A RETROSPECTIVE ASSESSMENT OF THE PORT ALFRED LINEFISHERY WITH RESPECT TO THE CHANGES IN THE SOUTH AFRICAN FISHERIES MANAGEMENT ENVIRONMENT

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ABSTRACT

Since the study on the Port Alfred/Kenton-on-Sea/Boknes linefishery by Hecht and Tilney (1989) there have been substantive changes to the linefish management environment in South Africa. Using the Port Alfred linefishery as a model, the aim of this study was to evaluate the effectiveness of the linefish management regulations that were implemented by Marine and Coastal Management (MCM) since 1992, and to assess the behaviour and 'status' of the fishery in response to these changes.

Changes to both the licensing structure and catch regulations have had a significant effect on the functioning of many aspects of the Port Alfred linefishery (fishing effort, catch composition, *cpue*) as well as on the structure of the fishery and its socio-economic profile. Overall commercial *cpue* decreased from 1985 to 1998. Since 1998 there has been a significant increase in *cpue* (from 2.3 $Kg.fisher^{-1}.hour^{-1}$ to a peak of 4.8 $Kg.fisher^{-1}.hour^{-1}$ in 2005). This was attributed to good catches of geelbek, particularly in 2005, 2007 and 2008 (during these years geelbek contributed an average of 35% to the total landings in comparison to a mean contribution of 11% between 1985 and 2004). However, size spectra analysis suggests that the increase in overall *cpue* since 1998 misrepresents the actual status of the fishery. If geelbek is excluded from the analysis on the grounds that it is the only species in the fishery that is highly migratory, susceptible to recruitment fluctuations and it does not contribute to the catches on a year round basis, then the results suggest that the fishery is still in a declining phase despite the 80% reduction in commercial effort and numerous stricter catch regulations (e.g. size/bag limits). Furthermore, the *cpue* of silver kob, which has been the "mainstay" species of the fishery, has consistently declined over the last 23 year period (from 1.69 $Kg.fisher^{-1}.hour^{-1}$ in 1986 to 0.86 $Kg.fisher^{-1}.hour^{-1}$ in 2007). The substantial reduction in commercial effort in the fishery from 33 vessels in 2001 to 13 in 2002 resulted in a shift from commercial to recreational fishing. The number of active commercial vessels in Port Alfred alone decreased from 29 in 1989 (Hecht 1993) to four in 2008. During the same timeframe, the number of regularly active recreational vessels had almost doubled (16 in 1989 to 26 in 2008). Despite the greater number of recreational boats in the fishery they only landed approximately half the average yearly tonnage of the commercial vessels (21.5 and 44.7 tonnes.annum^{-1}, respectively) between 2006 and 2008. This was ascribed to the differences in catch regulations for the two sectors. Furthermore, it was speculated that increasing operating costs and narrowing profit margins have contributed to lower levels of compliance in both the commercial and recreational sectors since 2006. For example, 16% of silver kob landed during 2006-08 were under the minimum size.

Despite the changes made to the regulations since 1998 and the 60.6% reduction in legislated commercial effort in the fishery between 2001 and 2002 it was concluded that the fishery has continued to decline. Except for the good recruitment of geelbek (which may be due to regulatory changes made in 1992) the changes in the management environment have had no measurable positive effect on this fishery. It is recommended that commercial effort should not be allowed to increase beyond the current number of active boats, that there should be an area restriction on all commercial linefish vessels, that the current recreational bag limit for silver kob should be re-assessed, there should also be a concerted and nationally funded effort to educate recreational anglers about the merits of catch and release, and the frequency of catch inspections of both sectors should be increased.

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"People are always good company when they are doing what they enjoy."

-Samuel Butler (1612 – 1680)

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CONTENTS

CHAPTER 1

THE LINEFISHERY AND THE LINEFISH MANAGEMENT ENVIRONMENT IN SOUTH AFRICA

THE SOUTH AFRICAN LINEFISHERY IN CONTEXT

The South African commercial fishing industry is comprised of 18 separately managed fisheries. In 2007 the total annual landings amounted to 653 407 tonnes (Sauer *et al.* 2003, Anon 2008), of which 10 007 tonnes (1.5%) was attributed to thelinefishery, making it the third most important fishery in South Africa in terms of tonnage landed. Snoek (*Thyrsites atun*) and five tuna species contributed 73% of the linefish catch in 2007, 8.5% was attributed to two sciaenid species geelbek (*Atractoscion aequidens*) and silver kob (*Argyrosomus inodorus*) and 81 other species made up the remainder (18.5%) of the landings (Anon 2008).

The coastline that supports these fisheries can be divided into three separate biogeographical provinces (Branch and Clark 2006). The Benguela province, which is characterized by cold nutrient rich water, extends up the west coast and supports a high biomass of fish yet has a low speciesdiversity (Branch and Clark 2006). This area is principally the focus of large-scale industrial fishing operations due to the high concentration of resources. In contrast, the KwaZulu-Natal province (situated on the northern part of the east coast) is characterized by relatively warm, nutrient deficient water from the Agulhas current. This region supports a high diversity of species but a lower biomass, and together with the density of surrounding human populations, is characterized by a dominance of recreational and subsistence fishers. The broadening of the continental shelf along the south east coast forces the Agulhas current offshore creating a dynamic intermediate zone influenced by both the cool Benguela system and the warm Agulhas current (Branch and Clark 2006).

These three biogeographical provinces play host to a combination of commercial, recreational and subsistence linefisheries. Furthermore, Griffiths (2000) draws reference to six separate linefish managerial regions within these three biogeographical provinces. These are presented in Figure 1.1. The Port Alfred linefishery is situated in the South-Eastern Cape.

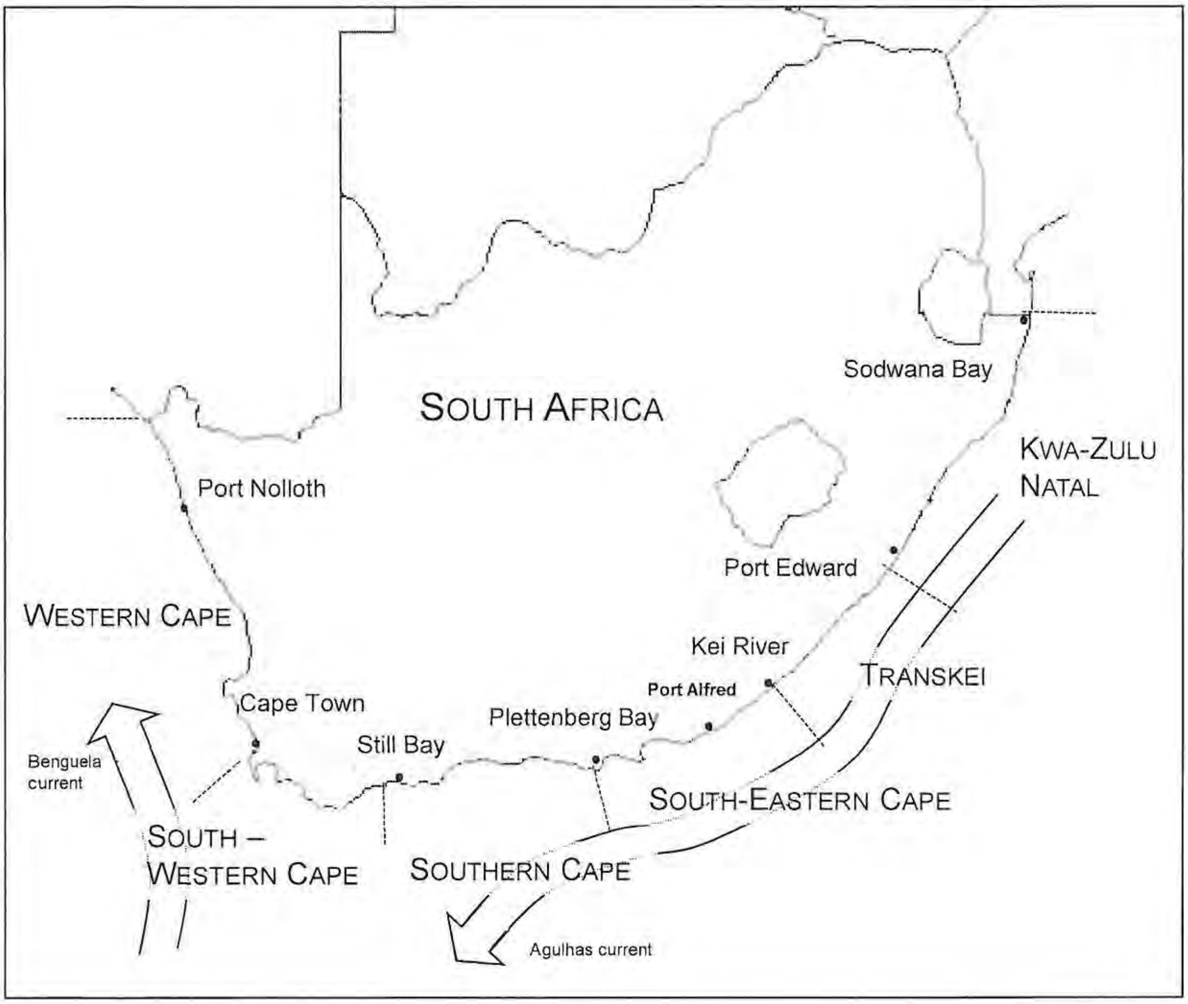

Fig. 1.1 The South African coastline divided into six fisheries management provinces (modified from Griffiths 2000).

The South African commercial linefishery consists primarily of small (3m to 8m) and medium (8m to 15m) ski-boats, 'bakkies'[1], 'chukkies'[2] and deck boats (Brouwer and Buxton 2002, Mather *et al.* 2003). As with the recreational fishers, they operate out of harbours, smaller fishing ports, river mouths and from beaches (often associated with urban nodes; Sauer *et al.* 2003).

In 2006 a total of 80 commercial linefish rights were allocated to the broader 'Eastern Cape' area from Cape Infanta to Port St. Johns (Augustyn 2006, DEAT 2006). Effectively this represented a reduction of around 80% of the commercial licences that were operative prior to 2002. However, it should be noted that even before the licence re-structuring it was estimated that less than 20% of the vessels caught more than 80% of the total landings (Sauer *et al.* 2003). Presently, the total applied commercial effort for the entire South African linefishery is capped at 455 vessels and 3450 crew (Anon 2008).

The recreational component of the South African marine linefishery consists of estuarine anglers fishing from boats or banks (over 50 000 countrywide), shore-based rock and surf anglers (around 853 500) and boat-based recreational anglers (ca. 31 860) (Leibold and van Zyl 2008). Exploiting similar species and using comparable vessels/gear to the commercial operators, the recreational ski-boat fishers (an estimated 1180 between Still Bay and Kei Mouth in 1997) account for an important economic component of the linefishery in the Eastern Cape, and prior to the reduction of commercial licences in 2002, landed an estimated 22% (410 tonnes) of what the commercial operators catch yearly (Brouwer and Buxton 2002, Mather *et al.* 2003).

The Eastern Cape linefishery is a multi-species fishery. A total of 46 teleost and 18 elasmobranch species have been recorded in the shore-based linefishery, while 34 teleost and 10 elasmobranch species comprise the bulk of the deep-sea linefishery (Brouwer and Buxton 2002). However, the greatest proportion of the catch in the various sectors of the linefishery was made up of only a few species. In the Eastern Cape only ten species (which included six sparids,two sciaenid species) contributed 75, 83 and 90 percent of the shore-based linefishery, recreational boat-based and commercial boat-based linefish catch, respectively (Brouwer and Buxton 2002).

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[1] 'Bakkie' – small rowing boat (3m) used on the west coast by traditional fishing communities

[2] 'Chukkie' – medium sized wooden/glass-fibre vessels (7 to 9m) with inboard engines, also used by the traditional fishing communities mainly along the Southern Cape coast.

Despite the management measures that were put in place in 1985 (bag limits, size limits, closed seasons and marine reserves) there is strong evidence of declining catch trends for both the shore-based and the commercial and recreational boat based linefisheries (Hecht and Tilney 1989, Penney *et al.* 1999, Griffiths 2000, Brouwer & Buxton 2002, Fennessy 2003, Mann *et al.* 2003). Moreover, there have been significant changes in the catch composition and the stocks of several species are considered to have collapsed or are highly vulnerable, and most of the species are overexploited (Hecht and Tilney 1989, Buxton 1993, McGrath 1997, Griffiths 1997c, Griffiths 2000, Brouwer and Buxton 2002). This can be attributed to three main factors namely, the previous non-research based fishing regulations, poor compliance and vulnerability of certain species, particularly 'K-selected' species with a long life span, slow growth and that are highly resident (Buxton 1990, Buxton 1993, Coleman 2000). Table 1.1 summarizes the status of nine of the more important species in the Eastern Cape linefishery and shows the poor condition in which the fishery found itself in more than a decade ago. With the exception of panga and yellowtail, the stocks of most of the commercially exploited species in the Eastern Cape had collapsed (Booth and Buxton 1997, Booth and Punt 1998, Griffiths 2000).

Table 1.1 Stock status of eight important commercial and recreational linefish species in the Eastern Cape boat-based linefishery.

Family	Species	Stock status		Reference
Sciaenidae	Geelbek (*Atractoscion aequidens*)*	Collapsed	Catch rate 1.46% of historical catch rates	Griffiths 2000
	Silver kob (*Argyrosomus inodorus*)**	Collapsed	2.9 – 9.8% of pristine spawner biomass	Griffiths 1997a, 2000
Sparidae	Carpenter (*Argyrozona argyrozona*)**	Collapsed	Catch rate 25.88% of historic catch rates	Griffiths 2000
	Dageraad (*Chrysoblephus cristiceps*)**	Collapsed	5% of pristine spawner biomass	Buxton 1993
	Panga (*Pterogymnus laniarius*)	Under-exploited	67% of pristine spawner biomass	Booth and Buxton 1997
	Roman (*Chrysoblephus laticeps*)**	Collapsed	Catch rate 4.65% of historical catch rates	Griffiths 2000
	Red stumpnose (*C. gibbiceps*)**	Collapsed	Catch rate 5.35% of historic catch rates	Griffiths 2000
	Seventy four (*Polysteganus undulosus*)**	Collapsed	Catch rate 0.22% of historic catch rates	Griffiths 2000

* Nomadic/ migratory species
** Resident species with home range

TRENDS IN FISHERIES MANAGEMENT THAT HAVE SHAPED LINEFISH MANAGEMENT IN SOUTH AFRICA

In light of the well documented decline in fish stocks worldwide, the last two decades have seen much scepticism and criticism surrounding resource management and governance of all natural marine resources across the globe (*inter alia* Caddy 1999, FAO 2004, Smith and Link 2005, Mahon *et al.* 2008). The shortcomings of traditional management strategies led to the realization that biological systems cannot be viewed as predictable and controllable (Mahon *et al.* 2008). It was found to be no longer sufficient to focus management on a single target species in a top-down regulatory manner and the impacts of fishing on the structure and functioning on the entire ecosystem needed to be considered as well (Gislason *et al.* 2000, Mahon *et al.* 2008). This new paradigm in fisheries management is now generally known as ecosystem-based fisheries management (EBFM) or the ecosystems approach to fisheries (EAF) (Pikkitch *et al.* 2004, Smith *et al.* 2007). These concepts in fisheries management are particularly pertinent to traditional marine linefisheries. Although there are desired or targeted species, the hook and line method of capture and the diverse and large areas over which fishing takes place make them relatively unselective (Goetz *et al.* 2007). Traditional linefisheries also support a broad user group with different requirements and in many areas linefisheries contribute to food security or towards the income of local areas and or communities (Branch *et al.* 2002, Isaacs 2006).

In 1982 the United Nations Convention on Law of the Sea (UNCLOS; the widely applied basic legal framework governing the use of the oceans and seas) outlined provisions for the protection and preservation of marine ecosystems (United Nations 1983). Although it did not implicitly call for an ecosystem approach to fisheries management, it paved the way for the more fisheries focused agreements such as the 1995 UN Fish Stocks Agreement (UNFSA), the FAO Code of Conduct for Responsible Fisheries in 1995 and the Jakarta Mandate on Coastal and Marine Biodiversity (1996) all of which encompass the broader goals of an EBFM (Parsons 2004).

An important element of an ecosystem approach is the recognition of the human factor as an ecosystem component. Specifically, the dependence of coastal communities on marine resources, the impact they have on the systems and their role in the management and restoration of those systems (McCay 2003). Worldwide, the influence of social scientists on marine fisheries policy is evident in several ways; firstly, there is a greater emphasis on the democratization of the policy process, thereby giving a greater voice to the fishers and members of the fishery dependent communities. Secondly, an evident emphasis on community level impacts. Thirdly, the assignment of fishing rights to particular entities, the so-called 'Rights Based Movement'. Lastly, social scientists have presented a critique of the prevailing scientific paradigms and the use of science in fisheries management (Jentoft *et al.* 1998, McCay 2003). McCay (2003) suggested a new paradigm of 'ecosystem management' that challenged the dominant scientific paradigm of a predictable, determinable relationship between catches and the abundance of fish. Due to high levels of uncertainty and variability, she calls for a humbler role of science in fisheries management (McCay 2003). Jentoft *et al.* (1998) held similar views, pointing out that fisheries management is science based, yet not science governed. It is immediately evident that the implementation of an EBFM system calls for a multi-disciplinary approach to resource management, drawing skills from the fields of science, economics, sociology and politics (Bowen 1997, Lane and Stephenson 1998).

Despite its apparent popularity, an EBFM approach is not without its flaws. Mace (2001) suggests that ecosystem-based management is impeded by greater information requirements, greater complexity, numerous alternative hypotheses on ecosystem structure and function, lack of operational objectives, lack of widely applicable performance measures and a possible reduction of future stock sizes of key species. More importantly, it was suggested that ecosystem-based management strategies cannot be implemented in isolation, and an integration of traditional and ecosystem based approaches should be the way forward (Gislason *et al.* 2000, Mace 2001).

The ecosystem approach to fisheries management has won favour in many countries with well established fisheries policies. Australia released an Oceans Policy in 1998, in Canada the Oceans Act was passed in 1996 (acclaimed to be the first comprehensive ocean management legislation in the world; Parsons 2004) and the EU adopted a form of an ecosystems approaches to fisheries management in the North Sea. These basic principles of ecosystem-based management, in particular social aspects, have to a large extent shaped the development of the post-apartheid fisheries management environment in South Africa (Hutton *et al.* 1997, Hutton *et al.* 2001, Kleinschmidt *et al.* 2003).

THE EVOLUTION AND CURRENT STATUS OF THE SOUTH AFRICAN LINEFISH MANAGEMENT ENVIRONMENT

The first recorded accounts of linefishing in South Africa dates back to the indigenous Khoi and European seafarers in the 1500's (Thompson 1913 cited by Griffiths 2000). Restrictions were first put in place in the second half of the 17^{th} century by van Riebeeck (circa. 1656; Thompson 1913). These restrictions prohibited the sale of fish in order to encourage agriculture.

Minimum size limits for linefish in South Africa were first introduced in 1940 (Sauer *et al.* 2003). These limits were only for selected species and based solely on subjective information, with little knowledge of the life-histories or actual stock status of the species (Griffiths 2000, Sauer *et al.* 2003). It was not until the early 1960's that research into a few species of economic importance was initiated, notably the work on seventy-four (Ahrens 1965). Apart from a few minor restrictions (bag limits, size limits and closed seasons for elf,snoek and kob) no action was taken until 1985 (Sauer *et al.* 2003).

The first comprehensive linefish management framework in South Africa was based on a document compiled by Garratt and van der Elst (1984) . However, this framework was based largely on the perceived vulnerability of economically exploited commercial species and lacked a solid scientific basis (Griffiths 2000). The framework set about revising minimum size limits, bag limits, and commercial species restrictions as well as capping the commercial effort at the 1984 level (Penney 1990a, Griffiths 2000).

For several reasons this framework failed to have any measurable impact on the sustainability of linefish resources (Griffiths 2000). Firstly, the fishery had essentially been an open-access fishery and in an attempt to inhibit the expansion of the fishery, the management framework advocated the introduction of commercial licenses. However, almost all applications for commercial licences were successful as either full-time fishers (A-license) or part-time fishers (B-license). Part-time commercial operators consisted largely of recreational fishers looking to subsidize their sport by selling their catch or subsidizing their income to varying degrees. Although effective in capping effort, the damage had already been done prior to the implementation of licenses (Griffiths 2000). Secondly, the framework was compromised by unclear management guidelines and strong lobby groups which resulted in conciliations between managers and user groups that had little benefit for fish stocks (Sauer *et al.* 2003). One of the elements that were incorporated into the management framework was the implementation of the National Marine Linefish System (NMLS) which made it mandatory for all commercial operators to report their daily catches on a monthly basis and this is ongoing to the present day (Penney 1993, C.Wilke, Marine and Coastal Management, Cape Town, pers. comm. 2007). Although not without its problems, the NMLS has proven itself to be a very valuable data resource.

After the 1994 democratic elections the South African government completely restructured the policies surrounding fisheries governance and management (Kleinschmidt *et al.* 2003, Mather *et al.* 2003, Isaacs 2006, Witbooi 2006, van Stittert *et al.* 2006). Previously, marine resource use in South Africa was governed by a 'top-down', resource based approach (Britz *et al.* 2001, Branch and Clark 2006) and the history of institutionalized racism had left a legacy of inequality amongst users (Hutton *et al.* 1997, van Stittert *et al.* 2006). Although the new fisheries policy, which culminated in the Marine Living Resources Act (Act 18 of 1998; MLRA), took heed of international trends in fisheries management, the policy more importantly attempted to redress injustices of the past and achieve equity in fisheries (van Stittert *et al.* 2006). The management of marine living resources consequently underwent a shift from a traditional 'resource' centred approach to a more holistic 'ecosystem' and 'people' centred approach (Britz *et al.* 2001, van Stittert *et al.* 2006).

The national body responsible for the governance of fisheries in South Africa, Marine and Coastal Management (MCM; formerly the Sea Fisheries Directorate) had the daunting task of facilitating the equitable restructuring of the fishing industry and addressing the developmental needs of the coastal communities with access to marine living resources. This task was aided by the economic and sectoral study of the South African fishing industry (Mather *et al.* 2003, Sauer *et al.* 2003). This study revealed that most marine fisheries were poorly managed (including the linefishery), that marine living resources were not being utilized to their full potential and that economic and social benefits were not being realized (Mather *et al.* 2003). It was suggested that the reasons for these failings included weak institutional structures and coordination, a lack of planning around policy implementation and inadequate compliance (Britz *et al.* 2001, Mather *et al.* 2003).

The shift from a traditional resource-based management approach to a 'people' centred approach inevitably led to numerous problems, and the capacity of MCM to implement the MLRA and ultimately to achieve policy goals came under question (Mather *et al.* 2003, Kleinschmidt *et al.* 2003, Isaacs 2006, van Stittert *et al.* 2006, Witbooi 2006). MCM was traditionally staffed with 'resource' scientists; managers and administrators and staff with 'people' orientated development skills were lacking nationally (van Stittert *et al.* 2006) and particularly in the Eastern Cape (Britz *et al.* 2001). It is no surprise that the introduction of the MLRA effectively started an institutional crisis within MCM and their initial attempt to restructure the fishery failed. In the same vein as previous management frameworks, the initial implementation of the MLRA focused on a 'command and control' style with regards to the distribution of access rights and resource management issues (Mather *et al.* 2003, Witbooi 2006). Among other things the initial implementation of the MLRA lacked user participation in the re-structuring of some fisheries, economic analysis and scenario planning, capacity building and support of new entrants into the fishing industry (Mather *et al.* 2003).

Despite the hiatus within MCM at the time, a new Linefish Management Protocol (LMP) was developed in 1999. Taking the form of a simplified Operational Management Procedure (OMP), the LMP set out to implement management strategies for each important linefish species through a predetermined system of monitoring, assessing and continuous adaptation of management regulations (Griffiths *et al.* 1999b). The LMP had the specific goals of facilitating the recovery and maintaining linefish stocks at optimum levels, ensuring user participation in the management procedure and guaranteeing that the process of granting access rights to the fishery was fair and equitable (goals which were consistent with the MRLA) (Griffiths *et al.* 1999b).

At the end of 2000 and in terms of the Marine Living Resources Act of 1998, the Minister of Environmental Affairs and Tourism declared that linefish stocks were in a state of crisis (Sauer *et al.* 2003). Amongst other measures, this resulted in a dramatic reduction of commercial linefishing rights in the following years, during which the number of commercial licenses was reduced from 2600 to 780 (Sauer *et al.* 2003). Commercial operators holding more than one type of commercial license as well as those fishers who were not solely dependent on linefish for an income (but merely supplementing their income with fishing) came under scrutiny (Griffiths 2000, MCM 2005). Given the economic contribution by linefishing to livelihoods (Leibold and van Zyl 2008), but notwithstanding the need for the drastic reduction in the number of licenses, the author is of the opinion that the economic effects would have been felt both directly and indirectly in coastal fishing communities and that the changes could lead to poor compliance.

In 2000 and 2001, MCM re-addressed broader fisheries policy issues and the implementation thereof (Branch and Clarke 2006). In an attempt to stabilize the linefishery the process of annually allocating linefishing rights was abolished. In 2002 medium-term commercial rights were issued for a period of 4 years. A complete revision of minimum size limits, bag limits, catch restrictions and closed seasons was promulgated in April 2005and in 2006 long-term rights (of up to 15 years) were allocated (Augustyn 2006, DEAT 2006).

The most pertinent changes in the linefisheries management environment since 1985 are summarized in Table 1.2, while Table 1.3 summarises the changes in bag and size limits, closed seasons and catch restrictions applicable to the more important species in the Eastern Cape linefishery.

Table 1.2 Summary of pertinent changes to the commercial and recreational linefishery management environment in South Africa between 1985 and 2007.

Year	Change to the Management Environment
1985	Revised minimum size limits, bag limits and commercial ban on certain species
	Commercial effort capped at 1984 levels
	Allocation of commercial linefish permits (A license - full-time commercial and B license - Part-time commercial)
	Full implementation of the National Marine Linefish data collection System (NMLS)
1992	Revision of minimum legal size limits for certain species
1994	Democratic elections in South Africa
1995	Appointment of a Fisheries Policy Development Committee (Output: Marine Fisheries Policy of 1997)
1998	Marine Living Resources Act (MLRA) of 1998
1999	Development of the Linefish Management Protocol (LMP)
	Introduction of annually allocated fishing rights
2000	Linefishery declared to be in a state of crisis (in terms of MLRA)
2001	One year moratorium on the annual rights allocation process (commercial licenses carried over from 2000)
2002	Medium-term commercial rights allocated (4 year tenure) – commercial effort reduced by ± 80%
	Revision of minimum legal size and bag limits, species catch restrictions, closed seasons
2005	Complete revision of minimum legal size limits, bag limits, species catch restrictions and closed seasons
2006	Allocation of long term rights (15 year tenure)

Table 1.3 Summary of the catch restrictions (minimum legal size, bag limits and closed seasons) for 12 species in the commercial (Comm.) and recreational (Rec.) linefish sectors as they were in the years 1985, 1992, 2002 and 2006. The restrictions applicable to the commercial A- and B-licences in 1985 and 1992 are presented as comm. A and comm. B, respectively.

Species	Size limit (mm TL)				Bag limit (Number of fish/person/day)									
	1985	1992	2002	2006	1985			1992			2002		2006	
					Rec.	Comm.		Rec.	Comm.		Rec.	Comm.	Rec.	Comm.
						B	*A*		*B*	*A*				
Geelbek (*Atractoscion aequidens*)	400	**600**	600	600	-	-	-	**10**	-	-	10	-	2	-
Silver kob (*Argyrosomus inodorus*)	400	400	400	**500**	10	-	-	10	-	-	**5**^{Ŧ}	-	5^{M}	-^{M}
Carpenter (*Argyrozona argyrozona*)	250	250	250	**350**	-	-	-	**10**	-	-	10	-	**4**	-
Dageraad (*Chrysoblephus cristiceps*)	250	**300**	300	**400**	5	5	-	5	5	-	5	-	**1**	**1**
Panga (*Pterogymnus laniarius*)	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Poenskop (*Cymatoceps nasutus*)	250	**500**	500	500	5	5	-	**2**	**2**	**2**	2	2	1	**1**
Roman (*Chrysoblephus laticeps*)	250	**300**	300	300	5	5	-	5	5	-	5	-	2	-
Red steenbras (*Petrus rupestris*)**	250	**400**	400	**600**	5	5	-	**2**	**2**	**2**	2	2	1	**1**
Red stumpnose (*C. gibbiceps*)	250	**300**	300	300	5	5	-	5	5	-	5	-	1	-
Santer (*Cheimerius nufar*)	250	**300**	300	300	5	5	-	**10**	-	-	10	-	5	-
Seventyfour (*Polysteganus undulosus*)	250	**400**	$p^{Ŧ}$	*p*	5	5	-	**2**	**2**	**2**	$p^{Ŧ}$	$p^{Ŧ}$	*p*	*p*
Elf (*Pomatomus saltatrix*)*	300	300	300	300	5	5	-	5	5	-	5	-	4	-

*	Closed season for elf implemented in 1985: 1 September to 30 November; amended in 2006: 1 October to 30 November
**	Closed season for red steenbras implemented in 1985: 1 September to 30 November; amended in 2006: 1 October to 30 November
Ŧ	Regulations were amended in 1998
-	Catch of species is unlimited
M	No more than one fish > 1100mmTL per day
p	Catch of species is prohibited

A HISTORICAL PERSPECTIVE OF THE PORT ALFRED LINEFISHERY

Hecht and Tilney's (1989) work in the mid 1980's was the first time the Port Alfred commercial fishery was studied as a whole. Amongst others, the study outlined the changes that had occurred in the fishery since the 1940's, described the species composition, catch, effort and temporal changes in non-standardised *cpue*, and expressed the fishery in terms of its contribution to the total South African linefish landings. Information Box 1.1 presents the summarized key findings of the work by Hecht and Tilney (1989).

OBJECTIVES AND STRUCTURE OF THE STUDY

The primary aim of this study was to assess the current status of the Port Alfred linefishery (including Kenton-on-Sea and Boknes) and to use the available historical datasets (outlined in Chapter 2) to retrospectively examine the fishery with a suite of both traditional and ecosystem-based metrics (catch rates, catch compositions, length frequency distributions, dominance curves and size spectra analyses). Comparisons could then be drawn between the past and the present to evaluate the effectiveness of the changes in the management environment.

The thesis is structured as follows. In this chapter an attempt has been made to place the South African linefishery into perspective, to sketch the institutional changes in fisheries management in South Africa, highlight the changes in specific linefish management control measures that have been applied since 1985, and to provide a thumbnail sketch of the Port Alfred linefishery as it was in the past. This provides the means with which to contextualize the present investigation. Chapter 2 provides an overview of the datasets used, the general methods applied and the study area. Using the National Marine Linefish System (NMLS) data, Chapter 3 investigates the changes that have occurred in the fishery with respect to effort, *cpue* and catch composition over the 23 year period between 1985 and 2007. Chapter 4 is a comparative investigation of the commercial and recreational sectors of the Port Alfred fishery. CCTV monitoring and access point surveys during 2006 to 2008 were used to calculate fishing effort, catch rates, total landings and catch compositions. These results were compared to those of Hecht and Tilney (1989). Chapter 5 uses the historical length data and the length data collected during the period April 2006 to December 2008 to investigate the effect of management changes on the size structure of selected species in the linefishery. Furthermore, in line with an EAF, community-based metrics were used to assess the effect of management changes on the fish assemblages. Chapter 6 draws general conclusions on the efficacy of management approaches and regulations, highlights a few management considerations and provides management recommendations based on the findings of the thesis.

Information Box 1.1 Summarized description of the Port Alfred linefishery based on the findings of Hecht and Tilney (1989)

- From the 1940's to the 1960's the catch was largely determined by market demand. Dageraad (with an average weight of 3.5kg) comprised the majority of the catch and together with other K-selected reef-dwelling species (sparids and yellowbelly rockcod) contributing roughly 95% of the total catch. A market for silver kob only developed in 1949. The study also noted a significant decrease in commercial catches between 1943 and 1987.

- There was a significant decrease in the population size structure of the various species. The average recorded weights of dageraad and kob dropped from 3.5 kg and 5kg in the period 1940-1960 to 650g and 828g in 1984-1987, respectively. This was coupled with a decrease in the size of the hooks used from 6/0 - 12/0 in 1943 to between 3/0 and 6/0 in 1987.

- In the 1980's the Port Alfred fishery was primarily based on commercial ski-boats that operated between Cape Padrone in the west and Great Fish River Point in the east. There were a total of 54 commercial fishing vessels registered in Port Alfred; 13 class A-licenses and 41 class B-licenses. Of these, 28 operators fished commercially on a full-time basis in 1987, while the other 26 vessels fished on an irregular basis without a commercial crew. The majority of these vessels were 6.7m catamaran vessels, which accommodated a crew of four to five fishers. These vessels operated between 10 to 18 days per month and stayed at sea for an average of 6.5 to 8 hours a day.

- Possibly due to technological advances and declining catches, the area fished increased over time. There was a significant shift from fishing the near shore reefs in the 1940's, to fishing the shallow banks (5 to 6 nautical miles offshore) in the 1960's, and as from the 1980's fishing operations extended up to 14 nautical miles offshore.

- In 1987 the commercial fishery (and its auxiliary service providers) was the third most important industry in Port Alfred. Moreover, during the period 1985 to 1986 the Port Alfred commercial linefishery was the third biggest single-port based linefishery along the South African coast (excluding snoek and yellowtail). The study also showed that between 1980 and 1987 the landings of linefish species increased from 100t to 380t.

- The study pointed out that the submission of commercial catch returns (which was made mandatory in 1985) had inherent problems; these included the lumping of 'redfish' species, lack of detailed effort data and the absence of length frequency data. This made it impossible to assess the status of the many 'lumped' sparid species based purely on a catch and effort basis.

- Between 1982 and 1987 the catch rates of all species declined and concern was expressed on the states of all the species in the fishery. A call was made even then for the serious reduction in fishing effort and better control of access to the fishery.

CHAPTER 2

GENERAL METHODS AND THE SPATIAL EXTENT OF THE FISHERY

INTRODUCTION

This chapter delineates the spatial extent of the fishery, outlines the origins of the various datasets, discusses the limitations of these datasets and presents the methods that were applied.

SPATIAL EXTENT OF THE FISHERY

The Port Alfred/Kenton-on-Sea/Boknes commercial and recreational linefishery spans roughly 90 km of coastline and extends from Cape Padrone (33° 46'S, 26° 28'E) in the west to Great Fish Point in the east (33° 21'S, 27° 6'E). Recreational operators occasionally made trips to Bird Island (33° 49'S, 26° 2'E), which was therefore included in the study area (Fig. 2.1). The geographic extent mirrors the situation as was described by Hecht and Tilney (1989). Private farm land limits access points to urban nodes, notably Port Alfred, Kenton-on-Sea and Boknes. Other, smaller beach launch sites and points are infrequently used. Fishing takes place to a depth of up to 120m and roughly 14 nautical miles offshore.

For the purposes of this study, the fishery was divided into four separate regions representing the extreme west, west, east and extreme east (Fig. 2.1). The regions are roughly equidistant in terms of kilometres of coastline (region A, however, includes Bird Island at the extreme western end of the fishery).

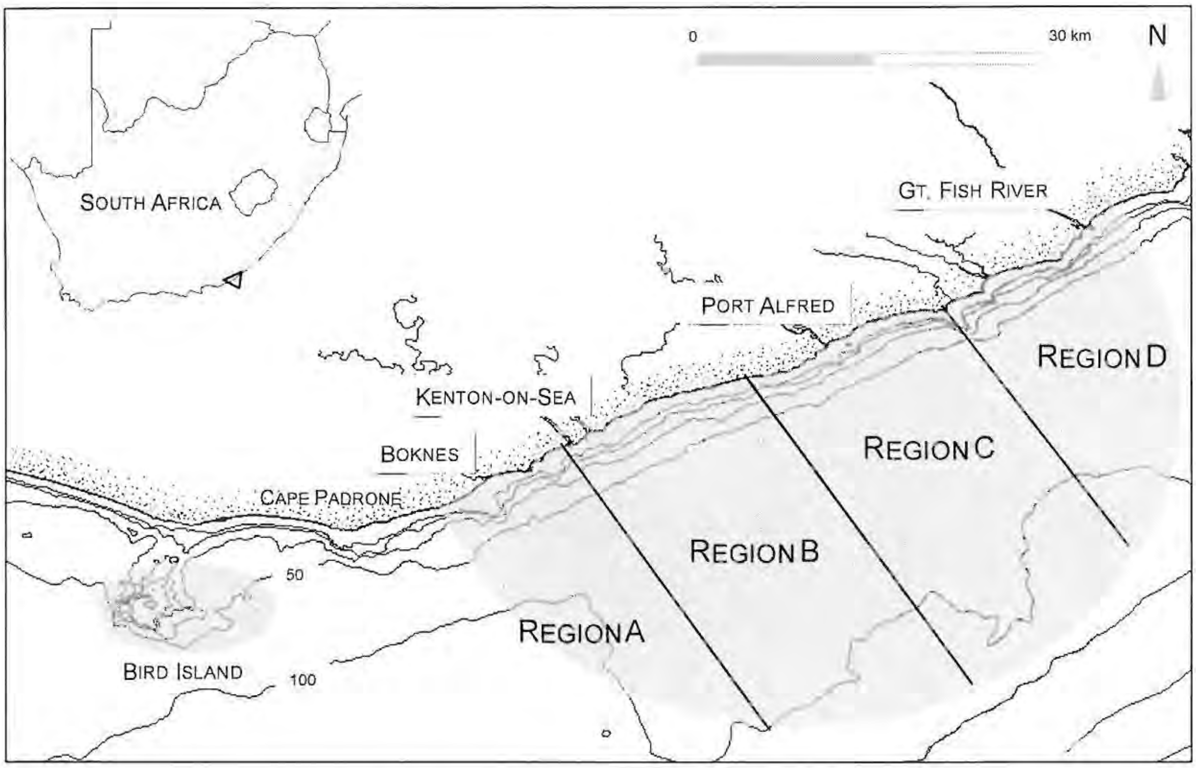


Fig. 2.1 The spatial extent of the Port Alfred/Kenton-on-Sea/Boknes linefishery, and the four regions that were used in the analytical work.

DATASETS AND DATA COLLECTION

This study used four major data sources and two subsidiary sources. These include: (i) catch, effort and fish length data collected from the Port Alfred linefishery by Hecht and Tilney for the period 1980 to 1987, (ii) Catch, effort and length data collected for the period 1990 to 1999 (commissioned by Marine and Coastal Management), (iii) data collected from the fishery by the author for the period 2006 to 2008 and (iv) catch and effort data from the National Marine Linefish System (NMLS) for the period 1985 to 2007. The two subsidiary datasets, viz. commercial invoice data and sea/weather condition data, were used to complement the four primary datasets. The type of data and collection procedures are set out in the following paragraphs.

Historical datasets

Firstly, the original data as used by Hecht and Tilney (1989) were obtained from Prof. T. Hecht of Rhodes University. These consisted of fish length data, fishing effort, catch composition and the original catch returns for the period 1980 to 1987.

Secondly, the data for the period 1990 to 1999 was obtained from MCM (C. Wilke, MCM). These data were collected during a sampling programme of the Port Alfred commercial and recreational fishery between 1990 and 1999. This programme was commissioned by the SFRI (now Marine and Coastal Management) and collected by C. Pittaway under the supervision of Prof. T. Hecht. The data for the 10 years consisted of length-frequency data, catch composition, total catch and effort. These were extracted from the NMLS database. The sampling procedure was identical to that of Hecht and Tilney (1987).

Commercial catch return data for the period January 1985 to December 2007 were extracted from the NMLS. Data was reported in kilograms of each species landed per day from each vessel. Inherent problems with the data system, which are discussed later, include the misidentification of species, over- and under-reporting and the 'lumping' of species such as 'redfish'. However, the NMLS data has been found to be fairly accurate for the Port Alfred/Kenton-on-Sea/Boknes commercial linefishery (Hecht and Tilney 1989, Sauer *et al.* 1997) and continues to form a relatively solid base with which to assess the quantity of fish landed and the species composition of the catches.

Data collection from 2006 to 2008

For the first 6 months of the study, July 2006 to December 2006, the commercial and recreational sectors of the fishery were sampled on a pre-determined, randomized, bi-weekly sampling schedule.

However, this system was found to be unsuitable because it did not provide adequate coverage of recreational and commercial sea trips. For the remainder of the sampling period (January 2007 to September 2008) the sampling procedure was changed. Recreational vessels were sampled whenever the vessels went to sea. Sampling of the recreational catch took place mainly at the Port Alfred River and Ski-boat Club (PARSC) fish cleaning facility, the small boat harbour and the Halyards slipway (Fig. 2.2). With the exception of a few local vessels and 'out-of-town' holiday vessels utilizing the Halyards slipway, the majority of the recreational vessels were moored in the small boat harbour. Fish cleaning services offered by former commercial crew at the PARSC jetty (and a beer in the pub) made it a popular stopover before mooring in the small boat harbour, and hence provided a good place for sampling recreational catches. A fair number of recreational vessels were moored in private boat-houses/jetties on the Port Alfred Marina and access to these vessels for sampling purposes was problematic. However, apart from one vessel, most of these vessels were inactive and their contribution to the fishery was assumed to be negligible.

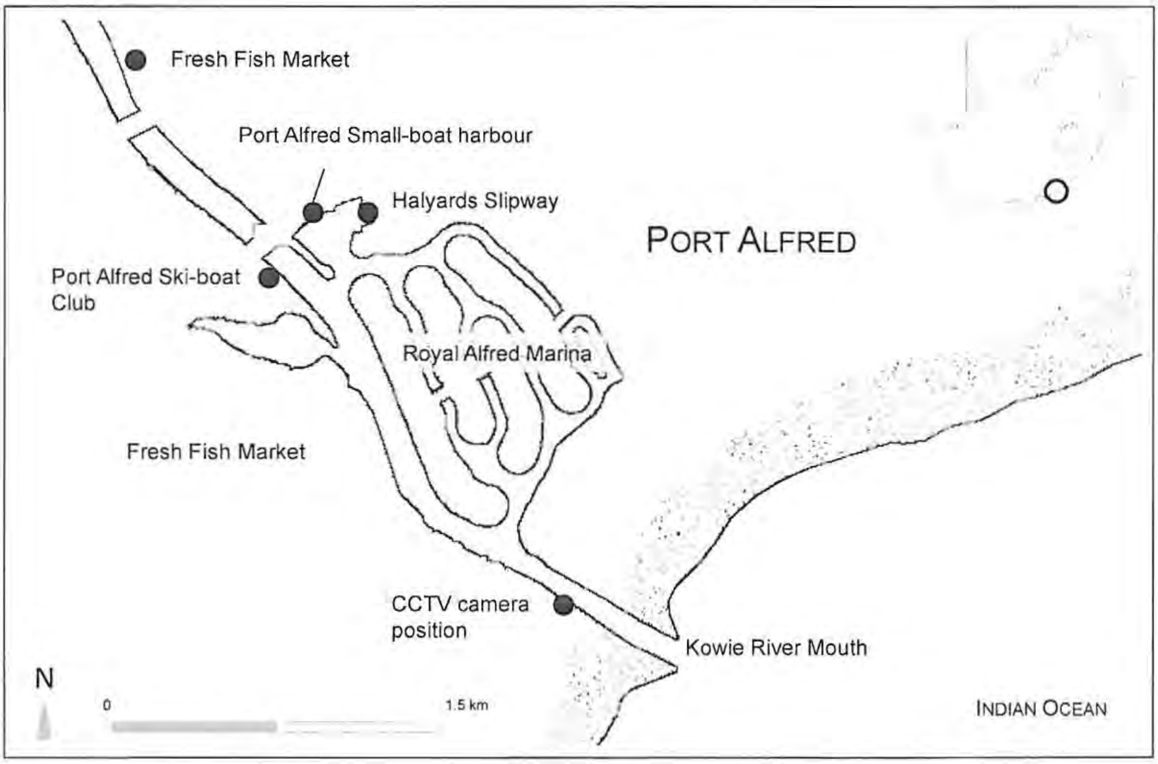


Fig. 2.2 Sites used for sampling the commercial and recreational linefishery within Port Alfred.

The following data were collected from each vessel: (i) Fish lengths (mmFL or mmTL, depending on the species) (ii) duration of fishing trip (iii) Number of crew on board (iv) and where possible, fish weight in grams. Under circumstances where catches were particularly large or when there were time constraints due to vessels returning at more or less the same time then only a proportion of the catch was sampled. Depending on the situation the sub-sampled proportions were recorded and represented as 50, 20, or 10 percent of the vessels' total catch. All data were captured by the author using a digital voice recorder.

In addition, randomized weekly trips were made to the launch sites at Boknes, Kenton-on-Sea and Cannon Rocks (Fig. 2.1) and a count of trailers was undertaken as a means of estimating the number of vessels at sea. Length data and catch data were collected whenever possible.

The commercial catch was originally sampled in the same way as the recreational catch. However, over 95% of the commercial catch from Port Alfred/Kenton-on-Sea/Boknes is purchased and processed by one wholesale/retail outlet[3]. Hence, this outlet provided a practical point at which to sample the catch. A system was set up whereby the weight and length of each fish in the catches of individual vessels was recorded whenever the boats went to sea. For a period of one month (June 2008 to July 2008) an East London registered commercial vessel fished extensively out of Port Alfred. The catch of this vessel was sampled at its mooring in the small boat harbour whenever it returned from sea.

From March 2007 until September 2008, a motion-activated CCTV monitoring camera was installed at a strategic site at the mouth of the Kowie River. This camera took 5 second video clips of each vessel moving in and out of the river mouth. These data were used to calculate effort by counting the number of fishers on each vessel and by determining time spent at sea for the commercial as well as recreational boats. The operation of the camera and the calculation of effort from the data will be discussed further in Chapter 4.

Subsidiary datasets

Commercial invoice data

Because more than 95% of the commercial catch in Port Alfred and surrounding area is processed at one outlet, the purchase invoices from that outlet provided backup information on total catch and effort for the entire study period. As with the NMLS data however, the species

[3] The remaining five percent is either kept for personal consumption or purchased by a number of informal, local buyers.

were 'lumped' into certain categories. This applied mainly to the 'smalls' category which included a variety of species. More than 90% of the fish lumped into the 'smalls' category were panga, *Pterogymnus laniarius* (C. McClleland, Fresh Fish Market, pers. comm. 2007).

Sea condition data

Data obtained from Windguru (*www.windguru.cz/int/index.php?sc=53100*) were used as a means to interpret the behaviour of the fishery with respect to prevailing sea conditions. Windguru is a weather forecasting service based on a Global Forecast System (GFS) predominantly for coastal areas. It is operated by the National Centre for Environmental Prediction (NCEP) which is a unit of the National Oceanographic and Atmospheric Administration (NOAA) and the National Weather Service (NWS), U.S.A. Despite using a globally driven model with a relatively low resolution (55km over a three day period), Windguru has been found to be fairly accurate for the Port Alfred region and is currently used for forecasting by most of the fishers in the area, including the commercial operators. Wind speed, wind direction, wave height, swell direction and swell period were drawn from the Windguru database on a daily basis for the entire study period[4].

SUMMARIZED HISTORY OF THE NMLS DATASET AND ITS LIMITATIONS

Catch data from commercial fishermen in the Cape and from recreational fishers in KwaZulu-Natal were collected from as early as the 1970's (Penney 1997). These two sources were largely independent initiatives undertaken by the former Sea Fisheries Directorate (Marine and Coastal Management) and the Oceanographic Research Institute (ORI). It was not until 1982 that these two data sets were combined forming the NMLS (Penny 1993, Sauer *et al.* 1997). With the introduction of A- and B- commercial licences in 1985, the submission of monthly catch returns were made compulsory for all commercial operators. Since then the NMLS has served as a central database for use in linefisheries research and management (Sauer *et al.* 1997).

The flexibility of the database allows for a diverse range of linefishery data to be analysed from a wide variety of sources (Penney 1993). Consequently, the NMLS has supported numerous

[4] Wave height (m) is the average height of the one-third highest swell; wind speed (km/h) at 10m above the surface; swell period of the dominant wave system in seconds.

scientific publications for the different linefisheries. These include explanations of spatial and temporal dimensions of catch composition, catch rate and seasonal investigations of effort (van der Elst and Penney 1995, Sauer *et al.* 1997, Pradevand and Govender 1999, Yemane 2005, Everett and Fennesy 2007).

Since the inception of the NMLS it was understood that the data captured by the system was never going to be completely accurate. It is accepted that a proportion of the data submitted by the fishers would be incomplete, exaggerated, under-reported or falsified (Penny 1997). The NMLS was primarily designed to provide indices of catch rates and effort in order to aid in the monitoring of major trends in the fishery and not intended for use where absolute accuracy was required (Penney 1997, Sauer *et al.* 1997).

Sauer *et al.* (1997) also highlighted some of the problems associated with the NMLS and these included the non-reporting of catches, non-reporting of certain species, and under- or over-reporting of catches. In addition to this, lumping of 'redfish' species makes species-specific analyses difficult. The most inaccurate data were reportedly those submitted by B-licence holders. The catch return data from the full-time (A-licence) fishers was suggested to be largely accurate apart from the under-reporting of 'fry' retained by the crew (Penney 1997, Sauer *et al.* 1997). Sauer *et al.* (1997) further suggested that catch returns submitted to the NMLS from the Port Alfred area fairly accurately reflected actual landings.

The comparability, integrity, and the temporal distribution of the data are the strongest points of the NMLS (Pradevand and Govender 1999). On the assumption that inaccuracies of the data are consistent on a temporal scale, the NMLS could, as suggested by Penney (1997), correctly reflect major trends in the fishery. For the purposes of this study it is assumed that the NMLS dataset represents a complete and accurate dataset for the fishery, even though it is known that the NMLS is predisposed to certain errors (Sauer *et al.* 1997). These errors were assumed to be random in nature and would therefore have little effect on the observed trends in the fishery.

THE USE OF GENERALIZED LINEAR MODELS FOR STANDARDIZING CATCH RATES FROM LONG TERM DATASETS

Catch rates, when used as indices of abundance, rely on the assumption that catch is proportional to the product of fishing effort and density of fish (Hilborn and Walters 1992), such that:

$$C = qEN \tag{2.1}$$

where C is the catch, E is the effort, N is the density and q, the 'catchability coefficient'.

In other words, C is the proportion of the density (N) that is caught with one unit of effort. When calculating catch per unit effort (C/E) it is implicitly assumed that the catchability coefficient (q) is constant and independent of time and space (Campbell 2004, Maunder and Punt 2004). However, it is known that in most situations this is not true, and q will vary according to many factors (such as the area fished, time of year fished, gear used, vessel type and even the skill of the skipper). In the past, the method used to account for q was to determine the relative fishing power of a vessel compared to a standard vessel. Although simple to apply, this method does not easily deal with multiple factors (for example the month or area that was fished). More importantly, there is no straightforward way to determine the precision of these standardised catch rates (often due to the non-linearity of fisheries data) (Maunder and Punt 2004).

Currently, generalized linear models (GLM's) are commonly used to standardise catch rates (Maunder and Punt 2004). GLM's are the statistical distributions of a particular response variable (in this case catch rate), and the linear relationships between the expected value of the response variable to a set of explanatory variables (such as the area or month that was fished) (Campbell 2004, Maunder and Punt 2004). The factors used in constructing a general catchability coefficient (q) for the whole fishery can thus be included on a per case basis. The catch rates in this study were standardized using these methods.

CHAPTER 3

AN ANALYSIS OF TRENDS IN COMMERCIAL FISHING EFFORT, *CPUE* AND CATCH COMPOSITION IN THE PORT ALFRED LINEFISHERY FROM 1985 TO 2007

INTRODUCTION

Access to long term historical data sources has allowed for the retrospective assessment of the Port Alfred linefishery as a means with which to assess how the fishery has behaved in response to management changes. The data used for this assessment was extracted from the National Marine Linefish System (NMLS) and covered the period January 1985 to December 2007. Analysis of the data (with recognition of the limitations of the dataset; refer to Chapter 2) provided an indication of the shifts that took place in the commercial linefishery with respect to effort, catch rates and catch compositions.

There is often very little data on the state of fish stocks prior to any exploitation (Baum and Myers 2004). Without this data, management measures have to be based on 'non-pristine' stock levels and this has the potential to be misleading (Myers and Worm 2003). This has been dubbed the 'missing baseline' problem. With continued exploitation the stock is depleted further and as a result management decisions are based on pristine levels that are constantly decreasing, this is referred to as the 'shifting baseline syndrome' (Pauly 1995, Baum and Myers 2004). Worldwide, historical datasets have been used to provide important information regarding the state of many fisheries (*inter alia* Holtzhausen 2001, Jackson *et al.* 2001, MacKenzie *et al.* 2002, Mendoza and Larez 2004, Lajus *et al.* 2005). In a South African context, Griffiths (2000) emphasized the value of historical fisheries data and the importance of such data in assessing the current status of a fishery. He also drew reference to the shifting base-line syndrome and noted that in the absence of long-term data fisheries managers, scientists and user groups have lost sight of the state of resources prior to their involvement. Using historical datasets from the Cape commercial linefishery for verification, it was found that virtually all commercially exploited species were overexploited (Griffiths 2000). Similarly, Yemane *et al.* (2004) found that over a hundred year period the *cpue* of linefisheries in the Cape Province had declined dramatically from 94 tonnes.vessel^{-1}.year^{-1} in the 1890's to 12 tonnes.vessel^{-1}.year^{-1} in the 1990's. These declines in *cpue* were the result of heavy commercial exploitation. Moreover, retrospective assessments of fisheries have also been found to be useful tools with which to assess the efficacy of changes to management strategies (Hecht and Tilney 1989, Attwood and Farquhar 1999, Penney *et al.* 1999, Chale-Matsua et al. 2001, Grandcourt *et al.* 2008). For example, Attwood and Farquhar (1999) found that the collapse of the linefishery between Hangklip and Walker Bay (on the Cape south coast) could be attributed to poor control and compliance. Similarly, Penney *et al.* (1999) suggested that the notable decline of catches in KwaZulu-Natal was a consequence of management efforts failing to reduce the fishing effort in the area. However, since these studies were done there have indeed been substantive reductions in commercial effort (as much as 80% countrywide since 1998). This chapter investigates the effect of these management changes on the Port Alfred linefishery using the historical commercial catch return data (NMLS).

The limitations of the long-term NMLS database were outlined in the previous chapter. By no means does this study attempt to validate the NMLS data. This study simply uses the historical data from the NMLS to retrospectively assess the changes in effort, catch compositions, GLM standardised *cpue* and species-directed *cpue* for the Port Alfred commercial boat-based linefishery between 1985 and 2007. The evident trends are juxtaposed with the changes that have taken place in the management environment in an attempt to assess how these metrics have responded.

MATERIALS AND METHODS

The 23 year period from 1985 to 2007 was divided into four separate management periods (Information Box 3.1). The criterion for determining the management periods was the *legislated* reduction in commercial fishing effort at several junctures. Each of the periods represents a change in the commercial licensing structure and, hypothetically, a significant decrease in commercial effort and structural change within the fishery. For the most part, amendments to the minimum legal sizes and bag limits coincided with the aforementioned changes to the licensing structure (Chapter 1, Table 1.3). Where applicable, the bag limits were considered when assessing the species-directed *cpue*.

Information Box 3.1 Four management periods between 1985 and 2007 selected to represent periods of significant change within the Port Alfred linefishery.

Management period A: 1985 – 1998

In 1985 full-time and part-time commercial licences were allocated (A- and B-licences, respectively). B-class licence holders largely acted as recreational fishers (who subsidized there fishing to some degree by selling their catch). Because B-class licence holders generated income from other sources and weren't solely dependent on the fishery they were governed by different economic pressures to the A-class licence holders. This resulted in different fishing patterns with respect to the total effort expended, catch rates and species targeted (Hecht, 1993). In 1992, there was a revision of the minimum size limits and bag-limits for the recreational and both B- and A-licensed commercial fishers (Chapter 1, Table 1.3). Notably, an increase in the size limits of geelbek (400mm to 600mm TL), roman, dageraad and santer (all 250mm to 300mm TL). Both the commercial A- and B-licence holders were limited to two poenskop, red steenbras and seventy four (up until 1998, where after a ban was introduced for the species) per day.

Management period B: 1999 – 2001

In 1999 the A- and B- commercial licences were abolished in favour of the allocation of annual fishing rights (Chapter 1, Table 1.2). The number of commercial licences was reduced and annual licenses were, in general, re-issued to previous A-licence holders. In 2001 there was a one year moratorium on the issuing of annual licences. The commercial licences that were issued in 2000 were re-issued in 2001.

Management period C: 2002 – 2005

In 2002, the numbers of licences were reduced further with the allocation of medium term fishing rights. At the same time, minimum legal size limits and bag limits for certain species were revised (Chapter 1, Table 1.3). These fishing restrictions however applied largely to the recreational fishers.

Management period D: 2006 -2007

Together with the allocation of medium and long-term fishing rights in 2006, there were significant changes to the size and bag limits for the recreational and commercial operators. Most notable, were the increases in the minimum legal size of kob from 400mmTL to 500mmTL, red steenbras from 400mm to 600mmTL, carpenter from 250mm to 350mmTL and dageraad from 300mm to 400mm. The bag limit of geelbek was reduced to two per recreational fisher.day^{-1}, that of carpenter was reduced to four per recreational fisher.day^{-1} (refer to Chapter 1, Table 1.3).

Fishing effort

To assess long-term commercial effort trends the mean number of vessel-days[v] per month were estimated from the catch returns submitted to the NMLS database for the years between 1985 and 2007. Non-parametric bootstraps (1000 replicates) were run for each year to generate 95% confidence intervals (Efron 1982, Efron and Tibshirani 1986).

Although catch returns did not record fishing trips with zero catch, it was assumed that due to the nature of the multi-species fishery, this occurrence was infrequent. Based on this assumption, and the fact that error was consistent over the 23 year period, it was concluded that the NMLS provided a satisfactory dataset with which to crudely assess the trends in effort between 1985 and 2007.

It was hypothesized that effort in each management period would be significantly lower than the previous one due to the re-structuring of the commercial licences and, as such, each would represent a distinct management time-period in the fishery. The reason for calculating commercial fishing effort was two-fold. Firstly, as a means of quantifying the decrease in commercial effort and secondly, to determine whether the selected management periods did in fact represent discrete phases in the history of the Port Alfred fishery. The differences in fishing effort between each period were tested using a one-way Anova followed by Tukey's post hoc test.

Catch Composition

The monthly commercial catch returns submitted to the NMLS only record those species that were caught each day and the cumulative weight for each species or species group. Therefore, eviscerated mass per species (or species group) was used to construct catch compositions on a yearly basis between 1985 and 2007. The composition of the catches was calculated in three ways. Firstly, as a composition comprising the four most important species (kob, geelbek, carpenter, panga), a 'sparid' group (excluding carpenter, panga), a 'non-sparid' group (which excluded kob and geelbek), and an elasmobranch group. Secondly, two ancillary compositions were constructed for the sparid and non-sparid groups. These compositions were based on the

[v] A vessel-day is the number of boats active on a particular day.

proportional breakdown of the two groups, i.e. the mass of each species as a proportion of the landings of all the species in that particular group. The temporal changes in the species dominance structure of the commercial landings between 1985 and 2007 were assessed with other community-based metrics in Chapter 5.

Overall and species-directed cpue

As outlined by Pollock *et al.* (1994), the calculation of catch rate took the form:

$$catch\ rate = \frac{\sum_{i=1}^{n}\left(\frac{C_i}{E_i}\right)}{n} \quad (3.1)$$

where C_i is the catch (kilograms) on day *i*, and E_i is the effort (hours) on day *i*.

As such, the historical catch rate data were extracted from the NMLS database in terms of kilograms.fisher^{-1}.hour^{-1} on a daily basis for the period 1985 to 2007. Generalized linear modelling[vi] was used to standardise the catch rate with respect to the set of covariates 'year', 'month', 'area fished' and 'distance fished from shore' to get the standardised *cpue*. Although the NMLS provides a high resolution for the 'area fished', commercial vessels seldom operated in a single location for the duration of a trip. For this reason, it was sufficient to divide the fishery into four spatial units representing the extreme east, east, west and extreme west of the fishery (Chapter 2, Fig. 2.1). This reduced the total degrees of freedom in the model, as well as eliminating the high variability of catch rates in seldom fished areas.

Prior to modelling the *cpue* an Akaike information criterion (AIC; Akaike 1973) was used to establish the optimum combination of covariates that contributed to the best fit of the generalized linear model (GLM). It took the form:

$$AIC = -2\ln\left[L\left(\theta_{p|y}\right)\right] + 2p \quad (3.2)$$

where, $L\left(\theta_{p|y}\right)$ is the likelihood of model parameters given the data *y*, and *p* is the number of free parameters (Johnson and Omland 2004).

[vi] All statistical modelling and analyses were done with R©, Statsoft Statistica 8© and Systat Sigma-plot 11©

Neither the catch rate data nor the log-transformed catch rate data were normally distributed. Thus, the response of the independent variable (catch rate) to the set of covariates representing the best fit were modelled with ordinary regression using a gamma distribution and a logarithmic link function in the form of:

$$g(\pi) = \int \beta_0 + \beta_1(x_i) + \varepsilon \qquad (3.3)$$

where g is the link function, $\beta_1(x_i)$ is the coefficient and estimation of the i^{th} predictor variable and ε is the error of the model. The logarithmic link function is simply described by:

$$g(\pi) = log\ (\pi) \qquad (3.4)$$

Species-directed catch rates for silver kob, carpenter, geelbek, panga, Roman and dageraad were also calculated in terms of $kilograms.fisher^{-1}.hour^{-1}$. The data were not normally distributed and there was an abundance of zero values as each species was not caught on every fishing outing. Therefore, GLM's were used to standardise the zero skewed data and to model the covariates against the observed catch rates for each species.

The nature of a multi-species, multi-user linefishery makes it particularly susceptible to many instances of zero catches for a particular species (Punt *et al.* 2000). Not only do zero catches create computational difficulties, more importantly there could be sampling biases and this could result in inaccurate catch rate estimations (Maunder and Punt 2004, Cunningham and Lyndenmayer 2005, Fletcher *et al.* 2005). The easiest methods to deal with zero catches are to either ignore them or assign a constant value to all the catch values (Maunder and Punt 2005). However, these methods (widely used in the past) are prone to problems such as positively biasing the standardised catch rates to different extents over a temporal scale or, in the case of introducing a constant, the standardization of catch rates could be sensitive to the value of the constant (Maunder and Punt 2004). Currently, the most utilized method of dealing with zero catches is the Delta-X approach (Stefanssen 1996, Punt *et al.* 2000, Ortiz and Arocha 2004), whereby the probability of a 'non-zero' catch rate and the positive catch rates ($cpue_{pos}$) are modelled separately by means of GLMs. These two models are then combined to give a standardised catch rate (*cpue*) that has been amended by the probability of capture (PC) for a particular species. A graphical example using silver kob in the Port Alfred fishery between 1985 and 2007 is presented in Figure 3.1.

In accordance with the Delta-Gamma model of catch rate standardization, zero and non-zero catch rates in the fishery were modelled separately for each species. This allowed for the calculation of the probability of capture (PC) coupled to a suite of predictor variables ('year', 'month', 'area fished' and 'distance fished'). The positive catch rate ($cpue_{pos}$) was modelled with respect to the same predictor variables. AIC's were used to establish the optimum combination of parameters contributing to the fit of each model (as presented in equation 3.2).

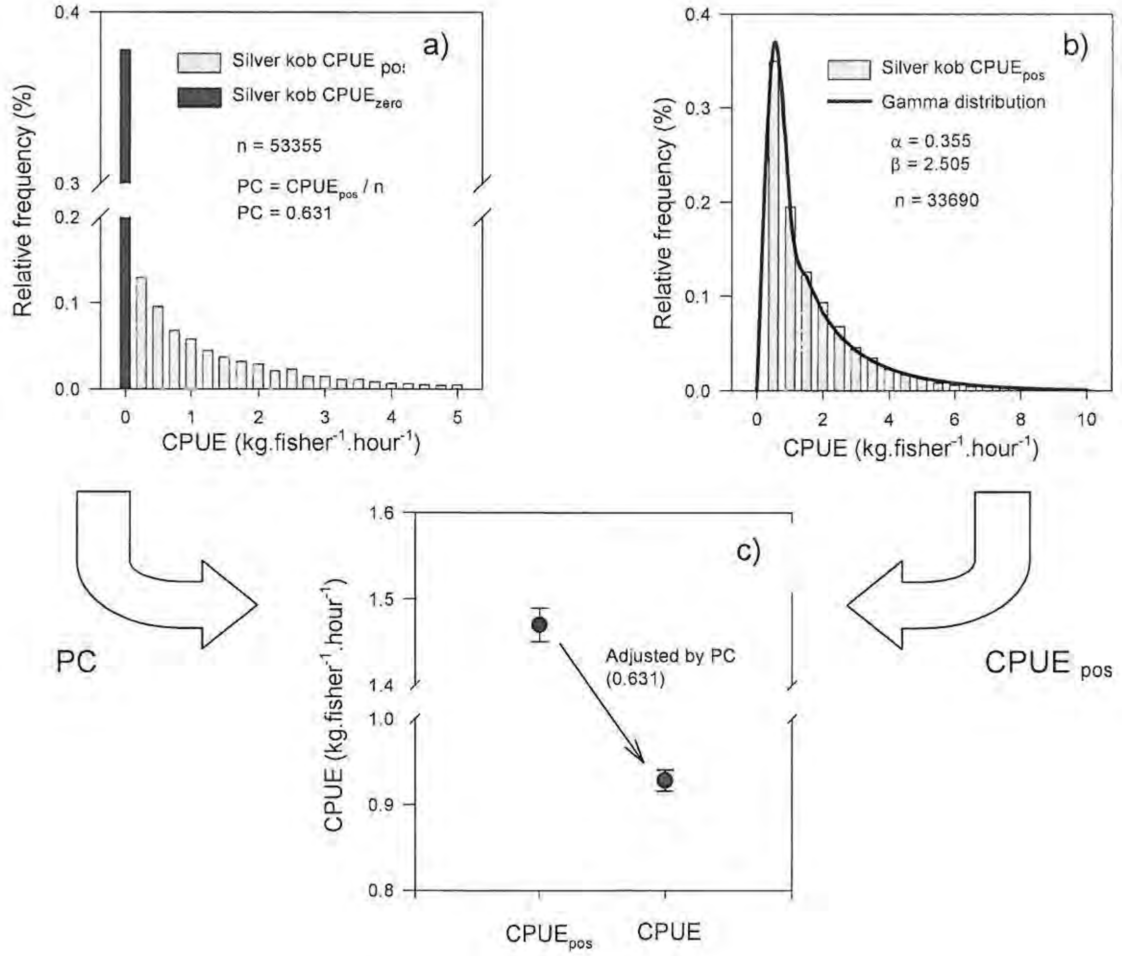


Fig. 3.1 An example of the Delta-X approach of standardising *cpue* using the catch rates of silver kob in the Port Alfred linefishery between 1985 and 2007. a) catch rates including zero catches used to calculate the probability of capture (PC), b) Positive catch rates only and, c) the standardised *cpue* amended by the PC.

Ordinary regression was used to model the positive catch data in terms of the explanatory variables (using a gamma distribution with a logarithmic link) and took the same form as equation 3.3. The presence/absence data was binomially distributed and a logit link function was used to model the data together with a binomial distribution (using logistic regression). The logit link took the form of:

$$g(\mu) = \log\left(\frac{p}{1-p}\right) \tag{3.5}$$

where, p is the continuous probability of the binary dependent variable, with a value between 0 and 1. Thus, modelling the presence or absence of a species in the catch took the form of:

$$\mu = \log\left(\frac{p(\text{present})}{1-p(\text{present})}\right) = \int \theta_0 + \theta_1(w_i) + \varepsilon \tag{3.6}$$

where $p(\text{present})$ is the probability of the species being present in the catch for the set of i covariates denoted by $\theta_1(w_i)$, and ε is the error of the model.

In order to standardise the catch rate for each species by incorporating the probability of capture (PC) the two models were combined as set out by Steffansson (1996):

$$cpue = \hat{\pi}.\hat{\mu} \tag{3.7}$$

where, $\hat{\pi}$ is the value for a particular data point predicted by the model for the positive catch such that,

$$\hat{\pi} = x'\hat{\beta} + \hat{\sigma}^2/2 \tag{3.8}$$

and $\hat{\mu}$ is the probability of a positive catch given the set of covariates for a particular data point such that,

$$\hat{\mu} = (w'.\hat{\theta})/\{1 + (w'\hat{\theta})\} \tag{3.9}$$

where $\hat{\sigma}^2$ is the residual mean square of the model (added to account for the year effect) and, $x'\hat{\beta}$ and $w'.\hat{\theta}$ are the vector coefficients and estimates of the explanatory variables for the positive catch and probability of capture, respectively.

Each datum point for the overall and species-directed catch rates was standardised (and amended by the PC in the case of the species-directed *cpue*). A Kruskal-Wallis One-way Anova on Ranks (K-W Anova) was then used to compare the *cpue* between each of the four management periods. A Dunn's multiple comparison procedure was used to isolate the difference in groups at a p level of 0.05.

For all the species examined (with the exception of panga) there was a change in the minimum legal size at some point during the 23 year period (Chapter 1, Table 1.3). Although these changes had the potential to affect the *cpue*, it was not possible to implicitly relate the increased size limits to changes in *cpue*. The effect of the size limits were thus ignored in this chapter but were dealt with in the length-based analysis in Chapter 5. On the other hand, there were changes to bag limits for Roman (1992) and dageraad (1992, 2006) that were applicable to the commercial operators (Chapter 1, Table 1.3). The implementation of bag limits would have a direct effect on the *cpue*. As such the species-directed *cpue* before and after these changes were compared with a K-W Anova followed by Dunn's multiple comparison procedure.

RESULTS

Changes in commercial fishing effort between 1985 and 2007

Between 1985 and 1991 effort increased from 181.1(*95% CI*: 162.3 – 199.3) to 378 (*95% CI*: 340.1 – 413.6) vessel-days.month^{-1} (Fig. 3.2). Since then effort has declined, and in particular a highly notable decrease from 176.28 (*95% CI*: 140.3 – 207.0) to 42.85 (*95% CI*: 35.3 – 50.1) vessel-days.month^{-1} in 2000 and 2002, respectively. The lowest fishing effort in the Port Alfred linefishery was recorded in 2007 with a mean of 34.21 (*95% CI*: 29.7 – 38.8) vesseldays.month^{-1}.

There was a significant difference in commercial fishing effort between the four management periods (*Anova*: $f = 35.64$, $p < 0.001$). As hypothesized, the four management periods represented distinct phases of significant change within the fishery (Fig. 3.2). Thus, these periods can be used as a framework for quantifying the effects of the management changes in the commercial linefishery.

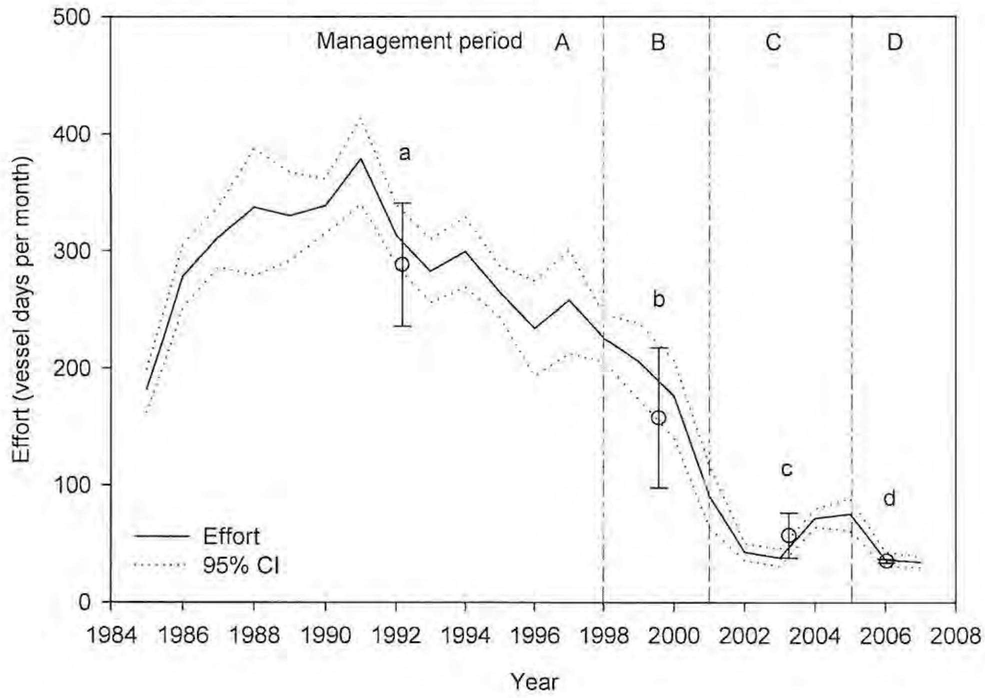


Fig. 3.2 Changes in commercial effort in the Port Alfred linefishery between 1985 and 2007. Box-and-whisper plots represent the mean effort during management periods A(1985-1998), B(1999-2001), C(2002-2005) and D(2006-2007). Lowercase letters denote significant differences at a *p* level of 0.05.

Shifts in catch composition between 1985 and 2007

The catch composition of the commercial linefishery for each year between 1985 and 2007 were calculated from the catch returns submitted to the NMLS database. They are presented separately for each of the dominant species (viz. silver kob, geelbek, panga and carpenter) and visually separated as per the aforementioned management periods (Fig. 3.3). Species that were reported as either 'redfish' or simply 'fish' were excluded from the analysis (over the 23 year period these 'lumped' species contributed a mean 2.59 ±0.9% of the total commercial landings).

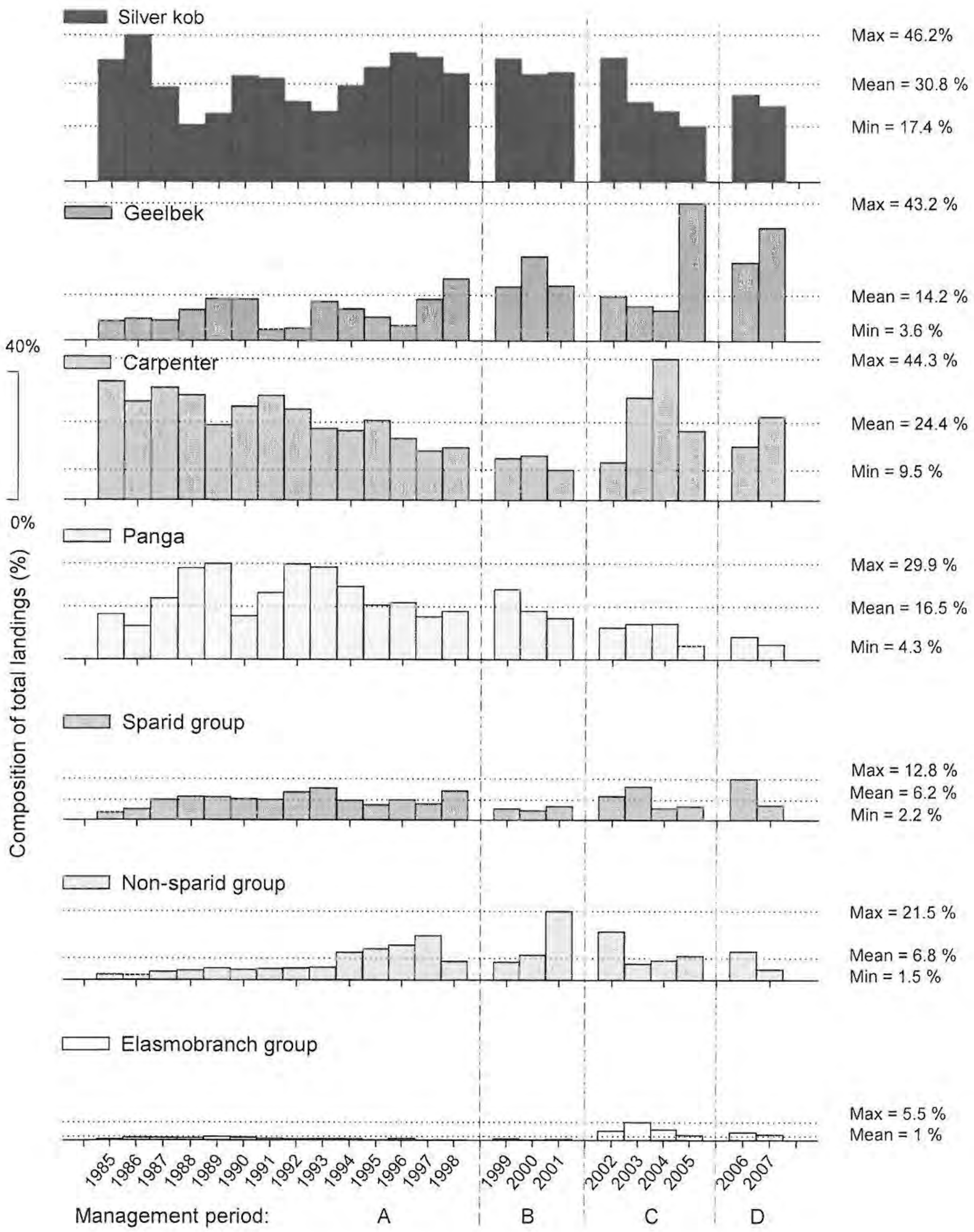


Fig. 3.3 The catch composition of the Port Alfred commercial linefishery between 1985 and 2007.

The consistently high contribution by silver kob (*Argyrosomus inodorus*) to the total annual landings, throughout the 23 year period, underscores the importance of the species in the fishery. On average silver kob contributed 30.8 ±7.9 % to the total commercial landings per year (maximum of 46.2% in 1986 and a minimum of 17.4% in 2005). There was no statistically significant difference in the proportion of silver kob caught between any of the four management periods (Anova: $df = 3, f = 1.34, p = 0.292$).

There was an increasing proportion of geelbek (*Atractoscion aequidens*) landed during the last two management periods. Between 2005 and 2007 geelbek contributed a mean of 35.06 ±8.4% per year as opposed to a mean of only 11.23 ±5.8% for the previous 20 years. There was a significant difference in the proportion of geelbek landed between the four management periods (*Anova:* $df = 3, f = 5.83, p = 0.005$). Notably, a significantly higher proportion in period D than period A (*Holm-Sidak:* $t = 3.53, p = 0.002$). The maximum contribution by geelbek to total landings (43.2%) was recorded in 2005.

The proportion of carpenter (*Argyrozona argyrozona*) steadily declined from 37.2% in 1985 to 9.5% in 2001 (management periods A and B). Between 2003 and 2007, the proportion of carpenter landed in the commercial sector increased substantially. In 2004, as much as 44% of the total commercial landings consisted of carpenter. The proportion of the total landings attributed to carpenter during period B was significantly lower than in period A (*Holm-Sidak:* $t = 3.86, p = 0.001$) and period C (*Holm-Sidak:* $t = 3.64, p = 0.002$). There was no significant difference between period D and the other periods.

In the late 1980's and early 1990's panga (*Pterogymnus laniarius*) comprised a substantial proportion of the catch with a maximum of 29.9% in 1989. Since then there has been a notable decline of panga in the commercial landings. Panga contributed a mere 4.3% to the total catch in 2005. There was a significant difference between management periods (*Anova:* $df = 3, f = 5.8, p = 0.006$). The contribution by panga to the total landings in period A was significantly higher than periods C (*Holm-Sidak:* $t = 3.23, p = 0.004$) and D (*Holm-Sidak:* $t = 3.124, p = 0.006$).

Over the 23 year period the elasmobranch group contributed a mean of 1.01 ±1.3% per year of the total landed catch. The largest proportion of elasmobranches was landed in 2003 (5.5%). There was a significant difference between the four management periods (*Anova:* $df = 3$, $f = 17.87$, $p < 0.001$). Commercial landings in period C had a significantly higher proportion of elasmobranches than period A (*Holm-Sidak:* $t = 6.81$, $p < 0.001$) and period B (*Holm-Sidak:* $t = 5.49$, $p < 0.001$). Period D had a greater proportion than period A (*Holm-Sidak:* $t = 2.75$, $p = 0.013$). Given the overall insignificant contribution made by sharks and rays the data were not analysed any further.

The annual contribution of the sparid group to the commercial landings between 1987 and 2007 remained fairly constant ($\bar{x} = 5.9 \pm 2.6\%$) and there was no significant difference between the management periods (*Anova:* $df = 3$, $f = 1.16$, $p = 0.35$). Similarly, there was no significant difference between the contributions of the non-sparid group for the four periods (*Anova:* $df = 3$, $f = 2.75$, $p = 0.071$) (however, the power of the test (0.374) was less than the desired 0.800, thus the results need to be interpreted with caution). In fact, the non-sparid group contributed as little as 1.5% to commercial landings in 1986 and as much as 21% in 2001. However, within the sparid and non-sparid groups there were significant shifts in the relative importance of the species over the 23 year period. The most notable of these changes were the decreasing proportion of dageraad (*Chrysoblephus cristiceps*) and increasing proportion of santer (*Cheimerius nufar*) in the commercial sparid group landings, and the decreasing proportion of shallow-water hake (*Merluccius capensis*) and increasing proportion of elf (*Pomatomus saltatrix*) in the commercial non-sparid group landings.

The compositions that make up the sparid and non-sparid groups for the period 1985 to 2007 are presented in Table 3.1. Dageraad, which made up a consistent proportion of around 28% of the sparid group in the management periods A and B (1985 to 2001) contributed a mere 1.48 ±0.2% in period D. There was an overall significant difference in the contribution of dageraad to the total sparid landings between management periods (*Anova:* $df = 3$, $f = 9.07$, $p < 0.001$). Period D had a significantly lower proportion of dageraad than periods A (*Holm-Sidak:* $t = 4.27$, $p < 0.001$) and B (*Holm-Sidak:* $t = 3.56$, $p = 0.002$). Similarly, Roman (*Chrysoblephus laticeps*) also contributed a lesser proportion to total sparid landings in period D (*Holm-Sidak:* $t = 4.46$, $p < 0.001$).

Table 3.1 The percentage contribution by selected species to the composition of the sparid and the non-sparid group landings in the Port Alfred commercial linefishery between 1985 and 2007.

Year	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007
Management Period							A									B			C			D	
Sparid group																							
Dageraad	17	31	21	30	30	36	29	24	26	21	22	26	39	42	37	30	18	30	11	2	5	1	2
Red Steenbras	53	27	40	30	14	20	27	24	20	21	10	4	3	6	7	8	5	1	9	18	9	5	14
Red Roman	21	16	10	10	22	14	16	11	11	13	21	17	12	15	20	19	22	18	14	15	13	5	14
Santer	0	4	10	12	13	13	13	32	35	25	26	22	25	23	25	36	42	38	49	51	59	15	57
Seventy-four	0	1	2	1	0	3	1	1	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0
Non-sparid group																							
Galeichthyes sp.	24	33	31	12	10	8	7	7	9	2	1	1	0	1	1	3	4	3	1	0	0	0	0
Elf	0	0	0	0	0	1	0	0	1	2	3	2	1	8	16	3	1	1	2	15	36	59	77
Hake	63	41	59	75	86	85	51	68	22	69	83	82	73	60	70	84	73	37	46	52	21	41	14
Yellowtail	1	17	1	1	2	2	29	1	54	7	5	6	17	5	2	2	2	3	7	0	33	0	2

Santer, which made minor contributions to sparid landings throughout management period A, made up a substantial proportion of the total landed sparids in the last two management periods (as much as 59% in 2005). There was a significant difference in the proportion of santer landed between the management periods (*K-W Anova:* $H = 11.92$, $df = 3$, $p = 0.008$). The sparid landings during period A had a significantly lower proportion of santer than the landings between 2002 and 2005 ($Q = 3.18$, $p < 0.05$).

The main contributor to the non-sparid group for the period 1985 to 2001 (management period A and B) was shallow-water hake. There was a significant decrease in the proportion of hake landed between the four management periods (*Anova:* $df = 3$, $f = 6.04$, $p = 0.005$). The contribution of hake during the periods C (38.9 ±13%) and D (27.5 ±19%) were significantly lower than the periods A and B, in which hake had mean contributions of 65.5 ±18% and 75 ±7% to the total non-sparid group landings, respectively. As the proportion of hake decreased, the proportion of elf increased. In management period D around 70% of the non-sparid group landings were attributed to elf.

Another notable contribution to the non-sparid group in the earlier years came from the two *Galeichthyes* species (*G. ater* and *G. feliceps*). Between 1985 and 1998 they contributed a mean of 10.5 ±11% to the total non-sparid landings. This was significantly higher than any of the other periods. In period D the *Galeichthyes* spp. made no contribution to the non-sparid group.

Trend in overall cpue between 1985 and 2007

The AIC model building function suggested that all the covariates (year, month, area fished and distance from the shore) should be included when modelling the overall *cpue* in the Port Alfred commercial fishery between 1985 and 2007. The standardised *cpue* plotted at the mean of the continuous covariate is presented in Fig. 3.4 (*Wald* X^2 $(22) = 3129.20$, $p = 0.000$).

The analysis revealed two distinct trends in *cpue* during the 23 year period. From 1985 to 1998 the *cpue* declined from 3.64 (95% CI: 3.5 - 3.8) to 2.28 (95% CI: 2.7 - 2.9) kg.fisher^{-1}.hour^{-1}. Since 1998 there has been a substantial increase in *cpue*, being particularly high during the years 2004 (4.79 kg.fisher^{-1}.hour^{-1}), 2005 and 2007. There was a significant difference in overall *cpue* between all the management periods (*K-W Anova:* $df = 3$, $H = 5700.3$, $p <= 0.001$) (Fig. 3.4

insert). Periods C and D had a significantly higher *cpue* than the first two management periods. The lowest observed *cpue* was during period B (1998 – 2001).

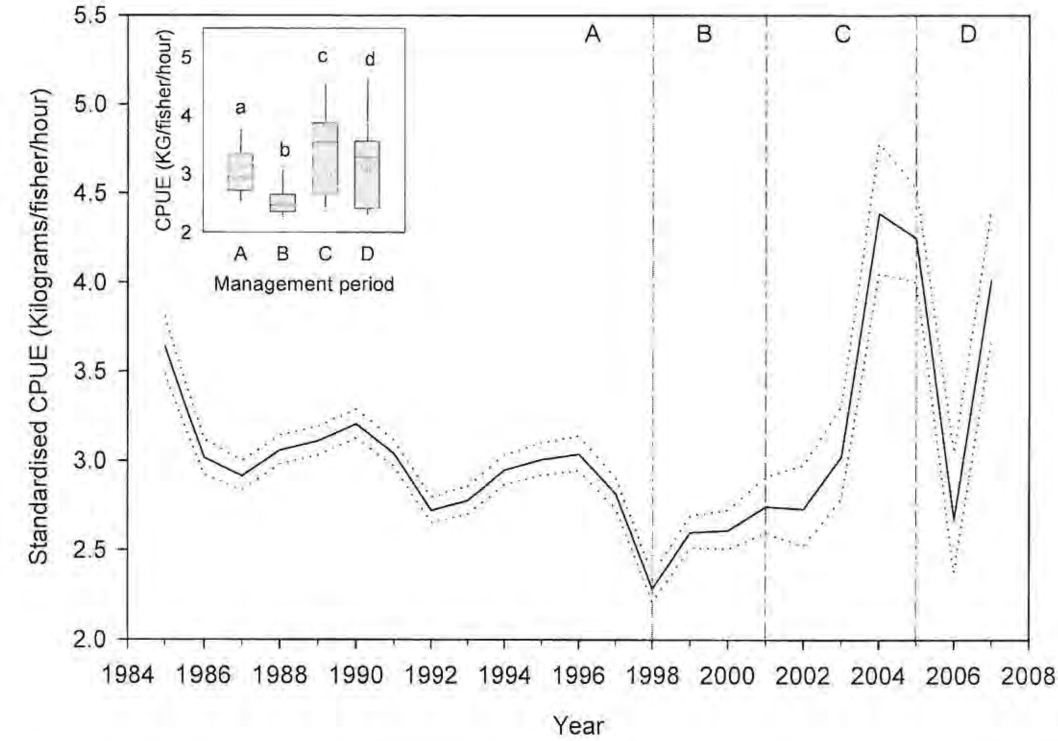


Fig. 3.4 *Cpue* for the Port Alfred commercial linefishery from 1985 to 2007. Upper and lower 95% confidence intervals are presented as broken lines around the mean. Insert: *cpue* during the management periods A(1985-1998), B(1999-2001), C(2002-2005) and D(2006-2007). Superscripts denote significant difference at a *p* level of 0.05.

Trends in the Probability of Capture (PC) and Species-directed cpue between 1985 and 2007

The presence and/or absence of the four dominant species in the commercial catches (viz. silver kob, carpenter, geelbek and panga) and two reef associated species (viz. roman and dageraad) were modelled with respect to the covariates year, month, area fished and distance fished from shore. The model provided the continuous probability of the binary dependent variable as a number between 1 and 0, or in other words, the probability of capture (PC) of a particular species given a specific set of covariates. An AIC test (Equation 3.3) run prior to modelling the PC and *cpue* indicated that all of the covariates should be included to give the model the best fit for each of the species (Table 3.2 and Table 3.3).

Table 3.2 Summary statistics of the GLM for the probability of capture (PC) of seven key species in the Port Alfred commercial linefishery between 1985 and 2007.

Species	Intercept (*df* = 1)		Year (*df* = 22)		Months (*df* = 11)		Location (*df* = 3)	
	Wald X^2	*p*	*Wald* X^2	*p*	*Wald* X^2	*p*	*Wald* X^2	*p*
Kob	853.29	*<0.001*	896.97	*<0.001*	327.4	*<0.001*	16.6	*<0.001*
Geelbek	660.93	*<0.001*	1646.81	*<0.001*	1215.72	*<0.001*	25.98	*<0.001*
Carpenter	206.08	*<0.001*	2030.65	*<0.001*	656.79	*<0.001*	25.07	*<0.001*
Panga	8.7	*0.003*	2031.94	*<0.001*	1247.56	*<0.001*	9.42	*<0.001*
Santer	432.18	*<0.001*	1718.04	*<0.001*	364.16	*<0.001*	25.48	*<0.001*
Roman	746.46	*<0.001*	374.24	*<0.001*	359.42	*<0.001*	17.85	*<0.001*
Dageraad	719.35	*<0.001*	768.32	*<0.001*	296.95	*<0.001*	37.76	*<0.001*

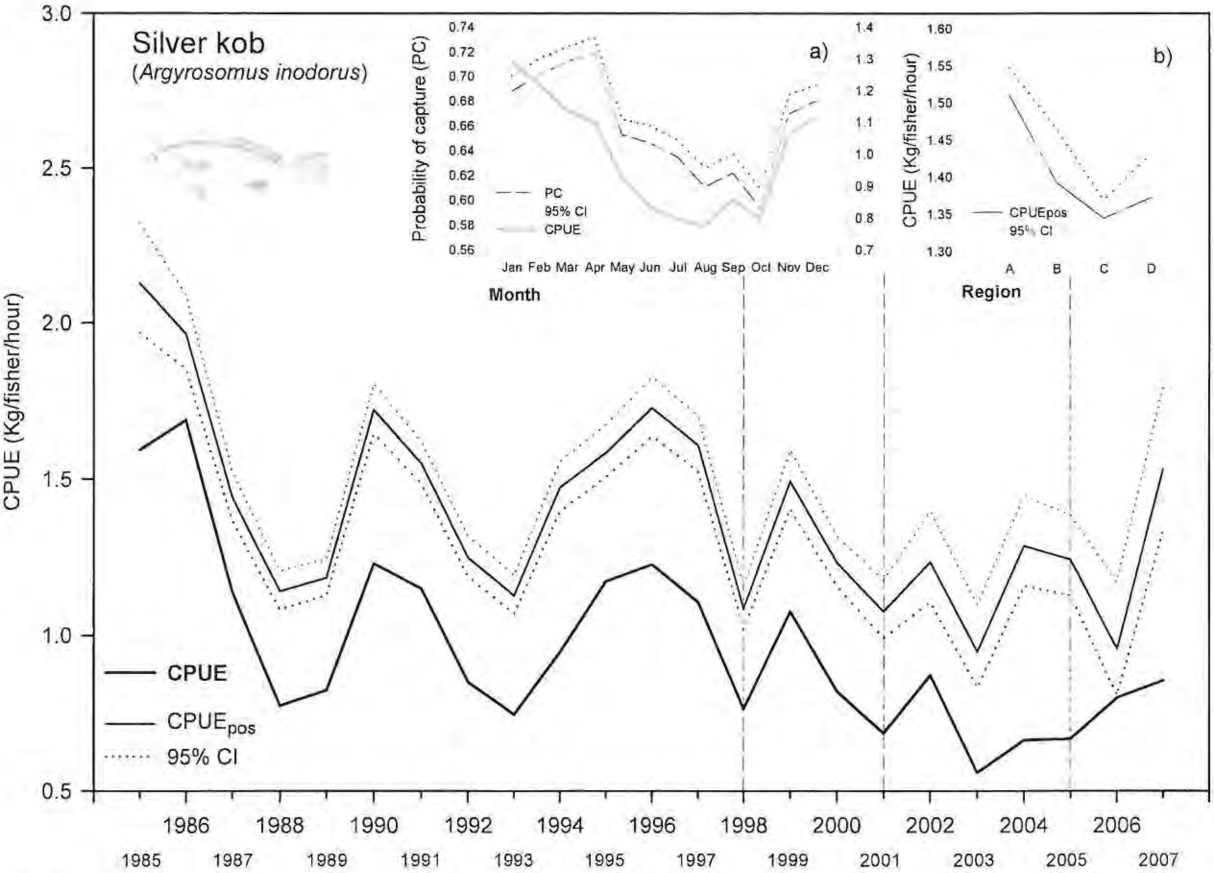
Whereas the model of the PC of a particular species predicts the probability of encountering that particular species, the model of the positive catch rates ($cpue_{pos}$) predicts the rate of capture only if a species is encountered. The standardised *cpue* is calculated by integrating the probability of capture with the positive catch rate (refer to Fig. 3.1). The $cpue_{pos}$ and the *cpue* for the dominant species (silver kob, geelbek, carpenter, panga) and two resident, reef associated species (Roman and dageraad) were plotted for the years from 1985 to 2007 (Figures 3.5 to 3.19). The associated PC of the species for each year are presented in the tables directly below each figure.

Table 3.3 Summary statistics of the GLM for the positive catch rates of seven key species in the Port Alfred commercial linefishery between 1985 and 2007.

Species	Intercept (*df* = 1)		Year (*df* = 22)		Months (*df* = 11)		Location (*df* = 3)	
	Wald X^2	*p*	*Wald* X^2	*p*	*Wald* X^2	*p*	*Wald* X^2	*p*
Kob	24.44	*<0.001*	1239.24	*<0.001*	528.53	*<0.001*	8.9	*<0.001*
Geelbek	93.29	*<0.001*	287.15	*<0.001*	238.4	*<0.001*	6.69	*<0.001*
Carpenter	603.42	*<0.001*	1211.7	*<0.001*	300.56	*<0.001*	17.6	*<0.001*
Panga	843.71	*<0.001*	1667.59	*<0.001*	377.49	*<0.001*	11.58	*<0.001*
Santer	411.47	*<0.001*	191.95	*<0.001*	69.51	*<0.001*	3.68	*<0.001*
Roman	1024.26	*<0.001*	295.57	*<0.001*	73.24	*<0.001*	4.19	*<0.001*
Dageraad	544.43	*<0.001*	410.36	*<0.001*	67.81	*<0.001*	3.64	*<0.001*

Silver kob

There was an overall declining trend in the *cpue* of silver kob for the 23 year period (*x-coefficient* = *-0.028*). The *cpue* decreased from a maximum of 1.69 kg.fisher^{-1}.hour^{-1} in 1986 to a minimum of 0.56 kg.fisher^{-1}.hour^{-1} in 2003 (Fig. 3.5). After 2003 the *cpue* gradually increased to a mean of 0.86 kg.fisher^{-1}.hour^{-1} in 2007, however, this was only marginally higher than the lowest *cpue* recorded prior to 1998.



PC (%)																							
Mean	72	82	74	63	64	68	70	63	61	60	70	67	65	65	68	62	59	66	54	48	50	76	53
Lwr CI	70	81	73	61	63	66	69	62	59	59	68	66	63	63	66	59	55	61	49	45	46	69	48
Upr CI	73	83	76	64	66	69	71	65	63	62	71	69	67	66	70	64	62	70	58	52	53	82	58

Fig. 3.5 *Cpue*$_{pos}$ and the PC amended *cpue* of silver kob (*Argyrosomus inodorus*) in the Port Alfred commercial linefishery between 1985 and 2007; Inserts a) monthly predicted PC (with 95% CI) and *cpue*, and b) the predicted *cpue*$_{pos}$ by region. The PC for each year is presented in the table below the plot (Lwr CI = lower 95% confidence interval, Upr CI = upper 95% confidence interval).

The high PC of silver kob over the 23 year period (a mean PC of 62 ±1%) indicates the importance of the species in the fishery. The lowest PC for kob was recorded in 2004 and 2005, when silver kob were still landed on 48% and 50% of the fishing trips, respectively. In the following year (2006) the PC for silver kob was 76% (the second highest of all the years).

There was a statistically significant difference between the four management periods (*K-W Anova:* $H = 363.3$, $df = 3$, $p =< 0.001$) (Fig. 3.6). However, there was no significant difference in the *cpue* between the management period A (1985 to 1998) and period B (1999 to 2002) ($Q = 0.278$, $p > 0.05$). Between 2003 and 2005 (period C) there was a significant decrease in *cpue* ($p < 0.05$). In period D (2006 to 2007) the *cpue* was significantly higher than that observed between 2003 and 2005 but still significantly lower than the mean *cpue* during the first two management periods ($p < 0.05$).

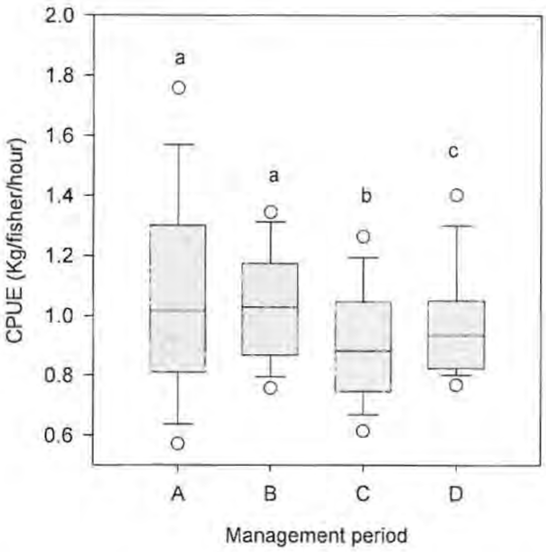
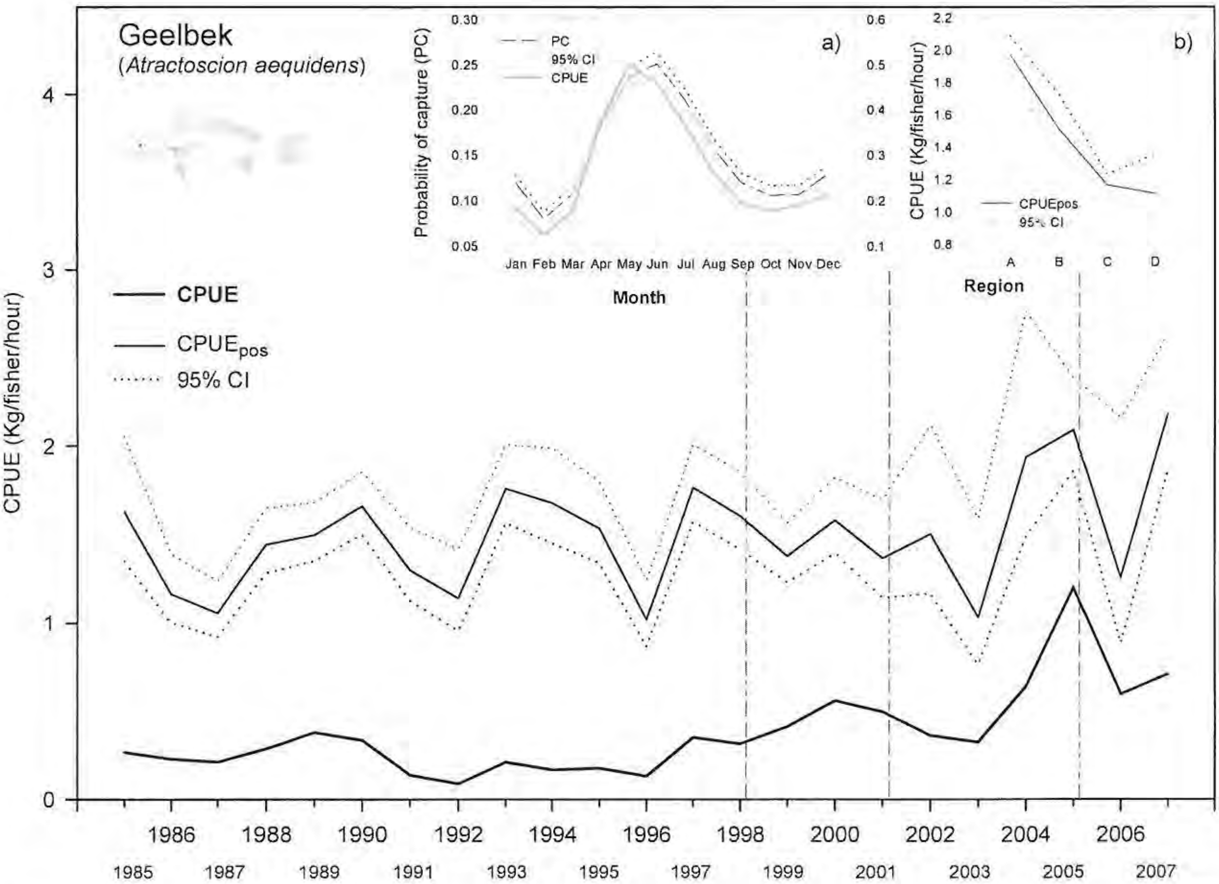


Fig. 3.6 *Cpue* of silver kob (*Argyrosomus inodorus*) during the management periods A(1985 -1998), B(1999 -2001) C(2002 -2005) and D(2006 -2007) in the Port Alfred commercial linefishery. Superscripts denote significant differences tested by Dunns' multiple comparison procedure at a *p*-level of 0.05.

Geelbek

The *cpue* of geelbek decreased from 0.27 kg.fisher^{-1}.hour^{-1} in 1985 to 0.09 kg.fisher^{-1}.hour^{-1} in 1992 (Fig. 3.7). Between 1996 and 2007 *cpue* increased steadily and particularly from 2004 and 2007. The highest *cpue* was recorded in 2005 (1.2 kg.fisher^{-1}.hour^{-1}) and this was more than double that of any previous year.



PC (%)

	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007
Mean	14	16	15	16	21	17	9	6	10	8	10	10	17	16	24	30	30	20	24	29	50	38	28
Lwr CI	12	14	14	15	20	16	8	5	9	8	9	9	16	15	23	28	27	17	20	25	46	31	24
Upr CI	15	17	17	18	22	18	9	7	11	9	11	11	18	18	26	32	33	24	29	32	54	45	33

Fig. 3.7 $Cpue_{pos}$ and the PC amended *cpue* of geelbek (*Atractoscion aequidens*) in the Port Alfred commercial linefishery between 1985 and 2007; Inserts a) monthly predicted PC (with 95% CI) and *cpue*, and b) the predicted $cpue_{pos}$ by region. The PC for each year is presented in the table below the plot (Lwr CI = lower 95% confidence interval, Upr CI = upper 95% confidence interval).

Prior to 2005, geelbek were caught on no more than 30% of fishing trips. However, in 2005 the PC of geelbek peaked at 50%. This was significantly higher than the mean PC for the entire 23 year period (20 ±10%). The lowest PC occurred in 1992 (6%) which corresponded with the low *cpue* as well as the increase of the minimum legal size limit from 400mmTL to 600mmTL during that year.

Both the *cpue* and PC of geelbek were highest during the winter months April, May, June and July (Fig. 3.7 insert a). *Cpue* was highest in the western end of the fishery, region A, and lowest in the most easterly region D (Fig. 3.7 insert b).

There was a significant difference in *cpue* for geelbek between management periods (*K-W Anova:* $H = 948.6$, $df = 3$, $p =< 0.001$). The *cpue* was lowest during period A and highest during period C (Fig. 3.8a). A Dunn's multiple comparison procedure failed to identify any significant difference between period B (1999 – 2001) and period D (2006 -2007) ($Q = 1.84$, $p > 0.05$) although the *cpue* for geelbek in period D fluctuated greatly.

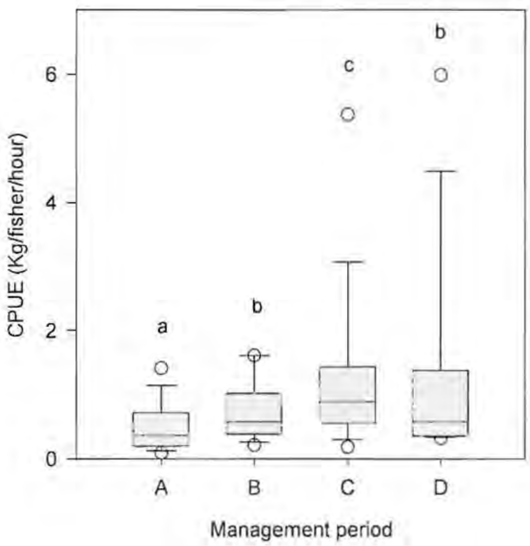
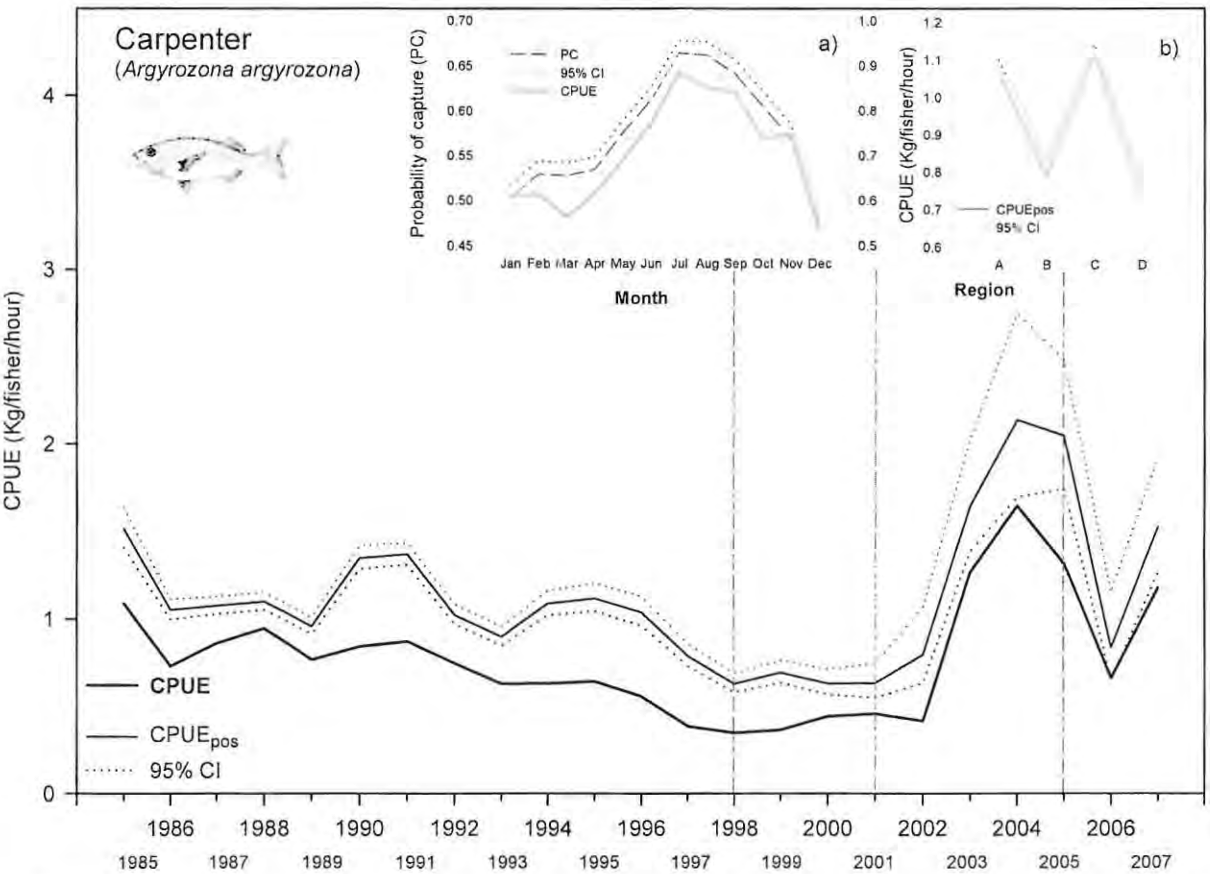


Fig. 3.8 *Cpue* of geelbek (*Atractoscion aequidens*) during the management periods A(1985 -1998), B(1999 -2001), C(2002 -2005), and D(2006 -2007) in the Port Alfred commercial linefishery. Superscripts denote significant differences tested by Dunns' multiple comparison procedure at a *p*-level of 0.05.

Carpenter

Between 1985 and 1998 the *cpue* of carpenter decreased from 1.09 to 0.35 kg.fisher^{-1}.hour^{-1}, respectively (Fig. 3.9). The *cpue* then increased significantly to 1.27, 1.65 and 0.66 kg.fisher^{-1}.hour^{-1} in 2003, 2004 and 2006. Despite the higher *cpue* in the latter years, there is only a marginally increasing trend in *cpue* over the entire 23 year period (*x-coefficient = 0.006*).



PC (%)	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007
Mean	64	59	68	74	67	55	56	62	58	50	49	45	40	43	41	54	56	42	69	80	59	65	69
Lwr CI	62	57	67	72	66	53	55	60	56	48	47	44	38	41	39	52	53	38	65	77	56	58	64
Upr CI	66	61	70	75	69	57	58	63	60	51	51	47	41	45	44	56	59	47	73	82	62	71	73

Fig. 3.9 $Cpue_{pos}$ and the PC amended *cpue* of carpenter (*Argyrozona argyrozona*) in the Port Alfred commercial linefishery between 1985 and 2007; **Inserts a)** monthly predicted PC (with 95% CI) and *cpue*, and **b)** the predicted $cpue_{pos}$ by region. The PC for each year is presented in the table below the plot (Lwr CI = lower 95% confidence interval, Upr CI = upper 95% confidence interval).

The mean PC of carpenter over the 23 year period was 58 ±1%. As with silver kob, this high PC indicates the importance of carpenter in the fishery. The lowest PC was in 1997 (40%), while in 2004 carpenter were landed on more than 80% of commercial fishing trips (significantly more than any of the other years). The PC and *cpue* were highest during the winter months June, July, August and September (Fig. 3.9 insert a). Region B had the lowest *cpue*, while region A was the most productive (Fig. 3.9 insert b).

There was a significant difference in *cpue* with respect to the four management periods (*K-W Anova:* $H = 4089.6$, $df = 3$, $p =< 0.001$) (Fig. 3.10a). The lowest *cpue* occurred during 1999-2002 (period B) and the highest was between 2003 and 2005 (period C).

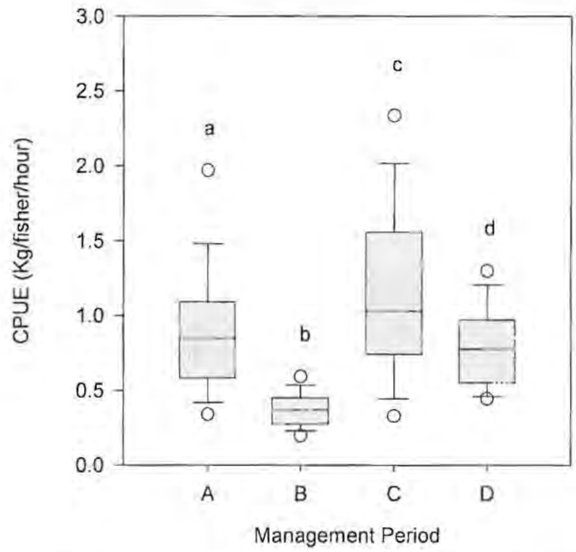
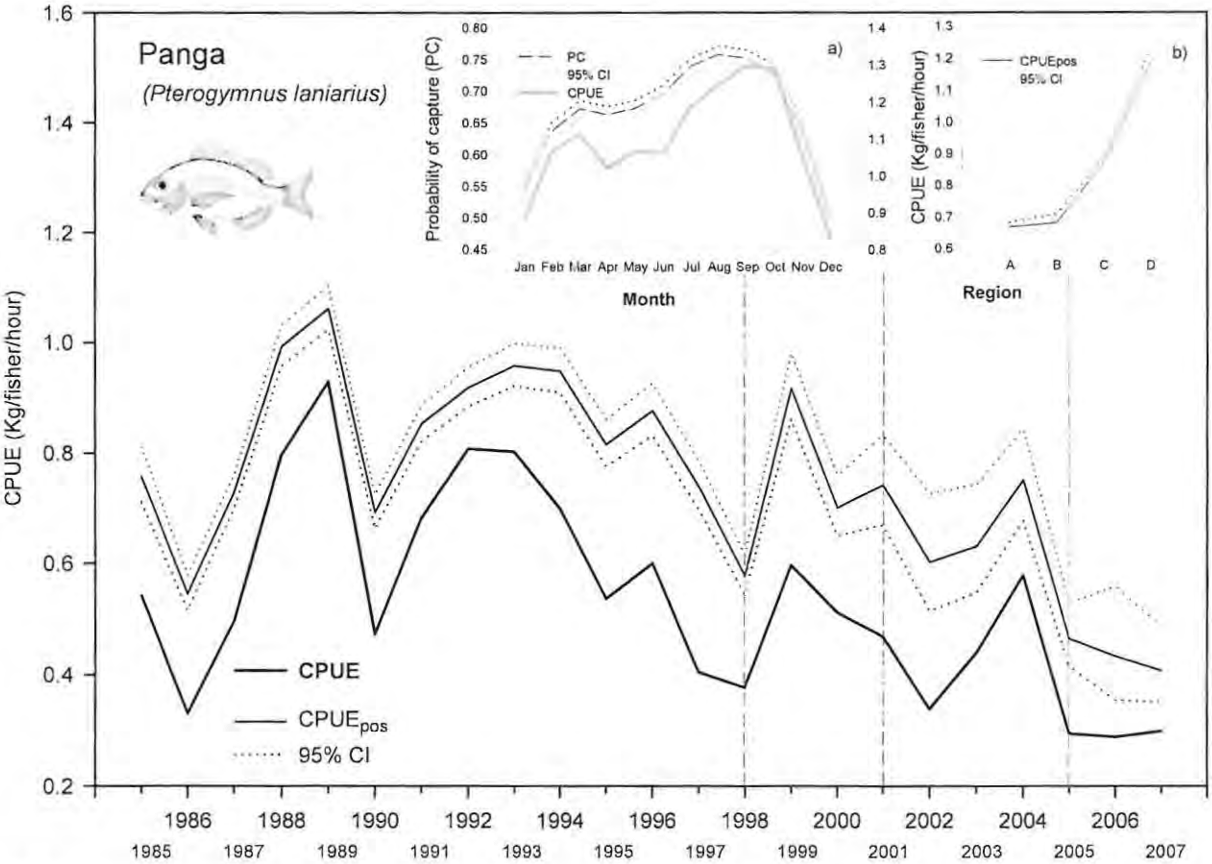


Fig. 3.10 *Cpue* of carpenter (*Argyrozona argyrozona*) during the management periods A(1985 -1998), B(1999 -2001), C(2002 -2005) and D(2006 -2007) in the Port Alfred commercial linefishery. Superscripts denote significant differences tested by Dunns' multiple comparison procedure at a *p*-level of 0.05.

Panga

The highest *cpue* of panga (0.93 $kg.fisher^{-1}.hour^{-1}$) was recorded in 1989 (Fig. 3.11), where after it declined to its lowest level in 2006 (0.29 $kg.fisher^{-1}.hour^{-1}$). Panga were most abundant in the commercial landings during the spring months (October and September) (Fig. 3.11 insert a). In contrast to the other species, the highest *cpue* for panga was recorded in the eastern end of the fishery (region D) (Fig. 3.11 inset b). The mean PC of panga over the entire 23 year period was 63 ±1% with a peak of 81% in 1989 and 1992. The lowest PC of 41% was recorded in 2002.



PC (%)																							
Mean	65	53	61	74	81	61	73	81	77	68	60	63	49	57	60	65	57	49	62	69	54	56	61
Lwr CI	62	51	59	73	80	59	72	79	76	66	58	61	47	55	58	63	54	45	57	66	50	49	56
Upr CI	67	54	63	76	82	63	74	82	79	69	61	64	51	59	62	67	60	54	66	72	57	63	66

Fig. 3.11 *Cpue*$_{pos}$ and the PC amended *cpue* of panga (*Pterogymnus laniarius*) in the Port Alfred commercial linefishery between 1985 and 2007; **Inserts a)** monthly predicted PC (with 95% CI) and *cpue*, and **b)** the predicted *cpue*$_{pos}$ by region. The PC for each year is presented in the table below the plot (Lwr CI = lower 95% confidence interval, Upr CI = upper 95% confidence interval).

There was a significant difference in *cpue* between all the management periods (*K-W Anova: H = 4482.9, df = 3, p =< 0.001*). The *cpue* was highest in the period A (1985-1998) and decreased consistently during each consecutive period (Fig. 3.12).

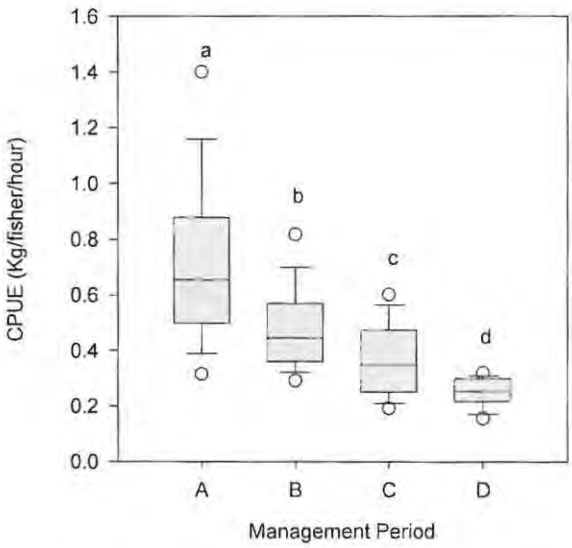
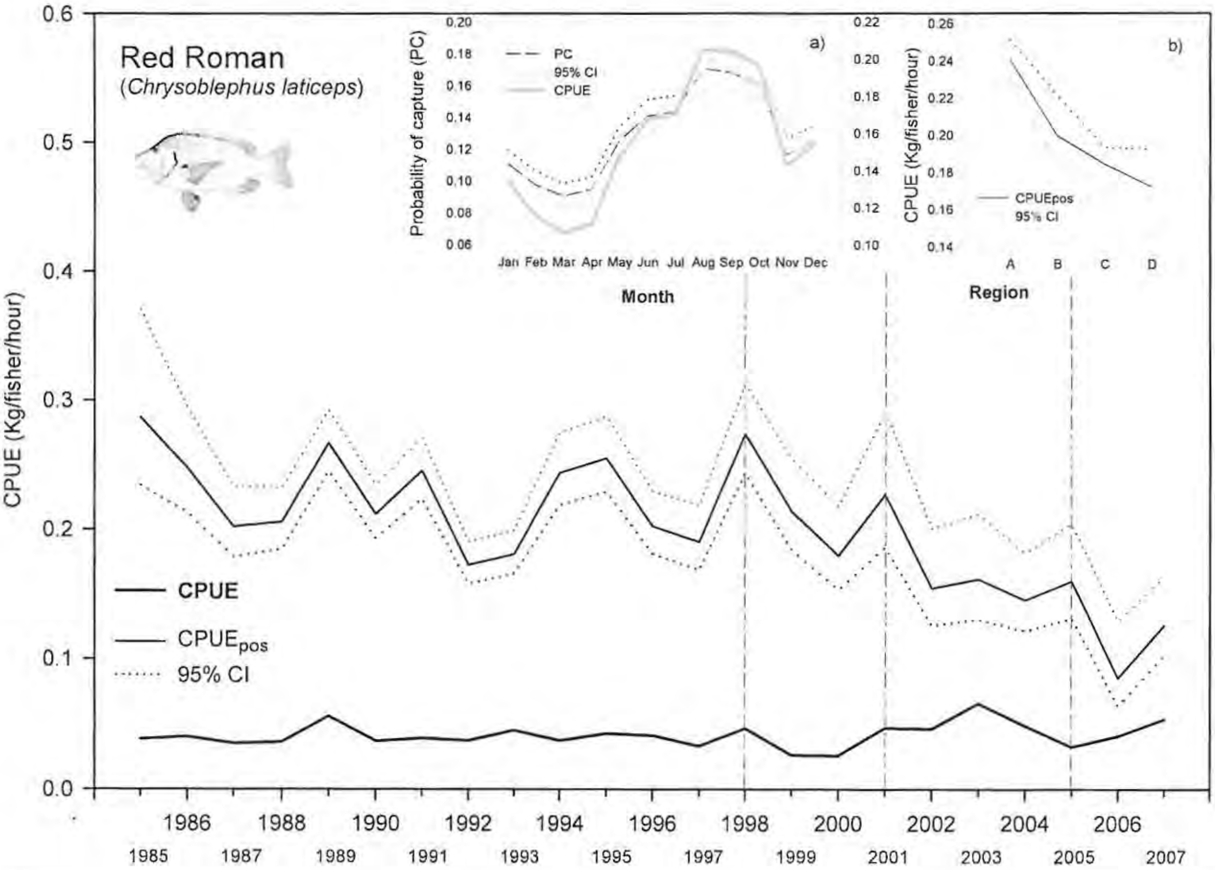


Fig. 3.12 *Cpue* of panga (*Pterogymnus laniarius*) during the management periods A(1985 -1998), B(1999 -2001), C(2002 -2005) and D(2006 -2007) in the Port Alfred commercial linefishery. Superscripts denote significant differences tested by Dunns' multiple comparison procedure at a *p*-level of 0.05.

Roman

The $cpue_{pos}$ (positive catch rates) of Roman remained fairly constant until 1998 and decreased from 1999 onwards (Fig. 3.13). However, due to the increase in PC from 2002 to 2007 the standardised *cpue* remained constant over the 23 year period (*x-coefficient = 0.0003*). *Cpue* in Period B was significantly lower than during the other periods (Fig. 3.14a; $p < 0.05$), although there was no significant difference in *cpue* for Roman between period D and periods A ($Q = 1.32, p > 0.05$) and C ($Q = 0.80, p > 0.05$).

From 1985 to 1998 the commercial B-licence holders were limited to a maximum of five roman per fisher.day^{-1}. In 1998, the A- and B-licence system was abolished and the remaining commercial operators that submitted catch returns were no longer limited by any bag limit for Roman. As such, during the 23 years there were two periods with different bag limits (1985-1998 and 1999-2007; Fig. 3.14b).



PC (%)	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007
Mean	10	12	12	12	15	12	11	14	16	11	12	14	11	13	8	9	14	18	25	20	12	22	24
Lwr CI	9	11	11	11	14	11	10	13	15	10	11	12	10	11	7	8	12	15	21	17	10	17	20
Upr CI	11	13	13	13	16	13	12	15	17	12	13	15	13	14	9	11	17	22	30	23	15	28	28

Fig. 3.13 *Cpue*$_{pos}$ and the PC amended *cpue* of Roman (*Chrysoblephus laticeps*) in the Port Alfred commercial linefishery between 1985 and 2007; **Inserts a)** monthly predicted PC (with 95% CI) and *cpue*, and **b)** the predicted *cpue*$_{pos}$ by region. The PC for each year is presented in the table below the plot (Lwr CI = lower 95% confidence interval, Upr CI = upper 95% confidence interval).

There was a significant difference in *cpue* between the two bag limit periods (*M-W U-test, p = < 0.001*). Counter intuitively, the standardised *cpue* of Roman was highest during 1985 to 1998 when the B-licence holders were subject to a bag limit, but dropped significantly from 1999 through to 2007 when the commercial fishers were no longer restricted by a bag limit.

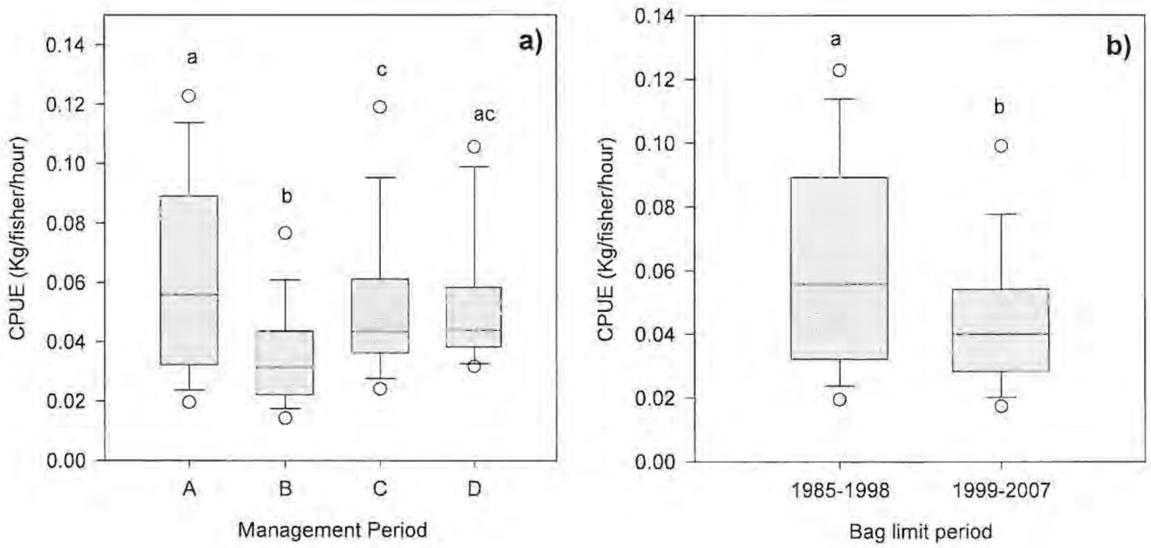
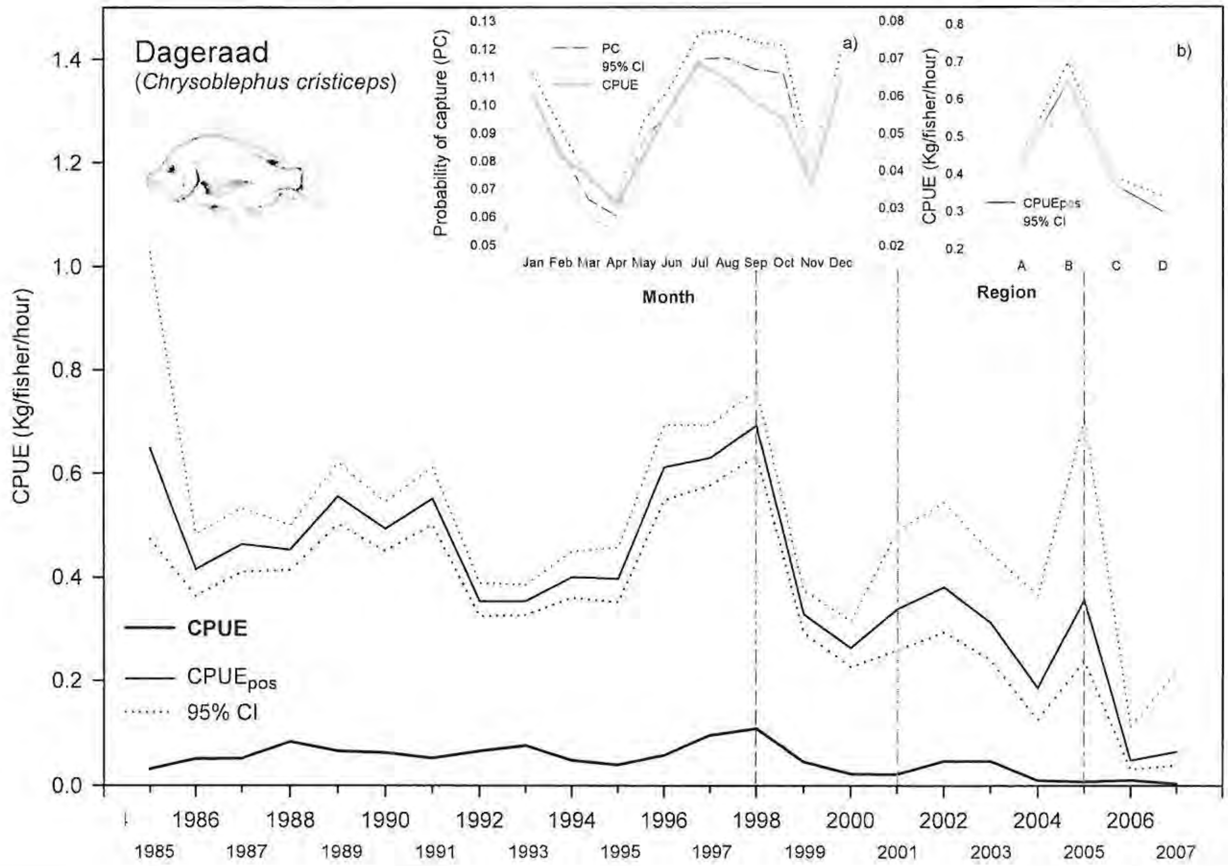


Fig. 3.14 a) *Cpue* of Roman (*Chrysoblephus laticeps*) during the management periods A(1985 -1998), B(1999 -2001), C(2002 -2005) and D(2006 -2007) in the Port Alfred commercial linefishery. **b)** The PC amended *cpue* of Roman with respect to periods of different bag limits: 1985 -1998 and 1999 -2007. Superscripts denote significant differences at a *p*-level of 0.05.

Dageraad

The $cpue_{pos}$ of dageraad increased from 0.42 kg.fisher^{-1}.hour^{-1} in 1985 to 0.69 kg.fisher^{-1}.hour^{-1} in 1998 (Fig. 3.15). The $cpue_{pos}$ and PC then decreased from 1999 to 2007. Similarly, the *cpue* gradually increases until 1998 and then decreases to almost zero kg.fisher^{-1}.hour^{-1} in 2007. The *cpue* in period A was significantly higher than the other periods (Fig. 1.16a; $p < 0.05$). There was no significant difference in *cpue* between the periods B and C ($Q = 0.47, p > 0.05$). Period D had a significantly lower *cpue* than all the other periods ($p < 0.05$).

Over the 23 years covered by the dataset there were three periods that had different commercial bag limits for dageraad. As was the case with roman, between 1985 and 1998 the B-licence holders were restricted to five dageraad per day. From 1999 to 2005, there was no bag limit for commercial fishers (due to the abolishment of the B-licences).



PC (%)	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007
Mean	4	9	9	14	10	10	8	14	16	9	8	8	13	13	10	6	5	9	11	3	2	6	2
Lwr CI	3	8	8	13	9	9	7	13	15	8	7	7	12	12	9	5	4	7	8	2	1	4	1
Upr CI	5	11	10	16	11	11	9	15	17	10	8	9	14	15	11	7	6	12	14	5	2	10	3

Fig. 3.15 *Cpue*$_{pos}$ and the PC amended *cpue* of dageraad (*Chrysoblephus cristiceps*) in the Port Alfred commercial linefishery between 1985 and 2007; **Inserts a)** monthly predicted PC (with 95% CI) and *cpue*, and **b)** the predicted *cpue*$_{pos}$ by region. The PC for each year is presented in the table below the plot (Lwr CI = lower 95% confidence interval, Upr CI = upper 95% confidence interval).

In 2006, commercial licence holders were limited to a single dageraad per fisher per day (which was coupled with a substantial 100mm increase in the minimum size limit). There was a significant difference in *cpue* between the three bag limit periods (*K-W Anova:* $H = 604.8$, $df = 3$, $p < 0.001$). Similar to Roman, the highest *cpue* was recorded during the 1985-1998 'bag limit period' after which the *cpue* decreased and particularly so during 2006-2007 (Fig. 3.16b).

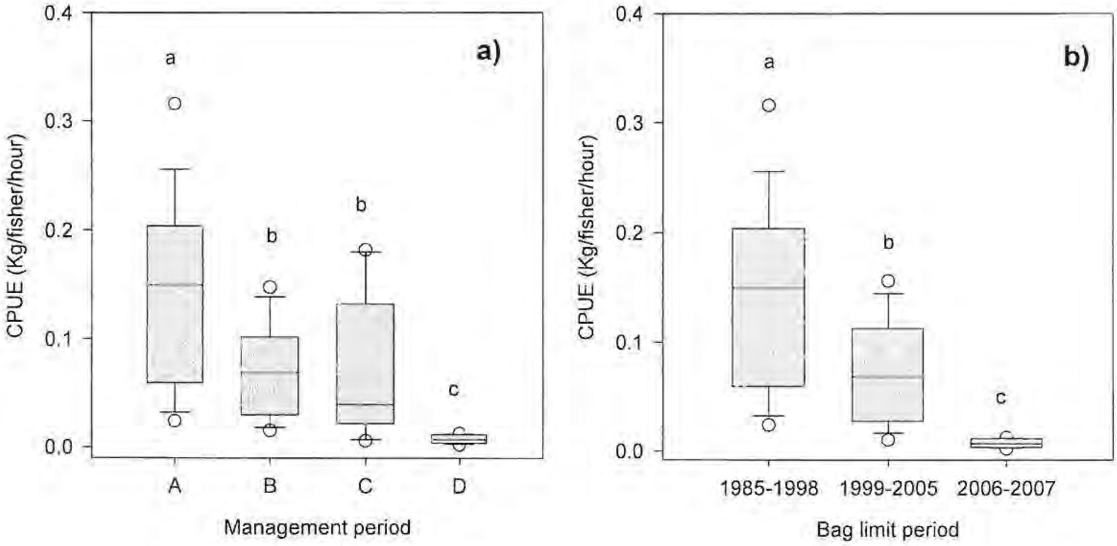


Fig. 3.16 a) *Cpue* of dageraad (*Chrysoblephus cristiceps*) during the management periods A(1985 - 1998), B(1999 -2001), C(2002 -2005) and D(2006 -2007) in the Port Alfred commercial linefishery. **b)** The PC amended *cpue* of dageraadwith respect to periods of different bag limits: 1985-1998, 1999-2005 and 2005-2007. Superscripts denote significant differences tested by Dunns' multiple comparison procedure at a *p*-level of 0.05.

DISCUSSION

Trends in fishing effort

To assess the effect of management changes on commercial catches with respect to composition, overall *cpue* and species-directed *cpue*, the 23 year period from 1985 to 2007 was separated into four management periods. These periods corresponded to major changes in the bag and minimum size regulations and the legislated reduction in fishing effort. These changes had a significant impact on fishing effort (in terms of the number of vessel outings per month) in a direct (Fig. 3.2) and indirect manner. However, within the four periods there were changes in effort that were not directly associated with the change in the licensing structure or catch restrictions. During management period A (1985 to 1998) there was an increase in total effort between 1985 and 1991 as new entrants entered the fishery, mostly as B-license holders.

These licenses were freely tradable, and this transferability resulted in a sustained high fishing effort as new unsuccessful entrants into the fishery tried to recoup their capital costs by selling their vessels (Penney 1999). However, from 1992 to 1998 there was a steady decline in effort, possibly because the fishery failed to produce expected returns. Penney (1999) noted that many of the B-licences were transferred from the Cape to KwaZulu-Natal (KZN) prior to this being prohibited in 1994. Although it is known that several Port Alfred vessels moved to KZN during the early 1990's, there is no indication of the number of licences that were transferred. This could partly explain the decrease in commercial effort in the Port Alfred fishery between 1991 and 1993. In 2004 and 2005 there was a marginal increase in effort and this was attributable to an abundance of geelbek and kob during those years, which made it economically viable for the remaining commercial operators to go to sea more frequently.

Although there were fluctuations, the commercial fishing effort during each of the four periods was significantly different and clearly represented distinct management periods. Effort was highest during period A (288.2 ±52 vessel-days.month^{-1}). After the abolishment of A- and B-licences in 1998 the commercial effort in Port Alfred was effectively reduced by 45%. At the end of 2001 the South African linefishery was declared to be in a state of crisis and in 2002 the commercial effort in Port Alfred was reduced by a further 63%. During management period D (2006 to 2007), the commercial effort (35.2 ±1.4 vessel-days.month^{-1}) was a mere 12% of the mean commercial effort between 1985 and 1998. The decline in commercial effort over time was largely mirrored by an increase in recreational vessels, although not in the same proportion (this will be discussed further in the following chapter).

During the four management periods that spanned the years from 1985 to 2007 there were significant changes in the commercial catch composition, overall *cpue*, and species-directed *cpue* in the fishery. However, the findings of this study must be interpreted within the limitations of data and with an acknowledgement of the behaviour of the fishers in response to the substantial regulatory shifts before drawing conclusions on the status of the fishery. These limitations are discussed in the following section.

Limitations of the data

The decrease in commercial effort and the consequent decrease in the number of submitted catch returns may increase the probability of the reporting errors not being randomly distributed. These errors could potentially affect the observed trends in the fishery. The reporting of by-catch and 'fry' in the Port Alfred fishery is a good example. While some commercial fishers fastidiously reported the 'unsold' catch others might have omitted them from their catch returns. Both Sauer *et al.* (1997) and Penney (1997) suggested that there is gross under-reporting of 'fry' retained by the crew. This is particularly notable with the *Galeichthyes* spp. and their contribution to the non-sparid group over the 25 year period in Port Alfred. There was a significant decrease in the contribution of *G. ater* and *G. feliceps* from 24% of the non-sparid group in 1985 to less than 1% in 1998, and are completely absent from the *reported* catch in the later years of the dataset. Hecht and Tilney (1989) estimated that less than 10% of sea catfish were reported and yet they contributed as much as 11% of the total catch in 1987. During the access point surveys of the Port Alfred fishery from 2006 to 2008 it was noted that the 'fry' retained by the crew were, in fact, excluded from catch returns.

Furthermore, as the number of commercial operators submitting catch returns decreases, the catch composition (and *cpue*) becomes more descriptive of the fishing practices of individual fishers and individual fishing trips. For example, between 1985 and 2002 the carpenter directed *cpue* gradually declined. However, from 2003 to 2007 there was a significant increase in the *cpue* of carpenter, which could be largely attributed to a *single* vessel that actively targeted the species in deeper water. With the few remaining number of commercial vessels in the fishery, the catch reported by this single vessel had a significant effect on the overall *cpue* during the last two management periods. Hecht and Tilney (1989) also reported that larger carpenter were targeted primarily by two larger commercial vessels operating in deeper water at the time. This was in fact reflected in the PC of carpenter in the late 1980's. However, the bias on the *cpue* would have been less due to the greater number of vessels submitting catch returns during that period. It is possible therefore that a certain, undetermined proportion of the variance in catch composition, *cpue* and species directed *cpue* may be due to the capacity and efficiency of individual vessels and evident trends could be the result of the fewer vessels reporting their catch to the NMLS database after 2002.

A further aspect that needs to be kept in mind is the 'dichotomy of the catch returns'. Up until 1999 there were two commercial licence categories (A and B). While all licence holders were required to submit catch data to the NMLS, the fishing practices of the two groups differed significantly. For the full-time operators (A-licence holders) commercial fishing was a primary source of income, while the part-time operators (B-licence holders) largely fished as recreational fishers or as commercial operators during particular good 'runs' of certain species (Hecht 1993, Penny *et al.* 1997, Sauer *et al.* 1997) In other words, the two licence holders were governed by different economic pressures (Hecht 1993). In addition to this, different fishing regulations applied to A- and B- licence holders for many of the targeted species (Chapter 1, Table 1.4). B-licence holders were limited to a maximum of 10 fish of any species per day (excluding silver kob, hake and carpenter) and a maximum of five species that were considered 'vulnerable' (these were mostly reef associated species, including Roman and dageraad). It would therefore be expected that the A- and B- licence holders had significantly different patterns of effort, catch compositions and catch rates. Since the analysis in this study included the catch returns submitted by both licences (for the purpose of highlighting the responses to management), the B-licence catch returns had the potential to dilute or inflate the trends evident for the full-time commercial operators. This would have an effect on the interpretation of effort, species composition, probability of capture, and *cpue* in the years prior to 1999.

Catch returns by B-licence holders also had an effect on the *cpue* for Roman and dageraad. From 1985 to 1998 the B-licence holders were limited to five Roman and dageraad per day. After the change to the licensing structure in 1998, the remaining commercial licence holders were no longer restricted by bag limits for these species and, counter-intuitively, from 1999 to 2005 the *cpue* of both Roman and dageraad declined. This could have been the result of the 'recreational tendencies' of the B-licence holders that submitted catch returns prior to 1998 and the remaining commercial operators not actively targeting these two species, or alternatively a decrease in abundance of these species, or a combination of both these factors

Trends in catch composition, cpue and species-directed cpue

The overall *cpue* in the Port Alfred commercial linefishery decreased by 37.4% from 1985 to 1998 (management period A). Similar trends were found by others working on commercial linefisheries in South Africa during the late 1990's and early 2000's (Attwood and Farquhar 1999, Penney *et al.* 1999, Griffiths 2000, Brouwer and Buxton 2002). Despite vessel and gear improvements, Penney *et al.* (1999) found that the catch rate for the KwaZulu-Natal commercial linefishery had been in constant decline since the inception of the fishery. Current catch rates at that time were about 10% of those at the start of the fishery (700 – 800 $kg.fisher^{-1}.year^{-1}$).

Similarly, Attwood and Farquhar (1999) found that the catch rate in the linefishery between Hangklip and Walker Bay in the Western Cape was a mere 25% of that recorded at the turn of the century.

From 1998/9 onwards there has been a substantial increase in the commercial *cpue* in the Port Alfred linefishery, in particular during management periods C and D. However, this significant increase in *cpue* is potentially misleading and should not be interpreted as a recovery of the fishery. The high *cpue* in 2004 and 2005 corresponded to high proportions of carpenter and geelbek in those years. Similarly, in 2007 the commercial catch was dominated by geelbek which also resulted in the high overall *cpue*. Without the diluting influence of the B-licence holders (who were limited to 10 fish.day^{-1}) and with the reduced number of submitted catch returns, the high *cpue* of individual fishing trips heavily influence the overall *cpue*. The investigation of the species-directed *cpue* sheds more light on the fishery trends as well as providing some indication of the status of the fishery.

Griffiths (1997a) suggested that silver kob could be the most important species caught by the linefishery between Cape Point and East London. Silver kob was, and still is, the mainstay species of the Port Alfred linefishery (Hecht 1993, Hecht and Tilney 1989). A constant, high proportion of the yearly total landings were attributed to the species for the entire 23 year period. During management periods A and B (1985 to 2001) silver kob made up the highest proportion of total landings. However, in periods C and D silver kob was the second most important species in terms of mass after geelbek (Fig. 3.3). Silver kob were landed on more than 50% of the fishing trips in most years and as often as on 85% of outings in some years (Fig. 3.5). The size limit of silver kob was only amended in 2006, and there have been no bag limits imposed on commercial operators. Furthermore, silver kob was targeted equally by A- and B-licence holders and the recreational sector (discussed further in the following chapter). Thus, in light of the significant changes to the management environment, silver kob could be considered the most viable indicator of the status of the Port Alfred linefishery when analysing the fishery on the basis of the fisheries dependent NMLS database.

Between 1985 and 2007 there has been a constant decline in the *cpue* of silver kob. This was consistent with the results of per recruit models applied by Griffiths (1997) for the three discrete silver kob subpopulations on the eastern seaboard of South Africa. He suggested that all stocks are heavily over-exploitedand there is limited potential for population growth. A decade ago it was suggested that the biomass of silver kob was less than 10% of the pristine spawner biomass (Griffiths 2000). Despite the suite of management changes that have been implemented since 1985, this study provides no indication of improvement of the silver kob stock in the Port Alfred area (which is part of the South Eastern Cape subpopulation; sensu Griffiths 1997a).

The other important sciaenid species in the fishery is geelbek. Geelbek on the eastern seaboard of South Africa can be divided into three size-related, sub-populations with well defined migratory patterns (Griffiths and Hecht 1995). In summary, juvenile fish of less than 500mm mainly occur in the 'nursery area' in the South East Cape (Port Alfred region). These fish then move south to the South Western Cape before joining the adult sub-population a year or two later. In autumn and early winter the adult sub-population (>810mm) migrate northwards to their spawning grounds in KwaZulu-Natal (Griffiths and Hecht 1995). The migratory nature of the species is evident in the significantly higher PC and *cpue* during the autumn and winter months (Fig. 3.7).

In 1992 the minimum size for geelbek was increased from 400mmTL to 600mmTL. This resulted in a significant reduction in the PC of geelbek (to a lowest point of 5% in the same year) as well as a decrease in *cpue* (Fig. 3.7). Although the possible impact of poor recruitment on the reduced PC during that time cannot be discounted, this suggests that a substantial proportion of geelbek landed by the commercial operators prior to 1992 were fish smaller than 600mmTL. In other words, prior to the increase in size limit in 1992, the juvenile sub-population was heavily targeted by commercial operators (this is substantiated in Chapter 5).

Since 1996, the PC and *cpue* of geelbek have been increasing (Fig. 3.7) and the contribution of geelbek to the total commercial landings increased significantly in the last two management periods. This was attributed to strong migratory pulses during the autumn and winter months (particularly in 2004, 2005 and 2007), and the highly directed effort by the remaining commercial operators. Despite being migratory and predominantly caught during the winter months, the annualised PC for geelbek was 50% in 2005 (i.e. geelbek were landed on half of the commercial fishing trips). During management period C, and particularly during 2005, geelbek was the most dominant species in the fishery, with larger fish (>800mmTL) being landed in autumn and winter and smaller fish <700mmTL caught in spring and summer. In fact, during

2004, 2005 and 2007 the overall *cpue* in the Port Alfred fishery was heavily influenced by the high abundance of geelbek.

Griffiths (2000) suggested an exponential decline in the abundance of geelbek in the South African linefishery (as much as 98% since 1900). The strong recruitment of geelbek in the last two management periods (from 2002 to 2007) in the Port Alfred fishery (and from hearsay evidence, the rest of the Eastern Cape) suggests that geelbek may be showing signs of recovery. This came 15 years after the increase in the minimum size limit and 5 years of significantly reduced commercial fishing effort. However, the close relationships between geelbek and their pelagic, shoaling prey (viz. *Sardinops sagax*, *Engraulis japonicas* and *Trachurus trachurus*) (Griffiths and Hecht 1995) and the erratic nature of these prey stocks (Armstrong *et al.* 1991, Barange *et al.* 1999, Swartzlose *et al.* 1999, Jablonski and Legey 2005) suggests that geelbek stocks have the potential to fluctuate greatly on a temporal basis. Therefore, geelbek should be considered a bonus species, and the increased *cpue* gives little to no indication of the overall health of the localised Port Alfred fishery.

Panga has played an important role in South African fisheries since the turn of the 19th century (summarised by Booth and Buxton 1997). It is an important catch of the trawl fishery and for a twenty seven year period (1964 to 1991) panga stocks were directly targeted by Japanese and Taiwanese trawlers with sustained landings as high as 18 000 tonnes per annum (Crawford *et al.* 1987). International vessels were excluded from the fishing grounds in 1992 and panga became an important species in the offshore linefishery (Booth and Buxton 1997). The shift was due to the declining abundance of larger, more valuable linefish species (Smale and Buxton 1985, Hecht and Tilney 1989, Badenhorst and Smale 1991). Studies done by Booth and Buxton (1997) suggested that panga stocks are in good health and could sustain substantial increases in fishing pressure of up to as five times the commercial effort of 1995.

In Port Alfred, panga currently contribute less than 5% of the total commercial landings despite being landed on more than 60% of the fishing trips in recent years. In congruence with the data presented by Booth and Buxton (1997), panga made substantial contributions to the Port Alfred commercial landings during the early 1990's (as much as 29% of the total landings by weight). Since then, the *cpue* of panga has steadily declined through each of the management periods. In light of the studies by Booth and Buxton (1997) this reduction in *cpue* is unlikely to be an indication of a decline in stock abundance but most likely a consequence of fisher behaviour. With increasing operating costs and the low market value of panga (2009 beach price of R14/kg; C. McClelland, Fresh Fish Market, pers. comm. 2009), commercial fishers were less likely to travel further to the deeper reefs to target the low value species.

Similarly, the decline in *cpue* of carpenter (the larger of which are also caught in deeper water; Hecht and Tilney 1989) may also be an effect of commercial operators fishing closer to the access points. As previously mentioned, a fewer commercial vessels were responsible for the high *cpue* in periods C and D. This would suggest that, like panga, trends in the carpenter directed cpue are representative of selective targeting and not necessarily an indication of stock abundance. In other words, this may suggest that economics has had a major impact on fisher behaviour and this could be greater than the impact of changes in output regulations and institutionally imposed reduction in effort.

Although Roman and dageraad make up a relatively small proportion of total commercial landings (less than 6.2% per annum throughout the 23 year period), it has been suggested that highly resident, long lived, late maturing, reef associated species such as roman and dageraad would be good indicators of the status and health of a fishery. The effect of changes in fishing pressure is most evident on these species, because of their late attainment of sexual maturity, lower natural mortality and greater longevity (Bruton 1990, Buxton 1993, Coleman 2000, Mann 2000, Newman 2000). Furthermore, they are more affected by lower levels of fishing pressure, are less resilient to fishing impacts and recovery is slower than species with more R-selected life histories (Russ 1991). However, changes in the minimum size and bag limit regulations that were instated for these species during the 23 year period had significant effects on their contribution to the catches and their *cpue* (Fig. 3.14b and Fig. 3.16b). For example, dageraad, which contributed a high proportion of the sparid group in previous years, decreased substantially in 1999. This corresponded with the abolishment of the A- and B-licences and could be related to the 'recreational tendencies' of the former B-licence holders. However, the most significant drop in *cpue* occurred in 2006 when the minimum legal size of dageraad was increased from 300mmFL to 400mmFL and the bag limit was reduced to a single dageraad per fisher per day for both the commercial and recreational fishers.

Although helpful in highlighting the effects of management changes, the usefulness of these species as indicators of fishery health (based on retrospective *cpue* analysis) is limited due to the regulatory changes that have taken place in South Africa.

The evidence presented on the basis of the analysis of the NMLS data suggests that changes to the South African linefishery management environment had a significant effect on effort, catch composition, overall and species-directed *cpue* in the Port Alfred/Kenton-on-Sea/Boknes commercial linefishery. In particular, the abolishment of the A- and B-licences in favour of annually allocated fishing rights in 1999, the reduction in commercial effort in 2002 and the amendments to minimum bag limits in 1992 and 2006. However, on closer examination it is necessary to highlight some anomalies. Firstly, the high overall *cpue* from 2004 to 2008 was related to strong migratory pulses of geelbek. Secondly, the changes in catch composition and *cpue* of many species reflect changes in fisher behaviour due to external, fishery independent factors such as rising operating costs and hence fishing closer to access points. To assess the current status of the Port Alfred fishery it therefore becomes necessary to juxtapose the direct impact of some of the management changes (e.g. the legislated reduction in effort) on the catch composition, overall and species directed *cpue* and PC, with the behavioural response of fishers to the changes in output regulations and their response to external factors such as rising operating costs.

Within the limitations of the NMLS, the species isolated as being the most unaffected by the management changes, and thus considered the 'canary in the coal mine', was silver kob. Evidence from this study suggests that, of all the species in the fishery, the *cpue* of silver kob would most accurately represent the actual stock abundance. Given the importance of silver kob to the welfare of the fishery, it is disquieting to find that the *cpue* of silver kob has declined steadily and significantly over the 23 year period. This could indicate that the management changes that have been implemented since 1985 have had little positive effect on the Port Alfred linefishery.

CHAPTER 4

A COMPARISON OF THE COMMERCIAL AND RECREATIONAL SECTORS IN THE PORT ALFRED LINEFISHERY

INTRODUCTION

Analysis of the fisheries dependent NMLS data in Chapter 3 clearly showed that there were significant changes in effort, catch rates, and in the catch composition in the Port Alfred commercial linefishery between 1985 and 2007. Many of the observed changes were attributed to the changes in linefish regulations and access management. In particular, commercial effort was reduced by changes to the commercial licence/rights structure in 1998, 2002 and 2006 (Table 1.3). The overall commercial *cpue* decreased from 3.6 to 2.3 kg.fisher^{-1}.hour^{-1} between 1985 and 1998. However, after 1998 the overall *cpue* increased to levels significantly higher than those at the start of the dataset (as high as 4.4 kg.fisher^{-1}.hour^{-1} in 2004). Amongst other reasons, the higher *cpue* was attributed principally to strong recruitment of the migratory geelbek in the fishery (particularly in 2004, 2005 and 2007) and the small number of remaining commercial operators who capitalized on the abundance of this species during those years. The *cpue* of silver kob (which can be considered as the mainstay of the fishery) continued to decline despite the significant reduction in commercial effort since 1985 (>80%), the increase in minimum size in 2006 (from 400mmTL to 500mmTL), and the recreational bag limit of 5 silver kob per fisher.day^{-1} that was imposed in 1998.

Despite the increasing trends in recreational effort that have been noted since 1985 (Smale and Buxton 1985, Hecht and Tilney 1989, Hecht 1993, Brouwer 1997, McGrath 1997, Sauer et al. 1997, Attwood and Farquhar 1999, Fennessy *et al.* 1999, Penney *et al.* 1999, Griffiths 2000, Brouwer and Buxton 2002), the NMLS has not accounted for the growth and influence of the recreational sector in many of the linefisheries in South Africa (Sauer *et al.* 1997).

For example, between 1975 and 1982, Smale and Buxton (1985) noted a 60% increase in the number of recreational vessels affiliated to a recreational ski-boat club in Port Elizabeth. In 1982, there were 645 recreational ski-boats associated with angling clubs (formal anglers) between Jeffrey's Bay and Kei Mouth (51 of which were registered in Port Alfred). Although not all of these vessels operated on a regular basis, they collectively landed an approximated 180 tonnes of fish per annum. This was equivalent to 37% of the commercial linefish landings in the Eastern Cape at the time (Smale and Buxton 1985).

Recreational efforthas continued to increase and, in 1995, there were an estimated 3444 recreational vessels operating with *regularity* along the South African coastline (excluding Transkei;Sauer *et al.* 1997). The majority of these vessels operated in KZN (85%), while 401 (11.6%) fished in the Eastern Cape (Sauer *et al.* 1997). The total number of formal and informal recreational vessels in the Eastern Cape was approximately 1100[vii] (Sauer *et al.* 1997, Brouwer and Buxton 2002). This equated to a 44% increase since the work of Smale and Buxton (1985) conducted in 1982. Furthermore, the recreational sector in the Eastern Cape landed approximately 410 tonnes in 1996 (Brouwer 1997, Sauer *et al.* 1997). Not only was this 22% of the commercial landings in that year, but more importantly, it represented an increase of more than 50% in total recreational landings since 1982 (Smale and Buxton 1985, Brouwer 1997). In 2007 the recreational ski-boat sector had effectively doubled since 1997, and there were an estimated 31 860 recreational deepsea anglers (9488 formal and 22372 informal) in South Africa, or approximately 6300 vessels[viii] (Leibold and van Zyl 2008).

In the last decade, except for the studies done by Brouwer and Buxton (2002) and Leibold and van Zyl (2008) there have been no studies quantifying the growth (or decline) of the recreational linefish sector in South Africa and its impact on the resources. McGrath (1997) suggested that even with decreasing catch rates and/or the implementation of more limiting catch regulations there would be no significant decline in recreational fishing effort. This was largely due to the high capital investment in the recreational sector (i.e. vessels and gear) and the need to maximise the use of equipment despite landing fewer fish per trip. It was hypothesized that the recreational

[vii] This figure was calculated from estimate of recreational effort from Still Bay to Kei Mouth (Brouwer and Buxton 2002) and the regional breakdown of recreational effort on the South African coastline provided by Sauer et. al (1997).

[viii] Calculated with an average of 5 anglers per vessel.

sector in the Port Alfred linefishery has continued to grow naturally and additionally, as a consequence of the reduction in commercial effort.This is putting increasing pressure on the already declining fish stocks (Chapter 1, Table 1.1).

There has been an increasing worldwide recognition that, in comparison to commercial fisheries, the impacts of recreational angling are not as benign as previously conceived (Post et al. 2002, Mcphee et al. 2002, Pradervand and Baird 2002, Schroeder and Love 2002, Coleman et al. 2004, Lewin et al. 2006). Without proper management measures in place, the recreational sector could have substantial negative impacts on exploited fish populations (Lewin et al. 2006). As a prelude to their work on the collapse of linefish stocks between Walker Bay and Hangklip in South Africa, Attwood and Farquhar (1999) suggest that combined fishing pressure by all sectors is likely to have an impact on fish stocks by substantially increasing the fishing mortality. The same is likely to be true for the Port Alfred fishery. With commercial effort capped in 2006 (Chapter 1, Table 1.2), the recreational sector will have a proportionately greater and greater impact on fish stocks, irrespective of whether or not further catch restrictions are put in place.

As was briefly discussed in the previous chapter, the dichotomy of commercial licences between 1985 to 1998 resulted in a cline gradation between 'full-time' commercial operators (A-licence holders) and commercial fishers that operated in the same manner as the recreationals (some of the B-licence holders) (Hecht and Tinley 1989, Hecht 1993, Sauer *et al.* 1997). The differences in economic motivation between A- and B-license holders not only blurred the lines of definition between the commercial and recreational sectors, but also complicated management measures (Sauer *et al.* 1997). Apart from minimum legal size limits[ix] for the A-licence holders there were a few other minor bag limit restrictions, the fishery was essentially open in terms of catch. The B-license holders on the other hand, were limited by a number of bag-limit regulations. The majority of these regulations were focussed on the more vulnerable, reef-associated species such as dageraad, Roman, poenskop and red stumpnose (Chapter 1, Table 1.3). Hecht (1993) noted that in Port Alfred between 1982 and 1988, the *cpue* of B-licence holders was significantly lower than that of the A-license holders and the duration of an average fishing day was more comparable to 'true' recreational fishers. The current recreational fishers are limited by an even

[ix] Minimum legal size limits were applicable to both types of commercial licence holders as well as the recreational fishers.

greater number of size/bag limit restrictions. The most pertinent change from the previous B-license restrictions to the currently applicable recreational regulations were the reduced bag-limits implemented for the two sciaenid species viz. silver kob (5 per fisher.day^{-1}) and geelbek (2 per fisher.day^{-1}) in 2006 (Chapter 1, Table 1.3).

In 1987 there were 54 commercially registered vessels in the Port Alfred fishery (13 A-licenses and 41 B-licences). All of the A-licensed vessels fished on a full-time basis, while only 15 of the B-licence holders fished commercially on a regular basis (Hecht and Tilney, 1989). In 1988 there were 82 were locally based recreational vessels and 47 short term visiting boats. An average of only 16 of the resident recreational vessels fished on a regular basis. The remainder fished sporadically or when there were good fish 'runs' (Hecht 1993). In terms of the active number of vessels, the Port Alfred linefishery during that time (1988 and 1989) was numerically dominated by commercial vessels. However, it was hypothesized that after the A- and B-licensing structure was abolished in 1998 (Chapter 1, Table 1.2) many, if not most of the former B-license holders remained in the fishery as recreational fishers, and there was a subsequent shift from commercial to recreational effort in the Port Alfred linefishery.

The aim of this chapter was to the identify and quantify the differences between the commercial and recreational sectors in the Port Alfred linefishery during 2006-2008 and in so doing to construct a descriptive snapshot image of the fishery (in terms of its structure, effort, catch composition, *cpue* and total landings). Where possible, the current metrics were compared with those estimated from the available historical data in order to assess the response of the fishery to changes in the linefish management environment.

MATERIALS AND METHODS

Fishing effort

Robust quantitative data are essential for the evaluation of the performance of a fishery and enables fishery managers to make rational and well-informed decisions pertaining to that fishery (Chen 2003, FAO 2006). Nominal commercial effort data is normally collected from logbooks (Hilborn and Walters 1992), port monitoring or more recently satellite-based systems such as VMS (Meaden and Kemp 1997). Broad continuous data sets collected from commercial fisheries (often through well-designed sampling programmes) tend to have a characteristically small error which is random in nature (Chen 2003). The quantity and quality of fisheries data that is collected is generally positively related to the economic or social value of that particular fishery (Chen 2003). In other words, large commercial fisheries tend to have more comprehensive data sets with a longer time series than other sectors.

In contrast, recreational (and small-scale commercial) catch and effort data is not often as easy to collect and there is potential for misrepresentation (Borlando-Machado 2006). Catch and effort data for recreational and small-scale commercial fisheries are most often collected by access-point surveys or mail surveys (Hilborn and Walters 1992, Pollock 1997). These sub-samples are then raised to an estimated effort based on a 'frame survey' of the number of active vessels (FAO 2006). As a result recreational fisheries data is often of a lower quality and lesser quantity, and the associated error tends to be non-random and biased (Chen 2003).

The main assumption of a 'frame survey' method is that fishing effort is evenly and predictably distributed on a temporal scale. In the Port Alfred linefishery adverse sea and weather conditions play a role in the temporal distribution of effort due to the exposed nature of the launch sites (Hecht 1993) and assuming that all recreational users behave in a similar fashion would clearly introduce significant bias. Effort calculations extrapolated from frame surveys are thus likely to be misrepresentative of the effort in this fishery.

To draw comparisons between the commercial and recreational sectors and obtain a comprehensive understanding of the fishery it was necessary to measure the effort of the two sectors in the same manner. This was done by installing a digital, motion-activated, CCTV camera at the mouth of the Kowie River, which is the primary access point of the fishery (Fig. 2.2, Fig. 4.1a). After a month of testing and calibration, the camera was in full operation at the beginning of March 2007. The camera continuously recorded the nominal effort of the commercial and recreational boats for a period of 12 months (March 2007 to February 2008).

Vessels that passed into the video frame of the camera triggered an integrated digital motion sensor and the camera recorded a 5 second video clip, which was stored on a digital video recording unit. The video clips were examined weekly and the best frame of the clip was converted into a single still image. These images recorded the time and date of occurrence, the identity of the vessel (by comparison to a photographic database of the vessels in the fishery), as well as the number of crew (Fig. 4.1b). Vessels were captured going in and out of the river mouth, hence providing an accurate assessment of time-based nominal effort.

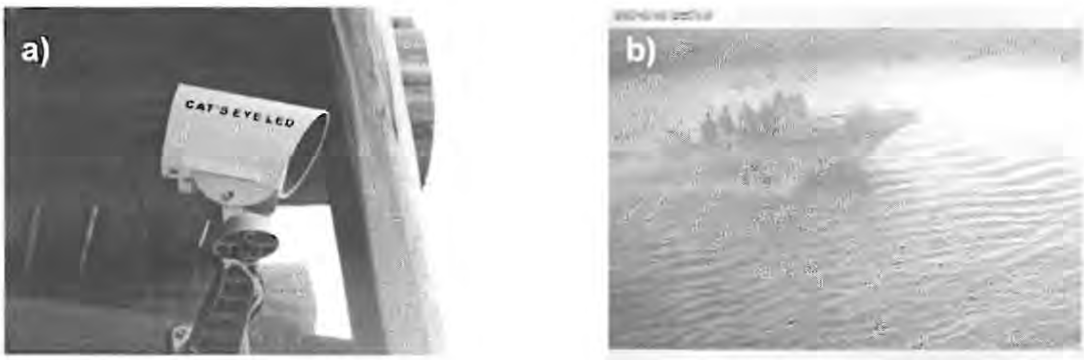


Fig. 4.1 a) The CCTV camera positioned at the mouth of the Kowie River in Port Alfred. b) A still image of a commercial vessel (5 crew) going to sea on the 2007/04/30 at 6.24 am.

The CCTV camera data were also used in combination with the data on weather/sea conditions (refer to Chapter 2) to investigate the effect of prevailing weather on fishing patterns in the fishery. A generalized linear model (GLM) was used to test the effects of wind speed, wind direction, swell period, swell size and swell direction on the temporal distribution of effort. The numbers of days with and without fishing activity were modelled with respect to the predictor variables using a binomial distribution with a logit link.

Prior to applying the GLM all the continuous predictor variables were entered into a correlation matrix. Correlated (i.e. 'redundant') variables were excluded in order to simplify the model (Crawley 1993). The optimal combination of the remaining predictor variables were found using an Akaike information criterion (AIC; Akaike 1973). Such that:

$$AIC = -2\ln[L(\theta_{p|y})] + 2p \quad (4.1)$$

where, $L(\theta_{p|y})$ is the likelihood of model parameters given the data y, and p is the number of free parameters (Johnson and Omland 2004).

The GLM was represented by the equation:

$$g(\pi) = \int \beta_0 + \beta_1(x_i) + \varepsilon \quad (4.2)$$

where g is the link function, $\beta_1(x_i)$ is the coefficient and estimation of the i^{th} predictor variable and ε is the error of the model. The logit link function is described by:

$$g = \log\left(\frac{p}{1-p}\right) \quad (4.3)$$

where, p is the continuous probability of the binary dependent variable, with a value between 0 and 1.

Catch composition and species diversity

The catch composition of the commercial and recreational landings in the fishery was investigated on a yearly basis for three equal time periods, each roughly a decade apart (1985 to 1987, 1996 to 1998, and 2006 to 2008). For each period three catch compositions were constructed for both the commercial and recreational fishers. As was done in Chapter 3, the first comprised of the main species in the commercial and recreational landings (silver kob, geelbek, panga, and carpenter), a 'sparid group', and a 'non-sparid' group (which included elasmobranches). It was hypothesized that, although the sparid and non-sparid groups would not account for much of the total landings, there would be a significant difference in the composition of these groups between the commercial and recreational catches. Therefore, the second and third catch compositions were constructed as the proportional breakdowns of the sparid and the non-sparid species to their respective groups (i.e. a percent of the total group weight).

For comparative purposes, the catch compositions for the main species of the commercial sector during each time period were constructed from data submitted to the NMLS. It was assumed that the former A-license holders operated in a similar fashion to the current long-term commercial rights holders. Thus, for the first two time periods (1985-87 and 1996-98) only the data submitted by the A-license holders was used to construct the catch compositions.

On the basis of evidence presented by Hecht and Tilney (1989) it was loosely assumed that some of the former B-licence holders in the Port Alfred fishery operated in a similar manner to the current recreational operators. On this assumption, and in the absence of any empirical data, the 'recreational' catch compositions for the two historical time periods (1985-87 and 1996-98) were constructed using catch return data submitted by B-license holders who landed less than ten tonnes over the respective three year sample periods. It must be pointed out that little inference will be drawn from these historical 'recreational' catch compositions due to lack of quantifiable robustness of the data (these data merely provide vague reference points in the absence of more descript data). The third period for which the recreational catch composition was constructed was for 2006 to 2008. This was based on the data collected during the regular sampling programme in this study.

Although suitable for constructing catch compositions of the main species in the fishery, the resolution of the NMLS data was not adequate to construct catch compositions of the sparid and non-sparid groups. These were constructed by using the access point survey data collected by Hecht and Tilney (1989) for the period 1985-87, C. Pittaway between 1996-98 (commissioned by MCM), and the data collected during the current study period (2006-08).

As shown in Chapter 3, from 2004 onwards commercial landings in Port Alfred fishery were dominated largely by a single species viz. geelbek. Unlike the commercial operators, the recreational fishers were subjected to bag limits for the more important species in the fishery such as geelbek (2 per $fisher.day^{-1}$), silver kob (5 per $fisher.day^{-1}$) and carpenter (4 per $fisher.day^{-1}$). Therefore, it was hypothesised that the species in recreational landings would, on a proportionate basis, be more evenly distributed. The dominance structure of the commercial and recreational sector landings will be investigated with the community-based analyses in Chapter 5.

Cpue

As outlined by Pollock *et al.* (1994), basic calculations of catch rates took the form:

$$catch\ rate = \sum_{i=1}^{n}\left(\frac{C_i}{E_i}\right) \Big/ n \tag{4.4}$$

where C_i is the catch and E_i is the effort on day i.

Catch rate was calculated per vessel on a daily basis for commercial and recreational operators for the periods 1985-87, 1996-98, and 2006-08. As for the catch compositions, estimations of catch rate for the 'recreational' sector during 1985-87 and 1996-98 were extracted from the NMLS (identified as B-licence holders landing less than ten tonnes during each of the three year sampling periods). The catch rates of both sectors for the 2006-08 period were calculated using data collected by the author during the current sampling programme.

The catch rates for each sector and time period were standardised with respect to two predictor variables viz. 'month' and 'area fished' by means of a GLM (equation 4.2). A gamma distribution with a logarithmic link function provided the best fit to the *cpue* data. The link function is simply described as:

$$g(\pi) = log\ (\pi) \tag{4.5}$$

Differences in the standardised *cpue* of the three time periods, as well as those of the commercial and recreational sectors, were examined by means of a Kruskal-Wallis One-Way Anova on ranks followed by Dunn's multiple comparison procedure.

Total landings

The total annual catch (kg) for the commercial and recreational sectors was estimated using the CCTV effort data (recorded as the average number of vessel-days fished per month) and *cpue* estimations for the period 2006 to 2008 (in $kg.vessel^{-1}.day^{-1}$). Because the camera only recorded fishing activity of vessels accessing the fishery through the Kowie river mouth, fish shop invoice data as well as the NMLS data were used to identify the proportion of commercial effort that occurred in the other areas of the fishery (Kenton-on-Sea, Boknes and Cannon Rocks). Although very crude, an indication of the proportion of recreational effort for other areas was estimated from counts of recreational trailers at launch sites (commercial trailers were identified and excluded from the count). Total landings (C_{total}) for each sector where then calculated as:

$$C_{total} = CPUE \times \frac{Effort}{p} \tag{4.6}$$

where, p is the proportion of effort that occurred in other areas of the fishery.

Non-parametric bootstrapping was used to generate lower and upper 95% confidence levels of the total catch for both the commercial and recreational sectors of the fishery (Efron 1982, Buckland 1984, Holye and Cameron 2003).

Estimates of total commercial landings for the two historical periods (1985-87 and 1996-98) were extracted from the NMLS. Calculations of total landings for recreational vessels during these historical periods were omitted from this study due to a lack of any reasonable estimate of total effort.

RESULTS

Structure of the fishery

In 2006-08 there were 161 registered sea-going vessels in the Port Alfred/Kenton-on-Sea/Boknes fishery (Port Alfred Small Boat harbour records), of which 153 were recreational boats. Although eight commercial rights were allocated to the area only four boats operated on a full-time commercial basis (two from Boknes/Kenton and two from Port Alfred). Only 26 of the recreational boats (17% of the total number) went to sea more than once a month (Table 4.1) while the remainder only went to sea occasionally. Within the recreational sector there were three charter vessels that actively operated in the area. These were predominantly larger catamaran vessels licensed to carry up to 10 fishers. These three vessels accounted for an estimated 55% of the recreational effort during a "working" week (Monday to Friday).

All the commercial and most recreational vessels (except one Bertrams displacement hull boat) are classified as ski-boats, between 6.5 and 9m in length, mainly catamaran type hulls, and powered by two outboard motors of between 60 hp and 250 hp. In addition to these, the recreational fleet also includes 3.5 to 5m inflatable ducks (11%) and smaller, 3 to 4m one-man 'skiffs' (3%). The latter operate within one nautical mile of the access points and are powered by a single outboard of between 20hp and 30hp.

During the 2006-2008 sampling period the only capital investment in to the Port Alfred fishery (new vessels and new fishing gear) occurred in the recreational sector. In contrast, the commercial vessels were substantially older and visibly very little money was spent on maintenance and upkeep (Fig. 4.2). Similarly, the commercial crew fished predominantly with un-maintained rods, 'Scarborough' reels and older tackle.

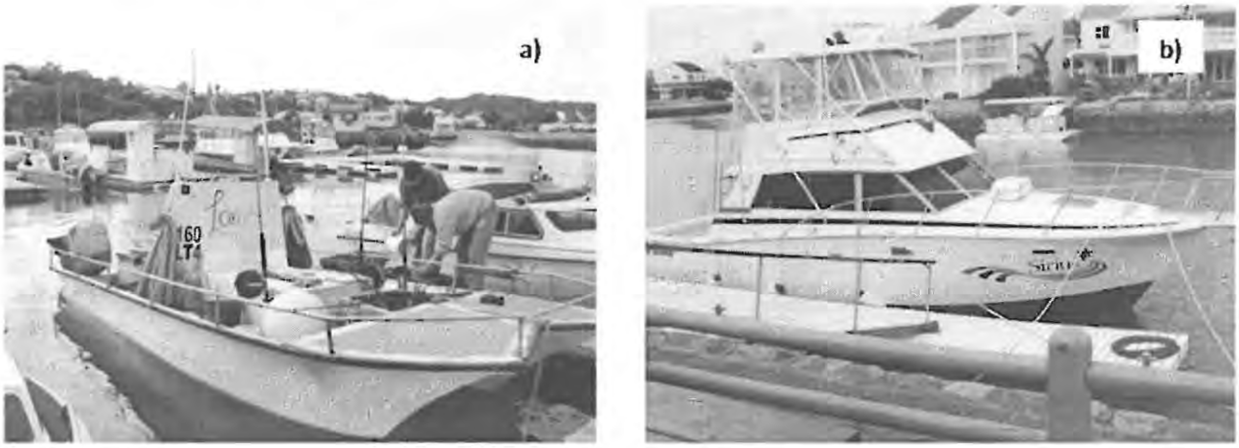


Fig. 4.2 a) The commercial ski-boat 'Lamont' in the small-boat harbour and **b)** 'Sirius' one of the many recreational vessels operating out of Port Alfred.

Table 4.1 Summarised profile of the commercial and recreational sectors of the Port Alfred fishery between 2006 and 2008.

		Commercial	Recreational	
			Individual	Charter
Total number of vessels		8	149	4
Number of active vessels		4	23	3
Average number of Crew		4 to 8 ($\bar{x}$= 6.2)	2 to 8 ($\bar{x}$ = 4.7)	up to 10 (n/a)
Vessel specs.:	***Vessel size***	6.5 – 8.2 m	3.5 – 10.4 m	6.7 – 8.2m
	Vessel type	Catamaran hull	Catamaran hull (71%), mono-hull (15%), inflatable (11%) and one-man skiffs (3%)	Catamaran hull
	Vessel age	10 to 25 years	New to 30 years	> 15 years
	Engines	2 x 60 - 115 HP	2 x 60 - 250 HP and single engines of 20 - 60 HP	2 x 120 to 250 HP
Gear type:	***Reels***	Scarboroughs	Multipliers, scarboroughs and coffee-grinders	Multipliers and Scarboroughs
	Bait	Pilchards	Pilchards, chokka and plastic lures	Pilchards
	Hooks	Multiple hooks (up to 4) 5/0 and 6/0	Single or double hooks 3/0 to 7/0	Single hooks 3/0 to 7/0
	Navigation	Echo-sounders and GPS	Echo-sounders and GPS	Echo-sounders and GPS

Both the commercial and recreational vessels use echo-sounders and global positioning systems (GPS). However, as in the past, the regular commercial operators (and a handful of the recreational fishers with knowledge of the fishery) often used landmarks to position themselves in well-known fishing areas.

Fishing effort

Table 4.2 summarizes the parameters of nominal effort of the commercial and recreational vessels in Port Alfred. It should be noted that the recreational effort refers only to the 26 active recreational vessels. The others that went to sea between one and ten times per year were excluded from this exercise. The data showed that the total nominal recreational effort over the year (2468 hours) was 56% higher than the commercial nominal effort (1382 hours). On average, the two commercial operators only fished on between five and ten days per month. This was 5 days less per month than the recreational fishers. In other words, the commercial fishers did not go out to sea on every available day. However, when the commercial vessels did put to sea they fished an average of 3 hours longer than the recreational vessels (8.3 ± 3.2 hours compared to 4.9 ±1.2 hours, respectively). The commercial vessels also carried a larger number of crew (6.2 ±0.4) than the recreational vessels (4.7 ±1.1). Despite the higher commercial crew number, the effective recreational effort (966.6 fisher days.year^{-1}) was still 26.2% higher than the commercial effort (714.0 fisher days.year^{-1}).

Fish purchase invoice data from local retail outlets were used to estimate the proportion of commercial effort that occurred at Kenton on Sea and Boknes. The proportion of recreational effort from these areas was estimated by recreational trailer counts. These were undertaken on a random basis for the two year period between June 2006 and April 2008.

During 2006 to 2008, the Fresh Fish Market was the main retail outlet/distributor of fish in the Port Alfred area. Approximately 95% of the commercially landed catch in the fishery were sold to this retailer (C. McClelland, Fresh Fish Market, pers. comm. 2008). The purchase invoice data from the retail outlet indicated that 56.8% of the commercial fishing trips during 2006 to 2008 were undertaken at Kenton-on-Sea/Boknes/Cannon Rocks. In contrast, 77.2% of the recreational fishing trips were out of the Kowie River mouth in Port Alfred. The other launch sites/access points made relatively small contributions to overall recreational effort (Table 4.3).

Table 4.2 The nominal effort of commercial (*n*=2) and recreational vessels (*n*=26) in the Port Alfred linefishery as gathered from the CCTV monitoring of the fishery (Kowie River Mouth: March 2007 to February 2008).

	Fishing time (hr:min / month)	Duration of fishing trip (hr:min / day)	Vessel days/ month	Number of days fished/ month	Number of crew
		Commercial (Total fishing time = 1382h 07min)			
Mean	106:19	8:14	13.2	7.7	6.2
St. Dev	41:54	3:09	2.0	1.4	0.4
Min	56:47	3:47	10	5	3
Max	206:49	14:46	17	10	10
n	171	171	171	100	99
		Recreational (Total fishing time = 2468h 39min)			
Mean	189:53	4:55	39.2	12.5	4.7
St. Dev	84:54	1:11	14.5	3.2	1.1
Min	92:04	1:02	16	7	1
Max	401:05	8:58	61	16	10
n	509	509	509	162	322

Table 4.3 Recreational trailer counts at the various access points/ launch sites in the Port Alfred fishery.

	No. days sampled	Trailer count	Vessel days /month	Proportion of recreational effort (%)
Port Alfred	n/a	n/a	39.2*	77.2
Kenton-on-Sea	51	15	8.2	16.1
Boknes	37	2	1.5	3.0
Kleinemonde	104	4	1.1	2.2
Cannon Rocks	36	1	0.8	1.6
Fish River Mouth	4	0	0	0.0
Kasouga	2	0	0	0.0
Total	234	22	50.81	100

* mean value extracted from CCTV camera data

The influence of weather patterns on the temporal distribution of fishing effort out of the Kowie River mouth was investigated. It was found that wind speed was auto-correlated with both swell size (*r = 0.417, p < 0.001*) and swell period (*r = -0.246, p < 0.001*). These two factors were thus excluded from the AIC analysis (Table 4.4). In terms of weather/sea conditions, wind speed was the best predictor of fishing activity in the Port Alfred fishery. The mean wind speed on days with fishing activity ($\bar{x}$ = 19.15 ±7.9 km.hour^{-1}) was significantly lower than days with no fishing activity ($\bar{x}$ = 22.77 ±8.84 km.hour^{-1}).

Table 4.4 Results of AIC and GLM analyses determining the influence of weather/sea conditions on fishing effort in Port Alfred between March 2007 and February 2008.

	AIC analysis				**GLM analysis**			
Parameters	*df*	AIC	Likelihood ratio (x^2)	*p*-value	Effect	*df*	Wald (x^2)	*p*
Wind speed	1	962.10	31.75	< 0.001*	Intercept	1	10.79	< 0.018*
Wave direction + Wind speed	9	971.99	37.86	< 0.001*	Wind speed	1	28.11	<0.001*
Wind direction + Wind speed	16	972.49	51.36	< 0.001*				
Wind direction + Wave direction + Wind speed	24	982.30	57.55	< 0.001*				
Wind direction	15	986.09	35.77	0.002**				
Wind direction + Wave direction	23	995.11	42.74	0.007**				
Wave direction	8	1001.09	6.76	0.562 ns				

ns = Not significant

* p < 0.05

** p < 0.01

Catch composition

A total of 75 species in 24 families were recorded on more than one occasion during the three time periods (Appendix A). Despite the high number of species recorded, only four regularly contributed more than 10% to the commercial and recreational total landings. The composition for the periods 1985-87, 1996-98, and 2006-08 for both the commercial and recreational sectors are presented in Figure 4.3.

In terms of the contribution to total landings (commercial and recreational), silver kob was, and still is, the most important species in the fishery. During 1985-87, 1996-98 it contributed 37.1 ±9% and 33.2 ±6% to the total commercial catch, respectively. In 2006-08 the contribution by silver kob dropped to 24.7 ±7%, and was the second largest contributor after geelbek. Although the proportion of silver kob in the recreational total landings decreased from 46.4 ±5% in 1985-87 to 28.9 ±10% in 2006-08, it remained the dominant species during all three periods. The contribution by silver kob to total landings decreased at a similar rate for both the commercial and recreational sectors from 1985-87 to 2006-08, testament to the fact that silver kob is equally targeted by the commercial and recreational sectors.

During both of the earlier time periods geelbek contributed less than 15% to both the commercial and recreational landings (Fig.4.3). However, since 2004 the contribution by geelbek to the total commercial and recreational landings increased significantly and in 2006-08, geelbek was the largest contributor to the commercial catch (32.7 ±5%). Although proportionately less than in the commercial landings, geelbek also made a significant contribution (20.2 ±17%) to the recreational catch in 2006-08.

Carpenter was the second largest contributor to the total commercial landings in both 1985-87 and 1996-98 contributing 34.4 ±4% and 18.5 ±2%, respectively. Despite increasing in proportion from the previous two periods, in 2006-08 carpenter was the third largest contributor to the commercial landings (23.9 ±4%) after geelbek and silver kob. Within the recreational sector carpenter contributed substantially in 1985-87 and 1996-98. However in 2006-08, it made an insignificant contribution (3.1 ±1%) to the recreational catch. The proportion of carpenter in the recreational landings was lower than that of the commercial sector during all periods, particularly during 2006-08.

Panga was the third largest contributor to the commercial landings in the two earlier periods, during which it contributed 15.5 ±5% and 15.3 ±3%, respectively. In 2006-08, panga only contributed 5.3 ±2% and 5.8 ±2% to the commercial and recreational total landings, respectively.

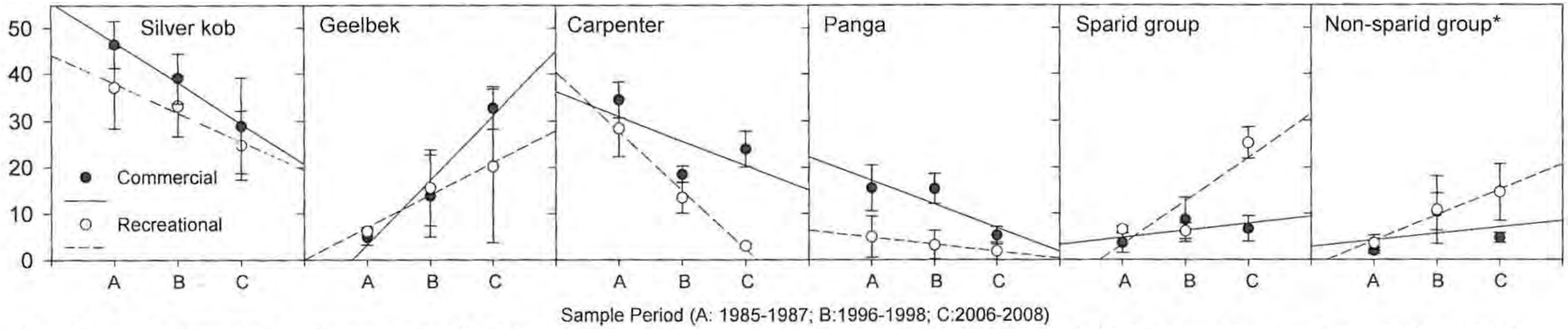


Fig. 4.3 Catch composition (percentage of total landings) of the main species and groups in the Port Alfred commercial and recreational sectors during the periods 1985 to 1987, 1996 to 1998 and 2006 to 2008. *The non-sparid group includes elasmobranches.

Table 4.5 Catch composition of selected species (percent of total group mass) in the sparid and 'non-sparid' groups in the Port Alfred commercial and recreational linefishery sectors during the periods 1985 to 1987, 1996 to 1998 and 2006 to 2008.

		1985 to 1987		1996 to 1998		2006 to 2008	
		commercial	recreational	commercial	recreational	commercial	recreational
Sparid Group							
	Red roman	11.2 ± 4.9%	22.1 ± 3.0%	10.8 ± 2.6%	22.4 ± 4.6%	9.5 ± 8.5%	11.4 ± 25.7%
	Red steenbras	44.0 ± 11.9%	15.3 ± 18.8%	3.3 ± 0.9%	4.6 ± 2.8%	9.9 ± 3.0%	5.1 ± 2.9%
	Red stumpnose	18.0 ± 12.6%	4.4 ± 3.4%	21.1 ± 16%	4.1 ± 2.1%	4.8 ± 5.1%	13.9 ± 4.2%
	Dageraad	21.7 ± 3.5%	28.8 ± 2.3%	44.8 ± 12.2%	27.3 ± 5.3%	1.5 ± 2.9%	3.6 ± 2.1%
	Santer	3.3 ± 3.2%	17.2 ± 9.7%	19.0 ± 1.8%	32.9 ± 3.8%	36.6 ± 21.6%	46.4 ± 25.4%
	Other	1.9 ± 1.4%	12.2 ± 11.5%	0.9 ± 0.5%	8.7 ± 4.5%	37.7 ± 15.1%	19.5 ± 7.3%
Non-Sparid Group							
	Hake	59.1 ± 20.2%	25.8 ± 3.6%	77.5 ± 49.6%	64.7 ± 33.7%	25.9 ± 3.6%	24.7 ± 20.1%
	Galeichthyes spp.	30.6 ± 2.6%	31.9 ± 22.3%	1.0 ± 0.0%	1.4 ± 0.9%	4.6 ± 0.7%	7.6 ± 1.4%
	Elf	0.0 ± 0.0%	0.7 ± 0.8%	0.6 ± 0.7%	3.6 ± 3.0%	65.8 ± 25.6%	3.4 ± 2.4%
	Yellowtail	0.5 ± 0.8%	27.2 ± 30.6%	10.4 ± 4.1%	20.0 ± 22.4%	1.1 ± 1.1%	52.9 ± 6.0%
	Other	9.7 ± 6.3%	14.4 ± 19.8%	10.5 ± 9.3%	10.3 ± 6.5%	2.6 ± 4.3%	11.5 ± 8.5%

During 2006-08, the sparid group was the second largest contributor to the recreational landings (25.3 ±3%). This was significantly higher than in the previous two periods and also higher than in the commercial catch composition for the same period (6.7 ±3%). Similarly, the contribution by the non-sparid group to the recreational landings was more than double that of the commercial sector during 2006-08.

Table 4.5 presents the contribution of a number of species in the sparid and non-sparid groups to the commercial and recreational landings. There were significant shifts in the relative importance of all the species in each of the historical sampling periods as well as between the commercial and recreational landings. As was evident in chapter 3, the two most notable changes were the decreasing proportion of dageraad and the increasing importance of santer in the commercial and recreational landings.

In 1985-87 red steenbras dominated the commercial sparid group landings by weight (44.4%). However, since then it has made an insignificant contribution to total landings. The second most important species in 1985-87 was dageraad (21.7%), while santer made an insignificant contribution (3.3%). The proportion of the species in the recreational sparid group landings during 1985-87 was more evenly distributed than in the commercial landings. Dageraad was the largest contributor (28.8%).

During 1996-98 dageraad was the main species in the commercial sparid group landings (44.8%), while red stumpnose and santer each contributed around 20% (21.1% and 19.0%, respectively). The three most dominant species in the recreational sparid group landings at that time were santer (32.9%), dageraad (27.3%), and Roman (22.4%).

In 2006-08, santer was the biggest single contributor to the commercial sparid group landings (36.6 %). The 'other species' group contributed 37.7% (this was largely due to the contribution by blue hottentot). Dageraad, which made significant contributions to the commercial sparid group landings in 1985-87 and 1996-98, only made up 1.5% of the group in 2006-08. Santer contributed 46.4% of the recreational sparid group landings in 2006-08 which equated to roughly 10% of the total recreational landings during that period.

During 1985-87, the commercial non-sparid group landings were dominated by shallow-water hake (59.1%), followed by two *Galeichthyes* species (*G. feliceps* and *G. ater*) which collectively contributed 31% of the commercial non-sparid group landings. These two sea-barbel species were also the largest contributor to the recreational non-sparid group in 1985-87 (31.9%).

During 1996-98 commercial and recreational non-sparid group landings were also dominated by hake (more than 60% for both sectors). During this period the contribution by the *Galeichthyes* species became insignificant.

In 2006-08 hake contributed a significantly lower proportion (roughly 25% for both sectors). During this most recent period the commercial non-sparid group landings were dominated by elf (65.8%), while the recreational non-sparid group landings were dominated by yellowtail (53%). In all the time periods yellowtail featured more prominently in the recreational catches than in the commercial landings.

Cpue

The catch rates for the commercial and recreational sectors in 1985-87, 1996-98 and 2006-08 were standardised with respect to the month and area fished ($Wald\ X^2(2) = 25.3$, $p < 0.001$) (Fig. 4.4). There was a significant difference in the *cpue* of commercial and recreational fishers, as well as between the three sampling periods (*K-W Anova:* $H = 60.479$; $df = 2$, $p = <0.001$).

The mean *cpue* for the commercial operators between 1985 and 1987 was 3.57 $kg.fisher^{-1}.hour^{-1}$. This was significantly higher than in 1996-98 (3.07 $kg.fisher^{-1}.hour^{-1}$; $Q = 22.792$, $p < 0.05$) and 2006-08 (3.24 $kg.fisher^{-1}.hour^{-1}$; $Q = 11.066$, $p <0.05$). There was no significant difference between the commercial *cpue* during 1996-98 and that during 2006-08 ($Q = 2.048$, $p = 0.653$).

The mean recreational *cpue* of 1.03 $kg.fisher^{-1}.hour^{-1}$ in 2006-08 was only 32% of the commercial *cpue* for the same period. The mean recreational *cpue* was also significantly lower than in 1985-87 (2.78 $kg.fisher^{-1}.hour^{-1}$; $Q = 15.012$, $p <0.001$) and in 1996-98 (2.50 $kg.fisher^{-1}.hour^{-1}$; $Q =13.582$, $p <0.05$).

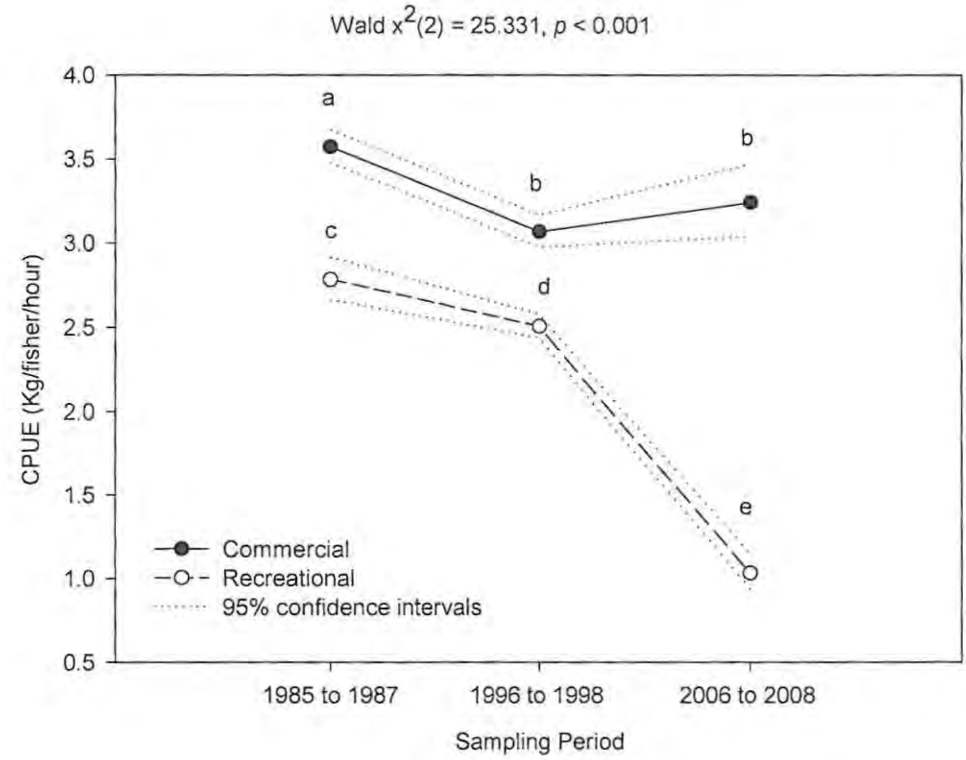


Fig. 4.4 *Cpue* of the commercial and recreational sectors in the Port Alfred linefishery during the periods 1985 to 1987, 1996 to 1998 and 2006 to 2008. Superscripts indicate significant difference at a *p* level of 0.05.

Total landings

During 1985-87 the annual total linefish landings for the Port Alfred linefishery fluctuated around 389.6 ± 66.9 tonnes (Fig. 4.5a). With increasing effort during the late 1980's and early 1990's (Chapter 3 Fig. 3.3), the commercial landings almost doubled by 1996-98 to 640.5 ± 413.0 tonnes per annum. Total annual commercial landings during the 2006-08 period were understandably lower due to the significantly reduced commercial effort (Fig. 4.5b), and varied between 22.6 (*95% CI*: 13.0 - 33.6) tonnes in 2006 and 60.7 (*95% CI*: 24.5 - 103.1) tonnes in 2008 (Fig. 4.5b). The mean annual total catch of the commercial fishers for the three year period was 44.7 ± 19.8 tonnes (a mere 7% of the annual landings during 1996-98). Total commercial landings as estimated from the NMLS for 2006 and 2007 lie within the 95% confidence intervals of the total catch calculated using effort and *cpue* data collected by the author.

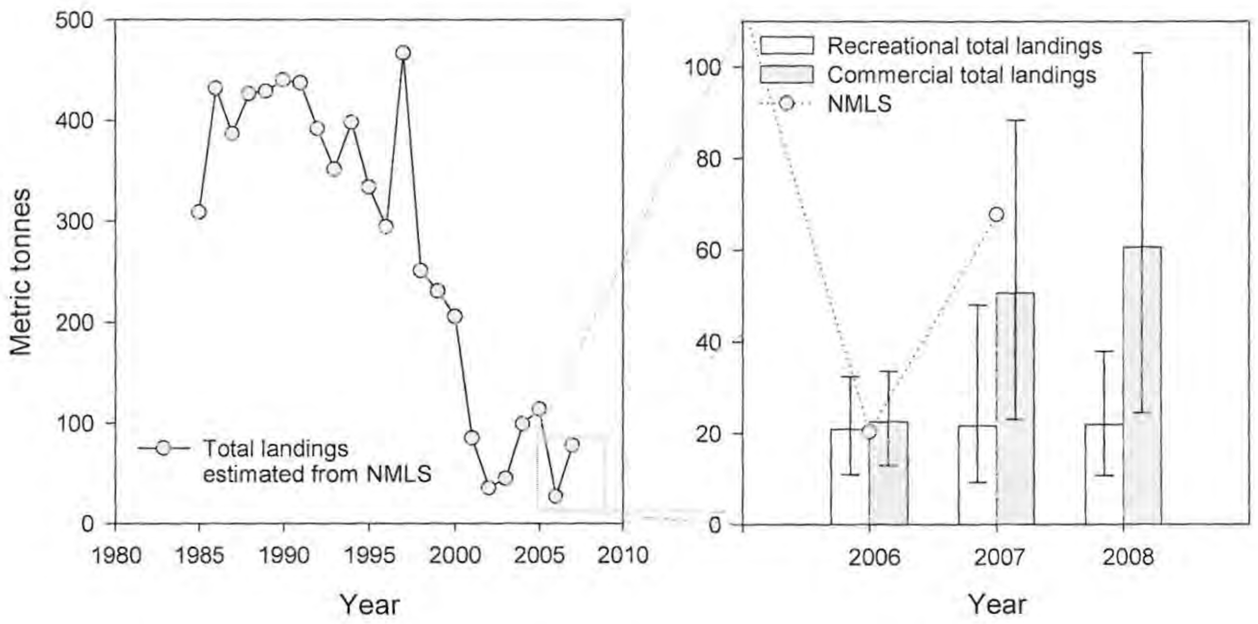


Fig. 4.5a) Total commercial landings of Port Alfred estimated from the NMLS (1985-2007) **b)** Commercial and recreational total landings calculated for the Port Alfred linefishery from 2006 to 2008. Error bars denote 95% confidence intervals.

The total recreational landings for the three year period were lower, yet more consistent than the commercial landings; 20.9 (*95% CI*: 11.0 - 32.5) tonnes in 2006, 21.7 (*95% CI*: 9.3 - 48.1) tonnes in 2007, and 22.0 (*95% CI*: 10.8 - 38.0) tonnes in 2008 (Fig. 4.5b). In 2006 the landings of the two sectors were approximately equal while in 2007 and 2008 the recreational landings were 59.2% and 63.8% lower than the commercial catch, respectively. These significant differences were ascribed to the high abundance of geelbek in 2007/8 and the recreational bag limit on the species.

DISCUSSION

There were 128 recreational vessels in the Port Alfred fishery between 1988 and 1989 (Hecht 1993) and in 2006-08 there were 153. Therefore, the recreational sector grew at a compounded 0.86% per annum between 1989 and 2008. However, by considering only those recreational boats that were highly active (16 in 1998/9 and 26 in 2007/8) implies that the active recreational fishery had grown at an annualised rate of 1.5% per annum. By contrast, commercial fishing effort was reduced to less than 20% of the peak in 1991, due to changes in the licensing structure since 2002 (Chapter 3).

In 1995 access point surveys revealed that the Eastern Cape coast had a ratio of commercial to recreational vessels of 1:1.3 (Sauer *et al.* 1997). Similarly, the ratio of commercial to recreational vessels for the Transkei linefishery was 1:2 (Fennessy 2003). These were higher than the rest of the Cape coast, but notably lower than the KZN region where the number of recreational vessels heavily outweighed commercial vessels (Penney *et al.* 1999). In Port Alfred, the ratio of registered commercial to recreational vessels was 1:2.4 in 1989 (Hecht 1993) and increased to 1:13 during 2006-08. In light of the 80% reduction in commercial effort (Chapter 3, Fig. 3.2), the ratio in 2006-08 is understandably higher than that observed by Hecht (1993). If the ratio calculated for the fishery in 1989 is amended to account for the reduction in commercial effort the resulting ratio is 1:12 (commercial to recreational vessels) which is similar to that calculated in this study. In other words, the switch from commercial to recreational effort was a result of the reduction in commercial effort in 2002 and not as a consequence of substantial increases in the number of new recreational vessels.

Furthermore, in 1989 the ratio of regularly active commercial vessels to active recreational vessels was 1.75:1 (Hecht 1993), in other words the active fishery was dominated by commercial boats. During 2006-08, the fishery was dominated by recreational vessels at a ratio of 1:6.5 (regularly active commercial to recreational vessels). As hypothesized, it stands to reason that many of the previously commercial licence holders remained in the Port Alfred fishery as recreational fishers (in particular the former B-licence holders). It was mainly these vessels that accounted for the 1.5% annual increase in the active recreational sector from 1989 to 2008. Possibly as a consequence of being former commercial licence holders, a high proportion of the recreational vessels (an estimated 35%) illegally sell their catch to a number of informal buyers as a means of 'subsidising their sport'.

Fishing effort in the Port Alfred linefishery was strongly influenced by prevailing weather patterns and sea conditions, in particular wind speed and swell size (Fig. 4.2). As was noted by Hecht (1993), the exposed nature of the Kowie river mouth (and other launch sites) have a limiting effect on the total effort in the fishery. These weather factors were largely unpredictable and not easily accounted for when using frame-surveys to estimate effort. Although not without its problems, CCTV monitoring was an effective way of measuring the nominal effort in the fishery. However, the one major problem was the necessary use of '*ad-hoc*' methods to estimate fishing effort at the other access points. During 2007, almost 60% of the commercial fishing effort in the Port Alfred fishery occurred out of Boknes and Kenton-on-Sea. Despite this, the study clearly shows the potential value of closed circuit TV cameras to remotely monitor fishing effort.

Between 1984 and 1987, the commercial fleet went to sea between 10 and 18 days per month (Hecht and Tilney 1989). Given the size of the commercial fishery at that stage (28 vessels), it can be assumed that the commercial operators went to sea on every possible sea day. In contrast, between March 2007 and February 2008 the two commercial vessels operating out of Port Alfred only went to sea on five to seven days per month. However, during the same period the recreational fishers operated between seven and 16 days per month. In other words, the two commercial operators fishing from Port Alfred did not capitalise on the maximum number sea days each month. Amongst other things, this was attributed to increasing operating costs, in particular fuel which accounted for 50.7% of the costs of running a commercial vessel (including depreciation)[10] and the reluctance to go to sea in the absence of information on fish 'availability'.

The average duration of a commercial fishing trip has not changed since 1988. Hecht (1993) estimated that the commercial vessels spent an average of 7.5 hours at sea per fishing day. Between 1994 and 1996, commercial operators between Still Bay and Kei Mouth fished for an average of 8.3 ± 2.3 hours each sea day (Brouwer and Buxton 2002). During the period March 2007 to February 2008, the average trip duration of the commercial operators in Port Alfred was 8.3 hours. This was marginally higher than that observed by Hecht (1993) but identical to that observed by Brouwer and Buxton (2002).

On the other hand, from March 2007 to February 2008 the average duration of a recreational fishing trip in Port Alfred was 4.9 hours which was about 50 minutes less than that estimated by Hecht (1993) in 1988 (5.7 hours). It was also substantially lower than the average duration of a recreational fishing trip observed between Still Bay and Kei Mouth between 1994 and 1996 (7.2 ±2.5 hours; Brouwer and Buxton 2002). There are a number of possible reasons for this decrease.

[10] This data is extracted from a concurrent study on the socio-economics of the fishery that was undertaken by the author and Prof. T. Hecht.

Firstly, due to increasing fuel prices, the recreational fishers are fishing closer to the access point hence reducing travelling time. Secondly, using improved navigational equipment such as GPS and echo-sounders would also reduce the time spent travelling and searching for fish. Thirdly, recreational fishers were limited by the increased catch restrictions and returned to port sooner as their bag limits (or bag limits of a particular species) were met. Evidence for the latter suggestion was observed on numerous occasions between 2006 and 2008 during times when geelbek were abundant.

In comparison to the historical data presented by Hecht and Tilney (1989) the commercial fishery is shrinking in geographic terms. Rapidly rising operating costs and a slow increase in the market value of fish have forced the commercial operators in Port Alfred/Kenton-on-Sea/Boknes to operate closer to the access points (G. Dickinson, commercial licence holder, pers. comm. 2008). Previously, the commercial vessels would travel up to four times further than the recreational vessels on an average fishing trip (Hecht 1993). This was qualified by Griffiths (2000) who suggested that during the period 1986-1998, technological advances resulted in the shift of relative importance of offshore species, e.g. larger carpenter. However, during 2006-2008 the commercial and recreational vessels operated in similar areas. While a few recreational vessels (3%) frequented Bird Island (in the extreme west of the fishery) to target game-fish, the majority of the recreational and commercial vessels operated in close proximity to the access point and targeted bottom fish.

The Port Alfred linefishery was, and still is, dominated by a few species (Hecht and Tilney 1989). Of the 75 species recorded in the fishery only four made considerable contributions to the commercial and recreational landings in 2006-08 (Fig. 4.3). These were silver kob, carpenter, geelbek and santer. Panga, which made significant contributions to the commercial landings in 1985-87 and 1996-98, contributed very little to the total commercial landings between 2006 and 2008 (Fig. 4.3). The main reason for this also seems to be related to operating costs and the low value of the species (G. Dickinson, commercial licence holder, pers. comm. 2008).

Griffiths (1997a) illustrated the importance of silver kob to the commercial linefishery between Cape Point and East London. Furthermore, Griffiths and Heemstra (1995) suggested that recreational kob catches could be as high as the commercial landings on the eastern seaboard of South Africa. In fact, during 1988 and 1989, recreational silver kob landings in Port Alfred were estimated to be half that of the commercial sector (Hecht 1993). At that stage the recreational anglers were limited to 10 silver kob per fisher.day^{-1}. Despite the reduction of the daily bag limit to 5 silver kob per recreational angler in 2002, silver kob remained the most important species in the recreational sector contributing the largest proportion of the total landings during 2006-08 (28.9%). Simple calculations from the estimated total landings and proportion of silver kob in the landings suggest that despite the changes to bag limits the recreational landings of silver kob (6.2 tonnes.annum^{-1}) were still roughly half that of the commercial silver kob catch (11.0 tonnes.annum^{-1}). Considering that the effective recreational effort has increased (Table 4.2) and the commercial effort has decreased (Chapter 3, Fig. 3.2), suggests either that the bag limit is effectively limiting the catch or that the stocks have declined. Although it is possibly a combination of both, the silver kob directed *cpue* (Chapter 3), provides evidence for the latter. Therefore, despite the suite of management measures that were implemented, both the commercial and recreational sectors continue to put pressure on silver kob in the Port Alfred linefishery and no doubt throughout the distribution of this silver kob subpopulation (sensu Griffiths 1997b).

In 2006-08 strong pulses of migratory geelbek were targeted by commercial operators and during this period geelbek was the most important contributor (32.7%) to the total landings. Similarly, recreational sector also targeted geelbek during 2006-08 (20.2%). Although the proportion of geelbek in the recreational total landings had increased substantially since 1985-87 (6.2%) and 1996-98 (15.6%), the landings in 2006-08 were ameliorated largely by the bag limit of two geelbek per fisher.day^{-1} (Fig. 4.3).

Between 1985 and 2007 there was a shift in the relative importance of the 'sparid species' group in both the commercial and recreational catch compositions (Fig. 4.3). It would appear that this may be due to two factors, viz. the imposition of bag limits and the increasing cost of fishing (in particular fuel costs). In the Port Alfred fishery, larger carpenter are generally caught in deeper water, further off-shore (Hecht and Tilney 1989, Brouwer and Buxton 2002). During 1985-1987, carpenter contributed substantial proportions to the landings in both the commercial and the recreational sectors (34.4 and 28.5%, respectively). In 1992, the recreational bag limit for carpenter was reduced to 10 fish per angler.day^{-1} and in 2006 was reduced further to four fish per angler.day^{-1}. Consequently, the proportion of carpenter in the total recreational landings decreased after each amendment to the bag limit. Although, carpenter continued to contribute a large proportion (23.9%) to the commercial landings in 2006-08, the recreational fishers were less likely to expend the effort and cost of travelling further distances to target carpenter if they were limited by a daily bag limit of four fish. Carpenter only contributed 3.1% of the recreational landings in 2006-08.

Owing to its small size panga has never been considered a recreational species (Hecht 1993, Booth and Buxton 1997) and during the three periods, panga did not make any significant contributions to the recreational landings. As was also evident in the previous chapter, the proportion of panga to the total commercial landings in 2006-08 decreased significantly (Fig. 4.3). This may be attributed to its low market value of species, small size, as well as the fact that they are more abundant in deeper water, further offshore (Hecht and Tilney 1989). Similarly, hake (which is also caught predominantly in deeper water) has decreased proportionately in both the commercial and recreational landings (Table 4.5).

As a result of increased operating costs, as well as the imposition of stricter bag limits for some species (e.g. carpenter), the recreational fishers targeted a substantially higher proportion of shallow water, reef-associated sparids than in the past (Fig. 4.3, Table 4.5). These species were generally caught in close proximity (< 5km) to the access points. In 2006-08 this shift in effort to shallower water, combined with the reduced bag limits for many species (most notably one dageraad and two Roman per angler.day^{-1}), has resulted in a greater proportion of the recreational landings being attributed to santer which has a bag limit of five per day (Table 4.5). In fact, santer contributed a significant 10% of the *total* recreational landings in 2006-08.

A declining *cpue* in commercial linefisheries in South Africa has been well documented prior to 2002 (Attwood and Farquhar 1999, Penney *et al.* 1999, Griffiths 2000, Brouwer and Buxton 2002). However, subsequent to the 80% reduction in commercial effort since 2002, the *cpue* of the remaining commercial operators in Port Alfred during 2006-08 was not significantly different to the *cpue* during 1996-98 (Fig. 4.4). However, as was concluded in Chapter 3, the high commercial *cpue* during 2006-2008 was due to the abundance of geelbek and not necessarily indicative of an improvement in the fishery.

What is of interest is the *cpue* ratio between the commercial and recreational sectors. In 1988, Hecht (1993) found that the commercial *cpue* (22.6 $kg.fisher^{-1}.day^{-1}$) was 2.6 times higher than the recreational *cpue* (8.7 $kg.fisher^{-1}.day^{-1}$). Similarly, Brouwer and Buxton (2002) found that the *cpue* of commercial operators in the Eastern Cape between 1994 and 1996 was 2.3 times higher than that of their recreational counterparts (21.5 ±35.4 $kg.fisher^{-1}.day^{-1}$ and 9.4 ±14.7 $kg.fisher^{-1}.day^{-1}$ respectively). However, in 2006-08 the commercial *cpue* (26.5$kg.fisher^{-1}.day^{-1}$)[11] in Port Alfred was 5.4 times greater than that of the recreational sector (4.9 $kg.fisher^{-1}.day^{-1}$)[12]. Considering that there was no significant difference between the commercial *cpue* in 1996-98 and 2006-08, the apparent crash of the recreational *cpue* in 2006-08 was simply due to compliance with the new bag limits in 2006 and in particular, the bag limit of two geelbek per recreational $fisher.day^{-1}$.

This suggestion is supported by the similar landings of the commercial and recreational sectors in 2006. During that year geelbek was less abundant in the area relative to 2007 and 2008 (Fig. 4.5b). The imposition of stricter bag limits has thus resulted in the relatively constant total catch of the recreational fishers despite the abundance of geelbek in the fishery. From this it can be concluded that management measures have been effective in limiting the *cpue* and total landings in the recreational sector. This, together with the increasing importance of the recreational sector, highlights the necessity for proper monitoring, control and enforcement of the regulations in both sectors the Port Alfred linefishery.

[11] Calculated as 3.2 $kg.fisher^{-1}.hour^{-1}$ (Fig. 4.4) at an average of 8.3 hours fishing per day (Table 4.2).
[12] Calculated as 1.01 $kg.fisher^{-1}.hour^{-1}$ (Fig. 4.4) at an average of 4.9 hours fishing per day (Table 4.2).

CHAPTER 5

LENGTH- AND COMMUNITY-BASED ANALYSES OF THE PORT ALFRED LINEFISHERY

INTRODUCTION

With increasing recognition of the importance and value of an ecosystem based fisheries management (EBFM) worldwide, there is a need for management decisions to be based on fishery indicators that are focused on an understanding of the broader effects of fishing at the community level (Gislaison and Rice 1998, FAO 2003, Butterworth and Plaganyi 2004, Garcia and Cochrane 2005, Branch and Clark 2006). Commonly used fishery indices (such as per-recruit analyses and species-directed *cpue*) focus on a target species and pay little attention to the fishing effects on the entire fish assemblage or the surrounding environment (Gislaison and Rice 1998, Daan et al. 2005, FAO 2003). It is generally agreed that combining a selection of multiple community-based metrics and traditional fisheries indices in a single study is necessary to obtain a more complete picture of how communities are changing (Rice 2000, Shin *et al.* 2005, Branch and Clarke 2006).

Diversity indices, ordination methods, size spectra, dominance curves and ecosystem mass-balance models have all been used to assess the effects of fishing pressure on many fish assemblages and their associated ecosystems (Greenstreet and Hall 1996, Rice and Gislaison 1996, Gislaison and Rice 1998, Bianchi *et al.* 2000). However, the large majority of these metrics have only been empirically and theoretically applied to demersal trawl fisheries and long-line fisheries (*inter alia* Duplisea *et al.* 1997, Jennings *et al.* 1999, Bianchi *et al.* 2001, Blanchard *et al.* 2004, Jouffre and Inejih 2005, Blanchard *et al.* 2005, Piet and Jennings 2005, Shin and Cury 2004, Yemane *et al.* 2005, Greenstreet and Rogers 2006). However, some have fairly recently been applied to in-shore linefisheries using fisheries dependent data (Yemane *et al.* 2004).

The selection of which community-based metric to use involves an understanding of the sensitivity of the ecosystem properties that they set out to describe, as well as which particular properties are relevant to the study (Rice 2000, Hauge *et al.* 2005, Jennings 2005). In addition to this, there must be an objective way to determine whether the differences found in the values of the selected metrics are meaningful (Rice 2000, Rochet and Trenkel 2003, Jennings 2005, Link 2005, Shin *et al.* 2005, Greenstreet and Rogers 2006). The following paragraphs summarise the selection process of to determine which of the community metrics would be most appropriate to evaluate the effects of fishing on the Port Alfred fish assemblage. For the purposes of this study the selection process was conducted on the assumption that changes in the management environment had negligible effects on the fish assemblage.

Rice (2000) classified and evaluated the sensitivity of some of the more commonly used metrics of community structure. He investigated ordination methods, emergent property metrics, diversity indices and aggregate indicators (which include size-spectra and dominance curves). Ordination methods (e.g. Principle Component Analysis and Correspondence Analysis) are essentially complex matrices reduced to a smaller number of orthogonal gradients (Field *et al.* 1982). Ordination methods can be misleading if there are several correlated factors that influence the species distributions (Rice 2000). Furthermore, although these methods are sensitive to changes in community structure, they fail to indicate the direction of change (either positive or negative; Yemane *et al.* 2005). More importantly, ordination methods fail to identify trends when most species are not present in the full range of sampled sites (for principle component analysis), when the dataset includes generalist and specialist species (for correspondence analysis), or species in the dataset are subject to substantial sampling errors (Rice 2000). Fisheries dependent sampling of the multi-species, multi-user Port Alfred linefishery fails to meet these data requirements and thus ordination methods have not been considered further.

Emergent property indices are not based on direct representations of occurrences or abundance. Instead, these metrics are based on hypotheses pertaining to the trophic interactions between species and species groups within a community (Pauly *et al.* 1998, Pauly *et al.* 2000, Gascuel 2005, Gascuel *et al.* 2005). A requirement of these models is data that are representative of all trophic levels (Jarre-Teichman 1998). Such data are often not available for many marine systems and more development needs to be done in order to assess the impacts of linefisheries using these metrics (Rice 2000).

Univariate diversity metrics (such as Shannon-Weiner diversity index, Pielous' evenness and Margalefs index) have been extensively used as indices in community ecology and incorporate the number of species (richness), abundances of each species (evenness) and the relative importance of species (dominance) in a single community. However, these indices have received much criticism due to the fact that there is no reason to expect a relationship between community stability, or environmental well-being and the diversity index (Green 1979, Keough and Quinn 1991, Greenstreet and Rogers 2006). Consequently, it has been suggested that these metrics have no biological meaning (Washington 1984, Keough and Quinn 1991). In communities where species are not competitively equal, disturbances such as fishing pressure could actually increase diversity (Green 1979). The relationship between the indicator and the anthropogenic impact needs to be well understood in order to show causality between predictor and dependent variables (Vos *et al.* 2000, Rochet and Trenkel 2003). Furthermore, information about the community is lost when reducing the information about that community to a single summary statistic (Green 1979, Rice 2000, Yemane *et al.* 2005). For these reasons diversity indices have not been used as indicators of fishing pressure in this study.

Amongst the various community metrics, aggregate indicators (such as size spectra and dominance curves) have received a great deal of attention from fisheries scientists (Greenstreet and Hall 1996, Rice and Gislaison 1996, Gislaison and Rice 1998, Bianchi *et al.* 2000, Yemane *et al.* 2004, Yemane *et al.* 2005). The basic premise behind these metrics is that fishing is always size selective and larger more valuable fish are generally targeted through spatio-temporal fishing strategies and fishing gear (e.g. hook size in a linefishery) (Beverton and Holt 1957, Shin *et al.* 2005, Goetz *et al.* 2007). The effects of fishing on fish assemblages are cumulative and over time larger fish become less abundant. Moreover, it is well known that the targeting of larger fish modifies the size structure of fish assemblages both directly and indirectly (Beverton and Holt 1957, Buxton and Smale 1989, Gislaison and Rice 1998, Daan *et al.* 2005, Dulvy *et al.* 2004, Jennings *et al.* 2005, Shin *et al.* 2005). It has therefore been suggested that size-based indicators (such as size-spectra analyses and dominance curves) can be used to provide a description of the responses of communities to exploitation in that they consider a range of fishery impacts on the entire fish assemblage (Bianchi *et al.* 2000, Yemane *et al.* 2004, Shin *et al.* 2005, Yemane *et al.* 2005).

Rice and Gislaison (1996) suggested that it would be possible to estimate the relative development of fishing intensity from the overall change in the size compositions of the fish fauna with time for a particular area. To quantify these changes in size compositions, size spectra can be created by plotting the logarithm of the overall abundance per size class against the logarithm of the median of the size class (Rice and Gislaison 1996, Bianchi *et al.* 2000). The slope and the intercept of the size spectrum would be directly proportional to the changes in overall fishing mortality if the fishery characteristics remain constant over time (i.e. gear type, species and size directivity) (Gislaison and Rice 1998, Bianchi *et al.* 2000). The slopes of the size spectra were found to be dependent on the fishing pressure (Rice and Gislaison 1996, Shin and Cury 2004), while the intercepts are related to the overall productivity of the system (Gislaison and Rice 1998, Bianchi *et al.* 2000, Bianchi *et al.* 2001).

Dominance curves present the cumulative abundances of species in a community ranked by their abundances (Lambshead *et al.* 1983). K-dominance curves, whereby the cumulative dominance in biomass is plotted against the log of the species rank, are useful in picking up patterns of relative species abundance without having to reduce the information to a summary statistic such as a diversity index (Rice 2000, Yemane *et al.* 2005). The basic, and proven, hypothesis behind the use of dominance curves to assess the effect of fishing pressure is that shifts in environmental stresses cause a subset of species to tolerate the perturbations better than others and thrive, while intolerant species become rarer (Clarke 1990, Greenstreet and Hall 1996). K-dominance curves of more perturbed communities lie above and to the left of the curves of unperturbed communities (Clarke 1990, Rice 2000).

As an extension to K-dominance curves, Abundance Biomass Comparison (ABC) curves were first proposed by Warwick (1986) as an indicator of disturbance to benthic invertebrate communities. ABC curves draw from the fundamentals of evolutionary r- and k-selection and have only fairly recently been used as a measure of fishing pressure (Bianchi *et al.* 2001, Blanchard *et al.* 2005, Yemane *et al.* 2005). Undisturbed communities tend to be dominated by the slow-growing, late maturing and larger K-selected species. When comparing the plots of dominance in terms of abundance and biomass in an undisturbed community the biomass curve tends to lie above the abundance curve. With increased stress on a community the k-selected species become scarcer and the more tolerant R-selected (fast-growing, small and opportunistic) species become dominant and the abundance curve lies above the biomass curve (Clarke and Warwick 1994, Yemane *et al.* 2005). The W-statistic, which provides a quantifiable indication of the relationship between abundance and biomass, is represented by the area between the two curves. A negative sign indicates the biomass curve lies below the abundance curve and this suggests that the target community has been disturbed (Warwick and Clarke 1991). Theoretical ABC curves indicating undisturbed, moderately disturbed and heavily disturbed communities are presented in Fig. 5.1.

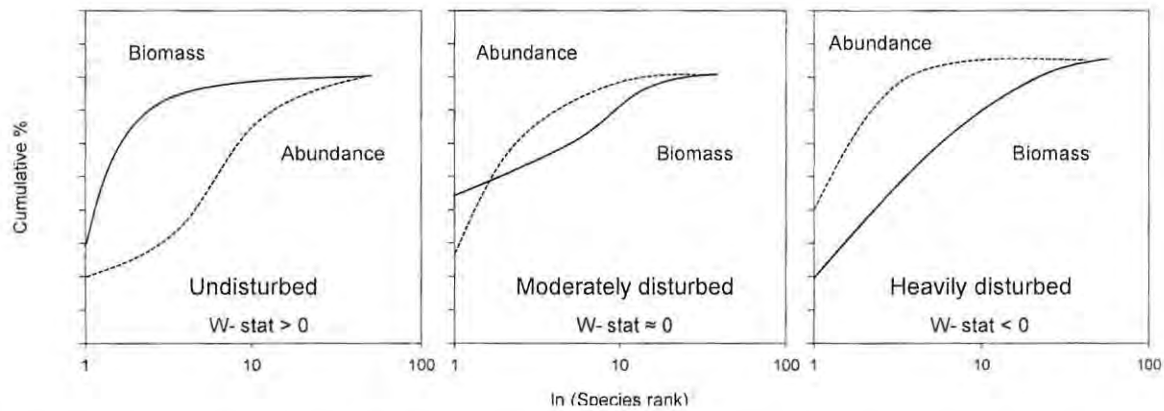


Fig. 5.1 Theoretical ABC curves for undisturbed, moderately disturbed and heavily disturbed communities (Modified from Clarke and Warwick, 1994).

An important element of ABC curves is that the biomass is compared with the abundance at the same time and place, upon which the community status can be evaluated without the need for a spatial or temporal control (Clarke and Warwick 1994, Yemane et al 2005). This makes ABC curves a useful metric for 'snapshots' of the fishery at different time periods when a continuous dataset is not available, such as in the Port Alfred linefishery.

In this chapter an attempt was made to relate the changes in size-frequency distributions, mean sizes, size spectra and dominance curves (including ABC curves) to the transformations that have occurred in the South African linefish management environment, as well as to assess the viability of using community-based metrics to assess multi-user, multi-species linefisheries using long-term datasets in light of management changes (Chapter 1, Tables 1.2 and 1.3).

MATERIALS AND METHODS

The species length data collected from the Port Alfred fishery for two historical time periods (1985-1987 and 1996-1998) and the current period (2006-2008) were used to construct size-frequency distributions of individual species, size spectra, K-dominance curves (both abundance and biomass) and ABC curves. These metrics were used to assess the temporal changes to species sizes and the fish assemblage over a 23 year period (1985 to 2007). Table 5.1 provides the numbers of fish lengths recorded in the each sample period. Furthermore, as some of the community metrics used have a theoretical basis in r- and k- selection, the life-histories of the predominant species are also summarised in Table 5.1 for reference purposes. These species are ranked by their perceived order on the r- and k- selectivity continuum in relation to each other.

Size-frequency distributions and trends in mean length

Using the species length data available from the Port Alfred fishery, size-frequency distributions of the more dominant species (silver kob, geelbek, carpenter and panga) were constructed for the equal time periods from 1985-1987, 1996-1998 and 2006-2008. The size-frequencies from the commercial and recreational sectors during the study period (2006-08) were compared with each other as well as with the lengths observed in the 1985-87 and 1996-98 datasets by means of Kolmogorov-Smirnov Two-sample tests (this not only accounts for differences in mean size, but also the differences in the shape of the size-frequency distribution).

The long-term trends in the mean sizes of these species were investigated for the years from 1985 to 2008 (except for 1989, 1990 and 2000-2005 due to the lack of data for those periods). Where necessary, differences in the mean lengths between the time periods were tested with a Kruskal-Wallace ANOVA on ranks followed by Dunn's multiple comparison procedure as a post hoc test in order to isolate significantly different cases.

Table 5.1 Numbers of length measurements of the important species in the Port Alfred linefishery during 1985-87, 1996-98 and 2006-08; and the summarised life-history parameters of each species as well as their perceived ranking on the r- and k- continuum in relation to each other.

	Species	Dataset (n)			Summarised life-history parameters						References
		1985-1987	1996-1998	2006-2008	50% maturity	Max. length	Max. weight (Kg)	Age at 50% maturity	Max. age	Reproductive style	
R*	**Silver kob** *(Argyrosomus inodorus)*	1683	6614	1472	290♂ 310♀	1450 mmTL	36.3	2.4♂ 1.3♀	25	Gonochorist	Griffiths 1996, 1997a, 1997b; Griffiths and Heemstra 1995
	Geelbek *(Atractoscion aequidens)*	2554	382	974	900	1300 mmTL	25	5	9	Gonochorist	Griffith and Hecht 1995; Smith and Heemstra 1986; Hutton 2001
	Carpenter *(Argyrozona argyrozona)*	2067	8454	1946	275♂ 262♀	608 mmFL	3.62	-	27	Gonochorist	Brouwer and Griffiths 2004
	Santer *(Cheimerius Nufar)*	130	629	1276	340	750 mmTL	7**	3 - 4	22	Gonochorist	Coetzee and Baird 1981; Garratt 1985
	Panga *(Pterogymnus laniarius)*	1601	7220	1725	204	405 mmFL	1**	4.3	16	Late gonochorist	Booth and Buxton 1997
	Red steenbras (*Petrus rupestris*)	60	79	15	575	130 mmFL	52.2**	7.2	33	Late gonochorist	Smale and Punt 1991
	Red stumpnose *(Chrysoblephus gibbiceps)*	38	17	110	unknown	750 mmTL	8.2**	unknown	unknown	Late gonochorist	Smith and Heemstra 1986
	Blue hottentot *(Pachymetopon aeneum)*	76	23	230	200- 250♀	345 mmFL	-	4.5 - 6	12	Protogynous hermaphrodite	Buxton and Clarke 1996
	Roman *(Chrysoblephus laticeps)*	118	593	432	172♀	512 mmFL	4.2**	2.5♀	17	Protogynous hermaphrodite	Buxton 1993
K*	**Dageraad** *(Chrysoblephus cristiceps)*	201	1354	92	375♀	700 mmTL	8.7**	9.5♀	23	Protogynous hermaphrodite	Buxton 1993

* It is recognised that the ranking of species on the r- and k- continuum is problematic and very subjective. The ranking was determined as a collective exercise by a group of 7 scientists: viz. Prof. T. Hecht, Prof. T. Booth, Dr O. Weyl, Dr W. Potts, Mr T. Richardson, Mr B. Donovan and Ms. M. Kruger

** Estimated by angling records (van der Elst 1988)

Size spectra analyses

It is not possible to assume that the fish length data collected during the three separate time periods (1985-87, 1996-98 and 2006-08) were collected at the same frequency. Therefore, the total weight (W_{ij}) for species *i* in year *j* was extracted from the NMLS. The proportion of the sampled weight (w_{ij})[13] to the total weight was used to calculate the overall abundance of each species in each 5cm size class (from 25cm to 100cm) during each year (N_{ijs}):

$$N_{ijs} = \frac{n_{ijs.}W_{ij}}{w_{ij}} \qquad 5.1$$

where, $n_{ijs.}$ were the numbers of species *i* in size class *s*, sampled in year *j*.

Size spectra were constructed by plotting the natural logarithm of the abundance of fish (N_{ijs}) in each size class against the natural logarithm of the corresponding class-mark (Bianchi 2000, Shin and Cury 2004, Yemane 2004). The height of the linear function representing the size-spectra, which has been suggested to reflect overall productivity of the fishery, is highly correlated with the slope (Gislaison and Rice 1998, Bianchi *et al.* 2001). To obtain meaningful trends in the heights the correlation needs to be removed. This was done by estimating the height at the mean value of the independent variable and fitting the linear regression to the plot of the natural logarithm of abundance against the deviation from the mean size class (Daan 2003, Rochet and Trenkel 2003). The slopes and heights of the size spectra were plotted for each year between 1985 and 2008 (except for 1989, 1990 and 2000-2005 due to data constraints).

Due to the nature of a multi-species fishery, in particular the relatively high proportion of smaller species, Shin and Cury (2004) suggested fitting a quadratic function to the size spectra instead of a linear function in order not to overestimate the numbers of smaller fish. However, in order to simplify the size spectra for comparative purposes in this study (Shin *et al.* 2005), a linear function was plotted from the descending limb of the frequency distribution as per the method described by Bianchi *et al.* (2000).

[13] The weight of each individual fish was calculated from the species length-weight relationships estimated by Atkin and van der Elst (1991).

Furthermore, Yemane *et al.* (2004) found that the dominance of snoek (*Thyrsites atun*) in Western Cape linefish landings resulted in a shallower slope of the size spectra and masked the effects of other species. Snoek undergo both longshore and offshore migrations, they are short-lived, grow fast, mature early and relative to other linefish species they are less vulnerable to overexploitation (Attwood and Farquhar 1999, Yemane 2004). In recent years the Port Alfred fishery has been dominated by geelbek (Chapters 3 and 4). Apart from maturing later and hence being more vulnerable to exploitation, geelbek have similar life-history traits to snoek (Table 5.1). The dominance of the migratory geelbek in later years could potentially skew the size-spectra and, like snoek in the Western Cape, result in a misrepresentative picture of the actual fishery status. As such, size spectra for the Port Alfred fishery were constructed both with and without geelbek for comparability across the years.

Dominance and Abundance Biomass Comparison (ABC) curves

K-dominance curves, in terms of both abundance and biomass, were constructed using the same up-calculated data as was used for constructing the size spectra. The data was grouped into three year periods each roughly a decade apart: 1985-87, 1996-98 and 2006-08. During each period the species landed were ranked in terms of their abundance and biomass separately. The natural logarithm of the rank of the species was plotted against the cumulative percent of that species for both abundance and biomass (Clarke 1990). In addition, K-dominance curves (in terms of biomass) were constructed in the same way for the recreational sector in Port Alfred for the period 2006 to 2008.

ABC curves are nothing more than the abundance and biomass K-dominance curves plotted on the same axis. The W-statistic, which describes the amount of disturbance to the fish assemblage, represents the degree of separation between the abundance curve and the biomass curve (Fig. 5.1) such that:

$$W = \sum_{i=1}^{S}(B_i - A_i)/(50 \times (s-1)) \qquad 5.2$$

where, s is the number of species, B_i the biomass and A_i the abundance of species i.

In order to investigate the progressive change of the dominance curves, the W-statistic was plotted for each year from 1985 to 2008 (excluding the data limited years 1989, 1990 and 2000 to 2005). As with the size spectra, the W-statistic was calculated both with and without geelbek.

RESULTS

Size-frequency distributions

The summarised parameters and statistical differences in the size-frequency distributions of the nine species examined for the three periods are presented in Tables 5.2a and 5.2b. The length-frequency histograms for the more abundant species (viz. silver kob, geelbek, carpenter and panga) are presented in Fig. 5.2 through to Fig. 5.5. Over the 23 year period the minimum legal size was increased for seven of the nine species. These changes significantly altered the size structure of the landings in the fishery and were predictably evident in the size-frequency distributions. For all species examined there were significant differences in the size distributions during the three periods (1985-1987, 1996-1998 and 2006-2008).

The mean size of silver kob decreased from 548mmTL in 1985-87 to 508mmTL in 1996-98 (Table 5.2a). In 2006, the minimum legal size for silver kob was increased from 400mmTL to 500mmTL and this is reflected in an increase in mean size to 604mmTL in 2006-08 (Table 5.2a and Fig. 5.2). Despite the length at 50% recruitment (595mmTL) being almost 100mm larger than the minimum legal size, 16% of the silver kob landed in the Port Alfred fishery were undersize.

The mean size of geelbek increased significantly from 611mmFL in 1985-87 to 762mmFL in 1996-98, and to 819mmFL in 2006-08. In 1985-87 geelbek first recruited into the fishery at a size of 390mmFL (413mmTL), at which stage the legal size limit was 400mmTL. In 1992 the minimum size for geelbek was increased from 400mmTL to 600mmTL and in 1996-98 the size at first recruitment was 600mmFL (equivalent to 636mmTL). In 2006, although the mean size had increased to 819mmFL, first recruitment occurred at 465mmFL (493mmTL) which is significantly smaller than the minimum legal size. It should be noted that the minimum size is substantially lower than the length at 50% sexual maturity of 900mmFL (954mmTL; Griffiths and Hecht 1995).

Table 5.2a Length frequency analyses for silver kob, geelbek, carpenter and panga in the Port Alfred linefishery during the periods 1985 to 1987, 1996 to 1998 and 2006 to 2008.

Species	Period	Mean (mmFL)Ŧ		Kolmogorov-Smirnov two-sample test, *D* and *p-values*			n	1st recruitment (mmFL)Ŧ	50% recruitment (mmFL)Ŧ	50% maturity (mmFL)Ŧ	Minimum legal size (mmTL: mmFL)	Illegal landings (%)
				85-87	96-98	06-08						
Silver kob	1985 - 1987	548.40 ±171.4	a	x	D = 0.109	D = 0.365	1683	390	475	290 ♂, 310♀*	400 : -	7.5%
(*Argyrosomus inodorus*)	1996 - 1998	507.51 ±109.2	b	*p < 0.001*	x	D = 0.415	6614	400	530	290 ♂, 310♀*	400 : -	3.9%
	2006 - 2008	603.55 ±114.6	c	*p < 0.001*	*p < 0.001*	x	1472	440	595	290 ♂, 310♀*	500 : -	16.0%
Geelbek	1985 - 1987	610.78 ±230.5	a	x	D = 0.625	D = 0.487	2554	390	480	900**	400 : 378	2.2%
(*Atractoscion aequidens*)	1996 - 1998	762.17 ±97.8	b	*p < 0.001*	x	D = 0.338	382	600	760	900**	600 :567	0.0%
	2006 - 2008	818.9 ±176.0	c	*p < 0.001*	*p < 0.001*	x	974	465	855	900**	600 :567	12.5%
Carpenter	1985 - 1987	323.57 ±54.4	a	x	D = 0.091	D = 0.548	2067	240	320	275 ♂, 262♀***	250 :225	0.6%
(*Argyrozona argyrozona*)	1996 - 1998	328.42 ±40.7	b	*p < 0.001*	x	D = 0.526	8454	260	320	275 ♂, 262♀***	250 :225	0.0%
	2006 - 2008	366.14 ±39.1	c	*p < 0.001*	*p < 0.001*	x	1946	295	360	275 ♂, 262♀***	350 :320	4.1%
Panga	1985 - 1987	285.88 ±32.6	a	x	D = 0.180	<u>**D = 0.016**</u>	1601	220	280	204****	n/a :n/a	n/a :n/a
(*Pterogymnus laniarius*)	1996 - 1998	286.88 ±19.9	b	*p < 0.001*	x	D = 0.179	7220	260	280	204****	n/a :n/a	n/a :n/a
	2006 - 2008	285.14 ±32.4	a	<u>***p = 0.986***</u>	*p < 0.001*	x	1725	220	290	204****	n/a :n/a	n/a :n/a

Ŧ All length measurements are fork length (mmFL) except silver kob which are measured as total length (mmTL)

* South Eastern Cape (Griffiths 1997)

** South Eastern Cape (Griffiths and Hecht 1995)

*** South Eastern Cape (Brouwer and Griffiths 2004)

**** Combined sexes in the South Eastern Cape (Booth and Buxton 1997)

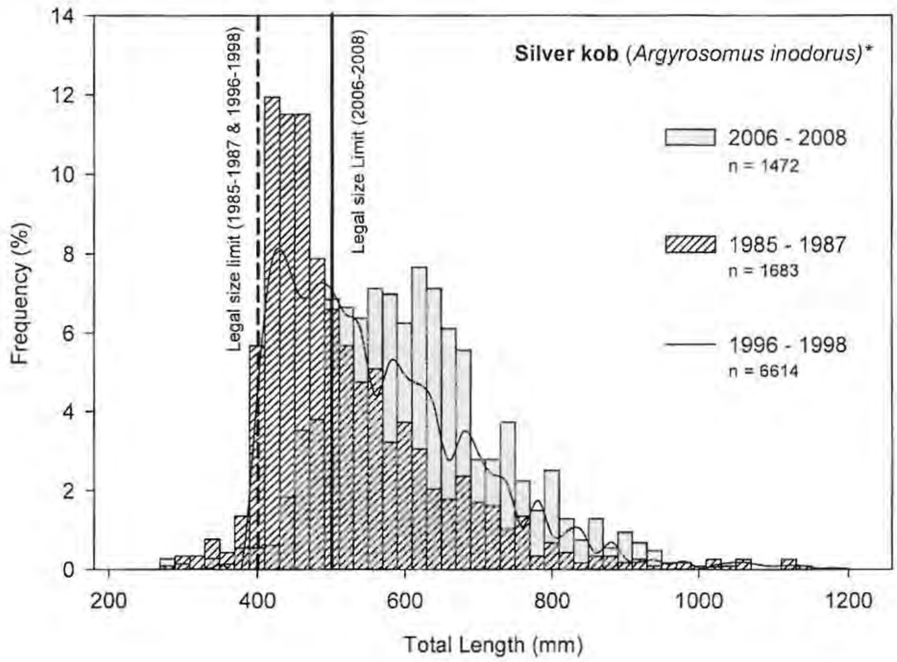


Fig. 5.2 Length-frequency distributions of silver kob (*Argyrosomus inodorus*) in the Port Alfred linefishery during the periods 1985 to 1987, 1996 to 1998 and 2006 to 2008. The minimum legal sizes (mmTL) are indicated by vertical lines. *Silver kob was wrongly reported as *Argyrosomus hololepidotus* in the 1985-1987 dataset.

The length-frequency histograms of geelbek landed in the Port Alfred fishery during 1985-87 and 2006-08 were strongly bimodal (Fig. 5.3). Prior to the change in size limit, a large proportion of the geelbek landed in 1985-87 were between 400mmFL and 600mmFL and the length at 50% recruitment (480mmFL) occurred in the smaller-sized modal group. Subsequent to the increase in size limit, the majority of geelbek landed were in the larger-sized modal group (the length at 50% recruitment was 855mmFL). Despite this, during the 2006-08 period 12.5% of landed geelbek were below the minimum legal size.

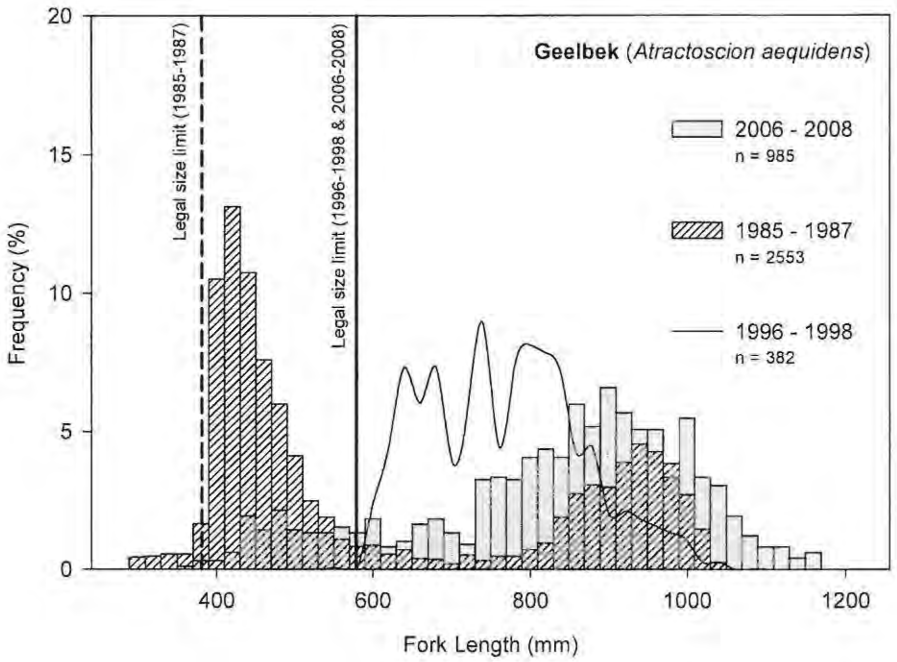


Fig. 5.3 Length-frequency distributions of geelbek (*Atractoscion aequidens*) in the Port Alfred/Kenton-on-Sea/Boknes linefishery during the periods 1985 to 1987, 1996 to 1998 and 2006 to 2008. The minimum legal sizes (mmFL) are indicated by the broken vertical lines.

Before 2006, the minimum size for carpenter was 250mmTL (204mmFL). In 2006 the minimum size was increased to 350mmTL (321mmFL). An insignificant number of the carpenter measured during the periods 1985-87 and 1996-98 were below the legal size limit. However, after the increase of the size limit in 2006, 4.1% of landed fish were smaller than the minimum size (Fig. 5.4).

There was no significant difference in the size-frequency distribution of panga landed in 1985-87 and those landed in 2006-08 (Kolmogorov-Smirnov; D=0.00158, p = 0.986; Fig. 5.5). Although the mean size was similar during all three time periods (290mmFL), in 1996-98 panga recruited into the fishery 40mm larger than during the other two periods. Panga are not subject to a size limit.

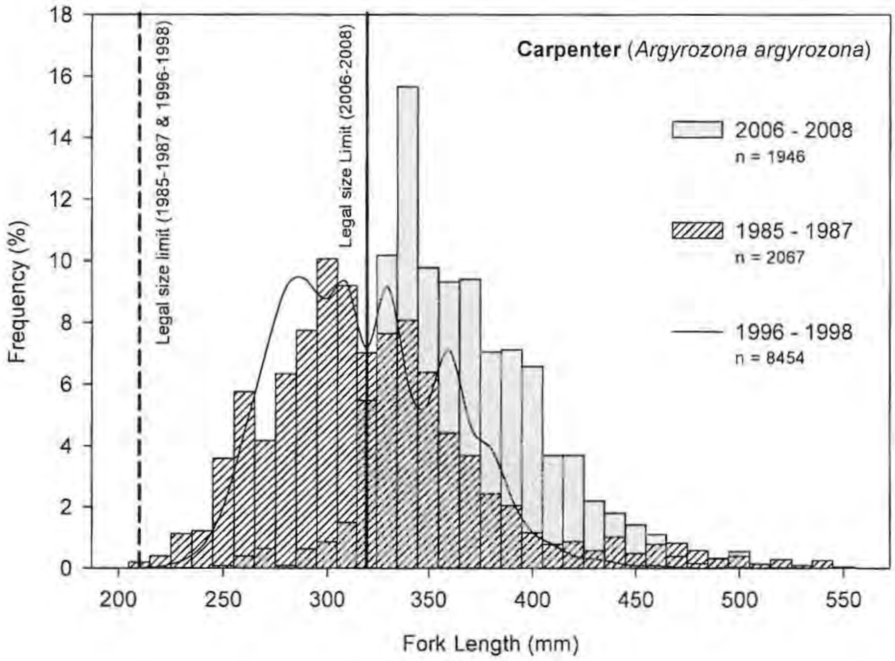


Fig. 5.4 Length-frequency distributions of carpenter (*Argyrozona argyrozona*) in the Port Alfred/Kenton-on-Sea/Boknes linefishery during the periods 1985 to 1987, 1996 to 1998 and 2006 to 2008. The minimum legal sizes (mmFL) are indicated by the broken vertical lines.

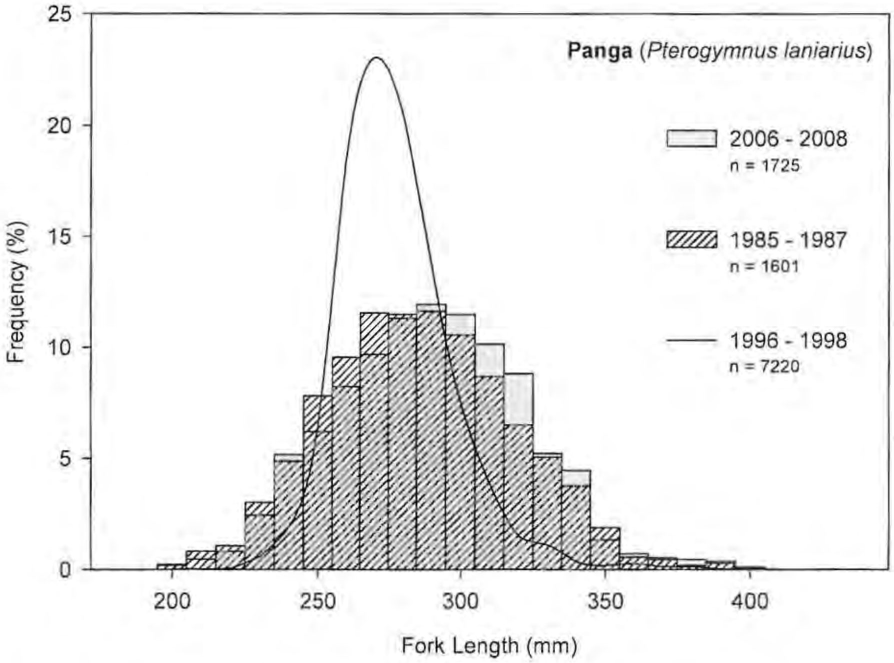


Fig. 5.5 Length-frequency distributions of panga (*Pterogymnus laniarius*) in the Port Alfred/Kenton-on-Sea/Boknes linefishery during the periods 1985 to 1987, 1996 to 1998 and 2006 to 2008.

In 1992, the size limit for santer was increased from 250mmTL to 300mmTL (Table 5.2b). Despite the increase there was no significant differences in the size-frequency distribution of santer landed between 1985-87 and 2006-08 (Kolmogorov-Smirnov; D = 0.109, p = 0.122). The mean size of santer caught during 1996-98 was similar to the mean size landed in 2006-08, however, the length at 50% recruitment had increased by 30mm. Santer first recruited into the fishery at a length of 230mmFL which was smaller than the minimum legal size limit of 264mmFL (300mmTL) and the size at 50% sexual maturity (340mmTL; Coetzee 1983).

The mean size of Roman between all three periods was similar (Table 5.2b). However, subsequent to an increase in the minimum legal size limit from 250mmTL (227mmFL) to 300mmTL (272mmFL) in 1992, Roman first recruited into the fishery at a 60mm larger size than in the past. However, there was no increase in the length at 50% recruitment.

As for panga, there is no legal size restriction for blue hottentot. There has been a significant decrease in the mean size of blue hottentot (Table 5.2b). In 2006-08, the lengths at first and 50% recruitment were between 50mm and 60mm smaller than in 1985-87 and 1996-98.

The minimum legal size for dageraad was increased twice in the 23 year period. These changes were reflected in the size at first recruitment. In 1992 the minimum size was increased from 250mmTL to 300mmTL and there was a 30mm increase in both the mean size and length at first recruitment. In 2006 the minimum size was increased to 400mmTL (353mmFL) and the size at first recruitment increased by 30mm. However, the mean size of dageraad landed during 2006-08 was smaller than those landed in 1996-98. In 2006-08 20.7% of dageraad landed in the fishery were under the legal size.

The trends in changes in mean size of the main component species of the Port Alfred fishery for the period 1985 to 1999 and the period 2006 to 2008 are presented graphically in Fig. 5.6. Amendments to the minimum legal sizes (Chapter 1, Table 1.4) were clearly evident in the increasing trends in the mean sizes of kob and geelbek. Despite the increases to the size limits, the mean size of dageraad only marginally increased, while that of Roman decreased between 1985 and 2008. The two most notable decreasing trends in mean length were panga and blue hottentot, for which there are no minimum legal sizes.

Table 5.1b Length frequency analyses for the sparids santer, Roman, dageraad, red stumpnose and blue hottentot in the Port Alfred linefishery during the periods 1985 to 1987, 1996 to 1998 and 2006 to 2008.

Species	Period	Mean (mmFL)		Kolmogorov-Smirnov two-sample test, *D* and *p-values*			n	1st recruitment (mmFL)	50% recruitment (mmFL)	50% maturity (mmFL)	Minimum legal size (mmTL: mmFL)	Illegal landings (%)
				85-87	*96-98*	*06-08*						
Santer	1985 - 1987	369.10 ±72.0	a	x	D = 0.193	**D = 0.109**	130	220	340	340*	250:218	0.8%
(*Cheimerius nufar*)	1996 - 1998	356.24 ±46.3	b	p < 0.001	x	D = 0.154	629	240	330	340*	300:264	0.9%
	2006 - 2008	359.75 ±63.4	a	**p = 0.122**	p < 0.001	x	1276	230	360	340*	300:264	3.4%
Roman	1985 - 1987	343.56 ±54.6	a	x	D = 0.165	D = 0.163	118	220	350	172**	250 :227	0.85%
(*Chrysoblephus laticeps*)	1996 - 1998	348.11 ±47.8	b	p < 0.01	x	**D = 0.072**	593	260	350	172**	300 :272	3.17%
	2006 - 2008	345.42 ±39.6	b	p < 0.025	**p = 0.141**	x	432	280	340	172**	300 :272	0.46%
Dageraad	1985 - 1987	309.81 ±55.4	a	x	D = 0.558	D = 0.208	201	220	310	375**	250 :221	0.00%
(*Chrysoblephus cristiceps*)	1996 - 1998	341.17 ±41.6	b	p < 0.001	x	D = 0.350	1354	250	350	375**	300 :266	0.31%
	2006 - 2008	321.08 ±60.6	c	p < 0.001	p < 0.001	x	92	280	350	375**	400 :353	20.7%
Red stumpnose	1985 - 1987	446.58 ±59.0	a	x	D = 0.418	D = 0.548	38	310	440	-	250 : - Ŧ	0.0%
(*C. gibbiceps*)	1996 - 1998	493.53 ±129	b	p < 0.05	x	**D = 0.344**	17	330	555	-	300 : - Ŧ	0.0%
	2006 - 2008	526.99 ±51.1	b	p < 0.001	**p = 0.062**	x	110	320	410	-	300 : - Ŧ	0.0%
Blue hottentot	1985 - 1987	317.76 ±31.3	a	x	D = 0.421	D = 0.201	76	260	320	200- 250***	- :-	-
(*Pachymetopon aeneum*)	1996 - 1998	356.96 ±60.7	b	p < 0.05	x	D = 0.536	23	260	365	200- 250***	- :-	-
	2006 - 2008	302.58 ±30.5	c	p < 0.025	p < 0.001	x	230	210	300	200- 250***	- :-	-

* South Eastern Cape(Coetzee 1983)

** Females from Port Elizabeth (Buxton 1993)

*** South Eastern Cape (Buxton and Clark 1986)

Ŧ Length FL: length TL conversion unknown for red stumpnose, however, all the measured FL were greater than the TL minimum legal size limit for all periods

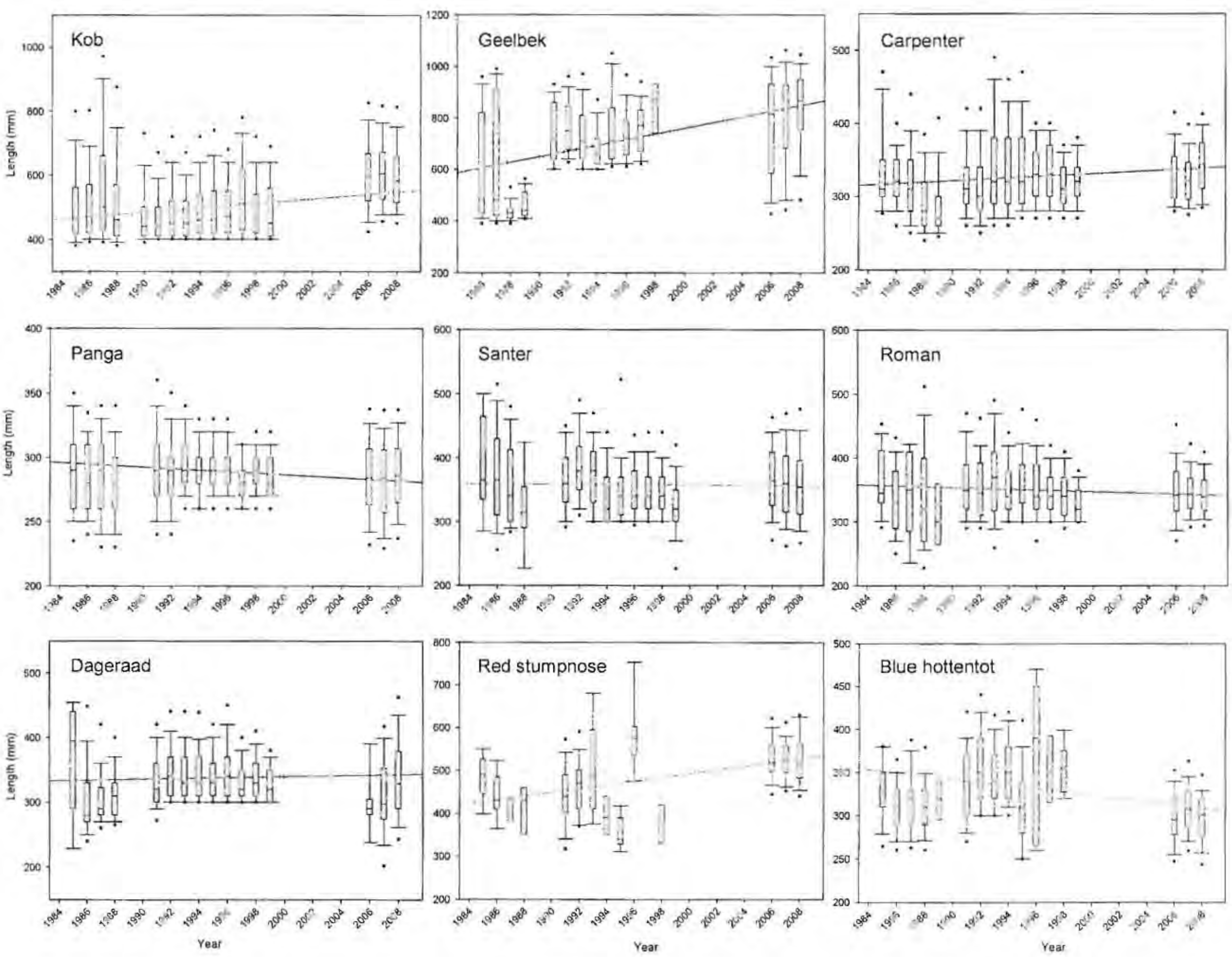


Fig. 5.6 Trends in mean size of nine selected species in the Port Alfred linefishery between 1985 and 2008.

Size-spectra analyses

The slope of the size spectra is suggested to be quasi-linearly related to fishing pressure (Shin and Curry 2004). Height is representative of the overall productivity of the fish assemblage (Bianchi *et al.* 2000). Increased fishing pressure results in an increasing steepness of the slope (i.e. the slope becomes more negative) and a decrease in the height of the size spectra (Bianchi *et al.* 2000, Yemane *et al.* 2004). Fig. 5.7 shows the trend in the slopes and heights of the size spectra in Port Alfred between 1985 and 2007. There was a significant difference between the slopes of the size spectra constructed with and without geelbek (Students' T-Test, $p < 0.01$). When geelbek was included in the size spectra there was an increasing trend in the slope (*x*-

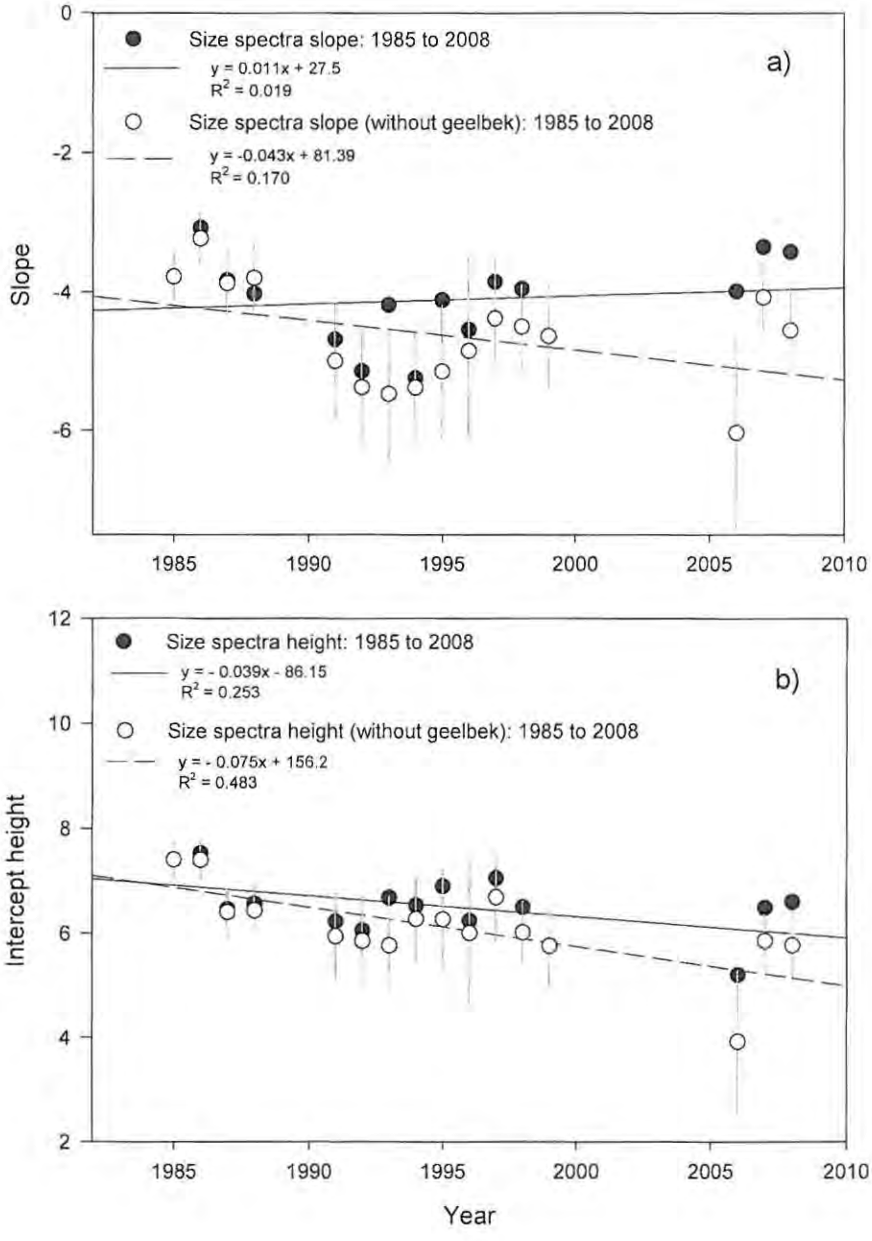


Fig. 5.7 Trends in **a)** the slope and **b)** the height of the size spectra in the Port Alfred linefishery between 1985 and 2008.

coefficient = 0.01; Fig. 5.7a). In other words, the landings were dominated by larger and, supposedly more K-selected species. However, without geelbek the slope steepens (*x-coefficient* = -0.04) and the height decreases over the 23 year period (*x-coefficient* = -0.08; Fig. 5.7b). This suggests that the commercial landings are dominated by relatively smaller fish, and the overall productivity of the fishery is lower than suggested by the size spectra with geelbek included.

The differences between the size spectra with and without geelbek were greatest during 2006-2008. Figure 5.8 shows the increasing influence of geelbek on the size spectra for the years 1987, 1997 and 2007. Although there was a relatively high abundance of geelbek in the fishery in 1987 (Fig. 5.8a), there was no significant difference in the size spectra with or without geelbek. During 1997 (Fig. 5.8b), and particularly in 2007 (Fig 5.8c), geelbek had a greater influence on the slope of the size spectra.

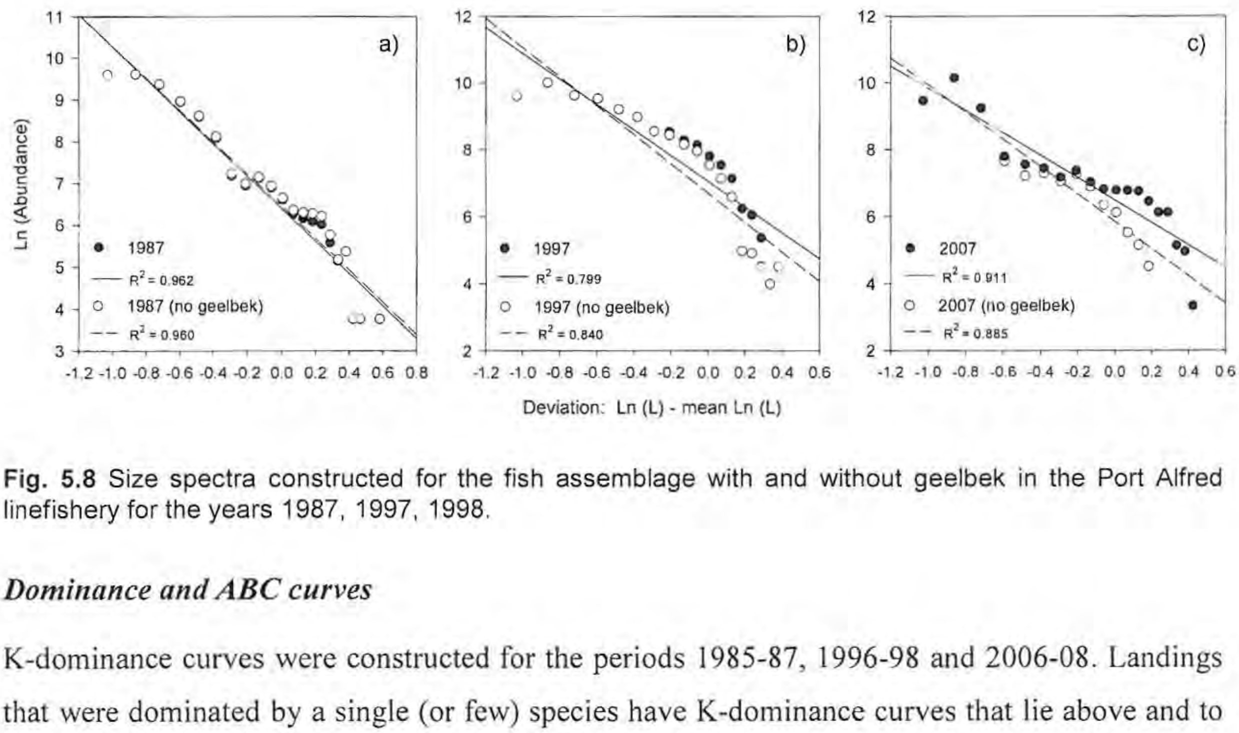


Fig. 5.8 Size spectra constructed for the fish assemblage with and without geelbek in the Port Alfred linefishery for the years 1987, 1997, 1998.

Dominance and ABC curves

K-dominance curves were constructed for the periods 1985-87, 1996-98 and 2006-08. Landings that were dominated by a single (or few) species have K-dominance curves that lie above and to the left of curves constructed from landings with a more even species distribution (Rice 2000). In terms of biomass, the K-dominance curve of the commercial landings during 1985-87 lay above and to the left of the curves for the two other periods (Fig. 5.9a). This was due to substantial proportions of silver kob and carpenter, which made up almost 80% of the commercial landings in 1985-87.

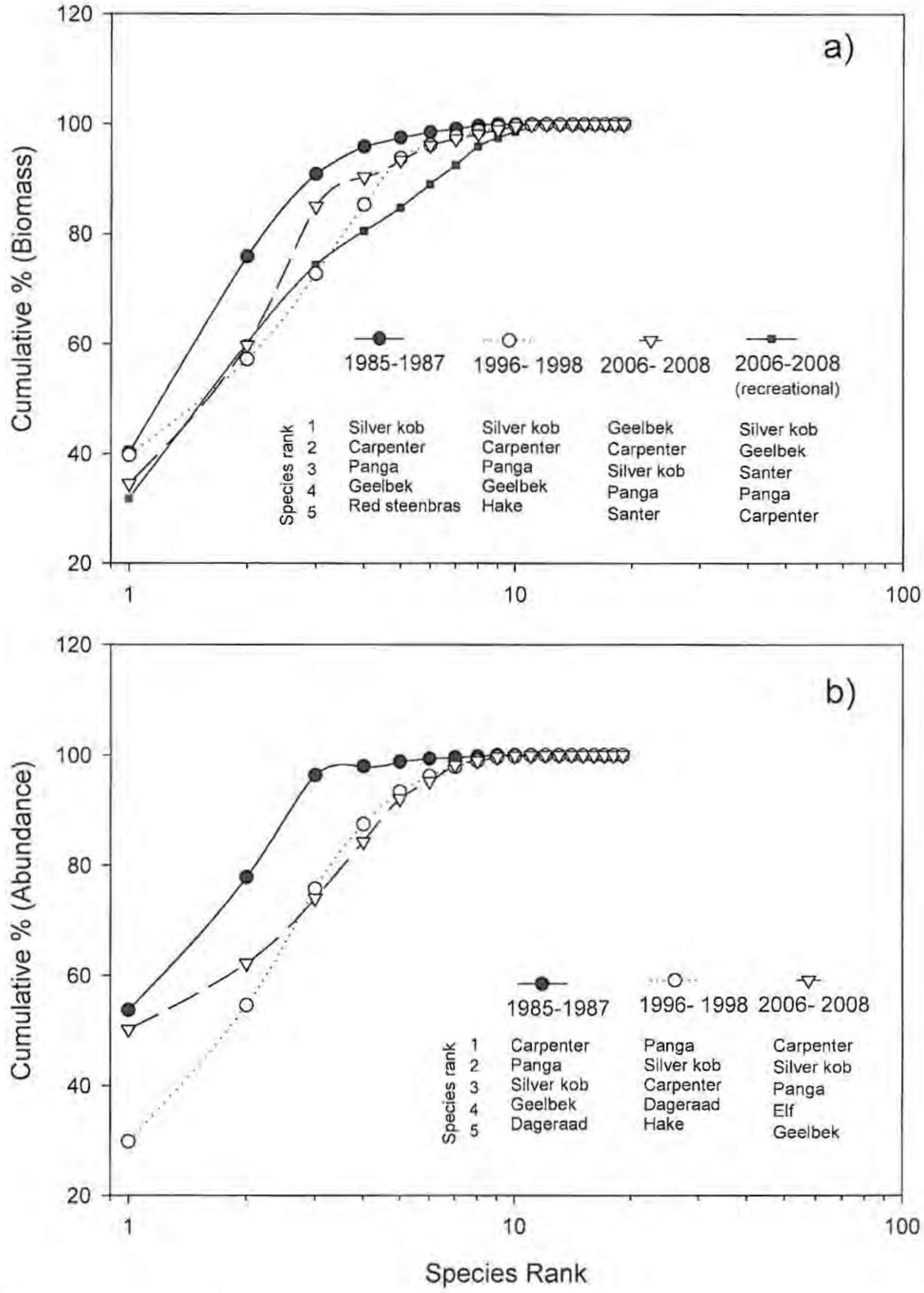


Fig. 5.9 a) Biomass and **b)** Abundance K-dominance curves in the Port Alfred linefishery for the periods 1985-1987, 1996-1998 and 2006-2008.

During 1996-98, silver kob contributed roughly the same proportion of the total commercial landings as it did in 1985-87. However, the proportion of carpenter, panga and geelbek in the commercial landings were similar (roughly 15% each) and the K-dominance curve suggested that during 1996-98 the percent contribution of each species to the commercial landings were more evenly distributed than in the other two periods (Fig. 5.9a).

In 2006-08, the K-dominance curve of the recreational landings lay below and to the right of the commercial K-dominance curve for the same period. This indicates that the commercial catch consisted of a few species that were dominant, while the proportions of the species in the recreational landings were more evenly distributed. Geelbek was the most dominant species in the commercial landings during 2006-08. The recreational landings had roughly equal proportions of geelbek, silver kob and santer.

In terms of abundance, the three periods were dominated by carpenter, panga and silver kob (Fig. 5.9b). In 1985-87, these three species made up more than 90% of the catch, while in 1996-98 and 2006-08 they contributed roughly 75% of the landings by number.

The W-statistic, or effectively, the area between the abundance and biomass K-dominance curves (plotted on the same axis), is suggested to provide an indication of fishing pressure (Fig. 5.1). Positive W-statistic value indicates that the abundance curve lies below the biomass curve indicating a predominance of larger species in the commercial landings. This state is indicative of a less stressed community structure. A negative W-statistic indicates a lower abundance of K-selected species and represents a more disturbed state (Warwick and Clarke 1991, Yemane *et al.* 2005). Figure 5.10 presents the trend in the W-statistic in the Port Alfred fishery between 1985 and 2008. During all the years (with the exception of 1997, 1999 and 2006) the W-statistic was marginally below zero. This indicates that the biomass curve lies below the abundance curve and there was an abundance of smaller species. The fishery was considered to be in a 'moderately disturbed' state during all years (refer to Fig. 5.1).

However, there was an increasing trend in the W-statistic over the 23 year period which would indicate that the landings are becoming progressively more dominated by larger species (*x-coefficient* = 0.003). This was due to lower values of the W-statistic in the late 1980's and early 1990's which were related to the targeting of panga (a comparatively smaller fish) in those years.

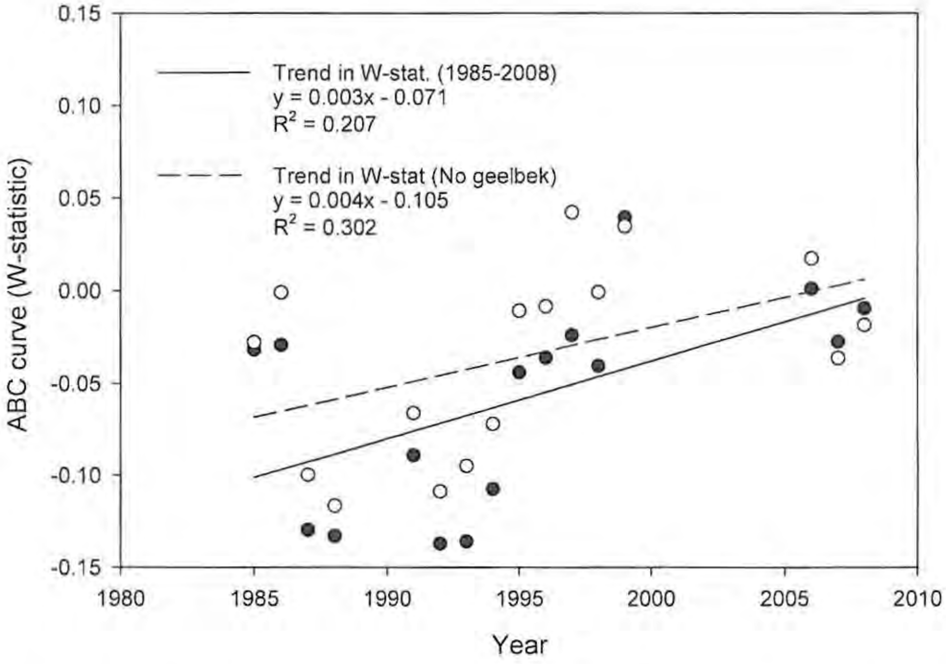


Fig. 5.10 Trends in the W-statistic calculated from the Abundance Biomass Comparison (ABC) curves of the Port Alfred linefishery the period 1985 to 2008 with and without geelbek (*Atractoscion aequidens*).

When geelbek were excluded from the ABC analysis, the W-statistic still showed an increasing trend (*x-coefficient* = 0.004). The W-statistics calculated without geelbek are generally larger than those calculated with geelbek (Fig. 5.10). This is due to the fact that one less species was included in the calculation each year and does not indicate the general presence of larger species in the fishery (refer to Equation 5.2). Although the overall trend in the W-statistic was not influence by geelbek to any great extent, during 2007 and 2008 the W-statistic was lower without geelbek than with geelbek (this is once again evidence of the dominance of geelbek in the Port Alfred linefishery in recent years).

DISCUSSION

Interpretation of the length-frequency distributions and mean lengths with respect to the changes in the management environment

Griffiths (1997b) isolated three separate silver kob stocks between Cape Point and the Kei River, and he treated the silver kob South Eastern Cape (Port Alfred region), South Western Cape and Southern Cape as discrete stocks. Per-recruit modelling of these three discrete subpopulations suggested that all silver kob stocks are heavily over-exploited, severely depleted and with limited potential for population growth. A minimum legal size of 400mmTL has been in place for silver kob since 1940. Despite being arbitrarily assigned it was found by Griffiths (1997a) that this was above the size at which the species reaches 50% maturity. Griffiths (1997a) also found that silver kob were not as resilient to exploitation as previously expected and kob stocks have collapsed despite a minimum size limit larger than the size at 50% maturity being in place for half a century (well more than twice the lifespan of the species).

Hecht and Tilney (1989) expressed concern over the large proportion of smaller silver kob (recruits) being landed, and the declining directed *cpue* in the Port Alfred fishery between 1982 and 1987. After the minimum size was increased from 400mmTL to 500mmTL in 2006, the mean size of silver kob increased but a significant proportion of smaller silver kob (<500mmTL) were still being landed by the fishery during 2006-08 (Table 5.2a, Fig. 5.2). Reports from the fisherman during 2006-08 indicate that there was a large abundance of silver kob of between 450mmTL and 500mmTL present in the fishery. Disconcertingly, the length-frequencies of landed silver kob were no longer knife-edged around the minimum legal size limit (as they were in previous time periods) but are more normally distributed over the size classes between 400mmTL and 700mmTL. This would indicate poor compliance by the fishers in Port Alfred. In fact, a significant proportion (16%) of silver kob landed during the current period (2006-08) was below the legal size limit. This was a marked increase from the illegal landings by commercial operators during the 1985-87 and 1996-98 periods when the size limit was 400mmTL (7.5% and 4%, respectively).

Since 2004, there has been abundance of geelbek in the Port Alfred/Kenton-on-Sea/Boknes fishery (Chapters 3 and 4) and other areas in the Eastern Cape (C.Wilke, MCM, pers. comm.

2008). During 2006-08 commercial linefishery has almost solely been sustained by these strong winter recruitments of migratory geelbek (Chapters 3 and 4). Geelbek stocks on the eastern seaboard consists of three size related subpopulations with well defined migratory patterns. Geelbek spawn in KZN during the spring and summer months. The juvenile fish occur in the 'nursery area' in the South Eastern Cape. These fish then migrate south to the South Western Cape for approximately one year before joining the adult sub-population. In Port Alfred larger geelbek are caught in autumn and early winter when adults follow the sardines on the way to the spawning grounds (Griffiths and Hecht 1995). This theory is supported by the length-frequency distributions of geelbek as well as the significantly larger size of geelbek landed during the winter months during this study period.

The strongly bimodal size distribution of geelbek from the 1985-87 period incorporated the juvenile fish as well as adult populations moving to spawning grounds (Fig. 5.3). Geelbek of less than 500mm made up the dominant mode of the bimodal length frequency distribution in 1984-87. Subsequent to the increase in minimum legal size from 400mmTL to 600mmTL in 1992, the geelbek landed in the latter two periods had a larger mean size and had a significantly different size-distribution to those caught in 1985-87. The geelbek from the smaller modal group were mostly absent from the commercial and recreational landings. As with silver kob, reports from the fishermen indicated a notable abundance of geelbek just less than 600mmTL present in the fishery during 2006-08. This was evident in the fact that 12.5% of geelbek landed in 2006-08 were under the legal size (in 1985-87 only 5% of the geelbek landed were undersize).

It is tempting to suggest that the imposition of the 600mmTL size limit in 1992 may have been the reason for the observed increase in abundance of geelbek since 2004. No such claim will be made but it is strongly recommended that the stock of geelbek be re-assessed to test this tentative hypothesis.

The minimum legal size for carpenter was increased from 250mmTL to 350mmTL in 2006. Therefore it is not surprising that carpenter caught in the current sampling period were significantly larger than in the previous periods (Table. 5.2a, Fig. 5.4). While none of the carpenter measured in the first two sample periods were undersize, 3.5% of the carpenter measured in the current period (2006-08) were smaller than the legal limit. When examining the 1985-87 and 1996-98 length-frequency distributions it is quite apparent that the previous minimum size for carpenter (250mmTL) served no purpose at all in the Port Alfred fishery. The mean sizes of carpenter caught during all the periods (320-360mmFL) were significantly larger than the previous size limit (250mmTL: 204mmFL). Size-frequency data for carpenter during 2006-08 would suggest that the new size limit is having an effect on fisher behaviour and landings. Compliance for carpenter is 'relatively' good as only 4.1% of the landed carpenter were undersize.

Yemane *et al.* (2004) noted that the mean size of many of the species in the Cape linefishery decreased dramatically between 1897 and 1998. In conjunction with declining *cpue*, they came to the conclusion that the reduction in mean size of a species was due to heavy exploitation. They further pointed out that the implementation of minimum size limit regulations (in 1940, 1985 and 1992) had a significant effect on the mean size of fish landed and that mean length alone would not provide a reliable index of the impact of fishing on the stocks. Between 1985 and 2008, there have been substantial increases in size limits for most species (Table 1.3). The trends in the mean size of silver kob and geelbek landed in Port Alfred increased substantially between 1985 and 2008 as a direct consequence of the increases to the size limits (Fig. 5.6).

However, despite the increases in size limits for Roman, dageraad and santer (Chapter 1, Table 3.1), the mean sizes of these species have either decreased (as in the case of Roman) or had little or no increase (as with dageraad and santer) between 1985 and 2008 (Fig. 5.6). This indicates one of two things, either the size limits are ineffective at limiting the catch of the particular species or alternatively, there is poor compliance resulting in substantial landings of undersize fish. In the case of dageraad, the latter seems to be the case. During the 2006-08 period 20.7% of dageraad landed in Port Alfred were undersize. As a result there was only a marginal increase in the mean size for dageraad despite the cumulative 150mm increase in size limit over the 23 year period. There was only a 10mm difference between the mean size of dageraad landed in 1985-87 and those landed in 2006-08. The increase in the size limit to 400mmTL (353mmFL) was based on the fact that dageraad are a slow growing, late maturing and highly vulnerable species (Griffiths 2000). During 2006-08, the size at first recruitment, 280mmFL, was almost 100mm below the size at 50% maturity (375mmFL; Buxton and Clarke 1986). These poor levels of compliance will continue to have dire consequences for the species.

Compliance with the size limit for santer was relatively good during 2006-08 with only 3.4% of catch being undersize (Table 5.2b). However, the data show that there was a marginal decrease in the size of santer landed between 1985 and 2008, despite the increase in the size limit from 250mmTL to 300mmTL in 1992 (Fig. 5.6). The mean size of santer landed during all three periods (360mmFL) and the size at 50% recruitment (330mmFL to 360mmFL) were substantially higher than the current minimum size limit (300mmTL:264mmFL). This means that the current size limit for santer in the Port Alfred linefishery serves no purpose. Moreover it is of concern that the current size limit is below the length at 50% maturity (340mmTL). The evidence presented in Chapter 4 suggests that santer stocks are being increasingly targeted by recreational fishers in the Port Alfred linefishery. In light of these facts it would be prudent to increase the minimum legal size of santer 350mmTL, corresponding to the size at greater than 50% maturity (340mmTL; Coetzee and Baird 1981).

Similarly, the size limit of Roman was increased from 250mmTL to 300mmTL (262mmFL) in 1992 and the mean size of roman landed during each of the three periods was roughly 350mmFL (almost 100mm larger than the minimum legal size). There was a decreasing trend in the size of Roman between 1985 and 2008 (Fig. 5.6). As with santer, the minimum legal size limit was therefore ineffective at limiting the landings of Roman in the fishery. However, the current size limit is well above the length at 50% maturity for Roman (172mmFL; Buxton 1993) and the proportion of illegal Roman landings during 2006-08 were low. Furthermore, the bag limit of two Roman per fisher.day^{-1} was shown to be effective in reducing the *cpue* of the species (Chapter 3, Fig. 14b). Therefore, although the size limit is ineffectual, there is no immediate concern with regard to the stock status of Roman in the Port Alfred linefishery.

Panga and blue hottentot are not subject to a minimum size limit. The mean lengths of both of these species declined between 1985 and 2008, in particular that of blue hottentot (Fig. 5.6). The resilience of the panga stocks was outlined in the previous chapter and warrants no further comment. On the other hand, the mean size of blue hottentot decreased substantially from 356mmFL in 1996-98 to 302mmFL in 2006-08. Blue hottentot is a slow growing late maturing species that, like many other of the more K-selected reef species, is particularly vulnerable to over exploitation (Coleman *et al.* 2000). Furthermore, in Chapter 3, it was found that both the commercial and recreational sectors are fishing closer to the shore and could have a substantial impact on the near-shore reefs and species such as blue hottentot. Consideration should therefore be given to setting a minimum legal size for this species at 250mmTL, roughly equivalent to the best estimate of size at 50% sexual maturity (200-250mmFL; Buxton and Clarke 1986).

Community metrics as tools for evaluating the Port Alfred linefishery

On the basis of size spectra analysis, Yemane *et al.* (2004) suggested that there was a long-term shift towards smaller fish in the South-Eastern Cape commercial catch from 1890 to 1998. They suggested that this was a result of both a decrease in the abundance of larger fish of all species and the differential over-exploitation of k-selected species. Furthermore, they pointed out (from evidence in the Western Cape) that the slope and height of the size spectrum was easily influenced by the presence of a large dominant species (e.g. snoek) The presence of such a species would result in the slope of the size spectra being shallower and masking the changes in the size composition of other species (Bianchi *et al.* 2000, Yemane *et al.* 2004).

In contrast with the findings of Yemane *et al.* (2004), there was an increasing trend in the slope and a decreasing trend in the height of the size spectra in the Port Alfred linefishery between 1985 and 2008 (Fig. 5.7a). In theory this would suggest that the fishery was gradually becoming more dominated by larger, more K-selected species and the overall productivity of the fishery was increasing as a consequence of reduced fishing pressure. In reality however, the large migratory geelbek contributed a substantial proportion of the landings during 2006 to 2008 (Chapter 3 and Chapter 4). Of the species examined, geelbek was ranked as the most R-selected species after silver kob (Table 5.1). In this case (with geelbek included), the assumption that the more k-selected species are the larger species breaks down, and the slope and height of the size spectra would not indicate the level of fishing pressure. Without the masking effects of geelbek, the slope of the size spectra has an opposite, and possibly more representative, trends over the 23 year period. The slope exhibits an increasing trend (Fig. 5.7a) while the height decreases (Fig. 5.7b). Therefore, the size spectra indicated that there was dominance of smaller fish in the landings and the overall productivity of the fishery decreasing due to fishing pressure. This was indeed the case.

Furthermore, from 1985 to 1999 the size spectra accurately represented the increased fishing pressure both with and without the inclusion of geelbek. It was only in the last period, 2006-

2008, that geelbek had a substantial effect on the size spectra (Fig. 5.8). This could possibly be due to the fact that the size limit of geelbek was increased by 200mmTL in 1992 and the mean size of geelbek increasing (Fig. 5.6). Unlike the years before 1992, this would result in geelbek being absent from the smaller size classes in the size spectra and have a greater influence on the slope and height.

Therefore, despite the suite of management measures that have been put in place, the increasing trend in the slope (coupled with the decreasing trend in the height) of the size spectra over the past 23 years suggest that the Port Alfred linefishery is still under substantial pressure and is in a declining phase.

In contrast to the size-spectra analysis, the ABC curves suggest that the Port Alfred fishery is gradually becoming more dominated by larger species. Even with geelbek excluded from the analysis, there was an increasing trend in the W-statistic over the 23 year period. It is hypothesized that this was largely due to the substantial commercial effort focussed on panga during the late 1980's and early 1990's. This selective targeting had a significant biasing effect on the trend in the w-statistic over the 23 year period. It could be argued that panga were targeted solely due to the declining abundance of larger, more K-selected species (Hecht and Tilney 1989, Badenhorst and Smale 1991). However, in the current fishery, the commercial and recreational fishers are influenced by increasing operating costs, and targeting low value and small panga neither economically viable nor recreationally satisfying. As mentioned in Chapter 4, instead of targeting low value species, the commercial vessels would actually just not go to sea on days when there were no geelbek or silver kob. Therefore the ABC curve could be heavily influenced by the behaviour of fishers. Overall it was concluded that ABC curves do not accurately represent the status of the Port Alfred linefishery.

Changes to the South African linefish management environment have had a significant effect on the size structure of the species within the Port Alfred commercial and recreational linefishery, in terms of both mean sizes and size-frequency distributions. Furthermore, the behaviour of the fishers has also changed in response to the management measures and external factors such as increasing operating costs. This highlights the necessity of using fishery independent data to accurately assess the response of the entire fish assemblages to changes in fishing effort using a full suite of community-based metrics.

CHAPTER 6

GENERAL CONCLUSIONS

INTRODUCTION

Changes to the linefishery management regulations in South Africa had a substantial effect on the fishery metrics investigated in this study. In Chapter 3 it was shown that the significant shifts in commercial fishing effort, catch composition, overall *cpue* and the directed *cpue* for many species were related to the changes that have occurred in the management environment between 1985 and 2007. In Chapter 4 it was shown that the differences in effort, catch composition, *cpue* and total landings between the commercial and recreational sectors in the Port Alfred linefishery were not only related to different economic pressures, but also governed by the differences in fishing regulations between the two sectors. Chapter 5 related the shifts in the size distribution of the dominant species to changes in the commercial and recreational fishing regulations. It also explored the validity of using community-based metrics as tools for assessing the 'status' of the fishery and found that fisher behaviour had a significant effect on these metrics. The summarised findings of all chapters are presented in Table 6.1. The following chapter draws from the findings of the previous chapters to highlight a few management considerations and puts forward management recommendations based on these findings.

MANAGEMENT CONSIDERATIONS

Understanding the user behaviour in response to management changes is essential when examining a fishery, particularly when historical, fisheries dependent data are used to draw conclusions on its status and making management decisions. Since the work done on the Port Alfred/Kenton-on-Sea/Boknes linefishery by Hecht and Tilney (1989) there were four, broadly categorised, factors that have governed the spatial and temporal distribution of effort, catch composition and *cpue* in the fishery, namely legislative, regulatory, biological and external

Table 6.1 Summarised findings of the retrospective study of the Port Alfred linefishery from 1985 to 2008. (▲▼ signify increasing and decreasing trends, respectively).

		Reference Chapter *Chapter 3: NMLS trends (1985 to 2007)*					*Chapter 4: Commercial vs. Recreational* *Chapter 5: Length-based analysis*		
		1985-1998	1999-2001	2002-2005	2006-2007	Overall	1985-1987	1996-1998	2006-2008
General fishery metrics									
Effort (vessels/month)	*Overall*	▼	▼	▲	▼	▼	-	-	-
	Commerci	-	-	-	-	-	-	-	39.45 ▼
	Recreation	-	-	-	-	-	-	-	58.73 ▼
Catch rate (kg/fisher/hour)	*Overall*	▼	▲	▲	▲	▲	-	-	-
	Commerci	-	-	-	-	-	3.57	3.07 ▼	3.24
	Recreation	-	-	-	-	-	2.78	2.50	1.03 ▼
Total catch (tonnes)	*Overall*	▼	▼	▲	▲	▼			
	Commerci	-	-	-	-	-	389.6	640.5 ▲	44.7 ▼
	Recreation	-	-	-	-	-	-	-	21.5
Community-based									
Size spectra analysis	*Slope*	-	-	-	-	▲	No trend	▲	▲
	Height	-	-	-	-	▼	▼	No trend	▼
ABC curves	Not viable as an indicator of fishing pressure in the Port Alfred linefishery								
Species related metrics									
Species Composition (%)	*Kob*	▲	▲	▼	▼	▼	37.1	33.2 ▼	24.7 ▼
	Geelbek	▲	▼	▲	▲	▲	4.9	13.8 ▲	32.7 ▲
	Carpenter	▲	▼	▲	▲	▼	34.4	18.5 ▼	23.9 ▲
	Panga	▼	▼	▼	▼	▼	15.5	15.3	5.3 ▼
	Sparid	▼	▲	▼	▼	▼	3.7	8.7 ▲	6.7 ▼
	Non-sparid	▲	▲	▼	▼	▲	0.8	2.3 ▲	3.4 ▲
Mean size (mm)	*Kob*	-	-	-	-	▲	548.4	507.1 ▼	603.6 ▲
	Geelbek	-	-	-	-	▲	610.8	762.2 ▲	818.9 ▼
	Carpenter	-	-	-	-	▲	323.6	328.4 ▲	366.1 ▲
	Panga	-	-	-	-	▼	285.9	275.3 ▼	285.1 ▲
Probability of capture	*Kob*	▼	▼	▼	▼	▼	-	-	-
	Geelbek	▼	▲	▲	▼	▲	-	-	-
	Carpenter	▼	▲	▲	▲	▼	-	-	-
	Panga	▼	▼	▲	▲	▼	-	-	-
	Roman	▼	▲	▼	▲	▲	-	-	-
	Dageraad	▲	▼	▼	▼	▼	-	-	-
Cpue (kg/fisher/hour)	*Kob*	▼	▼	▼	▲	▼	-	-	-
	Geelbek	▼	▲	▲	▲	▲	-	-	-
	Carpenter	▼	▲	▲	▲	▲	-	-	-
	Panga	▲	▼	▼	▲	▼	-	-	-
	Roman	▼	▲	▼	▲	▼	-	-	-
	Dageraad	▼	▲	▼	▲	▼	-	-	-

factors. These factors, or a combination of the factors, affected both the commercial and recreational sectors to varying degrees (Fig. 6.1). Legislative factors (the reduction of commercial licences) resulted in the shift from commercial to recreational effort in the fishery. Catch and effort in the recreational sector were governed largely by regulatory factors, such as size and bag limits. In contrast, catch and effort of the commercial operators were governed principally by external factors (such as operating costs and the beach price of fish) as well as biological factors such as the increase/decline in abundance of certain species.

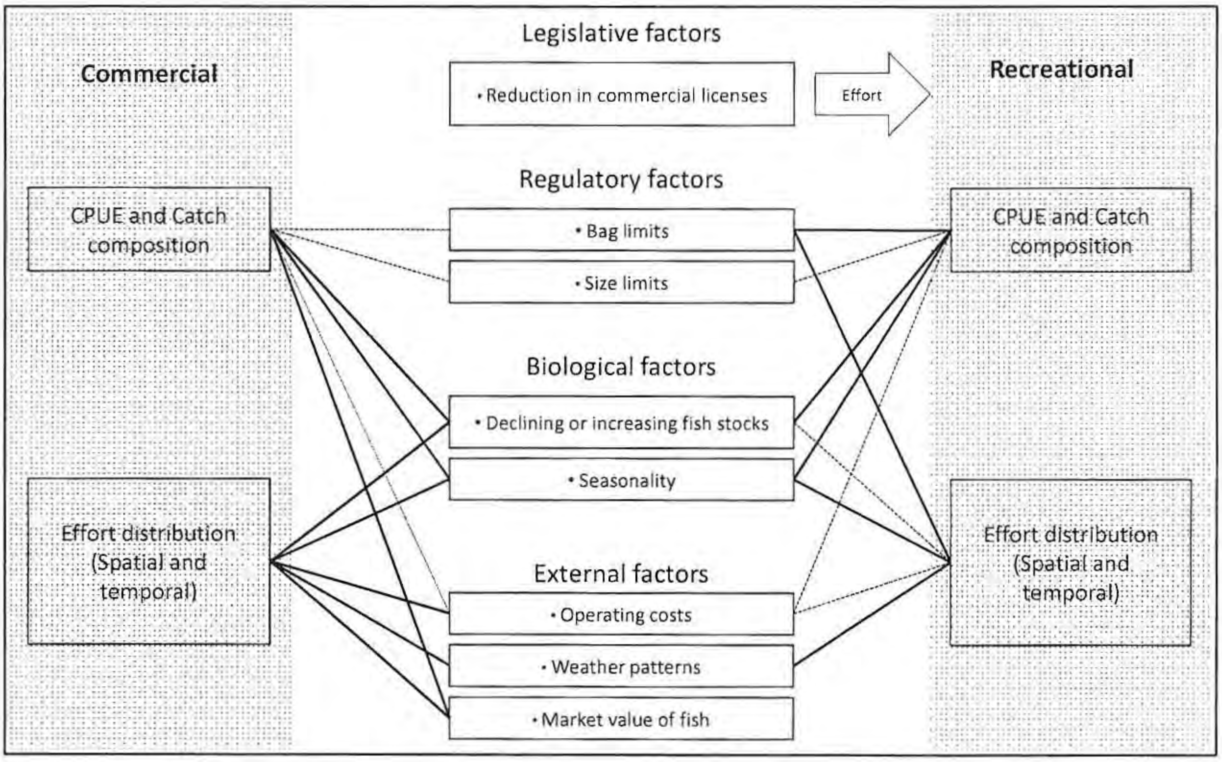


Fig. 6.1 Factors that govern the Port Alfred commercial and recreational linefishery subsequent to the reduction of commercial licences in 2002. Strong linkages are represented by thicker solid lines while weaker linkages are represented as broken lines.

Substantial changes were made to the linefish licensing structure in South Africa in 1998, 2002 and 2006 (Table 1.3). In 1998, the A- and B-license system was abolished in favour of annually allocated fishing rights (and later, medium- and then long-term fishing rights). Since 1999, monthly catch returns of the former B-licence holders were no longer submitted to MCM. This had a significant effect on the interpretation of the catch compositions and *cpue*. The B-licence holders, a proportion of whom operated largely as recreational anglers, (Hecht 1993, Sauer *et al.* 1997) had the potential to dilute, or inflate trends in catch composition, *cpue* and species-directed *cpue*. Although the catch returns of the B-licence holders were included in this study as a means of highlighting the effect of management changes, these catch returns *must* be excluded from any analysis aiming to assess historical trends using data from the NMLS.

Furthermore, with the reduced number of licences, the catch returns became more representative of individual vessels and fishing trips. For example, the *cpue* of carpenter gradually decreased from 1985 to 1999. However, the significantly higher carpenter directed *cpue* during 2004 and 2005 could be attributed largely to the landings of a single vessel operating in deeper water (Fig. 3.9) and not the result of a shift in the commercial effort as a whole. This factor should also be considered in future assessments of fisheries in South Africa when using the NMLS.

In 1989, the Port Alfred fishery was numerically dominated by 29 commercially active vessels (Hecht 1993). As a result of the 60.6% reduction in commercial effort from 33 vessels in 2001 to 13 vessels in 2002, there was a shift from commercial to recreational effort. In 2008 there were only four 'full-time' commercial vessels (despite eight licences allocated to the area) and the 'active' fishery was dominated by recreational vessels at a commercial to recreational ratio of 1:6.5. However, this was not the result of new recreational entrants into the fishery, but rather a function of the former commercial licence (B and some A-license) holders remaining in the fishery and operating as recreational fishermen. Although the licences had been removed, there was simply a displacement of effort from the commercial to the recreational sector. This emphasises the importance of assessing the impacts of both the commercial and recreational components on the fishery.

The recreational *cpue* (1.03 $kg.fisher^{-1}.hour^{-1}$) was less than a third of the commercial cpue (3.24 $kg.fisher^{-1}.hour^{-1}$; Fig. 4.4). This was solely due to the bag limits imposed on recreational fishers (Table 1.3). In particular, the bag limit of two geelbek per $fisher.day^{-1}$, which was implemented in 2006. Consequently, while the commercial landings were dominated by geelbek during the current sampling period, the recreational landings were dominated by silver kob, despite the abundance of geelbek during 2006-08 (Fig. 4.3, Fig. 5.9b). Therefore the recreational sector has the potential to have a significantly greater impact on fish stocks if it is not monitored. This stresses the importance of effective monitoring, control and surveillance of regulations in both sectors of the fishery.

Amendments to bag limits since 1985 have resulted in changes in the catch composition of the commercial and recreational landings (Fig. 4.3, Table 4.5). Bag limits for many of the more targeted and vulnerable species were reduced to less than five fish per $fisher.day^{-1}$ (e.g. roman to two per fisher.day^{-1} and dageraad to one per fisher.day^{-1}). As a consequence, the recreational landings were less dominated by a single species as the fishers maximised their overall bag limit of ten fish.day^{-1} by targeting a wider variety of species (Fig. 5.9a). Santer (which had a bag-limit of five per fisher.day^{-1}) become proportionately more prevalent in the recreational landings. Indeed, santer contributed as much as 10% of the total recreational landings in 2006-08, significantly more than the 1980's and 1990's (Table 4.5). In Chapter 5 it was proposed that the size limit of santer be increased from 300mmTL to 350mmTL.

Although the commercial operators are not subject to as many bag limits as the recreationals (Table 1.3), changes to the catch regulations for many of the reef-associated species also had an effect on commercial catches. In 2006, subsequent to the increase in size limit of dageraad from 300mmTL to 400mmTL and the reduction of the bag limit to a single fish per commercial fisher.day^{-1}, the dageraad directed *cpue* declined significantly (Fig. 3.16). There is uncertainty as to whether the decline in *cpue* was related to fishing regulations, a decline in abundance of dageraad or a combination of the two factors. Similarly the *cpue* of Roman was also influenced by size and bag limits (Fig. 3.14). As such, the comparison of historical and current *cpue* estimates provides little conclusive information on the stock health of these species. However, the mean size of both dageraad and Roman decreased over the 23 year period despite increases of the minimum size limits (Fig. 5.6). This would imply that there was either increased fishing pressure or poor compliance with size limits and that the stocks of both species are under severe pressure. Only 3.2% of Roman landed during 2006-08 were undersize (Table 5.1b), suggesting that the decrease in the mean size of roman was related to increased fishing pressure and not poor compliance. On the other hand, a substantial 20.7% of all dageraad landed during 2006-08 were undersize (Table 5.1b) and this was reflected in the decreasing trend in mean size since 1985.

Compliance with size limits was also low for some other important species in the fishery. In 2006 the size limit for silver kob was increased from 400mmTL to 500mmTL and during the 2006-08 period a substantial proportion (16%) of the silver kob landed in Port Alfred were undersize (Table 5.1a). Similarly, 12.5% of geelbek landed in 2006-08 were undersize (Table 5.1a). There appears to be a general reluctance by the recreational, and particularly the commercial operators to abide by recently implemented size limit changes and this undermines the effectiveness of the management measures. These compliance issues are related to poor enforcement and also possibly driven by the narrow profit margins due to the exponentially increasing operating costs and the linear increase in the beach price of fish (Fig. 6.2).

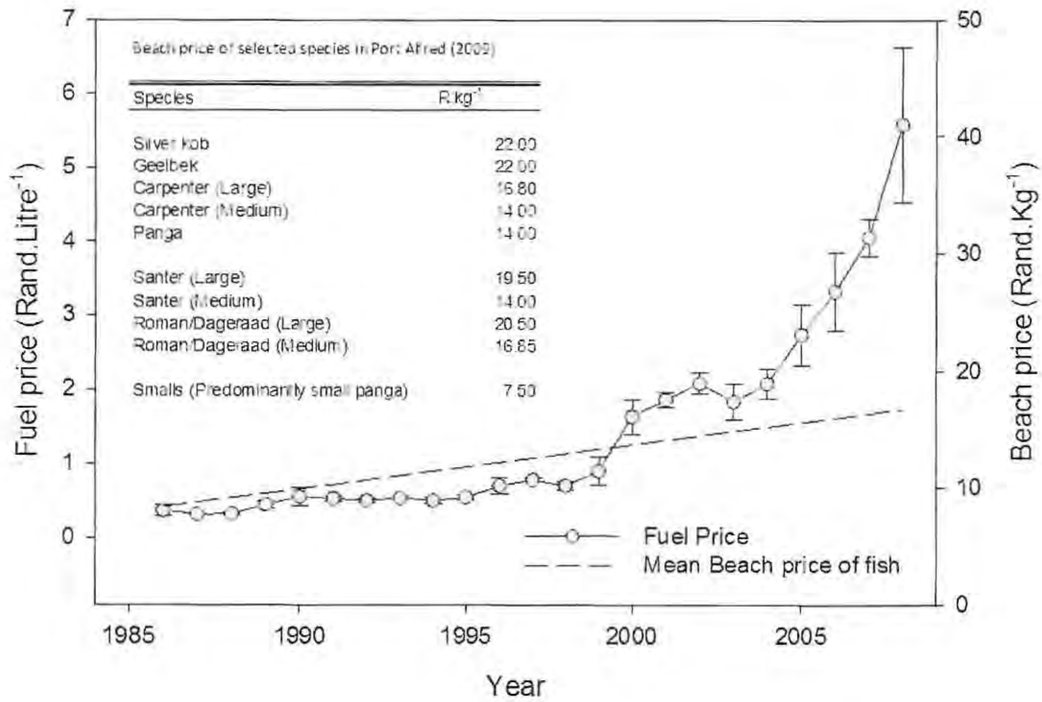


Fig. 6.2 Increase in fuel price in South Africa between 1985 and 2008, and the beach price of selected species in the Port Alfred fishery in 2008.

Fuel costs contribute the largest proportion to the operational costs of a commercial vessel. It was calculated that fuel contributed 50.8% to the total commercial operating costs on a yearly basis (including vessel depreciation)[14]. Although the fuel price has increased exponentially over the last two decades (in the order of 1500%), the beach value of fish has only increased by approximately 100% since 1985 (C. McClelland, Fresh Fish Market, pers. comm. 2009; Fig. 6.2). This has been the driving factor behind many of the trends observed in this study.

[14] Calculated from data collected in a concurrent study on the socio-economics of the Port Alfred fishery undertaken by the author and Prof. T. Hecht

Griffiths (2000) suggested that during the period 1986-1998, technological advances resulted in the shift of relative importance of species further offshore. Carpenter, panga and hake became more important (although geelbek and kob still contributed substantially to the landings). However in light of increasing overhead costs, during 2006-08 the commercial operators in the Port Alfred linefishery were fishing closer to access points and almost exclusively targeted the sciaenid species (particularly geelbek). Panga, which was targeted on the deeper low-profile reefs in the late 1980's and 1990's, did not make a significant contribution to either the commercial or recreational landings in 2006-08 (Fig. 3.11, Fig. 4.3). Due to the combination of increased overhead costs and the reduction in bag limit, carpenter (which was historically targeted by both the recreational and commercial sectors) no longer contributed any notable proportion to the recreational landings (Fig. 4.3). Recreational fishers were not willing to expend the time, effort and cost of target the deeper water species if they were limited to four carpenter per fisher.day^{-1}. These shifts not only placed an increased importance and reliance on the high valued sciaenid species (viz. silver kob and geelbek) but also focussed fishing pressure on the near-shore, reef-associated species.

STATUS OF THE FISHERY AND MANAGEMENT RECOMMENDATIONS

The high commercial *cpue* during 2004 to 2007 was related primarily to strong pulses of migrating geelbek. In fact, during 2006-2008 geelbek contributed substantial proportions of the total landings for both the commercial and recreational sectors (32.7% and 20.2%, respectively), significantly more than in the past (Fig. 4.3). It could be argued that between 2004 to 2008 the Port Alfred/Kenton-on-Sea/Boknes commercial linefishery was in fact sustained by these strong seasonal pulses of geelbek.

In Chapter 3 it was suggested that there may be signs of a possible recovery of the geelbek stock. It was hypothesized that this may possibly be related to the increase in the minimum size limit from 400mmTL to 600mmTL in 1992 and the reduced recreational bag limit in 2006.

There is an uncanny similarity between the observations of geelbek and the response of the white seabass (Sciaenidae: *Atractoscion nobilis*) to changes in the management environment. The white seabass is also species with a clearly defined migratory pattern and one of the more important commercial and recreational species on the Southern Californian and Mexican coast (Allen *et al.* 2007). Larger individuals are predominantly caught in the near-shore during breeding aggregations in the months June, July and August. The juveniles remain in shallow waters while the adults of the species migrate offshore feeding on squid and fish (Thomas 1968 cited by Allen *et al.* 2007). Due to over-fishing white seabass stocks collapsed between 1980 and 1981 (catches were 10% of the historically recorded landings) and catches remained low throughout the 1990's. In 1994, fishing for white seabass was limited by depth restrictions, which protected the spawning aggregations and the juvenile fish. In 2005, natural populations of white seabass off the Californian coast had appeared to have recovered (Allen *et al.* 2007, Pondella and Allen 2008). The commercial and recreational cpue had increased significantly and attained levels comparable to those prior to the collapse in the early 1980's (Allen *et al.* 2007, Pondella and Allen 2008). Although the management method for white seabass was spatially-based, it had the same effect as increasing the minimum legal size limit for geelbek in South Africa. Both management measures (which coincidentally were implemented at roughly the same time in the 1990's) limited the exploitation of juvenile fish and it appears that both have had similar effects on the stocks. However, the recovery of white seabass was not only attributed to the management measures that were put in place in 1994 but also the succession of warm water events between 1983 and 1998 (culminating in the El Niño event in 1997-1998) (Allen *et al.* 2007). In conclusion to their work Allen *et al.* 2007 emphasised the importance of careful management of fish stocks that appear to have 'recovered' and, moreover, the importance of external factors that played a critical role in the recovery of the species.

Along the eastern seaboard of South Africa, adult geelbek have a close relationship with their pelagic prey species, *Sardinops sagax* (Griffiths and Hecht 1995). Significant fluctuations in recruitment of these species in response to oceanographic-atmospheric phenomenon (such as the El Niño southern oscillation event) have been well described worldwide (Schwartzlose *et al.* 1999, Jablonski and Legey 2005). These fluctuations have been noted to have a significant effect on the abundance of avian top predators such as the African penguin (*Spheniscus demerus*) and Cape gannet (*Morus capensis*) (Crawford and Dyer 1995, Crawford *et al.* 1998, Verlande *et al.* 2004). It is reasonable to assume that, like other top predators, fluctuating abundance of small pelagic prey species will also affect the recruitment geelbek. Therefore, despite the seeming health of the geelbek, it is essential that stocks be re-assessed coupled with an analysis of the magnitude, spatial and temporal extent of the annual 'sardine run' up the eastern seaboard. Because of the potential for recruitment fluctuation and migratory nature of the species, geelbek should not be used as an indicator of the health of the Port Alfred fishery.

Furthermore, Penney *et al.* (1999) suggested that catch rates in the KZN commercial linefishery have been largely maintained by the sequential shifting of effort from one target species to an alternative species, as the abundance of the preferred species declines. Similarly, in the absence of geelbek, commercial and recreational effort in the Port Alfred linefishery is directed towards silver kob.

Silver kob was, and still is, the mainstay of both the commercial and recreational sectors of the Port Alfred linefishery. In light of the management changes, silver kob was considered the most suitable species with which to assess the status of the Port Alfred commercial linefishery. Of all the important species in the fishery, the *cpue* of silver kob was least affected by changes in the management environment. This is because there has been no commercial bag limit and because the size limit was only recently amended (from 400mmTL to 500mmTL in 2006) and silver kob are targeted equally by the commercial and recreational sectors. Despite contributing a similar proportion of the total catch between 1985 and 2007, the *cpue* of silver kob has constantly declined over the 23 year period (Fig. 3.5).

Although the size limit of 400mmTL was in place since the first attempt at fisheries management in the 1940's, silver kob stocks have shown very little signs of improvement. The decline in the *cpue* for silver kob does not bode well for both the Eastern Cape silver kob stock (sensu Griffiths 1997b) and the fishery as a whole. It is recommended that the three silver kob stocks on the eastern seaboard be urgently re-assessed. Ultimately, this assessment should facilitate the re-evaluation of the current recreational bag limit andthe implementation of a bag-limit for commercial operators. However, this could effectively be the death knoll of the commercial line-fishery in the area.

Furthermore, the possibility of a silver kob 're-stocking' programme should be explored. Not to be confused with long-term 'stock enhancement' (whereby wild stocks are continually augmented through the input of cultured juveniles/adults at current levels of fishing pressure), 're-stocking' is the introduction of cultured juvenile fish into the fishery combined with the reduction of fishing effort with the aim of rebuilding the natural spawning stock to a level that can once again be self-sustaining and provide regular yields (Bell 2006). Although the success stories of stock enhancement and re-stocking are few and far between and those that were a success were mostly sedentary marine invertebrates (Gardner 2006, Uki 2006, Wang 2006, Lui and de Mitcheson 2008), technological advances in recent years have addressed the pitfalls of re-stocking and refuelled the interest in both stock enhancement and re-stocking programmes for fin-fish (Kitada and Kishino 2006, Palmer 2007). A preliminary study on the feasibility of stock-enhancement for dusky kob (*Argyrosomus japonicus*) in South Africa revealed that not only is it a feasible option, but there was a general willingness-to-pay by recreational anglers (Palmer 2007). Although there will undoubtedly be numerous problems (such as promoting diseases or degrading the genetic diversity of the population; Palmer 2007), the application of silver kob 're-stocking' is not too farfetched and in light of the decline of silver kob stocks over the last decade and the failure of traditional management measures to improve the stocks, such a step could be worth serious investigation.

Brouwer and Buxton (2002) suggested that commercial operators across South Africa appeared to target species that would produce the highest yield. They would however, work on a trade-off between high catches of low value fish such as carpenter and panga (Fig. 6.2) and lower catches of high value fish such as silver kob and geelbek (Fig. 6.2). In the Port Alfred linefishery the abundant, low value species (viz. panga) are generally caught on the deeper, low profile reefs further offshore. In the past, and particularly in the late 1980's and early 1990's panga were targeted due to the declining abundance of larger, more valuable species (Hecht and Tilney 1989, Badenhorst and Smale 1991). However, in 2006-08 the commercial operators would no longer travel any great distances due to high operating costs. In fact, the commercial operators operating out of Port Alfred did not go to sea on every available sea day during 2007 to 2008 (Table 4.2). In other words, the commercial fishers would simply not go to sea if they were not guaranteed catches of high value species. Panga and carpenter were seldom targeted.

Studies of panga on the Agulhus bank suggested that the stocks could sustain a considerably higher level of fishing effort and are currently underutilised (Booth and Buxton 1997, Booth and Punt 1998, Booth *et al.* 1999). Therefore, from 1985 to 2007 the decline in panga *cpue* (Fig. 3.11) was not so much indicative of a decline in actual stock abundance but merely a function of economic pressures forcing commercial operators to fish closer to the access points. Similarly, larger carpenter are also more abundant in deeper water (Hecht and Tilney 1989, Brouwer and Buxton 2002) and when the commercial vessels could afford to target them the *cpue* was high (Fig. 3.9). This suggests that, like panga, the carpenter stocks are seemingly healthy and currently underutilised due to economic constraints.

In light of this, an effort should be made to focus commercial fishing effort on these two deep water species. As economic factors are the main driving forces behind the targeting of panga and carpenter, economic tools should be considered to encourage fishing effort directed at these species. Possible solutions would be to increase the current fuel subsidies or subsidize the purchasing of 4-stroke outboard motors for the commercial vessels. This would effectively reduce operating costs and make it more viable for the commercial operators to fish in deeper water. Furthermore, in combination with a commercial bag limit for silver kob, this would help to mitigate the increasing levels of fishing pressure on the near-shore environment. An additional solution would be to harness consumer power through marketing strategies and increase the beach price of the plate-sized redfish species such as panga and carpenter.

The combination of changes to regulations, higher operating costs and the shift from commercial to recreational fishing effort has resulted in increased fishing pressure in the near-shore area with a mean depth of 10 to 40m and a greater effort on reef-associated species. The life-history of these species makes them particularly susceptible to over-exploitation (Buxton 1993, Coleman 2000). Due to the significant changes in the management environment, the long-term historical catch data (NMLS) and data collected during the various sampling programmes between 1985 and 2008 failed to accurately monitor the response of these species to management measures. The *cpue* data for these species provided little information on the actual stock abundance, and the status of these species in the Port Alfred fishery remains an unknown.

However, the declining trend in mean size of the Roman (despite increases in size limits) and blue hottentot (in the absence of a size limit)[15] suggests that there is substantial fishing pressure directed on these shallow water, reef-associated species, the extent of which still needs to be assessed.

As a means to this end, a long-term programme should be implemented that focuses on the collection of fisheries independent data (i.e. data that is not influenced by regulatory or external economic factors; Burger *et al.* 2006) with particular emphasis on the reef-associated species. This would allow for the application of more robust quantitative modelling and analyses of the fishery in response to fishing pressure. In line with an ecosystems approach to fisheries management, a more accurate representation of the fish assemblages response to fishing pressure could be obtained, as well as allowing for the further development of community-based metrics for a multi-species, multi-user fishery such as Port Alfred.

A vital tool that is needed to accomplish this would be the promulgation of one or more marine protected areas (MPA's) in the Port Alfred region. An MPA is essentially a 'no-take' reserve in which all extractive activities (such as fishing and collecting) are prohibited (Russ 2002, Begg *et al.* 2004, Hilborn *et al.* 2004). Without going into too much detail, the implementation of MPA's and effectiveness thereof, has received both praise (Buxton and Smale 1989, Russ and Alcala 1996, Cowley *et al.* 2002, Gell and Roberts 2003) and criticism (Willis *et al.* 2003, Hilborn *et al.* 2004). Like many fisheries management tools MPA's are not always applicable to every situation and have many pitfalls (Hilborn *et al.* 2004), not least of which is the supposed in-effectiveness to protect highly mobile or migratory fish stocks (Gell and Roberts 2003, Hilborn *et al.* 2004). However, irrespective of these uncertainties, the MPA's in the Port Alfred region would serve the sole purpose of providing a scientific reference point with which to monitor the impacts of fishing pressure (in particular on the reef-associated sparids). Enhancement of the fish stocks in the adjacent, unprotected areas through either adult 'spillover' or 'larval dispersal' effects and the overall biodiversity protection role would be considered a bonus.

[15] This study provides good grounds for the implementation of a size limit for blue hottentot. In Chapter 5 it was recommended that a size limit of 250mmTL, corresponding to the size at 50% maturity (Buxton and Clarke 1986) be imposed on both the commercial and recreational fishers.

When looking at the fishery as a whole there is very little empirical evidence to suggest that the changes in the management environment have had direct positive effects on the Port Alfred linefishery. In fact, over the 23 year period, the slope of the size spectra gradually got steeper suggesting that the fishery was generally becoming more dominated by smaller fish than in the past (Fig. 5.7). This was despite the increases to the size limits for most of the important species in the fishery. Together with the declining *cpue* of silver kob (Fig. 3.5) this is good evidence to suggest that, with the exception of geelbek, the fish stocks in the Port Alfred linefishery are possibly still in a declining state. Furthermore, management outcomes have been negated to some extent by the shift from commercial to recreational effort in the fishery. This is not to say that the management measures put in place have been completely ineffectual. In fact, if it were not for the reduction of effort and improved regulations the fishery would probably be in a far worse state than it currently is.

However, in light of this study it is suggested that there should be no further increase in commercial fishing effort in the Port Alfred fishery. Furthermore, movement by commercial operators between fishing areas should be forbidden or strictly controlled. As a means of achieving these ends, the first step would be to limit the commercials to only landing catch at the port of registration.

To mitigate the impact of recreational fishing, MCM should consider implementing a substantive national education programme over a three to four year timeframe which aims to educate fishers on the methods and value of catch-and-release. This could be conducted in collaboration with NGO's and the visual media.

Integral to the implementation of any of these recommendations would be increased levels of monitoring, control, surveillance and in particular enforcement of the regulations. The importance of monitoring the daily bag-limit restrictions should not be understated (in particular for the recreational sector). It is recommended that fisheries inspectors increase the frequency of inspections, have a good understanding of the regulations (and the scientific rationale behind them) and possibly make use of CCTV monitoring to enforce compliance in both the commercial and recreational sectors.

APPENDICES

Appendix A. Positively identified species recorded on more than a single occasion in the Port Alfred fishery during the periods 1985-1989,1990-1994,1995-1999 and 2000-2008.

	Species	**Common name**	**1985-1989**	**1990-1994**	**1995-1999**	**2000-2008**
Osteichthyes						
Sciaenidae	*Argyrosomus inodorus*	Silver kob	X	X	X	X
	Argyrosomus japonicus	Dusky kob	X	X	X	X
	Atractosion aequidens	Geelbek	X	X	X	X
	Umbrina robinsoni	Baardman	X	X	X	X
	Argyrosomus thorpei	Squaretail kob		X		X
Sparidae	*Argyrozona argyrozona*	Carpenter	X	X	X	X
	Boopsoidea inornata	Fransmadam	X	X	X	X
	Cheimerius nufar	Soldier/Santer	X	X	X	X
	Chrysoblephus anglicus	Englishman		X		
	Chrysoblephus cristiceps	Dageraad	X	X	X	X
	Chrysoblephus gibbiceps	Red Stumpnose	X	X	X	X
	Chrysoblephus laticeps	Roman	X	X	X	X
	Chrysoblephus puniceus	Slinger		X	X	X
	Cymatoceps nasutus	Poenskop	X	X	X	X
	Dichistius capensis	Galjoen	X		X	
	Diplodus cervinus	Zebra/wildeperd	X	X	X	X
	Diplodus sargus	Dassie/blacktail	X	X		X
	Gymnocrotaphus curvidens	John Brown	X	X		
	Lithognathus lithognathus	White steenbras		X		
	Lithognathus mormyrus	Sand steenbras		X		
	Pachymetopon aenum	Blue hottentot	X	X	X	X
	Pachymetopon blochii	Hottentot	X	X	X	X
	Pachymetopon grande	Bronze bream	X	X	X	
	Petrus rupestris	Red steenbras	X	X	X	X
	Polyamblyodon germanum	German			X	
	Polysteganus praeorbitalis	Scotsman	X	X	X	X
	Polysteganus undulosus	Seventy four	X	X	X	X
	Porcostoma dentata	Dane	X	X	X	X
	Pterogymnus laniarius	Panga	X	X	X	X
	Rhabdosargus globiceps	White stumpnose				X
	Rhabdosargus holubi	Cape stumpnose	X	X		X
	Sarpa salpa	Strepie	X	X	X	X
	Sparodon durbanensis	White musselcracker	X	X	X	
	Spondyliosoma emarginatum	Steentjie		X		X
	Polysteganus coeruleopunctatus	Trawl soldier		X		
	Argyrops spinifer	King soldierbream			X	
	Pagellus bellottii natalensis	Sand soldier	X	X	X	X
Cheilodactylidae	*Chirodactylus grandis*	Bank steenbras				X

Appendix A. cont.

	Chirodactylus brachydactylus	Twotone fingerfin/butterfish		X	X	X
Serranidae	*Epinephelus marginatus*	Yellowbelly rockcod	X	X	X	X
	Epinephelus andersoni	Catface rockcod				X
	Acanthistius sebastoides	Koester	X	X	X	X
	Epinephelus flavocaeruleus	Yellowtail rockcod				X
	Epinephelus chabaudi	Mustache rockcod				X
Polyprionidae	*Polyprion americanus*	Wreckfish	X	X	X	
Pomatomidae	*Pomatomus saltatrix*	Elf	X	X	X	X
Carangidae	*Trachurus trachurus capensis*	Horse mackerel	X	X	X	X
	Seriolia lalandi	Yellowtail	X	X	X	X
	Decapterus russelli	Indian scad				X
Scombridae	*Scomber japonicus*	Chub mackerel	X	X	X	X
	Katsuwonus pelamis	Skipjack tuna	X	X	X	X
	Sarda orientalis	Striped bonito	X	X	X	X
	Scomberomorus commerson	King mackerel				X
	Sarda sarda	Atlantic Bonito	X	X	X	X
	Thunnus albacares	Yellowfin tuna		X	X	X
Merluccidae	*Merluccius capensis*	Shallow water hake	X	X	X	X
Trichiuridae	*Lepidopus caudatus*	Buttersnoek	X			X
Triglidae	*Chelidonichthys capensis*	Cape gurnard	X	X	X	X
Haemulidae	*Pomadasys commersonni*	Spotted grunter	X	X	X	
Coryphaenidae	*Coryphaena hippurus*	Dorado	X	X	X	X
Gempylidae	*Thyrsites atun*	Snoek	X	X		X
Zeidae	*Zeus faber*	John Dory	X	X		X
Ophidiidae	*Genypterus capensis*	Kingklip	X	X	X	X
Platycephilidae	*Platycephalus indicus*	Bartailed flathead			X	
Ariidae	*Galeichthyes feliceps*	White seacatfish/Barbel	X	X	X	X
	Galeichthys ater	Black seacatfish/barbel	X	X	X	X
Scorpaenidae	*Helicolenus dactylopterus*	Jacopever		X		
Lutjanidae	*Pristipomoides filamentosus*	Rosy jobfish				X
	Lutjanus argentimactulus	River snapper		X		
Chondrichthyes						
Sphyrnidae	*Sphyrna zygaena*	Smooth hammerhead shark			X	
Lamnidae	*Isurus oxyrinchus*	Shortfin mako shark	X			X
Triakidae	*Mustelus mustelus*	Smoothhound				X
	Galeorhinus galeus	Soupfin shark	X			X
	Triakis megalopterus	Spotted gullyshark	X			X
Squalidae	*Squalus megalops*	Shortnose spiny dogfish	X	X		X
Odontaspididae	*Carcharius taurus*	Spotted ragged-tooth	X			X
Callorhincchidae	*Callorhynchus capensis*	St Joseph shark	X	X	X	X

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