish *Galeichthys trowi* (Ariidae), all of which were tagged in this study.

Finally, some recommendations for future long-term monitoring and management of the Pondoland Marine Protected Area are discussed along with suggestions for future research.

PREFACE

The experimental work described in this dissertation was carried out in the School of Biological & Conservation Sciences, University of KwaZulu-Natal, Durban, from January 2009 to December 2010, under the supervision of Mr Bruce Q. Mann, Prof Rudy van der Elst (Oceanographic Research Institute) and Dr Paul D. Cowley (South African Institute for Aquatic Biodiversity).

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.

DECLARATION 1 - PLAGIARISM

I, ……………………………………………………………………………………… declare that

1. The research reported in this dissertation, except where otherwise indicated, is my original research.

2. This dissertation has not been submitted for any degree or examination at any other university.

3. This dissertation does not contain other persons' data, pictures, graphs or other information, unless specifically acknowledged as being sourced from other persons.

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Research presented in this dissertation has not yet contributed to any publications (in preparation, submitted, in press or published) other than those listed below:

Popular article

Maggs, J.Q. 2010. Marine Protected Areas. *Ultimate Spearfishing* **1** (2): 12-13.

Signed: …………………………………………………………

They that go down to the sea in ships,
that do business in great waters;
these see the works of the Lord,
and his wonders in the deep.

- Psalm 107: 23-24

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CHAPTER 1 INTRODUCTION

1.1 THE STATE OF THE KWAZULU-NATAL LINE-FISHERY

South African fisheries are exceptionally diverse and productive (Gell *et al.* 2003). Marine living resources include, amongst others, demersal and pelagic fish, crustaceans and molluscs, which are harvested by means of trawling, long-lining, purse-seining, poling, spear-fishing and line-fishing. The line-fishery that operates in estuarine, coastal and offshore marine environments, includes all forms of fishing that use hook and line but excludes long-lining (Mann 2000). This economically and socially important fishery is comprised of three distinct sectors, namely commercial, recreational and subsistence. The latter sector was officially recognised in 1998, and is restricted to shore-fishing.

Commercial boat-based line-fishing began off KwaZulu-Natal (KZN) in about 1905 and effort was initially limited to areas in close proximity to the Durban harbour due to technological and logistical constraints (Penney *et al*. 1999). This provided natural areas of refuge further away, but major advancements in technology took place during the 20th century. Motorised vessels, onboard freezing facilities, monofilament line, trailable ski-boats, outboard motors, radios, echo-sounders, off-road vehicles, Global Positioning Systems and cellular phones allowed fishers to shift their effort further out to sea and away from harbours to coastal areas (Penney *et al*. 1999, Griffiths 2000). Soon the entire continental shelf off KZN became accessible for exploitation.

Data on the KZN commercial line-fishery, collected since 1910 show interesting trends in catch composition (Penney *et al*. 1999). Large endemic sparids such as seventy-four *Polysteganus undulosus*, red steenbras *Petrus rupestris*, scotsman *Polysteganus praeorbitalis* and poenskop *Cymatoceps nasutus*, were heavily exploited in the 1920s and accounted for half the annual catch between 1922 and 1933 (van der Elst 1989). However, their limited distribution and complex life-histories make them particularly vulnerable to over-fishing. They grow slowly and mature late, while some change sex and display great site-fidelity. By 1983, these large endemic fishes favoured by fishers contributed no more than 5% to annual catches (van der Elst 1989).

The seventy-four was heavily targeted since the inception of the KZN line-fishery. An annual spawning migration brought seventy-four to KZN where massive aggregations on the Illovo Banks were intensively targeted by commercial fishers. The annual KZN landing of seventy-four in 1910 amounted to 1550 tons, some 70% of total landings (Penney *et al*. 1989, Penney *et al*. 1999). The stock could not sustain the level of harvesting and collapsed in the 1960s, but fishing continued (Chale-Matsau *et al*. 2001). By 1985, only 3.5 tons of this species were landed annually in KZN, leading to a closure of the seventy-four fishery by managers in 1998. After a ten year moratorium, a recent study by Mann *et al.* (2007) could find little evidence of stock recovery. The fishery for this species remains closed. Many other sparids have been similarly over-fished and their spawning stock biomass remains well below 25% of unfished levels. Stocks below the 25% threshold reference point are considered to have collapsed (Griffiths *et al*. 1999).

Historical data, collected by Natal Fisheries Officers in Durban, suggest that commercial catches were highest soon after the inception of the fishery with about 2000 metric tons landed in 1910. From then, catches gradually declined and by the 1960s, it was evident that some fish stocks had become substantially depleted. Some fishers abandoned the fishery because of poor catches, but those who persisted benefited from the reduction in total effort and attempted to maintain catch rates by switching to less desirable species (Penney *et al*. 1999). Dusky kob *Argyrosomus japonicus* and geelbek *Atractoscion aequidens* are demersal migrants that occur seasonally in KZN, and which have helped sustain catches as more popular species became less abundant. Today, dusky kob (Griffiths 1997) and geelbek (Griffiths 1988) have also been over-fished, and it is the smaller sparids, such as slinger *Chrysoblephus puniceus* and santer *Cheimerius nufar* which now sustain the commercial line-fishery.

In contrast to the KZN commercial fishery, reliable data on the recreational ski-boat line-fishery is only available from 1980, despite this fishery dating back to the early 1900s. In this relatively short time-span, recreational fishers also reported declines in seventy-four, red steenbras and scotsman (Penney *et al.* 1999). Today, slinger account for 30% of the recreational catch, but focus of this sector has shifted towards pelagic game-fish species such as king mackerel *Scomberomorus commerson*, yellowfin tuna *Thunnus albacares* and dorado *Coryphaena hippurus*. These species are more difficult to target because they only appear seasonally and are highly migratory or nomadic and, are thus more resilient to localised depletion.

Declining catch rates and changes in species composition are hallmarks of over-fishing and appear to be the dominant trends in the KZN line-fishery. The state of the KZN line-fishery is representative of the entire country, and in 2000, the Minister of Environmental Affairs and Tourism declared a state of emergency in South Africa’s line-fishery. This situation is not unique to South Africa. Globally, over-fishing has caused the collapse of many fish stocks (Kelleher and Kenchington 1992, Attwood *et al*. 1997a, Pauly *et al*. 2002, Pauly *et al*. 2005), with some species driven to the point of local extinction in certain areas (Roberts 1997). Actual declines in fish stocks could be more severe than indicated by falling catch rates because constantly improving technology enables more effective exploitation of limited stocks (technology creep), thereby masking actual stock levels (Pauly *et al*. 2002, Götz 2006). It is also difficult to contextualise the current state of fisheries because the pristine or baseline state of fish communities is often unknown (Dayton *et al.* 1998). This has become known as the 'missing baseline' problem (Myers and Worm 2003). In continuously declining fish stocks, successive generations of people become subject to the 'shifting baseline syndrome', believing that the abundance and size of fish in their youth represents baseline or pristine levels (Pauly 1995, Baum and Myers 2004, Pinnegar and Engelhard 2008).

1.2 FAILURE OF CONVENTIONAL FISHERIES MANAGEMENT

South Africa, like many other countries, has claimed an exclusive economic zone (EEZ), which is an area of ocean surrounding the land and extending 200 nautical-miles out to sea. This area of ocean was claimed in terms of the United Nations Conference on the Law of the Sea (UNCLOS). As a signatory to the UNCLOS, South Africa is obliged to manage its EEZ responsibly and in an ecologically sustainable manner.

Commercial line-fishing permits were introduced for the first time in 1985 and the total number of permits was capped. That is, no new permits were issued and the number of boats and crew were limited to the 1985 level. Although capped, the permits could be transferred within KZN, which meant that the number of vessels remained constant but the cost of permits has had to be repetitively recouped by new entrants into the fishery (Penney *et al.* 1999).

Recreational effort was not capped and continued to grow, surpassing the commercial sector significantly. Additionally, a number of fishers in KZN who were prevented from registering commercially, continued to sell their catch illegally causing tension between the two sectors (Penney *et al*. 1999). Commercial fishers are required to purchase costly permits and derive the majority of their income from fishing, which is expected to curtail effort as catch is depleted. However, recreational fishers can afford to continue fishing "commercially" in spite of poor catches because they are employed in other sectors. This effectively results in a subsidised fishery, which can continue fishing beyond the bio-economic equilibrium (Clark 1985). Recreational fishers also periodically drive down fish prices by flooding markets when fishing is good. This is especially evident with good "runs" of geelbek. The majority of the proceeds from illegally sold catch go toward covering the costs of recreational outings, such as fuel and bait. Unlimited participation, inexpensive fishing permits and illegal sale have resulted in substantial catch and effort by the recreational sector.

Catches in both sectors are governed by species-specific bag limits, minimum size-limits and closed seasons (Griffiths *et al*. 1999). Bag limits reduce the catch by individual fishers with the aim of reducing fishing mortality and ensuring that the catch is more fairly distributed. Minimum sizes are normally set at size-at-50%-maturity. This ensures that 50% of fish have an opportunity to spawn before being recruited into the fishery, which reduces growth and recruitment over-fishing. Closed seasons provide temporary protection to species when they are particularly vulnerable, for example, during spawning aggregations. Much greater restriction is placed on recreational than on commercial catches where unlimited exploitation is permitted on a number of species. The rationale is that any fisher should be able to catch a few fish for sport, but a limited number of commercial fishers should be able to take large catches, supply the market and thus derive a living.

Fishing effort is widely distributed along the coast and it is therefore difficult to enforce species-specific regulations but even with strict compliance, such regulations do not necessarily ensure successful management. The problems associated with managing multi-species, multi-user fisheries are well described (Russ 1991, Roberts and Polunin 1993). Attwood *et al*. (1997a) lists identification, measuring and weighing of organisms as problems associated with compliance monitoring. Unlimited participation in the recreational sector further reduces the effectiveness of catch limiting mechanisms. In most cases fishers do not achieve their bag limits (Bennett *et al*. 1994, Attwood and Bennett 1995, Mann 2000, Cowley *et al*. 2002). Fish that do not meet the legal requirements are returned to the water, but the survival rate of such fish is questionable in certain species (Attwood *et al.* 1997a, Sumpton *et al.* 2008, Sumpton *et al.* 2010). Barotrauma in deep water sparids may cause immediate death while rough handling may cause injury and also lead to death.

Existing bag limits and sometimes minimum size limits are based on stock assessments, usually using per-recruit prediction models. Such models assume that fish recruitment is constant although not specified, stocks exist in a steady state, and fishing and natural mortality rates are constant during the exploited phase. Environmental fluctuations, beyond human control could render these assumptions and the associated predictions invalid (Sparre and Venema 1998, Rodwell and Roberts 2004). Such single-species approaches consider stock structure merely for calculation of potential harvest, while little attention is given to the effect that the removal of large individuals has on recruitment potential. Stock assessments are also costly and time-consuming. In South Africa's multi-species fishery, stock assessments have focused on a handful of commercially important species leaving many others unaccounted for.

Conventional fisheries management has been useful to slow down the decline in South Africa's fish stocks, but is unable to cope with the present demand on South Africa's multi-species marine resources (Götz 2006). Little allowance has been made for management failure arising from a lack of knowledge. By itself, conventional fisheries management, considering species in isolation, has proven inadequate, allowing a crisis to develop in South African as well as global fisheries (Roberts and Hawkins 2000, Watson and Pauly 2001, Halpern 2003, Roberts *et al*. 2005). Climate change and increasing pollution levels may further aggravate the current situation. These failures are neither sustainable nor acceptable.

1.3 THE ROLE OF MARINE PROTECTED AREAS IN FISHERY MANAGEMENT

The failure of single-species fishery management has initiated a global paradigm shift towards an ecosystem based management approach (Halpern *et al.* 2010, Pomeroy *et al.* 2010), which considers entire ecosystems, multiple sectors and management objectives (Sink and Attwood 2008, McLeod and Leslie 2009). The importance of complete ecosystem protection has been noted in Emanuel *et al.* (1992), Attwood *et al*. (1997a), Gell and Roberts (2003b), and Sink and Attwood (2008). Many international conventions and agreements have set guidelines and targets for sustainable resource management (e.g. Convention on Biological Diversity, 1992; World Summit on Sustainable Development, 2002; World Parks Congress, 2003). South Africa has ratified its future commitment to the sustainable use of marine resources by being a signatory to these agreements, which call for the protection of representative ecosystems. In October 2010, the 10th Conference of the Parties (COP 10) to the Convention on Biological Diversity (CBD) set a target for protected area coverage. It was agreed that 10% of all marine and coastal eco-regions should be included in protected areas by 2020, which is an extension on the original 2012 deadline set by the CBD in 2005.

Marine protected areas (MPAs) or reserves have become extremely popular in many countries for such marine resource management (Attwood *et al*. 1997a, Kaplan 2009, Gaines *et al*. 2010) and were defined by Kelleher and Kenchington (1992) as, "*any area of intertidal or subtidal terrain, together with its overlying waters and associated fauna, flora, historical and cultural features, which has been reserved by legislation to protect part or all of the enclosed environment*". However, the definition does not explain exactly what protection means. MPAs can provide very different levels of protection (Unsworth *et al.* 2007). Areas may merely be zoned for future management or be excluded from selected environmentally harmful activities, or where only one or a few vulnerable species are protected. The highest level of protection is found in absolute no-take (no-go) sanctuaries, which were defined by Unsworth *et al.* (2007) as, "*an area where any form of fishing or exploitation is prohibited*".

In the absence of complete ecosystem understanding a no-take area provides a precautionary approach to resource conservation and insurance against management failure (Buxton 1993b, Bohnsack 1996, Lauck *et al*. 1998). There are also management benefits to having MPAs. Illegal fishing in a no-take area is relatively easy to detect and automatically represents non-compliance, whereas enforcing bag limits, minimum size limits and prohibited species requires considerable manpower and training. Multi-species resources managed within a MPA also reduce the reliance on empirical data (Attwood *et al.* 1997a), which is deficient in South Africa (Buxton 1993b).

On the one hand, MPAs can potentially provide effective localised protection (Attwood *et al.* 1997a), but on the other hand, they tend to concentrate effort elsewhere (Parrish 1999). Thus, it is imperative that open areas continue to be managed with existing catch limiting mechanisms such as size limits, bag limits and closed seasons (Allison *et al.* 1998). MPAs should not be viewed as an alternative but rather as an additional tool to complement existing fisheries management methods (Russ 2002, Roberts *et al*. 2005).

Today, South Africa has 22 recognised MPAs providing varying levels of protection. The Tsitsikamma National Park on the Cape south coast, proclaimed in 1964 was the first MPA to be established on the African continent. Sink and Attwood (2008) report that, although 9% of South Africa's coastline is fully protected in no-take MPAs, only 0.16% of South Africa's EEZ is protected, far short of the 10% set out by COP 10. Coverage is mostly on the east coast with little or no protection on the west coast and offshore environments, neglecting many biogeographic regions and habitats.

1.4 BENEFITS OF MARINE PROTECTED AREAS – THE RESERVE EFFECTS

With the proclamation of a MPA which prohibits fishing, certain changes in fish populations are expected to take place (Fig 1.1). The most noticeable changes will be localised, taking place within the immediate boundaries of the no-take area. For example, previously exploited species are likely to show population recovery (growth) as a direct result of protection from fishing, but in time the entire fish community may be affected indirectly. The prohibition of fishing in a no-take area may also have a regional influence on adjacent fisheries through the spill-over of fish biomass and seeding of eggs and larvae.

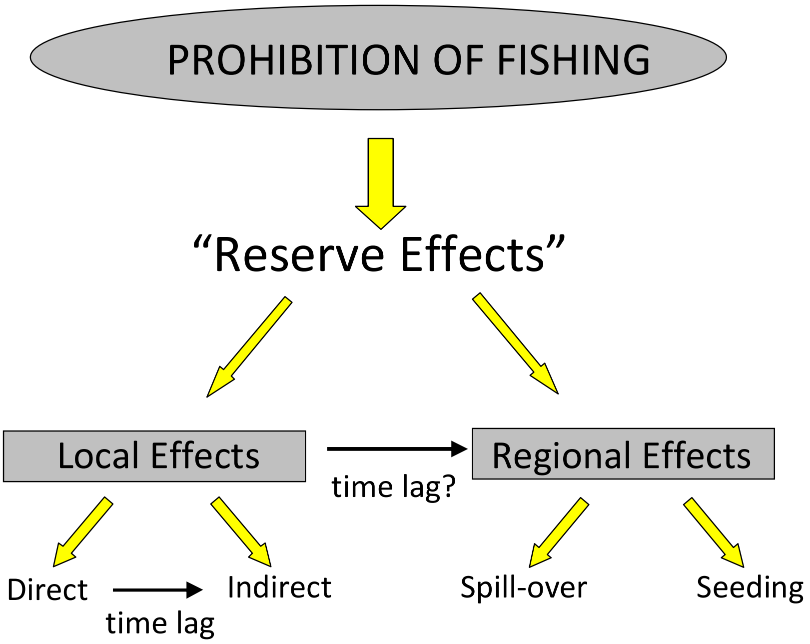


Figure 1.1. Theoretical outline of the reserve effects, following prohibition of fishing.

Direct effects on target species (localised)

Within the boundaries of a no-take area, protection from exploitation is expected to directly allow an increase in abundance and survivorship (biomass) in previously exploited species. Such changes are most relevant to the conservation of critical spawning stock. There are numerous local (Buxton and Smale 1989, Bennett and Attwood 1991, Brouwer 2002, Cowley *et al*. 2002, Götz *et al*. 2009) and international (Francour 1991, Russ and Alcala 1996b, Babcock *et al*. 1999, Edgar and Barrett 1999, Halpern 2003) accounts of the conservation benefits that take place within no-take areas.

Natural sex-ratios in protected areas: Fishing can skew natural sex-ratios and thus compromise reproductive potential. Many of South Africa's most important line-fish are believed to be sequential hermaphrodites (Buxton and Clarke 1989, Garratt *et al*. 1993, Fennessy and Sadovy 2002, Mann *et al*. 2005, Fennessy 2006). For example, it has been shown that all slinger *Chrysoblephus puniceus* (Sparidae) are born female and later change into males at a certain age (Garratt 1986, Garratt *et al.* 1993). Recreational fishermen usually target larger individuals and are encouraged to do so by minimum size limits. Such size selective exploitation can skew natural sex-ratios by removing older fish which are males (Buxton 1993a, Garratt *et al.* 1993). In an exploited population off Park Rynie, on the KZN south coast, Garratt (1993b) suggested that there was only one male slinger for every 55.5 females in 1992. The reproductive potential of such a population is compromised (Bannerot 1984, Buxton 1993a, Garratt *et al*. 1993, Coleman *et al*. 1996). The elimination of fishing mortality is expected to allow natural age and sex structures to be re-established. An unexploited population of slinger in the St Lucia Marine Reserve, in northern KZN, was found to contain five young female slinger for every older male and in southern Mozambique, the ratio was 2.3 females for every male (Garratt 1993b). This is assumed to be closer to the natural sex-ratio for this species. Similar differences between protected and exploited populations have been noted in other sex changing sparids such as roman *Chrysoblephus laticeps* and dageraad *Chrysoblephus cristiceps* (Buxton 1992, 1993a, Kerwath 2005).

Enhanced egg production: In areas where a fish species is known to spawn, an increase in abundance due to protection should further boost overall reproduction. Successful mating encounters also increase with higher density of individuals. However, protected individuals are able to reach maximum size, which is believed to have further benefits. Larger females have been found to produce exponentially more eggs than smaller individuals (Berkeley *et al*. 2004, Field *et al*. 2008). Older individuals are also believed to produce more resilient, genetically fitter offspring than younger fish (Hislop 1988, Marteinsdottir and Steinarsson 1998, Berkeley *et al*. 2004). Fished populations typically have a lower age structure, and in some cases, a reduced age-at-maturity (Buxton 1993a). Not only does exploitation reduce reproductive output by removing large individuals, it may also lead to production of weak offspring by unnaturally young parents. Protection of spawner-biomass in no-take areas thus provides insurance against recruitment failure.

Preservation of genetic diversity: Fisheries tend to select large, aggressive, resident fish. This favours a population with small body size, slow growth, timidity and higher mobility (Attwood *et al.* 1997a, Parsons *et al.* 2010). Protected areas can provide protection against loss of genetic diversity caused by over-fishing (Attwood *et al.* 1997a, Hauser *et al.* 2002).

Indirect effects on the community (localised)

Following a prohibition of fishing, it is expected that previously targeted fishes will become larger and more abundant, potentially increasing the competition between these and other non-target species. Additionally, most of South Africa's line-fish are significant predators in the ecosystem, preying on small reef-fish species, which in turn, prey on small invertebrates (Toral-Granda *et al.* 1999). As such, the prohibition of fishing may have a delayed, indirect influence on the entire fish community through cascading trophic interactions (Babcock *et al.* 2010). In the absence of exploitation, it is expected that the fish community may tend towards a more natural state, which has many benefits.

Provision of a natural benchmark: In the absence of pristine areas, a no-take area can provide a benchmark or control for scientific research (Attwood *et al*. 1997a, Roberts and Hawkins 2000, Babcock *et al*. 2010). Undisturbed fish populations can be compared with populations in exploited areas to place the effects of fishing in context (Griffiths 2000), and can be used to calculate parameters such as natural mortality. Additionally, the natural variability in undisturbed populations provides an opportunity to assess the effects of climate change in long-term monitoring projects.

Tourism: The benefits of natural ecosystems are becoming increasingly evident. Such areas may be attractive to tourists providing visitors with a wilderness experience. Accordingly, non-consumptive users, such as whale-watchers and SCUBA divers, may be willing to pay a great deal to visit such places. In some cases, the revenue generated from tourism can help to finance the management of the protected area (Depondt and Green 2006).

Spill-over to adjacent fisheries (regional)

The success of MPAs in conserving biodiversity has led to their rapid increase on a global scale (Gell and Roberts 2003b). However, their localised conservation benefits have little immediate benefit to adjacent fishing communities and opposition from these communities remains the principal barrier to area closure (Gell and Roberts 2003b; T. McClanahan, Wildlife Conservation Services, 2010, pers. comm.). A community in opposition to a no-take area may hinder conservation efforts by poaching (Gell and Roberts 2003b). Reasons for opposition include loss of fishing grounds, removal of livelihood, lack of involvement of local people and dissatisfaction with past failures (Gell and Roberts 2003b).

No-take MPAs can benefit adjacent fisheries through export of adult and sub-adult fish, and seeding of eggs and larvae (Palumbi 2001). Enhancement of adjacent fisheries may be accepted as compensation for loss of fishing ground and may even provide incentive for the fishing community to assist with monitoring of illegal fishing (Johannes 1978, 1982, Alcala and Russ 2006). With the proclamation of more MPAs comes the increasing demand for proof of benefits to fisheries (Kramer and Chapman 1999, Gell and Roberts 2003b), hence research and monitoring should be an integral part of the initial planning phase. No-take areas are expected to have the following regional influences:

Seeding of adjacent areas: Broadcast spawning by adults in protected areas can continuously provide large amounts of eggs and larvae to adjacent fished areas. Dispersal of offspring is believed to be dependent on ocean currents although dispersal may not always be passive (Beckley 1993). If positioned correctly, with knowledge of larval dispersal patterns, a no-take area can provide adjacent areas with a constant supply of new recruits (Tilney *et al.* 1996).

Biomass export*:* Further support may be gained through the promise of biomass export (Zeller 1997) – this is post-larval fishes moving from the protected no-take area into the adjacent exploitable areas (Rowley 1994). A well protected no-take area has the potential to provide a continuous source of certain fish species to adjacent fisheries (Alcala and Russ 1990, Attwood and Bennett 1994, Russ and Alcala 1996a).

1.5 THE OBJECTIVES OF THE PONDOLAND MARINE PROTECTED AREA

The Pondoland MPA was proclaimed on 4 June 2004 in Government Notice number 694 (Government Gazette No. 26430) under section 43 of the Marine Living Resources Act (Act 18 of 1998). The 800 km^2 protected area is situated in the Eastern Cape, South Africa, near to the southern border of KZN. Within the MPA, there are two controlled zones, which allow boat-based exploitation, but no industrial fishing, while a restricted no-take area is closed to all boat-based extractive exploitation (see Fig 2.3, Chp 2). In the proclamation of the Pondoland MPA, the following objectives are stipulated:

- "*Protect and conserve marine ecosystems and populations of marine species*;
- *Protect the reproductive capacity of commercially important species of fish, including shellfish, rock lobster and traditional linefish and to allow their populations to recover*;
- *Promote eco-tourism within the Marine Protected Area*."

With regard to ichthyofauna, the primary objectives of the MPA are therefore to rebuild depleted line-fish stocks, which are commercially important to KZN and the Eastern Cape and establish an undisturbed fish community. However, no specific mention is made of regional benefits. Although "reproductive capacity" is mentioned in point 2, the importance of such an objective to adjacent areas is omitted. Evaluations of MPA effectiveness are imperative (Attwood *et al*. 1997b, Pomeroy *et* *al*. 2004). An evaluation of the Pondoland no-take area in achieving its objectives was thus the primary focus of this study.

1.6 KEY QUESTIONS AND RESEARCH PLAN

The aim of this study was to evaluate the effectiveness of the Pondoland MPA in protecting key line-fish species, establishing an undisturbed fish community, and its potential benefit to adjacent fisheries. In this regard, specific emphasis is placed on the no-take area.

Key questions

1. Is the abundance and biomass of key South African line-fish species greater in the Pondoland no-take area than in the adjacent exploited area?

2. Is the fish community structure in the Pondoland no-take area different from that in the adjacent exploited area?

3. Is there potential for the Pondoland no-take area to benefit adjacent fisheries by the export of important line-fish species?

Thesis layout and research plan

Chapter 2 presents a general overview of the Pondoland region, and more specifically, the study area and sample sites. Chapter 3, 4 and 5 address the key questions of this study. Chapter 3 investigates key line-fish species by comparing abundance and biomass of fishes in the no-take area with that in the adjacent exploited area using a controlled fishing experiment. Chapter 4 investigates the fish community in both areas by means of an underwater visual census (UVC) and Chapter 5 evaluates the potential for regional fisheries benefits using a tag-recapture study. Finally, the results are synthesised and concluded in Chapter 6 with some recommendations for future monitoring, research and management. The key questions are answered in the following way:

Rebuilding depleted line-fish stocks (Chapter 3)

Line-fishing is an effective and reliable means of sampling a line-fish population. Captured fish can be accurately identified, measured and safely returned to the water, procedures not possible when using destructive methods such as trawling or ichthyocides. Samples can also be taken in turbid conditions that prevent other frequently used non-destructive methods from being used such as underwater visual census (UVC) or underwater video (Perrow *et al*. 1996, Bennett 2007).

Line-fishing can, however, be very selective with certain species and/or certain sized individuals being excluded from catches (Roberts 1995). Generally, herbivorous and corallivorous fishes will not take a baited hook and hook-size selectivity may cause skewed length frequencies in some species (Perrow *et al*. 1996). While being useful for comparisons of abundance and biomass of line-fish species, line-fishing does not adequately sample the entire fish community. By "sub-sampling" the community, line-fishing therefore provides a limited means of detecting changes in the overall species composition.

Changes in the fish community (Chapter 4)

UVC, in which an assessment of the wider fish community or population is made by direct observation, has been widely used since the 1950s (Brock 1954, Bohnsack and Bannerot 1986, Thresher and Gunn 1986, Bennett 2007). In spite of certain limitations, which are discussed in more detail in Chapter 4, UVC provides a simple, non-destructive method which is independent of line-fishing methods (Samoilys and Carlos 2000). Direct underwater observation also allows for the collection of a more representative sample of the entire fish community than line-fishing.

Used together, line-fishing and UVC were found by (Bennett 2007, Bennett *et al*. 2009) to be effective complementary methods of sampling fish populations in long-term monitoring studies, especially in MPAs where destructive sampling is not acceptable.

Regional fishery benefits (Chapter 5)

Mark-recapture techniques have proven a reliable method for showing dispersal of fishes from no-take areas (Brouwer *et al*. 2003, Attwood and Cowley 2005). With this method, suitable species are caught by fishers and tagged with uniquely numbered plastic "spaghetti" tags. The location and date of capture and length of fish is recorded with the unique tag number. Recaptured fish provide scientists with information on movement patterns, home-range sizes and growth, which can be used to investigate the potential biomass export to adjacent fisheries.

CHAPTER 2 DESCRIPTION OF THE PONDOLAND REGION AND SELECTION OF STUDY SITES

2.1 REGIONAL CONTEXT

Pondoland is situated on South Africa's east coast in the upper reaches of the Eastern Cape, between the Mtamvuna (31° 04.8' S, 30° 11.7' E) and Mzimvubu Rivers (31° 37.7' S, 29° 32.9' E). This 90 km coastline is located directly adjacent to the southern border of KwaZulu-Natal (KZN) and is part of a greater region between the Mtamvuna and Great Kei Rivers known as the Wild Coast (Fig 2.1.).

Pondoland is believed to be part of a broad transition zone between two major biogeographic regions (Stephenson and Stephenson 1972). It has been well established that the warmer subtropical Natal Bioregion extends southwards from about Cape Vidal on the KZN north coast (Jackson 1976), while the cooler warm-temperate Agulhas Bioregion extends eastwards from Cape Point (Turpie *et al.* 2000). However, there has been little agreement regarding the division between the two bioregions (Sink *et al.* 2005), possibly because of the transitional nature of the Wild Coast. Emanuel *et al.* (1992) conducted a study on rocky-shore invertebrates and suggested a distinct break between the two bioregions just north of East London. In contrast, Turpie *et al.* (2000) reported a gradual turnover of coastal fishes from Cape Point eastwards with no clear break between the Agulhas and Natal bioregions. These authors acknowledged limitations in the data from that area, but suggested that the Wild Coast was more similar to the Agulhas Bioregion. More recently, the Natal Bioregion was extended to include the Pondoland coast and now stretches from the Mbashe River in the south to Cape Vidal in the north (Lombard *et al.* 2004).

Much of the Pondoland coast is rural and the rugged shoreline is mostly inaccessible by road (Fig 2.2). The coastal geology is dominated by the Msikaba Formation (Thomas *et al*. 1992) and the coastline is generally rocky with small sandy bays and estuary mouths (Mann *et al.* 2003). The rocky outcrops are high profile in places and three rivers enter the sea as waterfalls. There are 23 estuaries along the Pondoland coast, of which 17 are temporary open/closed systems, five are permanently open to the sea and one, the Mzimvubu, is classified as a river mouth (Whitfield 2000). Many of the rivers carry large silt loads due to poor farming practices in the catchment areas (Morant and Quinn 1999).

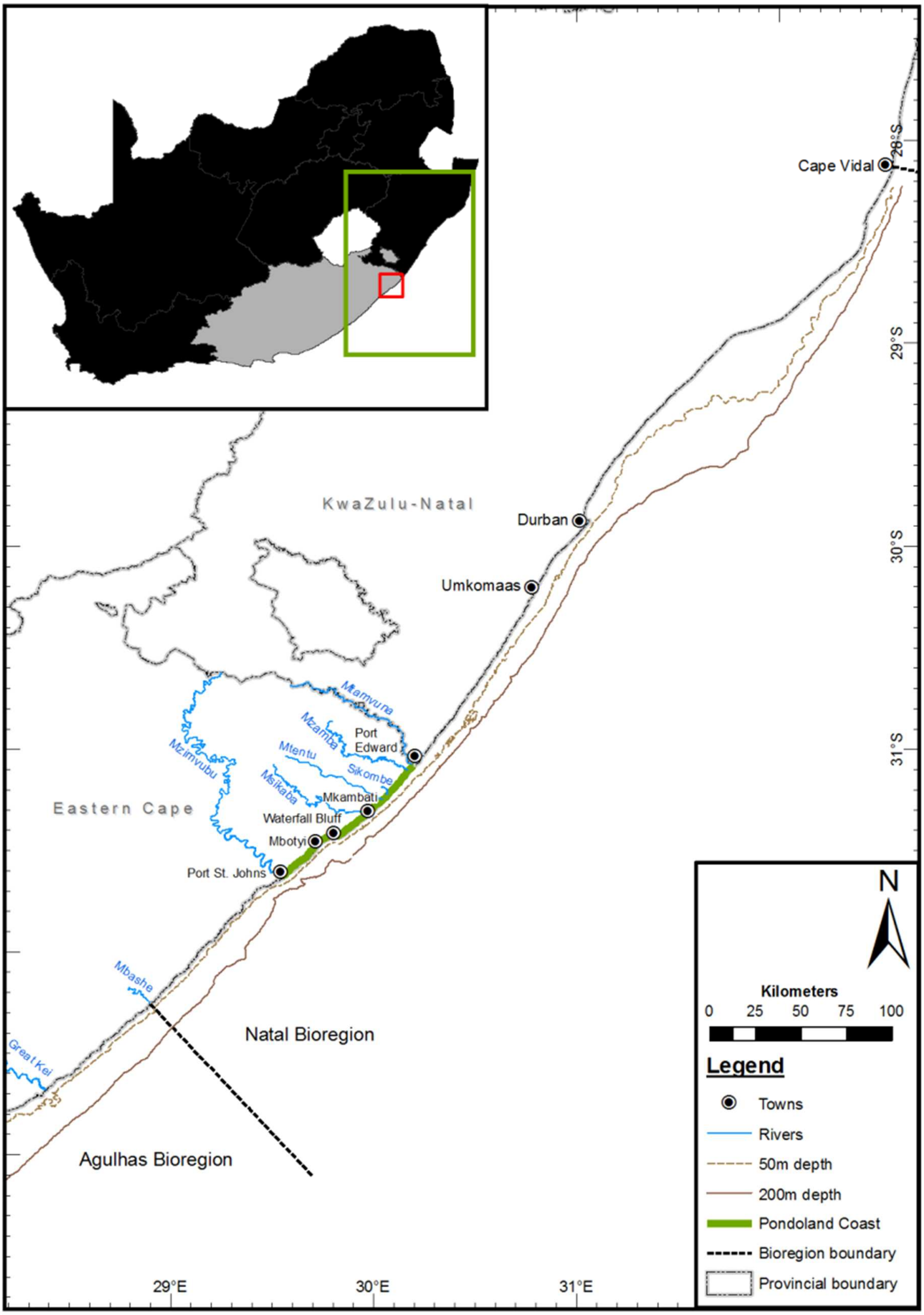


Figure 2.1. Regional map of the lower KZN and upper Eastern Cape coast. *Map compiled by the Oceanographic Research Institute, data sources: Council for Geoscience, Municipal Demarcation Board, South African National Biodiversity Institute, South African Navy Hydrographic Office.*

Figure 2.2. Waterfall Bluff (31° 26.071' S; 29° 49.354' E) on the Pondoland coastline.

2.2 CLIMATIC CONDITIONS

The Pondoland climate is mild, subtropical – transitional between the hot and humid KZN coast, and the cool and wet southern Cape. Relative humidity ranges from 60% in the winter up to 85% in the late summer with an annual average of 76%, while average air temperature on the coast is 18° C in the winter and 22° C in the summer (de Villiers and Costello 2006). Winds blow predominantly from the south-west or north-east, parallel to the coast and can be fairly strong at times. The region receives an average of 1200 mm of rainfall annually, mainly in the spring and summer months with about 732 mm falling between September and February. It is driest in winter between May and August, but a minimum of 50 mm of rain is expected every month (de Villiers and Costello 2006).

2.3 OCEANOGRAPHY

The oceanography on South Africa's east coast is dominated by the Agulhas Current, which brings warm water down the coast from the tropics (Schumann 1988, Beckley and van Ballegooyen 1992). This western boundary current transports about 70 million m^3 of seawater per second (Schumann 1988) and follows the edge of the continental shelf (Roberts *et al.* 2010). The shelf varies in width but is particularly narrow along the Wild Coast (Beckley *et al.* 2002), where it reaches its maximum velocity (Shannon 1989). Between Umkomaas on the KZN south coast and Mkambati, the Agulhas Current flows close to the shore and has a pronounced influence on the inshore shelf waters, where flow is predominantly south-westward and averages ~0.75 m.sec^{-1} (Roberts *et al.* 2010). Along the Pondoland coast, the shelf widens to 14 km offshore at Waterfall Bluff, before narrowing again to 6 km slightly north of Port St Johns (Birch 1996). A cyclonic eddy occurs in the area between Waterfall Bluff and Port St Johns, which results in a north-eastward counter-current and associated upwelling (Roberts *et al.* 2010).

The seafloor morphology between Port Edward and Port St Johns has been surveyed (Birch 1996), but limited information exists on the location of inshore reefs, shallower than 30 m (Mann *et al*. 2006, Celliers *et al*. 2007). In general, the Pondoland sub-tidal environment consists of shelving reef complexes running parallel to the coast, which is characteristic of the entire Natal Bioregion (Flemming 1978, Birch 1981, Ramsay 1991, Birch 1996, Ramsay 1996, Bosman *et al*. 2005). Birch (1996) also reported that five submarine canyons cut into the continental shelf: the Mtamvuna, Mtentu, Egosa, Mbotyi and Mzimvubu canyons. The structure of the sub-tidal reefs varies between 1.2 and 3.6 km in width and between 1 and 26 m in vertical relief (Birch 1996, Mann *et al.* 2006).

The Pondoland coastline is exposed and often subjected to adverse sea conditions. In the summer, north-east winds prevail and periodically swing to south-westerly with the passing of coastal low pressure systems. In the winter, cold fronts passing eastwards along the southern Cape coast cause strong south-west winds to blow on the east coast. Occasionally these fronts bring large south-westerly swell up towards the Eastern Cape and KZN. During periods of heavy rainfall, silt-laden rivers flow into the sea and reduce the seawater visibility. However, sea conditions are generally calmer and the water visibility is clearer during early winter, between May and June. Inshore sea temperature on the KZN coast varies between 20° C in the winter and 25° C in the summer, although localised upwelling events can cause rapid reductions in temperature (Schumann 1988).

2.4 BIOGEOGRAPHY

The Agulhas Current allows the intrusion of tropical Indo-Pacific species into high latitudes along South Africa's east coast (Lutjeharms 2006). The sub-tidal reefs off the Pondoland coast therefore provide habitat for a mix of subtropical and temperate fauna with many endemic fish and invertebrate species (Emanuel *et al*. 1992, Turpie *et al*. 2000, Mann *et al*. 2006, Celliers *et al*. 2007). The algal-dominated benthic community reflects the turbid conditions and sedimentation on the reefs, caused by the large amounts of riverine input (Lombard *et al.* 2004). Consequently, the benthic flora consists mostly of sand-algae base cover, coralline and turf algae (Celliers *et al.* 2007). Besides seasonal migrants, which include pilchard *Sardinops sagax* and associated predators, there are many endemic resident line-fish species which are recreationally and commercially important.

2.5 THE PONDOLAND MPA

Commercial line-fishing boats have been operating in the northern region of Pondoland (area of this study) since the 1970s. Large, non-shoaling sparids such as scotsman *Polysteganus praeorbitalis*, englishman *Chrysoblephus anglicus* and seventy-four *Polysteganus undulosus* were heavily exploited. The smaller shoaling sparid, slinger *Chrysoblephus puniceus*, was also targeted intensively with the introduction of echo-sounders (fish finders). Large catches were taken close to launch-sites at first but declines in catches were already evident by the late 1970s and skippers began travelling further to maintain catch rates (Rob Allen, Dive Factory, 2010, pers. comm.).

Prior to the proclamation of the Pondoland MPA, <3% of the Natal Bioregion's coastline was protected by no-take MPAs (Mann *et al.* 2006). Due to the unique attributes of the Pondoland coast and in terms of its rich marine biodiversity, it was considered to be an important gap in the existing network of MPAs along the South African coast (Attwood *et al*. 1997b, Turpie *et al*. 2000, Mann *et al*. 2006, Celliers *et al*. 2007).

The former Mkambati MPA was originally proclaimed in 1991 under South Africa's Sea Fisheries Act (No. 58 of 1973). During independence of the former Transkei homeland, the Mkambati MPA was recognised in terms of the Transkei Fisheries Regulations (Special Gazette No. 81, 9 December 1992) under the Environmental Conservation Decree (No. 9 of 1992). The Transkei was reintegrated into South Africa, and this MPA was again recognized under South African law on 1 November 1997 under the Sea Fisheries Act (No. 12 of 1988) and later under the Marine Living Resources Act (No. 18 of 1998, Government Gazette, Vol 426, No. 1429). The aim of this MPA was to protect important offshore line-fish populations between the Mtentu River in the north and the Msikaba River in the south. Although this MPA was located approximately 25 km from the nearest launch-site, ski-boat anglers were known to travel up to 40 km south from Port Edward (Kevin Cox, KZN Sharks Board, 2009, pers. comm.). These outings took them through the Mkambati MPA as far south as Msikaba and because the MPA was poorly managed, poaching continued unabated (Mann 1998). Furthermore, a launch-site at Msikaba was frequently used during holiday periods by boat owners belonging to the Msikaba Cottage Owners Association.

The new, much larger Pondoland MPA was proclaimed in June 2004 and includes approximately 800 km^2 of marine environment, replacing the Mkambati MPA (Fig 2.3). This MPA includes 80 km of coastline (straight line distance) and extends 10-15 km out to sea, to the 1000 m depth contour. The northern boundary is represented by a 128° line from the southern head of the Mzamba River. The southern boundary is a 128° line from the northern head of the Mzimvubu River. The no-take area of the MPA spans 40 km of coastline from the Sikombe River in the north to the Mbotyi River in the south, also extending out to the 1000 m depth contour. The no-take area represents 6.8% of the Natal Bioregion in terms of coastline length.

Shore-based fishing and spear-fishing are only permitted in certain areas throughout the entire MPA. Areas which are closed to such shore-based exploitation are indicated by red stripes in Fig 2.3. No boat-based exploitation is permitted anywhere in the offshore no-take area (i.e. between Sikombe and Mbotyi), but there are two buffer zones on the outer edges of the no-take area (indicated by yellow shading in Fig 2.3) where boat-based line-fishing and spear-fishing are permitted. No industrial fishing, such as trawling or long-lining, is permitted anywhere in the entire MPA.

2.6 SELECTION OF STUDY SITES

The launching of small craft is impossible along virtually the entire length of the Pondoland coastline due to the combination of inaccessibility, topography and frequent large surf. The most accessible, safe launch-site which can be used to access the Pondoland region is at Port Edward (31° 2.678’ S; 30° 13.719’ E), 25 km from the northern boundary of the no-take area. The Msikaba launch-site has not been used since the proclamation of the Pondoland MPA in June 2004.

The locations of the shallow inshore reefs were obtained from the KZN Sharks Board and from commercial fishers in the area as well as previous biodiversity studies (Fielding *et al*. 1994, Mann *et al*. 2006, Celliers *et al*. 2007). Four 2 x 1 km study sites containing scattered reef habitats within 10-30 m water depth were selected as study sites (Fig 2.4). Two of these study sites are located to the north of the Sikombe River in the exploited area (Casino & Mnyameni) and two are in the no-take area (Mtentu & Mkambati). All sites are located within 350-1500 m of the shoreline. Although Casino is outside the northern boundary of the MPA and Mnyameni inside the northern buffer zone (exploited area) of the MPA, both sites are similarly open to boat-based line-fishing and spear-fishing. Casino is thus referred to as an ‘exploited area’ study site.

Areas shallower than 30 m were selected to avoid excessive barotrauma to fishes while angling and to allow for divers to perform underwater visual censuses without having to undergo decompression. The study sites were accessed using a small, 6 m, surf-launched vessel from the Port Edward launch-site.

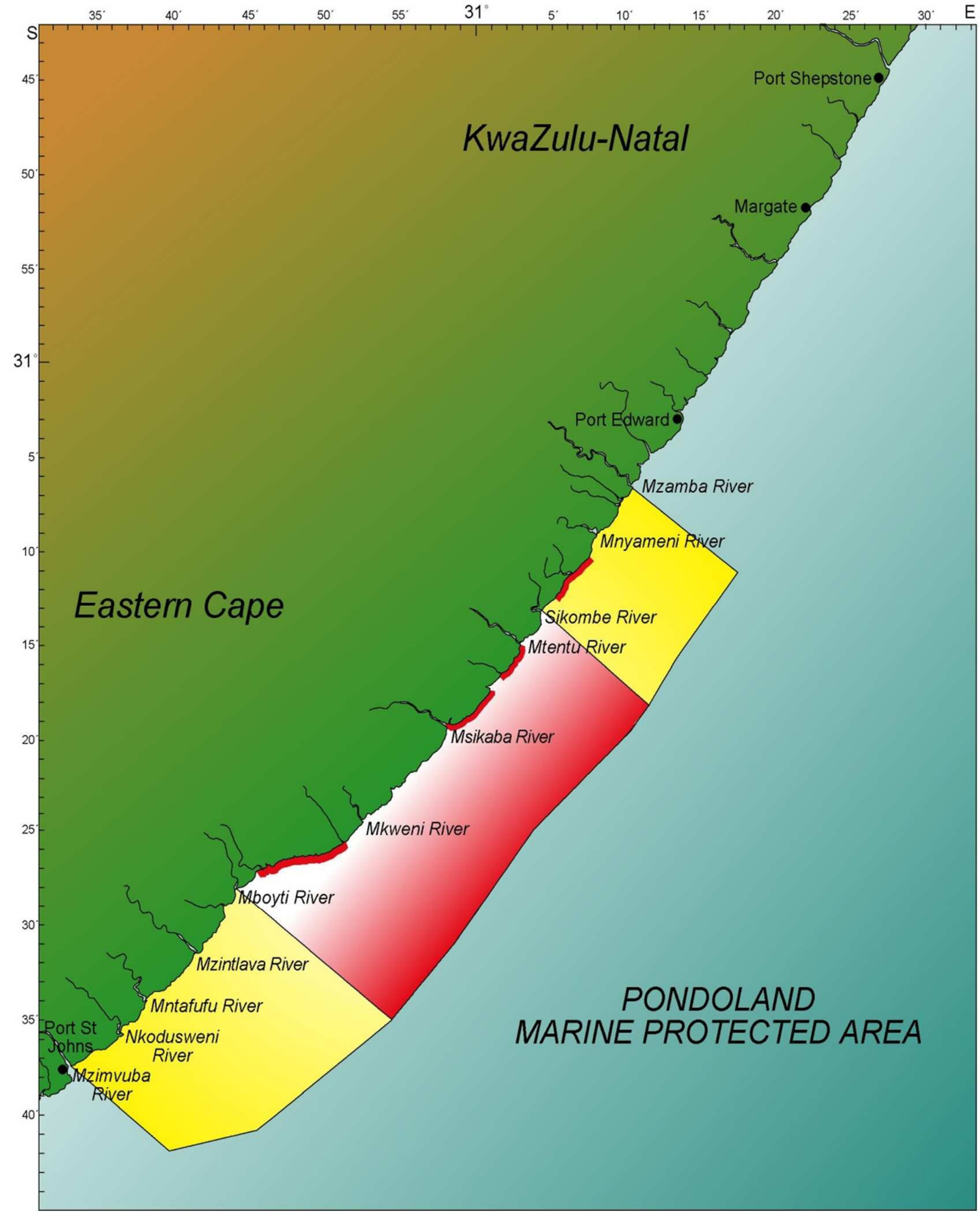


Figure 2.3. The Pondoland Marine Protected Area on the Eastern Cape coast of South Africa. Yellow shading indicates buffer zones where certain exploitation is permitted. The red shaded area (~400 km^2), between the Sikombe and Mbotyi Rivers, represents a no-take area for boat-based exploitation. Red stripes on the coastline indicate no-take areas for shore-based exploitation. Map used with permission from Department of Environmental Affairs.

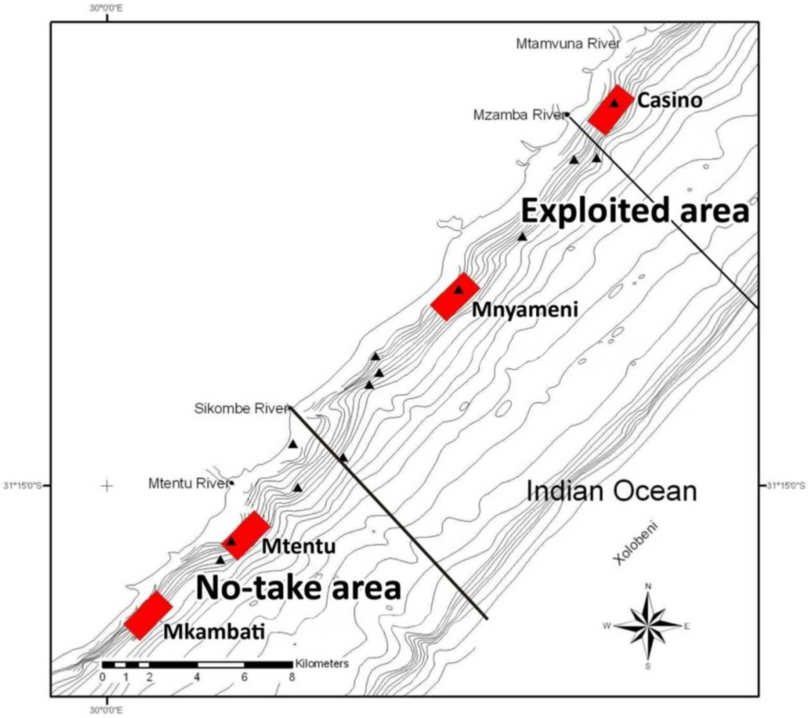


Figure 2.4. Study sites in the exploited (Casino & Mnyameni) and no-take areas (Mtentu & Mkambati) of the Pondoland MPA. Triangles represent sites dived during the 2002-2003 biodiversity survey conducted by *Mann et al*. (2006).

The area sampled in the current study, which is to the north of Msikaba was reported by Celliers *et al*. (2007) to be algal-dominated and relatively low in benthic diversity; the area to the south having higher diversity with suspension feeding invertebrates becoming more abundant. Celliers *et al*. (2007) attributed this shift away from macro-algae to low light levels associated with increased turbidity; and the shift towards suspension feeding invertebrates, south of Msikaba, they attributed to increased supply of nutrients from nearby rivers. However, these authors did not mention the potential effects of the current-eddy circulating between Waterfall Bluff and Port St Johns with its associated upwelling. All the sampling sites in the current study are located north of Msikaba and are therefore not expected to be influenced by these confounding factors.

The study sites are thus considered similar in benthic composition being algal-dominated and due to the close proximity of the sites to each other (within 30 km), all are assumed to be oceanographically similar and therefore comparable. Although poaching does take place in the no-take area, most illegal fishing occurs further offshore on deeper reefs (B.Q. Mann, Oceanographic Research Institute, 2010, pers. comm.) and thus the effect of poaching in the selected no-take study sites was considered to be negligible. Any differences in the fish community between the no-take and exploited areas can thus be largely attributed to the difference in fishing effort.

CHAPTER 3 AN ASSESSMENT OF THE DIRECT EFFECTS OF PROTECTION ON IMPORTANT FISHERY SPECIES

3.1 INTRODUCTION

Fishing reduces the abundance of target species and selects larger individuals (Pauly *et al.* 2002) reducing mean size and age of species (Pauly and Watson 2005). When previously exploited areas are closed to fishing, as in marine protected areas (MPAs), stock recovery of targeted fish species is anticipated (Jennings 2000). There have been numerous accounts of such recovery from across the globe in tropical and temperate ecosystems.

In the Philippines, Russ *et al*. (1996b) correlated increases in density and biomass of previously exploited fish stocks with protection in the Sumilon and Apo Island MPAs. Roberts (1995) reported a rapid build-up of fish biomass in two small Caribbean MPAs relative to nearby fished areas, recording increases in density and individual body size of important fishery species. Following the establishment of the Mombasa Marine National Park in Kenya, McClanahan and Kaunda-Arara (1996) found that most species of fish showed recovery. Willis *et al.* (2003) reported that large snapper *Pagrus auratus* (Sparidae) were 14 times more abundant in three New Zealand MPAs than adjacent fished areas.

In South Africa, the effects of MPAs have been measured in numerous controlled fishing experiments. In a shore-based fishing study, Cowley *et al*. (2002) showed that four important shore-angling species were more abundant and of a larger mean size in the protected Tsitsikamma National Park (protected since 1964) than in the adjacent exploited area. Following the proclamation of the De Hoop Marine Reserve on the southern coast of South Africa, a notable recovery was recorded in previously exploited shore-angling fish species (Bennett and Attwood 1991, Bennett and Attwood 1993). Götz *et al.* (2009) evaluated the functioning of the Goukamma MPA by means of a boat-based fishing study and found that protected roman *Chrysoblephus laticeps* were more abundant and of a larger mean size than they were in the nearby open area where they were heavily fished.

By means of a controlled fishing survey, this chapter asks one key question: is the abundance and biomass of key South African line-fish species greater in the Pondoland no-take area than in the adjacent exploited area? Thus it could be established whether the Pondoland MPA is achieving its primary objective of biodiversity conservation and rebuilding of depleted fish stocks, and whether there is a basis for achieving the secondary objective – export of fish biomass to adjacent fisheries.

3.2 METHODS AND MATERIALS

3.2.1 Data collection

Field trips, consisting of two angling days, were conducted quarterly from April 2006 until January 2010. During each field trip, one day was spent fishing at the two 2 km^2 study sites in the no-take area (Mtentu and Mkambati) and the other day at the two 2 km^2 study sites in the adjacent exploited area (Casino and Mnyameni) (see Chp 2). Each day, approximately three hours were spent fishing in each of the 2 km^2 sites. Fishing stations (GPS coordinates) within each 2 km^2 site were randomly selected and standardised angling was conducted by three anglers on a ski-boat. Most angling was conducted while at anchor although drift fishing was sometimes used when there was no current. Approximately one hour was spent fishing at each fishing station but this was sometimes affected by current and wind strength. Later in the study, angling events were limited to a maximum of one hour and a minimum of 15 minutes to ensure a spread of fishing effort within each site.

Besides the three anglers, a fourth person processed the catch, recording all catch and effort data and tagging selected fish species. Each fishing station was recorded as a fishing event, during which time, locality, catch (species and length) and effort (angler-hours) were recorded. A Garmin 76 handheld GPS, with an accuracy of at least 15 m, was used for all GPS positioning.

Catch-and-release line-fishing can be stressful to fishes and the severity of harm inflicted is strongly associated with the depth of water from which a fish is caught, the time that the fish spends out of the water and how well it is handled (Bennett 2007, Sumpton *et al*. 2010). Individuals may also suffer from barotrauma or injury to the gills or viscera caused by hook ingestion (Willis *et al*. 2000, Bennett 2007).

Every attempt was made to mitigate these effects. Standardised fishing traces were used, consisting of a 1.5 m monofilament mainline terminating in a lead sinker (10 – 16 ounce). Using three-way swivels, two circle hooks were attached to the mainline using a 0.5 – 1 m snoot of monofilament line. Circle hooks were used to reduce the incidence of gut-hooking (Zimmerman and Bochenek 2002, Cooke and Suski 2004, Bennett 2007), and barbs were removed to allow for rapid hook removal and to reduce injury (Diggles and Ernst 1997, Schaeffer and Hoffman 2002, Parsons *et al*. 2003, Bennett 2007). A larger circle hook (size 8/0 – 10/0) was used at the bottom of the trace to target larger fishes, while a smaller circle hook (size 4/0 – 6/0) was used at the top of the trace to target smaller fishes, thus targeting a representative sample. Bait was standardised to pilchard *Sardinops sagax* and squid *Loligo vulgaris reynaudii*.

To reduce the effects of barotrauma, fishing was limited to a maximum depth of 30 m and great care was taken to return each fish to the water as quickly as possible while minimising the stress imposed on the animal. The person processing the catch placed each fish caught on a vinyl stretcher with a built-in stainless-steel ruler ensuring that the surface was always wet. The stretcher was also cushioned from the deck by a 150 mm-thick sponge. A damp cloth was then placed over the fish, taking care to cover the eyes so that stress and sun exposure was minimised.

The hook was then removed, but if a fish was hooked in the oesophagus or gills, or if the hook was difficult to remove, the trace-line was cut leaving the hook in the fish and this was noted in the tag-release data. Attempting to get the hook out of the fish increases the chances of mortality, and not doing so has been found to minimise mortality in fishes which have ingested hooks (Butcher *et al.* 2010). Numerous recaptures of tagged fishes where the hook had been left behind indicated that these hooks, having no barb, were often shed and caused no obvious harm to the fish. If barotrauma was evident, the swim bladder was deflated or *vented* (St John and Syers 2005) by removing a scale and inserting a 12-gauge hypodermic needle through the body wall into the swim bladder at the point where the swim bladder is in close contact with the body wall (Buxton 1990, Bennett 2007). In large species such as the poenskop *Cymatoceps nasutus*, where venting is difficult, fish were released using a weighted down-rigger system (Mann and Kyle 2009, Sumpton *et al*. 2010).

For all fishes caught, the species, its fork length (FL) or total length (TL) to the nearest mm, and GPS coordinates were recorded providing a measure of abundance, length frequencies and catch composition. Selected species of fish and only those individuals of 300 mm or greater were tagged with plastic dart tags, supplied by the ORI Tagging Project and manufactured by Hallprint Pty Ltd, Australia (see Chp 5).

3.2.2 Data analysis

Relative abundance, in terms of number and mass of fishes was estimated from a CPUE index in the form of fish.angler-hour^{-1} and kg.angler-hour^{-1} respectively. This was done for overall catches and then separately for the eight most abundant species. For each of these species, length frequency analysis was used to compare the size structure of populations between the four study sites and between areas (exploited vs. no-take area with data from sites pooled). It was assumed that CPUE was related to relative abundance and takes the form CPUE = Nq, where N is abundance and q is the catchability factor that mediates the relationship (Bennett and Attwood 1991).

The use of CPUE in this way has been criticised (Maunder *et al.* 2006) as changes in abundance, fishing gear and cyclical environmental conditions may indirectly affect q (Bennett and Attwood 1991). In this study, each fish was assumed to have an equal catchability irrespective of abundance as large, rapid changes in abundance that might affect catchability were not known to occur. Fishing gear, bait and the skill level of anglers was kept as constant as possible throughout the study. Additionally, the study duration (approximately four years) was sufficient time to mitigate short-term environmental fluctuations. CPUE, calculated in this manner, has been used in similar long-term monitoring studies of MPAs (Bennett and Attwood 1991, Cowley *et al*. 2002, Attwood 2003, Götz *et al*. 2009). Bennett *et al*. (2009) optimised an offshore boat-based sampling protocol for the Tsitsikamma National Park and found that controlled angling, in which CPUE was used to measure fish abundance, was effective in sampling line-fish species. Biomass was calculated by converting fish lengths to weights using published length-weight regressions (Mann 2000) and from FishBase (Froese and Pauly 2010).

All univariate data was tested for normality using a Shapiro-Wilk test. Data deviating from a normal distribution and heteroscedastic data (unequal variance) were initially transformed to meet the assumptions of parametric testing procedures. Data of this nature was mostly right skewed, having many zero values. A square root transformation was applied before conducting t-tests and a log-transformation before simple linear regression analysis (Zar 1999). However, transformations had only limited success in achieving normality and equal variance but still had no effect on the outcome of the statistical procedure. That is, non-parametric testing on heteroscedastic data produced the same outcome as parametric testing on transformed, homoscedastic data. Therefore all univariate transformations were abandoned and non-parametric procedures were used for all data which did not meet the assumptions of normality and equal variance.

SigmaPlot (v11), Primer (v6), Genstat Discovery (v7) and Microsoft Excel (2003) were used for statistical procedures. A significance level (α) of 0.05 was used for all tests. Dispersion about the mean is reported as standard deviation (SD) which is suitable for a comparison of samples.

Overall abundance and biomass of fishes

A non-metric, multi-dimensional scaling (MDS) analysis, based on the Bray-Curtis similarity coefficient, was conducted using Primer (v6) (Clarke and Warwick 2001). This was to identify whether replicates at each of the four study sites – Casino, Mnyameni, Mtentu and Mkambati – displayed any multivariate grouping or separation in terms of overall CPUE (fish.angler-hour^{-1}).

To evaluate whether the Pondoland no-take area is achieving its objective of biodiversity conservation in terms of rebuilding depleted line-fish stocks, a spatio-temporal comparison was made between the no-take and adjacent exploited area. A spatial comparison, with data from all years pooled, was made using a non-parametric Mann-Whitney Rank Sum Test. To examine whether these spatial differences were linked to the prohibition of fishing with the proclamation of the no-take area, a time-series analysis, using simple linear regression, was used to detect changes over time. For the time-series analysis, data was aggregated for each 2 km^2 study site visit, equating to three fishing stations in most cases.

Zar (1999) warns against taking the mean Y at each value of X for linear regression because information is discarded. However, both averaging and using all data produced similar results, but the method which averages Y was simpler to display graphically.

Key species

Descriptive analysis of the catch data found that eight species accounted for 90% of the total catch and warranted further investigation. All these species except the Natal seacatfish *Galeichthys trowi* are important recreational and commercial line-fish in KZN and the Eastern Cape. The SIMilarity PERcentages (SIMPER) procedure, available in Primer (v6), was used to verify the species selection. Prior to analysis, a square root transformation was applied to lightly down-weight the contributions made by highly abundant species.

3.3 RESULTS

During four years of sampling, between April 2006 and January 2010, 16 field trips were conducted, taking place in January, April, July and October of each year. Five hundred and fifty-three angler-hours were spent on 241 fishing events with 286 angler-hours being spent in the exploited area and 267 angler-hours being spent in the no-take area. The maximum effort expended on an event was 6.33 angler-hours and the minimum was 0.33 angler-hours, while the average was 2.99 angler-hours. That is, the average angling event lasted one hour with three anglers fishing.

Catch composition

A total of 3720 fishes were caught between April 2006 and January 2010, comprising 49 species from 19 families (Appendix 1, Fig 3.1). The most speciose families were Sparidae (14 species) and Serranidae (5 species). Of the 3720 fishes caught, 1482 individuals from 43 species were caught in the exploited area and 2238 individuals from 40 species were caught in the no-take area. Teleosts accounted for 38 species and elasmobranchs for 11 species.

Eight species accounted for 90% of all fishes caught and 76% of the overall biomass. These species, in order of numerical abundance, were slinger *Chrysoblephus puniceus*, Natal seacatfish *Galeichthys trowi*, scotsman *Polysteganus praeorbitalis*, blue emperor *Lethrinus nebulosus*, yellowbelly rockcod *Epinephelus marginatus*, catface rockcod *Epinephelus andersoni*, halfmoon rockcod *Epinephelus rivulatus* and poenskop *Cymatoceps nasutus*.

A multivariate analysis of the catch data using the SIMPER procedure in Primer (v6) confirmed the dominance of these species. The eight species selected for further analysis were good discriminating species, and accounted for 66% of the separation between the replicates in the no-take area and those in the exploited area. In other words, these eight species were influenced the most by the prohibition of fishing and warranted further investigation.

In terms of catch composition, there was a greater abundance and biomass of slinger, scotsman and yellowbelly rockcod in the no-take area, while blue emperor, catface rockcod and halfmoon rockcod had a higher abundance and biomass in the exploited area. Natal seacatfish and poenskop exhibited only small differences between the two areas. All species not included in the top eight were categorised as “other”, and in terms of abundance and biomass, were greater in the exploited area than in the no-take area (Fig 3.1a, b).

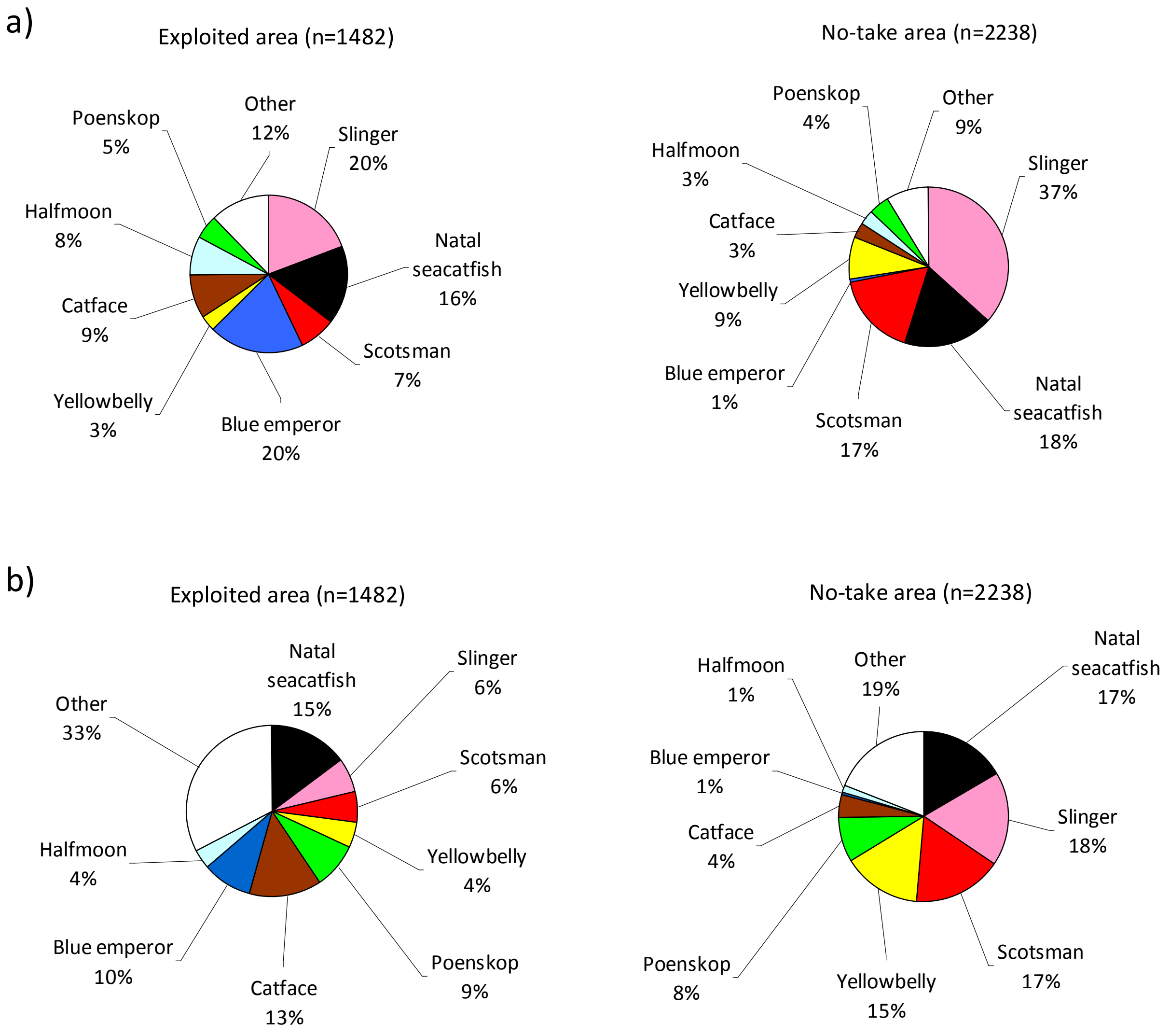


Figure 3.1. Catch composition in the exploited and no-take areas of the Pondoland MPA between April 2006 and January 2010. Percentages are given in terms of (a) numbers and (b) biomass (kg) of fish caught. Yellowbelly = yellowbelly rockcod, Catface = catface rockcod, Halfmoon = halfmoon rockcod.

Overall abundance and biomass of fishes

A multivariate analysis of the CPUE data using the MDS procedure showed a grouping of replicates from each of the four sites – Casino, Mnyameni, Mtentu and Mkambati (Fig 3.2). Furthermore, replicates grouped at a lower resolution, showing a considerable separation between the no-take (Mtentu and Mkambati) and adjacent exploited area (Casino and Mnyameni). The separation between the two areas was considered sufficient to pool data from the two sites within each area for further analysis.

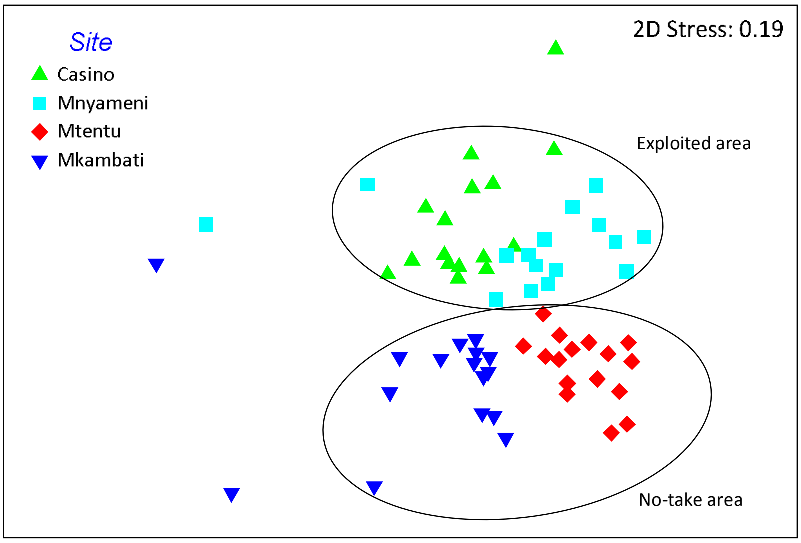


Figure 3.2. Non-metric, multi-dimensional scaling (MDS) analysis of square root transformed fish community data (fish.angler-hour^{-1}) recorded at the four Pondoland MPA study sites.

The overall CPUE in the no-take area (8 fish.angler-hour^{-1}) was significantly (P<0.001) greater than in the exploited area (4.5 fish.angler-hour^{-1}) (Fig 3.3). The overall CPUE in terms of weight was also significantly (P<0.001) greater in the no-take area (9.7 kg.angler-hour^{-1}) than in the exploited area (4.7 kg.angler-hour^{-1}).

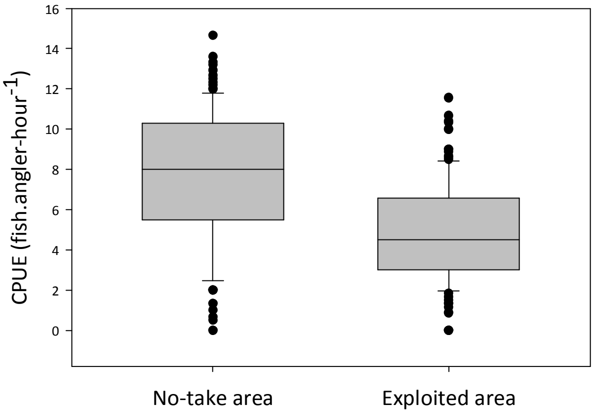


Figure 3.3. A comparison of the CPUE (fish.angler-hour^{-1}) recorded in the exploited and no-take areas between April 2006 and January 2010. Midline marks the median, box indicates the 25th and 75th percentiles, whiskers indicate the 10th and 90th percentiles and dots are outliers.

The overall CPUE revealed a steady increase over the study period, between April 2006 and January 2010 (Fig 3.4), in both the no-take (39%) and exploited areas (43%). However, a linear regression analysis of the two trends indicated that only the rising trend in the no-take area was significant

($P<0.001$). A time-series analysis of overall CPUE in terms of weight showed increases in the no-take (35%) and exploited areas (24%) but neither trend was significant.

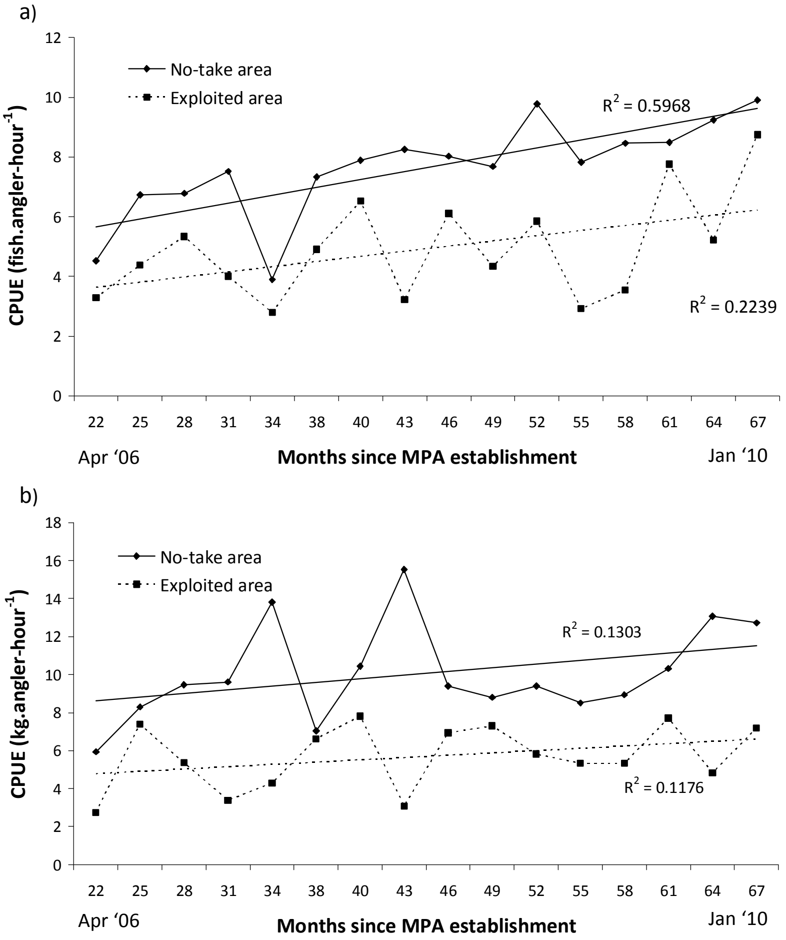


Figure 3.4. Temporal trends in overall CPUE in the no-take and exploited area between April 2006 and January 2010 beginning 22 months after establishment of the Pondoland MPA, a) fish.angler-hour^{-1} and b) kg.angler-hour^{-1}.

Seasonality

CPUE was highest in spring (October) in terms of fish numbers, after which it declined slightly in the summer (January), was lowest in autumn (April) and increased again in the winter (July) (Table 3.1).

Table 3.1. Seasonal variation in CPUE between April 2006 and January 2010 with data from all years pooled. Data presented as mean CPUE in terms of fish numbers (fish.angler-hour^{-1}). n = number of fishing events, standard deviation included in brackets.

	N	Summer	Autumn	Winter	Spring
No-take	116	8.30 (3.19)	5.92 (3.35)	7.48 (3.36)	8.38 (3.14)
Exploited	125	4.42 (2.84)	3.78 (1.99)	5.16 (2.25)	5.66 (2.58)
Overall	241	6.26 (3.57)	4.85 (2.94)	6.34 (3.08)	6.87 (3.13)

Key species

Slinger *Chrysoblephus puniceus*

A total of 289 slinger was caught in the exploited area, while 819 were caught in the no-take area. Overall, the CPUE of slinger in the no-take area was 3-fold higher (P<0.001) than in the exploited area. Slinger CPUE showed a significant (P=0.016) upward trend in the no-take area during the four study years, but no significant (P=0.528) change was evident in the exploited area (Fig 3.5a, Table 3.2).

The overall mean size of slinger in the no-take area (285.9 mm FL) was 19% larger (P<0.001) than in the exploited area (241.1 mm FL) (Fig 3.5b). Furthermore, the mean size of slinger in the no-take area showed a significant (P=0.003) upward trend during the four years of study, but no significant (P=0.397) change was apparent in the exploited area (Fig 3.5c; Table 3.3). The mean size of slinger in both areas was above the minimum legal size (250 mm TL = 213 mm FL).

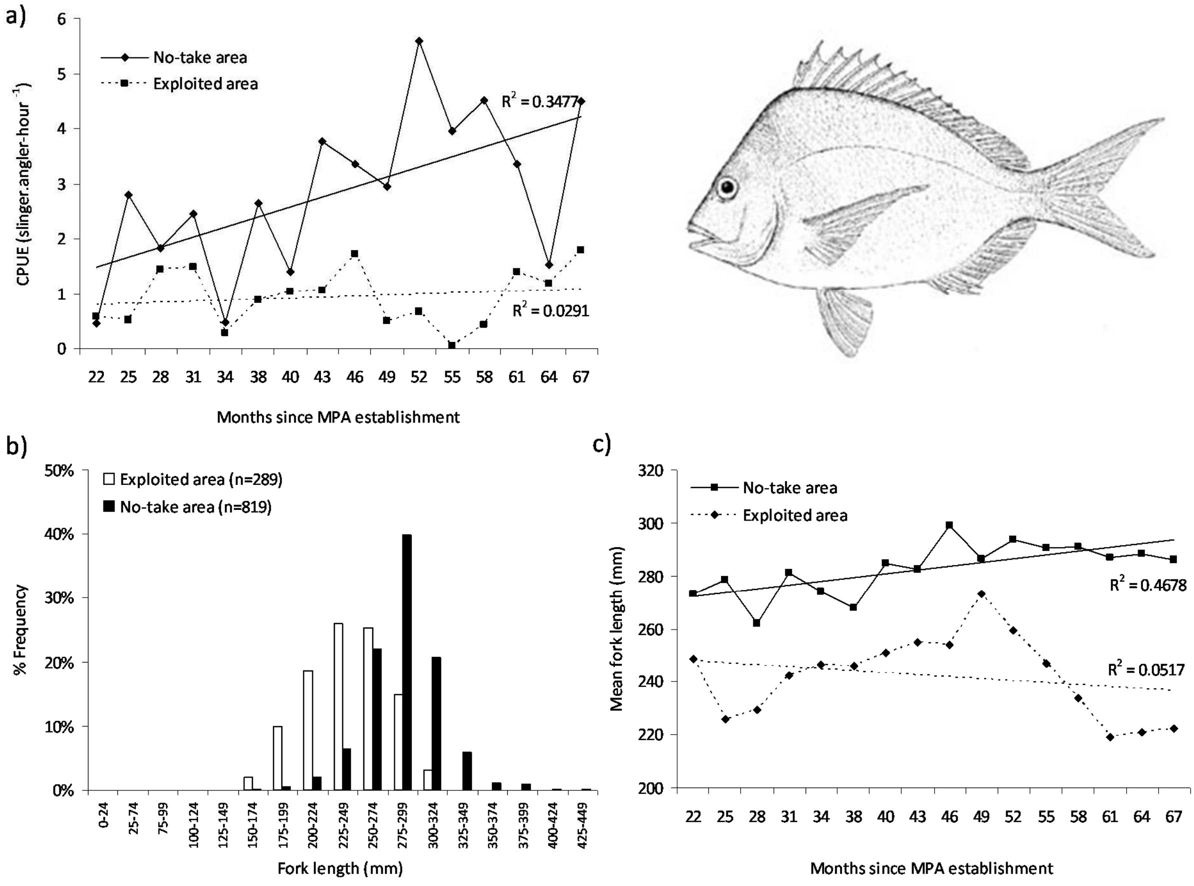


Figure 3.5. A comparison of slinger catches between the no-take and exploited areas with a) a time-series analysis of CPUE, b) an overall length frequency distribution and c) a time-series analysis of mean lengths.

Natal seacatfish *Galeichthys trowi*

A total of 236 Natal seacatfish was caught in the exploited area, while 408 were caught in the no-take area. Overall, the CPUE of Natal seacatfish in the no-take area was 1.8-fold higher (P=0.008) than in the exploited area. However, no significant changes were observed in Natal seacatfish CPUE in the no-take (P=0.365) or exploited areas (P=0.137) during the four study years (Fig 3.6a, Table 3.2).

The overall mean size of Natal seacatfish in the no-take area (407.9 mm FL) was not significantly (P=0.055) different from that in the exploited area (401.3 mm FL) (Fig 3.6b). There were also no significant changes in the mean size of Natal seacatfish in the no-take (P=0.161) or exploited area (P=0.365) during the four years of study (Fig 3.6c; Table 3.3). There is no minimum legal size limit for this species.

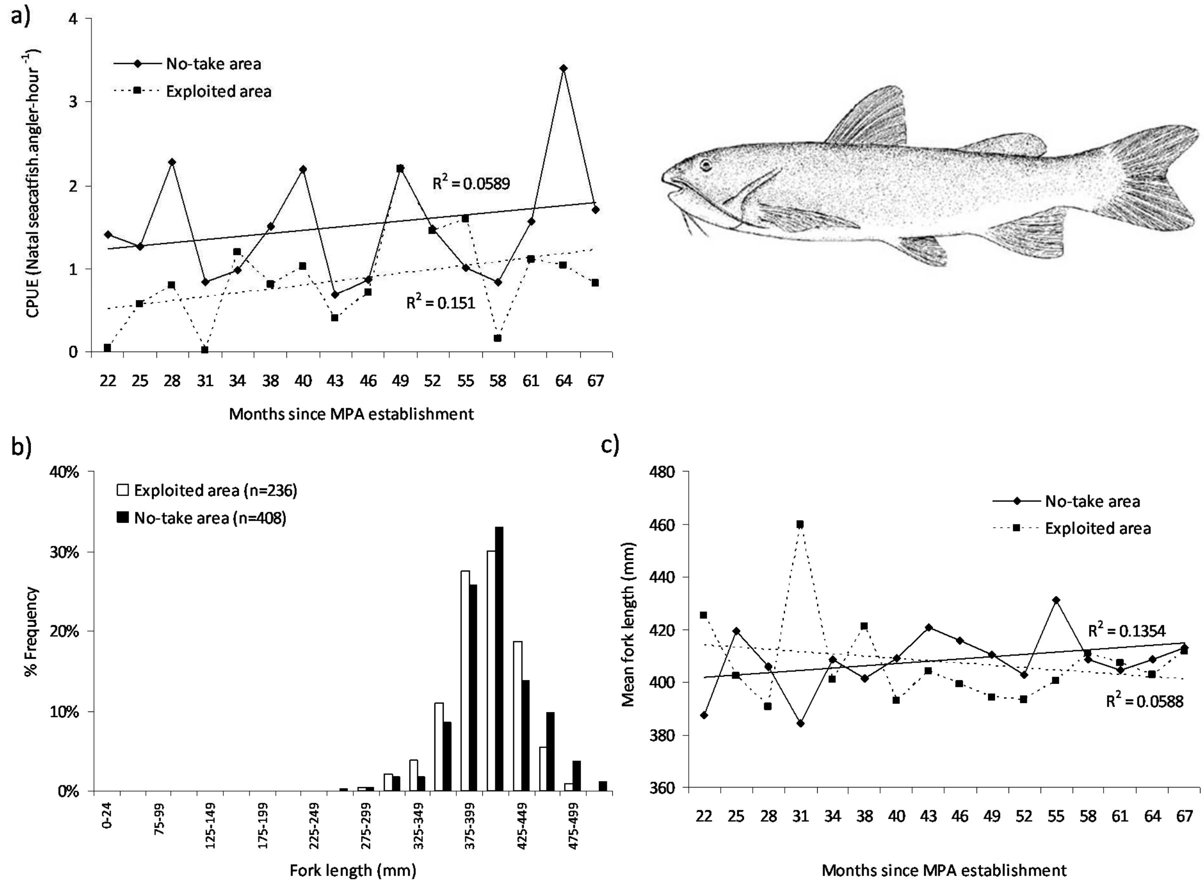


Figure 3.6. A comparison of Natal seacatfish catches between the no-take and exploited areas with a) a time-series analysis of CPUE, b) an overall length frequency distribution and c) a time-series analysis of mean lengths.

Scotsman *Polysteganus praeorbitalis*

A total of 110 scotsman was caught in the exploited area, while 375 were caught in the no-take area. Overall, the CPUE of scotsman in the no-take area was 3.9-fold higher ($P<0.001$) than in the exploited area. However, no significant changes were observed in scotsman CPUE in the no-take ($P=0.581$) or exploited areas ($P=0.727$) during the four study years (Fig 3.7a, Table 3.2).

The overall mean size of scotsman in the no-take area (405.5 mm FL) was 13% larger ($P<0.001$) than in the exploited area (358.4 mm FL) (Fig 3.7b). Furthermore, the mean size of scotsman in the no-take area showed a significant ($P=0.012$) upward trend during the four years of study, while no significant ($P=0.093$) change was apparent in the exploited area (Fig 3.7c; Table 3.3). The mean size of scotsman in both areas was above the minimum legal size (400 mm TL = 330 mm FL).

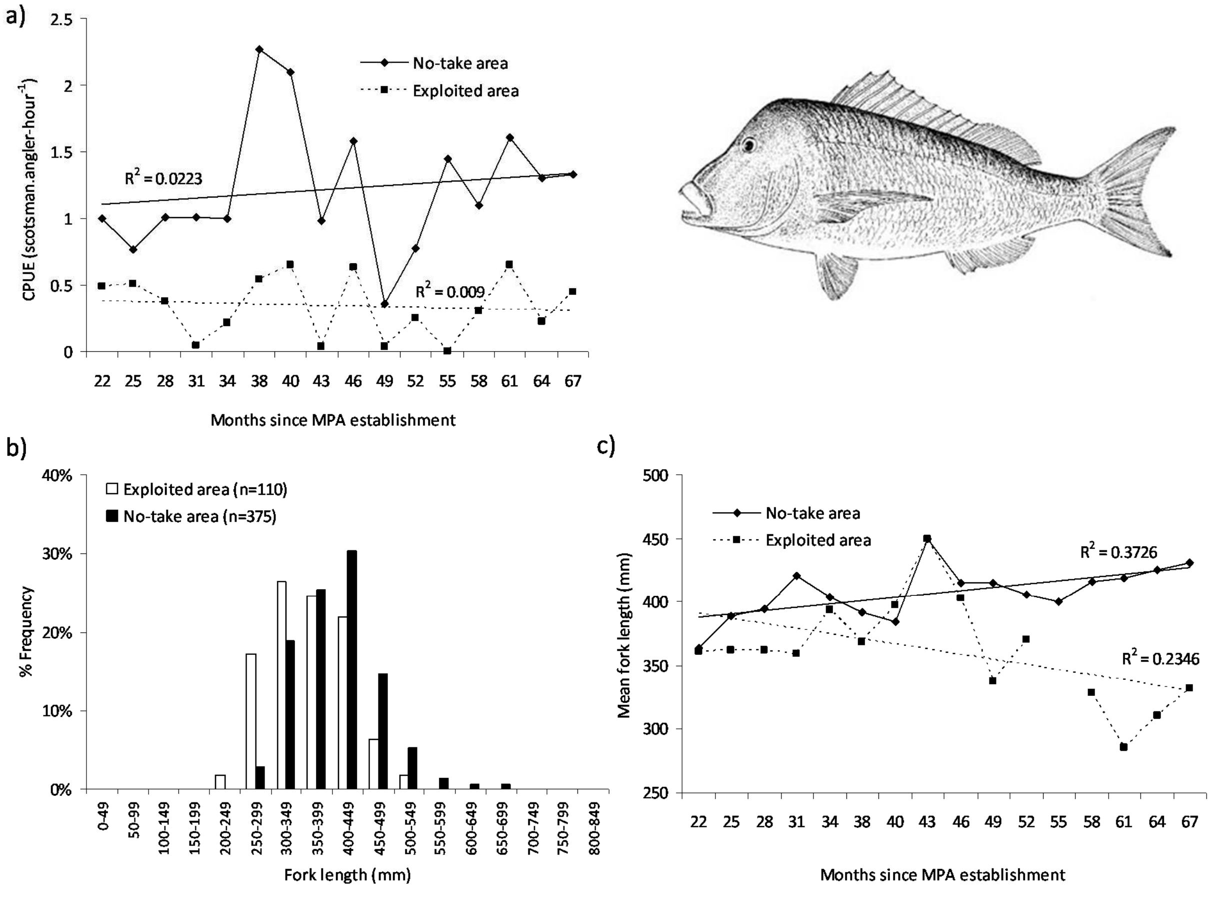


Figure 3.7. A comparison of scotsman catches between the no-take and exploited areas with a) a time-series analysis of CPUE, b) an overall length frequency distribution and c) a time-series analysis of mean lengths. Missing data points are the result of no fish being caught for that particular site visit.

Blue emperor *Lethrinus nebulosus*

A total of 290 blue emperor was caught in the exploited area, while only 22 were caught in the no-take area. Overall, the CPUE of blue emperor in the exploited area was 14-fold higher (P<0.001) than in the no-take area. No significant changes in blue emperor CPUE were observed in the no-take (P=0.108) or exploited areas (P=0.181) during the four study years (Fig 3.8a, Table 3.2).

The overall mean size of blue emperor in the no-take area (299.7 mm FL) was 18% larger (P<0.001) than in the exploited area (253.3 mm FL) (Fig 3.8b). However, there were no significant changes in the mean size of blue emperor in the no-take (P=0.872) or exploited areas (P=0.100) during the four study years (Fig 3.8c; Table 3.3). There is no minimum legal size limit for this species.

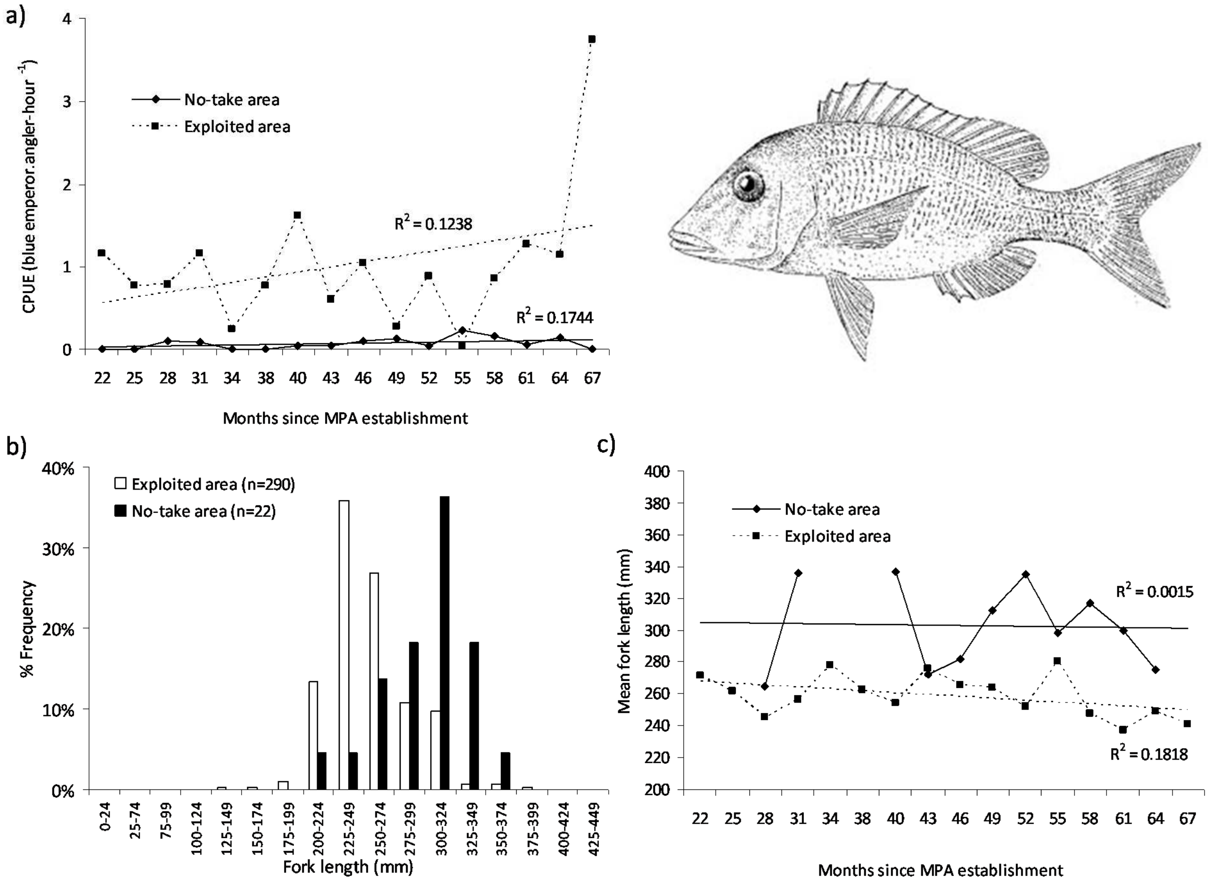


Figure 3.8. A comparison of blue emperor catches between the no-take and exploited areas with a) a time-series analysis of CPUE, b) an overall length frequency distribution and c) a time-series analysis of mean lengths. Missing data points are the result of no fish being caught for that particular site visit.

Yellowbelly rockcod *Epinephelus marginatus*

A total of 48 yellowbelly rockcod was caught in the exploited area, while 191 were caught in the no-take area. Overall, the CPUE of yellowbelly rockcod in the no-take area was 4-fold higher (P<0.001) than in the exploited area. However, no significant changes in yellowbelly rockcod CPUE were observed in the no-take (P=0.355) or exploited area (P=0.182) during the four study years (Fig 3.9a, Table 3.2).

The overall mean size of yellowbelly rockcod in the no-take area (490.9 mm TL) was 11% larger (P=0.001) than in the exploited area (440.9 mm TL) (Fig 3.9b). Furthermore, the mean size of yellowbelly rockcod in the exploited area showed a significant downward trend (P=0.047) during the four study years, while no significant (P=0.354) trend was apparent in the no-take area (Fig 3.9c; Table 3.3). The mean size in both areas was lower than the minimum legal size (600 mm TL).

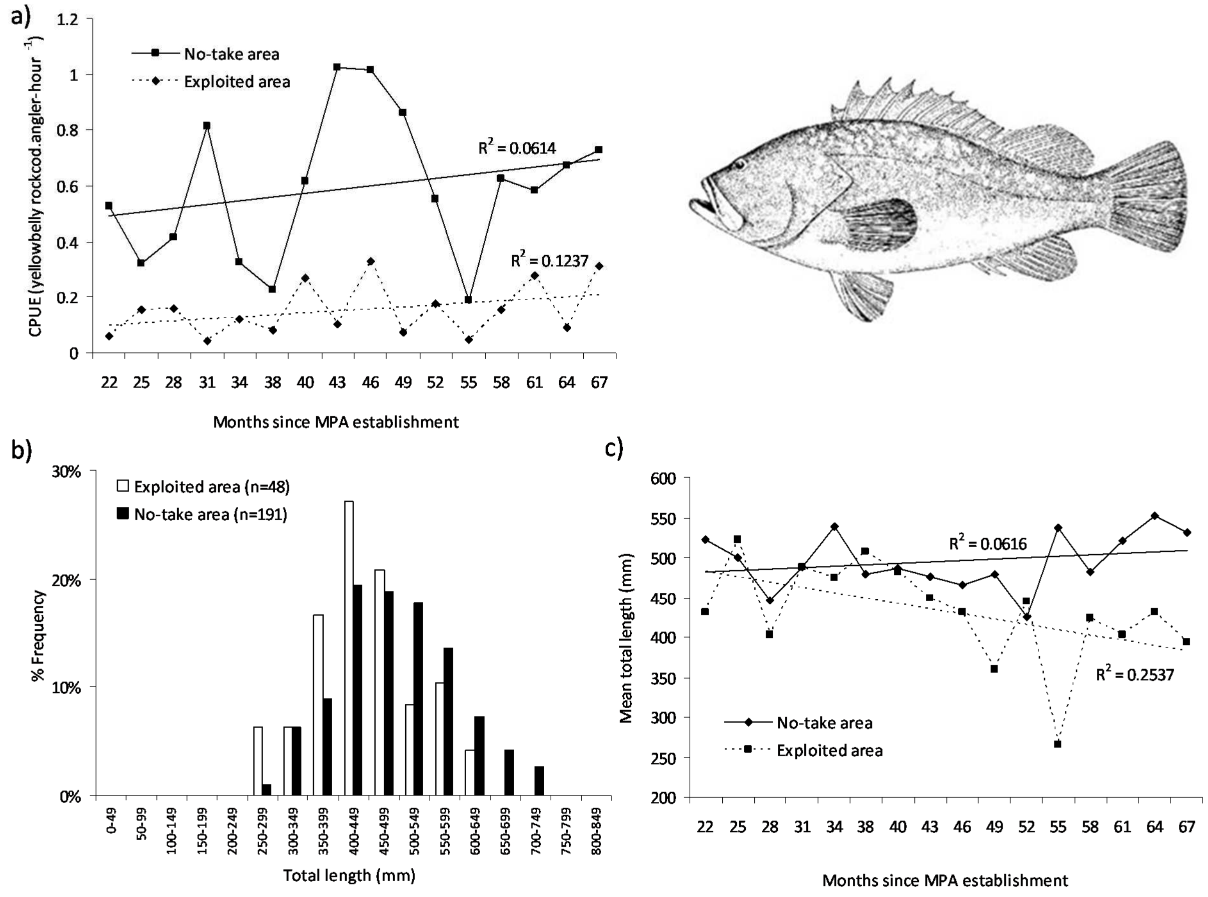


Figure 3.9. A comparison of yellowbelly rockcod catches between the no-take and exploited areas with a) a time-series analysis of CPUE, b) an overall length frequency distribution and c) a time-series analysis of mean lengths.

Catface rockcod *Epinephelus andersoni*

A total of 135 catface rockcod was caught in the exploited area, while only 70 were caught in the no-take area. Overall, the CPUE of catface rockcod in the exploited area was 2-fold higher (P=0.006) than in the no-take area. However, no significant changes in catface rockcod CPUE were observed in the no-take (P=0.959) or exploited areas (P=0.640) during the four study years (Fig 3.10a, Table 3.2).

The overall mean size of catface rockcod in the no-take area (518.4 mm TL) was not significantly (P=0.282) different from that in the exploited area (501.7 mm TL) (Fig 3.10b). However, the mean size of catface rockcod in the exploited area showed a significant downward trend (P=0.005) during the four study years, but no significant (P=0.068) trend was apparent in the no-take area (Fig 3.10c; Table 3.3). The mean size in both areas was greater than the minimum legal size (500 mm TL).

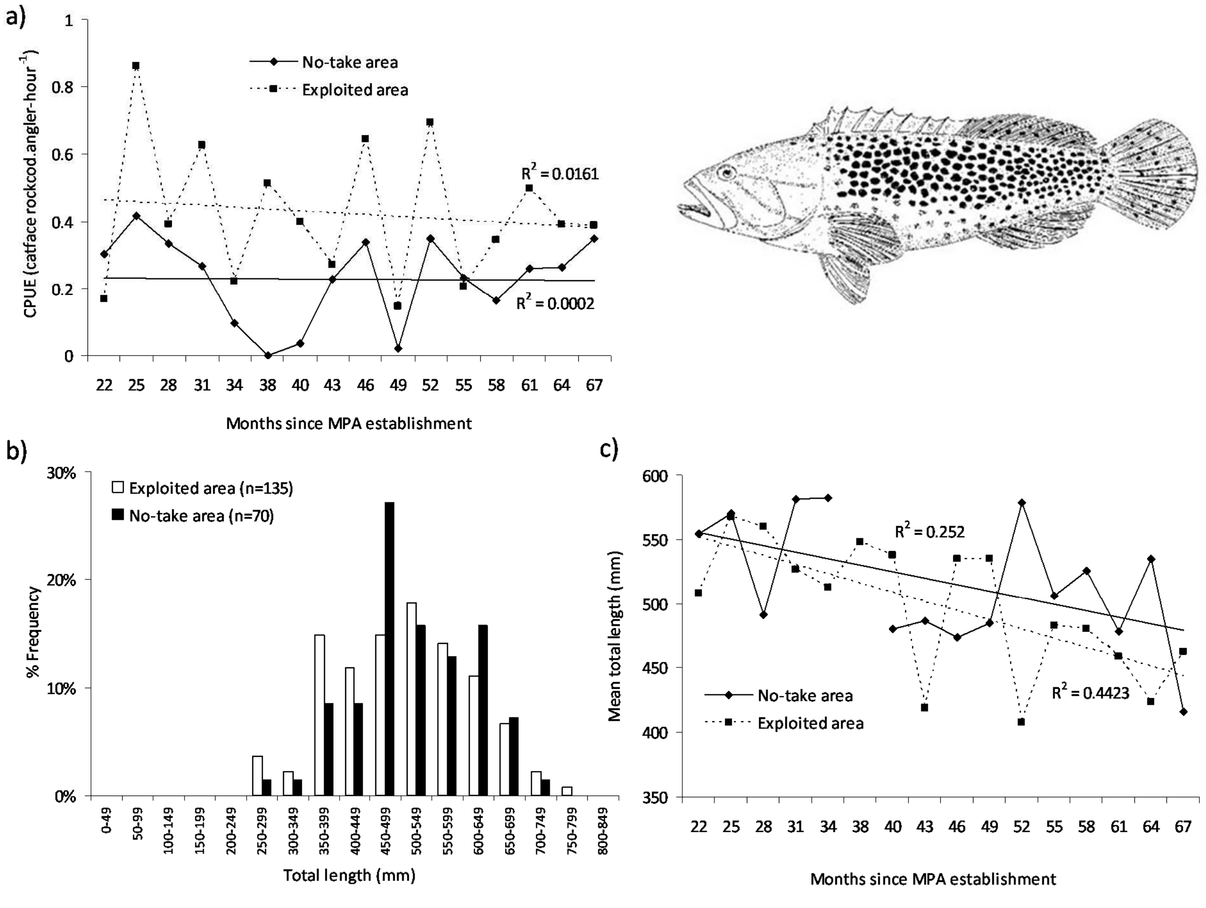


Figure 3.10. A comparison of catface rockcod catches between the no-take and exploited areas with a) a time-series analysis of CPUE, b) an overall length frequency distribution and c) a time-series analysis of mean lengths. Missing data points are the result of no fish being caught for that particular site visit.

Halfmoon rockcod *Epinephelus rivulatus*

A total of 117 halfmoon rockcod was caught in the exploited area, while 65 were caught in the no-take area. Overall, the CPUE of halfmoon rockcod in the exploited area was 1.5-fold higher (P=0.004) than in the no-take area. No significant changes were observed in halfmoon rockcod CPUE in the no-take (P=0.555) or exploited areas (P=0.743) during the four study years (Fig 3.11a, Table 3.2).

The overall mean size of halfmoon rockcod (326.9 mm TL) in the no-take area, was not significantly (P=0.198) different from that in the exploited area (320.3 mm TL), but there were a higher frequency of the larger size classes in the no-take area (Fig 3.11b). No significant changes were observed in the mean size of halfmoon rockcod in the no-take (P=0.202) or exploited areas (P=0.959) during the four study years (Fig 3.11c; Table 3.3). There is no minimum legal size limit for this species.

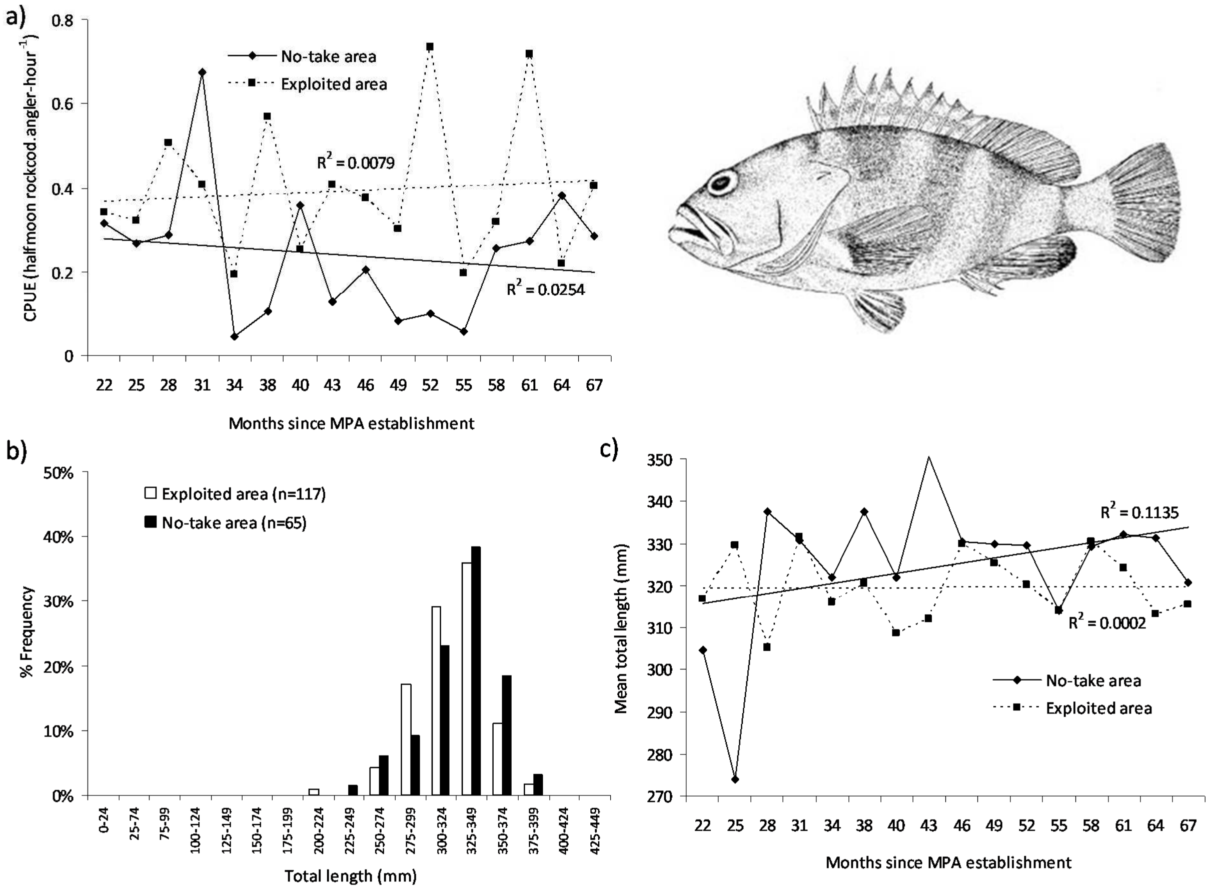


Figure 3.11. A comparison of halfmoon rockcod catches between the no-take and exploited areas with a) a time-series analysis of CPUE, b) an overall length frequency distribution and c) a time-series analysis of mean lengths.

Poenskop *Cymatoceps nasutus*

A total of 76 poenskop was caught in the exploited area, while 94 were caught in the no-take area. Overall, the CPUE of poenskop in the exploited area was not significantly (P=0.121) different from that in the no-take area. There were also no significant changes in poenskop CPUE in the no-take (P=0.548) or exploited areas (P=0.366) during the four study years, (Fig 3.12a, Table 3.2).

The overall mean size of poenskop in the no-take area (419.3 mm FL) was 10% larger (P=0.012) than in the exploited area (380.5 mm FL) (Fig 3.12b). No significant changes were observed in the mean size of poenskop in the no-take (P=0.401) or exploited areas (P=0.661) during the four study years (Fig 3.12c; Table 3.3). The mean size in both areas was smaller than the minimum legal size limit (500 mm TL = 445 mm FL).

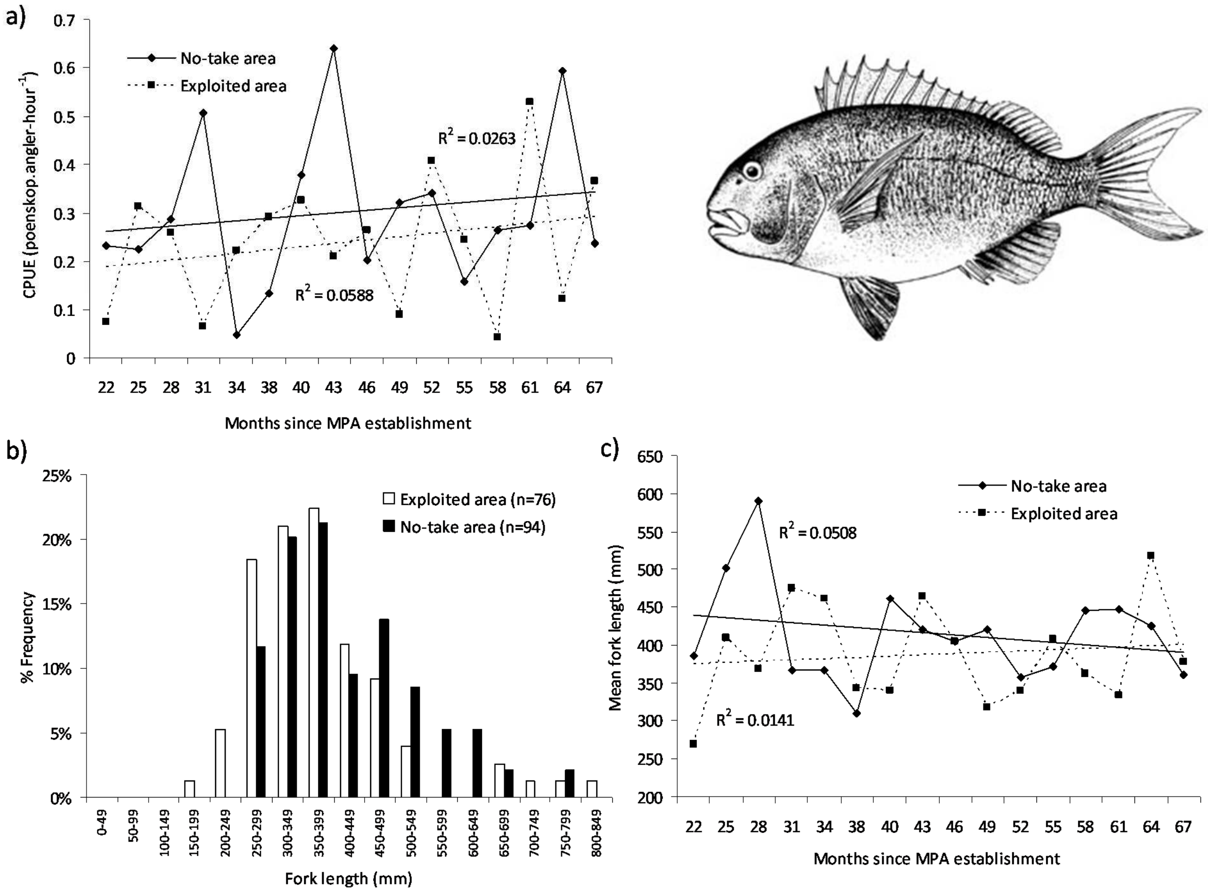


Figure 3.12. A comparison of poenskop catches between the no-take and exploited areas with a) a time-series analysis of CPUE, b) an overall length frequency distribution and c) a time-series analysis of mean lengths.

Table 3.2. A spatio-temporal summary of species–specific CPUE (fish.angler-hour^{-1}) for important line-fishes caught in the Pondoland no-take area and adjacent exploited area between April 2006 and January 2010. Values are means ± standard deviation, * indicates significance ($P<0.05$), n = number of fishing events, ↗ **–** increasing trend, ↘ – decreasing trend.

	SPATIAL CPUE				TEMPORAL CPUE	
Species	**Exploited area**		**No-take area**	**Statistics**	**Exploited area**	**No-take area**
Slinger	0.92 ± 1.26	<	2.73 ± 3.00	**n = 241, P < 0.001***	↗ ($F_{[1,14]}$ = 0.419, P = 0.528)	↗ **($F_{[1,14]}$ = 7.463, P = 0.016)***
Natal seacatfish	0.86 ± 1.22	<	1.53 ± 1.87	**n = 241, P = 0.008***	↗ ($F_{[1,14]}$ = 2.490, P = 0.137)	↗ ($F_{[1,14]}$ = 0.877, P = 0.365)
Scotsman	0.32 ± 0.56	<	1.24 ± 1.55	**n = 241, P < 0.001***	↘ ($F_{[1,14]}$ = 0.127, P = 0.727)	↗ ($F_{[1,14]}$ = 0.319, P = 0.581)
Blue emperor	0.98 ± 1.17	>	0.07 ± 0.22	**n = 241, P < 0.001***	↗ ($F_{[1,14]}$ = 1.979, P = 0.181)	↗ ($F_{[1,14]}$ = 2.957, P = 0.108)
Yellowbelly rockcod	0.14 ± 0.26	<	0.57 ± 0.65	**n = 241, P < 0.001***	↗ ($F_{[1,14]}$ = 1.976, P = 0.182)	↗ ($F_{[1,14]}$ = 0.915, P = 0.355)
Catface rockcod	0.43 ± 0.54	>	0.22 ± 0.34	**n = 241, P = 0.006***	↘ ($F_{[1,14]}$ = 0.228, P = 0.640)	↘ ($F_{[1,14]}$ = 0.003, P = 0.959)
Halfmoon rockcod	0.38 ± 0.46	>	0.25 ± 0.41	**n = 241, P = 0.004***	↗ ($F_{[1,14]}$ = 0.112, P = 0.743)	↘ ($F_{[1,14]}$ = 0.365, P = 0.555)
Poenskop	0.23 ± 0.40	<	0.29 ± 0.44	n = 241, P = 0.121	↗ ($F_{[1,14]}$ = 0.875, P = 0.366)	↗ ($F_{[1,14]}$ = 0.379, P = 0.548)

Table 3.3. A spatio-temporal summary of species–specific mean lengths (mm) for important line-fishes caught in the Pondoland no-take area and adjacent exploited area between April 2006 and January 2010. Values are means ± standard deviation, * indicates significance ($P<0.05$), n = number of individual fish, ↗ **–** increasing trend, ↘ – decreasing trend.

	SPATIAL LENGTHS				TEMPORAL LENGTHS	
Species	**Exploited area**		**No-take area**	**Statistics**	**Exploited area**	**No-take area**
Slinger (FL)	241.1 ± 32.94	<	285.9 ± 30.92	**n = 1108, P < 0.001***	↘ ($F_{[1,14]}$ = 0.763, P = 0.397)	↗ **($F_{[1,14]}$ = 12.308, P = 0.003)***
Natal seacatfish (FL)	401.3 ± 33.43	<	407.9 ± 36.96	n = 644, P = 0.055	↘ ($F_{[1,14]}$ = 0.875, P = 0.365)	↗ ($F_{[1,14]}$ = 2.193, P = 0.161)
Scotsman (FL)	358.4 ± 63.26	<	405.5 ± 65.15	**n = 485, P < 0.001***	↘ ($F_{[1,13]}$ = 3.277, P = 0.093)	↗ **($F_{[1,14]}$ = 8.313, P = 0.012)***
Blue emperor (FL)	253.3 ± 32.86	<	299.7 ± 35.40	**n = 312, P < 0.001***	↘ ($F_{[1,14]}$ = 3.110, P = 0.100)	↘ ($F_{[1,9]}$ = 0.027, P = 0.872)
Yellowbelly rockcod (TL)	440.9 ± 87.32	<	490.9 ± 97.97	**n = 239, P = 0.001***	↘ **($F_{[1,14]}$ = 4.760, P = 0.047)***	↗ ($F_{[1,14]}$ = 0.919, P = 0.354)
Catface rockcod (TL)	501.7 ± 109.14	<	518.4 ± 96.38	n = 205, P = 0.282	↘ **($F_{[1,14]}$ = 11.104, P = 0.005)***	↘ ($F_{[1,13]}$ = 3.973, P = 0.068)
Halfmoon rockcod (TL)	320.3 ± 28.45	<	326.9 ± 30.73	n = 182, P = 0.198	↗ ($F_{[1,14]}$ = 0.003, P = 0.959)	↗ ($F_{[1,14]}$ = 1.792, P = 0.202)
Poenskop (FL)	380.5 ± 124.30	<	419.3 ± 117.19	**n = 170, P = 0.012***	↗ ($F_{[1,14]}$ = 0.201, P = 0.661)	↘ ($F_{[1,14]}$ = 0.750, P = 0.401)

3.4 DISCUSSION

Empirical proof of the "reserve effects" associated with areas closed to fishing has been established through many studies. These can be categorised as direct effects, where there are changes in abundance and size of individuals (Palumbi 2001) and indirect effects, when changes in the community take place (Babcock *et al*. 1999, Friedlander *et al*. 2003, Graham *et al*. 2003, Götz 2006). Collectively, these reserve effects have been documented across the globe in tropical and temperate MPAs (Halpern 2003). However, critics of MPAs suggest, correctly, that fishery closures have the potential to concentrate effort elsewhere (Parrish 1999), which may cause an accelerated decrease in biomass and number of target species in open areas adjacent to no-take areas.

Equilibrium and optimum carrying capacity in fish populations should be the goal of MPA managers especially where biodiversity preservation and the concomitant provision of ecosystem services are the primary objectives. Total recovery of fish populations after prohibition of fishing is, however, dependent on a complex assemblage of ecological factors. These "recovery factors" mostly relate to population growth suggesting that recovery is a cumulative process (Babcock *et al.* 2010), beginning with the initial size of the population (number of fish) at the time of fishery closure and the intrinsic rate of natural increase (*r*).

Jennings (2000) elaborates on the following factors that may influence the recovery of depleted fish populations once fishing has been stopped. Spawning individuals in a heavily depleted fish population may compensate by producing more gametes than they would under normal population sizes. However, at low densities males may not be able to find females to mate with or broadcast spawning may be ineffective because of inadequate sperm concentration. In such cases, the overall egg production of the population would be greatly reduced and is referred to as depensation. After fertilised eggs are produced, variable patterns of connectivity and larval dispersal between sub-populations can influence the rates of recruitment (Carr and Reed 1993, Cushing 1996, Chambers and Trippel 1997, Allison *et al*. 1998). Once settled, fishes may move around in search of food and suitable habitat. These movements may take fishes into adjacent fished areas where they would be available to the fishery. While spill-over is desirable, a closed area, which is situated in an area of marginal habitat and food availability, may export biomass too quickly for recovery to take place within the closed area.

The effect of species longevity, competition, ecological succession and reserve size are also factors influencing the rate of recovery in areas closed to fishing (Denny *et al*. 2004, McClanahan and

Graham 2005, McClanahan *et al*. 2007, Babcock *et al*. 2010). The recovery trajectory is therefore not a steady increase, is difficult to predict and can vary vastly between studies. These factors are mostly associated with pre- and post-settlement life stages, and are consequently dependent on species-specific life-history parameters.

Halpern and Warner (2002) in a meta-analysis of 112 closed areas suggested that only 1 – 3 years was necessary for full recovery, reporting that time was found to be a non-significant factor. McClanahan and Graham (2005) pointed out that this is "biologically unrealistic unless rapid immigration into recently closed areas is the dominant process of recovery". In their own study, also using space-for-time substitution samples, McClanahan and Graham (2005) reported complete recovery (assumed by stability in fish populations) in biomass of tropical reef fishes after more than 20 years. Russ and Alcala (2004), using logistic modelling techniques, estimated that 40 years would be necessary for complete recovery of fish populations at a closed area in the Philippines. Polunin and Roberts (1993) reported direct effects in fish communities four years after protection from fishing while McClanahan and Kaunda-Arara (1996) measured an increase in lethrinids after three years following MPA establishment.

Space-for-time substitution is frequently used due to the difficulty in obtaining a long time-series but is not very useful for site-specific evaluations (McClanahan *et al.* 2009). True temporal analyses, in which a continuous time-series analysis is used, as used in this study, are more reliable than space-for-time substitution samples (Babcock *et al.* 2010). These authors assessed fish population recovery in multiple MPAs, reporting the average time for initial detection of direct effects in target species to be 5.13 years, while indirect effects took, on average, 13.1 years. These authors also reported only temporary equilibrium in populations, with some cases of instability in populations even after 40 years.

The Pondoland MPA

In Pondoland, the smaller Mkambati MPA existed prior to proclamation of the Pondoland MPA, but poaching took place due to poor law enforcement (Mann 1998). Two ski-boat launch-sites, situated at Port Edward and Mbotyi, are located within 26 km of the former Mkambati MPA. This distance would have limited outings to good weather-days, which are not commonplace on the Pondoland coast; however, a third launch-site at Msikaba is situated on the southern border of the former Mkambati MPA. Ski-boats regularly launched from this site during holiday periods and as they had easy access to the MPA, poaching probably took place quite frequently (B.Q. Mann, Oceanographic Research Institute, 2010, pers. comm.).

This study covered a four-year period commencing nearly two years after proclamation of the Pondoland MPA in June 2004. At this time, the area of protection was greatly increased and management was improved. Consequently, the effective exclusion of fishing is assumed to have begun with the proclamation of the Pondoland MPA.

Catch composition

The top eight species in this study, with the exception of Natal seacatfish, are important recreational and commercial line-fish in the KZN and Eastern Cape provinces of South Africa. The greater abundance and body size of targeted species in the no-take area is consistent with the direct effects, which are expected following a fisheries closure. Indirect or community effects resulting from changes in abundance and body size of predators are to be anticipated with a short amount of lag time (Babcock *et al.* 2010).

The number of species sampled in the no-take area (40) was similar to that in the exploited area (43). The small difference was due to a few species in the exploited area where only a few individuals were captured. Similarly, García-Rubies and Zabala (1990) counted 43 species in a closed area in the Mediterranean and 44 species in the adjacent fished area. The small difference in the number of species sampled by angling is therefore considered insignificant. The fish community is given greater focus in Chapter 4.

The top eight species, all semi-resident and strongly reef-associated, contributed 90% to the numerical catch composition. The relative contribution of “others” in terms of abundance and biomass was greater in the exploited area than in the no-take area. Collectively, target species accounted for a greater proportion of the community in the no-take area where fishing is prohibited (Fig 3.1a, b).

Not all of the target species were responsible for the greater abundance in the no-take area. Blue emperor, catface rockcod and halfmoon rockcod were more abundant in the exploited area. These differences between the two areas may, therefore, not be attributable directly to protection from fishing (Bennett and Attwood 1991) and require a further analysis of species-specific ecological factors (see Key species).

Overall abundance and biomass of fishes

The Pondoland no-take area contained a significantly higher relative abundance and biomass of fishes than the exploited area. This was in spite of a greater number of large flapnose houndsharks *Scylliogaleus quecketti* and a few large poenskop, which inflated the overall biomass in the exploited area. The relatively low catch rate in the exploited area provided enough time for fish to be processed by the fourth person, but the catch rate in the no-take area was very high, which often led to fishes being queued while others were being processed. This reduced the actual time that baited hooks were in the water. The catch rate, as an indicator of fish abundance, was always higher in the no-take area than in the exploited area but may have been even higher if no queuing took place.

Temporal changes in abundance and biomass may come from recruitment, redistribution, predation, and increased survivorship at age (leading to growth) due to reduced fishing mortality (Jennings 2000). Thus, temporal change in abundance and biomass may be calculated as follows:

Δ abundance = **recruitment + immigration – emigration – predation**

Δ biomass = **(recruitment + immigration – emigration – predation)** x individual growth

From these simple models, it follows that abundance drives biomass but that biomass has the potential to increase more rapidly than abundance when growth of individuals is factored in. This was not the case in Pondoland where abundance in the no-take area appeared to be increasing more rapidly than biomass (Fig 3.4). Possible reasons for this may be a combination of slow growth of many of the target species, the possibility of a northward migration to spawning grounds of larger adults of some species and external larval supply. Increased abundance and size of piscivorous predators in the no-take area may also result in greater predation on new recruits compared to that in the exploited area (Tupper and Juanes 1999, Heinlein *et al*. 2010). Nevertheless, an assessment of overall recovery is a vast generalisation as different species have greatly differing recovery factors (Roberts and Polunin 1993) and may respond to MPA protection very differently (Mosquera *et al*. 2000, Gell and Roberts 2003b). It should also be noted that, strictly speaking, samples taken at successive time intervals were not independent as evidenced by the high recapture rate of fishes discussed in Chp 5 (auto-correlation in residuals). It is acknowledged that this violation of the assumptions of parametric data analysis may have compromised the validity of the inferences derived from the time-series analysis.

Key species

CPUE is known to perform best as an index of abundance when results are determined for individual species (Richards *et al.* 1986) although still subject to the limitations of non-independence of samples in this study. Five of the eight most abundant species had a higher CPUE in the no-take area, of which four were significantly so. All top eight species were larger in the no-take area, five of which were significantly so. However, only one species showed a significant increase in CPUE during the study period while two species increased significantly in size within the no-take area and two decreased in size significantly in the exploited area. Although there were some spatial differences, trends were less convincing implying that disparities were evident before the proclamation of the no-take area.

The response of the following eight species to protection from fishing (reserve effects) and their contribution to enhancement of adjacent fisheries (regional effects) is investigated in the context of pre- and post-settlement factors (recruitment and movement).

Slinger *Chrysoblephus puniceus*

Slinger are a semi-resident, reef-associated species belonging to the sparid family. The species is endemic to the south-east coast of Africa, extending from southern Mozambique down to the Eastern Cape (Garratt 1993b), possibly as far as East London (B.Q. Mann, Oceanographic Research Institute, 2010, pers. comm.). It is the most important species in the KZN commercial boat-based line-fishery – contributing about 30% of the annual catch between 1986 and 1995 (Penney *et al.* 1999). To a lesser degree, this species is also important to the KZN recreational boat-based fishery. A complex life history including protogynous hermaphroditism makes this species vulnerable to over-fishing. Not surprisingly, this study strongly suggests that slinger have responded well to fishery closure in the Pondoland MPA.

Large-scale spawning of slinger is believed to occur in southern Mozambique and northern KZN where reproductively-active adults are found in abundance (Garratt 1993b). Moving south along the KZN coast, spawning tapers off and no reproductively active adults are known to occur along the southern KZN and Pondoland coast (Garratt 1993b). It is assumed that eggs and larvae of slinger move southward on the continental shelf with the inner edge of the Agulhas Current (Beckley 1993) and that recruitment of juveniles occurs predominantly along the coast south of Durban. Interestingly, newly recruited slinger, <50 mm TL, were first observed along the Pondoland coast during a diving survey undertaken between 2002 and 2003 (Mann *et al.* 2006), despite extensive previous diving surveys to locate the whereabouts of juveniles (P.A. Garratt, Two Oceans Aquarium, 2010, pers. comm.). Once recruited in the Pondoland area, it is apparent that there must be some northward movement of slinger to their adult spawning area north of Durban, although tagging data have shown them to be fairly resident (Garratt 1993b).

Recruitment of juvenile slinger into the no-take area is thus likely to take place from the north. Since the southern distributional limit of slinger is not far south of Pondoland, immigration of juveniles and sub-adults from the south is believed to be low. It is thus postulated that the increase in abundance and individual size of slinger in the no-take area is largely due to larval supply from the north and possibly increased survivorship due to protection (Polunin and Roberts 1993, Jennings 2000).

Another reason for the favourable response of slinger within the no-take area may be due to the initial population size, not in the no-take area but in the area of large-scale spawning to the north. In northern KZN, the establishment of the large St Lucia and Maputaland MPAs have prohibited the capture of bottom dwelling reef-fish species since 1979 and 1986 respectively (Garratt *et al.* 1993). It is postulated that this management intervention has protected part of the adult slinger population and may be why large catches of slinger by commercial and recreational anglers in KZN have been sustained (Penney *et al.* 1999).

The Pondoland no-take area is providing a refuge for slinger to increase in abundance and individual size and therefore achieves its primary objectives of localised biodiversity conservation and rebuilding of depleted line-fish stocks. However, to be successful as a fisheries management tool, a MPA must contribute to regional ecosystems (Palumbi 2001). The Pondoland no-take area clearly does not contribute slinger larvae to the meta-population, but it provides a refuge for recruits to grow large before potentially beginning a northward migration. The no-take area thus potentially achieves its secondary objective of enhancing adjacent fisheries by exporting adult fish biomass.

Natal seacatfish *Galeichthys trowi*

The taxonomy of the Natal seacatfish has only recently been established (Kulongowski 2009). The life-history characteristics are poorly understood and are thus assumed to be similar to *G. ater*, a closely related species occurring in the southern parts of the Eastern Cape (Tilney 1990). The Natal seacatfish occurs in KZN and the northern parts of the Eastern Cape and is caught in abundance mostly as by-catch (Mann 2000). It is not popular amongst recreational anglers due to its unpopular appearance and poisonous spines, and most are released as undesirable by-catch. The number of Natal seacatfish retained in the commercial boat-based line-fishery is, however, increasing due to sequential target switching as other more popular species decline (Penney *et al*. 1999, Mann 2000).

Natal seacatfish are also popular amongst crew on commercial ski-boats as they are good eating and are frequently given to the crew as “take home” fish. Nevertheless, it is unlikely that the stock of Natal seacatfish has been greatly affected by such fishing pressure.

Natal seacatfish do not appear to be responding in any way to the closure of fishing. This was anticipated, given the minimal effort targeted at this species. Natal seacatfish were found to be more abundant in the no-take area but if not linked to fishing pressure, the spatial difference is likely to be due to other ecological factors, which may affect other species similarly.

Not much is known concerning the reproduction of the Ariidae family, to which the Natal seacatfish belongs, except that they are paternal mouth brooders (Tilney 1990). Since there is no pelagic larval phase, juveniles do not disperse by drifting in currents. The spatial pattern must be due to post-settlement factors if not due to exploitation.

Differences in abundance are not always linked to exploitation (Bennett and Attwood 1991). Micro-scale habitat preference may lead new recruits to redistribute on reef systems (Jennings 2000). In this way, Natal seacatfish are patchily distributed, being closely associated with complex reef structure such as caves (pers. obs.). This may suggest that preferential habitat is located in the no-take area where Natal seacatfish were found to be more abundant.

Natal seacatfish are not currently a popular species but with the current trend of target switching, are highly likely to increase their contribution to overall catches, especially in the commercial sector. The Pondoland no-take area achieves its primary objective of biodiversity conservation, by protecting an abundance of Natal seacatfish that may otherwise be targeted in the future. As there is no pelagic larval phase, enhancement of adjacent fisheries will be limited to export of adults and sub-adults if Natal seacatfish in open areas experience increased fishing pressure in the future.

Scotsman *Polysteganus praeorbitalis*

One of the larger members of the sparid family, the scotsman is a semi-resident reef-associated predator. It is endemic to the south-east coast of Africa, occurring from Beira in northern Mozambique down to Algoa Bay on the south coast of South Africa (Smith and Heemstra 1986). Within this range, scotsman are most abundant on the lower KZN south coast and upper Eastern Cape coast (Garratt *et al.* 1994, Mann 2000) where it is highly esteemed by commercial and recreational boat-based anglers (van der Elst 1993). It is a relatively long-lived species, reaching a maximum age of more than 20 years (Mann *et al.* 2005) and is believed to be a protogynous hermaphrodite, therefore making it extremely vulnerable to over-fishing. CPUE of scotsman has dropped by 90% since 1934-41 along with significant decreases in mean size (Garratt *et al.* 1994). The stock is now considered to have collapsed as supported by a spawner-biomass-per-recruit study (Mann *et al.* 2005).

Scotsman were significantly more abundant and larger in the no-take area, where they were also increasing in mean size. During the four years of this study, the mean size of scotsman caught in the no-take area has risen above the observed size at first maturity (405 mm FL – Mann *et al*. 2005). This response appears, at this stage, to be limited to increased survivorship as no significant change in abundance was recorded.

Researchers (Garratt *et al.* 1994, Mann *et al.* 2005) have been unable to reliably determine reproductive and migratory habits of scotsman because of small sample sizes as a consequence of the few remaining individuals. Garratt *et al*. (1994) found only 13 scotsman in spawning condition, along the length of the KZN coast. Meta-population connectivity and movement of scotsman, and thus reasons for observed patterns in abundance, remain speculation.

This study of the Pondoland reefs only sampled in a depth range of 10 – 30 m – relatively shallow for scotsman, which can occur to depths of 120 m (Garratt *et al.* 1994). It is therefore possible that sampling did not detect significant changes which may have occurred in deeper water. According to Garratt *et al*. (1994), juvenile scotsman prefer shallower water between 10 – 30 m, while adult scotsman favour deeper water between 50 – 120 m. A range of sizes of scotsman, between 210 – 656 mm FL, were sampled in this study, but it is likely that adults move into deeper water and were thus not detected in this study. It also suggests that an increase in abundance due to recruitment should have been noticeable in the depth range of this study.

The Pondoland no-take area is providing a refuge to a greater amount of larger scotsman than the exploited area and therefore achieves its primary objective of biodiversity conservation. Furthermore, scotsman appear to be responding to no-take status by increased survivorship as indicated by the increase in mean size.

Considering the popularity of this species, surprisingly little is known regarding its life-history. Garratt *et al*. (1994), sampling along the KZN coast and into Mozambique, only obtained 375 specimens. Only 13 of these fish were mature or in the process of maturing. Similarly, Mann *et al*. (2005) were only able to obtain 187 scotsman in their study, 79% of which, were reproductively inactive.

Consequently, neither spawning season nor length at sexual maturity has been determined with any certainty. However, both studies found strong evidence of protogynous hermaphroditism and collected a greater proportion of large males on the KZN north coast, implying a northward movement of older fish. Although scotsman are not known to be common in the St Lucia and Maputaland MPAs in northern KZN (Chater *et al.* 1993), the current stock distribution should be interpreted with caution because of the critically depleted nature of this stock.

Northward movement is a trait believed to be characteristic of other sparids in the region, for example, poenskop (Buxton and Clarke 1989, Maggs 2009) and red steenbras *Petrus rupestris* (Brouwer 2002). Tag recaptures of scotsman support this idea (see Chp 5), implying export from the Pondoland no-take area. This could also explain why a significant increase in numbers of scotsman in the no-take area has not yet been recorded.

Blue emperor *Lethrinus nebulosus*

Blue emperor are a widely distributed lethrinid occurring throughout the Indo-West Pacific. It is a tropical species, abundant in Mozambique and KZN, extending as far south as Algoa Bay (Smith and Heemstra 1986). Blue emperor are becoming increasingly important in the South African line-fishery as other preferred species decline in number (Penney *et al.* 1999). It has become a very popular line-fish amongst commercial and recreational anglers and is occasionally taken by spear-fishers (van der Elst 1993, Mann 2000).

Blue emperor have been found to respond positively to area protection from fishing. In closed areas in Western Australia, blue emperor were found to be more abundant and of a larger mean size (Westera *et al*. 2003, Watson *et al*. 2007, Watson *et al*. 2009) inside a closed area relative to an adjacent fished area. Similarly, this species was larger in the Pondoland no-take area than in the exploited area, yet it was not possible to attribute this directly to protection from fishing. The reason for this is that blue emperor were substantially more abundant in the exploited area and did not show a response in either abundance or mean size since closure of the no-take area. Ecological effects or indirect community effects of protection from fishing may therefore offer an alternative explanation for the observed pattern in abundance.

The reported biological parameters of this species vary considerably throughout its distribution (Marriott *et al.* 2010) but little is known about the South African stock. The maximum size of blue emperor according to van der Elst (1993) is 750 mm with individuals reported to mature between 453 mm TL (Froese and Pauly 2010) and 600 mm (van der Elst 1993). Observation of commercial line-fish catches suggest that the majority of blue emperor caught in KZN are juvenile fish and most adults are only found in northern KZN and Mozambique (pers. obs; B.Q. Mann, Oceanographic Research Institute, 2010, *pers. comm.*). The largest individual caught in the no-take area was 355 mm FL, and in the exploited area, 375 mm FL. This is well below the length at sexual maturity, which precludes spawning on shallow reefs in the Pondoland area. Pelagic eggs and larvae of blue emperor are widely distributed with some having been recorded off KZN, presumably being distributed southwards inshore of the Agulhas Current (van der Elst 1993). If the supply of blue emperor to the Pondoland coast is primarily by egg and larval dispersal, the no-take and exploited areas would experience similar rates of recruitment, assuming similar environmental settlement cues in both areas. The large disparity between the two areas remains uncertain but may also be attributed to predation or movement.

With a greater abundance and individual size of potential predators such as scotsman and yellowbelly rockcod (see below) in the no-take area, it is possible that newly recruited blue emperor are a preferred prey item (Polunin and Roberts 1993). In this case, the low abundance of blue emperor would suggest a more natural community in this extremity of its distribution. Habitat preference is also a possible explanation, and potentially the habitat in the Pondoland MPA is not homogenous. As this species is known to form small shoals (van der Elst 1993), a patchy distribution of blue emperor with preference for certain habitat types, such as sandy edges on low relief reefs, may not have been adequately sampled.

Regarding the primary objective, it is difficult to evaluate whether an ecologically beneficial state lies in a greater or lesser abundance of blue emperor, given that biological parameters and trophic standing of the local stock are not well understood. The extremely limited quantity of individuals and assumed absence of sexually mature adults, strongly suggests that the Pondoland no-take area has a limited contribution of blue emperor larvae or adult biomass to the meta-population in adjacent areas. However, it is likely that the St Lucia and Maputaland MPAs in northern KZN provide protection to adults of this species thus highlighting the importance of a network of well-placed MPAs.

Yellowbelly rockcod *Epinephelus marginatus*

A large, wide ranging serranid, the yellowbelly rockcod occurs along, but is not limited to, the entire southern African coastline (Heemstra and Randall 1993). Yellowbelly rockcod were considered by Smith and Heemstra (1986) to be the most common serranid between KZN and Knysna. In this region, serranids are heavily targeted by all fishery sectors and are responsible for an increasing contribution to the total catch in the KZN commercial sector due to sequential target switching (Penney *et al.* 1999). This species is thus subject to heavy fishing pressure. It is a resident, long-lived species with strong evidence supporting protogynous hermaphroditism and thus it is highly vulnerable to over-fishing (Fennessy 2006). This author reported a decline in mean length of yellowbelly rockcod in KZN between the mid-1980s to the mid-1990s and considered the stock to be over-exploited with the spawner-biomass-per-recruit at 33% of pristine.

Ojeda-Martinez *et al*. (2007) reported increasing abundance of yellowbelly rockcod off Spain in an area closed to fishing and Francour (1994) also reported favourable responses to MPA protection in the Mediterranean. In the Pondoland no-take area, yellowbelly rockcod were significantly more abundant and of a larger mean size than in the adjacent exploited area. Since closure to fishing in the Pondoland no-take area, there has been a gradual decrease in mean size in the exploited area. This is possibly associated with a concentration of fishing effort in this region following fisheries closure as Parrish (1999) warned.

Varying levels of spawning take place along the KZN and Mozambique coasts, but it is suspected that spawning of yellowbelly rockcod south of KZN may be restricted by lower sea temperatures (Fennessy 2006). In the Pondoland study, the mean size in the no-take area remained below the size at 50% sexual maturity (620 mm TL) and none were greater than the size at sex change (860 mm TL) estimated by Fennessy (2006).

Despite yellowbelly rockcod being significantly larger in the no-take area than in the exploited area, even the largest individuals, sampled in the no-take area, were probably immature females, characteristic of the current KZN population (Fennessy 2006). This implies limited spawning potential of the Pondoland no-take area; but larger mature individuals including males possibly occur in deeper water of the no-take area due to ontogenetic habitat shift. It is believed that adults may also move northwards where the majority of spawning occurs and most juveniles recruiting into the no-take area are likely to originate from pelagic eggs spawned to the north (Beckley 1993).

The Pondoland no-take area is providing refuge for a greater abundance of large yellowbelly rockcod than the exploited area – achieving its primary objective of biodiversity conservation. With regard to the objective of fisheries management, no increase in abundance in the no-take area is yet apparent precluding any speculation regarding the export of adult fish biomass. Yet, it is likely that adult fish are moving out of the sample area onto deeper reefs and are potentially exporting larvae, albeit a limited amount.

Catface rockcod *Epinephelus andersoni*

The catface rockcod is a relatively fast-growing, medium-sized serranid, which is endemic to the south-east coast of Africa and Madagascar (Heemstra and Randall 1993). In South Africa, it is also heavily targeted by all fishery sectors and is the most common serranid in commercial and recreational ski-boat catches (Fennessy 2000). Catface rockcod are thus also likely to comprise a large component of the increasing contribution of serranids in commercial catches. With evidence strongly supporting atypical protogynous hermaphroditism, the life-history is complex, making it potentially vulnerable to over-fishing (Fennessy and Sadovy 2002). However, Fennessy (2000) considered this species to be optimally exploited, reporting the spawner-biomass-per-recruit to be at 42% of unfished levels with no change in mean lengths between the mid-1980s and mid-1990s.

Surprisingly, catface rockcod were found to be less abundant in the no-take area than in the exploited area. However, a downward trend in CPUE in the exploited area suggests that the concentration of effort in this area may have influenced the abundance of this species as suggested for yellowbelly rockcod.

Spawning of catface rockcod is believed to occur in northern KZN with no spawning known to take place off the Pondoland coast (Fennessy and Sadovy 2002). Recruitment of this species is thus likely to take place from the north (Beckley 1993) and is not likely to be a contributing factor for the disparity in the distribution of this species.

Little is known of the migratory habits of the catface rockcod but it is assumed to be highly resident, although it is possible that individuals also migrate from south to north (Fennessy and Sadovy 2002). It is also possible that small-scale transferral of individuals out of the no-take area has always taken place because of competition with other larger predatory species such as yellowbelly rockcod and scotsman. Alternatively, catface rockcod may be a roaming pioneer species, suggesting a degree of mobility. It has been observed that on several occasions, this species has been among the first to arrive on newly created artificial reefs (B.Q. Mann, Oceanographic Research Institute, 2010; S.A. Chater, uShaka Marine World, 2010; S. Bailey, South African Environmental Observation Network, 2010 pers. comm.). Catface rockcod may be moving out of the no-take area to take up niche space created in the adjacent exploited area because fishing has removed other more resident species such as yellowbelly rockcod.

Although it appears that the no-take area would be ineffective for conservation of catface rockcod and thus enhancement of adjacent fisheries (Fennessy and Sadovy 2002), a no-take area may still provide a temporary refuge for roaming fishes. This effectively reduces the probability of being captured, similar to a reduction in total effort and allows an increase in local stock biomass. In this way, a degree of conservation is available as well as enhancement of local fisheries but not to the extent observed in some other species in this study. Chapter 5 investigates the movement of catface rockcod further, but the possibility of individuals migrating north towards adjacent fisheries (Fennessy and Sadovy 2002) cannot be classed as beneficial fishery enhancement if the abundance of this species is greater in the exploited area.

The St Lucia and Maputaland MPAs in northern KZN were not considered to provide suitable harvesting refuges for catface rockcod (Fennessy 2000). Although these MPAs protect all bottom dwelling reef-fish species from boat-based harvesting, catface rockcod were not known to occur on the offshore reefs in abundance (Chater *et al.* 1993, Chater *et al.* 1995).

Halfmoon rockcod *Epinephelus rivulatus*

The halfmoon rockcod is a relatively small, faster-growing serranid, occurring throughout the Indo-West Pacific (Heemstra and Randall 1993) and is abundant off the south-east coast of southern Africa (van der Elst 1993). In South Africa, it is the second most common serranid in commercial ski-boat catches (Fennessy 2000) and is thus possibly making an increasing contribution due to sequential target switching. This species is of lower importance in all recreational sectors due to its small size (van der Elst 1993, Fennessy 2000, Mann 2000). Froese and Pauly (2010) consider the halfmoon rockcod to have a high resilience to fishery exploitation. Although it is a protogynous hermaphrodite, it grows fast, matures early and since it only reaches a maximum size of 435 mm TL, males are probably not selectively removed (Fennessy 2000, Mackie 2000). Fennessy (2000) reported an increase in mean size of halfmoon rockcod between the mid-1980s and mid-1990s and concluded that the stock was only lightly exploited, with the spawner-biomass-per-recruit estimated at 87% of unfished levels.

Chater *et al*. (1993) and Chater *et al*. (1995) found halfmoon rockcod to be abundant in northern KZN in the St Lucia and Maputaland MPAs but made no quantitative comparison with adjacent fished areas. Halfmoon rockcod were found to be more abundant in the Pondoland exploited area than in the no-take area. Although there was clearly a higher frequency of larger individuals in the no-take area this was not statistically significant and no temporal response to fisheries closure was evident. This is perhaps not surprising given the favourable state of the population as reported by Fennessy (2000). A stock that is not being greatly affected by fishing pressure would not be expected to respond substantially to fishery closure.

It is unlikely that recruitment is a factor responsible for the disparity in abundance along the Pondoland coast. The nature of their egg distribution is not clear, but halfmoon rockcod are assumed to follow larval dispersal patterns common in other serranids in the region (Beckley 1993). Spawning is also widespread, reported to occur along their entire southern African distribution (Fennessy 2000).

Halfmoon rockcod are considered to be highly resident and since spawning is widespread (Fennessy 2000), it suggests that there is no need for a spawning migration. As with catface rockcod, it is possible that small-scale transferral of individuals out of the no-take area has always taken place because of competition with other larger predatory species such as yellowbelly rockcod and scotsman.

At this stage, the Pondoland no-take area may only provide a temporary refuge for roaming halfmoon rockcod, although there is no evidence to suggest roaming behaviour. However, it is expected that the no-take area is exporting halfmoon rockcod larvae, but import of larvae into the no-take area is also likely to be taking place. Although apparently not important now, with the current trend of sequential target switching in worldwide fisheries, a less desirable species like halfmoon rockcod may, in the near future, depend on closed areas for larval export.

Poenskop *Cymatoceps nasutus*

The poenskop is one of the largest members of the sparid family and is endemic to South Africa, occurring from KZN to the Cape (Smith and Heemstra 1986), where it is highly sought after as a trophy fish by all recreational sectors. However, it is of lesser importance to the commercial fishery because of its low abundance (Buxton and Clarke 1989, Mann 2000). The poenskop is highly resident and has a very complex life-history (Buxton and Clarke 1989, Maggs 2009). Slow growth (reaching 45 years), late maturity and protogynous hermaphroditism make it extremely susceptible to over-fishing (Mann 2000). Although data are sparse, it is believed that poenskop are contributing progressively less to catches as it becomes less abundant – similar to other slow growing, endemic sparids (Penney *et al.* 1999). Relatively little is known about the biology and ecology of this species, including stock status, reproduction and movement (Mann 2000). Along with red steenbras, poenskop is one of South Africa's iconic species, believed to be heavily over-exploited and for which, MPAs have been suggested as the most plausible management option (Mann 2000, Maggs 2009).

Poenskop were larger in the no-take area than in the exploited area but abundance was no different between areas. The reason for this is unclear. No temporal response to fisheries closure is yet apparent but this is not surprising given that this species lives for 45 years (Buxton and Clarke 1989).

The reproductive habits of poenskop are poorly understood but it is known that reproductively active individuals occur in the upper Eastern Cape and possibly KZN but are absent from populations in the south-western and south-eastern Cape (Buxton and Clarke 1989). Larvae settling in these regions of the Cape are likely to have originated in areas to the north implying a north to south larval movement.

Unpublished tagging data, derived from the ORI Tagging Programme, show that juvenile and adult poenskop are highly resident. Yet if individuals located in the western reaches of the Cape are to spawn, then it is possible that an easterly migration takes place (Buxton and Clarke 1989).

The Pondoland no-take area is providing refuge for larger individuals of this species thereby fulfilling its primary objective of biodiversity conservation. A temporal response in abundance and size is much anticipated but larger specimens, already occurring within the no-take area, are likely to fulfil the objective of fisheries enhancement by supplying larvae to the adjacent fished areas. Recent observations by local ski-boat fishermen suggest that the number of smaller poenskop, <500 mm, is increasing in the exploited area of the Pondoland MPA. This suggests increased seeding from adult fish protected within the no-take area.

3.5 CONCLUSION

This chapter confirms that some of South Africa’s most important line-fish species are more abundant and of a larger size in the Pondoland no-take area, consistent with the reserve effects reported by other researchers studying the effects of areas closed to fishing in South Africa. However, due to the limitations of the experimental design, it was not possible to conclusively attribute the observed patterns to the proclamation of the MPA because the cessation of fishing was confounded with biogeography. The exploited area was located north-east of the no-take area and there are known latitudinal differences in fish (Mann *et al.* 2006) and invertebrate species distribution (Celliers *et al.* 2007) due to the transitional nature of the Pondoland ecosystem. Although desirable, it was not logistically possible to implement a rigorous before-after-control-impact (BACI) design or a stratified configuration of study sites. Furthermore, no data exist prior to the establishment of the MPA and a scarcity of trends during the study period also suggests that the current situation of greater abundance and biomass could have existed before the proclamation of the MPA. It is thus possible that, even though the Pondoland no-take area was exploited in the past, the level of exploitation never matched that in easily accessible areas, closer to launch-sites. Alternatively, the observed patterns may have been caused by biogeographical effects. Whatever the case may be, the Pondoland no-take area is providing refuge for depleted line-fish stocks, thereby achieving its primary objective and establishing the basis for the enhancement of adjacent fisheries. Possibly the most significant finding of this chapter was the rebuilding of the slinger population, which is currently the most important commercial and recreational line-fish in KZN.

CHAPTER 4 AN ASSESSMENT OF THE INDIRECT EFFECTS OF PROTECTION ON THE PONDOLAND FISH COMMUNITY

4.1 INTRODUCTION

It is evident that key line-fish species in the Pondoland no-take area are benefiting from the absence of fishing (see Chp 3). Such species are typically carnivorous and susceptible to a line-fishery because they readily take a baited hook (Russ 1991). An increase in their abundance and size, following protection from fishing, may lead to an increase in predation and competition in the no-take area. This may have an indirect effect on the entire fish community, including non-target species (Russ and Alcala 1989, Pauly *et al*. 1998, Götz *et al*. 2009).

Almany and Webster (2006) showed that fish communities are strongly influenced by predation, especially during the juvenile settlement phase and suggested that the removal of predators could indirectly affect fish community dynamics. In French Polynesia, Heinlein *et al*. (2010) found that coral reef fish recruits were 74% less abundant on patch reefs exposed to predators than on reefs where predators had been experimentally removed. In Australia and the Bahamas, Almany and Webster (2004) showed that generalist predators had a greater relative effect on less abundant species and therefore reduced local species richness. In the Goukamma Marine Protected Area (MPA), Götz *et al*. (2009) showed that the dominant predator, roman *Chrysoblephus laticeps* (Sparidae), was more abundant in the absence of fishing; however, other non-target species which shared the same diet as roman were more abundant in adjacent fished areas. These authors suggested that the abundance of non-target species in the protected area was controlled by competition with roman.

Making use of an underwater visual census (UVC) method, this chapter investigates aspects of the Pondoland MPA fish community. One key question is asked: is the fish community structure in the Pondoland no-take area different from that in the adjacent exploited area?

4.2 METHODS AND MATERIALS

4.2.1 Data collection

UVC data were collected annually for four consecutive years (2007 – 2010). Sampling was conducted over two days in May/June of each year. One day was spent diving at the two study sites in the Pondoland no-take area (Mtentu and Mkambati) and the other day was spent at the two control sites in the adjacent exploited area (Casino and Mnyameni) (see Fig 2.4, Chp 2). A single diving station (fixed GPS coordinate) within each study site was revisited once a year onboard a 6.5 m semi-rigid inflatable vessel launched at Port Edward. A Garmin 76 handheld GPS, with an accuracy of $\leq$15 m, was used for all GPS positioning.

Point-count methodology adapted from Samoilys (1997) and Thresher and Gunn (1986), was used in this study to assess relative fish abundance, biomass and community structure. At the diving station, two observers using SCUBA, descended onto the reef. The observers then swam together in a randomly chosen direction for approximately 30 fin-stokes before conducting the first point-count. The exact locality (GPS coordinate) of each point-count was recorded on the vessel after the observers signalled their position by tugging on a surface marker-buoy rope. For safety reasons, the two observers needed to remain in close contact with one another. A 10 m tape-measure was then used to separate the two observers, after which, each observer undertook a stationary five minute point-count. When the point-count was completed, the observers moved off in a random direction for approximately 30 fin-strokes and repeated the process. Three random point-counts were conducted by each observer during each dive (i.e. six point-counts per dive).

Point-counts recorded all fish within a 5 m radius (~79 m^2) around the observer including those fish moving into or through the area during the count. The circular area of the point-count also included the water column above the diver visualised as a cylinder, and all point-counts were conducted in water depths ranging from 12 to 24 m. A minimum water visibility of 5 m was required for the counts to be successfully conducted. The same two observers performed all the counts during the four successive years of the study and had extensive experience in local fish identification. During counts, each observer used a pencil and a Perspex slate to record the density and estimated length of all fish species observed within the point-count area. Fish length (total length) was estimated in 50 mm size classes following underwater training as recommended by Samoilys and Carlos (2000).

To obtain a representative sample, the diving stations were arranged so that in each area (exploited and no-take) there was one deep and one shallow station. The water depth of diving stations in the exploited area was 13-17 m at the Casino study site and 19-22 m at the Mnyameni site. In the no-take area, the Mkambati diving station was situated in 12-16 m water depth, while the Mtentu site was in 21-24 m. The topography of the reef which was sampled, consisted of rocky ridges and sandy gullies at the Casino site, flat sandy shelves at the Mnyameni site, ledges and overhangs at the Mtentu site, and boulders interspersed with sandy gullies at the Mkambati site. The habitat at all diving stations was dominated by red algae, sponges and bryozoans.

Every attempt was made to minimise sampling variability and error typically associated with UVC so that measured variability represented the effects of fishery closure as far as possible. For example, abundant supra-benthic fish species such as pinky *Pomadasys olivaceum* and blue hottentot *Pachymetopon aeneum* are attracted to divers, which can cause an over-estimation of their numbers. To avoid this, these species were recorded at the start of each count. Although small cryptic species were inevitably underestimated using the method described, an attempt was made during the last minute of each count to carefully search the reef substrate. During this time any semi-cryptic species such as halfmoon rockcod *Epinephelus rivulatus* and comber *Serranus cabrilla* were counted. The UVC technique was kept constant throughout the study and any error arising from the inherent limitations of UVC was thus kept constant.

4.2.2 Data analysis

The spatial proximity of point-counts conducted at each of the study sites posed a risk of auto-correlation (Götz 2006). Some fish species are inquisitive and known to follow a moving diver (pers. obs., Smith 2005) and are likely to be re-counted in subsequent counts. This increases the chances of pseudo-replication where replicates are not independent of each other, thereby violating the assumptions of statistical data analysis.

This problem was resolved by pooling all replicates collected during a single dive, thereby eliminating the possibility of auto-correlation. However, pooling of the data significantly reduced the number of data points, which increased the chances of statistical over-fitting where univariate data were concerned. For this reason, univariate data analysis was applied to un-pooled data, but was mostly limited to abundance and biomass testing with the intention of validating the findings of Chapter 3 (the fishing survey). Inherent biases in the estimation of fish density and length by an observer had the potential to bias the analysis of unpooled replicates. For example, one observer may consistently overestimate fish length, inflating biomass estimates in certain replicates. This was investigated by using a t-test to compare all replicates conducted by one observer with that of the other observer in terms of overall density and biomass of fishes per replicate. Pooling of replicates was considered acceptable for the detection of patterns in the fish community using a multivariate approach, which was the focus of this chapter.

All univariate data was tested for normality using a Shapiro-Wilk test. All data, violating the assumptions of normality and constant variance made by parametric analyses, were tested using non-parametric procedures. Prior to multivariate analysis, a square root transformation was applied to lightly down-weight the contributions made by highly abundant species. Univariate analyses were conducted using Microsoft Excel (2003) and SigmaPlot (v11). Genstat Discovery (v12) was used for generalised linear models (GLMs). Primer (v6) was used for all multivariate data analysis, including Abundance Biomass Comparison (ABC), Multidimensional Scaling Analysis (MDS), cluster analysis, Analysis of Similarities (ANOSIM), Biological-Environmental+STepwise (BEST) and SIMilarity PERcentages (SIMPER). A significance level (α) of 0.05 was used for all tests.

Indices of diversity in fish communities

To compare the overall fish community in the exploited area with that in the no-take area, multivariate replicates were reduced to univariate indices of diversity. Two indices commonly used for comparisons between biological communities (Clarke and Warwick 2001) are (i) Margalef's index of species richness (d) (Margalef 1958):

$$d = (S-1) / \log N \qquad \text{(eq 4.1)}$$

where S is the number of species and N is the number of individuals, and (ii) the Shannon diversity index (H') (Shannon 1949):

$$H' = -\sum_i p_i \log_e(p_i) \qquad \text{(eq 4.2)}$$

where p_i is the proportion of the total count arising from the ith species. Margalef's index is a measure of species richness which incorporates the number of individuals, while the Shannon diversity index is a measure of diversity which factors in dominance of species.

Overall density and biomass of fishes

Biomass was calculated by converting fish lengths to weights using published length-weight regressions (Mann 2000) and from FishBase (Froese and Pauly 2010). Relative abundance (fish.100 m^{-2}) and biomass (kg.100 m^{-2}) was compared between the four study sites using the Kruskal-Wallis one way analysis of variance (ANOVA) on ranks. Then, with data pooled according to protection status (i.e. exploited and no-take areas), density was tested with a Mann-Whitney Rank Sum Test. The Tukey Multiple Pairwise Comparison Test was used for *post hoc* separation of significantly different sites.

To validate these results, count data (number of fish & mass per replicate) were modelled using a GLM, which is able to isolate the effects that selected environmental variables have on dependent variables (Bennett 2007). GLMs are also suitable for data, which violate the assumptions of parametric analyses (Quinn and Keough 2002).

Akaike's Information Criterion (AIC) (Akaike 1973) was used prior to GLM analysis to identify the best subset of variables to include in the GLM. Decreasing the number of variables included in the GLM, reduced the chance of statistical over-fitting (Götz 2006). Among model selection criteria, AIC is favoured, enabling multiple models to be compared simultaneously, considering both fit and complexity (Johnson and Omland 2004).

To model point-count data, the response variable (relative fish abundance/biomass) was defined as a count variable with a Poisson distribution and the log-link function was used (McCullagh and Nelder 1995). The following GLMs were applied:

$$\log(Abundance) = \beta_0 + \beta_1(Status) + \beta_2(Depth) + \beta_3(Year) + \varepsilon' \qquad \text{(eq 4.3)}$$

$$\log(Biomass) = \beta_0 + \beta_1(Status) + \beta_2(Visibility) + \beta_3(Year) + \varepsilon' \qquad \text{(eq 4.4)}$$

where β_i values were the estimated coefficients and ε the error. Point-count radius and duration were kept constant over the duration of the study and were thus not influential factors. A linear relationship was assumed between effort (point-count radius and duration) and count data (abundance or biomass).

Abundance Biomass Comparison dominance curves

To investigate the magnitude of the effect that the absence of fishing has had on the fish community, Abundance Biomass Comparison (ABC) curves were plotted. These are species dominance curves that measure stress in a community by relating biomass to abundance. For example, a community may be ecologically stressed when a few large-bodied, long-lived species are removed by fishing (Clarke and Warwick 2001), causing a relatively small change in abundance, but a large change in biomass.

Spatial distribution of fish communities

The ANOSIM procedure was used to determine the similarity between fish communities at each of the study sites and according to protection status. Non-metric multidimensional scaling (MDS) and cluster analysis was used to graphically display the similarities in fish communities suggested by the ANOSIM procedure. The ANOSIM, MDS and clustering procedure were computed from a resemblance matrix based on the Bray-Curtis similarity coefficient (Bray and Curtis 1957). Hierarchical clustering using group average linking was used to overlay the MDS plot and grouping samples at the $\geq$50% level of similarity.

Linking spatial distribution of fish communities to environmental variables

Depth, visibility and latitude were measured for each sample. To match the observed spatial patterns in fish communities to measured environmental variables, the BEST procedure was used. This procedure was run using the spearman rank correlation and BIOENV method based on a Euclidean distance resemblance matrix. The BIOENV procedure considers combinations of environmental variables at increasing levels of complexity (i.e. each variable in isolation, then two variables, three variables, etc). A rank correlation value (p_w) is computed for each combination indicating the degree to which that variable or combination explains the observed patterns in the fish community. All environmental variables were normalised prior to analysis.

Trophic structure

Each fish species was assigned to one of six trophic levels based on available diet information (van der Elst 1993, King 1996, Branch *et al*. 2002, King and Fraser 2002, Froese and Pauly 2010). Fishes that feed primarily on seaweed were assigned to the herbivore category and those species with small invertebrates and seaweed in their diet were considered omnivores. Species feeding on plankton were classified as planktivores. The carnivorous category included carnivores that were not primarily piscivorous. A distinction was made between higher and lower carnivores (Floros 2010), where lower carnivores were small-bodied species such as *Plagiotremus rhinorhynchos*, but also included other larger species that feed primarily on sessile invertebrates such as *Oplegnathus robinsoni*. Higher carnivores were larger-bodied species that included invertebrates and some fish in their diet. Large species that feed primarily on fishes were included in the piscivorous category.

Key species

The SIMPER procedure was used to determine which species were responsible for the differences detected between the fish communities in the exploited and no-take areas. Five of the top discriminating species, which are also important fishery species in the region, were selected for length frequency analysis.

Seasonality

As only one field trip was conducted each year, it was not possible to detect any seasonality in count data and insufficient replication prevented the analysis of temporal trends.

4.3 RESULTS

A total of 4616 individual fishes from 91 species were recorded during the 96 point-counts conducted between 2007 and 2010. There were 2625 fishes from 82 species counted at the two exploited area sites (Casino and Mnyameni), and 1991 fishes from 55 species counted at the two no-take sites (Mkambati and Mtentu) (Appendix 2).

In the exploited area, 82 species were recorded belonging to 28 families, with the Labridae being the most speciose family (16 species) followed by the Sparidae (12 species). Five families accounted for 77% of the abundance in the exploited area. These were the Sparidae (33%), Haemulidae (16%), Serranidae (14%), Pomacentridae (8%) and the Lethrinidae (7%) (Fig 4.1a). The top five species, accounting for 51% of the fishes in the exploited area, were slinger *Chrysoblephus puniceus*, sea goldie *Pseudanthias squamipinnis*, dusky rubberlips *Plectorhinchus chubbi*, blue emperor *Lethrinus nebulosus* and striped grunter *Pomadasys striatum*.

In the no-take area, 55 species were recorded belonging to 17 families with the Sparidae being the most speciose family (15 species) followed by the Labridae (eight species). Five families accounted for 88% of the fishes in the no-take area. These were the Sparidae (47%), Haemulidae (30%), Serranidae (5%), Sciaenidae (4%) and the Pomacanthidae (3%) (Fig 4.1b). The top five species, accounting for 55% of the fishes in the no-take area, were slinger, dusky rubberlips, pinky *Pomadasys olivaceum*, Cape stumpnose *Rhabdosargus holubi* and scotsman *Polysteganus praeorbitalis*.

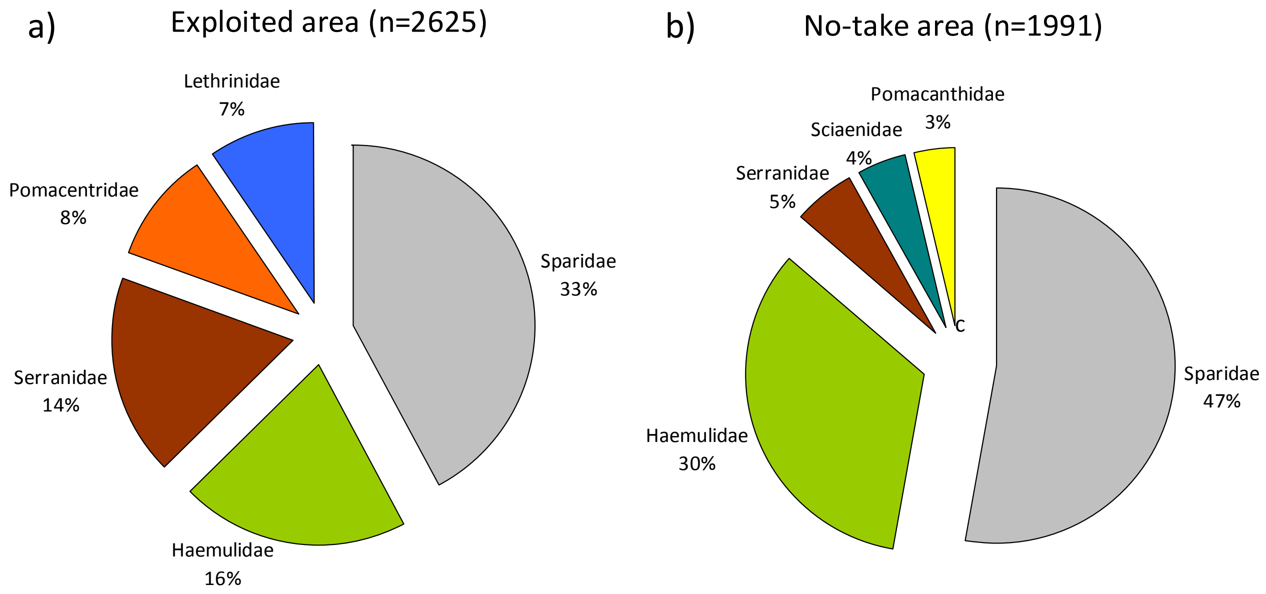


Figure 4.1 The top five fish families by number recorded by UVC in the a) exploited and b) no-take area of the Pondoland MPA between May 2007 and June 2010.

Diversity indices

A univariate analysis of the Pondoland fish community showed that un-pooled replicates taken in the exploited area consistently contained a greater number of individuals and species than replicates taken from the no-take area (Table 4.1). However, differences in Margalef's species richness and Shannon diversity were not significant (P>0.05).

Table 4.1 A summary of mean fish diversity indices sampled by UVC in the Pondoland MPA. Indices calculated on un-pooled, untransformed data. Means ± standard deviation.

	Exploited area (n=48)		No-take area (n=48)	P
Number of individuals (N)	54.7 ± 25.7	>	41.5 ± 26.3	**0.004**
Total species (S)	14.8 ± 3.3	>	12.4 ± 3.1	**<0.001**
Margalef's species richness (d)	3.527 ± 0.72	>	3.234 ± 0.8	0.061
Shannon diversity (H')	2.096 ± 0.4	>	1.963 ± .05	0.129

Overall density and biomass of fishes

An investigation of potential bias in observer estimation showed that no significant difference could be found between the two observers' estimates of fish density (P=0.907) or biomass (P=0.059). The density of fishes in point-counts ranged from 15 to 166 fish.100 m^{-2}. A Kruskal-Wallis one way ANOVA on ranks showed that a significant ($P<0.001$) difference existed amongst the study sites in terms of density (fish.100 m^{-2}) and biomass (kg.100 m^{-2}). *Post hoc* testing showed that the Casino study site was significantly different ($P<0.05$) from all other sites in terms of density, but in terms of biomass it was only the Mtentu site that was significantly different ($P<0.05$) (Fig 4.2).

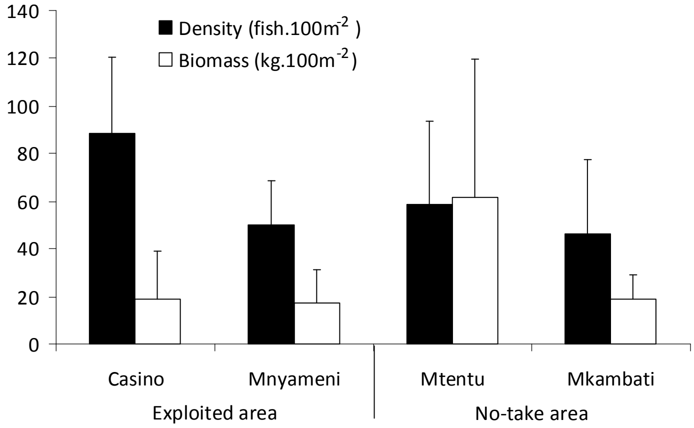


Figure 4.2 Average fish density (fish.100m^{-2}) and biomass (kg.100m^{-2}) values obtained from the underwater point-counts at each of the four study sites. Error bars indicate standard deviation**.**

With site data pooled into the exploited (Casino and Mnyameni) and no-take areas (Mtentu and Mkambati), a Mann-Whitney U test showed that the median density of fishes in the exploited area (63.3 fish.100 m^{-2}) was significantly ($P=0.004$) greater than the density in the no-take area (41.7 fish.100 m^{-2}). However, the average biomass of fishes recorded in the UVC was still significantly greater ($P<0.001$) in the no-take area (21.8 kg.100 m^{-2}) than in the exploited area (12.5 kg.100 m^{-2}).

The results of the GLM analysis were consistent with the Mann-Whitney U Tests, confirming that there was indeed a higher abundance of fishes in the exploited area ($P<0.001$) but that biomass was greater in the no-take area ($P<0.001$) (Fig 4.3).

Abundance Biomass Comparison

In both the exploited (Fig 4.4a) and no-take areas (Fig 4.4b) of the Pondoland MPA, the entire biomass curve lay above the abundance curve, indicating that biomass was concentrated in relatively few species, whereas abundance was more evenly spread across the various species. Although this pattern was evident in both areas, the *W*-statistic, which is a measure of the separation between the curves, was greater in the no-take area ($W=0.115$) than in the exploited area ($W=0.069$). This suggests that there are a greater number of large-bodied, possibly long-lived, species where fishing is prohibited – in agreement with the findings of Chapter 3.

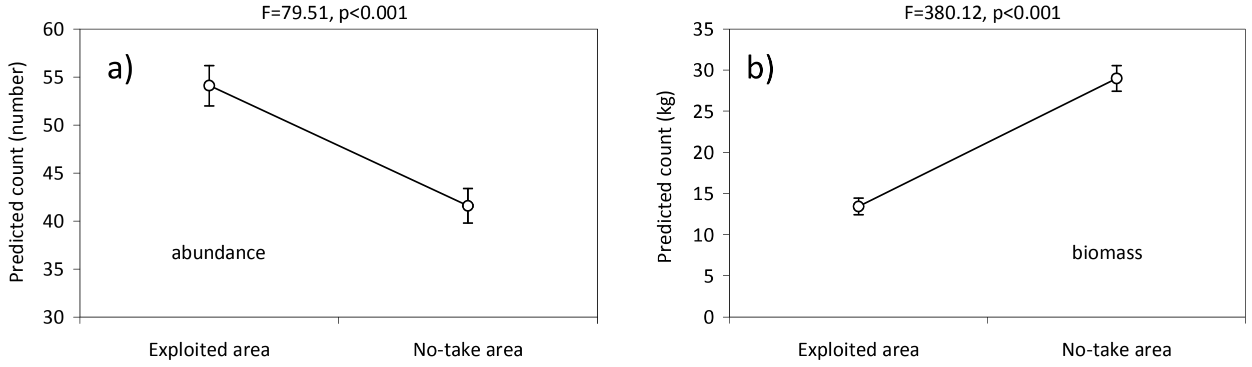


Figure 4.3 Results of the GLM analysis showing predicted UVC count data for a) fish per replicate and b) mass per replicate in the exploited and no-take areas of the Pondoland MPA. Error bars indicate 95% confidence intervals.

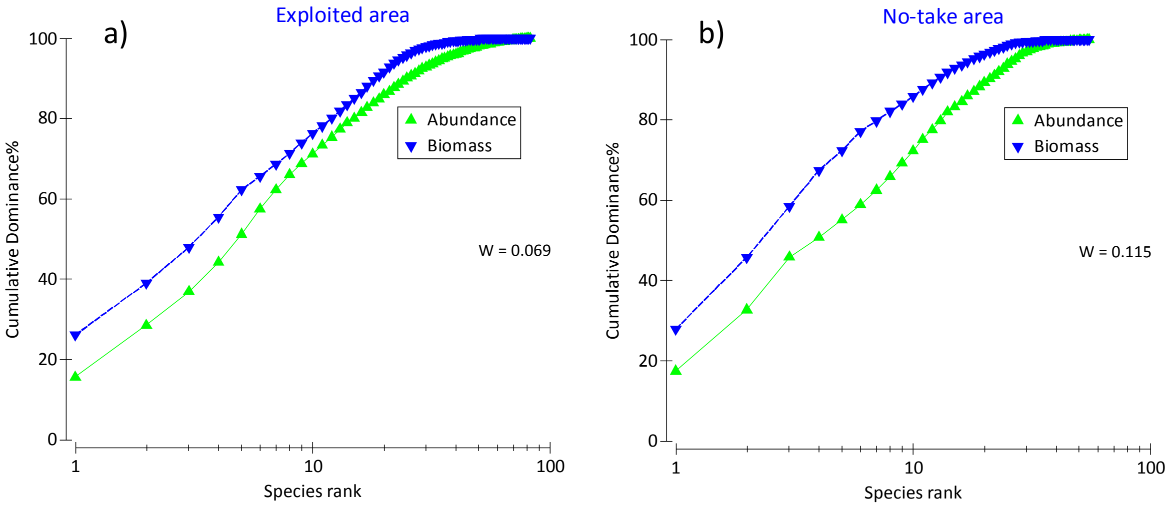


Figure 4.4 Abundance Biomass Comparison (ABC) dominance curves for the Pondoland MPA a) exploited area and b) the no-take area.

Spatial distribution of fish communities

A multivariate analysis showed that the four study sites were not just different in terms of total abundance and biomass but also had significantly different community structures (ANOSIM, R=0.752, P<0.001). An R-statistic approaching a value of one indicates an increasing degree of difference in the communities. Pairwise tests (Table 4.2) showed that the communities at Mkambati and Casino were the most different (R=0.969, P<0.05), while the communities at Mnyameni and Mtentu were the least different (R=0.51, P<0.05).

Table 4.2 A summary of R-statistics computed by the ANOSIM procedure for fish communities at the four study sites in the Pondoland MPA. Values approaching one indicate a high degree of difference. All values were significant (P<0.05). Protected sites are in bold.

	Casino	Mnyameni	**Mtentu**
Mnyameni	0.667		
Mtentu	0.854	0.51	
Mkambati	0.969	0.927	0.865

MDS was used to illustrate the ANOSIM site results (Fig 4.5). The uniqueness of a community at a given site is often depicted by isolation of that site's replicates (e.g. Mkambati), whereas similarity between communities at different sites is often depicted by the proximity of their replicates (e.g. Mtentu and Mnyameni). Encircled areas enclose samples with ≥50% community similarity according to the Bray-Curtis similarity coefficient. The MDS plot largely confirms the ANOSIM results, and shows a change in the community from left to right. This corresponds with an increase in latitude (and distance from the Port Edward boat launch-site). Casino is the furthest north closest to Port Edward launch-site followed by Mnyameni, Mtentu and Mkambati to the south.

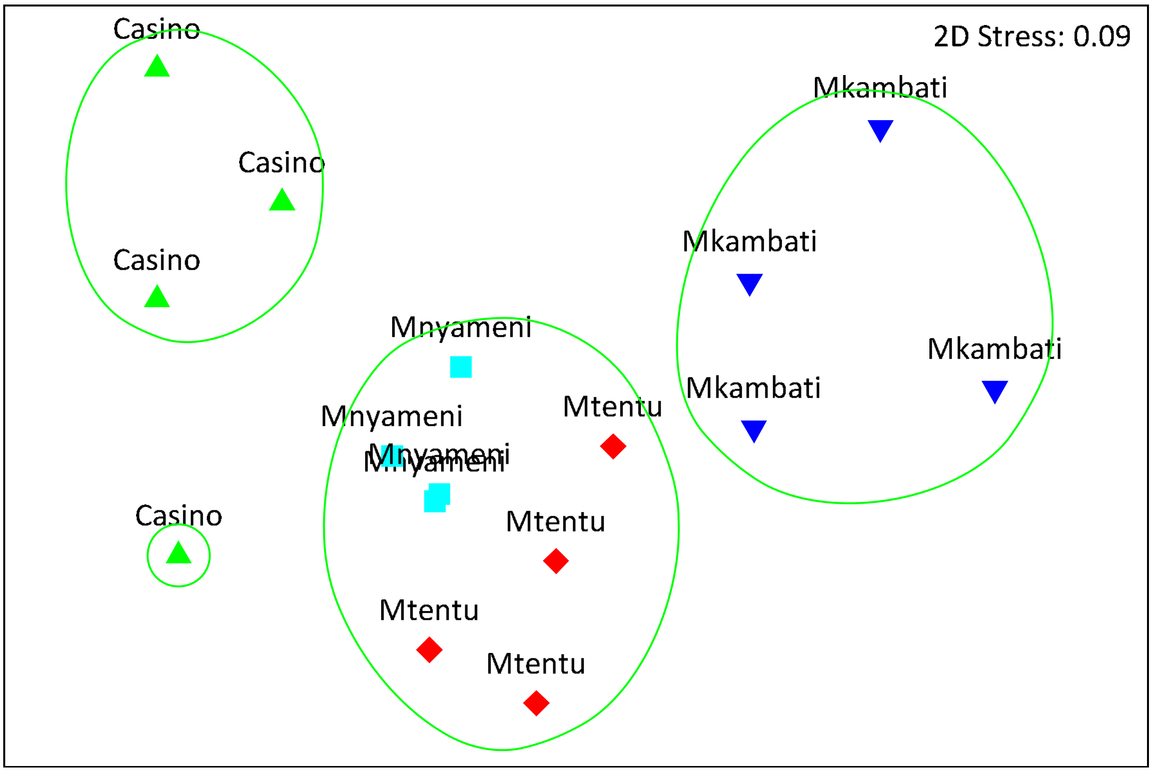


Figure 4.5 Non-metric MDS analysis of square root transformed fish community data recorded by UVC at the four Pondoland MPA study sites between 2007 and 2010. Exploited sites: Casino and Mnyameni; no-take sites: Mkambati and Mtentu. Encircled areas indicate clustering at the ≥50% resemblance level according to overlaid Bray-Cutis similarity coefficient.

Visibility, depth and latitude were tested with the BIOENV method to identify any correlations with the observed patterns in the fish community. It was confirmed that latitude singularly produced the best correlation (p_w=0.645) with the observed patterns in the fish community. The combination of latitude, depth (p_w=0.271) and visibility (p_w= 0.053) as explanatory variables did not improve the overall correlation (p_w=0.63).

With data factored into exploited and no-take status, ANOSIM showed that there was a moderate separation (R=0.423, P=0.002) between the fish community structure in the exploited area and that in the no-take area. Cluster analysis was then used to illustrate the ANOSIM results (Fig 4.6), and so gain a different perspective of the separation between the communities in each area.

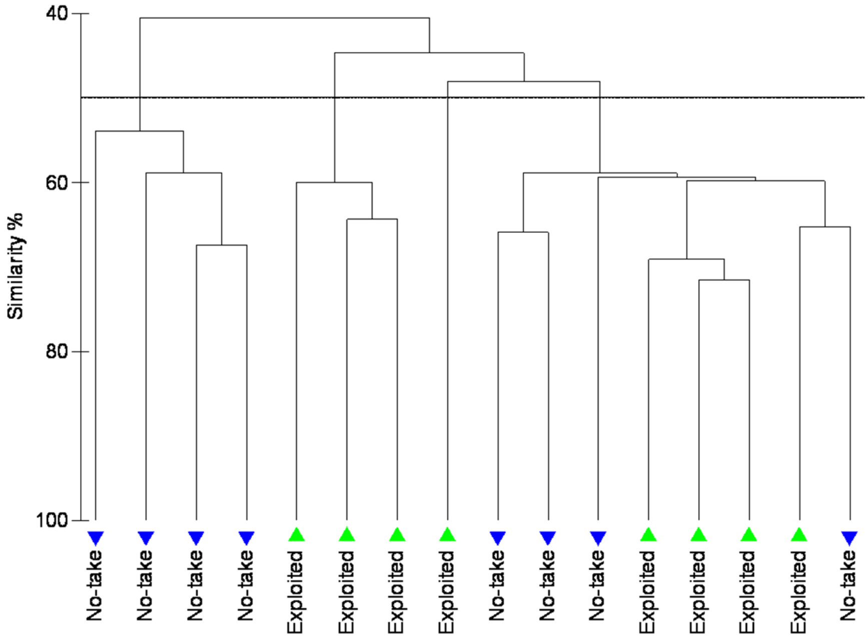


Figure 4.6. Bray-Curtis cluster analysis based on square root transformed fish abundance data recorded by UVC in the exploited and no-take areas of the Pondoland MPA between 2007 and 2010. Slice indicates clustering at the ≥50% resemblance level.

Trophic structure

The trophic structure of the fish community in the exploited area differed from that in the no-take area most importantly by the contribution of planktivores and piscivores (Fig 4.7). Small planktivores, mostly the sea goldie and the blue-spotted damsel *Chromis dasygenys*, contributed 25% to the fish assemblage in the exploited area, whereas planktivorous fishes, the majority of which were sea goldies contributed only 2% in the no-take area.

Larger piscivorous fishes, mostly scotsman and dusky kob *Argyrosomus japonicus*, made a relatively high contribution in the no-take area, whereas piscivores, mostly the small-bodied halfmoon rockcod *Epinephelus rivulatus*, contributed only 3% to the community in the exploited area.

There was also a marked difference in the contributions made by omnivores to the two areas. Omnivorous fishes, mostly Cape stumpnose and fransmadam *Boopsoidea inornata*, accounted for 15% of the community in the no-take area, whereas omnivores, mostly blacktail *Diplodus capensis* and fransmadam, contributed only 7% in the exploited area.

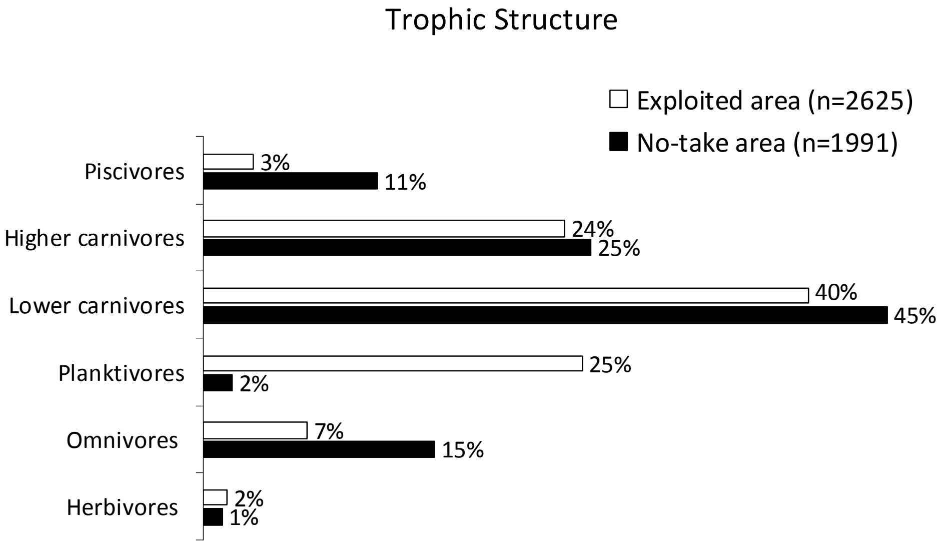


Figure 4.7 A comparison of the fish community trophic structure sampled by UVC in the Pondoland exploited (open bars) and no-take areas (black bars) between 2007 and 2010.

Key species

The SIMPER procedure was used to identify those species which contributed the most to the difference in fish communities between the exploited and no-take areas. This gives an indication as to which species are being affected the most either directly or indirectly by the prohibition of fishing. It was found that blue emperor, dusky rubberlips, sea goldie, slinger and striped grunter contributed 24% to the overall dissimilarity between areas in terms of numbers. However, in terms of biomass, it was dusky rubberlips, slinger, scotsman, yellowbelly rockcod *Epinephelus marginatus* and blue emperor that contributed 34% to the overall dissimilarity.

Blue emperor, dusky rubberlips, slinger, scotsman and yellowbelly rockcod are important fishery target species in KZN and the Eastern Cape, and were selected for further investigation. Sea goldies and striped grunter are small-bodied species, which are not important to the fishery. The disparity in their numbers according to the UVC counts is probably due to indirect effects following the removal of predators in the exploited area by fishing.

Blue emperor *Lethrinus nebulosus*

Blue emperor contributed the highest to the numerical dissimilarity in communities between the exploited and no-take areas (5.72%) and in terms of biomass, this species contributed 5.65%. A total of 193 individuals were counted in the exploited area where the average size according to UVC counts was 182.4 mm. This species was, however, absent from all point counts in the no-take area (Fig 4.8).

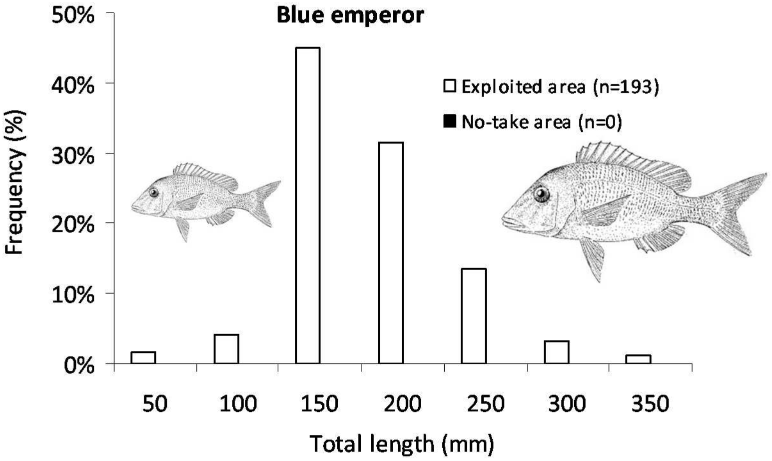


Figure 4.8 Length frequency distribution of blue emperor *Lethrinus nebulosus* in the Pondoland exploited (open bars) and no-take areas (black bars) according to UVC point-counts between 2007 and 2010.

Dusky rubberlips *Plectorhinchus chubbi*

Dusky rubberlips were more abundant in the no-take area and contributed 5.29% to the numerical dissimilarity in communities between the exploited and no-take areas. Regarding biomass, this species contributed the most (9.34%) to the dissimilarity in communities. A Mann-Whitney rank sum test showed that the median size in the exploited area (350 mm) was significantly ($P<0.001$) smaller than in the no-take area (400 mm) (Fig 4.9).

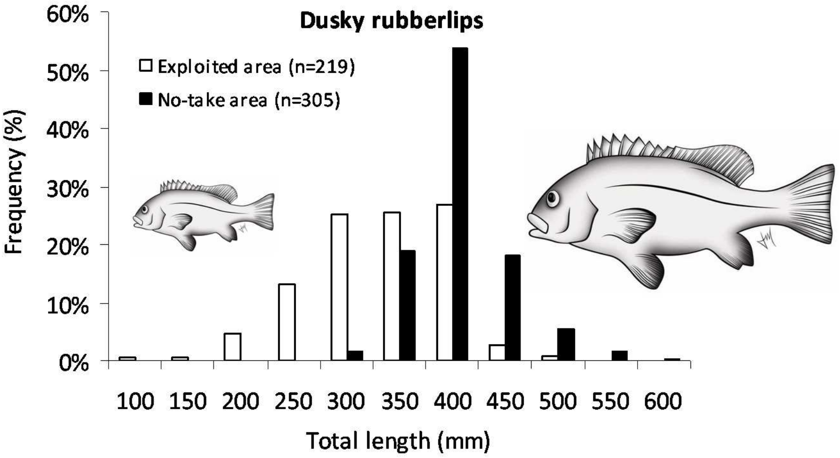


Figure 4.9 Length frequency distribution of dusky rubberlips *Plectorhinchus chubbi* in the Pondoland exploited (open bars) and no-take areas (black bars) according to UVC point-counts between 2007 and 2010.

Slinger *Chrysoblephus puniceus*

Slinger were more abundant in the exploited area, contributing 4.25% to the numerical dissimilarity in communities and 7.07% to the biomass dissimilarity between areas (second highest). A Mann-Whitney Rank Sum Test showed that the median size in the exploited area (200 mm) was significantly (P<0.001) smaller than in the no-take area (300 mm) (Fig 4.10).

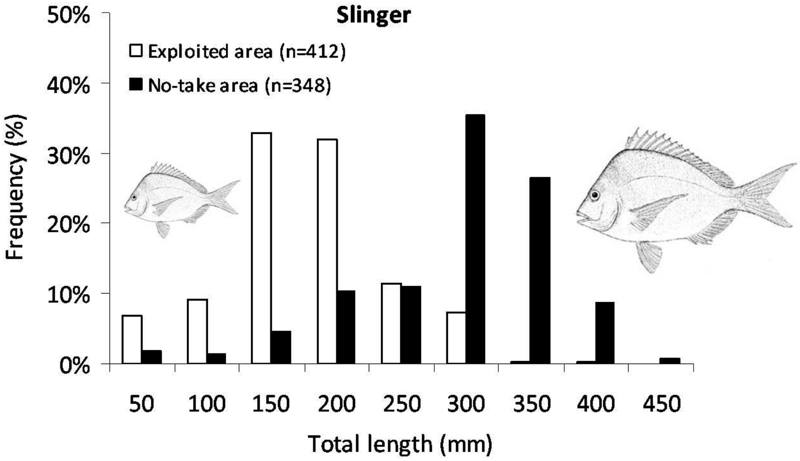


Figure 4.10 Length frequency distribution of slinger *Chrysoblephus puniceus* in the Pondoland exploited (open bars) and no-take areas (black bars) according to UVC point-counts between 2007 and 2010.

Scotsman *Polysteganus praeorbitalis*

Scotsman were more abundant in the no-take area, contributing 2.8% to the numerical dissimilarity in communities but 5.86% to the biomass dissimilarity between areas. A Mann-Whitney Rank Sum Test showed that the median size in the exploited area (300 mm) was significantly (P<0.001) smaller than in the no-take area (400 mm) (Fig 4.11).

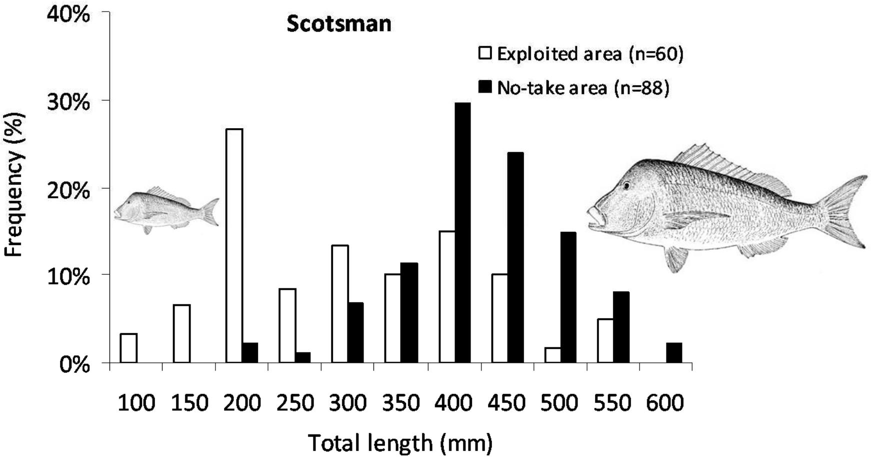


Figure 4.11 Length frequency distribution of scotsman *Polysteganus praeorbitalis* in the Pondoland exploited (open bars) and no-take areas (black bars) according to UVC point-counts between 2007 and 2010.

Yellowbelly rockcod *Epinephelus marginatus*

Yellowbelly rockcod contributed only 2.37% to the numerical dissimilarity in communities but 5.66% to the biomass dissimilarity between areas. A total of 47 individuals were counted in the no-take area, where the average size was 420.8 mm. Only one yellowbelly rockcod (500 mm TL) was counted in the exploited area during this study but was omitted from the length frequency distribution to aid display of other data (Fig 4.12).

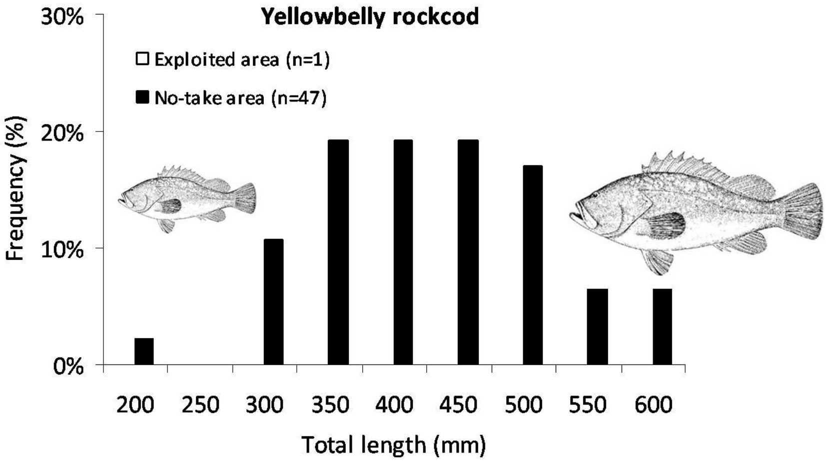


Figure 4.12 Length frequency distribution of yellowbelly rockcod *Epinephelus marginatus* in the Pondoland exploited (open bars) and no-take areas (black bars) according to UVC point-counts between 2007 and 2010.

4.4 DISCUSSION

The UVC method has certain inherent limitations. Primarily, this method is reliant on good visibility and favourable sea-going weather. Fish need to be identified at a distance making it difficult to separate species that are similar in appearance (Brock 1982, Samoilys and Carlos 2000). Cryptic species are usually underestimated (Brock 1982, Kulbicki 1998, Willis 2001) as are highly abundant species (Samoilys and Carlos 2000, Edgar *et al*. 2004). Size estimation of individuals is subject to error especially when a wide range of sizes is present (Bellwood and Alcala 1988). Size estimation has been reported to be most accurate for fish lengths around 300 mm, with accuracy diminishing for smaller and larger fishes (Edgar *et al.* 2004). Inconsistency of an observer and between observers further complicates these factors (Watson and Quinn 1997).

Despite the above-mentioned drawbacks of UVC, the method remains very popular and is used extensively for sampling in MPAs (Buxton and Smale 1984, Alcala 1988, Buxton and Smale 1989, Russ and Alcala 1989, García-Rubies and Zabala 1990, Polunin and Roberts 1993, Chater *et al*. 1995, Jennings *et al*. 1995, Roberts 1995, Rakitin and Kramer 1996, Russ and Alcala 1996a, Jennings and Polunin 1997, Graham *et al*. 2003, Bennett *et al*. 2009, Götz *et al*. 2009, Floros 2010, Maggs *et al*. 2010). The distinct advantages of this method are that surveys are relatively inexpensive (Watson and Quinn 1997), do not require specialist equipment and are particularly suitable for sensitive areas where destructive methods are unacceptable.

Species composition

The dominance of sparids in the Pondoland MPA was evident in terms of both the species diversity and abundance of individuals. Only the labrids exceeded the sparids in the number of species in the exploited area. Of the 15 sparid species recorded in the no-take area, 13 were endemic and all except german *Polyamblyodon germanum* contributed a greater proportion to the abundance in the no-take area. This emphasises the value of the Pondoland no-take area to regional marine biodiversity conservation and its role in protecting typically slow-growing, long-lived fishery species.

Mann *et al*. (2006) conducted a comprehensive survey of the sub-tidal reef-fish communities between Port Edward and Port St Johns recording 121 species from 43 families and noted a number of possible new distribution records. The current study recorded 91 species (29 families), 19 of which were not recorded by Mann *et al*. (2006). It should be noted that the current study was intended as an evaluation of the ecological effects of the no-take area and not as an assessment of the overall fish diversity.

Species recorded in this study, which were identified by Mann *et al*. (2006) as possible new distributions, are included in Appendix 2, although it was acknowledged that these could be classified as vagrants. Eight of the species recorded in the current study but not by Mann *et al*. (2006) possibly represent a southward range extension according to distribution ranges given by Smith and Heemstra (1986), King (1996) and King and Fraser (2002). These species are the blacktail goby *Ptereleotris heteroptera*, bluespotted tamarin *Anampses caeruleopunctatus*, lined tamarin *A. lineatus*, queen coris *Coris frerei*, redcheek wrasse *Thalassoma genivittatum*, blue pete *Pomacentrus caeruleus*, silverstreak goldie *Pseudanthias cooperi* and dusky rabbitfish *Siganus luridus*. It should be noted that older distributional limits of some species may have been conveniently set at the political border between South Africa and the former Transkei (Turpie *et al.* 2000). Thus new distributional records do not necessarily indicate range extensions.

Diversity indices

There were a consistently greater number of individuals and species in the exploited area counts compared with that in the no-take area. However, Margalef's species richness and the Shannon diversity index which compensate for number of individuals and dominance respectively, showed no significant difference between the exploited and no-take areas. This suggests that the low number of species in the no-take area may have been compensated for, by the high number of individuals in those few species. The lower species richness in the no-take area may be explained by selective predation on certain species. Alternatively, indiscriminate predation may also have effectively eradicated certain species, which are less abundant.

Hixon and Beets (1993) correlated piscivore abundance with non-selective reduction and elimination of common and rare species respectively. Heinlein *et al*. (2010), showed that predators reduced prey abundance and used rarefaction analysis to show that reduced species richness in a protected area was associated with fewer overall individuals. This indicated indiscriminate foraging by predators and not selective predation on certain species. It is suggested that predators in the Pondoland MPA probably operate in a similar fashion, being opportunistic rather than specialist prey consumers.

Overall density and biomass of fishes

The disparity in abundance between the UVC and the fishing survey was not unexpected. The selectivity of line-fishing has already been discussed. Using UVC, a broader view of the fish community was obtained including numerous small-bodied, non-targeted (possibly prey) species such as the labrids, chaetodontids and pomacentrids and very small individuals of target species such as slinger. These species, which were not caught by line-fishing were mostly more abundant in the exploited area. The difference in the abundance of non-target species is likely due to indirect factors such as a reduced competition or predation in the exploited area following the removal of target species (Pinnegar *et al*. 2004, Ashworth and Ormond 2005, Götz *et al*. 2009, Babcock *et al*. 2010).

Despite the greater abundance of smaller fishes in the exploited area, biomass was still greater in the no-take area – in agreement with the fishing survey (see Chp 3). This indicated that the biomass in the no-take area cannot be explained by fish abundance alone and is therefore better attributed to greater average body size, primarily of targeted reef-associated fishery species. The feeding preferences of these large–bodied species targeted by the line-fishery are the key to understanding indirect effects at work in the Pondoland MPA (see Trophic Structure).

Depth and protection status were found to be significant predictors of abundance, similar to the findings of Götz *et al*. (2009) and Smale (1987). This may explain why, in the no-take area, the deeper Mtentu site had a greater abundance and biomass of fishes than the more remote Mkambati site, which is further from the no-take boundary and should therefore be less influenced by edge-effects associated with no-take boundaries (Willis *et al*. 2000, Salomon *et al*. 2002).

Abundance Biomass Comparison

Under stable or unexploited conditions, long-lived, large-bodied fishes usually dominate a community in terms of biomass (Clarke and Warwick 2001, Friedlander and DeMartini 2002, Stevenson *et al*. 2007). Under such conditions the biomass is usually concentrated in only a few species, while numerical abundance is more evenly distributed among species (DeMartini *et al.* 2008). However, larger species are usually the first to be reduced by fishing with progressively smaller and less desirable species being targeted (Koslow *et al*. 1988, Jennings and Polunin 1996, Russ and Alcala 1996b, Jennings and Polunin 1997, Pauly *et al*. 1998, Pitcher 2001). This reduces the dominance of the overall biomass by large individuals. Under substantial levels of exploitation, small opportunistic species tend to dominate the community in terms of abundance (Clarke and Warwick 2001).

Clarke and Warwick (2001) explain that an ABC curve is useful in evaluating the stress level experienced by a community without the need for a control (the two curves act as an internal control). Nonetheless, they suggest that reference to a spatial or temporal control is still desirable. The situation in both the exploited and no-take areas suggests low exploitation but more so in the no-take area indicating only slight differences in exploitation. However, Natal fingerfin *Chirodactylus jessicalenorum*, which is a large-bodied species not readily caught by line-fishing, was more abundant in the exploited area. This species may have masked the effect of fishing on other large-bodied species. This highlights the value of no-take areas as benchmarks for scientific study in open areas (Dayton *et al.* 1998).

Besides a reduction in the dominance of biomass by large species, a considerable change in the abundance biomass relationship requires numerical dominance by a few small opportunistic species (Clarke and Warwick 2001). The situation in the Pondoland exploited area appears to be one of reduced target species biomass with a high number of species – none of which were numerically dominant except for the blue emperor. This situation is more consistent with a range of smaller species reaching their southernmost distributional limit in the exploited area rather than prey release due to reduced predation, although it is likely to be a combination of the two factors.

Spatial distribution of fish communities

Patterns in abundance, biomass and community suggested that each study site was unique with little resemblance between the two exploited sites (Casino and Mnyameni), as was the case with the two no-take sites (Mtentu and Mkambati). Moreover, a pattern emerged which suggested a latitudinal shift in the fish communities. This was apparent in the MDS and ANOSIM results and was confirmed by the BIOENV procedure. However, the study sites were arranged in a north-south orientation, parallel to the shore and all sites were to the south of the nearest boat launch-site (Port Edward). It was thus difficult to determine whether the changing fish community reflected the gradient in latitude or in fishing pressure.

In a study of the benthic invertebrates associated with sub-tidal reefs between Port Edward and Port St Johns, Celliers *et al*. (2007) found a distinct change in community composition with increasing latitude. Mann *et al*. (2006) also reported a shift in fish species composition with increasing latitude as many species reach their southernmost distributional limit along the Pondoland coast. The two above-mentioned studies covered a greater area (~90 km of coastline) than the current study (~30 km of coastline) and also included more boat launch-sites, suggesting community changes were related to latitude, as well as to a gradient/change in fishing pressure.

Trophic Structure

It is well established that in most cases, fishing can directly alter a fish community by the removal of piscivores and higher carnivores (Russ and Alcala 1989, Jennings and Polunin 1997, Pauly *et al*. 1998, Myers and Worm 2003). However, predator removal may indirectly affect other non-target species, disrupting the trophic structure (Stallings 2008). Piscivorous fishes are reported to be the most significant consumers of fish biomass (Grigg *et al*. 1984) even surpassing biomass removed by fishing (Jennings and Lock 1996). It has been suggested that the removal of predatory fishes, may lead to an increase in the abundance of prey species (Grigg *et al*. 1984, Hixon 1991, Graham *et al*. 2003, Stallings 2009). Yet, the hypothesis of prey-release at lower trophic levels has been contentious (Hixon 1991, Russ 1991, Caley 1993). Watson *et al*. (2007) reported that the effects of fishing on non-target species are dependent on their role in the ecosystem.

Studies of the effects of predator removal vary vastly in their methodology and geographical location with many contradictory reports existing in the literature. In some studies, no association between predator and prey abundance could be made (Russ and Alcala 1989, Jennings *et al*. 1995, Jennings and Polunin 1997, Russ and Alcala 1998), while others suggested that a predator-prey relationship did exist (Caley 1993, Graham *et al*. 2003, DeMartini *et al*. 2008, Heinlein *et al*. 2010).

Burger (1990) found a higher abundance of the large piscivorous red steenbras *Petrus rupestris* in the Tsitsikamma National Park than in the adjacent fished area. In contrast, this author found that the smaller fransmadam *Boopsoidea inornata* was more abundant in the adjacent fished area where there were less red steenbras. Götz *et al*. (2009) recorded a higher abundance of roman in the temperate Goukamma MPA but also showed that abundance of non-target species (as opposed to prey species) was higher in the adjacent exploited area. These authors suggested that diet overlap among the two groups allowed non-target species to proliferate in the exploited area due to reduced competition. The abundance of the planktivorous pomacentrid, *Acanthochromis polyacanthus* was significantly negatively correlated to that of the piscovorous serranid, *Plectropomus leopardus* on the Great Barrier Reef (Thresher 1983a, b, Graham *et al*. 2003). DeMartini *et al*. (2008) similarly found that piscivores and planktivores varied inversely between fished and protected areas in the central Pacific.

In the current study, carnivore dominance, especially piscivores was strongly evident in the no-take area providing support for direct trophic restructuring in the absence of fishing. Similarly, planktivores were much more abundant in the area of reduced predators (exploited area). The sea goldie and the blue-spotted damsel contributed the bulk of the planktivore abundance in the exploited area but were virtually absent in the no-take area. The blue-spotted damsel is near its southern distributional limit in Pondoland, and in this transitional area, may not reach the no-take area. On the other hand, the sea goldie reaches Algoa Bay, 400 km south-west of the Pondoland MPA but was notably scarce in the no-take area. It is strongly suspected that sea goldie abundance in the no-take area is controlled by resident piscivores such as the scotsman and yellowbelly rockcod but this remains to be confirmed, for example, by stomach content analysis.

The notable scarcity of herbivores (~2% of the community) across the study area was unexpected given the algal dominated benthic environment (Celliers *et al*. 2007). Götz *et al*. (2009) related algal dominance in shallow water to high productivity and suggested this as a possible reason for higher fish abundance on shallow reefs. If fish abundance in the Pondoland MPA is linked to benthic productivity, the bulk of energy from the benthic primary producers (algae) probably flows to larger predators via a route that does not include herbivorous fish species.

Energy could potentially enter the fish community via planktivorous species but their scarcity in the no-take area would be unable to support the high piscivore biomass. Alternatively, given the high overall carnivore dominance (98%) in the Pondoland MPA, herbivorous invertebrates may be the first link between primary producers and higher predators in the Pondoland MPA. Toral-Granda *et al*. (1999) linked commercially important South African reef-fish to invertebrate groups primarily via small reef-fish. The only small carnivore occurring in great abundance in the no-take area was the pinky. Unless, piscivores fed primarily on pinky they would have to switch to invertebrates as a source of energy, supporting the indiscriminate foraging idea. Nonetheless, generalised predation as is possibly the case in the Pondoland MPA makes it difficult to detect trophic linkages (Polis and Strong 1996).

For the above reason, it is recommended that future monitoring of the Pondoland MPA would benefit from including an invertebrate component.

Key species

Blue emperor *Lethrinus nebulosus*

Blue emperor were entirely absent from point-counts in the no-take area, similar to the results of the fishing survey where only 22 individuals were captured in four years. This is contrary to what Watson *et al*. (2007) found in a transition zone in Western Australia. These authors found greater abundance of blue emperor inside a protected area but found depth preference was variable within the area of protection, possibly related to seasonal spawning. This highlights the importance of depth stratification.

The mean length of blue emperor in the exploited area obtained by UVC was smaller than that in the fishing survey (Chp 3). This is most likely due to hook size selectivity in the fishing survey, favouring larger individuals and possible diver avoidance by larger individuals. However, the absence of blue emperor >400 mm (TL) in both the fishing and diving surveys suggests that the Pondoland region acts primarily as a nursery area for this species with adults only being found further north in northern KZN and Mozambique (pers. obs., B.Q. Mann, Oceanographic Research Institute, 2010 pers. comm.). Mann *et al*. (2006) found successively less blue emperor as they moved southwards and it is concluded that the region south of the Sikombe River becomes sub-optimal for settlement of blue emperor recruits.

Dusky rubberlips *Plectorhinchus chubbi*

Very little is known about the life-history of dusky rubberlips but it is known to occur down to 80 m and range across the WIO from India to the Eastern Cape, South Africa (Smith and Heemstra 1986, van der Elst 1993, King and Fraser 2002) and is also found off the Australian coast (Froese and Pauly 2010). Although relatively common, it is only occasionally landed as it does not freely take bait (van der Elst 1993) as was evident in this study.

Dusky rubberlips represented a significant proportion of the difference between the fish communities in the two study areas and were more abundant and larger in the no-take area. Assumedly not under great pressure from line-fishing, this species is, however, a popular target species in the growing spear-fishing sector (pers. obs.).

Thus, it can be assumed that the Pondoland MPA is providing a refuge for a greater abundance of large dusky rubberlips and is therefore achieving its primary objective. However, the larval and adult fish contribution of the Pondoland MPA to regional ecosystems is unknown regarding this species. It is suggested that priority be given to the study of the biology and life-history of this species in the near future.

Slinger *Chrysoblephus puniceus*

Slinger were more abundant but of smaller mean size in the exploited area than in the no-take area. These findings were consistent with the fishing survey (Chp 3). During the UVC, 49% of the slinger in the exploited area were 150 mm or smaller, whereas no slinger <150 mm were caught during the fishing survey. This finding was indicative of hook size selectivity and suggests that large numbers of small slinger, too small to catch, are present in the exploited area.

Scotsman *Polysteganus praeorbitalis*

Scotsman recorded by UVC exhibited similar patterns to that recorded in the fishing survey. Although the length frequency distribution in the no-take area was typically bell-shaped, there was an absence of individuals <200 mm. No individuals <100 mm were recorded in the current study, but scotsman <50 mm were commonly recorded by Mann *et al*. (2006). Although their study was conducted in the same season as the current study (May/June), their survey area covered the entire Pondoland coast (90 km) as far south as Port St Johns and consisted of 261 point counts over a range of reef habitats including deeper reefs (>25 m).

Yellowbelly rockcod *Epinephelus marginatus*

UVC sampling of yellowbelly rockcod was largely consistent with the results of the fishing study but only one individual was observed in the exploited area point-counts.

4.5 CONCLUSION

This chapter supported the findings of the fishing survey (Chp 3) and provided strong evidence that fishing in the exploited area has directly altered the fish community by the removal of larger, higher trophic level fishes. Evidence is also given for differences in the fish community at lower trophic levels. However, due to the limitations of the experimental design, it was not possible to conclusively attribute the observed patterns to the proclamation of the MPA because the cessation of fishing was confounded with biogeography. The exploited area was located north-east of the no-take area and there are known latitudinal differences in fish (Mann *et al.* 2006) and invertebrate species distribution (Celliers *et al.* 2007) due to the transitional nature of the Pondoland ecosystem. Although desirable, it was not logistically possible to implement a rigorous before-after-control-impact (BACI) design or a stratified configuration of study sites. Nonetheless, the no-take area is protecting a unique fish community whether this community is a result of the cessation of fishing or biogeography.

CHAPTER 5 MOVEMENT PATTERNS OF IMPORTANT FISHERY SPECIES OFF THE PONDOLAND COAST USING A TAG-RECAPTURE STUDY

5.1 INTRODUCTION

Conservation was the primary purpose for establishing many of South Africa's marine protected areas (MPAs) in the past (Gell *et al.* 2003), but a growing body of evidence, globally, is showing that fully protected no-take areas provide valuable fishery benefits (Gell and Roberts 2003a, Gell and Roberts 2003b). Following the build-up of depleted fish stocks in a no-take area, it has been predicted that competition may drive adult and sub-adult fishes away, providing density-dependent spill-over into nearby fisheries (Bohnsack 1998). However, such biomass export is difficult to prove and conclusive evidence is scarce (Tilney *et al*. 1996, Palumbi 2003, Unsworth *et al*. 2007).

Where protection has led to a build-up of biomass within a no-take area, spill-over into adjacent fisheries has been inferred by increases in catch-per-unit-effort (CPUE) and fish densities close outside no-take boundaries. In the Caribbean, Roberts *et al*. (2001) recorded substantial increases in CPUE of fish traps in close proximity to a network of four MPAs (no-take areas). McClanahan and Kaunda-Arara (1996) and McClanahan and Mangi (2000) used the same concept to investigate spill-over from a no-take area in Kenya. At the much studied Apo Island in the Philippines, Russ and Alcala (1996a), using underwater visual census (UVC), recorded significantly higher densities of large predatory fish within 300 m outside of the no-take boundary. In a later study, Maypa *et al*. (2002) reported an increase in the overall CPUE of the Apo Island fishery over a twenty-one year period. A substantial increase in the CPUE, density and modal size of bignose unicorn *Naso vlamingii* was recorded close outside the Apo Island no-take area, but no changes had taken place further away from the no-take boundary (Russ *et al*. 2003, Abesamis and Russ 2005). Concentration of fishing effort around MPA boundaries has also been used to suggest spill-over (McClanahan and Mangi 2000, Murawski *et al*. 2005, Goñi *et al*. 2008), but this may just reflect the perceptions of fishers that more fishes are likely to occur close to these boundaries (Willis *et al.* 2003).

Most studies suggesting spill-over from MPAs lack before and after controls, and are therefore unable to conclusively attribute improved catch rates and density to protection. However, rising tension between resource users and conservationists, is creating a demand for quantitative proof of fisheries benefits from no-take areas (Gell and Roberts 2003b). Foundational to this proof is an understanding of the nature of fish movements in relation to protected areas. In South Africa, Attwood and Cowley (2005) used a tag-recapture study to show convincing evidence of dispersal in galjoen *Dichistius capensis* from MPAs into fished areas.

Using the results of a tag-recapture study, this chapter investigates the movement patterns of seven important line-fish species in the Pondoland MPA. One key question is asked: Does the Pondoland no-take area enhance adjacent fisheries by exporting post-larval fishes? This question is answered by investigating the balance between biomass retention in the no-take area and its export to adjacent fisheries. In other words, what is the degree of residency among fishes? For some species, the first known estimates of home-range size are presented.

5.2 METHODS AND MATERIALS

5.2.1 Data collection

To assess the movement patterns of important fishery species in the Pondoland MPA, a tag-recapture study was undertaken. Data was collected quarterly from April 2006 to July 2010 in tandem with the fishing survey described in Chapter 3. All fishes tagged were those captured within the four 2 km^2 study sites in the fishing survey (see Fig 2.4, Chp 2). Consequently, the methods used to capture fishes for the purpose of tagging are described in Chapter 3. In addition to the fishes recaptured by the research team within the study sites, recaptures made outside the no-take area were also reported by the public.

Selected fish species were tagged by removing a single scale and inserting a plastic dart tag (Hallprint Pty Ltd, Australia) in the flesh. The tag was anchored in the pterygiophores with the barb facing toward the tail of the fish (Fig 5.1). Tag placement varied with some species such as Natal seacatfish *Galeichthys trowi*, but was generally at a point halfway between the lateral line and the base of the dorsal spines, dorso-ventrally, and halfway between the operculum and the caudal peduncle laterally. The angle of insertion was such that the tag faced upward and backward, similar to the angle of the dorsal spines, and at a 45° angle from the body of the fish so that no contact was made between the tag and the body.

Only selected species and individuals were tagged. For example, the muraenids (eels) were not tagged, and no individuals of any species <300 mm fork length (FL) or total length (TL) were tagged. The length restriction excluded small species such as fransmadam *Boopsoidea inornata* and dane *Porcostoma dentata* from being tagged. Although large numbers of blue emperor *Lethrinus* *nebulosus* and slinger *Chrysoblephus puniceus* were caught, the majority were not tagged as they were generally <300 mm FL. Slinger in the no-take area were tagged more frequently towards the end of the study due to the increasing occurrence of larger individuals >300 mm (see Chp 3). It was not possible to land very large sharks and rays with the fishing gear and the 6.5 m rigid hull ski-boat used in this study. These individuals were thus not tagged. Furthermore, Natal seacatfish were only tagged opportunistically when other more popular line-fish species were not being caught.

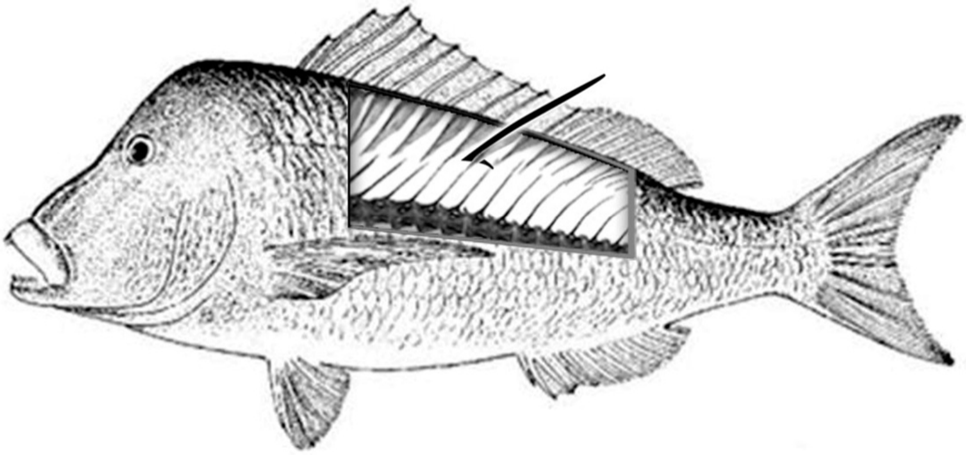


Figure 5.1. Method of tag placement used for the majority of fish tagged in this study. Scotsman *Polysteganus praeorbitalis*.

A small D-type tag, 85 mm long and 1.6 mm wide, was used for fishes 300-599 mm (FL/TL) long and a larger A-type tag, 114 mm long and 1.6 mm wide, for fishes >600 mm (FL/TL). Tags were marked with an alpha-numeric code and the postal address of the Oceanographic Research Institute in Durban. For example, "Tag No. D101881, ORI, P.O. Box 736, Durban". Once captured and tagged, the species, length, date, time and condition of the fish were recorded against the tag-number before being released. Additionally, the locality of each tag-release was recorded using a Garmin 76c, global positioning system (GPS), accurate to within 15 m. The same procedure was repeated for each subsequent recapture.

Sumpton *et al*. (2010) found that the depth at which a fish was captured, how it was hooked and its condition when released were significant factors affecting recapture rate, which was suggested to be an indicator of post-release survival. To mitigate these factors, fishing was limited to depths shallower than 30 m and barbless circle hooks were used. By fishing in shallow water, barotrauma was minimised but was more evident in some species such as slinger and poenskop *Cymatoceps nasutus* than in others such as yellowbelly rockcod *Epinephelus marginatus*. Specimens that suffered barotrauma, being obviously bloated, buoyant or having the stomach protruding from the mouth, were deflated by insertion of a 12-guage hypodermic needle through the body wall into the swim-bladder. In some cases, bigger individuals were returned back to the sea-floor using a weighted down-rigger system (Mann and Kyle 2009, Sumpton *et al*. 2010). The use of circle hooks resulted in a very low incidence of gut- or throat-hooking. In cases when a fish was gut- or throat-hooked, the fishing line was cut as close as possible to the hook and this was noted against the recorded tag number.

According to Danylchuk *et al.* (2007), fishes that survive the capture and release process may sustain irreversible damage leading to a delayed death or might be more vulnerable to predation. Consequently, every effort was made to minimise the stress imposed on a fish by careful handling, which included covering the fish with a moist cloth. Handling time out of the water was also kept to a minimum, normally less than 60 seconds.

5.2.2 Data analysis

Tag-recaptures that were made by the public within 2 km of the study sites were not included in the analyses. The exact location of such recaptures could not be determined with sufficient accuracy, either because the angler did not record the GPS location or was unwilling to disclose it. These localised movements represented only a fraction of the overall recaptures and would merely have strengthened the conclusions presented. Fine-scale accuracy was not required for long-distance movements reported by the public and these records were therefore included.

Nature of movement

Dingle (1996) provides a simple classification which is suitable for contextualising the movement patterns of fishes in the current study (Table 5.1). Station-keeping is a good indicator of the potential for biomass retention within a protected area, while ranging or migratory behaviour may indicate the potential for export to adjacent fisheries. The movement data obtained from recaptured fishes was used to investigate the balance between s*tation-keeping* (foraging within a home-range), and *ranging* (long-distance dispersal) behaviour. Movements which took place within a study site were considered to be localised and a reasonable indicator of foraging home-range behaviour, while those which left the study site were considered to be ranging, possibly migratory. From the localised movements, home-range size was estimated to further quantify the degree of residency.

Table 5.1. Classification of animal movement behaviour proposed by Dingle (1996).

Movement type	Characteristics
Station-keeping:	
Kinesis	Movements that serve to keep an animal stationary
Foraging	***Movements within a home-range***
Commuting	Diel movements between day & night locations
Territoriality	Territorial defence & aggression, non-overlapping home-ranges
Ranging	***Exploratory movements over wide areas in search of resources***
Migration	Persistent, directed, non-exploratory, predictable, physiological adaption

Station-keeping (foraging within a home-range)

A home-range can be defined statistically as the area within the space-utilisation contour that encompasses 95% of a fish’s movement (Anderson 1982). However, to calculate this area at least three points are required. In other words, the initial capture and at least two subsequent recaptures are required. Telemetry supplies high resolution movement data by frequently pin-pointing an individual’s locality and is more suitable for such calculations. The majority of data derived from tag-recaptures are from one-time recaptures, thus providing only a single dimension, which prevents calculation of area. Nevertheless, in this study the linear distance of a fish’s home-range rather than the area was most relevant to the question asked: will fish movements go beyond the no-take boundary?

A method for estimation of the linear distance of a home-range was adapted from Griffiths and Wilke (2002) who calculated travel range length (TRL) movements for five sparid species. However, Griffiths and Wilke (2002) analysed movements on a scale of kilometres and included large-scale (39-1000 km) movements in their calculation of TRL. Thus, their results were not representative of station-keeping. TRL was calculated by taking the distance encompassing 95% of *all* recorded movements and was referred to as radius. Referring to the distance between tag-release and recapture as a radius implied that the determined distance represented only half of a fish’s potential range (Fig 5.2 a). For example, a fish capable of moving a given distance between point B and C could also cover the same distance between point B and A.

The methods used in this study differed in two ways. To estimate a home-range which assumes 95% temporal utilisation (Anderson 1982), only intra-study site movements were used. Similar to a study by Attwood and Cowley (2005), all ranging movements were excluded from the home-range calculation. Secondly, the current study assumed that a fish’s home-range centre was somewhere between the two capture localities and thus linear distances represented the diameter of a home-range and not the radius (Fig 5.2 b).

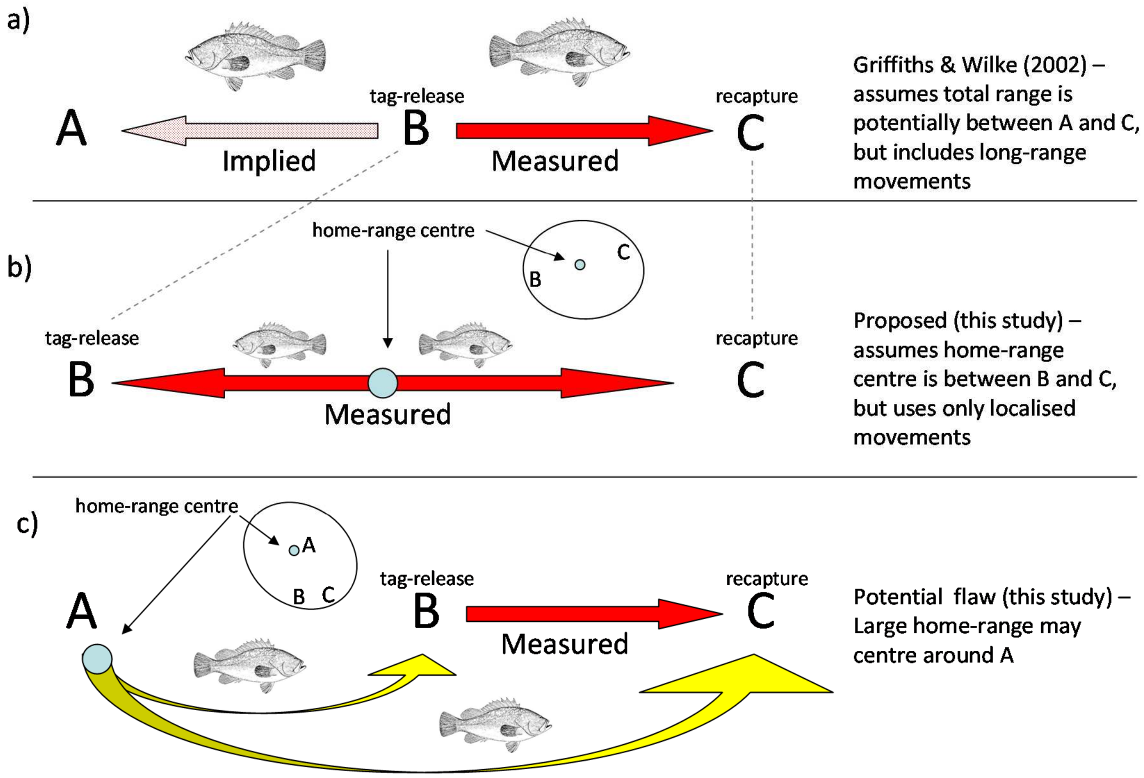


Figure 5.2. Possible explanations of movement distances from tag-recapture observations. a) assumption by Griffiths and Wilke (2002): movement distance (travel range length) equals twice the observed distance, b) assumption of this study: movement distance only equal to observed localised movement and c) movement distance potentially flawed (underestimated) by unknown movements.

Regarding localised movements, the hypothesis of a tag-recapture (i.e. tag-release at B and a recapture at C) representing a radius assumes that at first capture the fish was always at the centre of its home-range, which would be unrealistic for station-keeping behaviour. Fishes are curious and are attracted from some distance away to a point where there is bait and feeding activity as noted in baited underwater video methods (Harvey *et al*. 2007). It follows that a fishing station could attract some fishes away from the centre of their home-range. However, the assumption of this study is potentially flawed because a fish could have a home-range that centres on point A but also includes B and C (Fig 5.2 c). In other words, if fishing stations are clustered in an area smaller than the home-range of a fish, then the estimate of home-range may be a function of the distance between fishing stations rather than area utilisation by the fish. This may result in an underestimation of home-range diameter. The probability of this occurring was tested (Mann-Whitney U Test) by comparing the mean distance between random pairs of fishing stations, within each study site, with the mean distance of fish movements for each species.

Using only localised intra-study site movements in the current study, the linear distance of a home-range was estimated for each of the key species by taking the 95th percentile of tag-recapture movements (linear distance between tag-release and tag-recapture). The resulting estimate of home-range length is referred to as *single* linear distance (SLD) in this study (Fig 5.3 a). Assuming that a fish is randomly drawn from within the boundaries of its home-range at first capture (tagged) and then re-drawn from that same home-range at a later stage (recapture), the Euclidean distance between the two points (SLD) can be considered to represent a proportion of the length of the actual home-range. Repeating this several times, with different individuals of the same species increases the probability of obtaining the mean length of the home-range for that species.

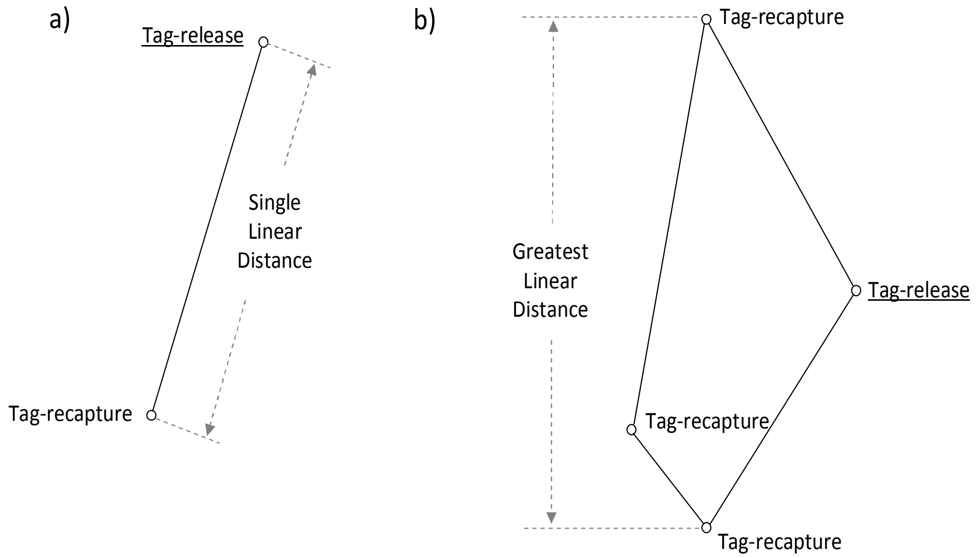


Figure 5.3. Calculation of home-range length used in this study. a) single linear distance (SLD): distance between release and recapture and b) greatest linear distance (GLD): widest distance of polygon.

A small proportion of fishes, after being tagged and released, were recaptured on multiple occasions. Such “multiple recaptures” were also incorporated into the calculation of SLD by using only the distance between the tag-release and first recapture. Subsequent recaptures of an individual were omitted from the calculation of SLD to prevent pseudo-replication. For example, a highly resident individual would be prone to being caught repeatedly in a small area and could bias the estimate of mean home-range size.

Multiple recaptures, having three or more capture points, provided a two-dimensional representation (polygon) of area utilisation. An alternative estimate of home-range size was therefore calculated by taking the 95th percentile of the *greatest* linear distance (GLD) across the interior of the polygon (Fig 5.3b). The resulting estimate was used to validate the SLD calculated above.

In addition to the above methods, Kramer and Chapman (1999) proposed the following equation for calculating the home-range length of a fish:

$$\text{Home-range length (m)} = 0.000178\ (\text{fork length (mm)})^{2.35} \qquad \text{(eq 5.1)}$$

Body length explained 73% of the variation in home-range size, but their data was biased towards tropical species. Nevertheless, estimations of home-range length in the current study (SLD and GLD) were compared to that predicted by Kramer and Chapman (1999), based on the maximum body length of a particular species recorded in this study.

Ranging (long-distance dispersal)

The potential of the no-take area to export biomass was evaluated using records of long-distance ranging movement for each species (i.e. tagged fish leaving the study site). The opportunistic recovery of ranging fishes originated exclusively from recreational and commercial fishers, whose fishing effort is widely distributed along the South African coast. However, only a limited number of ranging movements were reported, while the potential for non-reporting also existed (Brouwer 1997, Everett 2004, Dunlop 2010). Consequently, a purely descriptive analysis of ranging movement is given.

The Oceanographic Research Institute cooperative line-fish tagging programme

In 1984 the Oceanographic Research Institute (ORI) initiated a national cooperative line-fish tagging programme (van der Elst and Bullen 1993) which is still in operation. This programme is aimed at promoting voluntary tag and release angling among recreational anglers and the collection of valuable movement and growth data of priority species. Despite a membership of over 4000 anglers, there are only about 400-500 anglers, which are annually active on the ORI Tagging Programme. Several thousand fish are tagged annually, generating a large volume of data, but this suffers from certain inherent biases associated with the non-scientific collection of data by recreational anglers (Govender and Bullen 1999). The locality of tag-releases and recaptures is given as a number based on the kilometre distance from the northern border of Mozambique around the southern tip of Africa up to northern Namibia. For this reason, the data has a resolution of approximately 1 km at best. Nevertheless, these data are used to contextualise the results obtained in the current study.

5.3 RESULTS

A total of 2124 fishes were tagged during 18 field trips, between April 2006 and July 2010. Of these, 1396 were tagged in the no-take area, while 728 were tagged in the exploited area. The overall recapture rate was 23% for fishes released at the no-take sites and 29% for fishes released at the exploited sites. These recapture rates also include individuals that were recaptured more than once (i.e. multiple recaptures) as well as those that were recaptured outside the no-take area and reported by the public (Attwood 2002). Excluding multiple recaptures, 17% of all tagged fishes were recaptured and seven species accounted for 92% of the recaptures (Table 5.2). With the exception of blue emperor, these were the same key line-fish species investigated in Chapter 3 (scotsman *Polysteganus praeorbitalis*, yellowbelly rockcod *Epinephelus marginatus*, Natal seacatfish *Galeichthys trowi*, catface rockcod *E. andersoni*, poenskop *Cymatoceps nasutus*, halfmoon rockcod *E. rivulatus* and slinger *Chrysoblephus puniceus*). The movements of these seven species were investigated further.

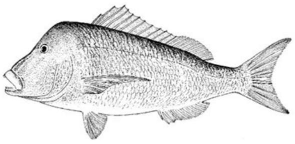
Movement patterns

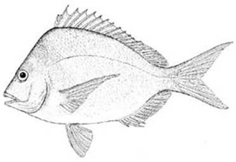
Movements were categorised as either station-keeping, if the fish was recaptured within the 2 km^2 study site, or ranging if the fish was recaptured beyond the boundaries of the study site. Ninety-six percent of all fishes were recaptured in the same study site where they were originally tagged. Of those recaptured within the original study site, 90% were recaptured within 250 m of the release site, varying between 72% for scotsman and 97% for yellowbelly rockcod (Fig 5.4). The majority of fishes, which had left the study site, moved more than 150 km, suggesting a distinct separation between foraging and ranging movements.

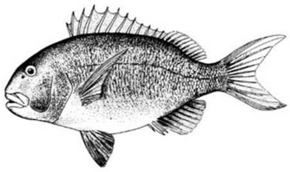
Little association could be made between the distance moved and fish length for most species. However, the longest distances moved by yellowbelly and catface rockcod were by larger individuals. A similar trend was apparent in scotsman, but none of these trends were significant (Fig 5.4). Movements could also not be correlated with the time a fish was at liberty. Some individuals moved away rapidly from the site of release while others remained resident for extended periods (Fig 5.5).

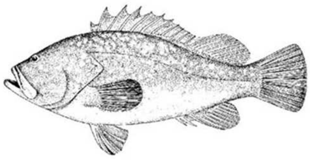
Table 5.2. A summary of tag and recapture data recorded for seven important fishery species in the Pondoland MPA between April 2006 and July 2010.

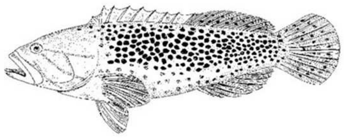
	Overall	Exploited area	No-take area
Scotsman *Polysteganus praeorbitalis*			
Number of individuals tagged	385	80	305
Number of individuals recaptured	98 (25%)	24 (30%)	74 (24%)
Number of individuals recaptured more than once	23	2	21
Most times an individual was recaptured		4	5
Overall recapture rate	**36%**	**39%**	**35%**
Slinger *Chrysoblephus puniceus*			
Number of individuals tagged	308	12	296
Number of individuals recaptured	26 (8%)	0	26 (9%)
Number of individuals recaptured more than once	None	n/a	n/a
Most times an individual was recaptured		n/a	1
Overall recapture rate	**8%**	**0%**	**9%**
Poenskop *Cymatoceps nasutus*			
Number of individuals tagged	156	56	100
Number of individuals recaptured	37 (24%)	16 (29%)	21 (21%)
Number of individuals recaptured more than once	5	3	2
Most times an individual was recaptured		3	2
Overall recapture rate	**28%**	**36%**	**23%**
Yellowbelly rockcod *Epinephelus marginatus*			
Number of individuals tagged	161	33	128
Number of individuals recaptured	51 (32%)	8 (24%)	43 (34%)
Number of individuals recaptured more than once	22	4	18
Most times an individual was recaptured		5	9
Overall recapture rate	**61%**	**61%**	**61%**
Catface rockcod *Epinephelus andersoni*			
Number of individuals tagged	168	117	51
Number of individuals recaptured	45 (27%)	32 (27%)	13 (25%)
Number of individuals recaptured more than once	18	12	6
Most times an individual was recaptured		5	5
Overall recapture rate	**45%**	**43%**	**49%**
Halfmoon rockcod *Epinephelus rivulatus*			
Number of individuals tagged	138	85	53
Number of individuals recaptured	30 (22%)	24 (28%)	6 (11%)
Number of individuals recaptured more than once	8	6	2
Most times an individual was recaptured		3	2
Overall recapture rate	**30%**	**39%**	**15%**
Natal seacatfish *Galeichthys trowi*			
Number of individuals tagged	175	76	99
Number of individuals recaptured	50 (29%)	26 (34%)	24 (24%)
Number of individuals recaptured more than once	18	10	8
Most times an individual was recaptured		4	5
Overall recapture rate	**49%**	**58%**	**41%**

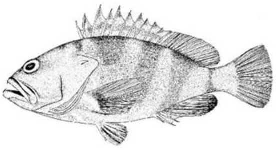


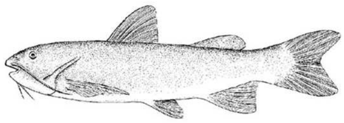












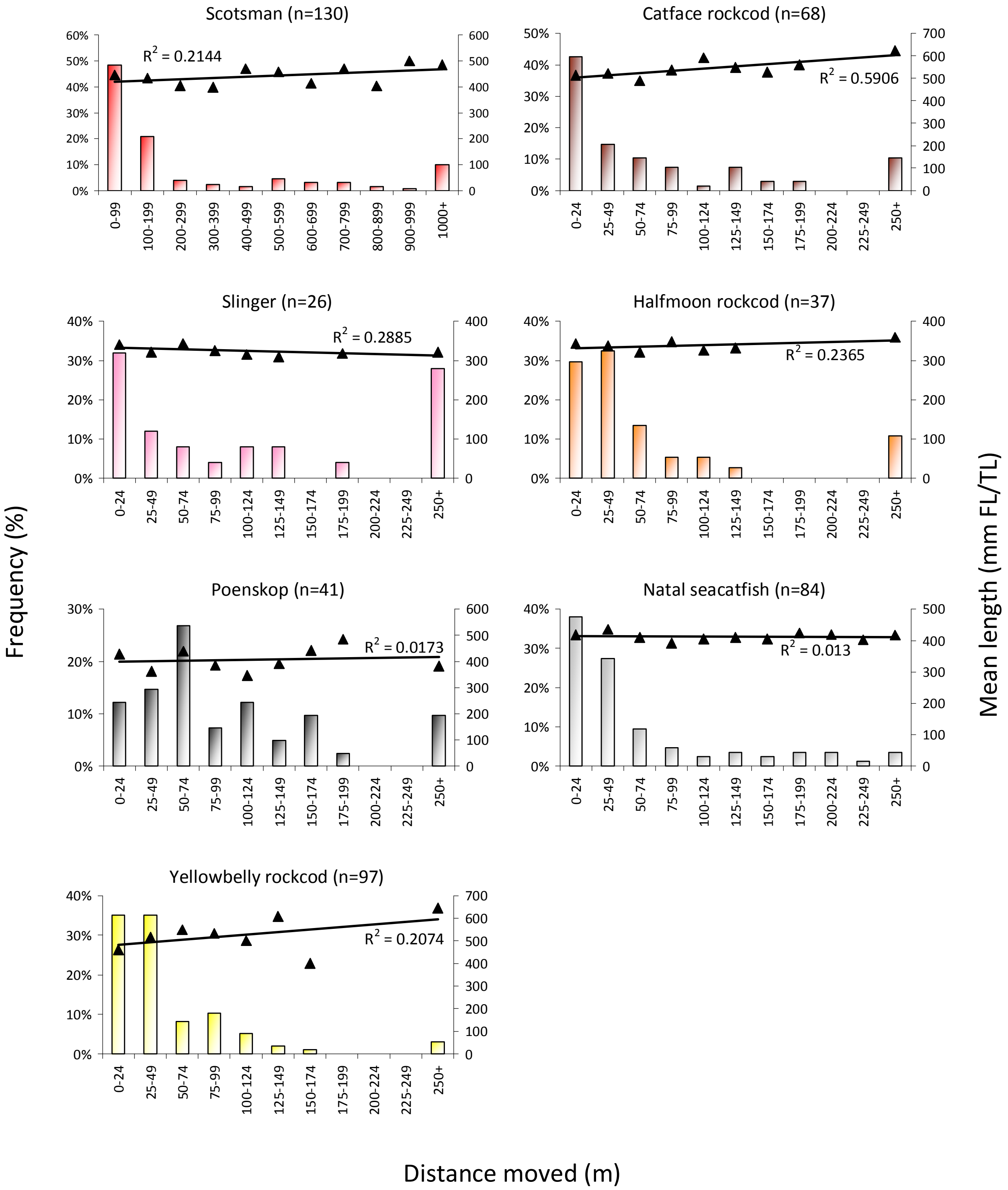


Figure 5.4. Frequency histograms of all movements (station-keeping and ranging) made by seven important fishery species tagged between 2006 and 2010 in the Pondoland MPA. ▲ indicates mean fork/total length of recaptured fishes. Non-compatible measurements reported by the public were omitted, for example, where fork length was measured at release and total length was measured at recapture.

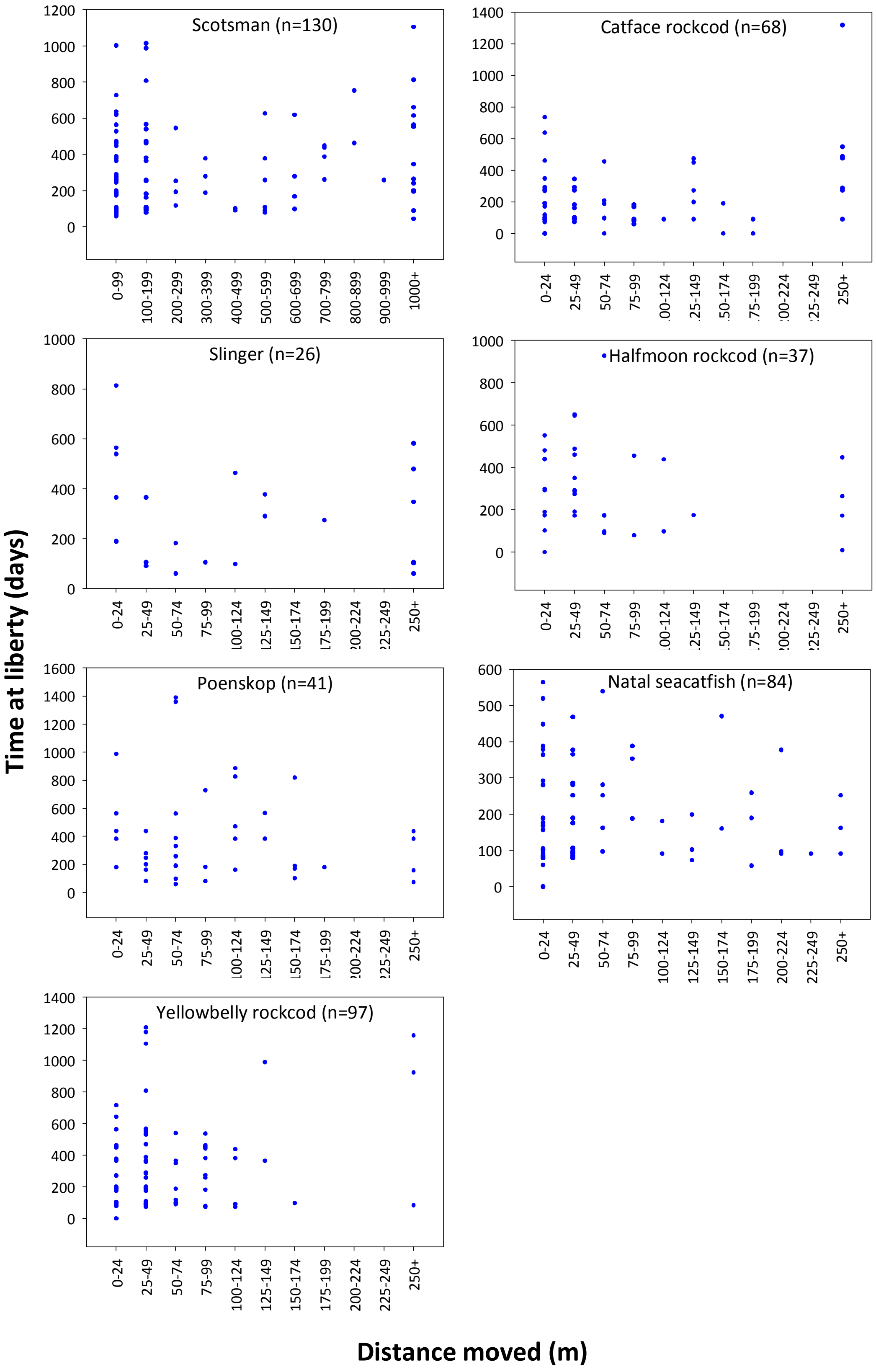


Figure 5.5. Scatterplot relationship between distance moved (m) and the number of days that recaptured fish were at liberty for seven important fishery species tagged between 2006 and 2010 in the Pondoland MPA.

Station-keeping (foraging within a home-range)

In all species and in all four study sites, it was found that the mean distance between fishing stations was significantly ($P<0.05$) greater than the mean distances moved by each fish species. This means fishing stations within each 2 km^2 study site were scattered over an area which was greater than the mean recorded movement distance of the tagged fish. This provided confidence in the home-range estimates.

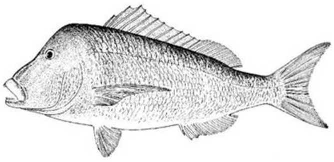
Single linear distance (SLD) estimates of home-range length (95th percentile of all localised first time recaptures) varied among the species, and the estimate for each species differed between the exploited and no-take areas (Table 5.3). Estimates of home-range length ranged from 73 m for catface and halfmoon rockcod in the no-take area up to 764 m for scotsman, also in the no-take area.

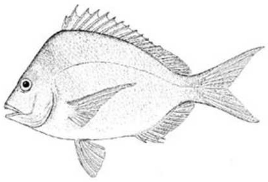
Multiple recaptures were relatively scarce but provided further insight into the home-range size of the species under investigation. The greatest linear distance (GLD) estimates obtained from individuals recaptured multiple times could be compared to SLD estimates, which used only two capture localities (Table 5.4). For this comparison, observations from both the exploited and the no-take areas were pooled as there were only a limited number of multiple recaptures (GLDs). SLD estimates of home-range length for catface rockcod and poenskop were substantially greater than their GLD estimates. Estimates for scotsman did not differ greatly, while those estimates for Natal seacatfish were identical. Yellowbelly and halfmoon rockcod had a GLD estimate greater than the SLD.

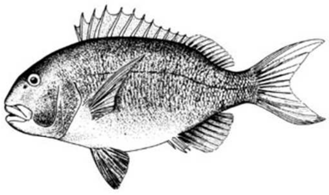
Based on the maximum body size recorded in this study for each species, home-range sizes predicted by Kramer and Chapman (1999) were mostly greater than estimates obtained by SLD and GLD values (Table 5.4). The SLD for scotsman was similar to the Kramer and Chapman (1999) prediction, while the SLD of slinger was substantially greater. Still, all home-ranges were considerably smaller than the size of the no-take area (Fig 5.6).

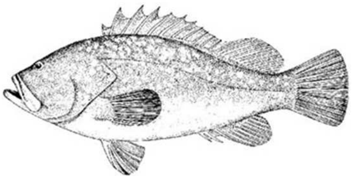
Table 5.3. Summary of localised movement statistics (intra-study site recaptures only) using single linear distances (SLD) for seven important fishery species tagged in exploited and no-take areas of the Pondoland MPA between 2006 and 2010. 95th percentile = SLD.

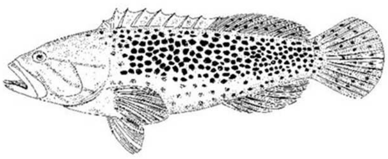
		Exploited area	**No-take area**
Scotsman *Polysteganus praeorbitalis*			
Number of observations		13	66
Distance moved (m)	median	211	60
	mean	249 (± 216)	164 (± 245)
	maximum	620	987
	95th percentile	**572**	**764**
Slinger *Chrysoblephus puniceus*			
Number of observations		0	24
Distance moved (m)	median		58
	mean		181 (± 254)
	maximum		709
	95th percentile		**696**
Poenskop *Cymatoceps nasutus*			
Number of observations		14	21
Distance moved (m)	median	70	70
	mean	119 (± 120)	121 (± 157)
	maximum	390	636
	95th percentile	**363**	**510**
Yellowbelly rockcod *Epinephelus marginatus*			
Number of observations		8	41
Distance moved (m)	median	29	34
	mean	44 (± 50)	59(± 108)
	maximum	154	699
	95th percentile	**127**	**118**
Catface rockcod *Epinephelus andersoni*			
Number of observations		26	12
Distance moved (m)	median	24	11
	mean	78 (± 111)	21 (± 27)
	maximum	418	79
	95th percentile	**292**	**73**
Halfmoon rockcod *Epinephelus rivulatus*			
Number of observations		20	6
Distance moved (m)	median	29	52
	mean	45 (± 55)	45(± 18)
	maximum	252	78
	95th percentile	**125**	**73**
Natal seacatfish *Galeichthys trowi*			
number of observations		25	24
Distance moved (m)	median	23	53
	mean	58 (± 88)	69 (± 82)
	maximum	356	372
	95th percentile	**229**	**196**

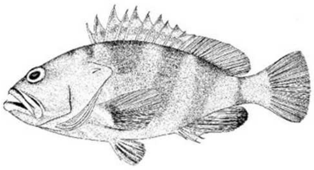












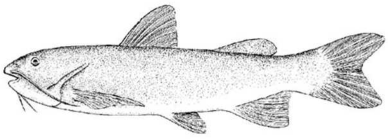


Table 5.4. A summary of single linear distance (SLD) and greatest linear distance (GLD) values (95th percentile) obtained in the current study compared with values obtained from the equation by Kramer and Chapman (1999), which were based on the maximum size of recaptured individuals in the current study. All values in meters.

	SLD (95th percentile)	GLD (95th percentile)	Kramer & Chapman prediction (max)
Scotsman	748	642	742
Slinger*	696		212
Poenskop	426	162	835
Yellowbelly rockcod	125	154	969
Catface rockcod	270	164	1016
Halfmoon rockcod	109	231	232
Natal seacatfish	223	223	359

*Home-range estimate limited to SLD observations obtained in no-take area

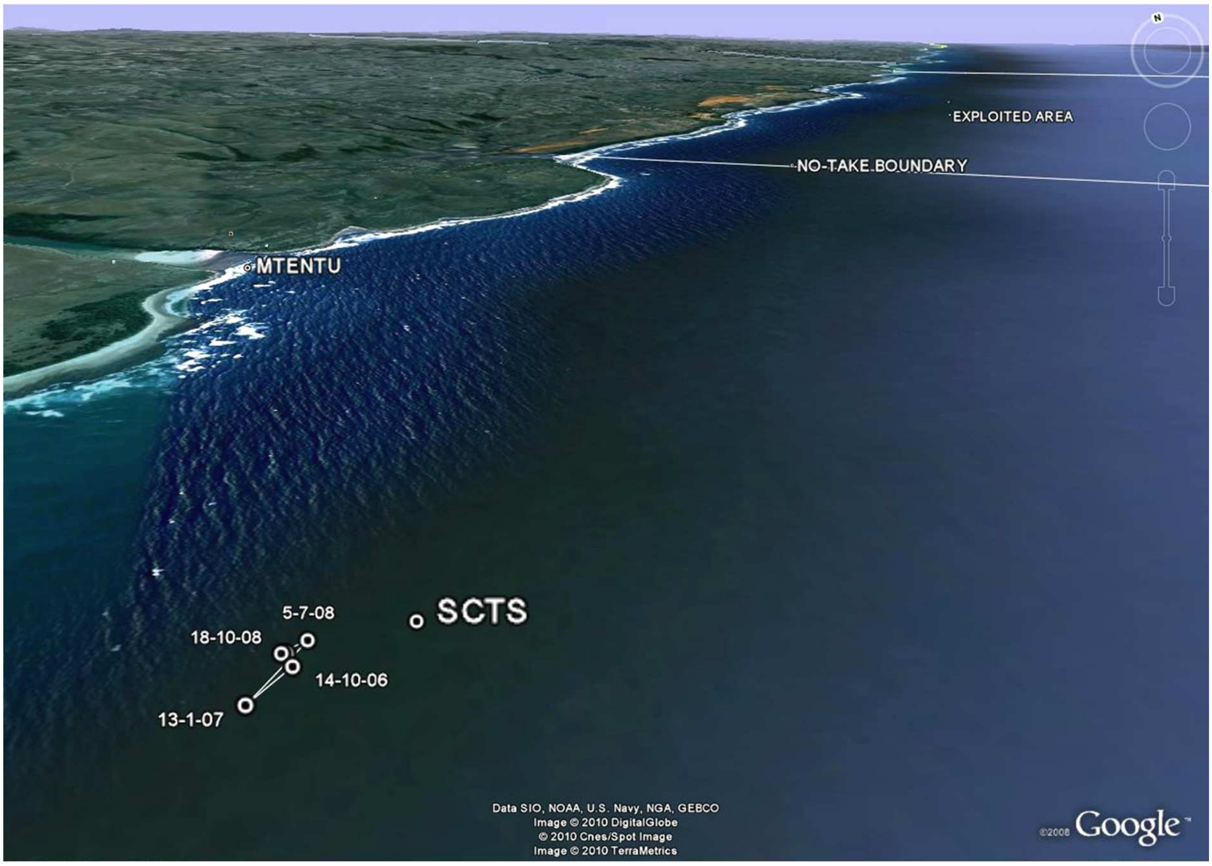


Figure 5.6. An example of a scotsman *Polysteganus praeorbitalis* tagged on the 14 October 2006 (31° 16.466’ S; 30° 2.93’ E) and subsequently recaptured on the 13 January 2007, 5 July 2008, 18 October 2008, 11 January 2009 and 23 January 2010. Not all dates could be displayed. *Image source: Google Earth™*.

Ranging (long-distance dispersal)

A very low proportion (4%) of recaptures for all studied species was recorded outside the 2 km^2 study sites in which they were tagged and released. Scotsman had the longest home-range and also the greatest proportion of ranging individuals of all seven species. No ranging movements were recorded amongst Natal seacatfish, poenskop or halfmoon rockcod. Although one halfmoon rockcod was recaptured 7 km south of its release site, this individual was suffering from barotrauma and was observed floating away when it was released. It is assumed that this individual was swept away from its home-range area by the southward flowing current.

With only two exceptions, fishes moving beyond the boundaries of the 2 km^2 study sites headed in a north–easterly direction, up the east coast of KwaZulu-Natal (KZN). One scotsman moved 7 km south from Mnyameni in the exploited area to Mtentu in the no-take area. Similarly, a catface rockcod moved 6 km south from Mnyameni to Mtentu. Most ranging movements took fishes out of the no-take area of the Pondoland MPA and into deeper water, as indicated by members of the public who had reported these recaptures.

It should be noted that the evaluation of dispersal patterns are hampered by the distribution of fishing effort in the region. No fishing is allowed for approximately 30 km south of the Mkambati study site because this area is protected by the Pondoland no-take area. Consequently, the probability of tagged fish being recovered from this area was zero. Although some poaching takes place in the no-take area, it is highly unlikely that an illegally recaptured fish would be reported. For these reasons, a thorough evaluation of dispersal patterns was not considered. Nonetheless, a summary of recaptured individuals of four species that abandoned their home-ranges are given below.

Six scotsman were recaptured nearby but outside of the 2 km^2 study site in which they were tagged. It is not clear whether these individuals, which had assumedly abandoned their home-range, were beginning a ranging movement or undertaking a movement of an intermediate nature, for example, a foraging sortie (Attwood 2002). One individual was recaptured only 56 m from where it was released 188 days earlier; but a subsequent recapture showed that this individual had abandoned its home-range and moved 320 km in 615 days. Besides the six scotsman recaptured close outside the study sites, another six undertook ranging north-eastward movements, travelling between 153 and 357 km from where they were originally tagged. Four of these originated from the no-take area.

The greatest movement recorded during this study was that of a slinger which covered nearly the entire distributional range of the species. This fish was tagged at Mtentu and recaptured by a commercial fishing vessel in Quissico, Mozambique after having travelled 1059 km in 582 days. Another slinger moved to Warner Beach on the KZN south coast, 163 km away. Both slinger had moved north-eastwards and were tagged in the no-take area.

Two yellowbelly rockcod undertook ranging north-eastward movements. One moved 187 km up the coast to Durban and the other, 342 km to Richard's Bay. Both were tagged in the no-take area. Three catface rockcod undertook ranging movements in a north-eastward direction. One of these moved 411 km from Mkambati in the no-take area to Cape St Lucia. Another two individuals moved from the Casino study site in the exploited area to Durban's No. 1 Reef (165 km) and to Blythedale (230 km) respectively.

5.4 DISCUSSION

This study has unequivocally shown that certain important fishery species are more abundant and larger inside the Pondoland no-take area compared to the adjacent exploited area (see Chps 3 & 4). As a consequence of their higher abundance, 48% more fishes were tagged in the no-take area than in the exploited area. Furthermore, because tagging was restricted to individuals over 300 mm, it also indicates a greater frequency of larger individuals in the no-take area. Yet, the recapture rate was greater in the exploited area. This was probably due to the reduced abundance of fishes in the exploited area and thus the probability of recapture was higher.

Eighty-three percent of all the fishes tagged in this study have not yet been recovered. Besides the fishes that remain at liberty, there are many reasons why a tagged fish may not be recovered. If these reasons are spatially or temporally governed, movement patterns may be confounded. Factors that may prevent a tagged fish from being recovered include immediate capture-induced mortality of the fish (directly or via opportunistic predation), delayed mortality (latent handling injury or natural), tag shedding and non-reporting by the public (Gillanders *et al*. 2001, Shirakihara and Kitada 2004).

Tag-induced mortality is extremely counter-productive. However, mortality probably has more to do with the entire process of capturing a fish rather than just the actual tag insertion. In this study, every effort was made to minimise the stress imposed on the fishes that were tagged. A significant finding of this study was that gut- and throat-hooked fishes, which had been released with the barbless hook still lodged inside, had often shed the hook suffering little obvious harm (Butcher *et al.* 2010).

It this study, it is likely that some recaptures made by the public were not reported. In a questionnaire survey of the KZN recreational line-fishery, Dunlop (2010) found that 42% of the tagged fishes caught by the public were never reported. Similarly, Brouwer *et al*. (1997) recorded a 44% non-reporting rate of fishes recaptured along the SA coast. In a tag-recapture study of ragged-tooth sharks *Carcharias taurus* in South Africa, Dickens *et al*. (2006) found that the rates of tag-reporting varied amongst provinces. Although the proportion of interviewed anglers which had caught tagged ragged-tooth sharks was lowest in KZN and highest in the Eastern Cape, tag-recoveries were more frequently reported in these two provinces than anywhere else.

The reasons for non-reporting may be due to a lack of knowledge or could be deliberate. For example, an angler is unlikely to report a tagged fish if it was retained illegally (Gillanders *et al.* 2001) or a community may be in opposition to a study because of an association between research and catch-restrictions. Contempt for the Pondoland MPA may cause anglers not to report tagged fish and in the worst case, to deliberately misreport recaptured fish. To make this problem worse, the anglers who are located closest to the MPA are the most likely to recapture fishes tagged in the MPA. These anglers are affected the most by the loss of fishing grounds and, consequently, are the most likely to have contempt for the study. In an attempt to alleviate this problem and create awareness, guest anglers were invited on field trips and presentations about the project were given at local ski-boat angling clubs.

Movement patterns

The stock recovery of over-exploited fishery species is dependent on the retention of fishes within no-take areas, whereas the benefits to adjacent fisheries rely on the dispersal (export) of fishes from the no-take area (Palumbi 2001, Zeller *et al.* 2003). Thus, two main objectives of area closure are potentially in conflict (Tolimieri *et al.* 2009). A balance between retention and export is needed for the long-term success of a no-take area (Kramer and Chapman 1999). If the spawner biomass is retained and limited larval exchange occurs, fish stocks will recover within the no-take area but the lack of benefits to nearby fisheries and support for the closure will rapidly diminish. Conversely, too much export will rapidly deplete spawner biomass within no-take areas and stock recovery will be compromised.

The concept of density-dependent spill-over implies, ideally, that fish biomass would build up within the boundaries of a no-take area and would only supply nearby fisheries with the "excess" biomass. Chapter 3 established that certain targeted line-fish species were more abundant and larger in the Pondoland no-take area, providing the prerequisite for spill-over (Gell and Roberts 2003b). However, the concept of spill-over is very simplistic and gives little attention to fish behaviour. It has been well established that many fishes maintain home-ranges (Attwood 2002) in which they may spend more than 95% of their time (Anderson 1982), while others abandon the home-range for a variety of reasons.

There was a clear division in the nature of movements in this study. Highly localised station-keeping behaviour was common and far outweighed occasional long-distance ranging movements in all species. The reason for this is uncertain but has been noted in other similar fish movement studies undertaken along the southern Cape coast of South Africa, particularly in the De Hoop MPA and the Tsitsikamma National Park.

Attwood (2002) found that 80% of recaptured galjoen *Dichistius capensis* were at the original release site, while a substantial fraction moved great distances (750 – 1300 km). The distance moved could not be associated with time at liberty but, with a few exceptions, fish which moved away from the release site were generally larger. All recaptures of tagged roman *Chrysoblephus laticeps* and dageraad *C. cristiceps* were reported by Buxton and Allen (1989) to have been within 2 km of the original release-site, with none leaving the study area. Kerwath *et al*. (2007) subsequently showed that the probability of recapturing tagged roman within a 100 m^2 cell exceeded 90% in two South African MPAs. Brouwer (2002) found that all juvenile red steenbras *Petrus rupestris* were recaptured on the same reef as where they were tagged, while all adults moved up the coast in a north-easterly direction and generally into deeper water. Brouwer *et al*. (2003) also found that carpenter *Argyrozona argyrozona* were mainly resident with a small proportion (7%) leaving the Tsitsikamma National Park. Cowley *et al*. (2002) showed that amongst four shore-angling species, 96% of recaptures were made within a 5 km research fishing area in the Tsitsikamma National Park. Although Brouwer (2002) found zero displacement among juvenile red steenbras and ranging behaviour exclusively amongst adults, none of these studies could find any clear correlation between the distances moved and fish size or time at liberty.

Griffiths and Wilke (2002) found that five sparid species; red steenbras, carpenter, santer *Cheimerius nufar*, roman and red stumpnose *Chrysoblephus gibbiceps* were highly resident, while a minority of individuals moved considerable distances. With the exception of carpenter, which moved offshore with growth, movement could not be related to fish size, sexual maturity or time at liberty, in contrast to the findings of Brouwer (2002).

In northern Australia, Russell and McDougall (2005) found that a small proportion of river snapper *Lutjanus argentimaculatus* had moved up to 315 km, while most remained close to the site of release. Residency was more noticeable among juveniles and ranging offshore movements were associated with time at liberty and larger fish. Along the east coast of Australia, Gillanders *et al*. (2001) found that most yellowtail *Seriola lalandi*, a wide-ranging migratory species, were recaptured within 50 km of the release site, while a small proportion made movements of 500-3000 km. Smaller fish showed less movement than large fish and it was indicated that fish at liberty for the longest, moved further.

In a multi-species mark-recapture study at Lizard Island on the Great Barrier Reef, Zeller *et al*. (2003) found that fish movements were highly localised and rarely took the fish away from the original release-site. In contrast to the other Australian studies, no clear relationship existed between distance moved and time at liberty. However, Zeller *et al*. (2003) did not record any ranging movement (maximum distance moved = 250 m) and thus their conclusions were likely based on station-keeping behaviour. It is highly doubtful that any correlations would be found between time at liberty and home-range utilisation.

Despite fishes moving out of the Pondoland no-take area and becoming available to the adjacent fishery, there was no evidence to suggest that this was caused by density-dependent spill-over. Many studies fail to discriminate between spill-over and individual variability in movement patterns (Zeller *et al.* 2003). Spill-over of excess fish biomass from a no-take area would imply that there is a gradual diffusion of individuals away from a high concentration of large fishes (Attwood 2002). This was not the case in the current study where some fish stayed resident for years and others moved long distances shortly after being released. No correlation existed between body size and distance moved, although movements over 150 km were generally undertaken by larger fishes. Furthermore, ranging movements were undertaken by fishes tagged in both the exploited and no-take areas, implying that export was not density-dependent and probably associated with individual variability.

Could this recurring pattern of movement differentiation (mostly resident, some nomadic) be linked to sample design? No research fishing was conducted outside of the two study sites in the Pondoland no-take area. No angling is allowed there and any recaptures obtained by poachers are unlikely to be reported. Therefore, scientific data collection would record localised movements within the study sites but not those movements that left the study site but remained in the no-take area. Only once a fish moved across to the exploited area would a recapture be possible. However, the same pattern (mostly resident, some nomadic) emerged from those recaptures which were originally tagged in the exploited area where fishing effort is well distributed. Telemetry studies are not reliant on recapture of tagged fish as continuous passive tracking gives a comprehensive picture of the everyday movements of fish tagged with transmitters. Using acoustic telemetry to study movement patterns of elf *Pomatomus saltatrix* in the Western Cape, Hedger *et al*. (2010) found that behaviour varied among individuals. Some elf remained resident and even spawned locally while others migrated great distances towards KZN. Many other studies using telemetry have shown similar movement patterns, with a high degree of station-keeping and a smaller element of nomadic behaviour (Egli and Babcock 2004, Kerwath *et al*. 2007, Childs *et al*. 2008).

Attwood (2002) proposed two models to explain such behaviour observed in galjeon. The polymorphic model, developed by Attwood and Bennett (1994), proposes that within a species, most individuals are highly resident while the remainder are nomadic and move continuously. The reason for the differentiation is uncertain but may be genetic or dependent on social or environmental conditioning (Swingland 1984, Dingle 1996). The tourist model was used by Craig and Hulley (1994) to describe sunbird movement and was subsequently adapted by Attwood and Cowley (2005) as an alternative explanation for galjoen movement. This model predicts that fishes will temporarily abandon their home-range to feed elsewhere but later return.

The difference in these two models is that with polymorphism, only certain individuals may become available to the adjacent fishery, whereas with the tourist model, all individuals may, at some time, become available to the fishery. Both of these models could account for the differentiation in movement observed in the present study but the study design was not able to discern between them. The possibility of both models being at work in a population should also not be discounted; certain individuals could be highly resident, some nomadic, while some maintain temporary home-ranges.

Should the differentiation be due to polymorhism, the effect of differential selection on different movement behaviours could pose a problem (Attwood 2002, Parsons *et al.* 2010). Fishing in an exploitable area would selectively remove fishes displaying resident behaviour while also opportunistically removing nomads leaving no-take areas. This would select for residency in no-take areas and nomadic behaviour in exploitable areas. The tourist model, where all individuals move, would not suffer from differential selection in the same way.

The experimental design in this study was able to adequately detect movement out of the no-take area but was weak in detecting movements into the no-take area. Immigrants may originate from elsewhere, which could obviously not be sampled in this study. For this reason, net export could not be quantified. Nevertheless, the observed differentiation in movement patterns appears to provide support for the objectives of the Pondoland MPA. The no-take area has the potential to continually export fish biomass to the adjacent fished area because the majority of the biomass is conserved in the no-take area. In the event of polymorphism, biomass would be retained in the no-take area, with a small proportion being exported. However, if movements are better explained by the tourist model, the majority of time spent in the no-take area would reduce the incidence of capture. In either model, an increase in abundance would most likely result in a greater occurrence of ranging movements, despite there being no evidence to suggest density-dependent spill-over.

Mounting evidence suggests that certain South African fish species periodically move up the east coast towards the northern extremities of their local distribution. Tag-recaptures in the ORI Tagging Programme have shown that the vast majority of ranging individuals in the studied species have been recaptured north-east of the original release-site. Reproductive studies have shown that in species displaying this behaviour, spawning is generally restricted to the northern regions of their distribution. This has been indicated by the general absence of reproductively active individuals in southern regions where there is a contrasting abundance of juveniles. It has thus been postulated that larger adults may migrate up the coast towards warmer water to spawn. However, bearing in mind the general southward movement of larvae toward nursery areas (Beckley 1993), this may also prevent eggs and larvae from drifting beyond southern distribution limits where environmental conditions are intolerable.

Many of the southern African endemic sparid species are believed to follow this pattern, including slinger (Buxton 1992, Garratt 1993b), poenskop (Buxton and Clarke 1989), red steenbras (Brouwer 2002), white steenbras *Lithognathus lithognathus* (Bennett 1993) and white musselcracker *Sparodon durbanensis* (Buxton and Clarke 1991, Watt-Pringle 2009). Along the east coast of Australia, the eastern Australian yellowfin bream *Acanthopagrus australis* (Sparidae) is also believed to follow a similar dispersal pattern (Roberts and Ayre 2010). In that region it is the East Australian Current which is thought to facilitate southward larval dispersal, while adults actively swim north to natal sites. There has also been indication that some serranids including yellowbelly rockcod and catface rockcod may follow a similar pattern in South Africa (Fennessy 2000).

Scotsman *Polysteganus praeorbitalis*

Scotsman displayed mostly resident station-keeping behaviour, maintaining small home-ranges with an element of ranging behaviour, which tended towards the north-east. This study presents the first known estimates of home-range for scotsman (642-748 m), which are very similar to those acoustic telemetry estimates derived for snapper *Pagrus auratus* (Sparidae) in New Zealand (650-903 m) (Parsons *et al.* 2003, Parsons *et al.* 2010), but much larger than for roman in South Africa (1 hectare) (Kerwath *et al.* 2007). Kramer and Chapman's (1999) prediction of 742 m, based on the largest scotsman captured in this study, is also very similar. Scotsman had the largest estimate of home-range in this study, yet this was still only a small portion of the length of the no-take area, thus encouraging biomass retention.

In addition to the station-keeping majority, six intermediate movements ranging between 2716 and 14129 m were observed, which are more difficult to explain. It is possible that the shorter movements were home-range relocations or that the species exhibits a bimodal home-range as noted in snapper (Parsons *et al.* 2010). The longer movements, on the other hand, may represent the beginning of ranging movements. A further six observations, between 153 and 357 km in length, represented true ranging movements. These 12 observations (9% of all observed) provide support for the export potential of scotsman from the Pondoland no-take area.

Slinger *Chrysoblephus puniceus*

Slinger similary displayed mostly resident station-keeping behaviour, maintaining small home-ranges with an element of ranging behaviour, which tended towards the north-east. This study presents the first known estimates of home-range size for slinger (696 m) but is limited to a small sample size (n=24). This estimate is considerably larger than the 212 m predicted by Kramer and Chapman (1999) for the largest slinger captured in this study.

Slinger tend to concentrate in large numbers in certain areas (pers. obs.) with few males and many females constituting a shoal. Also a protogynous hermaphrodite, females change into males at approximately 240 mm FL (Garratt 1985) and larger males appear to defend a territory during the breeding season (Garratt 1993a). It is possible that slinger maintain relatively small home-ranges (perhaps territories) as predicted by Kramer and Chapman (1999) but that individuals may periodically relocate to other shoals in response to social cues. Younger developing males are perhaps driven from the shoal by older more dominant males or else leave in search of less competition.

Two ranging movements were recorded amongst the tagged slinger. Both originated in the no-take area and travelled 163 and 1059 km respectively in a north-eastward direction. These observations (7% of all observed) provide support for the export potential of slinger from the no-take area. In addition, the study by Garratt (1993b), implies that the MPAs in northern KZN and the Pondoland MPA may work in tandem to successfully manage a valuable fishery resource. It is strongly suspected that there is a northward movement of adult slinger with more juveniles tending to recruit in the south towards Pondoland.

Poenskop *Cymatoceps nasutus*

Only station-keeping behaviour was recorded for poenskop; however, the ORI Tagging Programme also suggests an element of nomadic behaviour. This study presents the first known estimates of poenskop home-range size (162-426 m). The estimate using SLD (426 m) was more than twice that calculated by GLD (162 m). The GLD, which uses multiple recaptures is more reliable but perhaps the limited number of multiple recaptures used in the calculation of GLD (n=5) did not detect the true home-range usage. It is also possible that poenskop maintain atypical home-ranges (bimodal?), which skew the SLD estimate. Kramer and Chapman's (1999) prediction (835 m) is substantially larger than both estimates indicating a higher degree of site-fidelity than typically expected. Continued tagging of poenskop and the use of telemetry would shed more light on home-range usage by poenskop. Nevertheless, the home-range of this species is not expected to be substantially different from the estimates obtained.

The distribution of poenskop is from Cape Agulhas on the Cape south coast up to St Lucia on the KZN north coast (Smith and Heemstra 1986), although some may reach as far north as Maputo (van der Elst 1993). Moving up the east coast, large poenskop also tend to prefer progressively deeper water as suggested for other sparids. Along the lower Eastern Cape, large poenskop occur in water <25 m (Buxton and Smale 1984), in the upper Eastern Cape shallower than 40 m, whereas in KZN they are mostly caught much deeper between 40 and 80 m (Buxton and Clarke 1989). Reproductively active individuals are only known to occur in the upper Eastern Cape (Buxton and Clarke 1989) and possibly in southern KZN. Considering the absence of reproductively active individuals west of Port Elizabeth, Buxton and Clarke (1989) suggested that mature individuals may migrate up the east coast to spawn on deep offshore reefs resulting in an element of dispersal. It is therefore considered highly likely that the no-take area of the Pondoland MPA protects a large proportion of reproductively active, adult poenskop, especially on reefs deeper than those sampled in this study. These individuals would be capable of enhancing adjacent exploited populations through seeding of eggs and larvae.

Yellowbelly rockcod *Epinephelus marginatus*

Yellowbelly rockcod displayed mostly resident station-keeping behaviour, maintaining very small home-ranges with an element of ranging behaviour, which tended towards the north-east. This species displayed the highest degree of site-fidelity in this study, possibly due to its territoriality (Froese and Pauly 2010) and is thus the most susceptible to localised depletion by fishing. In the Mediterranean Sea, Lembo *et al*. (2002) used acoustic transmitters to estimate the home-range *area* of juvenile yellowbelly rockcod. Assuming a perfectly circular home-range, the diameter (i.e. linear distance) of such estimates would range between 46 and 154 m. The shape of a home-range would depend largely on the reef topography, making this an unrealistic assumption. Nevertheless, this estimate provides some measure of confidence to those obtained in this study (125-154 m). Kramer and Chapman's (1999) prediction of home-range size is likely to be an overestimation in the case of yellowbelly rockcod (969 m), suggesting higher than average site-fidelity.

Although yellowbelly rockcod were reported by Fennessy (2006) to spawn along the whole KZN coast, only limited spawning was reported to occur south of KZN possibly due to low water temperatures. This may provide the motivation for individuals to migrate north to spawn. Two movements by yellowbelly rockcod were ranging and both were in a north-easterly direction up the KZN east coast originating in the no-take area. These two observations (2% of all observed) represent the export potential of yellowbelly rockcod from the Pondoland no-take area.

Catface rockcod *Epinephelus andersoni*

Catface rockcod displayed mostly resident station-keeping behaviour, also maintaining very small home-ranges with an element of ranging behaviour, which tended towards the north-east. This study presents the first known estimates of catface rockcod home-range size (164-270 m). Similar to yellowbelly rockcod, Kramer and Chapman's (1999) prediction of 1016 m for the largest individual recorded in this study is likely to be unrepresentative of catface rockcod.

Chapter 3 cited evidence that catface rockcod may be a roaming pioneer species, but tag-recapture data presented in this chapter does not provide enough evidence to support this idea. It is interesting to note, however, that the majority of catface rockcod recaptures were only at liberty for a relatively short period (Fig 5.5). It is thus still possible that these results reflect temporary residency. Anecdotal evidence suggests that this species aggregates to spawn and it is possible that adults leave their home-range and return later. Perhaps the movement patterns of catface rockcod are more consistent with the tourist model; individuals may be highly resident, but periodically scout areas further away (consistent with diffusion – random independent movements). This would possibly lead them to unoccupied niches left by other species removed by fishing. Catface rockcod are thus a good candidate for telemetry studies.

No spawning of catface rockcod is known to occur in the Pondoland region possibly due to low water temperatures (Fennessy and Sadovy 2002) and this may provide the motivation for individuals to migrate north to spawn. Three movements by catface rockcod were ranging and all moved in a north-eastward direction up the KZN east coast. One originated in the no-take area and the other two in the exploited area. These three observations (4% of all observed) indicate the export potential of catface rockcod from the no-take area.

Halfmoon rockcod *Epinephelus rivulatus*

Only station-keeping behaviour was recorded for halfmoon rockcod in this study as well as in the ORI Tagging Programme. In Western Australia, Mackie (1998) estimated home-range *area* of halfmoon rockcod at 58-200 m^2 (max 389-434 m^2) using a tag-resighting study. The diameter of such areas is approximately 9-16 m (max 22-24 m). Estimates of home-range length obtained in the current study were considerably longer (109-231 m) but more similar to the prediction by Kramer and Chapman (1999) of 232 m. This indicates a lower degree of site-fidelity than observed among the other serranids in this study.

Natal seacatfish *Galeichthys trowi*

Only station-keeping behaviour was recorded for Natal seacatfish, providing the first known estimates of home-range size for this species (223 m). This is smaller than the prediction by Kramer and Chapman (1999) of 359 m, suggesting higher than average site-fidelity. Diving observations suggest that Natal seacatfish are slow-moving predators often associated with high-relief reefs including caves and overhangs. This habitat preference may result in this species having smaller home-ranges on the scattered reefs found along the Pondoland coast.

5.5 CONCLUSION

The most important finding of this chapter is that the Pondoland MPA is achieving its primary objectives. Based on their movement behaviour, exploited fishery species are afforded sufficient protection to ensure stock rebuilding. The no-take area also has the potential to sustainably contribute fish biomass to adjacent fisheries. In some cases exported fishes covered hundreds of kilometres, which appeared to be independent of protected fish density, size and time at liberty. No evidence was found to support density-dependent spill-over, but the tag-recapture data provided conclusive proof that fishes crossed the boundaries of the no-take area.

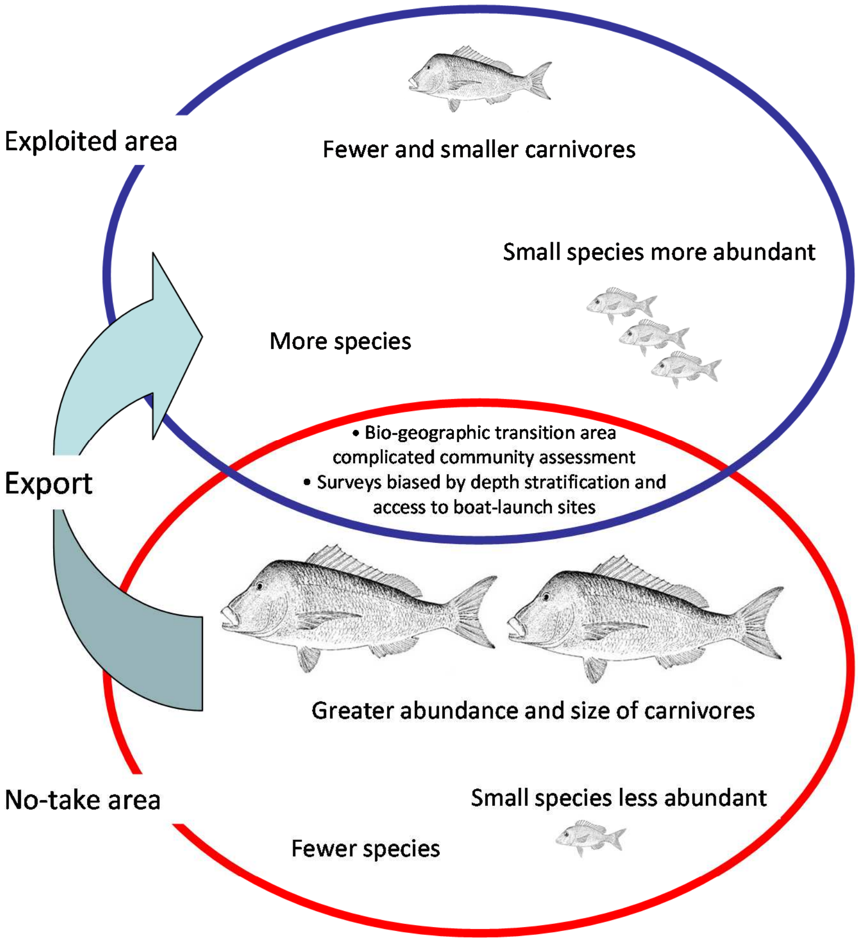
CHAPTER 6 GENERAL CONCLUSION

6.1 OVERVIEW OF FINDINGS

Notwithstanding any habit differences, there were some distinct differences between the fish communities in the Pondoland exploited and no-take areas (Fig 6.1). Some important line-fish species were significantly more abundant and of a larger mean size in the no-take area, which was expected following the closure to fishing. Many of these species are important predators, consuming both invertebrates and small fishes. As such, their greater abundance and body size in the no-take area was expected to have an influence on other species through competition and predation. Some important line-fish species were unexpectedly more abundant in the exploited area. These species may be more opportunistic and benefit from the reduced competition in the fished areas where the removal of dominant predators has made prey and habitat available. The abundance of carnivores, particularly piscivores, in the no-take area also appeared to have influenced other non-target species through predation. Thus, the assemblage of non-target fish species differed markedly between the two areas. Most importantly, small planktivores were abundant in the exploited area but contrastingly scarce in the no-take area.

Babcock *et al*. (2010) evaluated the recovery of fish stocks following fishery closure, reporting that most target species continued to increase, while some levelled off and others decreased. They attributed decreases to natural fluctuations, fishing effects outside of MPAs and indirect trophic effects. Species which show no trends following fisheries closure, may be lightly fished and therefore not influenced by protection, or the effect is undetectable using the chosen methods (Polunin and Roberts 1993). Weak trends in recovery can result from complex ecological factors (Jennings 2000, Halpern 2003, McClanahan *et al*. 2009) which may delay the detection of effects of fishery closure. Detection of reserve effects is further complicated by a lack of comparability between study areas due to habitat type, depth or topographic complexity (Bennett and Attwood 1991, Willis *et al*. 2003). Such problems are common in comparisons between no-take and exploited areas (Russ 2002).

In the current study, there were no clear changes over time between the fish communities in the exploited and no-take areas. It is possible that the sampling design was weak in detecting temporal trends in some fish species, or that the observed differences between the exploited and no-take areas may have already existed before the proclamation of the Pondoland MPA. It was established that due to their remoteness the reefs protected by the no-take area may have always experienced less fishing pressure than reefs closer to active boat launch-sites. It is also possible that ski-boat anglers fishing south of Sikombe River before the proclamation of the MPA, favoured deeper reef areas not sampled in this study. Alternatively, the four-year study period may have been too short to detect significant increases in abundance and body size in the Pondoland fish community where the majority of species are long-lived. For example, poenskop *Cymatoceps nasutus* may live for as much as 45 years. Protection is also less likely to have a great effect on populations that are not heavily exploited such as halfmoon rockcod *Epinephelus rivulatus*.



Controlled fishing

Advantages
- Less dependent on sea conditions
- Accurate length measurements and identification of fishes

Disadvantages
- Biased towards line-fish species
- Restricted to shallow water to reduce barotrauma

Underwater visual census

Advantages
- Direct underwater observation
- Less biased towards line-fish species

Disadvantages
- Dependent on water visibility
- Restricted to no-decompression depth for SCUBA divers
- Fish sizes are estimated
- Fish identification sometimes difficult

Figure 6.1. Schematic illustration of the main findings of the controlled fishing and underwater visual census surveys in the Pondoland MPA between 2006 and 2010.

A major pattern emerging from this study, which is consistent with most other endemic reef-associated fishery species, was the distinct *station-keeping* behaviour by all dominant species. The seven most abundant species in this study all occupied surprisingly small home-ranges (<1 km). However, a small proportion of these resident populations displayed *ranging* behaviour, as they abandoned their home-ranges and moved long distances up the KwaZulu-Natal east coast. While some species such as the seventy-four *Polysteganus undulosus* are known to undertake an annual spawning migration, it is suspected that in other species, individuals may move in isolation, for example scotsman *Polysteganus praeorbitalis*, and slinger *Chrysoblephus puniceus.*

When certain areas exploited by a fishery are closed to fishing as in a no-take area, an overall reduction of yield is expected, but fishes moving out of a closed area are expected to contribute to the yield in adjacent areas (Botsford *et al*. 2004). This begs the question of whether the volume of fishes moving north from the Pondland no-take area would be sufficient to compensate for the loss of fishing ground (and therefore yield) enclosed in the no-take area (Hilborn *et al*. 2004). However, this is perhaps a short-sighted question to ask. Most MPAs nowadays are proclaimed in response to declining fish stocks caused by over-fishing. It is unrealistic to expect a MPA to improve the yield in the adjacent area to a level achieved while over-fishing was taking place (Gell and Roberts 2003b). If the Pondoland no-take area is able to contribute to the yield in the adjacent fished area through the constant supply of adult fishes moving northward, this would in itself be a major achievement.

Another question arises as to whether the size of the Pondoland MPA is optimal. Although the minimum size of a protected area should be greater than a fish's home-range size for effective protection, it is very difficult to use home–range size to calculate optimum MPA size. For example, Griffiths and Wilke (2002) suggested that a no-take area should encompass three 'travel range lengths' – the radius containing 95% of all recapture movement. However, this does not take into consideration the total biomass protected or the available reef habitat in an area proposed for protection. The benefits of protection to a fish population depend on total biomass, not on the preservation of limited individuals. Protecting 10% of marine and coastal eco-regions as set out by the 10th Conference of the Parties (COP 10) to the Convention on Biological Diversity (CBD) provides a more holistic means of determining MPA size. The critical issue in this regard is to ensure suitable habitat protection within this 10% target.

This study has provided strong evidence showing that the Pondoland MPA is instrumental in conserving biodiversity and rebuilding depleted line-fish stocks. Export of fishes showed that the MPA also benefits adjacent fisheries and that it works well in conjunction with other MPAs (i.e. St

Lucia and Maputaland MPAs). This highlights the importance of an effective MPA network, which was seen as a priority at COP 10.

6.2 FUTURE MONITORING AND PROPOSED SAMPLING PROTOCOL

Surveys of the seafloor are extremely costly, but knowledge of the shallow sub-tidal environment along the Pondoland coast is imperative for proper monitoring and management of this MPA. It would be preferable that the entire continental shelf in the Pondoland region is bathymetrically mapped, but failing this a reef-mapping exercise of the four study sites is considered essential.

An optimised sampling protocol was developed for the warm-temperate Tsitsikamma MPA on the Cape south coast (Bennett 2007, Bennett *et al*. 2009). These authors suggested that controlled fishing and UVC were effective when used in conjunction. They found that during controlled fishing, two angler-hours per fishing station were sufficient to achieve low catch variability, high catch-per-unit-effort (CPUE) and a representative catch. It was also found that UVC transects were superior to point-counts in terms of efficiency, variability and bias (Bennett *et al.* 2009). It is suggested that this protocol would be the most suitable for the Pondoland marine environment. The following configuration is therefore suggested for future sampling and assessment of the effectiveness of the Pondoland MPA.

Layout of study sites

The rearrangement of study sites depends on the acquisition of accurate reef-mapping information. Ideally, replicate study sites should be situated in a line perpendicular to the coast. This would make fished sites equidistant from the current boat launch-site and protected sites equidistant from the no-take boundary, therefore reducing variability. However, the narrow continental shelf in the Pondoland region and the scattered nature of the reef in the area means that it is unlikely that sites situated perpendicular to the coast would be of similar depth and topography and therefore would not be comparable. It is therefore suggested that the current selection of study sites should remain; however, preliminary analysis should continue to consider the sites individually before pooling, as done in this study.

Monitoring trends in key line-fishes (Boat-based fishing)

- Quarterly sampling (January, April, July, October)
- Two ski-boats (alternating sampling between no-take and exploited areas)
- On each ski-boat, two people fishing, one person processing the catch

- Stratification (equal number of fishing stations in shallow, <20m and deep 20-30 m)
- Randomly chosen fishing stations within sampling sites and strata (based on reef-mapping)
- Maximum one hour, minimum 15 minute fishing stations
- Minimum of three hours fishing per study site
- All fishing at anchor
- Two barbless circle hooks per trace (1 x 4/0 and 1 x 8/0)
- Bait – pilchard *Sardinops sagax* and squid *Loligo vulgaris reynaudii*

Environmental parameters to record

- Wind direction
- Wind speed
- Water temperature (at depth of reef)
- Current direction
- Current speed

Monitoring trends in fish communities (UVC)

- Annual sampling (may/June)
- One ski-boat
- Randomly chosen diving stations (based on reef-mapping)
- Fixed diving depth and habitat type
- Three replicate strip transects (50x5 m) per 2 km^2 study site
- Two divers (one observer and one buddy diver)
- Two counts per transect (observer counts supra-benthic fishes while moving in one direction and then semi-cryptic benthic fishes on the return swim)
- Record substratum cover by means of photo-quadrats along each transect

Environmental parameters to record

- Water temperature (at depth of reef)
- Water visibility
- Reef relief/topography /substrate

Fish movement and export of fish biomass (tag-recapture study)

- Quarterly
- In tandem with boat-based fishing (see point 6.3.1)
- Tag indicator species with plastic “spaghetti” tags (key line-fish)

- Record GPS position of each fish tagged and recaptured
- Continue to encourage members of the angling public to report the capture of tagged fishes
- Annual sampling beyond the borders of the study sites in the no-take area to monitor fish movements which leave the sites but remain in the no-take area

Environmental parameters to record

- None

6.3 FUTURE RESEARCH

The assumption that small parts of an ecosystem closed to fishing automatically revert to pristine conditions is no longer valid (Botsford *et al*. 2009). Recovery of fish populations is a highly complex process and complete recovery may never take place. Recovery is dependent on many factors (Jennings 2000) but as the use of MPAs in fisheries management gains popularity, more in-depth investigations are focusing on how ecology and behaviour of fishes relates to no-take areas (Ormond and Gore 2005), with specific emphasis on movement. A long-term telemetry study in the Pondoland MPA would be most useful in gaining insight into real-time movements.

An understanding of meta-population connectivity through larval and adult dispersal is fundamental to the planning and evaluation of MPA networks. Special attention should be given to research on the location of spawning sites and nursery areas for key line-fish species in South Africa. The inclusion of these areas in MPAs should be seen as a priority.

The data collected in this study provides an opportunity for the calculation of valuable fish population parameters, especially natural mortality which is unknown in some important endemic line-fish species. The tag-recapture data also presents an opportunity to calculate population estimates. The publication of such valuable information in the future is warranted.

Some consideration should be given to the use of underwater video cameras, which can sample in much deeper water than fishing or SCUBA diving. Diving is constrained to depths shallower than 30 m due to legal restrictions and deep water fishing increases the chances of barotrauma injuries in captured fishes. Sampling in deeper water would shed light on the ontogenetic movements which are believed to take larger fishes into deeper water.

The KZN recreational boat-based fishery is becoming increasingly dependent on pelagic game-fish such as king mackerel *Scomberomorus commerson*, yellowfin tuna *Thunnus albacares* and dorado *Coryphaena hippurus.* In the future, the potential benefits of the Pondoland MPA to these more mobile species should be evaluated.

6.4 RECOMMENDATIONS FOR MANAGEMENT

Fishing can rapidly deplete a fish stock but recovery is a slow cumulative process dependent on longevity and life-histories of the species concerned (McClanahan 2000). MPAs with variable compliance can elevate fish biomass above levels typically found in fished areas, but recovery may be weak (McClanahan *et al.* 2009). Poaching does take place in the Pondoland MPA but the extent of this is unknown. Uncontrolled poaching may severely reduce the effectiveness of MPAs (Maliao *et al.* 2004) and it is therefore imperative that the Pondoland MPA should have uninterrupted closure to fisheries and strict law enforcement.

As per the Pondoland Marine Protected Area Management Plan (Anon 2006), awareness of the MPA needs to be increased. Amongst other objectives, it was suggested that the MPA and the zonal boundaries should be properly demarcated using highly visible beacons on the shore. It was also suggested that signage, including maps showing zonation and regulations of the Pondoland MPA, should be erected at all local boat launch-sites and formal access points. At the time of writing, only limited steps had been taken to increase awareness. This is a serious shortfall of the current management of the Pondoland MPA, especially considering its great value as suggested by this study. Furthermore, very limited compliance monitoring has taken place due to a lack of management capacity. Importantly, a patrol vessel has recently been acquired by Eastern Cape Parks and Tourism Agency at Mkambati. This vessel should be used to maintain a regular presence in the no-take area of the MPA to deter poaching and potential infringements of the regulations.

In conclusion, although essential to biodiversity conservation and fisheries management, MPAs should not be seen as a panacea for fisheries management. Areas open to fishing should continue to be managed by means of catch and effort limiting mechanisms such as bag-limits, size-limits and closed seasons (Hilborn *et al.* 2006).

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APPENDICES

Appendix 1. Catch composition from controlled fishing experiment in the exploited area and no-take area of the Pondoland MPA between April 2006 and January 2010, listed in alphabetical order by family name. The number of individuals per species observed in both survey areas and their relative proportion (%) are also given.

Species	Common name	Exploited area		No-take area		Whole area	
		Number	%	Number	%	Number	%
Ariidae							
Galeichthys trowi	Natal seacatfish	236	15.92	408	18.23	644	17.31
Balistidae							
Sufflamen fraenatum	Bridle triggerfish	4	0.27	1	0.04	5	0.13
Carcharhinidae							
Carcharhinus brevipinna	Spinner shark	1	0.07	2	0.09	3	0.08
Carcharhinus obscurus	Dusky shark	3	0.20	2	0.09	5	0.13
Mustelus mustelus	Smooth-hound	9	0.61	1	0.04	10	0.27
Scylliogaleus quecketti	Flapnose houndshark	41	2.77	25	1.12	66	1.77
Coryphaenidae							
Coryphaena hippurus	Dolphinfish	3	0.20	4	0.18	7	0.19
Dasyatidae							
Dasyatis chrysonota	Blue stingray	1	0.07	1	0.04	2	0.05
Gymnura natalensis	Backwater butterflyray	1	0.07	1	0.04	2	0.05
Taeniura melanospilos	Round ribbontailray	2	0.13	2	0.09	4	0.11
Dinopercidae							
Dinoperca petersi	Cavebass	20	1.35	10	0.45	30	0.81
Haemulidae							
Plectorhinchus chubbi	Dusky rubberlip	4	0.27	7	0.31	11	0.30
Plectorhinchus schotaf	Minstrel	1	0.07	4	0.18	5	0.13
Pomadasys striatum	Striped grunter	1	0.07	0	0.00	1	0.03
Lethrinidae							
Lethrinus crocineus	Yellowfin emperor	1	0.07	4	0.18	5	0.13
Lethrinus nebulosus	Blue emperor	290	19.57	22	0.98	312	8.39
Lutjanidae							
Lutjanus argentimaculatus	River snapper	2	0.13	0	0.00	2	0.05
Mullidae							
Parupeneus rubescens	Blacksaddle goatfish	1	0.07	2	0.09	3	0.08
Muraenidae							
Gymnothorax favagineus	Honeycomb moray	2	0.13	0	0.00	2	0.05
Gymnothorax undulatus	Leopard moray	3	0.20	0	0.00	3	0.08
Plotosidae							
Plotosus nkunga	Eel-catfish	1	0.07	2	0.09	3	0.08
Pomatomidae							
Pomatomus saltatrix	Elf	1	0.07	3	0.13	4	0.11

Species	Common name	Exploited area		No-take area		Whole area	
		Number	%	Number	%	Number	%
Sciaenidae							
Argyrosomus japonicus	Dusky kob	0	0.00	6	0.27	6	0.16
Argyrosomus thorpei	Squaretail kob	0	0.00	24	1.07	24	0.65
Umbrina robinsoni	Baardman	0	0.00	1	0.04	1	0.03
Scombridae							
Scomberomorus commerson	King mackerel	1	0.07	1	0.04	2	0.05
Scyliorhinidae							
Haploblepharus kistnasamyi	Puffadder shyshark	1	0.07	0	0.00	1	0.03
Poroderma pantherinum	Leopard catshark	2	0.13	1	0.04	3	0.08
Serranidae							
Cephalopholis sonnerati	Tomato rockcod	1	0.07	0	0.00	1	0.03
Epinephelus andersoni	Catface rockcod	135	9.11	70	3.13	205	5.51
Epinephelus malabaricus	Malabar rockcod	1	0.07	0	0.00	1	0.03
Epinephelus marginatus	Yellowbelly rockcod	48	3.24	191	8.53	239	6.42
Epinephelus rivulatus	Halfmoon rockcod	117	7.89	65	2.90	182	4.89
Sparidae							
Boopsoidea inornata	Fransmadam	4	0.27	10	0.45	14	0.38
Cheimerius nufar	Santer	27	1.82	33	1.47	60	1.61
Chrysoblephus anglicus	Englishman	14	0.94	16	0.71	30	0.81
Chrysoblephus cristiceps	Dageraad	4	0.27	0	0.00	4	0.11
Chrysoblephus puniceus	Slinger	289	19.50	819	36.60	1108	29.78
Cymatoceps nasutus	Poenskop	76	5.13	94	4.20	170	4.57
Diplodus hottentotus	Zebra	4	0.27	7	0.31	11	0.30
Diplodus capensis	Blacktail	1	0.07	3	0.13	4	0.11
Pachymetopon aeneum	Blue hottentot	2	0.13	11	0.49	13	0.35
Pachymetopon grande	Bronze bream	3	0.20	1	0.04	4	0.11
Pagellus natalensis	Red tjor-tjor	0	0.00	1	0.04	1	0.03
Polyamblyodon germanum	German	11	0.74	2	0.09	13	0.35
Polysteganus praeorbitalis	Scotsman	110	7.42	375	16.76	485	13.04
Porcostoma dentata	Dane	0	0.00	2	0.09	2	0.05
Sphyrnidae							
Sphyrna lewini	Scalloped hammerhead	3	0.20	4	0.18	7	0.19
Total: 49 species		1482		2238		3720	

Appendix 2. Species composition during the underwater visual census in the Pondoland MPA between May 2007 and June 2010, listed in alphabetical order by family name. The number of individuals per species observed in both survey areas and their relative proportion (%) are also given. H, herbivore; O, omnivore; PL, planktivore; LC, lower carnivore; HC, higher carnivore; P, piscivore. * Endemic, ** possible new distribution or range extension.

Species	Common Name	Trophic Level	Exploited area		No-take area	
			Number	%	Number	%
Acanthuridae						
*Acanthurus blochii***	Tailring surgeon	H	4	0.15%		
*Acanthurus dussumieri***	Pencilled surgeon	H	4	0.15%	1	0.05%
Acanthurus nigrofuscus	Brown surgeon	H	2	0.08%		
*Zebrasoma gemmatum***	Spotted tang	H	6	0.23%		
Ariidae						
*Galeichthys trowi**	Natal seacatfish	HC	9	0.34%	44	2.21%
Balistidae						
*Sufflamen fraenatum***	Bridle triggerfish	LC	10	0.38%		
Blenniidae						
Parablennius pilicornis	Ringneck blenny	O	1	0.04%	2	0.10%
Plagiotremus rhinorhynchos	Twostripe blenny	LC	3	0.11%	1	0.05%
Plagiotremus tapeinosoma	Piano blenny	LC	6	0.23%	1	0.05%
Caesionidae						
Caesio sp.	Yellow striped fusilier	PL	61	2.32%		
*Caesio teres***	Beautiful fusilier	PL	30	1.14%		
Carcharhinidae						
*Scylliogaleus quecketti**	Flapnose houndshark	HC	1	0.04%		
Chaetodontidae						
Chaetodon blackburnii	Brownburnie	LC	36	1.37%	7	0.35%
Chaetodon kleinii	Whitespotted butterflyfish	LC	1	0.04%		
Chaetodon madagaskariensis	Pearly butterflyfish	LC	1	0.04%		
*Chaetodon marleyi**	Doublesash butterflyfish	LC			5	0.25%
Cheilodactylidae						
*Cheilodactylus pixi**	Barred fingerfin	LC			1	0.05%
*Chirodactylus brachydactylus**	Twotone fingerfin	HC	10	0.38%	14	0.70%
*Chirodactylus jessicalenorum**	Natal fingerfin	HC	95	3.62%	43	2.16%
Cirrhitidae						
Cyprinocirrhites polyactis	Swallowtail hawkfish	PL	1	0.04%		
Dinopercidae						
Dinoperca petersi	Cavebass	HC	6	0.23%	10	0.50%
Ephippidae						
Platax teira	Longfin batfish	O	10	0.38%		
Haemulidae						
Plectorhinchus chubbi	Dusky rubberlip	HC	219	8.34%	305	15.32%

Species	Common Name	Trophic Level	Exploited area		No-take area	
			Number	%	Number	%
Plectorhinchus flavomaculatus	Lemonfish	HC	1	0.04%	1	0.05%
Pomadasys commersonnii	Spotted grunter	HC	3	0.11%		
Pomadasys olivaceum	Pinky	LC			261	13.11%
Pomadasys striatum	Striped grunter	LC	186	7.09%	26	1.31%
Labridae						
*Anampses caeruleopunctatus***	Bluespotted tamarin	LC	2	0.08%		
*Anampses lineatus***	Lined tamarin	LC	1	0.04%		
*Anchichoerops natalensis**	Natal wrasse	LC	4	0.15%	2	0.10%
*Bodianus bilunulatus***	Saddleback hogfish	LC	13	0.50%	18	0.90%
*Bodianus perditio***	Goldsaddle hogfish	LC	8	0.30%	1	0.05%
Cheilio inermis	Cigar wrasse	LC	1	0.04%		
Coris caudimacula	Spottail coris	LC	20	0.76%	14	0.70%
*Coris frerei***	Queen coris	LC	2	0.08%		
*Halichoeres cosmetus***	Adorned wrasse	LC	2	0.08%		
*Halichoeres lapillus***	Jewelled wrasse	LC	16	0.61%	3	0.15%
Labroides dimidiatus	Bluestreak cleaner wrasse	LC	31	1.18%	1	0.05%
Stethojulis interrupta	Cutribbon wrasse	LC	5	0.19%	1	0.05%
Thalassoma amblycephalum	Twotone wrasse	PL	17	0.65%	1	0.05%
*Thalassoma genivittatum***	Redcheek wrasse	LC	4	0.15%		
Thalassoma herbraicum	Goldbar wrasse	LC	8	0.30%		
Thalassoma lunare	Crescent-tail wrasse	LC	1	0.04%		
Lethrinidae						
Lethrinus nebulosus	Blue emperor	HC	193	7.35%		
Lutjanidae						
Lutjanus argentimaculatus	River snapper	P			4	0.20%
Mullidae						
Parupeneus indicus	Indian goatfish	LC	2	0.08%		
*Parupeneus macronema***	Band-dot goatfish	LC	2	0.08%		
Parupeneus rubescens	Blacksaddle goatfish	LC	41	1.56%	17	0.85%
Oplegnathidae						
*Oplegnathus conwayi**	Cape knifejaw	LC	7	0.27%	20	1.00%
*Oplegnathus robinsoni**	Natal knifejaw	LC	19	0.72%	11	0.55%
Pomacanthidae						
*Apolemichthys kingi**	Tiger angelfish	LC	6	0.23%	3	0.15%
Centropyge acanthops	Jumping bean	LC	2	0.08%	1	0.05%
*Centropyge multispinis***	Dusky cherub	LC	1	0.04%		
Pomacanthus rhomboides	Old woman	LC	50	1.90%	59	2.96%
Pomacentridae						
*Chromis dasygenys***	Bluespotted chromis	PL	162	6.17%	8	0.40%
*Chromis dimidiata***	Chocolate dip	PL	12	0.46%	2	0.10%
Chromis nigrura	Blacktail chromis	PL	26	0.99%		

Species	Common Name	Trophic Level	Exploited area		No-take area	
			Number	%	Number	%
*Pomacentrus caeruleus***	Blue pete	PL	1	0.04%		
Priacanthidae						
Priacanthus hamrur	Cresent-tail bigeye	HC	2	0.08%	1	0.05%
Pseudochromidae						
*Pseudochromis dutoiti***	Dutoiti	LC	6	0.23%		
*Pseudochromis natalensis***	Natal dottyback	LC	2	0.08%		
Ptereleotridae						
*Ptereleotris heteroptera***	Blacktail goby	PL	5	0.19%		
Scaridae						
Scarus ghobban	Bluebarred parrotfish	H	4	0.15%		
Sciaenidae						
Argyrosomus japonicus	Dusky kob	P			61	3.06%
Argyrosomus thorpei	Squaretail kob	P			1	0.05%
Umbrina robinsoni	Baardman	HC	11	0.42%	17	0.85%
Serranidae						
*Epinephelus andersoni**	Catface rockcod	P	1	0.04%	2	0.10%
Epinephelus marginatus	Yellowbelly rockcod	P	1	0.04%	47	2.36%
Epinephelus rivulatus	Halfmoon rockcod	P	25	0.95%	22	1.10%
*Pseudanthias cooperi***	Silver-streak goldie	PL	1	0.04%		
Pseudanthias squamipinnis	Sea goldie	PL	336	12.80%	26	1.31%
Serranus cabrilla	Comber	HC	1	0.04%		
Siganidae						
*Siganus luridus***	Dusky rabbitfish	H	5	0.19%		
Sparidae						
*Boopsoidea inornata**	Fransmadam	O	51	1.94%	67	3.37%
Cheimerius nufar	Santer	P			3	0.15%
*Chrysoblephus anglicus**	Englishman	HC	25	0.95%	28	1.41%
*Chrysoblephus puniceus**	Slinger	LC	412	15.70%	348	17.48%
*Cymatoceps nasutus**	Poenskop	HC	25	0.95%	25	1.26%
*Diplodus hottentotus**	Zebra	O	34	1.30%	71	3.57%
*Diplodus capensis**	Blacktail	O	76	2.90%	64	3.21%
*Pachymetopon aeneum**	Blue hottentot	LC	2	0.08%	18	0.90%
*Pachymetopon grande**	Bronze bream	H	17	0.65%	19	0.95%
*Polyamblyodon germanum**	German	LC	129	4.91%	75	3.77%
*Polysteganus praeorbitalis**	Scotsman	P	60	2.29%	88	4.42%
*Porcostoma dentata**	Dane	HC	21	0.80%	17	0.85%
*Rhabdosargus holubi**	Cape stumpnose	O	5	0.19%	98	4.92%
Sarpa salpa	Strepie	H			4	0.20%
*Sparodon durbanensis**	White musselcracker	HC			1	0.05%
Tetraodontidae						
Arothron immaculatus	Blackedged blaasop	LC	1	0.04%		

Species	Common Name	Trophic Level	Exploited area		No-take area	
			Number	%	Number	%
Canthigaster rivulata	Doubleline toby	O	1	0.04%		
Zanclidae						
Zanclus canescens	Moorish idol	O	1	0.04%		
Total: 91 species			2625		1991	