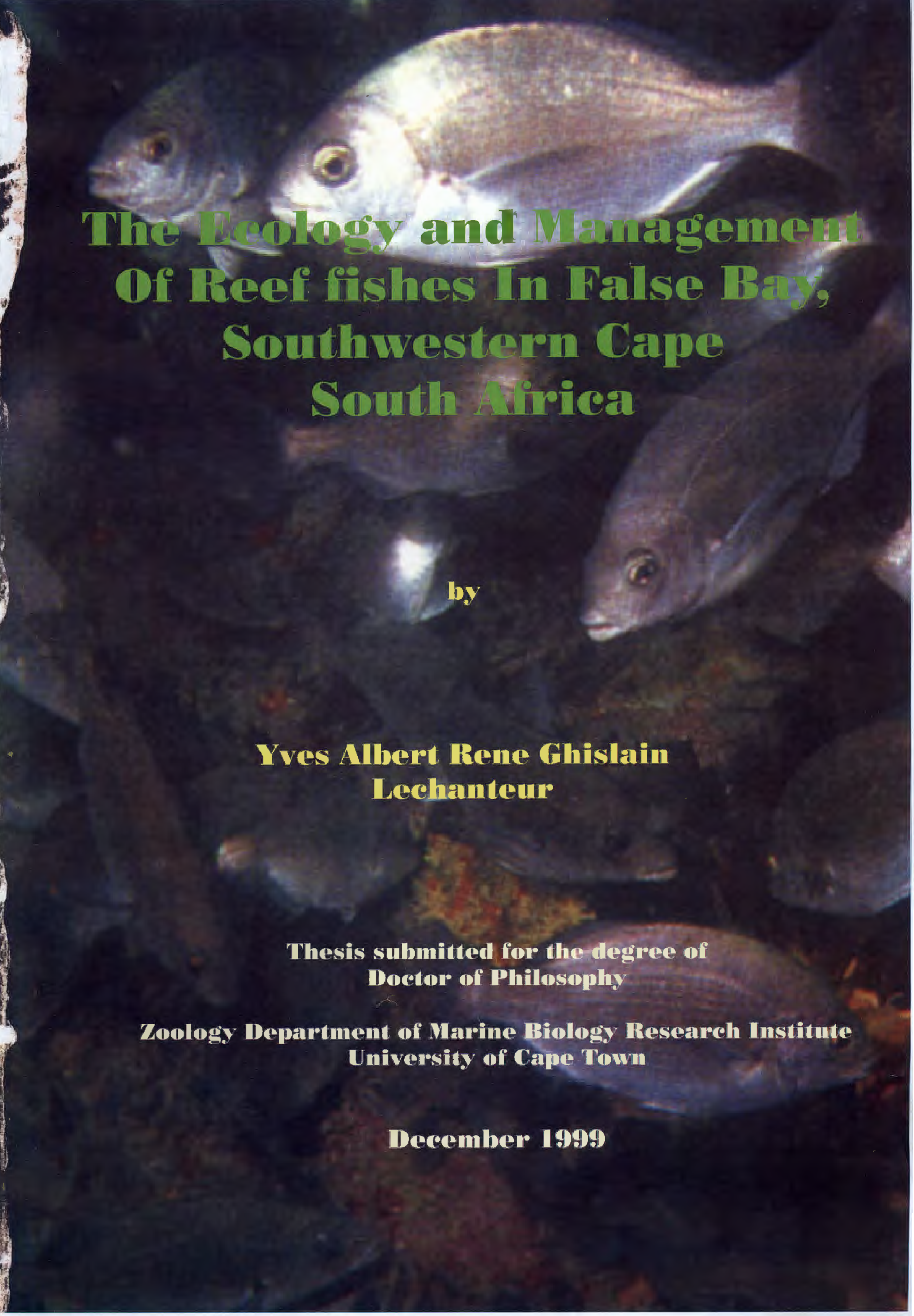

The Ecology and Management Of Reef fishes In False Bay, Southwestern Cape South Africa

by

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Thesis submitted for the degree of
Doctor of Philosophy

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University of Cape Town

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Declaration

This thesis is my own unaided work. It reports results of original research carried out in the Zoology Department, University of Cape Town, and has not been submitted for a degree at any other university.

Signed . . . . Signed . . . . Date . . 15/06/2000

Abstract

This thesis aimed to investigate the composition and seasonal variability of the False Bay suprabenthic reef fish assemblage, to determine which physical factors affect its composition, and to evaluate the fisheries management regulations and tools that are employed to manage reef fisheries within the Bay. The composition and seasonal variability of the suprabenthic reef fish assemblage was investigated by censusing approximately the same protected reef area monthly over a 14 month period. The five most abundant of the 26 demersal species (nine families) recorded (in decreasing order of abundance) were cape hottentot *Pachymetopon blochii* (30.6%), strepie *Sarpa salpa* (17.7%), fransmadam *Boopsoidea inornata* (16.1%), red roman *Chrysoblephus laticeps* (10.4%) and steentjie *Spondyliosoma emarginatum* (9.2%). Although the behavioural changes which some of the species undertook as water temperature dropped to 13°C or below resulted in their density appearing to vary seasonally, only the red stumpnose *Chrysoblephus gibbiceps* was found to migrate seasonally into False Bay. Water temperature, depth, the presence of large caves and vertical reef elevation, in isolation, all affect the spatial distribution and abundance of individual reef fish species and the reef fish assemblage as a whole. The investigation into the dietary requirements of 17 of the most abundant False Bay reef fish species revealed that most appear to feed within their own 'dietary niche', and that they consume predominantly reef-dwelling prey.

The total catch and catch composition of each of the five linefishery sectors active within the Bay were investigated to see which sector(s) impact which species most. The catch composition of the False Bay spearfishery has not previously been investigated in the Southwestern Cape. Both competitive and non-competitive spearfishing is practised within False Bay. Over 93% of the fish speared within the Bay are reef fishes. Factors affecting the catches of non-competitive spearfishers include whether spearfishers are boat- or shore-based, water temperature and experience. The catch per unit effort (cpue) of competitive boat-based spearfishers was almost twice that of non-competitive boat-based spearfishers. Using some of the information obtained above and assuming effort, the annual catch and catch composition of the spearfishery was estimated. This was combined with calculated catches for beach-seiners, commercial and recreational boat anglers, and shore anglers to estimate the annual total linefish catch taken from False Bay each year. Overall, an estimated 64 teleost and 12 cartilaginous species, along with additional species listed in 'unspecified' groups, were caught and retained annually between 1992 and 1995. Linefish species are targeted from all available habitats within the Bay, and many were shared by different sectors. However, the landings of only one fishery sector dominates catches of the most heavily impacted species (with respect to individual numbers landed). The value of some regulations enforced on False Bay fishers is discussed, along with the effectiveness of the enforcement. Some changes to regulations are proposed.

The last section of this thesis investigated the value of marine protected areas and artificial reefs as fishery management tools within False Bay. The management benefits provided by the 3.25km long Castle Rocks Marine Reserve were investigated by comparing the reef fish assemblage inhabiting the reserve to that inhabiting exploited reefs in the rest of False Bay. It was found that the reserve only benefits resident reef fish species, such as cape hottentot and red roman, although the protected stocks of many linefishery species comprised more large fishes. By comparing the cpue of non-competitive spearfishers active close to and further away from the reserve, it was also found that the reserve does not enhance linefish catches close to its borders by exporting large fishes. The fishery benefits provided by the reserve, therefore, are limited to the export of eggs and larvae, and juvenile fishes. Little is known about the former in this country. However, the management success of a small marine reserve within False Bay is dependent on the quality of reef it encompasses. The quality of reef found at a site is thus of utmost importance when considering where to establish small marine reserves within False Bay.

By comparing the fish assemblages inhabiting eight wrecks (six of which were sunk with the intention of enhancing reef fish productivity within False Bay) to those inhabiting exploited natural reefs, it was concluded that these wrecks provide little benefit to fisheries. It is proposed that this is because they were established in areas that are not optimal for the species they were intended to benefit. A number of points are proposed to maximise the fisheries benefits of future artificial reefs within the Bay. It is, however, hypothesised that the fishery benefits provided by a well-located artificial reef will not outweigh those provided by a well-located marine protected area. However, this should not be used as an excuse to not establish artificial reefs optimally for fishes in False Bay, as reef fisheries catches within the Bay are declining in quality.

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The collection of data during this thesis would have been near impossible without the help of a number of people. Thanks go to Paul Hanekom for always organising the required SCUBA and boating gear, at times instantly. More thanks go to the always keen Barry Clark and Cameron Smith, who regularly accompanied me at sea to undertake fish censuses. The fishes liked your chumming Cameron! Paul Hanekom, Adam West, Bruce Anderson and Steve Lamberth are also thanked for helping in this regard. Without the help of these people, data collection would have been far more difficult and time consuming.

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Prof. Charles Griffiths, Steven Mayfield, Dr Barry Clark and Charlotte Heijnis have all commented on rough drafts of some of the chapters in this thesis. They, at times, must have been frustrated with my language skills.

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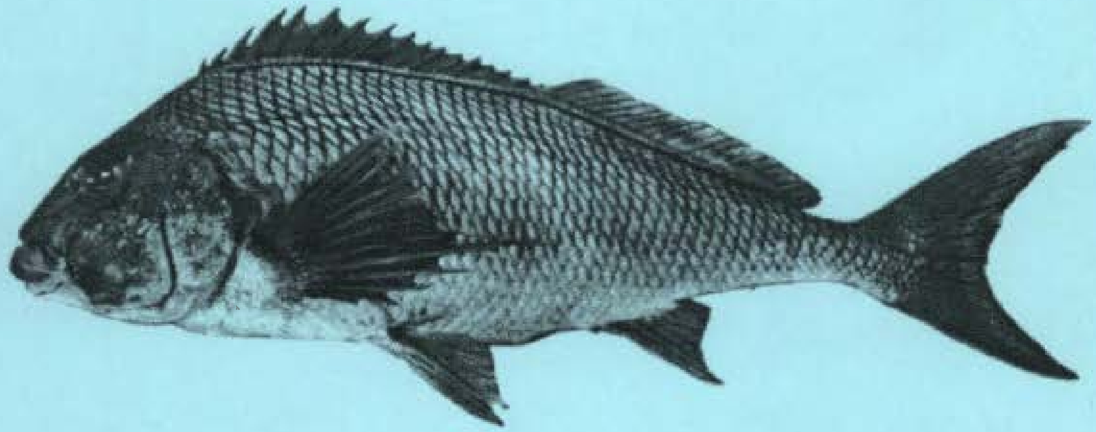
General introduction



Chirodactylus grandis

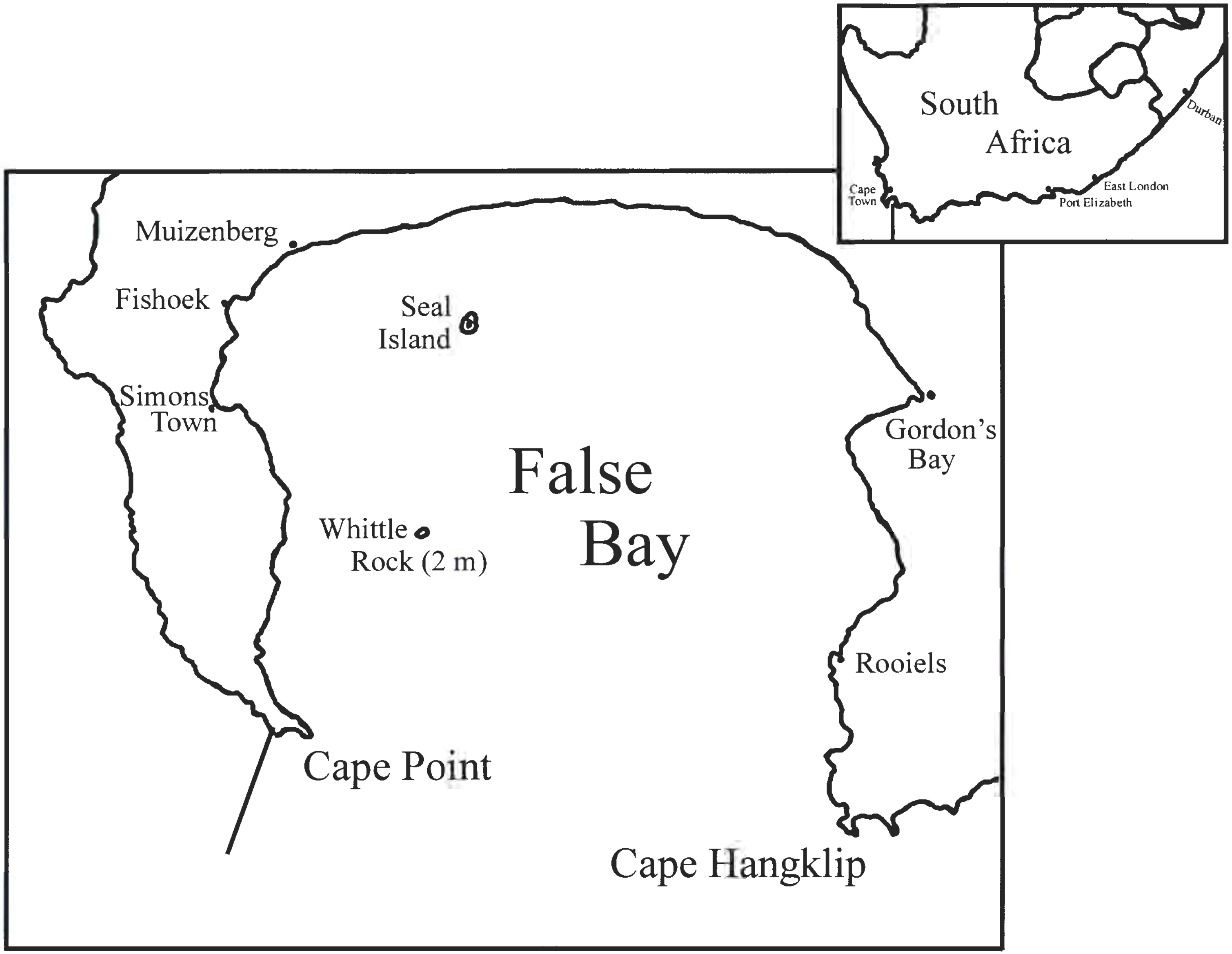


Figure 1 The location of False Bay along the South African coastline, and its salient features.

General introduction

This thesis is predominantly concerned with the composition, spatial distribution patterns, exploitation and management of the reef fish assemblage that inhabits False Bay reefs to a depth of 25 m. The Bay is located in the Western Cape Province of South Africa, between 34° 04' and 34° 23' south and 18° 26' and 18° 52' east, and for the purpose of this study is defined as the stretch of coastline between Southwest Reefs, just west of Cape Point, and Kleinmond, to the East of Cape Hangklip (Figure 1). Both its eastern and western shores are predominantly rocky, while its northern shore consists largely of sandy beaches (Jackson 1991). Most of the reefs on the western and eastern shores of the Bay extend from the shore to the sand interface at depths exceeding 15 m, while along the northern shore, most of the reefs are situated offshore.

Little information is available concerning the composition and abundance of the False Bay suprabenthic reef fish assemblage, and what is available has limitations. Van Herwerden (1989) reported the species composition of the reef fish assemblage encountered on a reef located on the west coast of the Bay. However, her study did not extend through the full depth range at her study site. The commercial and recreational catch records for the area (available from the National Marine Linefish System and published studies) are also of limited use, as they overestimate the abundances of species which are targeted, but underestimate those of other not. Chapter 1 of this thesis investigates the composition and size distribution of the suprabenthic fish assemblage that inhabits a stretch of reef (0-25 m deep) within the 3.25 km long Castle Rocks Marine Reserve located along the western shore of False Bay. The reefs encompassed within the reserve are representative of those of the region, and being protected from reef fisheries, are inhabited by the most pristine reef fish assemblage present in the Bay. By visually censusing the reef at monthly interval for 14 months, the study also aims to obtain an estimate of the seasonal variability of the assemblage.

Even less is known about which physical parameters affect the spatial distribution of False Bay reef fish species. Spearfishing (van Rooyen 1988) and angling (Schoeman & Schoeman 1990; Crous 1994) guides, along with published studies undertaken on the East coast of South Africa (Buxton & Smale 1989; Mann 1992; Mann & Buxton 1993) highlight the importance of depth, reef elevation and proximity of caves in affecting the spatial distribution of reef fish species. Furthermore, van Rooyen (1988), Buxton & Smale (1989) and Mann (1992) also report that water temperature affects the spatial distribution of some species. Chapter 2 sets out to investigate how important water temperature, depth, the size of caves available on reefs and vertical reef elevation are in affecting the composition and abundance of False Bay reef fish assemblages. How depth affects the distribution of individual species and the assemblage as a whole is tested by comparing the densities of species, reef teleosts, cartilaginous fishes and individual species obtained on reefs within different depth strata (to a maximum of 25 m). The importance of water temperature, the size of caves available on reefs and vertical elevation is tested in isolation of the other factors by comparing the above-mentioned relative abundances among reefs which only differ in the one factor whose importance is being investigated.

Another factor that could influence the spatial distribution of individual species is food availability (Buxton & Smale 1989; Mann 1992). Chapter 3 sets out to investigate the dietary requirements of 17 of the most abundant False Bay reef fish species to see whether each could have dietary requirements which influence its spatial distribution and abundance on reefs. Furthermore, by comparing the diets of these species, it also investigates niche separation among them. Finally, the dietary requirements of the suprabenthic reef fish assemblage are compared to those of the subtidal cryptic fish assemblage (Prochazka 1994) which co-occur on False Bay reefs.

Chapters 1, 2 & 3, therefore, investigate various aspects of the composition and reef requirements of the shallow water suprabenthic reef fish assemblage that inhabits False Bay.

The next two chapters of this thesis investigate the catch composition and total catch of the False Bay linefishery, and how successful fishery regulations are for managing it. False Bay is a very old South African fishing site, linefisheries having existed there since the early establishment of European settlements. Since then, linefishing effort within the Bay and along the remainder of the South African coast has increased considerably, resulting in linefish stocks coming under heavy fishing pressure. Presently, False Bay is the site of intense linefishing effort from three recreational (spearfishing, shore angling, recreational boat angling) and two commercial (commercial boat angling, beach-seining) sectors (van Rooyen 1988; Bennett 1991a; Penney 1991; Lamberth, Bennett & Clark 1994; Sauer, Penney, Erasmus, Mann, Brouwer, Lamberth & Stewart 1997). Effort is high as three commercial angling harbours (Kalk Bay, Simonstown, Gordon's Bay) and numerous slipways make the whole Bay accessible to boat-based fishers, while the roads that follow most of the Bay's shoreline make most of its coast available to shore-based fishers. Furthermore, the sandy beaches of the Bay are accessible to beach-seiners (Lamberth *et al.* 1994).

Chapter 4 investigates the catch composition and catch rates of the non-competitive and competitive spearfisheries active within the Bay, along with which factors affect their catches. It also discusses the effectiveness of the regulations that are enforced on False Bay spearfishers, and proposes changes to some of these if, based on the data obtained, these appear necessary to enhance fishery management and/or decrease the negative visual impact of this fishery sector.

In Chapter 5, the data from Chapter 4 are combined with information concerning catches from the other False Bay linefishery sectors (obtained either from published results or from reported catch returns) to obtain an estimate of the annual total linefish landings taken within False Bay. The composition of each sector's overall catch, in terms of fish numbers and/or mass, is calculated and compared to those of the other sectors to see which fishery sectors have the greatest impact (in terms of number of fishes landed) on particular species or groups of species. Based on the catch estimates obtained, changes to existing fishery regulations and enforcement methods are also proposed where these appear necessary.

Having investigated the annual landings of the False Bay linefishery, the last three chapters investigate the value of marine protected areas and artificial reefs as fishery management tools within False Bay. Four small (<5 km) and two larger (6.5 and 9 km) marine protected areas have been established within False Bay, these covering 24.5 km (16.7%) of the Bay's coastline. However, only within the Castle Rocks Marine Reserve and another small reserve are all reef fishes protected. The success of small Marine Protected Areas as fishery management tools has yet to be assessed, limiting our understanding of their potential as fishery management tools. Chapter 6, by comparing the protected fish assemblage inhabiting the 3.25 km long Castle Rocks Marine Reserve (situated on the west coast of False Bay) to that found on exploited False Bay reefs, investigates whether the protected assemblage is richer, more abundant and contains more large individuals than exploited assemblages. Furthermore, this chapter also tests the prediction that the reserve should enhance reef fish catches adjacent to its borders by exporting adult fishes, as does the larger De Hoop Marine

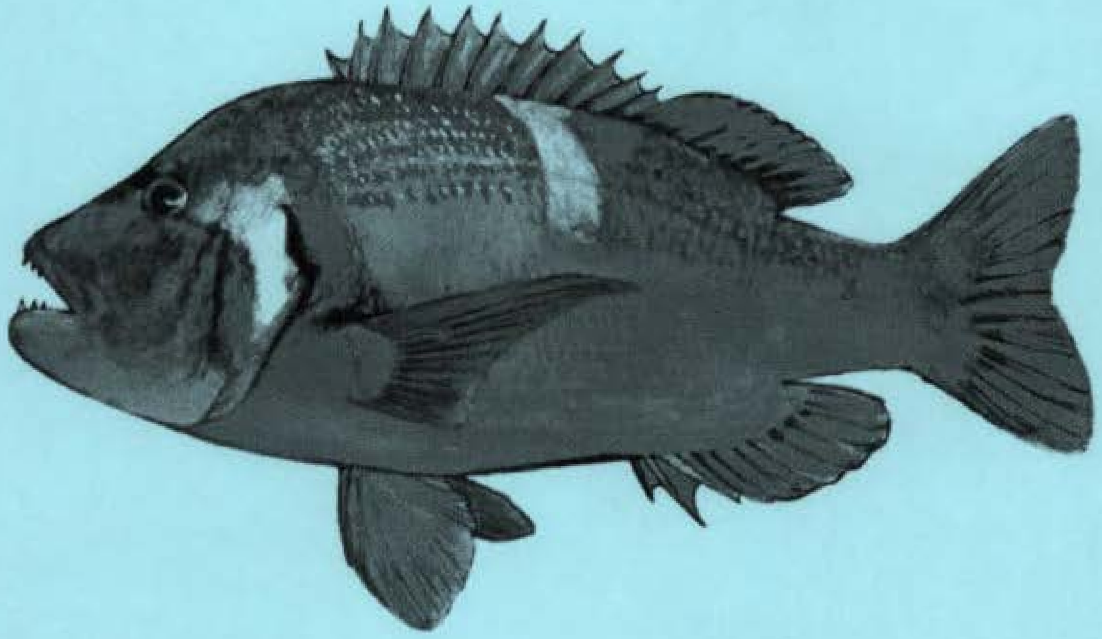
Reserve (Bennett & Attwood 1991, 1993a). This is done by comparing the catch-per-unit-effort (*cpue*) of non-competitive spearfishers active close to and further away from the reserve's borders. This chapter, by comparing the benefits of the Castle Rocks Marine Reserve to those of the larger (>45 km long) De Hoop Marine Reserve and Tstitsikamma National Park situated along the South and East coast of South Africa, also assesses the potential of small (<5 km) marine protected areas as fishery management tools. Changes in the size of, and regulations enforced within, the Castle Rocks Marine Reserve are proposed where this appears necessary.

Chapter 7 goes on to compare the protected reef fish assemblages that inhabit two separate protected areas. As these reserves encompass structurally different reefs, this chapter evaluates the importance of reef quality in affecting the species richness and relative abundance of reef fish assemblages, and the abundance and size of individual species on different reefs. The importance of reef quality in affecting the success of small marine reserves, and thus where these should be located within False Bay is also discussed.

The last chapter of this thesis investigates the value of artificial reefs as a fishery management tool within False Bay. Six ships have been scuttled within False Bay in the hope that they would enhance reef fish productivity in the area, while many others have sunk accidentally. Cliff (1983) reported on the fish colonisation of one of these wrecks during the 14 month period following its scuttling. His conclusion, concerning the use of artificial reefs for such a purpose, was rather negative. Chapter 8, by comparing the reef fish assemblages found on eight False Bay artificial reefs (all shipwrecks) to that which inhabits natural exploited reefs, investigates whether the artificial reefs are beneficial to the Bay's linefishery. The species richness, species and fish relative abundances, and size of fishes encountered on the artificial and natural reefs are compared. Based on the results of this and earlier chapters, the potential future use of artificial reefs as fishery management tools within False Bay is assessed, along with recommendations concerning where to establish future artificial reefs to maximise their benefits to reef fishes.

Chapter 1

Composition and seasonal variability of the suprabenthic fish assemblage in False Bay, South Africa



Chrysoblephus laticeps

Introduction

Catch records from recreational angling studies (*e.g.* Bennett, Attwood & Mantel 1994; Lamberth & Griffiths 1997), commercial catch records from the National Marine Linefish System, and a brief diving survey undertaken by Van Herwerden (1989) are the main source of information concerning the composition of the False Bay suprabenthic reef fish assemblage. However, the relative occurrences of targeted species tend to be overestimated by angling records compared to those that are not, while species that do not take baits (*e.g. Chirodactylus brachydactylus*, *C. grandis, Cheilodactylus fasciatus*) are also under-represented. The most efficient and unbiased method to investigate the composition and abundance of a reef fish assemblage is by undertaking a diving survey. Van Herwerden (1989) completed such a survey in False Bay, although she only undertook two daylong data collecting trips, and did not survey the full depth range at her study site.

Diving surveys of reef fish assemblages have been undertaken at several locations along the southern (*e.g.* Van Herwerden 1989), south-eastern (*e.g.* Zoutendyk 1982; Beckley & Buxton 1989; Buxton & Smale 1989; Burger 1991) and eastern coast (*e.g.* Berry, van der Elst, Hanekom, Joubert & Smale 1982; Chater, Beckley, Garrat, Ballard & van der Elst 1993) of South Africa. The surveys of Zoutendyk (1982), Van Herwerden (1989), Burger (1991), and Chater *et al.* (1993) did not consider seasonal variations in assemblage composition. Some of the other studies reported on seasonal assemblage variation, but focused only on a small portion of available reefs in the area (*e.g.* Berry *et al.* 1982), on a limited number of species (*e.g.* Buxton & Smale 1989) or recorded only presence/absence data (*e.g.* Beckley & Buxton 1989).

This study describes the species composition, abundance and seasonal variation of the suprabenthic reef fish assemblage on a protected False Bay reef. Investigating seasonal variation in composition is of particular importance as the potential seasonal variation in abundance of individual species could need to be taken into account during following chapters. The cryptic component of the assemblage, having been studied previously within close proximity of the study site by Prochazka (1994), was ignored during this study. The composition of the complete (cryptic and suprabenthic assemblages combined) fish assemblage is, however, elucidated by combining the data presented here with those of Prochazka (1994).

Methods

This study was undertaken in the Castle Rock Marine Protected Area on the west coast of False Bay (Figure 1.1). This is the only False Bay MPA within which no linefishing for reef fishes is permitted (Marine Living Resources Act 1998), resulting in its assemblage being the most undisturbed. It encompasses reefs to depths exceeding the 25 m depth limit of this study. To maximise the probability of encountering all the reef fish species present at the study site, each survey covered the full depth range from the inshore kelp bed and two sets of blinders/pinnacles. The offshore blinder/pinnacle is designated “Outer Castle” and rises from about 17 m almost to the surface, while the inshore one, designated “Pyramid Rock” rises from about 11 m and is more extensive than “Outer Castle” (Figure 1.2). No counts were undertaken below 25 m due to equipment, dive time and safety constraints.

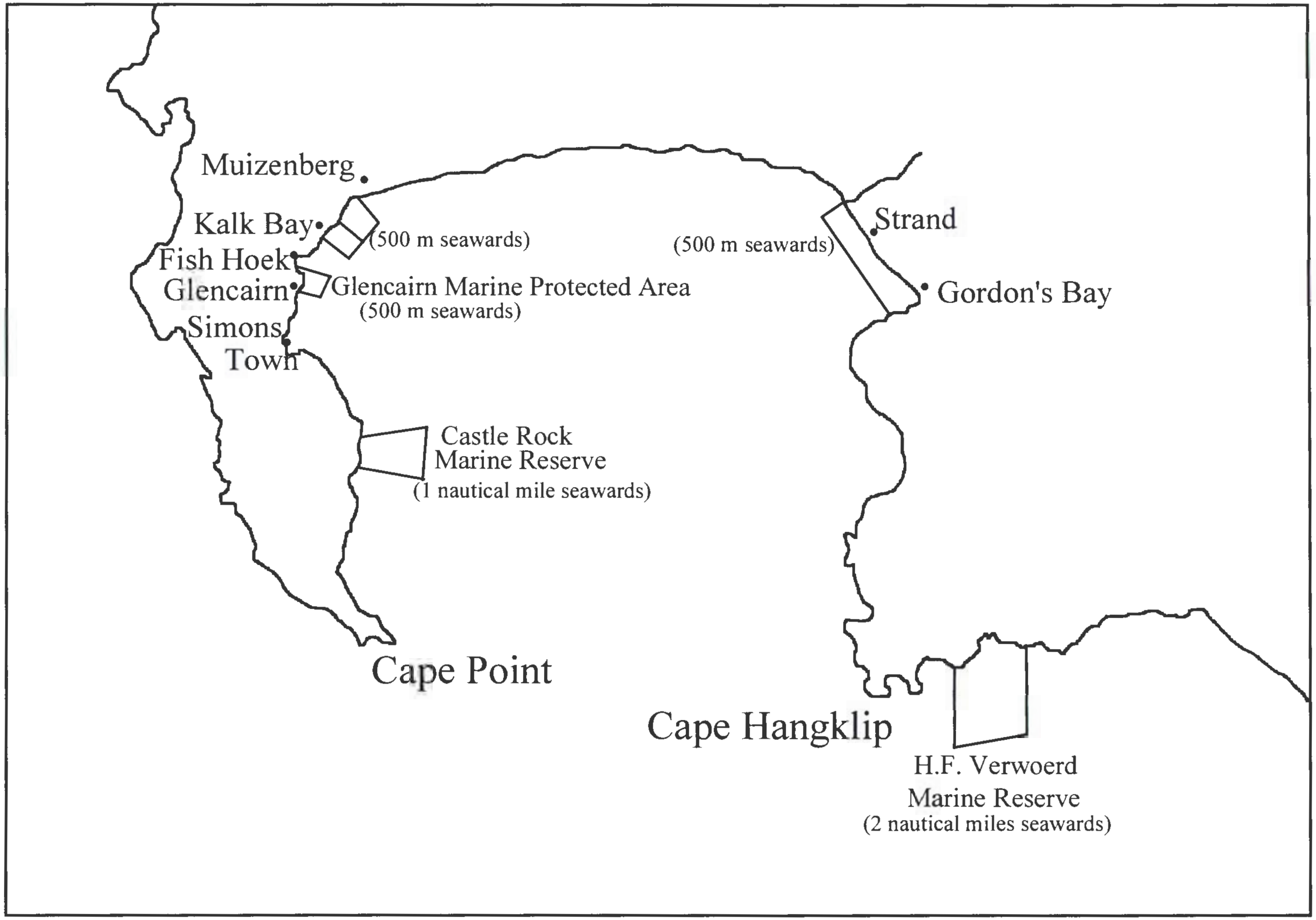


Figure 1.1 The location and offshore extent of the marine protected areas located within False Bay, after SFRI (1996).

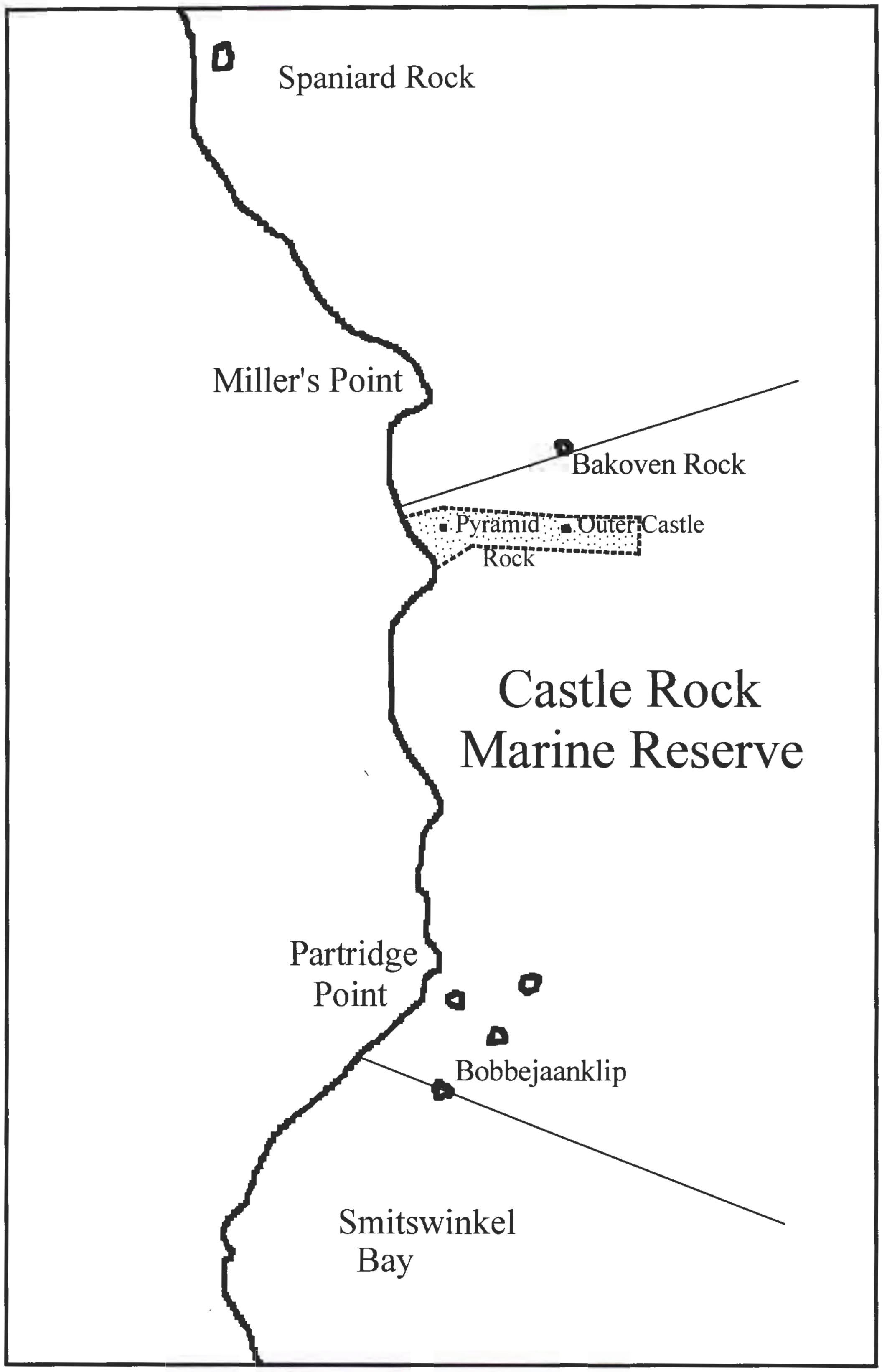


Figure 1.2 The location of the study reef (shaded area), in relation to physical features, within the Castle Rock Marine Reserve.

The fish assemblage inhabiting the study reef was censused once monthly from April 1993 to May 1994. Of the several non-destructive visual survey techniques that have been employed worldwide, we selected the instantaneous stationary point count census method, as it allowed for estimating the number of 'shy' (and thus scared of moving SCUBA divers) and 'inquisitive' reef fish species simultaneously. As with all approaches, there are advantages and disadvantages to this survey method, predominantly arising from the behaviour of the fish species being censused (Thresher & Gunn 1986; Buxton & Smale 1989; Burger 1991). Shallow reefs (<10 m depth) were censused mostly by breath-hold divers, while deeper reefs were censused by SCUBA divers. Each count was undertaken in the following way: a lone diver descended unobtrusively (not breathing in the case of SCUBA divers) onto the reef, and immediately identified and counted the fishes within his proximity. The diver noted fishes leaving the area as he descended onto the reef, before concentrating on 'inquisitive' species. Once counted, these were ignored while the diver searched for 'shy' species under rock cover. Fish size was estimated within 10 cm size classes. The accuracy of fish size estimation was tested on exploited reefs by estimating the size of individual fishes before spearing them and comparing estimated and actual sizes. Once fishes had been counted, the following reef parameters were also estimated from each site:

- Relative surface area of reef censused during that count
- Maximum vertical reef elevation within the area
- Size of caves (rock shelter within and/or under which a fish could hide) found within the area. These were categorized as none-to-small (shallow with only one exit) or medium-to-large (deep with more than one exit).
- Depth of the site (m)

In the case of the breath-hold diver, these parameters were noted either during the same dive or on a subsequent dive once the fishes had been censused. All information was recorded on pre-marked perspex slates. On completion of a count, the diver swam back to mid-water (or surface) and finned away far enough that the fishes encountered at the next site were unlikely to have been affected by his presence at the previous one. Counts were randomly distributed, rather than along a transect line with fixed points, to minimise fish disturbance. Individual counts took about three minutes to complete. At least 45 counts were completed during each census, 15 each from "Outer Castle", "Pyramid Rock" and the inshore kelp bed. The total reef surface area censused each month ranged from 1 000 - 1 500 m^2. Water temperature (on the bottom), weather and sea conditions were noted during each survey trip. Counts were subsequently categorised into the six depth strata described in Table 1.1.

Relative abundances (fish.100 m^{-2}) were calculated for each species encountered by dividing number of individuals by total reef area censused (all counts added together). In an attempt to elucidate seasonal changes in composition, monthly results were square-root transformed and compared using Bray-Curtis cluster analysis and multidimensional scaling techniques (Field, Clarke & Warwick 1982; Clarke 1993). Monthly relative abundances of individual species were also plotted to investigate seasonal shifts in abundance.

To evaluate the effect of depth on individual species, their relative abundances within each depth stratum was calculated and compared. To evaluate the effect of depth on the whole assemblage, the Shannon-Wiener overall index was calculated for each depth stratum using to the equation $H = -\Sigma(n_i/N)\ln(n_i/N)$

where N is the number of individuals of all species and n_i the number of individuals in each species.

Table 1.1 The six depth strata within which the 25 m depth limit of this study was sub-divided, along with a description of their sessile assemblage and susceptibility to wave-induced surge, and the occurrence of thermocline(s) within each

Stratum (m)	Dominant sessile macro-organisms	Susceptibility to wave-induced surge	Thermocline(s) occurrence
0-5	*Ecklonia maxima*	Often affected	Rare
>5-8	*Ecklonia maxima* *Laminaria pallida*	Less affected	Rare
>8-12	*Laminaria pallida* Filter- and deposit-feeders more abundant	Hardly affected by normal surge	Occasional
>12-16	Filter- and deposit-feeders *Laminaria pallida* present	Hardly affected by normal surge	Occasional
>16-20	Filter- and deposit-feeders	Only affected by surge during storms	Common
>20-25	Filter- and deposit-feeders	No surge	Common

Results

A total of 795 counts, covering 21 922 m^2 of reef, were completed over the 14 month study period. Water clarity ranged from 4->10 m. Altogether, 25 484 fishes representing 28 species and 11 families were counted (Table 1.2). *Liza richardsonii* and *Trachurus trachurus* were the only pelagic species encountered. The family Sparidae dominated suprabenthic species diversity (14 species) and total fish numbers (87.3%), followed by the family Cheilodactylidae, with four species totalling 7.5% of all counted fishes. After normalising for reef area from each depth stratum, the following were the most abundant species (in order of abundance): *Pachymetopon blochii* (30.6%), *Sarpa salpa* (17.7%), *Boopsoidea inornata* (16.1%), *Chrysoblephus laticeps* (10.4%), *Spondyliosoma emarginatum* (9.2%), *Cheilodactylus fasciatus* (4.6%), *Chirodactylus brachydactylus* (3.2%), *Diplodus sargus capensis* (2.1%), *Gymnocrotaphus curvidens* (1.5%), and *Chirodactylus grandis* (1%). Together, these made up over 93% of the reef-associated fish counted.

Eleven of the 26 demersal species censused were encountered each month, and a further five during more than 10 months. The remaining 10 species were less common, four only being encountered during one month (Table 1.2). Reef fish species richness varied between 18 and 20 each month, with no apparent seasonal trend.

Figures 1.3A & B show the results obtained from the cluster analysis and ordination performed on transformed monthly relative abundances. The relative abundance results first clustered at the 72% Bray-Curtis similarity level, highlighting the lack of obvious differences among months. Water temperature appears to have an effect on the apparent composition of the assemblage, however, as the first months to separate out (January 94, February 94, March

Table 1.2 Monthly species richness and fish relative abundances (fish.100 m^{-2}) encountered from April 1993 to May 1994 on the study reef located within the Castle Rocks Marine Reserve.

Species	Apr-93	May-93	Jun-93	Jul-93	Aug-93	Sep-93	Oct-93	Nov-93	Dec-93	Jan-94	Feb-94	Mar-94	Apr-94	May-94
Demersal teleost species														
Sparidae														
Boopsoidea inornata	30.6	28.7	30.2	15.9	25.7	20.5	18.1	9.7	16.7	9	10.8	3.2	13.3	28.4
Chrysoblephus gibbiceps	-	0.08	-	-	-	0.3	0.2	-	-	-	0.2	0.5	0.4	0.9
Chrysoblephus laticeps	12.3	11.5	12.9	13.2	16.8	17.1	14.5	8.7	9.8	3.7	3.4	8.1	7	11.7
Diplodus cervinus hottentotus	0.2	-	1.9	0.8	1.6	1.2	1.4	0.3	0.2	0.3	-	0.9	0.3	0.6
Diplodus sargus capensis	0.4	0.2	4.3	3.4	1.8	0.8	1.4	0.7	0.9	0.8	0.5	2.5	2.5	3
Gymnocrotaphus curvidens	4.4	2.2	2.1	1.9	2.5	1.6	1.5	0.5	1.2	1.4	0.2	0.3	1	3.3
Pachymetopon aeneum	0.3	1.4	0.1	0.1	1.1	0.2	0.4	0.2	0.1	0.1	0.07	-	0.1	0.5
Pachymetopon blochii	53.9	39.2	44	32.9	46.5	32.6	38.2	31.3	24.3	17	22.4	22.1	40	44.7
Petrus rupestrus	0.06	0.08	-	0.1	0.2	0.2	0.5	0.1	-	-	-	0.09	-	0.3
Rhabdosargus globiceps	-	-	-	-	-	-	-	0.07	-	-	-	-	-	-
Sarpa salpa	24.4	3.5	7.6	26	25.4	13.9	26.2	22.5	21.2	44.2	14.1	69.5	8.9	5.4
Spondyliosoma emarginatum	0.9	6.5	6.5	16	6	6.7	10.8	9.7	13.8	1.2	6.6	12.7	2.9	10.8
Cheilodactylidae														
Cheilodactylus fasciatus	7.9	4.3	5.7	4.3	4.6	8.8	3.7	1.7	3.5	1.5	1.1	2	2.1	6.8
Cheilodactylus pixi	-	-	-	-	-	-	-	-	-	-	-	-	-	0.09
Chirodactylus brachydactylus	0.5	5.1	4.2	5.7	5.2	5	2.6	0.9	2.6	1.8	0.7	1.4	2.1	5
Chirodactylus grandis	0.4	0.6	0.8	0.7	1.5	0.3	1.1	0.6	0.2	0.08	0.2	0.4	0.8	1.7
Coracinidae														
Dichistius capensis	0.6	0.7	0.5	0.5	1.5	1.2	0.2	-	1.2	0.2	-	0.8	0.2	1.6
Oplegnathidae														
Oplegnathus conwayi	0.6	0.7	0.2	1.3	0.5	0.8	0.08	-	0.6	0.08	-	-	0.09	0.1
Parascorpidae														
Parascorpis typus	0.6	0.3	0.5	0.8	0.7	1.2	0.9	0.3	0.6	0.1	0.1	0.3	0.3	0.6
Chaetodontidae														
Chaetodon marleyei	-	0.1	0.06	-	-	0.1	-	-	-	0.1	-	-	-	-
Pelagic teleost species														
Carangidae														
Trachurus trachurus	-	52.7	-	-	-	-	-	-	-	-	-	-	-	-
Mugilidae														
Liza richarsonii	-	-	-	-	-	-	-	-	-	-	-	-	9.9	-
Demersal cartilaginous species														
Carcharhinidae														
Mustelus mustelus	-	-	0.07	-	0.1	-	-	-	-	-	-	-	-	-
Triakus megalopterus	-	-	-	-	-	0.08	0.08	1.6	0.08	0.4	0.3	0.2	-	-
Hexanchidae														
Notorhynchus cepedianus	-	-	0.07	-	-	-	-	-	-	0.08	0.2	0.09	-	-
Scyliorhinidae														
Haploblepharus edwardsii	0.2	-	0.1	0.1	0.1	0.6	0.1	0.07	0.06	0.1	0.2	0.09	0.09	0.4
Haploblepharus pictus	0.4	-	-	0.05	0.1	-	0.1	0.05	-	0.09	0.1	-	-	-
Poroderma africanum	-	0.08	0.2	0.2	-	-	-	-	0.1	-	0.2	-	0.3	0.09
Water temperature (°C)	14.5	16.5	15	15	14.5	14.5	13.5	12.5	16.5	13.5	12	13	13	15
Overall totals														
Overall density of demersal fishes	144.7	104.7	122	124	142	113.2	122.1	89	97.1	82.2	61.4	125.2	82.4	126
Overall density of all fishes	144.7	157.4	122	124	142	113.2	122.1	89	97.1	82.2	61.4	125.2	92.3	126
Total number of species	18	19	20	19	19	20	20	18	19	20	18	18	19	20

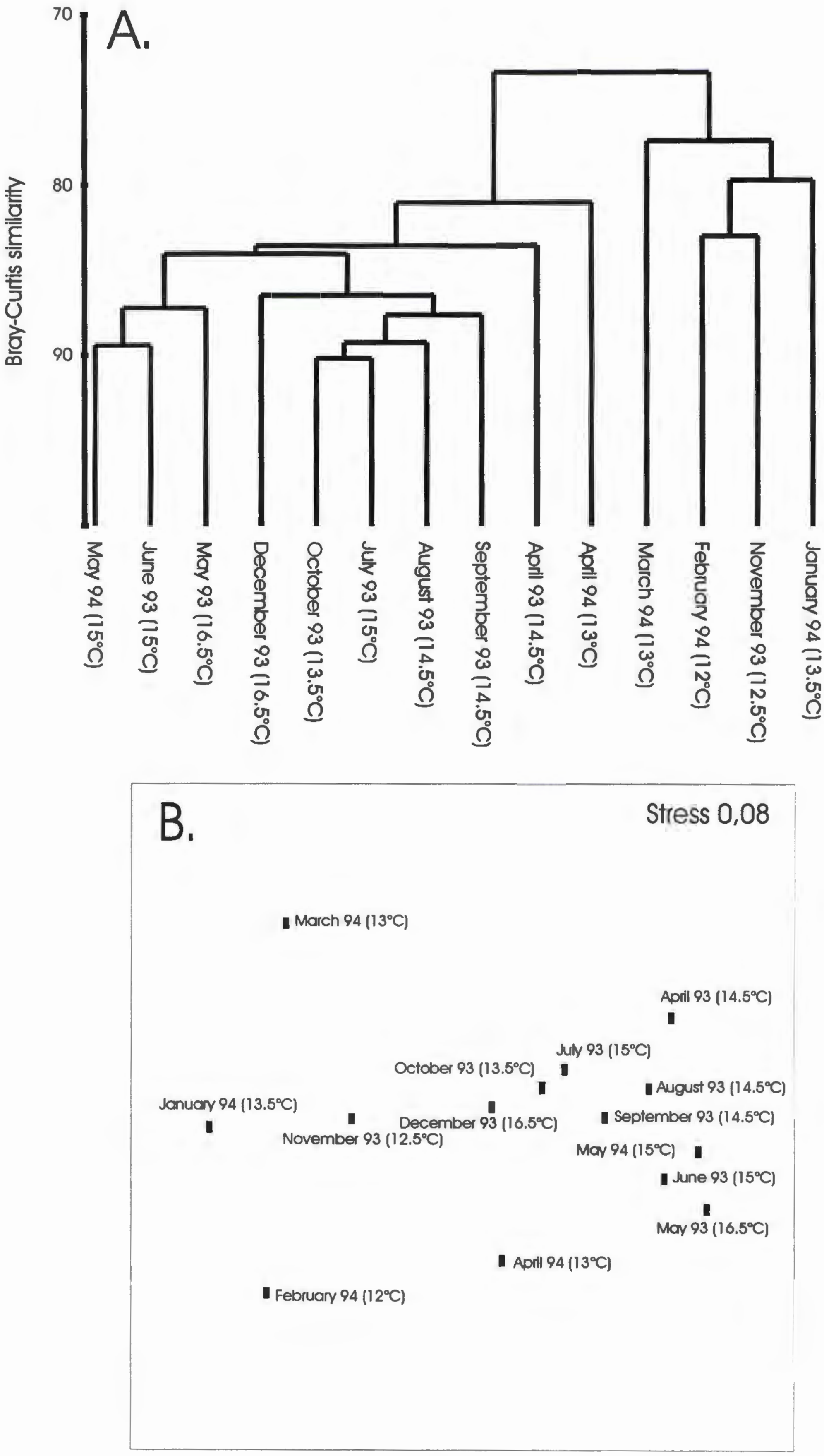


Figure 1.3. A. The Bray-Curtis similarity among the reef fish assemblages encountered each months (from April 1993 to May 1994) at the study site.

B. Multi-dimentional scaling plot depicting how each months assemblages are related to one another in space.

The water temperature encountered each month is shown in brackets.

94, and November 93) were characterized by colder than normal water. The maximum temperature difference between these and the other months was only 3.5°C, however.

Pachymetopon blochii, *Boopsoidea inornata*, *Cheilodactylus fasciatus*, *Chirodactylus brachydactylus*, *C. grandis* and *Chrysoblephus laticeps* appeared most abundant when water temperature was warmer than 13°C, while *Notorhynchus cepedianus* and *Triakus megalopterus* appeared most abundant when it was colder (Figure 1.4). Relative abundances of *Dichistius capensis* and *Spondyliosoma emarginatum* did not appear to vary with water temperature or seasonally (Figure 1.5A). The variations the relative abundance of *Chrysoblephus gibbiceps* did not appear to be related to water temperature. Large individuals (>40 cm TL) were present in March 1993 (just before the commencement of the study) but were only encountered again from February-May 1994, although small specimens were encountered in spring (Figure 1.5B).

Eleven of the 26 demersal species encountered were encountered at all depths, while a further six were encountered within five of the six depth strata. The number of species and relative abundance of each within each depth stratum are listed in Table 1.3. The abundance of such species as *Chrysoblephus laticeps*, *C. gibbiceps* and *Chirodactylus grandis* increased with depth, while those of *Diplodus sargus capensis* and *Sarpa salpa* were highest in the shallowest strata. *Pachymetopon blochii*, *Boopsoidea inornata* and *Dichistius capensis* were most abundant at intermediate depth. The Shannon-Wiener overall index value was smallest within the 0-5 m stratum and highest within the >5- 8 m and >12-16 m strata. The differences among strata were not great, however, the value of the 0-5 m stratum being 76.6% of the >5-8 and >12-16 m strata (Table 1.4).

Table 1.3 The relative abundances (fish.100 m-2) of suprabenthic fish species encountered within six depth strata.

	Depth zone (m)					
Species	0-5	>5-8	>8-12	>12-16	>16-20	>20-25
Demersal teleost species						
Sparidae						
Boopsoidea inornata	0.1	7.1	23.3	30.4	21.6	20.9
Chrysoblephus gibbiceps	-	-	-	< 0.1	0.8	0.8
Chrysoblephus laticeps	1.7	4.7	10.1	13.9	16.9	19.1
Diplodus cervinus hottentotus	0.2	0.5	0.5	0.9	0.7	3.2
Diplodus sargus capensis	7.8	1.4	0.9	2	0.8	0.4
Gymnocrotaphus curvidens	2.6	2	2.4	1.1	0.6	0.8
Pachymetopon aeneum	-	0.4	0.5	0.3	0.5	0.4
Pachymetopon blochii	7.6	27.9	46.1	45.3	29.7	39.3
Petrus rupestrus	-	0.1	0.2	0.2	0.2	0.1
Rhabdosargus globiceps	-	-	< 0.1	-	-	-
Sarpa salpa	51.4	13.2	30.9	14.1	3.9	-
Spondyliosoma emarginatum	-	1	4.9	8.6	19.2	25.2
Cheilodactylidae						
Cheilodactylus fasciatus	8.7	1.9	1.9	3.5	5.2	8.2
Cheilodactylus pixi	-	-	-	-	-	0.1
Chirodactylus brachydactylus	9	3.7	4.8	2.6	0.6	0.1
Chirodactylus grandis	-	-	0.2	0.9	1.4	3.8
Coracinidae						
Dichistius capensis	0.2	1.1	1.4	0.8	0.1	-
Oplegnathidae						
Oplegnathus conwayi	0.1	0.3	1.3	0.4	0.1	-
Parascorpidae						
Parascorpis typus	0.5	0.7	0.5	0.5	0.5	1
Chaetodontidae						
Chaetodon marleyei	0.1	-	0.1	< 0.1	-	-
Pelagic teleost species						
Carangidae						
Trachurus trachurus	-	-	-	19	-	-
Mugilidae						
Liza richarsonii	-	1.4	-	-	-	-
Demersal cartilaginous species						
Carcharhinidae						
Mustelus mustelus	0.1	< 0.1	-	-	< 0.1	-
Triakus megalopterus	0.6	1.1	< 0.1	-	-	-
Hexanchidae						
Notorhynchus cepedianus	-	0.1	0.1	0.1	-	-
Scyliorhinidae						
Haploblepharus edwardsii	0.1	0.2	0.1	0.1	0.1	0.3
Haploblepharus pictus	-	0.1	< 0.1	0.1	0.1	-
Poroderma africanum	0.6	< 0.1	0.1	< 0.1	0.1	0.1
Overall totals						
Total number of demersal fishes	91.3	67.4	130.1	125.5	102.9	124.1
Total number of fishes	91.3	68.8	130.1	144.4	102.9	124.1
Total number of species (all)	17	22	23	23	21	17

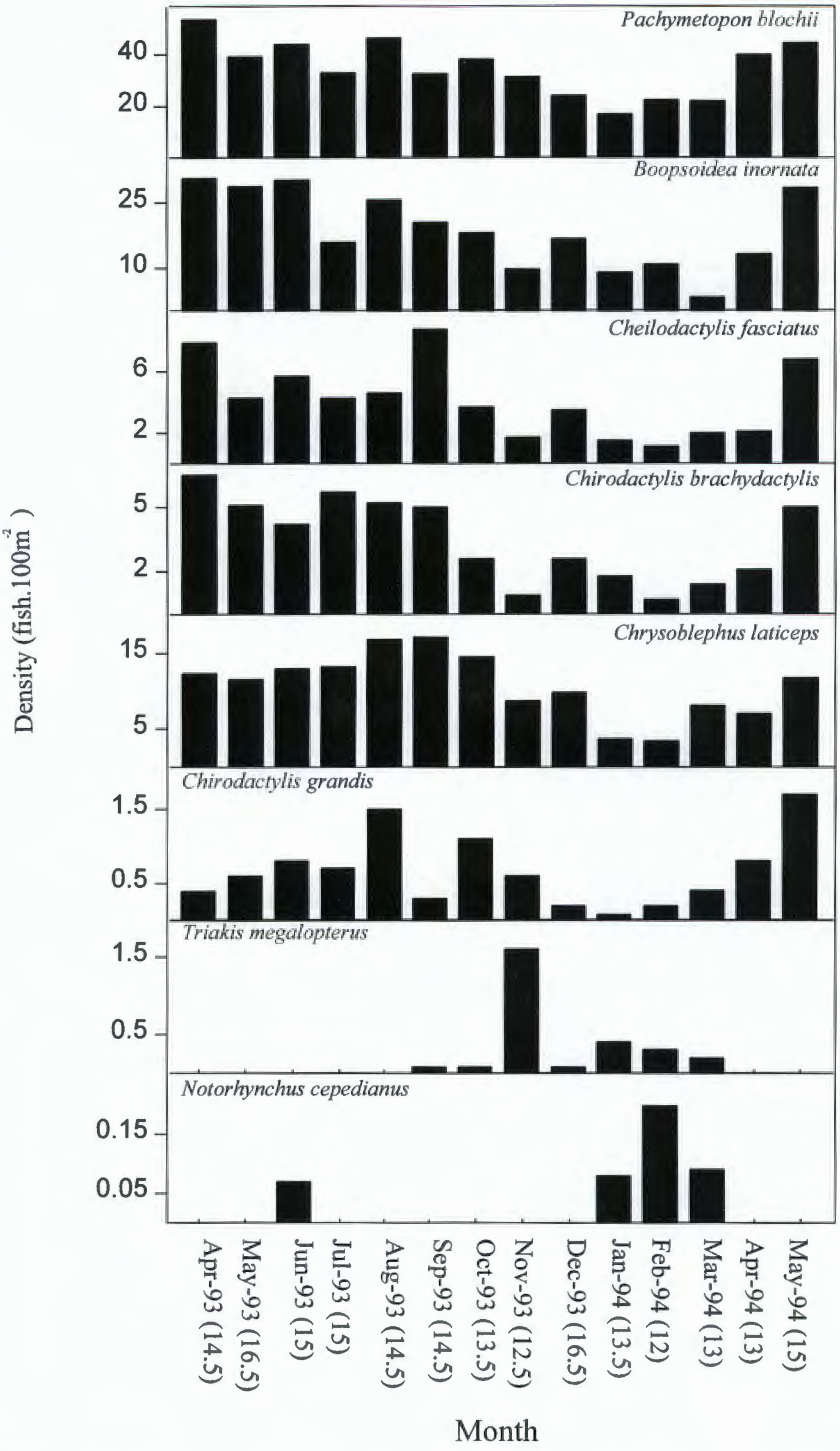


Figure 1.4 Apparent monthly density variation of reef fish species within the Castle Rock Marine Reserve.

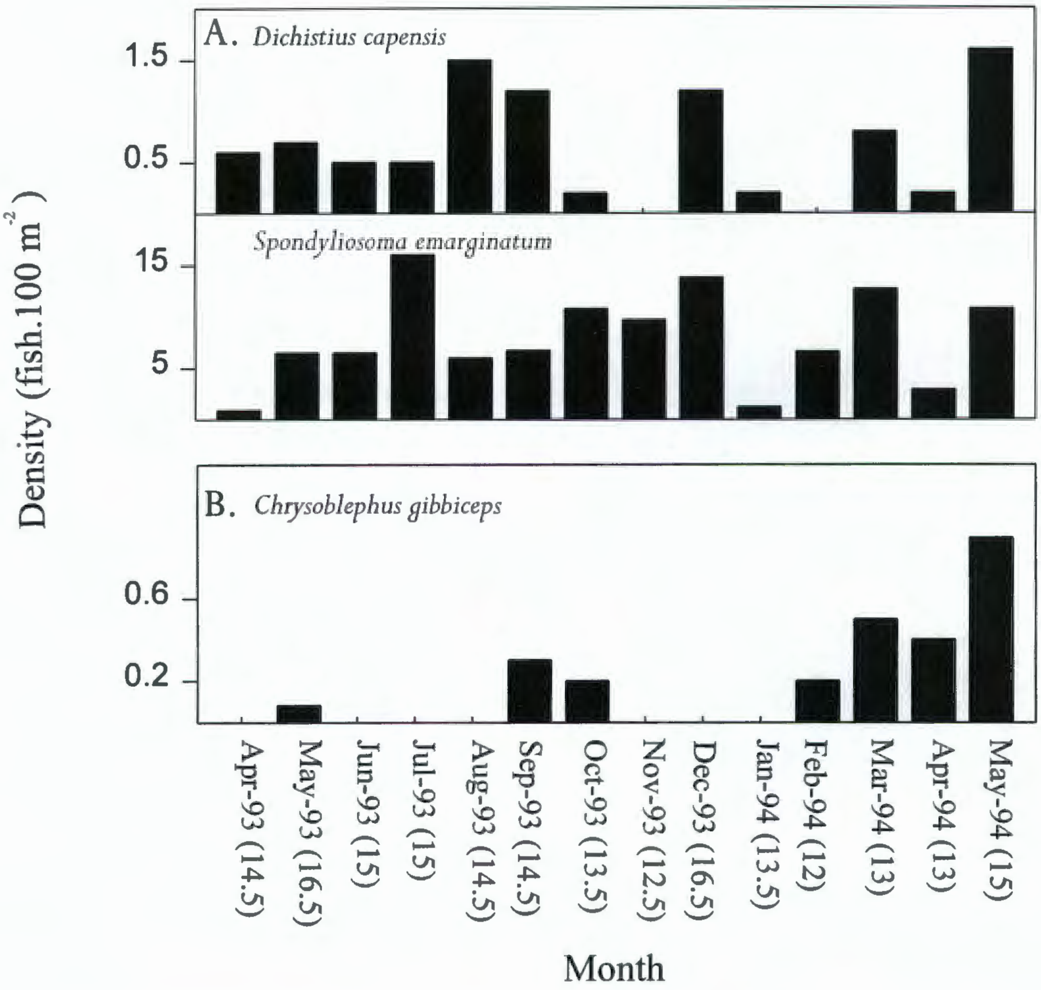


Figure 1.5.A. Monthly densities of two species whose density does not vary substantially with water temperature variations.

B. Monthly density variation of *Chrysoblephus gibbiceps*, the only encountered reef fish species whose density variations were seasonal, but not related to water temperature

Note, the value in brackets is the water temperature (°C) encountered that month.

Table 1.4 The values of the Shannon-Wiener overall index calculated from the fish assemblages encountered within the six depth strata censused. Note: bold values are maxima.

Depth Stratum (m)	Shannon-Wiener overall index
0-5	1.54
>5-8	**2.01**
>8-12	1.84
>12-16	**2.01**
>16-20	1.9
>20-25	1.85

The number of individuals seen together per sighting also varied among species. *Sarpa salpa* were most often encountered in shoals of >50 individuals, while *Spondyliosoma emarginatum*, *B. inornata* and *P. blochii* were seen in loose shoals of 15-30 individuals in water warmer than 13°C, but of >100 in colder water. *Argyrozona argyrozona*, *C. gibbiceps*, *Dichistius capensis*, *Diplodus cervinus hottentotus*, *D. sargus capensis*, *Mustelus mustelus*, *P. aeneum* and *Triakis megalopterus* were usually seen singly or in small loose groups (<10 individuals). Demersal non-shoaling species such as *Cheilodactylus fasciatus*, *Chirodactylus brachydactylus*, *Chrysoblephus laticeps* and *Gymnocrotaphus curvidens* were encountered in groups of 1-12, depending on the reef structure. Abundance was highest where cover was abundant. *Parascorpis typus*, *Petrus rupestris*, both *Haploblepharus* and *Poroderma* species, and *Notorhynchus cepedianus* were rarely encountered more than in pairs.

Few of the reef fish species making up the suprabenthic reef fish assemblage attain more than 50 cm total length (TL). Indeed, *B. inornata*, *C. fasciatus*, *Sarpa salpa* and *Spondyliosoma emarginatum*, species that rarely exceed 30 cm TL, dominated numbers. The largest teleosts encountered were (from largest species) *Chirodactylus grandis*, *Chrysoblephus laticeps*, *C.*

gibbiceps, *P. blochii* and *Gymnocrotaphus curvidens*. Demersal sharks were the largest fishes seen, of which *Notorhynchus cepedianus* attained over 3 m TL.

Discussion

This study reports on the suprabenthic reef fish assemblage within the Castle Rock Marine Protected Area. The study reef within the MPA was chosen since, because it encompasses reef with much structural variability, it is likely to harbour the majority of reef fish species found in False Bay. Secondly, due to the protection afforded to reef fishes by the reserve, it was assumed that the data would reflect the closest estimate of the size and species composition of an unexploited False Bay reef fish assemblage.

The species list from Table 1.2 is not an exhaustive list of the suprabenthic species that occur on False Bay reefs, although the majority of the reef fish species present within the Bay were noted during this study. Other Teleost species that are at times abundant over False Bay reefs include *Seriola lalandii* (a pelagic teleost predator), and pelagic planktivorous baitfish species such as *Sardinops sagax*, *Engraulis japonicus*, *Atherina breviceps* and *Decapterus spp.* (van der Elst 1988; Lamberth, Bennett & Clark 1995a, b; Chapters 6 & 7). Additional cartilaginous species also occur over reefs, although most (*e.g. Carcharhinus brachyurus*, *Dasyatis brevicaudata* and *Myliobatus aquila*) are only common during summer. The shallow water cryptic ichthyofauna of the Bay also boosts the species list substantially (Prochazka 1994).

Van Herwerden (1989) completed 40 stationary point counts within the Castle Rock Marine Protected Area during two separate survey trips. She encountered 13 suprabenthic reef fish species (4 families) including three 'dogfish'. *Pachymetopon blochii* was the most abundant, with sparids dominating diversity (46.2% of species). Zoutendyk (1982) surveyed the reef fish assemblage at two sites (0-12 m depth) in Plettenberg Bay over two consecutive days. Altogether, 22 suprabenthic species (8 families), including one elasmobranch, were encountered, of which *Pomadasys olivaceum* was the most abundant. Sparids dominated suprabenthic species diversity (59.1%). Both the study of Zoutendyk (1982) and Van Herwerden (1989) encountered fewer species than this study (28 species from 11 families), in all likelihood as a result of their smaller sampling regime (few counts and survey trips) within a limited depth range. Burger (1991) encountered 42 suprabenthic species (19 families), including seven elasmobranchs, within the Tsitsikamma National Park. At a site outside the Park, he encountered 30 species (9 families), including four elasmobranches, over two of the three reef categories censused within the Park. Overall, he reported *Sarpa salpa* as the most abundant species in the Tsitsikamma region, with the family Sparidae dominating species diversity both inside (42.9%) and outside (53.3%) the Tsitsikamma National Park. Beckley & Buxton (1989) reported on the fish assemblage inhabiting a site within Algoa Bay. They recorded 33 suprabenthic species (11 families), including two elasmobranchs, over a 14 months period. *Boopsoidea inornata* was the most regularly encountered species, while sparids dominated species number (57.6%). Berry *et al.* (1982) encountered 61 suprabenthic species (32 families), including two elasmobranchs, on a shallow reef in Durban. *Pomadasys olivaceum* was the most abundant species, and sparids still dominated species diversity, although they comprised a smaller proportion of the suprabenthic assemblage (9.8%), compared to on reefs censused further south and west along the South African coast. This is due to the presence of numerous species of more tropical origin off the KZN coast. The results of this study, Berry *et al.* (1982), Beckley & Buxton (1989) and Burger (1991) thus highlight an increase in suprabenthic reef fish diversity from west to east along the South African coastline, with sparids making up a smaller proportion of the assemblage in KZN than along the Cape Coast.

Water temperature affects the behaviour and/or spatial distribution of certain species. *Chrysoblephus laticeps*, *Cheilodactylus fasciatus*, *Chirodactylus brachydactylus* and *Gymnocrotaphus curvidens* recede into caves in cold water, while *P. blochii* and *B. inornata* congregate around high rising pinnacles (*e.g.* 'Outer Castle', 'Pinnacle Rock') at such times, resulting in these appearing less abundant (pers. obs.). The majority of the monthly relative abundance variations depicted in Figure 1.3 are due to behavioural changes associated with water temperature. However, the apparent monthly relative abundance variations of *Notorhynchus cepedianus* and *Chirodactylus grandis* are likely an artifact of the low number of counts undertaken and limited size of the study area, as their abundance appears roughly similar within False Bay throughout the year (pers. obs.).

False Bay suprabenthic reef fish species vary in their degree of residency. *Chrysoblephus laticeps* and *P. blochii*, as reported by Penrith (1972), van der Elst (1988), and van Rooyen (1988) appear resident. Recognisable *C. laticeps* individuals occupied certain caves for the full duration of the study, whereas *P. blochii* appears to reside in the vicinity of a prominant pinnacle/blinder around which it aggregates as water temperature decreases (pers. obs.). *Chirodactylus brachydactylus* and *G. curvidens* also appear resident within proximity of certain caves within False Bay. *Dichistius capensis*, as observed by Attwood & Bennett (1994) along the Southern Cape Coast, and *Diplodus sargus capensis* appear nomadic within the Bay. In the case of *D. sargus capensis*, the apparent nomadic bahaviour is in contrast to the findings of Attwood & Bennett (1995a). Both species do, however, retire into caves when water temperature falls drastically, resulting in their relative abundances appearing to decline at such times. The apparent seasonal variation in most species' relative abundances are thus more due to changes in their spatial distribution and/or behaviour within the same reef complex than seasonal migration to and from False Bay. *Triakus megalopterus* and adult *Chrysoblephus gibbiceps* appear to be the only suprabenthic reef fish species encountered that move into False Bay seasonally. Such movement appears to be linked to drops in water temperature in the case of *Triakus megalopterus* but not of adult *C. gibbiceps*. Although some adult *Chrysoblephus gibbiceps* may inhabit deeper False Bay reefs throughout the year, these are unlikely to represent a large proportion of the Bay's adult population, since experienced anglers and spearfishers report that this species' 'season' only extends from early March-May.

The comparison of Shannon-Wiener overall index values among the depth strata highlighted that the shallowest stratum censused is less diverse than the deeper ones, these all scoring more similarly, with the index being greatest in the >5-8 and >12-16 m strata. The relatively small differences amongst Shannon-Wiener values are partially explained by the fact that 17 of the fish species were encountered within at least five of the six depth strata.

A number of census techniques have been employed to investigate the composition of suprabenthic fish assemblages along the South African coast. They all have some advantages and disadvantages that are related to the behaviour of the species censused. The behaviour of fishes in the presence of SCUBA divers therefore needs to be taken into consideration when deciding on a census method. Certain reef fish species, and particular size classes of others, move away from approaching SCUBA divers within False Bay. These appear rare to such divers, although they appear more abundant to stationary, and thus quieter and more unobtrusive breath-hold divers (pers. obs.). This is why during this study, divers were instructed to move and breathe as little as possible while counting fishes, enabling them to census 'shy' species more accurately. However, it is felt that the stationary point count census method employed was not optimal for counting all species. Abundances of 'inquisitive' species may have been overestimated at all temperatures, and more cryptic suprabenthic species underestimated, especially in water 13°C or colder. Overall, it is felt that since secretive reef fish species are difficult to detect visually, their abundances estimates obtained when using any non-destructive visual census method are likely to be underestimates. In the case of *Cheilodactylus* species, collecting sufficient samples from randomly distributed rotenone stations, in the manner described by Prochazka (1994), may be the best way of obtaining realistic relative abundance estimates. Because scylliorhinids are easily attracted to bait, and can thus be captured, tagged and released, a tag and release study may be the most accurate method for determining their abundance.

Reef fish assemblages do not comprise solely suprabenthic species, however (Berry *et al.* 1982; Buxton & Smale 1984; Beckley & Buxton 1989; Burger 1991). While undertaking cryptic fish collections some 400 m from the study site, Prochazka (1994) reported the presence of 34 cryptic fish species (14 families). She reports this group's overall density (excluding three schylliorhinids and two *Cheilodactylus* species) to be 2.96 fishes.m^{-2} within the 0-20 m depth stratum, which is almost three times that of suprabenthic fishes within the same depth range. The composition of the total reef fish assemblage inhabiting the 0-20 m depth stratum can thus be estimated by combining the present results to those of Prochazka (1994). It comprises 60 species and has a density of 4 fishes.m^{-2} (Appendix 1). Overall, 74.1% of individuals (56.7% of species) are cryptic, and the remaining 25.9% (43.3% of species), including both *Cheilodactylus* species and all scylliorhinids, suprabenthic. However, despite their dominance, cryptic fishes are of little commercial or recreational angling value (Prochazka ,1994).

In conclusion, a False Bay suprabenthic reef fish assemblage was found to be relatively consistent within the 0-25 m depth stratum, with 16 of the 26 demersal species encountered being present throughout the year. Four of these species (*Chirodactylus brachydactylus*, *Chrysoblephus laticeps*, *Gymnocrotaphus curvidens*, *Pachymetopon blochii*) appear to be resident. Species <30 cm TL dominate the assemblage numerically, comprising over 50% of individual fishes. The diversity of the False Bay suprabenthic fish assemblage (28 species, 11 families) is inferior to that reported from sites along the East coast of South Africa, with species and family diversity being maximal off the KwaZulu-Natal coast. The overall False Bay reef fish assemblage (suprabenthic and cryptic combined) is dominated by cryptic species, these being difficult to detect and census with the use of non-destructive survey techniques. To investigate the composition of the complete fish assemblage at a site, both non- and destructive techniques have to be employed to cover both components of the assemblage optimally.

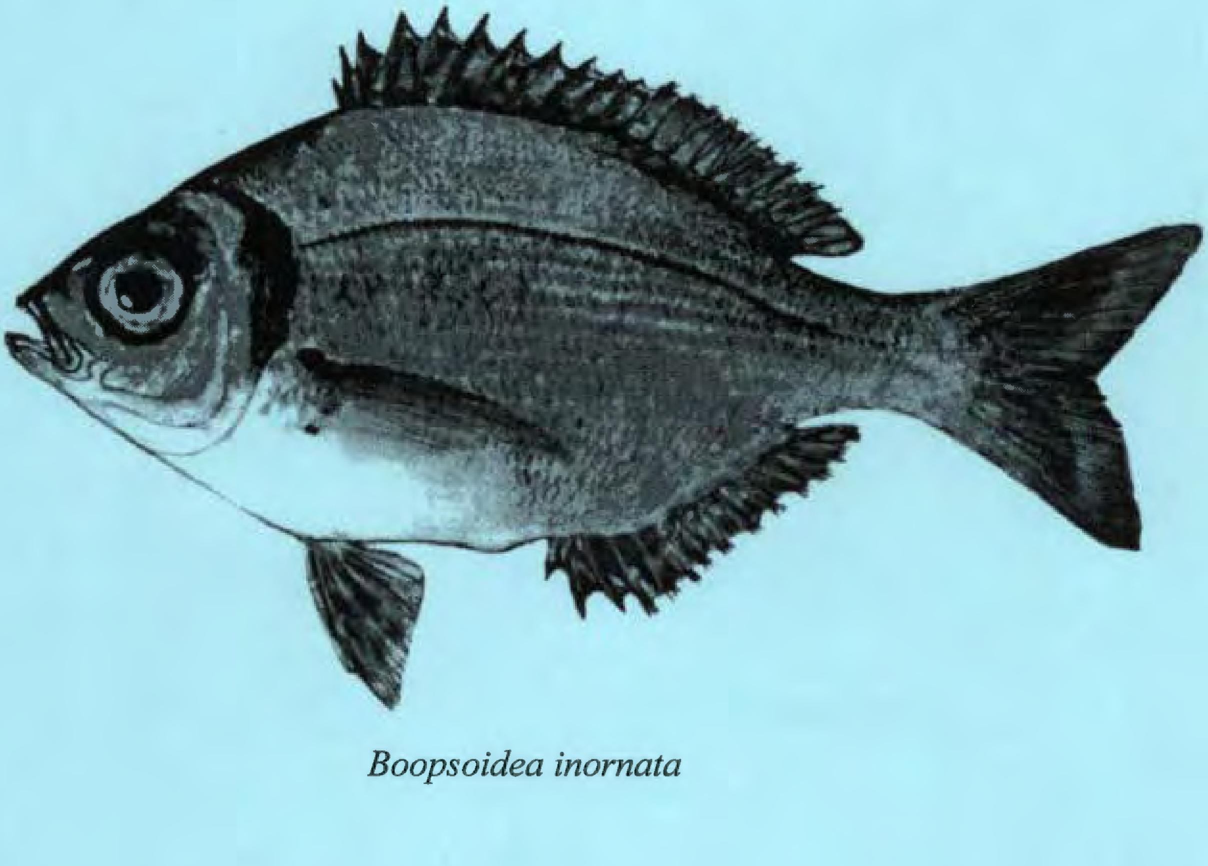
Appendix 1.1 Estimated relative abundance (fish.100 m^{-2}) of all species (suprabenthic and cryptic) which contribute to the composition of the fish assemblage found on False Bay reefs from the shore to a depth of 20 m. Cryptic fish density estimates are based on Prochazka (1994), while suprabenthic fish density estimates were obtained from this study.

Species	Relative abundance (Fish.100 m^{-2})
Suprabenthic species	
Demersal species	
Teleosts	
Sparidae	
Pachymetopon blochii	31.3
Pachymetopon aeneum	0.32
Chrysoblephus laticeps	9.5
Chrysoblephus gibbiceps	0.2
Diplodus sargus capensis	2.6
Diplodus cervinus hottentotus	0.6
Boopsoidea inornata	16.5
Spondyliosoma emarginatum	6.7
Sarpa salpa	22.7
Gymnocrotaphus curvidens	1.7
Petrus rupestris	0.1
Rhabdosargus globbiceps	<0.1
Cheilodactylidae	
Chirodactylus brachydactylus	4.1
Chirodactylus grandis	0.5
Cheilodactylus fasciatus	4.2
Cheilodactylus pixi (Prochazka 1994)	0.3
Coracinidae	
Dichistius capensis	0.7
Oplegnathidae	
Oplegnathous conwayi	0.4
Parascorpidae	
Parascorpis typus	0.5
Chaetodontidae	
Chaetodon marleyei	< 0.1
Condrichthians	
Carcharhinidae	
Triakis megalopterus	0.4
Mustelus mustelus	< 0.1
Hexanchidae	
Notorhynchus cepedianus	< 0.1
Scyliorhinidae	
Haploblepharus edwardsii	0.1
Haploblepharus pictus	0.1
Poroderma africanum	0.2
Pelagic species	
Carangidae	
Trachurus trachurus	3.8
Mugilidae	
Liza richarsonii	0.3
Cryptic species	
Ariidae	
Galeichthys ater	30.5
Batrachoididae	
Batrichthys apiatus	0.5
Batrichthys felinus	5
Bythitidae	
Bidenichthys capensis	23.5

Clinidae	
Blennioclinus brachycephalus	2
Blennophis anguillaris	3
Blennophis striatus	1
Cirrhobarbus capensis	2.5
Clinus cottoides	8.8
Clinus nematopterus	2.5
Clinus rotundifrons	6.5
Clinus superciliosus	6.3
Clinus taurus	1.3
Clinus venustris	3
Fucomenus mus	4.3
Muraenoclinus dorsalis	2.5
Pavoclinus graminis	1.5
Pavoclinus literofrontis	3
Pavoclinis maye	18.8
Pavoclinus pavo	19
Undescribed clinid	1
Congiopodidae	
Congiopodus torvus	0.5
Congrogadidae	
Halidesmus scapularis	89.8
Gadidae	
Gaidropsarus capensis	6.8
Gobiesocidae	
Chorisochismus dentex	6.3
Ecklonaichthys scylliorhiniceps	8.3
Undescribed gobiesocid	2.5
Gobiidae	
Caffrogobius agulhensis	4.8
Caffrogobius saldanha	20.8
Myxinidae	
Eptatretus hexatrema	0.3
Scorpaenidae	
Scorpaena scrofa	0.3
Syngnathidae	
Sygnathus acus	0.5
Tetrarogidae	
Coccotropsis gymnoderma	0.5
Trypterygiidae	
Chremnochorites capensis	32
Total numbers	
Suprabenthic reef fishes	103.8
Suprabenthic fishes	107.9
Cryptic fishes	296.4
Suprabenthic reef fish species	26
Suprabenthic fish species	28
Cryptic species	34
Number of reef fishes (all species)	400.2
Number of reef fish species (all species)	60

Chapter 2

Do physical factors affect the composition of reef fish assemblages in False Bay?



Boopsoidea inornata

Introduction

Studying the habitat requirement of important linefish species is important for their management (Rowley 1994). In the case of reef fishes, their distribution are known to be affected by a number of physical reef characteristics, including vertical relief, cover and depth, and water temperature. The significance of vertical relief to reef fishes has been demonstrated by several authors (*e.g.* Alevizon 1975; Gascon & Miller 1982; Thresher 1983; Stephens, Morris, Zerba & Love 1984; van Rooyen 1988; Shpigel & Fishelson 1989; West, Buckley & Doty 1994), some species being more abundant on high relief reef and others on flat reef (Love & Ebeling 1978; van Rooyen 1988; Buxton & Smale 1989; Crous 1994). In addition, pinnacles and blinders may become important aggregating sites for reef fishes when water temperature decreases sharply at a site (van Rooyen 1988; Buxton & Smale 1989).

Cover is also an important resource for fishes, as it provides protection from predators and/or turbulence (Larson 1980; Gascon & Miller 1982; Jones 1984; Lewis & Wainwright 1985; Shulman 1985; Caley & St John 1996). Cover may be provided in the form of caves and crevices (Penrith 1972; van Rooyen 1988), or by the three-dimensional structure of macroalgae (Levin & Hay 1996). Penrith (1972), Larson (1980), Gascon & Miller (1982), Buxton & Smale (1989), Mann (1992), and Mann & Buxton (1993) have all shown that the presence and sizes of holes are important factors affecting the composition of reef fish assemblages.

Depth also influences the spatial distribution of reef fish species (Gosline 1965; Smith & Smith 1966; Terry & Stephen Jr 1976; Love & Ebeling 1978; Hixon 1980; Bell 1983; van Rooyen 1988; Buxton & Smale 1989; Shpigel & Fishelson 1989; Schoeman & Schoeman

1990; Crous 1994). Depth may be related to a number of factors, one of which is turbulence or surge from wave action in shallower water. Because turbulence decreases in intensity with depth (Gosline 1965), species that are adapted to high turbulence are more abundant in shallow water assemblages, with species more sensitive to turbulence inhabiting deeper waters. Other factors that influence fish depth distibutions include favoured prey distribution (Bell 1983; Buxton & Smale 1989; Mann 1992; Mann & Buxton 1993), water temperature (Terry & Stephen Jr 1976) and the presence of competitors (Hixon 1980).

Water temperature can affect the spatial distribution of fish species at two scales. On a biogeographical scale, it plays an important role in delimiting geographical distributions, restricting individual species to regions where water temperature maxima and minima are rarely outside their tolerated temperature range. Within these areas, fishes can still control their metabolic rates by moving to water bodies whose temperature is more optimal. Such movement may alter the vertical and/or spatial distribution of such species (Rudstam & Magnuson 1985; van Rooyen 1988; Buxton & Smale 1989). Not all species behave this way, however, some remaining in the same area irrespective of temperature variations. The drop in metabolic rate that these species sustain may, however, affect their behaviour. For example, some may remain immobile in shelter, thereby appearing less abundant than in warmer water, while others could aggregate on other nearby portions of reef in cold water.

Each one of these physical factors alone may affect the spatial distribution of individual fish species, but as is reported by Gascon & Miller (1982), Lewis & Wainwright (1985), Walsh (1985), Holbrook, Swarbrick, Schmitt & Ambrose (1992) and Roberts (1996), the relative abundance and/or composition of reef fish assemblages may also be determined by a combination of physical factors. van Rooyen (1988), and Schoeman & Schoeman (1990) report this is the case for particular South African linefish species that are characteristic of reefs with different combinations of reef elevation, depth and/or cover.

This chapter sets out to investigate whether reef elevation, the presence and size of caves, depth, and water temperature influence the spatial distribution of individual reef fish species, and the composition and relative abundance of False Bay reef fish assemblages as a whole. The importance of the results is discussed with respect to the location of future marine reserves and artificial reefs within False Bay.

Methods

This study was undertaken within the Castle Rocks Marine Reserve because the reefs within its borders vary greatly in structural complexity, providing a wide variety of habitat types. Furthermore, due to the protection provided to reef fishes within the reserve, the spatial distribution of species within the reserve is not influenced by human exploitation.

Fish counts were undertaken from April 1993 - April 1995 using the same instantaneous stationary point count census method described in Chapter 1. No reefs at depths exceeding 25 m were censused. Water temperature was measured on the bottom with dive computers during each dive, but when a thermocline was present, it was measured both above and below this level. Counts were first subdivided into two categories according to water temperature. Counts obtained when the water was 13 °C or colder (termed cold) where grouped separately from those obtained when the water was warmer than 13 °C (termed warm). The cut-off point of 13 °C was based on behavioural observation of the different species in the field.

Within each temperature category, counts were also classified according to vertical elevation, cave size and depth categories. Vertical reef elevation was categorised into six elevation classes (0-1, >1-3, >3-5, >5-8, >8-12 and >12-16 m). These categories were also based on behavioural observations of fishes in the field. The size of caves were categorized as small (shallow with only one exit or absent) or as large (one or more deeper large holes/crevices with more than one exit). The depth categories were the same six employed in Chapter 1 (0-5, >5-8, >8-12, >12-16, >16-20 and >20-25 m).

Number of suprabenthic fish species and number of individuals from all reef teleost species combined per 100 m^{-2} were calculated, along with individual species' relative abundances, for each individual count. To investigate how depth affected the spatial distribution of individual species, and how species and reef teleost relative abundances varied with depth, these were calculated for all counts obtained within each depth stratum irrespective of elevation or cave size and compared. To investigate how important vertical reef elevation, cave size and water temperature were in affecting the composition of the reef fish assemblage on a particular reef, relative abundance values were compared between reefs that were similar to each other, except for the parameter being tested. However, since the counts were obtained randomly throughout most of the Castle Rocks Marine Reserve, equal number of replicate counts were not obtained for all the depth, vertical elevation, cave size and water temperature categories. Furthermore, censused reef surface areas were not the same for all counts. It was thus decided that to limit relative abundance variances, only counts which censused more than 7 m^2 would be employed. Furthermore, when investigating the effect of cave size and water temperature, only reef categories for which more than four replicate counts were obtained were compared. When investigating the effect of reef elevation, this was dropped to a minimum of four replicates. The Mann-Whitney U test was employed to compare relative abundances between reefs with different sized caves, or between similar reefs at different temperatures. When comparing relative abundances obtained from reefs containing different vertical elevations, a Kruskall-Wallis test was employed, as a comparison of more than two sets of relative abundance estimates was sometimes necessary.

Results

A total of 718 counts were obtained within the reserve, of which 525 (covering 16 887 m^2) were obtained when the water was warm and 193 (covering 6 069 m^2) when the water was cold.

Water temperature affects the relative abundance of a number of False Bay reef fish species. Table 2.1 shows that number of species per unit area, overall reef teleost relative abundances, and those of some individual species differed significantly with water temperature. In 35 of the 37 cases where differences were significant, relative abundance was greater in warm water. This was evident irrespective of depth and whether or not a reef contained large caves for five species that make use of large caves (*Chrysoblephus laticeps*, *Cheilodactylis fasciatus*, *Chirodactylis brachydactylis*, *Dichistius capensis* and *Diplodus cervinus hottentotus*), and two that rarely do (*Pachymetopon blochii* and *Boopsoidea inornata*). The two exceptions were *Spondyliosoma emarginatum* and *Triakis megalopterus*, two species which swim over reefs without making use of large caves.

Tables 2.2 & 2.3 show the relative abundances of species number, reef teleosts, cartilaginous fishes and various fish species by depth stratum in warm and cold water respectively. Although a number of the species did not appear to show preference for particular depth strata

Table 2.1 Relative abundance (n.100 m^{-2}) and standard deviation of species (number of species), reef teleost and individual species encountered on similar reefs (with respect to depth, vertical elevation and size of available cave) in cold (13 °C or colder) and warm (warmer than 13 °C) water. Only values obtained from five or more replicates and which differed significantly ($p<0.05$) from each other at different temperatures are presented. If the relative abundance values obtained for a group did not differ significantly, but followed the same trend as that of the first values reported in a column, this is shown by the sign s. If the relative abundance values follow the opposite trend to that of the first recoded values in a column, this is marked by the sign ns.

Depth stratum (m)	Elevation (m)	Cave size	Water temperature	Species	Reef teleost	*Boopsoidea inornata*	*Cheilodactylus fasciatus*	*Chirodactylus brachydactylus*	*Chrysoblephus laticeps*	*Dichistius capensis*	*Diplodus cervinus hottentotus*	*Pachymetopon blochii*	*Spondyliosoma emarginatum*	*Triakus megalopterus*
0-5	0-1	Small	Cold Warm	4.55 (1.95) 9.02 (5.29)	ns		s	0 3.07 (5.3)	s	s		s		s
>5-8	0-1	Small	Cold Warm	5.53 (3.48) 8.05 (4.56)	s		s	s	s	s		s		ns
>5-8	>1-3	Small	Cold Warm	6.93 (4.55) 11.16 (6.27)	s		s	s	s	ns		ns		0.41 (0.95) 0
>5-8	>5-8	Large	Cold Warm	ns	s		s	s	s	s		s		ns
>8-12	>1-3	Large	Cold Warm	s	ns	ns	s	3.02 (5.91) 6.75 (6.98)	s	s	s	s	ns	
>8-12	>3-5	Large	Cold Warm	13.7 (6.22) 18.03 (4.78)	67.33 (45.7) 151.36 (194.02)	s	s	s	7.78 (5.28) 13.09 (7.51)	0 1.64 (2.03)	s	s	ns	
>12-16	>1-3	Large	Cold Warm	s	ns	13.36 (11.79) 32.18 (24.48)	1.39 (2.7) 5.27 (5.8)	0.69 (15) 3.55 (2.91)	s	s	s	s	s	
>12-16	>3-5	Large	Cold Warm	14.66 (7.77) 23.82 (7.76)	76.99 (55.27) 125.78 (80.24)	s	s	0.53 (1.3) 7.36 (6.3)	13.58 (7.22) 22.62 (9.8)	0 1.78 (2.69)	s	s	ns	
>16-20	>1-3	Large	Cold Warm	s	s	s	s		ns	ns	s	20.84 (20.4) 43.15 (38.1)	48.23 (40.82) 32.44 (69.41)	
>16-20	>1-3	Small	Cold Warm	9.57 (4.39) 16.15 (7.65)	26.68 (27.72) 89.47 (47.01)	5.95 (5.74) 20.36 (25.01)	s		5.14 (4.74) 11.18 (5.92)			7.13 (6.84) 22.4 (16.11)	5.81 (12.79) 25.43 (28.22)	
>16-20	>3-5	Large	Cold Warm	13.58 (4.89) 21.98 (8.54)		9.11 (11.22) 23.96 (20.98)	2.89 (3.29) 12.03 (13.75)		10.45 (5.58) 21.98 (14)		0 1.21 (2.15)	ns	s	
>20-25	>3-5	Large	Cold Warm	s	68.3 (70.13) 186.3 (105.91)	s	3.17 (3.42) 16.02 (16.02)		s		ns	s	0 77.22 (126.87)	
>20-25	>5-8	Large	Cold Warm	15.4 (7.25) 25.75 (6.47)	75.34 (44.22) 171.13 (95.03)	s	2.96 (32.9) 14.71 (7.68)		s		s	s	ns	

Table 2.2 The mean (and standard deviation) relative abundance (n.100 m^{-2}) of 22 reef fish species, reef teleosts and cartilaginous fishes that were obtained within the different depth strata of the Castle Rocks Marine Reserve when the water was warmer than 13 °C. Note, only the relative abundance of individual species which were encountered more than 20 times during the study are included in the list, but all species, reef teleost and cartilaginous individuals were included when calculating their relative abundance

	Depth strata (m)											
Species	0-5		>5-8		>8-12		>12-16		>16-20		>20-25	
Reef teleosts												
Boopsoidea inornata	0.47	(2.19)	4.83	(13.93)	16.20	(24.98)	27.91	(25.42)	21.13	(20.46)	16.11	(18.67)
Cheilodactylus fasciatus	3.21	(5.52)	2.53	(4.82)	2.89	(4.71)	4.88	(6.39)	7.37	(9.21)	9.37	(9.12)
Chirodactylus brachydactylus	4.86	(6.57)	5.29	(8.23)	5.72	(6.06)	3.48	(4.82)	1.13	(3.15)	-	
Chirodactylus grandis	-		-		0.12	(0.59)	0.94	(2.38)	1.36	(2.34)	3.11	(4.31)
Chrysoblephus gibbiceps	-		-		-		0.02	(0.20)	0.63	(2.82)	0.90	(2.45)
Chrysoblephus laticeps	2.54	(3.85)	5.94	(5.84)	12.19	(7.56)	15.02	(9.72)	16.85	(10.92)	17.26	(12.13)
Dichistius capensis	3.00	(12.33)	1.23	(3.45)	1.50	(2.88)	0.87	(2.29)	0.05	(0.47)	-	
Diplodis cervinus hottentotus	1.34	(6.38)	0.55	(3.69)	0.60	(2.27)	8.22	(2.55)	0.87	(2.84)	1.89	(4.20)
Diplodus sargus capensis	5.99	(15.46)	1.18	(3.04)	1.78	(5.25)	2.05	(6.90)	0.88	(4.42)	0.71	(2.59)
Gymocrotaphus curvidens	2.17	(4.75)	2.87	(6.31)	3.18	(4.43)	1.38	(2.49)	1.08	(2.50)	0.67	(1.77)
Oplegnathus conwayi	-		0.27	(1.47)	1.19	(3.11)	0.32	(0.32)	0.51	(0.37)	-	
Pachymetopon aeneum	0.03	(0.25)	0.32	(1.83)	0.48	(2.31)	0.24	(1.52)	0.72	(2.28)	0.45	(2.61)
Pachymetopon blochii	9.23	(13.47)	35.73	(99.08)	47.16	(39.27)	44.24	(38.10)	35.05	(33.67)	36.50	(28.95)
Parascorpis typus	0.40	(1.09)	0.65	(1.61)	0.57	(1.51)	0.66	(1.42)	0.75	(2.14)	1.56	(1.85)
Petrus rupestris	0.13	(0.77)	0.05	(0.42)	0.22	0.79	0.24	(1.01)	0.22	(0.89)	-	
Sarpa salpa	38.74	(121.83)	10.87	(59.16)	33.95	(146.40)	9.19	(57.14)	-		-	
Spondyliosoma emarginatum	0.10	(0.65)	1.03	(3.90)	7.26	(18.85)	8.46	(20.01)	20.49	(42.67)	88.40	(185.15)
Cartilaginous species												
Haploblepharus spp.	0.17	(0.79)	0.31	(1.08)	0.13	(0.79)	0.15	(0.15)	0.31	(0.96)	0.68	(1.50)
Mustelus mustelus	0.04	(0.32)	0.04	(0.35)	0.03	(0.28)	-		0.02	(0.25)	-	
Notorhynchus cepedianus	-		0.07	(0.46)	-		-		-		-	
Poroderma africanum	0.21	(0.70)	0.06	(0.37)	0.07	(0.46)	0.02	(0.02)	0.05	(0.35)	0.13	(0.76)
Triakus megalopterus	0.20	(0.82)	0.60	(2.89)	-		-		-		-	
Overall relative abundance												
Reef teleosts (all species)	72.20	124.41	73.34	(132.20)	135.02	(162.18)	120.70	(88.51)	108.71	(65.21)	178.37	(175.92)
Cartilaginous fishes (all species)	0.62	1.26	1.09	(3.12)	0.23	(0.94)	0.17	(0.70)	0.38	(1.03)	0.81	(1.62)
Species (all)	11.34	6.94	15.09	(9.36)	18.57	(6.56)	18.70	(7.77)	18.33	(7.89)	21.29	(8.98)

Table 2.3 The mean (and standard deviation) relative abundance (n.100 m^{-2}) of 22 reef fish species, reef teleosts and cartilaginous fishes that were obtained within the different depth strata of the Castle Rocks Marine Reserve when the water was 13 °C or colder. Note, only the relative abundance of individual species which were encountered more than 20 times during the study are included in the list, but all species, and reef teleost and cartilaginous individuals were included when calculating their relative abundance

	Depth strata (m)					
Species	0-5	>5-8	>8-12	>12-16	>16-20	>20-25
Reef teleosts						
Boopsoidea inornata	-	8.53 (21.18)	18.31 (24.13)	18.07 (20.04)	10.63 (12.94)	12.58 (19.39)
Cheilodactylus fasciatus	0.33 (0.92)	0.47 (1.56)	0.96 (2.17)	1.54 (2.9)	2.43 (3.66)	3.37 (28.57)
Chirodactylus brachydactylus	-	1.16 (3.61)	2.22 (4.48)	0.40 (1.13)	0.29 (1.28)	0.27 (0.96)
Chirodactylus grandis	-	-	0.15 (0.67)	0.41 (1.23)	1.13 (2.36)	2.05 (4.06)
Chrysoblephus gibbiceps	-	-	-	-	0.80 (2.57)	4.40 (1.59)
Chrysoblephus laticeps	0.31 (0.83)	2.90 (3.96)	8.62 (7.06)	11.49 (7.05)	11.58 (10.99)	10.90 (9.38)
Dichistius capensis	3.72 (14.14)	0.32 (1.46)	1.10 (5.56)	0.31 (1.31)	-	-
Diplodis cervinus hottentotus	-	0.11 (0.69)	0.22 (1)	0.17 (0.72)	0.45 (2.82)	3.06 (7.1)
Diplodus sargus capensis	16.67 (5.16)	0.81 (2.85)	1.13 (2.95)	1.44 (5.69)	0.11 (0.78)	0.77 (2.77)
Gymocrotaphus curvidens	-	1.20 (5.58)	1.45 (4.19)	0.52 (1.31)	0.15 (0.74)	-
Oplegnathus conwayi	-	0.84 (3.56)	0.09 (0.58)	0.10 (0.56)	-	-
Pachymetopon aeneum	-	-	0.36 (1.42)	0.31 (1.05)	0.21 (1.08)	0.31 (1.11)
Pachymetopon blochii	4.14 (5.57)	22.64 (31.26)	33.96 (30.91)	42.69 (43.34)	27.11 (34.9)	27.77 (20.72)
Parascorpis typus	-	1.23 (4.77)	0.24 (0.88)	0.37 (1.04)	0.45 (1.16)	0.99 (2.73)
Petrus rupestris	-	0.08 (0.46)	0.07 (0.43)	0.35 (1.17)	0.16 (0.81)	0.62 (2.22)
Sarpa salpa	5.02 (110.99)	6.76 (41.1)	56.91 (193.84)	1.43 (85.15)	6.13 (43.76)	-
Spondyliosoma emarginatum	-	0.30 (1.83)	4.71 (24.97)	7.74 (12.36)	25.57 (36.29)	3.70 (7.42)
Cartilaginous species						
Haploblepharus spp.	-	0.24 (0.82)	0.10 (0.63)	0.10 (0.6)	0.05 (0.35)	0.45 (1.63)
Mustelus mustelus	-	-	-	-	-	-
Notorhynchus cepedianus	-	-	0.12 (0.78)	0.13 (0.13)	-	-
Poroderma africanum	-	-	-	-	-	-
Triakus megalopterus	0.60 (1.85)	0.38 (1.26)	-	-	-	-
Overall relative abundance						
Reef teleost (all species)	75.39 (157.72)	47.33 (76.5)	130.48 (196.6)	100.22 (106.14)	87.19 (81.7)	66.82 (50.06)
Cartilaginous fishes (all species)	0.94 (1.96)	2.65 (7.51)	0.32 (1.15)	0.23 (0.79)	0.05 (0.35)	0.45 (1.63)
Species (all)	4.40 (1.88)	11.09 (15.27)	14.57 (8.4)	14.91 (7.27)	13.31 (6.49)	16.50 (8.88)

(*e.g.* *Oplegnathus conwayi*, *Parascorpis typus*, *Petrus rupestris* and *Notorhynchus cepedianus*), the remainder did. *Diplodus sargus capensis* and *Dichistius capensis* were most abundant within the shallowest stratum, while *Mustelus mustelus*, *Triakis megalopterus* and *Poroderma africanum* were also most common in shallow (0-8 m) water. Five teleost species (*Chirodactylis grandis*, *Chrysoblephus laticeps*, *Chrysoblephus gibbiceps*, *Cheilodactylis fasciatus* and *Spondyliosoma emarginatum*) and two cartilaginous species (*Haploblepharus edwardsii* and *H. pictus*) were most abundant within the deeper depth strata. However, most species, including *Chirodactylis brachydactylis*, *Pachymetopon blochii* and *Gymnocrotaphus curvidens* were most abundant within the >8-12 or >12-16 m depth strata, irrespective of water temperature.

The large number of *Spondyliosoma emarginatum* (88.4 ± 185.15) encountered within the >20-25 m depth stratum resulted in overall reef teleost relative abundance being greatest (178.37 ± 175.92) within that stratum when the water was warm (Table 2.2). The second highest teleost relative abundance recorded in warm water was within the >8-12 m depth stratum. This depth stratum was also that within which teleost relative abundance was greatest (130.48 ± 196.6) when the water was cold (Table 2.3). Species density was maximal within the >20-25 m stratum in warm (21.29 ± 8.98) and cold (16.5 ± 8.88) water. Cartilaginous fishes were most abundant within the >5-8 m depth stratum in warm (1.09 ± 3.12) and cold (2.65 ± 7.51) water (Tables 2.2 & 2.3).

Table 2.4 reports species richness, reef teleost, cartilaginous fish and individual species' relative abundances on reefs that differed only with respect to the size of cave(s) available. Only values which differed significantly ($p<0.05$) are presented. For 22 of the 23 cases for which this was the case, the relative abundance on reefs that contained large caves was

Discussion

False Bay sea surface temperatures range from 11-20 °C (Atkins 1970a), while those of deeper layers also vary, albeit within a narrower temperature range. The warmest waters are present in summer during windless conditions, or when a warm water current enters the Bay (Atkins 1970a; Cram 1970; Day 1970). The coldest temperatures occur when a strong south-easterly wind results in the upwelling of cold (11-12 °C) water along the eastern shore (Cram 1970) during summer, or when a cold water current intrudes within False Bay (Atkins 1970b; Day 1970). The declines in number of species, reef teleost and most individual species' relative abundances when the water was cold highlights that water temperature has an important influence on the False Bay reef fish assemblage. In the case of *Chrysoblephus laticeps*, *Cheilodactylis fasciatus*, *Chirodactylis brachydactylis*, *Dichistius capensis* and *Diplodus cervinus hottentotus*, relative abundance declines are related to the behavioural change which these species undergo as the water cools to 13 °C or below. As is reported by van Rooyen (1988), and Buxton & Smale (1989) for *Chirodactylis brachydactylis* and *Chrysoblephus laticeps*, these species retire into caves when the water is cold, due to a decline in their activity. This explains why the abundance of these species decreases on reefs that provide no or small caves when the water is cold, as they have moved to nearby reefs that provide large caves. However, the secretive behaviour of these species within caves when the water is cold results in them being more difficult to detect, thus resulting in them still appearing less abundant on reefs with large caves in cold compared to warm water. In the case of *Pachymetopon blochii* and *Boopsoidea inornata*, the relative abundance differences noted at different temperatures are not related to these species retiring into large caves as the water cools. Both species shoal in the water column and aggregate on different nearby reefs as water temperature varies. This will be explained in greater detail when the importance of

Table 2.4 Differences in the relative abundance (n.100 m^{-2}) of species (species number), reef teleosts, cartilaginous fishes, and individual species encountered on reefs which are similar with respect to depth and vertical elevation, but which provide fishes with cave(s) of different sizes. Only those which differed significantly from each other are presented. Where the difference in relative abundance followed the same pattern as that shown by the values above that sign, or by the first values shown in a column, this is represented by the sign s, while ns shows where the difference followed the opposite trend. Note, only when more than five replicate counts were obtained from reef with a particular architecture were differences compared.

Water temperature	Depth stratum (m)	Elevation (m)	Cave size	Species	Reef teleosts	Cartilaginous fishes	*Boopsoidea inornata*	*Chirodactylus brachydactylus*	*Gymnocrotaphus curvidens*	*Chrysoblephus laticeps*	*Pachymetopon blochii*	*Spondyliosoma emarginatum*
Warm	>5-8	>1-3	Small	11.16 (6.27)	31.17 (34.42)	0.12 (0.6)		s	0.81 (1.69)	s	10.54 (10.08)	
			Large	7.17 (17.82)	58.98 (49.01)	1.29 (2.35)			2.71 (2.99)		20.67 (20.87)	
Warm	8-12	>1-3	Small	13 (5.47)	s	s	s	s	ns	8.51 (7.75)	s	ns
			Large	20.17 (6.33)						14.51 (6.76)		
Warm	>12-16	>1-3	Small	s	s	s	13.66 (12.35)	1.4 (3.12)	s	s	s	s
			Large				32.18 (24.48)	3.55 (2.91)				
Warm	>12-16	>3-5	Small	12.45 (2.3)	s	ns	s	0	0	10.72 (6.59)	s	s
			Large	23.82 (7.76)				7.36 (6.3)	2.41 (2.77)	22.62 (9.8)		
Warm	>16-20	>1-3	Small	s	s	ns	s	s	s	11.18 (5.92)	22.4 (16.11)	s
			Large							16.27 (8.23)	43.15 (38.1)	
Warm	>16-20	>3-5	Small	15.16 (3.28)	74.56 (60.96)	s	s	s	s	10.29 (5.42)	s	ns
			Large	22 (8.54)	110.52 (53.84)					22 (14)		
Cold	>16-20	>1-3	Small	9.57 (4.39)	26.68 (27.72)	ns	s	s	ns	5.14 (4.74)	7.15 (6.84)	5.81 (27.9)
			Large	16.5 (6.61)	107.39 (59.81)					17.59 (14.76)	20.84 (20.4)	48.23 (40.82)

greater than that on reefs providing no or only small caves. The only exception was for species density on >1-3 m high reefs within the >5-8 m depth stratum in warm water. Of the species for which a significant difference in relative abundance was obtained, three (*Pachymetopon blochii*, *Boopsoidea inornata* and *Spondyliosoma emarginatum*) do not habitually enter caves, while the remaining three (*Chrysoblephus laticeps*, *Chirodactylis brachydactylis* and *Gymnocrotaphus curvidens*) are regular cave dwellers.

The last physical reef parameter investigated was vertical elevation. No significant differences in species richness, reef teleost, cartilaginous fish or individual species relative abundances were obtained between reefs which differed only in elevation. However, species richness and the relative abundances of a number of individual species were positively related to reef elevation (Table 2.5). Within the >12-16 and >16-20 m depth strata, species richness was greatest on reefs with a vertical elevation of >3-5 m. Similar results (maximum species density on reefs with >3-5 m elevation) were obtained within the other depth strata, although these positive relations are not listed within Table 2.5, as the differences were either not as obvious, or were observed from insufficient counts. *Chirodactylis brachydactylis*, *Diplodus sargus capensis*, *Dichistius capensis* and *Gymnocrotaphus curvidens* were most abundant on reefs whose elevation ranged from >1-5 m, while others such as *Chirodactylis grandis*, *Pachymetopon blochii* and *Diplodus cervinus hottentotus* were most abundant on reefs rising >5-8 m off the sea floor. Overall, the density of many of the species censused, including that of all the important linefish species, was greater on reefs with some vertical elevation than on flat (0-1 m high) reef.

Table 2.5 The mean relative abundance (n.100 m^{-2}) and standard deviation of species (species number) and individual teleost species (of fishery importance) that were found on reefs which only differed in their vertical elevation. Only species whose relative abundance appears to be positively related to reef elevation are presented. Note, only relative abundances obtained from reef categories from which more than three replicates counts were obtained were considered. If too few such replicates were obtained, this is represented by -.

Water temperature	Depth stratum (m)	Category or individual species	Caves size	Elevation (m)			
				0-1	>1-3	>3-5	>5-8
Warm	>12-16	Species	Large	-	17.13 (6.88)	23.82 (7.76)	23.18 (4.52)
Warm	>16-20	Species	Large	-	16.66 (6.15)	21.98 (8.54)	21.79 (7.14)
Warm	0-5	*Chirodactylus brachydactylus*	Small	3.07 (5.3)	5.63 (5.82)	-	
Warm	>5-8	*Chirodactylus brachydactylus*	Small	1.32 (2.87)	4.05 (5.8)	-	-
Warm	>12-16	*Chirodactylus brachydactylus*	Large	-	3.55 (2.91)	7.36 (6.3)	4.19 (5.98)
Warm	>20-25	*Chirodactylus grandis*	Large	-	-	0.56 (1.36)	4.9 (4.67)
Warm	>12-16	*Chrysoblephus laticeps*	Large	-	13.72 (10.48)	22.62 (9.8)	18.5 (9.12)
Warm	>16-20	*Chrysoblephus laticeps*	Large	-	16.24 (8.23)	21.98 (14)	24.32 (11.13)
Warm	0-5	*Dichistius capensis*	Small	1.53 (7.63)	8.32 (22.02)	-	-
Cold	0-5	*Dichistius capensis*	Small	0.24 (0.83)	14.17 (28.33)	-	-
Warm	>5-8	*Dichistius capensis*	Small	0	0.69 (2.41)	-	-
Cold	>5-8	*Dichistius capensis*	Small	0.2 (0.72)	0.55 (1.59)	-	-
Warm	>12-16	*Dichistius capensis*	Large	-	0.62 (2.23)	1.78 (2.69)	1.14 (16.2)
Warm	>16-20	*Diplodus cervinus hottentotus*	Large	-	0.56 (2.27)	1.21 (2.15)	3.56 (7.18)
Warm	>20-25	*Diplodus cervinus hottentotus*	Large	-	-	0.56 (1.36)	4.35 (6.88)
Warm	0-5	*Diplodus sargus capensis*	Small	4.31 (10.9)	10.34 (23.46)	-	-
Warm	>5-8	*Diplodus sargus capensis*	Small	0	1.11 (3.85)	-	-
Cold	>5-8	*Diplodus sargus capensis*	Small	0	0.53 (1.59)	-	-
Warm	>12-16	*Gymnocrotaphus curvidens*	Large	-	1.54 (2.7)	2.41 (2.77)	0.83 (2.36)
Warm	>8-12	*Pachymetopon blochii*	Large	-	44.27 (33.41)	51.41 (38.27)	70.38 (58.96)
Warm	>16-20	*Pachymetopon blochii*	Large	-	43.15(38.1)	26.06 (17.4)	60.86 (44.28)
Cold	>16-20	*Pachymetopon blochii*	Large	-	20.84 (20.4)	33.09 (33.6)	74.39 (56.46)

vertical reef elevation is discussed below. As for *Spondyliosoma emarginatum* and *Triakis megalopterus*, the occasional opposite trends in their significant relative abundance differences do not appear to be of great importance. In the case of *S. emarginatum*, some significant differences followed opposite trends to others with respect to water temperature, while in the case of *T. megalopterus*, the species was only censused in cold water. None of the relative abundance differences obtained with respect to water temperature were attributable to recruitment events.

It is noteworthy that because many of the common reef fish species either retire into caves or aggregate onto different portions of a reef when the water is cold, most False Bay reefs appears devoid of fish life at such time. Only when caves and crevices are investigated closely are some fishes seen. This highlights that water temperature influences the apparent abundance and/or spatial distribution of a number of False Bay reef fish species, stressing that comparisons of certain species' relative abundances among sites within False Bay are only valid if the census results are obtained within the same water temperature range.

The fact that depth influences the spatial distribution of certain False Bay reef fish species is not surprising. Many reef fish species are reported to be distributed within particular depth ranges (Larson 1980; Thresher 1983; Stephens *et al.*, 1984; Lewis & Wainwright 1985), with the presence of congeneric species (Alevizon 1975), favoured prey (Buxton & Smale 1989; Mann 1992; Mann & Buxton 1993), the inability to inhabit turbulent conditions (Gosline 1965) and water temperature (Terry & Stephen Jr 1976) restricting depth distribution. It is noteworthy, however, that both in cold and warm water species density was maximal within the >20-25 m depth range, with the second highest species density being recorded within the >12-16 m depth stratum. Although reef teleost relative abundance within the >20-25 m stratum was high compared to within the other strata (especially in warm water), this was due mainly to the large number of *Spondyliosoma emarginatum* that were encountered within this stratum. The second greatest reef teleost relative abundance was recorded within the >8-12 m stratum in warm water, the same stratum within which reef teleost relative abundance was maximal in cold water. In the case of cartilaginous fishes, their greatest relative abundance was recorded within the >5-8 m depth stratum irrespective of water temperature. This is largely due to the large number of scylliorhinids and larger demersal sharks (*e.g. Triakis megalopterus*, *Mustelus mustelus* and *Notorhynchus cepedianus*) encountered within this stratum.

This study, therefore, highlights that for a False Bay site to have maximal species diversity and reef fish abundance, it must encompass reefs in shallow (0-8 m), intermediate (>8-16 m) and deep (>16 m) water. It is likely that the importance of the >20-25 m stratum is exaggerated when investigating the results of this study, as relatively few counts were obtained within it compared to the other strata, and most of these only covered reefs with large caves. The importance of this depth stratum for deeper water reef fish species such as *Chrysoblephus laticeps*, *Chirodactylis grandis*, *Chrysoblephus gibbiceps* and *Spondyliosoma emarginatum*, however, is noteworthy.

The results obtained concerning the influence of available cave sizes on the reef fish assemblage highlight that this physical reef parameter reflects reef quality for fishes. For four of the five reef categories on which species relative abundances were significantly different, it was maximal on reefs that contained at least one large cave, while the relative abundance of all reef teleosts combined was significantly greater on reefs providing large caves for the three reported cases. These results agree with Caley & St John (1996) who state that refuge availability increases species richness and reef fish abundance on coral reefs. It is likely that within False Bay, greater species, reef teleost and cartilaginous fish relative abundances are present on reefs containing large caves as such reefs provide the reef requirements of both species that do and do not require large caves. The only exception to this pattern was that recorded on >1-3 m high reefs within the >5-8 m depth stratum. It is likely that the shallower location of these reefs is important in explaining this, as when the water is warm, such species as *Chrysoblephus laticeps*, *Triakis megalopterus*, *Chirodactylis brachydactylis* and *Mustelus mustelus* appear to enter kelp beds to feed from reefs that have no large caves. However, due to surge, most of the large caves within this depth are not inhabited by these species, most individuals of these species rather residing in slightly deeper water.

Within the deeper depth strata, *Chrysoblephus laticeps*, *Chirodactylis brachydactylis* and *Gymnocrotaphus curvidens* are more abundant on reefs with large caves because they, like many other species (Larson 1980; Jones 1984; Lewis & Wainwright 1985) use these as a source of cover, especially when the water is cold. It is likely that they use large rather than small caves for a combination of reasons, one of which is predator escape. Because large caves have more than one exit, these are likely to be transient rather than permanent refuges (as defined by Caley & St John 1996) from small predators within False Bay, although they are permanent refuges from large predators such as large sharks and seals.

In the case of *Pachymetopon blochii*, *Boopsoidea inornata* and *Spondyliosoma emarginatum*, however, large caves are not used as shelter from predators. It is more likely that these species frequent reefs with large caves because their favoured prey are most abundant on such reefs.

Vertical reef elevation is reported to play an important role in affecting the spatial distribution of a number of reef fish species (Gascon & Miller 1982; Thresher 1983; Stephens *et al.* 1984; Shpigel & Fishelson 1989 West *et al.* 1994). It was thus surprising that compared to the numerous significant relative abundance differences obtained when investigating the importance of available cave size and water temperature, none were obtained among similar reefs differing with respect to vertical elevation. This is likely due to the limited reef surface area covered by individual counts, as this resulted in many counts not having recorded particular species. The standard deviations obtained around these species' mean relative abundance were, therefore, large. Furthermore, the limited number of replicate counts obtained from certain reef categories would have also resulted in large standard deviations around mean relative abundances. Vertical reef elevation does, however, appear to be a physical reef parameter that affects the spatial distribution of individual species. The relative abundance of a number of important linefish species appeared to be positively related to vertical reef elevation, while most of the species censused were more abundant on reefs with elevation of at least one meter than on flatter reef. The importance of reef elevation is most apparent in the case of *Pachymetopon blochii*, however. Whereas this species may be common on both flat and high relief reefs in warm water, should the water temperature drop to 13 °C or less, the species aggregates around the high pinnacles/blinders in the immediate area, remaining there until the water warms up again. *Boopsoidea inornata* undertakes similar movements as water temperature declines to 13 °C or less.

The results of this study thus highlight the potential importance of reef elevation for some False Bay reef fish species. It is, therefore, likely to also be an important factor determining reef quality for False Bay reef fishes. However, since flat and high elevation reefs are located within close proximity of one another within the Castle Rocks Marine Reserve, the influence of vertical elevation on False Bay reef fish species is underestimated by this analysis. Its influence will, however, be investigated further in Chapter 7, where the fish assemblages inhabiting reserves that encompass structurally different reefs are compared.

The above discussion highlights the influence that each physical reef parameter has on individual False Bay reef fish species and the reef fish assemblage as a whole. However, as is reported by Lewis & Wainwright (1985), Buxton & Smale (1989) and Holbrook *et al.* (1992), the relative abundance of individual species is usually affected by a combination of different physical reef parameters that operate simultaneously. The same appears to be true within False Bay, as species (number of species) and reef teleost relative abundance found on reefs that provide a combination of large caves and some vertical elevation of over 3 m are greater than those found on flat reefs that provide only small caves. This combination of large caves and vertical elevation thus has the same affect on False Bay's reef fish assemblages as a reef's structural complexity is reported to have by Potts & Hulbet (1994) and Roberts (1996). Within False Bay, therefore, structural complexity is likely to be a qualitative measure of a combination of available cave size and vertical elevation (relief).

The findings of this study are important in that they highlight some physical reef parameters that influence False Bay reef fish assemblages as a whole, while also affecting the spatial distribution of important linefish species. These factors should be taken into account when deciding where to establish future marine reserves (Rowley 1994) or deploy artificial reefs within False Bay. The behavioural and spatial distribution changes which some species undertake when water temperatures drop to 13 °C or less must be considered so that marine reserves encompass reefs that provide the reef requirements of these species at all temperatures. Similarly, the data obtained concerning the depth distribution of individual species, and species and teleost relative abundances within different depth strata, highlight that if the benefits of a reserve are to be maximised, it must be established at a site that contains reefs within all of the depth strata studied here. Finally, the presence of large caves and some reef elevation of over 3 m should also be viewed as a necessity for any future marine reserve established with the aim of benefitting reef fishes within False Bay, especially if this reserve is to be of a limited size. The value of marine reserves and artificial reefs as fishery management tools within False Bay is investigated in greater detail in later chapters of this thesis.

Chapter 3

Dietary separation among common suprabenthic fishes from False Bay, South Africa

Chirodactylus brachydactylus

Introduction

The diets of fishery species determine some aspects of their spatial distribution and ecological requirements (Mann 1992). Dietary studies involving one or a few South African reef fish species are common in the literature (*e.g.* Christensen 1978; Buxton 1984; Bennett & Griffiths 1986; Buxton & Clarke 1986; Coetzee 1986). These are of limited value in understanding the trophic structure of the reef fish assemblage at a site, or in explaining how different food resources are exploited by the assemblage as a whole (Burger 1991). To investigate these aspects, the diets of as many as possible of the reef fish species present at a site have to be studied. This study attempts this by analysing the diets of 17 of the most abundant suprabenthic reef fish species found between 0-25 m in False Bay (Table 3.1). All are present within False Bay throughout the year (Chapter 1).

The False Bay suprabenthic reef fish assemblage is less diverse than that inhabiting the warmer waters of the East coast of South Africa (Buxton & Smale 1984; Beckley & Buxton 1989; Burger 1991; Chater *et al.* 1993; Chapter 1). The 17 species whose diets are reported herein constitute over 98% of the suprabenthic reef fish individuals, and almost 71% of the suprabenthic reef fish species recorded from a protected site within False Bay (Chapter 1). The comparatively low reef fish diversity within False Bay, in combination with existing information on the diets of some species (based in all cases, except *Sarpa salpa*, on at least some stomachs collected from within False Bay), allows for an analysis of the trophic structure of the reef fish assemblage. Where some species' feeding biology has been investigated elsewhere in the country, the False Bay data provide an opportunity to study regional differences in diet.

Table 3.1 The 17 suprabenthic reef fish species whose diets were investigated during this study, along with where samples were obtained. The references noted indicate the source of information employed for the seven species from which no stomachs were analysed during this study.

Scientific name	Source of information	Source of samples	Number of stomachs with content
Cheilodactylidae			
Cheilodactylus fasciatus	This study	False Bay	50
Chirodactylus brachydactylus	This study	False Bay	50
Coracinidae			
Dichistius capensis	Bennett & Griffiths (1986)	False Bay	119
Parascopidae			
Parascorpis typus	This study	False Bay	51
Scylliorhinidae			
Haploblepharus edwardsii	Bertolini (unp. data)	Southwestern Cape	213
Haploblepharus pictus	Bertolini (unp. data)	Southwestern Cape	135
Poroderma africanum	Bertolini (unp. data)	Southwestern Cape	49
Poroderma pantherinum	Bertolini (unp. data)	Southwestern Cape	22
Sparidae			
Boopsoidea inornata	This study	False Bay	50
Chrysoblephus laticeps	This study	False Bay	51
Diplodus cervinus hottentotus	This study	False Bay	24
Diplodus sargus capensis	This study	False Bay	43
Gymnocrotaphus curvidens	This study	False Bay	50
Pachymetopon aeneum	This study	False Bay	50
Pachymetopon blochii	Pulfrich & Griffiths (1988)	Southwestern Cape	633
Sarpa salpa	Joubert & Hanekom (1980)	KwaZulu-Natal	187
Spondyliosoma emarginatum	This study	False Bay	50

Methods

Information concerning the diets of seven species was derived from published and non-published results (Table 3.1). In all cases but one (*Sarpa salpa*), these studies were based on at least some specimen collected within False Bay. In the case of *Sarpa salpa*, it was assumed that False Bay specimen fed predominantly on algae, as is reported from elsewhere in South Africa (Christensen 1978; Joubert & Hanekom 1980). The data from Joubert & Hanekom (1980), based on Kwazulu-Natal specimen, were thus used for this study. Data on diets of the remaining 10 species were derived from stomach content analysis of individuals collected within False Bay between March 1993 and June 1994. Samples were collected by spearfishing throughout the study period. Collecting stomachs by spearfishing was advantageous as all species, including those that do not take bait, could be collected wherever present, and individuals could be selected by size. Having said this, some *S. emarginatum* and *B. inornata* were collected using handlines from a boat when the water was too dirty to allow for spearfishing. The contents of these stomachs were, however, compared to those collected by spearfishing to see if there were marked differences in dietary composition.

Each individual was measured (total length - TL) to the nearest mm and then dissected. Stomach contents were removed and fixed in 4-10% formalin, and subsequently examined under a stereo dissecting microscope. Prey items were identified to the lowest possible taxon. Prey taxa were later grouped to enable comparison with data sets from other studies. The volume of small prey items was estimated by placing them on a 1 mm grid using the method of Bennett (1989). For larger prey items, volume was estimated by displacement. Bait used to capture specimens on handlines was ignored if found in the stomachs.

Dietary analysis was based on frequency of occurrence and percentage volume (Berg 1979). This study aimed to investigate interspecific dietary relationships, rather than presenting an exhaustive account of the diet of each individual species. Because of this, relatively few stomachs were collected from each species, as it was deemed likely that overlaps in dietary composition among species would be noticeable from relatively few stomachs. No variations in diet due to season or fish size were considered, although samples were collected throughout the year, and specimens were collected spanning the size range within which each species were regularly observed on natural reefs. The percentage volumes for the various prey items consumed by each species were compared using cluster analysis and multi-dimensional scaling (Field *et al.* 1982; Clarke 1993). Percentage volume data were square root transformed to prevent super-abundant prey species from dominating the analysis.

Results and Discussion

A total of 469 stomachs containing prey items were analysed. Numerous other analysed stomachs were found to be empty, especially from individuals collected from water 13°C or colder. Indeed, of the ten species analysed, only some *Diplodus sargus capensis* individuals fed in such water, the other species often being inactive inside caves at such time. In addition, all reproductively active *Spondyliosoma emarginatum* individuals (nest guarding males or mature females collected above nests) had empty stomachs, irrespective of water temperature. The percentage occurrence and volume of identified prey categories, as well as summaries of the published and unpublished results obtained for the other seven species, are listed in Tables 3.2 & 3.3. The diet of each species is discussed separately below.

Table 3.2 The dietary composition (% occurrence) of 17 of the common reef fish species that inhabit False Bay reefs

	Parascorpis typus	*Poroderma africanum*	*Poroderma pantherinum*	*Haploblepharus edwardsii*	*Haploblepharus pictus*	*Chrysoblephus laticeps*	*Sarpa salpa*	*Gymnocrotaphus curvidens*	*Pachymetopon aeneum*	*Diplodus cervinus*	*Diplodus sargus*	*Dichistius capensis*	*Cheilodactylus fasciatus*	*Chirodactylus brachydactylus*	*Boopsoidea inornata*	*Spondyliosoma emarginatum*	*Pachymetopon blochii*
Algae	2		4.6	0.9	11.1	-	100	72	14	4.2	9.3	47.1	-	4	12	8	34.2
Green		-	-		-	-	49.7	30	8	-	4.7		-	2	10	4	-
Brown		-	4.6	0.9	11.1		3.7	28	2	-	2.3	-	-		2		-
Red	2	-	-	-	-		85.6	62	10	4.2	7	-	-	2	6	4	-
Ascidiacea		-	-	-	-	7.8	15.5	54	28	-	32.6	6.7	2	6	20	10	1.8
Compound ascidians		-	-	-	-	2	-	52	28	-	30.2	-	-	4	8	10	-
Solitary ascidians			-	-	-	5.9	-	6	-	-	4.7	-	2	4	18	-	-
Crustacea	100	-	-	-	-	64.7	-	60	94	79.2	90.7	-	96	100	98	96	-
Amphipoda	23.5	-	-	-	0.7	11.8	45.5	56	94	79.2	81.4	33.3	96	100	86	96	-
Gammaridae	20		-	-	-	11.3	-	32	68	75	74.4	-	94	100	84	92	60.2
Sand dwelling	7.8	-	-	-	-	-	-	2	-	41.7	25.6	-	64	88	32	18	-
Reef dwelling	19.6		-	-	-	11.8	-	30	-	75	67.4	-	94	100	84	90	-
Caprellidae	7.8	-	-	-	-	2	-	46	88	16.7	32.6	-	54	82	32	86	41.1
Tanaidacea		-	-	-			1.6	4	-	4.2	9.3	11.8	16	28	-	2	0.4
Isopoda	2	-	-	0.9	-	3.9	2.1	4	2	29.2	24.6	58	38	88	40	22	14.8
Ostracoda	13.7			-	-	3.9	-	-		16.7	4.7	8.4	16	58	50	22	9.6
Leptostraca	-	-		-	-	-	-		-	8.3	-	1.7	-	10	8	-	-
Cumacea		-	-	-	-			-	-	-	-	0.8	2	4	-	2	-
Stomatopoda		2	-	-	0.7		-	-	-	-	-	-	-	-	-	2	2.6
Mysidacea	100		-	-	-		1.6	-		-		1.7		-	16	10	9.3
Crab		6.1	-	10.3	15.6	45.1	-	-	-	4.2	2.3	1.7	6	8	6	4	-
Adult		6.1	-	10.3	15.6	45.1	-		-	4.2	2.3	1.7	6	8	6	4	4.5
Megalopa				-	-	-	-		-				-	-		-	2.9
Hermit crab		-	4.6	2.4	-	11.8	-			12.5	2.3	-	4	-	-	-	
Jasus lalandii		32.7	9.1	0.9	13.3	5.9			-			-		-			
Cirripedia Rock barnacles			-	-	-	2	5.4	4	-	-	34.9	45.4	-	2	12	10	2.6
Pycnogonida			-	-	-		1.6	-	-		2.3	3.7	-		2		2.2
Copepoda		-	-	1.4	-		14.4	-	-			7.6	2	2	2		
Penaid prawn			-	-						-					2		-
Upogebia sp.				2.4	-				-	-			-				-
Euphausid	-		-	-	-		0.5	-		-						-	-
Unidentified			4.6	6.6	10.4		0.5		-							-	-
Echinodermata			-	-	-	45.1	-	78	2	41.7	11.6		10	26	22	6	-
Holothuroidea			-	-	-	7.4		-	-	-	2.3						0.1
Crinoidae		-	-	-		27.5		78	2	4.2		2.5	4	4	22	6	2.3
Asteroidea			-	-	-	5.9	-	-	-			-		-		-	-
Ophiuroidea		-	-	-	-	3.9		2	-	37.5	9.3	5.9	6	24			2.1
Echinoidea		-	-	-	-	9.8		-	-	-	-	0.8	-	-		-	0.5
Porifera		-	-	-	-		0.5	42	-	-	-	4.2	-	-	-	-	-
Mollusca	5.9	-	-	-		45.1	-	-	2	25	81.4	-	10	48	10	28	30
Cephalopoda	3.9	-	-	-		19.6		-	-	-	-	-	-	-	-	0	
Octopus spp.	-	-	9.1	0.5	3	19.6	-	-	-	-	-	-	-	-	-	-	-
Sepia spp	-	8.2	18.2	2.4	0.7	-	-	-	-	-	-	-	-	-	-	-	-
Squid (juveniles)	3.9	-		-	-	-	-		-	-	-	-	-	-	-	-	0.5
Unidentified	-	8.2	-	0.5	8.2	-	-	-	-	-	-	-	-	-	-	-	-
Bivalvia	-	-	-	-	-	11.8	14.4	4	2	4.2	34.9	94.1	-	24	2	6	-
Gastropoda	2	-		-	-	15.7	-	4	-	20.8	67.4	-	10	38	8	28	0.1
Whelk like	2	-	-	-	-	2	-	4	-	20.8	60.5	79.8	8	36	4	28	-
Limpet like	-	-	-	-	-	13.7	-	-	-	-	32.6	0.8	2	24	6	-	0.1
Hydrozoa Hydroids	-	-	-	-	-	-	61	78	86	4.2	4.7	5	2	-	6	22	29.6
Actinaria Sea anemone	-		-	-	-	9.8	-	-	-	-	-	3.4	-	-	-	-	0.5
Gorgonacea	-	-	-	-	-	-	-	2	16	-	-	-	-		-	-	-
Soft coral	-	-	-	-	-		-	-	20	8.3	16.3	-	-	2	20	16	-
Polychaeta	-		-	18.8	5.9	25.5	13.9	12	8	70.8	37.2	16.8	20	28	18	6	14
Errantia	-		-	-	-	19.6	-	2	6	50	32.6	-	18	26	16	4	-
Sedentaria	-		-	-	-	5.8	-	10	4	66.7	16.3	-	4	8	2	2	-
Unidentified	-	10.2		-		-	-	-	-	-		-		-	-		-
Bryozoa	-	-	-				1.6	44	2	-	15.3	4.2	-	2	2	4	-
Flat		-	-			-	-	26	2	-	15.3	-	-	-	-		-
Branching	-		-	-			-	26	-	-	-	-	-	2	2	4	-
Vertebrata Fish	7.8		-	-	-	2	1.1		-	-	-	-			2	6	0.1
Pelagic Whole	7.8		-	-			-	-	-	-	-	-			2	-	
Klipfish Whole	-		-	-	0.7	2	-	-	-	-	-			-		-	-
Skate Whole		2	-	-	-	-	-	-	-	-	-		-	-	-	-	
Merluccius spp.	-	-	4.6	0.5	-		-	-	-	-	-	-	-	-	-	-	-
Chunks	-	12.2	27.3	5.6	17	-	-	-	-	-	-	-	-	-	-	2	0.1
Eggs	-	-	-	-	-	-	1.1		-	-	-	-	-	-	-	4	-
Eggs from unknown source		-	-	-	-	-	-	-	-	8.3	-	-	-	-	-	-	2.4
Echiuroids		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0.3
Nematoda	-		-	-	-	-	-	-	-	-	-			-		-	2.9
Sipunculida			-	-	-		0.5	-	-	-	-	-	-			-	1.9
Unidentified	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	45.2
Insecta	-	-	-	-	-	-	-	-	-	-	-	0.8	-	-	-	-	-
Number of stomachs with content	51	49	22	213	135	51	187	50	50	24	43	119	50	50	50	50	633

Table 3.3 The dietary composition (% volume) of 17 of the common reef fish species found on False Bay reefs

		Parascorpis typus	*Poroderma africanum*	*Poroderma pantherinum*	*Haploblepharus edwardsii*	*Haploblepharus pictus*	*Chrysoblephus laticeps*	*Sarpa salpa*	*Gymnocrotaphus curvidens*	*Pachymetopon aeneum*	*Diplodus cervinus*	*Diplodus sargus*	*Dichistius capensis*	*Cheilodactylus fasciatus*	*Chirodactylus brachydactylus*	*Boopsoidea inornata*	*Spondyliosoma emarginatum*	*Pachymetopon blochii*
Algae		0.1	-	1	-	-	-	-	10.7	2.7	0.1	0.2	12.4	-	0.1	2.7	1.74	18.7
Green		-	-	-	-	-	-	4.9	1.3	2.1	-	0.1	-	-	0.1	1.7	1.64	-
Brown		-	-	1	1	3.6	-	0.4	2.9	0.1	-	0.1	-	-	-	0.1	-	-
Red		0.1	-	-	-	-	-	88.1	6.4	0.5	0.1	0.1	-	-	0.1	1	0.1	-
Ascidiacea		-	-	-	-	-	7.2	0.5	8.3	9.3	-	7.3	1.7	0.3	1.6	7.7	4.42	0.7
Compound ascidians		-	-	-	-	-	0.3	-	7.4	9.3	-	6.9	-	0.3	1.3	1.2	4.42	-
Solitary ascidians		-	-	-	-	-	6.9	-	0.9	-	-	0.5	-	-	0.3	6.5	-	-
Crustacea		94.5	48.1	12	43.1	74.7	33.2	0.9	0.9	9.6	28.4	38.7	50.4	89.5	87.2	63.4	68.1	42.1
Amphipoda		1.2	-	-	-	0.1	0.1	0.4	0.9	9.5	21.6	9.3	20.5	57	72.2	30.6	61.8	29.8
	Gammaridae	1.1	-	-	-	-	0.1	-	0.3	2.3	21.4	7.8	18.6	49.1	68	26.2	28.9	20.5
	Sand dwelling[1]	0.2	-	-	-	-	-	-	0.1	0.1	5	2.9	-	11.3	24.4	2.6	1.1	-
	Reef dwelling[1]	0.8	-	-	-	-	0.1	-	0.3	2.3	16.4	4.9	-	37.8	43.7	23.7	27.9	-
	Caprellidae	0.2	-	-	-	-	0.1	-	0.5	7.2	0.2	1.5	1.8	7.9	4.2	4.4	32.9	9.3
Tanaidacea		-	-	-	-	-	-	0.1	0.1	-	0.1	0.1	-	0.2	0.3	-	0.1	0.1
Isopoda		0.1	-	-	0.3	-	0.1	0.1	0.1	0.1	1.4	0.6	4	3.8	9.8	9.4	0.8	1.3
Ostrocoda		0.7	-	-	-	-	0.1	-	-	-	0.2	0.2	-	0.4	1.61	14.4	0.5	0.3
Leptostraca		-	-	-	-	-	-	-	-	-	0.1	-	-	-	0.3	0.2	-	-
Cumacea		-	-	-	-	-	-	-	-	-	-	-	-	0.1	0.1	-	0.1	-
Stomatopoda		-	1.8	-	-	1.3	-	-	-	-	-	-	-	-	-	-	0.1	2
Mysidacea		92.6	-	-	-	-	-	0.1	-	-	-	-	-	-	-	0.6	4.6	3.5
Crab		-	1.8	-	21.8	31.7	25.2	-	-	-	0.5	0.1	0.9	24.2	3.1	1.4	0.9	1.8
	Adult	-	1.8	-	21.8	31.7	25.2	-	-	-	0.5	0.1	0.9	24.2	3.1	1.4	0.9	-
	Megalopa	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1.8
Hermit crab		-	-	3.2	3.1	-	1.3	-	-	-	4.6	0.1	-	3.9	-	-	-	-
Jasus lalandii		-	44.5	7.8	1.7	25.6	6.5	-	-	-	-	-	-	-	-	-	-	-
Cirripedia	Rock barnacles	-	-	-	-	-	0.1	0.1	0.5	-	-	28.6	25	-	0.1	1.6	0.13	0.6
Pycnogonida		-	-	-	-	-	-	0.1	-	-	-	0.1	-	-	-	0.1	-	0.2
Copepoda		-	-	-	0.6	-	-	0.3	-	-	-	-	-	0.1	0.1	0.1	-	-
Panaid prawn		-	-	-	-	-	-	-	-	-	-	-	-	-	-	5	-	-
Upogebia sp		-	-	-	5.1	-	-	-	-	-	-	-	-	-	-	-	-	-
Euphausid		-	-	-	-	-	-	0.1	-	-	-	-	-	-	-	-	-	-
Unidentified		-	-	1	10.5	16	-	-	-	-	-	-	-	-	-	-	-	2.6
Echinodermata		-	-	-	-	-	10.6	-	35.3	0.1	2.5	13.1	0.6	1.5	6.2	8.5	13.5	3.8
Holothuroidea		-	-	-	-	-	0.5	-	-	-	-	8.4	-	-	-	-	-	0.1
Crinoidae		-	-	-	-	-	4.6	-	35.2	0.1	0.8	-	-	0.7	0.7	8.5	13.5	2.8
Asteroidea		-	-	-	-	-	2.4	-	-	-	-	-	-	-	-	-	-	-
Ophiuroidea		-	-	-	-	-	0.1	-	0.1	-	1.7	4.7	0.6	0.9	5.5	-	-	1
Echinoidea		-	-	-	-	-	3.1	-	-	-	-	-	-	-	-	-	-	0.1
Porifera		-	-	-	-	-	-	0.1	19.2	-	-	-	3.7	-	-	-	-	-
Mollusca		3.1	31	35.8	13	3.7	39.3	0.1	0.1	0.1	1.9	17.8	28.4	0.6	1.5	0.4	0.3	1.9
Cephalopoda		3.1	31	35.8	13	3.7	33.7	-	-	-	-	-		-	-	-	-	0.4
	Octopus spp.	-	-	4.9	3.9	2.6	33.7	-	-	-	-	-	-	-	-	-	-	-
	Sepia spp	-	27.7	30.9	8.9	0.1	-	-	-	-	-	-	-	-	-	-	-	-
	Squid (juveniles)	3.1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0.4
	Unidentified	-	3.3	-	0.1	1	-	-	-	-	-	-	-	-	-	-	-	-
Bivalvia		-	-	-	-	-	3.7	0.1	0.1	0.1	0.1	5.1	25.9	-	0.7	0.1	0.1	-
Gastropoda		0.1	-	-	-	-	1.9	-	0.1	-	1.8	12.7	2.5	0.6	0.8	0.3	0.3	1.5
	Whelk like	0.1	-	-	-	-	0.1	-	0.1	-	1.8	6.8	2.5	0.1	0.3	0.1	0.3	1.4
	Limpet like	-	-	-	-	-	1.8	-	-	-	-	6	-	0.5	0.4	0.3	-	0.1
Hydrozoa	Hydroids	-	-	-	-	-	-	4.8	9.2	73.6	0.1	0.2	-	0.1	-	0.3	0.8	11.4
Actinaria	Sea anemone	-	-	-	-	-	5.3	-	0.1	-	-	-	1.7	-	-	-	-	0.4
Gorgonacea		-	-	-	-	-	-	-	-	1.8	-	-	-	-	-	-	-	-
Soft coral		-	-	-	-	-	-	-	-	-	1.2	1.8	-	-	0.1	-	2.9	-
Polychaeta		-	-	-	32.3	4.3	1.9	0.2	0.3	0.2	65.2	23.4	-	8.1	3.1	10.8	0.9	3.8
Erantia		-	-	-	-	-	1.8	-	0.1	0.1	26.6	9.4	-	3.6	2.1	10.3	0.1	-
Sedentaria		-	-	-	-	-	0.1	-	0.2	0.2	38.7	14	-	4.5	1	0.5	0.8	-
Unidentified		-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Bryozoa		-	-	-	-	-	-	0.1	6.6	2.1	-	1.8	-	-	0.1	0.1	0.1	-
Flat		-	-	-	-	-	-	-	4.2	2.1	-	1.8	-	-	-	-	-	-
Branching		-	-	-	-	-	-	-	2.4	-	-	-	-	-	0.1	-	0.1	-
Vertebrata	Fish	2.3	20	55.4	10.5	13.9	2.5	0.1	-	-	-	-	-	-	-	1	6.6	0.1
	Pelagic Whole	2.3	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-
	Klipfish Whole	-	-	-	-	0.6	2.5	-	-	-	-	-	-	-	-	-	-	-
	Skate Whole	-	18.1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	Merluccius spp.	-	-	5.1	0.7	-	-	-	-	-	-	-	-	-	-	-	-	-
	Chunks	-	1.9	50.3	9.8	13.3	-	-	-	-	-	-	-	-	-	-	6.2	0.1
	Eggs	-	-	-	-	-	-	0.1	-	-	-	-	-	-	-	-	0.4	-
Eggs from unknown source		-	-	-	-	-	-	-	-	-	0.8	-	-	-	-	-	-	0.1
Echiuroids		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0.1
Nematoda		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0.1
Sipunculida		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0.1
Unidentified		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	17
Insecta		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Number of stomachs with content		51	49	22	213	135	51	187	50	50	24	43	119	50	50	50	50	633

Chirodactylis brachydactylis

Fifty stomachs obtained from fishes 173-441 mm total length (TL) were analysed. This species possesses large fleshy lips and very fine undeveloped teeth, and is thus adapted to feeding on small, soft and mobile prey. Gammarid amphipods dominated the diet both in term of percentage occurrence (100%) and volume (68%). Reef- (*e.g. Podocerus spp, Jassa spp* and *Amaryllis sp.*) and sand-dwelling species (*e.g. Ampelisca palmata* and *Paraphoxus sp.*) contributed 43.7 and 24.4% of volume respectively. Other important food items included isopods (9.8% volume) and small ophiuroids (5.5% volume). Neither large, hard-shelled and tough prey, nor sessile invertebrates, featured in the diet.

Chirodactylis brachydactylis is, therefore, a small-benthic-invertebrate carnivore that feeds on and next to reefs. It is unlikely to feed on sand-dwelling prey away from reefs, as this species was never observed away from reef in False Bay, and is not caught by beach-seiners, the one fishery sector that cannot operate over reef (Lamberth *et al.* 1994).

Burger (1991) investigated the diet of this species within the Tsitsikamma National Park. He also reports gammarid amphipods to be important in the diet, although large quantities of hard-shelled prey such as bivalves and gastropods were also consumed. His results are based on very few samples, however.

Cheilodactylis fasciatus

Fifty full stomachs were analysed from fishes 75-317 mm TL. This species possesses similar feeding adaptations to *Chirodactylis brachydactylis*, and is thus not adapted to feed on large, hard-shelled or sessile prey. Gammarid amphipods dominated the diet, occurring in 94% of the fishes analysed, sand-dwelling species contributing 11.3% of volume and rock-dwelling ones 37.8%. The low proportion of sand-dwelling gammarids consumed by this species correlates with observations that this species were never observed over sand away from (>10 m) from reef. Although small benthic crustaceans dominate the diet of *C. fasciatus*, this species may be more voracious than *Chirodactylis brachydactylis*, as some individuals had consumed relatively large prey, including both hermit crabs (3.9% volume) and crabs (24.2% volume). The latter value is, however, largely attributed to a single moulting adult *Plagusia chabrus* (volume 2.3 ml) consumed by a 257 mm fish. Other important prey items included caprellid amphipods (7.9% volume), errant and sedentary polychaete worms (8.1% volume combined) and isopods (3.8% volume).

Based on very few samples, *Cheilodactylus fasciatus* are also reported to feed predominantly on gammarid amphipods and other small benthic crustaceans within the Tsitsikamma National Park (Burger 1991). *Cheilodactylis fasciatus* is thus a small-benthic-invertebrate carnivore that feeds on or close to reefs.

Boopsoidea inornata

Fifty full stomachs were obtained from fishes 152-265 mm TL. *Boopsoidea inornata* possesses a small terminal and slightly protractile mouth armed with small, sharp teeth (van der Elst 1988). It feeds on a wide range of animal prey, gammarid amphipods being the most important (84% occurrence, 26.2% volume), reef-dwelling ones contributing 23.7% volume. Sand-dwelling prey were also consumed, including ostracods (50% occurrence) and sand-dwelling gammarids (32% occurrence), these together contributing 17% of gut content volume. Such prey are likely to be consumed close to reef, however, as *B. inornata* was never encountered out of sight of reef. Although numerous other benthic crustaceans were also consumed, only large isopods (*e.g. Paridotea reticulata* and *Cirolana spp.*) contributed substantially to dietary volume (9.4%). *Boopsoidea inornata*'s cutting teeth enable it to feed on sessile invertebrates, including soft coral (5.2% volume), small solitary ascidian pods (6.5% volume) and the tips of crinoid arms (8.5% volume). The species does occasionally also feed in the water column, as some mysids, juvenile fishes and one penaid prawn were also consumed.

Trow (1982) studied the diet of *Boopsoidea inornata* from the Transkei and Algoa Bay region. He also reports that it consumes small crustaceans and ascidians regularly, although he concludes that the latter were eaten mainly for their epiphytes. False Bay individuals, however, appear to consume ascidian pods for their actual content, as the pods were always empty in the latter parts of the digestive tract. Overall, *Boopsoidea inornata* is a small-benthic-invertebrate carnivore that feeds on a wide range of small sand- and reef-dwelling prey.

Spondyliosoma emarginatum

Fifty full stomachs were collected from fishes 146-262 mm TL. This species possesses small jaws armed with fine, elongate teeth (van der Elst 1988). Small reef-dwelling crustaceans dominated its diet. *Spondyliosoma emarginatum* seems capable of effectively targeting caprellid amphipods, these contributing almost a third of dietary volume, and occurring in all the stomachs analysed. Reef-dwelling gammarids were also important, occurring in 90% of stomachs and comprising over 27% of volume. The species, however, also consumes sessile organisms, including crinoid arms (13.5% volume), compound ascidians (4.4% volume) and soft coral (2.8% volume). The presence of mysids (4.6% volume) in the stomachs of some individuals suggests that this species is capable of feeding in the water column. The fish pieces contained in one stomach appeared to be bait, and so were not considered as part of the natural diet.

Spondyliosoma emarginatum is, therefore, a small-benthic-invertebrate carnivore that feeds predominantly over reefs.

Pachymetopon aeneun

Fifty full stomachs were obtained from fishes 190-367 mm TL. This species possesses a small terminal mouth with small but very sharp cutting incisors (van der Elst 1988), and is thus well adapted to clip sessile prey off the substratum. Hydroids were the most important food source (86% occurrence, 73.6% volume), with compound ascidians also being important (9% volume), although to relatively fewer individuals (28% occurrence). Individuals larger than 240 mm TL appeared to feed in two different manners. Most consumed vast quantities of hydroids, either clipping off mouthfuls at their base or chopping off pieces of algae on which the hydroids grew, with ingested caprellid and gammarid amphipods appearing to be a by-catch of the hydroid-based diet. Others, however, appeared to have targeted large caprellid amphipods (*e.g. Caprella spp.* and *Caprellina spp.*), which reside on hydroids. In these cases, the stomachs were packed with a mixture of short hydroid tips (about 30% volume) and large caprellids (70% volume), giving the impression that the fish had bitten off individual tips of hydroids on which the caprellids were found. Individuals smaller than 240 mm TL fed predominantly on reef-dwelling gammarid amphipods. In False Bay, therefore, the species appears to switch from being a small-benthic-invertebrate carnivore to a hydroid 'grazer/browser' as it grows.

Buxton & Clarke (1986) report on the diet of *P. aeneum* in Algoa Bay. Hydroids contributed just over 25% of the dietary volume at that site, considerably less than in this study, while ascidians (over 25% volume) and octocorals (17% volume) were consumed in greater amounts.

Gymnocrotaphus curvidens

Fifty full stomachs were obtained from fishes 175-414 mm TL. The species possesses a terminal mouth armed with outwardly flaring canines and some inner conical teeth (van der Elst 1988). Whole or fragmented crinoids were found in 78% of stomachs, contributing over a third of volume. Chunks of sponges (42% occurrence, 19.2% volume), compound ascidians (52% occurrence, 7.4% volume), hydroids (78% occurrence, 9.2% volume) and soft coral (50% occurrence, 9.8% volume) all contributed substantially to the diet. The majority of small invertebrates consumed by *G. curvidens* appear to have been taken as a by-catch of its sessile-invertebrate-based diet. Although some algal pieces were consumed (10.7% volume), they were covered with either hydroids or flat bryozoans, pointing to these probably being the main target.

Overall, *G. curvidens* is a generalised benthic 'grazing/browsing' carnivore that feeds almost exclusively on chunks of sessile invertebrates.

Diplodus sargus capensis

Forty-three full stomachs were analysed from fishes 168-334 mm TL. This species has a small terminal mouth with jaws armed with up to eight large cutting teeth (incisors) and several rows of molars (van der Elst 1988). A large proportion of the diet consisted of hard-shelled prey, with barnacles contributing 28.5% by volume and molluscs 17.8%. It fed on numerous other organisms, however, including polychaete worms (23.3% volume), holothuroids (over 8% volume), gammarid amphipods (7.8% volume) and compound ascidians (6.8% volume). *Diplodus sargus capensis*, therefore, appears to be a generalised benthic carnivore.

There appears to be a shift in the diet of *D. sargus capensis* along the South African coast. In Kwazulu-Natal, it is omnivorous, consuming large quantities of algae in addition to barnacles and sponges (Joubert & Hanekom 1980). These authors suggest that algae may, however, be consumed for the epiphytic diatoms that grow on them. Algae are also an important component of the diet in Algoa Bay (Coetzee 1986). South of Algoa Bay, the species appears to switch its diet, with fishes from the Tsitsikamma region (Mann 1992) and False Bay (present study) consuming mainly molluscs, barnacles and other hard-shelled prey, even though there is no shortage of algae at either site.

Diplodus cervinus hottentotus

Twenty-four full stomachs were collected from fishes 190-410 mm TL. The species possesses fleshy lips, used to manipulate prey, long, inclined and stout incisors, and some small molars (van der Elst 1988). It feeds on a wide range of sand- and reef-dwelling crustaceans, with gammarid amphipods occurring in 79.2% of the stomachs analysed and contributing 21.4% of the volume (one quarter of this being sand-dwelling individuals). Polychaete worms, swallowed whole or in large pieces, however, dominated its diet, with sedentary worms contributing 38.7% and errant worms 26.6% of volume. Hermit crabs were the only other prey item contributing to the diet (4.6% volume), the cephalothorax of these appearing to having been ripped out of the shells and consumed.

The diet of *D. cervinus hottentotus* within False Bay is very similar to that reported by Mann (1992) from the Tsitsikamma region, where individuals >150 mm TL also fed predominantly on polychaete worms and gammarid amphipods. This highlights that *D. cervinus hottentotus* is a benthic-invertebrate carnivore.

Chrysoblephus laticeps

Fifty-one full stomachs were collected from fishes 162-532 mm TL. The species possesses well-developed jaw muscles, long, stout and very pointed canines, and sharp molars (Buxton 1984). Large prey dominated the diet, including motile ones such as whole octopus (33.7% volume) and crabs (25.2% volume), and sedentary ones such as whole molluscs (*e.g.* mussels, keyhole limpets and whelks), sea anemones, echinoids, crinoids and the contents of *Pyura stolonifera* pods. The stomach contents of this species, unlike those of the other species so far discussed, contain but a few large items at a time. Prey can be very large indeed, some ingested octopus being almost as long as the fish itself (*e.g.* a 360 mm fish had consumed a 330 mm long octopus). *Chrysoblephus laticeps* was never observed feeding over sand away from reef.

Two other studies have investigated the diet of *Chrysoblephus laticeps*. Nepgen (1982) analysed specimens from False Bay, but only reported numerical and percentage occurrence information, while Buxton (1984) worked on specimens from Algoa Bay. Both studies confirm the results of this study, and support the findings that sub-adult to adult *C. laticeps* individuals are benthic macro-predators that feeds on large benthic prey. This species is presently the only abundant reef-dependent teleost in this feeding category within False Bay, although other species (e.g. *Petrus rupestris* and *Chrysoblephus gibbiceps*) would have also fitted within this category in the early 1900s, before they became over-exploited (Biden 1954).

Parascorpis typus

Fifty-one full stomachs were collected from fishes 80-334 mm TL. This species possesses a large, wide mouth and long gill-rakers that enable it to strain zooplankton prey from the water (Smith & Heemstra 1988). It is a very specialised carnivore that feeds almost exclusively on the kelp mysid *Mysidopsis major* (100% occurrence, 92.6% volume). The only other food items occasionally consumed were also pelagic, including small (late stage larvae or tiny juveniles) pelagic fishes (2.3% volume) and juvenile squids (3.1% volume). *Parascorpis typus* is, therefore, a midwater small-invertebrate carnivore.

Dietary information obtained from the literature

Pachymetopon blochii possesses small incisors (van der Elst 1988), and is omnivorous, feeding on a wide variety of algae and small benthic invertebrates (Nepgen 1977; Pulfrich & Griffiths 1988a).

Sarpa salpa is reported to be herbivorous in KwaZulu-Natal, algae comprising over 90% of its diet (Joubert & Hanekom 1980). It was assumed to also be herbivorous within False Bay, although the fact that the diet of *Diplodus sargus capensis* switches from being algal-based to animal-based along the South African coastline casts doubt on the accuracy of this. The assumption is defended, however, as unlike *D. sargus capensis*, *Sarpa salpa* has body features typical of a herbivore, including a small mouth with small but very sharp cutting teeth, and a long gut that enables it to digest seaweed.

Dichistius capensis possesses stout incisors and molariform teeth on the pharyngeal plates. This species feeds primarily on barnacles and mussels on inshore False Bay reefs, although reef-dwelling gammarid amphipods are also consumed in large quantities (Bennett & Griffiths 1986).

Haploblepharus edwardsii, *H. pictus*, *Poroderma africanum* and *P. pantherinum* all possess small teeth adapted for grasping prey (Bass *et al.* 1975; Compagno, Ebert & Smale 1989). Prey are usually swallowed whole or in large chunks, and include large crustaceans (crabs and *Jasus lalandii*), cephalopods (*Octopus spp.* and cuttlefish), and both cryptic and demersal fishes. *Haploblepharus edwardsii* also feeds on large polychaetes (Bertolini unpublished data). These demersal sharks are thus all benthic macro-predators that feed on large motile benthic prey. On the few occasions when *P. africanum*, *P. pantherinum* and *H. edwardsii* were observed feeding, they were visibly searching for prey. Once found, they attacked it (*Octopus vulgaris*) by grasping a leg and rotating themselves while holding it. This could last for more than five minutes, before a portion was twisted off and ingested.

Dietary overlap

The results of the dietary analyses are based on relatively small sample sizes, and do not take into account the dietary requirements of juveniles. Juveniles of numerous species, however, are likely to consume different prey compared to adults (*e.g.* Mann 1992). Dietary variation due to fish size did not appear to affect results much, however, since *Pachymetopon aeneum* was the only species that appeared to switch diet within the size range analysed. The obtained information was thus deemed sufficient to undertake a preliminary investigation of dietary overlap amongst species. Tables 3.2 & 3.3 show that the 17 species analysed exploit a wide spectrum of prey items, and that there is some degree of overlap in the diets of most species. For example, seven species feed regularly (>10% of dietary volume) on gammarid amphipods. Indeed, most of the species analysed share a number of prey items with other species. However, most do have a combination of important food items that is peculiar to them. These, therefore, appear to have their own 'dietary niche' (*e.g.* caprellid amphipod specialist - *Spondyliosoma emarginatum*, hydroid 'grazer' - *Pachymetopon aeneum*, mysid specialist - *Parascorpis typus*) within the suprabenthic assemblage.

This is supported by the results of the cluster and ordination analyses comparing the diets (% volume) of the different species (Figure 3.1a & b). Both *Poroderma* and *Haploblepharus spp.* grouped together with *Chrysoblephus laticeps* into a benthic macro-predator cluster. These species feed on cephalopods, large crustaceans and fish. *Parascorpis typus*, the only midwater small-invertebrate carnivore, formed a separate isolated group. The diets of *Gymnocrotaphus curvidens*, *Pachymetopon aeneum* and *Sarpa salpa* are sufficiently similar for them to have grouped together as 'benthic grazers'. However, of the three, *S. salpa* is the only specialised herbivore, whereas *G. curvidens* and *P. aeneum* are both carnivorous, *P. aeneum* feeding principally on hydroids, and *G. curvidens* on a variety of sessile colonial invertebrates. The remaining eight species formed a single grouping on the ordination plot, the stress (0.012) of which was relatively high (Field *et al.* 1982). The program was thus re-run using only this last group of species in an attempt to get a clearer clustering pattern of these (with less stress). The stress of the resulting plot (0.06) was greatly reduced, while the clustering pattern obtained was also clearer (Figure 3.1c). *Diplodus sargus capensis* and *Dichistius capensis* clustered from the other species due to the abundance of both mussels and barnacles in their diets. The small-benthic-invertebrate-carnivores (*B. inornata*, *C. fasciatus*, *Chirodactylis brachydactylis* and *S. emarginatum*) and the only omnivore (*P. blochii*) grouped together, while *Diplodus cervinus hottentotus* clustered separately, due to the

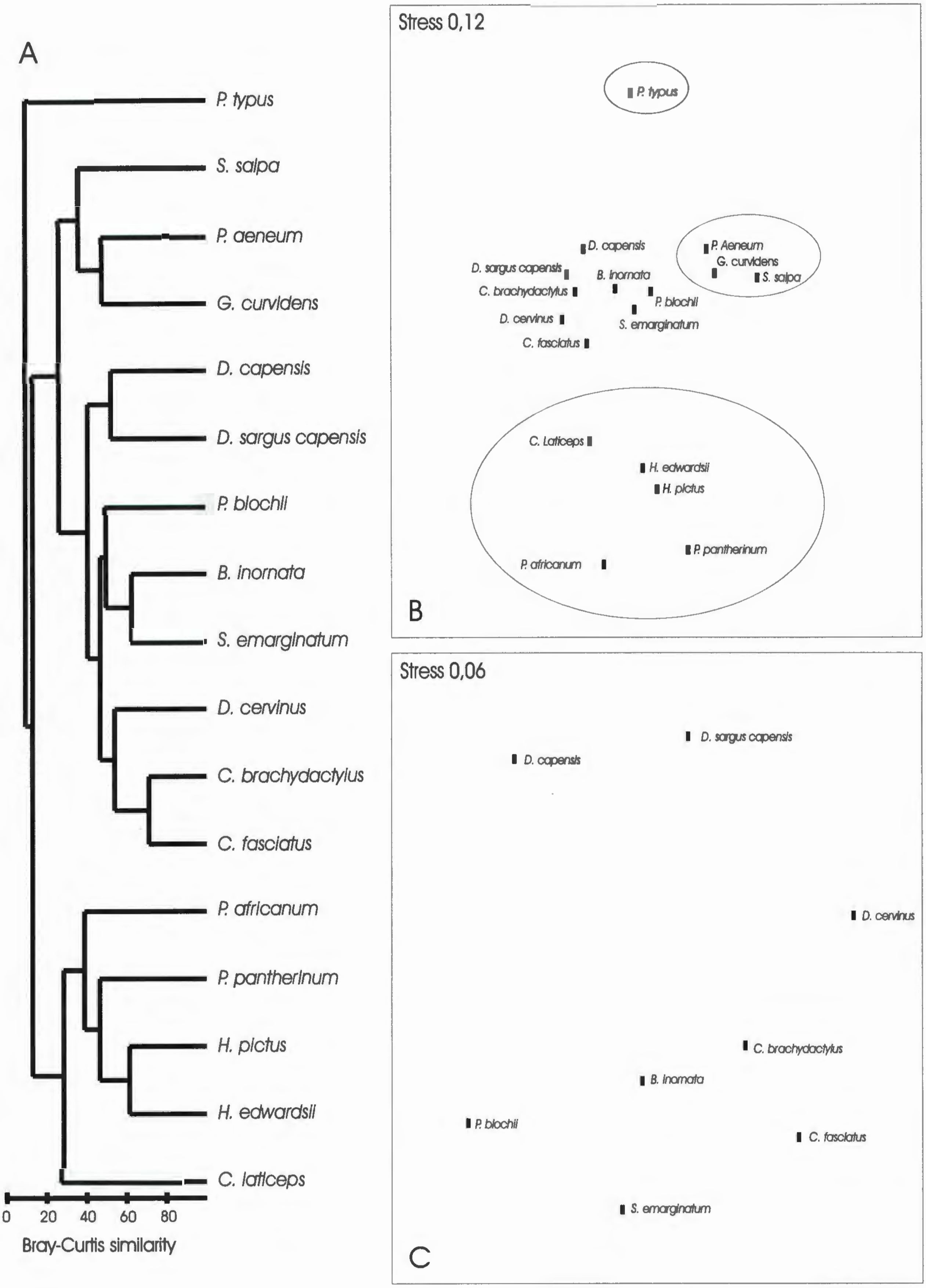


Figure 3.1.A. Dendogram showing the Bray-Curtis similarity between the dietery composition (based on %volume) of the different reef fish species.

B. Multi-dimentional scaling plot showing how these species' dietary compositions are related in space.

C. Multi-dimentional scaling plot showing how the dietary compositions of the species importance of polychaetes in its diet. Based on these results, the 17 False Bay reef fish species examined are classified into the seven feeding groups listed in Table 3.4. It must be remembered, however, that these categories are only based on the diet of sub-adult to adult individuals, with the diets of juveniles of most species likely to be different.

Table 3.4 The seven feeding groups into which the species investigated seperated into, according to their dietary composition.

Feeding group	Species
Generalised benthic carnivore	*Dichistius capensis* *Diplodus sargus capensis*
Benthic 'grazer'	*Gymnocrotaphus curvidens* *Sarpa salpa*
Hydroid 'grazer'	*Pachymetopon aeneum*
Small-invertebrate midwater predator	*Parascorpis typus*
Benthic macro-predator	*Chrysoblephus laticeps* *Haploblepharus edwardsii* *Haploblepharus pictus* *Poroderma africanum* *Poroderma pantherinum*
Small-benthic-invertebrate carnivore	*Boopsoidea inornata* *Cheilodactylis fasciatus* *Chirodactylis brachydactylis* *Pachymetopon blochii* *Spondyliosoma emarginatum*
Polychaete 'specialist'	*Diplodus cervinus hottentotus*

Interspecific competition for food among the species analysed is reduced since many of them have a combination of important food items peculiar to them. It is further reduced as some are distributed differently with respect to depth, availability of cave and/or reef elevation (Chapter 2). However, it is not known whether food is limiting within False Bay, or whether interspecific competition for food alone limits spatial overlap amongst them, as factors such as turbulence and prey distribution could also affect their spatial distribution in False Bay. It is likely that past and present food chains on False Bay reefs are very different. The over-exploitation of certain species (*e.g. Chrysoblephus gibbiceps*, *C. laticeps* and *Petrus rupestrus*) in the past could have released others from predation pressure and/or competition for space, thereby potentially increasing competition for smaller prey taxa (*e.g.* amphipods). Such ecosystem effects are, however, very difficult to quantify.

The 17 species analysed here make up almost 70% of the suprabenthic reef fish species, and over 98% of the suprabenthic individuals inhabiting shallow (0-25 m deep) False Bay reefs (Chapter 1). Suprabenthic fishes, however, form but one part of the False Bay reef fish assemblage, the other part comprising small cryptic fish species. Prochazka (1994) examined the diets of 20 False Bay reef-dwelling cryptic species and reports that amphipods and isopods dominate their diets. Suprabenthic and cryptic species shared numerous amphipod species, including sand- and reef-dwelling species, highlighting that these groups may compete for food to some extent. This competition for food is likely to be more intense between cryptic and juvenile suprabenthic fishes, however.

In conclusion, this study reports on the dietary composition of 17 suprabenthic reef fish species common on False Bay reefs. Of the ten species analysed, only *Diplodus sargus capensis* was found to have fed in water 13°C or colder. Reef-dwelling prey dominated the diets of all species studied. Consumed sand-dwelling prey are likely to be taken within close proximity of reef. Although some dietary overlap exists among the species, most have some combination of important food item consumed predominantly by them. Each species, therefore, appears to have its own 'dietary niche' within the assemblage.

Chapter 4

The False Bay spearfishery: Its catch composition, catch rate, and management

Sarpa salpa

Introduction

The spearfishery is a unique fishery sector as participants target individual fish visually (Bohnsack 1981; Denis 1988), experienced divers selecting prey by size, species, and sometimes even sex (Maas 1995). Because they select larger species and individuals within each species, spearfishers affect the composition of reef fish assemblages at popular spearfishing sites (Bohnsack 1981; Denis 1988). In South Africa, the spearfishery is the smallest of the linefishery sectors, with about 7 000 participants in 1996, increasing in numbers at over 6% per annum (Mann, Scott, Mann-Lang, Brouwer, Lamberth, Sauer & Erasmus 1997). These estimates, however, are based primarily on annual spearfishing licence sales in Kwazulu-Natal (KZN) and club:non-club ratios, with very little information concerning the spearfishery being available outside of KZN (Mann *et al.* 1997). From the start of this study till the end of 1998, spearfishing licence were only required in KZN (van Rooyen 1988; Cook 1990), although subsequently all linefishers (including spearfishers) have had to obtain a permit to exploit fish resources (Living Marine Resources Act 1998). The total number of spearfishers, however, remains unlimited.

This is the first study of the spearfishery of an area within the Western Cape, and although competition results from the Western Cape Province are available on the National Marine Linefish System (NMLS), little is known about non-competitive catches from the region (Mann *et al.* 1997). Because of this, the catch composition and catch rate of the regional spearfishery are unknown, as are its impacts on target species. Both non-competitive and competitive spearfishing is practised, both categories being controlled by the same minimum sizes, closed seasons and equipment restrictions. Competitive spearfishers, however, are further restricted by self-imposed competition rules (Cook 1990). The regulations and competition rules that control spearfishing, along with the physical restrictions of the sport, are as follows:

Non-competitive spearfishing

Important regulations include those which prohibit spearfishers using any breathing apparatus other than a snorkel, artificial light sources, thunderheads (powerheads) or other explosive devices, gaffs, baton and handspears to facilitate the spearing of fish (van Rooyen 1988; Cook 1990; Mann *et al.* 1997). Spearfishing is also prohibited outside of daylight hours (van Rooyen 1988), in estuaries, within most Marine Protected Areas, and in certain restricted areas such as harbours (Sea Fisheries Act No. 12 of 1988). The closed seasons and bag limits for spearfishers are the same as those enforced on all South African recreational linefishers (van Rooyen 1988; Cook 1990; Mann *et al.* 1997). The minimum sizes enforced on spearfishers outside KZN are the same as those enforced on recreational and commercial fishers outside that region (Cook 1990).

Spearfishing is a very physically demanding sport (Cook 1990; Theunissen 1998). Being breath-hold divers, spearfishers are not able to spearfish at all depths, with only fit, experienced participants capable of diving deeper than 20 m. Most spearfishers rarely dive deeper than 12 m. Other important physical restrictions include water clarity and temperature, swell, site accessibility, and wind direction and strength. The fear of sharks and shallow-water blackout further restricts the effort of prospective spearfishers (van Rooyen 1988; Cook 1990).

Competitive spearfishing

Competitive spearfishing involves divers, or team of divers, competing against one another according to additional competition rules. Two types of competitions are dived within False

Bay, one involving contestants who are restricted to operating from the shore, the other involving divers allowed to dive from boats. Shore-based competitions usually involve contestants from one club competing amongst themselves, although during the study period, the main event was the annual "Allrounders Competition", dived by most Cape Town underwater clubs. Boat-based competitions are those where contestants compete for provincial colours (Boland or Western Province), and thereby a right to represent their province at the annual National Spearfishing Championships. Within False Bay, shore-based competitions are rare, and their cumulative annual attendance low compared to that of boat-based competitions.

Apart from the physical restrictions and most of the regulations that restrict non-competitive spearfishers, competitive divers are also restricted by additional self-imposed competition rules. These include a pre-determined dive time (usually 4-5 hours), a limited competition area and species-dependent weight limits. During most competitions, a diver may present a maximum of two fish per species at the weigh-in, and may not spear more than two individuals per species, although he/she is not restricted in the number of species presented. Because this could contravene certain enforced recreational daily bag limits, organisers of provincial and national competitions apply to have these wavered. Although shore-based competition organizers did not apply for this exemption, participants in fact also shot fish in excess of daily bag limits. The diver scores points if a speared fish weighs more than the minimum weight stipulated for that species (see Table 4.1), although he/she is not penalised if it is too small to qualify for points. Fish speared during competitions may not be smaller then the minimum legal size for the species concerned, however. The species-dependent weight limits set during different spearfishing competitions are listed in Table 4.1, and are mostly well above the minimum legal size restrictions. A diver is awarded one point per fish above

Table 4.1 The minimum size restrictions enforced on all spearfishers within False Bay, and the minimum weight/size restrictions set during the different spearfishing competitions held within False bay.

	Minimum size limits (*) enforced on all spearfishers	Competition weight limits (fish < than this are not weighers)			
		Shore-based competitions		Boat-based competitions	
Species	Shore and boat based Minimum size (cm TL) (&)	Allrounders 96 (cm TL)	Other allrounders () value for ladies kg	Western Province kg	Boland kg
Lithognathus lithognathus	60 (1.79 kg)	60	3	3	3
Petrus rupestris	40 (0.93 kg)	40	3	3	3
Chirodactylus grandis	none	none	1-0.8 (0.6)	1	1
Chrysoblephus gibbiceps	30 (0.46 kg)	30	1-0.8 (0.6)	1	1
Chrysoblephus laticeps	30 (0.45 kg)	30	1-0.8	1	1
Dichistius capensis	35 (0.78 kg)	35	1-0.8	1	1
Gymnocrotaphus curvidens	none	none	1-0.8 (0.6)	1	1
Seriola lalandi	none	none	1-0.8 (0.6)	1	1
Pachymetopon aeneum	none	none	0.8 (0.5)	0.8	1
Pachymetopon blochii	22 (0.2 kg)	22	0.8 (0.5)	0.8	1
Chirodactylus brachydactylus	none	none	0.8 (0.5)	0.8	1
Diplodus cervinus hottentotus	30 (0.42 kg)	30	0.8	0.8	1
Boopsoidea inornata	none	none	0.4	0.4	1
Cheilodactylus fasciatus	none	none	0.4	0.4	1
Diplodus sargus capensis	20 (0.13 kg)	20	0.4	0.4	1
Rhabdosargus globiceps	25 (0.18 kg)	25	0.4	0.4	1
Sarpa salpa	15 (0.05 kg)	15	0.4	0.4	1
Spondyliosoma emarginatum	none	none	0.4	0.4	1
Oplegnathus conwayi	none	none	1-0.8 banned 1994	banned 1994 (1)	1
Parascorpis typus	none	none	0.8 banned 1994	banned 1994 (0.8)	1

* - Based on regulations listed in SFRI (1996)

& - Mass of individuals that size, based on length/weight regressions listed in van der Elst & Adkin (1991) that minimum weight, and one point per kilogram (up to a maximum of three kilogram) for those fish that qualify. The winner is the diver awarded the most points.

This study reports on the catch composition (species, size, number of fish) and catch rate of non-competitive and competitive spearfishers in False Bay (Southwestern Cape). The information reported in this chapter will be used in Chapter 5 to compare the catch composition of the spearfishing sector to those of the other False Bay linefishery sectors.

Methods

Information concerning non-competitive spearfishers was collected between January 1992-2000 by interviewing divers as they came out of the water, or through personal contact. Information noted from each diver included: total time spent spearfishing, composition of catch by species, number and size of fish (total length – TL in mm for demersal teleosts, or mass for pelagic fish and sharks), area dived, and estimated water temperature on the bottom. Temperature estimates were subsequently categorised into "warm" (>13°C) or "cold" (13°C or colder). To estimate which areas were most popular with non-competitive and competitive spearfishers, the False Bay coastline was divided into the nine different areas shown in Figure 4.1. Intensity of utilisation was calculated from the number of non-competitive dives and competitions held within each area during the study period. The borders of each Area were set according to known reef availability, reserve boundaries, competition area boundaries and occurrence of *Seriola lalandi*.

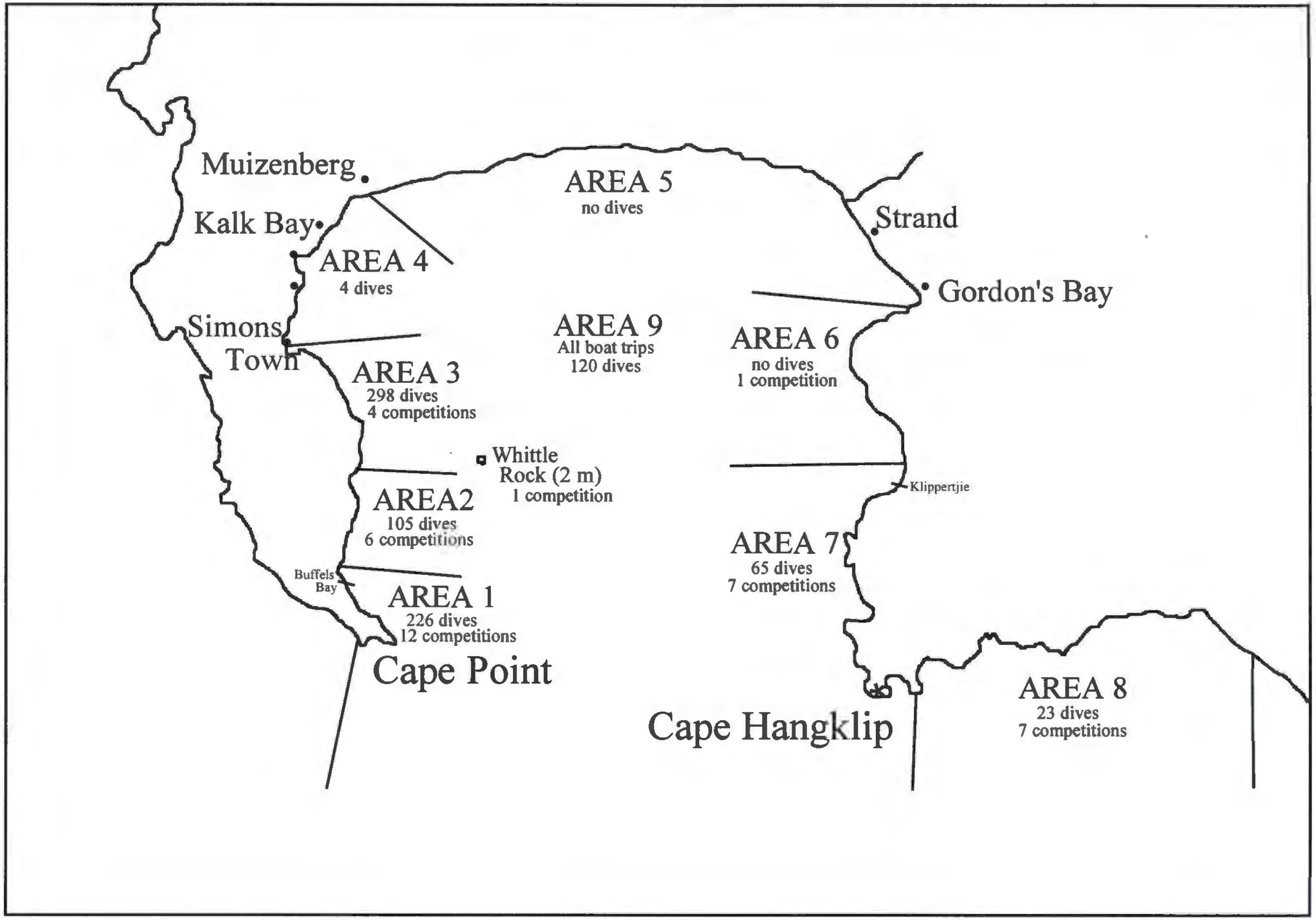


Figure 4.1 The nine spearfishing areas delineated within False Bay, and the number of inspected non-competitive dives and spearfishing competitions (shore- and boat-based) undertaken within each. Area 9 represents all areas dived from a boat by non-competitive spearfishers, irrespective of whether it is within reach of shore-based divers.

The equipment, experience, ability (e.g. maximum depth dived), level of knowledge, and spearfishing competition experience of each diver were also noted. This was used to categorise each diver into one of three classes. The success rates of individuals were then evaluated according to these classes. The three diver classes used were:

Class 1 - These divers are inexperienced, own unsophisticated diving equipment, dive at irregular intervals, have little knowledge of the areas they dive, nor of the fish they target, and cannot dive repeatedly to a depth exceeding 12 m. Although they may participate in shore-based competitions in False Bay, they do not participate in provincial competitions.

Class 2 - These divers are more experienced, own some sophisticated equipment designed for spearfishing (e.g. pair of long fins), usually dive more regularly than Class 1 divers, have some knowledge of the identity and whereabouts of the more common species they target, and can dive repeatedly to depths exceeding 16 m. They are experienced enough to participate in provincial competitions, although they do not score well during these.

Class 3 - These divers are the most experienced, own very sophisticated spearfishing equipment (e.g. long fins, special spearfishing wetsuit, good low-volume mask) and often more than one new speargun. They dive regularly, even when the water is "cold", have a good understanding of the behaviour and whereabouts of the species they target in all conditions, know most of the different reefs in False Bay, and can dive repeatedly to depths exceeding 20 m. Many of them participate in provincial competitions where they score well.

Some divers forwarded information from dives recorded in a diary. This information was only accepted if it reported information on all the dives undertaken by the diver during the time period recorded. When fish sizes were recorded by mass, these were converted to length by comparing previously weighed and measured fish of the same species, or by using available length-weight regressions. If a diver had not recorded the size or mass of his catch, but had noted all other required information, this information was only employed to calculate catch composition and catch-per-unit-effort (*cpue*) of spearfishers in False Bay. All *cpue* values were calculated as the cumulative number of fish shot per 100 hours by a particular group of divers.

Data were collected for competitive dives between January 1992-December 1996. The catch landed during 42 boat-based competitive dives and 44 shore-based competitive dives was obtained by analysing the catch of divers as they exited the water. This enabled the authors to note the number of 'extras' (fish shot that were not presented at the weigh-in, and/or that did not qualify for points) speared by these divers. Another set of information utilised was the spearfishing competition results of the Western Province Underwater Union (WPUU). These recorded the total number of 'weighers' (fish presented to the weigh-master that were larger than the minimum weight) shot during all 38 competitions organised by the Western Province Underwater Union from 1992-1996. The species composition of the catch was only available for 33 of these. The competition results, along with the diver inspections, provided information concerning the number of accepted 'weighers' shot. Furthermore, 15 of the Western Province competitions also recorded the number of 'non-weighers' (presented fish that weighed less than the weight limit) for each of the species shot. The proportion of 'non-weighers' to 'weighers' could thus be calculated for boat-based competitive divers from inspected dives and competition results. The greatest of the two proportions of 'extras' to 'weighers' for each species was then employed to estimate the total number of fish shot during the 33 competitions. The total number of fish shot during the 38 competitions held during this study was estimated by multiplying the proportion of ‘extras’ to ‘weighers’ obtained from the 42 inspected boat-based dives by the total number of ‘weighers’ shot during the 38 competitions. Due to insufficient available information concerning the number of ‘weighers’ and ‘extras’ speared during the 1992, 1993, 1995 and 1996 “Allrounders Competitions”, and the different minimum sizes and bag limits enforced during these (Table 4.1), the overall number of ‘extras’ shot during shore-based competitions was not estimated.

Whenever sufficient fish measurements were obtained for different species, the size distribution of fish speared by different groups of divers were compared using the non-parametric Kolmogorov-Smirnov two-sample test.

Results

Non-competitive spearfishery

Information from 875 shore-based (1 985.1 hrs) and 167 boat-based (450.5 hrs) dives was obtained. Of the 218 divers inspected, 57.8% (including seven of the eight ladies) were Class 1, 28% Class 2 and 14.2% Class 3 divers.

Figure 4.1 shows the spatial distribution of diving effort. Overall, 41% of the inspected dives were undertaken within Area 3, highlighting its popularity with shore-based divers. The majority of dives were undertaken on the western shore of the Bay, although the east coast of False Bay is dived more regularly than Figure 4.1 shows. Spearfishers target mainly *Seriola* *lalandi* off Buffels Bay (western shore) and Klippertjie (eastern shore of the Bay - Figure 4.1), whereas the rest of the Bay is dived mainly in the hope of shooting reef fish.

The average dive time of Class 3 divers diving from the shore was longer than that of Class 2 and Class 1 divers respectively, irrespective of water temperature (Table 4.2). Overall, the total dive-time of Class 1 divers dominated the False Bay non-competitive shore-based spearfishery, especially when the water was "warm", although that of Class 3 spearfishers was considerable, especially when the water was "cold" (Table 4.2). The mean dive-time of shore-based spearfishers was 2.27 hours.

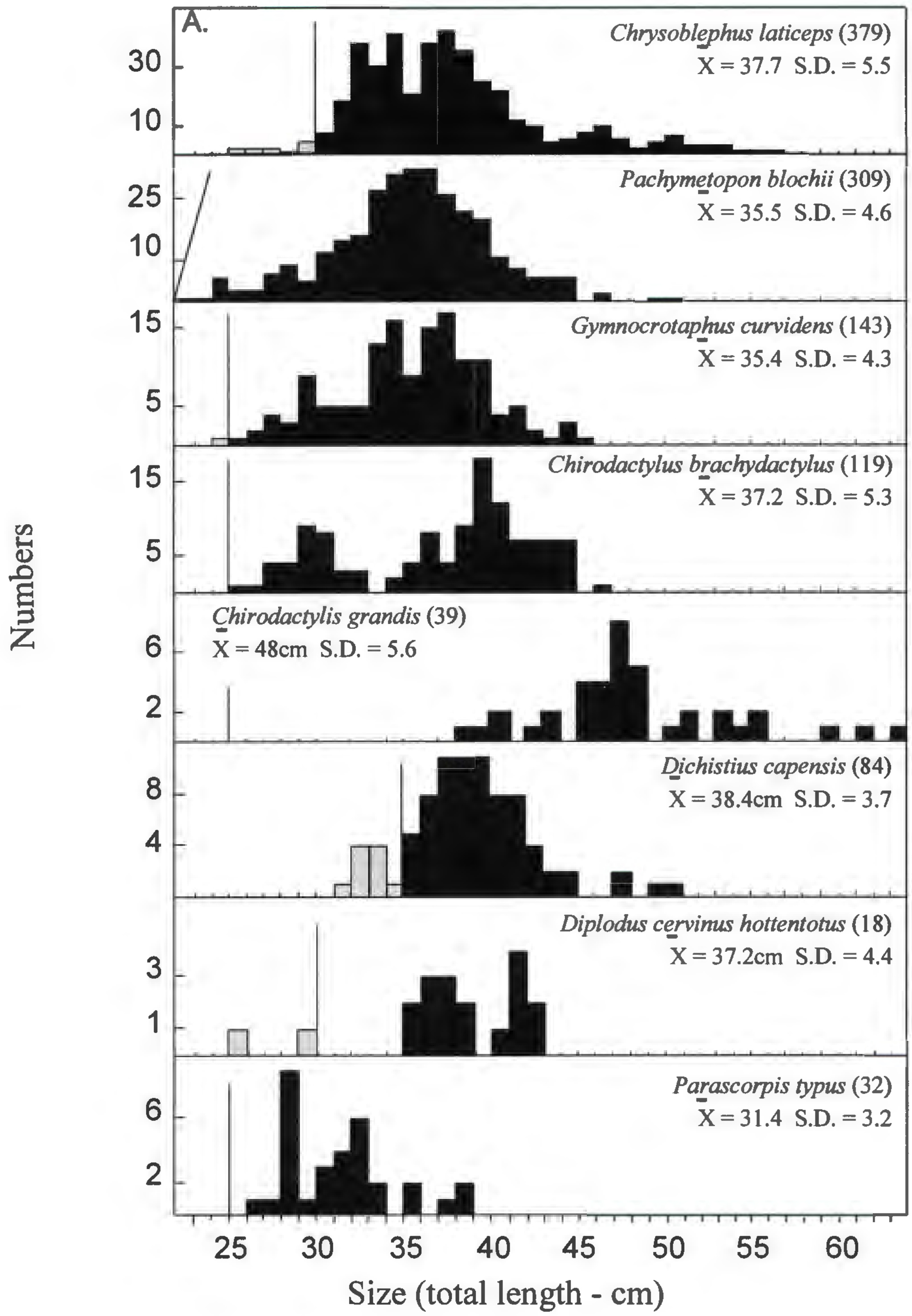
Of the 167 boat-based dives inspected, 25 were undertaken by Class 1, 58 by Class 2 and 84 by Class 3 divers (Table 4.3). The mean dive-time (excluding boat ride) of Class 3 divers was greater than that of the other classes in "cold" water, although that of Class 1 divers was greatest when the water was "warm" (Table 4.3). Overall, the mean dive-time of boat-based spearfishers was 2.7 hours, which is greater than that of shore-based spearfishers (Tables 4.2 & 4.3).

The *cpue* of spearfishers was affected by a combination of factors. The overall *cpue* of shore-based divers was 60.4 fish/100hrs (Table 4.2), which was greater than that of boat-based divers (84.8 fish/100hrs - Table 4.3). It also varied with water temperature, being lower in "cold" compared to "warm" water (Tables 4.2 & 4.3). *Cpue* was also affected by diver experience, with that of the most experienced (Class 3) divers being more than double that of Class 1 or 2 divers in both "warm" and "cold" water (Tables 4.2 & 4.3).

Altogether, non-competitive spearfishers landed 23 species (including 4 chondrichthians), 22 of which were shot by shore-based divers (Table 4.2) and 12 by boat-based divers (Table 4.3). All chondrichthians, *Chrysoblephus gibbiceps*, *Lithognathus lithognathus*, *Parascorpis typus*, *Petrus rupestris* and *Sarda sarda* were only shot by shore-based divers, while *Rhabdosargus globiceps* was only shot by boat-based divers. The overall catch this sector was dominated by reef-dependent species, these comprising over 93% of fish numbers and 75% of species. The five numerically most important reef dependent species were *C. laticeps* (31.4%), *Pachymetopon blochii* (22.6%), *Gymnocrotaphus curvidens* (10.3%), *Dichistius capensis* (7.9%) and *Chirodactylus brachydactylus* (7.3%). Of the four chondrichthian species shot, *Mustelus mustelus*, *Triakis megalopterus* and *Notorhynchus cepedianus* were shot in appreciable quantities (over 700 kg from only eight dives during which these were landed). *Seriola lalandi* was the only pelagic species shot regularly (over 6% of fish numbers).

The size distributions, by length (cm TL) or mass (kg), of species commonly speared by spearfishers are depicted in Figure 4.2. More than 93% of the reef fish were longer than 30 cm TL. However, it is important to note that the different classes of spearfishers contributed different portions to these size distributions, with Class 3 divers spearing significantly more large and Class 1 divers more smaller individuals ($P<0.05$) (Figure 4.3).

Divers speared fish for one or more of the following reasons: good eating quality, trophy size, practice for spearfishing competitions, and illegal sale. The first two reasons apply to all non-competitive spearfishers, the third to relatively few Class 2 and 3 divers and the fourth mainly to Class 3 divers. The majority of *Cheilodactylis fasciatus*, *Diplodus cervinus hottentotus*, *D.*



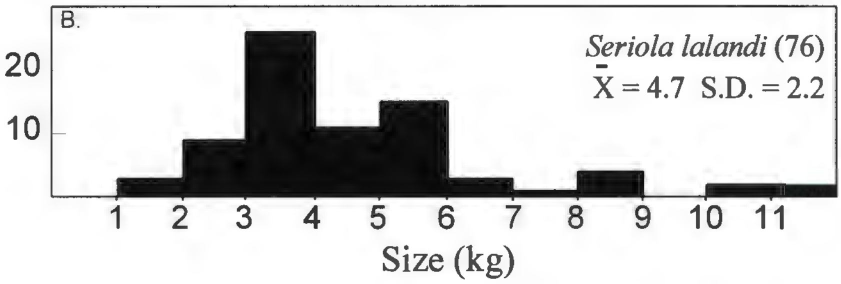


Figure 4.2A Size distribution plus mean size (total length - cm) and standanrd deviation of species speared within False Bay by non-competitive spearfishers.

B Size distribution (mass - kg) of *Seriola lalandi* shot within False Bay by non-competitive spearfis

Note, only species for which more than 14 measurements were obtained are shown. Line represents enforced minimum size restriction for the species concerned. Bars to the left of it represent undersiz individuals.-

Table 4.2 The % catch composition and *cpue* of the different classes of shore-based non-competitive spearfisher active within False Bay. The letter in brackets reports under which species list' each species is listed in SFRI (1996)

	Diver class						All divers		Overall total (all divers)	
	Class 1		Class 2		Class 3					
Water temperature	Warm	Cold	Warm	Cold	Warm	Cold	Warm	Cold	All temperatures	
	% numbers	% numbers	% numbers	% numbers	% numbers	% numbers	% numbers	% numbers	Numbers	% numbers
Species										
Chrysoblephus laticeps (a)	30.62	28.36	37.86	54.70	26.62	25.00	30.55	29.50	363	30.28
Pachymetopon blochii (c)	28.23	42.28	18.45	16.98	19.38	24.64	21.45	26.77	279	23.27
Gymnocrotaphus curvidens (b)	13.40	4.48	4.37	13.21	9.30	9.78	9.10	9.34	110	9.17
Chirodactylus brachydactylus (c)	10.53	13.43	6.80	1.89	11.89	4.71	10.22	5.81	106	8.84
Dichistius capensis (b)	5.26	2.99	8.74	1.89	11.37	7.97	9.10	6.31	98	8.17
Chirodactylus grandis (c)	0.96		1.94	5.66	2.84	4.35	2.12	3.79	32	2.67
Mustelus mustelus (c)						10.87		7.58	30	2.50
Cheilodactylus fasciatus (c)				1.89	4.39	1.09	2.12	1.01	21	1.75
Diplodus sargus capensis (b)	0.96		0.97		2.07	2.90	1.50	2.02	20	1.67
Parascorpis typus (c)	3.83	7.47	3.40				1.87	1.27	20	1.67
Diplodus cervinus hottentotus (a)	2.39		0.49	3.77	0.78	1.09	1.12	1.26	14	1.17
Triakis megalopterus (c)						3.26		2.28	9	0.75
Pachymetopon aeneum (a)					1.81	0.73	0.87	0.51	9	0.75
Oplegnathus conwayi (b)	0.96		1.46				0.62		5	0.42
Notorhynchus cepedianus (c)						1.81		1.26	5	0.42
Chrysoblephus gibbiceps (c)			0.97		0.26	0.36	0.37	0.25	4	0.33
Petrus rupestris (d)			0.49				0.13		1	0.08
Umbrina canariensis (b)						0.36		0.25	1	0.08
Epinephelus marginatus (a)					0.26		0.13		1	0.08
Sparodon durbanensis (b)	0.48						0.13		1	0.08
Lithognathus lithognathus (b)					0.26		0.13		1	0.08
Seriola lalandi (c)	2.87		12.14		8.53		7.98		64	5.34
Sarda sarda (c)			1.46				0.37		3	0.25
Trachurus trachurus (e)			0.49				0.13		1	0.08
Carcharhinus brachyurus (c)			0.49				0.13		1	0.08
Total number of fish	209	67	207	53	387	276	803	396	1199	
Total number of hours	504.94	210.16	420.55	181.86	361.40	305.70	1286.89	697.72	1984.61	
Cpue	41.39	31.88	49.22	29.14	107.08	90.29	62.40	56.76	60.42	
Number of dives	247	108	174	85	134	127	555	320	875	
Number fish/dive	0.85	0.62	1.19	0.62	2.89	2.17	1.45	1.24	1.37	
Mean dive time (hrs)	2.04	1.94	2.42	2.14	2.70	2.41	2.32	2.18	2.27	

a - Restricted species
b - Recreational species
c - Exploitable species
d - Critical species
e - Bait species

Table 4.3 The % catch composition and *cpue* of the different classes of boat-based non-competitive spearfishers active within False Bay. The letter in brackets reports under which 'species list' each species is listed in SFRI (1996)

	Class 1		Class 2		Class 3		All divers		All divers (overall total)	
Water temperature	Warm	Cold	Warm	Cold	Warm	Cold	Warm	Cold	All temperatures	
	% numbers	% numbers	% numbers	% numbers	% numbers	% numbers	% numbers	% numbers	Numbers	% numbers
Species										
Chrysoblephus laticeps (a)	30.77	10.00	40.91	50.00	38.75	23.74	38.71	25.45	126	32.98
Pachymetopon blochii (c)	53.85	80.00	18.18	31.25	16.25	29.50	18.89	32.73	95	24.87
Gymnocrotaphus curvidens (b)	15.38		9.09	6.25	16.25	7.19	14.75	6.67	43	11.26
Chirodactylus grandis (c)			4.55		6.25	21.58	5.53	18.18	42	10.99
Dichistius capensis (b)			9.09		5.00	7.91	5.53	6.67	23	6.02
Chirodactylus brachydactylus (c)		10.00			3.75	4.32	2.76	4.24	13	3.40
Cheilodactylus fasciatus (c)				12.50	1.88	2.16	1.38	3.03	8	2.09
Pachymetopon aeneum (a)					1.88	0.72	1.38	0.61	4	1.05
Diplodus cervinus hottentotus (a)			2.27		1.25		1.38		3	0.79
Diplodus sargus capensis (b)					0.63	1.44	0.46	1.21	3	0.79
Rhabdosargus globiceps (c)					1.88		1.38		3	0.79
Seriola lalandi (c)			15.91		5.00	1.44	6.91	1.21	17	4.45
Thunnus alalunga					1.25		0.92		2.00	0.52
Number of fish shot	13	10	44	16	160	139	217	165	382	
Number of hours	43.5	21.25	78.25	66.25	115	126.25	236.75	213.75	450.5	
Cpue	29.89	47.06	56.23	24.15	139.13	110.10	91.66	77.19	84.79	
Number of dive trips	14	11	32	26	44	40	90	77	167	
Catch/trip	0.93	0.91	1.38	0.62	3.64	3.48	2.41	2.14	2.29	
Mean dive time (hrs)	3.11	1.93	2.45	2.55	2.61	3.16	2.63	2.76	2.70	

a - Restricted species
b - Recreational species
c - Exploitable species

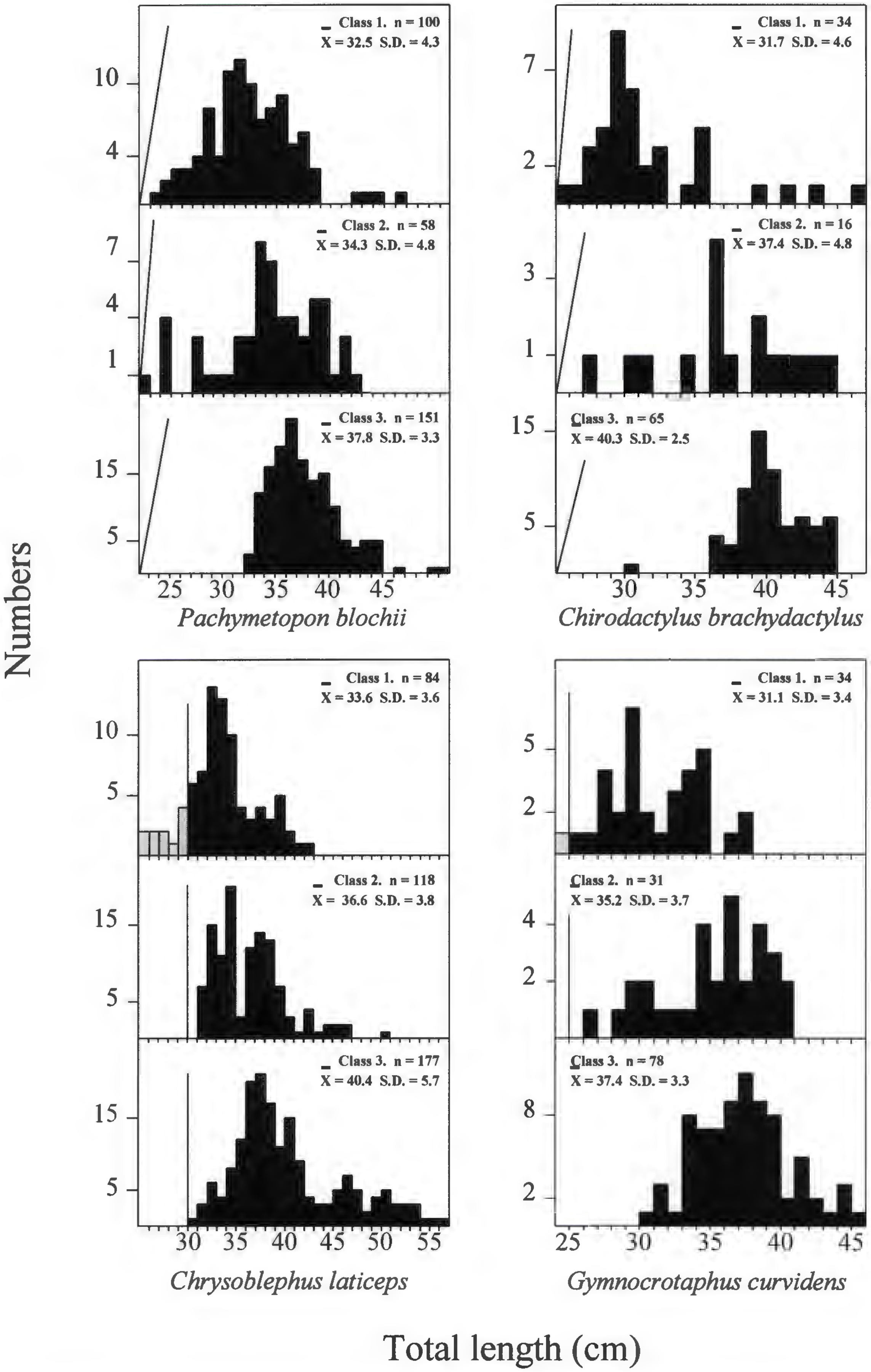


Figure 4.3 Size distribution plus mean size and standard deviation of the four species shot most regularly by the three non-competitive spearfisher classes within False Bay. Line represents the enforced minimum size restriction for the species concerned. Bars to the left of it represent undersize individuals.

sargus capensis and *Pachymetopon aeneum* landed by Class 2 and 3 divers (Tables 4.2 & 4.3) were speared by divers practising for competitions.

Competitive spearfishery

Information from 92 shore-based (311 hrs) and 656 boat-based (2700 hrs) competitive dives was obtained. Numerous ladies took part in the shore-based competitions as both sexes competed within separate categories during these, whereas none participated in boat-based competitions. Figure 4.1 shows that most competitions were organised from the shore or boats off Area 1 (32% of shore- and boat-based competitions combined).

Shore-based competitions lasted for a minimum of three hours, which is longer than the mean dive-time of shore-based non-competitive spearfishers (2.27 hrs). Provincial (Boland and Western Province) boat-based competitions had a minimum duration of four hours, which is also longer than the mean dive-time of boat-based non-competitive divers (2.7 hrs).

The *cpue* ("weighers" only) of shore-based participants was 50.8 fish/100hrs (Table 4.4), which is less than the *cpue* of non-competitive shore-based spearfishers (60.4 fish/100hrs – Table 4.2). It is also much less than that of boat-based divers (77 fish/100hrs). However, if the estimated number of "extras" speared by boat-based participants is taken into account when calculating their *cpue*, this increases to 117.5 or 161.8 fish/100hrs depending on the data employed (Table 4.4). These estimated *cpue*s are considerably greater than that of non-competitive boat-based divers (84.8 fish/100hrs – Table 4.3).

Table 4.4 The catch composition of shore- and boat-based competitive spearfishers active within False Bay. The numbers in brackets indicate the number of dives or competitions from which the values were calculated. The letter in brackets reports under which 'species list' each species is listed in the regulations explained in SFRI (1996)

	Diver class					All divers
	BBCD				SBCD	
Species	Weigher number (N) (33 competitions)	% extras (NW) (*) (42 dives)	Presented NW/W (15 competitions)	Est. overall total Highest % for 33 comp (&)	% numbers (44 dives)	% catch composition ($)
Pachymetopon blochii (c)	477	66.67	31.75	30.40	18.35	30.20
Chrysoblephus laticeps (a)	377	34.1	27.17	19.33	17.09	19.52
Gymnocrotaphus curvidens (b)	357	40	19.36	19.11	15.19	19.20
Chirodactylus brachydactylus (c)	297	32.26	21.26	15.02	5.06	14.69
Chirodactlylus grandis (c)	78		35.48	4.04	1.27	3.95
Diplodus sargus capensis (b)	64		43.75	3.52	3.80	3.59
Cheilidactylus fasciatus (c)	60	43.48	42.86	3.29	0.63	3.19
Dichistius capensis (b)	37	5.26	106.25	2.92	1.90	2.91
Oplegnathus conwayi (b)	8		175	0.84		0.81
Pachymetopon aeneum (a)	9		80	0.62		0.59
Diplodus cervinus hottentotus (a)	1		300	0.04	3.16	0.33
Boopsoidea inornata (d)	1			0.04	2.53	0.18
Epinephelus marginatus (c)	4			0.15		0.15
Clinus superciliosus (c)					1.27	0.07
Chrysoblephus gibbiceps (a)	2			0.08		0.07
Cafrogobius caffer (c)					0.63	0.04
Cafrogobius saldanae (c)					0.63	0.04
Naso sp. (c)	1			0.04		0.04
Parascorpis typus (c)	1			0.04		0.04
Unidentified new species (c)	1			0.04		0.04
Rhabdosargus globiceps (c)	1			0.04		0.04
Seriola lalandi (c)	7	25		0.33		0.32
Number of fish	4 369 (38 comp)			2615	(all dives) 158	
Number of weighers (we)	2 079 (38 comp)				(all dives)103	
Number of extras (% of we num)	2 290 (38 comp)	52.41% (42 dives)	31.33%		(44 dives) 53.39	
Number of dives	656 (38 comp)				92	
Number of hours	2 700 (38 comp)				311	
Cpue (fish/100 hours)	161.81				50.80	
Mean dive time (hrs)	4.12				3.38	

* - extras are defined here as fish which were not presented at the scale, presented non-weighers not taken into account
& - to allow for the estimation of the maximum catch by divers
$ - based on results from the 33 competitions of column b
a - restricted species
b - recreational species
c - exploitable species
d - bait species

Competitive spearfishers landed 22 species, 13 of which were landed by shore-based divers (4 only by them), and 18 by boat-based participants (3 only by them). The catch of competitive spearfishers was dominated by reef-dependent species (over 98% of individuals, 94% of species), the five numerically most important being *P. blochii* (30.2%), *C. laticeps* (19.5%), *G. curvidens* (19.2%), *Chirodactylus brachydactylus* (14.7%), and *C. grandis* (4%).

During over 90% of non-competitive dives inspected, two species or less were landed (Table 4.5). However, during 76.2% of the boat-based competition dives, the diver had speared three or more species, this value dropping to 68% for inspected shore-based competitive dives. Similarly, Table 4.6 shows that during only 1.3% of the 1 042 non-competitive dives inspected had more than ten fish being speared. By contrast this had been achieved during 28.4% of the competition dives inspected.

The size distributions of species speared by boat-based and shore-based competitive divers are presented in Figures 4.4 & 4.5 respectively. Over 95% of fish shot during boat-based competitions were longer than 30 cm TL (Figure 4.4), whereas proportionally more fish smaller than 30 cm TL (18.2%) were speared during shore-based competitions (Figure 4.5). For the species for which size distributions are presented for both non-competitive (in Figure 4.2) and boat-based competitive divers (in Figure 4.4), these were only significantly different for *G. curvidens* and *P. blochii* ($p<0.05$). In both cases, boat-based competitive divers speared more large individuals.

Table 4.5 Number of species in the catch of non- and competitive spearfishers in False Bay (1992-1997), expressed as a % of dives during which that number of species were present in the catch.

	Non-competitive divers	Competitive divers	
Species number	Shore- and boat-based combined	Shore-based	Boat-based
0	48.46	-	2.38
1	29.27	20	4.76
2	13.63	12	16.67
3	5.47	20	16.67
4	1.82	12	21.43
5	0.77	20	23.81
6	0.29	12	4.76
7	0.19	4	4.76
8	0.10	-	4.76
Number of dives	1042	25	42

Table 4.6 Number of fish in the catch of non- and competitive spearfishers in False Bay (1992-1997), expressed as a % of dives during which that number of fish were present in the catch.

	Non-competitive divers	Competitive divers	
Fish number	Shore- and boat-based combined	Shore-based	Boat-based
0	48.5	-	2.4
1	19.6	-	2.4
2	13.8	20	4.8
3	8.9	8	4.8
4	3.6	8	7.1
5	2.0	12	11.9
6	0.9	12	4.8
7	0.6	12	7.1
8	0.2	12	2.4
9	0.4	12	11.9
10	0.2	0	4.8
>10	1.3	4	42.9
Number of dives	1042	25	42

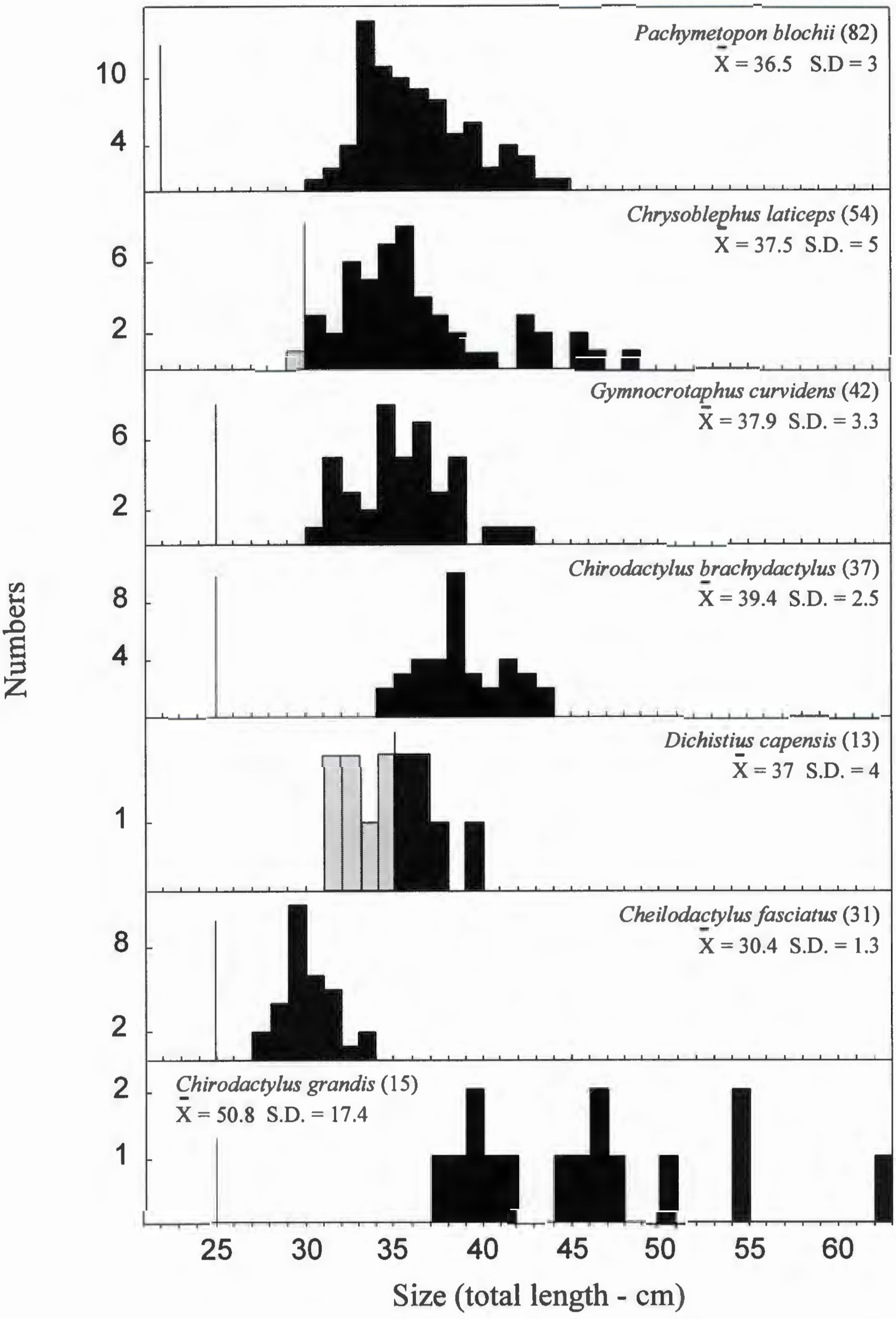


Figure 4.4 Size distribution plus mean size and standard deviation (total length - cm) of fish species shot within False Bay during provincial boat-based spearfishing competitions. Note, only species for which more than 14 measurements were obtained are shown. Line represents the enforced minimum size restriction for the species concerned. Bars to the left of it represent undersize individuals.

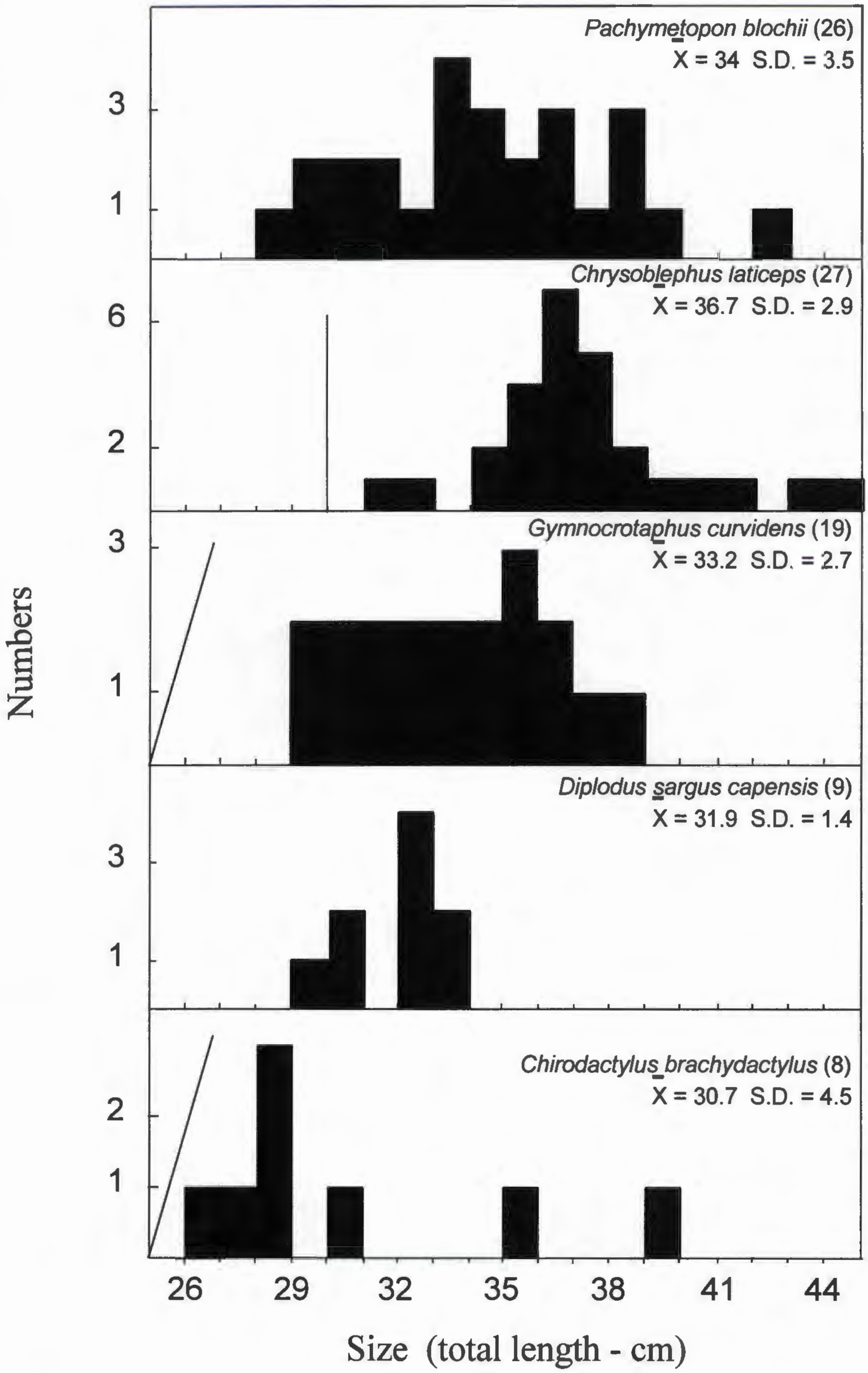


Figure 4.5 The size distribution plus mean size and standard deviation (total length cm) of species shot during shore-based spearfishing competitions within False Bay. Note, only species for which more than seven measurements were obtained are shown. Line represents the enforced minimum size restriction for the species concerned.

Discussion

Non-competitive spearfishery

Although Class 3 spearfishers comprised only 14.2% of inspected non-competitive divers in False Bay, this proportion is much higher than in KZN, where top spearfishers only make up approximately 1% of spearfisher numbers (van der Elst 1989). This may be partly linked to differences in sampling, although the warmer conditions and greater availability of large pelagic fish in KZN waters (van Rooyen 1988) are also likely to encourage additional inexperienced divers to partake in the sport.

The popularity of the eastern shore of False Bay to non-competitive spearfishers is underestimated in Figure 4.1. However, it is likely that the proportions of dives inspected within each area along that part of the coast show a true reflection of where shore dives are undertaken there, as very few access sites exist within Area 6, compared to within Areas 7 and 8.

Although the dive-time of Class 1 divers made up the largest proportion of the False Bay non-competitive spearfishery, Class 3 divers dominated boat-based spearfishing dive-time and contributed over 27% of the total dive-time of inspected dives (Tables 4.2 & 4.3). Considering that they contributed only 14.2% of non-competitive spearfishers encountered, this highlights the fact that Class 3 spearfishers dived more regularly, and for longer, than individual Class 1 and 2 divers. Class 3 spearfishers also landed almost 61% of the fish shot by non-competitive divers, further highlighting their importance. This is because they dive more regularly than other classes and are capable of finding the different species in "cold" and "warm" water, highlighting the fact that such knowledge was important factor in separating experienced (Class 3) from inexperienced (Class 1 and 2) spearfishers.

Shore-based non-competitive spearfishers had a lower *cpue* (60.4 fish/100hrs) than their boat-based counterpart (84.8 fish/100hrs). This is probably because boat-based spearfishing is less tiring and dangerous than shore-based spearfishing, along with the fact that boat-based spearfishers have access to more reefs per trip, including those inaccessible to shore divers. Water temperature also affected the *cpue* and catch composition of non-competitive spearfishers (Tables 4.2 & 4.3). This is because the spatial distribution and/or behaviour of a number of targeted reef fish species are different in "cold", compared to "warm" water. In cold conditions, species such as *C. laticeps*, *G. curvidens* and *Chirodactylus brachydactylus* retire into caves, while *P. blochii* congregate around offshore pinnacles, making them more difficult to find (Chapter 1). Overall, the *cpue* achieved during all inspected non-competitive dives was 64.93 fish/100hrs, which is considerably greater than that reported from KZN (33 fish/100hrs), or the Cape Province (45 fish/100hrs) by Mann *et al.* (1997). The *cpue* values reported by Mann *et al.* (1997) were obtained from relatively few outings, however.

The catch composition of non-competitive spearfishers included more species than that of competitive spearfishers (Tables 4.2, 4.3 & 4.4). In the case of some species (*e.g. Sparodon durbanensis*, *Umbrina canariensis*, *Petrus rupestris* and *Lithognathus lithognathus*), this is due to their low abundance on reefs and/or small size. In the case of all sharks and *Parascorpis typus*, this is due to them being banned from competitive catches.

Competitive spearfishery

A combination of reasons explains why the *cpue* of shore-based competition divers (50.8 fish/100hrs) was less than that of non-competitive shore-based divers (60.4 fish/100hrs). These include the fact that the number of "extras" speared during shore-based competitions was not estimated, the large number of inexperienced spearfishers (including numerous ladies) that participated, and the competition rules. The greatest estimate of the *cpue* of boat-based competitive spearfishers (161.8 fish/100hrs), however, was almost double that of non-competitive boat-based spearfishers (84.8 fish/100hrs). This is because competition rules encourage divers to spear as many species as possible and to ignore bag limits, although the experience of participating divers also plays a part.

Over 99% of the catch of competitive divers comprised reef-dependent fish. Because competitive divers shoot fish for points, they target the most abundant species available, which explains why the composition of their catch is different to that of non-competitive spearfishers. For example, *P. blochii* dominated their total catch numerically, rather than *C. laticeps*, while *D. capensis* was rarely speared during competitions.

Management

Over 93% of the catch landed by non-competitive spearfishers comprised reef dependent species, this increasing to over 99% for competitive divers. This is partly because reef-dependent species are more abundant than pelagic gamefish species within False Bay, although this pattern is still obvious within KZN (Mann *et al.* 1997), where gamefish are more abundant. Many of the reef fish species landed by the False Bay spearfishery are characterised by life history traits such as slow growth, residency, endemicity, and/or some form of sex change (van der Elst 1988). Furthermore, most are also targeted by other fishery sectors, which further complicates their management (Mann *et al.* 1997; Chapter 5).

As reported by Mann *et al.* (1997), the daily bag limits enforced onto recreational fishers do not restrict the catches of non-competitive spearfishers. The limit of five individuals from the 'restricted species list' was only transgressed once during the 1 042 inspected non-competitive dives, while the limit of 10 fish from the 'recreational species list' (but only five per species) was only contravened five times. Non-competitive spearfishers contravened no other daily bag limit regulation. Competitive spearfishers, however, regularly transgressed daily bag limits. Although during provincial competitions, contestants are legally permitted to spear two individuals from any number of species, they may not contravene other daily bag limits. This was, however, not made clear to contestants, nor enforced, however, as transgressions of the other daily bag limits were common. This led to the spearing of many 'extras' during competitions, a fact that has not been reported before.

The minimum size restrictions also did not limit the catches of spearfishers, as the size of fish speared both outside and during competitions was large compared to minimum legal sizes. The only species shot fairly regularly below its minimum legal size was *D. capensis*, although some undersize individuals of species for which the minimum legal size is 30 cm TL or greater were also speared. Inexperienced divers, who did not know the identity of fish and/or their minimum size, speared the majority of these undersize fish. The fact that the size distributions of species speared during competitions, during which different weight limits were set, were different, highlights the selectivity of the spearfishery as a whole.

Spearfishers are not always popular with line anglers within False Bay. This is because the *cpue* and quality of fish taken by spearfishers are often superior to those of anglers. For example, the *cpue* of all False Bay non-competitive spearfishers combined (64.9 fish/100hrs) is almost three times greater than the 23.3 fish/100hrs (retained fish only) for shore anglers in the same area (Lamberth & Griffiths 1997). The *cpue* of spearfishers participating in Western Province competitions within was even higher, at 161.8 fish/100hrs. Furthermore, as on the East coast of the country (van der Elst 1989; Mann *et al.* 1997), False Bay spearfishers land species that are not readily available to line anglers, although most speared species are also landed by other fishery sectors. Overall, however, the total catch of the False Bay spearfishery is much smaller than that of line anglers in the same area (Chapter 5), supporting Mann *et al* (1997), who states that there is little evidence of overfishing by the South African spearfishery. Effective management of all False Bay reef fishery sectors (including the spearfishery) is thus required to redress the overfishing of the False Bay reef fish resource.

Fuentes (1985) and Clark, Causey & Bohnsack (1989) report that spearfishing can impact linefish stocks negatively. Fuentes (1985) reports that Chilean spearfishers caused a decline in the abundance of sheephead (*Semicossyphus maculatus*), while Clark *et al.* (1989) report that the abundance of species targeted by spearfishers increased on a reef once spearfishing was banned. Although most False Bay spearfishers believe that their impact is small, some believe that spearfishing can have a detrimental affect on some species. This is supported by the alarming decline in number and size of *Parascorpis typus* (a species available only to spearfishers) speared during the last 25 years (Lechanteur unpublished data).

It is proposed that more restrictive daily bag limits should be enforced on non-competitive and competitive spearfishers, and that competition organizers and enforcement agencies should enforce these more efficiently (Mann *et al.* 1997). Proposed daily bag limits for competitive divers should include a predetermined maximum number of fish shot by individual divers, and the presentation of only one individual per species at the weigh-in. This predetermined maximum number of fish would depend on the number of species available at a site, and the weight limit for each. Limiting contestants to presenting one individual per species at the weigh-in is successful in limiting the numbers of 'extras' shot by competitive spearfishers in South and Southeastern Australia (Johnson 1985a, b; Lincoln Smith, Bell, Pollard & Russel 1989).

Although illegal in South Africa (van Rooyen 1988; Cook 1990; Mann *et al.* 1997), the selling of speared fish by individual spearfishers is common. Numerous experienced False Bay spearfishers sell speared teleosts to cover trip expenses and/or equipment repairs/acquisitions (Mann *et al.* 1997). The size of this 'commercial catch' is not known, but is likely to be in excess of 50% of the fish measured during this study (competitive and non-competitive catch combined). The sale of speared sharks is a more recent venture practised by fewer divers. They target demersal species, and their annual catch is likely to be large as four divers shot over 700 kg of sharks during only eight dives. This should not be encouraged, as legalising the sale of speared sharks would encourage divers to also sell teleost species. To limit the sale of speared fishes, the following propositions are forwarded:

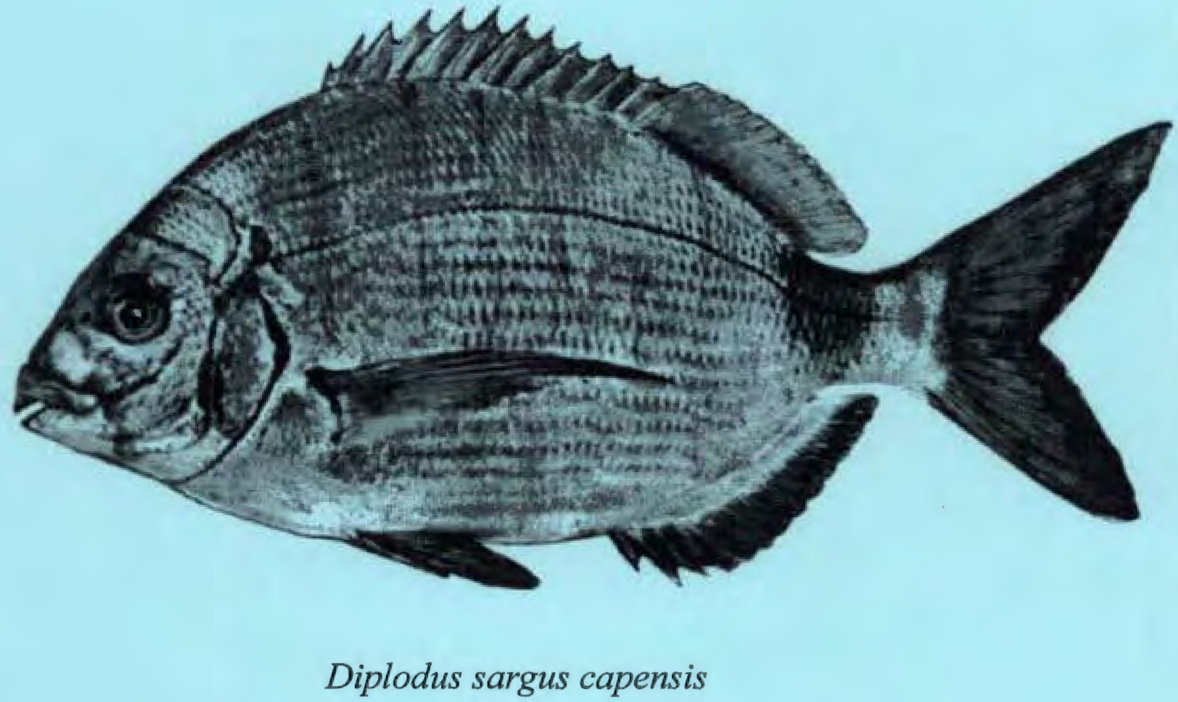
- The inspection rate of divers' catches should be increased substantially.
- Fish shops and seafood restaurants in the area should also be inspected, as these buy large quantities of speared fishes (pers. obs.).

It is hypothesised that increasing the inspection of commercial premises selling fishes would decrease the sale of speared fishes, as should an example be made of any found guilty, these and others would refrain from doing so, thereby restricting spearfisher's markets and ultimately their catch intended for sale.

The spearfishery is one of five linefishery sectors active within False Bay. The overall impact of this fishery sector on False Bay linefish stocks will be compared to that of the other sectors in the next chapter.

Chapter 5

Linefish under siege: Catch composition of False Bay linefishery sectors 1992 - 1995. Is fishery management adequate?



Diplodus sargus capensis

Introduction

Linefisheries have existed within False Bay since early European settlements were established, and since then, linefishing effort has increased, fishers being encouraged by good linefish catches, the proximity of False Bay to Cape Town and the availability of sheltered launching sites along its coastline. Presently, five linefishery sectors are active within the Bay, each targeting linefish by different techniques. This study sets out to compare catch composition and effort information available on each False Bay linefishery sectors to:

I. Estimate the total annual linefish landings taken from within False Bay.

II. Highlight which linefish species are most heavily exploited by the different sectors and overall.

III. Note the species over which there is competition among the different sectors.

Historical changes in catch compositions and fishery management regulations are also discussed. In view of the fact that linefishery management is to be based on the Linefish Management Protocol (LMP) of Griffiths, Attwood & Thomson (1999), changes in regulations are likely to be passed within the next few months. With this in mind, changes in present fishery regulations, based on the results of this study, are recommended when thought necessary.

The five False Bay linefishery sectors can be categorised as either commercial or recreational fisheries, with the latter being prohibited from selling their catch. Recreational fishers participate in a fishery mainly to relax and escape from daily stresses (Claverie 1990; Iverson 1996; Lamberth & Griffiths 1997) and usually target trophy fishes (Butterworth, Punt, Borchers, Pugh & Hughes 1989; Clarke & Buxton 1989; Garratt & van der Elst 1990; Iverson

1996), especially edible ones. Commercial fishers fish for a living (Iverson 1996) and are more interested in large volume catches than catches composed of a few trophy fishes of smaller overall mass (Butterworth *et al.* 1989; Hecht 1990). The total effort and catch of commercial and recreational fishers are controlled differently. The total effort of recreational sectors is not restricted, and their total cumulative catch is only restricted by daily bag limits enforced on each participating fisher. The effort of the commercial sectors is limited by restricting entry into the fisheries, a form of 'limited access' management (Iverson 1996). The daily catch limits in this sector are more lenient than those enforced on the recreational sectors (Table 5.1). The history, fishing method, and regulations controlling each of the False Bay linefishery sectors between 1992 and 1997 are described below.

Recreational shore angling

Shore angling, because it is the most accessible angling activity in terms of participation, dominates the South African linefishery (Guastella & Nellmapius 1993). Recreational shore angling is an open fishery sector, with the number of participants increasing (Clarke & Buxton 1989) at a rate of 6% per annum (calculated for the whole country - Bennett 1991b). Four categories of shore anglers are active in the False Bay region, including I. Sports (competition) anglers, who because they target mainly cartilaginous species during competitions have a negligible impact on edible species. Taylor (1993a) estimated that some 5 000 such anglers were active along the Cape coast during 1993. Because they represent top anglers, their impact on targeted species is likely to be high (Taylor 1993a). II. Recreational club (competition) anglers, who target only edible species (Bennett *et al.* 1994). Taylor (1993a), in contrast with Bennett *et al.* (1994), reports that their impact on edible species is likely to be high, as they represent more experienced anglers. The total number of recreational club anglers along the Cape coast was estimated to be 5 000 in 1993 (Taylor 1993a). III. Non-club anglers, who constitute the largest group of shore anglers (Bennett *et*

Table 5.1 The daily bag limits enforced on the different angling sectors (including spearfishers) within False Bay, based on SFRI (1996). Note: only species which are common within the Southwestern Cape are included.

	Critical list	Restricted list	Exploitable list	Recreational list	Bait list
	Carcharodon carcharias * *Cymatoceps nasutus* $ *Petrus rupestris* #	*Chrysoblephus gibbiceps* *Chrysoblephus laticeps* *Diplodus cervinus* *Pachymetopon aeneum* *Pomatomus saltatrix* Rock cods	*Argyrosomus spp.* *Argyrozona argyrozona* *Atractoscion aequidens* *Cheimerius nufar* *Chelidonichthys capensis* Elasmobranchs (except *C. carcharias*) *Merluccius spp.* *Pachymetopon blochii* *Pagellus bellottii natalensis* *Pterogymnus laniarius* *Rhabdosargus globiceps* *Seriola lalandi* *Thyrsites atun* Tunas All other species not listed under the other lists in SFRI 1996	*Dichistius capensis* *Diplodus sargus* *Gymnocrotaphus curvidens* *Lithognathus lithognathus* *Oplegnathus conwayi* *Rhabdosargus holubi* *Sparodon durbanensis* *Umbrina spp.*	Anchovies *Boopsoidea inornata* Mullets Sardines *Sarpa salpa* Scads *Scomber japonicus* *Spondyliosoma emarginatum* *Trachurus trachurus*
Recreational fishers	* None # and $ 2 per person per day	5 per person per day in total	10 per person per day in total	10 per person per day in total, but only 5 of the same species	Unlimited
Commercial anglers					
B-licence holders	* None # and $ 2 per person per day	5 per person per day in total	Unlimited	10 per person per day in total, but only 5 of the same species	Unlimited
A-licence holders	* None $ 2 per person per day # 5 per person per day	Unlimited for all species in the Cape	Unlimited	10 per person per day in total, but only 5 of the same species	Unlimited

al. 1994) in the Southwestern Cape. They account for the largest portion of the resource use amongst shore anglers (Bennett *et al.* 1994), even though their catch-per-unit-effort (*cpue*) is less than that of the two other shore angler groups (Taylor 1993a). IV. Subsistence anglers, who rely on fishing as an important source of food. Although difficult to quantify (Lamberth, Mann, Brouwer & Sauer 1996), their total number of participants is estimated to total 3% of the rock and surf anglers in the country (Brouwer, Mann, Lamberth, Sauer & Erasmus 1997), with their numbers likely to increase in the coming years (Hutchings 1993). Minimum sizes and daily bag limits (Tables 5.1 and 5.2) are employed to limit the catches of shore anglers (Attwood & Bennett 1995b). In addition, angling is prohibited within some marine protected areas, and certain species are subject to closed seasons (Living Marine Resources Act 1998).

Spearfishing

The spearfishery is one sector of the South African linefishery that has not been studied intensively. It involves a free diver shooting fish underwater (van Rooyen 1988; Cook 1990). Both non-competitive, where spearfishers shoot fishes as a hobby, and competitive spearfishing, where individual divers (or teams thereof) compete in accordance to additional spearfishing competition rules, is practised within False Bay (Chapter 4). Being a recreational fishery sector (van Rooyen 1988; Cook 1990; Mann *et al* 1997), the spearfishery has the same daily bag limits (Table 5.1) and minimum size restrictions (Table 5.2) enforced on it as other recreational fishery sectors. Numerous other regulations or self-imposed restrictions regulate the spearfishery (Chapter 4), making the spearfishery the most regulated sector of the South African linefishery (Mann *et al.* 1997).

Table 5.2 The minimum size restrictions enforced on the different linefishing sectors active within False bay, based on (SFRI 1996). Note: only species found in the Southwestern Cape are included.

Size (total length)	Species	Size (kg)	Species
15 cm -	*Sarpa salpa*		
20 cm -	*Diplodus sargus* *Rhabdosargus holubi*	3.2 kg -	*Thunnus albacares* *Thunnus obesus*
22 cm -	*Pachymetopon blochii*		
25 cm -	*Argyrozona argyrozona* *Rhabdosargus globiceps*	6.4 kg -	*Thunnus maccoyii* *Thunnus thynnus*
30 cm -	*Chrysoblephus gibbiceps* *Chrysoblephus laticeps* *Diplodus cervinus* *Pomatomus saltatrix*	25 kg -	*Xiphias gladius*
35 cm -	*Dichistius capensis*		
40 cm -	*Argyrosomus spp.* *Epinephelus marginatus* *Petrus rupestris*		
50 cm -	*Cymatoceps nasutus*		
60 cm -	*Atractoscion aequidens* *Lithognathus lithognathus* *Sparodon durbanensis* *Thyrsites atun*		

Boat angling

Boat anglers fish from boats using either a fishing rod or handline (Pulfrich & Griffiths 1988b; Schoeman & Schoeman 1990). This angling sector is divided into two categories:

1. Recreational boat angling. Information concerning the South African recreational boat-based linefishery is very limited, with the study of Sauer*et al.* (1997) being the first to analyse

this sector. Lamberth *et al.* (1996) estimated that 80 000 recreational boat anglers were active in South Africa. They are numerous in the Southwestern Cape, with their numbers increasing (Penney 1991; Sauer *et al.* 1997). Being recreational fishers, their total number of participants is not restricted (Winch 1990a; Lamberth *et al.* 1996), but their catch cannot be sold. The catch of this sector is regulated by the same regulations enforced on other recreational fishers (Tables 5.1 & 5.2). Most recreational boat angling is undertaken off ski-boats within False Bay (Sauer *et al.* 1997). Because they are not legally required to hand in catch returns, recreational boat anglers' catches are not well documented. This sector's total linefish catch was estimated at 12 780 kg for the Southwestern Cape in 1995 (Sauer *et al.* 1997), a small amount compared to that taken by the commercial sector in the same area. This sector's catches have declined over the last few years in the Southwestern Cape, with overfishing one of the major reasons (Sauer *et al.* 1997).

2. Commercial boat anglers. The catch of commercial boat anglers is limited by restricting entry into the fishery (Sauer *et al.* 1997). The advent of improved navigational (*e.g.* GPS) and fish finding equipment (*e.g.* echo-sounder) has provided False Bay commercial boat anglers with improved fishing possibilities (Garratt & van der Elst 1990), even though handlines are still employed to catch linefish (Pulfrich & Griffiths 1988b). Commercial boat angling is practised off either deck-boats (>20 m long) that hold crews of more than five, or ski-boats (<10 m long) that hold smaller crews. Deck-boats are based at the commercial fishing harbours of Kalk Bay, Gordon's Bay and Simonstown (Figure 5.1), have a limited range and are restricted to angling within False Bay and immediate surrounding areas. Ski-boats can be launched from any of the slipways distributed along the False Bay coast (Figure 5.1), and are thus capable of following large concentrations of fish (Penney 1991). Commercial boat anglers are further divided into either A-licence holders, who rely on

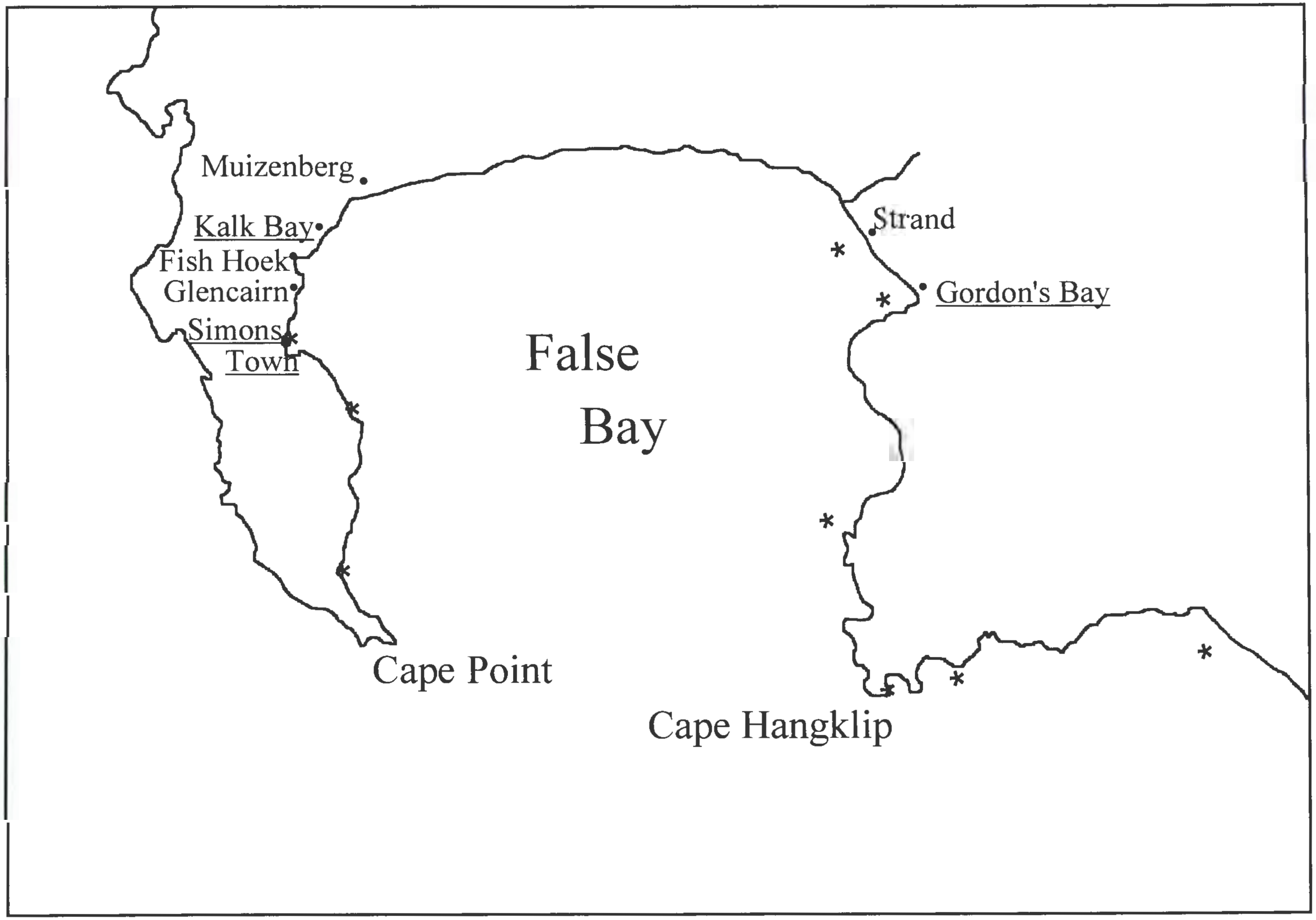


Figure 5.1 The location of slipways (*) and commercial fishing harbours (underlined town names) along the False Bay coastline.

angling for more than 50% of their income, or B-licence holders, who rely on angling for less than 50% of their income. Many of the latter are in reality recreational anglers who have purchased the B-licence so that they can sell their catch (Lamberth & Griffiths 1997). B-licence holders only fish when profitable species (*e.g. Atractoscion aequidens*, *Thyrsites atun*, *Argyrosomus inodorus*, *Seriola lalandi*) abound in the Bay (Penney 1991). A-licence holders also target these species when available, but being more reliant on their catches for their livelihood, also target less profitable species when profitable ones are unavailable. Pulfrich & Griffiths (1988b) estimated that the effort expended by commercial boats targeting demersal species was 454 210 man-hours in 1985. The total number of commercial boats active within False Bay was approximately 700 (83 A-licences) during the 1987-1988 season (Penney 1991), with many boats moving into False Bay from elsewhere when fishing was good. Commercial boat anglers are thus numerous and increasing in numbers (Penney 1991; Sauer *et al.* 1997) in False Bay, and their effort is substantial. South African commercial boat anglers are subject to the same minimum size restrictions as recreational fishers (Table 5.2), although their daily bag limits are more lenient (Table 5.1). Moreover, the daily bag limits are different for the two commercial boat angling sectors, with those enforced on A-licence holders being less restrictive than those enforced on B-licence holders (Table 5.1). The overall catch of B-licence holders is substantial, and was greater than that of A-licence holders for the time period 1978-1988 (Penney 1993). Indeed, that study reports that the B-licence holders' total catch exceeded that of the A-licence holders for all species except *Pachymetopon blochii* and *Rhabdosargus globiceps* during that time period. The total catch of False Bay commercial boat anglers is likely to be substantial, as the total linefish catch landed by this sector in the Southwestern Cape was 14 054 634 kg in 1996 (Sauer, *et al.* 1997). Commercial boat anglers are prohibited from selling species listed in the 'recreational list' (Table 5.1), catching certain species during their closed seasons and angling within some marine protected areas (SFRI 1996).

Commercial beach-seining

The False Bay beach-seine fishery is the oldest commercial fishery in the country (Lamberth *et al* 1994), having been active for more than 300 years (Lamberth & Bennett 1993). This fishery can only operate from beaches, which restricts beach-seiners to catching species that are encountered over sandy substrata (Lamberth *et al.* 1995a). The effort and total catch of beach-seiners is restricted by a licensing system that controls the number of entrants into the fishery (Lamberth *et al.* 1994). The method used by beach-seiners is simple: a shoal of fish (or a promising area) is located visually and a net set around it from a rowing boat launched from the shore. Once set, the net is pulled in (Lamberth *et al.* 1994). There were 14 beach-seining operators (29 nets) in False Bay in 1988 (Penney 1991), but by 1991 this had been halved (Lamberth *et al.* 1994). Lamberth *et al.* (1994) estimated that from January 1991 to December 1992, approximately 1 000 hauls were made by beach-seiners within False Bay. Management regulations enforced on this fishery sector include a restricted season (1 November - 30 April) during which 'angling species' may be retained (Penney 1991; Lamberth & Bennett 1993), and a site restriction prohibiting beach-seining within 500 m of estuaries (Clark, Bennett & Lamberth 1994a, b). The minimum size restrictions and closed seasons enforced on beach-seiners are the same as those that apply to all fishery sectors.

Methods

Because False Bay was only defined as the fishing area located between two points on the coast, fishing grounds located far offshore between these two sites still fit within False Bay, by definition. Species landed predominantly from these grounds are thus listed below as having been landed within False Bay, although most would have only been encountered many miles offshore of the coast. Information concerning the catches of the different False Bay linefishing sectors was obtained from a number of different sources. These are discussed below.

Recreational shore angling

Because no information concerning the catches of recreational shore anglers specifically from False Bay is available, their catch composition and *cpue* could only be estimated from Lamberth & Griffiths (1997), who investigated these anglers during the same time span within the Southwestern Cape. It was thus assumed that the angler effort of 5.47 anglers/km/day recorded by Lamberth & Griffiths could be applied to the 147 km shoreline of False Bay, as could the *cpue* of 23.25 fish/100hrs (retained catch). The composition of the estimated False Bay catch was also assumed to be the same as that reported by Lamberth & Griffiths (1997) for shore anglers in the Southwestern Cape. The data obtained from Lamberth & Griffiths (1997) takes into account fishes landed during angling competitions.

Spearfishing

Information concerning the False Bay spearfishery was obtained from Chapter 4, where the catch composition and *cpue* of both non-competitive and competitive spearfishers was calculated for the period 1992-1997. The total catch of the False Bay non-competitive spearfishery was estimated by assuming that I. A total of 520 spearfishers are active within False Bay (roughly twice the number of divers encountered on one coast within False Bay – Chapter 4). II. Of these divers, 10% (52) completed 35 shore dives annually within the Bay, while 20% (104) undertook 20, 20% (104) undertook 10, and the remaining 50% (260) undertook five. III. Thirty divers completed 15 boat dives, while 15 undertook five, and a further 20 undertook two. The above assumptions were based on average dive frequency reported by divers interviewed while collecting data for Chapter 4. Thus, the total number of non-competitive dives undertaken in False Bay was estimated at 6 805 per year (6 240 shore dives). The catch-per-dive of boat- and shore-based non-competitive spearfishers (irrespective of diver class or water temperatures) obtained in Chapter 4 was then multiplied by the estimated number of dives undertaken by each group to obtain estimates of the number of fishes shot annually. These estimate were then broken down into fish numbers per species for each group by multiplying each species' percentage contribution for each group (from Chapter 4) by the estimated number of fishes shot by each..

An estimate of the total number of fishes speared by shore-based competition divers in False Bay was obtained by multiplying the catch-per-dive estimate of Chapter 4 by an estimate of the participation of shore-based competitions. This estimate was obtained by assuming that only one shore-based competition was organised each year in False Bay, and 38 participants (mean number of participants during the 1992 and 1996 "Allrounders Competition" – Chapter 4) took part. The total number of fishes speared was subdivided into component species using the proportions of each species reported in Chapter 4. For boat-based competition divers, catch information was obtained from the Western Province competition results presented in Chapter 4. Estimated annual catch (all species combined) was calculated by assuming that I. The total number of fishes shot during Western Province competitions were the average of the total number of fishes shot per competition between 1992 and 1996, multiplied by the average number of competitions organised by Western Province each year (7.6). II. Boland divers undertook two competitions within False Bay each year (the same amount as was organised during one season during which the senior author participated in competitions organised by the Boland Underwater Union). III. The catches landed during the latter competitions were the same as those landed during Western Province competitions, except for the small species (400 g weight limits) speared by Western Province divers. It was assumed that none of these were speared by Boland divers, as their smallest minimum weight limit is one kilogram (Chapter 4). The estimated total catch of the boat-based competitive spearfishing sector was broken down to species using the proportion that each species made up in the Western Province catches reported in Chapter 4. The total estimated catch of the False Bay spearfishery was obtained by adding the total catch of the non-competitive and competitive sectors together.

Boat angling

1. Recreational boat angling. Since recreational boat anglers are not required to provide catch returns, and insufficient voluntary catch returns were available from the National Marine Linefish System (NMLS), the total catch of this sector had to be estimated. The following was assumed to achieve this: I. The ratio of recreational to commercial launches in False Bay is the same as that reported by Sauer *et al.* (1997) for the whole of the Southwestern Cape (1:81.8). Based on information obtained from the NMLS for the period 1992-1995, 15 248 commercial boat outings (52.2% by B-licence holders) were undertaken in False Bay. II. The *cpue* of this sector is 9 kg per boat day, the same as that of recreational boat anglers in the Southwestern Cape (Sauer *et al.* 1997). III. The catch composition (by mass) of this sector is the same as that of the B-licence commercial boat anglers active in False Bay. This was based on Lamberth & Griffiths (1997), Penney (1991) and Sauer *et al.* (1997) who propose that many False Bay 'recreational' boat anglers actually own a B-licence, and only go fishing when good quality fishes are available. IV. The size distribution of each

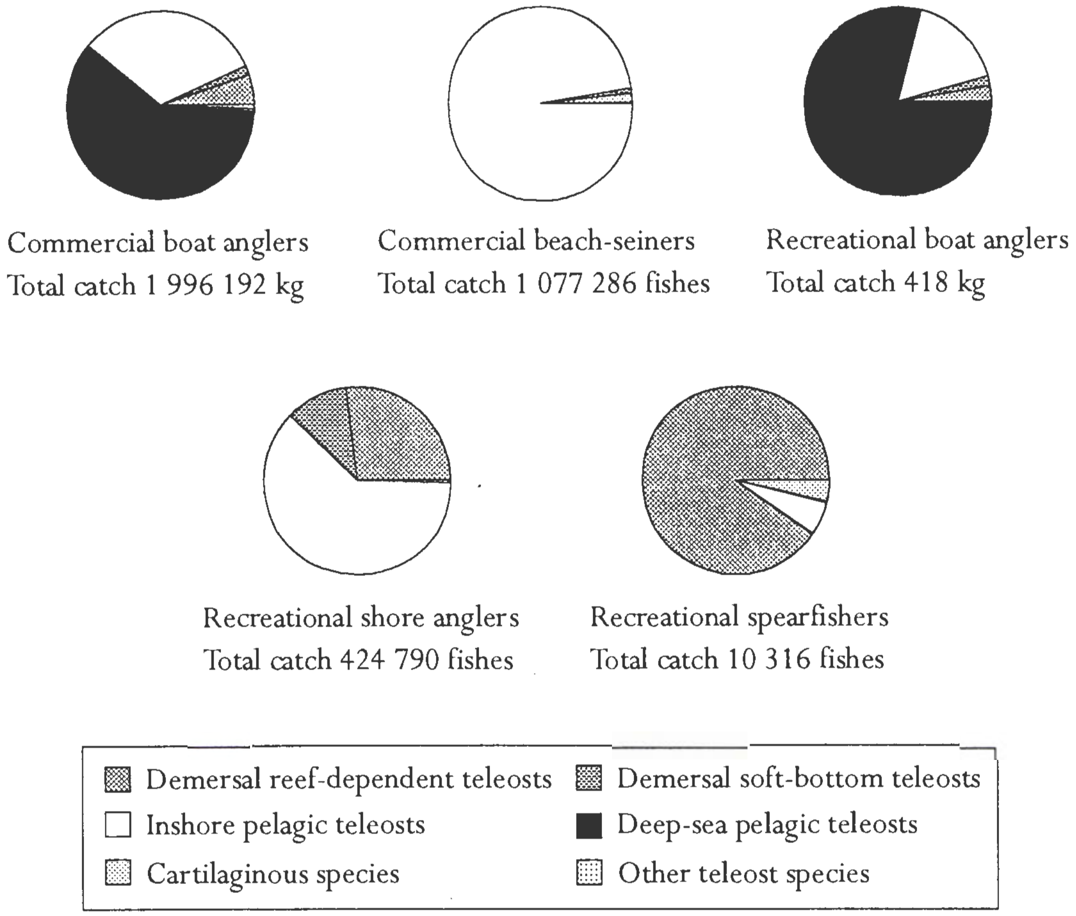
species caught by this sector was the same as that measured for the commercial boat angling sector (A- and B-licence combined). The catch (by mass) was calculated for each species, and converted into fish numbers for some either by sub-division into the size distributions obtained for commercial boat anglers, or by assuming that certain proportions of the landed mass were made up of fishes of a particular size. Fish mass assumptions were the same as those listed below for commercial boat anglers' catches.

2. Commercial boat angling. Effort, total reported catch, and catch composition of this sector was obtained from the catch returns which commercial skippers are required, by law, to report. These catch returns were obtained from the NMLS from four years worth of catch returns (January 1992 - December 1995) for both A- and B-licence holders separately. The catch returns consisted of the total mass (kg) of each species (or group of species). This was supplemented with a breakdown of the size distribution of a sub-sample of the total catch (A- and B-licence combined) of some species caught during the same time period (also obtained from the NMLS). These reported catch returns, however, are known to be inaccurate as commercial boat anglers under- and over-report their catches for reasons ranging from concerns with tax returns to wilful breaking of regulations (Sauer *et al.* 1997). The under-reporting is very severe in the case of *Seriola lalandi* (3.5 times) and *Thyrsites atun* (2.8 times) (Lamberth & Griffiths 1997). Catch returns for these species were thus corrected using the published correction values. Although it is likely that the catches of many other species are also under-reported, no other correction values were employed since the magnitude of under reporting is unknown. The obtained size distributions of a sub-sample of some species' catches were used to convert the mass data to an estimate of total fish number, assuming that fishes caught by A- or B-licence holders had the same size distribution. However, for those species for which no size distributions were obtained, the following

assumptions were employed to obtain an estimate of how many individuals of each of were landed: I. Sixty percent of the landed mass of *Seriola lalandi* comprised 3.5 kg individuals, the remainder weighing 5.5 kg. II. Twenty percent of the landed mass of *Thyrsites atun* comprised 2.5 kg individuals, 20% 5 kg individuals, while the remainder weighed 3.5 kg. III. Sixty percent of the landed mass of *Argyrosomus inodorus* comprised 2.5 kg individuals, while the remainder weighed 5 kg. IV. The entire landed mass of *Pomatomus sultatrix* consisted of 0.65 kg individuals, *Scomber japonicus* 0.33 kg individuals, *Atractoscion aequidens* 3 kg individuals, *Merluccius capensis* 2 kg individuals, and *Sarpa salpa* 0.075 kg individuals. These assumptions are based on observed commercial boat anglers' catches, or on an average size estimate for the species concerned if no catches had been observed. The estimated catch of both commercial boat angling sectors were added to obtain an overall estimate of the annual commercial boat anglers' linefish catch within the Bay.

Commercial beach-seining

Although beach-seiners have to report their catches, these reported catches are not accurate as numerous species are under-reported in catch returns (Lamberth *et al.* 1994). Thus, information from the study by Lamberth *et al.* (1994) was used to estimate the catch composition of the False Bay beach-seine fishery. This study reported the catch composition (by numbers) of 311 beach-seine hauls made within False Bay from January 1991 - December 1992. Only data concerning the numbers of retained fishes were employed for the purpose of this study. The total annual catch of the False Bay beach-seine fishery was calculated by multiplying the catch-per-haul estimate by 500, the estimated number of hauls made annually in False Bay (Lamberth *et al.* 1994).



Commercial boat anglers
Total catch 1 996 192 kg

Commercial beach-seiners
Total catch 1 077 286 fishes

Recreational boat anglers
Total catch 418 kg

Recreational shore anglers
Total catch 424 790 fishes

Recreational spearfishers
Total catch 10 316 fishes

Figure 5.1. The composition (by numbers or mass) of the annual retained catches of the five False Bay linefishing sectors.

Results

The total estimated annual catches (by numbers and/or mass) of the five False Bay linefishery sectors are shown in Table 5.3. Altogether, 64 teleost and 12 cartilaginous species were caught and retained by the different linefishery sectors in False Bay. Added to these totals would fall 'unspecified' species recorded as either 'unspecified rays', 'unspecified sharks', 'unspecified red', 'unspecified fish' and 'tuna' by commercial boat anglers. It is likely, however, that many of the species in the 'unspecified' lists are already listed in Table 5.3. Figure 5.2 shows the total catch and composition (by numbers or mass for the major linefish classes) of the five linefishing sectors within False Bay. The estimated effort of each, along with their total catch and its composition are discussed below.

Shore angling

The annual effort expended by shore anglers in False Bay is high, averaging 293 493 angler days (1 809 874 angler hours). Their total annual catch was 420 796 fishes, 61.9% of which were inshore pelagic teleosts, and 26.9% demersal reef dependent teleosts (Figure 5.2). Table 5.3 shows their catch to comprise 30 species. Fourteen of these were demersal reef-dependent teleosts, of which the most important (% total catch by numbers) were *Dichistius capensis* (12.6%), *Diplodus sargus capensis* (5.2%), *Sarpa salpa* (1.9%), *Diplodus cervinus hottentotus* (0.8%), *Pachymetopon blochii* (0.7%), sea catfish (0.4%), and *Clinus superciliosus* (0.4%). Two of the other important species (*Pomadasys olivaceum*, *Sparodon durbanensis*) are rare within False Bay (van der Elst 1988; Chapters 1, 6 & 7). Six demersal soft-bottom teleosts were caught by shore anglers, totalling over 45 000 individuals a year. Of these, *Argyrosomus inodorus* (5.1%), *Lithognathus lithognathus* (4.1%) and *Chelidonichthyes sp* (0.9%) were the most important (by numbers). Another species that was

Table 5.3 The annual catch composition (by numbers and/or mass) of the different linefishery sectors active within False Bay during particular years.

	Boat anglers (92-95)			ALL BOAT ANGLERS	ALL BOAT ANGLERS	Shore anglers	Spearfishers (92-97)		ALL SPEARFISHERS	Beach-seiners	OVERALL TOTAL
Species Scientific name	A-licence (kg/yr)	B-licence (kg/yr)	Recreational (kg/year #)	TOTAL CATCH **(kg/yr)**	TOTAL CATCH **(Number/yr)**	Retained fishes (Number/yr)	Competitive (Number/yr *)	Non-competitiv (Number/yr &)	TOTAL CATCH **(Number/yr)**	Retained fishes (Number/yr ^)	RETAINED FISHES **(Number/yr)**
Teleost species											
Reef-dependent species											
Argyrozona argyrozona	772	606	-	1378	5437	-	-	-	-	-	5437
Boopsoidea inornata	43	-	-	43	-	970	2		2	-	972
Cheilodactylus fasciatus	-	-	-	-	-	-	30	174	204	-	204
Cheimerius nufar	1	-	-	1	-	-	-	-	-	-	
Chirodactylus brachydactylus	-	-	-	-	-	-	171	710	881	-	881
Chirodactylus grandis	-	-	-	-	-	-	45	283	328	-	328
Chrysoblephus cristiceps	-	5	-	5	-	-	-	-	-	-	
Chrysoblephus gibbiceps	35	53	-	88	48	-	-	25	25	-	73
Chrysoblephus laticeps	2491	923	-	3414	6736	-	230	2881	3111	-	9847
Clinus superciliosus	-	-	-	-	-	1616	1	-	1	-	1617
Dichistius capensis	7	8	-	15	16	53664	34	740	774	-	54454
Diplodus sargus capensis	28	46	-	74	154	21983	34	157	191	2244	24572
Diplodus cervinus hottentotus	9	-	-	9	9	3556	3	112	115	-	3680
Epinephelus marginatus	-	-	-	-	-	-	1	-	1	-	1
Galeichthys spp.	-	-	-	-	-	1616	-	-	-	55	1671
Genypterus capensis	40	8	-	48	-	-	-	-	-	-	
Gymnocrotaphus curvidens	-	-	-	-	-	323	225	931	1156	-	1479
Merluccius spp.	1152	6985	2	8139	-	-	-	-	-	-	
Oplegnathus conwayi	-	-	-	-	-	-	9	41	50	-	50
Pachymetopon aeneum	26	92	-	118	214	-	6	83	89	-	303
Pachymetopon blochii	72314	3243	7	75557	192245	2910	353	1999	2352	-	199686
Parascorpis typus	-	-	-	-	-	-	-	25	25	-	25
Petrus rupestris	6	54	-	60	56	-	-	8	8	-	64
Pomadasys olivaceum	3	-	-	3	-	8082	-	-	-	-	8082
Popyprion americanus	-	113	-	113	-	-	-	-	-	-	
Pterogymnus laniarius	6146	1501	1	7648	26547	-	-	-	-	-	26547
Sarpa salpa	8	17	-	25	333	8082	-	-	-	13079	21494
Sebastes capensis	19	25	-	44	-	323		-	-	-	323
Sparodon durbanensis	-	-	-	-	-	3556	-	-	-	-	3556
Spicara axillaris	20	-	-	20	-	-	-	-	-	-	
Spondyliosoma emarginatum	2031	253	-	2284	12755	970	-	-	-	161	13725
Umbrina canariensis	-	-	-	-	-	6466	-	-	-	1838	8304
Unspecified red	408	29.00	-	437	-	-	-	-	-	-	
Total for reef-dependent species	85556	13959	10	99523	244550	114118	1144	8169	9313	17376	387375
Soft-bottom species							-				
Argyrosomus inodorus	4146	9101	3	13250	4240	21660	-	-	-	3640	29540
Atractoscion aequidens	2203	8499	3	10705	-	-	-	-	-	-	
Austroglossus microlepis	-	-	-	-	-	-	-	-	-	2	2
Chelidonichthys capensis	-	2	-	2	-	3879	-	-	-	48	3927
Lithognathus lithognathus	19	18	-	37	7	17457	-	8	8	4450	21922
Lithognathus mormyrus	-	-	-	-	-	647	-	-	-	2	649
Pomadasys commersonnii	-	-	-	-	-	1616	-	-	-	63	1679
Rhabdosargus globiceps	5657	2061	1	7719	19709	323	1	4	5	1042	21079
Rhabdosargus holubi	-	-	-	-	-	-	-	-	-	6	6
Solea fulvomarginata	-	-	-	-	-	-	-	-	-	2	2
Total for soft-bottom species	12025	19681	7	31713	23956	45582	1	12	13	9254	78806
Inshore pelagic species											
Elops machnata	-	-	-	-	-	-	-	-	-	5	5
Liza richardsonii	-	-	-	-	-	7435	-	-	-	996413	1003848
Mugil cephalus	-	-	-	-	-	1616	-	-	-	42	1658
Pomatomus saltatrix	2548	1372	-	3920	6021	247956	-	-	-	10984	264961
Sardinops sagax	-	-	-	-	-	-	-	-	-	11908	11908
Scomber japonicus	9032	2750	1	11783	35706		-	-	-	2	35708
Seriola lalandi	56217	50433	17	106667	26042	-	4	574	578	12285	38905
Stromateus fiatola	-	-	-	-	-	-	-	-	-	320	320
Thyrsites atun	345828	148681	51	494560	203476	-	-	-	-	-	203476
Trachurus sp.	-	-	-	-	-		-	-	-	17592	17592
Trachurus trachurus	574	286	-	860	-	5819	-	8	8	-	5827
Total for inshore pelagic species	414199	203522	69	617790		262826	4	582	586	1049550	1584208
Deep sea pelagic species											
Acanthocybium solandri	2	9	-	11	-	-	-	-	-	-	
Brama sp.	333	223	-	555	-	-	-	-	-	-	
Coryphaena hippurus	48	123	-	171	-	-	-	-	-	-	
Katsuwonus pelamis	243	687	-	930	-	-	-	-	-	-	
Makaira indica	-	18	-	18	-	-	-	-	-	-	
Sarda sarda	-	14	-	14	-	-	-	25	25	-	25
Tetraplurus audax	-	15	-	15	-	-	-	-	-	-	
Thunnus alalunga	135817	723133	248	859198	-	-	-	-	-	-	
Thunnus albacares	2660	26353	9	29022	-	-	-	-	-	-	
Thunnus obesus	94	1921	-	2015	-	-	-	-	-	-	
Thunnus spp	49673	214571	74	264318	-	-	-	-	-	-	
Xiphias gladius	50	161	-	211	-	-	-	-	-	-	
Total for deep sea pelagic species	188918	967226	330	1156478		-	-	25	25	-	25
	-	8391	-	8391	-	-	-	-	-	-	
Unspecified fish	236	123	-	359	-	-	-	-	-	-	
Total Teleost catch	700934	1212902	416	1914252		422526	1149	8788	9937	1076180	2050414
Cartilaginous species											
Sharks											
Callorhinchus capensis	-	-	-	-	-	323	-	-	-	1039	1362
Carcharhinus brachyurus	-	-	-	-	-	647	-	8	8	-	655
Galeorhinus galeus	1471	197	-	1668.00	-	-	-	-	-	-	
Isurus oxyrinchus	120	310	-	430.00	-	-	-	-	-	-	
Mustelus mustelus	-	-	-	-	-	323	-	256	256	-	579
Notorhynchus cepedianus	-	-	-	-	-	-	-	41	41	-	41
Poroderma africanum	-	-	-	-	-	323	-	-	-	-	323
Prionace glauca	-	78	-	78.00	-	-	-	-	-	-	
Triakis megalopterus	-	24	-	24.00	-	323	-	74	74	-	397
Unspecified shark	10458	2274	1	12733.00	-	-	-	-	-	-	
Total for shark species	12049	2883	1	14933.00		1940	-	379	379	1039	3357
Skates and rays											
Dasyatis marmarota	-	-	-	-	-	-	-	-	-	-	
Gymnura natalensis	-	-	-	-	-	-	-	-	-	29	29
Myliobatis aquila	-	-	-	-	-	323	-	-	-	-	323
Raja alba	-	-	-	-	-	-	-	-	-	37	37
Unspecified ray	76	-	-	76.00	-	-	-	-	-	-	
Total for skates and rays	76	-	-	76.00	-	323	-			66	389
Total cartilagenous catch	12125	2883	1	15009		2263	- -	379	379	1105	3746
Overall total catch (numbers)					539751	424789	1149	9167	10316	1077285	2054160
Overall total catch (kg)	713059	1215785	417	1929261		469078					
Number of trips	1321	1991	47	3359		293694		13230		500	
Number of man-hours	76292	135329				1811112		39324			
Total man-days	13596	18796				293694		13230			

\# - Catch composition assumed to be the same as that of the B-licence commercial anglers

$ - Based on the catches of shore anglers from the whole of the Southwestern Cape, as reported by (Lamberth & Griffiths 1997)

* - Based on the assumption that 1 shore based and 9.6 boat based competitions (7.6 Wester Province and 2 Boland) are held within False Bay each year

& - Based on the assumption that 6 805 non-competitive dives are undertaken by spearfishermen each year

^ - Based on results reported by Lamberth *et al.* (1994)

Table 5.4 The contribution (%) of the ten most important species (by numbers or mass) landed by the five False Bay linefishery sectors. The value in bracket indicates what percentage of the total catch of that species (by numbers) landed in False Bay is landed by that particular fishery sector. Percentage values for deep-sea species are not included as this analysis concentrates only on what is caught closer to shore within False Bay.

	Commercial fishers		Recreational fishers		
Species	Boat anglers (% mass)	Beach-seiners (% numbers)	Boat anglers (% mass)	Shore anglers (% numbers)	Spearfishers (% numbers)
Umbrina canariensis		0.2 (22.1)		1.5 (77.9)	
Chirodactylus brachydactylus					8.5 (100)
Chirodactylus grandis					3.2 (100)
Chelidonicthys capensis				0.9 (100)	
Pachymetopon blochii	3.8 (97.2)		1.7 (<0.05)		22.8 (1.2)
Diplodus sargus capensis	0.5 (0.6)	0.2 (9.1)		5.2 (89.5)	1.9 (0.8)
Pomatomus saltatrix		1 (4.2)		58.4 (93.6)	
Dichistius capensis				12.6 (98.6)	7.5 (1.4)
Atractoscion aequidens	0.5 (100)		0.7 (<0.05)		
Merluccius spp.	0.4 (100)		0.5 (<0.05)		
Liza richardsonii		92.5 (99.3)		1.8 (0.7)	
Trachurus sp.		1.6 (100)			
Gymnocrotaphus curvidens					11.2 (78.2)
Argyrosomus inodorus	0.7 (14.4)	0.3 (12.3)	0.7 (<0.05)	5.1 (73.3)	
Trachurus trachurus				1.4 (100)	
Scomber japonicus	0.6 (100)		0.7 (<0.05)		
Pterogymnus laniarius	0.4 (100)		0.2 (<0.05)		
Sardinops sagax		1.1 (100)			
Chrysoblephus laticeps					30.2 (44.7)
Mustelus mustelus					2.5 (44.2)
Thyrsites atun	24.8 (100)		12.2 (<0.05)		
Cheilodactylis fasciatus					2 (100)
Sarpa salpa		1.2 (60.9)		1.9 (37.6)	
Unspecified shark	0.6 (100)		0.7 (<0.05)		
Lithognathus lithognathus		0.4 (20.3)		4.1 (79.6)	
Rhabdosargus globiceps	0.4 (93.5)		0.7 (<0.05)		
Seriola lalandi	5.3 (66.9)	1.1 (31.6)	4.1 (<0.05)		5.6 (1.5)
% of overall catch	38	99.6	22.2	92.9	95.4
Deep-sea pelagics	57.9	-	78.8	-	0.2

unexpectedly important to shore anglers was *Pomadasys commersonnii*. Shore anglers also caught over 262 000 inshore pelagic teleosts, the most important being *Pomatomus sultatrix* (58.4%), *Liza richardsonii* (1.8%), *Trachurus trachurus* (1.4%), and *Mugil cephalus* (0.4%). Although some cartilaginous species were also caught, most were released. Table 5.4 shows that the 10 most important species (% total catch by numbers) caught by shore anglers were *Pomatomus sultatrix* (58.4%), *Dichistius capensis* (12.6%), *Diplodus sargus capensis* (5.2%), *Argyrosomus inodorus* (5.1%), *Lithognathus lithognathus* (4.1%), *Sarpa salpa* (1.9%), *Liza richardsonii* (1.8%), *Umbrina canariensis* (1.5%), *Trachurus trachurus* (1.4%), and *Chelidonichthyes sp* (0.9%). These species constituted 92.9% of the number of fishes retained, of which the top five made up 85.4%.

The species targeted by shore anglers in False Bay are caught in different areas. Demersal reef-dependent (*e.g. Dichistius capensis*, *Diplodus sargus capensis*) and pelagic species (*e.g. Seriola lalandi*) are targeted along the rocky western and eastern shores of the Bay (Biden 1954; Schoeman & Schoeman 1990; Crous 1994). However, since catches of *Dichistius capensis* have declined in the last 50 years (Bennett 1991), and improved fishing tackle, prawn pumps and better off-road vehicles became available, many shore anglers started fishing off the northern shores of False Bay. There, they target *Argyrosomus inodorus*, *Pomatomus sultatrix*, *Umbrina canariensis*, and *Lithognathus lithognathus* (Penney 1991; Lamberth & Bennett 1993; Taylor 1993a). Because of their movement onto the northern shore beaches of the Bay, their catch includes many species that are shared with other fishery sectors, especially the beach-seiners. This has led to much conflict between these two groups (Lamberth & Bennett 1993).

Spearfishing

The total effort of the non-competitive spearfishery was 6 805 dives (6 240 shore dives) per annum, during which 9 167 fishes from 24 species (four chondricthians) were speared. The majority of these fishes (almost 90%) are reef-dependent demersal teleosts (Table 5.3). The only commonly speared pelagic teleost was *Seriola lalandi* (6.3%). The targeting for demersal sharks for illegal sale led to an annual catch of over 370 sharks. This is likely to be an underestimate. The total annual catch of the competitive spearfishery was 1 149 teleosts (16 species), which represented only 11.1% of all fishes speared within False Bay. Overall, the catch composition of competitive spearfishers is even more dominated by demersal reef-dependent teleosts (99.6%) than that of non-competitive divers (Table 5.3).

Overall, the total estimated catch of the False Bay spearfishery (competitive and non-competitive combined) was 10 316 fishes (26 species). This catch was dominated by 17 demersal reef-dependent teleost species (90.3% by numbers), with inshore pelagic teleosts (mostly *Seriola lalandi*) contributing 5.6%. Cartilaginous fishes contributed 3.7% of the fishes shot (Figure 5.2), of which the majority (almost 98%) was taken with the intention of illegal sale. The 10 most important species (% total catch by numbers) landed by the spearfishery are listed in Table 5.4, these constituting 95.4% of the spearfishery's total catch. Of these, only *Seriola lalandi* (5.6%) is a pelagic teleost, the remaining nine being reef-dependent teleosts. *Cheilodactylis fasciatus*, *Parascorpis typus*, *Chirodactylis brachydactylis*, *Chirodactylis grandis*, *Oplegnathus conwayi* and *Gymnocrotaphus curvidens* are targeted mainly by spearfishers. False Bay spearfishers, however, share many species with other linefishing sectors in False Bay, including *Dichistius capensis* and *Diplodus sargus capensis* with shore anglers, and *Chrysoblephus laticeps*, *Pachymetopon blochii* and *Pachymetopon aeneum* with boat anglers. Indeed, spearfishers shot 31.6% of the *Chrysoblephus laticeps* landed within False Bay (Table 5.4), although, it is hypothesised that the commercial boat anglers' take of this species is larger than reported. *Seriola lalandi* is shared with boat anglers, shore anglers (Schoeman & Schoeman 1990; Crous 1994) and beach-seiners. The catch-per-unit-effort (*cpue*) of spearfishers was over 60 fish per 100 hrs. This was almost three times that of recreational shore anglers (Lamberth & Griffiths 1997), and is more comparable to that of commercial boat anglers.

Boat angling

1. Recreational Boat anglers. The annual effort of the False Bay recreational boat angling sector was only 46.5 launches. Consequently, the total annual catch of this sector was also small, totalling 418 kg of fish. This is likely to be an underestimate, which is supported by Lamberth & Griffiths (1997) who, having calculated the total catch of the Southwestern Cape recreational boat angling sector using the same method, also hypothesised that their value was an underestimate. Figure 5.2 shows that the catch of this sector comprised mainly deep-sea teleosts (79% by mass), followed by inshore pelagic teleosts (16.8%). The overall catch comprised 14 teleost species and two 'unspecified species groups ('unspecified shark' and 'tuna') (Table 5.3). Table 5.4 shows that the catch of the 10 most important species likely to be taken within the Bay by this sector was small, totalling only 22.2% of its annual landing. This is because a large proportion of their estimated catch comprised tuna species (over 220 kg). Inshore, this sector targeted mainly *Seriola lalandi* and *Thyrsites atun*, two inshore pelagic teleosts.

2. Commercial boat anglers. The annual effort expended by A-licence holders in False Bay averaged 76 292 man/hours (1 321 boat outings, 13 596 man/days), while their *cpue* averaged 371.5 kg per 100 hrs. The annual effort of the B-licence holders averaged 135 329 man/hours (1 991 boat outings, 18 796 man/days). Their *cpue* averaged 500.8 kg per 100 hrs. The

overall effort of the commercial boat angling sector thus totalled 211 621 man/hours (3 312 boat outings) from a mean of 246 (55 A-licence) boats. A number of differences in the catch composition of the two commercial boat angling sectors are apparent (Table 5.3). A-licence holders caught greater quantities of small and/or unappetising species than did B-licence holders. For example, they landed over five times more *Spondyliosoma emarginatum*, *Pterogymnus laniarius*, *Pachymetopon blochii*, 'unspecified red', *Boopsoidea inornata*, *Spicara axillaris*, *Galeorhinus galeus* and 'unspecified ray' than did B-licence holders (only species whose reported catch was greater than 10 kg are mentioned). B-licence holders, on the other hand, appeared to target species of good eating quality (*e.g. Argyrosomus inodorus*, *Atractoscion aequidens*, *Merluccius capensis*, *Petrus rupestris*, *Popyprion americanus* and tuna species - see Table 5.3).

The estimated total annual catch of the False Bay commercial boat angling sector was over 1 480 tonnes. The most important fish groups (in terms of mass) were deep-sea pelagic teleosts (57.9%), most of which are landed many miles offshore of the coast, inshore pelagic teleosts (31%) and demersal reef dependent teleosts (5%) (Figure 5.2). Their catch consists of 48 (43 teleosts) species and five 'unspecified species' groups (three 'unidentified' teleost groups). These catches are, however, likely to be under-reported for many species, particularly demersal reef dependent-ones. Of the 43 identified teleosts, 21 are demersal reef-dependent species (totalling over 45 tonnes), 11 deep sea pelagic species (over 1 156 tonnes), five demersal soft-bottom species (over three tonnes) and five inshore pelagic species (over 223 tonnes). One reported species (Tristan bluefish), of which over eight tonnes were reported, is unlikely to have been landed within False Bay. A further 265 tonnes of 'unspecified' fish are added to these identified teleost catches, 99.7% of which are 'tuna' species (Table 5.3). The total annual cartilaginous catch of the commercial boat angling sector was over 15 tonnes, and consisted of four shark species, along with 'unspecified sharks' and 'unspecified rays'. Of the shark species, *Galeorhinus galeus* and *Triakis megalopterus* are demersal, the first of these constituting 1 668 kg (11.1%) of the cartilaginous catch. The other two shark species are pelagic, inhabiting deep offshore waters, and are not important to the fishery (3.4% of cartilaginous catch). It is hypothesised that the 'unspecified shark' category (totalling 12 tonnes and 84.84% of the cartilaginous catch) comprises much *Carcharhinus brachyurus*, *Mustelus mustelus* and *Triakis megalopterus*, as many observed False Bay shark catches consisted of these species. Skates and rays did not contribute greatly (only 75.5 kg – 0.5%) to the cartilaginous catches. Table 5.4 shows the 10 most important species (by numbers) landed by the commercial boat angling sector within False Bay (deep-sea species excluded). *Thyrsites atun* (25.4%) and *Seriola lalandi* (5.3%), both inshore pelagic species, are the most important, although *Pachymetopon blochii* is also important (3.8%). Overall, however, these 10 species contribute to only 38% of this sector's total landing (by mass).

Of the species reported by commercial boat anglers, a number are shared with other linefishing sectors (Table 5.3). For example, *Argyrosomus inodorus*, *Rhabdosargus globiceps*, *Lithognathus lithognathus*, *Sarpa salpa*, *Pachymetopon blochii* and *Diplodus sargus capensis* are shared with shore anglers and beach-seiners, *Chrysoblephus laticeps* with spearfishers and *Seriola lalandi* with all sectors. For many of these shared species, however, the majority of the catch is estimated to be caught by commercial boat anglers. Commercial boat anglers also catch species that are not important to other sectors, either due to their small size, offshore distribution and/or unappealing appearance (*e.g. Boopsoidea inornata*, *Argyrozona argyrozona*, *Pterogymnus laniarius*, *Merluccius capensis*, *Genypterus capensis*, *Popyprion americanus*, *Spicara axillaris*, *Sebastes capensis*).

Beach-seining

Lamberth *et al.* (1994) reported that seven beach-seine permit-holders were registered to operate in False Bay during the period of their study. Their catches included 26 species, including three cartilaginous ones (Lamberth *et al.* 1996). Table 5.3 shows that inshore pelagic and soft-bottom teleosts dominated beach-seiners' landings. The dominant catch (% of total catch by numbers) was *Liza richardsonii* (92.5%), followed by *Trachurus sp.* (1.6%). However, *Seriola lalandi* (1.1%), *Pomatomus sultatrix* (1%) *Lithognathus lithognathus* (0.4%), *Argyrosomus inodorus* (0.3%) and *Umbrina canariensis* (0.2%) were an important part of the catch, especially by mass. The latter four species, along with *Diplodus sargus capensis*, are species shared with shore anglers, which has caused much conflict between the two sectors. Most *Seriola lalandi* caught by beach-seiners are landed from the western shore of the Bay (Smitswinkel Bay, Simonstown, Fish Hoek), while the majority of *Liza richardsonii* catches are landed from the northern shores. Forty-three percent of the *Lithognathus lithognathus* catch comes from Simonstown, with the remaining 57%, and most *Argyrosomus inodorus* and *Pomatomus sultatrix*, being caught from the northern shore of the Bay (Penney 1991; Lamberth *et al.* 1994).

Table 5.3 shows that of the species landed by beach-seiners, *Liza richardsonii*, *Trachurus sp.*, *Sardinops sagax* and *Callorhinchus capensis* are seldom taken by other sectors, while most of the other species caught by them are shared by other linefishery sectors. Beach-seiners' catches were, however, smaller than those of shore anglers for all species except *Liza richardsonii*, *Sarpa salpa*, *Rhabdosargus globiceps* and *Callorhinchus capensis*.

Combined catches of all sectors

Of the 76 identified species and five 'unspecified' species groups landed by the different linefishing sectors, the following demersal reef-dependent teleost species are the most heavily

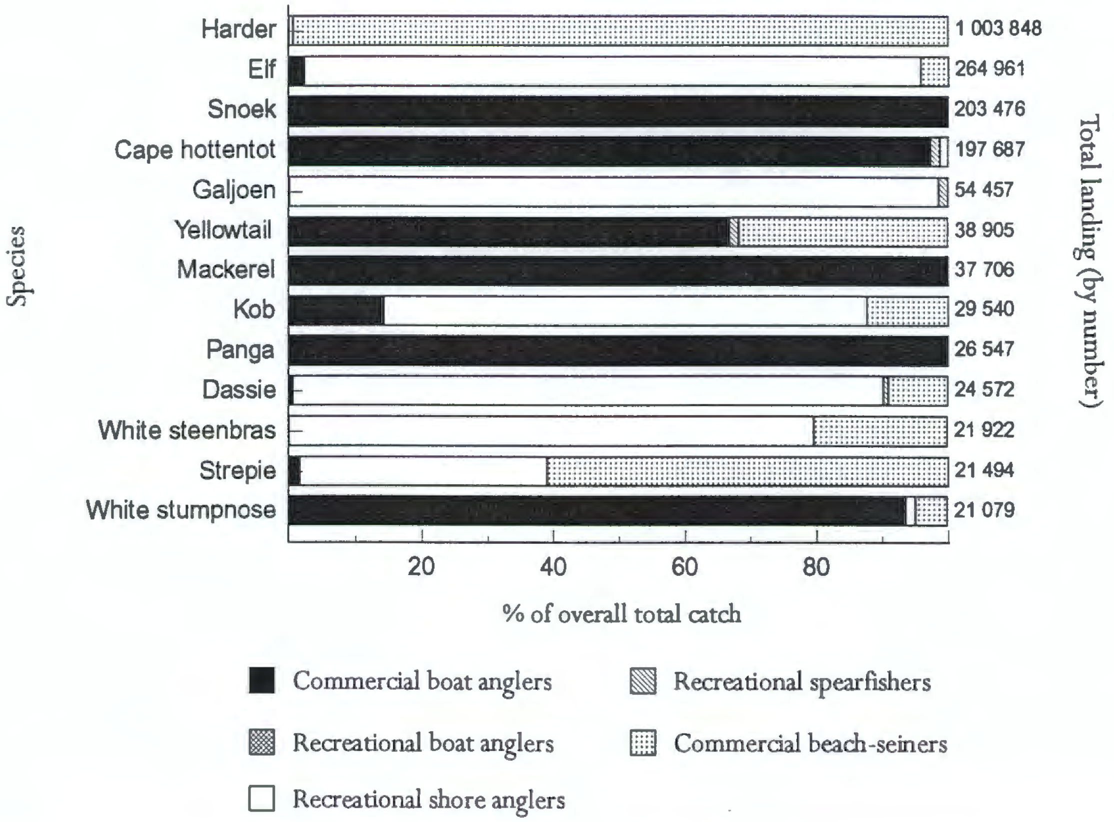


Figure 5.2. The 13 most caught linefish species (by numbers) in False Bay (excluding deep-sea pelagic species), with a percentage of the annual catch that each linefishing sector landed.

exploited (by numbers): *Pachymetopon blochii* (197 687), *Dichistius capensis* (54 454), *Pterogymnus laniarius* (26 547), *Diplodus sargus capensis* (24 572), *Sarpa salpa* (21 494) and *Spondyliosoma emarginatum* (13 725). *Argyrosomus inodorus* (29 540), *Rhabdosargus globiceps* (21 079) and *Lithognathus lithognathus* (21 922) are the most exploited the demersal soft bottom teleost species caught. *Liza richardsonii* (1 003 848), *Thyrsites atun* (203 476) and *Seriola lalandi* (38 905) are the most impacted inshore pelagic teleosts. Of the 1 156 473 kg of deep-sea teleosts landed each year, 99.7% comprise deep-sea tuna species. Of the cartilaginous species retained, *Galeorhinus galeus* (1 668 kg) and 'unspecified shark' (12 733 kg) are the most important. Figure 5.3 shows the 13 species that are numerically the most impacted by the overall linefishery (all sectors) within False Bay, as well as the proportion of the catch that is landed by each sector. Of these species, over 85% of *Thyrsites atun*, *Pterogymnus laniarius*, *Scomber japonicus*, *Pachymetopon blochii* and *Rhabdosargus globiceps* are landed by the commercial boat angling sector. Shore anglers land over 85% of *Dichistius capensis*, *Pomatomus sultatrix* and *Diplodus sargus capensis*, while beach-seiners land 99.3% of the *Liza richardsonii* catch. The remaining four species are more equally shared between the different sectors. Table 5.4, however, shows that of the 10 species most important (by numbers) to each of the fishery sectors, shore anglers still dominate the overall catch of *Lithognathus lithognathus*, *Argyrosomus inodorus*, *Umbrina canariensis*, *Chelidonichthyes sp* and *Trachurus trachurus*. Commercial boat anglers land the majority of *Atractoscion aequidens* and *Merluccius capensis* catches, while *Sardinops sagax* is the only other important species whose catch is landed mostly by beach-seiners. Four of the species most important to the spearfishery (*Chirodactylis brachydactylis*, *C. grandis*, *Cheilodactylis fasciatus*, *Gymnocrotaphus curvidens*) are landed mostly exclusively by this sector, while *Chrysoblephus laticeps* is more equally shared between the boat and spearfishing sectors.

Discussion

The species list in Table 5.3 highlights that within False Bay, the linefishery targets species from all available subtidal habitats. That table also shows that all False Bay linefishery sectors are multispecies fisheries. Commercial boat anglers (A- and B-licence combined) caught 46 species and 5 'unspecified' species groups, shore anglers 30 species, beach-seiners 26, spearfishers 26, and recreational boat anglers 14 species and 2 'unspecified' groups. This is explained by the fact that many species of good eating quality, similar size and with similar diets are available to the fisheries. Table 5.3 also shows that many of the species caught in False Bay are shared between the different sectors, although for most species, the majority of the overall catch is landed by one. The values in Table 5.3 agree with Lamberth *et al.* (1994) who estimated that shore anglers land the majority of *Lithognathus lithognathus*, *Diplodus sargus capensis*, *Umbrina canariensis* and *Dichistius capensis* caught in False Bay. In contrast, the catch composition for shore anglers does not agree with Lamberth *et al.* (1994), as it also suggests that shore angler's catches dominate landings of *Argyrosomus inodorus* and *Pomatomus sultatrix* in the area. In agreement with Lamberth *et al.* (1994), Tables 5.3 & 5.4 show that *Rhabdosargus globiceps* catches are dominated by commercial boat anglers.

Table 5.3 shows that the estimated effort of recreational boat anglers is a lot smaller than that of commercial boat anglers. This may be explained by the fact that, as proposed by Lamberth & Griffiths (1997), many of the 'recreational' boat anglers active within False Bay actually own a B-licence. This provides further support for the assumption that the catch composition of recreational boat-based fishers is the same as that of B-licence holders.

The total catch and catch composition of the False Bay linefishery has changed greatly since its inception. In 1910, commercial boat anglers targeted mainly *Thyrsites atun*, *Atractoscion aequidens* and *Argyrosomus inodorus* (Penney 1990). *Seriola lalandi* was not targeted then, as more profitable species abounded. Similarly, during the first half of this century, *Chrysoblephus gibbiceps* was targeted by anglers fishing for reef-dependent species and adult *Petrus rupestris* were landed regularly. However, during the first half of this century, *Atractoscion aequidens* were so abundant within False Bay that, in order to save space for packing these more profitable fishes, large (>25 kg) *Petrus rupestris* were not always loaded into the boats if caught (Biden 1954). *Chrysoblephus laticeps* was very common at that time and regarded as a pest (Biden 1954; Schoeman & Schoeman 1990). Shore anglers landed *Sparodon durbanensis* and also caught adult *Petrus rupestris*. Penrith (1972) reports that during the 1960s, the Southern Cape (Cape Point to Tsitsikamma) linefishery also targeted the same predatory species as before, although *Seriola lalandi* was now also targeted. To tide over the times between these species' seasons, however, linefishermen targeted demersal sparids. In False Bay, such species included *Chrysoblephus laticeps* (the most economically important of these winter species), *Chrysoblephus gibbiceps*, *Pachymetopon blochii*, *Argyrozona argyrozona* and *Rhabdosargus globiceps*. *Boopsoidea inornata* and *Spondyliosoma emarginatum* were targeted mainly as baitfish. This demersal-sparid-targeting fishery was not profitable. The present study indicates that commercial boat anglers still target *Thyrsites atun*, *Seriola lalandi*, *Argyrosomus inodorus*, *Atractoscion aequidens*. Indeed, Penney (1990) reports that the 1988 *Thyrsites atun* national catch was greater than that in 1910, although the total 1988 linefish catch only totalled 75% of that of 1910 (excluding tuna and squid, which were not targeted then). Moreover, the species composition and size distribution of catches has changed in the last few decades, especially those of reef fishes (Penney, Buxton, Garratt & Smale 1989). Catches of *Chrysoblephus gibbiceps*, *C.*

laticeps, *Pterogymnus laniarius* and *Argyrozona argyrozona*, species which are usually taken when the more profitable predatory species are not available (Pulfrich & Griffiths 1988b; Penney 1991), have declined substantially (Crawford & Crous 1982; Penney *et al.* 1989; Penney 1990). The decline in the availability of most linefish species has encouraged many commercial anglers to target different species (Pulfrich & Griffiths 1988b; Penney 1990; Attwood, Harris & Williams 1997a). For example, *Seriola lalandi* is now an important commercial angling species (unlike in 1910), while certain demersal species (*e.g. Boopsoidea inornata*, *Chelidonichthyes sp*, *Pachymetopon blochii*, *Spondyliosoma emarginatum* and sharks) are now targeted as food-fish. Indeed, this is supported by Pulfrich & Griffiths (1988b) and Penney (1991) who report that *Pachymetopon blochii* is presently more targeted in False Bay, especially by A-licence holders, than it was in the past. However, the most important of the new targeted species are tunas (Penney 1990), these now contributing a substantial portion of False Bay commercial anglers' catches (Table 5.3).

It is noteworthy that boat anglers target profitable species until the landings of these are negligible, after which they target the next profitable species that is available. This is termed 'pulse overfishing' by Claverie (1990) and 'serial overfishing' by Bohnsack & Ault (1996), and is a worldwide problem that impacts many slow growing, late maturing and/or endemic species. The effects of this fishery practice are difficult to overcome by management, as once such species have been overfished, they are unable to recover quickly. Indeed, in addition to the above mentioned life history characteristics, Buxton (1990a, 1993a), Garratt & van der Elst (1990), Hecht (1990), and Attwood *et al.* (1997a) report that longevity, migration, residency, sex change, barotrauma, and handling injury suffered by landed fishes, are the main management problems facing the South African linefishery. All of these are applicable to the False Bay region. The multispecies nature of the Bay's different fishery sectors is another factor that makes the management of the region's linefishery difficult, as it eliminates the possibility of managing each fish species in isolation (Buxton 1990b; Hecht 1990; Bennett 1991b; Lamberth *et al.* 1994; Iverson 1996; Lamberth & Griffiths 1997; Sauer *et al.* 1997). The situation is worsened by the fact that the spearfishery is the only fishery sector that can select its catch visually. Management is further complicated by the large number of species that are shared between different sectors, as individual sectors cannot be managed in isolation either. Bundy (1996), Iverson (1996) and Tilzey (1996) report that the sharing of species between different fishery sectors is a common management problem worldwide, with each sector wanting allocation rights to a portion of the shared stocks, particularly when these are decreasing. In False Bay, this sharing of species has led to clashes between recreational and commercial operators (Sauer *et al.* 1997), especially since these sectors' catches have declined. A typical example is the conflict between beach-seiners and shore anglers (Penney 1991; Lamberth & Bennett 1993; Lamberth *et al.* 1994), with beach-seiners being blamed by shore anglers for the depletion of the stocks of 'angling species', and the deaths of countless juvenile and inedible fishes (Lamberth & Bennett 1993). Although such clashes between different sectors over shared species are not unusual worldwide, they remain an important problem to overcome if linefish management is to be successful (Brokensha 1990).

A number of regulations are enforced to manage False Bay linefish stocks. These are intended to minimise total catch, thereby ensuring future catches of the species. For example, all linefish species fall into one of the five 'species lists' shown in Table 5.1. The total catch of each is, therefore, limited to some extent by daily bag limits. Similarly, minimum size restrictions (Table 5.2) and closed seasons, during which the particular species cannot be retained if caught, are also enforced. Marine reserves are also employed to help manage the fisheries of the area. However, fisheries regulations are often transgressed in South Africa.

Within the recreational sectors, this is mostly due to a lack of knowledge concerning the regulations (Lamberth *et al.* 1996; Sauer *et al.* 1997; Chapter 4), although some fishers do so intentionally. Reasons for wilful transgressions include jealousy between recreational and commercial fishers due to the more lenient restrictions enforced onto commercial fishers (Bennett 1991b; Sauer *et al.* 1997), and the selling of catches (Sauer *et al.* 1997; Chapter 4) to defray fishing expenses (Hutchings 1993; Lamberth *et al.* 1996). In the case of commercial boat anglers, regulation transgressions are usually intentional (Bennett 1991b; Taylor 1993b; Lamberth *et al.* 1996; Brouwer *et al.* 1997; Sauer *et al.* 1997). The reason why transgressing fishers escape apprehension lies in the inefficient enforcement of fishery regulations. In the Southwestern Cape (Brouwer *et al.* 1997; Lamberth & Griffiths 1997; Sauer *et al.* 1997), enforcement is very inefficient. For example, shore anglers in the region are estimated to be inspected only once in more than 25 years (Bennett 1991a; Lamberth & Griffiths 1997). Fishery regulation enforcement should thus be increased substantially in the region (Lamberth *et al.* 1996; Lamberth & Griffiths 1997; Sauer *et al.* 1997), along with the penalties for transgressions (Brokensha 1990). Furthermore, fishers should be educated about the value of the fishery regulations, thereby encouraging them to comply (Winch 1990b; Coetzee 1993; Taylor 1993b; Attwood, Beaumont, Branch, Densham, Dye, Feely, Harris, Heydorn & Hockey 1997b; Attwood, Mann, Beaumont & Harris 1997c).

The linefishery regulations enforced have failed to provide reasonable resource protection, and many linefish species have been over-exploited, resulting in other species being targeted. The fact that linefishing effort in the False Bay region, particularly recreational linefishing (Penney 1991), is likely to increase considerably in the future is an extra reason for concern. The fish resource of the area will have to be further sub-divided between the increasing number of fishers. How this will be achieved is being investigated presently, but is likely to involve making the regulations first implemented in 1985 more restrictive so as to achieve the goals of the new Linefish Management Protocol. Changes in enforced regulations are likely to be introduced for the majority of the reef fish species that are exploited within False Bay, including *Dichistius capensis* and *Pachymetopon blochii*, species for which management regulations have been supported previously by Bennett (1988) and Pulfrich & Griffiths (1988b). Jennings & Lock (1996) report that to optimise management strategies in the hope of maintaining a fishery's yield and catch composition, its effects on stocks need to be known. The impact of each fishery sectors on the False Bay stocks is not completely understood, however, and need further research. Notwithstanding that, it is proposed that the following recommendations could, along with other changes in regulations, help optimise fishery management under the new Linefish Management Protocol in the future: I. The presently enforced minimum size restrictions should be increased, thereby making them more restrictive. Most of the presently enforced minimum sizes are set at a size corresponding to 50% sexual maturity (Mann 1992; Buxton 1995), giving half of the individual fishes of a population a chance to reproduce before they are retained if caught (Buxton 1990a). Should this minimum size be increased (say to the 60-65% maturity level), the protection afforded to linefish populations would be increased, providing the species is not susceptible to barotrauma injuries and survives capture and release. II. The daily bag limits enforced onto the different linefishery sectors should be made more restrictive. These were imposed with the intention of protecting enough of the parent stock to help maintain catches at a sustainable level (Buxton 1990a). They are, however, not restrictive enough (Bennett 1988; Attwood & Bennett 1995b; Lamberth & Griffiths 1997; Chapter 4) and thus provide little protection (Mann 1992). Furthermore, they are not species specific. Attwood & Bennett (1995b) state that there are no advantages to multispecies daily bag limits, especially if the species within a single list have different conservation problems. To make the South African daily bag limits be more restrictive, the following changes are proposed for recreational fishers: a) a maximum daily bag of two individuals per species, and b) a maximum daily bag of four fishes. The first of these would apply to most of the important linefish species, except those whose spawner-biomas-per-recruit levels are below the threshold reference point of 25% set in the new Linefish Management Protocol. For such species, the daily bag limit should be one. The maximum daily bag limit of four fishes would only apply for species not listed within the 'bait list' in SFRI (1998), for which the daily catch would remain unlimited. III. Closed seasons, if thought necessary, should span a time period during which the species concerned would benefit optimally from the resulting decline in fishery effort (Buxton 1990a). Although usually spanning the species' breeding season (Buxton 1995), they rarely improve recruitment through the protection of breeding individuals of long-lived species (Mann 1992). The closed season for *Dichistius capensis* coincides with its breeding season, when the species is in the poorest condition and least catcheable (Attwood & Bennett 1990). Based on the last two reasons, Attwood & Bennett (1990) proposed that the species' closed season is not an effective restriction, and that it could be abolished. However, considering the rapid increase in recreational shore angling effort, a closed season could still be beneficial, especially if set to coincide with a time period when the species is more targeted due to its good condition and increased catcheability. IV. The management measures concerning the marine protected areas situated in False Bay could also be changed to make these protected areas more effective. No linefishing should be permitted within them, while those that were established in sub-optimal areas should be re-located. Increasing their size is likely to also enhance their fishery management tool potentials. Overall, it is hypothesised that the proposed changes to fishery regulations, along with an increase in the enforcement of all fishery regulations, would enhance fishery management in False Bay. Reproductive output of some targeted species would increase, while recreational catches would be more effectively controlled. However, as pointed out in Figure 5.3, commercial boat anglers land the majority of many of the most impacted species of the region. Furthermore, the practice of 'pulse/serial overfishing' by commercial boat anglers has led to a decline in the catches of the majority of the linefish species targeted previously. Considering that these commercial boat anglers are now more effective (better navigational equipment, sounding equipment, boats), it is likely that the presently targeted reef fish species will also become overfished, even though the regulations concerning their landings may be more appropriate than they were in the past. This problem is worsened by the fact that there are few reef fish species that are still abundant within False Bay. Should *Chrysoblephus laticeps*, *Pachymetopon blochii*, *Spondyliosoma emarginatum* and *Boopsoidea inornata* become overfished, it is not known which reef fish species would be targeted next by the commercial boat angling sector. Considering that restricting the total landings of individual commercial boat anglers by setting much stricter daily bag limits would hinder their livelihood, it is proposed that the total number of commercial permits made available each year should be decreased substantially.

Chapter 6

Small marine reserves: are they beneficial in fisheries management? A case study from False Bay, South Africa

Chrysoblephus gibbiceps

Introduction

Traditionally, fisheries have been managed using two forms of control measures: management of catch, and management of effort (Bohnsack & Ault 1996). Measures regulating catch include daily bag limits, catch quotas, size restriction and gear restrictions, while those regulating fishery effort include closed seasons, closed areas and limiting the number of entrants into a fishery (Buxton 1993b; Roberts & Polunin 1993; Pollard 1996). The goal of these measures is to allow the fishery to harvest 'excess production', while leaving the reproductive capacity of the stock intact (Dugan & Davis 1993). These classical management measures, however, were designed to manage single species/single gear fisheries rather than multi-species/multi-gear reef fisheries (Butterworth *et al.* 1989; Buxton 1993b; Roberts & Polunin 1993; Pollard 1996), for which they are less successful (Buxton 1993b; Pollard 1996). Reef fishes tend to be K-selected species, are characterised by slow growth, and thus slow recovery from exploitation. Fisheries targeting such species suffer from growth overfishing, as fishes are caught before they can grow (Buxton 1993b; Carr & Reed 1993; Bohnsack & Ault 1996; Attwood *et al.* 1997a). This is further complicated because reef fisheries often target the larger individuals of a species, which may also result in the species suffering from recruitment overfishing, when the number of fish entering the stock is reduced due to overexploitation of the spawning stock (Carr & Reed 1993; Bohnsack & Ault 1996; Attwood *et al.* 1997a). Reducing these two forms of overfishing is not easy, as measures set to minimise the one often conflict with those minimising the other (Dugan & Davis 1993).

The shortcomings of the classical fisheries management measures discussed above can be summarised as follows:

1. They require much information concerning the biology of the linefish stock. This information is usually difficult and expensive to obtain (Butterworth *et al.* 1989), and as a result, management decisions are often made based on insufficient information (Siegfried & Davies 1982; Roberts & Polunin 1993; Bohnsack & Ault 1996).
2. They are not suitable for the management of multispecies fisheries, because they are based on single species rather than multispecies population dynamics (Butterworth *et al.* 1989; Buxton 1993b; Roberts & Polunin 1993; Pollard 1996).
3. They are difficult and expensive to enforce (Roberts & Polunin 1993; Attwood *et al.* 1997a).
4. They may not be capable of dealing with pulse/serial overfishing and release mortality (Bohnsack & Ault 1996).

These problems have led scientists and fishery managers to focus increasingly on the use of marine reserves, where exploitation is restricted, for the protection and/or enhancement of fishery stocks (Butterworth *et al.* 1989; Carr & Reed 1993; Bohnsack & Ault 1996; Clark 1996; Attwood *et al.* 1997a).

Marine reserves are viewed as useful linefish management tools for a number of reasons. In particular, they protect all the exploited and unexploited species found within them (Buxton 1990b; Roberts & Polunin 1991; Mann 1992; Attwood *et al.* 1997a, b, c), *i.e.* both linefish and the communities which sustain them. They also protect the genetic diversity of targeted stocks, potentially an important commodity for the species concerned (Roberts & Polunin 1991; Dugan & Davis 1993; Bohnsack 1996a,b; Attwood *et al.* 1997a, b).

Marine reserves are also directly beneficial to fisheries. Firstly, they maximise the chance of a large number of reproductive propagules being produced by fishery species. This is because within marine reserves, targeted species reach greater relative abundances compared to on exploited reefs (Bell 1983; Buxton & Smale 1989; Roberts & Polunin 1991; Rowley 1994; Bohnsack 1996b; Attwood *et al.* 1997a), and are capable of growing to their maximum size, which is important since fecundity is positively related to body size (Roberts & Polunin 1991). These propagules can then enhance recruitment on adjacent exploited areas, if they are distributed to appropriate habitat by ocean currents (Roberts & Polunin 1991, 1993; Buxton 1993b, 1995; Bohnsack 1996a, b; Jennings & Lock 1996; Russ & Alcala 1996a, b; Attwood *et al.*, 1997a, b). However, the degree of residency of the species concerned, and the size of the reserve, affects the benefits species receive from reserves (Buxton 1993b; Attwood *et al.* 1997a). Only resident species are likely to escape fisheries targeting them outside a reserve (Bohnsack 1996b), especially if the reserve is small. The possibility that a reserve may be a source of recruits for exploited reefs is particularly important when exploited populations suffer from recruitment overfishing (Buxton 1990a, b; Dugan & Davis 1993), and/or when selective removal of larger individuals from sequentially hermaphroditic species has resulted in the skewing of their sex ratio (Buxton 1993a). Overall, marine reserves established to conserve reef fishes provide an area within which all reef fish individuals are safe from exploitation. Such areas have become increasingly necessary, as present-day linefishermen are more effective than they were in the past (due to improved navigational and angling equipment) and angling effort has increased substantially (Attwood *et al.* 1997a). The role of marine reserves in supplying recruits to exploited areas has rarely been evaluated, however (Roberts & Polunin 1991, 1993; Tilney, Nelson, Radloff & Buxton 1996; Attwood *et al.* 1997a). Secondly, marine reserves may enhance fisheries active near their borders by exporting adult fishes to these exploited areas (Roberts & Polunin 1991, 1993; Rowley 1994; Bohnsack 1996a; Attwood *et al.* 1997b). The export of shore angling species from the De Hoop Marine Reserve has been reported by Bennett & Attwood (1991, 1993a) and Attwood & Bennett (1994), while Russ & Alcala (1996b) report the same concerning tropical reef fishes in the Philippines. This benefit, however, is only apparent close to a reserve's borders (Roberts & Polunin 1991; Bennett & Attwood 1993a; Russ & Alcala 1996b). Thirdly, because marine reserves protect adult stocks of some targeted species, they also provide fisheries with an insurance against the failure of traditional management measures enforced in exploited areas (Roberts & Polunin 1991; Buxton 1993a; Bohnsack 1996a, b; Attwood *et al.* 1997a, b).

Over and above these benefits, additional advantages of marine reserves are:

- The legislation is easy to enforce, as a person fishing within a reserve is transgressing the law (Bennett 1991b; Roberts & Polunin 1991, 1993; Bohnsack 1996a, b; Bohnsack & Ault 1996; Attwood *et al.* 1997a, b).
- They do not require expensive annual data collection and assessment effort once their initial success has been tested (Roberts & Polunin 1991, 1993; Carr & Reed 1993; Bohnsack 1996a, b). Indeed, if proved effective and large enough, a reserve will remain effective even when the surrounding areas are subjected to very high levels of exploitation (Bennett 1991b).
- They are supported by the general public because they can still be used in non-consumptive ways, such as nature watching, underwater photography and diving (Bohnsack 1996b; Attwood *et al.* 1997b), and because their purposes are easy to understand. Furthermore, Brouwer *et al.* (1997), Mann *et al.* (1997) and Sauer *et al.* *(*1997) report that South African fishers support the use of marine reserves.
- They are useful for conservation education (Siegfried & Davies 1982; Bohnsack 1996b; Attwood *et al.* 1997a, b).

- They provide useful research areas (Siegfried & Davies 1982; Bohnsack 1996b; Attwood *et al.* 1997b) within which linefish research and the testing of management strategies can be undertaken (Hecht 1990).

The success of some South African marine reserves has been documented. Two large (more than 45 km long) marine reserves (Tsitsikamma National Park and De Hoop Marine Reserve), both situated on the South coast, are reported to be successful in conserving some linefish stocks. They both contain more individuals of targeted species, and proportionally more large individuals (Buxton & Smale 1989; Bennett & Attwood 1991). Furthermore, the De Hoop Marine Reserve is reported to enhance linefisheries outside of its borders due to the export of adult fishes (Bennett & Attwood 1993a, b), while the Tsitsikamma National Park exports sparid larvae to exploited reefs (Tilney *et al.* 1996). However, the establishment of such large marine reserves often leads to a conflict of interest, with fishers often opposing the establishment of the reserve (Beaumont 1997), often with some political backing (Roberts & Polunin 1993). South Africa has not escaped from this. After the establishment of the De Hoop Marine Reserve and Tsitsikamma National Park, fishermen wanted access to the preserved stocks within the new reserves (Attwood *et al.* 1997c). Due to *ad hoc* local pressure, a lack of predetermined criteria and planning (Hockey & Branch 1997), and the above-mentioned difficulties regarding the establishment of large marine reserves, many more small marine reserves (<10 km long) have been established within South Africa than large ones. As South Africa moves towards the implementation of the new fisheries policy drafted into the Marine Living Resources Bill, the goals of which include the redistribution of resources to broaden access to new entrants, consideration of 'subsistence' fisheries and resource co-management, an increase in demands for access to coastal resources is likely to result in management problems. Due to this, large reserves are unlikely to be supported by the general public as this will restrict fishers' access to large stretches of coastline. This may explain why Attwood *et al.* (1997b) report that in the future, it is likely that the establishment of numerous small marine reserves will be more feasible than the establishment of fewer large ones. This certainly applies for False Bay, as its coastline is only about 140 km long. Should 40 km of this be designated as a marine reserve (extending 5 km offshore), a large proportion of the reefs currently fished by commercial and recreational fishers would become unavailable, leading to conflict between fishers and managers. This could be resolved through the use of numerous small marine reserves, however, as their use is socially more acceptable (Attwood *et al.* 1997a). Four such small reserves (<5 km) and two larger ones (6.5 and 9 km) have indeed been established within False Bay (SFRI 1996). However, only within two of the small ones are all reef fishes protected

To date, the potential benefits provided to linefisheries by such small marine reserves have not been investigated, highlighting the important gap in management knowledge discussed by Attwood *et al.* (1997c). This chapter sets out to test whether one of the two small reserves within which all reef fishes are protected (the 3.25 km long Castle Rocks Marine Reserve situated on the western shore of False Bay - Figure 1.1) achieves the following:

- The protection of targeted and non-targeted species' populations, resulting in them being more abundant than on exploited reefs.
- These protected populations containing proportionately more large individuals compared to exploited populations.
- The enhancement of adjacent fisheries due to the export of adult fishes from the reserve.

The ability of the reserve to achieve the first two was tested by comparing the reef fish assemblage within the Castle Rocks Marine Reserve to that on exploited reefs in the Bay. To test whether the third benefit was achieved, in accordance with Roberts & Polunin (1991), Bennett & Attwood (1993a) and Russ & Alcala (1996b), the *cpue* of fishers active within close proximity of, and further away from, the reserve were compared.

Based on the results of these comparisons, recommendations are made concerning the use of such small marine reserves as a linefish management tool.

Methods

Study site

The Castle Rocks Marine Reserve is situated between Bakoven Rock (south of Miller's Point) and Bobbejaanklip (south of Partridge Point) on the western shore of False Bay (Figure 6.1). Because it stretches one nautical mile offshore (Crous 1994), the reserve encompasses reef extending to depths in excess of 25 m. Furthermore, numerous blinders and small islands (including a small seal colony) are located within its borders. It fits the definition of a marine reserve, as defined by Attwood *et al.* (1997b), as some form of linefishing, in this case the targeting of *Thyrsites atun* by boat anglers (Crous 1994), is permitted within its borders. This has caused some controversy (Winch 1990b), as it encourages these anglers to fish illegally for bottom fish within the reserve (Van Herwerden 1989). The Castle Rocks Marine Reserve is one of the only two False Bay marine reserves within which no reef fish may be taken (Crous 1994). No shellfish or algae may be collected within its borders either (Attwood *et al.* 1997b).

Experimental procedure

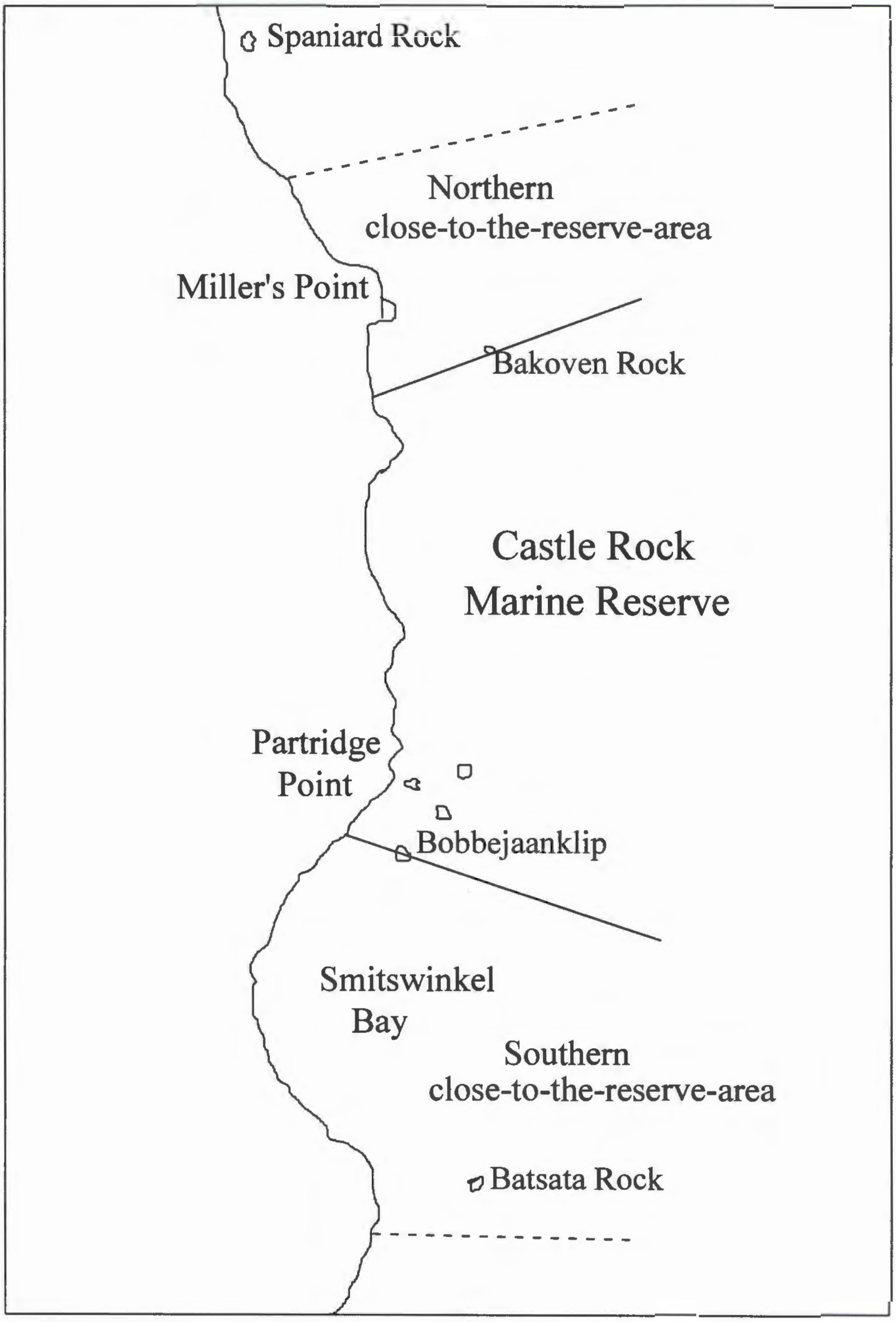


Figure 6.1 The location of the northern and southern close-to-the-reserve-areas in relation to the reserve borders.

Reef fish assemblages were surveyed within the Castle Rocks Marine Reserve and on exploited reefs using the same survey technique discribed in Chapter 1. To test whether the linefish stocks within the Castle Rocks Marine Reserve were denser than those outside its borders, fish relative abundance estimates obtained within and outside the reserve were compared using the Mann-Whitney U Test (Zar 1984). Much of the exploited reef located within the rest of False Bay was censused to allow for a comparison between the protected and exploited fish stocks.

This comparison was undertaken for each of the six depth strata discussed in Chapter 1, and within two separate temperature ranges: warm (warmer than 13 °C) and cold (13 °C or colder). For this comparison, all counts obtained within each of these depth strata were employed, providing that they covered a reef surface area larger than 7 m^2.

The size distributions of the different linefish species encountered within and outside the reserve borders were compared using the Kolmogorov-Smirnof Goodness of Fit test (Zar 1984), to test whether the protected populations contained proportionally more large individuals of targeted species than did exploited ones.

To test whether the reserve is exporting substantial numbers of adult reef fishes to exploited areas, the catch-per-unit-effort (*cpue*) of non-competitive spearfishers active close to, and further away from, the reserve were compared. Due to its selectivity for larger fishes, the spearfishery was deemed the most likely to show any increase in fish catch rates due to the export of adult fishes from the reserve. Should only small fishes be exported, this benefit will not be discernible by this *cpue* comparison. It was assumed that spearfishing effort is similar close to and further away from the reserve. Caravans Reef (located just north of Miller's Point) was designated as the northern boundary of this close-to-the-reserve area, while its southern boundary was the reef situated 200 m south of Batsata Rock (Figure 6.1). The *cpue* values compared included reef fish catches (all species combined) and total catches (all speared fishes irrespective of species group), and were calculated for all spearfishers combined irrespective of water temperature. These *cpue* values were calculated using the data reported in Chapter 4.

Results

Twenty-five reef fish species were encountered during the study period (Tables 6.1 & 6.2). All were seen when the water was warm (Table 6.1), while *Poroderma pantherinum* was the only species not encountered in cold water (Table 6.2). *Notorhynchus cepedianus* was only seen inside the reserve, while *Poroderma pantherinum* was only encountered outside of it (Tables 6.1 & 6.2).

Eighteen of the species encountered were teleosts, of which seven are important to boat anglers (*Boopsoidea inornata*, *Chrysoblephus gibbiceps*, *C. laticeps*, *Pachymetopon aeneum*, *P. blochii*, *Petrus rupestris* and *Spondyliosoma emarginatum*), five to shore anglers (*P. blochii*, *Dichistius capensis*, *Diplodus sargus capensis*, *D. cervinus hottentotus*, *Sarpa salpa*), and seven to spearfishers (*Chirodactylis brachydactylis*, *C. grandis*, *P. blochii*, *D. capensis*, *Gymnocrotaphus curvidens*, *C. laticeps*, *Cheilodactylis fasciatus*). Four of the seven cartilaginous species (*Haploblepharus edwardsii*, *H. pictus*, *Poroderma africanum*, *P. pantherinum*) are not targeted by False Bay fishers, while *Mustelus mustelus*, *Notorhynchus cepedianus* and *Triakis megalopterus* are targeted by commercial boat anglers, shore anglers and spearfishers (Chapter 5).

Tables 6.1 & 6.2 show that the reef fish species richness within the Castle Rocks Marine Reserve was significantly greater ($P<0.05$) than on exploited reefs, for all depths strata

Table 6.1 The relative abundance (fish.100 m^{-2}) of the species encountered in warm water (>13°C) within and/or outside the Castle Rocks Marine Reserve, within different depth strata. The significance ($p<0.05$) of all differences was tested using the Man Whitney U test.

	0-5 m			>5-8 m			>8-12 m			>12-16 m			>16-20 m			>20-25 m		
	Inside	Outside		Inside	Outside		Inside	Outside		Inside	Outside		Inside	Outside		Inside	Outside	
Species																		
Reef teleost species																		
Boopsoidea inornata	0.47	0.74	&	4.76	3.04	$	15.70	1.32	$	27.91	2.40	$	20.94	7.57	$	16.11	7.94	$
Cheilodactylus fasciatus	3.21	2.35	$	2.75	3.75	&	7.16	4.11	*	4.88	7.92	@	10.49	9.47	*	9.54	7.23	*
Cheilodactylus pixi	-	-	-	-	-	-	-	-	-	-	-	-	-	0.07	&	0.24	0.48	&
Chirodactylus brachydactylus	4.86	3.95	*	5.99	4.58	*	8.14	3.94	$	3.48	4.42	&	1.12	2.33	@	-	1.95	@
Chirodactylus grandis	-	-	-	-	-	-	-	-	-	0.94	0.489	*	1.34	0.17	$	3.11	0.13	$
Chrysoblephus gibbiceps	-	-	-	-	-	-	-	-	-	0.02	-	*	0.62	-	$	0.90	-	*
Chrysoblephus laticeps	2.54	0.22	$	5.85	1.58	$	12.93	4.81	$	15.02	9.17	$	16.70	10.02	$	17.26	10.23	$
Dichistius capensis	3.00	3.68	&	1.21	1.42	&	1.45	1.29	$	0.87	0.39	$	0.05	-	*	-	0.24	&
Diplodus cervinus hottentotus	1.34	-	$	0.55	0.40	*	0.58	0.62	&	0.82	0.65	*	0.87	0.53	*	1.89	1.00	*
Diplodus sargus capensis	5.99	2.02	$	1.16	2.22	&	1.72	3.51	&	2.05	1.50	*	0.88	1.43	&	0.71	5.81	&
Gymnocrotaphus curvidens	2.12	0.63	$	3.48	0.78	$	5.16	1.53	$	1.38	1.48	&	1.07	1.18	&	0.67	0.84	&
Oplegnathus conwayi	-	-	-	0.39	0.03	$	1.15	0.04	$	-	0.03	@	0.05	-	*	-	-	-
Pachymetopon aeneum	0.03	-	*	-	-	-	0.47	0.17	$	0.24	0.47	&	0.72	0.07	$	0.45	-	*
Pachymetopon blochii	9.23	3.49	$	35.17	14.55	$	63.09	22.66	$	44.24	24.68	$	34.73	19.21	$	36.95	25.93	*
Parascorpis typus	0.40	0.31	*	0.77	2.45	&	0.55	0.50	*	0.66	0.58	*	0.75	0.79	&	1.56	2.10	&
Petrus rupestris	-	-	-	0.05	0.04	*	0.21	0.01	$	0.24	0.03	$	0.22	0.04	*	-	-	-
Sarpa salpa	38.74	44.24	&	10.70	29.87	&	32.91	6.12	$	9.19	19.13	&	-	4.57	@	-	-	-
Spondyliosoma emarginatum	-	-	-	-	-	-	7.38	7.57	&	8.46	19.76	&	20.31	14.52	*	88.40	14.56	$
Cartilaginous species																		
Haploblepharus spp.	0.17	0.27	&	0.31	0.24	*	0.13	0.17	&	0.15	0.50	&	0.31	0.35	&	0.68	0.25	*
Mustelus mustelus	0.04	-	*	0.04	0.01	*	0.03	0.01	*	-	0.01	&	0.03	-	*	-	-	-
Notorhynchus cepedianus	-	-	-	0.07	-	$	-	-	-	-	-	-	-	-	-	-	-	-
Poroderma africanum	0.21	0.10	$	0.06	0.12	&	0.06	0.05	$	0.02	0.07	&	0.05	0.13	&	0.13	-	*
Poroderma pantherinum	-	0.14	&	-	0.02	&	-	-	-	-	-	-	-	-	-	-	-	-
Triakis megalopterus	0.20	0.05	*	0.60	0.01	*	-	0.03	&	-	-	-	-	-	-	-	-	-
Overall totals																		
Number of species	11.33	9.07	$	15.77	10.27	$	21.59	11.26	$	18.70	14.57	$	18.62	13.76	$	21.29	16.28	$
Number of reef fishes	72.20	62.19	$	74.16	65.68	$	158.72	59.22	$	120.70	92.70	$	110.91	72.23	$	178.37	78.56	*
Number of soft-bottom fishes	-	0.10	&	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Number of cartilaginous fishes	0.62	0.59	*	1.07	0.42	$	0.22	0.28	&	0.17	0.59	@	0.38	0.51	&	0.81	0.25	*
Number of pelagic fishes	-	15.43	&	1.33	9.14	&	0.10	0.52	&	11.65	2.98	*	-	-	-	-	-	-
Total number of fishes	72.82	78.30	@	76.56	75.23	$	159.04	60.02	$	132.52	96.26	$	111.29	72.73	$	179.18	78.81	$
Number of seperate counts	63	136		127	264		96	235		103	176		110	73		33	19	

$ - Significantly more fishes within the reserve
* - More fishes within the reserve, but difference is not significant
@ - Significantly more fishes outside the reserve
& - More fishes outside the reserve, but the difference is not significant

Table 6.2 The relative abundance (fish.100 m^{-2}) of each of the species encountered in cold water (13 °C or colder) within and/or outside the Castle Rocks Marine Reserve, within different depth strata. The significance ($p<0.05$) of all differences was tested using the Man Whitney U test.

	0-5 m			>5-8 m			>8-12 m			>12-16 m			>16-20 m			>20-25 m		
	Inside	Outside		Inside	Outside		Inside	Outside		Inside	Outside		Inside	Outside		Inside	Outside	
Species																		
Reef teleost species																		
Boopsoidea inornata	-	0.45	&	8.31	0.31	$	17.45	2.74	$	18.07	3.01	$	10.43	1.28	$	12.58	0.90	$
Cheilodactylus fasciatus	0.33	0.84	&	0.45	2.38	&	3.24	2.57	*	1.54	3.54	&	2.39	6.33	@	3.37	6.36	&
Cheilodactylus pixi	-	-	-	-	-	-	-	-	-	0	-	-	-	0.14	&	-	0.67	&
Chirodactylus brachydactylus	-	5.41	@	3.76	2.66	*	7.93	3.22	*	0.40	2.40	@	0.28	1.18	@	0.27	1.03	&
Chirodactylus grandis	-	-	-	-	-	-	-	-	-	0.41	-	$	1.11	0.16	$	2.05	-	$
Chrysoblephus gibbiceps	-	-	-	-	-	-	-	-	-	-	-	-	0.78	-	$	0.44	-	*
Chrysoblephus laticeps	0.31	0.05	$	2.82	0.54	$	9.77	3.32	$	11.49	4.16	$	11.35	5.88	$	10.90	4.42	*
Dichistius capensis	3.72	0.13	*	0.31	0.30	*	1.05	0.70	*	0.31	0.33	&	-	-	-	-	-	-
Diplodus cervinus hottentotus	-	0.27	&	0.11	0.16	&	0.21	0.63	&	0.17	0.02	$	0.44	0.07	*	3.06	0.22	*
Diplodus sargus capensis	16.67	1.63	*	0.79	0.71	*	1.07	1.37	&	1.44	1.56	&	0.11	0.16	&	0.77	1.11	&
Gymnocrotaphus curvidens	-	1.03	&	2.49	0.17	*	4.09	1.05	*	0.52	0.27	*	0.15	0.52	&	-	0.14	&
Oplegnathus conwayi	-	-	-	0.81	-	*	0.09	0.06	*	0.10	-	*	-	-	-	-	-	-
Pachymetopon aeneum	-	-	-	-	-	-	0.34	0.32	*	0.31	0.49	&	0.21	-	*	0.31	-	*
Pachymetopon blochii	4.14	5.45	&	23.36	15.12	*	35.48	23.50	$	42.69	23.54	$	26.58	17.29	*	27.77	14.97	$
Parascorpis typus	-	0.11	&	1.19	0.27	*	0.23	0.35	&	0.37	0.53	&	0.44	0.29	*	0.99	1.83	&
Petrus rupestris	-	-	-	0.07	-	*	0.06	0.04	*	0.35	-	$	0.16	-	*	0.62	-	*
Sarpa salpa	50.23	14.66	$	6.58	15.03	&	54.26	2.24	*	14.29	9.06	*	6.01	-	*	-	-	-
Spondyliosoma emarginatum	-	-	-	0.29	0.86	&	4.49	4.38	*	7.74	6.79	*	25.08	5.76	$	3.70	26.97	&
Cartilaginous species																		
Haploblepharus spp.	-	0.08	&	0.23	0.23	&	0.09	0.17	&	0.10	0.11	&	0.05	0.10	&	0.45	0.13	*
Mustelus mustelus	-	0.04	&	-	0.02	&	-	-	-	-	-	-	-	-	-	-	-	-
Notorhynchus cepedianus	-	-	-	-	-	-	0.11	-	*	0.13	-	*	-	-	-	-	-	-
Poroderma africanum	-	0.04	&	-	0.13	&	-	0.18	&	-	0.13	&	-	0.05	&	-	-	-
Poroderma pantherinum	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Triakis megalopterus	0.60	-	$	0.37	-	$	-	-	-	-	-	-	-	-	-	-	-	-
Overall totals																		
Number of species	4.40	6.66	@	14.75	6.25	$	23.58	8.65	$	14.91	8.44	$	13.31	8.58	$	16.50	11.06	$
Number of reef fishes	75.39	30.21	*	51.35	38.51	*	139.91	46.48	$	100.22	55.70	$	87.19	40.51	$	66.82	58.89	*
Number of soft-bottom fishes	-	-	-	-	0.02	&	0.08	-	*	-	-	-	-	-	-	-	-	-
Number of cartilaginous fishes	0.94	0.24	$	2.58	0.39	$	0.30	0.35	&	0.23	0.27	&	0.05	0.25	&	0.45	0.13	*
Number of pelagic fishes	-	-	-	-	0.14	&	-	2.63	&	-	0.04	&	-	-	-	-	-	-
Total number of fishes	76.32	30.45	*	53.93	39.06	*	140.29	49.46	$	100.45	56.01	$	87.24	41.11	$	67.27	59.02	*
Number of seperate counts	16	61		38	122		16	136		16	132		16	75		16	18	

$ - Significantly more fishes within the reserve
* - More fishes within the reserve, but difference is not significant
@ - Significantly more fishes outside the reserve
& - More fishes outside the reserve, but the difference is not significant

(irrespective of water temperature), except in the 0-5 m stratum when the water was cold. At such times, that particular stratum's species richness was significantly richer on exploited compared to protected reefs (P<0.05).

Tables 6.1 & 6.2 also show that reef fish relative abundance (all species combined) was greater within, compared to outside, the reserve. This difference was significant within all but the deepest depth stratum when the water was warm (Table 6.1), but was only so within the >8-12, >12-16 and >16-20 m depth strata when the water was cold. This is likely to be due to the smaller number of counts obtained from within the reserve when the water was cold.

Table 6.1 shows that when the water was warm, some species of fishery importance (*e.g. Chirodactylis grandis*, *Pachymetopon blochii*, *Boopsoidea inornata*, *Chrysoblephus laticeps*) were significantly more abundant within the reserve ($p<0.05$) compared to outside (within the same depths stratum). Two additional species (*Petrus rupestris*, *Chrysoblephus gibbiceps*) were also always more common (but not always significantly so) within the reserve. When the water was cold, all the above-mentioned species, along with *Triakis megalopterus* and *Oplegnathus conwayi*, were more common within the reserve, although the differences were not as regularly significant (Table 6.2).

Two groups of reef fish teleost species do not appear to benefit from the protection afforded by the reserve. The first comprises species that are not targeted by the linefishery, including *Cheilodactylis fasciatus*, *C. pixi*, *Parascorpis typus* and *Spondyliosoma emarginatum*, while the second comprises species that appear to be nomadic, including important linefishery species such as *Dichistius capensis*, *Diplodus cervinus hottentotus*, *D. sargus capensis* and *Sarpa salpa*. Additionally, the reserve does not appear to benefit cartilaginous or pelagic species to any great extent (Tables 6.1 & 6.2).

Of further interest are the significantly greater relative abundances of *Chirodactylis brachydactylis* and *Cheilodactylis fasciatus* outside the reserve within some of the depth strata, particularly when the water was cold. This is interesting as these species (particularly the first - Chapter 4) are targeted by spearfishers within False Bay, and appear to be resident (pers. obs.) although this has yet to be investigated.

Figure 6.2 shows the size distributions of species encountered within and outside the reserve. It shows that the protected stocks of some important fishery species contain proportionately more large individuals than do unprotected stocks. This was particularly the case for resident species (*e.g. Chrysoblephus laticeps*, *Pachymetopon blochii*), or species suspected of being resident (*e.g. Gymnocrotaphus curvidens*). Furthermore, for many of these species, the protected population contained the largest individuals encountered. Of further interest is the fact that the protected stocks of some of the species which were not always found to be more abundant within the reserve (*e.g. Diplodus sargus capensis*, *Dichistius capensis*, *Cheilodactylis fasciatus*, *Spondyliosoma emarginatum*) still contained proportionally more large individuals than did exploited stocks. This was even the case for *Chirodactylis brachydactylis*, a species which was significantly more abundant outside the reserve within certain depth strata, especially when the water was cold.

To test whether the reserve enhances catches adjacent to its borders by exporting adult fishes, the *cpue* of non-competitive spearfishers active close to, or further away from, the reserve was compared. Within close proximity of the reserve, the *cpue* of the non-competitive spearfishers was 57.31 fish per 100 hours for all species combined (reef and pelagic). That of spearfishers active elsewhere in False Bay was virtually the same, at 57.93 fish per 100 hours. When only reef fish catches are compared, however, the *cpue* of divers active close to the

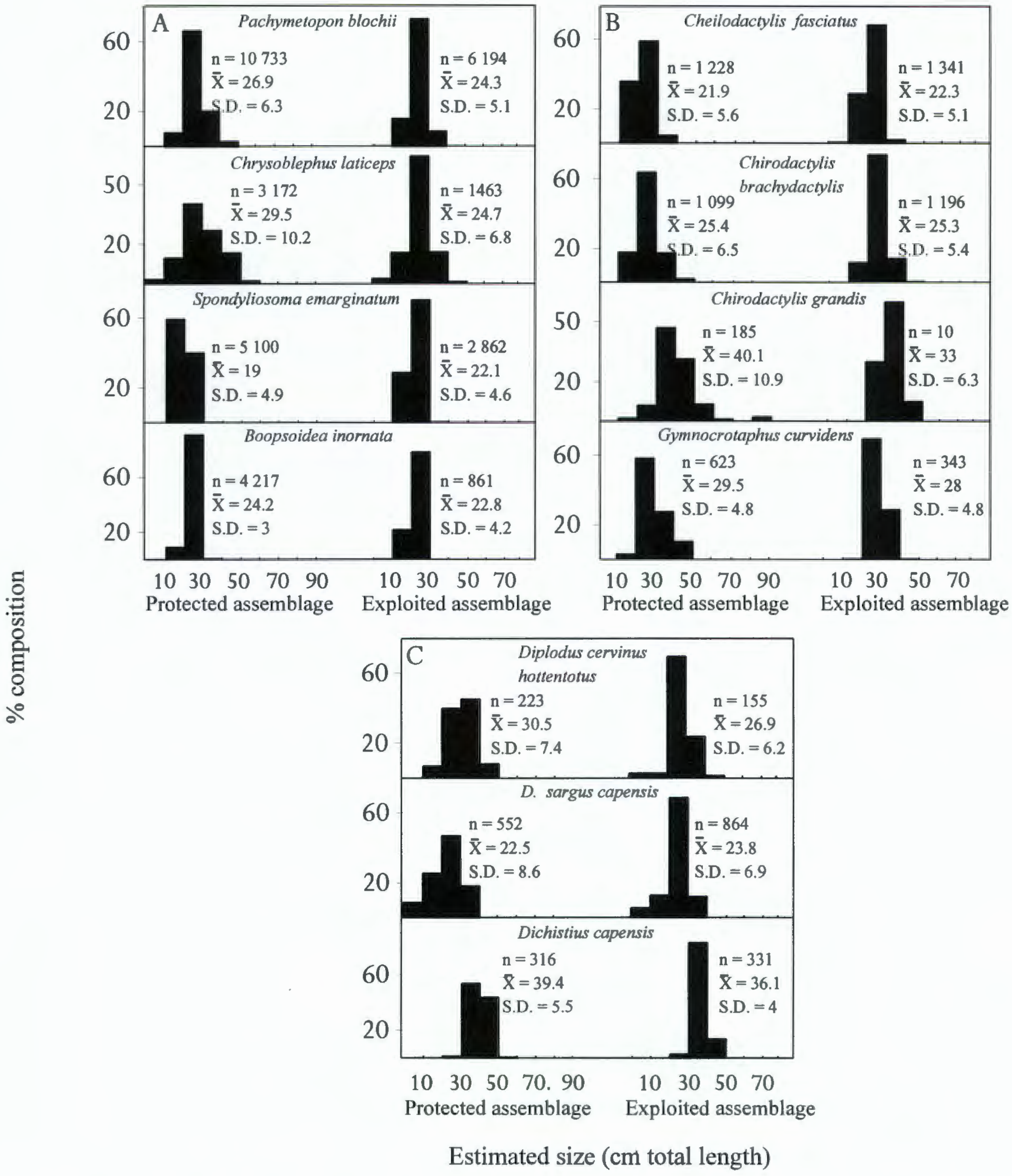


Figure 6.2 The size distribution and mean estimated size (plus standard deviation) of reef fish species of A. predominantly boat angling, B. spearfishing and C. shore angling importance that had significantly different size distributions (p<0.05) within, compared to outside, the Castle Rocks Marine Reserve. All encountered individuals of each species, irrespective of water temperature or depths, are included in the figure.

reserve (56.4 fish per 100 hours) is 7.5% higher than that of divers in the rest of False Bay (52.4 fish per 100 hours), although this difference is not significant.

Discussion

Many articles support the fact that marine reserves are potentially beneficial to fishers and fishery managers (*e.g.* Roberts & Polunin 1991, 1993; Bohnsack 1996a, b; Attwood *et al.* 1997a, b, c). As in other parts of the world, marine reserves have been viewed as a potentially useful linefish management tool in False Bay, as their protected stocks, by exporting eggs and larvae and some adult fishes, may enhance recruitment and fisheries catches on adjacent exploited reefs. Furthermore, not being a species-specific management tool, marine reserves are a useful addition to traditional fishery management measures, since because their regulations apply to all fishery sectors, they can resolve conflict between competing fishery sectors controlled by different fishery regulations (Bohnsack 1996b). Chapter 5 of this thesis, which discussed the catch composition and management regulations enforced on the different False Bay linefishery sectors, discussed how considerable this conflict can be.

This study supports the fact that fish assemblages inhabiting marine reserves are usually more abundant, and contain a higher proportion of large fishes, than do exploited fish stocks (Attwood *et al.* 1997a). In the case of the Castle Rocks Marine Reserve, it is proposed that this is due to the availability of reef attributes (*e.g.* blinders, pinnacles rising from deep water, large caves and crevices) that characterise good quality reefs in False Bay, and to the depth to which reefs extend. The reserve thus contains the small-scale variations in physical conditions that Emanuel, Bustamante, Branch, Eekhout & Odendaal (1992) deem necessary for reserves to be beneficial. The importance of reef quality in affecting reserve benefits will, however, be investigated further in Chapter 7.

Although Beuttler (1994) reports that there are benefits to keeping marine reserves small, there are also disadvantages to this practice. This study highlights that the Castle Rocks Marine Reserve's small size is an important shortcoming. It is only successful at protecting resident species such as *Chrysoblephus laticeps* and *Pachymetopon blochii*, while nomadic ones such as *Dichistius capensis*, *Diplodus cervinus hottentotus*, *D. sargus capensis* and *Sarpa salpa* do not appear to benefit to the same extent. This nomadic lifestyle explains why these species do not benefit optimally from the protection afforded to them by the Castle Rocks Marine Reserve, as only when they are within its borders are they protected. Since their movement patterns take them out of the reserve at other times, they are vulnerable to fishers. However, nomadic species such as *Dichistius capensis* can benefit from marine reserves, as Bennett (1988), and Attwood & Bennett (1994) report that the De Hoop Marine Reserve is beneficial for that species. However, the De Hoop Marine Reserve is 42.5 km longer than the Castle Rocks Marine Reserve, and is thus large enough to protect most individuals throughout their life, unlike the Castle Rocks Marine Reserve.

However, it is noteworthy that the size distribution of *Dichistius capensis*, *Diplodus cervinus* hottentotus and *D. sargus capensis* (which are thought to be nomadic) encountered within the reserve contained proportionally more large individuals than did exploited populations. This may be due to the protection provided by the reserve, although it is likely that chance also played an important role. This is supported by Tables 6.1 & 6.2, which show that these species were not always more abundant within the reserve. It is proposed that these species are protected to a similar extent as are the other potentially 'nomadic' species encountered (*e.g. Sarpa salpa*, *Pachymetopon aeneum*, *Petrus rupestris* and *Oplegnathus conwayi*). Judging from its distribution in the water column (usually more than 1 m above the reef) and random swimming direction, *Chirodactylis grandis* also appears to be a nomadic species. However, Tables 6.1 & 6.2 show that the species was significantly more abundant within the reserve compared to outside. Similarly, Figure 6.2 shows the size distribution of *Chirodactylis grandis* within the reserve to include significantly more large individuals than the exploited population. These results are similar to those obtained for the *Chrysoblephus laticeps*, and point to the possibility of *Chirodactylis grandis* (of the size seen within False Bay) being resident. Very little is known about the movement patterns of this species, however.

The higher relative abundances and larger mean sizes of fishes within the reserve highlight the fact that the protected populations of some species (especially resident ones) could enhance recruitment on adjacent exploited reefs by exporting eggs and/or larvae. This was not investigated, and has rarely been investigated worldwide. However, Tilney *et al.* (1996) investigated the potential of reef sparids within the Tsitsikamma National Park seeding areas adjacent to the park with larvae, and report that this is taking place. They also discuss the importance of the vertical distribution of larvae in the water column, highlighting that should larvae move to the surface layers, their dispersal distance would be increased substantially. It is likely that if the Castle Rocks Marine Reserve does enhance recruitment on exploited reefs, it is the resident species (*e.g. Chrysoblephus laticeps*, *Pachymetopon blochii*) that are likely to benefit the most, as these species are significantly more abundant, and larger, within the reserve. However, it has to be investigated whether the reserve in question is actually positioned optimally to enhance recruitment on adjacent reefs. This involves studying the geographical location of the reserve with respect to current patterns in the area (Attwood *et al.* 1997a). The seeding of adjacent reefs by a marine reserve (the source) will only be possible if the currents of the area export recruits to areas which lie within the geographical distribution of the species concerned (the sink). If this is not the case, the reserve will not enhance recuitment on exploited reefs, and thus be less beneficial to fisheries (Rowley 1994). As highlighted by Tilney *et al.* (1996), and Attwood *et al.* (1997a), the potential success of the Castle Rocks Marine Reserve in seeding exploited reefs within False Bay also depends on the larval behaviour of the species concerned. Based on the different False Bay circulation patterns reported by Atkins (1970b), it is proposed that if eggs and larvae remain close to the bottom and larvae settle within ten days of hatching the reserve could seed exploited False Bay reefs. On the other hand, should these larvae swim to the surface, and remain there for a few weeks, it is likely that they would be unable to recruit on appropriate inshore habitat, as they would be transported too far offshore by this time. However, since little is known concerning the larval dispersal of most South African reef fish species, much work still needs to be undertaken in this field. Such work is of great importance since understanding the physical/biological factors that affect the successful settlement of larvae could allow for more effective management of linefish populations through the use of optimally located marine protected areas (Botsford, Wing & Largier 1998). The fact that the Castle Rocks Marine Reserve is situated at the very westerly edge of the distribution range of most of the reef fishes found within it (van der Elst 1988), however, is likely to limits the reserve's potential in enhancing recruitment to reefs west of it. It appears, therefore, that it is not a foregone conclusion that the Castle Rock Marine Reserve seeds adjacent exploited reefs with recruits.

Another important point that needs to be discussed is the viability of the protected stocks located within the Castle Rocks Marine Reserve (Attwood *et al.* 1997a). Should the reserve's populations not be self-sustaining, the reserve will be of limited use to fishery managers and the region's linefishery as its protected stocks would not provide an insurance against a fishery collapse outside of the reserve. Nor would it provide an insurance against pollution threats to the whole region (Attwood *et al.* 1997a, b). Protected resident linefish populations are likely to be viable within the Tsitsikamma National Park and De Hoop Marine Reserve, as these protected areas are more than 40 km long. However, because the Castle Rocks Marine Reserve is only 3.25 km long, this is less likely to be the case, and needs to be investigated further.

Another way in which a marine reserve can enhance fisheries outside its borders is by exporting adult fishes (Roberts & Polunin 1991, 1993; Bohnsack 1996a; Attwood *et al.* 1997a, b). The comparison of the *cpue* of non-competitive spearfishers active within close proximity to, or further away from, the Castle Rock Marine Reserve showed that, although the reserve could be exporting some adult reef fishes to exploited reefs, this benefit is not substantial. Although this result could be affected by different spearfishing effort within the two reef areas, this is unlikely to be the case. Like all other popular spearfishing sites within False Bay, reefs close to the reserve are visited by spearfishers because of their potential fish-holding capacity. None of the spearfishers encountered close to the reserve spearfished there due to the presence of the reserve. Instead, the apparent lack of substantial fish export from the reserve is likely to be related to the movement patterns of the linefish species concerned, and its small size. For example, an individual *Chrysoblephus laticeps* is resident (Penrith 1972; van Rooyen 1988; Buxton & Allen 1989) and thus unlikely to move outside the reserve if it has recruited within it. The same is also likely to apply to other resident species such as *Pachymetopon blochii* and potentially *Gymnocrotaphus curvidens*. The negative effect of the reserve's small size is best illustrated by comparing the *Dichistius capensis* conservation success of this small reserve to that of the De Hoop Marine Reserve. Bennett & Attwood (1991, 1993b), and Attwood & Bennett (1994) report that the De Hoop Marine Reserve is successful in conserving *Dichistius capensis* stocks in the Southern Cape, and that it exports sufficient *D. capensis* individuals to enhances fisheries close to its borders. Compared to this, the present study found that the Castle Rocks Marine Reserve does not conserve *Dichistius capensis* stocks optimally, in all likelyhood due to its limited size, and is thus unlikely to enhance *Dichistius capensis* catches outside of its borders by exporting excess specimens. The finding that the Castle Rocks Marine Reserve does not enhance adjacent fisheries substantially by exporting adult fishes is further supported by the fact that reefs situated within the close-to-the-reserve area are not fished regularly by commercial or recreational boat anglers, nor by shore anglers (pers. obs.).

However, the comparison of non-competitive spearfishers' *cpue* only tested for the export of large fishes from the reserve. It did not test the possibility that the reserve could be exporting sub-adult fishes, as small fishes are not targeted by spearfishers. It is, therefore, possible that the reserve does enhance fish stocks on adjacent exploited reefs by exporting such fishes from its borders, although if it does, this benefit is likely to be very localised.

The Castle Rocks Marine Reserve encompasses quality reefs that provide most adult reef fishes with their reef requirements. However, it also appears to encompass habitats required by some species' juveniles (*e.g. Chrysoblephus laticeps*, *Pachymetopon blochii*, *Parascorpis typus*, *Diplodus sargus capensis*). It is likely that recruits of other species also inhabit the very shallow waters (<50 cm deep) of the reserve, although no fish counts or collections were undertaken there. The reserve, therefore, encompasses habitats used by certain reef fish species as nursery areas. However, juveniles of most of the species that inhabit the reserve are also likely to recruit to areas outside the reserve. Indeed, some species (*e.g. Dichistius* *capensis*, *Diplodus sargus capensis*) are likely to recruit mostly outside of the reserve, as juveniles of these are common in the surf zone along the northern shore beaches of False Bay (Clark *et al.* 1994a). Overall, however, the limited size of the Castle Rocks Marine Reserve is likely to limit the benefits it provides as a nursery area, as only a limited number of recruits are protected within its borders. This is a less important shortcoming of the reserve, however, as long as juvenile fishes can recruit elsewhere in the Bay, and sufficient recruits settle within the reserve to self-sustain its reef fish assemblage. Furthermore, juvenile reef fishes are not landed in large numbers by the False Bay linefishery.

Overall, the results of this study indicate that within False Bay, a small marine reserve is capable of providing protection to resident linefish species such as *Chrysoblephus laticeps* and *Pachymetopon blochii*. Both these species were significantly more abundant within, compared to outside, the reserve, and protected populations comprised significantly more large individuals. The reserve also appears to be used by some species as a nursery area. However, due to its small size, nomadic species do not appear to benefit substantially from the Castle Rocks Marine Reserve, while its nursery function is likely to also be very limited. Finally, the reserve's small size results in concerns about whether the linefish stocks within its borders are self-sustaining.

Attwood & Bennett (1995a) report that in South Africa, the establishment of numerous small marine reserves is likely to be more feasible than the establishment of fewer large ones in the future. This is because small reserves have numerous advantages over large ones. Amongst others, they enjoy local public support, are easy to establish, manage and enforce, and also provide good research opportunities (Beuttler 1994; Attwood *et al.* 1997b). Small reserves are thus not a waste of conservation effort. Indeed, Attwood *et al.* (1997b) report that a combination of small and large marine reserves is optimal for the protection of the environment and linefish species, and to maintain fisheries. Attwood & Bennett (1995a), however, question whether these small reserves will achieve the same conservation success as those achieved by the large De Hoop Marine Reserve and Tsitsikamma National Park. To maximise the protection that can be afforded to nomadic species by the Castle Rocks Marine Reserve, it is proposed that it should be increased in length, although it is difficult to know by how much (Attwood *et al.* 1997b). To maximise its chance of significantly improving the protection afforded to such nomadic species as *Dichistius capensis* and *Diplodus sargus capensis*, a reserve should be sufficiently long to contain enough individuals of nomadic species throughout their movements. Tagging studies could elucidate the movement range of these species, although such studies are time consuming and require substantial effort (Attwood *et al.* 1997a). However, Attwood & Bennett (1995a) propose that the protection that *Dichistius capensis* and *Diplodus sargus capensis* receive from marine reserves could be increased by increasing the total marine reserve area in the region. Within False Bay, this could be achieved by establishing additional small marine reserves and re-locating some of the present ones to more appropriate areas. Furthermore, shore angling should be forbidden within reserves as *D. capensis* and *Diplodus sargus capensis* are landed primarily by this sector (Chapter 5).

Chapter 5 discussed the inefficient enforcement of fisheries regulations within False Bay, and indeed the Southwestern Cape. The ease of marine reserve regulation enforcement (Bohnsack 1996a; Attwood *et al.* 1997a, b) and the other advantages discussed earlier thus makes the use of marine reserves an attractive management option in the region. The poaching of protected resources within marine reserves, however, is still occurring in South Africa (Brouwer *et al.* 1997), especially in the Southwestern Cape (Attwood *et al.* 1997a, b).

Anglers are often tempted to fish within marine reserves because these contain more and larger fishes. Some of these anglers are ignorant of the effect of their actions (Attwood *et al.* 1997a, b) as they do not realise how important protected assemblages are to their fishery. Claverie (1990), Taylor (1993b), Attwood *et al.* (1997b, c), and Sauer *et al.* (1997) propose that such anglers should have the importance of protected stocks explained to them, as once they understand the use of marine reserves as a fishery management tool, they will understand how beneficial they are. This could not only potentially stop many of them from poaching within South African marine reserves, but could also result in some unofficially enforcing reserve regulations on other anglers by peer pressure. Claverie (1990) maintains that such 'peer pressure enforcement' could be a very important source of enforcement. In addition to this type of enforcement, Attwood *et al.* (1997a) propose that 'interpretative enforcement' (Causey 1995), where a transgressor is educated about the reason for regulations if these are not known. Bohnsack (1996b) proposes that as more and more people become familiar with marine reserves, they are likely to be more easily accepted. However, a combination of 'interpretative' and 'peer pressure enforcement' is unlikely to put an end to all poaching. Other anglers poach within marine reserves to make money (Attwood *et al.* 1997b), even though they may know this to be wrong (Attwood *et al.* 1997a). For example, Attwood *et al.* (1997b) mention that the poaching of reef fishes is a cause of concern within the Castle Rock Marine Reserve. This is supported by the fact that commercial boats were seen catching reef fish within the reserve during the present study, especially when weather and sea conditions were bad enough to prohibit them from angling further offshore. All these anglers were likely to be commercial anglers, as they were active off large deck boats similar to those based at Kalk Bay. No ski-boat anglers were ever seen targeting reef fishes inside the reserve. When confronted by the author at sea, the transgressing anglers explained that they were actually targeting *Thyrsites atun*, which they pointed out was legal. To make matters worse, in False Bay, intentional poachers usually escape prosecution because as Attwood *et al.* (1997b, c) report, enforcement agencies in the area are understaffed, lack marine trained specialists, and do not have sea-going vessels. The poor demarcation of marine reserve's sea boundaries is also a major problem that needs to be overcome (Attwood *et al.* 1997c). Although the poaching level within the reserve is unquantified, however, the benefits provided by the reserve to highly resident species are still obvious, highlighting that poaching is low enough to not impact these species substantially. This should not be used as a reason to ignore the problem, however, as poaching protected stocks limits the management benefits that such stocks can provide to fisheries and managers in the future. Furthermore, future poaching activities will only be deterred if present poachers are arrested and prosecuted. The following propositions could help to minimise the impact of poachers:

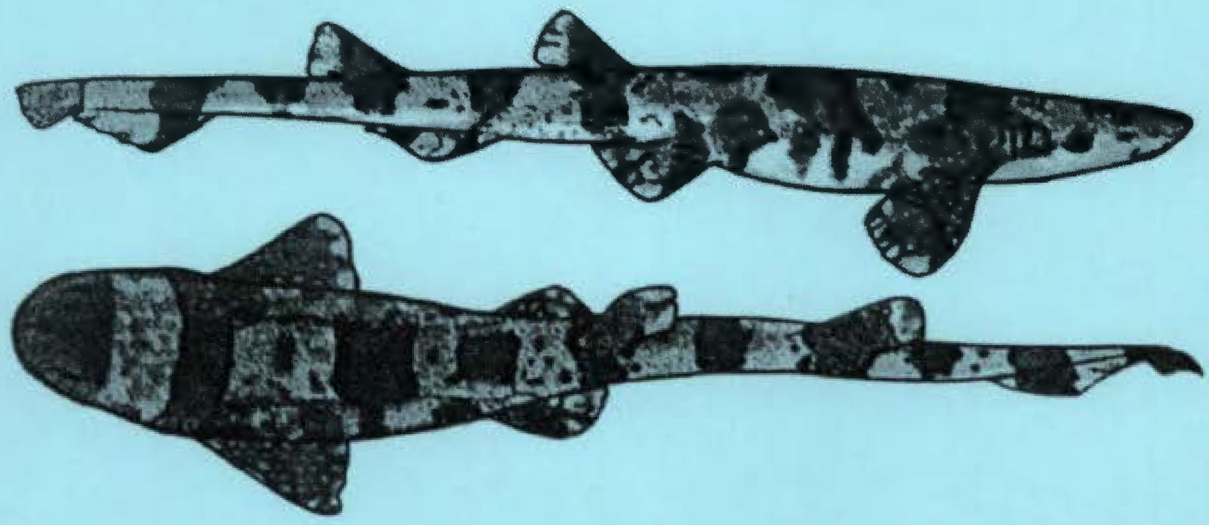
- The staff of marine reserve regulation enforcement agencies should be increased (Brokensha 1990; Winch 1990b; Coetzee 1993; Attwood *et al.* 1997b, c).
- One or more sea-going crafts should be available to enforce regulations at sea, along with more vehicles for land based enforcement (Brokensha 1990; Attwood *et al.* 1997b, c).
- Arrested poachers should be very severely dealt with, providing a more effective deterrent (Winch 1990b; Coetzee 1993).

These propositions should make reserve regulation enforcement more efficient in the region. This is necessary if marine reserves are to adequately protect the resources found within them, and thus be successful fishery management tools (Clark 1996; Attwood *et al.* 1997b).

In the case of the Castle Rocks Marine Reserve, it is proposed that to put an end to the poaching of reef fishes by boat anglers, the targeting for *Thyrsites atun* within the reserve should be banned. Even though this is likely to result in intense pressures from the fishing industry for access to this species within the reserve (Attwood *et al.* 1997c), this would make the reserve's regulations that much easier to enforce, rendering the excuse that anglers are targeting *Thyrsites atun* while actually targeting reef fishes within the reserve useless, and minimising the temptation of angling within the reserve altogether. Finally, marine reserve regulation enforcement must be improved in the region if the benefits of marine reserves are to be maximised. As discussed by Attwood *et al.* (1997a, c), the enforcement of marine reserve regulations must be an integral part of marine reserve management (Hemming & Pierce 1996) in False Bay, not an afterthought.

Chapter 7

Does the quality of reef within a small marine protected area affect its value as a fishery management tool? A case study from False Bay, South Africa



Haploblepharus edwardsii

Introduction

The benefits of marine reserves are numerous. Chapter 6 highlighted that by protecting linefish species and their environment, marine reserves may be a useful addition to the traditional single-species linefish management measures including minimum sizes, daily bag limits and closed seasons. Reserves are also a valued resource for scientists, particularly those who study linefish management measures (Buxton 1995; Attwood *et al.* 1997a). This is because the protected fish assemblage can be compared to an exploited assemblage, allowing for fishing effects to be investigated and management measures changed, should this be required (Jennings & Lock 1996). Marine reserves can also benefit linefisheries in adjacent areas by exporting eggs and larvae and/or adult fishes. Furthermore, marine reserves can promote and support recreational activities such as SCUBA diving and underwater photography, while also potentially becoming important ecotourism sites (Buxton 1995; Ward 1996; Attwood *et al.* 1997a). Marine reserves are also favoured by law enforcement agencies, as reserve regulations are easy to enforce (Bohnsack 1990). In the Southwestern Cape, this is an important benefit, as fishery regulations enforcement is very inefficient in the region (Attwood *et al.* 1997a, b; Brouwer *et al.* 1997; Lamberth & Griffiths 1997; Sauer *et al.* 1997).

Numerous marine reserves of different sizes have been created in South Africa (Payne & Crawford 1989; Attwood *et al.* 1997c), some with the above-mentioned advantages clearly in mind, although in many cases the proposed functions are unclear or vague (Attwood *et al.* 1997c). Some reserves, such as the De Hoop Marine Reserve, Tsitsikamma Coastal National Park, St Lucia Marine Park and Maputuland Marine Reserve, are longer than 45 km (Bennett & Attwood 1991; Burger 1991; Chater *et al.* 1993), but many, like some situated within False Bay, are very short (<5 km).

As pointed out in Chapter 6, two existing large marine reserves (De Hoop Marine Reserve and Tsitsikamma Coastal National Park) have been shown to be beneficial in conserving linefish species (Buxton & Smale 1989; Bennett & Attwood 1991, 1993b), while the De Hoop Marine Reserve has been shown to enhance adjacent fisheries by exporting adult fishes (Bennett & Attwood 1993b). It is, however unlikely that additional large marine reserves will be supported by the general public in most areas. The fact that the use of small marine reserves is socially more acceptable may explain why Attwood *et al.* (1997b) report that the establishment of numerous small marine reserves will be more feasible than the establishment of fewer large ones in the future. Chapter 6 highlighted that resident species such as *Pachymetopon blochii* and *Chrysoblephus laticeps* can benefit from such small reserves. However, being small, these reserves are unlikely to contain the physical reef requirements of all the species in the region. This begs an important question. Is the abundance of reef fishes influenced by reef architecture? The answer to this question is important for two reasons. Firstly, if it is not, small marine reserves can be located on the basis of social/political criteria. However, if reef architecture is important, reserves must be cited on the basis of reef characteristics, and current marine protected areas that are not suitable should be relocated to more appropriate sites.

Two small, similarly sized marine protected areas, the Glencairn Marine Protected Area – GMPA, and the Castle Rocks Marine Reserve – CRMR are located within False Bay. Although the two are situated within 15 km of each other (Figure 1.1), their reefs differ greatly in architecture. This chapter compares the reef fish assemblages inhabiting the CRMR and GMPA. Based on the results from Chapter 2, it is predicted that the fish assemblages inhabiting the two sites will vary on the basis of different reef structure. The suitability of both sites as marine reserves is discussed, along with the importance of considering reef architecture when selecting the location of future small marine reserves.

Methods

The fish assemblages at the two sites were investigated using the stationary point count method described in Chapter 1. Each count was categorised into one of the depth zones described in Chapters 1 & 2 (0-5, >5-8, >8-12, >12-16, >16-20 and >20-25 m). Reef architecture within each count's search area was classified as follows. Reef elevation was estimated in metre intervals, and each count then categorised within the same elevation ranges employed in Chapters 1 & 2 (0-1, >1-3, >3-5, >5-8, >8-12 and >12-16 m). The size of any available caves was classified as either large (if at least one cave was deep and had more than one exit) or small (if no or only shallow caves with only one exit were present). Data for the CRMR was collected between March 1993 and March 1995, and that for the GMPA between March and June 1995. A χ^2 Goodness of Fit test (Zar 1984) was used to compare the frequency of counts obtained where reef with different cover (large or small cave) and elevation was encountered. During the test, the observed frequencies were those obtained within the GMPA, while the predicted ones were those obtained within the CRMR. Since Chapter 2 emphasised how water temperature affected reef fish distribution patterns, only counts obtained in water 11-13 °C were compared.

The fish assemblages encountered within the two sites were compared to investigate the importance of reef architecture in affecting reef fish community structure. Because the censuses completed within each site were undertaken in 'cold' (11-13 °C) water (Chapter 2), both sites are nearby, affected by similar current patterns, and have similar wave exposures and water temperature ranges, it was assumed that the differences in reef profile and architecture at each site would alone explain differences in fish assemblage.

Overall reef fish (all reef fishes combined), individual species', and species richness (number.100 m^{-2}) were compared within the different depth strata between the two reserves. Because data collected within the CRMR included counts obtained within two depth strata (>16-20 and >20-25 m) within which no reefs were found in the GMPA, the overall species composition and fish relative abundance (for all counts obtained) of both sites was also compared. Replicates were obtained for each depth zone at each site (and for each site as a whole) by adding successive separate counts together (irrespective of reef elevation and presence of caves) until the censused reef surface area totalled 144-252 m^2. As many replicates as available data allowed were calculated. Comparisons were undertaken using the non-parametric Mann-Whitney U Test, as not all groups of relative abundance estimates were normally distributed (Zar 1984). The total number of species encountered at each site was also noted.

The size distributions of 18 species encountered within each protected area (irrespective of depth) were compared using the non-parametric Kolmonogov-Smirnof Test (Zar 1984).

Results

A total of 299 counts covering 11 822 m^2 of reef were obtained from the GMPA and 352 counts covering 11 218 m^2 from the CRMR. Reef was encountered within all six depth strata in the CRMR, but only within the shallower four within the GMPA. Furthermore, the reefs encompassed within each protected area differed in their architecture. Table 7.1 shows that within the four depth zones encompassing reef in the GMPA, significantly more counts ($p<0.01$) encompassing large caves were obtained within the CRMR than GMPA (each occurrence is one count, irrespective of the number of caves encompassed within its search area). The same result (and significance level) was found when all counts obtained within each protected area (irrespective of depth) were included in the analysis.

Table 7.1 Breakdown of the number, and % breakdown, of counts obtained within the Glencairn Marine Protected Area (GMPA) and Castle Rocks Marine Reserve (CRMR), which covered reefs with no or only small caves (small cave) or at least one large cave (large cave). A Chi-square Goodness Of Fit Test was used to test the significance of differences (df = 1). Note: the predicted frequencies were those obtained within the CRMR and the observed ones those within the GMPA.

Depth stratum	Cave size	CRMR (Numbers)	GMPA (Numbers)	CRMR (% number)	GMPA (% number)	Significance level (*)
0-5m	Small	28	40	56	93	$P<0.01$
	Large	22	3	44	7	21.552
5-8m	Small	34	52	61.8	72.2	$P<0.01$
	Large	21	20	38.2	27.8	9.577
8-12m	Small	34	108	38.2	63.9	$P<0.01$
	Large	55	61	61.8	36.1	161.713
12-16m	Small	23	12	37.7	80	$P<0.01$
	Large	38	3	62.3	20	37.498
All depths	Small	148	212	42.2	70.9	$P<0.01$
	Large	203	87	57.8	29.1	93.961

* - Value of Chi-square

Table 7.2 shows the breakdown of the frequency of reef elevations encountered at the two sites. In the GMPA, the reef encompassed within each depth zone is significantly flatter ($p<0.01$) than that within the CRMR. The same held ($p<0.01$) if all the counts obtained within each protected area were included in the analysis. Based on the results from Chapter

Table 7.2 Breakdown of the number and % breakdown of counts that covered reefs with different reef elevation within the Castle Rocks Marine Reserve (CRMR) and Glencairn Marine Protected Area (GMPA). A Chi-square Goodness Of Fit Test was used to test the significance of differences. Note, the frequencies obtained within the CRMR were used as the predicted frequencies, while those from within the GMPA were the observed ones.

Depth stratum	Reef elevation (m)	CRMR (Numbers)	GMPA (Numbers)	CRMR (% number)	GMPA (% number)	Significance level (*)
0-5 m	0-1	20	32	40	72.7	*P*<0.01
	>1-3	11	12	22	27.3	df = 2
	>3-5	19	-	38	-	26.291
>5-8 m	0-1	15	27	27.3	37.5	*P*<0.01
	>1-3	17	42	30.9	58.3	df = 3
	>3-5	8	3	14.5	4.2	64.49
	>5-8	15	-	27.3	-	
>8-12 m	0-1	21	74	23.6	44.1	*P*<0.01
	>1-3	29	82	32.6	48.8	df = 4
	>3-5	26	12	29.2	7.1	251.162
	>5-8	10	-	11.2	-	
	>8-12	3	-	3.4	-	
>12-16 m	0-1	13	12	21.3	80	*P*<0.01
	>1-3	20	2	32.8	13.3	df = 5
	>3-5	18	1	29.5	6.7	42.333
	>5-8	3	-	4.9	-	
	>8-12	2	-	3.3	-	
	>12-16	5	-	8.2	-	
All depths	0-1	83	145	23.6	48.5	*P*<0.01
	>1-3	112	138	31.9	46.2	df = 5
	>3-5	100	16	28.5	5.4	178.404
	>5-8	42	-	12	-	
	>8-12	9	-	2.6	-	
	>12-16	5	-	1.4	-	

* - Value of Chi-square

2, therefore, the reef within the GMPA is of poorer quality than that within the CRMR, with respect to the requirements of reef fishes.

A total of 1 955 fishes from 21 species (18 reef fish species) were counted within the GMPA, and 7 174 fishes from 23 species (21 reef fish species) were observed within the CRMR. Within all depth zones, species richness, and the relative abundances of reef teleosts, soft-bottom teleosts and cartilaginous fishes were all greater in the CRMR than in the GMPA, these differences being significant ($p<0.05$) in the case of species richness and reef teleosts relative abundance (Table 7.3). Of the 27 species observed, 13 were more abundant within the CRMR across each of the four depth ranges covered within the GMPA. The relative abundance differences of three of these (*Pachymetopon blochii*, *Boopsoidea inornata* and *Triakis megalopterus*) were significant ($p<0.05$).

Irrespective of depth, species richness within the CRMR (4.1 ± 1.3) was significantly greater ($p<0.01$) than within the GMPA (2.2 ± 1), while overall, 23 species were encountered within the CRMR compared to 21 within the GMPA. Reef teleosts relative abundance within the GMPA (12.5 ± 14.9 fish.100 m^{-2}) was significantly lower ($p<0.01$) than within the CRMR (65.8 ± 45.5 fish.100 m^{2}). The relative abundances of soft-bottom teleost and cartilaginous fish were also greater within the CRMR, although the difference was not significant. Pelagic teleosts were the only fish group more abundant within the GMPA, although this difference was not significant.

When relative abundances were compared irrespective of depth, 19 species were more abundant (13 significantly so) within the CRMR, while seven species were more abundant (but only one significantly so) within the GMPA. Only two of the latter species were reef teleosts, however. Five species (*Chirodactylis grandis*, *Petrus rupestris*, *Notorhynchus*

Table 7.3 The mean and standard deviation (value in brackets) of the relative abundance (number/100 m^{-2}) of species, groups of species, and individual species encountered within different depth strata in the Glencairn Marine Protected Area (GMPA) and Castle Rocks Marine Reserve (CRMR). Differences were tested for significance using the Mann-Whitney U test.

	0-5 m			>5-8 m			>8-12 m			>12-16 m			All depths		
Species	GMPA	CRMR		GMPA	CRMR		GMPA	CRMR		GMPA	CRMR		GMPA	CRMR	
Teleosts															
Reef fishes															
Boopsoidea inornata	0	0	$	0.1 (0.4)	4.1 (5.9)	$	0	8.1 (7)	$	0	10.8 (7.7)	$	<0.1	7.5 (7.1)	$
Cheilodactylus fasciatus	0.4 (0.6)	4.4 (7.7)	#	0.7 (0.7)	0.4 (0.6)	*	0.7 (0.9)	0.8 (0.9)	#	0.1 (0.2)	1.4 (0.9)	$	0.6 (0.8)	1.6 (2.6)	$
Chirodactylus brachydactylus	0.3 (0.3)	6.1 (9.6)	#	1.4 (1.4)	0.7 (0.8)	*	1 (1.1)	1.3 (1.2)	#	0.2 (0.3)	0.2 (0.3)		1 (1.1)	1.1 (3.2)	#
Chirodactylus grandis	0	0		0	0		0	0.1 (0.3)	#	0	0.5 (0.5)	#	0	0.4 (0.7)	$
Chrysoblephus gibbiceps	0	0		0	0		<0.1	0	*	0	0		<0.1	0.2 (0.5)	#
Chrysoblephus laticeps	0	0.6 (0.2)		1.8 (1.5)	2.3 (1.4)	#	7 (4.8)	6.1 (3.4)	*	5.3 (5.9)	8.6 (5.1)	#	4.8 (4.8)	6 (4.7)	#
Dichistius capensis	1.5 (4)	1.7 (3.5)	$	0	0.3 (0.3)	$	0	0.2 (0.3)	$	0	0.6 (1.3)	#	0.2 (1.5)	0.4 (1.2)	$
Diplodus cervinus hottentotus	0	0		0	0.2 (0.3)	#	0.7 (2.3)	0.2 (0.4)	*	0	0.1 (0.2)	#	0.4 (1.8)	0.3 (0.8)	*
Diplodus sargus capensis	0	8.6 (9.1)	$	0.1 (0.3)	1 (1.5)	#	1 (3.7)	1.8 (2.9)	$	0	1 (2)	#	0.6 (2.9)	1.7 (3.8)	$
Gymnocrotaphus curvidens	0	1.6 (2.2)	#	0.5 (1)	0.6 (1.3)	#	0.1 (0.2)	0.6 (1.1)	$	0	0.4 (0.7)	#	0.1 (0.5)	0.5 (1.1)	$
Oplegnathus conwayi	0	0		0	0.1 (0.5)	#	<0.1	0.1 (0.3)	#	0	0.1 (0.1)	#	<0.1	0.1 (0.3)	#
Pachymetopon aeneum	0	0		0.1 (0.3)	0	*	0	0.2 (0.3)	$	0	0.1 (0.3)	#	<0.1	0.1 (0.3)	$
Pachymetopon blochii	0.3 (0.6)	3.8 (1)	$	1.5 (2.3)	25.9 (22.6)	$	1.9 (2.2)	22.2 (14.7)	$	1 (1.1)	24.9 (14.2)	$	1.6 (2.1)	24 (18.9)	$
Parascorpis typus	0	0.2 (0.3)	#	0.6 (0.6)	1 (1.1)	#	0.9 (1.5)	0.2 (0.3)	*	0.8 (1.3)	0.3 (0.3)	*	0.7 (01.2)	0.5 (0.7)	*
Petrus rupestris	0	0.1 (0.2)	#	0	0.1 (0.2)	#	0	0.4 (0.5)	$	0	0.1 (0.3)	#	0	0.2 (0.4)	$
Sarpa salpa	0	39.6 (69)	$	0	6.3 (15.2)	#	2.9 (11.8)	21 (44)	#	0	8.1 (24.2)	#	1.7 (9.1)	13.1 (34.1)	$
Spondyliosoma emarginatum	0	0		0.1 (0.4)	0.3 (1)	#	1.1 (2.5)	6.1 (11.3)	$	1.1 (1.8)	8.6 (4.8)	$	0.7 (2)	7.5 (11.8)	$
Soft-bottom fishes															
Amblyrhynchotes honkenii	0	0		0	0		<0.1	0	*	0	0		<0.1	0	*
Rhabdosargus globiceps	0	0		0	0		0	<0.1	#	0	0		0	<0.1	#
Pelagic fishes															
Liza richardsonii	5.8 (16.4)	0	*	0	1.6 (5.9)	#	0	0		0	0		0.8 (6)	0.4 (2.9)	*
Sardinops sagax	0	0		0	0		2 (12.1)	0	*	25.8 (44.8)	0	*	2.5 (13.6)	0	*
Cartilaginous fishes															
Haploblepharus edwardsii	0.5 (0.6)	0.1 (0.2)	*	0.3 (0.5)	0.2 (0.2)	*	0.3 (0.4)	<0.1	@	0	0.1 (0.1)	#	0.3 (0.5)	1 (0.2)	@
Notorhynchus cepedianus	0	0		0	0		0	0.2 (0.3)	$	0	0.1 (0.2)	#	0	0.1 (0.2)	$
Poroderma africanum	0.1 (0.2)	0.5 (1.2)	#	0	0		0.1 (0.3)	<0.1	*	0	0		<0.1	0.1 (0.4)	#
Poroderma pantherinum	0.1 (0.2)	0	*	0	0		0	0		0	0		<0.1	0	*
Triakis megalopterus	0	0.5 (0.3)	$	0	1 (1.3)	$	0	0.1 (0.2)	$	0	0		0	0.3 (0.8)	$
Overall relative abundances															
Species	1.3 (0.5)	3.2 (0.8)	$	2.3 (1)	3.9 (1.7)	$	2.4 (1)	4.2 (1.2)	$	1.3 (1.2)	4.3 (1.5)	$	2.2 (1)	4.1 (1.3)	$
Reef teleosts	2.5 (3.7)	66.6 (69.4)	$	6.8 (4.7)	43.3 (32.7)	$	17.2 (17.3)	69.3 (52.3)	$	8.4 (10.2)	65.8 (36.1)	$	12.5 (14.9)	65.1 (45.5)	$
Pelagic teleosts	5.8 (16.4)	0	*	0	1.6 (5.9)	#	2 (12.1)	0	*	25.8 (44.8)	0	*	3.3 (14.7)	0.4 (2.9)	*
Soft-bottom teleosts	0	0		0	0		<0.1	<0.1	#	0	0		<0.1	<0.1	#
Cartilaginous fishes	0.6 (0.5)	1.1 (1.2)	#	0.3 (0.5)	1.2 (1.3)	$	0.4 (0.5)	0.4 (0.4)	#	0	0.2 (0.3)	#	0.4 (0.5)	0.5 (0.9)	#
Total number of species	8	13		11	17		16	21		7	18		21	23	

$ - Significantly more abundant within the CRMR
# - More abundant within the CRMR, but difference not significant
@ - Significantly more abundant within the GMPA
* - More abundant within the GMPA, but difference not significant

cepedianus, *Triakis megalopterus*, *Rhabdosargus globiceps*) encountered within the CRMR were not seen within the Glencairn site, of which the first four were significantly more abundant ($p<0.05$) within the CRMR. Three species (*Amblyrhynchotes honkenii*, *Sardinops sagax*, *Poroderma pantherinum*) were only seen within the GMPA.

The size distributions of eight of the 18 species encountered within both sites were significantly different ($p<0.05$). Figure 7.1 shows that in all these cases, the size distributions of those inhabiting the CRMR contained more large individuals, even though the mean length of just six of the eight species was greater within the CRMR. The largest censused individuals of the other two species (*Chirodactylis brachydactylis*, *Diplodus sargus capensis*) were still encountered within the CRMR.

Discussion

When reef fish assemblages inhabiting two separate reef systems located within one geographical area are compared, it is usually to investigate the impact of fishing by comparing protected and exploited assemblages inhabiting 'similar' reefs (*e.g.* Bell 1983; Russ & Alcala 1989; Burger 1991; Mann 1992). Such studies, therefore, never investigate how reef architecture affects the composition of reef fish assemblages. In South Africa, marine reserves have often been established in an *ad hoc* fashion without sufficient information, and with little monitoring of their success after their establishment (Attwood *et al.* 1997b, c; Hockey & Branch 1997). This appears to be particularly true in the case of small marine reserves. However, since Attwood *et al.* (1997b) report that in South Africa, it is likely to be easier to establish numerous small reserves rather than a few large one in the

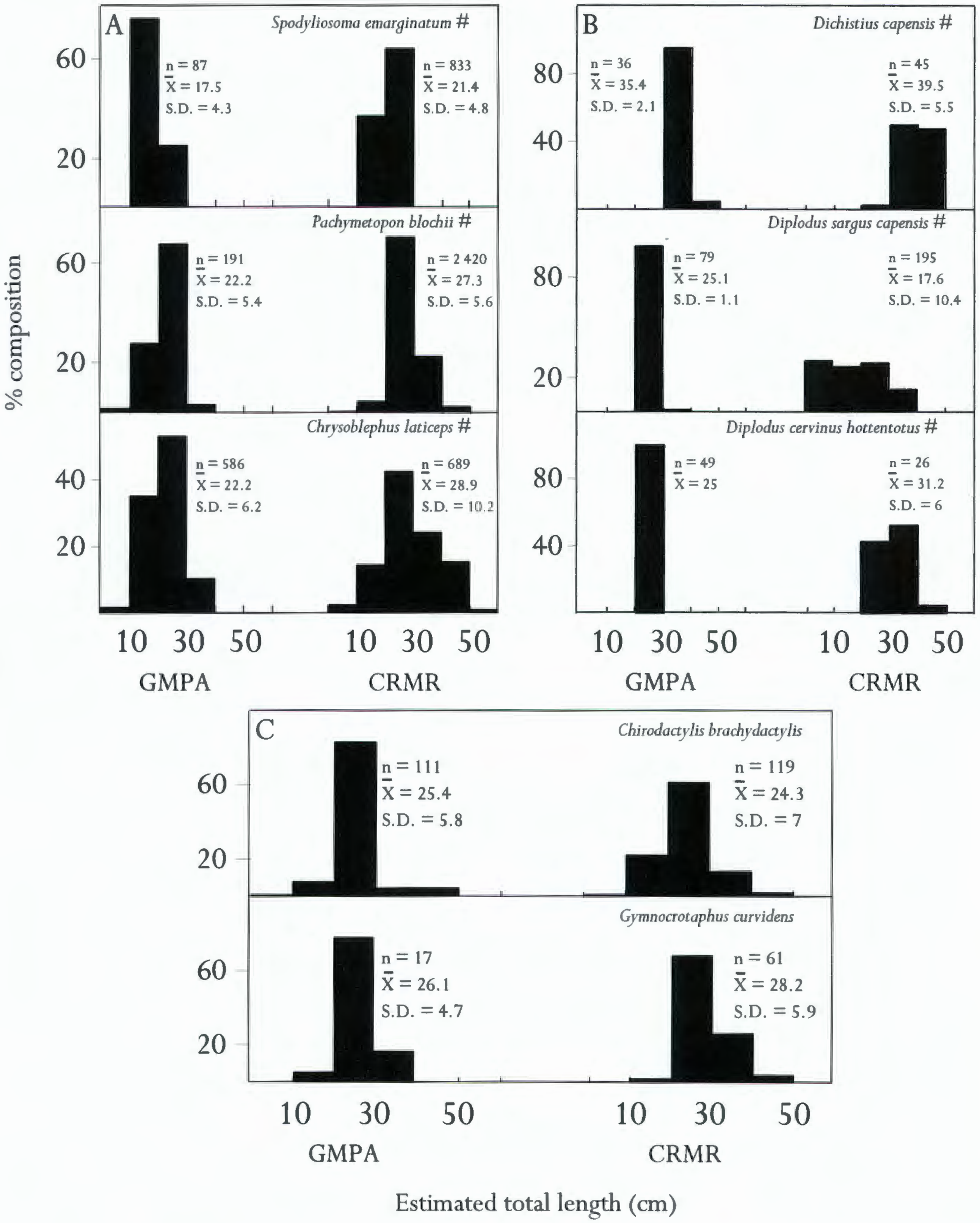


Figure 7.1. The size distribution of A. Species targeted predominantly by boat anglers, B. shore anglers and C. spearfishers, that were encountered within the Glencairn Marine Protected Area (GMPA) and Castle Rocks Marine Reserve (CRMR). Note, only species for which more than ten individuals were encountered within both MPAs are included. # indicates that the two size distributions differed significantly from each other.

future, the importance of a small reserve's location in affecting its potential success must be understood. Although such work has not been undertaken in this country, the need for it is highlighted by Davidson & Chadderton (1994) who report that reef type is important in affecting the composition of fish assemblages in New Zealand.

Chapter 2 investigated the effect that a number of physical reef parameters have on the spatial distribution and abundance of False Bay reef fish species. By comparing the protected assemblage discussed in that chapter to a second protected one inhabiting a structurally different reef located at a nearby site it was hoped to elucidate just how important physical reef parameters are in affecting the composition of a fish assemblage. The results highlight the fact that physical reef characteristics, such as vertical elevation and/or presence of large caves, are important physical factors affecting the composition of reef fish assemblages. Species richness and reef teleosts relative abundances were significantly greater within the CRMR compared to the GMPA. The relative abundances of many individual species were also significantly greater within the CRMR. The majority of the relative abundance differences observed can be explained by a shortage of the different species' reef requirement at the Glencairn site. For example, reefs with vertical elevations in excess of five meters, which provide aggregating points for such species as *Pachymetopon blochii* and *Boopsoidea inornata* when the water is 13 °C or colder, are lacking within the GMPA. This is probably one of the main reasons explaining the significantly lower abundance of these two species, as well as *Gymnocrotaphus curvidens*, *Diplodus sargus capensis* and *Dichistius capensis*, at that site compared to within the CRMR.

The lower abundances of *Sarpa salpa* and *Triakis megalopterus* within the GMPA could be due to GMPA kelp beds being narrower than CRMR ones (GMPA reefs are more than 8 m deep closer to shore), since these are rarely seen outside of kelp beds (Chapters 1, 2 & 6).

This is likely to also be an important reason explaining the lower relative abundances of *Diplodus sargus capensis* and *Dichistius capensis* within the GMPA.

However, some of the species that require large caves as a source of cover (*e.g. Chrysoblephus laticeps*, *Chirodactylis brachydactylis*), did not seem to be as negatively affected by the flat nature of the reef within the GMPA. Although both were still more abundant within the CRMR, both species' relative abundance differences were not significant. Indeed, within the >5-8 m stratum, *Chirodactylis brachydactylis* was more abundant (but not significantly so) within the GMPA, while it was similarly abundant at both sites within the >12-16 m stratum. Similarly, *Chrysoblephus laticeps* was more abundant (but not significantly so) within the GMPA in >8-12 m of water. It is likely that the similarity in the abundances of both species at the two sites is partly due to reef elevation not being such an important reef requirement for them.

The limited depth range of the reef situated within the GMPA is likely to minimise reef fish species diversity. Of the species that were most abundant at depths exceeding 12 m within the CRMR, only four (*Chrysoblephus gibbiceps*, *Spondyliosoma emarginatum*, *Pachymetopon aeneum* and *Boopsoidea inornata*) were encountered within the GMPA, of which only *C. gibbiceps* was not significantly more abundant within the CRMR. In addition, *Chirodactylis grandis*, a species that reaches maximal abundance at depths exceeding 16 m (Chapter 2), was not encountered within the GMPA.

Only *Parascorpis typus*, *Diplodus cervinus hottentotus* and *Haploblepharus edwardsii* were more abundant within the GMPA, although the differences in *P. typus* and *D. cervinus hottentotus* relative abundances were not significant. Furthermore, the GMPA relative abundance estimate of *D. cervinus hottentotus* was greatly inflated by one large shoal (35 fishes) encountered. *Diplodus cervinus hottentotus* was only encountered four times within the GMPA, compared to 12 times within the CRMR. *Haploblepharus edwardsii* was the only species whose abundance was significantly greater within the GMPA.

Therefore, based on overall species numbers, species richness and the relative abundances of fish (reef teleosts or individual species), the fish assemblage inhabiting the structurally more complex reef encompassed by the CRMR is superior in quality to that inhabiting the flatter reef encompassed by the GMPA. However, when the size distributions of the different species censused at the two sites are compared, additional important differences are observed. The assemblage inhabiting the GMPA contains significantly fewer large individuals than that in the CRMR. Predictably, this was the case for species such as *Pachymetopon blochii*, *Diplodus sargus capensis*, *Dichistius capensis* and *Spondyliosoma emarginatum*, which were significantly more abundant within the CRMR. However, this was also the case for *Chirodactylis brachydactylis*, *Chrysoblephus laticeps* and *Diplodus cervinus hottentotus*, all of which were more abundant (but not significantly so) within some depth strata in the GMPA.

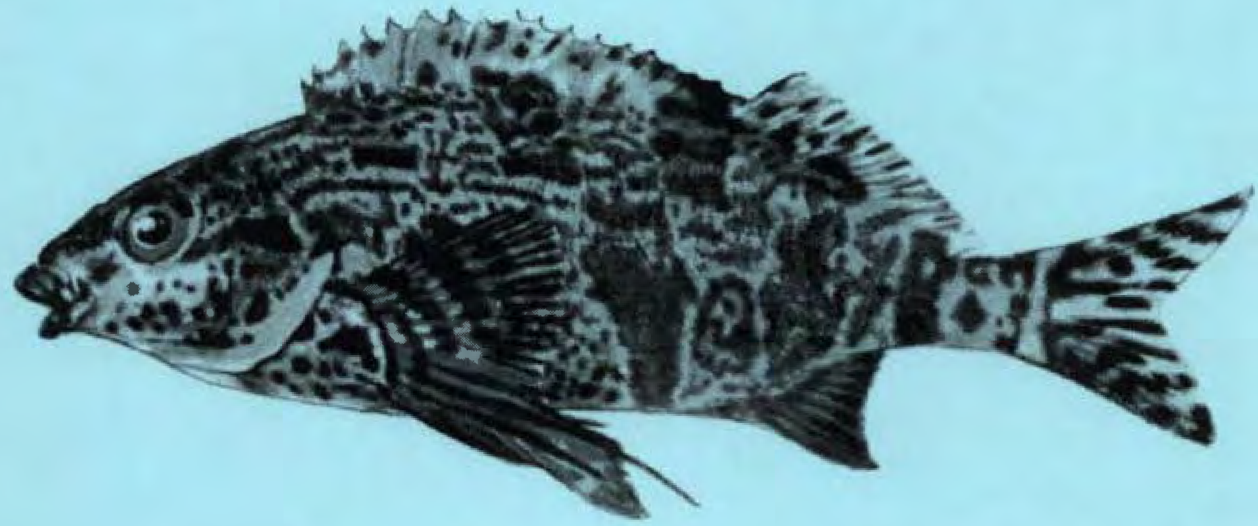
Overall, therefore, the structurally poorer reef situated within the GMPA is inhabited by fewer large individual fishes, and fewer fishes as a whole, than the structurally complex reef encompassed by the CRMR.

Although different poaching levels within the two reserves may explain some of the differences discussed above, the lack of cover and/or sufficient food within the GMPA is likely to be the most important factor. Within False Bay, a reef provides two resources to most reef fishes: food (Chapter 3) and shelter (Chapter 2). For a reef fish species to colonise reefs in appreciable numbers and grow optimally, therefore, reefs have to provide both resources. The lack of either resource on a reef could affect the composition of the fish assemblage inhabiting it. For example, even though the flat reefs within the GMPA are potentially adequate feeding grounds for *Pachymetopon blochii* and *Boopsoidea inornata* when the water is warm, the GMPA is not adequate for these species as it encompasses no high-rising pinnacles onto which they can congregate when the water is cold. The opposite situation (cover available, but little food) could also result in a poorer assemblage for different species. Although less common than within the CRMR, some large caves are available to species requiring these within the GMPA. Predictably, therefore, species whose distributions are affected more by the availability of large caves than by reef elevation (*e.g. Chrysoblephus laticeps*, *Chirodactylis brachydactylis*), were found in comparable relative abundances at the two sites. However, the size distribution of *Chrysoblephus laticeps* within the GMPA was significantly different from that in the CRMR, with the latter population including many more large individuals. Since both reserves are of similar age, the difference in size distribution between the two sites cannot be explained by their date of declaration. Instead, the reason could be twofold. Firstly, because the reef within the GMPA is flat, it is likely that prey consumed regularly by *Chrysoblephus laticeps* (*e.g.* crabs and cephalopods - Chapter 3) are rarer there than within the CRMR. *Chrysoblephus laticeps* inhabiting the Glencairn site, therefore, may have to survive on potentially poorer quality food, or less food altogether. Secondly, the differences in size distributions between the two sites could also be related to intraspecific competition for food. As was shown earlier, the reef within the GMPA contains significantly fewer large caves than that within the CRMR, while the abundance of *Chrysoblephus laticeps* and *Chirodactylis brachydactylis* at the two sites are comparable. This means that more *Chrysoblephus laticeps* have to inhabit each cave than is the case within the CRMR. However, because *Chrysoblephus laticeps* is resident around its set of caves (Penrith 1972; van der Elst 1988; van Rooyen 1988; Buxton & Allen 1989), it is likely to feed mostly within close proximity to these. Should this be the case, it could result in increased interspecific competition for food, and a shortage of adequate food, within close proximity of large caves within the GMPA. This could impact the growth rate of *Chrysoblephus laticeps*, resulting in smaller fishes dominating the GMPA population.

The comparison of the reef fish assemblages inhabiting the two sites clearly indicates that the conservation benefits offered by the two reserves are vastly different, and provides some of the information required to score each using Hockey & Branch (1997)'s COMPARE (Criteria for Marine Protected ARea Evaluation) methodology. The reef fish assemblage inhabiting the CRMR is more abundant, and contains significantly more important linefish species and large individuals than does the GMPA one. Being flatter, providing less large caves and not extending to a depth greater than 15 m, the reef within the GMPA is unlikely to provide the reef requirements (food and/or cover) of most of the False Bay reef fish species in need of some form of conservation. This points to the CRMR being the more successful marine reserve of the two as a linefish management tool (Hockey & Branch 1997), and supports the recommendations of Emanuel *et al.* (1992) concerning the necessity of small scale variations in physical variations within marine reserves. This highlights the importance of physical reef structure in affecting the composition of the reef fish assemblage inhabiting a site within False Bay. It is, therefore, an important factor that needs to be considered when selecting sites for the establishment of marine reserves in the hope of conserving reef-dependent fishes. Only by establishing a marine reserve within an area capable of providing all the reef requirements (shelter and food) of the species it is aimed to conserve will it be beneficial to them (Hockey & Branch 1994; Rowley 1994). Although Attwood *et al.* (1997a) state that the process of site-selection for marine protected areas intended to improve a population's protection or the fishery for a species requires an understanding of the life history of the target species and oceanography of the area, this study highlights that the quality of the environment at a site is also of paramount importance. This is especially important if a number of small marine reserves, rather than one large one, are to be established to conserve reef fishes, as is the case within False Bay. It also highlights that the structural architecture of reefs encompassed by some of the small marine protected areas already present within False Bay should be investigated to assess these protected areas' potential as fishery management tools. Should they, like the GMPA, encompass reefs of poor quality for reef fishes, they may have to be relocated to increase the benefits which reef fishes and their fisheries obtain from them.

Chapter 8

The value of artificial reefs as reef fishery enhancement tools in False Bay: Are they worth the effort?



Cheilodactylus fasciatus

Introduction

The deployment of artificial reefs is reported to be a promising linefishery management strategy (Bohnsack & Ault 1996), providing additional habitat onto which fishes can recruit and be available to fisheries. They have been deployed to increase the availability of reefs in certain areas, as well as enhance catches in previously overfished and/or unproductive areas (Buckley 1982; Cliff 1983; Polovina 1989). Most structures have been readily colonised by demersal reef and transient fish species (*e.g.* Chater 1994; West *et al.* 1994). Such fishes utilise artificial reefs as a source of shelter, food and living space, as well as a point of physical orientation (Bohnsack 1989; Meier 1989).

Along the South African coastline, adverse weather conditions have resulted in many accidental shipwrecks, creating the majority of artificial reefs (Turner 1988). More than 30 such wrecks are found within False Bay alone (Turner 1988). However, within False Bay, and off KwaZulu-Natal (Chater 1994), a number of absolete ships and/or barges have also been deliberately scuttled to create artificial reefs. Within False Bay, these include the *SAS Good Hope*, *Oratava*, *Princess Elizabeth*, *Rockeater* and *SAS Transvaal* in Smitswinkel Bay, and the *SAS Pietermaritsburg* north of Miller's Point. These were deployed with the intention of a) creating artificial reefs to enhance reef fish productivity (Cliff 1983), and b) providing additional dive sites for SCUBA divers (P. Hanekom U.C.T., pers. com.). The only studies that has been undertaken on these wrecks are those of Fricke, Koop & Cliff (1982), who reported on the benthic invertebrates colonising the *SAS Transvaal*, and Cliff (1983), who reported on the reef fish colonisation of the same wreck. In addition, Cliff (1983) compared the fish assemblage inhabiting the *SAS Transvaal* to that on the *Rockeater*.

This study aimed to evaluate eight of the False Bay wrecks with respect to their potential for enhancing reef-fish productivity and reef-fish catches. Based on the results obtained, the use of artificial reefs as fishery management tools within False Bay is discussed, and guidelines are proposed that should maximise the benefits of future artificial reefs in the region.

Methods

Four False Bay sites encompassing eight wrecks were sampled (Figure 8.1). Information concerning the location, year of sinking, depth and substratum on which each of the censused wrecks lies is listed in Table 8.1, along with the date when each fish assemblage was censused. The oldest wreck is the *Clan Stuart,* while the most recent is the *SAS Pietermaritzburg*. All except the *Ispahan* lie isolated on sand, while the two shallow-water wrecks are the only ones that sank accidentally.

An estimate of the composition and relative abundance of each wreck's fish assemblage was obtained using a non-destructive visual census technique. A lone diver counted fishes along three transects swum at slow and constant speed along the length of each wreck (one along each side and one over the centre). Due to their shallow depths, the *Ispahan* and *Clan Stuart* were sampled using snorkelling gear, while the remaining wrecks were censused by SCUBA divers. All demersal and pelagic fishes were counted. Cryptic species, because they are difficult to census non-destructively, were ignored. The number of fish species, and the number of individuals per species, encountered on each wreck were divided by the total search area to obtain estimates of relative abundance (n.100 m^{-2}). This was done for each wreck separately, for shallow-water (those located at depths up to 8 m) and deep-water

Table 8.1 The name, location, year sunk, condition, depth and substratum on which each of the eight wrecks from which fish assemblages were censused, sank.

Name	Reason for sinking	Location	Year sunk	Condition	Maximum vertical elevation (m)	Depth	Substratum sunk on	Date when visited
Shallow-water wrecks								
Ispahan	Accidental	Holbaai Point		Broken up	2-3	<6 m	Reef	June 1993
Clan Stuart	Accidental	Simonstown Beach	1914	Broken up	6	<8 m	Sand	May 1993
Deep-water wrecks								
SAS Pietermaritzburg	Scuttled	Between Miller's Point and Spaniard Rock	1994	Good	>15	22 m	Sand	August 1996
Smitswinkel Wrecks								
Rockeater	Scuttled	Smitswinkel Bay	1972	Good	Up to 13	>30 m	Sand	July 1995
SAS Goodhope	Scuttled	Smitswinkel Bay	1978	Good	Up to 13	>30 m	Sand	July 1995
SAS Transvaal	Scuttled	Smitswinkel Bay	1978	Good	Up to 13	>30 m	Sand	June 1995
Oratava	Scuttled	Smitswinkel Bay	After 1978	Good	4-5	>30 m	Sand	June 1995
Princess Elizabeth	Scuttled	Smitswinkel Bay	After 1978	Good	4-5	>30 m	Sand	June 1995

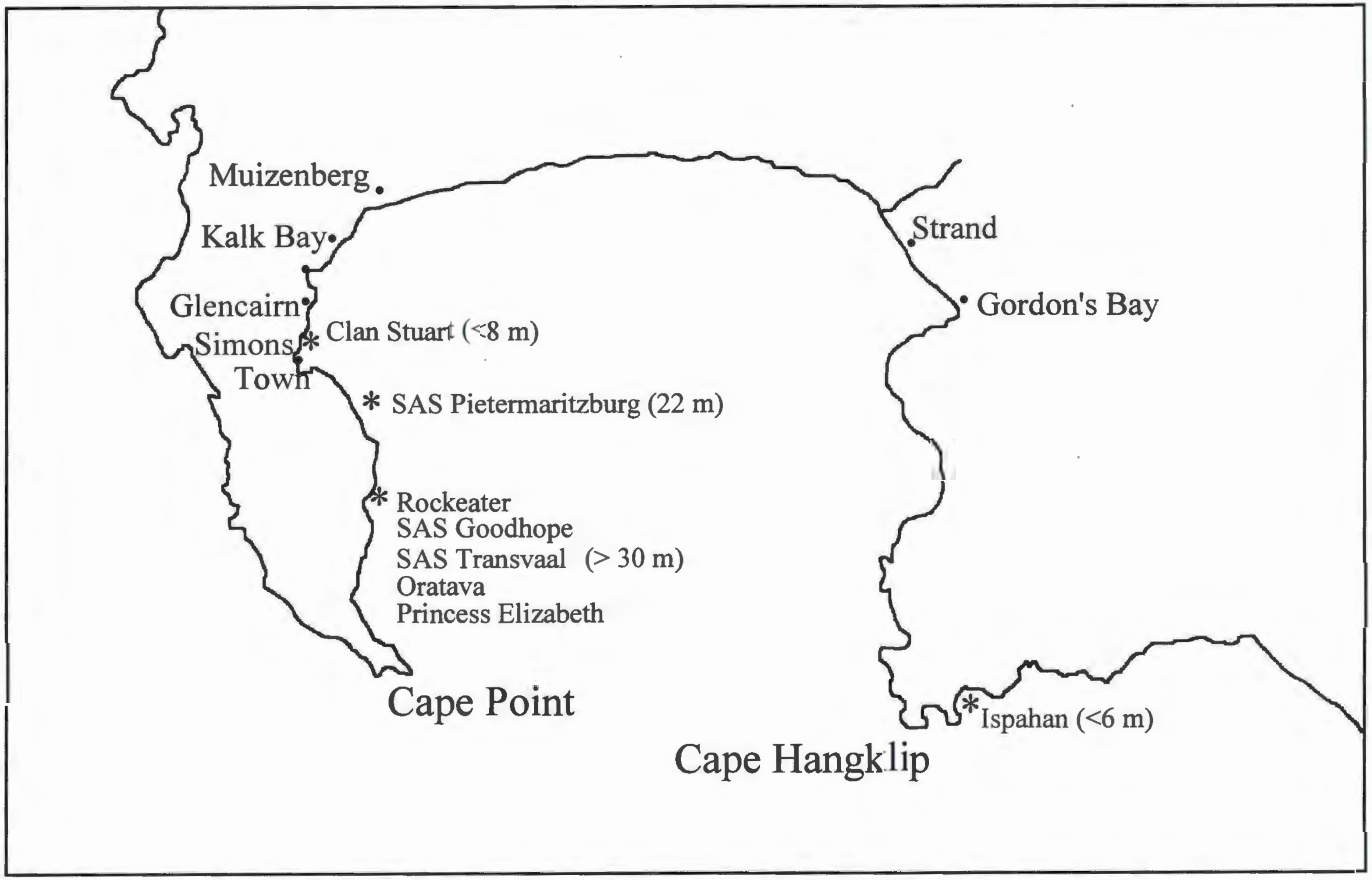


Figure 8.1 The location and depth of the eight wrecks censused within False Bay.

wrecks (those located at depths >16 m) water, and for all wrecks combined. To obtain estimates of the relative abundance of each species, particular groups of species, and of the overall fish assemblage inhabiting different wreck categories, data from different wrecks were combined.

An Analysis of Variance (ANOVA) was run to compare fish relative abundances obtained from shallow- or deep-water wrecks with both low relief (up to 2 m elevation, no large caves) and high relief (all reefs with large caves) natural reefs, within the same depth stratum. A Tukey HSD test for Unequal Sample Size (Spjotvoll & Stoline test) was run to find which mean(s) varied from the other(s). Data concerning fish relative abundance on natural reefs were obtained from Chapter 6, where 1 451 fish counts (covering 52 229 m^2) were completed on exploited reefs throughout False Bay (maximum depth 25 m), using the stationary-point-count census technique described in Chapter 1. To make relative abundance estimates obtained on the wrecks (obtained from search areas 455-2 000 m^2) comparable to those obtained on natural reefs (obtained from point counts of <50 m ^{2}), fish assemblages encountered during separate point counts were summed till the resulting search area totalled 800-1 200 m^2.

Results

A total of 3 411 fishes from 18 non-cryptic species (seven families), were recorded from the 24 transects undertaken on the wrecks (Table 8.2). Mean fish relative abundance (± standard deviation) on all censused wrecks was 50.2 ± 48.9 fishes/100 m^2. Highest counts were obtained on the *Oratava* (144 fishes/100 m^2), one of the Smitswinkel Bay wrecks, and lowest counts on the *SAS Pietermariztburg* (5.44 fishes/100 m^2), the most recently scuttled wreck

Table 8.2 The composition (species and number of fishes belonging to each) of the fish assemblages encountered along the three transects undertaken on each of the wrecks censused. Teleost and cartilaginous species are ranked in order of abundance.

	Smitswinkel Bay wrecks (SBW)						Other wrecks			All wrecks together	
	Depth >30 m						Depth 22 m	Depth <8 m		All depths	
	SAS Transvaal	*Oratava*	*Princess Elizabeth*	*Rockeater*	*SAS Goodhope*	Total (num) (all SBW)	*SAS Pieter-maritzburg*	*Clan Stuart*	*Ispahan*	Numbers	% composition
Species richness	11	8	8	8	6	8.2	6	6	5	7.25	
Species											
Reef teleosts											
Spondyliosoma emarginatum	1 200	600	100	31	310	2 241	25	-	-	2 266	66.43
Cheilodactylus fasciatus	82	118	54	55	177	486	14	31	27	558	16.36
Pachymetopon blochii	63	22	61	18	200	364	3	3	14	384	11.26
Diplodus sargus capensis	-	-	-	-	-	-	-	30	-	30	0.88
Chrysoblephus laticeps	8	8	5	4	4	29	-	-	-	29	0.85
Pterogymnus laniarius	11	-	10	5	-	26	-	-	-	26	0.76
Diplodus cervinus hottentotus	-	-	-	-	-	-	-	25	-	25	0.73
Parascorpis typus	5	2	5		1	13	2	-	-	15	0.44
Chirodactylus brachydactylus	3	-	1	-	-	4	-	-	7	11	0.32
Chirodactylus grandis	3	3	1	3	-	10	-	-	-	10	0.29
Cheilodactylus pixi	3	2	-	-	4	9	-	-	-	9	0.26
Umbrina canariensis	-	-	-	8	-	8	-	-	-	8	0.23
Argyrozona argyrozona	-	-	-	7	-	7	-	-	-	7	0.21
Dichistius capensis	-	-	-	-	-	-	-	-	3	3	0.09
Soft-bottom teleost											
Rhabdosargus globiceps	1	-	-	-	-	1	-	-	-	1	0.03
Cartilaginous species											
Haploblepharus edwardsii	-	-	-	-	-	-	4	17	2	23	0.67
Poroderma africanum	2	1	-	-	-	3	1	-	-	4	0.12
Notorhynchus cepedianus	-	-	-	-	-	-	-	2	-	2	0.06
Overall total	1 381	756	237	131	696	3 201	49	108	53	3 411	

(Table 8.3). The family Sparidae dominated total fish numbers (81.1%), followed by the Cheilodactylidae (17.2%). *Rhabdosargus globiceps* was the only soft-bottom species recorded, and no pelagic species were encountered.

Table 8.2 shows the total number of individuals of each species encountered on each wreck, as well as all the wrecks combined. Although 3 382 teleosts from 15 species were encountered, *Spondyliosoma emarginatum* (2 266 fishes – 67%), *Cheilodactylus fasciatus* (558 fishes – 16.5%) and *Pachymetopon blochii* (384 fishes – 11.4%) were the most abundant species, together comprising 94.9% of all fishes encountered.

The greatest fish diversity (11 species) was encountered on the *SAS Transvaal*, while the mean (± standard deviation) species density recorded from the Smitswinkel Bay site was 8.2 ± 1.8 species/100m^2. Six species were encountered on the *SAS Pietermaritzburg* and *Clan Stuart*, and five on the *Ispahan* (Table 8.2). Table 8.3 shows that overall, shallow-water wrecks had a mean species richness of 1.05 ± 0.07 species/100 m^2, almost the same as that of deep-water wrecks (1.01 ± 0.43). However, both values are lower than those for natural low relief reef in shallow (1.15 ± 0.21) and deep water (1.28 ± 0.27), and natural high relief reef in shallow (1.81 ± 1.40) and deep water (1.53 ± 0.11).

Of the 18 species recorded, only *Pachymetopon blochii* and *Cheilodactylus fasciatus* were encountered on all the wrecks, while six (*Dichistius capensis*, *Diplodus cervinus hottentotus*, *D. sargus capensis*, *Rhabdosargus globiceps*, *Umbrina canariensis* and *Notorhynchus cepedianus*) were encountered on only one wreck. Deep-water reef-fish species (*Argyrozona argyrozona*, *Cheilodactylus pixi*, *Chirodactylus grandis*, *Pterogymnus laniarius* and *Spondyliosoma emarginatum*) were only encountered on deep-water wrecks, while *Dichistius*

Table 8.3 The relative abundance (fish.100 m^{-2}) of each of the species making up the fish assemblages inhabiting of the different wrecks censused, as well as their overall relative abundances as encountered on shallow- and deep-water wrecks, and all eight wrecks combined. Note, three transects were completed on each wreck

	Smitswinkel Bay wrecks (SBW)					Other wrecks			Shallow-water wrecks	Deep-water wrecks	All wrecks
	Depth >30 m					Depth 22 m	Depth <8 m		0-8 m	>16 m	All depths
Species	*SAS Transvaal*	*Oratava*	*Princess Elizabeth*	*Rockeater*	*SAS Goodhope*	*SAS Pietermaritzburg*	*Clan Stuart*	*Ispahan*			
Reef teleosts											
Spondyliosoma emarginatum	60.00	114.29	19.05	3.88	41.33	2.78	-	-	-	40.2 (42.5)	30.2 (40.5)
Cheilodactylus fasciatus	4.10	22.48	10.29	6.88	23.60	1.56	5.17	5.93	5.6 (0.5)	11.5 (9.4)	10 (8.4)
Pachymetopon blochii	3.15	4.19	11.62	2.25	26.67	0.33	0.50	3.08	1.8 (1.8)	8 (9.9)	6.5 (8.9)
Diplodus sargus capensis	-	-	-	-	-	-	5.00	-	2.5 (3.5)	-	0.6 (1.8)
Chrysoblephus laticeps	0.40	1.52	0.95	0.50	0.53	-	-	-	-	0.7 (0.5)	0.5 (0.5)
Pterogymnus laniarius	0.55	-	1.90	0.63	-	-	-	-	-	0.5 (0.7)	0.4 (0.7)
Diplodus cervinus hottentotus	-	-	-	-	-	-	4.17	-	2.1 (3)	-	0.5 (1.5)
Parascorpis typus	0.25	0.38	0.95	-	0.13	0.22	-	-	-	0.3 (0.3)	0.2 (0.3)
Chirodactylus brachydactylus	0.15	-	0.19	-	-	-	-	1.54	0.8 (1.1)	0.1 (0.1)	0.2 (0.5)
Chirodactylus grandis	0.15	0.57	0.19	0.38	-	-	-	-	-	0.2 (0.2)	0.2 (0.2)
Cheilodactylus pixi	0.15	0.38	-	-	0.53	-	-	-	-	0.2 (0.2)	0.1 (0.2)
Umbrina canariensis	-	-	-	-	-	-	-	-	-	0.2 (0.4)	0.1 (0.4)
Argyrozona argyrozona	-	-	-	0.88	-	-	-	-	-	0.2 (0.4)	0.1 (0.3)
Dichistius capensis	-	-	-	-	-	-	-	0.66	0.3 (0.5)	-	0.1 (0.2)
Soft-bottom teleost											
Rhabdosargus globiceps	0.05	-	-	-	-	-	-	-	-	<0.1	<0.1
Cartilaginous species											
Haploblepharus edwardsii	-	-	-	-	-	0.44	2.83	0.44	1.6 (1.7)	0.1 (0.2)	0.5 (1)
Poroderma africanum	0.10	0.19	-	-	-	0.11	-	-	-	0.1 (0.1)	0.1 (0.1)
Notorhynchus cepedianus	-	-	-	-	-	-	0.33	-	0.2 (0.2)	-	<0.1
All species	69.05	144.00	45.14	15.38	92.80	5.44	18.00	11.65	14.8 (4.5)	62 (51.7)	50.2 (48.9)
Area (m^2)	2 000	525	525	800	750	900	600	455			

capensis and *Diplodus sargus capensis*, both shallow-water species (Chapters 1 & 2), were only recorded on shallow-water wrecks.

In shallow water, reef teleosts were less abundant on the censused wrecks (13.02 ± 2.56) than on both low (44.02 ± 28.72) and high relief natural reef (58.42 ± 24.15). Relative abundances of reef teleosts obtained on deep-water wrecks (61.99 ± 51.61), however, were almost the same as those recorded from high relief natural reefs in deep water (62.92 ± 20.77), but greater than that obtained from low relief natural reefs (40.57 ± 20.64). Because few cartilaginous fishes and soft-bottom teleosts were encountered on the wrecks or natural reefs, the above patterns are very similar to those recorded for overall fish relative abundances.

Discussion

The analysis of fish relative abundances obtained on the wrecks and natural reefs compared relative abundance estimates obtained using two different census techniques. Stationary point counts were not attempted on the wrecks, as individual counts would have needed to be undertaken too close to one another to obtain sufficient area coverage on each wreck. This could have affected the behaviour of the fishes, and hence the accuracy of the census. However, since care was taken not to count fishes that were attracted to the diver more than once, and fishes inhabiting the wrecks were not disturbed by the diver, relative abundance estimates obtained from the two census techniques were deemed comparable, even though the census methods were different.

Based on the composition of the fish assemblages encountered on the wrecks, it appears as if the sinking of the ships has resulted in the enrichment of the fish assemblage found at the site prior to their sinking. Of the 17 species encountered on the wrecks lying on sand, *Rhabdosargus globiceps* is likely to have been the only one present at the sites prior to the sinking of the ships. However, the 18 species encountered during this study represent but a small proportion of those which frequent False Bay natural reefs (Chapters 1, 6 & 7). Indeed, *Sarpa salpa*, *Boopsoidea inornata* and *Gymnocrotaphus curvidens*, abundant species on natural reefs, were not recorded at all. Although the absence of these and most of the other species can be partly explained by the limited surface area covered by the wrecks, and the fact that each wreck was only censused once, other reasons are also likely be important. These are discussed below.

The study of Cliff (1983), describing the colonisation of the *SAS Transvaal* by reef fishes during the 14 months following its scuttling, is the only one concerning the colonisation of artificial reefs by reef fishes in False Bay. He also compared the fish assemblage found on the *SAS Transvaal* two years after its deployment, to that inhabiting the *Rockeater*, scuttled nearby nine years earlier. Sixteen fish species were recorded from these two wrecks, including nine reef fish, three non-reef-associated, and four cryptic species. *Pachymetopon blochii*, *Sarpa salpa* and *Spondyliosoma emarginatum* were the numerically dominant reef sparids, while *Trachurus trachurus* was the most abundant non-reef-associated species. Only six of the nine reef fish species reported by Cliff (1983) were encountered at the Smitswinkel wrecks site during this study, although an additional six reef fish species and one non-reef-associated species were seen. An additional four reef fish species were encountered on the shallower wrecks.

The fish assemblage encountered on the five Smitswinkel Bay wrecks comprised few known resident species. *Pachymetopon blochii* and *Chrysoblephus laticeps* are the only known resident species (van Rooyen 1988; Buxton & Allen 1989), although *Cheilodactylus fasciatus*, *C. pixi* and *Chirodactylus brachydactylus* also appear resident (pers. obs.). Reef fish diversity at the Smitswinkel Bay wreck site has changed since the study by Cliff (1983), with seven new reef fish species being recorded during this study and three of those recorded by Cliff (1983) being absent. Also, reef fishes are less abundant than reported by Cliff (1983), who reported over 900 fish on the *SAS Transvaal* alone, compared to the 3 021 fishes censused on all of the Smitswinkel Bay wrecks during this study.

Spondyliosoma emarginatum, *Cheilodactylus fasciatus* and *Pachymetopon blochii* were the three most abundant species encountered on the wrecks, these together contributing over 94% of the assemblage. These are, however, also very common on natural reefs in False Bay (Chapters 1 & 6). The assemblage on the wrecks does not contain many species attractive to the False Bay linefishery. Although 2 802 (82.9%) of the 3 382 teleosts inhabiting the wrecks are of some importance to Fishers, *Spondyliosoma emarginatum*, a small species, makes up 80.9% of that total, highlighting the fact that very few individuals of other important linefish species were encountered. On the Smitswinkel Bay wrecks, although seven linefish species were encountered, only *Pachymetopon blochii* and *Spondyliosoma emarginatum* were abundant and large enough to be targeted by boat anglers. These wrecks are not popular angling sites (Cliff 1983), which points to them having failed to enhance linefish catches substantially in False Bay. The same applies for the three shallower wrecks censused, for although three additional linefish species were recorded on these, too few individuals were encountered to significantly enhance stocks in the area.

The censused wrecks do not appear to provide optimal habitat for reef fish as compared to adjacent natural reefs. Neither do they significantly enhance reef fish stocks in the area. This is not surprising considering their limited surface area compared to that of natural reefs in False Bay. However, it is proposed that the location of the censused wrecks also limits their potential as management tools. For example:

1. The Smitswinkel Bay wrecks are located too deep for them to accommodate maximal numbers of reef fish species, many of which (*e.g. Pachymetopon blochii*, *Chirodactylus brachydactylus*, *Gymnocrotaphus curvidens* and *Dichistius capensis*) are more abundant in shallower water (Chapters 1 & 6).
2. The shallow-water wrecks censused lie too shallow to accommodate many important linefish species. Furthermore, due to their shallow location, they have been broken down by wave action.
3. Seven of the wrecks (including all of those scuttled purposefully to create artificial reefs) lie on sand far from natural reefs. Therefore, species must recruit from the plankton, or swim over sand as post-recruitment juveniles, to colonise these wrecks. Such artificial reefs are inaccessible to the post-recruitment juveniles of many False Bay reef fish species, since these do not appear to swim large distances over sand.
4. Although sinking ships on sand provide fishes with new reefs (Guastella & Smith 1997), the small size of the resulting reefs limits the number of reef fishes that can make use of them as a source of both shelter and food.

Although these wrecks have had limited success in enhancing fisheries, careful planning could enhance the usefullness of future such artificial reefs. Having ruled out the areas where such structures would endanger shipping, the following points need to be considered when deciding where to deploy future wrecks:

1. The ships should be scuttled where they are least exposed to wave action and sand inundation.
2. They should be deployed within the depth zone where the species they are meant to enhance are most abundant.
3. As many as possible of the reef requirements of colonising fishes, including relief, cover and food, should be provided by the artificial reef and surrounding habitat. This means that within False Bay, such artificial reefs should be established alongside natural reef, as the natural reef would then provide food for additional fishes that would use the artificial reefs solely as shelter.
4. Artificial reefs should be positioned within a marine reserve, thereby counteracting the fact that artificial reefs are known to aggregate fishes (Buckley 1989; Bohnsack 1996b), making them more vulnerable to fishers.

Wrecks, however, are not only used by marine organisms, as they are also popular angling and/or dive sites (Brock 1994; Chater 1994; Guastella & Smith 1997). False Bay spearfishers regularly visit the *Ispahan* when targeting *Dichistius capensis*, while SCUBA divers are the most numerous human benefactors of wrecks in False Bay. The goal of scuttling the *SAS Pietermaritzburg* and the five wrecks within Smitswinkel Bay to create additional dive sites for SCUBA divers was thus achieved. These two wreck sites are the most popular dive sites for boat-based SCUBA divers along the west coast of False Bay (pers. obs.). However, it is hypothesised that any ship sunk within easy reach in False Bay will become a popular SCUBA dive site, irrespective of where it is sunk. Scuttling ships so as to optimally benefit

False Bay reef fishes should thus not adversely affect the popularity of wrecks for SCUBA divers.

One final potential use of wrecks in False Bay could be to use them as net fouling structures deployed to prevent beach-seiners from operating more than a certain distance offshore from beaches. The wrecks would then have a similar function as they do when deployed to prevent trawling, as proposed by Seaman, Buckley & Polovina (1989). Sinking wrecks off the high-energy beaches where beach-seiners work, however, would minimise their benefits to reef fishes and SCUBA divers, and is thus not recommended.

General conclusion

Diplodus cervinus hottentotus

Summary and conclusion

This thesis set out to investigate three aspects concerning the reef fish assemblage and fisheries of False Bay:

1. The composition of the suprabenthic reef fish assemblage (up to 25 m deep), and which physical and biological factors may affect its composition at a site.
2. The impact that each of the five False Bay linefishery sectors have on the Bay's linefish resource, and the effectiveness of the regulations set to manage them.
3. The value of small marine protected areas and artificial reefs as fishery management tools within False Bay.

The analysis was broken down into eight chapters.

Chapter 1 showed the False Bay reef fish assemblage (inhabiting reefs to a depth of 25 m) to be dominated by sparids (53.8% of species and 87.3% of individuals), followed by cheilodactylids (15.4% of species and 7.5% of individuals). The assemblage was dominated by relatively few species, the ten most abundant making up over 93% of the reef fish numbers encountered. The majority of the abundant species rarely reach 30 cm total length (TL), and few of the others 50 cm TL. Chapter 1 also showed that 17 of the 26 reef fish species encountered appeared to inhabit reefs indefinitely, individuals of some of these being resident on particular reef patches for the duration of the study. Only two of the reef fish species encountered were seasonal migrants, highlighting that the composition of the False Bay reef fish assemblage is stable throughout the year.

Chapter 2 investigated how water temperature, depth, the availability of different sized caves and vertical elevation affected species richness, reef teleosts, cartilaginous fishes and individual species' abundances. The importance of water temperature was highlighted as at different temperatures, some species' behaviour changed, resulting in them either moving onto different portions of a reef or retiring into large caves. When abundances were compared between similar reefs (same depth range, cave size and vertical elevation) at different temperatures, species richness and reef teleosts relative abundances were usually greater in warm compared to cold water. This highlighted the necessity for only comparing census results obtained from within the same water temperature ranges (either warm or cold) in subsequent chapters. Chapter 2 also highlighted the importance of depth in affecting the diversity and abundance of reef fishes at a site, as some species were most abundant within different depth strata. The size of caves available on a reef was also found to be an important factor affecting the composition of reef fish assemblages. Species richness, the abundance of a specific reef teleost species and total reef teleost abundance were usually significantly greater on reefs providing some large caves than on reefs providing none. The potential importance of vertical reef elevation was also noted, although no significant differences in the abundances of any species or groups of species' were detected among reefs differing only in their elevation. Overall, however, it is noteworthy that the distribution patterns of most of the common False Bay reef fish species are affected by a combination of depth, cave size and vertical elevation.

Chapter 3 investigated the dietary composition of 17 of the most abundant False Bay reef fish species and revealed that the majority of these fed predominantly on reef-dwelling prey, with the few sand-dwelling prey consumed likely to be taken within close proximity of reef. For these species, sand does not appear to be favoured substratum above and from which prey are consumed. By comparing the diet of each species to those of the others, this chapter also showed that although there was some dietary overlap among species, most consume particular food items, or combination of these, resulting in them appearing to have their own 'dietary niche' within the assemblage.

The next two chapters were related to linefishery impact and management, and investigated the second aspect of this thesis. Chapter 4 investigated the catch composition and catch rate of the False Bay spearfishery, along with which factors affect its efficiency. It showed that shore- and boat-based non- competitive and competitive spearfishing is practised within the Bay. In the case of both the non-competitive and competitive spearfishery, mean catches taken by shore-based spearfishers were always smaller than those of boat-based spearfishers. Non-competitive spearfishers' mean catches (fish numbers) were always greater in warm compared to cold water, while the catch-per-unit-effort *cpue* (with respect to fish number) of the most experienced spearfishers (Class 3) was always superior to those of the less experienced. The *cpue* of competitive spearfishers was greater than that of non-competitive divers. Although the percentage composition of the catches of non-competitive and competitive spearfishers differed, reef-dependent species dominated the catches of both spearfishery sectors. Chapter 4 also assessed the value of the different fishery regulation enforced on spearfishers, and reported that the minimum size and daily bag limit restrictions enforced on spearfishers are not sufficiently restrictive. Some changes to these regulations are proposed that would limit fish mortality and/or the negative visual impact of this fishery sector.

Chapter 5 combined the data obtained from Chapter 4 with information concerning the *cpue*, catch composition and effort of the other four linefishery sectors active within False Bay, to estimate the total annual linefish catch landed within the Bay. It estimated that annually, over 76 linefish species were landed between 1992 and 1997, and that these were targeted from all available subtidal habitats. It found that although some species are shared by a number of sectors, usually only one sector impacts each species significantly. This is an important finding, as it facilitates decisions concerning which linefishing sector needs to be restricted, should a particular species' overall catch need to be limited. Chapter 5 also highlighted the lack of additional linefish species that could be targeted, should those presently targeted become seriously overfished. The difficulties in managing the False Bay linefishery, including the biology of some species, the multispecies nature of each fishery sector and the sharing of different species among different sectors are discussed, along with some proposed changes to linefishery regulations. The inefficient enforcement of fishery regulations in the Southwestern Cape and the leniency of the prosecuting system towards transgressors are noted as important management problems that need to be overcome.

Having quantified the impact of the False Bay linefishery and assessed the value of the traditional management measures set to manage it, the last three chapters investigated the value of marine protected areas and artificial reefs as fishery management tools within the Bay. Chapter 6, by comparing the protected assemblage inhabiting a small (3.25 km) marine reserve to that inhabiting exploited False Bay reefs, assessed whether such a small reserve provided any benefits to fishes and managers. It found that the protected fish assemblage's species richness and reef teleosts abundance were greater than those of the exploited assemblage. Some individual reef fish species were also significantly more abundant within compared to outside the reserve, while for many species, the protected stocks were found to contain more large individuals than exploited stocks. However, because of its small size, the reserve provides limited benefits to False Bay reef fisheries, as it only benefits resident species. Furthermore, its small size restricts the viability of its stocks should exploited stocks become overfished. By comparing the catch rates of non-competitive spearfishers close to and further away from the reserve, Chapter 6 also found that the reserve does not enhance adjacent fisheries significantly by exporting adult fishes. Finally, this chapter also noted that the behaviour of the larval phase of False Bay linefish species needs to be investigated, as such information is necessary to assess whether fish larvae produced within the Bay's reserves are able to recruit on exploited reefs.

Chapter 2 highlighted that a reef's architecture is important in affecting the composition of the reef fish assemblage that will make use of it, and should be considered when deciding where small marine reserves are to be established. Chapter 7, by comparing the reef fish assemblages inhabiting protected reefs of different quality, set out to investigate whether reef quality was important in affecting the composition of a reef fish assemblage at a site. The comparison showed that the assemblage inhabiting good quality reef (within the Castle Rocks Marine Reserve) was superior to that inhabiting poor quality reef (within the Glencairn Marine Protected Area) with respect to species richness, fish abundance and size. This chapter also provides some of the necessary information required to score each protected areas using Hockey & Branch (1997)'s COMPARE (Criteria and Objectives for Marine Protected Area Evaluation) methodology. Of the two marine protected areas, the Castle Rocks Marine Reserve is the most successful fishery management tool. This highlights the fact that reef quality is important in affecting the composition of a reef fish assemblage at a site, and thus, the benefits that a small marine reserve can provide to fisheries and their managers.

Finally, by comparing the fish assemblages encountered on eight False Bay wrecks to that inhabiting exploited reefs, Chapter 8 investigated the fishery and management value of artificial reefs deployed within False Bay. The assemblages inhabiting wrecks were poorer in species density, and at times reef fish abundance, than those inhabiting 'flat' (<2 m elevation and no large caves) and 'good' (any reef with large caves) exploited reefs. However, most of the species encountered over both wrecks and exploited reefs had similar size distributions. The lack of desirable linefish (by species and/or size) inhabiting the wrecks, compared to what is available on exploited reefs, explained why the censused wrecks do not appear to have substantially benefited linefisheries in False Bay. Altogether, the possibility of using artificial reefs to enhance reef fish productivity within False Bay was not rejected, however, although it was felt that for future ones to be more beneficial to reef fishes, decisions concerning where to establish them need to consider the reef requirements of the species concerned.

Chapters 4 & 5 highlighted the fact that reef fish resources within False Bay are heavily exploited by the linefishery, that many species have already suffered from pulse/serial overfishing, and that there are few remaining reef fish species that are not maximally exploited in the Bay. These chapters also highlighted that the traditional management regulations enforced onto the linefishery are not sufficiently restrictive to minimise linefishery-related mortality, nor are they enforced efficiently. This points to the need for stricter regulations and additional fishery management measures to redress the fishery decline. The establishment of small marine protected areas (<10 km) and artificial reefs are two such measures that have been employed within False Bay. Chapters 6 & 7 showed that the fishery management benefits provided by small marine reserves are limited, while Chapter 7 highlighted that the reef requirements of the species concerned need to be considered when deciding where to establish marine reserves within the Bay. The findings of that chapter also highlight that the quality of reefs encompassed within False Bay's other small marine protected areas should be assessed, as it is a quick way of indirectly assessing their reef fish assemblage (based on the findings of Chapter 2), and thus potential as a linefish management tool.

However, the results of Chapters 6 & 7 highlight that a large (>35 km), or numerous small, marine reserves are required to benefit nomadic reef fish species within False Bay. A large reserve would be socially unacceptable due to the limited size of the Bay, however, so it is proposed that some of the small marine protected areas which are already present within the Bay be relocated, and additional ones established, in areas which encompass appropriate environment. This would result in these being inhabited by more abundant and diverse reef fish assemblages (including nomadic species), thereby increasing their fishery and management benefits. Furthermore, no linefishing should be permitted within these protected areas. Finally, the poor enforcement of reserve regulations within False Bay is highlighted as an aspect in need of improvement.

As for the use of artificial reefs as linefish management tools, it is felt that the fishery benefits provided by well-located artificial reefs the size of shipwrecks are unlikely to outweigh those provided by well-placed marine reserves. However, this should not negate their use as it is felt that they could still enhance reef fish assemblages significantly at a site if deployed optimally with respect to the reef requirements of the species concerned. Wrecks are in any case very popular SCUBA dive sites, and likely to be planned again in the future with this in mind. Because wrecks, if scuttled within easy access, will become popular SCUBA dive sites irrespective of the substratum on which they lie, it is possible to scuttle them to benefit both reef fishes and SCUBA divers simultaneously. The important thing to note is that for this to be achieved, the reef requirements of the reef fish species concerned are of paramount importance when deciding where to sink them.

References

Dichistius capensis

References

Alevizon, W.S. (1975). Spatial overlap and competition in congeneric surfperches (Embiotocidae) off Santa Barbara, California. *Copeia* **2,** 352-356.

Atkins, G.R. (1970a). Thermal structure and salinity in False Bay. *Transactions of the Royal Society of South Africa* **39,** 117-128.

Atkins, G.R. (1970b). Wind and current patterns in False Bay. *Transactions of the Royal Society of South Africa* **39,** 139-148.

Attwood, C.G. and Bennett, B.A. (1990). A Simulation model of the sport-fishery for galjoen *Coracinus capensis:* an evaluation of minimum size limit and closed season. *South African Journal of marine Science* **9,** 359-369.

Attwood, C.G. and Bennett, B.A. (1994). Variation in dispersal of galjoen (*Coracinus capensis)* (Teleostei: Coracinidae) from a marine reserve. *Canadian Journal of Fisheries and Aquatic Science* **51,** 1247-1257.

Attwood, C.G. and Bennett, B.A. (1995a). Modeling the effect of marine reserves on the recreational shore-fishery of the South-Western Cape, South Africa. *South African Journal of marine Science* **16,** 227-240.

Attwood, C.G. and Bennett, B.A. (1995b). A procedure for setting daily bag limits on the recreational shore-fishery of the South-Western Cape, South Africa. *South African Journal of marine Science* **15,** 241-251.

Attwood, C.G, Harris, J.M. and Williams, A.J. (1997a). International experience of marine protected areas and their relevance to South Africa. *South African Journal of marine Science* **18,** 311-332.

Attwood, C., Beaumont, J., Branch, G.M., Densham, D., Dye, A., Feely, J.M., Harris, J., Heydorn, A. and Hockey, P. (1997b). Towards a new policy on marine protected areas for South Africa. Pretoria: SANCOR. 127pp.

Attwood, C.G., Mann, B.Q., Beaumont, J. and Harris, J.M. (1997c). Review of the state of marine protected areas in South Africa. *South African Journal of marine Science* **18,** 341-367.

Bass, A.J.; D'Aubrey, J.D. & Kistnasamy, N. 1975. Sharks of the east coast of southern Africa. II. The families *Scyliorhinidae* and *Pseudotriakidae. Investigational Report*, Oceanic Research Institute, Durban, 37: 63pp.

Beaumont, J. (1997). Community participation in the establishment and management of marine protected areas: a review of selected international experience. *South African Journal of marine Science* **18,** 333-340.

Beckley, L.B. and Buxton, C.D. (1989). Underwater observation of reef fish in and around Algoa Bay, South Africa. *Transactions of the Royal Society of South Africa* **47,** 29-38.

Bell, J.D. (1983). Effects of depth and marine reserve fishing restrictions on the structure of a rocky reef fish assemblage in the north-western Mediterranean Sea. *Journal of Applied Ecology* **20,** 357-369.

Bennett, B.A. (1988). Some considerations for the management in South Africa of galjoen *Coracinus capensis* (Cuvier), an important shore-angling species off the South-Western Cape. *South African Journal of marine Science* **6,** 133-142.

Bennett, B.A. (1989). The diets of fish in three South-Western Cape estuarine systems. *South African Journal of Zoology* **24,** 163-177.

Bennett, B.A. (1991a). Long term trends in catches by shore anglers in False Bay. *Transactions of the Royal Society of South Africa* **47,** 683-690.

Bennett, B.A. (1991b). Conservation in the marine environment: some problems with the management of shore-angling in the Southwestern Cape. *South African Journal of Aquatic Science* **17,** 12-18.

Bennett, B.A. and Attwood, C.G. (1991). Evidence for the recovery of a surf-zone fish assemblage following the establishment of a marine reserve on the southern coast of South Africa. *Marine Ecology Progress Series* **75,** 173-181.

Bennett, B.A. and Attwood, C.G. (1993a). In defence of the De Hoop Marine Reserve. In *Second South African Marine Linefish Symposium.* Special Publication No 2 (ed. L.E. Beckley and R. van der Elst), pp. 175-176. Durban: Oceanic Research Institute.

Bennett, B.A. and Attwood, C.G. (1993b). Shore-angling catches in the De Hoop Nature Reserve, South Africa, and further evidence for the protective value of marine reserves. *South African Journal of marine Science* **13,** 213-222.

Bennett, B.A., Attwood, C.G. and Mantel, J.D. (1994). Teleost catches by three shore-angling clubs in the Sou7th-Western Cape, with an assessment of the effect of restrictions applied in 1985. *South African Journal of marine Science* **13,** 11-18.

Bennett, B.A. and Griffiths, C.L. (1986). Aspects of the biology of the galjoen *Coracinus capensis* (Cuvier) off the South-Western Cape, South Africa. *South African Journal of marine Science* **4,** 153-162.

Berg, J. (1979). Discussion of methods of investigating the food of fishes, with reference to a preliminary study of the prey of *Gobiusculus flavens* (Gobiidae). *Marine Biology* **50,** 263-273.

Berry, P.F., van der Elst, R.P., Hannekom, P., Joubert, C.S.W. and Smale, M.J. (1982). Density and biomass of the ichthyofauna of a Natal Littoral reef. *Marine Ecology Progress Series* **10,** 49-55.

Beuttler, T.M. (1994). Marine nature reserves in France: legal framework, management, and comparative notes for the U.S. Marine Sanctuary Program. *Coastal Management* **22,** 1-24.

Biden, C.L. (1954). *Sea-angling fishes of the Cape.* Cape Town: Juta and Co. 303pp.

Bohnsack, J.A. (1981). Effects of piscivorous predator removal on coral reef fish community structure. In *Third Pacific Workshop* (ed. G.M. Gaillet and C.A. Simenstad), pp. 258-267. Pacific Grove, California: University of Washington.

Bohnsack, J.A. (1989). Are high densities of fishes at artificial reefs the result of habitat limitation or behavioral preference? *Bulletin of Marine Science* **44,** 631-645.

Bohnsack, J.A. (1990). The potential of marine fishery reserves for reef fish management in the U.S. southern Atlantic. Miami: NOAA, 2pp.

Bohnsack, J.A. (1996a). Marine reserves, zoning, and the future of fishery management. Fisheries **21,** 14-16.

Bohnsack, J.A. (1996b). Maintenance and recovery of reef fishery productivity. In *Reef Fisheries* (ed. N.V.C. Polunin and C.M. Roberts), pp. 283-313. London: Chapman & Hall.

Bohnsack, J.A. and Ault, J.S. (1996). Management strategies to conserve marine biodiversity. *Oceanography* **9,** 73-82.

Botsford, L.W., Wing, S.R. and Largier, J.L. (1998). Population dynamics and management implications of larval dispersal. *South African Journal of marine Science* **19,** 131-142.

Brock, R.E. (1994). Beyond fisheries enhancement: artificial reefs and ecotourisms. *Bulletin of Marine Science* **55,** 1181-1188.

Brokensha, M.M. (1990). Problems associated with the implementation of linefish management measures. In *South African National Scientific Programmes Report* No 167, 112-114.

Brouwer, S.L., Mann, B.Q., Lamberth, S.J., Sauer, W.H.H. and Erasmus, C. (1997). A survey of the South African shore-angling fishery. *South African Journal of marine Science* **18,** 165-177.

Buckley, R. (1982). Marine habitat enhancement and urban recreational fishing in Washington. *Marine Fisheries Review* **44,** 28-37.

Buckley, R. (1989). Artificial reefs should only be built by fishery managers and researchers. *Bulletin of Marine Science* **44,** 1054-1056.

Bundy, A. (1996). Conflicts in fisheries: an integrative approach, with management options. In *2nd World Fisheries Congress*, vol. 1 (ed. D.A. Hancok and J.P. Beumer), Brisbane, Australia.

Burger, L.F. (1991). The distribution patterns and community structure of the Tsitsikamma rocky littoral ichthyofauna. MSc thesis, Department of Ichthyology and Fisheries Science: Rhodes University. 116pp.

Butterworth, D.S., Punt, A.E., Borchers, D.L., Pugh, J.B. and Hughes, G.S. (1989). A manual of mathematical techniques for linefish assessment. *South African National Scientific Programmes Report*, No 160. 89pp.

Buxton, C.D. (1984). Feeding biology of the roman *Chrysoblephus laticeps* (Pisces: Sparidae). *South African Journal of Marine Science* **2,** 33-42.

Buxton, C.D. (1990a). Problems associated with the management of reef fishes. *South African National Scientific Programmes Report,* No 167, 41-45.

Buxton, C.D. (1990b). Marine reserves – prospects for the future. *South African National Scientific Programmes Report,* No 167, 102-106.

Buxton, C.D. (1993a). Life-history changes in exploited reef fishes on the east coast of South Africa. *Environmental Biology of Fishes* **36,** 47-63.

Buxton, C.D. (1993b). Marine reserves – the way ahead. In The Second South African Marine Linefish Symposium, Special Publication No 2 (ed. L.E. Beckley and R.P. van der Elst), pp. 170-174. Durban: Oceanographic Research Institute.

Buxton, C.D. (1995). Reef fish management. *S A Commercial Marine* **4,** 10-11.

Buxton, C.D. and Allen, J.C. (1989). Mark and recapture studies of two reef sparids in the Tsitsikamma Coastal National Park. *Koedoe* **32,** 39-45.

Buxton, C.D. and Smale, M.J. (1984). A preliminary investigation of the marine ichthyofauna in the Tsitsikamma Coastal National Park. Koedoe **27,** 13-24.

Buxton, C.D. and Smale, M.J. (1989). Abundance and distribution patterns of three temperate marine reef fish (Teleostei: Sparidae) in exploited and unexploited areas off the Southern Cape coast. *Journal of Applied Ecology* **26,** 441-451.

Buxton, C.D. and Clarke, J.R. (1986). Age, growth and feeding of the blue hottentot *Pachymetopon aeneum* (Pisces: Sparidae) with notes on reproductive biology. *South African Journal of Zoology* **21,** 33-38.

Caley, M.J. and St John, J. (1996). Refuge availability structures assemblages of tropical reef fishes. *Journal of Animal Ecology* **65,** 414-428.

Causey, B.D. (1995). Enforcement in marine protected areas. In *Marine Protected Areas: Principles and Techniques for Management* (ed. S. Gubbay), pp. 119-148. London: Chapman & Hall.

Carr, M.H. and Reed, D.C. (1993). Conceptual issues relevant to marine harvest refuges: examples from temperate reef fishes. *Canadian Journal of Fisheries and Aquatic Science* **50,** 2019-2028.

Chater, S. (1994). The ORI artificial reefs. *Sea Tales* 18-19.

Chater, S.A., Beckley, L.E., Garrat, P.A., Ballard, J.A. and van der Elst, R.P. (1993). Fishes from offshore reefs in the St Lucia and Maputuland Marine Reserves, South Africa. *Lammergeyer* **42,** 1-17.

Chritensen, M.S. (1978). Trophic relationships in juveniles of three species of sparid fishes in the South African marine littoral. *Fisheries Bulletin* **76,** 389-401.

Clark, C.W. (1996). Marine reserves and the precautionary management of fisheries. *Ecological Applications* **6,** 369-370.

Clark, B.M., Bennett, B.A. and Lamberth, S.J. (1994a). Assessment of the impact of commercial beach-seine netting on juvenile teleost populations in the surf zone of False Bay, South Africa. *South African Journal of marine Science* **14,** 255-262.

Clark, B.M., Bennett, B.A. and Lamberth, S.J. (1994b). A comparison of the ichthyofauna of two estuaries and their adjacent surf zones, with an assessment of the effects of beach-seining on the nursery function of estuaries for fish. *South African Journal of marine Science* **14,** 121-131.

Clark, J.R., Causey, B. and Bohnsack, J.A. (1989). Benefits from coral reef protection: Looe Key Reef, Florida. *Coastal Zone* **4,** 3076-3086.

Clarke, J.R. and Buxton, C.D. (1989). A survey of the recreational rock-angling fishery at Port Elizabeth, on the south-east coast of South Africa. *South African Journal of marine Science* **8,** 183-194.

Clarke, K.R. (1993). Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology* **18,** 117-143.

Claverie, M.F. (1990). Keynote address. *South African National Scientific Programmes Report*, No 167, 1-8.

Cliff, G. (1983). Early colonization by fish of an artificial reef in False Bay, South Africa. *Transactions of the Royal Society of South Africa* **45,** 63-71.

Coetzee, C. (1993). Law enforcement as a management technique in Natal coastal fisheries. In *The Second South African Marine Linefish Symposium* (ed. L.E. Beckley and R.P. van der Elst), pp. 177-178. Durban: Oceanographic Research Institute.

Coetzee, P.S. (1986). Diet composition and breeding cycle of blacktail, *Diplodus sargus capensis* (Pisces: Sparidae), caught off St Croix Island, Algoa Bay, South Africa. *South African Journal of Zoology* **21,** 237-243.

Compagno, L.J.V., Ebert, D.A. and Smale, M.J. (1989). *Guide to the sharks and rays of Southern Africa,* 160pp. Cape Town: Struik Publishers.

Cook, P.A. (1990). The effect of spearfishing on the marine linefishery in South Africa. *South African National Scientific Programs Report,* No 167, 16-20.

Cram, D.L. (1970). A suggested origin for the cold surface water in central False Bay. *Transactions of the Royal Society of South Africa* **39,** 129-137.

Crawford, R.J.M. and Crous, H.B. (1982). Trends in commercial handline catches of redfishes along the Southern Cape coast, Republic of South Africa. *Koedoe* **25,** 13-31.

Crous, H. (1994). *Saltwater fishing in Southern Africa*, 238pp. Cape Town: Struik Publishers.

Davidson, R.J. and Chadderton, W.L. (1994). Marine reserve site selection along the Abel Tasman National Park coast, New Zealand: consideration of subtidal rocky communities. *Aquatic Conservation: Freshwater and Marine Ecosystems* **4,** 153-167.

Day, J.H. (1970). The biology of False Bay, South Africa. *Transactions of the Royal Society of South Africa* **39,** 211-221.

Denis, G.D. (1988). The impact of spearfishing on the abundance of large predatory coral reef fishes. Report. Puerta de Tiarra: Department of Marine Science, University of Puerto Rico.

Dugan, J.E. and Davis, G.E. (1993). Applications of marine harvest refugia to coastal fisheries management. *Canadian Journal of fisheries and Aquatic Science* **50,** 2029-2042.

Emanuel, B.P., Bustimante, R.H., Branch, G.M., Eekhout, S. and Oodendaal, F.J. (1992). A zoogeographic and functional approach to the selection of marine reserves on the west coast of South Africa. In *Benguela Trophic Functioning.* (ed. Paine, A.I.L., Brink, K.H., Mann, K.H. and Hillborn, R.), *South African Journal of marine Science* **12,** 341-354.

Field, J.G., Clarke, K.R. and Warwick, R.M. (1982). A practical strategy for analysing multispecies distribution patterns. *Marine Ecology Progress Series* **8,** 37-52.

Fricke, A.H., Koop, K. and Cliff, G. (1982). Colonization and viability of an artificial steel reef in False Bay, South Africa. *Transactions of the Royal Society of South Africa* **44,** 499-511.

Fuentes, H.R. (1985). Notes concerning the impact of spearfishing on a population of sheephead *Semicossyphus maculatus* (Perez 1886). In *Joint International Scientific Diving Symposium,* pp. 151-164. La Jolla, California.

Garrat, P.A. and van der Elst, R.P. (1990). Status of the fishery in Natal and Transkei. South African National Scientific Programmes Report, No 167, 27-31.

Gascon, D. and Miller, R.A. (1982). Space utilization of temperate reef fishes inhabiting small experimental artificial reefs. *Canadian Journal of Zoology* **60,** 798-806.

Gosline, W.A. (1965). Vertical zonation of inshore fishes in the upper water layers of the Hawaiian islands. *Ecology* **46,** 823-831.

Griffiths, M.H., Attwood, C. and Thomson, R. (1999) (in press). New management protocol for the South African linefishery. SANCOR occasional report series.

Guastella, L. and Smith, A. (1997). Artificial reefs. *Divestyle* **8,** 54-58.

Guastella, L.A. and Nellmapius, S.J. (1993). Marine recreational angling in Natal. In *The Second South African Marine Linefish Symposium,* Special Publication No 2 (ed. L.E. Beckley and R.P. van der Elst), pp. 113-117. Durban: Oceanographic Research Institute.

Hecht, T. (1990). Fundamentals of marine linefish research in Southern Africa. *South African National Scientific Programmes Report*, No 167, 75-81.

Hemming, B. and Pierce, B. (1996). Fisheries enforcement: our last fisheries management frontier! In *2nd World Fisheries Congress,* vol. 1 (ed. D.A. Hancock and J.P. Beumer), pp. 143-144. Brisbane, Australia.

Hixon, M.A. (1980). Competitive interactions between California reef fishes of the genus *Embiotica*. *Ecology* **61,** 918-931.

Hockey, P.A.R. and Branch, G.M. (1997). Criteria, objectives and methodology for evaluating marine protected areas in South Africa. *South African Journal of marine Science* **18,** 369-383.

Holbrook, S.J., Swarbrick, S.L., Schmitt, R.J. and Ambrose, R.F. (1992). Reef architecture and reef fish: correlations of population densities with attributes of subtidal rocky environments. In *Proceedings of the Second International Temperate Reef Symposium,* (Ed. C.N. Battershill), pp99-106. Auckland, New Zealand.

Hutchings, L. (1993). Closing comments. In *The Second South African Marine Linefish Symposium*, Special Publication No 2 (ed. L.E. Beckley and R.P. van der Elst), pp. 203-205. Durban: Oceanographic Research Institute.

Iverson, E.S. (1996). *Living marine resources: their utilization and management*, 403pp. New York: Chapman & Hall.

Jackson, W.P. (1991). False Bay 21 years on – an environmental assessment. *Transactions of the Royal Society of South Africa* **47,** 363-784.

Jennings, S. and Lock, J.M. (1996). Population and ecosystem effects of reef fishing. In *Reef Fisheries* (ed. Polunin, N.V.C. and C.M. Roberts), pp. 193-218. London: Chapman & Hall.

Johnson, J.E. (1985a). Spearfishing competitions in South Australia (1983/1984). 1. Shore and boat events, Report Number 12. Adelaide: Department of Fisheries.

Johnson, J.E. (1985b). Spearfishing competitions in South Australia (1983/1984). II. Australian Skindiving Convention, Report Number 13. Adelaide: Department of Fisheries.

Jones, G.P. (1984). The influence of habitat and behavioural interactions on the local distribution of the wrasse, *Pseudolabrus celidotus*. *Environmental Biology of Fishes* **10,** 43-58.

Joubert, C.S.W. and Hanekom, P.B. (1980). A study of feeding in some inshore reef fish of the Natal coast, South Africa. *South African Journal of Zoology* **15,** 262-274.

Lamberth, S. and Bennett, B.A. (1993). When is an "angling" fish not an angling fish? Evidence from beach-seine catches in False Bay. In *The Second South African Marine Linefish Symposium*, Special Publication No 2 (ed. L.E. Beckley and R.P. van der Elst), pp. 153-156. Durban: Oceanographic Research Institute.

Lamberth, S.J., Bennett, B.A. and Clark, B.M. (1994). Catch composition of the commercial beach-seine fishery in False Bay, South Africa. *South African Journal of marine Science* **14,** 69-78.

Lamberth, S.J., Bennett, B.A. and Clark, B.M. (1995a). The vulnerability of fish to capture by commercial beach-seine nets in False Bay, South Africa. *South African Journal of marine Science* **15,** 25-31.

Lamberth, S.J., Bennett, B.A. and Clark, B.M. (1995b). Seasonality of beach-seine catches in False Bay, South Africa, and implications for management. *South African Journal of marine Science* **15,** 157-167.

Lamberth, S.J., Mann, B.Q., Brouwer, S. and Sauer, W. (1996). An evaluation of fisher's perceptions, and management, of the South African marine linefishery. In 2^{nd} *World Fisheries Congress*, vol. 1 (ed. D.A. Hancock and J.P. Beumer), pp. 146. Brisbane, Australia.

Lamberth, S.J. and Griffiths, C.L. (1997). The distribution of total catch and effort between all sectors of the linefishery in the Southwestern Cape. Report. Cape Town: University of Cape Town.

Larson, R.J. (1980). Territorial behavior of the black and yellow rockfish and gopher rockfish (Scorpaenidae, *Sebastes*). *Marine Biology* **58,** 111-122.

Levin, P.S. and Hay, M.E. (1996). Responses of temperate reef fishes to alterations in algal structure and species composition. *Marine Ecology Progress Series* **134,** 37-47.

Lewis, S.M. and Wainwright, P.C. (1985). Herbivore abundance and grazing intensity on a Caribbean coral reef. *Journal of Experimental Marine Biology and Ecology* **87,** 215-228.

Lincoln Smith, M.P., Bell, J.D., Pollard, D.A. and Russel, B.C. (1989). Catch and effort of competition spearfishermen in southeastern Australia. *Fisheries Research* **8,** 45-61.

Love, M.S. and Ebeling, A.W. (1978). Food and Habitat of three switch-feeding fishes in the kelp forests off Santa Barbara, California. *Fishery Bulletin* **76,** 257-271.

Maas, T. (1995). *Bluewater Hunting and Freediving*, 214pp. Ventura: Bluewater Freedivers.

Mann, B.Q. (1992). Aspects of the Biology of Two Inshore Sparid Fishes (*Diplodus sargus capensis* and *Diplodus cervinus hottentotus*) off the South-East Coast of

South Africa. MSc thesis, Department of Ichthyology and Fisheries Science: Rhodes University. 125pp.

Mann, B.Q. and Buxton, C.D. (1993). The biology and management of blacktail (*Diplodus sargus capensis*) and zebra (*Diplodus cervinus hottentotus*) off the South-eastern Cape coast. In *The Second South African Marine Linefish Symposium*, Special Publication No 2 (ed. L.E. Beckley and R.P. van der Elst), pp. 26-27. Durban: Oceanographic Research Institute.

Mann, B.Q., Scott, G.M., Mann-Lang, J.B., Brouwer, S.L., Lamberth, S.J., Sauer, W.H.H. and Erasmus, C. (1997). An evaluation of participation in and management of the South African spearfishery. *South African Journal of marine Science* **18,** 179-193.

Meier, M.H. (1989). A debate on responsible artificial reef development. *Bulletin of Marine Science* **44,** 1051-1057.

Nepgen, C.S.de V. (1977). The biology of the hottentot *Pachymetopon blochii* (Val.) and the silverfish *Argyrozona argyrozona* (Val.) along the Cape South-west coast. *Sea Fisheries Branch Investigational Report*, No 105, 1-35.

Nepgen, C.S.de V. (1982). Diet of predatory and reef fish in False Bay and possible effects of pelagic purse-seining on their food supply. *Fisheries Bulletin of South Africa* **16,** 75-93.

Payne, A.I.L. and Crawford, R.J.M. (1989). *Oceans Of Life Off Southern Africa*, 380pp. Cape Town: Vlaeberg Publishers.

Penney, A.J. (1990). Long- and short-term catch trends in the South African commercial linefishery. *South African National Scientific Programmes Report*, No 167, 38-41.

Penney, A.J. (1991). The interaction and impact of net and linefisheries in False Bay. *Transactions of the Royal Society of South Africa* **47**, 663-682.

Penney, A.J. (1993). The National Marine Linefish System. In *The Second South African Marine Linefish Symposium*, Special Publication No 2 (ed. L.E. Beckley and R.P. van der Elst), pp. 68-72. Durban: Oceanographic Research Institute.

Penney, A.J., Buxton, C.D., Garrat, P.A. and Smale, M.J. (1989). The commercial marine linefishery. In *Oceans of Life off Southern Africa* (ed. Payne, A.I.L. and R.J.M. Crawford), pp. 214-229. Cape Town: Vlaeberg Publishers.

Penrith, M.J. (1972). The behaviour of reef-dwelling sparid fishes. *Zoologica Africana* **7**, 43-48.

Pollard, D.A. (1996). Marine harvest refugia as a tool for inshore fish population enhancement. In *2nd World Fisheries Congress*, vol. 1 (ed. D.A. Hancock and J.P. Beumer), pp. 82. Brisbane, Australia.

Polovina, J.J. (1989). Should anyone build reefs? *Bulletin of Marine Science* **44**, 1056-1057.

Potts, T.A. and Hulbet, A.W. (1994). Structural influences of artificial and natural habitats on fish aggregations in Onslow Bay, North Carolina. *Bulletin of Marine Science* **55**, 609-622.

Prochazka, K. (1994). Habitat partitioning in shallow-water cryptic ichthyofaunal communities in the Western and Southwestern Cape, South Africa. PhD Thesis, Zoology Department: University of Cape Town. 108pp.

Pulfrich, A. and Griffiths, C.L. (1988a). Feeding biology of the hottentot, *Pachymetopon blochii* (Val.), with an estimate of daily ration. *South African Journal of Zoology* **23**, 196-207.

Pulfrich, A. and Griffiths, C.L. (1988b). The fishery for hottentot *Pachymetopon blochii* Val. In the South-Western Cape. *South African Journal of marine Science* **7,** 227-241.

Roberts, C.M. and Polunin, N.V.C. (1991). Are marine reserves effective in management of reef fisheries? *Reviews in Fish Biology and Fisheries* **1,** 65-91.

Roberts, C.M. and Polunin, N.V.C. (1993). Marine reserves: simple solutions to managing complex fisheries? *Ambio* **22,** 363-368.

Roberts, C.M. (1996). Settlement and beyond: population regulation and community structure of reef fishes. In *Reef Fisheries* (ed. N.V.C. Polunin and C.M. Roberts), pp. 85-112. London: Chapman & Hall.

Rowley, R.J. (1994). Marine reserves in fisheries management. *Aquatic Conservation: marine and Freshwater Ecosystems* **4,** 233-254.

Rudstam, L.G. and Magnuson, J.J. (1985). Predicting the vertical distribution of fish populations: analysis of Cisco, *Coregonus artedii*, and yellow perch, *Perca flavescens*. *Canadian Journal of Fisheries and Aquatic Science* **42,** 1178-1188.

Russ, G.R. and Alcala, A.C. (1989). Effects of intense fishing pressure on an assemblage of coral reef fishes. *Marine Ecology Progress Series* **56,** 13-27.

Russ, G.R. and Alcala, A.C. (1996a). Marine reserves: rates and patterns of recovery and decline of large predatory fish. *Ecological Applications* **6,** 947-961.

Russ, G.R. and Alcala, A.C. (1996b). Do marine reserves export adult fish biomass? Evidence from Apo Island, central Philippines. *Marine Ecology Progress Series* **132,** 1-9.

Sauer, W.H.H., Penney, A.J., Erasmus, C, Mann, B.Q., Brouwer, S.L., Lamberth, S.J. and Stewart, T.J. (1997). An evaluation of attitudes and responses to monitoring and management measures for the South African boat-based linefishery. *South African Journal of marine Science* **18,** 147-163.

Schoeman, F. and Schoeman, S. (1990). *Strike! The book on Salt Water Fishing in Southern Africa*, 616pp. Somerset West: Strike Publications & Promotions.

Seaman, W., Buckley, R.M. and Polovina, J.J. (1989). Advances in knowledge and priorities for research, technology and management related to artificial aquatic habitat. *Bulletin of Marine Science* **44,** 527-532.

SFRI (1996). Marine Conservation Do's & Don'ts. Tandym Print.

SFRI (1998). Marine Conservation Do's & Don'ts. Tandym Print.

Shpigel, M. and Fishelson, L. (1989). Habitat partitioning between species of the genus *Cephalopholis* (Pisces, Serranidae) across the fringing reef of the Gulf of Aquaba (Red Sea). *Marine Ecology Progress Series* **58,** 17-22.

Shulman, M.J. (1985). Coral reef fish assemblages: intra-interspecific competition for shelter sites. *Environmental Biology of Fishes* **13,** 81-92.

Siegfried, W.R. and Davies, B.R. (1982). Conservation of ecosystems: Theory and practice. *South African National Scientific Programmes Report*, No 61, 97pp.

Smith, M.M. and Heemstra, P.C. (eds.) 1988. *Smith's Sea Fishes*, 1048pp. Johannesburg: Southern Book Publishers.

Smith, J.L.B. and Smith, M.M. (1966). *Fishes of the Tsitsikamma Coastal National Park*, 161pp. National Parks Board of Trustees of the Republic of South Africa.

Stephens, J.S., Morris, P.A., Zerba, K. and Love, M. (1984). Factors affecting fish diversity on a temperate reef: the fish assemblage of Palos Verdes Point, 1974-1981. *Environmental Biology of Fishes* **11,** 259-275.

Taylor, V. (1993a). The effect of recreational rock and surf angling on fish resources along the Cape coast. In *The Second South African Marine Linefish Symposium*, Special Publication No 2 (ed. L.E. Beckley and R.P. van der Elst), pp. 186-189. Durban: Oceanographic Research Institute.

Taylor, R.H. (1993b). Carrying capacity: the limit to sport fishing. In *The Second South African Marine Linefish Symposium*, Special Publication No 2 (ed. L.E. Beckley and R.P. van der Elst), pp. 165-169. Durban: Oceanographic Research Institute.

Terry, C.B. and Stephens Jr, J.S. (1976). A study of the orientation of selected embiotocid fishes to depth and shifting seasonal vertical temperature gradients. *Bulletin of the Southern California Academy of Sciences* **75,** 170-183.

Theunissen, T. (1998). The spearo's reply to the bubble blowers. *Out There* **26,** 57-59.

Thresher, R.E. (1983). Environmental correlates of the distribution of planktivorous fishes in the One Tree Reef lagoon. *Marine Ecology Progress Series* **10,** 137-145.

Thresher, R.E. and Gunn, J.S. (1986). Comparative analysis of visual census techniques for highly mobile, reef-associated piscivores (Carangidae). *Environmental Biology of Fishes* **17,** 93-116.

Tilney, R.L., Nelson, G., Radloff, S.E. and Buxton, C.D. (1996). Ichthyoplankton distribution and dispersal in the Tsitsikamma National Marine Park marine reserve, South Africa. *South African Journal of Marine Science* **17,** 1-14.

Tilzey, R.D.J. (1996). Allocation issues between recreational and commercial fishers: the Australian experience. In *2^{nd} World Fisheries Congress*, vol. 1 (ed. D.A. Hancock and J.P. Beumer). Brisbane, Australia.

Trow, B.E. (1982). A preliminary study on the diet and feeding of the "frans madame", *Boopsoidea inornata* (Sparidae: Pagellinae). Honours project, Department of Ichthyology and Fisheries Science: Rhodes University. 81pp.

Turner, M. (1988). *Shipwrecks & Salvage in South Africa – 1505 to the present,* 224pp. Cape Town: C Struik.

van der Elst, R.P. (1988). *A Guide to the Common Sea Fishes of Southern Africa*, 398pp. Cape Town: Struik Publishers.

van der Elst, R.P. (1989). Marine recreational angling in South Africa. In *Oceans of Life off Southern Africa* (ed. A.I.L. Payne and R.J.M. Crawford), pp. 164-176. Cape Town: Vlaeberg Publishers.

Van Herwerden, L. (1989). Human recreational activity and its impact on a metropolitan coastline. MSc thesis, Zoology Department: University of Cape Town. 88pp.

van Rooyen, P. (1988). *Diving and Spearfishing in South Africa*, 157pp. Johannesburg: Southern Book Publishers (Pty) LTD.

Walsh, W.J. (1985). Reef fish community dynamics on small artificial reefs: the influence of isolation, habitat structure, and biogeography. *Bulletin of Marine Science* **36,** 357-376.

Ward, C. (1996). Ecotourism and the sea. Responsible travel in the big blue. In *National Marine Day*, Sea Fisheries. pp. 1-2.

West, J.E., Buckley, R.M. and Doty, D.C. (1994). Ecology and habitat use of juvenile rockfishes (*Sebastes* spp) associated with artificial reefs in Puget Sound, Washington. *Bulletin of Marine Science* **55,** 344-350.

Winch, G. (1990a). Deep sea fishing; structure, economics and tournaments. *South African National Scientific Programmes Report*, No 167, 9-12.

Winch, G. (1990b). The angler's view of linefish management and research. *South African National Scientific Programmes Report*, No 167, 109-111.

Zar, J.H. (1984). *Biostatistical analysis.* Englewood Cliffs, N.J.: Prentice-Hall.

Zoutendyk, P. (1982). A diving survey of two reefs in Plettenberg Bay. In *Agulhas Bank Studies*, Report No. 9, 1-9.